

Climate change and the potential impact on breeding waders in the UK

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Wader biologists have long been aware of the importance of climatic conditions for wader populations throughout the annual cycle. During the breeding season, climate change and associated changes in sea level may have potentially dramatic effects on wader populations both directly, through the loss of breeding habitat, and indirectly, through processes such as changes in prey populations, habitat quality, timing of migration and so on. The relative importance of each of these processes is as yet unclear. Here we review the variety of mechanisms through which climate change may influence breeding wader populations in the UK. Most UK breeding wader populations are currently declining and there is therefore an urgent need to assess the additional threats that climate change and sea level rise may pose in the future.

INTRODUCTION

There can be little doubt that climate is changing on a global scale. However, climate change predictions at a national and regional level are more uncertain (Newton 1998, Watkinson *et al. in press*). What we cannot be certain about is the severity of the predicted changes and the likely impacts of climate change on habitats and species. Current models predict that by 2080, average temperatures in the UK will rise by between 2 and 3.5°C (Hulme *et al.* 2002). High summer temperatures and heavy winter precipitation may become more frequent, summers may become drier and very cold winters may become rarer (Hulme *et al.* 2002). The effects of climate change will vary depending on geographical location; for example, southern and eastern counties of the UK may see the greatest increases in temperature, as much as 5°C in summer (Hulme *et al.* 2002).

Climate change impacts on sea levels as a result of increased temperatures causing the expansion of seawater and the melting of ice (Watkinson *et al. in press*). It is predicted that relative sea level will continue to rise in addition to an increase in the frequency of extreme sea levels (Hulme *et al.* 2002). The greatest impact of sea level rise may be evident in East Anglia in south-east England for two reasons: (i) the coastline and land mass are already low-lying (Sharpe & Huggett 1998, Norris & Buisson 1994) and (ii) sea level rise will be amplified by the continuation of downward isostatic land movements (Sharpe & Huggett 1998) and a rise in sea level of between 26 cm (best case scenario) and 86 cm (worst case scenario) by 2080 is predicted for this area (Hulme *et al.* 2002).

The effect of climate change on breeding wader populations in the UK is a multifaceted subject and will be dependent upon the complex interactions between a range of ecological, social and economic factors (Norris & Pain 2002). However, climatic conditions are known to affect survival and reproduction in birds (Newton *et al.* 1993, Hollands &

Yalden 1991). Climate change may have a direct influence on a number of ecological factors such as habitat area, land-use, vegetation structure, water availability, predation, prey abundance and parasites. These factors will indirectly affect habitat quality, which may influence the distribution and productivity of breeding waders. In some cases, it may be difficult to predict whether these effects will be positive or negative. It is likely that the impacts of climate change on the 19 species of wader which regularly breed in the UK could be dependent on three things: (i) breeding distribution, (ii) habitat requirements and (iii) breeding range. For example, redshank *Tringa totanus* are widely distributed and breed in a range of habitats. For this species, it is possible that a negative effect of sea level rise on coastal breeding redshank could be compensated for by a positive effect on wet grasslands such as a warmer climate leading to increased invertebrate abundance for chicks. In contrast, dunlin *Calidris alpina* are narrowly distributed in northern Scotland and are restricted to one habitat, peat bogs (Lavers & Haines-Young 1996, Lavers *et al.* 1996). If climate change were to have a detrimental effect on peat bogs there is no alternative breeding habitat and therefore no likely compensatory effect.

PHENOLOGY

There is an increasing body of evidence suggesting that climate change is having an impact on the timing of life cycles in birds (Crick & Sparks 1999, Winkler *et al.* 2002). This has a number of potential implications for avian ecology. Early arrival (in migrants) and trends towards earlier laying-dates may lead to a mis-match in timings between chick rearing and peaks in prey abundance (Crick & Sparks 1999, Winkler *et al.* 2002). Earlier egg laying dates may increase the length of the breeding season, resulting in some species either laying more clutches or having the option to re-lay following clutch loss (Green 1986). This may be especially advantageous to relatively late nesting species such as redshank and

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black-tailed godwits *Limosa limosa* (Kruk *et al.* 1996). Although the trend for early laying dates has not been quantified in wader species, the hatch dates of both redshank and black-tailed godwit vary annually and are associated with a measure of spring warming (Kruk *et al.* 1996). Analyses of long term nest record data in the UK found that 19 out of 36 species studied (mainly passerines) showed long term trends towards earlier laying in recent years (Crick & Sparks 1999). In addition, 17 of these species also showed significant weather effects, with warmer temperatures in March and April associated with earlier laying (Crick & Sparks 1999). The authors predict that by 2080, laying dates in the UK, for the species studied, may be on average 8 days earlier, with a maximum of 18 days advancement.

SHIFTING DISTRIBUTION PATTERNS

Populations of species occurring within the UK at the southern and western limits of their range may decrease or disappear completely as a breeding species, for example, dotterel *Charadrius morinellus*, whimbrel *Numenius phaeopus*, wood sandpiper *Tringa glareola* and greenshank *Tringa nebularia* (Cadbury 1995). Of these species, greenshank and dotterel are largely confined to northern Scotland; greenshanks mainly occur in peat bog habitat and dotterel in a montane environment (Gibbons *et al.* 1993). It is estimated that a 3°C rise in temperature would be sufficient to reduce the cover of peat bogs by 20% and montane habitat within the UK would become extinct (Briggs & Hossell 1995), therefore having a likely detrimental effect on populations of these species.

LOSS OR CHANGE IN QUALITY OF BREEDING HABITAT

Direct habitat loss resulting from sea level rise is likely to be confined to low-lying coastal areas such as saltmarshes, beaches and coastal grazing marsh (Watkinson *et al. in press*). In contrast, habitat losses resulting from changing climatic conditions (increased temperature and winter precipitation and decreased summer rainfall) are more likely to affect terrestrial areas such as wet grassland and montane habitats through changes in land-use management, vegetation characteristics and water balance (Briggs & Hossell 1995). It seems likely that where losses to one habitat occur through environmental change, increases in an alternative habitat will be gained. For example, coastal realignment may produce increases in the area of saltmarsh at the cost of a loss of coastal grazing marsh (Watkinson *et al. in press*). Changes in habitat quality, whether positive or negative, could be more widespread than habitat loss and could be applicable to all habitats. The consequence of habitat loss or changes in habitat quality depends on the quality of the habitat affected (Sutherland & Dolman 1994, Pettifor *et al.* 2000). Predicting these effects for breeding waders requires (i) a knowledge of the likely proportional changes in the population as a result of habitat deterioration or improvement and (ii) the density-dependent birth and death rates in the breeding and wintering grounds (Sutherland 1996, 1998). An excellent example of these processes is described by Liley's (1999) study of breeding ringed plovers *Charadrius hiaticula* in the East Anglian coastal zone. Breeding success in this population was strongly influenced by beach width (narrower beaches having higher levels of nest predation) and human

disturbance (through avoidance of heavily disturbed areas and trampling of nests). Beach structure in East Anglia is likely to be greatly influenced by future sea level rise and increases in summer temperatures may influence the number of tourists visiting these beaches (Norris & Buisson 1994). The effect of such changes on the ringed plover population was explored through the development of a population model that incorporated the density-dependent effects of changing beach structure and levels of tourism. This model predicted that habitat loss resulting from a rise in sea level of 50 cm, or deterioration of habitat quality arising from a 50% increase in human disturbance across the study site, would result in a decline of around 11% in the ringed plover population.

A major potential consequence of sea level rise is the large-scale erosion of coastal habitats, particularly saltmarsh (Norris & Buisson 1994). This can be exacerbated by coastal squeeze, the process by which the presence of hard sea defences around the coastline prevents landward migration of coastal habitats (Norris & Buisson 1994, Sharpe & Huggett 1998). Coastlines are highly dynamic systems and have undergone periods of erosion and accretion over geological timescales. The presence of hard sea defences clearly limits the dynamic capacity of these systems.

Saltmarshes are extremely valuable habitats, supporting nationally and internationally important breeding populations of waders. For example, redshank breeding on UK saltmarshes represent about 45% of the British population (Brindley *et al.* 1998) and 10% of the NW European population (Piersma 1986). English saltmarshes are by far the most important for redshank in the UK (86% of the total) owing to their greater area and higher redshank densities (Brindley *et al.* 1998). The East Anglian saltmarshes support about half of the saltmarsh breeding redshank population (Cadbury *et al.* 1987) and Norris & Buisson (1994) estimated that 27% of the UK population was at threat from sea level rise. Two surveys of redshank breeding density have been carried out, in 1986 and 1996 (Brindley *et al.* 1998). These provide detailed information on breeding densities on saltmarsh and show a decline of 22.9% in this habitat over the ten-year period. In addition, Cook *et al.* (1994) compared the 1986 data for Essex saltmarshes with a further survey in 1993 and reported a decline of 44%.

On the Wash saltmarshes, redshank breeding densities are related to grazing pressure, with high densities where grazing is light and low densities where it is heavy (Norris *et al.* 1997). It is therefore argued that changes in grazing pressure rather than habitat loss resulting from sea level rise has driven many of the declines reported in East Anglia (Norris *et al. in press*). However, the Essex sites are all un-grazed so the declines there may be the result of erosion of the high shore, saltmarsh breeding habitat, coupled with increased levels of nest inundation (Cook *et al.* 1994).

Efforts to mitigate the loss of saltmarshes have focussed on recreating the habitat through coastal realignment (see Atkinson this volume) but an additional mitigation mechanism may be appropriate management and creation of freshwater grazing marshes. However, in some areas decisions have to be made about whether to protect internationally important coastal wetlands or to maintain existing important intertidal habitats (Sharpe & Huggett 1998).

In addition to saltmarsh, redshank and other breeding waders (e.g. lapwing *Vanellus vanellus*, snipe *Gallinago gallinago*, pied avocet *Recurvirostra avocetta* and Eurasian



oystercatcher *Haematopus ostralegus*) also breed on coastal freshwater grazing marshes, a habitat that is also threatened by rising sea levels. This creates a situation in which (a) grazing marshes provide a means of mitigating for saltmarsh habitat loss if the management of grazing marshes can be optimised for breeding waders and (b) the trade-off between any increases in saltmarsh habitat at the expense of grazing marshes would need to be assessed (Sharpe & Huggett 1998). However, optimisation of the grazing marsh habitat and, in particular, the water regime, may become progressively more difficult to manage as a result of increasing summer temperatures and a predominance of drier springs and summers (Briggs & Hossell 1995). This may be further exacerbated by an increasing need for irrigation in adjacent intensive agricultural areas leading to increased water extraction and an overall lowering of the water table (Briggs & Hossell 1995).

Other habitats may be similarly affected; for example, peat bogs occur in wet, cold areas where acidic conditions facilitate the build up of peat (Briggs & Hossell 1995). Although an increase of winter rainfall is predicted for such areas, drier summers with raised temperatures may lead to greater evapotranspiration and a lowering of the peat water table. This could result in encroachment of birch and pine and a tendency for blanket bogs to be replaced by heathland (Briggs & Hossell 1995, Cadbury 1995). This might happen in the Flow Country of Scotland where breeding dunlins could be adversely affected by loss of peat-bog habitat. Dunlins are strongly associated with peat bogs and are totally absent from wooded areas (Lavers & Haines-Young 1996). Moreover, within peat bogs, dunlin density declines with increasing distance from pool systems (Lavers & Haines-Young 1996, Lavers *et al.* 1996). The quality of dunlin breeding habitat also increases with distance from the edge of conifer plantations (Lavers & Haines-Young 1997). There are thus a number of consequences of the potential drying out of peat bogs arising from increased temperatures. First, the number and distribution of important pool systems is likely to decrease through the lowered water table. Second, when bogs become drier the nutrient status increases (Briggs & Hossell 1995) and ground conditions may improve resulting in an increase in the suitability of the area for forestry plantation (Lavers & Haines-Young 1997). This may further reduce dunlin populations that are already breeding in a landscape fragmented by forestry plantations (Lavers & Haines-Young 1997).

LOSS OR CHANGE IN QUALITY OF WINTERING HABITAT

In their winter quarters, many wader species forage almost entirely in intertidal habitats. These are already under threat from human-related factors, such as land-claim and development, with an annual rate of intertidal habitat loss in the UK of 0.2–0.7% (Davidson *et al.* 1991). By 2014, it is estimated that 4% of intertidal flats may be lost (Norris & Buisson 1994). Intertidal habitat may also be lost through climate change, but this will only occur if sedimentation rates are too slow to compensate for sea level rise (Beukema 2002). Any loss or change to intertidal habitats could have consequences for overwinter survival (Atkinson *et al. in press*) and the condition of adult birds at the beginning of the breeding season. In geese, individual body condition (measured at the end of spring at staging sites) was found to be a

good predictor of the probability of breeding successfully (Spaans *et al.* 1993).

One possible advantage of climate change for wintering waders is that severe cold winters may become less frequent (Hulme *et al.* 2002). Cold weather may affect waders directly by mortality from hypothermia or indirectly by reducing prey availability or profitability (Newton 1998, Clark 2002). Species such as redshank and grey plover *Pluvialis squatarola* may be particularly vulnerable to severe cold weather, so warming may benefit some populations through increased winter survival in less severe weather (Clark 2002). The effect on intertidal invertebrates of increasing temperatures and less severe winter weather may be positive or negative and appears to be species dependent (Honkoop & Beukema 1997, Dekker & Beukema 1999, Atkinson *et al. in press*). In general, cold weather depresses the activity of intertidal invertebrates (Evans 1976, Esselink & Zwarts 1989). Therefore warmer winters may result in increased availability of prey for waders. Bivalve molluscs, which are common wader prey, such as cockles *Cerastoderma edule*, mussels *Mytilus edulis* and baltic tellin *Macoma balthica*, typically lose body mass throughout the winter. However, increased temperature increases the loss of body mass (Honkoop & Beukema 1997) and this may result in decreased profitability of intertidal prey in a warming climate. In contrast, some species of intertidal prey populations collapse in severe winters and gain benefits from increased survival and reproduction in warmer temperatures (Dekker & Beukema 1999).

PREY ABUNDANCE

Climate change may also have implications for prey abundance during the breeding season: (i) through the direct effect of changing weather patterns on prey seasonality and abundance and (ii) through the indirect effects of changes in habitat such as vegetation structure or hydrological changes. Geographical variation in climate change may have potential repercussions for migrant species (Stensteth & Myrsterud 2002). Migrants may have no way of forecasting weather on arrival at breeding grounds (Norris & Pain 2002) and this could lead to a mis-match in timing of migration relative to optimal timing of breeding and the seasonal availability of food (Evans 1997, Stensteth & Myrsterud 2002). For example, common sandpipers *Actitis hypoleucos* suffer higher annual mortality in years when snowstorms occur in the days following their arrival in the breeding grounds and this results in reduced breeding density in that year (Hollands & Yalden 1991).

Most wader species forage on invertebrate prey and changes in invertebrate abundance could operate at two levels: (a) effects on adult condition and (b) the implications for chick foraging and survival.

(a) Potential effects of changing prey abundance on adult birds

Spring temperatures may have an effect on soil fauna development (Kruk *et al.* 1996) and accessibility of soil fauna as prey for waders is influenced by water levels and soil saturation (Green & Cadbury 1994). These factors may influence the condition of the female, her egg laying ability and the timing of egg laying (Kruk *et al.* 1996). Most wader species lay a maximum of 4 eggs (Wallander & Andersson 2002) and these can account for a high proportion of female body



mass. For example, redshank may lay a total egg weight of 88 g in four days, which amounts to approximately 70% of female body weight. There are also advantages in being able to produce larger eggs that are more likely to hatch and fledging weight increases with egg size (Perrins 1996). In redshank, larger eggs produce larger young (Thompson *et al.* 1990) and Galbraith (1988a,b) showed that lapwing chicks hatching from larger eggs have better growth and survival rates.

If adults arriving on breeding grounds are in poor condition because of food availability on the wintering grounds, or if foraging conditions are poor on the breeding grounds, this can have a detrimental effect on the reproductive output (Newton 1998, Pettifor *et al.* 2000). Not only does food abundance affect the reproductive success of the adult (Newton 1998), but it can also influence the future reproductive success of the chicks (Haywood & Perrins 1992, Newton 1998). In great tits *Parus major*, environmental conditions affecting the food intake rate of chicks, such as territory quality and timing of food abundance, can influence the clutch size laid in subsequent years by those chicks (Haywood & Perrins 1992).

(b) Potential effects of changing prey abundance on chick foraging and survival

Perrins (1991) argues that chick food supply (in passerines) is the most important resource during the breeding cycle and that this is the selective factor to which timing of breeding is adjusted. However, in ground nesting species such as waders, timing of breeding may be adjusted in response to the risk of predation. Changes in phenology may lead to a mis-match of timings between chick hatching and chick food supply in seasonal environments (Visser *et al.* 1998). Even when the timing of reproduction in both predator and prey populations are synchronous, initiation of reproduction in invertebrate prey may take place at the correct time but the development time of invertebrates could be shorter, resulting in a food supply that does not last for the entire chick development period (Visser *et al.* 1998).

An example of the potential effect of climate on prey populations is the study of the arthropod prey of breeding red knot *Calidris canutus* (Tulp *et al.* 1998). The emergence of these is affected by daily temperature and there is a clear relationship between the growth rate of knot chicks and the abundance of arthropod prey. In addition, the prey are more active in warm weather so it is likely that they are easier to locate and catch when it is warmer (Tulp *et al.* 1998). The amount of time available for chick foraging is also an important factor. When temperatures are warmer, young chicks often spend less time being brooded, resulting in more time spent foraging (Beintema *et al.* 1991, Tulp *et al.* 1998). Habitat characteristics such as wetness and vegetation height can also affect the availability of prey. Tulp *et al.* (1998) found that wetter sites had a higher abundance of Diptera prey. Furthermore, the slope of the temporal decline of arthropods was lower in wetter sites resulting in a higher prey availability for wader chicks towards the end of the season when compared to dry sites. These differences were mirrored in the distribution of wader families, which was biased towards wet sites.

There is some evidence that wader chicks may change prey as they get older and it has been hypothesised that this may be because the relative profitability of different prey

types changes with chick age. For example, lapwing chicks increase the number of earthworms in the diet as they get older (Beintema *et al.* 1991). However, in wet grassland systems the availability of earthworm prey can decrease through the season as reduced rainfall results in a drop in the water table and the worms become out of reach to wader chicks (Green 1986). If climate change predictions for warmer, drier summers are correct then there may be some advantages for young chicks in terms of increased abundance and activity of invertebrates but older chicks may find it increasingly difficult to obtain profitable prey suitable for maintaining the growth rates necessary to reach fledging weight.

PREDATION AND PARASITES

Predation, both of clutches and chicks, is a key process in determining reproductive output in wader species (Green 1986, Martin 1993a, Grant *et al.* 1999, Valkhama *et al.* 1999, Johansson 2001) perhaps as a result of their ground nesting habits (Newton 1994). Climate change may have an influence on predation in a number of ways: (a) it may have a direct influence on the breeding productivity and survival of the predators resulting in increased abundance (Green 1986, Grant *et al.* 1999, Koivula & Rönkä 1998) and wider distribution, (b) it may influence the availability of alternative (non-wader) prey for predators (Green 1986), (c) changes in habitat structure may change the effectiveness of anti-predator strategies (Koivula & Rönkä 1998), (d) habitat generated changes may affect the species composition and abundance of the avifauna, including some species which may provide benefits in terms of advance warning of predators (Green 1986, Koivula & Rönkä 1998, Johansson 2001, Šálek & Šmilauer 2002) and (e) habitat fragmentation can lead to increased predation rates resulting from edge effects (Koivula & Rönkä 1998, Grant *et al.* 1999, Johansson 2001, Šálek & Šmilauer 2002).

If the direct effects of climate change on factors such as vegetation structure and prey availability act to reduce populations of breeding waders, then the reduced predation rates afforded by semi-colonial nesting (Green 1986, Koivula & Rönkä 1998, Johansson 2001, Šálek & Šmilauer 2002) may be lost. Furthermore, habitat loss and fragmentation increases the edge effect, which is caused by a decrease in habitat patch size leading to an increase in the relative amount of edge habitat. Edge effect has been shown to influence predation rates, with nests in wider sites (less edge) suffering less predation than those in narrow sites (more edge) (Koivula & Rönkä 1998, Johansson 2001).

In species that rely on nest concealment, climate related changes leading to more growth and reduced predation of clutches, particularly by avian predators, may increase nesting success (see Thyen & Exo, this volume). Species that may benefit in this way include Temminck's stint *Calidris temminckii*, which nests on coastal, open and short vegetated habitats. It has been suggested, however, that deterioration of the breeding habitat through overgrowth can lead to a reduction in the early detection of predators and early departure from nests (Koivula & Rönkä 1998).

Parasites can have implications for the population dynamics of birds (Rätti & Alatola 1993, Dobson & Hudson 1995, Newton 1998, Newborn & Foster 2002) and parasite infected individuals can be more susceptible to predation (Dobson & Hudson 1995). Although wader species do harbour parasites, the effects on host fitness and fecundity are poorly under-



stood (Van Oers *et al.* 2002) and the effects of changing climate on parasite populations have yet to be investigated. However, Tinsley (1999) argues that experimental evidence suggests that internal conditions within the host present a more severe environment than fluctuating environmental conditions during other stages of the life cycle.

CONCLUSIONS

There are many uncertainties surrounding the magnitude of global and local climate change scenarios, but we can be certain that climate change and the resulting rise in sea level is already happening. There are many studies that address the effects of ecological factors (prey abundance, predation, habitat loss and changes in habitat quality) on wader breeding success but most have only been examined over small spatial scales. Climate change and sea level rise will operate over much bigger scales, so there is an increasing need for wader studies that address these issues at larger and more appropriate scales than in the past. It may, for example, be necessary to adopt an integrated approach in which populations are monitored at an international or flyway scale covering a whole range of ecological impacts. Moreover, there are relatively few studies of the interactions between climate and those factors that are important determinants of breeding productivity. Examples include: (i) the interactions between climate, invertebrate prey and chick growth, (ii) the population response of predators to a warmer, drier climate and (iii) the population response of breeding waders to habitat loss and fragmentation.

In the UK, most breeding waders are already declining and a significant proportion of remaining populations are confined to nature reserves (Ausden & Hirons 2002). Climate change is presenting the conservation bodies with a further challenge. Hopefully they will be able to optimise the potential of existing sites and increase the area of habitat for breeding waders. Management of water resources and vegetation may become increasingly difficult and land managers may have to adopt new techniques to overcome these problems. These new techniques could be approached in an experimental way allowing ecologists the rare opportunity to study the effects of manipulated ecological factors in a changing climate. Ultimately this may improve our understanding of the impacts of climate change on breeding waders in a way that ensures their survival.

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