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The Effects of Climate Change on Migratory Waterbirds within the African-Eurasian Flyway

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EXECUTIVE SUMMARY

1. Climate change: past and future

1.1 It is now unequivocal that our climate is warming. Observations of increases in global average air and ocean temperatures, widespread melting of snow and ice and rising global average sea levels all point directly to a warmer planet. Temperatures are predicted to increase by anywhere between 0.9 and 3.8°C by 2100, with the highest warming occurring near the poles, particularly in the northern hemisphere.

1.2 Precipitation has, and is generally predicted to increase in the tropics and polar regions, but decrease at mid-latitudes, with the Mediterranean basin and South Africa experiencing considerably lower rainfall in the future.

1.3 During the 20th century sea-level rose at an average rate of about 1.7 mm yr^{-1}, but since 1990 this rate has increased and it has been rising at a rate of around 3 mm yr^{-1}. Estimates of future sea-level rise vary considerably, but if the Greenland and Antarctic ice sheets melt, sea-levels could rise by several metres over the course of the next century.

2. Climate change impacts

2.1 This report describes the past and likely future impacts of climate change on waterbirds within the African-Eurasian Flyway. Waterbirds are likely to be amongst the organisms most adversely affected by climate change as they are associated with a habitat that is very vulnerable to changes in rainfall, evaporation and human-demand and because they migrate between different areas and could thus be deleteriously affected in any one of those areas. Coastal waterbirds will be also be affected by sea-level rise.

2.2 Both the breeding and wintering ranges of waterbird species covered by AEWA are moving poleward or shifting upwards. Some coastal waterbird species in north-west Europe have shifted their wintering grounds by more than 100 km in the last 20 years. Poleward shifts in breeding distributions are less dramatic, but nevertheless well documented. Such shifts are expected to occur and accelerate in the future.

2.3 At present, reductions in abundance at the lower-latitude extremities of species' ranges are often matched or exceeded by increases at the poleward edge. However, several waterbird species breed on the poleward margins of continental landmasses and have nowhere to move to. As species continue to move poleward, increasing numbers of species will face this problem. Similarly, shifts in coastal waterbirds may force them to utilize areas with lower tidal amplitudes and consequently less area for feeding.

2.4 Warmer temperatures have resulted in many species advancing aspects of their life cycle. Earlier arrival from wintering grounds and earlier onsets of breeding are well documented across many species and from numerous locations. Changes in the departure date from breeding grounds are less consistent, as warming temperatures enable earlier completion of breeding, but also reduce the risk of mortality due to cold temperatures in late autumn and early winter.

2.5 As climate changes, the cues used to advance breeding and arrival from wintering grounds can act as evolutionary traps, whereby former reliable signals might no longer serve to maximise benefits to waterbirds. For example, organisms often advance their life-cycles at different rates and there are well documented instances where mismatch between the timing of hatching and the timing of prey availability has occurred. Similarly, earlier nesting resulting from warmer temperatures in early spring can expose birds to higher rainfall, leading to increased chick mortality.

2.6 Waterbirds are likely to become more susceptible to such evolutionary traps as temperature increases accelerate, unless they can evolve sufficiently quickly to accommodate changes in climate. At present the speed at which such evolutionary responses can occur is poorly documented.

2.7 The impacts of climate change on the demography of waterbirds are not well studied, although there are some documented cases of long-term changes in survival and productivity in non waterbird species. The impacts of weather on the survival and productivity of waterbird species are very well documented, and consequently it seems reasonable to expect that climate change will affect demography. Both survival and productivity are generally unfavourably affected by cold temperatures. Thus warming temperatures are likely to boost the populations of species over-wintering in cold areas. It should be noted however that increases in one species are likely to result in decreases in competing species, so for example, migrants are likely to face greater competition from residents during the breeding season.

2.8 In general, indirect impacts on survival and productivity are less clearly understood. Higher temperatures may result in adverse indirect effects such as increased evapotranspiration of wetlands. Likewise rainfall may have mixed effects. High rainfall could increase the mortality of some waterbirds, particularly those that do not have fully waterproof feathers, such as newly hatched young, but might also increase the number of wetlands. The impacts of range shifts and changes in phenology on survival and productivity has received little attention.

2.9 Changes in survival and productivity resulting from climate change, may be at least partially compensated for because most populations are density-dependent. Thus, the population-level impacts of climate change remain poorly understood.

2.10 Waterbirds that persist in areas subject to ‘coastal squeeze’, whereby the landward movement of habitats is prevented by flood defences are also likely to be particularly vulnerable. Documented instances of the effects of ‘coastal squeeze’ on waterbirds are limited to a few examples of species loss in northwest Europe.

2.11 Climate change is likely to interact with other factors, such as habitat loss and fragmentation to worsen the impact of that factor. Other factors interacting in this way include alien species invasion and human demand for water resources and land as areas near the equator become less suitable.

3. Climate change adaptations

3.1 The magnitude of future impacts of climate change on waterbirds is likely to be considerably affected by human habitat modification, offering considerable opportunities for adaptive management. Adaptive management options include: site management, establishing a coherent network of protected areas, management of the wider countryside and minimisation of other impacts,

3.2 Site management could entail manipulation of vegetation structure to ensure cooler micro-climates or ‘managed-realignment’ and controlled flooding of coastal wetlands. Other site management options include manipulation of hydrology, but in so doing, due consideration should be given to potential knock-on effects that occur downstream.

3.3 Ensuring a coherent network of sites could buffer populations against climate change by increasing the overall available area of habitat, but also by facilitating dispersal between sites. Although birds are highly mobile and could move easily, the prey on which they depend may have more difficulty moving unless sites are located sufficiently close together. Highest priority should be given to establishing protected areas at the beginning or end or arduous migration stages or in areas where there is a paucity of protected areas at present. To this end, the creation of more protected areas immediately north and south of the Sahara is crucial.

3.4 Although networks of protected areas provide one means of aiding species dispersal, another way is to manage the wider countryside more favourably. In developed countries this is likely to be best achieved by strategically integrating this objective with existing land- and water-use policies such as into agri-environment schemes, the water-frameworks directive or into flood management plans. In developing countries this is likely to be best achieved by seeking win-win situations with people, as local people tend to exercise more power than governments over the local use of wetland resources. Minimisation of other impacts, serves to buffer waterbird populations against climate change, and particularly in the marine environment may be the most feasible adaptation measure.

4. Climate change vulnerability

4.1 We constructed a climate vulnerability index to identify which species are most at risk from climate change. Species with small populations, small ranges, with highly fragmented ranges, associated with threatened or vulnerable habitats and with specialist requirements are considered most at risk from climate change.

4.2 Using this index, two of the species listed on Annex 2 of the AEWA Agreement are judged to be critically threatened by climate change, a further seven are considered highly threatened, a further 14 considered moderately threatened, a further 61 judged to experience some threat from climate change, with the remainder (150) considered only negligibly threatened. Slender-billed Curlew *Numenius tenuirostris* and White-winged Flufftail *Sarothrura ayresi* are judged to be critically threatened and Cape Gannet *Morus capensis*, Crowned Cormorant *Phalacrocorax coronatus*, Bank Cormorant *Phalacrocorax neglectus*, Slaty Egret *Egretta vinaceigula*, Northern Bald Ibis *Geronticus eremita*, Madagascar Pratincole *Glareola ocularis* and Damara Tern *Sterna balaenarum* are considered highly threatened.

4.3 Species' populations listed on Table 1 of the Agreement are considered to be more threatened than species listed on Annex 2. Thirteen are judged to be critically threatened and a further 18 are considered highly threatened. Overall, even the least-threatened population of a given species is more threatened by climate change than the species itself. Although this is an inevitable conclusion given the criteria used to assess threat, it serves as a reminder that focus on species as opposed to populations may underestimate the degree of threat facing waterbirds.

5. Future research

5.1 We recommend that future climate-related waterbird research and conservation action be directed towards these vulnerable species and populations. However, we also recommend that future research should concentrate on broad themes that test avian responses to climate change as a whole rather than the response of just one species. Highest amongst such research priorities is whether species can evolve sufficiently quickly to keep pace with climate change. Other important questions include whether adverse impacts are compensated for by reduced density-dependent mortality, and the extent to which cues used to trigger departure from wintering grounds are correlated with factors that advance the timing of breeding and lead to phenological disjunction.

5.2 Although all areas of the African-Eurasian flyway are likely to be adversely affected by climate change, it is sub-Saharan Africa and central and southwest Asia that face the greatest challenges. These parts of the flyway have a higher diversity of waterbird species and host many migratory waterbirds during the boreal winter when mortality has the highest impact on populations. They are also likely to face the most pressure from factors that conflict with waterbird conservation interests and are where funding for conservation action is least adequate. Thus future research and conservation action should be focused on these regions.

1. INTRODUCTION

It is now unequivocal that our climate is warming. Observations of increases in global average air and ocean temperatures, widespread melting of snow and ice and rising global average sea levels all point directly to a warmer planet (IPCC 2007b). There is overwhelming evidence that humans are contributing to global warming. Most of the observed increase in temperatures since the mid-20th century is very likely to be due to the observed increase in anthropogenic greenhouse gas concentrations (IPCC 2007b). Discernible human influences now extend to other aspects of climate, including ocean warming and acidification, continental-average temperatures, temperature extremes and wind patterns. Anthropogenic greenhouse gas emissions are also one of the main contributing factors to sea-level rise (IPCC 2007b). Climate change is considered to be the greatest environmental challenge facing the world today (Defra 2007; GEO 2007; King 2004).

Climate change is likely to affect all ecosystems, but wetlands are particularly vulnerable. Not only are they the world's most threatened ecosystem (Reid et al. 2005), but their sensitivity to water regimes makes them especially susceptible to changes in precipitation or evapo-transpiration. Wetlands are among the world's most productive environments. They host a very high biological diversity, providing the water and primary productivity upon which countless species of plants and animals depend for survival (Ramsar 2007). However, it is waterbirds that are the most widely used tool to identify, designate and justify the protection of important wetlands (Stroud et al. 2001). Their sensitivity to environmental change, the relative ease with which they can be counted and their tendency to congregate at key locations make them effective proxies for aspects of wider biodiversity (Stroud et al. 2001).

Many waterbirds migrate between different areas to capitalise on seasonally available resources. During their migrations, these waterbirds cross political boundaries between nations; boundaries that have no inherent meaning for the birds, but which have a dramatic influence on their annual life-cycles and their individual survival chances, due to the great differences that exist between countries in conservation policy. Migratory species are dependent on the specific sites that they find at the end of their journey and along the way. Increasingly these sites are threatened both by climate change and other human disturbances and causes of habitat degradation. Historically, the plight of waterbirds has been aggravated by the fact that it has long been held that migratory species legally do not fall within the jurisdiction of one particular country which could be held responsible for any harm occurring to them. However, in 1972 the United Nations Conference on the Human Environment, recognized the need for countries to co-operate in the conservation of animals that migrate across national boundaries or between areas of national jurisdiction and the high seas. This recommendation resulted in the Convention on the Conservation of Migratory Species of Wild Animals (CMS), which came into being in 1979. The goal of the CMS is to provide a framework for the conservation for migratory terrestrial, marine and avian species over the whole of their range. This is very important, because failure to conserve these species at any particular stage of their life cycle could adversely affect any conservation efforts elsewhere. The fundamental principle of the CMS is that contracting parties acknowledge the importance of migratory species being conserved and of Range States agreeing to take action to this end whenever possible and appropriate, paying special attention to migratory species, with unfavourable conservation status, and taking individually or in co-operation appropriate and necessary steps to conserve such species and their habitat (CMS 2007).

A specific daughter agreement of the CMS designed to conserve waterbirds within the African-Eurasian flyway is the African-Eurasian Migratory Waterbird Agreement (AEWA), which came into being in 1995. The AEWA covers 235 species of birds ecologically dependent on wetlands for at least part of their annual cycle, including many species of divers, grebes, pelicans, cormorants, herons, storks, ibises, spoonbills, flamingos, ducks, swans, geese, cranes, rails, waders, gulls, terns and even the south African penguin. The Agreement provides for coordinated and concerted action to be taken by the Range States throughout the migration system of waterbirds to which it applies (AEWA 2007).

The African-Eurasian flyway (see Figure 1) is one of at least seven major flyways and geographically, covers 118 and the European Community (EC) from parts of Europe, Asia, Canada, the Middle East and Africa. The geographical area covered by the AEWA stretches from the northern reaches of Canada and the Russian Federation to the southernmost tip of Africa.

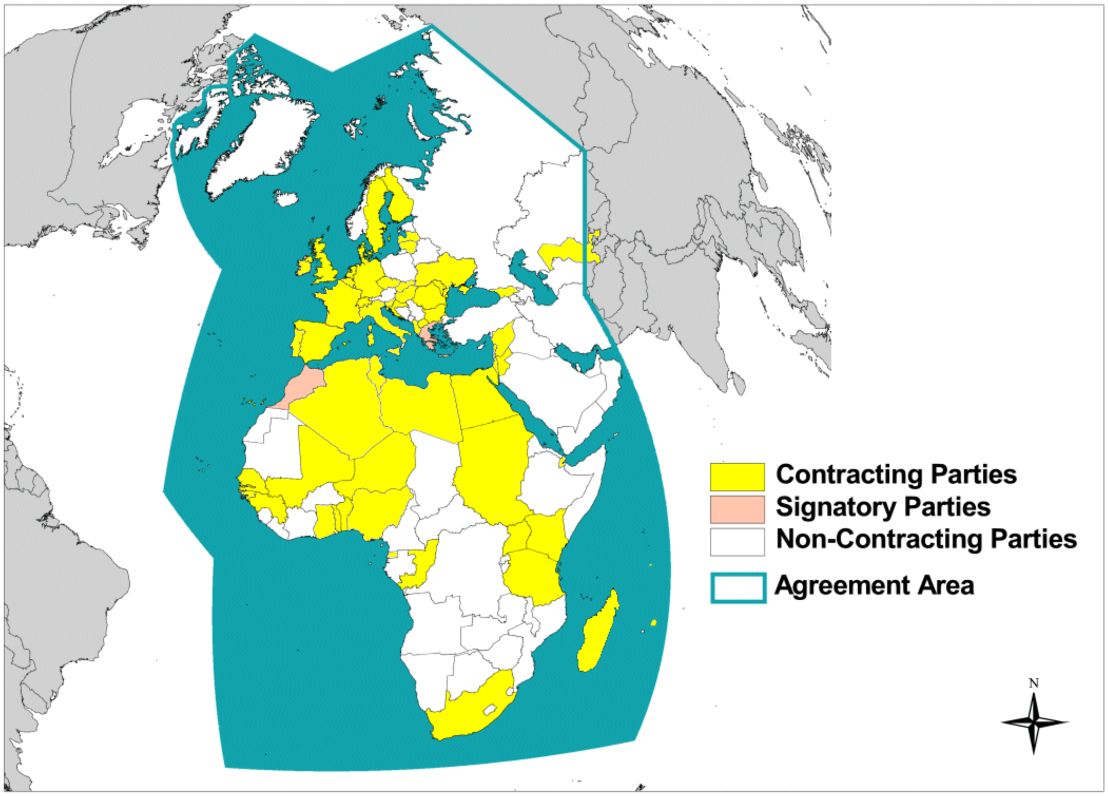


Figure 1. Area covered by the African-Eurasian Waterfowl Agreement. Source: (AEWA 2007)

2. CLIMATE CHANGE WITHIN THE AEWA AGREEMENT AREA

2.1. Temperature change

2.1.1. Global

Global mean temperatures have risen by 0.74ºC ± 0.18ºC when estimated by a linear trend over the last 100 years (1906-2005). The rate of warming over the last 50 years is almost double that over the last 100 years (IPCC 2007b). Following a period in the late-1800s, when temperatures were about 1.3ºC colder than today, temperatures rose to 0.8ºC colder than present in the late 1930s, prior to a slight cooling of 0.2ºC, which lasted up until the late-1970s, followed by rapid warming of about 1ºC during the last 25 years (Brohan et al. 2006; Hansen et al. 2001). Temperatures at present are warmer than at any point during the last thousand years. Eleven of the last twelve years (1995–2006) rank among the 12 warmest years in the instrumental record of global surface temperature, which began in 1850 (IPCC 2007b). Palaeoclimatic information suggests that current temperatures are unusual in at least the previous 1,300 years. Average Northern Hemisphere temperatures during the second half of the 20th century are very likely to be higher than during any other 50-year period in the last 500 years and likely to be the highest in at least the past 1,300 years. The last time the polar regions were significantly warmer than at present for an extended period was about 125,000 years ago (IPCC 2007b). Seasonally, warming is generally slightly higher in winter (IPCC 2007b). Globally, the number of frost-days has decreased by about 4 days since 1880 and the growing season has increased by 2 days. Seasonal global temperature changes between 1979 and 2005 are shown in Figure 2.

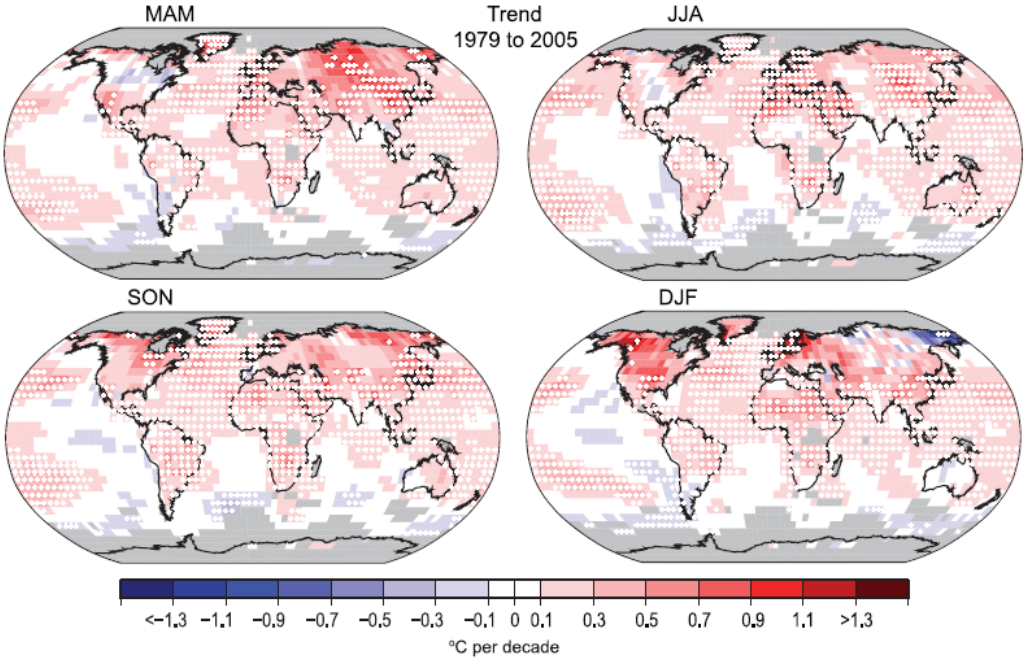


Figure 2. Linear trend of seasonal MAM, JJA, SON and DJF temperature for 1979 to 2005 (°C per decade). Areas in grey have insufficient data to produce reliable trends. Source: IPCC (2007b)

For the next two decades, a warming of about 0.2°C per decade is projected (IPCC 2007b). Even if the concentrations of all greenhouse gases and aerosols were to be reduced to their concentration at year 2000 levels, a further warming of about 0.1°C per decade would be expected. Continued greenhouse gas emissions at or above current rates will cause further warming and induce many changes in the global climate system during the 21st century that are very likely to be larger than those observed during the 20th century (IPCC 2007b). Predicted temperature increases vary according to scenarios of greenhouse gas emissions and there is also uncertainty associated with each projection such that overall, temperatures are predicted to increase by anywhere between 1.1 and 6.4°C by 2100 (IPCC 2007b). It is most plausible that temperature increases will be somewhere in the region of 2-3°C. Seasonal predictions of global temperature changes are shown in Figure 3.

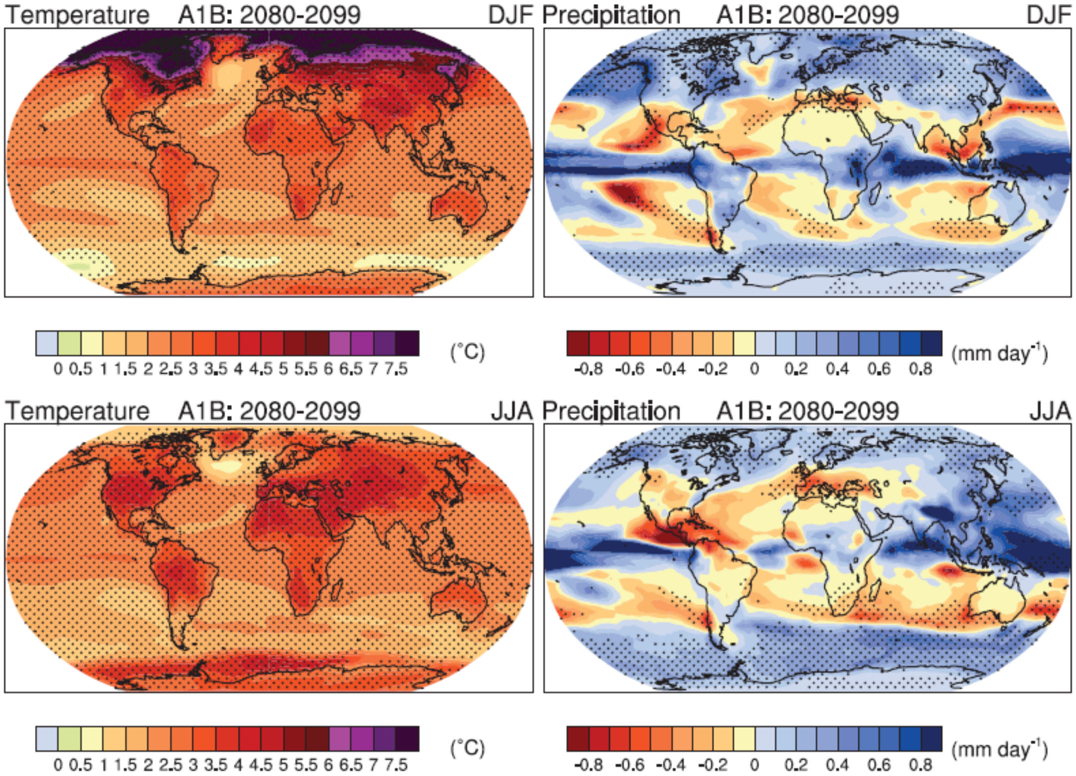


Figure 3. Multi-model mean changes in surface air temperature (°C, left) and precipitation (mm day–1, right) for boreal winter (DJF, top) and summer (JJA, bottom). Changes are given for the IPCC (2007b) SRES A1B scenario, for the period 2080 to 2099 relative to 1980 to 1999. Stippling denotes areas where the magnitude of the multi-model ensemble mean exceeds the inter-model standard deviation. Source: IPCC (2007b).

2.1.2. Africa

Mean temperature increase has been variable across Africa. The greatest warming has occurred in western regions, particularly just north and just south of the Sahara, where temperatures have increased by about 1ºC (linear trend estimate 1901-2005). In eastern Africa, particularly inland, temperature increases during the 20th century were moderate (c. 0.2ºC) and in some areas did not increase at all (Smith & Reynolds 2005). Differences in the degree of warming between seasons are relatively small. There is some evidence of more widespread warming in northern Africa in December, January and February and of less warming within the Sahara in June, July and August between 1979 and 2005 (Smith & Reynolds 2005).

It is very likely that all of Africa will be warmer this century than last and that the annual average warming throughout the continent will be higher than the global average (IPCC 2007b). The drier subtropical regions are expected to warm more than the moister tropics. By 2030, the western Sahara is expected to have increased in temperature by 1-1.5°C and the rest of Africa by 0.5-1°C. By 2065, temperatures in the Sahara and inland southern Africa are expected to have increased by 1.5-3°C and in the tropics and coastal northern Africa, by 1.5-2°C. By 2100, parts of the Sahara could be up to 5.5°C warmer and a warming of about 3.5°C is expected for all regions except the tropics, which are predicted to increase in temperature by about 2-3°C. In general, temperatures are likely to increase most in the boreal summer such that on average they will be about 1°C greater than the annual average (IPCC 2007b).

2.1.3. Europe and the eastern Nearctic

Mean temperature increase has been variable across Europe. The greatest warming has occurred in southern regions, particularly in the Alps north-east of Italy, where temperatures increased by 2ºC between 1901 and 2005. In northern and western Europe, temperature increases have been moderate over this period (c. 0.5ºC) and in the south-east of Greenland, temperatures have actually undergone a moderate decrease of up to 0.8ºC over this period (Smith & Reynolds 2005). There are some notable differences in the degree of warming between seasons between 1979 and 1997. Boreal winter temperatures in Fenno-Scandia have increased by more than 3ºC, whereas northern Spain has experienced slight cooling (Smith & Reynolds 2005). Boreal summer, temperature increases have been moderate in Fenno-Scandia (c. 0.5ºC) and highest in Central Europe where some regions have experienced a temperature increases in excess of 2ºC (Smith & Reynolds 2005).

Annual mean temperatures in Europe and the north-western Nearctic are predicted to increase by more than the global mean (IPCC 2007b). Predictions vary, but temperatures are typically expected to increase by between 2 and 5ºC this century. The warming in northern Europe and the western Nearctic is likely to be highest in the boreal winter and that in the Mediterranean area largest in the boreal summer. The lowest winter temperatures are likely to increase more than average winter temperature in northern Europe and the western Nearctic, and the highest summer temperatures are likely to increase more than average summer temperature in southern and central Europe (IPCC 2007b). Throughout north-eastern Europe and inland Greenland, winter temperatures are predicted to increase by almost 10ºC this century and around Svalbard, by more than 10ºC. In the Mediterranean region, temperature predictions are more moderate and are expected to be in the region of 2.5ºC. In summer, Mediterranean temperatures are predicted to increase by 5ºC, but northern European and western Nearctic temperatures by only 2-3ºC (IPCC 2007b).

2.1.4. Middle East and Asia Minor

Mean temperature increase has been variable across the Middle East and Asia Minor. The greatest warming has occurred in Iran and immediately to the east of the Caspian Sea where between 1901 and 2005, mean temperatures increased by more than 2ºC. Across the Arabian peninsula there was a gradient of warming with moderate increases in the east of about 1.3ºC, but negligible to very slight increases in the northwest (Smith & Reynolds 2005). There were few broad-scale seasonal differences, but some small-scale ones. Between 1979 and 2005, the area to the east of the Caspian Sea has experienced the greatest warming in the boreal winter, whereas the area to the south and across the north of the Arabian peninsula has experienced greater temperature increases in the boreal summer (Smith & Reynolds 2005).

Annual mean temperatures in the Middle East and Asia Minor are predicted to increase by slightly more than the global mean (IPCC 2007b). In general temperature increases are predicted to be greater in the boreal summer than in the boreal winter and greater inland than in coastal areas. In Asia Minor and the north of the Arabian peninsula, boreal summer temperatures are predicted to increase by between 5 and 7ºC this century. Temperatures in the south of the Arabian peninsula and in coastal areas are expected to decrease by less, typically by 2-4ºC. In Asia Minor and throughout most of the Arabian Peninsula and eastward, boreal winter temperatures are predicted to increase by between 2 and 3ºC this century. The southern Arabian peninsula inland, and eastern Iran are predicted to increase in temperature by 3-4°C.

2.2. Rainfall change and drought

2.2.1 Global

Globally, there was no significant trend in precipitation during the 20th century, but over land it has generally decreased. Between 1900 and 2005, precipitation has generally increased over land north of 30°N, but downward trends dominate the tropics since the 1970s. From 10°N to 30°N, precipitation increased markedly from 1900 to the 1950s, but declined after about 1970. Downward trends were recorded in the deep tropics from 10°N to 10°S, especially since the late-1970s (IPCC 2007b). It has become significantly wetter in northern Europe, but drier in the Sahel, the Mediterranean and southern Africa (IPCC 2007b). Overall, precipitation appears to be becoming more variable. There has been a substantial increases in the number of heavy precipitation events and there is some evidence of extreme (1 in 50 year return period) events increasing. However, droughts have also become more common, especially at lower latitudes, since the 1970s. Observed marked increases in drought in the past three decades arose from more intense and longer droughts over wider areas, as a critical threshold for delineating drought is exceeded over increasingly widespread areas. Decreased land precipitation and increased temperatures that enhance evapotranspiration and drying are important factors that have contributed to more regions experiencing droughts, as measured by the Palmer Drought Severity Index (Dai et al. 2004; IPCC 2007b; Palmer 1968).

Precipitation is generally predicted to increase in the tropics and polar regions, but decrease at mid-latitudes (IPCC 2007b). In East and Central Africa, Greenland, Northern Fenno-Scandia and Siberia for example, precipitation is expected to increase by 100-170 mm per year (2080-2099 relative to 1980-1999), whereas in the Mediterranean Basin it is predicted to decrease by 100-150 mm per year. The intensity of precipitation events is expected to increase, particularly at high-latitudes and in the tropics in areas where rainfall is predicted to be higher. However, even in areas predicted to have less rainfall, intensity is predicted to increase, but there would be longer periods between rainfall events. It is predicted that the centre of continents will be drier, leading to a greater risk of droughts in those regions. Seasonal predictions of global temperature precipitation changes are shown in Figure 3.

2.2.2. Africa

There are considerable temporal and regional differences in the extent to which rainfall has changed in Africa. In the Sahel region and other areas just south of the Sahara and in Egypt there was a catastrophic decrease in rainfall between 1901 and 2005, in some places close to 100%, but slight increases of about 15% in the upper Nile and Congo basins in central-east Africa (IPCC 2007b). By contrast, between 1979 and 2005 there were moderate increases in rainfall in the Sahel region of about 15% and at only a few sporadic locations, predominantly in southern Africa did rainfall decrease. Globally between 1901 and 2002 Africa has experienced the greatest increase in the severity of droughts. Decreases in the Palmer Drought Severity Index over this period, are at their maximum value of less than –4, for most of the Sahel region and other areas immediately south of the Sahara and also in many parts of southern Africa. However, the Ethiopian Highlands and a small region to the south of Lake Victoria have experienced less severe droughts, with in an increase in Index values of about 3 (Dai et al. 2004; IPCC 2007b).

Annual rainfall is predicted to decrease by 2100 (relative to the latter part of the 20th century) in much of Mediterranean Africa and the northern Sahara, with the likelihood of a decrease in rainfall increasing as the Mediterranean coast is approached. Overall, rainfall is expected to decrease by 20-50%. Drying will be most severe, but less widespread in the boreal summer, with the Mediterranean coast experiencing less than half the present annual rainfall, but the south-eastern Sahara actually experiencing an increase in rainfall. In southern Africa rainfall is likely to decrease most dramatically during June, July and August, with less than 50% of the present rainfall throughout most of inland southern Africa, but only very minor decreases in the austral summer. The tropics, particularly in east-central Africa are predicted to experience much higher rainfall (>30%) in the boreal winter, and moderate increases (5%) during the boreal summer. It is uncertain how rainfall in the Sahel, the Guinean Coast and the southern Sahara will evolve this century (Dai et al. 2004).

2.2.3. Europe and the north-eastern Nearctic

There are moderate temporal and regional differences in the extent to which rainfall has changed in Europe and the north-eastern Nearctic. Between 1901 and 2005 central Europe and Fenno-Scandia has experienced a moderate (20-30%) increase in rainfall, central and southern Europe has experienced little change and the Mediterranean basin has experienced moderate declines of c. 10%. Since 1979, the pattern has changed somewhat. Only Fenno-Scandia and the area around the Faeroe Islands have experienced higher rainfall (c. 10%), whereas central and south-eastern Europe have experienced decreases (IPCC 2007b). The Palmer Drought Severity Index reveals a substantial increase in the severity of droughts throughout central and southern Europe, particularly in south-eastern Europe, but a substantial reduction in severity in northern Fenno-Scandia and European Russia (Dai et al. 2004; IPCC 2007b).

Annual rainfall is very likely to increase in most of northern Europe and decrease in most of the Mediterranean area. In northern Fenno-Scandia, mean increases of 15-20% are expected by 2100, whereas in southern Spain and Greece, 15-30% decreases are expected. In general, increases will be more pronounced in the boreal winter, whereas decreases will be more pronounced in the boreal summer. Thus in central Europe, precipitation is likely to increase by 5-10% in winter but decrease by 5-10% in summer. In north-eastern Fenno-Scandia, winter increases will exceed 30%, whereas throughout the Mediterranean, rainfall in summer will decrease by more than 30%. Extremes of daily precipitation are very likely to increase in northern Europe. The annual number of precipitation days is likely to decrease in the Mediterranean area, as is the severity of drought.

2.2.4. Middle East and Asia Minor

There are some regional differences in the extent to which rainfall has changed in the Middle East and Asia Minor. Between 1901 and 2005, there was a c. 50% reduction in rainfall in the area around Israel and Jordan and minor reductions (<20%) towards the west of Asia Minor. The Caspian Sea region experienced moderate increases, but there were no significant changes on the Arabian peninsula (IPCC 2007b). Between 1979 and 2005 there were moderate decreases (15%) around Israel and Jordan, moderate increases (c. 5%) in the Crimea, but no significant changes elsewhere (IPCC 2007b). Throughout most of the region there has been an increase in the severity of droughts, but particularly in Iran. Only the Crimea has experienced a very slight reduction in the severity of droughts (Dai et al. 2004; IPCC 2007b).

Asia Minor is predicted to experience a significant reduction in rainfall, with a slightly greater decrease in the boreal summer. In June, July and August, a 20-30% reduction in rainfall can be expected, whereas in the boreal winter this is more likely to be about 15-25%. The northern part of the Middle East, particularly around the south of the Caspian Sea is likely to experience much drier conditions (30-50% reduction in rainfall) in the boreal summer, but no major change in the boreal winter. Southern parts of the Arabian peninsula are predicted to be generally wetter throughout the year, with a 5-20% increase in precipitation (IPCC 2007b).

2.3. Global changes in wind patterns

The circumpolar vortex of wind systems in the northern hemisphere significantly expanded between 1949 and 1970 (IPCC 2007b), but appears to have been contracting since then, consistent with a warming of the north pole (IPCC 2007b). In the southern hemisphere high latitudes, the largest changes have been the enhancement of circumpolar westerlies in the austral summer, again consistent with a warming trend. In both hemispheres, westerly winds appear to have strengthened at mid-latitudes, especially in the boreal winter. A number of recent studies suggest that cyclone activity over both hemispheres has changed over the second half of the 20th century. General features of this change include a poleward shift in storm track locations, increased storm intensity, but a decrease in total storm numbers (IPCC 2007b). There is also some evidence that tropical storms have increased in intensity and frequency, but such patterns are difficult to discern due to inter-decadal variability (IPCC 2007b).

Most recent published studies suggest an overall decrease in the number of tropical storms, but predictions suggest that increases in the peak intensity of future tropical cyclones are likely (IPCC 2007b). Models suggest fewer mid-latitude storms consistent with a poleward shift in their tracking locations, particularly notable in the southern hemisphere.

2.4. Global large-scale climate circulation patterns

It is difficult to discern consistent trends in large-scale circulation patterns such as the El Niño Southern Oscillation (ENSO) and North Atlantic Oscillation (NAO) as decadal variations considerably complicate the interpretation of climate change. The NAO has exhibited a trend towards its positive phase (strengthened mid-latitude westerlies) in the last four decades, although it returned to near its long-term mean state between 2000 and 2005 (IPCC 2007b). ENSO has exhibited considerable inter-decadal variability in the past century, but systematic changes in ENSO activity have also been observed, in particularly different evolution of ENSO events and enhanced ENSO activity since the late-1970s. Within the AEWA region, interpretation of these effects is complicated by interactions with Atlantic and Indian Ocean associated climate shifts, but overall it appears to have lead to warmer and wetter weather in Greenland and Iceland, anomalous easterlies along the equator in Africa and increasingly unusual weather in East Africa (IPCC 2007b).

ENSO inter-annual variability is predicted to continue no matter what changes in average background conditions occur. There is no consistent indication at this time of discernable changes in projected ENSO amplitude or frequency in the 21st century. By contrast there is a projected positive trend in the NAO, although the magnitude of this shift differs depending on the models used (IPCC 2007b).

2.5. Sea-level rise

Global sea level rose by about 120 m during the several millennia that followed the end of the last major ice age approximately 21,000 years ago, and stabilised between 3,000 and 2,000 years ago. After this it did not change significantly up until the late 19th century. During the 20th century it rose at an average rate of about 1.7 mm yr^{-1}, but since 1990 this rate has increased and it has been rising at a rate of around 3 mm yr^{-1} (IPCC 2007b). Sea-level rise has not been uniform throughout the globe. In the Baltic for example, mean sea-level has been rising by c. 4mm per-year between 1950 and 2003, whereas off the coast of East Africa and southeast of Greenland, sea-levels actually decreased (IPCC 2007b). Historic patterns of changes in extreme sea-level are masked by spatial variability and the few numbers of locations where such data are available. However, analysis of hourly sea levels from 141 stations around the world suggest that there is an increase in extreme (highest 1%) sea-levels since 1975, but this is broadly consistent with patterns of increase in mean sea-level.

Although overall, sea-level did not change between 3,000 and 2,000 years ago and the late 19th century, some land masses are still recovering from the effects of the ice ages and are undergoing isostatic rebound, whereby some areas, notably in northern Europe, Siberia and Canada, that were depressed by huge weights of ice are still rising. At some of these locations, notably in Great Britain, post-glacial rebound of Northern Britain is causing a subsidence of the southern half of the island.

Sea-level is projected to rise during the 21st century by between 18 and 59 cm. The average rate of rise during the 21st century is very likely to exceed the 1961 to 2003 average, with a central estimate of 4 mm per year by 2100 (IPCC 2007b). Thermal expansion is the largest component, contributing 70 to 75% of the central estimate in these projections, but glaciers, ice caps and the Greenland Ice sheet are also projected to contribute to sea level. However, it is very important to note that these estimates of future changes do not incorporate uncertainty in carbon cycle feedbacks nor do they include the full effects of changes in ice sheet flow (IPCC 2007b). Gregory et al. (2004), suggest that temperatures by 2100 will be sufficient to melt the Greenland ice sheet causing an increase in sea-level of 7 m spread over the next 1,000 years. Similarly, Overpeck et al. (2006) suggest that the rate of melting of the Antarctic ice sheet is faster that previously supposed. Thomas et al. (2004) suggest that glacier thinning rates near the coast of Antarctica during 2002–2003 were much larger than those observed during the 1990s and flow into floating ice shelves over bedrock up to hundreds of metres deeper than previous estimates. As such, the risk of sea-level rise should the Antarctic ice sheet collapse is greater than previously thought. It is also noteworthy that in areas of increased wind speed, such as at mid-latitudes, more extreme wave heights will result.

3. CURRENT EFFECT OF CLIMATE CHANGE ON WATERBIRDS

Migrant waterbirds are particularly vulnerable to climate change because they rely on a network of (often globally) dispersed sites and must travel between them. They are often strongly constrained by time, for example, having to time arrival to match periods of resource abundance. Their main habitat: wetlands, is the most threatened of all ecosystems (Reid et al. 2005) and is one of the habitats, along with Arctic tundra on which many waterbirds breed, likely to be most adversely affected by climate change (IPCC 2007a). Given the complexity of migratory systems, direct experimental evidence of impacts of climate change independent of other factors is almost impossible to obtain. Consequently, most of what is known has been inferred from other lines of evidence. Here we review published literature to determine the impacts for climate change on species listed on Annex 2 of the AEWA Agreement. We start by considering the effects of climate change on range and distribution. We then look at how the timing of life-cycle events has been affected (phenology). We then consider impacts on demography, primarily survival and productivity. We also consider the special implications that changing rainfall will have on waterbirds. Lastly, we consider how climate change can interact with other factors to result in indirect effects.

We make use of all literature that was available to us through standard literature search engines such as Web of Knowledge and Google Scholar. However, it should be borne in mind that it is very difficult to establish whether lack of information regarding climate change impacts on a particular species is because no such impact is present or simply because no research has been conducted on that particular species. The literature is always likely to be biased towards demonstrating effects rather than lack of effects. Studies that cannot demonstrate an effect are much less likely to be published, simply because it is difficult to establish whether failure to detect an effect is due to lack of effect or due to inadequacy in the study such that although an effect was there, it could not be detected. Moreover, due to the greater availability of financial resources for research in rich developed countries, there is a considerable bias towards studies conducted in north-west Europe. For this reason, although we focus on reporting instances where climate change has been shown to affect a species specifically listed on Annex 2 of the Agreement, if particular climate change impacts are well documented for a number of species, we report on these too, even if none are listed on Annex 2.

3.1. Changes in range and distribution

It is well known that temperature is one of the primary determinants of the location of biomes and habitats (Huntley 1991). It is therefore logical that species ranges also depend on temperature (Parmesan 1996; Root 1988a; Root 1988b). Rainfall patterns can also affect species ranges (Newton 1998), and this has led to some long-standing cultural associations with waterbirds. For example, White Storks *Ciconia ciconia* are known as rain birds in dry areas of Africa (Robinson et al. 2005; Sitters et al. 2004) and in parts of East Africa, the call of a Black Cuckoo *Cuculus clamosus* is seen as a harbinger of rain. Both breeding and wintering ranges of waterbirds have been affected by climate change in the latter half of the 20th century (Austin & Rehfisch 2005; Thomas & Lennon 1999).

3.1.1. Changes in breeding range

In the UK, Thomas & Lennon (1999) showed that between the periods of the two BTO Breeding Bird Atlases (1968-1972) and (1989-1991), bird species (including some waterbird species) extended their breeding ranges northwards by an average of 18.9 km, a trend that they attributed to warming temperatures. Such temperature increases are probably the most likely explanation for the colonisation of the UK by several new breeding bird species from the continent, including waterbird species such as Little Egret *Egretta garzetta* (see Figure 4a) (Musgrove 2002; Robinson et al. 2005). Other species such as Eurasian Spoonbill *Platalea leucorodia* and Cattle Egret *Bubulcus ibis* seem set to follow (Figure 4b). Range shifts can also be altitudinal. In Poland, White Storks are shifting their breeding range to higher altitudes, with both the maximum and upper-quartile altitudes of nests increasing significantly (Tryjanowski et al. 2005). There are concerns that upward altitudinal shifts in montane bird communities might occur in the UK (Watt et al. 1998). Predictive models, such as MONARCH (Harrison et al. 2001), suggest that high altitude species in Britain, will reduce in population size under current climate change scenarios. Indeed climate change may already have been responsible for the extinction of breeding Eurasian Dotterel *Eudromias morinellus* in England and Wales (Harrison et al. 2001). Overall, range shifts, although not widely documented in species listed in Annex 2 of the Agreement, are quite likely to be occurring for many waterbird species to a greater or lesser degree.

3.1.2. Changes in wintering range

In the UK, the distributions of eight out of nine common species of wader over-wintering on estuaries have shifted eastward with recent climate change (Figure 5), with smaller species showing the greatest temperature dependence. This is thought to be linked to the trade-off between better feeding and risk of temperature-induced mortality. In the UK the east coast is generally colder than the west, but because estuaries are muddier in the east, they have more productive feeding conditions. With global climatic change it appears that the advantage gained by waders wintering in the milder west to avoid cold weather-induced mortality is diminished (Austin & Rehfisch 2005). This pattern is similar to that which is occurring across Europe. Maclean et al. (in review) investigated shifts in the distribution of over-wintering waders throughout north-west Europe. All seven of the species studied are changing their abundance in response to temperature change and some underwent dramatic shifts in their range between 1980 and 2000, in excess of 150 km. Open coast species in the UK also appear to be moving northwards as winters become milder, with a general northward shift in the distribution of waders between the winters of 1984/85 and 1997/98 (Rehfisch et al. 2004). There is also some evidence that shifts in the over-wintering distribution of ducks are occurring, with an increasing proportion now using formally frozen lakes in Eastern and Central Europe (Maclean et al. 2006b). These findings have important implications for conservation. For example, the average number of Dunlin over-wintering on the Severn Estuary SPA is near to dropping below the 1% threshold used for designation (Figure 6).

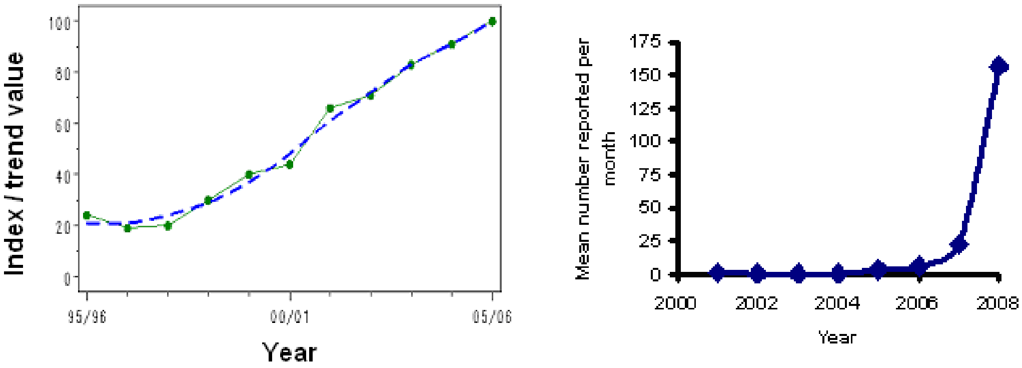


Figure 4. Wetland Bird Survey index (solid) and smoothed-trend (dashed) values of Little Egret *Egretta garzetta* over-wintering in Britain (left). Mean number of Cattle Egrets *Bubulcus ibis* reported in the Britain and Ireland every month (right). Source: Rare Bird Alert. Note: data from 2008 are 1 Jan – 19 Feb inclusive, but are considered to span two-months.

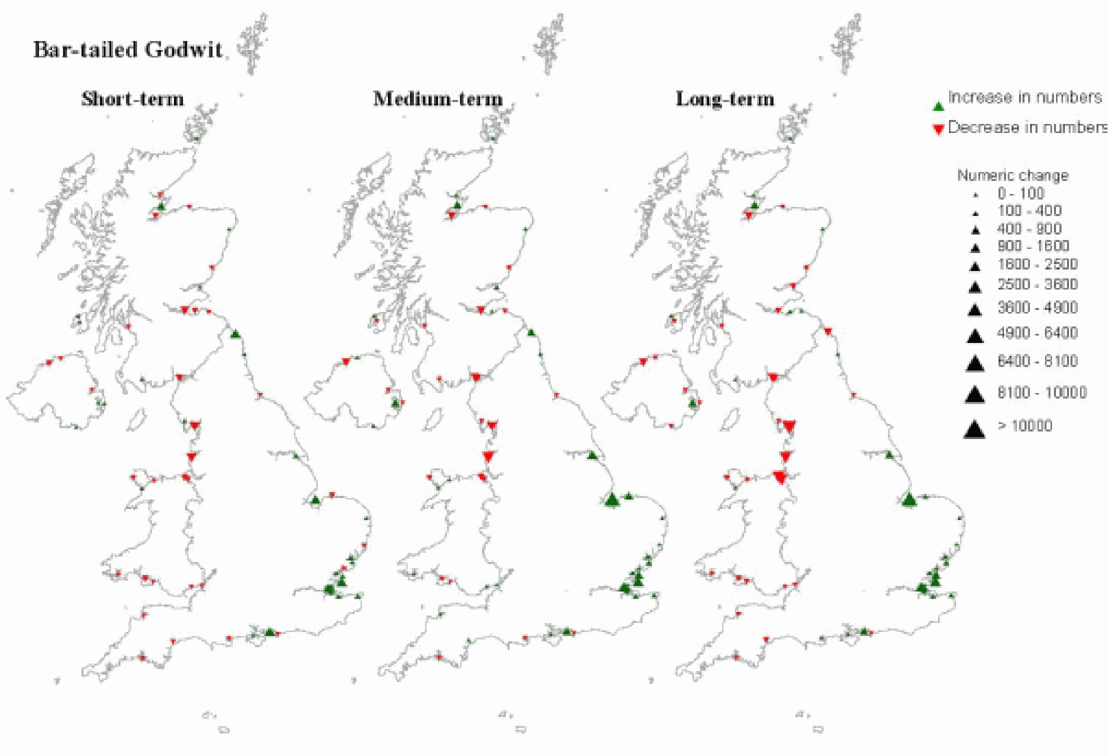


Figure 5. Distribution change in Bar-tailed Godwit *Limosa lapponica* between the winter of 1998/99 and 2003/04 (short-term), 1993/94 and 2003/04 (medium-term) and 1978/99 (long-term). The general shift eastward is evident. Source: (Maclean & Austin 2006).

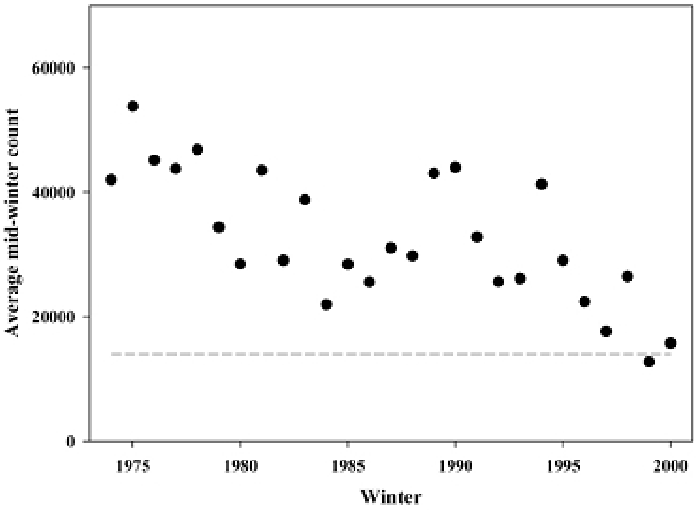


Figure 6. Mean number of Dunlin over-wintering on the Severn Special Protection Area (SPA). The dotted line represents 1% of the international population, the threshold used to designate this site as an SPA for this species. Waders on a number of estuaries in the west of the UK have declined such that they have or will soon drop below the threshold used for designation. Source: Austin & Rehfisch (2005).

3.1.3. Changes in migratory routes

Shifts in wintering range are partly a consequence of (or at least reflect) reduced migratory behaviour. A number of wader species may be travelling shorter distances between summer and winter than previously (Rehfisch et al. 2004), and open-coast species in the UK have shifted their winter distribution towards their breeding ground (Rehfisch et al. 2004). Similarly, Barnacle Geese *Branta leucopsis* may be short-stopping, using more northerly located islands in Norway as stop-over sites on spring migration due to warmer temperatures and enhanced grass-growth (Prop et al. 1998).

Although to date, most evidence is from non-waterbird species, one trend that seems set to continue, is that which has been recently been observed for Barn Swallows *Hirundo rustica* and House Martins *Delichon urbica*: an increasing proportion of the populations of these Palaearctic breeding species over-wintering north of the Sahara rather than risking the perilous journey across this ever expanding arid zone (Baillie & Peach 1992; Peach et al. 1991; Szep 1995; Winstanley et al. 1974),. Almost all Palaearctic - Afrotropical migrant waterbirds listed on Annex 2 of the Agreement have small over-wintering populations north of the Sahara (see Table 1). Others have a more sizable over-wintering population in Europe, but still have their main wintering quarters south of the Sahara. There is already some evidence that species such as Black-winged Stilt *Himantopus himantopus* are over-wintering for the first time on the Iberian peninsula (Tavares *pers. comm.*) and species such as Common Greenshank *Tringa nebularia* have undergone substantial increases in number in the West Mediterranean (Figure 7).

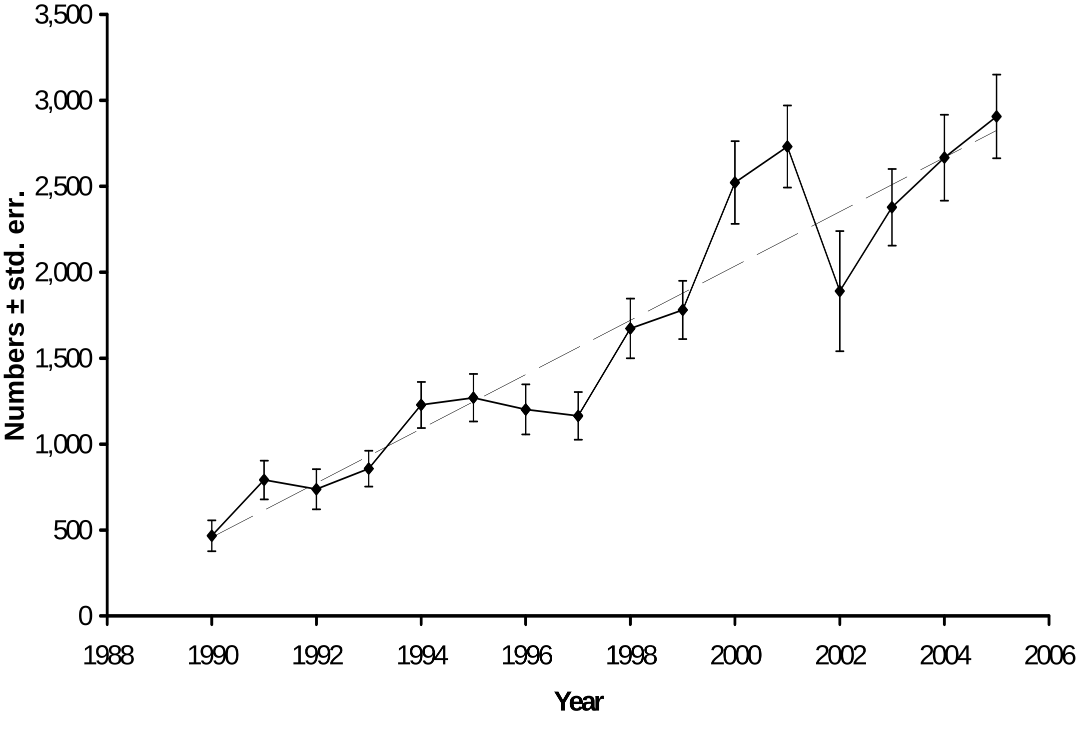


Figure 7. The number of Common Greenshank *Tringa nebularia* over-wintering in the West Mediterranean. Source Wetlands International. Numbers in both areas have increased recently, possibly in response to milder winters.

Table 1. Palaearctic-Afrotropical migrants with small wintering populations north of the Sahara. The location of the main northern wintering population is indicated by an X in the appropriate column. Some species (indicated by the absence on an X in any column) do not currently over-winter north of the Sahara, but are considered good candidates for doing so in the future.

Species	Iberia and /or adjacent North Africa	Nile delta area	South-eastern Europe and adjacent Asia	More widespread in Europe
Purple Heron *Ardea purpurea*		X		
Squacco Heron *Ardeola ralloides*		X		
Black-crowned Night-Heron *Nycticorax nycticorax*		X		
Little Bittern *Ixobrychus minutus*		X		
Bittern *Botaurus stellaris*				X
Black Stork *Ciconia nigra*	X			
White Stork *Ciconia ciconia*	X			
Eurasian Spoonbill *Platalea leucorodia*	X			
Glossy Ibis *Plegadis falcinellus*	X			
Marbled Duck *Marmaronetta angustirostris*	X			
Garganey *Anas querquedula*			X	
Demoiselle Crane *Grus virgo*				
Corn Crake *Crex crex*				
Little Crake *Porzana parva*	X			
Spotted Crake *Porzana porzana*	X			
Baillon's Crake *Porzana pusilla*	X			
Black-winged Stilt *Himantopus himantopus*	X			
Kentish Plover *Charadrius alexandrinus*				X
Great Snipe *Gallinago media*				
Common Snipe *Gallinago gallinago*				X
Jack Snipe *Lymnocryptes minimus*				X
Collared Pratincole *Glareola pratincola*				
Black-winged Pratincole *Glareola nordmanni*				
Black-tailed Godwit *Limosa limosa*	X			
Little Ringed Plover *Charadrius dubius*	X			
Common Greenshank *Tringa nebularia*				X
Common Sandpiper *Tringa hypoleucos*				X
Green Sandpiper *Tringa ochropus*				X
Wood Sandpiper *Tringa glareola*		X		
Marsh Sandpiper *Tringa stagnatilis*	X			
Terek Sandpiper *Tringa cinerea*				
Temmink's Stint *Calidris temminckii*			X	
Little Stint *Calidris minuta*				X
Ruff *Philomachus pugnax*				X
Whimbrel *Numenius phaeopus*	X			
Gull-billed Tern *Sterna nilotica*		X		
Whiskered Tern *Chlidonias hybridus*		X		
White-winged Tern *Chlidonias leucopterus*				
Black Tern *Chlidonias niger*				

3.2. Timing of biological events

3.2.1. Timing of migration

The results of several studies investigating the spring arrival times of migrant species in temperate latitudes have shown that birds are reaching their breeding grounds progressively earlier in the season as the climate becomes warmer (Parmersan & Yohe 2003; Root et al. 2003; Walther et al. 2002). For example, three studies on the spring arrival of migratory birds in the UK suggest that between 26% and 72% of the species recorded were arriving earlier in spring (Sparks & Mason 1999; Sparks 1999), with individual species arriving up to two weeks earlier over the past 2-3 decades (Sparks & Mason 1999). Similar advances in arrival dates of spring migrants have occurred in Europe (Lehikoinen et al. 2004) and are very pronounced in some species listed on Appendix 2 of AEWA. For example, the spring arrival dates of the first White Stork at many sites across Spain has advanced by about 40 days since the mid-1940s (Gordo & Sanz 2006).

Such advances appear to be related to two factors. Firstly, early arrival is necessary as birds need to compensate for advancement in the timing of peak food availability. Many migratory species are insectivores and migration has evolved to ensure young hatch concurrently with periods of high insect abundance (Lack 1968; Visser et al. 2004). Secondly, for most species there is a trade-off between enhanced breeding success and survival (Drent et al. 2006). The earlier a bird arrives on a breeding site, the higher its breeding success, but the greater the risk of thermally-induced mortality (Drent et al. 2003; Drent et al. 2006). Warmer temperatures cause a shift in this balance such that earlier arrival becomes a better strategy. This trade-off can differ between individuals however, demonstrating the complexity of the system. Male Bar-tailed Godwits *Limosa lapponica* with brighter breeding plumage for example, depart for breeding grounds 10-days later than others, because it is thought that the best males may be able to afford later arrival thus avoiding survival costs (Drent et al. 2003).

Departure dates from breeding grounds tend to show a less consistent trend with climate. For example, departure dates of White Storks in Spain have been variable, with no consistent overall trend, but evidence of being latest in the 1990s and as early in the first years of the 21st century as in the late-1950s (Gordo & Sanz 2006). The lack of trend is hardly surprising given the opposing effects climate change might have on migrant departure. On the one hand, the entire breeding cycle is likely to advance, which one would expect to result in earlier departure. On the other hand, temperatures are warmer and cues which trigger migration such as colder temperatures and a reduction in food availability (Elkins 2004) are likely to occur later in the year. Furthermore, it has also been suggested that departure is affected by day length, which is unrelated to climate change (Coppack & Both 2003; Coppack et al. 2003; Lawrence & Soame 2004).

3.2.2. Timing of breeding

In addition to changes in migration phenology, changing climate conditions have affected the breeding phenology of migratory (and non-migratory) birds. For example, (Crick et al. 1997) demonstrated that the laying dates of around one-third of 65 breeding bird species in the UK have advanced, on average, by nine days between 1976 and 1996. These species were not limited by taxonomic group. They include a number of species listed on Annex 2 of AEWA and both migratory and resident species exhibiting a range of ecologies. More detailed analysis of the data over a longer, 57 year, period indicated that laying dates were significantly related to either spring temperature or spring rainfall or both for 31 of the 36 species studied (Crick & Sparks 1999). Furthermore, it was possible to demonstrate that the recent advancement of laying dates for seven of these species could be explained solely by climate, as later laying occurred as spring temperatures decreased up to the 1970s, and laying only advanced when this temperature trend was reversed during the following decades.

Earlier laying dates have been reported in species throughout Europe, North America and Japan, with 79% of 57 species showing a negative relationship between laying date and air temperature (Dunn 2004). There are also a number of single-species studies examining the timing of breeding. For example, in Eurasian Golden Plover *Pluvialis apricaria*, the first-laying dates are negatively correlated with both March and April temperatures and the mean laying-date of the first clutch, additionally negatively correlated with March rainfall. The timing of final laying dates is correlated with April temperatures only and models suggest that they advanced their breeding phenology by 9-days during the course of the 1990s (Pearce-Higgins et al. 2005).

3.2.3. Mechanisms, evolutionary traps and constraints

There is evidence from a whole range of species suggesting that shifts in phenology have resulted in mistiming with respect to the availability of resources. For example, the migratory Pied Flycatcher *Ficedula hypoleuca* has advanced its laying date over the past 20 years. This temporal shift has been insufficient, however, as indicated by increased selection for earlier breeding over the same period (Both & Visser 2001). Similarly, although Great Tits *Parus major* can shift their laying dates earlier in response to earlier warm spring weather, often in parallel with the emergence of their caterpillar prey (Perrins 1991), they cannot significantly decrease their incubation period. Their caterpillar prey, however, can halve their development time in sufficiently warm weather and pupate earlier, leading to an early shortage of the prey that Great Tits feed to their young (Buse et al. 1999). Similarly, the departure of Arctic-breeding geese from wintering areas in northern central and western Europe is dependent on the quality (primarily protein content) of food plants. As a result of regional increases in temperature in Europe coupled with a slower advancement of spring phenology in Arctic shallow waters, the stopover arrival of geese in these areas does not match the (relatively) delayed vegetation growth. There is an even bigger mismatch on the Arctic breeding grounds where these species may arrive much too early for successful breeding (Lindström & Agrell 1999). Similarly, coastal invertebrates rely on synchronisation of photo-period and environmental cues to ensure successful reproduction and climate change is likely to cause uncoupling of such cues such that coastal waterbirds experience worse feeding conditions (Lawrence & Soame 2004).

Phenological mistiming has also been documented in migratory waterbirds such as Common Pintail *Anas acuta* (Drever & Clark 2007). Visser & Both (2005) reviewed all 11 examples (including one AEWA species: Eurasian Golden Plover), ranging from marine plankton to birds, where phenological shifts relative to food supply could be measured and found that for almost all species, the phenology of the focal species shifts either by too little (5) or too much (3). Similar mismatches can occur with components of weather. In Spain, most (91%) of White Stork nestling mortality occurs before they are 20 days old. These mortality rates are highly weather-dependent, with high mortality occurring during rainy periods. Recent advances in the timing of breeding in response to warmer temperatures has however, exposed chicks to more rain, as rainfall tends to be higher earlier in the year (Jovani & Tella 2004). In migrant species mismatched cues are particularly important, as departure from wintering grounds often depends on endogenous rhythms such as photoperiod that are not affected by climate change (Both & Visser 2001).

To understand the true implications of these mismatches, it is necessary to understand the endogenous mechanisms that drive these advancements. Essentially, these mechanisms could be twofold. First, there can be a micro-evolutionary (genetic) response to the selection pressures for earlier breeding. Second, the migrants can show a phenotypically plastic response (i.e. they modify their behaviour) to trends in weather or climate patterns on their wintering ground and/or along their migratory routes, whereby if spring arrives earlier on the wintering grounds, spring migration will also start earlier, facilitating earlier onset of breeding (Jonzen et al. 2006). It is important to draw a clear distinction between these two mechanisms at least insofar as understanding each. If responses to climate change are phenotypic, then they can happen almost instantaneously, but there is a limit to the extent to which phenology can be changed. Furthermore, in a rapidly changing environment the environmental cues used to trigger advancement, might no longer be associated with adaptive outcomes, and organisms can become “trapped” by their evolutionary response. If they are micro-evolutionary, then changes happen more slowly, but overall a slightly larger degree of change can occur and species are not "trapped" by their evolutionary response. Thus there are three major concerns about the future effects of climate change on the timing of biological events. Firstly, do the external cues used by birds to advance their behaviour at one stage in their annual-cycle correlate with actual benefits, such as peaks in food availability, later in their annual cycle? Secondly, do they possess sufficient phenotypic plasticity to cope with changes, and lastly, can they evolve quickly enough to keep pace with changes?

The fact that mistiming is occurring would suggest that evolutionary changes are not happening sufficiently quickly, as inappropriate cues only matter within a bird's life-time rather than through evolutionary time (Jonzen et al. 2006). Nevertheless, there is evidence that at least some changes in biological timing are evolutionarily rather than phenotypically driven. For example, Jonzen et al. (2006) show that long-distance migrants have advanced their spring arrival in Scandinavia by more than short-distance migrants, and that the beginning of their migratory journey is also earlier. Since the cues used by these migrants to trigger departure are uncorrelated with climate change on the breeding grounds, such advancement is demonstration that evolutionary changes have occurred. Visser et al. (1998) show that there has been differential selection of different phenotypes, such that an increasingly greater proportion of recruits in the following year come from early nesting individuals. The most parsimonious explanation overall is (1) that birds are responding phenotypically to climate change, (2) that individuals possess different phenotypes, some of which have 'earlier' traits, (3) that 'earlier' phenotypes are being preferentially selected and (4) that the rate of this selection is at present insufficient to keep pace with climate change. Interestingly there is also evidence from Dutch studies on Great Tits that 'more plastic' rather than just 'earlier' phenotypes are being selected for (Nussey et al. 2005). Although in the long-run, this will improve a population of bird species' ability to cope with climate change it also incurs disadvantages in that a smaller proportion of the population behaves in an optimal manner. It also demonstrates that the degree of climate change experienced by species is increasing through time.

3.3. Changes in demography

Although climate change causes changes in distribution and the timing of biological events, in a conservation context, it is at least equally important to understand how this will impact on populations. To do so, one needs to understand how climate change has affected demographic parameters.

3.3.1. Survival

It is well known that adverse weather affects bird survival and can do so either directly or indirectly. Direct effects result because the rate of heat loss from a bird's body increases as temperatures decrease and wetting of the feathers by rain reduces their effectiveness to act as insulation (Robinson et al. 2005). Although the latter is not a major problem for many adult waterbird species, which tend to have waterproof feathers, it can be a problem for juveniles. In cold, wet conditions birds therefore have to invest more energy in maintaining their body temperature, and individuals failing to meet these increased costs may die. This is true particularly of resident passerine species in Europe (Peach et al. 1995; Thomson et al. 1997; Robinson et al. 2004; Robinson et al. 2007), but also applies to migratory waterbirds listed on Annex 2 of the Agreement, such as Common Redshank (Insley et al. 1997) and Red Knot *Calidris canutus* (Wiersma & Piersma 1994). The former study found that survival rates of first-year Common Redshank were lower in cold winters and very wet winters, possibly due to increased thermoregulatory costs in the exposed environment that they inhabit. Under warmer conditions, such effects are likely to become less severe, with beneficial consequences for the survival of individual birds. The majority of bird species are limited in the amount of energy they are able to store as fat reserves, principally because excess weight increases the energetic cost of flight, as well as leading to increased predation risk as escape flight ability is impaired (Alerstam 1990; Gosler et al. 1995). Body weight (and hence condition) is thus maintained in an extremely dynamic fashion, with a number of factors influencing the bird's fattening decisions. Short, unexpected periods of adverse conditions can disrupt the delicate balance of trade-offs, causing large-scale mortality over very short periods of time. For example, Clark (2004) found that the number of dead, ringed individuals of several migratory wader species listed on Annex 2 of the Agreement, including Common Redshank, Grey Plover *Pluvialis squatarola* and Dunlin increased during winters with periods of extreme cold.

Indirect effects of climate and weather on survival are also important and occur primarily through influencing the abundance and availability of food supply. Just as climate and weather affects birds, the quality, availability and accessibility of prey is also influenced, affecting survival through altered intake rates. Such factors may be more important in the non-breeding period when many populations, particularly those that winter in the northern hemisphere, may be food-limited (Robinson et al. 2005). Extended periods of frost or snow cover may prevent ground-feeding birds from foraging, leading to increased rates of mortality and hence a reduction in population size (Robinson et al. 2007. Survival rates of many passerines that spend the winter in Europe are known to be related to winter weather conditions (Peach et al. 1995; Robinson et al. 2005) and this appears to be related to the accessibility of food sources (Robinson et al. 2005). Waterbirds, as the Grey Heron *Ardea cinerea*, and dabbling ducks are particularly susceptible to decreases in food availability during spells of cold winter weather at higher latitudes, when areas of open water freeze over (Besbeas et al. 2002) and low temperatures and high rainfall in winter adversely affect the foraging success of many migratory shorebirds (Goss-Custard 1970; Insley et al. 1997; Pienkowski 1983). A warming climate is likely to increase the survival rates of many waterbird species at higher latitudes, although it is probably more likely to benefit residents than migrants as the majority of migrant species spend the non-breeding season in warmer areas. Consequently there may be increased competition between residents and migrants during the breeding season.

Migratory waterbirds also require favourable conditions at stopover sites in order to successfully complete their journey. For example, climate change is thought to affect Svalbard Pink-footed Goose *Anser brachyrhynchus* survival, with warmer and wetter winters, enhancing survival due to increased food availability on Norwegian staging grounds (Kery et al. 2006). Climatic conditions on the poleward, spring journey are likely to be particularly important. This is for two reasons. Firstly, there is a greater evolutionary imperative to arrive early on the breeding grounds, as the first arrivals secure the best breeding territories and hence increased reproductive success (Kokko 1999). Secondly, mortality during spring migration has less chance to be compensated for by other mortality factors than on autumn migration. Mortality during the winter is often significant, and is likely to be density dependent, so that many individuals that die in the autumn may have died in the winter anyway, even if they had survived the autumn migration period, although this is less likely to be the case in long-lived species (Robinson et al. 2005).

There is increasing evidence that infectious diseases can influence the survival of bird populations (Hochachka & Dhondt 2000; Newton 1998; Tompkins et al. 2000). Changes in weather conditions may lead to shifts in the rate of reproduction of pathogens, or in the distribution of vectors carrying these pathogens (Faustino et al. 2004). For example, Epstein (2001) suggested that warm winters and spring droughts might have contributed to the spread of West Nile Virus through avian populations in Europe and North America. The distribution and incidence of other avian diseases, such as malaria and avian influenza, may also be dependent on climatic variables (Olsen et al. 2006) although little information is currently collected on diseases in wild bird populations and there are insufficient historical data to allow the assessment of trends in infection over time.

Overall, it is thought that weather conditions, particularly in winter, have an impact on bird survival. The trend towards warmer conditions at higher latitudes may decrease the mortality of resident species due to adverse conditions (although see section 3.6. for likely impacts of extreme events), but this may lead to migrants facing increased competition during the breeding season. Warming in tropical regions, and particularly lower rainfall (see section 3.4) will reduce over-winter survival for many long-distance migrants. To what extent density dependence in population processes will counteract these effects is unclear at present; this is an area that merits further research, and one for which much data is likely to be available, at least for commoner species (Robinson et al. 2005). Stopover sites on migration are key periods for migrating birds as they experience great physiological stress. Changes in conditions experienced and in available food resources could have substantial effects, either beneficial or detrimental, depending on the type of change. The impacts of climate change will be species-specific, depending on their ecology, although groups such as long- or short-distance migrants may show similar patterns of impact. The role of diseases and parasites in population ecology, has recently begun to be realised and should be the subject of further research.

3.3.2. Productivity

As with survival, climatic conditions can influence productivity both directly and indirectly. Direct effects result because young chicks may be particularly susceptible to chilling (hypothermia) due to their small size and undeveloped plumage, as may incubating adults under especially severe conditions. In warmer climates, e.g. the Mediterranean, they may also be susceptible to heat stress (hyperthermia). Indirect effects result because climatic conditions can also influence the amount of food that parents are able to collect for their offspring (Robinson et al. 2005).

Evidence of direct impacts of climatic conditions is most clear for passerines, raptors and game birds, with the former most affected by temperature, and the latter two by rainfall (Robinson et al. 2005). Several studies have also documented relationships between waterbird fledging success and temperature (Hötker & Segebade 2000; Schekkerman et al. 1996) with higher temperatures generally leading to higher productivity. Similarly, the proportion of juvenile Dunlin over-wintering in north Wales is strongly correlated with summer temperatures in the Arctic. The proportion of juveniles was highest at intermediate temperatures, suggesting that as temperatures increase, productivity may decline (Beale et al. 2006).

Indirect effects occur primarily because cold and/or wet weather can reduce the invertebrates available to birds as prey by reducing their overall abundance and levels of activity (Avery & Krebs 1984; Davies & Green 1976; Foster 1974; Morse 1989). However, such impacts will depend very much on the timing of rainfall in relation to young being in the nest, and may vary regionally (Robinson *et al.* 2005). Indirect effects can also occur as a result of more complex ecological interactions. For example, temperature affects the size of the lemming (*Lemmus* spp. and *Dicrostonyx* spp.) populations in the tundra (Soloviev & Tomkovich 2003; Summers & Underhill 1987). In years when lemmings are scarce, avian and mammalian predators such as skuas *Stercorarius* spp. and Artic Fox *Alopex lagopus* turn their attention to the contents of nests, reducing the breeding success of the ground-nesting wader species. Thus, predicted warmer climates are likely to have severe negative impacts on wader populations, most of which are migratory, particularly those that breed in the Arctic (Robinson et al. 2005).

Climate can also affect productivity by changing the time available for breeding. For example, populations of many waders breeding in the High-Arctic spend the winter in coastal areas of Europe and Africa. Although the implications of earlier arrival and breeding of northern waders are currently unexplored, it may provide waders with greater opportunities for multi-brooding or clutch replacement after failed breeding attempt (Rehfisch & Crick 2003). Furthermore, earlier breeding may provide juveniles and post-breeding adults with a longer period to grow and improve their body condition before migrating south at the end of the summer. However, there is the possibility that warmer temperatures may lead to increases in high latitude precipitation, such that snow cover is retained for longer. Thus, waders that attempt to arrive earlier on their breeding grounds may be faced by prolonged snow cover that could affect their productivity. Furthermore, increased snow cover and delayed melt could delay their breeding, which may be detrimental given the short ‘window of opportunity’ that they have for breeding at high latitudes before temperatures start to fall again (Robinson *et al.* 2005). There is some evidence of this occurring in geese. For example, the timing of Snow Goose *Chen caerulescens* arrival can affect their probability of breeding, with those arriving earlier less likely to breed. However declines in breeding success later in the season is such that Snow Geese will maximise their reproductive success by arriving a few days earlier than the median (Bety et al. 2004). However, across most species, late arrival leads to lower reproductive success, with birds balancing this against starvation risks (Drent et al. 2003).

Climate change can also influence adult condition at the start of the breeding season. Even if individuals are able to survive unfavourable winter conditions while on passage or on return to the breeding grounds, they may be left with depleted energy reserves that are insufficient to allow successful breeding during the following season. For example, ambient temperatures during the laying period have been found to positively influence the clutch size of a number of waterbirds in the High-Arctic (Ganter & Boyd 2000). There is more evidence from passerine species, including the migratory Pied Flycatcher and Common Redstart *Phoenicurus phoenicurus* (Jarvinen 1989; Winkel & Hudde 1997) and increased clutch sizes have been reported as a consequence of a trend towards warmer temperatures and earlier arrivals (Jarvinen 1996; Winkel & Hudde 1997). In addition, Nager & Zandt (1994) observed that Great Tits laid larger eggs in warmer temperatures, which are likely to have increased hatching success and chick survival (Both et al. 1999; Jarvinen & Vaisanen 1983). Broggi et al. (2004) and Ojanen (1983) showed that temperature acted as a constraint on egg-laying in the Great Tit due to conflicts with daily metabolism, so increased egg sizes are likely irrespective of calendar date of laying. Sanz's (2002) meta-analysis of Blue Tit *Cyanistes caeruleus* and Great Tit populations showed no trend towards increased clutch size, suggesting that the breeding season has just become earlier rather than following the relationship that currently exists between laying date and clutch size, and hence producing a larger clutch size at an earlier date (Crick 2004). This highlights a general problem with understanding the mechanisms of climate change – the relationships we observe now may not hold under changed climatic conditions. Understanding the mechanisms behind the relationships will be important if we are to accurately assess the impacts of a changing climate.

In summary, there is good evidence that changes in productivity have occurred as a result of changes in climate. In northern temperate regions, at least, productivity for many species may increase, provided benefits from warmer temperatures are not offset by increased predation. However, the density-dependent nature of population processes does not mean an increase in population size, as survival may decline in response to the increased competition between a greater number of individuals. There is evidence that productivity of open ground nesters may decrease in response to increased precipitation, though the timing of such events will be critical in determining the severity of impacts on population size. Changes in productivity in response to climate change are particularly well documented in seabirds, and such changes appear to be independent of other factors, such as fishing pressure. Reductions in seabird productivity seem to be driven by declines in prey distribution in response to warming sea temperatures (Robinson et al. 2005).

3.3.3. Population impacts

The size of all populations is essentially controlled, at a fundamental level, through the balance of two demographic rates: the rate at which new individuals are born into the population, and the mortality rate, i.e. the rate at which old individuals leave the population (Newton 1998), although immigration and emigration can be important at a local-level. A number of detailed population studies have shown that reasons for changes in population size can be explained by understanding variation in these demographic rates (Atkinson et al. 2003; Frederiksen et al. 2004; Goss-Custard et al. 1995a; Goss-Custard et al. 1995b; Peach et al. 1994; Potts 1986; Robinson et al. 2004; Siriwardena et al. 2000). Several studies also examine the effects of climatic variables on changes in bird population listed on Annex 2 of the Agreement (see Table 2).

Numbers of breeding Eurasian Wigeon *Anas penelope* in Iceland were related to the date of water becoming ice-free on their breeding lakes (Gardarsson & Einarsson 1997). It is unclear though, whether this simply represented dispersal elsewhere, possibly into a non-breeding population and so whether the observed change in numbers on the study lakes was related to the overall population size, or simply reflected a redistribution elsewhere. It would appear that the decline in the numbers of many species of Britain’s internationally important wader populations (Rehfisch et al. 2003a; Rehfisch et al. 2003b) is a result of a change in their distribution (Austin & Rehfisch 2005; Rehfisch et al. 2004) that has extended to Continental Europe (Maclean et al. in review). However, it is not known whether this is more likely that this is due to a redistribution of the population than due to demographic processes.

Generally, evidence of indirect effects of climate change on populations, through for example range shifts and changes in phenology, are lacking.

It is also noteworthy that climate change during one phase of a species' annual cycle can affect numbers during other phases. For example, numbers of Eurasian Golden Plovers breeding in upland England are related primarily to temperature on the wintering grounds (Yalden & Pearce-Higgins 1997) increasing two years after warm and moist winters, presumably because juveniles survive better (Forchhammer et al. 1998). Similar changes are likely to be exhibited by many of the migratory waders listed on Annex 2 of the Agreement, particularly for those that winter in northern temperate latitudes.

Table 2. Effects of weather variables on the survival, productivity or local population size of species listed on Annex 2 of the Agreement. Data on productivity are generally lacking

Species	Survival		Productivity		Population size		Reference:
	Season	Variable	Season	Variable	Season	Variable	
Grey Heron *Ardea cinerea*	W	T					(Besbeas et al. 2002)
					W	T	(Marchant et al. 2004)
Purple Heron *Ardea purpurea*					S	R	den Held (1981)
					S	R	Cave (1983)
Squacco Heron *Ardeola ralloides*					S	R	den Held (1981)
Black-crowned Night-Heron *Nycticorax nycticorax*					S	R	den Held (1981)
Mute Swan *Cygnus olor*					W	T	Bacon & Andersen-Harild (1989)
Eurasian Wigeon *Anas penelope*					S	T	(Gardarsson & Einarsson 1997)
Northern Lapwing *Vanellus vanellus*	W	T					(Peach et al. 1994)
Eurasian Golden Plover *Pluvialis apricaria*					W	T	(Yalden & Pearce-Higgins 1997)
Grey Plover *Pluvialis squatarola*	W	T					(Clark 2004)
Common Redshank *Tringa totanus*	W	T/R					(Insley et al. 1997)
	W	T					(Clark 2004)
Dunlin *Calidris alpina*	W	T					(Clark 2004)

Season: W=Winter, S=Spring; Variable: R=Rainfall, T=Temperature

3.4. Effects of changing rainfall patterns

The effects of high rainfall on birds are similar in many ways to those of low temperature and the effects of drought similar to high temperature. Consequently, many of the issues associated with changing rainfall have already been covered. However, a number of species groups are particularly affected by rainfall. For example, wet weather has been linked to chick mortality in game birds, possibly because their chicks leave the nest soon after hatching and are particularly susceptible to wetting when foraging in damp vegetation (Moss et al. 2001; Panek 1992; Summers et al. 2004). Some waterbird species, notably ducks also have chicks which leave the nest soon after hatching, but because of their close association with water one would expect them to be less vulnerable to wet weather. Indeed for waterbird species one might expect drought to be a more severe issue. For example, migratory waterbirds occurring in northern Europe and wintering in Africa south of the Sahara are likely to be greatly impacted by the state of vegetation, which is dependent mostly on rainfall rather than temperature. A useful measure for monitoring annual variation in this is the Normalised Difference Vegetation Index (NDVI), which has been shown to be correlated with populations of several migrant birds (Saino et al. 2004). Although resident at higher latitudes may face harsh conditions in winter, migratory species that spend the non-breeding period in the southern hemisphere may also suffer from adverse climatic conditions. For those spending the winter in sub-Saharan Africa, rainfall appears to be particularly important, with those over-wintering in the Sahel region especially vulnerable.

3.4.1. The Sahel region and crucial stop-over sites

Several waterbird species have been shown to be affected by rainfall in the Sahel region. For example, the number of breeding pairs of Purple Herons *Ardea purpurea* in The Netherlands, correlates closely with the extent of winter drought in the Sahel region (Den Held 1981), apparently as a result of a correlation between annual survival and drought severity (Cavé 1983). Similar correlations have been described between winter wetness in the Sahel and the annual survival of and the western population of White Storks (Kanyamibwa et al. 1993). Several passerine species in northern Europe has been shown to be especially affected by rainfall in the this region, suggesting that this is a widespread phenomenon. For example, during periods of Sahelian drought such that which occurred during the late-1960s, populations of migratory passerines such as Sedge Warbler *Acrocephalus schoenobaenus*, Barn Swallow, Common Whitethroat *Sylvia communis* and Sand Martin *Riperia riperia* declined precipitously (Baillie & Peach 1992; Peach et al. 1991; Szep 1995; Winstanley et al. 1974). A fairly large number of waterbird species (39) such as Purple Heron, Eurasian Spoonbill and Great Snipe *Gallinago media* follow a similar migratory route these passerine species and have similar wintering areas to the species for which links have been observed (Wetlands International 2006). An even larger number (51) are local migrants that rely on wetlands in this region. Due to their dependence on wetlands, all these species might be expected to be especially vulnerable to drought. Continent-wide analysis shows that populations of Afro-Palaearctic migrant birds have shown a pattern of sustained, often severe decline between 1970 and 2000. Trends of long-distant migrants were significantly more negative than short-distance migrants or residents. The negative trend appeared to be largely, but not entirely due to declines in species wintering in dry, open habitats in Africa, although analysis of closely related pairs of species, one a long-distance migrant the other not, revealed greater declines in long-distance migrants irrespective of breeding habitat (Sanderson et al. 2006).

3.5. Effects of sea-level rise

Rises in sea-levels will have both direct and indirect effects. Direct effects include flooding of nest sites. Even moderate rises in sea-levels can lead to a substantial shortening in the return period of storm surges. For example, in coastal areas of the Mediterranean a 20 cm rise in sea-level would increase the frequency of a 1 in 100 year event such that it happens 3-4 times a year, whereas a 50 cm rise would lead to 1 in 100 year events happening once or twice a week (Figure 8; Nichols et al. 1999). The shortening of the return rate of storms varies substantially in different areas. In the Netherlands for example, even a 50 cm rise in sea-level would only shorten the storm-surge frequency from 1 in 100 years to 1 in 70 years. Those areas most vulnerable to major shortening in the return periods of storm surges are those which do not normally experience severe storms, such as the Mediterranean, Black and Red Sea coasts. The nests of all waterbird species nesting in low-lying areas will be subject to increased risk of flooding. At present it is not known whether birds that choose to nest close to water minimize risk of flooding by nesting a certain height above mean high water, or whether some process enables them to assess flood risk and they base it on factors such as the return period of storms. If the former then species nesting close to the water on the Atlantic seaboard, such as Roseate Terns *Sterna dougallii* would be most at risk. If the latter, it is those nesting in the Mediterranean, Black and Red Sea, such as Kentish Plover *Charadrius alexandrinus* and Audouin’s Gull *Larus audouinii* that will be most at risk. However, regardless of the cues used by birds to select breeding sites, populations with localised breeding colonies are most likely to be at greatest risk.

Some coastal species are likely to suffer from rising sea levels due to habitat loss. Increases in sea level combined with 'coastal squeeze', whereby landward progression of coastal habitats is prevented by flood-defence works, could result in the loss of substantial areas of inter-tidal mudflats (Pethick & Crooks 2000), with severe implications for both wintering and breeding wildfowl and wader species. Saltmarshes are likely to be under threat as well (Hughes 2004; Pethick & Crooks 2000), which will have direct effects on birds such as Common Redshank which use this habitat for breeding, or Brent Geese *Branta bernicla* which use it for feeding. It will also have indirect effects for species in other coastal habitats, because saltmarshes are the base of estuarine foodwebs, supplying large amounts of organic matter to adjacent areas (Hughes 2004). The addition of flood defences also significantly modifies the existing habitats into which they are placed, which may have implications for the birds that use these habitats (Moschella et al. 2005).

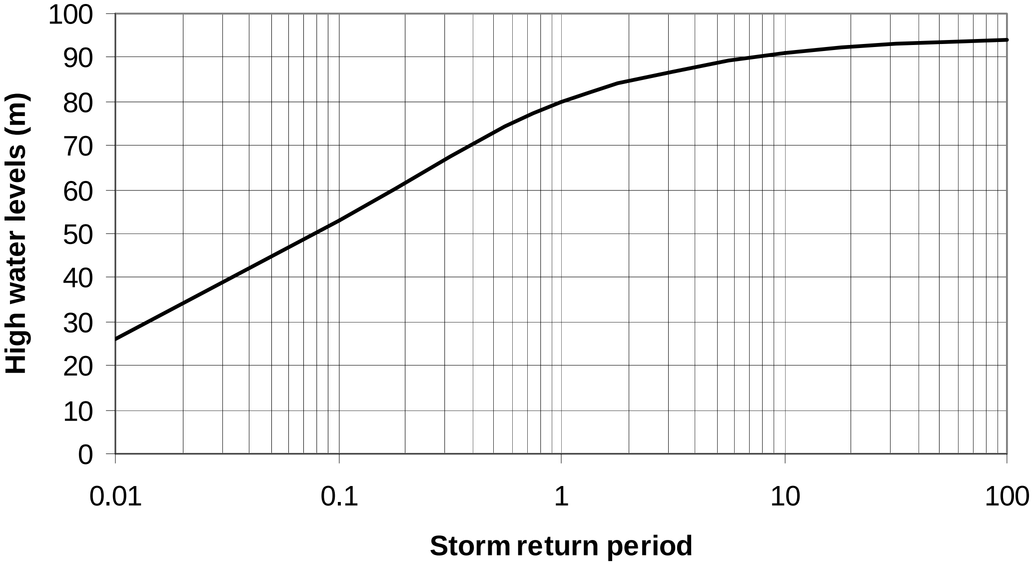


Figure 8. Storm return period for the coast of Egypt. A 20 cm rise would thus result in 1 in 100 year events occurring 4-5 times a year and a 50 cm rise would result in 1 in 100 year events occurring once or twice a week. Adapted from Nichols et al. (1999)

In some areas where these important habitats would theoretically be able to move inland as sea levels become higher, the land is being developed by conservationists to facilitate this process, creating areas of 'managed retreat' through 'coastal realignment' (Atkinson et al. 2004). Such adaptation measures are likely to be necessary to prevent declines in the overall numbers of waders using European estuaries. Other habitats, which are at risk from sea-level rise but could benefit from adaptation measures, are coastal freshwater or brackish lagoons and freshwater grazing marsh. Both habitats are extremely valuable for wildfowl and waders as feeding and nesting sites (Stroud et al. 2001), but are vulnerable to saltwater inundation as the invertebrate prey or vegetation on which birds feed can only tolerate a finite range in salinity or flooding conditions (Boorman 1992; Olff et al. 1988). Freshwater lagoons are at risk from any inundation. However brackish lagoons are often at risk from lack of inundation if coastal defence works are present, particularly in summer when evaporation can lead to hyper-saline conditions if no flooding occurs. In such instances, flood defence works could be designed to allow periodic inundation when necessary.

3.6. Wind, Storms and Hurricanes

Changes in predominant wind direction can have an effect on both the timing and ability of individuals to complete their migratory journey (Elkins 2004; Robinson et al. 2005), the impacts on population size may be relatively limited, and of concern only for species with small populations, such as Siberian Cranes *Grus leucogeranus* (Robinson *et al.* 2005). However, during peak migration periods, even relatively moderate uncharacteristic winds, if combined with poor visibility and / or rain, can lead to large displacements of birds from their normal migratory trajectory (Elkins 2004). In the UK for example, north-easterly winds combined with poor visibility conditions towards the end of August or early September can lead to thousands of migrants from Scandinavia on route to further south, occurring in bushes on the eastern seaboard (Elkins 2004). Whilst it is normally small passerines that are affected in this way, waterbirds are also susceptible to this phenomenon to some degree (Elkins 2004) and may be particularly vulnerable if unfavourable winds displace birds away from crucial stop-over sites towards the end of long stages in their migratory journey (Robinson et al. 2005). For example, the common tern colony on Bermuda was destroyed by a hurricane in 2003, reducing this extra-limital population from 20 to 12 individuals, all of which were females (Nisbet in litt).

Severe storms can also have a direct impact on an individual's survival; for example, Lens & Dhondt (1992) reported the death of 62% of their study population of crested tit *Parus cristatus* after a severe storm in Belgium and similar devastating effects have been observed for hurricanes (Askins & Ewert 1991; Wauer & Wunderle 1992). Although one might expect small passerines to be most at risk from severe storms in this way, very severe storms may also affect waterbirds particularly if they contribute to destroying habitat. Some areas, particularly those in the tropics, will face an additional threat in the form of elevated rates of erosion that have resulted from increases in the strength and frequency of winds and storm events. An increase in storm prevalence is also likely to increase the risk of nests flooding, which will be most deleterious to localised colonial species such as Roseate Terns (Robinson et al. 2005).

3.7. Indirect impacts of climate change

3.7.1. Climate change and land-use change

Although land-use changes resulting from recent climate changes are difficult to identify due to multiple non-climate driving forces and the presence of adaptation, effects have been detected in forestry and a few agricultural systems (IPCC 2007a). However, thus far, recent warming has been of limited consequence to land-use change in comparison with other factors. While there is now significant evidence of observed changes in natural systems in every continent, including Antarctica, as well as from most oceans, the majority of studies come from mid- and high latitudes in the Northern Hemisphere. Documentation of observed changes in tropical regions and the Southern Hemisphere is sparse (IPCC 2007a), although it is unclear whether this is due to publication bias or this is an adequate reflection of geographical variations in effects. Observed changes likely to have had impacts on waterbirds include small shifts in cereal production towards colder climes in Northern Europe, increased drainage to minimize flood risks (Hossell et al. 1996; NFU 2005; Parry et al. 1999) and the consequential loss of unimproved wet grassland, essential breeding and feeding grounds for many waders (Stroud et al. 2001).

3.7.2. Climate change and water-use change

Climate-related trends of some components of water-use during the last decades have already been observed (IPCC 2007a). For a number of components, for example groundwater, the lack of data makes it impossible to determine whether their state has changed in the recent past due to climate change. During recent decades, non-climatic drivers have exerted strong pressure on freshwater systems. This has resulted in water pollution, damming of rivers, wetland drainage, reduction in stream flow, and lowering of the groundwater table (mainly due to irrigation). In comparison, climate-related changes have been small (IPCC 2007a). Nevertheless, there is clear evidence that there have been changes in the way water resources are used, with increase in usage during warmer periods resulting in a lowering of the water table (CBD 2003; Holman & Loveland 2001; Holman et al. 2005a; Holman et al. 2005b; NFU 2005) and this is likely to have had at least some impact on waterbirds (UKBAP 2005).

3.7.3. Climate change and flood-defences

Climate change is already causing increased flood risk, both as a result of sea-level rise and due to more extreme rainfall events (IPCC 2007a). In coastal areas, sea-level rise has resulted in minimal loss in areas where re-alignment is managed such that there is no overall loss of important habitats (Crooks 2004; Hughes 2004). However, in some areas where coastal defence works have been placed around much of the coast, areas of important habitat have been lost as they are squeezed between rising seas and hard defences (Mitchell et al. 2007; UKBAP 2005). Although coastal defence works generally result in coastal squeeze, it is also important to note that lack of defences can result in the loss of important habitat. Two of the most important coastal habitats in Europe for waterbirds for example, are brackish lagoons and areas of coastal grazing marsh containing pools of freshwater used by breeding and feeding waders (BirdLife International 2007; Maclean & Austin 2006). With the lack of appropriate sea-defences, several such areas (e.g. the North Norfolk Coast in the UK and the Camargue in France) are threatened by sea-level rise.

3.7.4. Climate change and tourism and recreation

In some colder areas, there is evidence that recreational tourism has increased in response to warmer temperatures (Coombes et al. 2005). Certainly it is a very well established phenomenon that visits to coastal areas of Northern Europe increase when the weather is warm (Viner 2006). In addition to increased recreational disturbance there has been an observed increase in the length of the tourism season, increased tourism infrastructure (i.e. hotels, attractions, marinas), increased waste (i.e. sewage, solid waste) and increased destruction of natural habitat (Coombes et al. 2005; Viner 2006), factors which have had a detrimental impact on waterbirds in some of the important coastal habitats of the UK for example (Maclean & Austin 2006).

4. FUTURE EFFECTS OF CLIMATE CHANGE ON WATERBIRDS

4.1. A framework for modelling future changes: approaches and limitations

There are seven main approaches that are used to predict the ecological consequences of environmental change such as climate change. These are: extrapolation, experiments, phenomenological models, behaviour-based models, expert opinion, outcome-driven modelling and scenarios (Sutherland 2006). Each approach has different strengths and weaknesses and in practice several approaches are used. The merits and weakness of each are discussed in turn here. The text makes considerable use of material presented in Sutherland (2006), but the authors' own interpretation of drawbacks and benefits are also presented and other literature is drawn upon where necessary.

4.1.1. Extrapolation

This approach makes the assumption that a particular trend, be it values associated with one dataset or be it correlations between two or more variables (such as temperature change and changes in abundance) will continue. The great advantage of this approach is its simplicity. All that is needed is one or more time-series of data. However, its simplicity is also its disadvantage. If conditions change such that the trend or relationship itself changes, which is highly likely in the context of climate change, extrapolations are unlikely to be sensible (Sutherland 2006).

4.1.2. Experiments

Experiments probably provide the most convincing means of predicting responses to climate change. The problem is that they are usually almost impossible to carry-out in a fully realistic manner. There are some excellent experiments such as that which manipulated the temperature and rainfall patterns of existing plots of grassland (Grime et al. 2000) and the merits of such experiments are that they can reveal unexpected responses as a result of complex interactions, which may not interact in the same way under baseline conditions. However, the disadvantage of experiments is that they cannot be carried out at the spatial or temporal scale over which climate changes and organism interactions with other factors are expected.

4.1.3. Phenomenological models

Phenomenological models, those based upon empirically derived functions, are pivotal to understanding the impacts of climate change in so far as they are designed to examine causal relationships. Typically, this modelling approach entails understanding how demographic parameters are affected by climate change. The approach has several draw-backs however. Firstly, major components of population demography are difficult to estimate. Phenological models depend on three fundamental tenets of population dynamics: (1) that populations increase in the absence of competition (intrinsic population growth), (2) that at higher population levels this growth rate is reduced as a result of competition (density-dependence) and (3) that stochasticity (i.e. random events) can produce deviations from a deterministic equilibrium (i.e. a pre-determinable stable-state). However, field measurement of intrinsic population growth usually entails either looking at population changes over time or measuring demographic components and combining these. In so doing, intrinsic population growth is confounded with density-dependent growth. Similarly, measuring density-dependence is difficult. Doing so usually entails using a time-series to plot the change between each time interval against the total population size. Unless there is no census error, or census error can be measured exactly (which is rarely the case), then density-dependence will be overestimated (Freckleton et al. 2006). Stochasticity is difficult to measure simply because it relies on long runs of data collected in the same way. Another difficulty is that phenomenological models can potentially suffer from the same failings of extrapolation. When using phenomenological models to predict the consequences of climate change, one of the most common approaches is to correlate demographic parameters with components of climate change. If current relationships do not hold true into the future, as they may well not given the wide array of non-linear relationships observed in nature, then any projections are unlikely to hold true.

4.1.4. Behaviour-based models

The philosophy behind this approach is to understand the behavioural decisions individuals make and use this to derive demography to make predictions from first principles. The starting point is usually to assume that individuals will settle where they gain the highest – i.e. they follow an ideal-free distribution (Fretwell & Lucas 1970). The next step is to understand the negative feedbacks of population size (such as depletion) upon distribution and performance. This could be used to determine key questions about the consequences of habitat loss or modification that may result from climate change or sea-level rise on migratory waterbirds: will the individuals move elsewhere or are all alternative sites full? The critical question then is how much the population is likely to change as a result of habitat loss or modification? To answer this, it is necessary to derive population parameters and then incorporate these into models that link habitat loss and population size. Examples of such studies include Durell et al. (2006) and Sutherland (1996).

The advantage of the behaviour-based modelling approach is that it does not suffer from extrapolation problems to the same degree as other methods. Information is derived from first principles and while some factors, such as future food supply cannot be extrapolated without some risk of error, the extent to which birds attempt and have the means to maximise energy intake is less likely to change to a significant degree. However in applying such models it is often necessary to fit unknowns, thus reducing the model's predictive powers and often many parameters are not known with a high degree of confidence. There is a need for greater development and testing of this approach, although see Goss-Custard et al. (2006) for one non climate change example of where this was done.

4.1.5. Expert opinion

In principle, just asking someone what might happen in relation to climate change in the future might seem to be a poor way to make predictions, but in practise many of the decisions in applied ecology are based on the opinions of others (White et al. 2005). Moreover, particularly if scientific techniques cannot offer predictions with a high degree of confidence it is probably better to accept that opinion is a necessary technique. The advantage of this approach is that information is simple, cost-effective and relatively easy to obtain. The disadvantage of this approach is that experts are typically far more confident than is justifiable (Burgman 2005). Currently, when expert-opinion is used to guide climate change policy, it is usually obtained in an unstructured manner, for example by using a single expert. Policy-makers need to consider more rigorous means of obtaining expert opinion by devising specific questionnaires, asking a range of experts and determining the degree of consistency in expert answers (see Sutherland (2006) for a detailed methodology).

4.1.6. Outcome-driven models

The conventional approach to make predictions with regards to how climate change might affect biota is that described in the previous sections, and it is to calculate how the expected climate changes lead to ecological changes. This approach can be unrealistic if there are complex interacting processes, each with associated uncertainty. An alternative approach is to reverse the problem by considering a number of possible outcomes and assessing the likelihoods of each. Techniques such as Bayesian analysis, probabilistic reasoning and fuzzy logic, that attempt to find efficient solutions to problems with imprecision, uncertainty and partial truth are increasingly being used and developed. A full account is not given here but can be found in Shepard (2005) and Tettmanzi & Tomassini (2001). The disadvantage of this approach is that it is new, and as such relatively untried and tested and many of the methods have not been fully developed. A major advantage is that it is much better at dealing with uncertainty, does not give unrealistic estimates of the level of confidence and provides optimal solutions in the face of uncertainty.

4.1.7. Scenarios

Decision- and policy-makers are often faced with considerable uncertainty. Under such circumstances, conventional modelling approaches do not provide sufficiently meaningful results and in some instances other methods, such as outcome-driven models are prohibitively expensive or time-consuming. The scenario approach was pioneered by Royal Dutch Shell in the 1960s and now adopted by many organisations including NATO (Sutherland 2006). The process starts by considering the major issues for which there is uncertainty. In practice the two most important are usually used and for each the resulting combinations, outcomes are predicted and presented. This is the procedure used by the IPCC to generate climate predictions (IPCC 2007b). The advantage of this approach is that it enables a few simple messages to be conveyed to policy- or decision-makers in situations where there are many complex factors that lead to an outcome. The disadvantage is that the scenarios are not necessary predictions at least insofar as the likelihoods of each are not assessed. Thus it is often difficult to decide upon one single best solution to a problem or to determine the relative likelihood of one or other outcome occurring.

4.2. Future distribution and range changes

Since current distributions are affected by climatic variables and since there is widespread evidence that historic distributions have changed in response to warming temperatures and other climate changes, it is reasonable to assume that such changes will continue. Broadly therefore, one can expect species to continue to expand their breeding range poleward. In the UK for example, it seems likely that several new species of waterbird will colonise from the continent and spread northwards. The three most likely are Cattle Egret, Spoonbill and Great Egret *Casmerodius albus*. These species have already increased substantially in number in the Netherlands (van Roomen et al. 2005) as they have already undergone rapid northward expansions and have established a recent toehold in the UK (Figure 4b). However one can expect several species to retract their range, particularly at higher latitudes and altitudes. Such effects will be most pronounced in the Arctic where temperatures are predicted to increase by the most, thus affecting waders and wildfowl that breed there, which could see their habitat compressed towards the poles and conceivably disappearing altogether (Rehfisch & Crick 2003). Range shifts due to temperature in Africa are less certain partly as there is less information from this region, and partly because the upper temperature limit of species which occur in the tropics is unknown. Broadly one might expect similar results to those elsewhere with poleward expansion of species ranges. Much of the land mass in tropical East Africa is mountainous. Here one might also expect altitudinal shifts to occur, with lowland species increasingly occupying upland areas and those associated with higher altitude, such as the Ethiopian Highland population of White-winged Flufftail disappearing altogether. Dotterel are also likely to be highly vulnerable to climate change as almost the entire wintering population is confined to mountains in North Africa (Wetlands International 2006).

In winter, isotherms in Europe are aligned north-east to south-west rather than east to west, such that temperatures increase as one moves south-west rather than just south. Consequently, species on the whole are likely to move in a north-easterly direction rather than in a northerly direction. For waders shifting their over-winter distributions in Europe this could pose problems. Increases in the north-east may be in some instances matched by decreases in the south-west, suggesting a wholesale shift rather than a range expansions (Austin & Rehfisch 2005; Maclean et al. 2006a; Rehfisch et al. 2004). Provided it is shifts rather than range-expansions that are occurring, an increasing proportion of the population is likely to be over-wintering on the Baltic coast in the not too distant future. The tidal range in the Baltic is much smaller than in the North Sea and Atlantic, the areas of exposed mud are also much smaller and often related to wind direction rather than tidal flow. Consequently waders will have less area on which to feed and the availability of resources will become less reliable. Similar northward expansions are predicted on Britain's non-estuarine coasts, but the extent to which this might be detrimental is less clear. Based on the scenarios for Britain's climate in 2020 and 2080, it is predicted that the distributions of the waders will move away from the west. The non-estuarine coasts of Britain hold particularly high proportions of the international flyway populations of Common Ringed Plover *Charadrius hiaticula*, Sanderling *Calidris alba*, Purple Sandpiper *Calidris maritima* and Ruddy Turnstone *Arenaria interpres* that are all expected to show continuing declines in Britain to 2080 (Rehfisch et al. 2004). However, it is possible that these species may be able to make increasing use of coasts in Scandinavia. One area of uncertainty, but certainly a plausible outcome is that waterbirds that currently migrate across the Sahara may increasingly opt to over-winter in areas north of the Sahara. This may be advantageous in terms of reduced mortality whilst on migration, but would almost certainly lead to increased competition with other species already over-wintering on these sites unless these also move northward (Austin et al. 2000).

Several studies have also attempted to model changes in species distributions (e.g. Huntley et al. 2007). Perhaps the most widely used method of modelling the impact of climate change on species distributions is the bioclimatic envelopes approach (Beaumont et al. 2007). This method essentially delineates the upper and lower values of climatic variables based on current distributions and then projects future distributions by assuming that species will occupy the same 'climate space' in the future. Using this approach for example, Berry et al. (2001), Harrison et al. (2001) and Harrison et al. (2001) predict that Red-throated Divers breeding in northern Scotland will lose climate space and decline under the 2050s High scenario, where its population is currently stable (Berry et al. 2001). Conversely however, breeding Eurasian Oystercatchers *Haematopus ostralegus* are predicted to expand into more inland areas in eastern England (Berry et al. 2002; Harrison et al. 2001).This approach has also been extended throughout Europe. For example, Harrison et al. (2006) modelled the availability of suitable climate space across Europe for the distributions of 47 species chosen to encompass a range of taxa and to reflect dominant and threatened species from 10 habitats. The general pattern is of a south-west to north-east shift in suitable climate space, with gains balancing losses for many species. The future distribution of only two waterbird species listed on Annex 2 of AEWA was modelled however: Common Crane *Grus grus* and Corn Crake *Crex crex*. Both species lose more climate space than they gain under all climate scenarios. Common Cranes are predicted to become largely confined to Scandinavia, but not Denmark. Climate, however, is not the only factor determining species' distributions and currently Common Crane populations are increasing in Denmark, possibly through reduced hunting pressure (Harrison et al. 2006). At the European scale, Corn Crakes were the most sensitive of all the species modelled, especially after 2050, losing between 36% and 76% of climate space, disappearing from all of southern and most of central and western Europe. It is also predicted to be highly vulnerable in Italy, as only isolated high mountain areas remain climatically suitable by 2080 (Harrison et al. 2006). Using a broadly similar approach Austin et al. (2001), Austin & Rehfisch (2005) and Maclean et al. 2006a), related current distributions and abundances of waders to current climate and predict continued northward and eastward shift in the distribution of wader populations as numbers at colder sites increase (Austin et al. 2001; Austin & Rehfisch 2005; Maclean et al. 2006a; Maclean et al. in review).

Climate envelope modelling is also the approach taken in the climatic atlas of European breeding birds (Huntley et al. 2007). This atlas predicts that several predominantly Mediterranean or Black Sea species of waterbird such as Squacco Heron *Ardeola ralloides*, Great Egret, Purple Heron, Glossy Ibis *Plegadis falcinellus* Black-winged Stilt. Other species, such as Little Bittern *Ixobrychus minutus*, White Stork, Black Stork *Ciconia nigra*, Ferruginous Duck *Aythya nyroca*, Little Crake *Porzana parva* and Kentish Plover *Charadrius alexandrinus*, associated with central Europe will expand into Fenno-Scandia and Britain. It is Arctic breeders that appear to be most at risk. Many such as Bewick's Swan *Cygnus columbianus*, Barnacle Goose *Branta leucopsis*, King Eider *Somateria spectablis*, Long-tailed Duck *Clangula hyemalis*, Velvet Scoter *Melanitta fusca*, Smew *Megus albellus*, many of the *Calidris* waders and Pintail Snipe *Gallingo stenura* are likely to contract their range considerably due to loss of climate space

However, bioclimatic envelope modelling approaches have their limitations. The climate envelope approach tends to ignore local-scale effects such as species dispersal and local connectivity (del Barrio et al. 2006) and often does not consider land-use, changes in land-use under future climate scenarios and interactions with other species. One of the critical assumptions of this approach is that the limits of a species' range is set by climate variables, but it is actually well known that species distributions are usually limited, at least in one direction by competition with other species (Begon et al. 2003). For the envelopes to realistically represent a species' range, other explanatory variables such as land cover, biotic interactions and dispersal mechanisms need to be incorporated into species-climate models (Heikkinen et al. 2006). The accuracy of climate envelope approaches is not always tested, and uncertainties in the modelling process can have a pronounced effects on the projected distributions of the species modelled (Beaumont et al. 2007; Pearson et al. 2006). Nevertheless, where accuracies have been tested and comparisons with other methods made, the results are generally favourable. Huntley et al. (2004) used climate-envelopes to model the future distributions of 306 European species and by applying the model to fit current distributions and using goodness-of-fit measures to show that useful models were fitted for more 96% of species. Similarly, Thuiller (2003) and Thuiller (2004), compared bioclimatic envelope approaches with other methods of predicting species distributions and found predictions to be broadly similar. However, although these method of testing model suitability assess the extent to which factors other than climate affect species distributions, they do not assess the extent to which current relationships hold true under future scenarios. This is in fact a weakness of all of the commonly used approaches for predicting species distributions and results mainly because insufficient time has elapsed since such model predictions were generated for them to be accurately tested. Particularly with initiatives such as the 3rd BTO Atlas of breeding bird distributions underway, and the third survey of the UK's open coasts completed in 2007, it should be possible to test the validity of these types of approaches more thoroughly in the future.

4.3. Future changes in timing of biological events

General trends towards earlier spring arrival from wintering grounds in Palaearctic-Afrotropical migrants (Gordo & Sanz 2006; Lehikoinen et al. 2004; Sparks & Mason 1999; Sparks 1999) will probably continue. There may be a limit to how much birds can advance their arrival date by, as the cues used to trigger their departure from Africa, such as day-length, are unrelated to climate. The processes driving advancement are twofold: (1) phenotypic plasticity whereby an individual bird modifies its behaviour in response to external cues and (2) micro-evolutionary selection, whereby earlier arriving birds become more prevalent in the population because they survive better than late arriving ones. Lack of relevant cues triggering departure will nullify this first process unless the cues triggering departure in Africa are correlated with those benefiting early arrival, but the second process will drive earlier arrival irrespective of cues.

If birds do arrive earlier, the trend towards earlier breeding (Crick et al. 1997; Crick & Sparks 1999; Drent et al. 2006) is also likely to continue. As birds alter their phenology in response to warming temperatures, it will become increasingly likely that these critical biological events will not be timed optimally. This could occur either because weather conditions are unfavourable (Jovani & Tella 2004) or because the main food source on which chicks feed is advancing their phenology at a faster or slower rate. For example, although temperatures are and will continue to become warmer in early spring at higher latitudes, it may be that temperatures are more variable at this time, subjecting early nesters to sudden cold periods. Similarly, in many areas, rainfall is higher in winter than in summer, so earlier nesting may result in higher mortality due to exposure to rain (Jovani & Tella 2004), particularly if current trends towards higher winter, but lower summer rainfall continue (IPCC 2007b). The young of many migratory waterbirds are insectivores (del Hoyo et al. 1992-2008). Although birds can shift their laying dates earlier in response to warm spring temperatures, they cannot significantly decrease their incubation period. However, the insect prey on which they feed can hatch or pupate earlier and significantly shorten their development time, such that they emerge earlier than the chicks (Buse et al. 1999). Although this is probably a general trend across many species, some insects are predicted to advance their phenology by less than their predators. For example, (Pearce-Higgins et al. 2005) modelled the relationship between advancement in the breeding phenology of Eurasian Golden Plover and the timing of their adult tipulid prey emergence. Climate predictions for 2070-2099 predict that aspects of Eurasian Golden Plover breeding phenology will advance by between 13 and 25 days, whereas tipulid emergence is predicted to advance by only 12 days, thus causing mismatch in timing. It is clear that irrespective of whether birds shift their phenology more rapidly or more slowly than their prey, mistiming will occur, leading to lower chick survival. However, as with shifts towards early migration, mistiming will only occur if evolutionary changes cannot keep pace with climate change and it is phenotypic plasticity that is driving the earlier nesting. The extent to which rapid evolutionary changes can occur and whether this will be able to keep pace with climate change in the future is perhaps the most critical unanswered question in climate-change biology.

4.4. Future changes in predation and parasitism

Increases in temperature are likely to lead to colonisation and range expansion of several species of avian predator and parasite. This is especially true in the Arctic, where already the range of the Arctic Fox *Alopex lagopus* in Canada may have contracted to the benefit of the Red Fox *Vulpes vulpes* (Hersteinsson & McDonald 1992). Moreover it is possible that other wader predators, such as Hedgehog *Erinaceus europaeus*, Badger *Meles meles*, Weasel *Mustela nivalis*, Polecat *Putorius putorius*, Wild Cat *Felis catus* and Lynx *Lynx lynx*, may also start colonising more northerly wader breeding habitats. Already waterbirds have to cope with increasing raptor numbers(Butler & Vennesland 2000) and further colonisation of the far north by Sparrowhawk *Accipter nisus*, a major predator of waders (Cresswell 1994), could significantly increase predation pressure on waders. As conditions get milder in the far north, the present relationship between high lemming abundance and low depredation of wader nests (Underhill *et al.* 1993) may become less clear and waders may be faced by regular predation pressure from a wider variety of predators. Based largely on Piersma (1997), Lindström & Agrell(1999) suggested that the appearance of parasites in the northerly breeding habitats of waders expected with warming conditions could pose a serious threat to the survival of waders, largely as a result of the low immunological investment made by waders.

4.5. Future changes in demography

Current changes in survival in relation to warmer temperatures are likely to continue, although the extent to which this will impact at a population level is uncertain. Overall warmer weather, particularly in winter, seems set to continue to cause lower mortality, but patterns may be complex. For example, if a trend towards warmer winters causes large range shifts, as is occurring in common wader species in north-west Europe (Austin & Rehfisch 2005; Maclean et al. in review; Rehfisch *et al.* 2004), the extent to which warmer temperatures affect birds in this way will depend largely on temperature variability. Since Fenno-Scandia experiences more variable winter temperatures than the UK (the temperature of which is moderated by the gulf stream), short, unexpected cold periods may cause mass mortality (Clark 2004). Similarly any increases or decreases in a particular species, is likely to cause the opposite response in species that compete with that undergoing changes (Begon et al. 2005).

The future effects of high temperatures towards the warmer extremities of species' ranges are less easy to predict. Most species will not be that adversely affected by warmer temperatures directly, simply because most have the thermal-regulatory capacity to cope with the heat. However higher temperatures are likely to cause direct bodily water loss and cause wetlands to dry, thus increasing a bird's susceptibility to drying. Also, warmer temperatures may cause changes in community composition, which are likely to affect the survival of some species due to trophic interactions (Begon et al. 2005).

As with survival, changes in climatic conditions seem set to continue to influence productivity, but in an unpredictable manner. Towards the colder extremities of species' ranges, warmer temperatures are likely to increase hatching and fledging success, but may make species more susceptible to sudden, short-periods of adverse weather which cause high chick mortality (Robinson et al. 2005). In the warmer extremities of a species range, temperatures may directly adversely affect productivity, but this will depend partially on the extent to which prey items are adversely affected, when in fact the converse may be true in some instances (Robinson et al. 2005). The most likely adverse effect will result from increased evapo-transpiration causing wetlands to dry up. As with survival, changes in rainfall may have positive or negative effects on productivity. Higher rainfall may lead to some insect-prey or polychaete becoming less active and may cause chick mortality in some species as the chicks are forced to move through wet vegetation (Pienkowski 1981; Summers et al. 2004). It will however increase the geographic extent of wetlands and may result in improved conditions for other prey species.

4.6. Future effects of changing rainfall patterns

Changes in rainfall may have both beneficial and negative consequences for migratory waterbirds. Lower rainfall may lead to reduced mortality for those species such as Corn Crake or flufftails, which are only partially dependent on wetlands and likely to suffer chilling as they move through damp vegetation, or which feed on insects and other invertebrates that become more active during drier periods (Summers et al. 2004). However, there is likely to be a severe threat to waterbirds faced by changes in water regime. For example, of all migratory birds listed on the Convention on Migratory Species, 84% face some threat from climate change, almost half because of changes in water regime (Robinson et al. 2005). Waterbirds, being the most dependent on water, are likely to face an even greater threat. Historically, it has been trans-Saharan migrants that have suffered the most, predominantly due to the prevalence of droughts, particularly in the late-1960s (Baillie & Peach 1992; Peach et al. 1991; Szep 1995; Winstanley et al. 1974). Although it is uncertain how rainfall in the Sahel, the Guinean Coast and the southern Sahara will evolve this century, rainfall in the Mediterranean region is predicted to decrease dramatically (IPCC 2007b). Thus although the wintering grounds may be unaffected by drought, for some species, the length of the hazardous journey across dry regions during migration is set to increase. This may have major implications for many of the Palaearctic-Afrotropical migrant waterbird species.

Waterbirds are likely to be especially vulnerable to changes in rainfall due to their dependence on wetland habitats, the area of which can change rapidly in response to changing rainfall. Although one might expect that migratory birds are fairly well adapted to small inter-annual variation in rainfall, prolonged periods of drought are likely to be detrimental and prolonged periods of higher rainfall beneficial. Precipitation is generally predicted to increase in the tropics and polar regions, but decrease at mid-latitudes, with large increases in rainfall predicted and observed in East and Central Africa, Greenland and Northern Fenno-Scandia and Siberia for example, but substantial decreases predicted and observed in the Mediterranean basin and Southern Africa (IPCC 2007b). Consequently, species such as White-headed Duck *Oxyura leucocephala* or Slaty Egret, which are primarily confined to wetlands within these regions predicted to get drier (Wetlands International 2006) are likely to be the most vulnerable. Although a major proportion of only a few species are confined to these regions (Table 3), a much larger number of the biogeographic populations listed on Table 1 of the Agreement are associated with these drier regions.

4.7. Future effects of sea-level rise

The direct and indirect effects of sea-level rise on migratory waterbirds are likely to continue. The magnitude of these effects is highly dependent on the extent to which sea-levels will rise. Whilst the (IPCC 2007b) predict only moderate sea-level rises of 18-59 cm during the course of the next century, these predictions have a major caveat that the models used to date do not include the full effects of changes in ice sheet flow (IPCC 2007b). Recent published evidence suggests that sea-level rise could be in the order of several meters by 2100 (e.g. Gregory et al. 2004; Overpeck et al. 2006; Thomas et al. 2004). If this is indeed the case, then there will be major consequences for waterbirds. Not only will coastal nesting birds be much more susceptible to having their nests flooded, but important areas of habitat will be lost. In the UK for example, more than half of the internationally important wetlands

(Ramsar 2007) are situated on the coast or in low-lying areas and are thus threatened by sea-level rise (Stroud et al. 2001).

Table 3. Migratory waterbird species listed in Appendix 2 of AEWA, for which a substantial proportion breeds or winters range in wetlands either Southern Africa / Madagascar or the Mediterranean Basin and/or Caspian Sea area, two regions predicted to dry substantially. Source: Wetlands International (2006)

Species	**Mediterranean / Caspian**		**Southern Africa / Madagascar**	
	Breeding	Wintering	Breeding	Wintering
Dalmatian Pelican *Pelecanus crispus*	Y	Y		
Pygmy Cormorant *Phalacrocorax pygmeus*	Y	Y		
Madagascar Pond Heron *Ardeola idae*			Y	
Slaty Egret *Egretta vinaceigula*			Y	Y
Northern Bald Ibis *Geronticus eremita*	Y			
Lesser White-fronted Goose *Anser erythropus*		Y		
Red-breasted Goose *Branta ruficollis*		Y		
South African Shelduck *Tadorna cana*			Y	Y
Marbled Duck *Marmaronetta angustirostris*	Y	Y		
White-headed Duck *Oxyura leucocephala*	Y	Y		
White-winged Flufftail *Sarothrura ayresi*				Y
Black-winged Pratincole *Glareola nordmanni*	Y			
Madagascar Pratincole *Glareola ocularis*			Y	
Sociable Lapwing *Vanellus gregarius*	Y			
Eurasian Dotterel *Eudromias morinellus*		Y		
Chestnut-banded Plover *Charadrius pallidus*			Y	Y
Caspian Plover *Charadrius asiaticus*	Y			
Whiskered Tern *Chlidonias hybridus*	Y			

Rising sea-levels are predicted to affect the shape of estuaries generally, but particularly those in areas subject to subsidence since the end of the last ice-age, such as in the south-east of England (Harrison et al. 2001). Estuary morphology largely determines intertidal sediments that in turn influences the abundance and availability of invertebrate prey of waterbirds. More extensive intertidal flats are likely to become sandier, improving the habitat quality for species such as Eurasian Oystercatcher, but reducing it for species such as Common Redshank (Austin et al. 2001; Austin & Rehfisch 2003; Rehfisch et al. 2000). As a result of sea level rise, it is predicted that the densities of Eurasian Curlew, Dunlin and Common Redshank will decrease and that the density of Oystercatcher would increase on UK estuaries such as the Deben and Duddon (Austin et al. 2001; Austin & Rehfisch 2003). However, the total numbers of each species of wader that could be held by each of these sites is predicted to grow as a result of increasing estuary area with sea level rise more than compensating for any reduction in densities, provided that future flood-defences do not result in coastal squeeze (Austin et al. 2001; Austin & Rehfisch 2003). Similarly, large numbers of Red-throated Divers over-winter on the coasts of Britain and Ireland, where their preferred shallow inshore waters and sandy bays could be adversely affected by sea-level rise (Berry et al. 2001).

Other modelling approaches predict more universally adverse affects. For example, using individual-based models, the quality of Poole Harbour for five over-wintering shorebirds was assessed, namely Dunlin, Common Redshank, Black-tailed Godwit *Limosa limosa*, Oystercatcher and Eurasian Curlew. All five species are predicted to be seriously adversely affected by simulated sea-level rise (Durell et al. 2006).

4.8. Future indirect impacts of climate change

4.7.1. Future climate change and land-use change

Climate change is likely to lead to some major changes in the way land is used and this will have knock-on effects for migratory waterbirds. Although such changes are highly dependent on agricultural policy, which is hard to predict, there are some broad changes that are likely. For example, cereal production is likely to spread to parts of northern Europe where livestock grazing currently predominates (Hossell et al. 1996; NFU 2005; Parry et al. 1999). Where such areas comprise unimproved grassland, this is likely to have implications for breeding and over-wintering waders and dabbling ducks, and may also have implications for over-wintering geese and swans (Stroud et al. 2001). In warmer areas, double cropping may become feasible, which could have a high impact on ground nesting waterbirds associated with agricultural land such as Northern Lapwings *Vanellus vanellus* (NFU 2005). Changes in land use and management in response to climate change are also likely to include drainage to avoid increased risk of flooding. Increased drainage will be detrimental to many waterbird species (NFU 2005), particularly those, such as herons and egrets, dabbling ducks and breeding waders, which rely heavily on flooded lowland grassland. However, there may also be more crop irregation, which could benefit other species. In northern Europe, warmer temperatures may mean that animals would not necessarily need to be removed from higher ground during the winter. This could benefit waterbirds by reducing grazing on overgrazed lowland but if it increases the grazing intensity on moorland it will lead to a loss of *Calluna* and an increase in grass species, which is likely to be detrimental to upland breeding waders such as Eurasian Curlew (Mitchell et al. 2007; NFU 2005).

Major changes in land-use are likely to occur on coastal grazing marshes as a result of sea level rise. As inundation of these areas increases, they are likely to become less favourable for maintaining livestock. Often the value to waterbirds of such habitats is enhanced by short sward lengths and the presence of pools of freshwater maintained by livestock grazing and trampling (Hart et al. 2002; Norris et al. 1997; Tichit et al. 2005). Reductions in grazing in response to sea-level rise may have detrimental consequences for biodiversity in this habitat. Much, in Europe at least, will also depend on the extent to which environmental orientated agricultural policies are adopted. In England, for example, there is a target for 60% of England's agricultural land to be under the Environmental Stewardship scheme by December 2007. Agri-environment schemes encourage less intensive agriculture, and since the 2007 review, taking account of climate change has become a core objective, although at this stage it is not entirely clear what this means in practice (Mitchell et al. 2007).

4.7.2. Future climate change and water-use change

Climate change is likely to lead to some major changes in the way water resources are used and this will also have major implications for migratory waterbirds. As the climate warms there is initially likely to be an increase in demand for irrigation; this may cause low flow in rivers, over-abstraction of other surface waters, lowering of water tables, leading to degradation of water resources and aquatic ecosystems with general negative effects on biodiversity (CBD 2003; Holman & Loveland 2001; Holman et al. 2005a; Holman et al. 2005b; NFU 2005). As water scarcity increasingly becomes an issue this is likely to cause increased regulation and a decrease in irrigation of agricultural land. There may be an increase in the development of water storage facilities, or reservoirs to supply water for irrigation, which could lead to losses of natural or semi-natural habitats (Mitchell et al. 2007). There may also be increased attention to ditch management, which could be beneficial or detrimental to wildlife, depending on whether the method and timing of management included consideration of biodiversity interests.

Decreasing summer rainfall and increasing demand for water, will reduce water supplies with implications for wetland habitats (Mitchell et al. 2007), and consequently waterbirds. For example, in Europe, it may no longer be possible to maintain appropriate hydrological regimes for the protection of waterbirds in many of the small, scattered, discrete wetland nature reserves and large-scale wetland restoration schemes (Mountford et al. 2004). In Africa, particularly towards the south and near the Mediterranean, where it is predicted to get much drier, large-scale degradation of wetlands seems likely. Decreased availability of water leads to lower agricultural yields, which in turn increases poverty, thus making the farming of marginal land, such as wetlands an increasingly attractive option (Maclean 2004). In the south-west of Uganda for example, severe wetland drainage has occurred for these reasons, as the area has become drier (Maclean et al. 2003b & c) with detrimental effects on waterbirds (Maclean et al. 2003a & c; Mafabi 2000).

4.7.3. Future climate change and flood-defences

Climate change is likely to increase flood risk, both as a result of sea-level rise and due to higher rainfall in some areas. In developed countries, considerable effort is already devoted to dealing with predicted higher water levels and hence greater flood-risk, both within the coastal/estuarine zone and freshwater systems (Ramsbottom et al. 2005). Changes in flood defence that are already under way concentrate on confining excess waters within “safe” bounds. In river floodplains, actions include raising of flood-banks, greater demand for flood-storage and upgraded controls through barrages and sluices. Higher flood-banks ensure greater capacity within the channels and more control over flooding, not only of the agriculture, industry and housing, but also of floodplain wetlands. Unless engineering of the banks includes systems of slackers and more subtle regulation, then the probable impact of raised banks will be to further divorce the river from its floodplain, reducing water inputs to wetlands (Mitchell et al. 2007). However, policies that designate and protect floodplains and washlands can increase the areas available as potential habitat for waterbirds (Environment Agency 2005; OST 2004). Similarly, flood barriers, sluices and flood-banks, advocated to mitigate tidal or flood surges, may provide scope for targeted management that allows wetlands to coexist with intensively-used land, but such a flexible approach requires thorough planning and supervision (Haskoning 2006). For example, the floodplain of the Rijn in The Netherlands has been allowed to revert to more natural state in the past 10 years to increase its capacity to absorb floods, with concomitant benefits to biodiversity (van Geest et al. 2005)

In coastal areas, flood risk management policy could have an important impact on the extent to which coastal waterbirds can tolerate climate change. The impacts of sea-level rise will be minimal if sea level rise is viewed as an inevitable process and coastal areas are re-aligned through management such that there is no overall loss of important habitats (Crooks 2004; Hughes 2004). Should coastal defence works be placed around much of the coast, many areas of important habitat will be lost as they are squeezed between rising seas and hard defences (Rehfisch et al. 2005). In developed countries, managed realignment is an important coastal policy responding to climate change (Cobbold & Santema 2001; National Trust 2005; RSPB 2001). Some freshwater and brackish wetlands of international importance are threatened by sea-level rise and managed retreat, whereby low-lying coastal areas are allowed to be flooded. Two of the most important coastal habitats in Europe for waterbirds are brackish lagoons and areas of coastal grazing marsh containing pools of freshwater used by breeding and feeding waders (BirdLife International 2007; Maclean & Austin 2006; Rehfisch et al. 1994). With the presence of appropriately designed flood defence works, water-level and salinity regimes could be controlled to enhance their attractiveness to waterbirds.

4.7.4. Future climate change and recreational disturbance

In some colder areas, recreational tourism is likely to increase because of climate change.
This could lead to increased visitor pressure on water bodies, rivers and wetland habitats of importance to waterbirds. Coastal areas are likely to be amongst the worst affected. In additional to increased recreational disturbance, the longer tourism season, increased tourism infrastructure (i.e. hotels, attractions, marinas), increased waste (i.e. sewage, solid waste) and increased destruction of natural habitat (Coombes et al. 2005; Viner 2006), is likely to have a detrimental impact on waterbirds. Bird species such as the ground-nesting coastal Common Ringed Plover are likely to be amongst the worst affected due to increased recreational disturbance (Liley 1999). However, there are

also potential benefits such as an increased educational awareness and improved perceptions and attitudes to coastal and wetland environments (Brown & McLachlan 2002; Thompson et al. 2002).

5. POSSIBLE MEANS OF ADAPTING TO CLIMATE CHANGE

5.1. Site management

One of the most effective means of helping waterbirds to adapt to climate change, particularly in developed countries where sufficient resources are likely to be available, is active site management. For example, the character of a particular wetland is affected predominantly by its hydrology and by the dominant plant species present. Most plant species can survive at warmer sites than they typically occur in naturally (as can be observed in botanic garden collections of alpine plants). In natural conditions they are excluded by competition with species that grow faster and larger at higher temperatures (Morecroft & Paterson 2006). In some cases reducing competition may allow threatened species to persist, for example, by changing grazing regimes, preventing spread of competitor species or even direct removal of competitors in the vicinity of particularly threatened individuals. Manipulating microclimate, by modifying vegetation height or canopy structure is another option for manipulating invertebrate communities that constitute waterbird prey (Mitchell et al. 2007).

There are limits to the extent to which management can compensate for increasing temperatures, and such manipulations may also have knock-on effects such as increasing the risk of waterbird nest predation. However, particularly in human-altered landscapes, which dominate most developed countries and increasingly developing countries, water supply can be manipulated through changing drainage or damming which may offer potential to offset some of the effects of drier summers. Approaches like these will usually need to be considered and implemented at a local level, with an element of trial and error, at least for the immediate future (Mitchell et al. 2007). The is more scope to manipulate coastal sites at risk from sea-level rise, particularly in developed countries. Key habitats could be manipulated to control the amount of sea-water inundation are managed in such a way that priority habitats at risk of inundation, such as salt-marsh, are maintained through managed realignment (Mitchell et al. 2007).

5.2. Protected area networks

If species approaching their climatic limits cannot adapt to the new climate and cannot be maintained in their present locations by management, they will only survive if they move into new areas where the climate is suitable. Thus facilitating species dispersal is a valuable means of helping species to adapt to climate change. One of the most effective means of assisting species dispersal is to ensure that there is an adequate network of protected areas. Although the extent to which species can disperse through landscapes is partially dependent on the existence of small wetlands, ditches and rivers, the protection of a network of wetlands relatively close to each other is an essential means of adapting to climate change as not only will it aid species dispersal, but it will help ensure that species displaced by unsuitable climate space have somewhere to go (Mitchell et al. 2007).

To give due consideration to climate change, the identification of new wetlands to protect should follow four broad criteria (in addition to the current criteria such as the number of waterbirds present): (1) they should be situated towards the colder limits of species ranges they are designed to facilitate the protection of, (2) they should be targeted to areas where they can contribute most to developing an ecological network, for example by being situated between two existing wetlands (Catchpole 2006), (3) they should be located in areas predicted to become drier, but likely to remain within relevant flyways and (4) they should be located in areas which are towards the start or finish of long and arduous migration journeys such as to the north and south of the Sahara. Perhaps the biggest barriers to such common sense approaches are the legislative frameworks surrounding wetland protection. Although legislation such as the Ramsar Convention and EC Habitats and Birds directives are specifically designed to ensure a strategic, international approach to wetland conservation, the systems for designating sites tend to concentrate on waterbird numbers at the time of designation rather than on likely numbers in the future. Thus recent colonizers to a country, such as Little Egrets in the UK, often do not form designated features of any site within that country (Stroud et al. 2001). This is inevitable given that there is a need to rely on actual rather than predicted data, but highlights the need for a regular review of designated features and areas.

5.3. Management of the wider countryside

Although networks of protected areas provide one means of aiding species dispersal and ensuring adequate availability of habitat in areas likely to be colonized by waterbird species in the near future, perhaps a more effective means of aiding species dispersal is to ensure a ‘permeable landscape’, especially for species that have dispersed distributions and do not normally gather in large congregations. Such connectivity also increases resilience by allowing recolonisation following local one-off extinctions (for example, from fire or a pollution incident) and promotes outbreeding (Mitchell et al. 2007). The concept of connectivity across landscapes is well accepted but the best means of determining and achieving it will depend on specific circumstances and the waterbird species of interest. Three main strategies have been proposed to increase connectivity: ‘corridors’, ‘stepping stones’ and improving the quality of the matrix in which habitat patches are found. Corridors are linear and would directly link wetlands. Natural features such as ditches form corridors, as would river margins provided that they are vegetated and not hard engineered for flood management (Catchpole 2006). Stepping stones would be small areas of wetland or seasonal floods within otherwise unsuitable habitat, which reduce the distance between larger areas of a habitat. The matrix surrounding semi-natural habitat patches is important in promoting the dispersal of species between patches (Baum et al. 2004; Castellon & Sieving 2006; Catchpole 2006; Donald & Evans 2006). In practice, this approach is likely to include protection of any small wetland feature and a large-scale strategic approach to managing water-levels and drainage so that landscapes become more suitable for waterbird species.

In developed countries one of the best means of achieving appropriate management of the wider countryside would be to integrate such management into other policies that affect the use of the wider countryside, such as into agri-environment schemes, the EU Water Framework Directive (2000/60/EC) or into flood management plans (Mitchell et al. 2007). In developing countries, there is also scope to such management into other land-use policies. However, such policies are often not implemented effectively due to inappropriate governance and are often highly sectoral or conflicting (Bennun & Njoroge 1999). Furthermore, since livelihoods are more closely linked to natural resources, people living in the vicinity of wetlands often exercise more power than governments over the use of wetland resources (Gichuki 2000; Maclean 2004; Maclean et al. 2003b & c) and thus have greater influence over the way wetlands are used. Thus integrating local people’s needs into wetland conservation policy by seeking win-win situations is likely to be the most effective means of ensuring the conservation of wetland complexes at a landscape scale. There is considerable scope for doing this, as the types of policies which benefit waterbirds also benefit people (Maclean 2004; Maclean et al. 2006c & submitted). One of the major causes of wetland drainage in East Africa for example is poverty, as wetland drainage is limited by labour costs associated with draining the deeper wetlands. If people are poorer, they value their time less and are more inclined to engage in labour intensive activities that bring only minimal benefit (Maclean 2004; Maclean et al. 2003b & c). Policies that alleviate poverty and promote a diversity of income sources are likely to reduce drainage and thus improve the outlook for waterbirds in a changing climate (Maclean 2004).

5.4. Minimising other impacts

Adaptation to climate change should not be pursued in isolation from other pressures and the reduction of other pressures can increase the resilience of populations and communities against the impact of climate change (Hulme 2005). Populations and communities already threatened by one pressure are almost always more vulnerable to others. Some of these pressures are relatively well understood, subject to less uncertainty than climate change and can be addressed more directly than climate change impacts themselves. Some interactions between climate change and other pressures can have effects that are greater sum of the parts. For example, habitat loss and degradation will compound the effects of climate change by reducing the extent to which species can disperse (Mitchell et al. 2007).

Thus one of the major means of adapting to climate change will be to prevent wetland drainage and degradation. This will buffer waterbird species against hazards such as prolonged periods of drought or other climatic factors that might reduce population size. It will also ensure that species can disperse adequately as temperatures increase. Although birds are mobile, the prey items on which they feed may be less mobile and less able to disperse as the climate becomes unsuitable within the area they currently inhabit. This is likely to be most necessary for species such as Lesser Flamingos *Phoenicopterus minor*, which have specialized diets.

6. SPECIES ESPECIALLY VULNERABLE TO CLIMATE CHANGE

6.1. Criteria used to assess vulnerability

We use six criteria to establish the vulnerability to climate change of all species listed on Annex 2 of the Agreement and all populations listed on Table 1 of the Agreement. The approach is broadly based on that used by the IUCN to assess risk of extinction (IUCN 2007). The criteria were: population size, range size, area of occupancy as a measure of the extent to which meta-populations are fragmented, the susceptibility of its preferred habitat to climate change and the extent to which it has a specialist diet. Although we recognise that numerous other factors may cause a species to be at risk from climate change, many of this are likely to be highly species-specific and undocumented in the available literature. Our criteria for assessing vulnerability are thus based on information that was readily available for all of the species and populations listed in the Agreement.

Scores were assigned to each category as explained below and each factor was considered to add to a species' total vulnerability to climate change, so that the combined score was calculated by adding the relevant scores from each category as follows:

Total vulnerability score = Population score + range score + fragmentation score + habitat score + food score

Scores and weightings of each category were chosen such any given value in one category was equally likely to increase a species' probability of extinction as the same value in another category, as assessed using the authors' expert opinion. Justification and the precise score assigned for each category are given below. It should be borne in mind however, that the combination of factors in this way, and indeed the selection of the factors to use in the first place is fairly subjective. Consequently we suggest that our index is interpreted with a degree of caution. The vulnerability index is intended to be advisory so that, subject to interpretation, they can be used as a consistent basis on which to direct hypothesis future research and subsequent conservation efforts if required

6.1.1. Population size

Small populations were considered to be more vulnerable to climate change as such populations are inherently more vulnerable to extinction as a result of stochastic events such as drought (Begon et al. 2005) and are less likely to survive if adverse conditions drive them below a certain threshold (Begon et al. 2005), with the probability of extinction approximately inversely related to population size logarithmically transformed (Begon et al. 2005). Thus, scores were assigned as indicated in Table 4. Population estimates were derived from Wetlands International (2006) for waterbird species and from BirdLife International (2007) for the few species and populations listed on Annex 2 and Table 1 not covered by Wetlands International (2006).

Table 4. Contributing climate vulnerability score as a result of population size.

Population size (number of individuals)	Score
1-10	10
11-50	9
51-100	8
101-500	7
501-1,000	6
1,001-5,000	5
5,001-10,000	4
10,001-100,000	3
100,001-1 million	2
More than 1 million to 10 million	1
More than 10 million	0

6.1.2. Range score

Species with small ranges are on average more likely to be vulnerable to climate change as often the reason for their small range is that they are only able to survive within a rather narrow range of climatic conditions (Julliard et al. 2004). Indeed this is the fundamental assumption under-pinning climate-envelope analysis. Whilst we recognise that ranges are often restricted by interactions with other species or environmental conditions other than climatic variables, the precise limitations are rarely known, and assessment of these from the literature or otherwise was considered beyond the scope of this study. Nevertheless, at least in many instances, a small geographical range is likely to increase a species' vulnerability to climate and consequently, we assigned scores as listed in Table 5. As climatic requirements usually differ between the breeding and non-breeding seasons, we considered the breeding and non-breeding range separately and assigned a score according to which ever was smaller. Range sizes were taken from maps depicted in Wetlands International (2006) for most species and del Hoyo et al. (1992-2008) for the few seabirds listed on Annex 2, but not covered in Wetlands International (2006). The maps in Wetlands International (2006) are in any case derived from del Hoyo et al. (1992-2008). Range size was considered to be the area of the minimum convex polygon surrounding all records and categories assigned by visual estimation, backed up by plotting distributions using ArcView in instances where there was doubt as to which category should be assigned.

Table 5. Contributing climate vulnerability score as a result of range size.

Range size (km^2)	**Score**
1-25	10
26-100	9
101-625	8
626-2,500	7
2,501-10,000	6
10,001-62,500	5
62,501-250,000	4
250,001- 1,000,000	3
1,000,000-6,250,000	2
6,250,001-250,000,000	1
More than 250 million	0

6.1.3. Fragmentation score

Species and populations with highly fragmented ranges are likely to be more susceptible to climate change (Harrison et al. 2006). Migration between populations is generally more inhibited if such populations are isolated from one another (Begon et al. 2005), and each isolated population is likely to be more vulnerable to stochastic extinctions that may result from climate change, such as drought - induced deterioration of a particular wetland or series of wetlands (IPCC 2007a). As an approximate measure of this threat, we assigned each species a score based approximately on the species' area of occupancy as defined by IUCN (2007). The area of occupancy is essentially the sum of all individual occupied areas that make up a species' range. In general, the area of occupancy relative to total range of a species decreases as species or populations becomes more fragmented. As measures of a species' area of occupancy are highly dependent on the spatial scale at which they are measured, choosing a consistent scale across species is necessary. Since most areas of occupancy were assigned to categories using visual estimation from maps in del Hoyo et al. (1992-2008) and Wetlands International (2006), we used the pixel resolution of these maps as our unit of scale. In instances where scores could not be assigned with reasonable certainty, we used ArcView to calculate the precise area of occupancy. Scores were assigned as indicated in Table 6.

Table 6. Contributing climate vulnerability score as a result of fragmentation.

Area of occupancy (km^2)	Score
1-25	5
26-100	4.5
101-625	4
626-2,500	3.5
2,501-10,000	3
10,001-62,500	2.5
62,501-250,000	2
250,001- 1,000,000	1.5
1,000,000-6,250,000	1
6,250,001-250,000,000	0.5
More than 250 million	0

6.1.4. Habitat score

Species associated with habitats particularly vulnerable to climate change are more likely to be vulnerable to climate change themselves. We established the favoured breeding or non-breeding habitat of each species from del Hoyo et al. (1992-2008) and then assigned a habitat vulnerability score based partially on the extent to which particular habitats were at risk, but taking into consideration the degree of habitat specialisation. Although the exercise was somewhat subjective, with each species and population treated on its own merit and without precise quantification, the broad reasoning behind the approach was that the average risk to its habitat corresponded to high, medium, low and negligible risk, although generally a higher score was assigned to species with more specialist habitat requirements. High risk habitats (score = 6) are those very vulnerable to future climate change, such as seasonal and ephemeral wetlands in South Africa or the Mediterranean Basin, which are predicted to dry. The narrow strips of Arctic tundra located at the northern extremities of land masses and high altitude vegetation, which will almost certainly alter significantly as temperatures increase, were also assigned to this category. Medium risk habitats (score = 4) were considered to be seasonal or ephemeral wetlands situated outside areas predicted to get drier on average, but predicted to experience more variable rainfall conditions. Permanent, large wetlands located in the Mediterranean and South Africa were also considered medium-risk as was Tundra and Taiga generally and areas prone to coastal squeeze, such as salt marshes. Low risk habitats (score = 2) were permanent, large wetlands located outside the Mediterranean or South Africa, grassland and most other habitats. Negligible risk habitats (score=0) were those significantly altered by humans and this score was generally assigned to species such as Common Mallard *Anas platyrhynchos*, capable of occupying a wide variety of habitats, including those significantly altered by humans. Species were assigned separate scores for their breeding and non-breeding habitats if these differed, and the highest one taken as the actual score.

6.1.5. Food score

Species with more specialist food requirements will almost certainly be more vulnerable to climate change, as it is unlikely that species will alter phenology or shift their range in precisely the same manner as their prey (Visser & Both 2005). We established the degree of diet specialisation by referring to del Hoyo et al. (1992-2008) and then assigned a food vulnerability score based on the degree of specialisation. High scores (6) were assigned to species with a very specialist diet, such as Cape Gannet, which has a diet restricted to fewer than 10 species of fish, with one or two species contributing to more than 50% of the diet by weight. Medium-scores (4) were assigned to species which had a fairly varied diet, but generally confined to one group of organisms, such as tubiferous plants or fish. Low scores (2) were assigned to species who's diet was varied, but were nevertheless either carnivorous or plantivorous, rather than omnivorous. A score of zero was assigned to species whose diet was truly omnivorous.

6.2. List of vulnerable species

Species with a total score greater than 20 were considered critically threatened by climate change. Those with a score between 17.5 and 19.5 (inclusive) were considered highly threatened and those with a score between 15 and 17, moderately threatened. If the total score was between 10 and 15, species were considered to be experiencing some threat from climate change and those with a score lower than ten only minimal threat from climate change. A full list of the scores assigned to each species is given in Appendix 1. Those species critically or highly threatened are listed in Table 7.

Table 7. Assessment of vulnerability of all species listed on Annex 2 of the AEWA Agreement which are either critically threatened by climate change (🟥) or highly threatened by climate change (🟧).

Common name	**Species**	**Population size score**	**Range size score**	**Fragmentation score**	**Habitat score**	**Food score**	**Climate vulnerability score**
Cape Gannet	*Morus capensis*	2	2	3.5	6	6	19.5
Crowned Cormorant	*Phalacrocorax coronatus*	4	4	2	4	4	18
Bank Cormorant	*Phalacrocorax neglectus*	4	6	3	4	2	19
Slaty Egret	*Egretta vinaceigula*	5	4	2	6	2	19
Northern Bald Ibis	*Geronticus eremita*	7	4	3.4	4	0	18.5
White-winged Flufftail	*Sarothrura ayresi*	6	3	4	6	2	21
Madagascar Pratincole	*Glareola ocularis*	2	2	3.5	6	6	18.5
Slender-billed Curlew	*Numenius tenuirostris*	9	3	2.5	6	2	26.5
Damara Tern	*Sterna balaenarum*	3	3	1.5	6	4	17.5

6.2.1. Cape Gannet

The Cape Gannet is primarily confined to Africa, breeding on just six offshore Islands in South Africa and Namibia between 24°38'S and 33°50'S: Bird (Lambert's Bay), Malgas and Bird (Algoa Bay), Mercury, Ichaboe and Possession (BirdLife International 2007; Brown et al. 1982). However, it has recently colonised Australia, where it was first recorded in 1981 and 11 –16 pairs now breed. It may also have colonized Iles Amsterdam and St Paul (Marchant & Higgins 1996). Adults are fairly sedentary ranging within 500 km of breeding colonies. Young up to 2 years and less are more wide ranging, migrating up to the Gulf of Guinea and a few to Mozambique, occasionally as far north as Kenya, but usually within 100 km of land. Individuals show a very high fidelity to colonies and even to nest sites (Brown *et al.* 1982). In 1996, the global population numbered c.173,000 breeding pairs. The breeding population decreased by 31% between 1956 and 1996 and by 15% between 1978 and 1996. The colony at Possession Island now numbers only 750 pairs, and may soon be lost (BirdLife International 2007). Its preferred fishing haunts are the Benguela and Agulhas currents, seldom beyond the continental shelf. Its diet is very specialist, comprising mainly of pilchards *Sardinops ocellata* (51%), maasbankers *Trachurus trachurus* (20%) and anchovies *Engraulus capensis* (12%), with small amounts of other fish and cephalods taken on occasions (Brown *et al.* 1982), although BirdLife International (2007), suggest that sardine *Sardinops sagax*, saury *Scomberesox saurus* and offal discarded by fishing boats are also food sources, suggesting that diet may vary according to availability. Threats other than climate change include food shortage following the collapse of the Namibian sardine fishery, oil-spills, guano collection and displacement by Cape Fur Seals *Arctocephalus pusillus*.

This species is mostly threatened by climate change because of its specialist diet, the small number of locations at which it occurs and preferred foraging habitat. As waters off the coast of South Africa warm, there will be limited opportunity to move poleward because there is minimal land available for nesting between South Africa and the Antarctic Continent, posing a major problem for this species if its food cannot survive warmer temperatures. Seabirds are notoriously sensitive to climate change-induced changes in ocean currents and food availability (Aebischer et al. 1990; Viet et al. 1997). Massive declines and movements in many seabird species, especially apex predators occurred as a result of the changes in the California current, with a 90% local decline in Sooty Shearwater *Puffinus griseus* for example (Viet et al. 1997). Large changes in the Benguela and Agulhas currents are quite likely to occur in response to melting of the Antarctic Ice sheet, changing wind directions and altered water temperatures. The high site fidelity exhibited by adult Cape Gannets and their predisposition towards feeding relatively close to breeding colonies make this species extremely vulnerable to climate change, perhaps more so than the index suggests, although the recent colonisation of Australia might suggest they can shift their range to some degree. Future climate-orientated research on this species should focus predominantly on how its main food sources are likely to change in relation to climate change. In so doing, it would also be important to establish how readily this species switches prey in relation to availability. This is likely to be particularly important during the breeding season, when the distance that adults can feed away from the colony, may be constrained by the need to feed young. Another research priority would be to consider the array of intervention-management options, such as providing artificial nesting platforms or supplementary feeding, which could be adopted should climate change drive this species close to extinction.

6.2.2 Crowned Cormorant

The Crowned Cormorant is restricted to the west coast of southern Africa, breeding at 48 localities from Walvis Bay in Namibia to Cape Agulhas in South Africa (BirdLife International 2007; Brown et al. 1982; Harrison et al. 1997;). During 1977-1981, the population was estimated to be 2,665 breeding pairs. The most recent estimate is of 8,700 individuals (BirdLife International 2007). Although not comprehensive, available information suggests that the population is stable or increasing. Crowned Cormorants breed among mixed seabird colonies, rocky outcrops and islands and forage in shallow coastal waters and estuaries, favouring those washed by cold water (Brown et al. 1992; Harrison *et al.* 1997). They tend to feed on slow-moving benthic fish, particularly clinids, pipefish and sol and also on shrimps and isopods (Brown *et al.* 1982). Threats other than climate change include human disturbance, to which the species is very susceptible. Mortality due to entanglement in fishing-line at nests is also a concern (BirdLife International 2007).

The main threats from climate change to this species arise because of its specialised diet the habitat with which it is associated. As with the preceding species, warming waters off the coast of South Africa could pose a major problem for this species if its food cannot survive warmer temperature, as there is limited opportunity to move poleward because there is minimal land available for nesting between South Africa and the Antarctic Continent. Future research should thus focus on establishing the extent to which its feeding habitat is likely to alter under future climate scenarios, particularly with respect to the relative abundance of actual and potential food sources. Some of this species' favoured nesting locations are likely to be in low-lying areas and the species may thus be threatened by sea-level rise. Establishing the extent to which this is the case and assessing the need for adaptive management should also be a research priority.

6.2.3. Bank Cormorant

The Bank Cormorant is endemic to Southern Africa, breeding at 45 localities. It is a colonial nester, usually occupying the tops of small isolated pinnacles of rock or rounded granite islands, often far out to sea, between Swakopmund in Namibia to Cape Agulhas in South Africa (BirdLife International 2007; Brown et al. 1982). The non-breeding range extends from just south of Hoanibmond south to Die Walle. Several island populations in the west and north Cape Province have declined in recent years and seven former breeding localities have been vacated. The total number of breeding pairs fell from 8,672 in 1978-1980 to 4,888 in 1995-1997. Most losses during this period were from Ichaboe and Mercury Islands off Namibia, which previously supported 68% of the global population. The population is now believed to be 3,704 breeding pairs, with recent increases on Mercury Island due to removal of Cape Fur Seals. It occupies the same nesting sites year after year, almost always immediately adjacent to the sea, where high seas wash over them necessitating rebuilding. It rarely strays far from colonies, is marine, and forages among kelp *Ecklonia maxima* beds close to and rarely more than 10 km from shore. Indeed, its distribution broadly reflects that of kelp. Juveniles disperse up to 450 km along the coast, whereas adults are more sedentary and have not been recorded to move more than 150 km (BirdLife International 2007). The diet consists primarily of fish (especially Pelagic Goby *Sufflogobius bibarbatus*, crustaceans and cephalopods such as Cape Rock Lobster *Jasus lalandi* (Brown et al. 1982; BirdLife International 2007). Threats other than climate change include disturbance (which resulted in the loss of four colonies, and reductions in the populations at six others, between 1978 and 1997 (BirdLife International 2007), displacement by Cape Fur Seals, decreased food availability, oil spills, habitat destruction, both through guano collection and coastal developments and alien mammalian predators (BirdLife International 2007).

As with the preceding species, the main threats from climate change to this species arise because of its small population size, its small range and the fragmented nature of its breeding colonies. As waters off the coast of South Africa warm, there is no opportunity to move poleward because there is no land available for nesting between South Africa and the Antarctic Continent, posing a major problem for this species if the kelp beds on which it depends cannot survive warmer temperatures. Its predisposition for high nest-fidelity and for nesting close to the water may make it vulnerable to sea-level rise, although it is quite likely that the birds could modify their behaviour to respond to such changes. However, if some of its favoured nesting islands have unsuitable habitat further above the water or if the entire island is susceptible to flooding, then there may be a fairly serious risk posed by nest flooding in response to sea-level rises. Consequently future research should focus on how the availability of habitat for nesting will change in response to sea-level rises. Given this species' strong dependence on kelp beds and minimal scope for poleward range shift, future research should also focus on how susceptible the northern margins of kelp distribution are to warming temperatures. As with the two preceding species, another research priority would be to consider the array of intervention-management options, such as providing artificial nesting platforms or supplementary feeding, which could be adopted should climate change drive this species close to extinction.

6.2.4. Slaty Egret

Slaty Egrets are endemic to Southern Africa, where they have a restricted distribution in marshes and flood plains of the Okovango delta, Chobe National Park (Botswana), Caprivi Strip (Namibia), Kafue Flats, Zambezi River north to Bengweulu swamps and Nchelenge District (Zambia) and Mozambique (Zambezi Delta). It has been extirpated from South Africa as a breeding species, but occasionally wanders there and to the Democratic Republic of Congo, Zimbabwe and possibly Angola (Brown et al. 1982; BirdLife International 2007). The species is nearly always encountered in small numbers, suggesting that the world population is in the order of 3,000-5,000 (BirdLife International 2007; Delany and Scott 2007).

Its breeding habitats are poorly known, but breeding appears to be irregular, coinciding with high flood-levels. Its preferred feeding habitat appears to be extensive shallow inundation zones, especially where water level is falling, often foraging where there is tall grass to conceal birds. It does not frequent open water, but appears to undergo seasonal movements in relation to rainfall and will breed at temporary wetlands (Brown et al. 1982; BirdLife International 2007). Its diet comprises mainly small fish, but probably also takes dragonflies and occasionally snails (Brown et al. 1982). Threats other than climate change include flood regulation, water abstraction, land-claim for agriculture, reed-cutting, fire, rice production and tourism development, which affect its habitat. Also, breeding success is erratic, and can be significantly affected by human interference, poor floods and predation of nests and adults (BirdLife International 2007).

Its main threat from climate change is because of its small population size, limited range and vulnerability of its habitat. Its dependence on shallow flood plains and seasonal wetlands make it particularly vulnerable to the drier temperatures predicted for Southern Africa, not only because such wetlands will dry up and reduce in size, but because they may be threatened by increased water abstraction or wholesale conversion in response to agricultural demand. For example, the Okovango delta, the stronghold for these species, has a very large catchments area and changes considerably in size in response to changes in rainfall (IPCC 2007a). The close link between breeding success and rainfall may also make it especially vulnerable to drought. Thus future research should focus predominantly on its precise habitat requirements, particularly in relation to water-levels and how the availability of this habitat is likely to change in relation to changes in rainfall.

6.2.5. Northern Bald Ibis

This species formally bred in Morocco, Algeria and Turkey, with one breeding record from Egypt in 1921 (Brown et al. 1982) and was fairly widespread in North Africa as a non-breeder. In North Africa, this species now only survives in Morocco at Souss-Massa National Park (338 km^2; three colonies) and at nearby Tamri (one colony, almost half the breeding population)[2], with some movement of birds between these two sites. However, in contrast with the now extinct populations in Turkey and elsewhere in Morocco, the birds are not migratory and present in Souss-Massa National Park throughout the year although there are reports of it in Mauritania as a non-breeder. The wintering quarters of Turkish population were never known with certainty and tended to be erratic, occurring periodically in small numbers in Sudan, Eritrea and Ethiopia and perhaps in Somalia, Saudi Arabia and Yemen, with the most recent wintering record being of three adults in February 1997 in the Massawa area of Eritrea (Brown et al. 1982; BirdLife International 2007). The Turkish colony is now heavily managed; with birds taken into captivity after the breeding season to prevent them from migrating. A colony of three pairs and one adult was recently discovered in Talila, Syria and further breeding colonies may exist across the Syrian Steppe, although searches in 2003 proved fruitless. Post-1989 records in Saudi Arabia and Eritrea suggested that there was an undiscovered breeding colony remained in the Middle East, but whether in addition to the Syrian colony is uncertain (BirdLife International 2007). Several adults from the population in Syria have recently been satellite-tracked and the wintering grounds in the Ethiopian Highlands have been identified (RSPB 2008). En-route they pass through Saudi Arabia and Eritrea and it is likely that at least some of the records during the1990s in these regions are of birds from this Syrian breeding colony. In 1994, the Moroccan population was estimated at 300 individuals (59 breeding pairs). In 1998, it had declined to c.200 birds, following the mysterious death of 40 birds in 1996. In 1999, the population had increased slightly, and by 2006 there were around 277 adult birds, of which 102 pairs made nests (92 pairs laid eggs) (Armesto et al. 2006; BirdLife International 2007). Importantly, since 1980 there has been no overall decline in numbers at Souss-Massa NP. Growing numbers and good productivity in recent years (over 500 birds in the Moroccan population after the breeding season in recent years) gives cause for optimism that former colonies may soon be recolonised. The Turkish population, which is heavily managed, now numbers 86, and may have risen to 100 in 2006. The Syrian population has suffered a severe population decline in the past 30 years, and numbered just five adults in 2005 (Armesto et al. 2006; BirdLife International 2007).

Historically, its favoured breeding habitat was on cliff ledges or in caves on inland water courses, or among boulders on steep slopes and on top of old buildings, preferring shaded areas (Brown et al. 1982). Its preferred feeding habitat is fairly varied: it can be seen along dry river beds, rocky slopes and ledges, cultivated fields, high meadows, pastures and valleys of mountain streams and semi-arid ground with little vegetation. Its food is varied, but comprises mainly crickets, grasshoppers and locusts, with earwigs, beetles, ants, woodlice, spiders, scorpions, molluscs and occasionally frogs, tadpoles, lizards, fish, rodents, birds and mammals (Brown et al. 1982). Mortality during migration appears to be high among the Syrian birds; only one recruit joined the colony in 2004, out of 14 fledged and migrated juveniles. Breeding performance is highly variable from one year to the next but does not appear to be related to rainfall in the vicinity of the colonies (BirdLife International 2007). It is suggested that coastal fogs in the Souss-Massa region may buffer the adverse impacts of low rainfall and may in part account for the year-round residency of the birds. The Turkish population has been shown to be genetically distinct from the Moroccan population. Threats other than climate change include human persecution, especially hunting, but also the loss of steppe or unintensive agricultural areas (particularly in Morocco), pesticide poisoning, human disturbance, and dam construction. At Souss-Massa NP, the most recent causes of breeding failures have been loss of eggs to predators and, more importantly, poor chick survival as a result of starvation and predation (Armesto et al. 2006; BirdLife International 2007).

It is primarily identified as being at risk from climate change due to its very small population and area of occupancy. Beyond these factors, there is little to suggest that climate change adversely affects this species, except perhaps of the Syrian population, where high mortality may be a result of adverse rainfall conditions. It is thus perhaps less threatened by climate change than the vulnerability index would suggest. Future climate-related research should however focus on whether lack of rainfall and concomitant food shortage is a cause of mortality in the Syrian population both on breeding grounds and on wintering grounds. Beyond that, climate-related research initiatives are inline with those already ongoing and should focus on establishing whether undiscovered colonies persist in the Middle East and on the capacity for new colonies to be colonised in North Africa.

6.2.6. White-winged Flufftail

Very little is known about this species, but it is thought to be an endemic resident and intra-continental migrant to Africa. It has at least two disjunct populations. One in Ethiopia, where it is known from two sites in the central highlands and is the only known breeding area from this species. It was rediscovered in South Africa in the early-1980s in two marshes, where prior to that it was only known from four old specimens (Urban et al. 1986) and is now known from nine sites (BirdLife International 2007). There are also records from Zimbabwe (1988 and two records in the 1970s), where its status is unknown. Claimed records from Zambia and Rwanda are unproven (BirdLife International 2007). It is not known where the Ethiopian breeding population spend the non-breeding season, but possibly in SW Ethiopia rather than in South Africa. In South Africa, the total population is estimated to be 235 birds. In the Ethiopian highlands, 10-15 pairs have bred at Sululta annually since 1996 and c.200 pairs were discovered at a new breeding site in 1997 (BirdLife International 2007). It is probably migratory or nomadic, in search of transiently suitable habitat. Birds in Ethiopia are present (at 2,200-2,600 m) between June and October, while non-breeding birds in South Africa are present (mostly at 1,100-1,900 m) from November to March. Whether a single population migrates between these two countries, or each country hosts its own subpopulation, is not yet known (BirdLife International 2007)**.**

Very little is known about its breeding habits, as no nest or eggs have yet been found (Urban *et al.* 1986). Feeding habitats are also poorly known. In Ethiopia it was formally present at a small marsh with rushes and marsh orchids and ankle deep water and at a flat grassy plain partly inundated during the rainy season. In Zimbabwe, it is known from marsh with taller vegetation up to 1 m rank grass interspersed with *Cyperus* species. In South Africa it prefers long grass with reeds and bullrushes (Urban et al. 1986). BirdLife International describe it as inhabiting short, dense marsh vegetation, shallowly flooded in the wet season and dominated by sedges, grasses and forbs. Its diet is not known, but probably consists mainly of aquatic insects (Urban et al. 1986). Threats other than climate change include drainage of seasonal marshes for cultivation and forestry, flooding by dams, catchment erosion, water abstraction, human disturbance, too-frequent burning, and excessive trampling and grazing by livestock and cutting of marsh vegetation for fodder. The two Ethiopian sites are on state-run farms which are about to be privatised, which may lead to detrimental modification or drainage (BirdLife International 2007).

This species is primarily identified as being threatened by climate change because of its small population and area of occupancy. Its habitat is also susceptible to climate change. At its Ethiopian breeding grounds, it may be vulnerable to warming, as it inhabits high altitude wetlands, and the topography at higher altitudes may be such that there are no suitable wetlands. It should be a research priority to establish whether this is indeed the case. Its habitat is also likely to be highly threatened by changes in rainfall regime, particularly in South Africa, which is predicted to be drier in the future (IPCC 2007b). Seasonal wetlands are particularly vulnerable to lower precipitation, not only because such wetlands dry up completely or reduce in size, but because they may be threatened by increased water abstraction or wholesale conversion in response to agricultural demand. Establishing the extent to which this species can respond to changes in rainfall by moving between wetlands and the likely extent and locations of suitable wetlands should also be a research priority. In order to achieve this, more precise information on its habitat is required, as at present, even basic knowledge of its current movements is lacking. Establishing whether the Ethiopian population is genetically distinct from that in South Africa, would be a useful first step in addressing this issue. Additionally, baseline surveys are needed to establish the true status and distribution of this species.

6.2.7. Madagascar Pratincole

This species breeds in Madagascar, where it is found in groups of 10-50 in a variety of habitats across most of Madagascar except the extreme south-west (BirdLife International 2007). The global population is estimated to be between 5,000 and 10,000 individuals (Wetlands International 2006) It appears to be more resident in eastern Madagascar which probably constitutes its main breeding range, whereas on the west coast it is most often observed as a migrant. It migrates to East Africa during the austral winter (May-August) where it is mainly found near the coast between southern Somalia and Mozambique north of the Zambezi River, although large numbers have sometimes been recorded from inland sites (Urban et al. 1986; BirdLife International 2007). Urban et al. (1986) also suggest that this bird over-winters on the Comoros. It is locally, but ephemerally abundant on coasts and edges of lakes and rivers near the coast, particularly at Sabaki Creek in Kenya. Several wintering sites are very important, holding several thousand birds**,** whereas only two sites are known on Madagascar holding more than 100 individuals**.** It is known to breed in loose colonies on rocky islets in rivers, saltmarsh and coastal rocky areas. It is also found in short grasslands and on lake edges (BirdLife International 2007). Wetland areas such as river mouths and lakes are important wintering sites. Its preferred food is insects, mainly hymenopterans, neuropterans and beetles (Urban et al. 1986). Threats to this species other than climate change include anthropogenic modification of its habitat, especially its breeding areas on the east coast of Madagascar.

The main threats to this species resulting from climate change are due to its small population and relatively restricted range. Its breeding habitat is unlikely to be sufficiently specialist to be at major risk from climate change. Its over-wintering habitat may be threatened by sea-level rise, although its precise habitat requirements are not known. Future climate-related research should thus focus on establishing these precise habitat requirements and on determining the vulnerability of these to climate change. It may be that islands, such as those located on the mouth of Sabaki Creek in Kenya are favoured roosting sites as they are free from predators. The fact that almost the entire global population can be found in a single flock during the non-breeding season makes this species particularly vulnerable to stochastic events. Future research should thus focus on how it responds opportunistically to ephemeral peaks in food and on how these might be affected by future climate change.

6.2.8. Slender-billed Curlew

There is a high likelihood that this species is already extinct. The last known confirmed record was of an extra-limital bird at Druridge Pools, Northumberland in the UK in May 1998. This species has only been confirmed breeding near Tara, north of Omsk in Siberia, Russia, between 1914-1924. Formerly at least, it migrated west-south-west from its presumed breeding grounds in Siberia through central and eastern Europe, predominantly Russia, Kazakhstan, Ukraine, Bulgaria, Hungary, Romania and Yugoslavia to southern Europe, Greece, Italy, and Turkey, and north Africa, Algeria, Morocco and Tunisia. It has also been reported from Slovenia, Uzbekistan and Turkmenistan. Reports of birds wintering in Iran persist but require confirmation (BirdLife International 2007). Regarded as very common in the 19th century, it declined dramatically during the 20th century. Flocks of over 100 birds were recorded from Morocco as late as the 1960s and 1970s. However, between 1980-1990, there were only 103 records involving 316-326 birds, and from 1990-1999, this dropped to 74 records involving 148-152 birds^{-}, many of which are unconfirmed (BirdLife International 2007). The last confirmed record of this species in Morocco was of 1-5 bird regularly visiting the Merja Zerga lagoon in the winter of 1995/95. After that, the only confirmed record apart from that in Northumberland was of a flock of 19 birds wintering in Italy in the winter of 1994/95 (Serra et al. 1995). Rumours of unconfirmed or unphotographed birds persist.(Nankinov et al. 2003a; Nankinov et al. 2003b). For example 26 birds identified as this species were recorded in the Danube Delta in Ukraine between 1985 and 2004 all in autumn, with the most recent being in August 2004 (Zhmud 2005). There is also some evidence of birds seen on spring migration in Bulgaria, with 63 records involving an aggregated total of 178 bird recorded between 1992 and 2002, mainly from Bourgas wetland (Nankinov et al. 2003a and b). None of these birds have been photographed however and the authenticity of these records has been questioned.

The only known nests were recorded on the northern limit of the forest-steppe zone in habitat more typical of taiga marsh. On migration and in winter, a wide variety of habitats are used, including saltmarsh, steppe grassland, fishponds, saltpans, brackish lagoons, tidal mudflats, semi-desert, brackish wetlands and sandy farmland next to lagoons (BirdLife International 2007). Threats on the breeding grounds are unknown. Within its potential breeding range, the taiga has been little modified, the forest-steppe partially cultivated and much of the steppe modified by agriculture. Habitat loss in the wintering grounds is of unknown importance. There has been extensive drainage of wetlands in the Mediterranean and North Africa and potentially important areas in Iraq. Historically hunting was high and may have been the key factor in its decline (BirdLife International 2007).

This species is primarily identified as being threatened by climate change because of its very small population size and area of occupancy. Little is known about its breeding habitat, but it could be threatened by the expansion of agriculture into areas formally too cold for farming. Its winter habitat is likely to be threatened to a small degree by sea-level rise and potentially more so by drier conditions in the Mediterranean and Black Sea areas, which may reduce the area extent of wetlands. Future research, climate-change orientated or otherwise should focus on establishing the true status of this species. To this end, verifying the identification of those reported in Ukraine, Bulgaria and Iran by obtaining photographs should be a high priority. If the species does indeed occur in these areas as regularly as reports suggest there may be scope to satellite track individuals thus establishing where its favoured breeding and winter grounds are. Only after such basic information has been obtained can efforts be made to safeguard the species against climate change.

6.2.9. Damara Tern

Breeding Damara Terns are confined to the coast of Namibia south to the Cape Provinces in South Africa and north to Cabinda in Angola, with 98% of the population nesting between the Orange and Cunene Rivers (BirdLife International 2007). It disperses north after the breeding season and is recorded regularly along the coasts of Democratic Republic of Congo, Congo (Brazaville), Gabon, Nigeria, Benin, Ghana and Côte d'Ivoire (Urban et al. 1986). During 2002, the total population is estimated at 14,000 birds and may be declining (BirdLife International 2007). The majority of the population is concentrated in the central area of its range (around 23˚C) coinciding with the main spawning ground of fish species on which it preys (BirdLife International 2007). Little is known about its precise feeding habits, but it is thought to feed on small fish, squid and probably also crustaceans (Urban et al. 1986). The species favours inshore bays, estuaries, creeks, harbours, lagoons and salt-pans in the surf zone and roosts communally on beaches and sand banks, preferring inner beaches less frequented by predators (Urban et al. 1986; BirdLife International 2007). They breed in a variety of habitats**,** including rocky ledges and at rehabilitated diamond mines, preferring breeding sites that provide good visibility (Urban et al. 1986; BirdLife International 2007). They generally form loose aggregations of 4-60 pairs usually 10 or fewer and occasionally singly (Urban et al. 1986). Potential threats other than climate change include land claim, dredging and hotel construction, which threaten some feeding areas and recreational disturbance such as that caused by off-road vehicles may damage or cause desertion of nests.

This species is primarily threatened by climate change because of the habitat with which it is associated and because of its fairly specialist diet. As with many of the preceding species, its confinement to the southern margins of Africa make it especially vulnerable if waters off the coast of South Africa warm, because there would be no opportunity to move poleward due to the lack of land between South Africa and the Antarctic Continent. The species is notoriously site faithful (Urban et al. 1986), and low lying colonies could be threatened by sea-level rise. The availability of feeding habitat decrease if sea-levels rise substantially, but the extent to which this is likely to be the case will depend largely on how the coastal zone is managed and developed in the future. Future research should focus on establishing the extent to which its feeding habitat is likely to alter under future climate scenarios, particularly with respect to the relative abundance of actual and potential food sources. Future research should also identify which colonies are most at risk of flooding and identify the need for adaptive management such as the provision of flood-defence works around colonies.

6.3 List of vulnerable populations

In addition to individual species, several biogeographic populations are listed in the AEWA agreement (on Table 1). As with the listed species, the listed populations with a total score greater than 25 were considered critically threatened by climate change. Those with a score of over 20 were considered highly threatened and those with a score between 15 and 20, moderately threatened. A full list of the scores assigned to each population at least moderately threatened is given in Appendix 1. Those populations critically threatened are listed in Table 8.

6.3.1. White Stork – Southern Africa population

This species is fairly widespread throughout the AEWA region, with three distinct populations. That most likely to be threatened by climate change is a small population breeding in South Africa. Brown et al. (1982) refer to this population as breeding sporadically, but Wetlands International (2006) suggest that as many as 20 pairs breed. The favoured breeding location is mainly in southern regions of the Cape Province, especially the wheatlands of the Ruens (Hockey et al. 1989). Here, large flocks, including non-breeding Palaearctic migrants, gather to feed in lucerne fields and recently burned lands (Hockey et al. 1989). Breeding adults appear to remain in the year throughout the year, but offspring migrate to tropical Africa (Hockey et al. 1989), with one ringed young from South Africa recovered from the Tanzania / Zambia border, 3,300 km from its natal site (Brown et al. 1982). This species' breeding habits are not well known from Africa, but unlike in Europe where its nests are associated with buildings and artificial nest platforms, it is thought to nest exclusively in trees, mainly between September and December (Brown et al. 1982; Hockey et al. 1989).

Although the feeding habits of this population are not documented in detail, they are likely to be similar to other populations, which feed entirely on animals, including especially mice, small reptiles, amphibia, fish and large insects (especially Orthoptera). In Africa large flocks sometimes follow locust swarms. Threats other than climate change include the use of toxic sprays to control locusts and agricultural intensification (Brown et al. 1982). Threats associated with climate change are likely to result mostly from lower rainfall, which may affect the availability of food directly or due to more intensive management of water resources. Climate change could also affect the nature of agriculture in its main breeding location, which in turn could have major implications on food and habitat availability. Since the true status of this population is not really known and it appears to mix readily with non-breeding Palaearctic migrants, establishing the status of this population is probably the most urgent research priority. If the population is genetically isolated from Palaearctic populations than this species is indeed highly threatened by climate change. If, as is also possible, the breeding population represents sporadic breeders of Palaearctic origin, than not only is this population not truly a distinct population, but climate-change impacts are likely to be buffered. Other research priorities include establishing its dependence on agriculture and associated land-uses and determining how land-use is likely to be affected by lower rainfall and increased evapo-transpiration.

Table 8. Assessment of vulnerability of all populations listed on Annex 2 of the AEWA Agreement which are critically threatened by climate change.

Common name	**Species**	**Population**	**Population size score**	**Range size score**	**Fragmentation score**	**Habitat score**	**Food score**	**Climate vulnerability score**
White Stork	*Ciconia ciconia*	Southern Africa	9	5	2.5	6	0	24.5
Northern Bald Ibis	*Geronticus eremita*	South-west Asia & South Asia (win)	10	7	4	4	0	27
Northern Bald Ibis	*Geronticus eremita*	Morocco	7	7	4	4	0	24
Sacred Ibis	*Threskiornis aethiopicus*	*aethiopicus* - Iraq & Iran	7	4	2	6	0	21
Cape Teal	*Anas capensis*	Lake Chad basin	7	4	2.5	6	0	21.5
White-headed Duck	*Oxyura leucocephala*	Algeria & Tunisia	7	5	2.5	6	2	22.5
Siberian Crane	*Grus leucogeranus*	Iran (win)	10	5	2.5	6	4	25.5
Common Crane	*Grus grus*	Turkey & Georgia (bre)	7	3	2	6	0	20
Demoiselle Crane	*Grus virgo*	Turkey (bre)	9	5	3			
Demoiselle Crane	*Grus virgo*	Black Sea (Ukraine) / North-east Africa	6	4	2	6	0	20
White-winged Flufftail	*Sarothrura ayresi*	Ethiopia & Southern Africa	6	8	4	6	2	27
Chestnut-banded Plover	*Charadrius pallidus*	*venustus* - Eastern Africa	5	5	2.5	6	4	20.5
Slender-billed Curlew	*Numenius tenuirostris*	Central Siberia / Mediterranean & SW Asia	9	3	2.5	6	2	22.5

Win = Wintering; Bre = Breeding

6.3.2. Northern Bald Ibis – South-west Asia South Asia wintering population

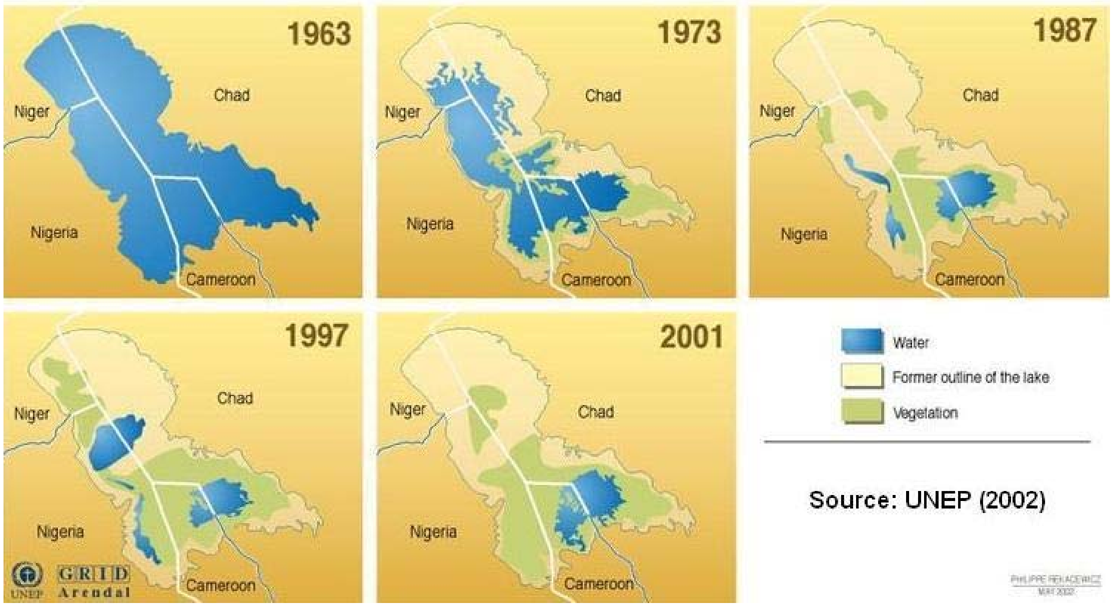
See species account (section 6.2.5)

6.3.3. Northern Bald Ibis – Morocco population

See species account (section 6.2.5)

6.3.4. Cape Teal - Lake Chad basin population

There are three distinct populations of Cape Teal. The East African Rift Valley population, the Lake Chad population and Southern African (North to Angola and Zambia) population. The population in Lake Chad numbers less than 500 (Wetlands International 2006). The highest individual count is of 300 in the early 1970s, but since then there has been no counts of more than 80 individuals (Brown et al. 1982; Cramp 1977) and this population may also include those birds recorded regularly from Kanu state in north-east Nigeria. The absence of any movements from other areas remains unproven, but the main dates in which it has been recorded here and in surrounding areas suggest birds undergo movements in response to changing water-levels. Although very little is known about this population, if indeed it is that, much can be inferred by the status of Lake Chad itself. During the 20th century, the extent of Lake Chad has fluctuated considerably as a result of cyclical drought and flooding and this is likely to have a direct effect on this population of Cape Teal. It has ranged from a surface area of almost 26,000 km^2 in June 1962 to 1,653 km^2 in April 1985 (see Figure 10). At its lowest level, the two basins have been physically separated by a dry ridge and most recently the north-western basin has dried out completely. The present lake is only a tiny remnant of the Paleo-Lake Chad, which first came into existence by tectonic subsidence 35 million years ago. Since the 1980s, despite higher rainfall the lake has continued to shrink primarily as a result of increased water abstraction and over-grazing, which has resulted in desertification (Coe & Foley 2001) (Figure 10).



Source: UNEP (2002)

Figure 10. The drying up of Lake Chad. The collection of maps is produced by UNEP (2002) and is based on a series of satellite images provided by the NASA Goddard Space Flight Centre available at: http://www.gsfc.nasa.gov/gsfc/environ/lakechad/chad.htm

The habits of this population are not well known, but that of birds from other parts of the range fairly well documented. Further south in Africa, this species prefers pools, lagoons, salt-pans, tidal mud-flats, brackish water and soda lakes. Its food comprises of animal and vegetable matter. In the cape 63% of its diet consists of insects, 10% of other invertebrates and tadpoles and 17% of plant matter, mainly *Potemogeton pectinatum*. The main threat to this population is likely to result from the drying-up of Lake Chad. However, as it seems to favour relatively shallow waterbodies and the population itself is quite small, it may not be heavily threatened until the lake becomes almost completely dry. The most urgent research priority is to establish the true, current status of this population, particularly whether individuals present in Lake Chad truly represent a genetically distinct population or whether they simply consist of occasional visiting flocks from elsewhere within this species' range. A further research priority is to establish its basic habitat and food requirements and to assess how these are likely to be affected by the continued reduction in the surface area of the Lake.

6.3.5. White-headed Duck - Algeria & Tunisia population

There are four distinct populations of White-headed Duck. The Algerian and Tunisian population is resident and numbers approximately 400-600 individuals (Wetlands International 2006). Its precise status is poorly known. In Algeria, it probably breeds at Lake Tonga, El Kala region, where 15 were counted in 1977 and 36 in 1978 (Urban et al. 1986). In Tunisia, it was fairly common, but appears to have declined. Up to 670 were recorded at Lake Tunis in 1973, now more than the entire population, with smaller numbers occurring elsewhere. Its breeding habits in Africa are poorly known. In Europe, it breeds in April-July in freshwater or alkaline, eutrophic lakes, with emergent vegetation. It is a diving duck, feeding on fish, frogs, worms, molluscs and crustaceans in Europe (BirdLife International 2007) and mainly on plant matter such as the leaves and seeds of pondweeds, the seeds of bullrush and on crustaceans (Urban et al. 1986). The main threats to this species other than climate change are from drainage of its habitat. Threats in Africa are unknown, but elsewhere it is threatened by drainage. Approximately 50% of its breeding habitat has been drained during the 20th century. Remaining sites are vulnerable to pollution. A 1989 study in the main Pakistani wintering lakes showed that suitable habitat had decreased because of lowered water levels due to reduced water supply, and that fisheries had increased disturbance. The European population is threatened by competition and hybridization with the introduced Ruddy Duck *O. jamaicensis*. Further threats include drowning in fishing-nets, hunting and ingestion of lead shot. Droughts in Kazakhstan and Uzbekistan may have caused recent poor breeding seasons**.**

This species has primarily been identified as critically threatened by climate change because of its small population and range size and because of the vulnerability of its habitat to climate change. It is likely to be severely threatened by future drier conditions in the Mediterranean basin (IPCC 2007b), as indicated by its vulnerability to drought elsewhere (BirdLife International 2007). Future research in relation to climate change should focus on finding out more about the basic ecology of this species, particularly on its breeding success and preferred habitat requirements in relation to water levels. Following on from this, determining the likely future water levels at its favoured breeding locations is of high priority.

6.3.6. Common Crane – Turkey & Georgia breeding population

There are several distinct populations of Common Crane. Most are relatively abundant and not seriously threatened by climate change, but a small population, numbering 200-500 birds breeds at two sites, one in Turkey, the other in Georgia (Wetlands International 2006). The Georgian breeding site is thought to host only 7-10 pairs, with numbers decreasing (BirdLife International 2004). In Turkey, isolated groups breed east and south of the Black sea mainly on the Central Plateau and in the east of the country (BirdLife International 2007; Cramp 1980; Beaman et al. 1975; Beaman 1978). The Common Crane appears to favour seasonally flooded or marshy plains and wetlands, with small numbers, 5 to 40 pairs, breeding at various sites throughout the country (Beaman 1978; Beaman et al. 1975). The precise migratory status of this population is not known with certainty with suggestions that is perhaps resident or makes short movements (Cramp 1980), perhaps to the Black Sea coastlands (Beaman 1978; Beaman et al. 1975) or that part of the population may migrate to north-east Africa (Meine & Archibal 1996; Wetlands International 2006). The specific food requirements of this population are poorly documented, but are likely to be similar to other populations. Elsewhere, Common Cranes are known to have varied diet, comprising mainly of plant material, including roots, rhizomes, tubers, stems, leaves, fruits and seeds of grass mainly, but also of crops and wild herbs. Animal prey are also taken and may predominate at certain times of year and includes Coleoptera, Orthoptera, Odonata and Diptera.

Knowledge on the general the size, distribution, status and movements of this population are extremely limited (Meine & Archibal 1996) and the most urgent research priority is probably to establish the true status of this population, particularly in relation to where it winters. Threats other than climate change are mainly to its habitat as a result of ongoing drainage and agricultural intensification (BirdLife International 2007). Climate-change induced threats are likely to be associated with increased evapo-transpiration and lower rainfall which is likely to result drier, more unfavourable habitat or changes in the nature of low-intensity farming.. Other priorities include the identification of main breeding sites so that these can be protected from agricultural intensification and to establish more specifically how its habitat and food requirements are linked to water availability.

6.3.7 Siberian Crane – Iran wintering population

The Siberian Crane breeds in Arctic Russia in Yakutia and western Siberia. Three regional populations were recognised, but one is likely to be extinct. The eastern population breeds between the rivers

Kolyma and Yana and south to the Morma mountains and migrates south to the lower reaches of the Yangtze River, especially Poyang Hu Lake, China (Kanai et al. 2002b). The western population breeds in the Tyumen District, Russia, and winters in Fereidoon Kenar and Esbaran in Iran (CMS 2004; Kanai et al. 2002a) and is the population most likely to be affected by climate change within the AEWA region. A central population of Siberian Cranes once nested in western Siberia and wintered in India. The last documented sighting of Siberian Cranes in India during the winter months was in 2002, but there have been recent unconfirmed reports of several (four to seven) in other areas within the breeding range and of a single crane in Uzbekistan and Pakistan along the migration route, and in elsewhere in India (CMS 2004). Nevertheless, there is a high probability this population has been recently extirpated (BirdLife International 2007). The global population is about 3,000, the majority of which belong to the eastern population and winter at Poyang Hu (Delany and Scott 2002). The population in Iran numbered between 9 and 14 birds until the late 1990s and usually included one or two juveniles (CMS 2004). The population has since started to decline: 9 birds recorded in the winters of 1996/97 and 1997/98, 7 in 1998/99, 5 in 2000/01 and just 3 adults in 2001/02 and 2002/03 and 3 adults and 1 juvenile in 2003/04 (CMS 2004) and is consequently highly vulnerable to climate change.

Siberian Cranes breed and winter in wetlands. Their favoured nesting habitats are bogs, marshes, and other wetland types of the lowland tundra, taiga/tundra transition zone, and taiga, preferring wide expanses of shallow fresh water with good visibility (Meine & Archibald 1996). Wintering habitat requirements are similar. In Iran, they winter in paddy-fields (Kanai et al. 2002a). During migration they favour large, isolated wetlands. Siberian Cranes eat a variety of food items. On the breeding grounds in spring, they eat cranberries, rodents, fish and insects (Meine and Archibald 1996). On migration and on the wintering grounds, they excavate nutrient rich roots and tubers from wetlands. They are predominantly vegetarian outside their breeding season (Kanai et al. 2002b). Threats other than climate change to the Iranian population are primarily from hunting and oil exploration (Kanai et al. 2002a).

This species is primarily identified as threatened by climate change because of its small population size and area of occupancy. However, its breeding and non-breeding habitats are also threatened by climate change. On the breeding grounds they favour areas with no trees as this ensures good visibility. The scarcity of trees in tundra and taiga habitats is primarily a consequence of the presence of permafrost, which is likely to be significantly reduced in northern taiga and southern tundra zones (Piersma & Lindstrom 2004). Increased variability in rainfall is predicted in their Middle Eastern wintering grounds. Such changes in rainfall patterns are likely to pose serious threats to the tuberiferous plant species on which they feed. Given the small number of wetlands, which they habitually occupy, they possess limited capacity to respond to seasonal changes in rainfall by moving to other wetlands.

6.3.8. Demoiselle Crane –Turkey breeding population

There is a very small, declining population of this species, numbering between 30 and 60 birds, breeding in Eastern Anatolia in Turkey (BirdLife International 2004). Its status as a migrant is rather poorly studied (Archibal & Meine 1996), but may be resident (Wetlands International 2006) or migrate south with the Black Sea and Kalmykia populations to Sudan and other areas of East Africa (Archibal & Meine 1996). It was first discovered breeding in 1971 12 km NW of Dogu Bayazit and since than has been recorded as a breeder sporadically in small numbers (Beaman et al. 1975; Beaman 1978; Archibal & Meine 1996). This population and the Black Sea population are separated by the Caucasus Mountains. In the past, both were probably part of a single contiguous population, now interrupted by local extirpations, that surrounded the Black Sea (Archibal & Meine 1996).

Little is known about the precise habitat and food requirements of this population, but elsewhere it requires ready access to drinking water, but otherwise frequents a wide range of habitats from shrubby steppe to coarse grassland interspersed with salt flats. It often makes use of extensive cultivation, but its nest sites are usually in adjoining undisturbed habitat (Cramp 1980). In the 1970s, the sporadic records of breeding were all from islands in Marshes (Beaman et al. 1975; Beaman 1978). Its food generally comprises of plant material, but invertebrates, particularly beetles are also taken in summer. Elsewhere, it is known to feed extensively on ripening cereal grain in late Summer and Autumn. Threats other than climate change include agricultural intensification and disturbance at its main breeding site. At its wintering site, although the area is important because of current land-use practices, agricultural intensification could lower the quality of this site. Given the paucity of knowledge about the status of this population, the most urgent research priority is to establish the true status of this population. Other research priorities include determining the habitat requirements of this population in particular in relation to water availability and agricultural practices.

6.3.9. Demoiselle Crane – Black Sea (Ukraine) / North-east Africa population

There is a small population of this species, numbering between 600 and 750 (Wetlands International 2006), possibly less than 500 (Archibal & Meine 1996), which breeds in the Black Sea & Ukraine area and winters in sub-Saharan Africa from Lake Chad to Ethiopia (Wetlands International 2006). The breeding grounds of this population are mainly in the Kerch peninsula of Crimea and other portions of south-eastern Ukraine, with the majority breeding at Syvach Bay, an extensive system of lagoons with sandy islands and shores, surrounded by steppe, in the coastal zone of the Black Sea and Sea of Azov (Archibal & Meine 1996; BirdLife International 2007). In general, this population frequents a fairly broad range of habitats from shrubby steppe to grassland and cultivations, especially for feeding, although nesting is usually in undisturbed habitat. Although probably relatively stable at present (BirdLife International 2004), This population has declined steadily since the 1950s, no longer occurring as a breeding bird in Romania, Moldova, or Bulgaria, nor as a wintering bird in Egypt (Archibal & Meine 1996). The precise wintering grounds of this population are unknown as they are probably joined by birds from further east during the non-breeding period. Birds from this population and elsewhere once over-wintered in a broad swath from NE Nigeria to Ethiopia (Urban et al. 1986). It has now been extirpated from much of its former range. The majority now winter in southern Sudan, primarily at just one location: Gezira. The Gezira is a large area of flood-plain between the Blue and White Niles to the south of Khartoum. The area is intensively farmed and particularly along the western bank of the Blue Nile south-eastwards to the town of Wad Madani and beyond is criss-crossed with networks of irrigation channels favoured by the Cranes (BirdLife International 2007). Threats other than climate change include agricultural intensification and disturbance at its main breeding site. At its wintering site, although the area is important because of current land-use practices, agricultural intensification could lower the quality of this site.

The main threats other than climate change are from habitat loss and degradation and disturbance (Archibald & Meine 1996; BirdLife International 2007). The breeding grounds in steppe habitat, especially those with nearby water sources, are highly attractive for agricultural development, resulting not only in conversion of habitat but increased pressures from grazing, disturbance, poaching, and other human activities (Archibald & Meine 1996). Although the relationship between farmers and Demoiselle Cranes has improved, changes in agricultural practices (such as spring ploughing and increasing use of pesticides) continue to have negative impacts on nesting and feeding behaviour (Archibal & Meine 1996). At its wintering sites changes in agricultural practices are likely to be having an adverse impact, despite the dependence this species has on agricultural land (BirdLife International 2007). The main threat from climate change results from the small size of this population and because its habitat is sensitive to changes in rainfall regimes. Any prolonged changes in rainfall are likely to result either a change in the type of crops grown or more intensive management of water resources, both of which are likely to have an adverse effect on this species. The most urgent research priority should thus focus on determining what changes in agricultural practices are likely to occur in response to future climate change at its main breeding and wintering sites.

6.3.10. White-winged Flufftail – Ethiopia & Southern Africa

See species account (section 6.2.6).

6.3.11. Chestnut-banded Plover – *venustus* - Eastern Africa population

There are two distinct populations of Chestnut-banded Plover: *pallidus* from Southern Africa and *venustus* from the East African Rift Valley in Kenya and Tanzania, the latter numbering 4,000-5000 birds (Wetlands International 2006) or 5,000-6,000 birds (BirdLife International 2007). This population occurs at a handful of sites in southern Kenya and northern Tanzania and is concentrated at just three sites in the northern Rift Valley with counts from Tanzania of 2,340 individuals at Lake Natron and 520 individuals at Lake Manyara, and 590 individuals at Lake Magadi in Kenya. The population is predominantly resident, but undergoes local movements in response to drying-up of breeding habitat. All three of its most favoured lakes are similar in that they are highly-alkaline and saline and fluctuate greatly in water-level in response to rainfall. Lake Natron is a shallow (maximum depth 50 cm) soda-lake in a closed basin on the floor of the Eastern Rift Valley, extending 58 km south from the Kenyan border with a mean width of 15 km, although sometimes much larger than this. There are considerable seasonal fluctuations in surface area between the middle of the dry season (November-December) and the end of the rains (April-May). The water is highly saline with chloride concentrations reaching 65,000 mg/litre and is unsuitable for direct human and livestock use (BirdLife International 2007). Lake Manyara is located 85 km south of Lake Natron and is approximately 40 km long and 13 km wide, although again water-levels affect its size. The lake is generally saline (pH levels vary with rainfall) with areas of fresher water where perennial streams and rivers flow into the lake. Lake Magadi, 85 km south-west of Nairobi is also a shallow alkaline lake with extensive surface water is present only after heavy rains over the local catchment, when run-off reaches the northern end of the lake via three wadis. Most of the lake is a vast expanse of solid sodium carbonate and allied salts, some 15-30 m thick, mined by a soda company. The species tends to feed along the waters edge on a variety of insect and crustacean species.

Threats other than climate change are mainly to Lake Natron, where the site is much threatened by the potential construction of a soda ash plant and hydroelectric/irrigation dam, which would increase water-flow to the lake and thus vastly reduce its salinity. The main reason this population is threatened by climate change is because of its small size and the small number of sites to which it is confined. All these sites are likely to be extremely vulnerable to changes in rainfall regime, which will affect water-levels and salinity directly and affect the degree to which the wider catchment is managed. Both higher and lower rainfall could potentially be damaging, with higher rainfall lowering salinity and lower rainfall leading to a reduction in the amount of suitable habitat available. Probably the most urgent research priority is to assess how management of water resources within the wider catchment of the three key sites is likely to be affected by changes in rainfall and evapotranspiration. Another key priority is to assess the extent to which the population of this species is sensitive to changes in salinity and alkalinity of its favoured feeding locations.

6.3.12. Slender-billed Curlew – entire population

See species account (section 6.2.8)

7. INTERNATIONAL RESEARCH NEEDS

In addition to species-orientated research, discussed at the and of each of the species accounts above, there are several research needs, which span all species and divide more conveniently by topic rather than by species or species group. Each of these is discussed below.

7.1. Can birds evolve fast enough to keep pace with climate change?

Migratory waterbirds, in common with other living organisms, are likely to respond to climate change in two ways: by altering their behaviour and by evolving. The first arises because all organisms possess at least some ability to change their phenotype in responses to changes in the environment (McNamara & Houston 1996). For example, they use external cues such as temperature, favourable wind conditions and clear skies to decide to depart on migration or might use temperature or vegetation growth to decide to begin breeding. Thus if temperatures are warmer in early spring or vegetation grows earlier, they chose to begin breeding earlier. The reason that they use these cues is that they have served them or their ancestors well in the past and they have thus evolved to make use of the array of signals that correlate best with the conditions that maximise their survival or reproduction (Stenseth & Mysterud 2002). However, if the environment changes rapidly as a result of climate change, formerly reliable cues might no longer be associated with adaptive outcomes. In such cases, organisms can become “trapped” by their evolutionary responses to the cues and thus experience reduced survival or reproduction (Schlaepfer et al. 2002). Examples of such mismatch include instances where birds nest earlier in response to warmer temperatures, but then become adversely affected by high rainfall (Jovani & Tella 2004), mistiming with respect to food resources because birds respond to temperature differently than their food (Perrins 1991), or instances where the cues used to depart from over-wintering grounds do not correlate with the timing of events on breeding grounds (Schlaepfer et al 2002).

The only successful way of avoiding such evolutionary traps is for birds to evolve sufficiently quickly to keep pace with climate change. There is some evidence that some birds can evolve quickly. Remarkably quick changes in migration directions have been witnessed in a number of *Sylvia* warblers such as Blackcaps *Sylvia atricapilla*, which now regularly over-winter in the UK (Helbig 1996)**.** Similarly, but perhaps more dramatically, a proportion of the population of Yellow-browed Warblers *Phylloscopus inornatus*, which breed in central and eastern Asia and generally migrates to south-east Asia during the boreal winter, are now almost certainly migrating south-west to over-winter in Africa, as demonstrated by the exponential increase of records of this species in the United Kingdom in September and October (Gilroy & Lees 2003). A demonstration of rapid evolution in response to climate change is that long-distance migrants have advanced their breeding by more than short-distance migrants suggesting an evolutionary rather than phenotypic response, as the cues triggering departure are said not to correlate with warm temperatures (Jonzen et al. 2006). However, the extent to which this pattern is prevalent in waterbirds is unknown, as is whether any bird species can evolve quickly enough under accelerated global warming. Establishing the pace of evolution in waterbirds can be done in two ways. Firstly by recording instances where birds have advanced their phenology or modified their behaviour in some way to respond to climate change, but then demonstrating that the cues used to allow such modified behaviour are uncorrelated with components of climate change. This is the approach underpinning Jonzen et al.’s (2006) study, albeit that the lack of correlation is assumed rather than tested. The second approach would be to use long time-series of data to demonstrate that the extent of mismatch is reducing through time. Reduced mismatch would suggest higher survival and reproduction of those individuals less prone to evolutionary traps. The unique long-term datasets available for many waterbird species make this group of organisms ideal candidates for testing such mechanisms.

7.2. Does climate change affect population sizes?

Many individual waterbirds are likely to experience deleterious effects as a result of climate change and sea-level rise. However the extent to which this will have an adverse impact on populations is largely unknown. It is a well known phenomenon in population ecology that the survival and reproductive rates of an organism depend on the number present in the first place (Ohman & Hirche 2001).This generally occurs because individuals compete for resources (Malthus 1798), although interference competition, whereby animals interfere with each other's ability to obtain resources can also occur (e.g. Creswell 1997)**.** Thus, if individuals of a population die as a result of climate change, or produce less offspring lower mortality and higher productivity as a result of reduced competition for resources may fully compensate for any losses due to climate change. A reduction in the overall population size will only occur if the reduction in survival or productivity due to climate change exceeds the extent to which it can be compensated for by altered density-dependent mortality and productivity or if the level of resources themselves are adversely affected.

There are a number of ways in which the population impacts of climate change and sea-level rise could be tested, but two approaches are likely to be most successful and allow the most insight to be gained, although the second would need to incorporate the first. The first is essentially the phenomenological modelling approach discussed in section 4.1.3. The second is the behaviour-based modelling approach discussed in section 4.1.4. The phenomenological modelling approach, would entail estimating demographic parameters under a range of densities and then determining intrinsic rates of population growth and the extent of density-dependent mortality or productivity. Coupled with an understanding of how climate change affects survival and productivity, a population model could be constructed to predict the impacts of climate change on species populations. This approach is likely to be particularly useful in instances where climate change is thought to affect survival and mortality rates directly rather than through altered resource availability. The second approach allows a better understanding to be gained of the impacts of changed resource availability on species populations. The starting point of this approach would be to assume that individuals follow an ideal-free distribution (Fretwell & Lucas 1970) and then incorporate the negative feedbacks of population size (such as depletion) upon distribution and performance. If population parameters are then incorporated into such models, derived in the way described in the first approach, an understanding of both the direct and indirect impacts of climate change on populations could be determined. Although this method has not been used to assess the impacts of climate change, it has been used to link habitat loss and population size (e.g. Durell et al. 2006; Sutherland 1996) and could readily be adapted to determine the impacts of climate change.

7.3. Climate change and migration cues

One concern about the impacts of climate change on migratory birds, is that the cues used to trigger departure from wintering grounds are not correlated with optimal conditions for breeding on their breeding grounds (Coppack & Both 2002; Coppack et al. 2003; Lawrence & Soame 2004; Lundberg & Atalo 1992; Robinson et al. 2005). The converse could also be true: cues triggering departure from breeding grounds may not be correlated with climate changes occurring on wintering grounds. Most studies suggesting that such mismatches might be occurring, cite day length as one of the primary signals controlling departure from wintering grounds (Coppack & Both 2003; Coppack et al. 2003; Lawrence & Soame 2004). Evidence that this is the case is inconclusive. Although one study in which captive birds were exposed to day lengths simulating migration to and from wintering grounds in southern Europe, demonstrated that birds considerably advanced their spring migratory activity and testicular development (Copack et al. 2003), it is not known whether this is the primary cue for departure from wintering grounds.

Departure from breeding grounds in the northern hemisphere is generally determined by weather favourable for migration: clear skies and winds in the desired direction of flying (Elkins 2004). Although this is likely to be a 'short-window' cue in the sense that it will affect the precise day or time of departure, but is unlikely to control the time-of-year of departure, it demonstrates that climatic conditions are at least partially responsible for triggering the onset of migration. Even 'long-window' cues, i.e. those that determine the time of year of departure may be affected by components of climate such as temperature directly or indirectly by being related to food availability (Elkins 2004). It is thus at least plausible that climatic conditions play some role in triggering departure from wintering grounds. Whilst it is still possible that climate changes in wintering grounds are not correlated with those on the breeding ground, the existence of large-scale climate circulation systems such as El-Nino and NAO and the fact that climate changes have a consistent directional trend, may mean that they are more correlated than previously supposed. If this is indeed the case, then the assumptions underpinning studies such as that by Jonzen et al. (2006), which is said to demonstrate evolutionary changes, are not valid.

Establishing the cues used to trigger departure would be best achieved by observing when birds depart over a number of years (or from a number of locations) and then correlating these with the various candidate cues (e.g. temperature, rainfall, photoperiod, food availability) to see which of this best explains departure date. The date of departure could be established either by direct observation, or through the use of some tracking system such as radar, radio-tags, GPS tags or satellite tags. The required precision and available financial resources should primarily guide the method adopted as the precision achieved is directly related to costs. Once the cues have been established, it should be possible to examine the extent to which they are correlated with factors that benefit survival and productivity on breeding grounds and determine the extent to which they are likely to become increasingly uncorrelated as the world's climate changes.

7.4. Climate variability in early spring

Another concern about the impacts of climate change on migratory waterbirds is that these birds may not advance their phenology by the same rate as their prey (Robinson et al. 2005). Consequently there may be a mismatch between chicks hatching, and the availability of food with which to feed them (e.g. Perrins 1991). In a sense this is an example of an evolutionary trap (Schlaepfer *et al.* 2002), whereby the cues used to make behavioural life-history decisions are inappropriate. While such biological mismatch is a well-observed phenomenon (see Robinson et al. 2005 and earlier in this report for reviews), the extent to which two different environmental determinants of survival or productivity can become mismatched has received far less attention, yet could actually be more important.

It is well known that adverse weather can affect bird survival and productivity during the early breeding season in Europe (Peach et al. 1995; Thomson et al. 1997). Indeed, chicks during the early stages of life often exhibit very high rates of mortality (upwards of 80%), with unfavourable weather the most important cause of chick mortality for most species (Robinson et al. 2005). Thus although warmer springs might benefit birds, if the earlier onset of warm weather results in earlier breeding, but at this time temperatures are more variable and/or rainfall is heavier, high chick mortality may occur. Surprisingly, the extent to which this is the case and whether patterns of variability differ geographically have not been explored within the context of nestling survival. It would be straightforward to undertake, merely relying on obtaining daily weather data and analysing them appropriately.

7.5. Sea-level rise and risk of nest flooding

Aside from habitat loss, one of the impacts of rising sea-levels is likely to be nest flooding of coastal waterbirds, such as terns and plovers (Robinson et al. 2005). At present however, the factors used to determine how close to the waters edge birds nest, are not known. At least four possibilities exist: (1) it could be related to the vertical or horizontal distance from the waters edge; (2) it might be indicative of past flooding: e.g. birds build their nests within a particular habitat type affected by average salinity conditions; (3) it could be base on past experience of flooding or (4) it might be based on fidelity either to a particular nest site or particular colony. It might be that multiple factors are used and that the factors depend on the species in question. However, the cue used will have a large bearing on the risk of nests flooding due to sea-level rise. If birds chose to nest a certain distance above or away from the waters edge, then there will only be a greater risk of flooding if water-levels become more variable. If birds exhibit high nest-site fidelity however, then there is a very high risk of flooding and the risk will be greatest in areas prone to variable water heights such as along the Atlantic coast. If some pre-learned or inherited sense of risk is used as a cue, then the risk of flooding will be highest in areas where a small rise in sea-level substantially decreases the return-rate of storm-surges, such as along the Mediterranean coast (Nichols et al. 1999). At present this is unknown, but could be ascertained by marking the location of nests relative to the waters edge at a large number of locations, recording topography and assessing the extent to which sea-levels will result in a shortening of the return rate of storm-surges.

8. CONCLUSIONS

There is evidence of a large number of climate change impacts on waterbird species and populations listed on Annex 2 and Table 1 of the Agreement, the majority of which are detrimental. An even larger number are likely to be occurring, but have not been documented. Impacts include poleward shifts in range, altered timing of biological events such that there is a mismatch with resources and a loss of habitat due to sea-level rise. Indirect effects are also important and include the loss of habitat and degradation of habitat due to flood-defence works, increased use of water resources in dry periods and increased recreational disturbance during periods of warm weather. The extent to which such impacts adversely affect populations is less certain as loss of individuals from a population may be compensated for by decreased density-dependent mortality and increased productivity. Nevertheless, there are documented incidences of climate change impacts at the population level. Future impacts are less certain. Although the type and direction of impact can often be determined by extrapolating from current documented impacts, multifarious non-linear relationships in nature and the complexity of species interactions are such that impacts can rarely be predicted or quantified. Although some attempts have been made to model the likely future impacts of climate change, rarely have these models been tested.

Nevertheless, many migratory waterbirds are likely to be increasingly adversely affected by climate change. In addition to non-climate related threats facing their habitat, in many areas their habitats are likely to deteriorate as rainfall regimes change and temperatures increase, disappear as sea-levels rise and face increasing pressure from higher demand for water resources. Migrants are especially vulnerable because impacts anywhere within their range can adversely affect them. Furthermore, the intensity of human exploitation of planetary resources GEO (2004) severely constrains the extent to which waterbirds can adapt to changing climatic patterns. Intense wetland drainage and increasingly inhospitable landscapes prevent dispersal and colonisation of new areas and act together with climate change to lengthen migratory routes. Habitat loss means that birds cannot find suitable areas to move to when the areas they are in become too warm or dry. The numerous other pressures that have already decreased the size of many populations means that these populations will have lost some of the genetic variability that could have helped them cope with the variable temperatures and climatic conditions of the future. These indirect effects are likely to become increasingly important in the future.

Species likely to be most vulnerable to climate change are those with small populations, those within small ranges in either the breeding or non-breeding season or during migration, those with specialist food or habitat requirements or those confined to especially vulnerable habitats such as mountain tops or seasonal wetlands in the Mediterranean basin, which are predicted to become much drier. Those also threatened by other factors are likely to be especially vulnerable. Of the 235 bird species listed on Annex 2 of the Agreement, our analysis suggests that two are critically threatened by climate change, a further seven are highly threatened and a further 14 are moderately threatened. Species populations listed on Table 1 of the Agreement are more threatened. Thirteen are judged to be critically threatened, a further 18 are considered highly threatened. Overall, even the least threatened population of a given species is more threatened by climate change than the species itself. Although this is an inevitable conclusion given the criteria used to assess threat, it serves as a reminder that focussing on species as opposed to populations may underestimate the degree of threat facing waterbirds.

Waterbird species associated with certain habitats are likely to be more vulnerable than others. Those associated with montane habitats and tundra on the poleward edges of land-masses are particularly vulnerable because their distributions cannot shift as temperatures increase. Species associated with seasonal wetlands in the Mediterranean and Southern Africa are also vulnerable, because these regions are predicted to become much drier and this habitat is highly sensitive to changes in rainfall. Although all areas of the African-Eurasian flyway are likely to be adversely affected by climate change, it is sub-Saharan Africa and central and southwest Asia that face the greatest challenges. These parts of the flyway have a higher diversity of waterbird species and host many migratory waterbirds during the boreal winter when mortality has the highest impact on populations. They are also likely to face the most pressure from factors that conflict with waterbird conservation interests and are where funding for conservation action is least adequate.

We recommend that future research and conservation action be directed towards the most vulnerable species and in particular their most vulnerable populations. However we also recommend that future research should concentrate on broad themes that can rapidly test avian responses to climate change as a whole rather than the response of just one species. Highest amongst such research priorities is whether species can evolve sufficiently quickly to match climate change. Other important questions include whether adverse impacts are compensated for by reduced density-dependent mortality and the extent to which cues used to trigger departure from wintering grounds are correlated with factors that advance the timing of breeding. We also recommend that international research priorities should address possible means of adaptation. Such measures include establishing an adequate network of protected areas to accommodate range-shifts and to manage the wider countryside in a manner that facilitates dispersal.

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APPENDICES

7.6. APPENDIX 1 – SPECIES VULNERABILITY TO CLIMATE CHANGE

Appendix 1. Assessment of vulnerability of all species listed on Annex 2 of the AEWA Agreement. 🟥 = critically threatened by climate change (score ≥ 20); 🟧 = highly threatened by climate change (score ≥ 17.5); 🟨 = moderately threatened by climate change (score ≥ 15); ▢ = some threat from climate change (score ≥ 10); 🟩 = minimal threat from climate change (score < 10).

Common name	**Species**	**Population size score**	**Range size score**	**Fragmentation score**	**Habitat score**	**Food score**	**Climate vulnerability score**
African Penguin	*Spheniscus demersus*	2	6	3	2	4	17
White-billed Diver	*Gavia adamsii*	3	0	0.5	2	2	7.5
Black-throated Diver	*Gavia arctica*	1	0	0	0	2	3
Great Northern Diver	*Gavia immer*	2	0	0.5	0	2	4.5
Red-throated Diver	*Gavia stellata*	2	0	0	0	2	4
Slavonian Grebe	*Podiceps auritus*	7	0	0.5	0	0	7.5
Great Crested Grebe	*Podiceps cristatus*	1	0	0	0	2	3
Red-necked Grebe	*Podiceps grisegena*	2	0	0.5	0	2	4.5
Black-necked Grebe	*Podiceps nigricollis*	1	0	0.5	2	2	5.5
Little Grebe	*Tachybaptus ruficollis*	1	0	0	0	2	3
Dalmatian Pelican	*Pelecanus crispus*	3	1	1.5	6	4	15.5
Great White Pelican	*Pelecanus onocrotalus*	2	0	1	4	2	9
Pink-backed Pelican	*Pelecanus rufescens*	3	1	0.5	0	4	8.5
Cape Gannet	*Morus capensis*	2	2	3.5	6	6	19.5
Cape Cormorant	*Phalacrocorax capensis*	3	2	1.5	6	4	16.5
Great Cormorant	*Phalacrocorax carbo*	1	0	0.5	0	2	3.5
Crowned Cormorant	*Phalacrocorax coronatus*	4	4	2	4	4	18
Bank Cormorant	*Phalacrocorax neglectus*	4	6	3	4	2	19
Socotra Cormorant	*Phalacrocorax nigrogularis*	2	2	2	6	2	14
Pygmy Cormorant	*Phalacrocorax pygmeus*	3	2	1.5	4	2	12.5
Grey Heron	*Ardea cinerea*	1	0	0	2	0	3
Black-headed Heron	*Ardea melanocephala*	1	0	0	2	0	3
Purple Heron	*Ardea purpurea*	2	0	0.5	2	0	4.5
Madagascar Pond-Heron	*Ardeola idae*	5	2	1.5	4	0	12.5
Squacco Heron	*Ardeola ralloides*	2	0	0.5	4	0	6.5
Rufous-bellied Heron	*Ardeola rufiventris*	3	2	1	2	0	8
Bittern	*Botaurus stellaris*	2	0	0	2	0	4
Cattle Egret	*Bubulcus ibis*	1	0	0	2	0	3
Great Egret	*Casmerodius albus*	1	0	0	2	0	3
Black Heron	*Egretta ardesiaca*	3	0	0.5	2	0	5.5
Dimorphic Egret	*Egretta dimorpha*	4	1	1.5	4	2	12.5
Little Egret	*Egretta garzetta*	1	0	0	2	0	3
Western Reef Heron	*Egretta gularis*	4	1	1	4	2	12
Slaty Egret	*Egretta vinaceigula*	5	4	2	6	2	19

Common name	Species	Population size score	Range size score	Fragmentation score	Habitat score	Food score	Climate vulnerability score
Little Bittern	*Ixobrychus minutus*	2	0	0	2	0	4
Dwarf Bittern	*Ixobrychus sturmii*	3	0	0.5	2	0	5.5
Intermediate Egret	*Mesophoyx intermedia*	2	0	0	2	0	4
Black-crowned Night-Heron	*Nycticorax nycticorax*	1	0	0	2	0	3
Shoebill	*Balaeniceps rex*	4	1	1	2	2	10
African Openbill	*Anastomus lamelligerus*	2	0	0.5	2	6	10.5
Abdim's Stork	*Ciconia abdimii*	2	1	0.5	4	4	11.5
White Stork	*Ciconia ciconia*	2	1	1	2	0	6
Woolly-necked Stork	*Ciconia episcopus*	3	0	0	2	0	5
Black Stork	*Ciconia nigra*	3	0	0.5	4	0	7.5
Marabou Stork	*Leptoptilos crumeniferus*	2	0	0.5	0	0	2.5
Yellow-billed Stork	*Mycteria ibis*	3	0	0	2	4	9
Northern Bald Ibis	*Geronticus eremita*	7	4	3.5	4	0	18.5
African Spoonbill	*Platalea alba*	3	0	0.5	2	0	5.5
Eurasian Spoonbill	*Platalea leucorodia*	3	0	0.5	4	2	9.5
Glossy Ibis	*Plegadis falcinellus*	1	0	1	4	0	6
Sacred Ibis	*Threskiornis aethiopicus*	2	0	0.5	2	0	4.5
Lesser Flamingo	*Phoenicopterus minor*	1	0	2	2	6	11
Greater Flamingo	*Phoenicopterus ruber*	4	0	1	4	0	9
Egyptian Goose	*Alopochen aegyptiacus*	2	0	0	0	0	2
Common Pintail	*Anas acuta*	1	0	0.5	2	0	3.5
Cape Teal	*Anas capensis*	2	1	0.5	2	0	5.5
Northern Shoveler	*Anas clypeata*	1	0	0	2	0	3
Common Teal	*Anas crecca*	1	0	0	2	0	3
Red-billed Duck	*Anas erythrorhyncha*	2	1	0.5	2	0	5.5
Hottentot Teal	*Anas hottentota*	2	0	0.5	2	0	4.5
Eurasian Wigeon	*Anas penelope*	1	0	0.5	2	2	5.5
Common Mallard	*Anas platyrhynchos*	0	0	0	0	0	0
Garganey	*Anas querquedula*	1	0	0.5	4	0	5.5
Gadwall	*Anas strepera*	1	0	0.5	2	2	5.5
Yellow-billed Duck	*Anas undulata*	2	1	0.5	2	0	5.5
Greater White-fronted Goose	*Anser albifrons*	1	0	0.5	4	2	7.5
Greylag Goose	*Anser anser*	2	0	0.5	0	2	4.5
Pink-footed Goose	*Anser brachyrhynchus*	2	2	1.5	6	2	13.5
Lesser White-fronted Goose	*Anser erythropus*	3	1	1.5	2	4	11.5
Bean Goose	*Anser fabalis*	2	0	0	0	2	4
Common Pochard	*Aythya ferina*	1	0	0.5	0	0	1.5
Tufted Duck	*Aythya fuligula*	1	0	0.5	0	0	1.5
Greater Scaup	*Aythya marila*	1	0	0.5	2	4	7.5
Ferruginous Duck	*Aythya nyroca*	2	1	1	4	2	10
Brent Goose	*Branta bernicla*	2	0	1	4	2	9
Barnacle Goose	*Branta leucopsis*	2	2	1.5	6	2	13.5
Red-breasted Goose	*Branta ruficollis*	3	2	2	4	2	13
Common Goldeneye	*Bucephala clangula*	1	0	0.5	0	2	3.5
Long-tailed Duck	*Clangula hyemalis*	1	0	1	2	2	6
Bewick Swan	*Cygnus columbianus*	2	0	1	4	2	9
Whooper Swan	*Cygnus cygnus*	2	0	0	0	2	4

Common name	**Species**	**Population size score**	**Range size score**	**Fragmentation score**	**Habitat score**	**Food score**	**Climate vulnerability score**
Mute Swan	*Cygnus olor*	2	0	0.5	2	2	6.5
Fulvous Tree-Duck	*Dendrocygna bicolor*	1	0	0	2	2	5
White-faced Tree-Duck	*Dendrocygna viduata*	1	0	0	2	0	3
Marbled Duck	*Marmaronetta angustirostris*	3	1	1.5	6	0	11.5
Velvet Scoter	*Melanitta fusca*	1	0	1	0	2	4
Common Scoter	*Melanitta nigra*	1	0	1	2	4	8
Smew	*Mergellus albellus*	2	1	1	0	0	4
Goosander	*Mergus merganser*	2	0	0.5	0	2	4.5
Red-breasted Merganser	*Mergus serrator*	2	0	1	0	0	3
Southern Pochard	*Netta erythrophthalma*	3	0	0.5	4	0	7.5
Red-crested Pochard	*Netta rufina*	2	1	1	4	2	10
African Pygmy-goose	*Nettapus auritus*	2	0	0	2	2	6
White-headed Duck	*Oxyura leucocephala*	3	1	2	6	2	14
Maccoa Duck	*Oxyura maccoa*	3	1	0.5	4	0	8.5
Spur-winged Goose	*Plectropterus gambensis*	2	0	0.5	2	2	6.5
Steller's Eider	*Polysticta stelleri*	2	0	1	6	2	11
Comb Duck	*Sarkidiornis melanotos*	2	0	0	2	0	4
Common Eider	*Somateria mollissima*	1	0	1	4	2	8
King Eider	*Somateria spectabilis*	1	0	1	4	2	8
South African Shelduck	*Tadorna cana*	3	2	1	4	0	10
Ruddy Shelduck	*Tadorna ferruginea*	2	0	0.5	4	0	6.5
Common Shelduck	*Tadorna tadorna*	2	0	0.5	4	4	10.5
White-backed Duck	*Thalassornis leuconotus*	3	0	0.5	2	2	7.5
Black Crowned-Crane	*Balearica pavonina*	3	1	1	4	0	9
Grey Crowned-Crane	*Balearica regulorum*	3	1	1	2	0	7
Wattled Crane	*Grus carunculatus*	4	1	1	4	2	12
Common Crane; Crane	*Grus grus*	2	0	1	4	0	7
Siberian Crane	*Grus leucogeranus*	5	1	2.5	4	4	16.5
Blue Crane	*Grus paradisea*	2	2	1.5	6	2	13.5
Demoiselle Crane	*Grus virgo*	2	1	1	6	0	10
Striped Crake	*Aenigmatolimnas marginalis*	4	1	0.5	4	2	11.5
Black Crake	*Amaurornis flavirostra*	4	1	1	2	0	8
African Crake	*Crecopsis egregia*	4	1	0.5	2	0	7.5
Corn Crake	*Crex crex*	1	2	1	4	0	8
Common Coot	*Fulica atra*	1	0	0	0	0	1
Crested Coot	*Fulica cristata*	2	1	0.5	2	0	5.5
Lesser Moorhen	*Gallinula angulata*	2	1	0.5	2	0	5.5
Common Moorhen	*Gallinula chloropus*	1	0	0	0	0	1
Allen's Gallinule	*Porphyrio alleni*	2	0	0	2	0	4
Little Crake	*Porzana parva*	2	1	1.5	4	2	10.5
Spotted Crake	*Porzana porzana*	2	1	0.5	4	0	7.5
Baillon's Crake	*Porzana pusilla*	4	0	0	6	2	12
Water Rail	*Rallus aquaticus*	2	0	0.5	4	0	6.5
Kaffir Rail	*Rallus caerulescens*	4	1	0.5	2	0	7.5
White-winged Flufftail	*Sarothrura ayresi*	6	3	4	6	2	21

Common name	Species	Population size score	Range size score	Fragmentation score	Habitat score	Food score	Climate vulnerability score
Streaky-breasted Flufftail	*Sarothrura boehmi*	4	2	1.5	4	4	15.5
Buff-spotted Flufftail	*Sarothrura elegans*	4	1	0.5	2	0	7.5
Crab Plover	*Dromas ardeola*	3	3	1.5	2	4	13.5
African Black Oystercatcher	*Haematopus moquini*	4	2	1	4	4	15
Eurasian Oystercatcher	*Haematopus ostralegus*	1	0	0.5	2	4	7.5
Black-winged Stilt	*Himantopus himantopus*	1	0	0	2	0	3
Avocet	*Recurvirostra avosetta*	2	0	0.5	2	0	4.5
Senegal Thick-knee	*Burhinus senegalensis*	3	1	0.5	2	2	8.5
Grey Pratincole	*Glareola cinerea*	3	2	1	2	4	12
Black-winged Pratincole	*Glareola nordmanni*	2	1	1.5	6	4	14.5
Rock Pratincole	*Glareola nuchalis*	3	1	0.5	2	4	10.5
Madagascar Pratincole	*Glareola ocularis*	4	3	1.5	6	4	18.5
Collared Pratincole	*Glareola pratincola*	2	1	0.5	4	4	11.5
Egyptian Plover	*Pluvianus aegyptius*	3	1	0.5	2	2	8.5
Kentish Plover	*Charadrius alexandrinus*	2	0	0.5	4	2	8.5
Caspian Plover	*Charadrius asiaticus*	2	2	1.5	6	2	13.5
Little Ringed Plover	*Charadrius dubius*	2	0	0.5	2	2	6.5
Forbes's Plover	*Charadrius forbesi*	3	1	0.5	2	2	8.5
Common Ringed Plover	*Charadrius hiaticula*	2	1	0.5	4	2	9.5
Greater Sand Plover	*Charadrius leschenaultii*	2	1	1	6	2	12
White-fronted Plover	*Charadrius marginatus*	3	0	1	4	2	10
Lesser Sand Plover	*Charadrius mongolus*	2	1	1	2	2	8
Chestnut-banded Plover	*Charadrius pallidus*	3	1	1.5	6	4	15.5
Kittlitz's Plover	*Charadrius pecuarius*	2	0	0.5	2	2	6.5
Three-banded Plover	*Charadrius tricollaris*	2	1	0.5	4	4	11.5
Eurasian Dotterel	*Eudromias morinellus*	2	2	1.5	6	2	13.5
Eurasian Golden-Plover	*Pluvialis apricaria*	1	1	1	2	0	5
Pacific Golden Plover	*Pluvialis fulva*	2	2	1	2	4	11
Grey Plover	*Pluvialis squatarola*	2	2	1	4	2	11
White-headed Lapwing	*Vanellus albiceps*	3	1	0.5	2	2	8.5
Crowned Lapwing	*Vanellus coronatus*	4	1	0.5	4	2	11.5
Sociable Lapwing	*Vanellus gregarius*	5	2	1	6	2	16
White-tailed Lapwing	*Vanellus leucurus*	3	1	1.5	6	2	13.5
Senegal Lapwing	*Vanellus lugubris*	3	1	1	2	2	9
Black-winged Lapwing	*Vanellus melanopterus*	4	1	1	6	2	14
Wattled Lapwing	*Vanellus senegallus*	2	1	0.5	2	2	7.5
Spur-winged Lapwing	*Vanellus spinosus*	2	0	0.5	2	0	4.5
Brown-chested Lapwing	*Vanellus superciliosus*	4	3	1.5	4	4	16.5
Northern Lapwing	*Vanellus vanellus*	1	0	0.5	2	2	5.5
Ruddy Turnstone	*Arenaria interpres*	2	1	1	4	4	12
Sanderling	*Calidris alba*	2	1	1	6	4	14
Dunlin	*Calidris alpina*	1	1	1	4	2	9
Red Knot	*Calidris canutus*	1	1	1	6	2	11
Curlew Sandpiper	*Calidris ferruginea*	1	2	1	4	2	10
Purple Sandpiper	*Calidris maritima*	2	2	1	4	2	11
Little Stint	*Calidris minuta*	1	2	1.5	6	2	12.5
Temminck's Stint	*Calidris temminckii*	2	1	1	4	2	10

Common name	**Species**	**Population size score**	**Range size score**	**Fragmentation score**	**Habitat score**	**Food score**	**Climate vulnerability score**
Great Knot	*Calidris tenuirostris*	2	2	1	4	4	13
Common Snipe	*Gallinago gallinago*	1	0	0	2	2	5
Great Snipe	*Gallinago media*	2	1	0.5	2	2	7.5
Pintail Snipe	*Gallinago stenura*	1	1	1	2	0	5
Broad-billed Sandpiper	*Limicola falcinellus*	2	1	1.5	4	2	10.5
Bar-tailed Godwit	*Limosa lapponica*	1	1	1	4	2	9
Black-tailed Godwit	*Limosa limosa*	2	0	1	2	0	5
Jack Snipe	*Lymnocryptes minimus*	1	1	0.5	2	2	6.5
Eurasian Curlew	*Numenius arquata*	2	1	0.5	2	0	5.5
Whimbrel	*Numenius phaeopus*	1	0	1	2	2	6
Slender-billed Curlew	*Numenius tenuirostris*	9	3	2.5	6	2	22.5
Grey Phalarope	*Phalaropus fulicaria*	1	0	1	4	2	8
Red-necked Phalarope	*Phalaropus lobatus*	1	1	1	2	4	9
Ruff	*Philomachus pugnax*	1	0	0.5	0	2	3.5
Eurasian Woodcock	*Scolopax rusticola*	0	0	0.5	4	2	6.5
Terek Sandpiper	*Tringa cinerea*	2	1	1	2	4	10
Spotted Redshank	*Tringa erythropus*	2	1	1	2	0	6
Wood Sandpiper	*Tringa glareola*	1	0	0	0	2	3
Common Sandpiper	*Tringa hypoleucos*	1	0	0	0	0	1
Common Greenshank	*Tringa nebularia*	1	1	0.5	0	2	4.5
Green Sandpiper	*Tringa ochropus*	1	0	0.5	0	0	1.5
Marsh Sandpiper	*Tringa stagnatilis*	2	1	0.5	2	2	7.5
Common Redshank	*Tringa totanus*	1	0	0.5	0	2	3.5
Whiskered Tern	*Chlidonias hybridus*	2	0	0.5	2	2	6.5
White-winged Tern	*Chlidonias leucopterus*	1	0	0.5	2	2	5.5
Black Tern	*Chlidonias niger*	2	0	0.5	2	2	6.5
Herring Gull	*Larus argentatus*	1	2	1.5	0	0	4.5
Armenian Gull	*Larus armenicus*	3	4	2	4	2	15
Audouin's Gull	*Larus audouinii*	3	2	1.5	6	4	16.5
Yellow-legged Gull	*Larus cachinnans*	1	1	1	0	0	3
Common Gull	*Larus canus*	1	0	0.5	0	0	1.5
Grey-headed Gull	*Larus cirrocephalus*	2	0	0.5	0	0	2.5
Kelp Gull	*Larus dominicanus*	1	0	0.5	4	0	5.5
Lesser Black-backed Gull	*Larus fuscus*	2	1	1	2	0	6
Slender-billed Gull	*Larus genei*	2	1	1.5	4	2	10.5
Iceland Gull	*Larus glaucoides*	2	1	1	6	2	12
King Gull	*Larus hartlaubii*	3	3	1.5	4	0	11.5
Sooty Gull	*Larus hemprichii*	2	1	1.5	4	0	8.5
Heuglin's Gull	*Larus heuglini*	2	2	1	2	0	7
Glaucous Gull	*Larus hyperboreus*	1	0	0.5	6	0	7.5
Great Black-headed Gull	*Larus ichthyaetus*	2	1	1	4	2	10
White-eyed Gull	*Larus leucophthalmus*	3	3	1.5	4	2	13.5
Great Black-backed Gull	*Larus marinus*	2	0	0.5	4	0	6.5
Mediterranean Gull	*Larus melanocephalus*	2	1	1	4	2	10
Little Gull	*Larus minutus*	1	0	1	0	2	4
Black-headed Gull	*Larus ridibundus*	1	0	0.5	0	0	1.5
Little Tern	*Sterna albifrons*	2	0	1	2	4	9

Common name	**Species**	**Population size score**	**Range size score**	**Fragmentation score**	**Habitat score**	**Food score**	**Climate vulnerability score**
Damara Tern	*Sterna balaenarum*	3	3	1.5	6	4	17.5
Lesser Crested Tern	*Sterna bengalensis*	2	0	1	2	6	11
Great Crested-Tern	*Sterna bergii*	4	0	1	2	6	13
Caspian Tern	*Sterna caspia*	2	0	1	2	4	9
Roseate Tern	*Sterna dougallii*	3	0	1	2	6	12
Common Tern	*Sterna hirundo*	1	0	0.5	0	2	3.5
Royal Tern	*Sterna maxima*	2	0	1	2	4	9
Gull-billed Tern	*Sterna nilotica*	2	0	1	2	2	7
Arctic Tern	*Sterna paradisaea*	1	1	0.5	6	4	12.5
White-cheeked Tern	*Sterna repressa*	2	1	1.5	4	4	12.5
Sandwich Tern	*Sterna sandvicensis*	2	0	1	2	6	11
Saunders's Tern	*Sterna saundersi*	3	1	1.5	2	2	9.5
Antarctic Tern	*Sterna vittata*	2	1	2	6	4	15
Sabine's Gull	*Xema sabini*	2	1	1.5	6	2	12.5
African Skimmer	*Rynchops flavirostris*	3	1	0.5	0	4	8.5

7.7. APPENDIX 2 – POPULATION VULNERABILITY TO CLIMATE CHANGE

Appendix 2. Populations listed on Table 1 of the AEWA Agreement, which are either critically threatened (score ≥ 20) by climate change (🟥) or highly threatened (score ≥ 17.5) by climate change (🟧). Win = winter, bre = breeding.

Common name	Species	Population	Population size score	Range size score	Fragmentation score	Habitat score	Food score	Climate vulnerability score
Gavia adamsii	White-billed Diver	Northern Europe (win)	6	4	2.0	4	2	18.0
Botaurus stellaris	Bittern	capensis - Southern Africa	5	3	2.5	6	0	18.5
Egretta vinaceigula	Slaty Egret	South-central Africa	5	4	2.0	6	2	19.0
Ciconia ciconia	White Stork	Southern Africa	9	5	2.5	6	0	24.5
Ciconia nigra	Black Stork	South-west Europe / West Africa	5	3	2.5	6	0	18.5
Ciconia nigra	Black Stork	Southern Africa	5	3	1.5	6	0	17.5
Geronticus eremita	Northern Bald Ibis	South-west Asia & South Asia (win)	10	7	4.0	4	0	27.0
Geronticus eremita	Northern Bald Ibis	Morocco	7	7	4.0	4	0	24.0
Platalea leucorodia	Eurasian Spoonbill	balsaci - Coastal West Africa (Mauritania)	4	6	3.0	4	2	19.0
Platalea leucorodia	Eurasian Spoonbill	leucorodia - West Europe / West Mediterranean & West Africa	3	5	2.5	6	2	18.5
Threskiornis aethiopicus	Sacred Ibis	aethiopicus - Iraq & Iran	7	4	2.0	6	0	21.0
Anas capensis	Cape Teal	Lake Chad basin	7	4	2.5	6	0	21.5
Anas hottentota	Hottentot Teal	Lake Chad Basin	5	4	2.0	6	0	19.0
Cygnus columbianus	Bewick Swan	bewickii - Northern Siberia / Caspian	6	4	3.0	4	2	19.0
Marmaronetta angustirostris	Marbled Duck	East Mediterranean	6	3	2.5	6	0	19.5
Marmaronetta angustirostris	Marbled Duck	West Mediterranean / West Medit. & West Africa	5	3	2.5	6	0	18.5
Melanitta fusca	Velvet Scoter	fusca - Black Sea & Caspian	5	4	2.0	6	2	19.0
Oxyura leucocephala	White-headed Duck	Algeria & Tunisia	7	5	2.5	6	2	22.5
Oxyura leucocephala	White-headed Duck	West Mediterranean (Spain & Morocco)	5	4	2.0	6	2	19.0
Grus grus	Common Crane	Turkey & Georgia (bre)	7	3	2.0	6	0	20.0
Grus leucogeranus	Siberian Crane	Iran (win)	10	5	2.5	6	4	25.5
Grus virgo	Demoiselle Crane	Turkey (bre)	9	5	2.5	6	0	24.5
Grus virgo	Demoiselle Crane	Black Sea (Ukraine) / North-east Africa	6	4	2.0	6	0	20.0
Fulica cristata	Crested Coot	Spain & Morocco	5	4	2.5	6	0	19.5
Sarothrura ayresi	White-winged Flufftail	Ethiopia and Southern Africa	6	8	4.0	6	2	26.0
Charadrius pallidus	Chestnut-banded	venustus - Eastern Africa	5	5	2.5	6	4	20.5

Common name	Species	Population	Population size score	Range size score	Fragmentation score	Habitat score	Food score	Climate vulnerability score
	Plover							
Vanellus gregarius	Sociable Lapwing	Central Asian Republics NW India	7	3	1.5	6	2	19.5
Vanellus gregarius	Sociable Lapwing	SE Europe & Western Asia / North-east Africa	6	3	1.5	6	2	18.5
Vanellus melanopterus	Black-winged Lapwing	minor - Southern Africa	5	3	1.5	6	2	17.5
Calidris alpina	Dunlin	schinzii - Baltic/SW Europe & NW Africa	5	4	2.5	4	2	17.5
Numenius tenuirostris	Slender-billed Eurasian Curlew	Central Siberia / Mediterranean & SW Asia	9	3	2.5	6	2	22.5

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