Understanding the Causes of Decline in Breeding Wetland Bird Numbers in England

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1.1   Introduction

In 2007, the UK Government agreed a set of Public Service Agreements (PSAs) for the period 2008-11, including one on the Natural Environment (PSA 28) with a commitment to safeguard, value and enhance biodiversity. To measure progress against this PSA, an indicator of wild birds has been adopted that comprises an aggregate indicator including indices for breeding farmland, woodland and wetland birds. The trend data for the PSA indicators are used in the England Biodiversity Strategy and many of the species included in the indicators are also on the revised UK BAP list of priority species. To meet the biodiversity targets of the PSA (28) there is an urgent need to develop a robust and comprehensive evidence base of the population status, species resource needs, and potential drivers of decline through a combination of literature reviews and the use of statistical modelling approaches. This report reviews evidence concerning known and likely causes of decline in nine species in the wetland bird indicator; Common Sandpiper, Curlew, Dipper, Lapwing, Redshank, Reed Bunting, Sedge Warbler, Snipe and Yellow Wagtail.

Birds of the Western Palaearctic and the search engines Google Scholar and Web of Knowledge were used to carry out a comprehensive literature search for each species.

1.2   Individual Species Declines and Distributions

There is consensus among survey results that Common Sandpiper numbers have been declining since the mid-1980s. Its European breeding population is large but the species has suffered widespread declines, and undergone a moderate decrease in numbers (>10%). In the UK, the Common Sandpiper is a migrant breeder and a passage/winter visitor. It is widespread in upland areas and there are approximately 24,000 pairs in Britain. Poor breeding success and reduced survival of first-year birds over winter in West Africa have been suggested as possible factors for population declines.

Despite an increase in Curlews nesting alongside waterways, most data point towards a long-term decline. It is a widespread breeder across much of northern Europe. Some smaller populations were stable or increased during 1990–2000 but key populations in the United Kingdom, Finland and Russia all declined, and overall, the species underwent a moderate reduction in numbers (>10%). Population declines are likely to be driven by the low production of young.

Dipper have shown fluctuations in their population but have in general shown a slight decline. It is a widespread but patchily distributed resident in the more mountainous areas of Europe, in the UK being widespread in the north and west. Breeding performance has improved strongly over time; broods are on average larger, and there has been substantial reduction in failure rates of nests at the egg stage but the most recent trends still show that Dipper populations are declining, suggesting some mechanism other than breeding success may be important.

Lapwing have declined in numbers in most habitats over the last few decades. It is a widespread breeder across much of Europe although has a 'vulnerable' status due to these population declines. Declines are likely to be driven by low productivity, in part due to changes in agricultural practice, resulting in breeding performance lower than that required to maintain population stability.

Evidence suggests that Redshank are undergoing a prolonged slow decline, both in lowland grassland and coastal habitats, and within the uplands. It is a widespread breeder across Europe but has a 'declining' status. The highest densities of Redshank in Britain occurred in coastal areas of East Anglia and north-western England. Breeding densities tend to be highest on saltmarsh and coastal grassland but breeding success here can be low, due to a combination of low hatching success

resulting from predation and tidal flooding, and poor chick survival. Adult survival rates can have a large effect on the productivity required to maintain a stable population.

Reed Bunting underwent steep declines during the 1970s, followed by a period of relative stability and in recent years the population has shown signs of increasing. Reed Buntings are widespread in Britain, although are much less common in upland areas. They were formerly restricted to marsh or riverine areas but are now found on rough ground in agricultural areas. First-year (and, to a lesser extent, adult) survival decreased during the late 1970s and the 1980s and these were sufficiently large to have caused the population decline, although low breeding success may have prevented subsequent recovery.

Although Sedge Warbler shows fluctuations in its population, most datasets indicate a long-term decrease in numbers. The Sedge Warbler is a migrant breeder and passage visitor to Britain, wintering in tropical and southern Africa. Much of the year-to-year variation in population size has been driven by changes in adult survival which, in turn, are related to changes in rainfall on their wintering grounds, just south of the Sahara Desert, in the West African Sahel.

In England, numbers of Snipe have fallen rapidly since the 1970s. BBS data suggest that the UK population may be showing signs of recovery although country population trends vary. The snipe is a migrant/resident breeder and a passage/winter visitor in the UK. It is widespread, but now relatively scarce on farmland. There are a number of potential drivers in the decline of breeding snipe with habitat quality, breeding season food availability and reduced breeding success being the most likely candidates.

In recent decades, Yellow Wagtail have experienced marked declines in abundance and range in Britain, and more widely across Europe. It is a summer visitor to Britain, wintering in sub-Saharan Africa. It appears that British Yellow Wagtails are declining in range and abundance in pastoral regions, but increasing in arable regions and showing at least population stability in mixed farming regions.

1.3   Resource Requirements

The diet of all species except Reed Bunting is composed of predominantly invertebrates year round. Reed Bunting diet consists of mainly invertebrates during the breeding season, although adults and young will consume a variable amount of seeds. During the winter, the diet is composed mainly of seeds when they feed on open ground and cultivated fields.

Common Sandpiper is usually associated with clear lakes, rivers or streams, particularly fast-flowing rocky upper courses with stony, shingly, or rocky edges although time is also spent feeding on wet grassy areas. Dippers are associated with fast-flowing streams and rivers with rocks, boulders, shingle, water-falls and rock outcrops with shallow water.

The waders (Curlew, Lapwing, Redshank and Snipe) have similar broad habitat preferences, favouring wet grasslands in the lowlands and in the uplands, poorly drained moors and rough grass fields. Lapwing, and to a lesser extent, Curlew, also breed in arable fields, while Redshank are also abundant on saltmarsh. Sward height preferences vary, with Curlew preferring medium to tall (10-54cm) tussocky vegetation, Lapwing favouring a short (<15 cm) tussocky sward, Redshank showing preference for tall swards of around 10 - 40 cm and Snipe requiring soft damp ground with a 'tussocky' sward of around 10-30 cm. All four species show positive correlations with site wetness and the provision of surface water and are more likely to persist in fields where the soil conditions are wet and soft, with more standing water and soil moisture.

Yellow Wagtails are often associated with water, but this is not an essential requirement and many birds breed on dry arable farmland in the UK, especially spring-sown crops. They breed in habitats as diverse as lowland wet grassland, arable crops, heathland, upland pastures and hay meadows,

requiring heterogeneous swards to provide the appropriate foraging and nesting habitats. Reed Bunting breed on farmland, semi-natural grassland and riparian habitats, often associated with wet margins and their nests are located in thick vegetation near the ground. Oil-seed rape fields have become a favoured arable habitat. Sedge Warblers in farmland breed around the fringes of water in ponds and water courses although they also breed in arable fields. They show a preference for dense vegetation generally along the water's edge, usually avoiding wetter reedbeds in standing water.

1.4   Drivers of Population Change

1.4.1   Climate change

Fluctuations in population levels and annual adult survival rates of Sedge Warbler since the late 1960s are strongly correlated with indices of wet season rainfall in the West African winter quarters. Mortality rates of wintering Sedge Warblers increased in years with poor rainfall in West Africa and habitat availability in the winter quarters has probably been the main factor limiting the size of the population in Britain during 1963-1988. Although Yellow Wagtail also winter in Africa, there is no good evidence to suggest that this factor has had a major contribution to the population decline in this species. However, there is speculation that failure of Sahel rainfall in some years may have caused poor over-winter survival in first year Common Sandpipers as a result of lower food supplies.

Climate, impacting to increase the frequency and severity of spring and summer flooding since the early 1980s, has been implicated in the decline of Snipe at the Ouse Washes, one of the key lowland wet grassland breeding areas for this species. Spring and summer flooding is thought to reduce breeding success by decreasing the amount of time available for re-nesting following clutch loss as flooding delays the onset of nesting by up to 70 days.

For Redshank, loss of coastal breeding grounds as a result of climate change leading to sea level rise is also an issue. Current loss of saltmarsh is estimated to be more than 100 ha per year in the UK, attributable to coastal squeeze.

1.4.2   Habitat loss and degradation

The main reason for the decline of grassland waders (Curlew, Lapwing, Redshank and Snipe) is thought to be habitat destruction and reduction in the quality of habitats that remain. Breeding habitat has been lost due to changes in agricultural practices, particularly drainage and the intensification of farming. Large areas of grassland have been drained and converted into arable land. The management of much of the remaining grassland has become far more intensive over recent decades and has included extensive drainage, increased use of pesticides and fertilizers, re-seeding, earlier and more frequent mowing and increased grazing pressure. The rapid changes in sward height associated with the intensification of grassland management may affect the foraging efficiency, the availability of nest sites and susceptibility to predation. Earlier spring grass growth, earlier cutting dates and higher stocking levels have increased egg and chick mortality and reduced relaying opportunities. A reduction in the area of mixed farming systems has also reduced the availability of high-quality foraging habitat (e.g. pasture) to birds breeding in arable areas, resulting in reduced breeding success. These changes are also likely to have had an impact on Yellow Wagtail and, to a lesser extent, Reed Bunting populations and may also have been responsible for preventing Sedge Warbler populations from recovering from population losses due to poor conditions in the wintering grounds.

Agricultural improvement (drainage, inorganic fertilizing and, in some cases, reseeding) of grasslands in the uplands has also progressed rapidly. Around 20% of upland heather moorland present in England and Wales in the mid 1940s has changed due to agricultural reclamation, high grazing pressures, bracken *Pteridium aquilinum* invasion, afforestation and peat extraction. The main effects of increased grazing pressure on upland breeding birds are likely to be loss of preferred vegetation types, and the alteration of food supplies and predation pressure.

For those species which nest on arable land (Lapwing and Yellow Wagtail and to a lesser extent Curlew and Reed Bunting) the change in the timing in the sowing of crops has had an impact on habitat suitability. Spring-sown cereals were once favoured nesting crops but these have been widely replaced by autumn-sown cereals. The use of autumn crops has resulted in the creation of apparently suitable sites that quickly become untenable as crops grow above acceptable height limits. The large-scale switch from spring-sown to autumn sown crops in the 1970s also resulted in a loss of winter stubbles, a preferred feeding habitat for over-wintering Reed Buntings.

For Redshank, loss of coastal breeding grounds is also an issue. Current loss of saltmarsh is estimated to be more than 100 ha per year in the UK, attributable to coastal squeeze. Inappropriate grazing on many of the remaining saltmarshes has caused further problems. Figures suggest that inappropriate grazing management affects an area of saltmarsh three times larger than the area estimated to be lost to erosion over the next 20 years.

The loss of small wet features such as ponds, field drainage, and the dredging and straightening of rivers and streams is likely to have reduced the suitability of large areas of farmland as a breeding habitat for Reed Bunting and Sedge Warbler.

1.4.3   Pollution

There is some concern about the possible indirect effects of ivermectins on birds that feed in, or around, animal dung, such as Yellow Wagtail but the scientific evidence for these effects are limited.

1.4.4   Predation

Predation is thought to be a major driver in the declines of breeding waders and recent empirical evidence suggests that levels of predation on wader nests are unsustainably high. In many cases, predation may be limiting the recovery of wader populations, where breeding habitat is otherwise favourable. An increase in numbers of predators and changes in habitat are thought to have made wader nests and chicks more vulnerable to predation.

1.4.5   Disturbance

For ground-nesting waders, loss of eggs and chicks due to farming operations is likely to be an important cause of failed breeding attempts although losses are thought to be generally small relative to losses to predation. Losses attributable to farming operations are especially pronounced in spring cereals and stubbles where agricultural activity often coincides with the peak nesting period of Curlew and Lapwing. Loss of nests to trampling is also an issue for ground-nesting waders on grassland. Although grazing is essential to produce swards of the requisite height and structure, the presence of livestock can depress productivity through nest trampling or disturbance. The frequency and date of grass cutting is important for birds nesting in agricultural grasslands, particularly in silage fields and the increase in cultivation procedures is likely to be responsible for increased destruction of Yellow Wagtail clutches.

1.4.6   Food availability

The increased intensity of farming is thought to have resulted in reduced food supplies for breeding waders as there is evidence to suggest that management intensity influences the size abundance, availability and diversity, of invertebrates. The loss of insects associated with cattle has been cited as a possible cause of the decline in Yellow Wagtails.

Acidification of upland streams resulting from afforestation and the resulting acid deposition is cited as the main factor contributing to the declines in Dipper populations. Adults spend more time foraging on more acidic streams because of the decreased abundance of food. At sites of high acidity, reduced brood sizes, low nestling survival and the low incidence of second breeding attempts result in

a significant reduction in total productivity at acidic compared with non-acidic sites. Although more recently there have been signs of recovery from acidification, the ecological effects are marginal and there is evidence that there is still significant episodic acidification in acid-sensitive areas of Britain, which probably explains the slow biological recovery in many locations.

The most likely cause of the decline in the British Reed Bunting population is the loss of suitable food and habitat on farmland resulting from changes in agricultural practices. Granivorous species such as Reed Bunting are most likely to be limited by the problem of a late-winter 'hungry gap' in food availability while the abundance of invertebrate prey may limit breeding productivity through its effect on the number and condition of chicks fledged.

Low rainfall in the wintering grounds is thought to have resulted in decreased food supplies in these habitats for Sedge Warblers.

2.   Introduction

In November 2007, the UK Government agreed a set of Public Service Agreements (PSAs) for the period 2008-11. This included one on the Natural Environment (PSA 28) with a commitment to safeguard, value and enhance biodiversity. To measure progress against this PSA, an indicator of wild birds has been adopted that comprises an aggregate indicator including indices for breeding farmland, woodland and wetland birds. Each sub-index is assessed separately as a Department Intermediate Outcome indicator. The trend data for the PSA indicators are used in the England Biodiversity Strategy and many of the species included in the indicators are also on the revised UK BAP list of priority species. In order to meet the biodiversity targets of the PSA (28) there is an urgent need to develop a robust and comprehensive evidence base of the population status, species resource needs, and potential drivers of decline through a combination of literature reviews and the use of statistical modelling approaches. There is a great deal of research information available on the causes of declines and options for conservation of farmland birds and, to a lesser extent, of woodland and wetland breeding birds. However, to date, information on the latter two groups of birds has not been collated and synthesised in a manner that can be used to assess the main potential drivers of declines and to identify priority actions to stem and ultimately reverse these declines.

This report reviews evidence concerning known and likely causes of each of the declining species in the wetland bird indicator.

2.1   Methodology

To undertake the literature review, Birds of the Western Palaearctic and the search engines Google Scholar and Web of Knowledge were used to carry out a comprehensive literature search for each of the declining species in the wetland bird indicator (Table 1). Web of Knowledge was searched using the species name and scientific name, with all returns examined. Google Scholar was searched by common name and scientific name and then each name together with 'ecology', 'conservation', 'decline' and 'habitat' and the first 100 returns were examined. Google Scholar searches entire documents and several thousand hits are often returned, many of which are not relevant. As literature sources are listed in descending order of relevance, it was thought that examining the first 100 would ensure that evidence was obtained in an efficient manner, with minimal risk of missing relevant studies. To pick up any additional relevant studies, the species' English and scientific names were paired with 'habitat degradation', 'habitat loss', 'climate change', 'predation', 'pollution', 'competition', 'diet', 'nest destruction' and 'resource', with the first 20 returns examined for relevance.

Table 1 Species in the new Water and Wetland Breeding Bird Indicator for England. Species in bold are those addressed by this report

Species	Species
Cetti's Warbler	Mallard
Common Sandpiper	Moorhen
Coot	Mute Swan
Curlew	Oystercatcher
Dipper	**Redshank**
Goosander	Reed Warbler
Great Crested Grebe	**Reed Bunting**
Grey Heron	Sand Martin
Grey Wagtail	**Sedge Warbler**
Kingfisher	**Snipe**
Lapwing	Teal
Little Egret	Tufted Duck
Little Grebe	**Yellow Wagtail**

Species reviews are listed alphabetically.

3. COMMON SANDPIPER *ACTITIS HYPOLEUCOS*

3.1 British Population Status and Trend

BOCC status: Amber

Indicator trend: The indicator trend shows a long-term moderate decline, from 1985.

Other trend data: BBS data between 1994 and 2007 from Scotland and the entire UK show a decline (-14% and -18% respectively; Risely *et al.* 2008). Sample size was too small to generate a trend for England. WBS data from 1975 to 2005 indicate a 22% reduction in numbers (Baillie *et al.* 2007). Combined data from WBS/WBBS for England from 1974 to 2007 show a long-term decline until 2004, after which numbers have begun to increase.

Summary: Common Sandpiper numbers have been declining since the mid-1980s.
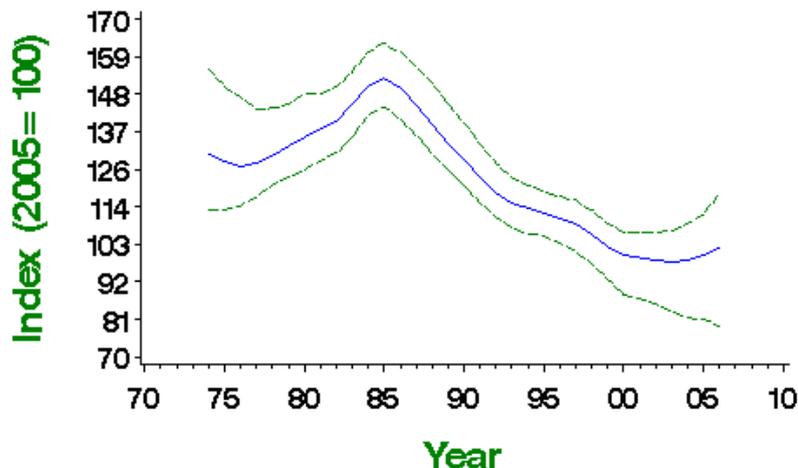
Figure 1 Waterways bird survey index 1974-2006 for Common Sandpiper. From Baillie *et al.* 2007

3.2 Distribution and Demographic Changes

3.2.1 Worldwide and European

The Common Sandpiper occurs in northern and central Europe and Asia, wintering in Africa, Asia and Australia, breeding throughout a wide band of latitudes and climatic and vegetation zones (Cramp and Simmons 1983). Its European breeding population is large at more than 720,000 pairs, and was stable between 1970 and 1990. Many European populations, including key ones in Russia and Norway, were stable or increased between 1990 and 2000 but the species has suffered widespread declines, particularly in Sweden and Finland, and undergone a moderate (>10%) decrease in numbers (Birdlife International 2004). In Estonia, numbers increased during the first half of the 1970s but then began declining, with steeper declines in the last 30 years (Tuule *et al.* 2005). Declines elsewhere in Europe during the 1990s have also been documented (Sanderson *et al.* 2006) and the European status of this species is no longer considered 'secure' (BirdLife International 2004).

3.2.2 Britain

The Common Sandpiper is a migrant breeder in the UK and a passage/winter visitor, arriving around late April, settling into territories and fledging chicks in early July. By mid-July most adults and young birds have left the breeding grounds (Holland *et al.* 1982a) and most migrate to West Africa, South of the Sahara (Holland and Yalden 2002b). There are estimated to be approximately 24,000 pairs in the Britain (Dougall *et al.* 2004). This species is widespread in upland areas north of a line from the Humber to the Servern estuaries (Ward *et al.* 1994).

The repeat Breeding Waders of Wet Meadows survey found that numbers of Common Sandpiper had undergone a small (12.5%) increase between 1982 and 2002 (Wilson *et al.* 2005). However, the range

of this species in Britain declined by 14% between the two breeding atlas surveys of 1968–72 and 1988–91, due to its disappearance from several areas peripheral to its range (Yalden 1993). In the uplands, Common Sandpipers increased in northeast Scotland, Lewis and Harris and the West Flows but decreased by 15% on the East Flows between 1980 and 2002 (Sim *et al.* 2005).

3.2.3   Changes in demographic parameters

Poorer breeding success and reduced survival of first-year birds over winter in West Africa have been suggested as possible factors for population declines (Holland and Yalden 2002a). UK clutch sizes have shown a slight decline since the 1960s but there was no change in the daily failure rates at the egg stage (Baillie *et al.* 2007).

3.3   Resource Requirements

3.3.1   Breeding season diet

The Common Sandpiper does not have any particular dietary specialisation (Yalden 1986a). It rarely probes, but locates prey visually, picking mainly from the ground, low vegetation, and from the faeces of mammals, such as sheep (Cramp and Simmons 1983, Yalden 1986a). The diet of this species mainly consists of immobile or free-flying adult invertebrates, particularly insects. Dietary analysis by Yalden (1986a) showed that 53% of species within the diet of adults consisted of 'terrestrial' groups of invertebrates, 20% were aquatic and 27% were ambiguous. Yalden also found that chicks took significantly more smaller prey species, such as prototrupoids, other hymenoptera and elaterids, whereas adults took more earthworms and aquatic prey such as mayfly nymphs, with earthworms being more important earlier rather than later in the season.

A wide variety of species may be taken, including beetles (Coleoptera), dipteran flies, bugs (e.g. Gerris, Corixa), earwigs (Forficula), grasshoppers and crickets (Acrididae, Gryllidae), stoneflies (Plecoptera), mayflies (Ephemeroptera), caddisflies (e.g. Phryganeidae), lepidopterans (e.g. *Aglais urticae*), dragonflies (Odonata) and hymenopterans (Ichneumonidae, Formicidae). It also takes spiders (e.g. water-spider Argyroneta), crustaceans, molluscs, annelid worms (e.g. earthworms Lumbricidae, nereid worms), small frogs and tadpoles (Rana), small fish, and occasionally plant material, including seeds (Scirpus, Polygonum) (Cramp and Simmons 1983 , Holland *et al.* 1982a, Yalden 1986a)

3.3.2   Winter diet

In Africa, the winter diet is similar to that during the breeding season, consisting mainly of small invertebrates. This species is known to associate with, and forages from the back of, hippopotamus (*Hippopotamus amphibious*) and crocodiles, e.g. *Crocodylus niloticus*, possibly also taking ectoparasites (Cramp and Simmons 1983). Arcas (2004) found that during the winter in Northwest Spain Common Sandpiper diet was mainly composed of three marine invertebrates, the amphipod *Talitrus saltator* (46.9%), the polychaete *Nereis diversicolor* (39.0%) and the decapod *Carcinus maenas* (6.3%) with insects being little represented, contrary to in the breeding or migratory seasons.

3.4   Habitat Associations

3.4.1   Breeding season

Common Sandpiper prefer clear lakes, rivers or streams, particularly fairly fast-flowing rocky upper courses with stony, shingly, or rocky edges (Cramp and Simmons 1983). The distribution of Common Sandpiper can be predicted by channel and flow characteristics (Buckton and Ormerod 1997, Brewin *et al.* 1998), reflecting the species' requirements for extensive areas of shoal and shingle on low-gradient streams (Yalden 1986b). Yalden (1986b) found that Common Sandpiper prefer the wider, more shingly, stretches of river. Marchant and Hyde (1980) reported that Common Sandpiper

densities were greater where gradients were below 5m/km. In some regions, this species uses water-bodies near sea-level, or even on marine inlets such as sea lochs with shingle bars, whilst elsewhere it is concentrated in uplands, with altitude being a secondary consideration (Cramp and Simmons 1983, Stillman and Brown 2002).

Fluctuations in water level can be an important factor in influencing habitat selection (Elgmork 2003) and in the Netherlands exceptionally low water levels in 1996 are thought to have contributed to relatively large numbers of breeding birds in that year (Erhart 1997). Although this species can tolerate more enclosed sites between tree stands, rocks or embankments, several studies have found that Common Sandpiper prefer streams surrounded by open areas and few trees, preferring rough pasture, moorland and riparian wetlands (Buckton and Ormerod 1997, Brewin *et al.* 1998, Vaughan *et al.* 2007).

A study in the Peak District recorded 190-210 breeding pairs of Common Sandpiper between 1977 and 1980; 78 pairs had territories beside rivers and 128 pairs were by reservoirs, indicating that both hill streams and reservoirs with shingly shores and islands were important breeding territories (Holland *et al.* 1982b). Territories are usually linear, along watercourses or lakesides. Nest sites are dry, often raised, but hidden sites, immune from flooding, normally within *c*. 50 m of water. Heavily vegetated and bare, rocky river banks tend to be avoided (Cramp and Simmons 1983), although Diez and Peris (2001) found no relationship between abundance of Common Sandpiper and vegetation coverage in Spain. A study in Estonia found that when the mowing of ditchsides ceased the number of Common Sandpiper in these areas decreased remarkably (Tuule *et al.* 2005).

Common Sandpiper spend much time feeding in grassland, although shingly river edges are also important, as are unvegetated bars (Vaughan *et al.* 2007). The feeding activity of young chicks is also largely confined to wet grassy areas where the chicks feed on short grass and wet flushes, but older chicks spend much more time feeding along the stony edges of rivers (Yalden 1986a and b). At the start of the breeding season, adults spend more time foraging on improved pasture but as the vegetation gets longer these become less suitable. Much of the foraging habitat of adults is made up of the grasslands associated with rivers (Yalden 1986b, Tuule *et al.* 2005). It has been suggested that rivers and reservoirs provide comparable feeding grounds for chicks in terms of chick weight gain in each habitat (Yalden and Dougall 1994).

Surveys in Great Britain have shown that the number of these birds is sensitive to the length of rivers and brooks (Yalden 1986a, Vickery 1988), since small brooks lack banks of pebbles or debris-rich material. However, this is also influenced by the surrounding landscape, for example Sandpipers can be seen at smaller brooks more often in Scotland than on lowland areas (Dougall *et al.* 2004).

3.4.2   Winter

The majority of this species winter in Africa although small numbers winter within the maritime climatic zone of Western Europe north to Britain and around the Mediterranean basin (Cramp and Simmons 1983). Outside breeding season Common Sandpiper will visit almost any kind of fresh water, including ditches and pools, as well as coastal lagoons, tidal creeks, mangroves, rocky shores, mud flats, sewage farms, harbours, and docks. On the coast, it uses mangroves, rocky headlands, beaches, estuaries, and refuse heaps (Cramp and Simmons 1983). In west-central Spain, higher numbers of non-breeding Common Sandpiper have been found on wider rivers (Diez and Peris 2001).

On migration, Common Sandpiper visit a wide range of shoreline habitats but appear to prefer freshwater habitats throughout their range and are scarcer in maritime habitats (Holland and Yalden 2002b).

3.5   Impacts Of Environmental Change

3.5.1   Climate change

There is speculation that failure of Sahel rainfall in some years may have caused poor over-winter survival rates of first year birds. However, two populations in the north of England were found to have different levels of recruitment and as these birds would be expected to over-winter in the same area, this suggests that other factors must also be involved; Dougall *et al.* (2005) state that a climatic influence do not fully explain why this species is in retreat.

In terms of the effects of climate change in the breeding habitat, Forchhammer *et al.* (1998) found a correlation between annual breeding numbers of Common Sandpiper and the North Atlantic Oscillation (NAO). Warmer, snowy winters were correlated with NAO and lower Common Sandpiper populations. Holland and Yalden (1991) found that late April snowstorms reduced apparent survival rates and Holland and Yalden (2002a) reported that adult survival was correlated with late April temperatures during the 1980s, although this was no longer true in the 1990s, implying a change in weather or the birds' response to it. Common Sandpiper have a short breeding season, constrained by the availability of aquatic and riparian prey (Yalden 1986a). Hatching dates can be significantly earlier when May is warmer (Dougall *et al.* 1995), which may allow birds more chance to lay replacement clutches. This suggests that they may be positive and negative effects of climate.

In Lithuania, Žalakevisius *et al.* (2006) found that moist habitat birds such as the Common Sandpiper, predominate among the birds with decreasing populations on the southern periphery of the species' range. They state that the species that are biologically linked with shallow water and moist habitats are likely to be highly threatened by climate warming.

3.5.2   Habitat loss and degradation

In Estonia, Tuule *et al.* (2005) reported that a large number of pastures have become overgrown with bushes up to the banks of rivers and as a consequence the number of Sandpipers has declined drastically. However, there is little other evidence of habitat loss and degradation being a key driver of Common Sandpiper declines.

3.5.3   Loss of nesting sites: No evidence.

3.5.4   Pollution

Aquatic pollution has been found to have little impact on Common Sandpiper as they are not totally dependent on aquatic prey. Vickery (1991) found no correlation between density of Common Sandpiper and the acidification of streams in southwest Scotland.

3.6   Impacts Of Biological Change

3.6.1   Predation

In the Borders, the poorer hatching success and chick survival in 1998 than 1999 was attributed to predation in a cooler wetter June (Mee 2001). However, few other studies discuss the role of predation in the decline of Common Sandpiper.

3.6.2   Hunting

Stiefel *et al* (1985) reported that 49% of recoveries in their study were of birds that had beens shot and only 22% had been recaptured. However, patterns in hunting may have changed in more recent times as Meissner (1997) found that 43% of recoveries in his study were recaptures and only 23% had been shot. There was no evidence assessing the impact of hunting on population dynamics but if

hunting pressure has declined it seems unlikely that this is currently a driver a population declines in Common Sandpiper.

3.6.3 Competition: No evidence.

3.6.4 Disturbance

In the Peak District, it has been argued that recreational disturbance on reservoirs is an issue (Yalden 1992). Birds in this study took flight 29% more than they would have if undisturbed and they were frequently forced to encroach on their neighbours' territories, causing far more fighting than done by nearby riverine birds. In some areas, Common Sandpipers avoided using the favoured angling beaches and as a consequence suffered from a reduction in the size of the breeding population, although breeding success was unaffected (Yalden 1992). A study in Bavaria found that reproductive success was lower in areas where pressure by human recreation was higher (Schol 2006).

Around large, varied reservoirs, some parts of the shoreline are so steep, rocky, or wooded that there are usually some refuges, and feeding birds can always fly across to the opposite shore of the reservoir so disturbance is less of an issue. However, for smaller, more open, reservoirs this is not the case, and, in the face of severe angling pressure, these reservoirs usually lack Common Sandpipers.

3.6.5 Food availability

A study in the Peak District found that reduced recruitment was responsible for hampering population recovery after a hard-weather event in 1989. Breeding success did not obviously differ, suggesting that over-winter survival of first year birds may have changed, possibly as a result of lower food supplies in wintering grounds (Holland and Yalden 2002a).

3.6.6 Summary

Poorer breeding success and reduced survival of first-year birds over winter in West Africa have been suggested as possible reasons for the population declines in this species (Holland and Yalden 2002a).

3.6.7 Possible Actions to aid recovery

Creation of vegetated islands, shoals and bars will provide nesting and feeding areas (Ward *et al.* 1994). Breeding records of Common Sandpipers in the Netherlands between 1994-96 were mostly from recently-initiated 'nature development-projects' in riverine areas where suitable breeding habitat had developed on slightly vegetated and partly wooded banks of rivers and gravelpits, indicating that habitat creation projects can be successful (Erhart 1997).

The provision of refuge areas (small, fenced, forestry enclosures, for example, or zoned sanctuaries) on reservoirs would enable this species to co-exist with recreational users.

More studies are needed to determine exactly what has been causing the population decline, including studies focussing on over-wintering populations in Africa.

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4. CURLEW *NUMENIUS ARQUATA*

4.1 British Population Status and Trend

BOCC status: Amber
Indicator trend: Declining
Other trend data: The long-term trend in England is one of probable decline and between 1970 and 2001 Curlews declined by 49% (Gregory *et al.* 2004). Combined CBC/BBS data indicate a 29% decline between 1967 and 2005 (Baillie *et al.* 2007). Curlews monitored by CBC were mostly in lowland habitats and may have been affected primarily by drainage of farmland (Gibbons *et al*. 1993). BBS data show that declines have been widespread; the short-term trend between 1994 and 2007 shows a 20% decline for England and a 36% decline for the UK (Risely *et al.* 2008). WBS data, in contrast, indicate a moderate increase in Curlews nesting alongside waterways during the 1980s, followed by a more stable period, resulting in a 21% increase between 1980 and 2005 (Baillie *et al.* 2007). Wintering Curlew abundance has shown a shallow (7%) long-term increase (Banks *et al.* 2006).
Summary: Despite an increase in Curlews nesting alongside waterways, most data point towards a long-term decline in this species.
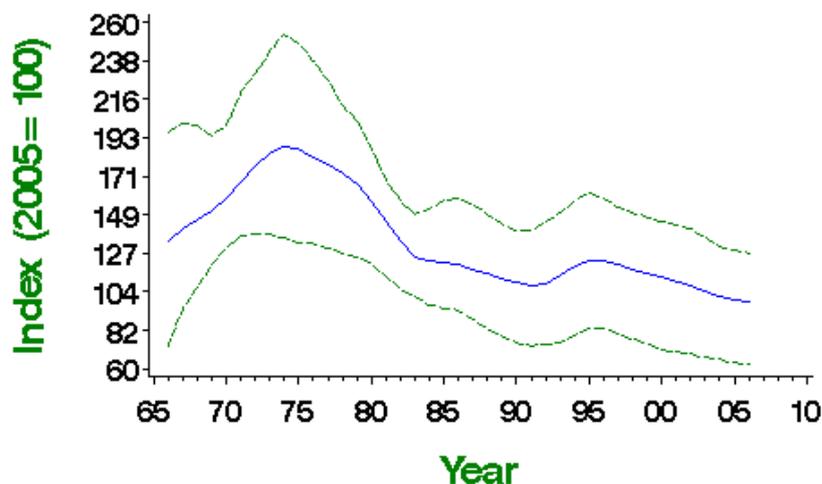
Figure 2 CBC/BBS trend for Curlew in England 1966-2006. From Baillie *et al.* 2007

4.2 Distribution and Demographic Changes

4.2.1 Worldwide and European

The Curlew is a widespread breeder across much of northern Europe, in upper middle latitudes, extending north as far as the sub-arctic, but otherwise is found in boreal, temperate, and steppe zones and occurring more patchily farther south (Cramp and Simmons 1983). Europe probably holds more than 75% of the global breeding population, at more than 220,000 pairs, although this underwent a moderate decline between 1970 and 1990 (based mainly on wintering data). Some smaller populations were stable or increased during 1990–2000 but key populations in the United Kingdom, Finland and Russia all declined, and overall, the species underwent a moderate reduction (>10%) in numbers (Birdlife International 2004). Around 40% of the wintering population has declined so reductions in breeding numbers may be underestimated (Tucker and Heath 1994). Declines have been reported in most European countries, linked with habitat changes, though in some areas, this may be partly offset by shifts to cultivated land.

4.2.2   Britain

Curlew underwent a marked range expansion in the 20th century especially in southern and eastern England but also in the Outer Hebrides and Ireland, with increased breeding in lowland areas, including cultivated farmland (Sharrock 1976). The UK population size was estimated to be 107,000 (99,500–125,000) pairs in 1985–99 (O'Brien 2005, Birdlife International 2004, Baker *et al* 2006).

A sample of sites surveyed in 1982 for the Breeding Waders of Wet Meadows survey were re-visited in 1989 and this showed no evidence of a change in numbers of Curlew (O'Brien and Smith 1992). However, surveys on wet meadows in England and Wales between 1982 and 2002 revealed a 39% decrease in breeding Curlew, and showed that their populations were mainly concentrated in the north and west of England, being virtually absent from Wales (Wilson *et al.* 2005). This survey also showed that occupancy of sites had decreased, from 19% to 12%. In Northern Ireland, a breeding decline of around 60% occurred between the mid 1980s and 1999 (Henderson *et al.* 2002). Curlew declined significantly in 50% or more of study areas in British uplands between 1980-1991 and 2000-2002 (Sim *et al.* 2005), although Henderson *et al.* (2004) found no change in Curlew status on upland CBC sites in Britain between 1968-80 and 2000.

4.2.3   Changes in demographic parameters

Population declines are likely to be driven by the low production of young (Berg 1992a, 1994a). Although samples are small, failure rate of nests at the egg stage have fallen slightly (Baillie *et al.* 2007).

4.3   Resource Requirements

4.3.1   Breeding season diet

Curlew principally forage on invertebrates. From the littoral zone, Curlew feed on a range of species including polychaetes (Annelida), crustaceans, bivalve molluscs and occasionally small fish, including *Pomatoschistus microp*s. Inland, it takes insects, especially adult and larval beetles (e.g. Carabidae, Histeridae, Staphylinidae, Hydrophilidae, Scarabaeidae), flies (Tipulidae larvae and pupae, Orthorrhapha and Cyclorrhapha), grasshoppers, locusts, crickets, and occasionally earwigs (Forficula), bugs (e.g. Corixa), lepidopterans (e.g. larvae of Hepialus), caddisflies (Trichoptera), dragonflies (Odonata), mayflies (Ephemeroptera), and ants (Formicidae). Curlew also feed on earthworms (Lumbricus, Allolobophora), freshwater shrimps (Gammarus), woodlice (Isopoda), spiders (Araneae), and occasionally vertebrates including small fish, frogs and toads, lizards, young birds and possibly eggs, and small rodents. Plants include mosses, horsetail Equisetum, seaweeds (Ulva), berries of blackberry (Rubus), bilberry (Vaccinium), cranberry (Oxycoccus), and crowberry (Empetrum), and leaves and grain of cereals and grasses (Cramp and Simmons 1983, Berg 1993).

Earthworms were found to be the most important food organisms during the pre-breeding period in Sweden, when a significantly higher number of earthworms were caught per minute in sown grassland than in tillage, while surface-living invertebrates played a minor role (Berg 1993). Significantly fewer earthworms were caught in the breeding period.

Feeding often occurs outside the breeding territory, either in the adjacent area, or at sites some distance away. A study in Sweden found that sown grassland was used significantly more than expected for foraging early in the season, possibly indicating the strong influence of the nutritional requirements in the pre-breeding period on territory establishment. The same fields were mostly preferred in the pre-breeding period also, suggesting that nests were built close to good foraging areas (Berg 1992b).

4.3.2 Winter diet

Seasonal variations in diet partly reflect the seasonal differences in habitat. During September to March, observations indicated diet composed chiefly of polychaetes (probably mainly *Nereis diversicolor* and *Cirriformia tentaculata*), and bivalve molluscs (*Scrobicularia plana*), with occasional crabs *Carcinus maenas* (Goss-Custard *et al.* 1977). In studies in France and Germany, *Scrobicularia plana* and ragworms were the main prey items in winter (Boileau *et al* 2002, Rippe and Dierschke 1997) while Curlew feeding on intertidal muddy areas in Cadiz Bay, Spain fed mainly on a crustacean decapod, the Shore Crab *Carcinus maenas*. Other prey included *Hydrobia* spp., fish, amphipods and seeds, but these were much less abundant. On one occasion, the remains of a small mammal, possibly a rodent, were found (Perez-Hurtado *et al.* 1997). *Carcinus niaenas* was the most important prey species in almost 50% of the diets; polychaetes formed about 25 % in a study from the German Waddensea (Peterson and Exo 1999).

Curlew wintering in the Banc D'Arguin, Mauritania, fed on the lower part of the beach and on the sandflats on small fiddler crabs *Uca tangeri*. Curlews on *Zostera* ate gobiid fish and small specimens of the Blue Crab *Callinectes marginatus,* both of which were taken from tidal pools (Zwarts *et al.* 1990).

4.4   Habitat Associations

4.4.1   Breeding season

Curlew favour moist poorly drained upland moors and rough grass fields, having dry nest-sites and damp feeding areas close together. Curlew also breed in traditionally managed hayfields, damp pastures overgrown with rushes, raised bogs, heaths, slacks among sand-dunes, and farmland sown to cereal crops, potatoes and turnips (Cramp and Simmons 1983, Berg 1992a, Bracken *et* al 2008). The nest site is on the ground, often on a tussock or low hummock or in grass or growing crops. Numbers on lowland wet grassland are a fraction of the UK population as this is essentially an upland-breeding wader (Wilson *et al.* 2005). Curlew prefer open landscapes with wide visibility, unbroken by forest or woodland and other features permitting surprise approach, although it tolerates low shrubs and coarse herbage, provided they are sufficiently spaced to permit ready movement.

On lowland wet grasslands in England and Wales, Wilson *et al.* (2005) found that Curlews were more likely to be present on northern sites and sites containing hay and a range of ungrazed grass. Dry sites with some hay and arable held the highest densities. At a landscape scale across English and Welsh farmland, Atkinson *et al.* (2002) found that breeding Curlew were most abundant in 10 x 10 km squares with more than 50% cover of grassland. In other countries, Curlew have also been found to prefer breeding areas with a high proportion of grassland, while dry tillage tends to be avoided (Berg 1992b, Valkama *et al.* 1998).

The height and density of vegetation, frequency and extent of wet patches, the nature and frequency of disturbance, and the gradualness of change in traditionally occupied territories are significant influences on habitat selection. Curlew prefer 'medium to tall' vegetation, between 10 – 45 cm tall with a tussocky sward and heterogeneous vegetation cover (Valkama *et al.* 1998, Pearce-Higgins and Grant 2006, Durant *et al.* 2008). In a study in the uplands, the density of breeding Curlew was positively correlated with rush density (Baines 1988) and Curlew showed positive associations with vegetation typical of wet conditions on moorlands in northern England and southern Scotland (Pearce-Higgins and Grant 2006). On grazing marshes in France, Curlews were found to occupy marshes with higher intensity of spring grazing, although they preferred fields with a lower than average annual grazing intensity overall (Titchit *et al.* 2005).

In Sweden, Curlews preferred to nest on grassland and fallow fields, where hatching success was higher than on tillage fields, and nests were situated further from forest edges than random sites (Berg 1992a). The highest density of Curlews was found at a mixed farmland site and the lowest density at a

modern arable site. These differences in bird densities were attributed to the effects of differences in the area of grasslands, which were important foraging and nesting habitats, and wetness between sites. In this study, Berg found that production of young Curlews on farmland dominated by cereal crops was too low to maintain a stable population and it was speculated that birds disperse to modern farmland from bogs, where reproductive success was much higher (Berg 1992a, Berg 1994a and b). However, Curlew abundance was found to be negatively related to the cover of bog in a study in the uplands in Scotland (Brown and Stillman 1993).

Tharme *et al.* (2001) found that Curlew occurred at higher densities on grouse moors than on other moors, and there was a positive effect of heather burning as a management tool. Density of Curlew on grouse moors may have increased where high sheep grazing has resulted in a change in the vegetation cover from heath to grassland (Jenkins and Watson 2001). Haworth and Thompson (1990) found that Curlew favoured intensively keepered moorland, possibly because of freedom from predation or disturbance from walkers.

Within the uplands of the South Pennines, Curlew were more abundant on low-altitude flat areas and showed associations with tall heather and wet flush (Brown and Stillman 1993, Stillman and Brown 1994), although Haworth and Thompson (1990) found that Curlew in a similar area favoured the high plateau.

4.4.2   Winter

Directly after the breeding season, Curlew shift to mainly marine coastal habitat, especially mudflats and sands extensively exposed at low tide, resting on adjoining saltmarshes, foreshores, and floodlands (Cramp and Simmons 1983). A study in the Orkney Islands showed Curlew preferred areas with muddy substrates and sheltered coasts (Summers *et al.* 2002).

4.5   Impacts Of Environmental Change

4.5.1   Climate change

The winter distribution of Curlew in the UK appears to have shifted in a northeasterly direction, and this was proposed to be as a response to changing winter temperatures (Maclean *et al.* 2008). Curlew have shown greater per annum declines towards the southern limit of their British breeding range, providing support for the hypothesis that climate change has influenced the abundance changes that have been detected. However, it is likely that several different factors have caused these changes (Sim *et al.* 2005) and furthermore, changes in distribution do not necessarily equate to changes in overall population numbers.

The growth of opportunistic green algae may be promoted by climate change; in the Ythan Estuary of northeast Scotland macroalgal blooms were correlated with higher spring temperatures (Hughes 2004). Algal mats have negative effects on mudflat invertebrates, particularly *Corophium*, and when algal mats were common, the food intake of Curlew *Numenius arquata* declined (Hughes 2004).

Although there is limited evidence that climate change has played a role in the decline of the Curlew population to date, changes in future climate may be important. Predicted climate change could result in loss of coastal grazing marsh to managed realignment of sea defences and increased frequency of spring floods or droughts, which are likely to have an impact on wader populations on lowland wet grassland in the future (Wilson *et al.* 2004). A modelling study of over-wintering Curlew in Poole harbour predicted that survival would be affected by lower temperatures and sea-level rise. A predicted 10 cm reduction in the height of intertidal patches reduced survival rates while a reduction of 40 cm meant that no birds at all survived the winter (Durell *et al.* 2006). Another study on the Humber estuary found that a 2 to 8 % reduction in intertidal area (the magnitude expected through sea level rise and industrial developments) resulted in decreased predicted survival rates of Curlew (Stillman *et al.* 2005).

4.5.2   Habitat loss and degradation

Curlew have undergone substantial habitat changes, most recently through drainage, land reclamation, and increasing human pressures (Cramp and Simmons 1983). The main reason for the decline of grassland waders in the EU is stated to be habitat destruction, as breeding habitats have been lost due to changes in agricultural practices, particularly drainage and the intensification of farming (Hotker 1991, Jensen and Lutz 2007). Habitat loss due to other factors, mainly enlargement of harbours, industrial developments, and road building, is the second most often mentioned reason for the decline of grassland waders (Hotker 1991).

Along with most other agricultural habitats, the management of grassland has become far more intensive over the last few decades and has included extensive drainage, increased use of pesticides and fertilizers, re-seeding, earlier and more frequent mowing and increased grazing pressure (see Fig. 3). The rapid changes in sward height associated with the intensification of grassland management may affect the foraging efficiency and the availability of nest sites for breeding Curlew (Vickery *et al.* 2001). Over the last 50 years, the area of land (both lowland and upland) under pasture has been reduced from 50% to less than 35% (Wakeham-Dawson and Smith 2000). Wilson *et al.* (2005) recorded a 27% decrease in the area of hay meadows and a 23% decrease in cattle grazing between 1982 and 2002. Earlier spring grass growth, earlier cutting dates and higher stocking levels may have increased egg and chick mortality and reduced relaying opportunities. Wilson *et al.* (2004) state that the declines in waders on lowland wet grasslands are intrinsically linked to drainage and other changes in grassland management.

Agricultural improvement (drainage, inorganic fertilizing and, in some cases, reseeding) of grasslands in the uplands has also progressed rapidly in recent years. Around 20% of upland heather moorland present in England and Wales in the mid 1940s has changed under agricultural reclamation, high grazing pressures, bracken *Pteridium aquilinum* invasion and afforestation. Of that remaining, 70% is estimated to be at risk of change, with at least 50% in 'poor' or 'suppressed' condition liable to further reductions and damage under sheep *Ovis aries* grazing (Thompson *et al.* 1995). In Northern England, densities of Curlew fell from 10 to 1 pair per km^2 between the years 1985 and 1987 as marginal farmland was converted to improved grassland (Baines 1988). Curlew declined by 82 and 36% following pasture and meadow improvement respectively (Baines 1988). Baines (1989) also found that the breeding success of Curlew was lower on improved grasslands, with 59% of birds hatching chicks on unimproved grassland and only 23% hatching chicks on improved grassland. Afforestation in the uplands has also resulted in habitat loss and Curlew has been displaced over substantial areas (Fuller and Ausden 2008). Afforestation can be detrimental not only through loss of moorland, but also by its fragmentation effect and by providing cover for predators (Avery and Leslie 1990, Thompson *et al.* 1995 in Sim *et al.* 2005).
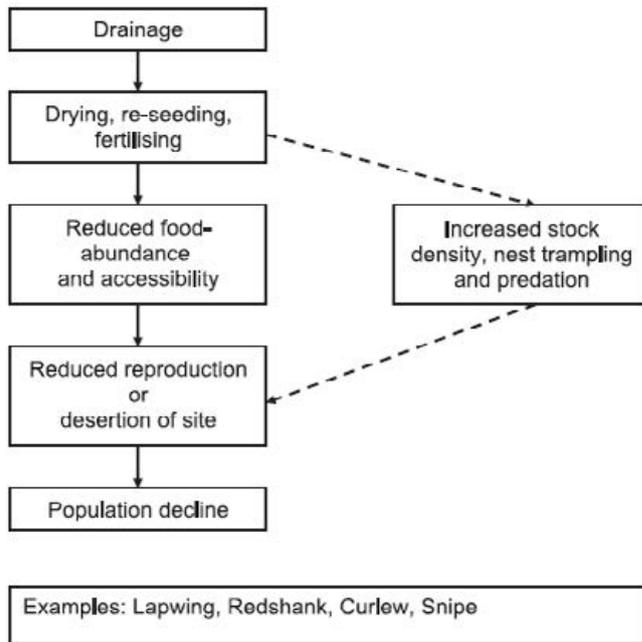
Figure 3 Proposed sequence of events through which the draining and re-seeding of wet grassland causes population declines in waders. From Newton *et al.* 2004.

There have been large changes in grazing practices, both in the lowlands and uplands, the most striking of which was the large increase in numbers of sheep, which more than doubled between 1950 and 1990 (Fuller and Gough 1999). The strongest increases were in Wales and northern England, with relatively little change in the Scottish Highlands (Fuller and Gough 1999). The resulting heavy grazing almost certainly reduced the habitat quality for ground-nesting birds such as Curlew (Fuller and Gough 1999, Fuller and Ausden 2008). There is also evidence of a long-term increase in Red Deer numbers in the Scottish highlands (Clutton- Brock *et al.* 2004 in Sim *et al.* 2005). The main effects of increased grazing pressure on upland breeding birds are likely to be loss of preferred vegetation types, and the alteration of food supplies and predation pressure. Within Britain, the increase in sheep numbers between 1969 and 1988 was highest in Wales, north England and south Scotland - regions in which some of the most serious wader declines were found, suggesting that increased grazing pressure may have had an adverse effect on upland breeding wader populations (Sim *et al.* 2005).

In Europe, the decline of the Swedish Curlew population since 1950 has also been attributed to changes in land use, resulting in decreased grassland area and increased habitat fragmentation, which have probably had negative affects on both breeding and foraging possibilities (Berg 1992b). Changes in land use and habitat fragmentation are probably the main reasons for the decline of this species in farmland, since they directly affect both food availability and reproduction (Berg 1994b). In Finland, Valkama *et al.* (1998) suggest that lack of suitable habitats in intensively cultivated farmland areas may be one reason for poor reproductive success and decline of Curlew populations.

In a study of over-wintering Curlew in Poole harbour, Curlew survival was predicted to be affected by loss of terrestrial habitat (Durell *et al.* 2006) although Curlew mortality and body condition were predicted to be unaffected by a 20% reduction in the area of mudflats on the Seine estuary (Durell *et al.* 2005).

4.5.3   Loss of nesting sites

Habitat loss and degradation (see above) have resulted in a reduction in the quantity and quality of nesting habitat.

4.5.4   Pollution

There is no evidence that pollution has been a major factor in the decline of Curlew populations. A study by Currie and Valkama (1998) found that prey biomass and abundance, and foraging success did not vary along a pollution gradient and there was no difference in adult weight during incubation or measures of breeding success between polluted and non-polluted areas. They thus concluded that there were minimal immediate effects of heavy metal pollution on foraging and breeding success in the Curlew. Denker and Buthe (1995) found only low levels of PCB in Curlew eggs in Germany and also concluded that this was not a problem.

4.6   Impacts of Biological Change

4.6.1   Predation

Several studies have found predation to be the most important reason behind failed breeding attempts (Berg 1992a, Grant *et al.* 1999, Valkama and Currie 1999, Valkama *et al.* 1999). In an area of arable farmland in southwest Finland, nest predation (81%) was the most important reason for failed breeding attempts (Valkama and Currie 1999, Valkama *et al.* 1999). Valkama *et al.* (1999) found that the majority of nests where predator type could be identified were predated by mammals. They also found that the intensity of nest predation varied between their two field sites, and suggested that area differences in predator densities and landscape features were probably important in determining this. In Northern Ireland, nest predation accounted for 85-97% of nest failures and 74% of chick mortality (Grant *et al.* 1999). The authors suggest that predation rates on Curlew nests in Northern Ireland may have increased in recent decades and suggest that increases in mammalian and avian nest predators may have contributed to this, or changes in habitat that may have made Curlews more vulnerable. Curlews have been found to be less successful in defending their nests against predators in short vegetation (van der Wal and Palmer 2008), suggesting that over-grazing may have an impact. Predation may also be higher in more southerly latitudes (Berg 1991).

Baines *et al* (2008) found that numbers of Curlew on a moor in Scotland decreased when grouse moor management was discontinued. Increased Crow numbers, together with an increase in Red Foxes *Vulpes vulpes*, probably contributed to the observed declines. Amar *et al.* (2008) found no evidence to suggest that raptor predation limited the population of Curlews on an upland sporting estate in Scotland and inferred that concerns over the impact of natural densities of hen harriers on vulnerable upland waders are unjustified.

In a farmland area in Finland, Curlew nest predation was lower close to Kestrel nests than further away, and Curlews tended to nest closer to Kestrel nests than expected by chance, suggesting that the breeding association of Curlews and Kestrels was a behavioural adaptation against nest predation (Norrdhal *et al.* 1995). Berg (1996) found that distance to forest edge did not influence the predation rate of nests but in terms of survival of chicks, chick predation was higher for clutches that hatched close to forest edges (Berg 1992a). Chick survival was found to be higher on tillage and lays than on meadows (Berg 1992a) and Berg attributed this to higher predation rates of chicks on meadows.

4.6.2   Hunting

The Curlew is listed on Annex II/2 of the EU Birds Directive as a species for which hunting can be permitted in Denmark, France, Ireland and UK. Since 1994, there have only been open seasons for Curlew in France, Ireland and Northern Ireland. Of the recoveries of birds with known causes in the Migration Atlas, 70% were categorized as being deliberately taken by man (Bainbridge 2002). Hunting is not considered to be a major cause for the decline in the EU but the shooting that takes place in France, where most Curlew are taken, may have consequences for those segments of the populations that pass through (Jensen and Lutz 2007).

Increases in the number of Curlews in the north and northeast of Scotland occurred after 1981/82, when the species stopped being a legal quarry species (Austin *et al.* 2000). Overall, the number of Curlew taken annually in the EU is estimated to be between 13,000 and 18,000 birds (3 – 4 % of the European/EU wintering population; Jensen and Lutz 2007). It has been argued that hunting during migration has been an important factor in the decline of the Fenno-Scandian Curlew populations since 1950 and about 20,000-50,000 Curlews and Whimbrels *Numenius phaeopus* (mainly juvenile curlews) were shot annually between 1950 and 1981, with a decrease in the number shot during the 1960s and 1970s (Berg 1994a). After postponement of the start of the hunting season to 1st September in 1982, the annual bag decreased by half, to about 9700 individuals (Henriksen, 1991). These changes in the Danish hunting, and the ban on hunting this species in Great Britain, may have increased the survival of Curlews.

4.6.3 Competition

No information

4.6.4 Disturbance

Destruction of eggs and chicks on agricultural grasslands is a problem for Curlew. In a study in Sweden, the second most important factor accounting for nest loss was destruction by farming practices, which was especially important in tillage early in season (Berg 1992a). In an area of arable farmland in southwest Finland, nest losses due to agricultural practices (16%) were an important reason for failed breeding attempts (Valkama and Currie 1999, Valkama *et al.* 1999). Trampling by cattle caused 7% of all nest failures in a study in Northern Ireland (Grant *et al.* 1999) and was an important factor in sown pasture where it destroyed 55% of the pasture nests in a study by Berg (1992a). Losses to farming operations and trampling are likely to have increased with higher stocking rates and agricultural intensification.

Potential disturbance by humans has been associated with the absence of breeding Curlew on uplands in the South Pennines (Haworth and Thompson 1990). Over winter, sustained disturbance associated with footpaths, roads, railroads and development can reduce local habitat quality for waterbirds and the carrying capacity of estuaries (Burton *et al.* 2002a and b). Numbers of Curlew were found to be significantly lower when a footpath was close to a count section in a study on six English estuaries (Burton *et al.* 2002a). Disturbance of feeding birds both day and night was predicted to have a significant effect on mortality and body condition, as did disturbance of roosting birds on the Seine estuary, France (Durell *et al.* 2005). The presence of hand harvesters on an estuary in northern Spain had a significant effect on foraging activity of Curlews but there were no significant differences in feeding rate or foraging speed between days with and without harvesting, suggesting that mud disturbance by harvesters working by hand did not affect Curlew fitness (Navedo and Masero 2007 and 2008). Habituation may be an important response to regular but benign disturbance, reducing the disturbance-induced decrease in feeding time to a level which may be compensated for, at least in summer within the normal late-tide foraging period (Fitzpatrick and Bouchez 1998).

4.6.5 Food availability

The increased intensity of farming may have resulted in reduced food supplies for Curlew as there is evidence to suggest that management intensity influences the size, as well as the abundance and diversity, of invertebrates. Intensive grassland management with high inputs of fertilizer and intensive grazing or mowing may be particularly detrimental to larger insect species (Beintema *et al.* 1990) and this will have an impact on foraging efficiency. Mowing greatly reduces invertebrate populations in ground and herb layers, and winter flooding decreases biomass of soil and epigeous invertebrates (Ausden Sutherland and James 2001), although soil macro-invertebrates benefit from organic fertilizers (Behrens *et al.* 2007).

The availability of surface-living invertebrates to Curlews can be affected by both invertebrate density and vegetation structure. Berg (1993) found that a large number of surface-living invertebrates were caught in tillage but here their biomass was relatively low and the vegetation sparse. In contrast, in sown grass (with relatively high and dense vegetation), the biomass of invertebrates was high, but the number caught by Curlews per minute was low, despite many foraging pecks and probes. Although the biomass may be less, the greater availability of food items in grasslands and flooded tillage than in dry tillage early in the season is probably partly responsible for the higher densities of Curlews in grasslands and flooded tillage (Berg 1993).

A study in Poole harbour found that wintering Curlew had low prey biomass densities and were likely to be affected by reductions in their food supply (Durell *et al.* 2006) and in a study on the Severn estuary, Curlew densities were correlated with the densities of their larger-sized prey (Goss-Custard *et al.* 1991). Mechanical harvesting of cockles results in the loss of a significant proportion of the most common invertebrates and causes a sufficiently large mortality of non-target invertebrates, as such, harvesters should be excluded from areas of conservation importance for intertidal communities such as invertebrates, fish and birds (Ferns *et al.* 2000).

4.6.6 Summary

Loss of breeding habitat through changes in land-use and agricultural practices and nest predation are considered the most significant causes responsible for the decline of Curlew. It is likely that these factors interact to reduce breeding success. Human disturbance on breeding and wintering areas is believed to be of secondary importance.

4.6.7 Possible Actions to aid recovery

Priority actions are the reduction in intensive farming to improve conditions on breeding habitats and protection of wintering habitat. Curlews are a relatively widespread species and to have an impact on the population of this species through a site-based scheme would be expensive, meaning that agri-environmental measures working through a targeted system, by improving conditions across farmland, is a more realistic approach (O'Brien and Bainbridge 2002, Wilson *et al.* 2004).Van de Wal and Palmer (2008) suggest that agri-environment schemes may reap the greatest benefits for waders through reducing stocking density, where avian predation pressure is high. Since the abundance of avian predators is largely outside individual farmers' control, sward height should not be reduced to the extent that predation risk is increased where otherwise good quality wader habitat is present. The problem of over-grazing of moorlands also needs to be addressed.

4.7 References

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5.   DIPPER *CINCLUS CINCLUS*

5.1   British Population Status and Trend

BOCC status: Green.
Indicator trend: The long-term trend for this species is a fluctuating trend, with most recent years seeing a decline.
Other trend data: BBS data for the entire UK show a 12% decline in numbers between 1994 and 2007 (Risely *et al.* 2008). WBS data show that Dipper populations have fluctuated over the last thirty years, but shown little overall trend (7% decline between 1975 and 2005; Baillie *et al.* 2007).
Summary: Evidence points to a slight overall decline in Dipper populations.
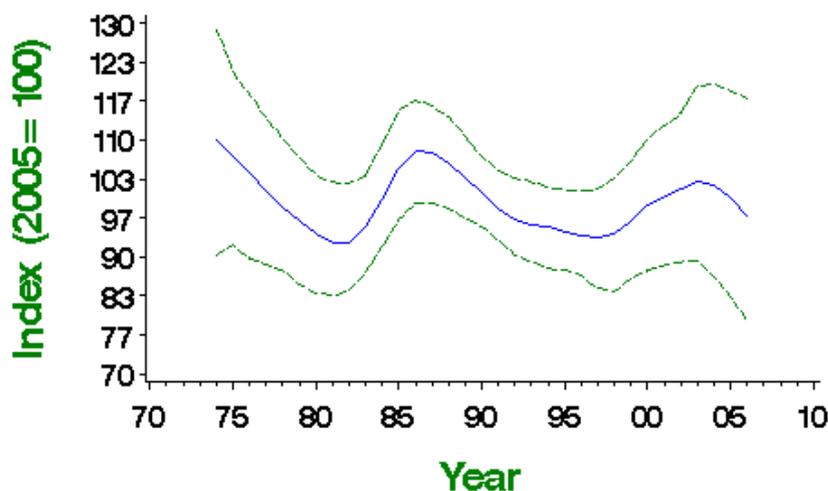
Figure 4 Waterways Bird Survey trend for Dipper in England 1974-2006. From Baillie *et al.* 2007

5.2   Distribution and Demographic Changes

5.2.1   Worldwide and European

The Dipper is a widespread but patchily distributed resident in the more mountainous areas of Europe, accounting for less than half of its global range. Its European breeding population is more than 170,000 pairs, and was stable between 1970 and 1990. Although there were declines in a few countries during 1990–2000, populations were stable across the majority of its European range, including the key one in Romania. Consequently, its European status is evaluated as 'Secure' (Birdlife International 2004).

5.2.2   Britain

Dipper are widespread in the north and west of the UK but are absent from England east of a line from the Humber to the Isle of Wight (Ward *et al.* 1994). The UK population size was estimated at 6,800–20,000 pairs in 2000 (Birdlife International 2004, Baker *et al.* 2006). Dippers decreased by 100% on upland CBC sites in Britain between 1968-80 and 2000 (Henderson *et al.* 2004).

5.2.3   Changes in demographic parameters

Breeding performance has improved strongly over time, and laying dates have become earlier, maybe because of climate change (Crick and Sparks 1999). Broods are now on average larger, and there has been substantial reduction in failure rates of nests at the egg stage but the most recent trends still show that Dipper populations are declining (Baillie *et al.* 2007).

5.3   Resource Requirements

5.3.1   Breeding season diet

This species feeds on large invertebrates, especially Caddis Fly larvae (Cramp 1988). Diet also includes the following; springtails (Collembola), Protura, two-pronged bristle-tails (Diplura), silverfish (Thysanura), adult and nymphal mayflies (Ephemoptera), dragonflies (Odonata), adult and nymphal stoneflies (Plecoptera), adult crickets and grasshoppers (Orthoptera), earwigs (Dermaptera), adult and nymphal bugs (Hemiptera), lacewings (Neuroptera), butterflies and moths (Lepidoptera), adult and larval flies (Diptera), adult and larval wasps and ants (Hymenoptera), adult and larval beetles (Coleoptera), spiders (Araneae), shrimps (Amphipoda), isopods (Isopoda), daphnia (Phyllopoda), crayfish (Astacus), millipedes (Diplopoda), centipedes (Chilopoda), leeches (Hirudinea), earthworms (Oligochaeta), flatworms (Platyhelmenthes), roundworms (Nematoda), snails (Gastropoda), bivalves (Bivalvia). They may also take fish (minnows *Phoxinus phoxinus*, loach *Cobites barbatulatus*, bullhead *Cottus gobio*) and eggs of salmon *Salmo salar* while plant material may be taken irregularly (Cramp 1988, Ormerod 1985a and b; Ormerod *et al.*. 1985, Ormerod and Tyler 1991, Breitenmoser-Wursten, 1997, Santamarina 1990)

The diet of young is similar to the adults', largely consisting of insect larvae supplemented by adult insects, crustaceans, molluscs, and fish. In southwest Scotland, adult birds showed a strong preference for caddisfly larvae when feeding nestlings between 4 and 10 days old (Vickery 1998). Mayfly nymphs predominate in the diet of chicks up to 9 days old with the proportion of caddis fly larvae increasing up to day 15 (Ormerod 1985a). Overall, caddis fly larvae are the most numerous prey in diet of young (Ormerod 1985a, Breitenmoser-Wursten, 1997, 1999). Yoerg (1998) found that young Dippers forage very differently from adults, specializing on small, stationary prey, such as simuliid larvae (Diptera). Large items such as caddis flies declined in adult diet but increased in nestling diet as the chicks grew, probably reflecting the growing energy demands of the brood (Ormerod *et al.* 1987).

5.3.2   Winter diet

Fish, molluscs and crustacea were important at circumneutral sites during the winter in a study in Wales and Scotland (Tyler and Ormerod 1992). Diet has been found to differ significantly between spate and base-flows during the winter (Taylor and O'Halloran 2001). During baseflow conditions, diet was dominated by trichopteran larvae, whereas during the spate period diet was composed primarily of ephemeropteran nymphs, and dipteran larvae and contained greater proportions of prey taxa of terrestrial origin. Enhanced hydrological cycles with greater precipitation, predicted under current climate change models, may have an effect on the diet of Dippers (Taylor and O'Halloran 2001).

5.4   Habitat Associations

5.4.1   Breeding season

Most Dipper populations are resident and are associated with fast-flowing streams and rivers with rocks, boulders, shingle, water-falls and rock outcrops with shallow water (Ward *et al.* 1994, Roche and D'Andurain, 1995, Cramp 1988). Cramp (1988) states that typical habitats tend to have little aquatic or bankside vegetation, although Buckton and Ormerod (1997) found that Dippers showed a preference for sites with 'tree-related features' while Logie *et al.* (1996) found a positive relationship between dipper density and bankside deciduous cover. Ormerod *et al.* (1986) found that sites without breeding Dippers tended to have more coniferous forests in their catchment.

Within south-west Scotland, the breeding density of dippers was positively correlated with pH and negatively correlated with gradient (Vickery 1991). High numbers of Dippers often occur on waters where there the gradients are steep and there is a large area of riffle (Marchant and Hyde 1980, Round

and Moss 1984, Vaughan *et al.* 2007). In Wales, over 40% of the variation in the abundance of Dippers was explained by gradient, with more Dippers being found on the steeper gradients (Ormerod *et al.* 1985). There were strong correlations between breeding abundance and combinations of variables describing stream-gradient and the relative abundances of larval Trichoptera and nymphal Plecoptera in a study in Wales (Ormerod *et al.* 1985).

Dipper nest sites are restricted to some extent in that they are almost always over rushing water. However, Dippers are extremely flexible and have been noted to use a range of man-made and natural sites, including rock faces, bridges, low mud banks on boulders in the middle of streams or nested in trees (Ward *et al.* 1994). Dippers may show a significant preference for streams with bridges and walls as these are often used as nesting or roosting sites (Buckton and Ormerod 1997).

5.4.2 Winter

As for summer

5.5 Impacts of Environmental Change

5.5.1 Climate change

Dipper populations in southern Norway appear to have been positively influenced by climate; a trend towards warmer winters appears to allow easier access to foraging streams and subsequent increases in population size. Fewer individuals were recruited after cold winters (Sæther *et al.* 2000). A stochastic model showed that an expected change in climate will strongly affect the dynamics of the population, leading to a nonlinear increase in the carrying capacity and in the expected mean population size (Saether, *et al.* 2000). Loison *et al.* (2002) also found an affect of climate. In their study, 80% of the variance in apparent survival rates was explained by winter temperature and population size for adults, and 48% by winter temperature for chicks.

According to historical data, the initiation of reproduction for Dipper on Lake Zurich now occurs earlier in the year and this is cited as being due either to global warming, or to an increase in water temperatures as a result of urbanisation. Warmer waters accelerate life cycles of aquatic arthropods and generate earlier emergences of insects that form the prey base of the Dipper. Since avian reproduction is cued by trophic resources, an earlier appearance of aquatic insects will induce an earlier breeding season for the Dipper (Hegelbach 2001).

Although evidence suggests that to date, Dipper populations appear to have benefited from climate change, in the future, spring macroinvertebrate abundance may decline by 21% for every 1°C rise (Durance and Ormerod 2007). This could dramatically reduce food supplies available to Dipper. Climate change may also have an impact directly, through altered survival during abnormally high or low flows in summer, when adult mortality peaks naturally and may be exacerbated by floods or droughts. Unfortunately, quantitative evidence about these effects is limited (Pers. Comms. Steve Ormerod).

5.5.2 Habitat loss and degradation

Afforestation has contributed to the acidification of streams, resulting in reduced food supplies (see sections below on 'Pollution' and 'Food supply').

5.5.3 Loss of nesting sites

This is not likely to be a factor in Dipper decline (Vickery 1991).

5.5.4 Pollution

Acidification

Acidification of upland streams as a result of afforestation and the resulting acid deposition is cited as the main factor contributing to the historic declines in Dipper populations (Fuller and Ausden 2008). Breeding density and productivity of Dippers are lower on acidic than more neutral streams and in streams with higher aluminium concentrations (Ormerod and Tyler 1991 Ormerod *et al.* 1986, Buckton *et al.* 1998, Logie 1995, Logie *et al.* 1996, Vickery 1991). Adult mass was significantly lower on acidic streams than on circumneutral streams and second clutches were absent (Ormerod *et al.* 1991). Adults have been found to spend more time foraging on more acidic streams, presumably because of the decreased abundance of food (O'Halloran *et al.* 1990, Logie 1995, Tyler and Ormerod 1992, Vickery 1992). Vickery (1992) reported that at sites of high acidity reduced brood sizes, low nestling survival and the low incidence of second breeding attempts resulted in a significant reduction in total productivity at acidic compared with non-acidic streams. Ormerod and Tyler (1993a) also found that broods at acidic sites were significantly smaller than at circumneutral sites and that Dippers at acidic sites bred significantly later, but post-fledging survival declined most rapidly through the season at the former. The pH-related differences in the invertebrate fauna of streams probably result in low prey availability for Dippers on acidic streams.

According to simulations carried out by Ormerod and Tyler (1989), only reductions in sulphate deposition by over 50% of 1984 levels would prevent a decline in the number of streams chemically suitable for Dippers in Wales. Greater reductions in deposition in the model permitted some recovery except where conifer forestry occupied acid sensitive catchments.

Acidification can also affect the eggs of Dippers. Dipper eggs in an area suffering from acidification in Norway were significantly thinner and lighter than eggs from less acidified areas (Nybø *et al.* 1997). This trend was also found by Ormerod *et al.* (1988) for Dipper eggs in Wales and Scotland, although neither study found any significant influence on hatching success.

Although more recently there have been signs of recovery from acidification, the ecological effects are marginal (Ormerod and Durance 2009) and there is evidence that there is still significant episodic acidification in acid-sensitive areas of Britain, which probably explain the slow biological recovery in many locations (Kowalik *et al.* 2007).

In contrast to studies in Scotland and Wales, a study in Sweden found no apparent effects of stream acidity on breeding Dippers (Alhund *et al.* 1999) and a repeat survey by Buckton *et al.* (1998) found that gains and losses of Dipper numbers were not related to habitat structure or acid-base status, but might be stochastic. In Ireland, afforestation was found to have no major impact on stream chemistry (Giller and O'Halloran 2004) and Dipper populations appear to be unaffected by acidification there (Smiddy *et al.* 1995).

Other pollution

A study in Italy found that in 93% of cases, Dipper occurred in unpolluted reaches and were absent from most polluted or strongly polluted streams. Water quality degradation in the study area, especially in the hill streams, was reported to be causing Dippers to contract to higher altitudes (Sorace *et al.* 2002).

A study in Wales found significant variation between catchments in pollution from the insecticides Aldrin and Dieldrin, which reflected land use. Levels of contaminants found in eggs increased significantly where river catchments had high sheep densities and these patterns are consistent with the former legal or latterly illicit use of Dieldrin in sheepdip (Ormerod and Tyler 1993b). However, neither laying dates, clutch-sizes, brood-sizes, nest failure nor hatching failure along a contaminated river in Wales differed significantly from those along adjacent reference rivers or Welsh rivers as a whole, suggesting that PCBs can occur in Dipper eggs without effects on breeding performance and survival (Ormerod *et al.* 2000). There is some evidence of low to medium levels of contaminants

found in Dipper eggs, but there is only slight evidence of any toxic effects. These include moderate shell thinning in relation to increasing DDE, and some evidence that contaminants had contributed to egg failure (Ormerod and Tyler 1992). Population effects by PCBs on European dippers are therefore unlikely.

5.6 Impacts of Biological Change

5.6.1 Predation

There has been relatively little work into the effect of predation on Dipper populations. In southwest Norway, nest predation was higher in nests visible from above than covered nests; nest survival was 49% for open nests and 68% for covered. It was estimated that 56% of the nests produced fledged young. Predation was the most important factor in reducing nesting success (Efteland and Kyllingstad 1984).

Predation by the alien mink has wiped out the population in some parts of Shropshire, and this too will affect the potential of the species to recover and re-colonise the local streams (Smith 2006). Domestic predators, such as cats, may also have an impact on Dipper numbers. It is reported in the Migration Atlas that 39% of recoveries of dead birds were due to domestic predators (Conway 2002). Another 11% were due to natural predators.

5.6.2 Competition

No information

5.6.3 Disturbance

No information

5.6.4 Food availability

As discussed above (see section on 'Acidification'), stream chemistry may have an indirect effect on the density of Dippers via changes in food quantity and/or quality as many of the invertebrates known to be important in the diet of Dippers are sensitive to pH (Ormerod 1985a, Ormerod and Tyler 1987, Ormerod et al. 1987, Vickery 1991). Abundance and diversity of invertebrate species within water courses increased with increasing pH in a study in south-west Scotland (Vickery 1991). Other studies have also reported that acidic streams exhibit a scarcity of mayflies, caddisflies, molluscs and crustaceans, and a predominance of stoneflies (ie see Hildrew et at 1984, Kimmel et al. 1985 in Vickery 1991). Preferred prey at acidic sites may be smaller (Tyler and Ormerod 1992). Sites that lost Dippers were those with the highest declines of Plecopteran abundance in a study contacted by Buckton et al. (1998) in Wales. Reduced food abundance has been suggested as one factor responsible for the scarcity of Dippers at acidic sites (Ormerod et al.. 1986; Ormerod and Tyler 1987, Tyler and Ormerod 1992) as there are particularly marked reductions at low pH in the density of fish, trichopteran larvae and ephemeropteran nymphs, all groups of importance in the diet of dippers (Ormerod and Tyler 1987).

5.6.5 Summary

Decreased food availability as a result of a decrease in the quality of riverine feeding habitats is the most likely cause of the population declines in Dipper.

5.6.6 Possible Actions to aid recovery

Providing pools and riffles with rocks and boulders will help create feeding areas. Food availability can be influenced by river regulation construction and this should be avoided during the breading

season of Dipper. Afforestation with coniferous trees should be avoided in areas known to be important breeding areas. There has been some work done looking at the effectiveness of liming acidic streams and the conclusion was that liming had limted success as a restoration measure (Ormerod and Durance 2009).

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6.   LAPWING *VANELLUS VANELLUS*

6.1   British Population Status and Trend

BOCC status: Red list
Indicator trend: The long term trend is of moderate decline.
Other trend data: BBS data indicate a shallow increase in England (5%) since 1994, but steep decline in Scotland (-38%) while data for the entire UK show an overall decline of 18% between 1994 and 2007 (Risely *et al.* 2008). Although CBC recorded some increase in its early years, Lapwings have declined continuously on lowland farmland since the mid 1980s. Henderson *et al.* (2004) reported no change in the status on upland CBC sites in Britain between 1968-80 and 2000. Combined BBS and CBC data for the UK show a 33% decline and a 9% decline for England between 1967 and 2005 while WBS data shows a 9% decline from 1980 to 2005 (Baillie et al 2007). Winter numbers counted by WeBS, mainly at coastal sites and omitting some big concentrations inland, have increased in Britain since the mid 1980s but decreased in Northern Ireland (Banks *et al.* 2006); these birds are mainly of continental origin.
Summary: Overall, this species appears to have undergone a long-term decline in numbers.
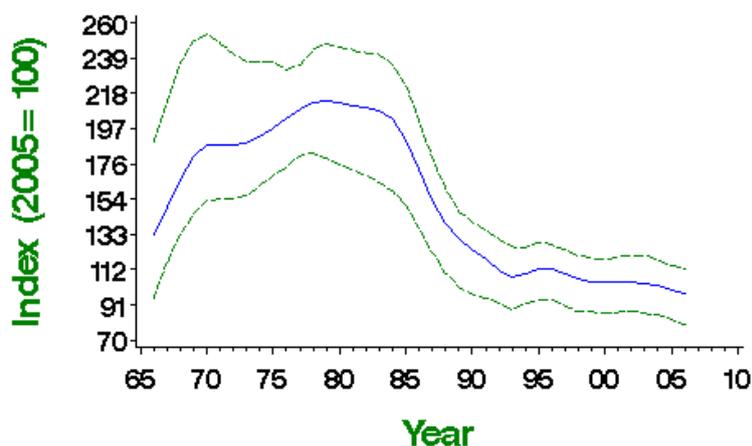
Figure 5 CBC/BBS trend for Lapwing in the UK 1966-2006. From Baillie *et al.* 2007

6.2   Distribution and Demographic Changes

6.2.1   Worldwide and European

This species occurs in upper and lower middle latitudes, in both continental and partially oceanic regions, from boreal and temperate to steppe and Mediterranean zones. It is a widespread breeder across most of Europe, which probably holds more than 50% of its global population. The European breeding population is very large (>1,700,000 pairs), and was stable between 1970 and 1990 (Birdlife International 2004), although declines were reported across some countries (e.g. Moller 1983, Hotker 1991). From 1990 to 2000 several small populations were stable or increased, but the species suffered declines across much of Europe, most notably in the UK, Netherlands and Russia, and underwent a large decline (>30%) overall. Consequently, this previously species is now evaluated as Vulnerable (Birdlife International 2004).

6.2.2   Britain

The UK population size was estimated at 156,000 (137,000–174,000) pairs between 1985 and 1999 (Birdlife International 2004, O'Brien 2005, Baker *et al.* 2006). Lapwing were found to be widely spread throughout Britain, being present in 83% of British 10 km squares during the 1988-92 Breeding Bird Atlas survey (Gibbons *et al.* 1993) and 50% of the 1 km squares surveyed in Scotland (O'Brien and Bainbridge 2002). Gregory *et al.* (2004) found that the UK population had declined by

41% between 1970 and 2001, while the shorter-term decline between 1994 and 2001 was 18%. Historically, Lapwing decreased during the 19th century, followed by some recovery from 1926 after improved protection (Cramp and Simmons 1983). There were widespread declines in 1940s, continuing in many areas, probably due to changes in agriculture (Cramp and Simmons 1983). The decreases have been chiefly in south-west England, west Wales and north-west Scotland. There was a 9% decrease in the number of 10 km squares occupied by Lapwing between the 1970 Atlas and the 1990 Atlas, but over this time there was a 47% decrease in the breeding population, indicating that although this species remained widespread, its density had decreased (Fuller *et al.* 1995).

In England and Wales, Nicholson (1938) estimated the Lapwing population as 175,000 pairs and this had declined to 123,000 pairs by 1987 (Shrubb and Lack 1991), with a further decline of 49% between 1987 and 1998 when the population was estimated as 62,923 pairs (Wilson *et al.* 2001). On lowland wet grassland sites in England and Wales, Lapwing declined by 38% between 1982 and 2002 and although present throughout these two countries, their populations were concentrated on a few key sites (50% of the population were recorded on less than 8% of the area; Wilson *et al.* 2005). In British uplands, Lapwing declined significantly in 50% or more of study areas between 1980-1991 and 2000-2002 (Sim *et al.* 2005) and experienced population declines of more than 50% over 15 years in Northern Ireland (Henderson *et al.* 2002), mirroring similar declines throughout grassland areas of Wales and southeast England (Wilson *et al.* 2001, 2005).

Over winter immigrants from the continent swell the British wintering population of Lapwings in late autumn, with an estimate 1.5 to 2 million individuals in 1981-92 (Baker *et al.* 2006). This number can vary markedly, as Lapwings make hard weather movements and large numbers may leave the country to avoid severe frost and snow (although local movements to southwest England are more usual).

6.2.3   Changes in demographic parameters

Adult and first-year survival rates show no trend through time (Peach *et al.* 1994, Catchpole *et al.* 1999). Mean clutch size increased significantly as the population size fell (Baillie *et al.* 2007) but there have been several very poor years for egg-stage survival since 1996 (Leech *et al.* 2007). Population declines are likely to be due to lower nest and chick survival, meaning that breeding productivity dropped below a sustainable level (Hudson *et al.* 1994, Siriwardena *et al.* 2000, Besbeas *et al.* 2002, Newton. 2004, Milsom 2005).

6.3   Resource Requirements

6.3.1   Breeding season diet

This species requires ready access to preferably moist soil carrying an appreciable biomass of surface or subsurface organisms (Cramp and Simmons 1983). The diet of Lapwings consists primarily of Mollusca, Annelida, Arachnida, Orthoptera, Hymenoptera, Diptera and Coleoptera. The most important prey groups are slugs (Mollusca), earthworms (Annelida), spiders (Araneae), Chironomidae (non-biting midges), leatherjackets (Tipulidae larvae), ants (Formicidae), ground beetles (Carabidae), weevils (Curculionidae), click beetles (Elateridae), dung beetles and chafers (Scarabeidae), grasshoppers (Acrididae), Buprestidae and Tenebrionidae (Beintema *et al.* 1991, Ausden *et al.* 2003). Soil invertebrates tend to be more important in the early part of the breeding season, but as the ground becomes harder Lapwings tend to switch to feeding upon surface invertebrates (Baines 1990).

Lapwing chicks feed on surface dwelling organisms and soil invertebrates and exploit fauna living inside cow pats, including most of the invertebrate classes listed above (Beintema *et al.* 1991). Sheldon (2002) found that the body condition of Lapwing chicks was correlated with the number of earthworm setae per gram of faecal sample, suggesting that earthworms were a desirable food for chicks. It is likely that larger chicks can not grow up on insects alone and need to switch to more profitable prey such as earthworms (Beintema *et al.* 1991). However, the proportion of aquatic invertebrates in the diet has been found to increase as the breeding season progresses, while that of

soil invertebrates decreased and aquatic invertebrates in shallow pools can be an important component of the diet of breeding Lapwing in May and June (Ausden *et al.* 2003).

6.3.2   Winter diet

For Lapwing feeding on arable land over winter, Gillings (2003) found that numerically, arthropods (mostly Carabids and millipedes) were the main diurnal prey types but by biomass, small earthworms were the major prey items. Invertebrates, including small adult and larval Carabidae, Staphylinid beetles, and millipedes, remain the most important food resource throughout the winter, with earthworms providing a large proportion of diet. Diurnal intake rates were highest during mild weather and on grass and sugar beet stubble fields; they were lowest on cereal crops, yet this was the habitat most consistently occupied (Gillings and Sutherland 2007).

6.4   Habitat Associations

6.4.1   Breeding season

Lapwing occupy characteristic natural sites such as fens, bogs, marshes, floodland fringes, and other wetlands, where the vegetation is short, as well as low heath or moor, especially with grass patches, and chalk, limestone, or other upland grasslands. Related semi-natural sites include badly drained or abandoned pastures, rough rushy grazings with bare patches, and permanent pasture, meadows, and hayfields (Cramp and Simmons 1983). Pressures on preferred habitats may partly account for efforts to colonize other novel types such as airfields, race courses, golf links, burnt grasslands, spoil deposits from mining, refuse dumps, and gravel-pits, in cases where indiscriminate human disturbance is prevented (Cramp and Simmons 1983).

In England and Wales, breeding Lapwings are closely associated with farmland, with 95% of the total population occupying this habitat (Wilson *et al.* 2001). Wilson *et al.* (2001) showed that Lapwings had a strong preference for spring tillage, while rough grazing was also favoured in some regions. Permanent grassland was used in proportion to its availability, but ley grassland and autumn tillage were largely avoided. In addition, habitat diversity was positively associated with Lapwing presence, reflecting the fact that mixed farming regimes provide the most suitable agricultural landscapes for this species (Wilson *et al.* 2001). It was also shown that whereas Lapwings occupied 18% of spring-tilled fields that adjoined grassland, only 5% of spring-tilled fields with no adjacent pasture were occupied (Wilson *et al.* 2001). Lapwings often favour proximate grassland habitat near their nesting site, because it is a preferred chick-rearing habitat (Wilson *et al.* 2001; but see Sheldon 2002). Higher densities of Lapwing were present on sites that contained more ungrazed grassland, sheep grazing and arable land than those with more hay and silage production (Wilson *et al.* 2005).

Lapwing are known to prefer nest-sites with short or no vegetation with rare or occasional tussocks (Cramp and Simmons 1983, Galbraith 1988; Berg *et al.* 1992, Milsom *et al.* 2000, 2002, Durant *et al.* 2008). Milsom *et al.* (2000) showed that the optimum sward height for Lapwing on coastal grazing marshes was 3-5 cm, although Vickery *et al.* (1997) found a greater range of vegetation in breeding fields of 6-14 cm. Sheldon *et al.* (2005) demonstrated that in an arable region of England, vegetation height at the nest differed significantly between crop types, with average sward height being greatest (7 cm) in winter cereals and shortest (8 cm) in spring-sown crops. In winter cereals, specifically, crop height at the nest was significantly shorter than at random points within the same field (Sheldon *et al.* 2005). Sheldon *et al.* (2005) suggested that winter cereal sward structure is generally too tall and dense at the onset of the Lapwing's breeding season to represent a suitable nesting habitat. Grey-brown or grey-green tints of spring vegetation apparently allow birds to discriminate in favour of fields where growth will not be too tall later in breeding season (Cramp and Simmons 1983). On grazing marshes in France, Lapwings occupied fields with a lower than average annual grazing index, a higher than average intensity of spring grazing and a higher than average autumn stocking rate, from which it is inferred that Lapwing habitat suitability is improved by intense autumn grazing (Tichet *et al.* 2005). Likewise, Lapwing breeding numbers increased significantly concurrent with the application of cattle grazing in a study in Denmark between 1990 and 2001, most likely due to an

increased number of nesting sites and/or improved predator detection (Olsen and Schmidt 2004). It may be that there is a requirement for an irregular vegetation cover to make nests less detectable to predators (Redfern 1982).

In the uplands, Lapwings also favour short, damp swards, rushy pastures and rough grazing. They avoid leys, improved grass and unenclosed heath/moorland (Galbraith *et al.* 1984; Baines 1988). These preferences reflect two mechanisms. First, short swards enhance the ability of the birds to see and capture prey, regardless of prey density. Secondly, damp habitats increase availability of earthworms. Indeed, Lapwings are associated with fields with areas of surface flooding or wet features (Vickery *et al.* 1997, O'Brien 2002, Milsom *et al.* 2000, 2002, Wilson *et al.* 2005, Durant *et al.* 2008, Eglington *et al.* 2008a, Kalhert *et al.* 2007). However, a fine balance needs to be achieved between a field that is damp enough that worms approach the soil surface, and a field that is so damp that earthworms are effectively 'drowned' (O'Brien 2002, Ausden *et al.* 2001).
Tharme *et al.* (2001) found that Lapwing occurred at higher densities on grouse moors than on other moors and that there was a positive effect of heather burning as a management tool. Density of Lapwing on grouse moors was found to have increased where high sheep grazing had resulted in a change in the vegetation cover from heath to grassland (Jenkins and Watson 2001).

Vegetation height is also important for chicks and Lapwing chicks on agricultural grasslands generally prefer to forage in short rather than long swards. In a study of chick foraging behaviour, Devereux *et al.* (2004) demonstrated that feeding rates declined significantly as sward height increased. As surface invertebrate abundance did not vary substantially between different sward heights in this study, the reduced prey intake rate of chicks in long swards was therefore interpreted as being a consequence of either diminished chick mobility or lower food accessibility (Devereux *et al.* 2004).

Another key requirement of breeding Lapwing relates to site wetness and Lapwing prefer wetter sites with more surface water (Vickery *et al.* 1997, O'Brien 2002, Milsom *et al.* 2000, 2002, Wilson *et al.* 2005, Durant *et al.* 2008, Eglington *et al.* 2008a, Kalhert *et al.* 2007). Studies on the North Kent Marshes ESA and the Broads ESA found that adult Lapwing nested closer to wet features and settling densities were highest in marshes with a greater extent of surface flooding (Milsom *et al.* 2002, Eglington *et al.* 2008a). Most adult Lapwing however, were seen feeding on open turf (Milsom *et al.* 2002). A study in the Netherlands found that differences in the number of Lapwing observations were largely accounted for by differences in soil moisture or groundwater depth (Verhulst *et al.* 2007). Lapwing chicks have a preference for feeding in wet features (Milsom *et al.* 2002, Joiner 2002, McKeever 2003, Eglington 2008), where their feeding rates can be three times higher than on open turf (Milsom *et al.* 2002), and they feed on aquatic invertebrates within these features (Ausden *et al.* 2003).

Winter flooding can also be important, as like grazing, this can reduce vegetation height and is also expected to create more wet foraging areas (Ausden *et al.* 2001). However, Olsen and Schmidt (2004) found that Lapwings were unaffected by winter flooding and hypothesized that the reason for this was most likely to be found in the short distances from the Lapwing breeding grounds to a shallow lake, that apparently held sufficient foraging areas (i.e. mud flats).

Other factors influencing the breeding distribution of Lapwing include the presence of power lines and field size. A modelling study undertaken by Milsom *et al.* (2000) found that the probability of a marsh being occupied increased linearly with its area and marshes under power lines or national grid cables were less likely to be occupied. In a study comparing conventional agriculture to organic agriculture in Finland, Lapwing density was significantly positively associated with landscape openness and area under organic management, and negatively with the presence of rural settlement and the number of barns. The preference for organic farming probably reflects the increased amounts of food there (Piha *et al.* 2007).

6.4.2   Winter

Generally, Lapwing winter in many of the similar areas to which they breed (Cramp and Simmons 1983) although large-scale changes in the wintering distribution of Lapwings have been witnessed in recent decades (Gillings 2003). Whereas previous winter surveys showed the greatest concentrations of Lapwings in central England (Lack 1986), there has been a shift in the predominant winter distribution towards the east such that many birds now winter in East Anglia (Gillings 2003). Amelioration of winter conditions appears to be driving these distributional changes, but they simultaneously have repercussions for broad patterns of winter habitat selection in this species due to the polarisation of farming across an east-west axis in Britain (Gillings 2003). Lapwing were previously known to winter largely in grassland habitats, but now most occur within the arable zone (Gillings 2003). In England and Wales, 25% of Lapwing are associated with arable crops in winter, 25% with grassland, and 15% with plough and harrow. Inevitably, there are regional disparities in habitat associations, with greater use of crops in east and west England, and more use of pasture in Wales and northern England. Relative to their availability at the national level, Lapwing exhibited the strongest positive selection for sugar beet stubbles, followed by grass pasture, other stubble and bare tillage, while there was only marginal selection for cereal crops. Cereal stubble and maize were avoided (Gillings 2003). Gillings (2003) found that the importance of cereal crops increases progressively during the winter following its increasing availability in the landscape. However, in relation to the availability of different habitats within the study site, cereals were neither positively nor negatively selected (Gillings 2003). Harrow/drilled soil was only selected during autumn and early winter. Gillings (2003) suggested that the majority of wintering Lapwings in the study area foraged in cereal fields, which is by far the most abundant habitat type, but make opportunistic use of ephemeral habitats such as harrow and sugar beet fields as they become available.

On the Orkney Islands, Lapwing showed a preference for muddy substrates and tended to avoid sand, boulders and gravel shores (Summers *et al.* 2002).

6.5   Impacts of Environmental Change

6.5.1   Climate change

Some studies have documented the negative effect of severe winters (Catchpole *et al.* 1999), and more specifically of low winter soil temperature and low winter rainfall (Peach *et al.* 1994), on Lapwing survival. The increase in Lapwing mean annual survival since 1960 (Peach *et al.* 1994) coincides with the general upward trend in the NAO index since then (see Hurrell *et al.* 2001). Other studies have found winter severity to be of variable importance in influencing Lapwing breeding numbers (Olsen and Schmidt 2004).

A study in the Netherlands found that warmer springs lead to faster grass growth and hence earlier mowing dates. However, it was also found that warmer springs lead to earlier hatching dates and that because of this, the negative effects of early mowing on the breeding success were less than expected (Kruk *et al.* 1996).

Beintema (1994) looked at rainfall patterns and annual variations in condition indices of chicks and found that chick condition was correlated with total rainfall in May. Similar fluctuations in chick survival were also correlated with rainfall in May suggesting a relationship between condition index and fledging success. 'Good' and 'bad' years tended to coincide with wet and dry springs.

Although there is limited evidence that climate change has played a role in the decline of the Lapwing population to date, changes in future climate may be important. Predicted climate change could result in loss of coastal grazing marsh to managed realignment of sea defences and increased frequency of spring floods or droughts, which are likely to have an impact on wader populations on lowland wet grassland in the future (Wilson *et al.* 2004).

6.5.2. Habitat loss and degradation

Over time, the natural habitat of Lapwing has been encroached upon by man, who has simultaneously created potentially suitable substitutes, largely by farming. However, increasingly these have been affected by changing technology and practice which have reduced the quantity and quality of breeding habitat available. For Lapwings, habitat losses to agriculture and intensification of farming are likely to have played a role in the decline of this species by reducing breeding productivity (Hotker 1991, Shrubb 2007, Fuller and Ausden 2008). Wilson *et al.* (2004) state that the declines in waders on lowland wet grasslands are intrinsically linked to drainage and other changes in grassland management.

Along with most other agricultural habitats, the management of grassland has become far more intensive over the last few decades and has included extensive drainage, increased use of pesticides and fertilizers, re-seeding, earlier and more frequent mowing and increased grazing pressure (see Figure 3, in Snipe review). Since the 1960s, high fertilizer inputs, drainage and silage production all increased the capacity of grassland to carry livestock (Fuller and Ausden 2008). Fertilisation has a significant effect on field occupancy by Lapwings and Durant *et al.* (2008) found that Lapwing prefer fields with no, or low level, fertilisation. Earlier spring grass growth, earlier cutting dates and higher stocking levels have increased egg and chick mortality and reduced relaying opportunities. The rapid changes in sward height associated with the intensification of grassland management may affect the foraging efficiency and the availability of nest sites for breeding Lapwing (Vickery *et al.* 2001). This loss in habitat quality has been accompanied by a reduction in quantity and over the last 50 years the area of land (both lowland and upland) under pasture has been reduced from 50% to less than 35% (Wakeham-Dawson and Smith 2000). Wilson *et al.* (2005) recorded a 27% decrease in the area of hay meadows and a 23% decrease in cattle grazing between 1982 and 2002. Mixed farming systems are now much scarcer than 50 years ago and polarization (concentration of arable land in eastern UK and pasture in western UK) has reduced the availability of high-quality foraging habitat, i.e. pasture, to birds breeding in arable areas, resulting in reduced breeding success (Hudson *et al.* 1994)

These processes have also occurred in the uplands and agricultural improvement of upland grasslands has progressed rapidly in recent years resulting in marked reductions in the density of, and in the proportion of fields used by Lapwing (Baines 1988). In the uplands, afforestation has also resulted in habitat loss. Fuller and Ausden (2008) state that there is no doubt that many species of open habitats, especially breeding waders such as Lapwing, have been displaced over substantial areas by afforestation in the uplands.

There have been large changes in grazing practices, both in the lowlands and uplands. The most striking of these changes has been the recent large increase in numbers of sheep, which have more than doubled between 1950 and 1990 (Fuller and Gough 1999). The strongest increases were in Wales and northern England, with relatively little change in the Scottish Highlands (Fuller and Gough 1999). The resulting heavy grazing almost certainly reduced the habitat quality for ground-nesting birds such as Lapwing (Fuller and Ausden 2008). Local extinction of this species was associated with increases in cattle (all 10 km squares and the pastoral region) and decreases in rough grazing (arable region; Chamberlain *et al.* 2000).

On arable land, spring-sown cereals were once favoured nesting crops but these have been widely replaced by autumn-sown cereals (Fuller and Ausden 2008). The use of autumn crops has resulted in the creation of apparently suitable sites which quickly become untenable as crops grow above acceptable height limits (normally c. 8 cm in dense stands and c. 15 cm in more open crops), although higher in some which are unusually well-spaced.

In other countries within Europe, a lack of management has resulted in degradation of breeding habitat. In the 20th century, grasslands were the primary nesting habitat for many wader species in Estonia with the highest diversity and abundance of waders on managed coastal meadows (Lilleleht 1998, in Pehlak and Lohmus, 2008). Concurrent with the overgrowth of meadows due to ceased

grazing, short-grass dwellers (such as Lapwing) have decreased since the 1950s, while numbers on inland mires have been gradually increasing (Pehlak and Lohmus, 2008)

6.5.3 Loss of nesting sites

Habitat loss and degradation have resulted in loss of nesting habitat (see above).

6.5.4 Pollution

No information.

6.6 Impacts of Biological Change

6.6.1 Predation

Increased nest predation has been suggested as a possible additional cause of Lapwing population declines and predation pressure is frequently implicated as a major cause of nest failure (Bellebaum 2002, Sheldon 2002; Chamberlain and Crick 2003, Milsom 2005, Bolton *et al.* 2007a). The proportion of nests lost to predators was substantially higher in the 1990s than in previous decades (Sharpe *et al.* 2008). Important predators of Lapwing nests include Red Fox *Vulpes vulpes*, Badger *Meles meles* and Carrion Crow *Corvus corone* (Bolton *et al.* 2007b). Recent empirical evidence suggests that levels of predation on wader nests are unsustainably high in many cases, even in some situations where breeding habitat is otherwise favourable (MacDonald and Bolton 2008a). A model put together by MacDonald and Bolton (2008a) suggested that for chick survival rates of 25%, rates of nest loss (to all causes combined) of up to 50% are sustainable, but in situations of lower chick survival, higher rates of nest survival must be achieved to maintain stable Lapwing populations.

Bolton *et al.* (2007a) recently described the effects of experimental predator control on the breeding success and population dynamics of Lapwing. The authors showed that, after controlling for site-specific differences in predator densities, predator control resulted in significantly higher rates of nest survival and breeding success (Bolton *et al.* 2007a). However, predator control did not influence the subsequent trends of Lapwing populations, but such an effect is not necessarily anticipated, given that large-scale immigration and emigration occurs between different populations that could potentially obscure its identification.

Lapwing generally adopt a semi-colonial nesting strategy, and several studies have shown that increasing nest densities are associated with diminished nest predation rates (Berg *et al.* 1992; Hart *et al.* 2002, Šálek and Šmilauer 2002, Seymour *et al.* 2003; Eglington *et al.* 2008b MacDonald and Bolton 2008b). This finding may reflect either that Lapwing can actively repel nest predators when nesting in high densities, or that they can determine areas characterised by low predation pressure, and nest in such habitats in high densities (MacDonald and Bolton 2008b).

On South Uist, the Lapwing population declined by 81% between 1983, before hedgehogs colonised the island, and 1995, several years after hedgehogs were introduced. Predation by hedgehogs is cited as being the cause of these declines (Jackson and Green 2000, Jackson *et al.* 2004). Between 1983 and 2000 Lapwing showed a moderate increase in the northern zone (24%) where hedgehogs were absent or at very low densities but decreased by 31% in the southern zone where hedgehogs were abundant (Jackson *et al.* 2004). Predation was the most important cause of all nest losses (44%) in a study on grasslands in the Netherlands (Beintema and Muskens 1987).

Within-field characteristics can influence the likelihood of nest predation. Sheldon (2002) and MacDonald and Bolton (2008a) both noted that nests further from the field boundary suffered a lower predation rate than those in closer proximity. Both authors accounted for this fact by invoking the predilection of predators to forage along linear features, and thus it was suggested that the probability of encountering a nest is less marked with greater distance from these features (Sheldon 2002;

MacDonald and Bolton 2008a). However, such effects are site specific and other studies have found no effect of distance to boundary (e.g. Eglington *et al.* 2008b, and see MacDonald and Bolton 2008a).

Arable fields have typically been found to have lower daily predation rates (DPR) than grass fields (Galbraith 1988, Baines 1990, O'Brien 2001, Seymour *et al.* 2003, Sharpe 2006, Sharpe *et al.* 2008, Sheldon *et al.* 2007). DPR on Lapwing nests in upland Britain was lower at sites where more arable land and improved grassland was present (O'Brien 2001). Nest DPR was also found to be higher in improved pasture than in unimproved pastures and meadows (Baines 1990, Fletcher *et al.* 2005). Evans (2004) states that the long-term Lapwing population declines may be explained partly by increased nest predation rates resulting from agricultural intensification increasing vegetation density. Lapwing select sparsely vegetated nest-sites as they can detect predators sooner and reduce predation rates by adopting mitigating action. Lapwing nest DPR on organic and conventionally managed arable fields in the Netherlands did not differ (Kragten and de Snoo 2007). Grazed coastal marshes, or marshes grazed at higher intensity, reported higher DPR (Hart *et al.* 2002).

In some cases, although predation may be high there is not necessarily an impact on nesting success as re-nesting can compensate for this, and Bientema and Muskens (1987) didn't consider predation to be a major threat to lapwing in Dutch meadows. Amar *et al.* (2008) found no evidence to suggest that raptor predation limited the Lapwing populations on an upland sporting estate in Scotland and inferred that concerns over the impact of natural densities of hen harriers on vulnerable upland waders are unjustified. However, Baines (2008) found that numbers of Lapwing on a moor in Scotland decreased when grouse moor management was discontinued and suggested that increased numbers of Crows and Red Foxes probably contributed to the observed declines.

6.6.2   Hunting

The Lapwing is still hunted in many European countries and although the hunting pressure on birds has declined all over Europe (Barbosa 2001) the annual harvest in the EU is estimated to be in the order of one million birds, amounting to 13-24% of the autumn population (European Commission 2006). Just over half of all recoveries reported in the Migration Atlas were categorized as having been deliberately taken by man (Appleton 2002). Hunting may be the main limiting factor in regions where it remains till early March (Tinarelli and Baccetti 1989). A study in Italy attributed the increase in Lapwing population during the 1970s to lower hunting pressure (Tinarelli and Baccetti 1989). Zidkova *et al.* (2007) found a significant positive relationship between the regional population trend and hunting pressure in a study in the Czech Republic. The impact of hunting on the Lapwing population in England is likely to be less however, as the proportion of shot British lapwings from all recoveries is ca 50% less than for the Czech population (Zidkova *et al.* 2007).

Although hunting is not thought to be the prime reason for the population declines, a harvesting of this magnitude is almost certainly incompatible with a restoration of the species to Favourable Conservation Status in Europe. Also, considering the very low annual productivity (in the order of 0.5 fledged young per pair), harvesting at any level cannot be regarded sustainable (European Commission 2006).

6.6.3   Competition

Olsen and Schmidt (2004) found that in a study of waders on wet grassland in Denmark, Lapwing experienced direct density-dependence, suggesting that intra-specific competition was important. They state that this is an expected result as this species exhibits strong breeding site fidelity. However, there is little other information on this issue.

6.6.4   Disturbance

In arable landscapes, farming operations such as ploughing and rolling constitute an important source of nest destruction, although the precise magnitude of nest losses varies between crop types according

to the nature and timing of management practices. In particular, losses attributable to farming operations are especially pronounced in spring cereals and stubbles where agricultural activity often coincides with the peak nesting period (Sheldon 2002). For example, Linsley (1999) demonstrated that the nest failure rate observed in unsown fields (56%) was substantially greater than that of sown fields (5%). In pastoral farmland, grazing density often detrimentally impacts Lapwing breeding success. Although grazing is essential to produce swards of the requisite height and structure, the presence of livestock depresses productivity through nest trampling or disturbing breeding Lapwing. Trampling was the second biggest cause of nest loss in Dutch pastures, accounting for 23% of all losses (Beintema and Muskens 1987). Beintema and Muskens (1987) modelled that the large increase in cattle numbers following fertilisation would have much more of an impact on nesting success than predation and found that young cattle are the worst tramplers (Bientema and Muskens 1987). A stocking rate of approximately 2·5 cows ha–1 for the whole of the incubation period leads to approximately 35% of lapwing nests being trampled (Beintema and Muskens 1987).

In the Netherlands, Lapwing were found in similar densities on mown and grazed fields and a high proportion (89%) of the early clutches hatched before mowing (Guldermond *et al.* 1993). The effect of nest destruction can be limited by the use of nest protection during mowing, which increases the hatching success despite of some predation and abandonment (Guldemond *et al.* 1993, Isaksson *et al.* 2007). Lapwings breeding on fields with the per-clutch payment scheme reportedly had higher hatching rates in the Netherlands (Musters *et al.* 2001) but Verhulst *et al.* (2007) found that this does not necessarily result in higher breeding densities as densities of did not increase during the 4 years of the study, although the hatching rate was found to be 25% higher in areas with nest protection.

Studies in the Netherlands have shown evidence of strongly reduced densities of several wader species, including Lapwing, in broad zones adjacent to busy roads (Reijnen *et al.* 1996). This density reduction is related to a reduced habitat quality due to increased noise from traffic. At 50,000 cars a day Lapwing populations were reduced by 14-44% up to 1500 m from the road. Reijnen *et al* 1996 that in areas with a dense network of extremely crowded motorways, traffic should be considered a serious threat to breeding bird populations in grasslands. However, there is little information available on the effects of traffic on population dynamics (i.e. nest success, survival).

6.6.5   Food availability

There is evidence to suggest that degradation of breeding habitats may have resulted in a decrease in the quality of foraging areas for Lapwing. Management intensity influences the size, as well as the abundance and diversity, of invertebrates. Intensive grassland management with high inputs of fertilizer and intensive grazing or mowing may be particularly detrimental to larger insect species (Beintema *et al.* 1990) and this will have an impact on foraging efficiency. Fertilization of grassland has been shown to decrease the average body mass of insects present, thus, this, combined with earlier retreat of earthworms due to improved drainage may render improved meadows unsuitable feeding areas for chicks (Beintema *et al.* 1990).

Grasslands with a long history of winter flooding can also contain much lower biomasses of soil macroinvertebrates. Ausden *et al.* (2001) found that introducing winter flooding to previously unflooded grassland greatly reduced soil macroinvertebrate biomass and that soil macroinvertebrates were slow to recolonize winter-flooded grassland when it was re-immersed in spring. However, pools of winter flood water that remained in spring and early summer provided a source of aquatic invertebrate prey for breeding wading birds and Ausden *et al.* (2001) suggest that optimal conditions for breeding lapwings will probably be provided by creating a mosaic of unflooded grassland, winter-flooded grassland and shallow pools.

6.6.6   Summary

Lapwing populations have declined partly because of habitat loss and degradation (including loss of unimproved grassland) and partly because breeding success has deteriorated as a result of the demise

of spring-sown crops and crop rotations, perhaps coupled with poorer food supplies (Fuller *et al.* 1995). Increasing disturbance, predation, nest destruction and creation of less suitable habitat due to higher stocking rates have been suggested as contributing factors, as has the 'improvement' of pasture land and consequent loss of rough grazing (Shrubb 2007, Hudson *et al.* 1994). Chamberlain and Crick (2003) suggest that recent population change may have been influenced by changes in clutch failure rates, perhaps mediated by an increase in grazing intensity in marginal uplands and by increased predation rates, possibly associated with habitat change. Sharpe *et al.* (2008) suggest that chick mortality is the main determinant of poor Lapwing productivity and population decline.

6.6.7   Possible Actions to aid recovery

Conservation of Lapwing in the wider countryside can only be achieved through agri-environment schemes (Wilson *et al.* 2004). To have an impact on the population of this species through a site-based scheme would be expensive, although Lapwing have been found to have either increased, or at least be less likely to have decreased in nature reserves as opposed to unprotected sites (Ausden and Hirons 2002). However, it is more likely that agri-environmental measures working through a targeted system, by improving conditions across farmland, would be a more realistic approach to aiding population recovery (O'Brien and Bainbridge 2002).

On grassland, ensuring strong or moderate grazing pressure the previous autumn to maintain short areas of sward and delaying grazing turnout in spring should help produce the required sward conditions whilst minimizing the effects of trampling (Devereux *et al.* 2004, Durant *et al.* 2008). As chicks may move to different fields after hatching it is not sufficient just to consider grazing regimes on specific fields, without considering the importance of adjacent or nearby fields for chick rearing (Durant *et al.* 2008).

Raising water levels, introducing surface flooding, and managing water levels to ensure that wet areas are maintained throughout the season is also key (Eglington *et al.* 2008a). Maintaining shallow wet features until later in the breeding season may be particularly important to ensure that there are plentiful food supplies for chicks (Ausden *et al* 2003, Kalhert *et al.* 2007).

On arable land, recent research (Sheldon 2002.) has indicated that an option within agri-environment schemes for an over-wintered cereal or linseed stubble followed by a spring/summer fallow may be beneficial for breeding Lapwings. Breeding success in this habitat was higher than in other habitat types, (although still below that thought necessary to maintain the population at present levels) and Lapwings demonstrated positive selection for nesting in these fields.

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7.   REDSHANK *TRINGA TOTANUS*

7.1   British Population Status and Trend

BOCC status: Amber listed
Indicator trend: The long-term trend for the UK is of a moderate decline.
Other trend data: BBS data from 1994 – 2007 show a 12% decline in numbers for the whole of the UK and a 29% decline for England (Risely *et al.* 2008). There was a shallow increase in recent years, but this assessment rests entirely upon an upturn recorded in 2004, which was followed by further declines. WBS results show a decline along waterways of 45% between 1975 and 2005, which apparently accelerated during the 1990s (Baillie *et al.* 2007). Winter abundance counted by WeBS has shown a broadly stable trend in Britain, with a very slight increase over the past decade (Banks *et al.* 2006).
Summary: Evidence suggests that this species is undergoing a prolonged decline**.**
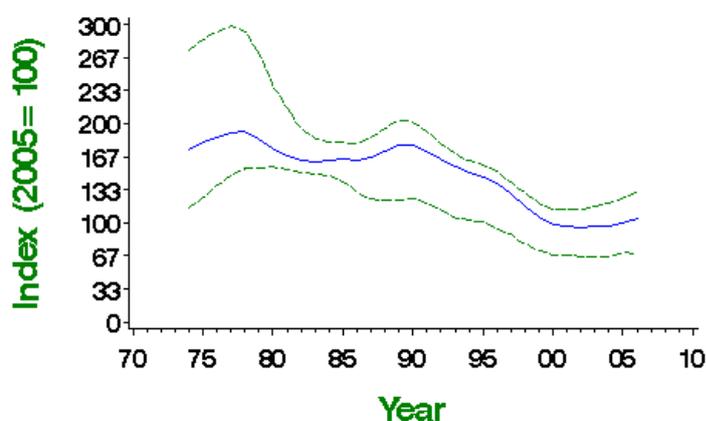
Figure 6 Waterways Bird Survey trend for Redshank, 1974 to 2006. From Baillie *et al.* 2007

7.2   Distribution And Demographic Changes

7.2.1   Worldwide and European

The Redshank is a widespread breeder across much of Europe, which constitutes more than 50% of its global breeding range. Its European breeding population is large (>280,000 pairs), but has undergone a moderate decline between 1970 and 1990. Although several populations, notably sizeable ones in Russia and Norway, were stable or increased during 1990–2000, the species continued to decline across much of its European range, and underwent a moderate decline (>10%) overall. Consequently, it is evaluated as Declining (Birdlife International 2004). Population trends in EC countries were reviewed by Hotker (1991) who reported that Redshank populations on grasslands were in decline.

7.2.2   Britain

It was estimated that there were 38,800 (31,400 - 44,400) pairs in the UK between 1985 and 1999 (O'Brien 2005, Birdlife International 2004, Baker *et al.* 2006). There were an estimated 16,433 pairs (11,291-21,943) on saltmarshes in 1996, representing c. 45% of the population of Redshank breeding in Great Britain. This equates to a 23% decline in the population since 1985 (Brindley *et al.* 1998). The highest densities of Redshank in Britain have been on the East Anglian and north-western coasts of England. During the 1980s and 1990s there were contractions in breeding range in north-eastern England, central Scotland, southern England and areas of Wales, both at coastal and inland sites (Griffin *et al.*, 1991, O'Brien and Smith, 1992; Gibbons *et al.*, 1993, Cook *et al.*, 1994, 1995), with declines in breeding densities ranging from 20 to 76%. The decline in breeding densities on saltmarsh has been mirrored by a contraction in breeding range within Britain across an array of habitats

(Gibbons, Reid and Chapman 1993). By 1988-91, breeding Redshank had disappeared from 40% of the 10 x 10 km squares they occurred in at the time of the 1968-72 breeding atlas (O'Brien and Self 1994). In upland areas of Britain, Redshank were found on less than half the number of plots on which they were originally recorded and decreased by 74% on upland CBC sites in Britain between 1968-80 and 2000 (Henderson *et al.* 2004). On wet meadows in England and Wales, surveys revealed a decrease of 29% in breeding birds between 1982 and 2002 with their populations being concentrated mainly in East Anglia, north Kent and parts of northern England (Wilson *et al.* 2005). Occupancy of sites also decreased, from 39% to 21% (Wilson *et al.* 2005).

Wintering populations (augmented by many Icelandic and some other northern European breeders) have been stable since the mid 1980s (Banks *et al.* 2006).

7.2.3   Change in demographic parameters

Breeding densities tend to be highest on saltmarsh and coastal grassland but breeding success here is very low, due to a combination of low hatching success resulting from predation and tidal flooding, and poor chick survival. Breeding densities tend to be lowest on inland grassland, but breeding success is much higher (Smart 2005). Adult survival rates can have a large effect on the productivity required to maintain a stable population (Otvall and Harding 2005). There has been a steep fall in the failure rate of eggs at the nest stage since the 1960s (Baillie *et al.* 2007).

7.3   Resource Requirements

7.3.1   Breeding season diet

Redshank take a wide-range of invertebrate prey from the soil, amongst the sward and from shallow pools and wet mud. Redshank feeding on coastal grazing marshes may take a high proportion of prey from the adjacent estuary (Ausden *et al.* 2003). The proportion of aquatic invertebrates in the diet has been shown to increase as the breeding season progresses, while that of soil invertebrates decreases (Ausden *et al.* 2003). Species found in the diet include Bibionid flies, the estuarine *Hediste* and *Caricinus maenus*, and a high proportion of Coleoptra larvae, particularly in May, along with *Chironomus salinarius* pupae and larvae, *Ephydridae* larvae and the beetle *Paracymus aenus* (Ausden *et al.* 2003, Sanchez *et al.* 2005).

Redshank chicks feed on surface dwelling organisms and consume fauna on vegetation (Beintema *et al.* 1991). Prey taxa include; Lumbridcidae, Gastropods, Chironomids, Diptera, Hymentoptera, Coleoptera, Araneida and Diptera and Coleoptera larvae (Beintema *et al.* 1991).

7.3.2   Winter diet

Analysis of pellets from areas in southern England suggested that most of the small items eaten were Corophium over 4 mm long. The small isopod *Cyathura carinata* was also sometimes found (Goss-Custard 1969, Goss-Custard 1977). Other studies have shown that the principal prey are insects, dipterans and coleopterans, including chironomids, ephydrids, *Potamonecte* spp. and *Enochrus*. Crustaceans are also eaten and include *Carcinus maenas* and *Crangon crangon*. Molluscs are also taken, but less frequently, and are mainly *Hydrobia ulvae*. The remains of amphipods, ostracods and small fish have also been found occasionally, as have plant remains (Perez-Hurtado *et al.* 1997, Peterson and Exo 1999). Polychaetes and molluscs have been found to dominate the diet in autumn and isopods in midwinter. Redshank diet shows much seasonal and spatial variation in southern Europe (Sanchez *et al.* 2005).

7.4 Habitat Associations

7.4.1 Breeding season

Redshank breed on both inland and coastal grazing marshes, building their nests in grass tussocks or patches of relatively tall vegetation. Around 60% of the British population nest on saltmarshes (Hughes 2004) and proximity to estuaries may be important in influencing the suitability of coastal grazing marshes (Ausden *et al.* 2003). At a landscape scale, Atkinson *et al.* (2002) found that breeding Redshank were most abundant in 10 x 10 km squares with more than 50% cover of grassland while Redshank showed a mixed preference for intensive and extensive grasslands in a study in Hungary (Baldi *et al.* 2005).

The two main predictors of nest site choice for Redshank are sward structure and surface water, although other factors could also been involved, such as field area and disturbance (Durant *et al.* 2008a). Redshank require tall swards of around 10 - 40 cm and are sensitive to mid-spring grazing, preferring light spring grazing that produces tall heterogeneous swards (Norris *et al.* 1997, Tichit *et al.* 2005, Durant *et al.* 2008a and b). On both inland and coastal marshes, Redshank select areas with tussocky vegetation. Areas with extensive silage production tend to hold significantly lower densities of Redshank as this produces unfavourable tall, dense swards (Wilson *et al.* 2004). In a study in Sweden, Redshank densities in individual fields were positively related to grazing intensity, although at the same time, a general increase in grazing intensity in this area paralleled declining wader numbers (Otvall and Smith 2006). In the Netherlands, Redshank were found more frequently on mown rather than grazed fields (Guldermond *et al.* 1993). In Iceland, Redshank significantly avoided heath and agriculture, and their presence was more likely in patches with 10–20 cm sward height, small and medium hummocks, large numbers of sedge-pools, low water-tables and low cover of hummocks (Gunnarsson *et al.* 2006).

Breeding location and breeding densities within fields have been found to be positively related to the presence of wet features and surface flooding (Vickery *et al.* 1997, Milsom *et al.* 2000, 2002, Smart *et al.* 2006) and adults have been found to select fields with areas of 5–50% surface water (Durant *et al.* 2008a and b). Redshank have a preference for feeding in wet features and Redshank probing in wet rills were found to capture prey three times as fast as those birds probing on open turf (Milsom *et al.* 2002). This has been suggested to be because they feed on aquatic invertebrates within these features (Ausden *et al.* 2003). In the Pennines, Haworth and Thompson (1990) found that redshank were closely associated with wetter bogs dominated by extensive stands of *Eriophorum*, maybe because of the high densities of tipulids and other invertebrates present. On inland sites in East Anglia, decreasing soil penetrability increased the probability of nest occurrence and hatching probability, whilst proximity to wet features and vegetation type at the nest were important on coastal sites (Smart *et al.* 2006). On saltmarshes around the Wash, Redshank were also associated with creeks and the tide edge (Goss-Gustard and Yates 1992) and fields under agri-environment schemes in Denmark which aimed to retain surface water had a positive effect on Redshank settlement densities (Kalhert *et al.* 2007).

Redshank are highly faithful to their natal area and a high proportion of birds that survive post-fledging mortality return to breed in their natal area. However, Redshank can learn from reproductive failure and react by shifting nest-site location the following season (Thompson and Hale, 1989).

7.4.2 Winter

Redshank winter on mud flats, estuaries, and wetlands. The majority of Redshank are site-faithful both within and between winters (Burton 2000). Redshank densities were found to be lower in estuaries with more consolidated sediments and were related to densities of *Neris diversicolor* and *Corophium volutator* (Goss-Custard *et al.* 1991). A study on Orkney round that Redshank preferred sheltered muddy shores in Orkney (Summers *et al.* 2002).

7.5 Impacts of Environmental Change

7.5.1 Climate change

A study in the Netherlands found that warmer springs lead to faster grass growth and hence earlier mowing dates. However, it was also found that warmer springs lead to earlier hatching dates and that because of this, the negative effects of early mowing on the breeding success were less than expected (Kruk *et al.* 1996).

Beintema (1994) looked at rainfall patterns and annual variations in condition indices of chicks and found that chick condition was correlated with total rainfall in May. 'Good' and 'bad' years tended to coincide with wet and dry springs.

Year-to-year variation in adult survival on the Moray Forth in Scotland was weakly negatively related to the number of snow days in winter. Year-to-year variation in first-year survival was non-linearly related to winter rainfall, with low survival during dry, cold winters, higher survival during winters with average rainfall and lower survival during wet winters (Insley *et al.* 1997).

The winter distribution of Redshank in the UK appears to have shifted in a north-westerly direction, and this was proposed to be as a response to changing winter temperatures (Maclean *et al.* 2008). A study of over-wintering Redshank in Poole harbour, predicted that simulated sea-level rise would seriously affect Redshank; a 10 cm reduction in the height of intertidal patches reduced survival rates while a reduction of 40 cm meant that no birds at all survived the winter (Durell *et al.* 2006). Stillman *et al.* (2005) predicted that a 2 to 8 % reduction in intertidal area (the magnitude expected through sea level rise and industrial developments) decreased survival rates of Redshank on the Humber estuary.

There is no clear evidence that saltmarsh losses due to sea-level rise have so far caused reductions in bird populations at a national level (Norris *et al.* 1998, Norris *et al.* 2004, Fuller and Ausden 2008). However, although there is limited evidence that climate change has played a role in the decline of the Redshank population to date, changes in future climate may be important as this may result in loss of coastal grazing marsh to managed realignment of sea defences and increased frequency of spring floods or droughts, which are likely to have an impact on wader populations on lowland wet grassland in the future (Wilson *et al.* 2004). There is also concern that as estuaries become wider due to land being claimed by the sea, sediments are expected to become sandier, and consequently they would be capable of supporting lower densities of species such as Redshank (Austin and Rehfisch. 2003).

7.5.2 Habitat loss and degradation

The declines in breeding Redshank are thought to be primarily due to the loss and degradation of important coastal breeding sites, combined with the intensification of agriculture affecting inland sites (Hotker 1991, Tucker and Heath, 1994; Brindley *et al.*, 1998; Norris *et al.*, 1998). Between 1938 and 1981, 82% of the coastal grassland in Essex was lost, mainly from conversion to arable farmland, and similar declines in the north Kent marshes (-48%), Romney marshes (-55%) and in the Norfolk Broads (-37%) have also been recorded (Williams and Hall 1987; Williams and Bowers 1990). Wilson *et al.* (2005) recorded a 27% decrease in the area of hay meadows and a 23% decrease in cattle grazing between 1982 and 2002. Considerable range contraction occurred from many areas of the UK by 1988–91, probably as a result of the drainage of farmland (Gibbons *et al.* 1993). Over the last 50 years the area of land (both lowland and upland) under pasture has been reduced from 50% to less than 35% (Wakeham-Dawson and Smith 2000) while current loss of saltmarsh is estimated to be more than 100 ha per year in the UK (Atkinson *et al.* 2004), with 40 ha/year being lost in southeast England, attributable to coastal squeeze (Hughes 2004).

Not only is a direct loss of habitat a problem, but a decline in habitat quality, such as that caused by the change in grazing practices is also important (Brindley *et al.*, 1998; Norris *et al.*, 1998). Redshank are visual feeders (Goss-Custard, 1969) and require swards of shorter vegetation with patches of

surface water (Vickery *et al.*, 1997; Milsom *et al.*, 1998, 2000). Consequently Redshank need a habitat of varied vegetation structure, characteristic of moderate cattle grazing (Norris *et al.*, 1997). Overgrazing and grazing by sheep tends to produce a short uniform sward whilst ungrazed saltmarshes promote a tall sward, both of which are unsuitable (Norris *et al.*, 1998). The increase in grazing intensity between 1985 and 1996 was found to be sufficient to account for the observed changes in Redshank density on saltmarshes in Britain (Norris *et al.*, 1998). Figures suggest that inappropriate grazing management affects an area of saltmarsh three times larger than the area estimated to be lost to erosion over the next 20 years and the grazing management of saltmarshes is changing from ungrazed/lightly grazed to moderately/heavily grazed at a more rapid rate than the loss of saltmarsh to erosion (Norris *et al.* 1998).

Intensification of grassland farming operations has occurred on the remaining grasslands since the 1970s (Vickery *et al.* 2001). There have been large changes in grazing practices on grassland sites in the lowlands and uplands. The most striking of these changes has been the recent large increase in numbers of sheep, which more than doubled between 1950 and 1990 (Fuller and Gough 1999). The resulting heavy grazing almost certainly reduced the habitat quality for ground-nesting birds such as Redshank (Fuller and Ausden 2008). Agricultural improvement, characterized by high fertilizer inputs, drainage and silage production, increased the capacity of grassland to carry livestock and this has contributed to the reduction in numbers of breeding Redshanks (see Figure 3 in Snipe review; Baines 1988, Fuller and Ausden 2008). The rapid changes in sward height associated with the intensification of grassland management may affect the foraging efficiency and the availability of nest sites for breeding Redshank (Vickery *et al.* 2001).

In the uplands, afforestation has been another factor and Fuller and Ausden (2008) claim that there is no doubt that many species of open habitats, especially breeding waders such as Redshank, have been displaced over substantial areas by afforestation in the uplands.

7.5.3   Loss of nesting sites

Habitat loss and degradation have resulted in loss of nesting habitat (see above).

7.5.4   Pollution

There is little information regarding the impact of pollution on breeding Redshank. The eggs of the Redshank breeding on the North Sea coast of Germany had a mean PCB concentration of 0.93 ppm, which was the lowest value of all investigated the waterbird species (Denker *et al.* 1994). As such, it is not envisaged that this is a big issue.

7.6   Impacts of Biological Change

7.6.1   Predation

Changes in land-use may have increased the suscepability of nests to predation and/or increased predator numbers in the landscape. Recent empirical evidence suggests that levels of predation on wader nests are unsustainably high in many cases, even in some situations where breeding habitat is otherwise favourable (MacDonald and Bolton 2008). In Dutch pastures, predation was an important cause of nest loss (26%) in Beintema and Muskens' (1987) study whilst Guldemond *et al.* (1993) found that 13% of nests under nest protectors predated and 18% of nests that had been mown around suffered loss to predation. In a study in East Anglia, predation was the main cause of nest failure (Smart 2005). On South Uist, the Redshank population declined by 36% between 1983, before hedgehogs colonised the island, and 1995, several years after hedgehogs were introduced; predation by hedgehogs is cited as the main cause of these declines (Jackson and Green 2000, Jackson *et al.* 2004). The decrease in nest survival rates on coastal meadows in south-east Sweden has been attributed to changes in predator densities and/or behaviours (Ottvall 2005). In the German Waddensea saltmarshes, hatching success of Redshanks was low due to high clutch predation,

probably by mustelids, rodents, and Carrion Crows. On average, only about 10% of clutches produced chicks and hatching and breeding success was too low to maintain the population size on a long-term basis (Thyen and Exo 2003, 2005). Nest protectors have been shown to increase the hatching success of Redshank nests, but this may be at the expense of adult predation, which was found to be higher on nests with enclosures (Isaksson *et al.* 2007).

Predation on adults is also an issue. In three winters, raptor predation, mostly by Sparrowhawks, was shown to be the most significant cause of mortality in Redshank. Over 50% of the total population and over 90% of the juvenile population were taken in two winters (Cresswell and Whitfield 1994). Winter mortality in both juvenile and adult redshanks was density-dependent, due to predation by Sparrowhawks (Whitfield 2003a) and predation risk declined with distance from cover (Whitfield 2003b). On a small Scottish estuary at low tide, Redshank on salt-marshes were 5 times more likely to be killed than on the mussel beds, but had a much higher foraging rate and overall energy intake rate. Therefore, adults on the mussel beds minimized risk of predation, rather than maximizing intake rate at low tide (Cresswell 1994).

7.6.2 Hunting

Of the recoveries reported in the Migration Atlas, 30% were deliberately taken by man (Clark 2002). There is little other information about the impact of hunting on this species but levels of hunting are thought to have decreased and thus it is unlikely that this is currently a cause of population decline.

7.6.3 Competition

Olsen and Schmidt (2004) found that in a study of waders on wet grassland in Denmark, Redshank experienced direct density-dependence, suggesting that intra-specific competition was important. They state that this is an expected result as this species exhibits strong breeding site fidelity. Food intake rates of over-wintering Redshank declined with increasing flock size, further indicating an effect of competition and that Redshank flocked to avoid predation rather than to increase their food intake rates (Whitfield 2003b). However, competition is not thought to be a major factor in Redshank population declines.

7.6.4 Disturbance

Redshank nests can be particularly vulnerable to trampling, and Beintema and Muskens (1987) found that a stocking rate of approximately 2·5 cows ha^{-1} for the whole of the incubation period lead to approximately 70% of Redshank nests being trampled, while 10% were destroyed by agricultural operations. Young cattle were the worst tramplers. However, trampling accounted for just 3% of nest failures in a study on saltmarshes in the German Waddensea (Thyen and Exo 2005) and 7 – 12% of nest losses in East Anglia (Smart 2005). In the Netherlands, Verhulst *et al.* (2007) found that hatching rate was 25% higher in areas with nest protection although Guldemond *et al.* (1993) found that 10% of nests were lost to trampling, despite the use of nest protectors. In the same study, Guldemond *et al.* (1993) found that 62% of clutches hatched before mowing. Given the dramatic increase in stocking rates in many areas, increases in trampling levels seem likely to have occurred, although there was no evidence documenting the effect of this on population dynamics.

Potential disturbance by humans has been associated with the absence of breeding Redshank on uplands (Haworth and Thompson 1990). In the Netherlands however, a study found no effect of traffic on densities of breeding meadow birds near roads (Reijnen *et al.* 1996).

Over winter, a study on the Severn Estuary, found that one foraging site was almost entirely avoided during the day, probably due to disturbance from an adjacent heliport, but was used by the majority of individuals at night when the heliport was closed (Burton and Armitage 2005). Numbers of wintering Redshank were also lower on sections of an estuary that had a footpath in close proximity and this study suggested that sustained disturbance reduced local habitat quality of estuaries. (Burton *et al.*

2002a). The impact of disturbance from construction work around Cardiff Bay was also found to have significantly reduced both the densities and feeding activity of Redshank on adjacent intertidal mudflats. Despite this, evidence suggests that the displacement of Redshank from these mudflats did not contribute to a decline in this species (Burton *et al.* 2002b).

Overall, there is little quantitative evidence detailing the impact of the various types of disturbance on Redshank survival and mortality.

7.6.5 Food availability

There is evidence to suggest that degradation of breeding habitats may have resulted in a decrease in the quality of foraging areas. Management intensity influences the size, as well as the abundance and diversity, of invertebrates. Intensive grassland management with high inputs of fertilizer and intensive grazing or mowing may be particularly detrimental to larger insect species (Beintema *et al.* 1990) and this will have an impact on foraging efficiency. Fertilization of grassland has been shown to decrease the average body mass of insects present, thus, this, combined with earlier retreat of earthworms due to improved drainage may render improved meadows unsuitable feeding areas for chicks (Beintema *et al.* 1990).

On wet grasslands, aquatic invertebrates in shallow pools can be an important component of the diet of breeding Redshank in May and June, but most management of grazing marsh set out in agri-environment schemes only specified retaining pools until the end of April (Ausden *et al.* 2003) meaning that later in the season, many wet areas dry up. Grasslands with a long history of winter flooding contain much lower biomasses of soil macroinvertebrates, and Ausden *et al.* (2001) found that introducing winter flooding to previously unflooded grassland greatly reduced soil macroinvertebrate biomass. However, pools of winter flood water that remain in spring and early summer provide a source of aquatic invertebrate prey for breeding wading birds and Ausden *et al.* suggest that optimal conditions for breeding Redshank will probably be provided by creating a mosaic of unflooded grassland, winter-flooded grassland and shallow pools.

Unlike most waders, adult Redshank take very small prey in relation to their body size and so must feed for long periods during each tidal cycle to achieve their daily energy intake needs. Thus they have little scope to extend their feeding time during severe weather. Both factors lead to more rapid depletion of fat reserves than in other species which have higher energy intake rates or lower total daily requirements (Mitchell *et al.* 2000).

Lack of food can also result in increased predation. Animals cannot always optimize anti-predation behaviour because anti-predation behaviours are usually incompatible with foraging behaviours to maximize intake rates. Increased starvation risk in midwinter means Redshank are forced to feed on highly profitable prey, the behaviour of which means that Redshanks are forced to feed vulnerably. The overall number of Redshank and the proportion of Redshank killed increased in cold months (Cresswell and Whitfield 2008). At the Tyninghame estuary, Scotland, Redshank chose to feed on either a mudflat or saltmarsh. Energy intake rates were 23% higher and thermoregulatory costs were 40% lower on the saltmarsh, but predation risk from Sparrowhawks (Accipiter nisus) was 21 times higher. When starvation risk increased on the mudflat, more Redshanks selected the saltmarsh (Yasue *et al.* 2003).

7.6.6 Summary

Agricultural intensification affecting breeding grounds is implicated in these population declines, mainly from the drainage of favoured wetland breeding habitats and inappropriate grazing regimes. The management of grassland has become far more intensive and has included extensive drainage, increased use of pesticides and fertilizers, re-seeding, earlier and more frequent mowing and increased grazing pressure. Earlier spring grass growth, earlier cutting dates and higher stocking

levels has increased egg and chick mortality and reduced relaying opportunities. On coastal sites, inappropriate grazing has also reduced habitat quality.

7.6.7 Possible Actions to aid recovery

Site designation has been found to be an important predictor of presence and density of Redshank (O'Brien and Smith 1992, Wilson *et al.* 2005); breeding Redshank increased by 58% on RSPB lowland wet grassland reserves between 1987-2000 (Ausden and Hirons 2002). Further to this, population trends were most favourable in the more expensive ESA options aimed at enhancing habitat, although the less expensive, habitat maintenance options also appear to have benefited Redshank (Wilson *et al.* 2007). Management on such areas includes raising water levels and introduction of surface flooding, reducing grazing levels during the breeding season and delaying mowing to reduce nest loss. Maintaining shallow wet features until later in the breeding season may be particularly important to ensure that there are plentiful food supplies (Ausden *et al* 2003). Targeting beneficial conservation activity to the sites with high densities of this species, such as through an agri-environment scheme, could have a substantial effect on the population at a relatively small cost (O'Brien and Bainbridge 2002, Wilson *et al.* 2004).

Norris *et al.* (1998) recommend that on saltmarshes, the maintenance of cattle grazing, at densities of about one animal per hectare or less should be encouraged, along with the re-introduction of cattle grazing to areas of marsh that were formerly grazed by cattle. The introduction of grazing to sites with no history of grazing should be discouraged and there should be an aim to reduce the stocking density of sheep, preferably by removing sheep grazing from certain marshes to permit the vegetation to recover. Cattle should, if possible, be put on to the marsh towards the end of the nesting season (i,e. late May/early June) to minimize this trampling risk (Norris *et al.* 1997).

In terms of minimising the impact of predation on Redshank, manipulating habitat structure to remove cover for predators, and ensuring profitable food is available for Redshank may be options (Quinn and Cresswell 2004).

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8. REED BUNTING *EMBERIZA SCHOENICLUS*

8.1 British Population Status and Trend

BOCC status: Amber
Indicator trend: Moderate decline
Other trend data: Both CBC/BBS and WBS indices declined rapidly during the 1970s, but Reed Bunting abundance subsequently remained remarkably stable (Baillie *et al*. 2007). CBC (1966 – 2000) and BBS (1994 – 2002) data showed a 48% decline (Gregory *et al.* 2002). There was a 62% decline on CBC plots between 1974 and 1999 (Crick *et al.* 2004) and WBS data from 1975 to 2005 indicate a 59% decline along waterways (Baillie *et al.* 2007). However, in recent years (1994 – 2007), BBS results indicate significant population increase; 19% in England, 31% for the entire UK (Risely *et al.* 2008). CES data for the 2007 season saw adult numbers at their lowest level ever (Coiffait *et al.* 2008) and CES data show a 47% decline for adults and a 70% decline for juveniles between 1984 and 2005 (Baillie *et al.* 2007).
Summary: Most data point towards a steep historical decline followed by a more recent period of stability/increase.
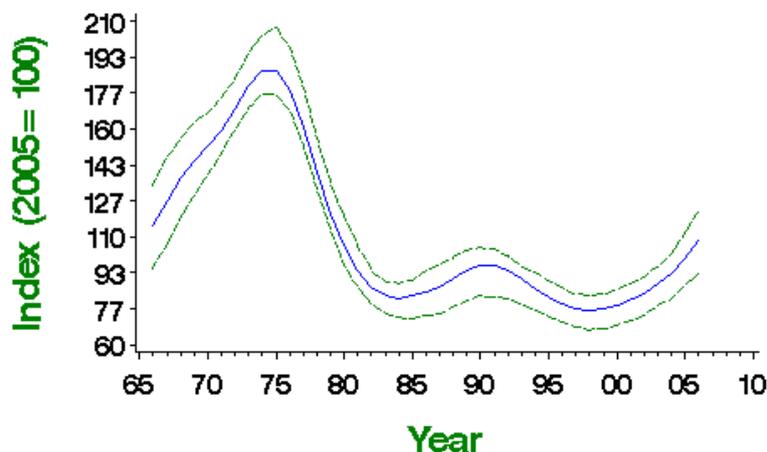
Figure 7 Combined CBC/BBS UK trend for Reed Bunting 1966 – 2006. From Baillie *et al.* 2007

8.2 Distribution and Demographic Changes

8.2.1 Worldwide and European

The European population of Reed Buntings as a whole declined severely during the 1970s, mainly because of habitat loss and agricultural intensification (Donald and Gailly 1997, PECBM 2006). By the late 1980s and early 1990s populations in many European countries were relatively stable (e.g., Norway, Poland, Sweden), others were declining (e.g., Germany, Belgium), and some increasing (e.g., Denmark, Ukraine; Donald and Gailly 1997). Reed Buntings breeding in north and central Europe are migratory, or partially migratory, mainly over-wintering in the Mediterranean area (Mezquida and Villaran 2006).

8.2.2 Britain

Reed Buntings are widespread in Britain, although are much less common in upland areas. They are present throughout year with a population size estimated to be between 192,000 and 211,000 territories in 2000 (Birdlife International 2004, Baker *et al.* 2006). They were formerly restricted to marsh or riverine areas but are now occasionally found on rough ground in agricultural areas (Ward *et al.* 1994). Reed Buntings increased in numbers between 1963 and 1975, expanding into less preferred farmland habitats as numbers increased. However, this increase was followed by a steep decline in numbers between 1975 and 1983, driven by reduced over-winter survival, possibly related to severity

of winters (Marchant *et al.* 1990) or loss of over-winter seed resources due to intensification of agriculture (Brickle and Peach 2004). During this period Reed Bunting declined by 58% on farmland and by 66% along linear waterways (Peach *et al.* 1999). Peach *et al.* (1998) found that catches of both adult and juvenile Reed Buntings on Constant Effort sites between 1983 and 1995 decreased by over 40%. Furthermore, linear trends in catch sizes decreased significantly more in dry (adults – 14%, juveniles – 26%) than in wet (adults – 3%, juveniles – 5%) habitats. Across farmland CBC plots in England and Wales, the population underwent a significant decline of 23% between 1970 and 1990 (Chamberlain and Fuller 2001) and suffered a 64% decrease in density (Chamberlain *et al.* 2001). There was a 59% decrease in breeding population of Reed Buntings over the period 1968-1991 (Fuller *et al.* 1995). Losses of Reed Bunting were significantly lower in 10 km squares from predominantly arable regions compared to pastoral or mixed farming regions (Chamberlain and Fuller 2000, Gates and Donald 2000).

In recent decades there has been a minor contraction of range, predominantly in the north and west (Gibbons *et al.* 1993). Following a slight recovery, during the 1990s the population in England experienced further significant declines (Baillie *et al.* 2007), possibly driven by reduced nest success (Peach *et al.* 1999). These declines have not just been confined to the lowlands and number of Reed Buntings breeding on upland farmland has also declined (Fuller *et al.* 2002). Population levels have remained relatively stable since the mid-1990s.

8.2.3   Changes in demographic parameters

There is strong evidence that first-year (and, to a lesser extent, adult) survival decreased during the late 1970s and the 1980s, the time during which the British population was declining. These declines in over-winter survival were sufficiently large to have caused the population decline and, given that nesting success was relatively high during this period, are likely to have done so. Low breeding performance could have prevented a population recovery for Reed Bunting after the species' decline in the 1970s (Peach *et al.* 1999, Siriwardina *et al.* 2000b) and a recent study suggests a decline in total seasonal productivity in some areas (Brickle and Peach 2004). There was a linear increase in daily nest failure rates at the egg stage between 1968 and 2005 (Baillie *et al.* 2005). Reed Bunting breeding performance fell on arable and mixed farms after 1976, but tended to rise on grazing farmland and declines were greater in northern Britain than in the southeast (Peach *et al.* 1999). Although temporal changes in breeding performance have been different in arable/mixed and grazing farmland this difference is due, primarily, to variation in patterns of change in the nestling period failure rate (Siriwardena *et al.* 2000a)

8.3   Resource Requirements

8.3.1   Breeding season diet

The diet during the breeding season consists primarily of invertebrates; adult diet may comprise of almost 80% invertebrates (Prŷs-Jones 1977, Holland *et al.* 2006), although both adults and offspring will consume a variable amount of seeds, particularly those of grasses (Brickle and Peach 2004). Precisely which prey is taken varies through the breeding season. Springtails (Collembola) and midges *Chironomidae* (Diptera) are important in adult diet early in the spring. In April and May caterpillars (Lepidoptera) (foraged for in hedges and trees) are the most important component, and by June-July spiders (Araneae) and dragonflies and damselflies (Odonata). Also taken are craneflies (Tipulidae), weevils (Curculionidae), horse flies (Tabanidae), mayflies (Ephemeroptera), sawfly larvae (Symphyta: Tenthredinidae), stoneflies (Plecoptera), grasshoppers, etc. (Orthoptera), bugs (Hemiptera), lacewings, etc. (Neuroptera), caddis flies (Trichoptera), adult and larval Hymenoptera (Tenthredinidae, ants Formicidae), beetles (Coleoptera), harvestmen (Opiliones), ticks (Acari), Crustacea, snails (Pulmonata), Bivalvia (Sphaeriidae).and Orthoptera (Cramp and Perrins 1994, Wilson *et al.* 1997a, Holland *et al.* 2006).

Nestlings are fed invertebrates (Cramp and Perrins 1994). Brickle and Peach (2004) found that caterpillars (Lepidoptera and Symphyta larvae – 41%) comprised the bulk of provisioned insect food to chicks, followed by spiders (Araneae – 29%), beetles (Coleoptera – c. 15%) and flies (Diptera – 12%) Adult Diptera and beetles decrease as proportion of diet towards end of season, while Orthoptera, adult Lepidoptera, and snails increase. Spiders were present in 80–90% of samples during July–August; in mid-season (June), mayflies, caterpillars, and larval Hymenoptera are the commonest prey. In samples from grassland habitats, the main component of diet was adult Diptera, especially crane-flies Tipulidae, and in those from areas with more trees and bushes, caterpillars, larval Hymenoptera, and adult beetles (Cramp and Perrins 1994)

8.3.2   Winter diet

Reed Bunting eat mainly seeds during the non-breeding season although are opportunistic throughout year. In winter, Reed Bunting feed mainly on the seeds of grasses and herbs, often taken from or near the ground. A diverse range of seeds are taken, including goosefoots (Chenopodiaceae), amaranths (Amaranthaceae), chickweeds and mouse-ears (Carophyllaceae), crucifers (Cruciferae), lupins *Lupinus* (Leguminosae) and the seeds of wild grasses such as meadow-grass *Poa*, millet *Setaria*, fescues *Festuca*, rye-grass *Lolium* and cockspur *Echinochloa*. Grass seeds are particularly important, representing approximately 68% of all seeds taken (Prŷs-Jones 1977). An analysis of droppings collected in winter on unused fields in south west Poland (fallow, stubble of root crop, vegetable and cereal) showed that the base of the diet were seeds of annual weed species*: Chenopodium album* (74% of all recorded seed species), *Amaranthus retroflexus* (16%), *Setaria vir*idis (10%), *Stellaria media* (0.21%), *Fumaria officinalis* (0.01%; Orlowski 2007). An experiment whereby supplementary food was provided over-winter showed that Reed Buntings fed intensively on cereal grains and selected maize over both barely and oats, but not wheat (Perkins *et al.* 2007).

Outside of the breeding season, Reed Bunting more often feed on ground in open countryside and cultivated fields, weedy areas, woodland clearings, uplands, etc., away from water, often in flocks with other seed-eaters (Cramp and Perrins 1994). In winter, they are frequently found at bird-tables in gardens, as well as in farmyards and they may make considerable use of game feeders within game cover (Brickle 1997). Unlike other buntings, cereal seeds are relatively unimportant to Reed Buntings, although they can form a larger proportion of the diet in the late winter when other seed stocks have been diminished.

Invertebrates are also taken during the non-breeding season, including spiders (Araneae), springtails (Colembola), Hemiptera, Diptera larvae and beetles, but they form only a small component (~ 5%) of the diet, (Cramp and Perrins 1994).

8.4   Habitat Associations

8.4.1   Breeding season

Reed Buntings like marshy areas, reedbeds, fringing emergent vegetation, hedgerows, ditches and rank vegetation, nesting in thick vegetation close to the ground (Ward *et al.* 1994, Cramp and Perrins 1994). More specialized habitats in Britain include salt-marshes on low coasts, open areas and moist slacks of coastal sand-dunes, gravel pits, and upland rivers. Woodlands may be occupied where there is water, or where clearings or young plantations create suitable conditions (Cramp and Perrins 1994). The apparent attachment to wetlands occurs indirectly, through dependence on their associated vegetation types rather than being linked with any special need for water (Cramp and Perrins 1994). Across Britain, Gregory and Baillie (1998) estimated that 51% of the total breeding Reed Bunting population was on farmland (mainly on tilled land), 25% on semi-natural grassland (mostly in water meadows and grazing marsh) and 12% on riparian sites (beside rivers, canals and small water bodies). Average densities were highest at riparian sites, with around 10 birds km^{-2}, 5 birds km^{-2} in semi-natural grassland and around 3 birds km^{-2} in grass/till, till and scrub.

Reed Buntings on farmland are typically associated with wet margin habitats. Mason and Macdonald (2005) found them to be associated with marginal vegetation on coastal borrow dykes in eastern England, although their population density in this region was found to be very low (Mason and MacDonald 2000). Brickle and Peach (2004) found that rank grassland and herbaceous vegetation, much of which adjoined wetland features, was the dominant habitat type (30%) within 100 m of nests, whereas farmed habitats, such as set-aside and cereals, were less important constituents. In agricultural areas of Poland, Reed Bunting breed mainly in marshland biotopes (Kosinski and Tryjanowski 2000, Surmacki 2001), with wet marginal habitats consisting of reeds (22%) being the principle habitat type in breeding territories, followed by herbaceous vegetation (13%) and meadows (9%; Surmacki 2004). Another Polish study documented breeding on abandoned arable fields with a well developed layer of perennial plants (Orlowski 2005). In Ireland, Reed Buntings were found to show a close affinity for fens (Braken *et al.* 2008). On peatlands in Eastern Finland, Reed Bunting were most abundant in shrub habitats (Kouki *et al.* 1992) while another study found that cover of mires, shoreline length and cover of marshland were all positively related to presence of Reed Buntings (Virkkala *et al.* 2005). Within reedbeds in north east Spain, the abundance of Reed Bunting was greatest in areas of drier soils and with a denser basal stratum (Martınez-Vilalta *et al.* 2002). Oilseed rape crops and small wetland features (such as ditches) now provide most nesting places in farmland (Newton 2004).

Nests are generally placed close to ground level (mean height 25 cm; Brickle and Peach 2004), particularly in dense, herbaceous vegetation, which affords a greater degree of concealment from predators (Brickle and Peach 2004, Surmacki 2004). Buntings rarely locate their nests in reeds, which instead serve as song-posts for territorial establishment and defence (Gordon 1972, Surmacki 2004). Where wetland features on agricultural farmland are scarce, oilseed rape appears to be a suitable alternative habitat for Reed Buntings and they have been commonly found breeding in rape (Burton *et al.* 1999, Gruar *et al.* 2006). Gruar *et al.* (2006) demonstrated that the density of bunting in oilseed rape fields was 4 times greater than the densities on wheat, barley and set-aside fields, and a study in Poland found a similar pattern (Sumaki 2001). It was speculated that rape might permit Reed Bunting to breed on arable farmland that would otherwise be unsuitable because of a dearth of wetland features, although rape fields that were close to wet features were more likely to be used (Gruar *et al.* 2006). Oil seed rape may provide greater nest concealment and higher invertebrate densities than cereals or set-aside (Sumaki 2001, Brickle and Peach 2004).

Reed Buntings principally forage in close proximity to the nest; Brickle and Peach (2004) showed that 87% of foraging sorties were within 100 m from the nest. Rank and emergent vegetation is a favoured habitat type in which to forage during the breeding season, presumably because these habitats are characterised by greater food abundance (Brickle and Peach 2004). Burton *et al.* (1999) found that on average, rape was used for 76% of foraging visits, although it only represented 39% of the available habitat. Wheat, grass and dykes were also used, but linseed and peas were avoided completely.

Robinson *et al.* (2001) found that in grassland landscapes, increasing the amount of arable cultivation is likely to be beneficial for granivorous bird species. The presence of more fallow land was associated with better breeding performance for Reed Bunting, as was the presence of more mixed farming land (Siriwardena *et al.* 2001).

8.4.2   Winter

There is little emigration from Britain during the winter, with less than 1% of birds leaving the country (Prŷs-Jones 1984). Aside from a withdrawal from more upland regions, Reed Buntings are found in most farmland areas during the winter. They are widely spread over farmland in small flocks, although they congregate in evening roosts in marshy areas such as reedbeds. Most birds do not move far from the breeding site in the winter (Prŷs-Jones 1984). Throughout the range, there is often a shift after the breeding season to drier and more open situations, more similar to habitats of other Emberizidae at this season (Brown and Atkinson 1996, Cramp and Perrins 1994).

Buckingham and Peach (2006) showed that Reed Bunting had a preference for ungrazed seeded plots over-winter, compared to plots that were grazed or mown. They have also been found to avoid ploughed fields and winter cereals, favouring weed-rich winter stubbles, as these provide more food (Orlowski 2005, Field *et al.* 2007). A study of lowland farms in central England showed that Reed Buntings preferentially selected intensive barley stubbles on which to forage during the winter, because these habitats were associated with greater densities of key dietary seeds, and also provided increased areas of bare soil, which apparently allowed more effective feeding (Moorcroft *et al.* 2002). In Poland, however, wintering buntings preferred to forage in densely weeded fields, while cereal stubbles were characterised by a relatively low abundance of birds (Orlowski 2005). Wild bird cover crops may also be used, particularly kale, quinoa and millet and crops containing a high abundance of weeds (Stoate *et al.* 2003, Henderson *et al.* 2004). Siriwardena *et al.* (2008) showed that Reed Bunting use feeding plots, especially after mid-February.

8.5 Impacts of Environmental Change

8.5.1 Climate change

The trend in first-year survival identified by Peach *et al.* (1999) does not seem to have been caused by any associated changes in the severity of winter weather and it seems unlikely that climatic factors could have caused the decline in Reed Bunting numbers in Britain. Although severe winter weather did cause a dramatic population decline in 1963, and may have contributed to smaller declines in 1977, 1979 and 1982, other cold winters had little apparent effect on numbers (Peach *et al.* 1999).

8.5.2 Habitat loss and degradation

The loss of small wet features such as ponds, field drainage, and the dredging and straightening of rivers and streams is likely to have reduced the suitability of large areas of farmland as a breeding habitat for Reed Bunting (Peach *et al.* 1999, Brickle and Peach 2004). However, this has probably occurred continuously during the 20th century and it is not clear how the loss of such habitat could have caused the relatively sudden population decline between 1976 and 1983 (Peach *et al.* 1999).

Modern grasslands are poor habitats for granivorous birds and drainage, reseeding, the switch from hay to silage and increased fertilization and stocking rates have all had an impact on habitat quality. The practice of autumn grazing and cutting of grass swards has reduced the value of seeded swards for birds as an over-wintering habitat as this prevents the grasses from setting seed (Chamberlain *et al.* 2001, Buckingham and Peach 2006). An increase in the number of sheep has been associated with increased probability of local extinction of Reed Bunting from 10 km squares and there has also found to be an increased likelihood of extinction with increasing altitude, suggesting losses have been more common in the uplands (Chamberlain *et al.* 2000).

The large-scale switch from spring-sown to autumn sown crops in the 1970s resulted in a loss of winter stubbles, a preferred feeding habitat for over-wintering Reed Buntings (Wilson *et al.* 1997 etc). However, despite the continued loss of spring-sown cereals since 1983, Reed Bunting numbers have remained relatively stable (Peach *et al.* 1999).

8.5.3 Loss of nesting sites

No information

8.5.4 Pollution

No information

8.6   Impacts of Biological Change

8.6.1   Predation

Relatively high nesting success in both farmland and wetland habitats during the period of most rapid population decline suggests that nesting success per breeding attempt was not an important factor driving the population decline of the late 1970s, although the decline in nesting success in northern Britain might have contributed to the larger, more prolonged population decline in that region (Peach *et al.* 1999).

Brickle and Peach (2004) found that of 152 nests, 35% were unsuccessful as a result of predation (21% of all nests at the egg stage and 10% at the nestling stage). They suggest that nest concealment may be a potentially limiting factor for farmland Reed Buntings as nest survival was positively related to nest concealment. Crick *et al.* (1994) found that predation accounted for a third of all nest losses in the UK and Schiegg *et al.* (2007) state that it was the main source of nest failure in their study area in Switzerland. Other studies have found between 53 and 93% of nests failures were due to predation, with predators including Carrion Crow *Corvus corone*, Magpie *Pica pica*, Weasel *Mustela nivalis*, Fox *Vulpes*, rat *Rattus*, White Stork *Ciconia ciconia* grass snake, vole Arvicola, Cat *Felix cattus* and man (Haukioja 1970, Hornby 1971, Ghiot 1976, Okulewicz 1989 in Cramp and Perrins 1994).

8.6.2   Hunting

The Migration Atlas reports that 34% of recoveries of dead birds with a known cuase of death were deliberately taken by man (Prŷs-Jones 2002). However, given that British Reed Buntings are resident to the UK, and that this is not a quarry species in this country, hunting is not likely to be a driver of declines in Reed Bunting.

8.6.3   Competition

No information

8.6.4   Disturbance

There is little information regarding the impact of disturbance on Reed Bunting populations. Survival of nests was found not to be influenced by spraying activities in fields of oilseed rape in east Scotland, although in some years, up to 50% of nests could be lost to swathing (Burton *et al.* 1999).

8.6.5   Food availability

The most likely cause of the decline in the British Reed Bunting population is the loss of suitable food and habitat on farmland as a consequence of changes in agricultural practices (Peach *et al.* 1999). The loss of hedges, ditches, ponds and uncultivated field margins coupled with the increased usage of herbicides probably removed much suitable Reed Bunting breeding habitat on farmland during the late 1970s. Granivorous species such as Reed Bunting are most likely to be limited by the problem of a late-winter 'hungry gap' in food availability (Siriwardena *et al.* 2008). A reduction in the abundance of invertebrates during summer might account for the lower nesting success of Reed Buntings in both farmland and waterside habitats since the mid-1980s, but this trend requires further investigation (Peach *et al.* 1999).

The abundance of invertebrate prey may limit breeding productivity through its effect on the number and condition of chicks fledged. The spraying of rotational set-aside with broad spectrum herbicides during late April or May is likely to have reduced the value of these fields as both nesting and foraging habitats for Reed Buntings (Brickle and Peach 2004). Further to this, the large-scale switch

from spring-sown to autumn sown crops in the 1970s resulted in a loss of winter stubbles, a preferred feeding habitat for over-wintering Reed Buntings (Wilson *et al.* 1997 etc).

A study looking at the importance of field boundaries found that Reed Bunting distribution was influenced by variables indicative of the importance of food supply which was associated with the presence of plants supplying seeds and insects (Sparks *et al.* 1996). Mean annual nesting success was significantly higher in years with mass outbreaks of *Epirrita* larvae in mountain birch forest in central Norway, although this didn't translate into a corresponding increase in population density (Hogstad 2005).

8.6.6   Summary

It is likely that the decline in Reed Bunting is related to agricultural intensification which has resulted in the loss of habitat through drainage of wet areas (reducing nesting habitat) and loss of winter food resources, especially weed-rich stubbles, their preferred feeding habitat (Newton 2004). This has resulted in decreasing survival rates, although a subsequent population recovery may have been prevented by increased nest losses (Wilson *et al.* 1996, Peach *et al.* 1999, Newton 2004). This is supported by a moderate decline in CES productivity and a significant increase in failure rates at the egg stage. Farmland densities are four times higher in oilseed rape than in cereals or set-aside and this crop is crucial in reducing the dependency of the species on wetlands (Gruar *et al.* 2006).

8.6.7   Possible actions to aid recovery

Installing small wet features, such as ponds, in farmland will provide foraging areas (Brickle and Peach 2004). It is also important to ensure that ditches remain wet all year round, and that the emergent rank vegetation is left in place as this will also provide a nesting habitat (Bradbury and Kirby 2006). Allowing grasses to set seed over-winter, rather than mowing them, will provide foraging areas for granivorous birds (Buckingham and Peach 2006).

Of the measures available under agri-environment schemes in England, the establishment of uncropped, tussocky grass field margins and wildlife strips, pollen and nectar mixes and uncropped field corners should all increase nesting and feeding opportunities for farmland Reed Buntings (Brickle and Peach 2004). However, revision of existing agri-environment measures is needed, including the extension of the period over which seed-rich habitats are retained into spring and adjustment of option content or management to promote seed retention at the same time.

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9. SEDGE WARBLER *ACROCEPHALUS SCHOENOBAENUS*

9.1 British Population Status And Trend

BOCC status: Green

Indicator trend: Fluctuating, with no long-term trend

Other trend data: BBS data for the UK 1994 – 2007 show an 8% increase while data for England show an 11% decrease (Risely *et al.* 2008). This moderate decline in England is slightly uncertain because the long-term changes are partly obscured by shorter fluctuations in numbers (Baillie *et al.* 2007). CBC/BBS data for England show a 33% decline between 1967 and 2005 while WBS data points to 20% decline from 1975-2005 (Baillie *et al.* 2007). The smoothed CBC/BBS and WBS trends show four troughs in population, related to years of poor West African rainfall, with a low point in 1984–85. The CES, which provides the biggest Sedge Warbler sample, also illustrates the large year-to-year fluctuations that occur in this species. CES data show a 19% decline in adults and a 36% decline in juveniles over the period 1984-2005 (Baillie *et al.* 2007). CES data for the 2007 season saw adult numbers at their lowest level ever (Coiffait *et al.* 2008).

Summary: Although this species shows fluctuations in its population, most datasets do point to a long-term decrease in numbers.
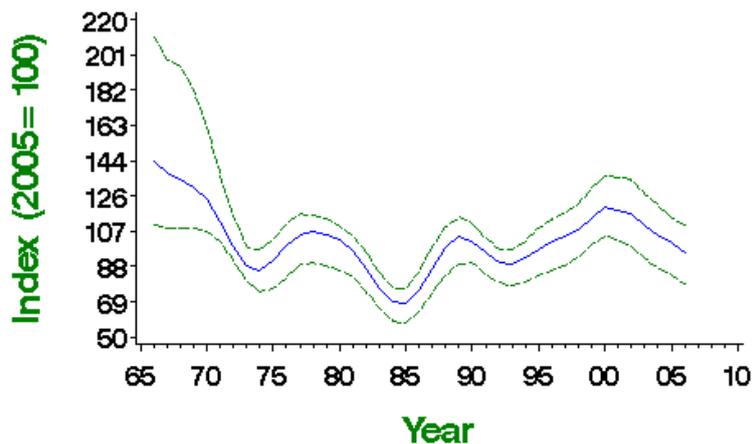
Figure 8 Combined CBC/BBS UK trend for Sedge Warbler 1966 – 2006. From Baillie *et al.* 2007

9.2 Distribution And Demographic Changes

9.2.1 Worldwide and European

This species is a widespread summer visitor in Europe, except in the south. Its European breeding population is very large (>4,400,000 pairs), and was stable between 1970 and 1990 (Birdlife International 2004). Russia holds a large proportion of European breeding population. Habitats include drier reedbed margins often with emergent scrub, overgrown ditches and cover away from water such as hedgerows. It winters widely in Africa south of the Sahara. There were rapid declines in Belgium and Germany 1970-1990 and in Finland during 1990–2000 (Birdlife International 2004). Sedge Warbler populations in The Netherlands declined dramatically, especially between 1973-75 and 1982-85 (Foppen *et al.* 1999). Overall the species has remained stable and consequently, it is evaluated as Secure.

9.2.2 Britain

The Sedge Warbler is a migrant breeder and passage visitor to Britain, wintering in tropical and southern Africa. This species is widespread and numerous throughout lowland Britain. The summer population was estimated to be between 297,000 and 321,000 territories in 2000 (Birdlife

International 2004, Baker *et al.* 2006). Lack (1989) reported that over a 25 year period, Sedge Warbler showed fluctuations in number but there was no significant overall change.

9.2.3   Changes in breeding parameters

Much of the year-to-year variation in population size has been driven by changes in adult survival rates which, in turn, are related to changes in rainfall on their wintering grounds, just south of the Sahara Desert, in the West African Sahel (Peach *et al.* 1991). Short-term variation can generally be explained by rainfall patterns in the Sahel region of Africa, as highlighted by CES results (Peach *et al.* 1991). Peach *et al.* (1991) found that daily nest survival rates at the egg and chick stage had increased, suggesting that breeding success had not declined. CES productivity data show a steep fall in the 1980s, followed by further shallow decrease (Baillie *et al.* 2007).

9.3   Resource Requirements

9.3.1   Breeding season diet

Sedge Warblers are insectivorous. They feed predominantly low down in dense vegetation, notably reeds and rushes, and also in cereal fields and periphery of bushes (Cramp 1992). The following have been recorded in the diet in the west Palaearctic: mayflies (Ephemeroptera), dragonflies and damsel flies (Odonata), grasshoppers (Orthoptera:), psocids (Psocoptera), adult and larval bugs (Hemiptera), adult and larval lacewings (Neuroptera), adult and larval Lepidoptera, caddis flies (Trichoptera), flies (Diptera), Hymenoptera, adult and larval beetles (Coleoptera), harvestmen (Opiliones), spiders (Araneae), slugs and small gastropod molluscs and earthworms (Oligochaeta; Cramp 1992, Chernetsov and Manukyan 1999).

The diet of the young is similar to that of adults and is largely comprised of soft-bodied insects such as Chironomidae and other Diptera (Tipulidae and Syrphidae), dragonflies and damsel flies, aphids, lacewings, and adult and larval Lepidoptera (Cramp 1992).

9.3.2   Winter diet

Sedge Warbler are mainly insectivorous but may take some plant material in winter. In late summer and autumn commonly taken invertebrates include aphids, chironomids, Hymenoptera, beetles, and spiders (Chernetsov and Manukyan 1999). Plant material includes the flowers and berries of *Salvadora persica*. Invertebrates taken in winter quarters are notably Chironomidae and other soft-bodied arthropods (Cramp 1992).

9.4   Habitat Associations

9.4.1   Breeding season

This species shows a preference for dense vegetation generally along the water's edge (but by no means exclusively near water) along rivers estuaries, lakes and ponds, utilising reeds, willow carr, scrub and bushes, although it usually avoids wetter reedbeds in standing water. It will breed in dry situations offering suitable cover, including neglected orchards, farm hedgerows, nettlebeds, fields under rice, barley, beans, oilseed rape, clover, and other crops, and even in dry young conifer plantations with trees up to c. 2 m high or in dense scrub of sea buckthorn *Hippophae rhamnoides* (Cramp 1992). It nests low down in dense bankside vegetation occasionally over water (Ward *et al.* 1994). The main habitat in farmland was found to be the fringes of water in ponds and water courses and farm CBC plots with running water held four times higher densities of Sedge Warblers than plots without (Lack 1989).

Sedge Warbler have been found breeding in oil seed rape in east Scotland, although they were found mainly close to dykes and it is thought they were nesting in the dykes rather than the rape itself

(Burton *et al.* 1999, Zajác *et al.* 2008). In Poland, Sedge Warbler occupied mid-field marsh patches and drainage ditches, preferring areas with a high proportion of reeds and meadows. In general, birds preferred to settle near fields of oil seed rape, while root crops and spring cereals were avoided (Surmaki 2005a). Adults were found to forage in oil seed rape and cereals, oil seed rape being visited mainly in May, while cereal fields were visited in June and July, showing that oil seed rape may be useful as a foraging habitat during the breeding season (Surmaki 2005b). Lack (1989) found that Sedge Warblers were commonest on arable farms of East England and had the lowest density in grass regions and Lack suggested that cereal fields may actually be more similar to their natural habitat grass fields, particularly those which are cut or grazed.

This species prefers areas with tall vegetation and often preferring areas that contain very tall sedges in fen habitats, which are especially important as song-posts (Bracken *et al.* 2008). A study in south west Slovakia found that the Sedge Warbler's habitat was represented mainly by littoral components of vegetation and that parameters of the littoral vegetation played an important role in habitat selection (Grubjarova *et al.* 2005). In reedbeds in Hungary, Sedge Warbler showed a strong preference for habitat edges and there was a significant edge (0-5 m) preference for reed edges (Báldi and Kisbenedek 1999). Sedge Warblers have been shown to show a preference for uncut reed; densities in uncut reed and other vegetation were 6-50 times higher than in cut vegetation in the Netherlands (1997, 1999). Uncut reed is more suitable for nesting as it allows earlier nesting - and thus to raise on average more clutches per season - and because the birds suffer a lower predation risk in uncut reed (1997, 1999). Anselin and Meire (1989) suggested that the presence of old reed is a possible cue for territory selection.

A study in England found that feeding sites utilized in breeding season were 47% marshland (chiefly Glyceria), 26% scrub, 21% field vegetation, 6% woodlands (Catchpole 1973b in Cramp 1992). In late summer, willowherb may be an important source of insect food (Prato and Prato 1977 in Cramp 1992).

9.4.2   Winter

In Africa, Sedge Warbler use a wide variety of habitats, including reeds over or next to water, papyrus, rank grass, sedge, reedmace, bushes, and tall elephant grass, often near lakes and may also occur in low trees and moist thicket (Cramp 1992).

9.5   Impacts of Environmental Change

9.5.1   Climate change

Fluctuations in population levels and annual adult survival rates since the late 1960s are strongly correlated with indices of wet season rainfall in the west African winter quarters. Peach *et al.* (1991) showed that mortality rates of wintering Sedge Warblers increased in years with poor rainfall in West Africa and habitat availability in the winter quarters has probably been the main factor limiting the size of the population in Britain during 1963-1988. Glue (1993) reported that a second year of improved rainfall in 1992 over parts of the drought-stricken Sahel region enhanced the survival and numbers of Sedge Warbler returning to breed in Britain in 1993. Population declines in the Netherlands were found to correlate with yearly rainfall in the western part of the Sahel-Sudan zone (Foppen *et al.* 1999) and similar patterns in adult abundance have been found in data from other countries in Europe (Peach *et al.* 1991).

Péron *et al.* (2007) found that Sedge Warbler advanced their migration date in years with warm springs and therefore does not take advantage of climate warming to spend more time in temperate breeding grounds. This suggests that the advance in departure dates is explained by fitness benefits from early arrival at wintering grounds.

9.5.2   Habitat loss and degradation

Drought in the wintering grounds in West Africa has had a negative impact on the quantity and quality of wintering habitats (Peach *et al.* 1999). However, habitat loss and degradation through drainage and increasing agricultural intensification in the breeding grounds may have been responsible for preventing Sedge Warbler populations from recovering (Peach *et al.* 1991). Foppen *et al.* (1999) found that breeding populations in relatively unfragmented landscapes showed a clear recovery after the 1984 population decline, but that those in heavily fragmented landscapes showed no recovery. Simulations showed that in fragmented landscapes (less than 1% marshland) the relative decrease in numbers was 50% higher than in less fragmented habitats. Furthermore, the recovery to initial numbers after a decrease in fragmented marshland landscapes would take about five times longer than in areas with more than 15% suitable habitat. They concluded that Sedge Warblers breeding in fragmented marshland habitats are more vulnerable to drought events in West Africa than those in unfragmented habitats (Foppen *et al.* 1999)

9.5.3   Loss of nesting sites

No information available.

9.5.4   Pollution

Berthold (1973) claimed that the decline was caused by pesticide-contamination in the areas of passage migration and/or in the winter quarters. However, other than this study, no evidence was found detailing the impact of pollution on Sedge Warbler declines.

9.6   Impacts of Biological Change

9.6.1   Predation

Twenty-six per cent of Sedge warbler nest attempts in wetlands in Poland failed due to predation. A study in the Netherlands found that there were no differences in clutch size or fledging success in non-depredated nests of Sedge Warblers between cut and uncut vegetation but the predation rate of nests in cut vegetation was 1.6 times as high as in uncut vegetation (Graveland 1997, 1999). Thirty-six per cent of recoveries of all dead birds reported in the Migration Atlas were categorized as being taken by domestic predators (Peach 2002). There is little other information regarding predation and it is unclear how much impact predation has had upon population declines.

9.6.2   Hunting

Peach (2002) states that 10 out of 12 recoveries from south of the Sahara with known cause of death were birds that were deliberately taken by man. Only 9% of recoveries of all dead birds reported in the Migration Atlas were categorized as being deliberately taken by man (Peach 2002). As hunting levels has decreased, it seems unlikely that this has been a driver in the declines of Sedge Warbler.

9.6.3   Competition

Populations of Sedge Warblers appear to be limited by competition for resources on the wintering grounds, probably because drought has decreased the quantity and quality of these habitats (Peach *et al.* 1991, Baillie and Peach 1992).

9.6.4   Disturbance

No information available.

9.6.5 Food availability

Low rainfall in the wintering grounds is thought to have resulted in decreased food supplies in these habitats (see above section on 'Climate Change'). The amount of food available can also influence migration strategies, which in turn can impact on arrival dates and fitness. For example, the rate at which birds gained weight varied from year to year at the same site from 0.40 g, 0.05 g and 0.55 g per day in 1973, 1974 and 1975, respectively (Bibby *et al.* 1976). In 1974, when the rate of weight gain was low, 84% of Sedge Warblers stayed more than 2 days. In 1973 and 1975, when the rate of gain was greatest, only 46% stayed two or more days. Slow rates of fattening, delay migration and migrants that reach breeding areas late or in poor condition may fail to breed that year (Newton 2004). This variation in stopover length between sites and years is thought to be because the supply of its preferred diet (reed aphids) is spatially and temporally unpredictable but can be superabundant (Schaub and Jenni 2001).

9.6.6 Summary

Sedge Warbler numbers have fluctuated, with a decrease in the early 1970s. The population trends do not show any general association with factors occurring in the breeding grounds, but the trends are strongly associated with where the species spend the winter (Lack 1989). Variation in losses between fledging and the following breeding season were correlated with conditions on the wintering grounds (Baillie and Peach 1992), particularly the amount of rainfall. Populations of Sedge Warblers appear to be limited by competition for resources on the wintering grounds (Baillie and Peach 1992), although degradation of breeding habitats through drainage and agricultural intensification may have hindered population recovery.

9.6.7 Possible actions to aid recovery

Retaining existing stands of emergent vegetation, in larger blocks where possible will help improve the breeding habitat for this species. Establishment of new stands by transferring dredged material to areas of shallow water will also help create new breeding habitat (Ward *et al.* 1994). Reducing the intensity of farming on breeding areas is also likely to help. Further studies of factors operating on the wintering grounds are also important.

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10. SNIPE *GALLINAGO GALLINAGO*

10.1 British Population Status And Trend

BOCC status: Amber
Indicator trend: The long-term trend for Snipe is a probable decline.
Other trend data: CBC data from lowland England show that numbers have fallen rapidly since the 1970s and these data show that Snipe underwent a 64% population decline between 1974 and 1999 (Gregory *et al.* 2002, Gibbons *et al.* 1993, Siriwardena *et al.* 2000). However, more recently, BBS data for the entire UK show a 38% increase in numbers between 1994 and 2007, with a 58% increase in Scotland and a 2% increase in England (Risely *et al.* 2008). The combined BBS/CBC/WBBS/WBS trend in England from 1974 to 2007 shows an initial decline followed by a recovery between 1976 and 1982 (Baillie *et al.* 2007). The population underwent a quite steep decline until the mid-1990s and numbers have not recovered since then although they have remained relatively stable. The trend in the upland and moorland strongholds of Snipe is not fully known, but the 1988–91 atlas documented range loss widely in Wales, Northern Ireland and Scotland, as well as lowland England, and a general decrease is therefore probable.
Summary: In England, numbers of Snipe have fallen rapidly since the 1970s although recently, the population appears to be showing signs of recovery.
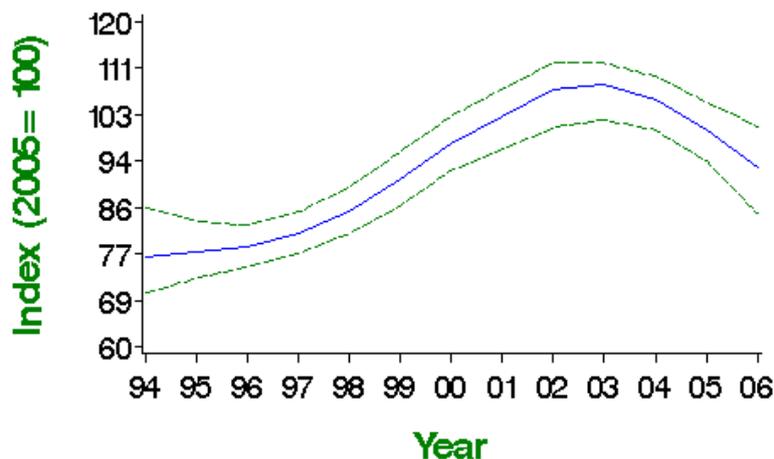
Figure 9 BBS trend for the UK for Snipe 1994 – 2006. From Baillie *et al.* 2007

10.2 Distribution and Demographic Changes

10.2.1 Worldwide and European: Snipe breed in high and middle latitudes, from the low Arctic throughout boreal and temperate zones but not in the Mediterranean, and only marginally overlaps steppe. It spans continental and oceanic level lowlands where drainage is impeded, with access to shallow fresh or brackish water (Cramp and Simmons 1983). Its European breeding population is large, at more than 930,000 pairs (BirdLife International 2004). Wet grasslands are the most important breeding habitats within the countries of the EC (Hotker 1991) but declines have been reported consistently across Europe (Hotker 1991, Nijland *et al.* 1996) mainly due to habitat changes, especially drainage (Cramp and Simmons 1983). Overall, declines in Europe have been estimated at more than 10% and this species has a 'declining' status (Birdlife International 2004).

10.2.2 Britain: The snipe is a migrant/resident breeder and a passage/winter visitor in the UK. It is widespread, but now relatively scarce on farmland (Ward *et al.* 1994). Sharrock (1976) estimated there to be 80,000–110,000 pairs, but current estimates are much lower than this at around 59,000 pairs (52,600–69,000 95% confidence intervals) (Birdlife International 2004, O'Brien 2005, Baker *et al.* 2006). The population experienced a decline during the 19th century, then some increase early in the 20th century, followed by a marked decline in central and southern England since c. 1950 and

local decreases elsewhere, possibly due to drainage (Cramp and Simmons 1983). Numbers have also seen reductions after hard winters (Sharrock 1976). Between the 1970s and 2000s, Snipe population numbers underwent declines of over 60% (Gregory *et al.* 2002, Wilson *et al.* 2005). The repeat Breeding Waders of Wet Meadows survey in 2002 estimated a 62% decline in Snipe in England and Wales since 1982, with the remaining birds becoming highly aggregated into a tiny number of suitable sites (Wilson *et al.* 2005). Smith (1983) found that 37% of the Snipe population on lowland wet grassland in England and Wales were on just two sites, the Ouse and the Nene Washes, while a later study by Wilson *et al.* (2005) also found these sites to be extremely important and that Snipe only occupied 8% of sites surveyed. In Scotland, 50% of Snipe were found on only 4% of the Scottish lowlands (O'Brien and Bainbridge 2002) and the number of 10 km squares occupied by Snipe in Britain declined by 19% between 1968-72 and 1988-91 (Gibbons *et al.* 1993). In Northern Ireland, a breeding decline of around 30% occurred between the mid 1980s and 1999 (Henderson *et al.* 2002).

Changes in Snipe numbers are variable across the UK and a recent study found significant population declines in the south and east of England but a 61% population increase since the early 1990s at the Nene Washes (Smart *et al.* 2008). Snipe also increased significantly in the Lake District and in the West Flows between the 1980s and 2002 (Sim *et al.* 2005). Snipe decreased by 74% on upland CBC sites in Britain between 1968-80 and 2000 (Henderson *et al.* 2004).

10.2.3 Changes in demographic parameters: It is thought that reduced breeding success is responsible for the overall decrease in population size, although further studies on this are needed. Daily nest failure rates at the egg stage appear to have halved (Baillie *et al.* 2007).

10.3   Resource Requirements

10.3.1 Breeding season diet

Snipe diet is mainly comprised of invertebrates, feeding chiefly on wet ground at the edge of water or in shallows, taking items by probing, or much less often from the surface or vegetation (Green 1984, Green 1988, Green, *et al.* 1990). Insects include, most frequently, larval and adult dipteran flies (Chironomidae, Tipulidae, Culicidae, Tabanidae, Syrphidae, Stratiomyidae) and beetles (Carabidae, Hydrophilidae, Dytiscidae, Staphylinidae, Cerambycidae, Elateridae, Scarabaeidae) but also ants (Formicidae), caddisflies (e.g. larvae of Phryganeidae), dragonflies and damsel flies (e.g. Libellula, Agriidae), mayflies (Ephemeroptera), bugs (e.g. Corixa, Ilycoris, Philaenus), lepidopterans (e.g. Larentia), oligochaete (e.g. Lumbricidae) and polychaete worms (e.g. Nereidae), leeches (Hirudinea), molluscs (Helicidae, Valvata, Viviparus, Bulimus, Planorbis, and Bithynia), isopods (e.g. Asellus), amphipods (e.g. Gammaridae), spiders (Araneae) and frogs (Rana). Plant material has been consistently recorded and includes vegetative debris (e.g. moss, leaves, roots and stems of grass, rushes, etc.) and seeds (Cramp and Simmons 1983, Green *et al.* 1990).

Hoodless *et al.* (2007) found that Snipe diet during April-June on moorland consisted mainly of earthworms and tipulid larvae, which accounted for 61 ± 7% and 24 ± 6% of the dry weight of prey items ingested respectively, although a wide variety of surface-active and aquatic prey were also taken, especially early in the season. There were two main differences between the findings of Hoodless *et al.* (2007) on moorland and Green *et al.*'s (1990) study on wet grassland; tipulid larvae were taken more frequently and more aquatic groups were included in the diet on moorland than on lowland wet grassland, where the only aquatic prey were chironomid larvae. Green *et al.* (1990) found that the amount of earthworms in the diet varies between years in relation to flooding in spring as excess flooding results in a decrease in earthworm abundance.

Snipe chicks are fed by the parents on invertebrates from the soil (Green 1984).

10.3.2 Winter diet

Quantitative data on winter diet are sparse but winter diet is thought to be similar to that in the breeding season (Cramp and Simmons 1983).

10.4   Habitat Associations

10.4.1 Breeding season

Snipe require soft damp ground with a 'tussocky' sward and large open fields without tall trees. The preferred sward height is 10-30 cm with tussocks of 50-80 cm, with a cover of around 30-40% short sward, less than 20% bare ground, 60-70% tussocks and up to 20-30% wet ground/shallow water (<10 cm) (Mason and MacDonald 1976, Green 1986, English Nature 2002). Baines (1988) found that Snipe were more abundant in upland grasslands with dense *Juncus*. Any tall or dense vegetation present needs to be separated by more open ground with tussocks or clumps of sedge, rushes and coarse grasses. Pure sedge is associated with unsuitably deep water and pure grass with ground too hard for its bill to probe (Cramp and Simmons 1983). Pearce-Higgins and Grant (2006) reported that Snipe abundance was positively correlated with more heterogeneous vegetation structure on moorlands in southern Scotland and Northern England and that Snipe were positively associated with plants that were indicative of wet conditions, such as sedge. Snipe densities on marshy grassland or acid flush were 3.4 times higher than on heath and heath/grass habitats in Northumberland (Hoodless *et al.* 2007). Shepherd and Stroud (1991) found the highest densities of Snipe in the inner Hebrides breeding on marsh/fen and also on marshy *Phragmites* beds where the reed was sparse and short. They found that wet machair, wet acidic grassland and course wet pasture were also important breeding habitats.

Breeding Snipe are now restricted to a few key places such as nature reserves and areas where management for breeding waders is implemented (Smart *et al.* 2008). The basic requirement of this species is soft accessible organic soil, rich in food organisms just below surface, with frequent patches vegetation cover affording good visibility of approaching danger, and preferably also provided with look-out posts 1 m or more above ground (Cramp and Simmons 1983). In temperate and oceanic regions, it typically frequents open wet ground such as rushy fields, water-meadows, washes, marshy edges of rivers and lakes, salt-marshes, and blanket bogs (Mason and MacDonald 1976, Sharrock 1976) and shows a preference for sites on peat soils (Wilson *et al.* 2004).

Snipe are more likely to persist in fields where the soil conditions are wet and soft, with more standing water and soil moisture (Brown and Stillman 1993, Green and Robins 1993, Stillman and Brown 1994, Vickery *et al.* 1997, Henderson *et al.* 2002, Smart *et al.* 2008). Smith (1983) noted from a survey of lowland wet grassland in England and Wales that Snipe were absent from coastal grazing marsh. He postulated that the alluvial soils of the coastal marshes were too hard for feeding. Incubating female Snipe have been seen to forage in both unflooded meadowland near the nest and on wetter areas at the edges of pools and ditches (Green *et al.* 1990). Green found that they fed near the nest where there was a moderate biomass density of invertebrates and the soil was soft and easy to probe but fed in the wetter habitats further from the nest if the invertebrate density near the nest was low or the soil had dried out (Green *et al.* 1990). On moorland in Northumberland, Snipe were flushed from locations that were closer to ditches or pools and characterized by wetter soil and more uneven swards than randomly selected points (Hoodless *et al.* 2007).

Snipe usually avoid hard, dry, or broken ground, closed woody vegetation, and unsheltered open areas (Cramp and Simmons 1983). However, Snipe can be found in places where the typical preferred habitat is lacking, such as on the St Kilda island group, Scotland, where it disperses widely over both moist and dry terrain, dry grassland, maritime grassland above cliffs, heather moor, patches of wood-rush, bracken and iris (Cramp and Simmons 1983). In Iceland it breeds on wet moorlands and marshes, but also on dry grassland and heath, and even in dense birch *Betula* woodland, up to 10 m tall (Gunnerson *et al.* 2006). In the uplands, Snipe were found to be positively associated with low-

altitude areas towards the edge of moorland blocks (Stillman and Brown 1994). Bracken and Bolger (2006) found that Snipe were positively associated with non-rotational set aside in Ireland. At a landscape scale, Atkinson *et al.* (2002) found the breeding Snipe were most abundant in 10 x 10 km squares with more than 50% cover of grassland.

Snipe are site faithful, so isolated patches of suitable area mean limited expansion of breeding pairs (Cramp and Simmons 1983) and means that re-colonisation of a rehabilitated habitat is slow.

10.4.2 Winter

Outside the breeding season, the habitat choice is similar, with more use of artefacts such as sewage farms, wet cattle pastures and ricefields (Cramp and Simmons 1983). Sage *et al.* (2006) recorded that Snipe used short-rotation willow coppice as an over-wintering habitat in Northern England.

Densities of foraging Snipe in winter are closely related to the density and biomass
of earthworms (Hoodless, Draycott and Tucker, 1998). Surface water is also important in wintering habitats and Barnett (2004) found a positive correlation between the amount of surface water on grassland and Snipe numbers. Barnett (2003) also found a negative relationship with grazing, meaning that Snipe were significantly more abundant on unimproved grasslands. Snipe were found to be positively associated with muddy substrates in coastal sections of Orkney (Summers *et al.* 2002).

10.5   Impacts of Environmental Change

10.5.1 Climate change

The decline of Snipe at the Ouse Washes is thought to be due to an increase in the frequency of spring and summer flooding since the early 1980s, reducing the breeding success of Snipe by decreasing the time available for them to lay replacement clutches as flooding delays the onset of nesting by up to 70 days (Green 1988). Nesting stops later when soil conditions remain soft and moist for feeding (Green 1998), but if the soil is harder and drier at the end of the season, laying is likely to stop earlier so drier summers are also likely to have reduced the length of the breeding season.

The extent and suitability of lowland wet grassland will face further pressure in years to come as a result of climate change as coastal marshes may be lost to managed realignment and increased spring floods or droughts are likely to impact on the habitat suitability (Wilson *et al.* 2004).

10.5.2 Habitat loss and degradation

Habitat loss and degradation are thought to be the main causes of decline in most species of wader (Hotker, 1991, International Wader Study Group, 2003). Drainage and related agricultural intensification, and changes in grassland management in the marginal uplands are cited as the main causes of the decline of Snipe populations (Fuller and Ausden 2008). Conversion of lowland wet grassland to arable has reduced the area of grassland available to breeding waders during the 20th century. However, there has been little further overall loss of grassland to arable and since the early 1980s waders have continued to decline (Wilson *et al.* 2005). Instead, intensification of agricultural management, such as re-seeding, increased use of fertilisers, a switch from hay to silage and drainage have decreased the suitability of grassland (See Figure 3 in Curlew review, and Wilson *et al.* 2004). Agricultural intensification and improvement is likely to result in drier, more homogeneous (i.e. less tussocky) fields and encourages large-scale uniformity of habitat (Shepherd and Stroud 1991). Over 97% of grassland in lowland Britain has been improved over the last 50 years (Fuller 1987, Vickery *et al* 1999). There has also been intensification of upland habitats and numbers of Snipe breeding in these areas have also decreased (Fuller *et al.* 2002). Baines (1988) found that Snipe had decreased by 99% on improved upland pastures and disappeared from improved upland meadows, and that densities of Snipe were highest on unimproved pastures. He also reported that Snipe numbers decreased by one pair per 100 ha^{-1} for every 6% land improvement. Hoodless *et al.* (2007) found that

Snipe densities on unimproved acid grassland were 2.7-times higher than on improved grassland in Northumberland. Moorland drainage during the 1970s and 1980s is likely to have reduced pockets of deep, wet peat and thus removed some feeding areas for Snipe. In Ireland, Henderson *et al.* (2002) reported an avoidance of improved grassland, arable land and upland rough grassland by Snipe.

There have been large changes in grazing practices, both in the lowlands and uplands. The most striking of these changes has been the recent large increase in numbers of sheep, which have more than doubled between 1950 and 1990 (Fuller and Gough 1999). The strongest increases were in Wales and northern England, with relatively little change in the Scottish Highlands (Fuller and Gough 1999). Increased grazing pressure resulting from higher stocking densities and increased deer numbers is also likely to have reduced the habitat quality of uplands by influencing the vegetation structure (Sim *et al.* 2005). The resulting heavy grazing has resulted in a shift from heather-dominated habitats to heather/grass mosaics in many upland areas and almost certainly reduced the habitat quality for ground-nesting birds such as Snipe (Fuller and Ausden 2008). Increased stocking densities may have also resulted in an increase in trampling rates and in greater numbers of avian nest predators such as Crows and Magpies. On lowland grasslands, increased densities of sheep has resulted in increased uniformity of swards (Vickery *et al.* 2001) and this reduces the availability of the heterogeneous tussocky swards favoured by Snipe.

In the uplands, afforestation is another issue as this results in the loss and fragmentation of moorland and can also lead to increased cover for predators such as Common Buzzards, Crows and Foxes (Sim *et al.* 2005). There is no doubt that many species of open habitats, especially breeding waders such as Snipe, have been displaced over substantial areas by afforestation in the uplands (Fuller and Ausden 2008).

However, a recent study conducted by Smart *et al.* (2008), found that lowland wet grassland fields in the UK are wetter and softer now than they were in the early 1990s due to changes in management through decreasing grazing pressure and increasing surface flooding. Variation in late summer soil conditions influence the length of the Snipe breeding season, as Green (1988) found that Snipe continue to initiate nests well into July when good feeding conditions exist and that breeding ceased when the penetration resistance had increased to above 5.8 kgf. Smart *et al.* (2008) found that penetration resistance was maintained at less than 6 kgf into June and this was not thought to be a major reason for the population declines. In spite of habitat condition being altered in a way that should have been beneficial to Snipe, numbers have continued to decline, suggesting that other factors must be involved. It is possible that the improved habitat conditions recorded by Smart *et al.* are relatively recent occurrences and thus habitat degradation in the past may have contributed to the population declines. Indeed, the drainage and improvement of wet pastures, and the increased trend towards arable farming are stated as being the chief causes of the decline of Snipe in lowland Britain by Mason and MacDonald (1976) and Baines (1988) attributes the virtual absence of Snipe from improved areas to drainage

Habitat loss and degradation has also been seen across Europe. For example, in Schleswig-Holstein, Germany, many wet meadows have been transformed into cultivated grassland and Snipe numbers have drastically declined from 13,000 pairs in 1970 to 1,500 pairs in 1992 (Busche 1994). Nijland *et al.* (1996) found that Snipe numbers on reserves in the province of Friesland, the Netherlands, were 13 times higher than on cultivated grassland.

10.5.3 Pollution

No information.

10.6 Impacts of Biological Change

10.6.1 Predation

Increases in predator numbers and changes to the habitat making nests and chicks more vulnerable to predation, may have increased predation rates, although detailed quantitive evidence on this are scarce. Recent empirical evidence suggests that levels of predation on wader nests are unsustainably high in many cases, even in some situations where breeding habitat is otherwise favourable (MacDonald and Bolton 2008). Studies of Snipe productivity on lowland wet grassland show that trampling and predation were key factors in limiting nest success (Mason and MacDonald 1976, Green 1988). Green (1988) found that nest failure was responsible for reducing productivity by around 40%. Predation is thought to be an important cause of decline in certain populations; a study on the island of Uist showed that there were large declines in wader numbers and nest success following the spread of the introduced hedgehog. The Snipe population declined by 43% on South Uist between 1983, before hedgehogs colonised the island, and 1995, several years after hedgehogs were introduced (Jackson and Green 2000). Predation of wader eggs by hedgehogs was frequent and this is cited as a cause of the decline in Snipe population on Uist (Jackson and Green 2000, Jackson *et al.* 2004).

10.6.2 Hunting

Snipe are a quarry species and hunting pressure in mainland Europe is high, with 680,000 Snipe hunted annually in France (Leray 1998). Most recoveries of Snipe ringed in Britain and Ireland from abroad have been killed by hunters along the Atlantic coasts of France and Iberia (Henderson 2002). The population in France is threatened by habitat loss and also by the very early opening of the hunting season in mid July, which affects the breeding birds. Dubois *et al.* (1991) advocate delaying the opening of the hunting season until September at least if this species is to be rescued as a breeding bird in France. Bregnballe *et al.* (2004) found that numbers of Snipe increased by 15-39 fold on an area where hunting was banned in a Danish coastal wetland, illustrating the value of hunting refuges for this species. With regards to the population of Snipe in the UK, it seems unlikely that hunting is a major driver, levels of hunting have been decreasing. However, quantitive evidence as to how hunting effects population levels is lacking.

10.6.3 Disturbance

Agricultural improvement has led to increased stocking densities (Wilson *et al.* 2004). At livestock densities of 2-3 cattle ha^{-1}, trampling rates have been found to be as high as early-season predation rates (Green 1988). Snipe nests seem particularly vulnerable to trampling, and a stocking rate of approximately 2·5 cows ha^{-1} for the whole of the incubation period leads to approximately 60% of snipe nests being trampled (Green 1986). However, Smart *et al.* (2008) reported a reduction in grazing intensity and suggested it is unlikely that increases in trampling are causing current reductions in productivity.

10.6.4 Food availability

The increased wetness of fields found by Smart *et al.* (2008) may have actually resulted in a decrease in the earthworm prey available within these fields as the introduction of winter flooding to previously unflooded grassland significantly reduces the abundance of terrestrial earthworm species and arthropods (Ausden *et al.* 2001). It is possible that the extensive flooding which has been necessary to provide soil soft enough for snipe to probe in on mineral soils has ultimately reduced their food supply to levels where it now limits populations of breeding snipe at these sites. In some cases the introduction of surface flooding can result in compaction and consolidation of the upper soil and therefore make it less easy for snipe to probe for prey (Ausden *et al.* 2001).

Agricultural intensification of moorlands and grasslands, including increased grazing intensity and drainage, may also have resulted in a decrease in the invertebrate food availability for chicks (Buchanan *et al.* 2006). The tall, dense swards associated with improved grassland are also likely to make it more difficult for Snipe to probe for invertebrate prey, and they may avoid uniform dense swards such as those managed for silage production as these offer poor chick foraging habitat (Vickery *et al.* 2001).

10.6.5 Summary

There are many potential drivers of the decline but reductions in habitat quantity and quality, driven by land management, are proposed as one of the main contributing factors in the historical declines of breeding Snipe (Newton 2004, Smart *et al.* 2008). This is likely to have resulted in reduced food supply and decreased breeding success. Wilson *et al.* (2004) state that declines are intrinsically linked to drainage and changes in grassland management.

10.6.6 Possible actions to aid recovery

The site fidelity of Snipe means that improving breeding success is a good way of increasing the breeding population. This may be done by improving insect prey at breeding sites, and improving grazing conditions. Prolonging the breeding period by maintaining moist soil conditions for longer should also help. Providing a network of ditches and varied surface topography will help provide abundant damp, penetrable areas even when the water table levels are low. This should provide good feeding habitat for Snipe, which in turn can help prolong the breeding season (Green 1988, Green *et al.* 1990).

There is evidence of a relationship between site protection and changes in Snipe populations (O'Brien and Smith 1992, Wilson *et al.* 2005). Snipe declined by more than 40% on unprotected sites between 1982 and 1989, but only on 15% on reserves (O'Brien and Smith 1992). Wilson *et al.* (2007) found 17 times as many Snipe on RSPB reserves than in the wider countryside. This is likely to be due to the improved management conditions on reserves which aim to increase site wetness. Snipe population trends have been found to be most favourable (increasing or declining less rapidly) in the more expensive ESA options aimed at enhancing habitat; the less expensive, habitat maintenance options, appear to have little benefit (Ausden and Hirons 2002, Wilson *et al.* 2005, Wilson *et al.* 2007).

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11.   YELLOW WAGTAIL *MOTACILLA FLAVA*

11.1   British Population Status and Trend

BOCC status: Red
Indicator trend: The UK trend shows a long-term rapid decline.
Other trend data: BBS data show a decline of 47% for the UK and a 46% decline for England between 1994 and 2007 (Risely *et al.* 2008). Trends derived from combined CBC and BBS data suggest that the Yellow Wagtail decreased by 68% between 1978 and 2003 in England and WBS data recorded a 96% decline between 1975 and 2005 (Baillie *et al.* 2007). Gregory *et al.* (2002a) documented a population decline of 40% between 1970 and 1999 in the UK. This species' range also contracted by 9% between 1968-72 and 1988-91, with birds disappearing from parts of coastal south England, and farmland in Dorset, Hampshire and Wiltshire (Gibbons *et al.* 1993).
Summary: In recent decades, this species has experienced marked declines in abundance and range in Britain, and more widely across Europe (PECBM 2006).
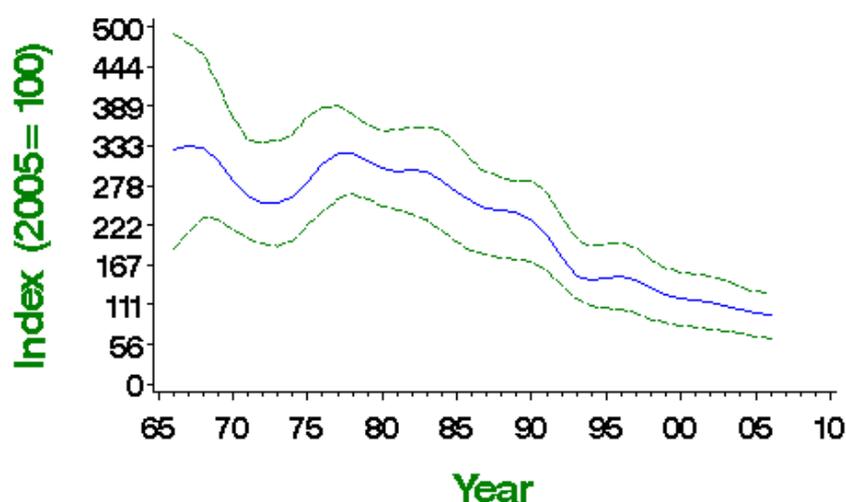

Figure 10 Combined CBC/BBS trend for the UK for Yellow Wagtail 1966 – 2006 From Baillie *et al.* 2007

11.2   Distribution and Demographic Changes

11.2.1 Worldwide and European

This species breeds in the west Palearctic from lower middle to high latitudes in arctic tundra and subarctic, boreal, temperate, steppe, and Mediterranean zones. It is mainly continental but marginally oceanic, largely on level or gently sloping lowlands in damp meadows on river banks and lake shores up to 2000–2500 m (Cramp 1988). In the west of its range (e.g. East and West Germany, Switzerland) it is confined to lowlands, avoiding mountains and broken, arid, sandy, stony, or bare ground, forests, and enclosed landscapes (Cramp 1988). It is a widespread summer visitor to most of Europe, with a very large (>7,900,000 pairs) breeding population, which was stable between 1970–1990 (Birdlife International 2004). Although there were widespread declines, most notably in Romania, during 1990–2000, other key populations (such as those in Russia, Belarus, Poland, Ukraine and Bulgaria) were stable, and the species probably underwent only a slight decline overall. Consequently, it is provisionally evaluated as Secure (Birdlife International 2004).

11.2.2 Britain

The Yellow Wagtail is a summer visitor to Britain, wintering in sub-Saharan Africa and returning to Britain in April (Cramp 1988). It is widespread, occurring as far north and west as southern Scotland, and south-west England respectively, but the greatest concentrations are found along the coastal

fringes of south-east England, in the east Midlands and the Fens of Cambridgeshire and Lincolnshire. It appears that British Yellow Wagtails are declining in range and abundance in pastoral regions, but increasing in arable regions and showing at least population stability in mixed farming regions (Chamberlain and Fuller 2001).

Wilson and Vickery (2005) found that Yellow Wagtails had declined by 65% between 1982 and 2002 on wet grasslands and that site occupancy had decreased from 41% to 16%; they suggest that declines on wet grassland sites have been steeper than elsewhere. Fuller *et al.* (1995) reported a 9% decrease in the number of 10 km squares occupied by Yellow Wagtail between 1970 and 1990. Chamberlain *et al.* (2001) found that losses of Yellow Wagtail were significantly lower in 10 km squares from predominantly arable regions compared to pastoral or mixed farming regions, and they actually expanded in arable regions. Yellow Wagtail had lower local extinction rates in 10 km squares that had increased most in total agricultural area, total tilled area and sugar beet area (Chamberlain *et al.* 2000) and had a population increase in mixed farm regions but decreased significantly on grassland by 85% between 1970 and 1990 (Chamberlain *et al.* 2001). The magnitude of the decline appears to vary between habitats, being especially dramatic in wet grassland and marginal upland areas (Henderson *et al.* 2004, Wilson and Vickery 2005).

These changes have prompted the Yellow Wagtail to be categorised as an Amber List species, and would now qualify for Red List Status (Gregory *et al.* 2002b).

11.2.3 Changes in demographic parameters

Although nest record sample sizes are small, there has been a notable reduction in brood size since the mid 1960s (Baillie *et al.* 2007). There is little other published information on changes in reproductive or mortality rates, or on possible effects on population levels of events in African wintering areas.

11.3 Resource Requirements

11.3.1 Breeding season diet

The diet of both adults and chicks consists almost entirely of small invertebrates (Davies 1977, Holland *et al.* 2006). The following have been recorded in the diet: larval and adult mayflies (Ephemeroptera), dragonflies (Odonata), stoneflies (Plecoptera), grasshoppers, etc. (Orthoptera), earwigs (Dermaptera), termites (Isoptera), bugs (Hemiptera), alder flies, etc. (Neuroptera), larval and adult butterflies and moths (Lepidoptera), larval and adult caddis flies (Trichoptera), larval and adult flies (Diptera), Hymenoptera (sawflies, ants, wasps), adult and larval beetles (Coleoptera), spiders (Araneae), harvestmen (Opiliones), Myriapoda, small molluscs, and worms (Cramp 1988).

In the early part of the breeding season (shortly after arrival from Africa) Yellow Wagtails in Oxfordshire took almost all Diptera, and 86% of the diet consisted of midges (Chironomidae). By May this proportion had dropped to 35%, with a further 44% of the diet consisting of Drosophilidae (Davies 1977). Diptera and Nematocera are important prey throughout the breeding season, as are spiders (Araneae) (Gibbons *et al.* 1993).

Davies (1977) found that nestlings were fed mainly on aphids (Homoptera) and a variety of Diptera, including blowflies (Calliphora), robber-flies (Asilidae), crane-flies (Tipulidae), dung flies (*Scathophaga stercoraria*, amongst others) and house-flies (Muscidae) and weevils (Curculionidae). Gilroy (2006) found that flies, beetles and damselflies (Zygoptera) were important constituents of the nestling diet in a population of Yellow Wagtail in Lincolnshire. This study also noted seasonal changes in the composition of nestling diet, with early broods being fed a greater proportion of beetles, while later broods were provisioned more damselflies (Gilroy 2006). Python (2006) noted that Yellow Wagtails in Switzerland largely fed their offspring with large caterpillars (46%), with Coleoptera, spiders and Nematocera all supplementing the diet.

Yellow Wagtail often feed in association with grazing cattle and sheep, taking insects disturbed by animals or blood-sucking species from animals themselves. Peck rates have been found to be higher when feeding next to cattle (Källander 1993). They are also known to hunt insects emerging from earth turned over by ploughs in fields (Cramp 1988).

11.3.2 Winter diet

In the Afrotropics, food consumption does not alter markedly through course of winter, but food may become increasingly difficult to obtain in late winter as the dry season progresses (Cramp 1988). The following were recorded in Afrotropical winter quarters: stick and leaf insects (Phasmida), cockroaches (Dictyoptera), bugs (Pentatomidae), Hymenoptera, Hemiptera, Orthoptera, larvae of moths (Sphingidae), beetles (Cerambycidae), crustaceans, berries of Salvadora, and small seeds (Cramp 1988).

11.4 Habitat Associations

11.4.1 Breeding season

Yellow Wagtails are often associated with water, but this is not an essential requirement and many birds breed on dry arable farmland in the UK. They breed in habitats as diverse as lowland wet grassland, arable crops, heathland, upland pastures and hay meadows (Snow and Perrins 1998). In England, the breeding population is strongly concentrated within c. 20 km of the coastline and on lower reaches of slow-flowing rivers, in water-meadows, grazed semi-marshland, and cattle pastures. They may also use areas such as sewage farms and reservoir margins (Cramp 1988). Preferred habitats include along river valleys, in water meadows (both freshwater and brackish) and damp fields, especially where the vegetation is low and there is shallow surface water nearby (Ward *et al.* 1994). Dry farmland, heath, moor, and upland areas generally support much lower densities than damper terrain (Sharrock 1976, Fuller 1982).

Mason and Lyczynski (1980) demonstrated that 52% of nests were close to water, and that 67% were in grassland. A study in Norway found a higher proportion (81%) of nests in grassland close to water (Paulsen 1993). In a grassland-breeding population on the Nene Washes, breeding territories were associated with fields previously subject to protracted winter floods and those containing small water features (i.e. ponds or ditches; Bradbury and Bradter 2004). The occurrence of winter flooding was seemingly favourable for Yellow Wagtails because it influenced the architecture of the grassland habitat, creating a sparse sward with a considerable proportion of bare earth interspersed between the sward (Bradbury and Bradter 2004). Areas of sparse, low vegetation are ideal foraging habitats for this species (Nelson 2001, Bradbury and Bradter 2004). In contrast, nests were generally associated with taller swards, presumably because they afford greater concealment (Bradbury and Bradter 2004). These findings suggest that Yellow Wagtails require heterogeneous sward heights within single fields to provide the appropriate foraging and nesting habitats (Bradbury and Bradter 2004). Foraging opportunities may also be provided by access to open water and presence of stock (Bradbury and Bradter 2004) and Flyckt (1999) found that the birds showed a preference for breeding in grazed areas (63-70%) in comparison with mowed meadows. Yellow Wagtail preferred intensive grasslands in a study in Hungary (Baldi *et al.* 2005), although other studies in Hungary have found them to occur at higher abundances on extensive fields (Batary *et al.* 2007). In Italy, Laiolo (2005) found arable to be an important breeding habitat.

Although traditionally viewed as a bird of wet grassland, the Yellow Wagtail has increasingly bred in arable farmland during the last century. Mason and MacDonald (2000) examined the habitat associations of farmland-breeding wagtails in Essex and showed that they favoured spring-sown crops (73% of territories), particularly potatoes, whereas autumn-sown crops and grassland were avoided. However, the situation may be more complex than portrayed in this study. Gilroy (2006) demonstrated a mid-season shift in the habitat associations of Yellow Wagtails in a population in East Anglia, with breeding territories being most closely associated with autumn-sown crops in early May,

but an increasing preference for potatoes with the progression of summer. This shift was interpreted as autumn-sown crops becoming less favourable as their increasing sward height and density limited ground access (Gilroy *et al.* 2008). Crop diversity at a landscape scale was preferred by Yellow Wagtails (Python 2006; Gilroy 2006), presumably because a mosaic of habitat types offers greater foraging and nesting opportunities. In addition, hedges were avoided, while peaty, organic soils were favoured over other soil types. Further investigating this latter finding, Gilroy (2006) showed a strong relationship between wagtail breeding distribution and soil penetrability, with softer soils having higher densities of territories. The precise mechanisms underlying this association are as yet unknown, but may be related to the fact that soil structure, possibly as a function of soil organic matter content, influences prey abundance or affects the ease of nest construction. In a small Swiss population of Yellow Wagtails, there was a greater density of breeding territories in courgette plantations, closely followed by potato fields, but cereal crops were avoided (Python 2006). A study in Denmark found no significant differences in the number of birds on organic and conventionally managed farms (Christensen *et al.* 1996).

Yellow Wagtails generally prefer to feed in open habitats with low vegetation. In arable farmland in eastern England, Yellow Wagtails demonstrated a strong preference for foraging along tracks or near ditches (Gilroy 2006). Individuals provisioning chicks during June used wheat, potatoes and bean crop approximately in proportion to their availability within the environment, but showed a strong aversion to set-aside, oilseed rape and sugar beet (Gilroy 2006). In contrast, there was a seasonal shift in foraging habitat preferences, with wagtails provisioning chicks during July favouring potato crops, while largely avoiding winter-sown wheat. This may be related to the growth stage of the winter wheat (leaves dying off and crop ripening) rendering it less suitable as a foraging habitat (Gilroy 2006). Foraging efforts are generally concentrated with a radius of 130 m of the nest-sites (Python 2006), although occasionally adults will travel up to 1 km to exploit foraging opportunities.

In a modelling study undertaken by Milsom *et al.* (2000) the distribution of Yellow Wagtail was correlated with surface topography of grazing marshes. The probability of a marsh being occupied increased linearly with its area. Marshes that were adjacent to arable land were four times more likely to be occupied than marshes not adjacent to arable.

11.4.2 Winter

This species winters in Africa and is often found in close association with large herbivores, both wild and domestic (Cramp 1988). The range of winter habitats is wide, extending from residual rain-pools and ricefields to parched acacia steppe, and up to an altitude of 2200 m. An essential requirement is for unobstructed ground, whether short grass or cultivation, including cassava plots and inside of a banana grove (Cramp 1988).

11.5   Impacts of Environmental Change

11.5.1   Climate change

Short-term and less severe changes in climate had a positive effect on the potential climate space of Yellow Wagtail with a predicted expansion of up to 19% westwards and northwards (Harrison *et al.* 2003). Cotton (2003) found Yellow Wagtail to be one of just a few species that showed later, rather than earlier trends towards arrival dates in a study in Oxfordshire.

There is no good evidence that Yellow Wagtails have been affected by periodic drought conditions in the Sahel (Marchant *et al.* 1990), although this does not rule out effects of long-term changes in climate and land use in the winter quarters.

11.5.2   Habitat loss and degradation

Drainage and related agricultural intensification of wet grasslands is likely to have contributed to declines in Yellow Wagtail numbers, along with changes in lowland grazing patterns and the change

from spring-sown to autumn-sown arable crops (Vickery *et al.* 2001, Newton 2004, Fuller and Ausden 2008). The replacement of grassland with arable land and the intensification of grassland management may explain the losses from some areas (Chamberlain and Fuller 2000; Vickery *et al.* 2001), Since the 1960s, high fertilizer inputs, drainage and silage production all increased the capacity of grassland to carry livestock (Fuller and Ausden 2008) and this has contributed to the reduction in numbers of breeding Yellow Wagtails (see Figure 3 of Curlew review).

Yellow Wagtails both nest and feed primarily on the ground, so land management practices affecting vegetation structure may also be important. Farmland drainage, the conversion of pasture to arable land and the change from spring to winter cereals have been cited as possible causes of the decline in Yellow Wagtails (Gibbons *et al.* 1993, Nelson *et al.* 2003). Abandonment of breeding areas or population declines is sometimes due to changes in land use or management, especially where denser or taller growth of vegetation during breeding season has resulted. As intensive management practices encourage the growth of dense, uniform swards, much agricultural grassland is now unsuitable as nesting and foraging habitat for Yellow Wagtails, and this loss of habitat may elucidate the especially pronounced declines of this species on wet grassland. Gilroy (2006) speculated that because autumn-sown crops dominate much of the arable landscape in Britain and Europe, a lack of suitable breeding habitat in late summer may curtail the breeding season of Yellow Wagtails, as is the case in Skylarks.

Sage e*t al.* (2006) found evidence that willow short rotation coppice plantations, as an energy crop, may displace Yellow Wagtails.

11.5.3 Pollution

There is some concern about the possible indirect effects of ivermectins on birds which feed in, or around, animal dung (McCracken and Bignal 1991). Research on other species suggests that breeding success and juvenile survival may be reduced in areas where ivermectins are used (McCracken 1993; McCracken, Foster and Kelly (1995). There is little other information available regarding the impact of pollution.

11.6   Impacts of Biological Change

11.6.1 Predation

Nest predation rates of Yellow Wagtail vary according to crop type, thus changes in cropping patterns are likely to have influenced predation rates. The likelihood of predation correlates with proximity to the nearest field edge, with more predation occurring at distances within 60 m from the boundary, than beyond (Gilroy 2006, Morris and Gilroy 2008). Correspondingly, Yellow Wagtails showed a strong avoidance of areas within 60 m of the field edge for nesting, while strongly preferring distances exceeding 100 m in both wheat and potato crops. Additionally, in autumn-sown wheat, the probability of predation was greater closer to tramlines than further away. This can be explained by the fact that predators frequently forage along tramlines, and nests in greater proximity to the edge are more likely to be detected (Gilroy 2006). However, Yellow Wagtails frequently place nests very close to tramlines - as these provide the only ground-access points within the otherwise dense and uniform crop. Hence crop structure restricts birds to nest in areas where predation risk is high. Nests in field bean crops experienced particularly high predation rates, compared to other crop types (Gilroy 2006). As bean crops gain height, abscission of the lower leaves results in an increase in horizontal visibility at ground level, which will facilitate detection by predation (Gilroy 2006). In contrast, visibility decreased with height in other crop types.

Mason and Lyczynski (1980) found that of 65 nest losses, 31% were due to desertion, 29% to predation, and 14% to weather. Bellebaum *et al.* (2002) reported that Yellow Wagtail breeding on wetland meadows in the Lower Oder Valley (Federal State of Brandenburg) experienced a loss rate of 57% to predation or weather, although this had no negative influence on the reproduction rate since re-nesting was possible and concluded that predators do not pose a threat. A study in Poland found

that fox presence negatively influenced the number of Yellow Wagtails (Tryjanowski *et al.* 2002), suggesting that the birds prefer to select areas with lower predation risk.

Overall, there is little quanititative evidence on how predation has impacted upon overall productivity and thus it is hard to know how important it has been in driving population declines.

11.6.2 Hunting

Hunting and other human-induced causes of death predominate in the ring-recovery data with 55% of recoveries deliberately taken by man (Wood 2002). There was little other information regarding the impact of hunting on population declines but as hunting levels has been decreasing it is unlikely that this is a major driver.

11.6.3 Competition

No information available.

11.6.4 Disturbance

The frequency and date of grass cutting is important for birds nesting in agricultural grasslands, particularly in silage fields and the increase in cultivation procedures is likely to be responsible for increased destruction of Yellow Wagtail clutches (Newton 2004). Wilson (1991) found that up to 33% of nests in a study site in the Yorkshire Dales failed due to grass cutting before nestlings had fledged, and recommended that delaying cutting by one or two weeks would have enabled this problem to have been avoided. Similarly, ADAS (1995) found that 25% of 83 nests studied in the Pennine Dales ESA were lost to cutting operations in June and July. Bellebaum *et al.* (2002) found that agricultural operations accounted for the loss of 22% of nests in Germany while Mason and Lyczynski (1980) found 15% of nests were lost to agricultural operation and another 8% trampling to cattle in their study. Despite showing a preference for grazed meadows in a study in Sweden, average breeding success in these areas was lower than in mowed areas (4.26 vs. 4.68 fledged young per nest). Out of 13 unsuccessful breeding attempts, 8 failures were directly caused by grazing cattle (Flyckt 1999). Another Swedish study attributed 42% of nest losses to destruction by mowing (Paulsen 1993).

Despite several studies quantifying nest losses, there is very little information regarding overall effects on producitivity, so the impact on population levels is difficult to assess.

11.6.5 Food availability

The losses of insects associated with cattle have been cited as a possible cause of the decline in Yellow Wagtails (Gibbons *et al.* 1993, Newton 2004, Nelson *et al.* 2003). Yellow Wagtails are obligate insectivores, so might be expected to have suffered from long-term invertebrate declines (Campbell *et al.* 1997, Benton *et al.* 2002). Increasing use of pesticides may reduce food supplies and Henderson *et al.* (2008) found that the density of Yellow Wagtails was significantly higher in zero pesticide plots in a study in the UK.

11.6.6 Summary

Farmland drainage, the conversion of pasture to arable land, the change from spring to winter cereals, and the loss of insects associated with more intensive farming have been cited as possible causes of population decline in Yellow Wagtail.

11.6.7 Possible actions to aid recovery

In the face of widespread invertebrate declines, Yellow Wagtail may benefit from provision of permanent water features that provide a source of invertebrates with aquatic larval stages. Indeed, Yellow Wagtails have responded with increased breeding density to raised water levels in some grassland nature reserves. Other actions include delaying mowing dates on grassland to reduce loss of nests and young.

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