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Current and potential threats to Nordic duck populations — a horizon scanning exercise

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We review the current and future threats to duck populations that breed, stage, moult and/or winter in the Nordic countries. Migratory duck species are sensitive indicators of their changing environment, and their societal value confirms the need to translate signals from changes in their distribution, status and abundance into a better understanding of changes occurring in their wetland environments. We used expert opinion to highlight 25 major areas of anthropogenic change (and touch briefly on potential mitigation measures through nature restoration and reserve management projects) that we consider key issues likely to influence Nordic duck populations now and in the near future to stimulate debate, discussion and further research. We believe such reviews are essential in contributing to development of successful management policy as well as stimulating specific research to support the maintenance of duck species in favourable future conservation status in the face of multiple population pressures and drivers.

Introduction

Migratory waterbirds, especially duck species, are of considerable economic and societal importance (e.g. as huntable quarry, as source of down, indicators of wetland health, Green & Elmberg 2014) and are protected under international legislation (e.g. Ramsar Convention, African-Eurasian Waterbird Agreement and EU Birds Directive). Many populations move between highly productive ecosystems to breed and survive (including marine and high latitude areas, Dalby *et al.* 2014) making them potentially more able to adapt to change than more sedentary species. However, their reliance upon far flung networks of linked suitable habitats along their migratory corridors also potentially makes them more sensitive to global change. The annual migrations between different bioclimatic regions increase the likelihood of decoupling climate variables that could affect the same birds at different points in their annual cycles. As well as suffering increasing anthropogenic pressures, duck populations increasingly face the potential consequences of climate change, which could include mismatches in the timing and availability of food or refuge from predation. Potentially, these factors could ultimately affect emigration and immigration, survival and fecundity and hence changes in distribution and population size, recently reviewed by Guillemain *et al.* (2013).

Populations of 19 common native duck species breed, moult, stage and winter in the Nordic countries, mixing with individuals from Russia and elsewhere in Europe to disperse throughout the continent and into Africa during other stages of the annual cycle. The aggregation and mixing of individuals from a wide breeding provenance, especially at moult and winter quarters, makes monitoring of discrete duck populations difficult. Historically, mid-winter counts (e.g. Delany *et al.* 1999) have provided time series of relative abundance data for common ducks at large spatial scales. However, wintering populations draw large numbers of individuals from low densities of breeding birds over vast tracts of nesting territory, confounding attempts to understand drivers of change, which may be geographically constrained during the summer period. Recent shifts in wintering distributions (e.g. Lehikoinen *et al.* 2013, Pavón-Jordan *et al.* 2015) and problems with count coverage have further compounded the use of changes in wintering numbers as sensitive indicators of population change.

Concerns for the well-being of duck populations in Nordic countries have been expressed but have hitherto been founded on scant information. For example, after many years of sustained increase, many dabbling and freshwater diving ducks that winter in Britain are showing very recent signs of decrease (Holt *et al.* 2012). The population of some breeding and wintering sea ducks in the Baltic have more than halved during the past two decades (Skov *et al.* 2011, Ekroos *et al.* 2012a, Waldeck & Larsson 2013). But are these changes due to reductions in overall population size, shifts in distribution, to changes in our ability to monitor their numbers or a combination of all three? Although we have very poor annual measures of vital rates for common duck species, the proportions of young birds shot, for example, amongst wigeon *Anas penelope* in Britain and Denmark has been falling since 1982 (Mitchell *et al.* 2008, Christensen & Fox 2014) and survival of young from breeding to wintering grounds is low (Guillemain *et al.* 2010a). Long term analyses of annual adult survival rate time series for ducks species are very scarce and typically from birds caught on the wintering grounds of mixed breeding provenance (e.g. Devineau *et al.* 2010). An adequate understanding of factors affecting the population dynamics of duck species will only be forthcoming by following discrete defined "populations" of individuals where the overall rate of change in numbers is known, together with long term regular demographic monitoring to understand and interpret which vital rates are responsible for observed changes in population size.

Maintaining and restoring duck populations, whether as prey for hunters or to meet biodiversity targets, requires focused actions to mitigate the effects of specific adverse drivers of population change. We have often been spectacularly unsuccessful in foreseeing such changes in the past and one way of preparing ourselves for change in the future is to undertake systematic reviews of potential threats as well as positive drivers to population change in a way that can contribute to our ability to deal with such change (the

"horizon scanning" of Sutherland & Woodroof 2009, already applied to wader species by Sutherland *et al*. 2012). In this review, we attempt to assess some of the most important prevailing environmental and anthropogenic factors acting upon populations of Nordic breeding ducks, try to assess their relative weight as drivers and the causes of the changes we may observe and make predictions about other likely threats in the future, starting with the effects of climate change.

What features of changing climate are likely to become threats and drivers?

Autumn, winter and spring temperatures have increased in Scandinavia in recent years, which have contributed, for instance, to the declining extent of the ice cover in the Baltic Sea (European Environmental Agency 2012). In contrast, summer temperatures are predicted to increase only moderately as compared with those of other seasons. According to climatic scenarios, temperatures in northern Europe during winter are expected to increase much more rapidly than summer temperatures (Jylhä *et al*. 2004, European Environmental Agency 2012). Increasing winter temperature will lead to decreasing amounts of ice in arctic areas and in the Baltic Sea (Meier 2002, Meier *et al*. 2012, Jylhä *et al*. 2008, European Environmental Agency 2012). Furthermore, precipitation is expected to increase by up to 40% throughout the year, but with most profound effects in winter (Jylhä *et al*. 2004, European Environmental Agency 2012). "Windiness" is projected to increase, both in terms of extended wind run and strength; indeed unpredictable extremes of wind and many other features of the weather are currently expected (European Environmental Agency 2012). In the following sections, we consider the specific effects of climate and other environmental change on duck species in the Nordic countries.

Climatic effects on abundance and distribution in the non-breeding season

Migration dates of ducks are known to be affected strongly by variation in climate (Lehikoinen *et al*. 2006, Rainio *et al*. 2006, Lehikoinen *et al*. 2013) and habitat (Guillemain *et al*. 2015). Delayed autumn migration and short-stopping have caused several duck species to reduce their migration distance, possibly in response to these changes in winter climate. For example, it is well documented that ducks migrate shorter distances in mild winters compared to cold ones (Ridgill & Fox 1990) and several studies have found decreasing trends in migration distance (Sauter *et al*. 2010, Gunnarsson *et al*. 2012). Consequently, the wintering distributions of several species (particularly diving ducks) have shown dramatic changes at the edges of their winter range since 1980 and these changes can be explained by the increase in temperatures in the northern parts of the flyway, especially in the Baltic where ice cover has been reduced in the last 25 years (Lehikoinen *et al*. 2013, Pavón-Jordan *et al*. 2015). Regional trends among dabbling ducks suggest that these birds may have also responded by changing their distribution: countries on the northeastern end of the wintering range show increasing trends while countries further southwest show decreasing trends (Calbrade *et al*. 2010, Nilsson & Månsson 2010, Hornman *et al*. 2011). However, detailed flyway analysis of mid-winter counts failed to detect major changes in the centre of gravity of dabbling duck wintering ranges across western Europe, and no clear correlations between distribution and January temperature and precipitation, suggesting little shift in their winter distributions in recent decades, except perhaps for declines in wigeon in Ireland and Iberia (Dalby 2013). Milder winters also result in earlier spring migration, possibly in part because birds are wintering closer to their breeding grounds (Vähätalo *et al*. 2004, Rainio *et al*. 2006, Lehikoinen *et al*. 2006, Jónsson *et al*. 2009).

Climatic effects on survival during the non-breeding season

Milder climate may enhance survival due to lower energetic stress in European wintering sites, and mallard *Anas platyrhynchos* ringed in Sweden during migration show higher survival

rates in the 2000s than in the 1960s–1980s (Gunnarsson *et al*. 2012). Correspondingly, the body condition of mallard and teal *Anas crecca* has improved from the 1950s to the 2000s in French wintering areas (Guillemain *et al*. 2010b). However, rapid northward winter range shifts may also increase mortality risk. Wintering ducks typically exhibit poorest body condition at the end of winter (Fox *et al*. 1992, Loesch *et al*. 1992), and thus they can be even more vulnerable to occasional cold spells (given that extreme events become more frequent) than while wintering further south (*see* Suter & van Eerden 1992). Nevertheless, there is some evidence that mallard pair earlier in milder winters with likely fitness consequences in such seasons (Raitasuo 1963, Jónsson & Garðarsson 2001).

Furthermore, as wintering conditions become more favourable in the north, the situation may be the reverse in southern Europe. Decreasing rainfall in the Mediterranean region is expected to decrease the extent and quality of available wetlands (European Environmental Agency 2012) where precipitation and winter NAO are known to influence local wintering duck abundance (Almaraz *et al*. 2014). Further south, garganey *Anas quequerdula* and large numbers of pintail *Anas acuta* winter in the Sahel region of western Africa, where precipitation has remained below average levels since the 1970s despite recent slightly increasing trends in rainfall (Zwarts *et al*. 2009). Changes in the breeding populations of garganey and pintail were positively correlated with the size of the wetlands in their Sahel wintering areas (Zwarts *et al*. 2009); hence climate change affecting their winter quarters may cause population declines in these species (e.g. Pöysä & Väänänen 2014).

Climatic effects on reproductive success during the breeding season

Changes in the timing of onset of spring and the potential for decoupling between the peak in food availability and peak in resource demand (when offspring hatch) have received considerable attention in the ornithological literature (e.g. Both *et al*. 2006 but *see* also Dunn *et al*. 2011). However, ducks have received little attention in this regard. Pöysä and Väänänen (2014) suggested that spring weather conditions largely drove between-year variation in garganey numbers breeding at the northwestern edge of the species' range, although the positive effects of warm springs may be offset by negative effects of drought on the winter quarters. However, the little available evidence suggests that dabbling ducks are sufficiently flexible in their breeding phenology to adjust for changes in the timing of spring (Drever & Clark 2007, Sjöberg *et al*. 2011). Studies of the emergent patterns of chironomids in typical boreal lakes suggest these are neither compressed nor predictable enough to be a major selective force on the timing of egg-laying (and therefore hatching) amongst dabbling ducks. Indeed, duckling food supply (in the form of emerging chironomids) was high from just before until long after the peak of duckling hatching, implying some variation in the timing of hatching may not affect access to food (Dessborn *et al*. 2009). Breeding performance of mallard and teal were not adversely affected by advancements in ice break-up in Fennoscandia (Arzel *et al*. 2014). Conversely, freshwater diving ducks, which breed later than most dabbling ducks, seem to be less flexible and therefore at risk of becoming increasingly mismatched (Drever *et al*. 2012). Early spring temperatures are predicted to show more rapid increases than summer temperatures due to asymmetric climate change (Jylhä *et al*. 2004). Some duck species have shown a strong phenological response to variation in spring climate, and thus there is a risk that breeding is initiated too early. However, since late spring and early summer temperatures have not increased at the same pace, ducklings may hatch into colder conditions than previously although this is not studied in ducks (*see* Ludwig *et al*. 2006, Lehikoinen *et al*. 2009 for case studies of black grouse *Tetrao tetrix* and common buzzard *Buteo buteo*).

Increasing rainfall and windiness have been shown to reduce common eider *Somateria mollissima* duckling survival (Mendenhall & Milne 1985), whereas common goldeneye *Bucephala clangula* duckling mortality was unaffected by weather conditions (Paasivaara & Pöysä 2007). On the other hand, an advancing spring phenology, associated with earlier ice break-up, may

allow females to breed in a better body condition, as shown for common eiders (Lehikoinen *et al*. 2006, Jónsson *et al*. 2009). Thus, changes in climate may decrease reproduction output of some but not all duck species.

Climate effects on freshwater systems

A doubling of atmospheric CO_2 is predicted to elevate summer temperatures by an average of 4 °C in summer and 9 °C in winter in arctic and subarctic North America (Rouse *et al*. 1997), mirrored in Eurasia. The loss of permafrost will remove the impervious frosted layer close to the surface that maintains many summer lakes and wetlands during the summer (Mackay 1992, Woo *et al*. 1992), physically removing staging and breeding habitats for ducks. These processes will increase nutrient, sediment, and carbon loadings to aquatic systems, resulting in both positive and negative effects on freshwater chemistry (Wrona *et al*. 2006). Projected effects on waterbirds include altered migration routes and timing, a potential increase in mortality and decreased growth and productivity from disease and/or parasites, and probable changes in habitat suitability and timing of availability (Wrona *et al*. 2006). Climate change will alter characteristics of lakes that formerly froze to the bottom, allowing fish to colonise and exploit invertebrate prey, formerly the food source of migratory ducks (such as long-tailed duck *Clangula hyemalis*) in summer, while extended growth seasons will change trophic relationships, mineral and nutrient availability in aquatic systems (Rouse *et al*. 1997, Matell *et al*. 2013). Climate change may also affect aquatic systems and ducks during their annual migrations and reproductive periods through modified thermal regimes, reduced ice cover in lakes, altered stream flow regimes, increased salinity and increased water-development activities in the form of canal and reservoir construction (Poff *et al*. 2002). Projected changes in air temperature and rainfall will affect river flows, contaminant flows and dilution. Increased runoff will affect flow strengths and sediment loads, potentially altering river morphology and lake sedimentation, impacting freshwater duck habitats throughout the annual cycle in lakes and floodplains (Whitehead *et al*. 2009). More frequent extreme precipitation events on breeding and wintering areas, such as summer droughts, flash-flooding, reduced velocities and higher water residence times in rivers and lakes will enhance the potential for toxic algal blooms in freshwater, reducing dissolved oxygen levels especially at waterbird breeding habitat. Storms that terminate drought periods will promote increased bottom and shore erosion as well as increasingly flush nutrients from urban and rural areas or generate pulses of damagingly low pH waters flushed from acidified upland catchments (Whitehead *et al*. 2009). Changes in climate, land use and water chemistry are enhancing levels of dissolved organic carbon into freshwater systems, causing browning and warming of water courses and lakes with adverse effects on the food chain (e.g. Kritzberg *et al*. 2014).

Climate effects on peatlands

With elevated temperatures, northern peatlands will suffer lowered summer water tables, increased desiccation, reduced surface patterning (especially of lakes and pools which are of great value to waterbirds) and unsaturated, oxic conditions at their surface, resulting in aerobic decomposition and larger releases of CO_2 (Holden *et al*. 2004). These conditions will encourage the growth of shrub-dominated communities (Weltzin *et al*. 2003) as the tree line expands northwards and in altitude in the tundra and alpine habitats (Kullman & Öberg 2009, but *see* MacDonald *et al*. 1998, 2008). While such higher primary production could potentially compensate for carbon loss from soils, their presence fundamentally and adversely changes the nature of the system for waterbirds. Falling water tables also facilitate easier drainage for peat extraction and ultimately for agriculture and forestry, so exacerbating anthropogenic peatland habitat loss in ways not currently possible.

Climate effects on snow melt and thaw

Warmer springs will affect altitudinal gradients associated with topography and aspect. Lesser

white-fronted geese *Anser erythropus* and Greenland white-fronted geese *A. albifrons flavirostris* exploit the same food items at progressively higher altitudes as they are progressively released by the summer thaw in mountainous terrains (Ekman 1922, Fox and Stroud 1981) and it seems likely that duck species exploit such gradients as well. Under current climate change scenarios, it might be expected that warming will both advance the start of such melt patterns and progressively reduce the delay between melt at lowest levels and that at highest altitudes, potentially releasing food resources earlier and reducing the benefit of staggered melt in the slow release of food through the summer season. With less snow cover during the early breeding period, more simultaneously-available breeding habitat could release density-dependence and lead to reduced predation pressure by ground predators, such as mustelids and foxes.

Climate effects on land use

Climate warming will extend cropping areas northwards in Europe, introducing new crop species and varieties, higher crop production and expansion of suitable areas for crop cultivation, enhancing nutrient leaching and the turnover of soil organic matter, enhancing loss and degradation of wetland habitats (Olesen & Bindi 2002). In southern areas, increases in water shortage and extreme weather events may reduce harvestable yields, although this may result in the abundance of reservoirs that provide permanent summer and winter water in otherwise increasingly arid areas (Olesen & Bindi 2002). These processes will likely reinforce the current trends of intensification of agriculture and urbanisation in northern and western Europe, as well as in the Mediterranean and southeastern parts of Europe, with associated effects on wetlands and waterbirds.

Climate effects on seawater salinity, acidity and temperature

Climate change can drive long-term changes in the quality of duck breeding habitat as exemplified in the marine environment by the Baltic population of the common eider, which is dependent on its main prey species, the blue mussel *Mytilus edulis* for accumulating body stores during winter for subsequent investment in reproduction (Laursen *et al*. 2009). Blue mussels require saline conditions (Westerbom 2006), which explains why both species become increasingly rare in areas of low salinity (although small numbers of common eiders still breed in the eastern Gulf of Finland and the Gulf of Bothnia despite the complete absence of blue mussels there, Valkama *et al*. 2011). Rainfall is predicted to increase in northern Europe, reducing the salinity of the Baltic Sea (Meier *et al*. 2006, Mackenzie *et al*. 2007) impacting mussels and the already declining Baltic common eider population (Westerbom 2006, Ekroos *et al*. 2012a) and other mussel-feeding duck species as well. Rising water temperatures reduce shell strength (Mackenzie *et al*. 2014) which could potentially benefit mussel-eating birds and acidification of inshore marine waters reduces the strength of byssal threads in *M. edulis*, reducing the robustness of substrate attachments of this important prey item, especially in the face of expected increased windiness (O'Connell *et al*. 2013). Projected changes in seawater temperature will also affect the quality of blue mussels and the ducks that feed upon them, since increasing winter water temperatures may severely reduce bivalve flesh content (Waldeck and Larsson 2013).

Climate effects associated with sea level rise

Coastal habitats are globally among the most important feeding, roosting and breeding areas to waterbirds in general and the Anatidae in particular. Shallow waters, mudflats and salt marshes offer good quality forage, undisturbed roosting sites and suitable nesting areas for many duck species (Bellrose 1980, Scott & Rose 1996). However, climate change-induced rises in sea level and associated coastal squeeze threaten to drastically reduce the area of available coastal habitats (Hughes 2004, FitzGerald *et al*. 2008). In the modern landscape, the majority of coast-

lines are bordered by coastal protection measures (dikes, dunes, elevated field boundaries) to effectively protect human agricultural interests but which prevent the natural inland migration of coastal habitats and result in net waterbird habitat loss when sea levels rise. Hence, the very large aggregations of waterbirds wintering in coastal, shallow wetlands are especially under threat from rising sea levels (Guillemain *et al.* 2013). For instance, Clausen *et al.* (2013) found that the area of salt marsh in Danish Special Protection Areas for birds will decline by 15%–44% under different climate change scenarios during the next century. Likewise, substantial areal loss has been modelled for both shallow water macrophyte habitats (up to 37%) and mudflats (up to 62%) as a result of sea level rise up to 1 m. (Clausen & Clausen 2014). Furthermore, potential nest losses may result from more frequent flooding (van de Pol *et al.* 2010) and saline intrusion into coast-near wetlands that are currently freshwater or brackish in nature (often exacerbated by drainage activities e.g. Holman *et al.* 2013). Sea level rise results in more frequent flooding of coastal areas that would otherwise be dry, which may positively affect the availability of plant seeds for filter-feeding dabbling ducks (Therkildsen & Bregnballe 2006) and potentially partially compensate for habitat loss discussed above.

The expected drop in available coastal habitats can be counteracted by means of more efficient salt marsh management (Clausen *et al.* 2013), and managed retreat of coastlines by removing existing sea walls in coastal areas (Pethick 1993, Clausen & Clausen 2014). Since such measures invariably interfere with traditional animal husbandry practices and/or involve the physical loss and change of privately owned property, they can only be expected to be implemented via subsidy schemes for local land owners with associated environmental impact assessment to secure duck habitat.

Effects of changing predation risk during breeding

Duck populations may face increased predation pressure (to which they are inherently vulnerable due to their predominantly ground nesting habits), both as a result of climate change and from other direct and indirect anthropogenic impacts. Firstly, climate change may affect reproduction by altering predator–prey interactions. Small rodents (voles and lemmings) used to show 3–5-year population cycles in northern Europe (Hanski *et al.* 1991), but in recent years these cycles have become more irregular and their amplitude has decreased (Kausrud *et al.* 2008, Cornulier *et al.* 2013, Nolet *et al.* 2013). Rodent cycles affect reproductive success and survival of ducks because predation pressure peaks during the low phase of the cycle (when ducks serve as alternative prey), but when small mammals are common predation pressure can be lighter (e.g. Iles *et al.* 2013). The evidence for covariation between vole cycle phase and duckling production mainly comes from arctic species (e.g. long-tailed duck, Steller's eider *Polysticta stelleri* and common eider, Quakenbush 2004, Zydelis *et al.* 2006, Hario *et al.* 2009, Iles *et al.* 2013), and this relationship has so far not been studied in the boreal zone. However, Nordic data on mallard (Dalby *et al.* 2013a) and wigeon (Christensen & Fox 2014) breeding success shows signs of cyclicity, which could potentially also be driven by predator abundance. A warming climate may facilitate the northward expansion of competitively dominant predator species, such as the red fox *Vulpes vulpes* and raccoon dog *Nyctereutes procyonoides* (e.g. Killengreen *et al.* 2007, Kauhala & Kowalczyk 2011), which will have as yet unknown, but potentially severe, consequences on waterbird population dynamics, particularly in northern Scandinavia (where a recent red fox cull programme changed 5% annual reductions in breeding numbers of lesser white-fronted geese to > 20% annual increases; T. Aarvak unpubl. data). However, recent studies of mammalian nest predator removal in the North American prairies did not enhance mallard productions to levels as previously expected and may therefore not be a viable or economic strategy for increasing duck productivity under all circumstances (e.g. Amundson *et al.* 2013).

Anthropogenic impacts through introduced or invasive predators may pose threats to duck populations. At the SPA/Ramsar site at Lake Engure, Latvia, all studied waterbirds (except

waders) during 1948–2011 experienced an initial phase of population growth, followed by recent declines. Besides eutrophication, one of the main drivers of the similar population fluctuations in these species was elevated predation pressure from two introduced alien mammal predators, the raccoon dog and American mink *Mustela vison*, as well as by the native red fox (Vîksne *et al.* 2011). Larger duck species such as common eiders may locally experience severe short-term population losses due to peaks in mammalian predation, particularly by raccoon dogs (Ekroos *et al.* 2012b). Since it was brought to Europe for fur farming in the 1920s, the American mink has spread widely and is considered a major predator of ducks and duck nests. Removal of mink from 60 Finnish islands proved highly successful and showed that mink presence affected nest-site selection of ducks, increasing the attractiveness of isolated breeding sites (Nordström & Korpimäki 2004) and that smaller duck species may be disproportionately affected by mink predation (Nordström *et al.* 2002). Recovery in numbers of breeding birds of a range of species following mink eradication (including tufted ducks *Aythya fuligula* and dabbling ducks) showed the impacts of mink predation on duck and other bird populations and, consequently, the eradication programme was extended to other parts of the archipelago (Banks *et al.* 2008).

Finally, the recent rapid recovery of large avian predators such as the white-tailed sea eagle (*Haliaeetus albicilla*) in the Nordic countries under protective legislation has reduced adult and offspring survival of ducks. Female common eiders are typically highly site faithful to breeding sites (e.g. Öst *et al.* 2011), but site fidelity may carry a substantial survival cost under conditions of rapidly changing predation risk (Ekroos *et al.* 2012b). Eagle predation significantly reduced common eider survival amongst females nesting on less vegetated islands compared to those on forested ones (and was the lowest survival recorded anywhere, Ekroos *et al.* 2012b). Increases in white-tailed sea eagle abundance has elevated common eider duckling predation, modified parental care strategies (Jaatinen *et al.* 2011, Jaatinen & Öst 2013), encouraged cooperative parental care strategies, increased female group sizes (Jaatinen *et al.* 2011) and increased the rate at which brood-tending associations form (Jaatinen & Öst 2013). The net effect of such behavioural adaptations on overall reproductive output of duck populations is unclear, but overly large brood-rearing groups may be more exposed to visually hunting predators, or increased disease transmission among ducklings (Jaatinen *et al.* 2011). Furthermore, elevated predator pressure on nesting common eider females may be the main driver of the increasingly male biased sex ratio in the Baltic/Wadden Sea flyway population (Lehikoinen *et al.* 2008), with potentially severe long-term consequences for population level productivity and viability.

Even if breeding ducks show positive responses to new predator regimes and pressures, such adaptation may be complicated by the fact that whereas elevated mammalian predation is expected to lead to selection for more isolated nest-sites (Nordström & Korpimäki 2004), increasing predation pressure from particularly white-tailed sea eagles may select for the opposite pattern, i.e. nesting on isolated and open islands may be associated with a higher predation risk (Ekroos *et al.* 2012b). Such opposing selection pressures may expose ducks to choose between the "devil and the deep blue sea". On the one hand, however, adaptation may still be possible over evolutionary time, considering that ducks, as has been shown in cavity-nesting common goldeneyes, are demonstrably able to assess nest predation risk and actively select safe nest-sites (Pöysä 1999, 2006). On the other hand, the extreme breeding philopatry of female ducks may constrain the selection of safe breeding habitat, as island-breeding ducks such as common eiders are reluctant to switch breeding islands even after repeated nest failure (Öst *et al.* 2011). In such cases, the ability of first-time breeders to move to safe breeding areas may play a key role in the possible adaptation to shifting predation pressures, and this question is open to further investigation (Ekroos *et al.* 2012b).

Effects of freshwater eutrophication

Changing land use and increasing atmospheric nutrient loads are contributing nutrients to

natural systems that are successively affecting the trophic status of wetlands, lakes and water courses from their source to the sea (e.g. Carpenter *et al*. 2011, Jarvie *et al*. 2012), all of which in turn affect conditions for waterbirds at all stages of their life cycles. Oligotrophic (nutrient poor) wetlands tend to support limited growth of plants, algae, invertebrates and fish, generally offering poor feeding conditions for ducks, which therefore use such lakes in smaller numbers. In contrast, under normal circumstances, eutrophic conditions in wetlands generally favour duck communities, because they support productive plant and associated invertebrate communities in shallow water for dabbling ducks and algae-zooplankton-arthropod/fish food webs in deeper water that provide food for diving ducks. The range of Nordic duck species that use freshwater wetlands tend to winter in eutrophic waters in southern parts of Europe, the Black Sea and North Africa, breed in low densities over vast areas of the tundra and boreal zones in northern parts, as in Iceland, Fennoscandia and Russia. Breeding areas tend to have low human population density and limited agricultural development, whereas staging and wintering areas lie in some of the most populated and intensively cultivated parts of the planet. This combination has created hyper-eutrophication problems because of nutrient enrichment as a result of fertilizer, sewerage and other human influenced runoff that contributes especially nitrogen and phosphorus in accessible form to biological systems, causing massive phytoplankton blooms, turbid waters and hypoxia. Such perturbations radically affect plant, invertebrate and fish communities with usually adverse effects on breeding, staging and wintering ducks. Eutrophication of freshwaters predictably increases algae biomass because of shifts in the flux of growth-limiting nutrients (Smith *et al*. 2006), can affect the extent of floating and emergent habitat (denying breeding ducks access to open water, e.g. Vîksne *et al*. 2011) and has generally been detrimental where hyper-eutrophication of eutrophic lakes has occurred (as on important duck breeding areas in Finland, Ekholm & Mitikka 2006, Pöysä *et al*. 2013). In combination with elevated temperature, such changes in water chemistry may also increase the frequency of outbreaks of disease, such as botulism (Anza *et al*. 2014). However, at the other extreme, reductions in runoff of agricultural nitrogen and phosphate into Lough Neagh (an important wintering lake for diving ducks in Northern Ireland) have apparently been associated with reductions in chlorophyll in the water and in the density and biomass of benthic invertebrates, which in turn have had deleterious effects on the wintering waterbirds (especially diving ducks) of the site (Tomankova *et al*. 2013, 2014).

Effects of brackish eutrophication

Adverse effects of hyper-eutrophication and other factors has been evident in inshore, estuarine and brackish lagoon waters in the UK and the Netherlands for many decades, and have resulted in mass reductions of eelgrass *Zostera* spp. beds and associated organisms (e.g. de Jonge *et al*. 2000, Garrard & Beaumont 2014). Much of the habitat loss occurred before the advent of waterbird monitoring schemes, hence the true impacts on staging and wintering duck populations remain anecdotal. In Denmark, large and important *Zostera* and other sea-grass communities remain, providing food for thousands of ducks, geese, swans and coots, although hyper-eutrophication problems have been evident since the mid-1970s. The first systematic waterbird surveys identified the Wadden Sea, Ringkøbing Fjord, Nissum Fjord, and Limfjorden as the most important staging areas for dabbling ducks in the country (Joensen 1974). Duck populations on the Tipperne reserve in Ringkøbing Fjord (monitored since 1928) numbered 20 000–80 000 dabbling ducks in autumn for more than half a century until the submerged vegetation collapsed following massive algae blooms in 1978 and 1979. Regular algae blooms, poor water quality and low plant and fish stocks have prevailed since then, so duck numbers have declined, and the lagoon ecosystem has so far only partly recovered despite massive investments in nutrient reduction programmes and changes in sluice-salinity management regimes of the lagoon (Petersen *et al*. 2008, Meltofte & Clausen 2011). Hyper-eutrophication problems have occurred in Nissum Fjord since the 1980s, some parts of

the Limfjord since the 1980s and other parts in the 1990s, reducing submergent plant stocks and associated herbivorous waterbirds (Clausen & Percival 1998, Clausen & Holm 2011).

Effects of marine eutrophication

Populations of birds are generally food limited, and human activities, especially increased use of fertilizers in agriculture, have also resulted in increases in nitrogen and phosphorus concentration in coastal waters (Møller *et al*. 2007). This has caused cascading effects on phytoplankton, zooplankton and, ultimately, biodiversity in the marine environment, although trophic relationships can be complex (Møller *et al*. 2015). Strong positive correlations between primary productivity and fish production (Nielsen & Richardson 1996, Nixon & Buckley 2002) have knock-on effects for breeding arctic terns *Sterna paradisaea*. In this species, increased clutch size, reduced recruitment rate and decreased longevity have been associated with increased fertilizer levels (Møller *et al*. 2006, 2007) so piscivorous ducks may show similar relationships. In Denmark and the Netherlands, there were strong positive relationships between fertilizer use and population size of some breeding and wintering waterbirds, but stronger negative relationships amongst herbivores and piscivores than in species feeding on the benthos (Philippart *et al*. 2007, Møller *et al*. 2015). It may be the case that in early times, severe organic pollution elevated marine benthic bivalve populations that supported much higher levels of feeding sea ducks than is now the case (e.g. in the Pomeranian Bay; Kube 1996). Indeed, large aggregations of diving ducks associated with distillery waste and sewerage outfalls have disappeared when the major organic fractions in such disposal sites have been removed (e.g. in Scotland; Campbell 1984). Furthermore, some of the adverse effects of eutrophication in the Baltic may be counteracted by the reduction of ice cover that enables mixing of surface waters and penetration of oxygen to lower levels in the water column (Eilola *et al*. 2013). Breeding common eider abundance at the Christiansø colony in the Baltic Sea was closely correlated with increasing nutrients in sea water during 1920–1996, also followed by a reduction in nutrient concentrations in the mid-1990s that gave rise to an ecosystem cascade effect that reduced mussel stocks, the staple food of common eiders (Laursen & Møller 2014).

Wetland restoration and habitat creation

Although the scientific community and some local authorities in Denmark have been concerned about water quality and the deteriorations of wetland quality in Denmark since the 1970s, actions were rapidly stepped up when summer hypoxia-induced mass mortalities of lobsters were reported by fishermen on prime-time television in 1986. Subsequently, Ringkøbing Fjord was listed on the Montreux record of wetlands showing unfavourable conditions under the Ramsar convention in 1990. Since then, three national action plans to improve the aquatic environment have been launched, primarily to reduce releases of phosphorus (from household sewage) and nitrogen (from agriculture). Reductions in these nutrient releases are well-documented (Carstensen *et al*. 2006, Windolf *et al*. 2012), and while some lake restoration effects have been promising, the lower parts of many catchments are not showing major improvements, because of the complex interplay of processes involved (e.g. sediment roles in nutrient availability, Søndergaard *et al*. 2013).

One of the first and best documented lake restoration schemes was Lake Veluwe in the Netherlands created in 1957 adjacent to the Ijsselmeer. In the early 1960s, it supported a rich density of submerged macrophytes (mainly *Chara* and *Potamogeton*) and zebra mussels *Dreissena polymorpha* attracting large numbers of feeding ducks. In the late 1960s, the system completely collapsed after major algae blooms and these continued to occur throughout the 1970s and 1980s (Van den Berg *et al*. 1999, Noordhuis *et al*. 2002, Ibelings *et al*. 2007). Restoration started in Lake Veluwe with a phosphorous removal system in 1979, but it took two decades before major returns of the macrophytes, mussels and waterbirds were observed. Although diving ducks and long-necked herbivorous swans returned, dabbling ducks did not, because food plant densities

in the shallowest parts accessible to these short-necked species remained low (Noordhuis *et al.* 2002). One of the largest wetland restorations in Europe took place in the lower Skjern River valley in west Jutland, where after 35 years of drainage and intensive arable tillage, the river was re-engineered to its original meanders and flooding regime, creating 22 km^2 of lakes, shallow wetlands and seasonally flooded grazed wet grassland costing 38 million euros. The primary motivation was to restore the sediment/nutrient retention capacity of the river valley to reduce eutrophication of Ringkøbing Fjord at its efflux. Breeding waterbird species richness and diversity increased after restoration and mean breeding numbers of all waterbird species increased from 134 ± 22.9 SE (n = 3 years) pairs to 1744 ± 153 SE (n = 5 years). Twenty-nine waterbird species returned to breed, 10 of national or international significance (Danish Red List/European Union Birds Directive Annex 1 species) now ranking Skjern River amongst the top 10 most important breeding and staging waterbird sites in Denmark (Bregnballe *et al.* 2014). These case studies, and others from elsewhere throughout Europe, show the possibility to create and restore wetlands, reverse the more adverse effects of eutrophication and create extensive habitats for ducks at all stages of their annual cycle. However, large scale projects are expensive and often require more bio-remediation than just nutrient load reductions (e.g. fish manipulations, mud-removal) to ensure success. Unfortunately, many restoration attempts are only successful in the short-term, with turbid-water and poor vegetation states returning later (Gulati & van Donk 2002, Søndergaard *et al.* 2007). Nonetheless, when well-planned in the longer term, they can be highly successful, as for example at the Maribo Lakes, Denmark, where the combined effects of nutrient reductions and fish-manipulations led to a major recovery of submerged water plants, and ultimately breeding and staging duck populations (Clausen *et al.* 2014).

Diseases/parasites

Climate change has the potential to affect life cycles and transmission of infectious diseases affecting ducks (Harvell *et al.* 2002, Altizer *et al.* 2013, Garrett *et al.* 2013), their ranges and those of disease vectors (Rogers & Randolph 2000, Zamora-Vilchis *et al.* 2012). Parasite communities may change in response to the distribution of parasites and intermediate hosts (Zamora-Vilchis *et al.* 2012, Pickles *et al.* 2013), which generally show pole-ward movements (Harwell *et al.* 2009, Lafferty 2009). However, the complexity of such changes are widely acknowledged considering species-specific differences in responses of infective agents and vectors to changes in temperature, precipitation, salinity, drought etc., compounded by potential mismatches which may arise between hosts, prey and vectors (Schweiger *et al.* 2008, 2012). New disease patterns may also arise through climate-induced changes in food occurrence and abundance, where changes in diet expose ducks to new parasite species or heavier parasite loads in alternative foods (Burek *et al.* 2008). Parasitism may also interact with eutrophication (*see* e.g. Green *et al.* 2011). Climate-induced changes in prey abundance and diversity, or in food quality, may also compromise duck immune competence when individuals fail to optimise body condition, rendering birds more susceptible to diseases and/or parasite infections (O'Hara & O'Shea 2001), contaminants and toxic effects of bioactive metabolites (Jenssen 2006, Noyes *et al.* 2009).

Migratory ducks exploit a chain of aquatic environments, which offer favourable conditions for virus and parasite transmission and survival in space and time (Webster *et al.* 1978, Lu *et al.* 2003). Surface-feeding species may be especially susceptible to transmission through oral-oral contact (respiratory route) and faecal-oral contact (e.g. concentrated cloacal virus excretion into the water; e.g. Sturm-Ramirez *et al.* 2005), and the prevalence of diseases may be maintained or reinforced through intra- and inter-specific aggregations of large numbers of birds at various multispecies stop-over and wintering sites. Virus prevalence is higher amongst long-distance migratory species, and lower in species foraging in marine habitats (Garamszegi & Møller 2007). The complexity of the interactions between climate change, host and vector species ecology and that of the diseases and parasites, makes it difficult to make species-specific pre-

dictions of effects, as climate change most probably will have adverse effects on the reproductive performance and survival in some species, but not on others. Consequently, monitoring and research are needed to determine relationships between disease occurrence and climate change, as the ecology and migratory behaviour of ducks makes them especially susceptible.

Fisheries by-catch and mussel fisheries

Birds drown in fisheries nets and equipment, but assessing the extent and population impacts can be challenging (Žydelis *et al*. 2009). Nevertheless, shifts in ranges of fish species favoured for human consumption can have major effects on survival of especially long-lived seaduck species that are susceptible to drowning in nets. Lumpfish *Cyclopterus lumpus* fisheries occur on inshore waters in Newfoundland, Greenland, Iceland, Norway, Germany and the Baltic States, which are also the feeding grounds of common eider, where drowning may threaten local eider breeding populations (Thorsteinsson & Marteinsdóttir 1992, Merkel 2004b). In Greenland and Iceland, the death toll is estimated to be in thousands of birds. In Iceland, 0.3% (Breiðafjörður Bay) and 1.3% (in NW Iceland) of all common eiders were estimated to have drowned in nets in the summer of 1987 (Thorsteinsson & Marteinsdóttir 1992). In Norway, lumpfish fisheries are declining where these previously drowned wintering Steller's eiders (Fox *et al*. 1997), partly due to variable availability and price, but also ironically due to increase in by-catch of red king crab *Paralithodes camtschaticus*, a generalist feeding decapod, introduced to the Barents Sea (Murmansk Fjord, Russia) during late 1960s (*see* below; Mikkelsen & Pedersen 2012).

Elevated market prices have supported the expansion of the mussel fishery during the late twentieth century in several west-European countries (e.g. the Netherlands, Denmark and England) where the resulting overexploitation of mussel stocks has had consequences for waterbirds. In the Netherlands, mass winter starvation of common eiders (Camphuysen *et al*. 2002) and oystercatcher (*Haematopus ostralegus*) occurred following intensive mussel fishery and in the Danish Wadden Sea, common eider numbers declined, their distribution changed and a significant decrease in body condition occurred (Laursen *et al*. 2009).

Artificial stocking of native species

Introduced and native species have the potential to impact ducks in a variety of adverse ways through competition for space and food as well as predators (*see* above). In the case of the mallard, release of large numbers of hand-reared birds for hunting purposes is a common practice in many European countries. In France, 1.4 million mallards are released each year (Mondain-Monval & Girard 2000), 400 000 annually in Denmark (Noer *et al*. 2008) and at least 250 000 in Sweden (Söderquist 2015), compared to 20 000–50 000 breeding wild pairs in Denmark (BirdLife International 2004) and 200 000 pairs in Sweden (Ottoson *et al*. 2012). Although introgression of hand-reared mallard DNA into the wild French mallard population was limited (Champagnon *et al*. 2013), hand-reared mallards released in Sweden have a significantly different genetic structure compared to the wild mallard population in the Nordic countries (Söderquist 2015). This genetic introgression of farmed mallard genes may have led to the change in bill morphology found in Nordic mallard populations, not only in the country of release but also in neighbouring release free countries (Söderquist *et al*. 2014). Releases may affect the migration of mallard (Söderquist *et al*. 2013) and also facilitate the spread of diseases, such as avian influenza (Handberg *et al*. 2010, Vittecoq *et al*. 2012) and may increase intra- as well as interspecific competition. More research is needed about the possible competitive effects of releases (Gunnarsson *et al*. 2013). However, since hunting mortality is by far the most common cause for recovered wild mallard (Gunnarsson *et al*. 2008), the large releases of hand-reared mallard must contribute to relieving some hunting pressure on the wild mallard population, as well encouraging land owners and managers to improve habitat and restore wetlands at release sites, which will benefit wild birds (Vildtforvaltningsråd 2006).

Spread of introduced/invasive species

Invasive alien species are considered to be one of the greatest current threats to natural systems and species diversity (Sala *et al*. 2000). It's estimated that they increasingly cost the European Union (EU) more than 12 billion euros per year, resulting in regulation that came into force on 1 January 2015 (Sundseth 2014). Invasive, alien ducks themselves can constitute a threat, as was the case for the North American ruddy duck *Oxyura jamaicensis*, which escaped from captivity in England. This species dispersed and began hybridising with the native white-headed duck *Oxyura leucocephala* in Spain, where it threatened the local and previously successful conservation efforts to restore the white-headed duck population from the brink of extinction (Hughes *et al*. 2006). By the winter of 2014/2015, an implemented eradication strategy was close to success, largely due to the concentrated campaign in the United Kingdom, at a cost of between 5 and 7.5 million Euros (Robertson *et al*. 2015).

Since the red king crab became established in the southern Barents Sea in the late 1970s, it has become invasive and spread widely from Kolguev Island along the coast of northern Norway to Sørøya (Britayev *et al*. 2010, Mikkelsen & Pedersen 2012). The crab is an active predator, feeding in deep soft-bottom environments. Studies carried out in Norway and Russia indicate reduced species richness, density and biomass of soft-bottom epifauna and infauna in crab-invaded areas (Britayev *et al*. 2010, Oug *et al*. 2011) which are the most important winter feeding habitats for wintering Steller's eiders and long-tailed ducks, the species most adversely affected by the crabs (Bustnes *et al*. 2000, Bustnes & Systad 2001a, 2001b). There is abundant evidence of competition between fish and ducklings in freshwater systems (e.g. Eriksson 1979, Hunter *et al*. 1986), so the introduction of trout species to upland lakes in Norway has been suspected as a cause of reduced reproductive success in breeding common scoter *Melanitta nigra* (I.K. Petersen pers. comm.). Other invasive species can provide a novel food resource and enhance wintering duck densities and diversity (e.g. Werner *et al*. 2005). Ducks may themselves also be vectors of alien invasive species, spreading propagules and dispersing plant and animal forms (e.g. Garcia-Álvarez *et al*. 2015). Some of these invasive species may offer a source of food for ducks, whilst others may have major adverse consequences for aquatic ecosystems (as in the case of the South American plant *Ludwigia grandiflora*, which has a detrimental effect on native plant and macroinvertebrate communities in Europe e.g. Stiers *et al*. 2011). Greater vigilance is required to monitor effects of such invasive/introduced species whose spread is often started by human intervention, but subsequently aided by climate change.

Overexploitation

Sustainable harvest of ducks has become a vital basis to management of hunting throughout the Nordic states, although balancing exploitation with favourable conservation status remains a challenge for some populations. All eider duck species have been harvested in some parts of their range, at least until recently, including king and common eider (Merkel 2004a, 2004b, Chaulk *et al*. 2005, Powell & Suydam 2012). Hunting was believed to have caused declines in king and common eiders in West Greenland in the 20th century, because declines were observed mostly nearby human settlements (Christensen & Falk 2001). In Labrador and Greenland, common eider populations were in decline until the end of the 20th century due to overharvest (Merkel 2004a, Chaulk *et al*. 2005), because both populations subsequently recovered quickly after shortening of the spring hunting season and implementation of a successful education programme (Chaulk *et al*. 2005, Merkel 2010). Recent declines in the Baltic-Wadden Sea common eider population have also precipitated innovative selective hunting regulation that protects females, with the aim of restoring the population to favourable conservation status without the need to totally close the hunt (Christensen & Hounisen 2014). Many duck species have relatively short generation times but are highly fecund, making them better suited to sustain a viable harvest than many other longer lived avian species and those with low reproductive success (Lebreton 2005). However, the

problem remains that the numbers of each species shot each year remains unknown, making it impossible to assess the sustainability of current harvest levels.

As well as mortality, hunting is widely acknowledged to cause disturbance which in autumn could potentially reduce the time spent by staging ducks at a given resort, pushing birds more rapidly down the flyway towards ultimate winter quarters (Madsen & Fox 1997). The designation of substantial areas of hunting-free protected areas within large autumn staging areas for ducks within Denmark resulted in elevated numbers of locally staging birds in a way consistent with prolonged local length of stay, rather than as a result of drawing other birds from local wetlands (Madsen 1998). The instigation of a nationwide programme of such hunting-free protected areas throughout Danish wetlands was planned specifically to elevate the carrying capacity of such wetlands (Madsen *et al*. 1998), which was likely to result in a slower passage of ducks passing down the flyway to wintering areas further, with potential consequences further down the flyway (Madsen & Fox 1997). If sufficient previously unoccupied capacity could be made available to autumn staging ducks by further reducing disturbance from hunting, such a network of reserves could support greater numbers of ducks even in mid-winter. This could potentially lead to a net short-stopping of the overall population in geographical areas short of those occupied in former times (*sensu* Elmberg *et al*. 2014, as we are now seeing as a result of climate change, *see* above).

Pollutants and contaminants

The survival and/or reproductive success of many seaduck populations can be compromised by pollution, particularly because of large-scale oil spills (Esler *et al*. 2000, 2002). Recent studies indicate that sea duck populations only begin to recover from such major spills within a range of 16 to 32 years, for best-case and worst-case predictions, respectively (Iverson & Esler 2010). In the Baltic Sea, hundreds of small-scale spills (less than 1 tonne each) are registered annually, causing chronic injury to seaducks (Larsson & Tydén 2005). Recent suggestions of cargo transport via the Arctic Ocean (e.g. oil transport between Europe and China along the Siberian coast) gives major future concerns for new oil spills in highly sensitive areas. Following declines of common eiders in North America, concentrations of persistent organic pollutants were studied in four duck species in Alaska and Siberia 1991–1995, which showed that quantities of cadmium, copper, lead, and selenium were higher in common eiders than other waterbirds (Stout *et al*. 2002). Monitored concentrations have increased rapidly, for example hepatic cadmium and zinc concentrations tripled in common eiders in the eastern Canadian Arctic between 1992 and 2008 for unknown reasons (Mallory *et al*. 2014). Even though effects of persistent organic pollutants on the health of the birds are generally poorly known, there are concerns about their build-up in tissues of sea ducks, particularly with regards to body condition and breeding success. Studies of the effects of local point pollution discharge have shown dramatic effects on breeding common eider populations, as well as recoveries in periods following cessation of discharges (e.g. polycyclic aromatic hydrocarbons in Norway; Bustnes 2013). Levels of mercury and cadmium become increasingly concentrated in the kidney and liver during periods of fasting (Wayland *et al*. 2002). This suggests that such chemicals are deposited in fat tissue prior to breeding and as females deplete fat stores during incubation, they increasingly leak into the bloodstream. Pollutants have been implicated in affecting seminiferous tubule diameter in male ducks, and causing damage to chromosomal structure (Matson *et al*. 2004, Pollock & Machin 2008) but much remains unknown and relationships seem dependent on individual traits of affected birds.

Increasing human disturbance — wind farms

In response to anthropogenic climate change caused by carbon emissions, many states throughout the world have set targets to meet increasing energy demands from renewable sources, such as wind turbines on land and at sea,

particularly in Asia, United States and Europe. However, wind farms may adversely impact bird populations as a result of habitat loss, barrier effects and collisions between birds and turbines (Fox *et al*. 2006). Habitat loss occurs when birds show avoidance to wind farms and their vicinity. Hötker *et al*. (2006) showed that six out of eleven studies (55%) conducted during the breeding season reported unchanged or increased duck densities near wind farms, whereas five (45%) reported lower densities. Outside the breeding season, 8% of studies reported similar or increased densities near wind farms, whereas in 92% (n = 22) densities were lower. On the basis of a meta-analysis of 19 surveys, Stewart *et al*. (2007) showed that duck densities were more adversely affected than other groups of species. Hötker (2006) also concluded that ducks showed amongst the highest avoidance distances outside the breeding season. "Habituation" to wind turbines, which may counteract a potential habitat loss over time, may occur for some species. Common scoters initially absent from within a Danish offshore wind farm post construction, were subsequently observed feeding between turbines, whereas at another site, long-tailed ducks occurred at lower densities between turbines than prior to construction, whilst densities elsewhere were stable or greater than prior to construction (Petersen *et al*. 2006). Wind farms may act as a barrier to seasonal migration or regular daily movements if birds change their flight direction when approaching them, which elevates energy expenditure but reduces the number of flying birds near turbines. Little is known about the extent to which wind farms may act as barriers to the movements of ducks, but common scoters and common eiders avoided offshore wind turbines (Christensen *et al*. 2004, Desholm & Kahlert 2005) and the relatively few migrating eiders entering into the wind farm flew between turbine rows at low altitude (Desholm & Kahlert 2005). Drewitt & Langston (2006) concluded that none of the barrier effects identified so far have significant impacts on populations. Collision rates between birds and wind turbines vary considerably between different wind farms. In Europe, wind farms near wetlands present a particularly high collision risk (Hötker *et al*. 2006). Despite ducks being active by night at dusk and dawn, collisions rates between ducks and wind turbines are low as compared with those of other species groups (Dürr 2013), consistent with their high avoidance rates at offshore wind farms (Cook *et al*. 2012).

Increasing human disturbance — recreation

Kayaking, kite-surfing and boating can adversely affect ducks depending on the trade-off between available fitness-related resources, e.g. food intake versus the perceived predation risk posed by human presence (Frid & Dill 2002, Bregnballe *et al*. 2009a, Laursen & Holm 2011). Responses range from minor changes in behaviour, such as increased vigilance that slightly reduces teal foraging efficiency following pedestrian disturbance to a wetland (Bregnballe *et al*. 2009b), to major impacts on fitness. Examples of more severe impacts include leisure boats increasing duckling mortality in common eider broods because they increase gull predation success (Åhlund & Götmark 1989) or forcing moulting common eiders to permanently abandon preferred feeding sites in the Wadden Sea (Laursen & Holm 2011). Ironically, with respect to the predicted future increase in predation pressure by white-tailed sea eagles on common eider in breeding areas, human presence and disturbance may in fact even have positive effects in reducing predation impacts. This is generally known as the scarecrow effect (e.g. Leighton *et al*. 2010) and is thought to already be manifest along the Finnish coast, where eider breeding numbers have recently dramatically increased at Bengtskär, a very popular tourist attraction outside Hanko, SW Finland, in stark contrast to the declining breeding numbers in adjacent areas (M. Öst unpubl. data). Baltic shipping traffic is known to cause permanent displacement of non-breeding common scoters and other sea ducks (Schwemmer *et al*. 2011). Disturbance can reduce habitat carrying capacity and locally reduce abundance. Ultimately, human presence may affect duck mortality. In Canada, more than 40% of eggs and hatchlings in a common eider colony were lost after human intrusion (Bolduc & Guillemette 2003). Based on bird flight dis-

tance measurements, recreational activities like walking or bait digging have limited disturbance effects compared to kite surfing, windsurfing and dogs running off the lead (Laursen *et al*. 2005, Liley *et al*. 2011). However, ducks can also habituate, adapt to different conditions and show ecological plasticity to cope with disturbances, for instance by increasing night-time foraging if disturbed by day (Bregnballe *et al*. 2009b) or by engaging in kleptoparasitism to compensate for lost foraging area due to human disturbance (Holm & Clausen 2009). Regular or predictable disturbance can result in habituation, where birds ultimately ignore recreational activity (Laursen & Holm 2011, Bouskila & Blumstein 1992, Berl 2013). Provision of sensitive routing and observation facilities for people (Guillemain *et al*. 2007) and establishment of disturbance-free reserves (Madsen *et al*. 1998) are effective in reducing the effects of human activities from the most important duck feeding, roosting and breeding sites in wetlands and at sea. Effects of disturbance like habitat loss, loss of feeding time and increased energy expenditure are easy to measure on a local scale, but impacts on population level are extremely difficult to assess, especially in the long term (Møller 2008). Nevertheless, it is important to identify bottlenecks in the annual life cycle (e.g. early in the brood-rearing period when ducklings are small and during the flightless moulting period) where ducks are particularly sensitive, and to continuously monitor disturbance effects as human recreational activities in wetlands are increasing.

Increased timber extraction in the boreal forest with increased growth

The doubling of atmospheric CO_2 is predicted to support a shift of northern forest northwards by 500 km, potentially covering 50% of existing tundra (Foley *et al*. 1998, Cramer *et al*. 2001), creating new opportunities for ducks of the boreal forest, but restricting habitat availability for obligate tundra nesters. Higher atmospheric CO_2 levels are likely to increase plant growth rates (Kirilenko & Sedjo 2007), especially in conjunction with expected increases in temperature (more so in winter than summer) and precipitation (up to 40% more in winter) in boreal forests, resulting in more rapid exploitation cycles and generally more access and disturbance in forests throughout the year. Lack of nesting holes for goldeneyes, goosander *Mergus merganser* and smew *Mergellus albellus* has been compensated for by the provision of nest-boxes by ornithologists and hunting organisations, although this is not a viable option in areas of sparse human settlement.

Designing protected area networks to accommodate range shifts

Northwards species range shifts due to climate change have been demonstrated in several studies (e.g. Parmesan 2006, Brommer *et al*. 2012, Lehikoinen *et al*. 2013, although range changes are not always consistent Brommer and Møller 2010) as species readjust distributions to match their habitat and energetic requirements to new climatic circumstances (Virkkala *et al*. 2013). Protected area networks protect species from human-induced habitat alterations (e.g. logging, drainage; Virkkala & Rajasärkkä 2011) but were designated according to the species distributions that prevailed at the time of designation. Accelerating climate change affects the rate at which species are changing their distributions, potentially pushing species of conservation importance out of protected areas. Most climate change scenarios forecast more rapid temperature increases at northern latitudes (IPCC 2007) and hence boreal species may show more rapid range changes than in biomes further south (Virkkala *et al*. 2013).

Assessments of the effectiveness of protected area networks are rare, but they generally suggest a clear benefit of SPAs network to bird populations (especially those species listed in Annex I of the Birds Directive) at a European scale (Donald *et al*. 2007), although this effect depends on habitat and latitude (Virkkala & Rajasärkkä 2011, Virkkala *et al*. 2013). Birds exploiting mires, mountains and marshlands are better protected by reserves against the effects of climate changes than those in forests (Virkkala *et al*. 2013). Thomas *et al*. (2012) found that protected area networks losing bird species due to climate

warming also served as new suitable colonising areas for species expanding northwards. Species benefit from the quality of the protected habitat and management to enhance or at least maintain habitat quality to support the establishment of new populations (Mawdsley *et al*. 2009, Thomas *et al*. 2012, Virkkala & Rajasärkkä 2012).

Waterbirds are particularly sensitive to changes in weather and strongly respond to, for instance, harsh winters by migrating further south along the flyway (Ridgill & Fox 1990). Many waterbird species have been short-stopping (reducing their migration distance, Elmberg *et al*. 2014) in the past decade (Dalby 2013, Lehikoinen *et al*. 2013, Pavón-Jordan *et al*. 2015), whilst others do not (Dalby *et al*. 2013b). Waterbirds are target species in the Natura 2000 network due to their ecological and socio-economic value (*see* Dalby *et al*. 2013b and Green & Elmberg 2014 and references therein) and therefore, such redistribution may affect the ability of protected area networks to fulfil their management objectives. Recent analysis of the wintering numbers of smew throughout the European flyway has revealed a shift northwards in the wintering range, as numbers decline at the southern and central parts of the wintering distribution and abundance correspondingly increased at the northeastern parts of the winter range (Pavón-Jordan *et al*. 2015). Importantly, at the newly colonised northeast of the winter range, numbers have increased inside and outside SPAs, but the numbers increased more inside SPAs (Pavón-Jordan *et al*. 2015). This confirms the importance of protected areas in supporting, assisting and protecting species in their range shifts under current climate change, but also the need for dynamic revision of protected site networks to keep up with changing needs (*see* also Thomas *et al*. 2012).

Changes in species interactions

As climate and environmental change affect the distribution and abundance of a range of taxa, we are expected to witness novel and changing species interactions as organisms that have never been sympatric increasingly overlap. Some such changes are already becoming manifest. For instance, duck nest densities are generally higher in or close to gull colonies (Koskimies 1957, Hilden 1965, Dwernychug & Boag 1972) where nest predation may also be lower than elsewhere (Bengtson 1972, Newton & Campbell 1975, Young & Titman 1986, Väänänen 2000, Blums *et al*. 2003). There are indications that black-headed gulls *Chroicocephalus ridibundus* and (in some areas) common gulls *Larus canus* are showing widespread declines throughout Europe, especially in areas where breeding duck populations are greatest (e.g. Denmark, Heldbjerg 2001; Sweden, Källander 1996; Finland, Asbirk *et al*. 1997; Latvia, Vîksne *et al*. 1996). Whatever their cause, these declines could contribute to local reductions in both breeding density and reproductive output amongst breeding duck populations.

Discussion

It is impossible in a review of this kind to cover all issues relating to current and future drivers affecting population change in Nordic breeding duck species and we have been less wide ranging in our assessment than for instance Sutherland *et al*. (2012) in an attempt to restrict our coverage to more likely major factors on the immediate predictive horizon. Nevertheless, we consider this exercise to be highly valuable for considering available expert opinion relating to realistic scenarios likely to impact on population demography in a way that affects the distribution and abundance of common duck species in the immediate future (i.e. the next 25–50 years). We consider this a vital exercise to stimulate debate, discussion and further thought and research relating to these and the next level of issues likely to be confronting duck species. We believe such reviews are essential for formulation of policy and policy tools as well as stimulating specific research to start to generate potential solutions to problems facing the maintenance of duck species in favourable conservation status now and in the future.

Despite the many and varied potential future changes to the ecosystems exploited by ducks, these migratory species remain wholly dependent upon a string of different sites that provide

(i) sources of water, energy and nutrition and (ii) safety from predation and disturbance which are strung out along their current and future flyways. The availability of such site networks supported by adequate habitat provision and management remains absolutely crucial to the survival and well-being of these populations and there is no doubt that while climate change may change their relative arrangement, the continued existence and relative availability of key habitats to support these populations remains the most urgent need to maintain now and in the future to support current levels of population and elevate those where necessary. Hence, no matter what happens in the future, the cornerstone of effective population management has to be the effective provision of coherent site safeguard networks that especially protect key sites that may be particularly important for large proportions of some populations even for relatively short periods in every stage of the annual cycle. It is, however, our task to second guess how pressures will affect these relationships and find the operational tools to perpetuate such populations.

One of the major unresolved questions is whether Nordic duck populations will be able to adapt fast enough to the changes that are occurring in their immediate environment. We know that within two decades, some wader species have shifted their entire migration routes in responses to changing circumstances within their flyway (Rakhimberdiev *et al*. 2011), while some diving duck species have dramatically shifted their winter quarters (Lehikoinen *et al*. 2013, Pávon-Jordan *et al*. 2015). This suggests that at least these species possess considerable phenotypic flexibility to adjust to such change in relatively short time spans (i.e. within generations). However, it is not clear how the effects of shifting predation and nutrient regimes and other pressures will affect other duck species. The adaptive responses to these and other pressures on breeding, staging and overwintering ducks may push the balance of phenotypic responses in different directions; however, the harsh reality is that we shall simply have to wait and see how these processes become manifest, unless we can vastly improve our capacity to model and forecast future population development into areas without prior experience. Nevertheless, this is a good reason to explore some of the more exciting prospects offered by implementing adaptive management frameworks, where the experimental demonstration of the relative effectiveness of many competing models offers prospects of progress where agreement on practical management may be difficult to obtain.

It is therefore essential to maintain the modest monitoring mechanisms presently in place to continue to monitor Nordic duck population size, trends and distribution at as many levels as possible to understand the basic patterns of change in abundance and distribution. These are vital if we are to be able to generate rudimentary hypotheses upon which to develop policy recommendations for improved management. In most cases, the breeding, staging and wintering numbers in the Nordic countries are well known. For some of these populations, the demographic drivers of trends in these populations are also monitored, for instance, in the rare instances where we have regular ringing data to generate annual survival estimates or breeding surveys to monitor reproductive success. However, this is far from always being the case, especially for those species that disperse to wintering areas that may not be so regularly covered by existing count networks and where collation of flyway level statistics still remains to be carried out. We urgently need to greatly improve the status of the current flyway monitoring and the extent and scope of site safeguard networks to be able to cope with the observed changes in distribution and abundance as new patterns emerge. This requires gathering of all interested parties at local, regional, national and supra-national levels to facilitate actions at the flyway levels.

Most important is that the process does not stop here. We need to enshrine the concept of "adaptive foresight" (Sutherland *et al*. 2012) that challenges all the appropriate stakeholders, agencies and policymakers to contribute to the construction of more robust and resilient policies in the face of change and a lack of knowledge. In this case, our task is made easier by the taxonomic focus (ducks) and the geographical scope (Nordic States) and we hope that this document forms some basis for further development, dependent on the nature of the stakeholders and their representative views and interests.

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