Trends in staging waders on the Tipperne Reserve, western Denmark, 1929-2014

with a critical review of trends in flyway populations

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Cover page: Several thousand Bar-tailed Godwits used to stage and refuel on
Tipperne Reserve in May, but numbers have declined particularly since the 1980s.
Water colour by Jens Gregersen, Bar-tailed Godwits and Dunlins on Tipperne in May.
Title page: Since the 1950s, up to about 1000 post-breeding Pied Avocets have
gathered on Tipperne Reserve during July-September. Drawing by Jens Gregersen.

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Trends in staging waders on the Tipperne Reserve, western Denmark, 1929-2014 with a critical review of trends in flyway populations

Hans Meltofte & Preben Clausen

(Med et dansk resumé: Udviklingen i forekomsterne af rastende vadefugle på Tipperne 1929-2014 med en kritisk gennemgang af udviklingen i flyway-bestandene)

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Abstract

Numbers of staging as well as breeding waterbirds on Tipperne Reserve in west Jutland have been monitored since 1928. This has resulted in one of the world's longest time series for analyses of bird abundance and trends in relation to changes in the environmental conditions and the total flyway-populations. The occurrence of staging waders (shorebirds in American English) up to 1982 has been reviewed previously. In this paper we update these results up to 2014 with focus on numerical changes in the Reserve and in the flyway populations in general.

During the first decades, the counts of staging birds in the Reserve were not very systematic. The observers recorded the numbers of waterbirds daily or at intervals of a few days. From 1973, however, censuses were organized by well-defined routines, which from 1978 focused on a census of birds in the entire Reserve once in every running five-day period year round. Since that time activities have been cut back several times: in 1998 counts in the winter months from December to February ended; from 2002 censuses were reduced to three per month in spring and two per month of autumn; and from 2012 censuses were further reduced to two per month even in spring.

Data are presented in three ways. Changes during the entire study period 1929-2014 are shown as running nine-year medians of maximum numbers recorded within selected periods during spring and autumn migration. For the more systematically covered period 1973-2014, we have calculated bird-days for the same periods during spring and autumn migration and analysed trends in these using polynomial regression. Finally, we present phenological graphs where we compare the occurrences during the first 15 years of the systematically covered period with the last 15 years and the entire 1973-2014 period.

The conditions for the birds in the Reserve are largely determined by regulation of the water level in the fjord by a sluice at Hvide Sande and by discharge of sediment and nutrients from the surrounding area, i.e. predominantly from the river Skjern Å. Conditions are also influenced by management of the lands of the Reserve by grazing and mowing meadows and reed beds.

The most notable changes in the occurrence of waders in the Reserve are associated with the draining of the Skjern Å delta in the middle of the 1960s, which considerably increased sediment discharge. At the same time, discharge of nutrients also increased sharply due to increased use of fertilizers in agriculture in the catchment area of the rivers. This led to highly increased densities of benthic invertebrates on the shallow flats and strongly increasing numbers of waders that prefer muddy sediments such as Common Snipe *Gallinago gallinago*, Spotted Redshank *Tringa erythropus*, Common Greenshank *Tringa nebularia* and Common Redshank *Tringa totanus*. This is in marked contrast to the 'collapse' of the occurrence of herbivorous waterfowl that happened in the fjord around 1979 as a result of plant death caused by algal blooms.

The main exception to generally positive long-term trends among waders on Tipperne was declining numbers of Bar-tailed Godwits *Limosa lapponica* up to around 1990. The reason for this may be a combination of higher water levels, reed and other plant overgrowth of the highest-lying flats, and depletion of ragworm stocks due to arrival of larger numbers of godwits and other waders early in the season.

A decline in mowing and grazing of the Tipperne meadows from the late 1950s to the 1970s led to the meadows becoming overgrown resulting in reduced numbers of staging Eurasian Golden

Plovers *Pluvialis apricaria* and Northern Lapwings *Vanellus vanellus*. With resumption of management of the meadows, numbers of these species recovered. Likewise, Common Snipes benefited from extensive mowing of reed beds etc. in the 1970s and 1980s.

The picture becomes less positive for a number of species when we consider the short-term changes (15 years). This applies primarily to a number of breeding species which peaked with extremely large breeding populations during the early 1990s, but whose numbers have since fallen to much lower levels primarily due to increased predation. Furthermore, for a number of species, the greatly reduced census frequency since 2002 has reduced the chances to 'catch' the often narrow peaks in staging numbers, which are typical for wader migration. Decreasing benthos densities since the 1990s as a result of reduced eutrophication may also have contributed to this decline.

There are few changes on Tipperne that can be related to the trends in the total flyway populations. The main reason is undoubtedly that bird numbers in a relatively small area like Tipperne are determined more by local conditions than by changes in the overall population sizes, which also often are very uncertain. We have therefore undertaken a critical review of the entries in the latest authoritative overviews from Wetlands International. Unfortunately, for the waders the results are often a plethora of conflicting statements based on partly different time periods.

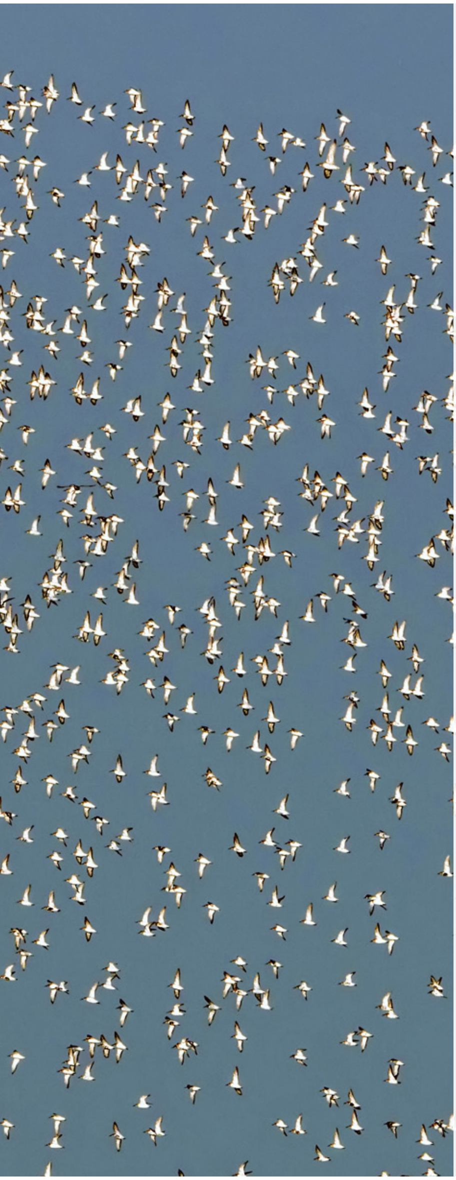
Our critical review suggests that while a number of the breeding 'meadow birds' in temperate western Europe have declined quite considerably (especially Black-tailed Godwit *Limosa limosa* and Ruff *Calidris pugnax*) due to agricultural intensification, most Arctic and boreal wader populations are thriving. Hence, among the 29 distinct wader populations considered here (predominantly on the East Atlantic flyway), 18-19 populations are either stable, possibly stable, fluctuating or increasing, while only four can definitely be said to be decreasing, and a further 6-7 possibly are decreasing.

An indication of improved conditions for waders is that a number of species now arrive several weeks earlier in spring than they did 80 years ago and a few also begin egg laying earlier. This is probably a result of milder winters and springs and thus earlier availability of benthic invertebrates on shallow flats, which makes it possible for the birds to move to an area like Denmark much earlier than in the past.

The few species whose numerical changes on Tipperne may at least partly be related to changes at the flyway level are Pied Avocet *Recurvirostra avosetta*, Grey Plover *Pluvialis squatarola*, Eurasian Curlew *Numenius arquata* and Curlew Sandpiper *Calidris ferruginea* in particular. The populations of the first three species were probably greatly reduced during the 19th and 20th centuries due to hunting and trapping either on the breeding grounds or along the flyway – or both – but have since recovered as a result of improved protection.

In summary, our critical analysis of the wader population trends in focus here suggests that the often stated widespread population declines are hardly as general as postulated. This includes that we question the validity of the listing of Eurasian Curlew and maybe even Northern Lapwing as Near Threatened on IUCN's Red List of Threatened Species. Our recommendation is that we focus conservation efforts on the species and populations which have well-documented problems such as European farmland birds and especially waterbirds in East Asia, many of which have serious problems due to wetland reclamation, hunting and trapping.

Photo: Torben Andersen, Dunlins.




Contents

Introduction

Numbers of breeding as well as staging waterbirds and several other bird species have been monitored on Tipperne Reserve in the shallow brackish lagoon of Ringkøbing Fjord (Fig. 1), west Jutland, ever since 1928. This monitoring has resulted in a number of major publications on the numbers, phenology, trends and other characteristics of the occurrence of specific breeding and staging bird species in relation to environmental and management changes within and outside of the Reserve (Madsen 1985, Meltofte 1987, 2010, Thorup 1998, Meltofte & Clausen 2011, Petersen *et al.* 2012, Meltofte & Amstrup 2013).

Changes in water level, salinity, eutrophication, sedimentation and vegetation height have had major impacts on the numbers of waders, swans and ducks staging on the meadows and shallow flats (Meltofte 1987, Meltofte & Clausen 2011). These environmental factors have had either direct or indirect impacts through effects on feeding conditions in the form of changes primarily in the occurrence of benthic invertebrates and submerged vegetation.

Numbers, geographical origin, phenology and population trends of 35 species of staging waders on the Reserve during 1928-82 were analysed by Meltofte (1987)

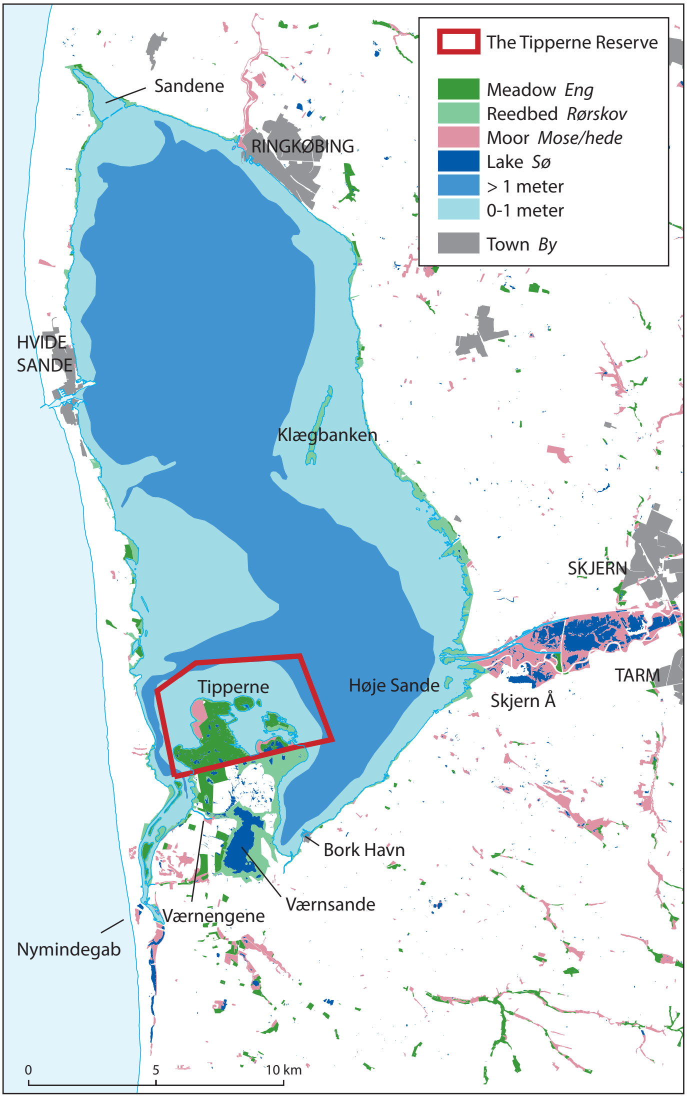
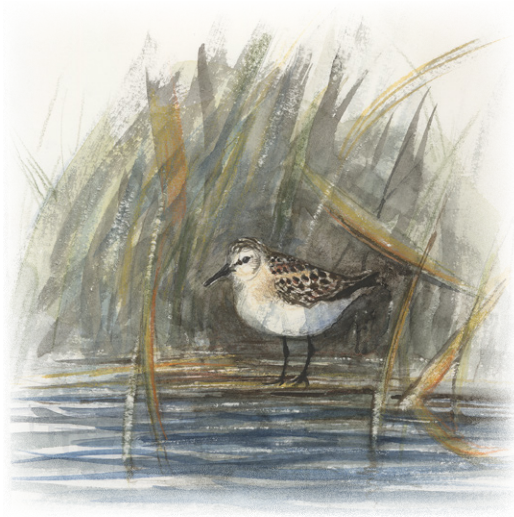


Fig. 1. Map of Ringkøbing Fjord with the Tipperne Reserve and habitats demarcated. Besides Tipperne, flats sufficiently shallow for waders are found only at Høje Sande, Sandene and around the northern end of Klægbanken, but all are much smaller than the available wader habitats on Tipperne. Habitat areas on land have been extracted from the national mapping of protected habitats (§3) from 1995 to 2010 (Danmarks Miljøportal). Depth distribution in the fjord is based on a survey conducted by the former Ringkøbing County.

Kort over Ringkøbing Fjord med Tipperreservatet. Udover Tipperne findes tilstrækkeligt lavvandede områder for vadefugle kun på Høje Sande, Sandene og omkring den nordlige ende af Klægbanken, men alle disse er meget mindre end de tilgængelige habitater for vadefugle på Tipperne. Habitaterne på land er uddraget fra den nationale kortlægning af beskyttede naturtyper (§3) fra 1995-2010 (Danmarks Miljøportal). Dybdefordelingen i fjorden er baseret på en kortlægning foretaget af det tidligere Ringkøbing Amt.

and supplemented by Meltofte (1993) with phenological data up to 1990. In this paper we continue the trend analyses until 2014 for the 25 most regular and numerous wader species in relation to local environmental conditions and overall population trends at the flyway level.

The aim of these concerted efforts on the entire range of species analysed was to identify likely causal relationships between environmental conditions and bird numbers in this 'intensively' monitored area. Such relationships should not only be used in the management of the Reserve, but could also be used in evaluations of similar changes documented in the many less intensively monitored areas in other parts of Denmark and abroad. Furthermore, the results of this extraordinary long time series may contribute to the evaluation of population trends at the flyway level, where Delany *et al.* (2009), Nagy *et al.* (2014, 2015) and van Roomen *et al.* (2015) have contributed with the most recent authoritative accounts.



Juvenile Little Stint. Water colour by Jens Gregersen.

Habitats and food resources

Tipperne Reserve is part of EEC Bird Protection Area no. 43 and Ramsar site no. 2, and the entire area is protected under the Danish nature protection legislation. The land area of Tipperne Reserve has been protected from shooting and general public access ever since 1898. But this protection was not effective until 1928, when the Reserve largely acquired its present size, i.e. with the inclusion of the extensive shallow flats (Fig. 1).

Detailed descriptions of habitats on the Reserve and the changes that have taken place over the last 80 years are given by Meltofte (1987), Thorup (1998) and Meltofte & Clausen (2011). Here, we present a brief review only of the main features based primarily on the latter account.

Tipperne Reserve constitutes the northern quarter of a low-lying peninsula with surrounding waters in the 285 km^2 coastal lagoon of Ringkøbing Fjord in west Jutland (Fig. 1). The Reserve covers 7 km^2 of brackish meadows and marshland surrounded by 18 km^2 of shallow water. The water level and salinity in the lagoon have been regulated by a sluice at Hvide Sande since 1931, which separated the lagoon from the North Sea. Prior to 1931, there was an open outlet near Nymindegab southwest of Tipperne, through which natural semidiurnal tides and storm surges could influence water levels in the lagoon. Management of the sluice at Hvide Sande has varied considerably over the years, as has the discharge of sediments and nutrients from the main river, Skjern Å, flowing into the lagoon.

Water levels in the lagoon have been monitored on a regular basis from around 1900. Here we present data from 1928 onwards, which is a combined dataset based on monitoring carried out by the national hydrology authorities at harbours in the lagoon, and by staff at Tipperne Reserve. Creating such a long time-series requires inter-calibration between different monitoring stations, as well as correction of apparent errors (e.g. after ice damage of meter poles or malfunction of automated meters) before merging datasets. Meltofte (1987) gives a detailed account of how he approached this challenge for the period 1928-1982, and Meltofte & Clausen (2011) describes our approach for the years 1973-2009. The methods from the latter have also been used for 2010-2014.

Prior to the establishment of the sluice, average water levels in Ringkøbing Fjord in 1928-1930 were much higher than they are today during winter, early spring, late summer and autumn, but were comparable to present day levels in late spring and early summer (Fig. 2). During some periods of these early study years, this might have meant that the mudflats around Tipperne were hardly available to staging waders, especially during the juvenile migration period. One should, however, recall that the Figure depicts average water levels but not tidal variations. Before the sluice, Ringkøbing Fjord was subject to tides with amplitudes of 0.6-0.8 meters in this part of the North Sea. These tides gave waders some daily access to feeding, at least during low tide, on

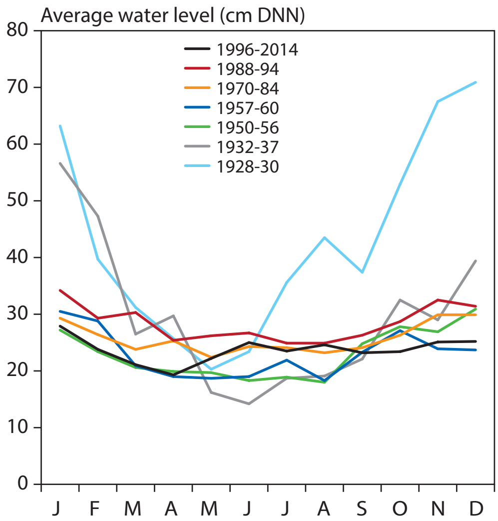


Fig. 2. Annual variation (monthly means) in water level at Bork Havn (4 km south of the Reserve, see Fig. 1) at the southern end of Ringkøbing Fjord prior to establishment of the sluice at Hvide Sande in 1931, and during six periods with different sluice management schemes after 1931 (details given in Tab. 1 in Meltofte & Clausen 2011). Years with changes in sluice management and years without simultaneous measurement of salinity (1938-49, 1961-69, and 1985-87) have been omitted.

Årsvariationen i middelvandstanden målt pr. måned ved Bork Havn (Kirkehøj) i perioden før slusen blev etableret samt i seks perioder med forskellig slusepraksis. For detaljer om ændret slusepraksis henvises til Tab. 1 i Meltofte & Clausen (2011). Kalenderår, hvor slusepraksis er ændret, er udeladt fra beregningerne, og perioderne 1938-49, 1961-69 samt 1985-87 er udeladt, fordi der ikke er samtidige målinger af salinitet. Vandstanden er angivet i forhold til Dansk Normal Nul (DNN), der kan omregnes til den nye kote Dansk Vertikal Reference 1990 (DVR90) med formlen DVR90 = DNN-11 cm (se Appendiks 1 i Meltofte & Clausen 2011).

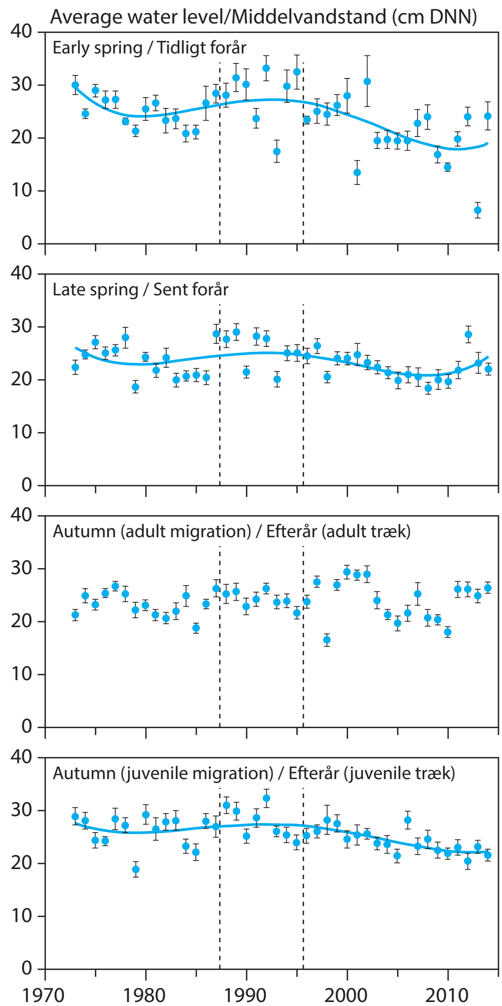


Fig. 3. Average water level (± 95% confidence intervals) at Tipperne 1973-2014 for four different seasons, i.e. early spring (1 March-20 April), late spring (21 April-19 June), and two seasons corresponding to the main autumn migration periods of adults (20 June-13 August), and juvenile birds (14 August-30 November), respectively. Water levels were monitored at Tipperne or estimated from measurements at Bork Havn for days where no measurements were available for Tipperne. Hatched lines indicate when sluice operations changed. For three of the seasons, significant 4th order polynomial curves are shown. The 4th order polynomial for early autumn did not differ significantly from a straight line.

Middelvandstanden (± 95 % konfidensintervaller) på Tipperne 1973-2014 fordelt på fire sæsoner, henholdsvis tidligt forår (1. marts-20. april) og sent forår (21. april-19. juni), samt to sæsoner, der repræsenter efterårstrækperioderne for henholdsvis adulte (20. juni-13. august) og juvenile fugle (14. august-30. november). Vandstanden er målt på Tipperne eller estimeret ud fra målinger ved Bork Havn på dage, hvor der ikke er foretaget måling på Tipperne. For tre af sæsonerne er signifikante 4.-gradspolynomier plottet. De stiplede linjer angiver tidspunkt for ændring af slusepraksis. Vandstanden er angivet i forhold til Dansk Normal Nul (DNN).

the mudflats under most situations except during storm surges that caused particularly high water levels.

From 1932 to 1937, the sluice was used to maintain a high winter water level (Fig. 2) in order to let the estuarine waters fertilize the surrounding meadows by flooding. This practice was abandoned in 1938, and since then water levels have been heavily regulated and generally kept at an average of between 15 and 40 cm (Fig. 2).

When we look in more detail at the years 1973-2014 in particular, it is apparent that even within this narrower band of water levels, sluice management has not kept water levels constant. During this period, there were three major management protocols for the sluice (listed in Tab. 1 of Meltofte & Clausen 2011), primarily aimed at regulating salinity, but with apparent impacts on water levels also. Average water levels in the early spring period were the most variable over the years, with averages generally above 20 cm or even 30 cm prior to 2000, but near or below 20 cm after 2002 (Fig. 3). We are not aware of a change in sluice regulations from 2003 that can explain this apparent and fairly abrupt change. For late spring, there was also a tendency for slightly lower water levels in most of the last study years, whereas water levels were stable with no major changes during the midsummer (adult migration) period 1973-2014 (Fig. 3). For the late summer-autumn (juvenile migration) period, average water levels gradually fell from near 30 cm in the 1970's to near 20 cm in the 2010's (Fig. 3).

A change in water level of just 10 cm might seem minute, but has an immense impact on accessible mudflats on Tipperne. We used depth data compiled by Jensen (1986) along eight transects used for vegetation mapping on the Reserve's mudflats and shallow waters, corrected these depths to 0 (Ordnance Datum = 0 cm), and subsequently modelled in ArcView to see whether the shallows would be accessible to all waders (water depth on the shallow less than 5 cm), medium-sized and large waders (water depth between 5 and 10 cm), large waders only (water depth between 10 and 15 cm), or inaccessible to waders (water depth more than 15 cm) at water levels ranging from 0 to 30 cm. The three size classes of waders generally correspond to Dunlin *Calidris alpina* and most other *Calidris* sandpipers (small waders), Common Redshank *Tringa totanus*, other *Tringa* species and Ruff *Calidris pugnax* (medium-sized waders), and godwits *Limosa* spp., curlews *Numenius* spp. and Pied Avocet *Recurvirostra avosetta* (large waders) (Thorup 1998).

The modelling exercise illustrates that at or below 0 cm, well over half of the shallows on the entire Reserve will be available even to the small waders (Fig. 4). At 10 cm, the entire Tippesande area (c. 2 km²), about 0.6 km² east of Opgrøden, and a smaller part north of the peninsula will be available to the small waders. But once levels reach 20 cm, the accessible area for the smallest waders shrinks to about 0.35 km² found between Opgrøden and Fuglepold. At 30 cm, no shallows are accessible to small waders, only small parts are available to medium-sized waders, and about 2 km² are available to large waders (Fig. 4).

Another important change to the environment of the lagoon was the drainage of the large delta of Skjern Å in the mid-1960s. This change together with increased

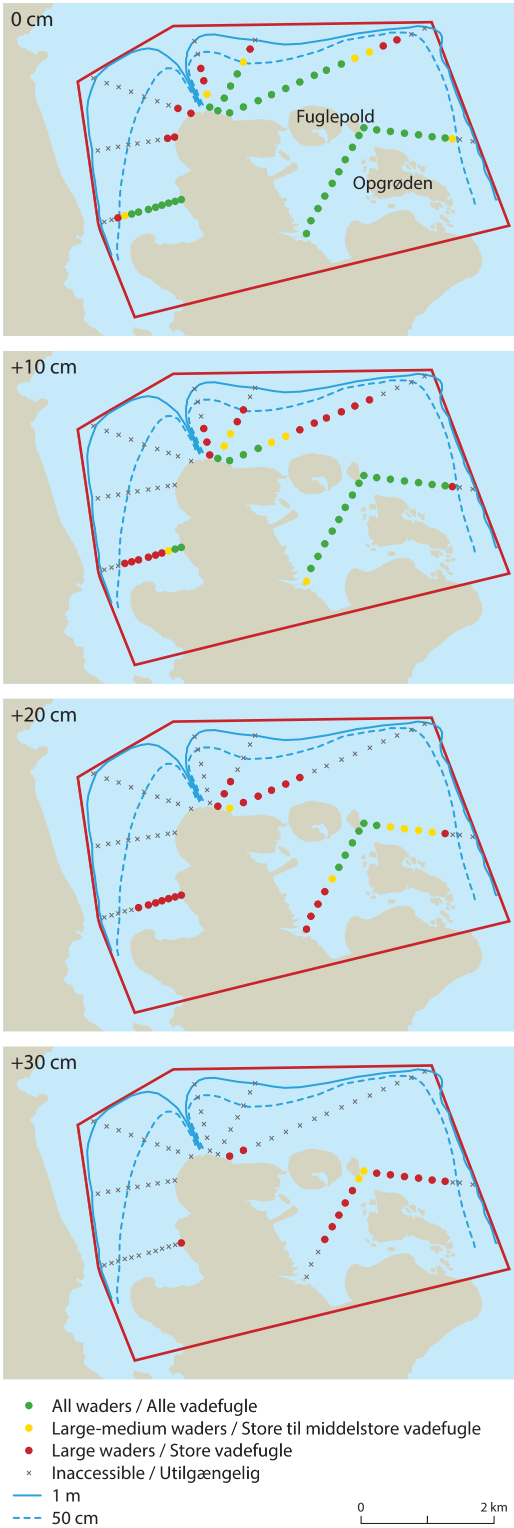


Fig. 4. Map of Tipperne Reserve showing accessibility of the shallow flats to three size classes of waders at four water levels. Two bathymetric contours are also shown.

Kort over Tipperreservatet med tilgængeligheden af vadefladerne for tre størrelsesklasser af vadefugle under fire vandstande. To dybdekurver er også angivet.

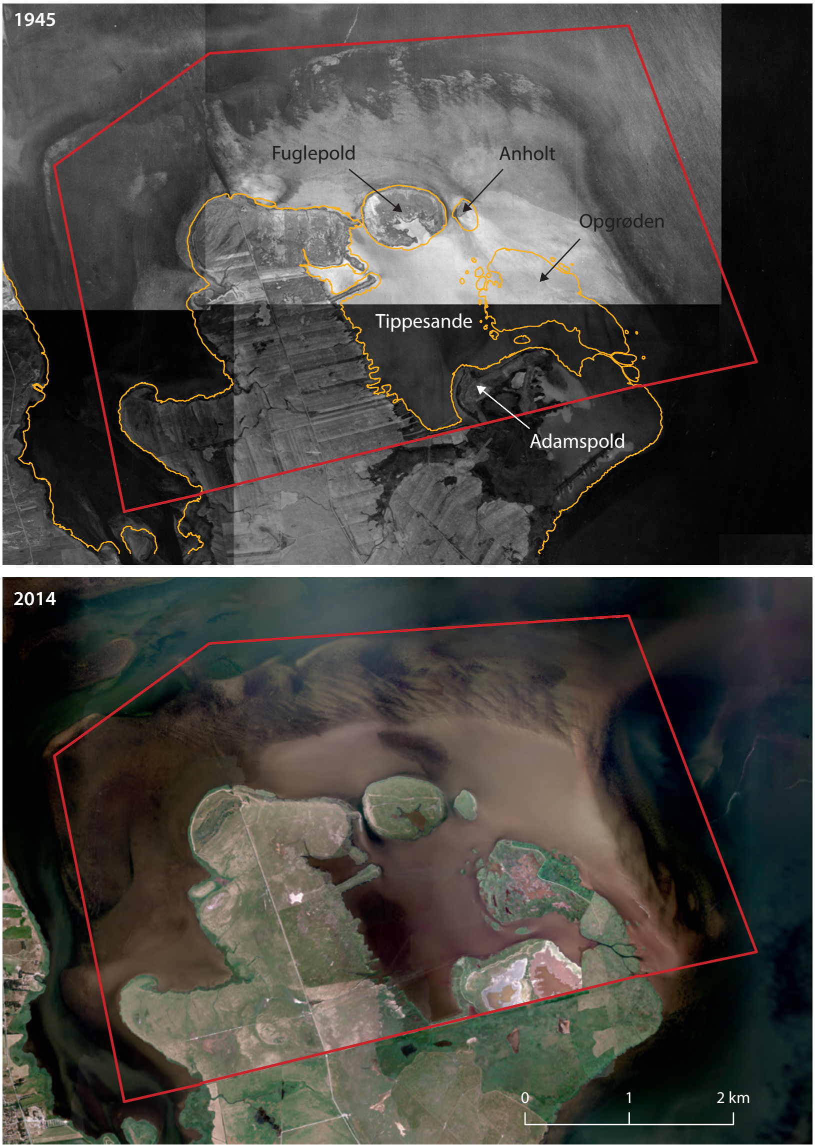


Fig. 5. Two aerial photographs of the Tipperne reserve. The upper photo collage is based on images taken by Royal Air Force in 1945, and the lower is based on the most recent orthophotos from summer 2014. On the upper photo the present coastline is given as a thin orange line, illustrating loss of previous feeding mudflats for waders in the eastern part of the Reserve. Note that the meadows were divided into hay cutting parcels during the first three decades of the study period as seen in the upper photo. Sources: 1945 reproduced with permission from I·GIS; 2014 DDO®land COWI A/S.

To luftfotos af Tipperreservatet. Den øverste fotokollage er baseret på fotos taget af Royal Air Force i 1945 og den nederste på de nyeste orthofotos fra 2014. På det øverste er den nutidige kystlinje anført med en tynd orange linje, der illustrerer tabet af fourageringsområder for vadefugle på mudderfladerne i den østlige del af reservatet. Bemærk at engene var inddelt i høparceller i de første tre årtier af undersøgelsesperioden, som det ses på det øvre foto. Kilder: 1945: gengivet med tilladelse fra I·GIS; 2014 DDO®land COWI A/S.

use of fertilizers in the agriculture-dominated catchment resulted in heavily increased sedimentation and eutrophication of the lagoon (see below). A large part of the delta was re-established as wetlands during 1999-2002, but the effect on the inflow of nutrients to the fjord has been limited (Andersen 2005, Nielsen 2007).

The sedimentation processes affected especially the eastern parts of the Reserve, where subsequent successional development of bulrush *Schoenoplectus maritimus & S. tabernaemontani*, common reed *Phragmites australis* and salt marshes resulted in loss of mudflats. This is most obvious in the area now known as Opgrøden, but areas were also been lost on the east side of Adamspold, along the south and west sides of Tippesande, around Fuglepold and Anholt, the rest of Adamspold and on the north side of the reserve (Fig. 5). The total area of these newly developed marshes is about 1.35 km^2, representing an effective loss of about one third of the mudflats that would have been available to small waders at 10 cm water level prior to the vegetation expansion.

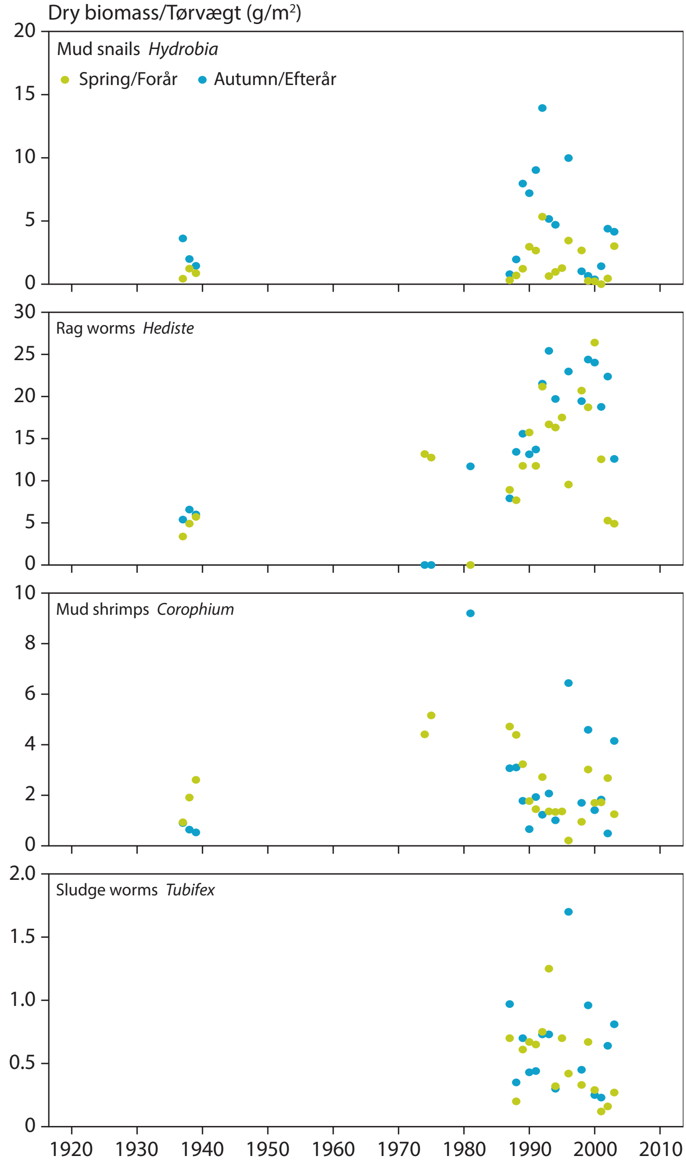


Fig. 6. Mean biomass per m² of mud snails *Hydrobia* spp./*Potamopyrgus antipodarum*, rag worms *Hediste diversicolor*, mud shrimps *Corophium volutator* and sludge worms *Tubifex* spp. and/or *Tubificoides benedii* in spring (March-June) and autumn (September-November) 1937-1939, 1974-1975, 1981 and 1987-2003 on the Tipperne flats. Summer months are excluded because recent data are not available from this period. The oldest samples were given in wet mass which here is transformed to dry mass based on wet/dry mass relations in more recent samples. Note that no sludge worms were found in the 1930s.
Sources: data from 1937-81 given by Meltofte (1987) together with data collected at the field station during 1987-2003.

Biomassen af de fire mest almindelige bunddyr på Tippernes lavvandede grund, henholdsvis dyndsnegle, børsteorme, slikkrebs og sadelbørsteorme fra prøver indsamlet henholdsvis forår og efterår 1937-39, 1974-75, 1981 og 1987-2003. Bemærk, at der ikke var nogen sadelbørsteorme i 1930erne.

Kiørboe (1980) described the vegetation composition of the shallows around Tipperne and in conjunction with this measured grain-size and composition in the upper 10 cm of the sediments. The main feeding areas of mudflat associated waders are all found in his vegetation community I area. This community has a low vegetation biomass (3.4 g/m²), median grain size of 187 µm, and sediment with 0.63% organic matter (i.e. less than on the flats with deeper water), 3.34% silt and clay, and the rest sandy soils.

From 1967 onwards, the salinity of the lagoon was planned to vary between 5‰ and 10‰. This management practice resulted in significantly more fresh and stagnant water than had been the case earlier. Together with the sedimentation and eutrophication from the 1960s onwards, this change severely reduced the water quality of the lagoon. To compensate for this, the desired salinity was raised to 7-12‰ in 1987 and raised again to a summer salinity of 12-15‰ and a winter salinity of at least 8‰ from late 1995 onwards. At the same time, the desired water level was raised according to the sluice management protocol and fixed to be allowed to vary between 10 cm and 35 cm above Danish Ordnance Datum. Despite this, we have seen significantly lower water levels since 2003, as mentioned earlier. This latest management practice has not only increased the salinity but has also resulted in larger water level variations, which are partly the result of large variations in precipitation in the 3477 km^2 catchment of the lagoon.

Waders primarily feed on a variety of invertebrates, but due to the brackish water the benthic fauna on the Reserve is relatively poor in species. Throughout the study period 1929-2014, the benthic fauna of the shallow flats have been dominated by the ragworm *Hediste diversicolor*, mud shrimp *Corophium volutator* and four different species of mud snails e.g. *Potamopyrgus antipodarum* and three *Hydrobia* spp. Furthermore, hypoxia tolerant sludge worms *Tubifex costatus* and *Tubificoides benedii* have been common apparently since the 1980s. Bivalves (common edible European cockle *Cardium edule*, blue mussel *Mytilus edulis*, Baltic tellin *Macoma balthica* and sand gaper *Mya arenaria*) were numerous on the mudflats during 1910-1915, a salt water period with a wide man-made opening to the North Sea, but this community has been absent during most of our study period. The last remaining bivalve, the sand gaper died out between 1939 and 1974, most likely after 1960 as a result of decreasing salinity (Meltofte 1987, Thorup 1998). The increased salinity after 1995 caused a massive increase in sand gaper in other parts of the lagoon (Petersen *et al.* 2008), but at least until 2003 (the last year with samples) the occurrence of sand gaper was insignificant at Tipperne.

As a result of variations in management of the inflow of salt water to the lagoon through the sluice together with heavy eutrophication during the latter half of the study period 1929-2014, the relative abundance among species has varied considerably. However, it is difficult to quantify the development in the benthic food resources for the waders, both because few quantitative data are available and because different sampling methods have been applied over the years (Meltofte & Clausen 2011). Nevertheless, there is no doubt that ragworms, mud shrimps and mud snails have increased in density and biomass on the mudflats around Tipperne from the 1930s to the 1990s in parallel with increased sedimentation and eutrophication. The biomass of mud shrimps increased tenfold from the 1930s to the 1970s, mud snails tripled from the 1930s to the early 1990s, and ragworms increased fourfold up to the late 1990s (see Fig. 13 in Meltofte & Clausen 2011). However, all three species declined after the mentioned peak populations, such that total biomass of invertebrates on the mudflats peaked in the mid-1990s (Fig. 6). Meltofte & Clausen (2011) compared invertebrate sample data from Tipperne with data from other Danish estuarine inlets and the Wadden Sea, and noted that total biomass was lower or comparable to the other sites.

The meadows of Tipperne, which are important feeding and roosting areas for some of the staging wader species and breeding sites for several others, were utilized for hay-cutting and late summer and for autumn grazing by cattle up to the 1950s. Between the 1960s and mid-1970s agricultural use was gradually reduced so that larger parts of the meadows and marshes became overgrown by reeds, and some higher lying areas were even colonized by *Salix*. This development has had significant negative impacts on breeding and staging birds that depend on the meadows. Grazing and hay cutting was therefore gradually re-introduced during the 1970s. Since then, most of the Reserve has been managed with the aim of maintaining a meadow and marsh landscape with good breeding and staging habitats for waders (see Thorup 1998).

Material and methods

Beginning in 1929, birds staging on the Reserve were counted on a daily basis or at intervals of a few days during varying time periods of the year, but generally at least from April/May to August/September (Meltofte 1987). In 1972, the monitoring was reorganised with the introduction of standardized procedures and reporting forms, and plotting of all bird flocks on maps with demarcated sectors of the Reserve was conducted from 1973 onwards (Meltofte & Clausen 2011). Two field workers were stationed at the observatory in the Reserve to provide year round coverage, which from 1978 onwards focused on surveying the entire Reserve at least once during each running five-day period (Fig. 7). This arrangement was maintained until 1997, after which bird counts were conducted only from early March until late November. Furthermore, from 2002, counts were reduced to three times per month in spring and two times per month in autumn (Fig. 7). This procedure was largely maintained up to 2011. In 2012 the programme was further reduced to two counts per month, and in 2012 and 2013 only one count was performed each year in the wader autumn migration peak season in July-August

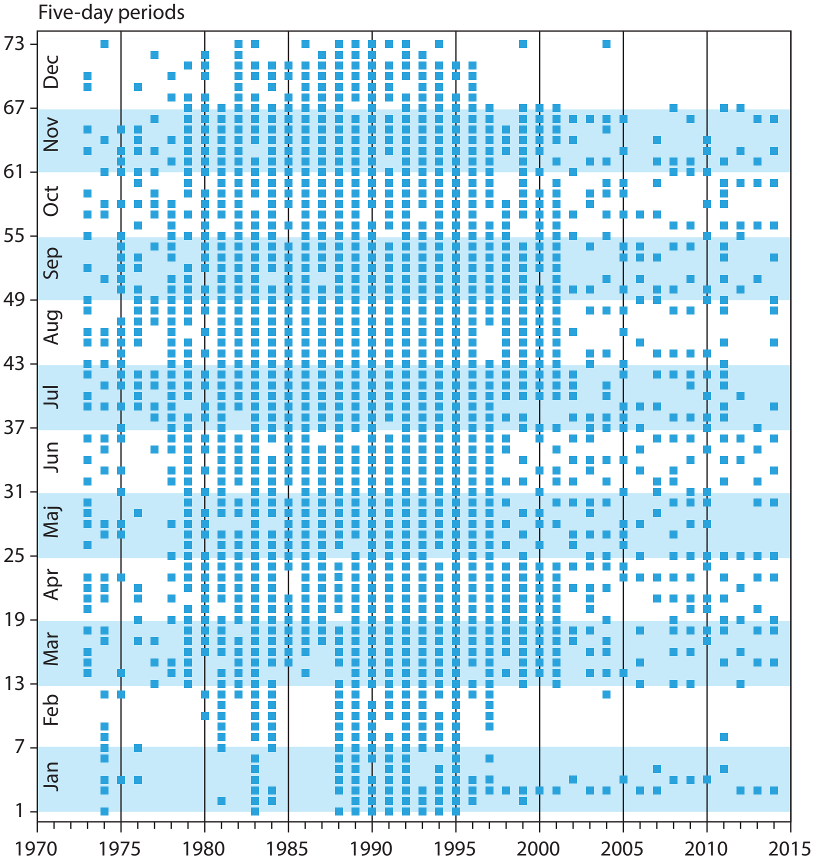


Fig. 7. Five-day periods with 'complete' censuses of waterbirds in the Tipperne Reserve since censuses were standardised in 1973.

Femdagesperioder med totaltællinger af vandfugle i Tipperreservatet siden standardiseringen af tællingerne i 1973.

(Fig. 7). In 2014, only one count was performed in July, but two in August. For this reason, data from these three years were omitted from the trend analyses for all species with peak migration of either adults or juveniles – or both – in July-August.

For species breeding in large numbers on the Reserve, 'staging' individuals were not counted during the breeding season up to and including 1968, when only the number of pairs multiplied by two was recorded in the journals for extended periods (see Meltofte 1987 and the discussion). Such data have been excluded from the present analyses. Data from before 1973 must also be considered of considerably lower quality both because of lack of standardization and because the survey coverage before May and after mid-August was often curtailed (Meltofte 1987). Furthermore, a number of very high 'counts' of certain species during the 1930s and particularly the 1940s should be treated with some caution (see the individual species and the discussion). The same applies to data on *Calidris* species apart from Dunlins before 1966, because these species were not recorded systematically until then.

In this account, we present both phenological graphs and trends in accumulated 'bird-days' (one bird in one day) during the standardized study period 1973-2014. Phenological graphs give average numbers per five-day period for the entire study period together with average numbers per 10-day period for the first 15 years (1973-1987) compared with the last 15 years (2000-2014). The phenology graphs were calculated by simple averaging of 'total' numbers for each five-day period over the two respective 15 year periods, leaving years where no count had been made in a given five-day period out of the calculations.

Trends in accumulated bird-days are given for selected parts of the annual phenology, i.e. spring migration separately from autumn migration, and adult autumn migration separately from juvenile autumn migration (Tab. 1). For a few species, even the spring migration is separated between an early and a late part according to the passage of more or less separate populations/subspecies. The dates of separation were chosen from data presented by Meltofte (1993), yet it is important here to note that these separations do not mean that all birds

Tab. 1. Dates separating sequences of the annual phenology given as the first date of the first five-day period in the beginning of each sequence.

Opdelingen af forårs- og efterårstrækket i perioder med forskellige sekvenser og aldersgrupper af hver art angivet som begyndelsesdatoen i den første femdagesperiode i hver sekvens.

Species / Phenological sequence	**Spring I-II Migration**	**Autumn/ad. migration**	**Juvenile migration**
Eurasian Oystercatcher *Haematopus ostralegus*	–	30.6	–
Pied Avocet *Recurvirostra avosetta*	–	25.6	–
Grey Plover *Pluvialis squatarola*	–	30.6	08.9
Eurasian Golden Plover *Pluvialis apricaria*	–	20.6	18.9
Common Ringed Plover *Charadrius hiaticula*	21.4	30.6	29.8
Northern Lapwing *Vanellus vanellus*	–	31.5	–
Whimbrel *Numenius phaeopus*	–	10.6	–
Eurasian Curlew *Numenius arquata*	–	31.5	30.7
Bar-tailed Godwit *Limosa lapponica*	–	20.6	29.8
Black-tailed Godwit *Limosa limosa*	–	31.5	–
Ruddy Turnstone *Arenaria interpres*	–	20.6	14.8
Red Knot *Calidris canutus*	–	30.6	19.8
Ruff *Calidris pugnax*	–	10.6	04.8
Curlew Sandpiper *Calidris ferruginea*	–	30.6	14.8
Temminck's Stint *Calidris temminckii*	–	20.6	09.8
Dunlin *Calidris alpina*	21.4	20.6	24.8
Little Stint *Calidris minuta*	–	30.6	14.8
Common Snipe *Gallinago gallinago*	–	20.6	–
Jack Snipe *Lymnocryptes minimus*	–	20.6	–
Common Sandpiper *Actitis hypoleucos*	–	20.6	04.8
Green Sandpiper *Tringa ochropus*	–	31.5	–
Spotted Redshank *Tringa erythropus*	–	31.5	04.8
Common Greenshank *Tringa nebularia*	–	10.6	04.8
Common Redshank *Tringa totanus*	26.4	31.5	30.7
Wood Sandpiper *Tringa glareola*	–	10.6	20.7

within these time periods were e.g. either adults or juveniles, only that the majority belonged to the indicated category.

The number of bird-days was calculated for each year by averaging numbers per five-day period counted on 'total' counts on the Reserve within a given observation period, and multiplying the average by the number of days in that period; e.g. in the case of adult Red Knot *Calidris canutus* that dominate the knot flocks from 30 June to 19 August (Tab. 1), data from ten five-days periods were used to calculate the average 'adult knot' number and the number of bird-days was calculated by multiplying the average by 50 days. All five-day periods without 'total' counts were omitted from these computations.

For the entire study period 1929-2014, nine-year running medians based on maximum numbers within the same annual time periods are given in separate graphs, since the data from the first 44 years do not allow calculation of bird-days. For the latter part of the study period, where counts were made more systematically, only maximum values from counts with 'total' coverage of the Reserve are included. This means that higher maximum values may exist from other counts. This is especially so for the years 1973-1977, when relatively few 'total' counts were performed, until full censuses were introduced in 1978 (Fig. 7).

Polynomial regression analyses (Zar 1984) were applied to describe the temporal trends in bird-days for 1973-2014 (or 2011). Five polynomial models with increasing degree of polynomial variables were run for each species. The first model was of the form Y = intercept, second model was $Y = intercept + \beta_1 \cdot X$, third model was $Y = intercept + \beta_1 \cdot X + \beta_2 \cdot X^2$, fourth model was $Y = intercept + \beta_1 \cdot X + \beta_2 \cdot X^2 + \beta_3 \cdot X^3$ and the fifth model was $Y = intercept + \beta_1 \cdot X + \beta_2 \cdot X^2 + \beta_3 \cdot X^3 + \beta_4 \cdot X^4$, where X represents the year, Y represents the number of birds, and β is the parameter estimates. F-tests were applied to compare the models. The polynomial model with the highest degree of polynomial variable which significantly improved the description at the 5% level was chosen. The statistical trend analyses were performed using the software R version 3.01 (R Core Team 2013).

We define short term trends here as statistically significant trends within the last 15 years, which is half the climatology period of 30 year averages as defined by the World Meteorological Organization. Similarly, we define long term trends as trends over time spans of more than the 30 year climatology period; however, here these trends are based on subjective evaluations of the median curves of maximum numbers from the entire study period on Tipperne. At a flyway scale, this means trends within the 20th century for most species. Fifteen years for short term trends was chosen in order to avoid too much influence of interannual variability, which can easily bias attempts to elucidate actual trends.

Increases are defined as > 25% increase, decreases as > 20% decrease, strong increases as > 100% increase, strong decreases as > 50% decrease.

Data on breeding bird numbers are based on Thorup (1998, updated *in litt.*). For the common wader species breeding on the Reserve these data are given in the graphs with nine-year running medians.

Species accounts

Eurasian Oystercatcher *Haematopus ostralegus*

In spring, numbers of Oystercatchers recorded on Tipperne reflect to a large extent developments in the breeding population, with a surplus of migrants in most years (Fig. 8; Meltofte 1987). This means that numbers were relatively high (up to 50-150 individuals and 40-60 pairs) during the 1930s and 1940s, but decreased quite markedly after the cessation of hay cutting on the Reserve during the 1950s and 1960s resulted in overgrowth of the meadows (Thorup 1998). Following a minimum of 0-8 pairs during the 1970s, the breeding population recovered to some extent after hay cutting and cattle grazing were increasingly re-established from the mid-1970s onwards. The breeding population reached a small maximum of up to around 20 pairs during the late 1980s and early 1990s, after which the population again decreased to below 10 pairs in the last 10 years (Thorup 1998 and *in litt.*). A further 3-6 pairs bred on Værnengene on the southern part of the Tipperne Peninsula, representing a reduction from around 5-15 pairs in the 1980s and 1990s (O. Amstrup *in litt.*).

In addition to local breeders, a larger proportion of staging migrants and probably also non-breeders seem to have occurred during the last 2-3 decades (Fig. 8). Indeed, numbers of 'excess birds' were higher in spring during 2000-2014 compared to 1973-1987 in spite of a decreasing breeding population (Fig. 9). This includes that numbers of bird-days both in spring and autumn increased up to the 2000s, only to decrease again over the last few years, possibly as a result of reduced counting frequency (Fig. 10).

Besides vegetation height on the meadows (which is only relevant for the breeders, because staging Oystercatchers feed only on the mudflats), the occurrence and availability of food for adults and young in the form of ragworms and bivalves (sand gaper) have also played a role (Thorup 1998). For example water levels in spring

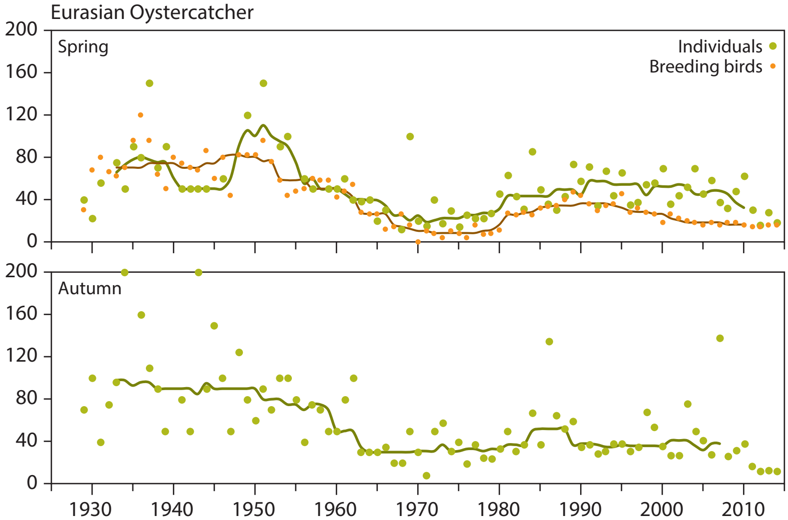


Fig. 8. Annual maximum numbers of Eurasian Oystercatcher per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Strandskade pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

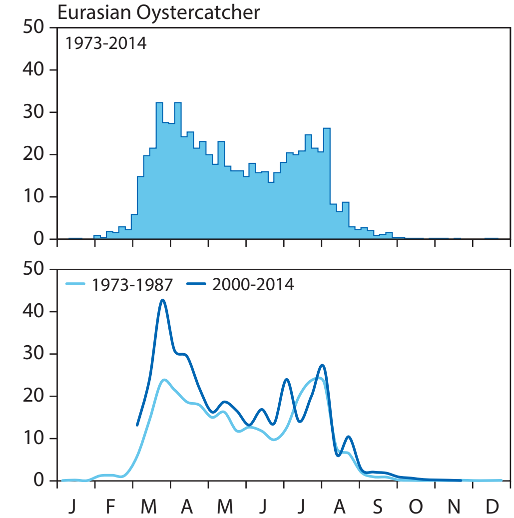


Fig. 9. The phenology of Eurasian Oystercatcher on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Strandskader på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

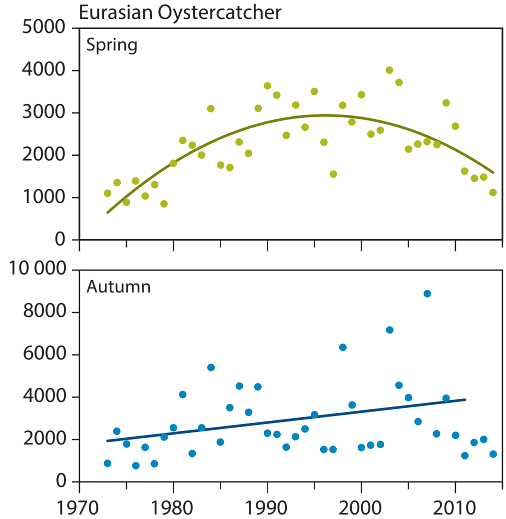


Fig. 10. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Eurasian Oystercatcher with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Strandskade pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

were particularly low during 2000-2010 (Fig. 3) when the numbers of Oystercatchers were also higher than accounted for by the breeding population (Fig. 8).

In autumn, numbers were also considerably higher during the 1930s, 1940s and 1950s with a majority of maximum records of between 50 and 100 individuals and a few records of up to 200 individuals (Fig. 8). However, during the last five decades numbers have not varied to the same extent as the spring totals (Fig. 8), in that the majority of maximum records have been between 30 and 70 individuals and a few of up to 140. Similarly, numbers of bird-days have remained at much the same level since the early 1980s (Fig. 10).

Numbers of Oystercatchers on Tipperne closely follow trends in the breeding population, yet with a surplus of visitors in spring and autumn. Photo: Ulrik Bruun.

The European *H. o. ostralegus* flyway population – now numbering between 850 000 and 950 000 individuals – has been declining since around 1990 (Delany *et al.* 2009, Meltofte *et al.* 2006, Nagy *et al.* 2014, 2015, van de Pol *et al.* 2014, Blew *et al.* 2015, van Roomen *et al.* 2015). This has resulted in listing of the Eurasian Oystercatcher as Near Threatened on IUCN's Red List of Threatened Species (BirdLife International 2015a). The recent decline was preceded by a strong increase during much of the 20th century in north-western Europe as a result of better protection (Cramp *et al.* 1983), but numbers on Tipperne show no correlation to this (Fig. 8). The Danish breeding population was estimated to number between 10 000 and 14 500 pairs at the turn of the century and has been in serious decline during recent years (Thorup 2006a, Nyegaard *et al.* 2015).

Conclusion: Numbers of Oystercatchers on Tipperne are generally small and 'entirely' governed by local conditions primarily in the form of food availability on the shallow flats and vegetation height for breeders on the meadows.

Pied Avocet *Recurvirostra avosetta*

Avocets have bred in highly varying numbers on Tipperne ever since monitoring began 86 years ago, and from around 1960 the Reserve developed into the second most important post-breeding staging and moulting area for this species in Denmark (Fig. 11; Meltofte 1987, 1993, Thorup 1998). In most years between the 1920s and mid-1940s, the breeding population numbered between 250 and 500 pairs, after which strong predation pressure and overgrowth of the meadows reduced the population to between five and 60 pairs in the early 1970s. Reintroduction of hay cutting and cattle grazing together with culling of foxes *Vulpes vulpes* led to the re-establishment of a population of 400-500 pairs during the 1980s and '90s, followed by a new decrease to only a few tens of pairs in recent years. Varying numbers of up to 55 pairs have also bred on Værnengene on the southern part of the Tipperne Peninsula (O. Amstrup *in litt.*).

Avocets now arrive almost one month earlier than they did at the beginning of the study period in 1929 (Meltofte 1987, Petersen *et al.* 2012). Avocets have also increasingly begun to remain on the Reserve in November and a few do so even into December (Fig. 12; Meltofte 1987).

Numbers recorded on the Reserve in spring closely resemble the breeding population, especially when birds from adjacent colonies in the fjord are taken into account (Fig. 11; Thorup 1998). In contrast to this, the development of the Reserve as a major moulting area in July-September happened during the 1950s and 1960s when there were relatively few breeders (Fig. 11; Meltofte 1987). Since round 1960, post-breeding flocks of up to between 500 and 2500 adults and juveniles have been recorded in most years (max. 3840 in late July 1965), but numbers have always been irregular, so that the numbers of bird-days decreased from a maximum around 1980 to a minimum in the mid-1990s, only to increase again up to the late 2000s – in spite of the very low breeding population and with the exception of the last few years (Fig. 11). Since 2003, this trend was probably facilitated by relatively low water levels in late summer and early autumn, while there were higher July water levels (Fig. 3) and few counts during the last few years (Fig. 7).

The slowly decreasing average numbers on the Reserve during spring and early summer (Fig. 12) probably reflect birds both departing for other breeding sites in the fjord and increasing vegetation height, making it increasingly difficult to see all birds. The pronounced difference between the early period 1973-1987, when most post-breeding birds remained on the Reserve during July-September, and 2000-2014 when many more birds used the Reserve for post-breeding moult (Fig. 12), is striking and resembles the situation in the early 1970s when more than 1000 Avocets from other breeding sites congregated on Tipperne to moult (Meltofte 1987). Immigration of large numbers of Avocets may occur even into late September and October.

Moulting birds apparently to some extent alternate between staying on the Reserve and in another reserve in the fjord, Klægbanken, 12 km northeast of Tipperne (Fig. 1), where up to 779 Avocets have been recorded in late summer (P. Clausen unpubl.). In years with no or only few moulting Avocets in the fjord, they probably moult together with the large congregations that gather 35-90

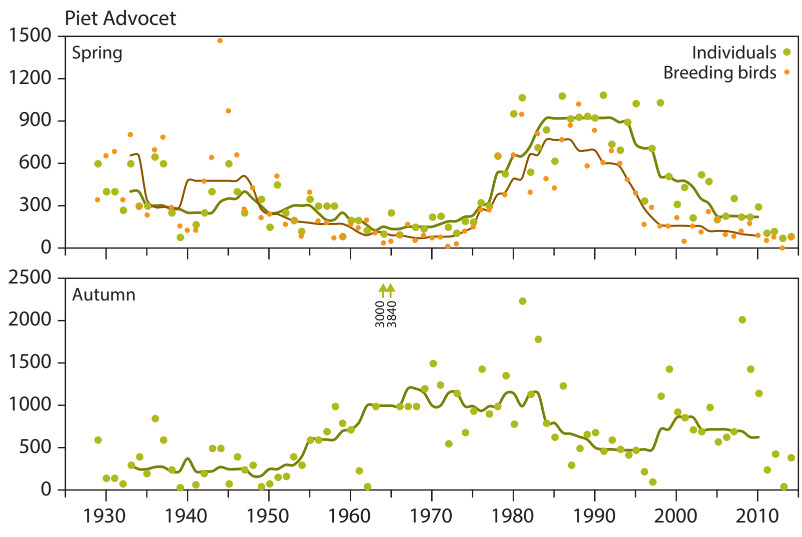


Fig. 11. Annual maximum numbers of Pied Avocet per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Klyde pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

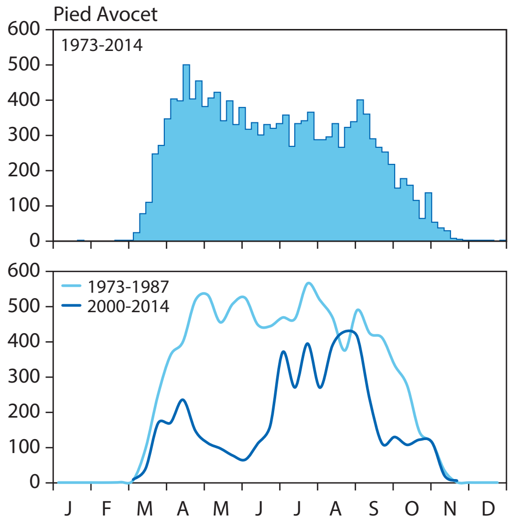


Fig. 12. The phenology of Pied Avocet on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Klyder på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

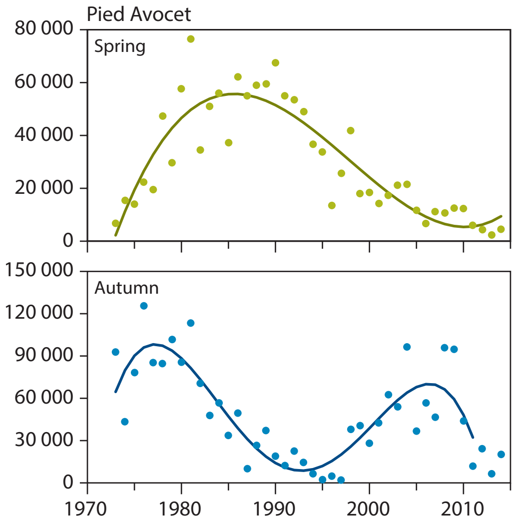


Fig. 13. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Pied Avocet with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Klyde pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Numbers of breeding Pied Avocets on Tipperne peaked during the 1980s and early 1990s and then dropped considerably. Photo: John Larsen.

km to the south in the Wadden Sea (see Laursen & Frikke 2013) or in Bøvling Fjord 60 km north of Tipperne, where up to 2500 birds have been recorded (Clausen *et al.* 2001) – i.e. far more birds than the local breeding population can account for (Christensen & Østergaard 2012). The possibility cannot be excluded that the reason for the low numbers of moulting Avocets on the Reserve in some of the periods with many breeders is the reduction in invertebrate densities on the mudflats that may result from intensive feeding during the preceding breeding season (Meltofte 1987).

The development of a moulting ground on Tipperne from around 1960 may be related to the pronounced increase in the breeding population in Denmark and southern Scandinavia during the preceding decades (Meltofte 1987). On a local scale, numbers of both breeders and post-breeders – recorded as bird-days (Fig. 13) – were lowest at the supposed peak in benthic prey on the mudflats during the mid-1990s. During this period, the extraordinary high water levels in summer and early autumn 1997-2002 (Fig. 3) may have played a role (cf. Thorup 1998, Meltofte & Clausen 2011). The extraordinary low numbers in the last few years may be the result of poor survey coverage.

The West European flyway population is estimated to number between 88 000 and 98 500 individuals (Nagy *et al.* 2015) following recovery during the 20th century from persecution and colder conditions (Cramp *et al.* 1983, Delany *et al.* 2009). However, in spite of a further increase from around year 2000 onwards (Nagy *et al.* 2014, van Roomen *et al.* 2015), numbers in the Wadden Sea have shown a decreasing trend during the last 25 years (Blew *et al.* 2015).

Like in other West European countries, the Danish Avocet population increased significantly during much of the 20th century and culminated at about 5000 breeding pairs around 1990 (Bregnballe *et al.* 2015). However, since then the population has declined to maybe only 2500 pairs in 2014, with the decline of the colony on Tipperne as a major contributor (Bregnballe *et al.* 2015).

Conclusion: Numbers of Avocets on the Tipperne Reserve in spring are closely related to the numbers of breeders in the fjord, while numbers of moulting post-breeders partly deviate from this pattern in that large numbers apparently may take advantage of previously 'un-tapped' resources in periods with few breeders and sufficiently low water levels. The initiation of Tipperne as a moulting area around 1960 may also have been related to the population increase in north-western Europe at a time when few birds were breeding on Tipperne. Furthermore, the arrival of the species on the Reserve is now about one month earlier than 80 years ago.

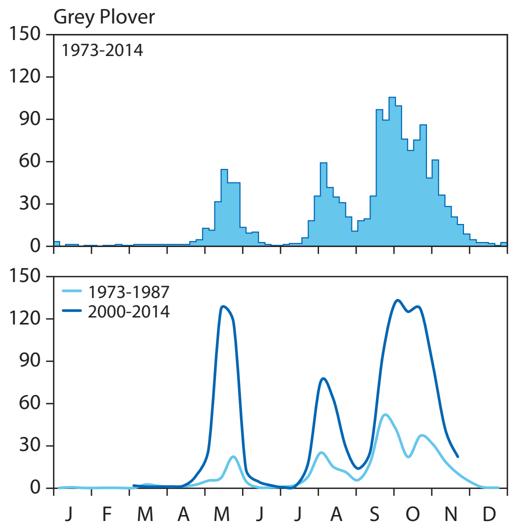


Fig. 14. The phenology of Grey Plover on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Strandhjejler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

Grey Plover *Pluvialis squatarola*

Grey Plovers occur in 'appreciable' numbers primarily on exposed flats on Tipperne Reserve during the spring, adult autumn and juvenile autumn migrations (Fig. 14). In recent decades, maximum numbers have reached 100-300 adults in both spring and autumn and 100-600 juveniles in most years (Fig. 15). These figures are significantly higher than those recorded before 1980, when maximum numbers were considerably below 100 in most years (Meltofte 1987). Exceptions to this general pattern in the form of higher numbers occurred in years with particularly low water levels, and for the juveniles also years with birds staging on the meadows – to the extent that these in the early decades were not misidentified Eurasian Golden Plovers *Pluvialis apricaria*! Like with several other species, the low numbers recorded in the very last years, both in maximum numbers and in bird-days (Fig. 16), are probably the result of reduced counting frequency.

The marked increase in Grey Plovers on Tipperne Reserve since the 1980s cannot be explained by water level changes (cf. Fig. 3), whereas increasing densities of benthos may have contributed. The increase parallels a substantial increase in numbers of Grey Plovers wintering in north-western Europe since the mid-1980s. This increase probably reflects a milder winter climate that is

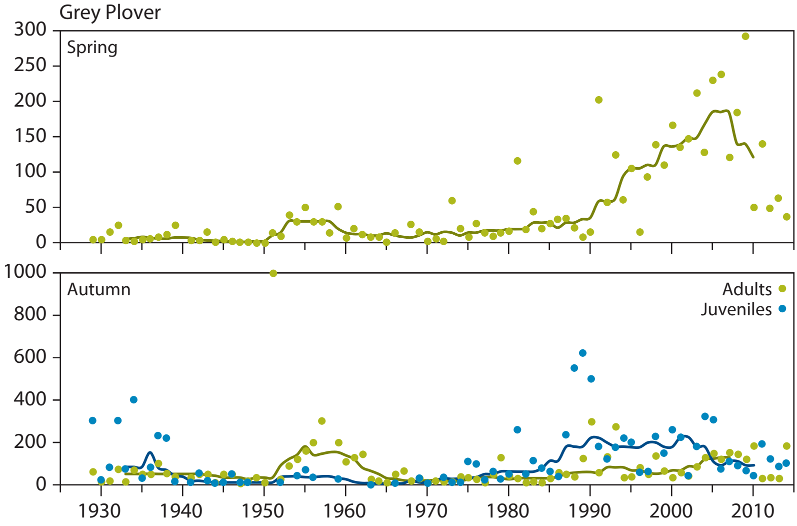


Fig. 15. Annual maximum numbers of Grey Plover per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Strandhjejle pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

thought to have resulted in a northward shift in the winter distribution of the East Atlantic flyway population (Delany *et al.* 2009; see also Bar-tailed Godwit *Limosa lapponica* below). Reduced hunting pressure in many West European countries may also have contributed to this northward shift (cf. Prater 1981: 111, Tubbs 1991, 1996). The population increase and westward expansion of the breeding range of the species in European Russia, which must have led to an increase in the European breeding segment of the flyway population that winters in West Europe (Hagemeier & Blair 1997, Engelmoer 2008), may also have contributed to this shift.

Delany *et al.* (2009) state that the midwinter population, which now numbers about 200 000 individuals distributed between north-western Europe and West Africa (Nagy *et al.* 2015), is declining, whereas Nagy *et al.* give it as stable or fluctuating. This ambiguity is also found in the most recent analyses of midwinter census data, which shows that a marked increase in the flyway population took place up to the mid-1990s followed by a moderately decreasing trend since then (Nagy *et al.* 2014, van Roomen *et al.* 2015). Year round data (i.e. including both the European and the African wintering segments) from the Wadden Sea show relatively stable

Water levels are important for the feeding conditions for waders on Tipperne; here a Bar-tailed Godwit and four Grey Plovers feeding on the flats. Drawing by Jens Gregersen.

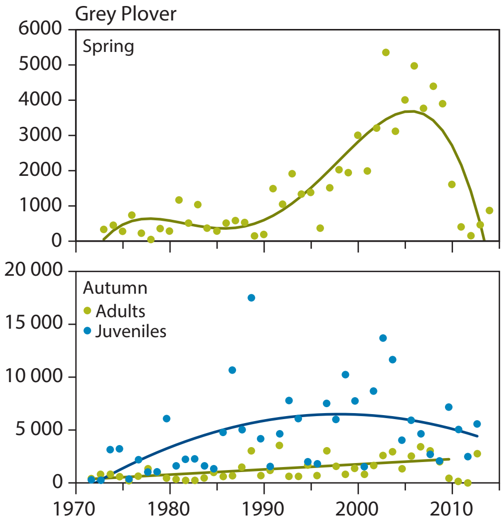


Fig. 16. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Grey Plover with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Strandhjejle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

numbers since the late 1980s, but again with a decreasing trend during the most recent years (Blew *et al.* 2015). In West Africa, wintering numbers have been relatively stable since 1980 except for a low count in 2014 (van Roomen *et al.* 2015). Fluctuating numbers culminating in the 1990s were also found among Grey Plovers passing Blåvandshuk south of Tipperne during 1964-2003 (Meltofte *et al.* 2006).

Arrival of Grey Plovers on the Reserve in spring has advanced more than any other species analysed, i.e. no less than 65 days since 1929 (Petersen *et al.* 2012). This advance is most likely the result of birds from the European wintering segment of the population moving up to the northern part of the Wadden Sea and adjacent areas earlier than they did before (cf. Meltofte 1993, Laursen & Frikke 2013). In the Wadden Sea, numbers of these European-breeding birds predominate up to early May, while numbers of Siberian breeders predominate in mid and late May after having wintered in West Africa (Meltofte *et al.* 1994, Engelmoer 2008).

Conclusion: The Grey Plover is one of the few wader species on Tipperne where trends in numbers on the Reserve clearly reflect overall flyway population trends, i.e. a marked increase in the numbers of birds wintering in north-western Europe during the second half of the 20th century that now arrive on Tipperne as early as March.

Eurasian Golden Plover *Pluvialis apricaria*

Large numbers of Golden Plovers utilize Tipperne Reserve as a staging, moulting and roosting site during both spring and autumn passage (Fig. 17). On the Reserve they can use short-grass meadows for feeding, particularly in spring, and short-grass meadows as well as exposed mudflats for roosting, particularly in autumn (Meltofte 1987). During 2011-2013, 1-2 pairs of Golden Plovers bred on the Reserve as one of the few breeding sites for this species in Denmark (Nyegaard *et al.* 2014).

In spring, numbers have varied quite a lot in that very few Golden Plovers utilized the Reserve up to around 1950, after which annual peak numbers grew to 2000-3000 in a number of years (Fig. 18; Meltofte 1987). Somewhat lower numbers were present in the period with tall vegetation on the meadows in the late 1960s and early 1970s. Again, very low numbers of Golden Plovers used the Reserve in spring during most of the 1980s, followed by higher numbers than ever from the late 1990s (Figs 17, 18 and 19). This latter development may be related to the massive increase in Barnacle Geese *Branta leucopsis* grazing the meadows in March-April and early May from the second half of the 1990s onwards (Thorup in print), thereby creating optimally low vegetation for the plovers. Furthermore, it is noteworthy that numbers now peak as early as in April, so that most birds have left

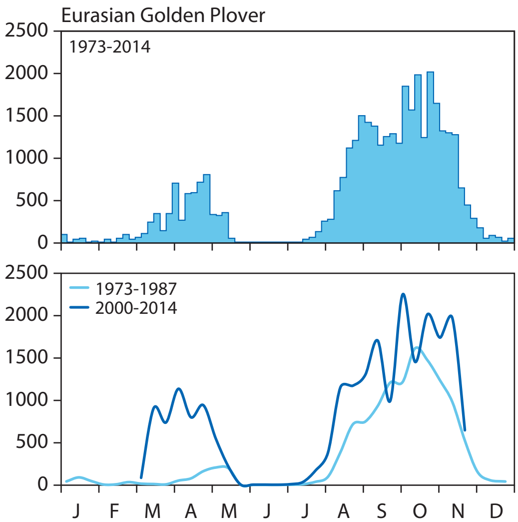


Fig. 17. The phenology of Eurasian Golden Plover on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Hjejler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

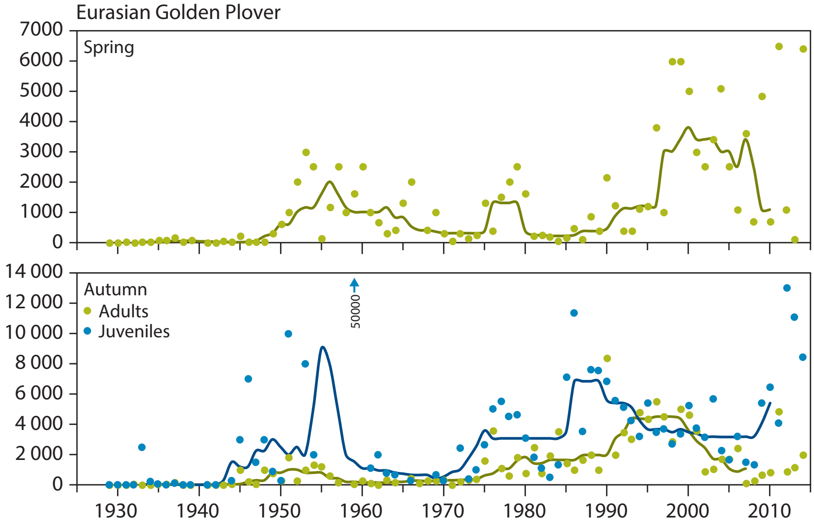


Fig. 18. Annual maximum numbers of Eurasian Golden Plover per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Hjejler pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

by the time numbers peak in west Jutland in general, and compared to when they peaked previously (Fig. 17; Meltofte 1987).

In autumn, numbers of Golden Plovers are dominated to a large extent by large flocks of day roosting birds which feed by night in surrounding areas during periods of full moon (Meltofte 1987; see also Northern Lapwing *Vanellus vanellus*). This means that numbers have been highly variable both within years and between the study years. Like in spring, very few adults were recorded during the 1930s, until concentrations of up to 300-1800 were seen in the late 1940s and early 1950s (Fig. 18; Meltofte 1987; see also Northern Lapwing). This was followed again by low numbers during the period with tall vegetation on the meadows, particularly during the 1960s, until management re-established short-grass meadows and numbers of adults increased during the 1970s and 1980s. Numbers peaked with maxima of 3000-6000 in most years during the 1990s, only to decrease again from around 2002, possibly due to reduced counting frequency (Fig. 18). With few exceptions, bird-days spent by adults on the Reserve mirror this decrease (Fig. 19). No correlation was found between numbers of adults and the wetness of the meadows (depth of the ground water table) during 1 August-20 September 1987-2014.

Since many adult Golden Plovers stay to moult in west Jutland (Meltofte 1993), numbers of 'juveniles' given in the graphs also include these birds. Like for the adults, high numbers were recorded between the mid-1940s and late 1950s (several years had up to between 3000 and 10 000 individuals, and even 50 000 stated for mid-November 1959) and again from the late 1970s (up to between 3000 and 7500 in most years and exceptionally 11 000-13 000; Fig. 18). During 1994-2010, autumn bird days on Tipperne correlated statistically significantly with index numbers of Golden Plovers in the Danish reserve network in general (Clausen *et al.* 2013). On Tipperne, particularly high maximum and bird-day numbers were recorded in the juvenile migration period over the last few years (Figs 18 and 19).

The relevant flyway population of *altifrons* Golden Plovers is estimated to have between 800 000 and 1.4 million individuals (Nagy *et al.* 2015). If the population development in Finland can be taken as indicative of this population in general, there was a marked increase during the middle of the 20th century (Cramp *et al.* 1983), an increase that can be related to reduced hunting mortality (Piersma *et al.* 2005). More recently, numbers of breeding Golden Plovers in Norway, Sweden and Finland have been relatively stable during 2002-2013 (Lindström *et al.* 2015), while the population development at flyway scale was given as possibly increasing by Delany *et al.* (2009) backed up by a quite substantial increase in the international midwinter counts (Nagy *et al.* 2014). This was 'downgraded' to stable by Nagy *et al.* (2015), while decreasing numbers have been recorded in the Wadden Sea (Gillings *et al.* 2012, Blew *et al.* 2015). This quite substantial uncertainty may be the result of varying numbers of Golden Plovers utilizing coastal habitats, where survey coverage is better than inland. This possibility is supported by large fluctuations in midwinter totals in recent years (Nagy *et al.* 2014). Numbers of birds passing Blåvandshuk south of Tipperne on active migration, i.e. unaffected by such bias, were relatively stable during 1964-2003 (Meltofte *et al.* 2006) like was found at

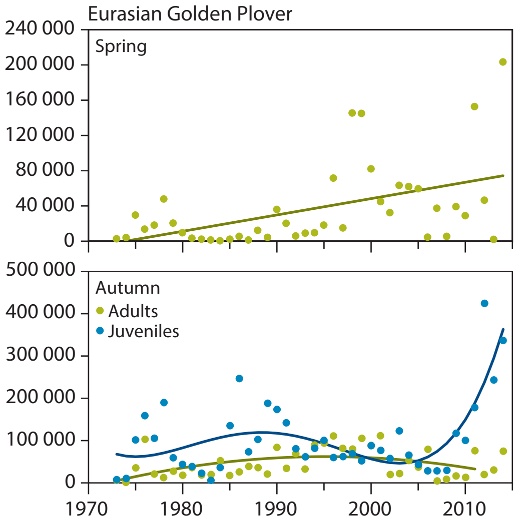


Fig. 19. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Eurasian Golden Plover with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Hjejle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

recent breeding bird censuses in Norway, Sweden and Finland as mentioned above.

Numbers staging during autumn in Denmark grew considerably after the species was protected from hunting in 1983, which allowed many more Golden Plovers (> 300 000 in October) to stay in Denmark during the autumn moult (Rasmussen *et al.* 2010). From a long term perspective, numbers on Tipperne have actually also gone up (Figs 18 and 19).

Conclusion: Golden Plovers are sensitive to height of the vegetation on the meadows and to water level, so that large numbers feed and roost on the meadows and the mudflats only when the vegetation is low and/or the mudflats are exposed. Numbers on the Reserve have varied a lot in relation to these conditions. These, however, do not explain all of the variation, and feeding possibilities in neighbouring night-time feeding areas together with intrinsic factors such as tradition for choice of day roost may also play a role.

Common Ringed Plover *Charadrius hiaticula*

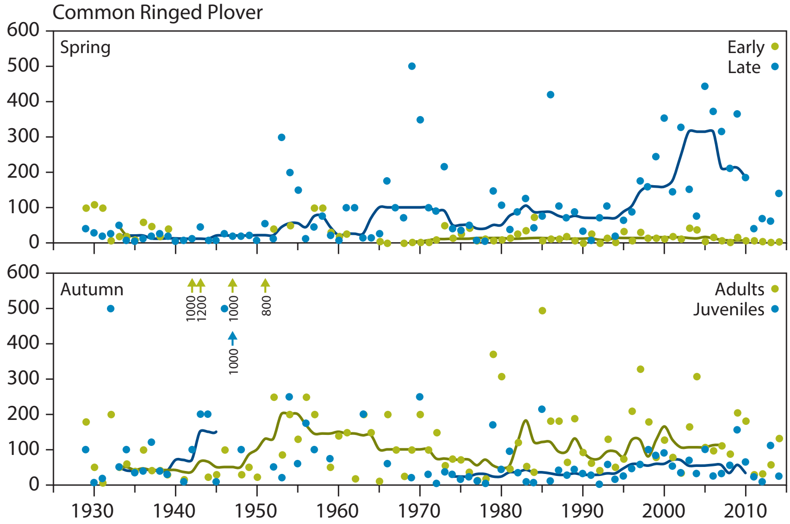
During recent decades in particular, up to several hundred Common Ringed Plovers have utilized Tipperne Reserve during spring migration (Fig. 20). Besides a modest spring peak of 'southern' *C. h. hiaticula* birds in March (Fig. 21), numbers in spring culminate during the migration of Arctic and sub-Arctic populations of *C. h. tundrae/psammodroma* in May (Meltofte 1987, 1993, Fischer & Meltofte 2015). These birds probably also dominate the adult autumn migration in August, when both maximum numbers and bird-days were relatively stable during the study period (Figs 20 and 22). Only a few pairs bred on the Reserve in the 1930s and 1940s (Thorup 1998).

Numbers vary considerably from year to year, partly depending on the occurrence of exposed flats on the Reserve (Meltofte 1987). For example, exceptionally low water levels in spring during 2000-2010 (cf. Fig. 3) may have contributed to the high maximum numbers as well as bird-days in this period (Figs 20 and 22). In contrast to the adults, numbers of autumn staging juveniles on Tipperne may have decreased since the relatively high

The occurrence of Common Ringed Plovers on Tipperne is dominated by northern 'tundra' populations in May and August-September. Photo: Peter Vadum.

Fig. 20. Annual maximum numbers of Common Ringed Plover per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Stor Præstekrave pr. sæson (jvf. Tab. 1) med niårs glidende medianer.



numbers that were recorded during the 1940s, 1950s and 1960s (Fig. 20). This may again have been caused by changes in water levels at the critical times of the year (Meltofte 1987), and numbers have been relatively stable during the last 40 years (Figs 20, 21 and 22).

Like spring staging 'tundra' Common Ringed Plovers on Tipperne, annual numbers of staging birds in the Wadden Sea increased over the last two decades (Blew *et al.* 2005, 2015). This is somewhat contrary to the statement by Delany *et al.* (2009) that the Greenlandic/Icelandic *psammodroma* population is decreasing, and that the temperate N European *hiaticula* population is "possibly decreasing". These populations currently number 240 000 individuals and between 55 600 and

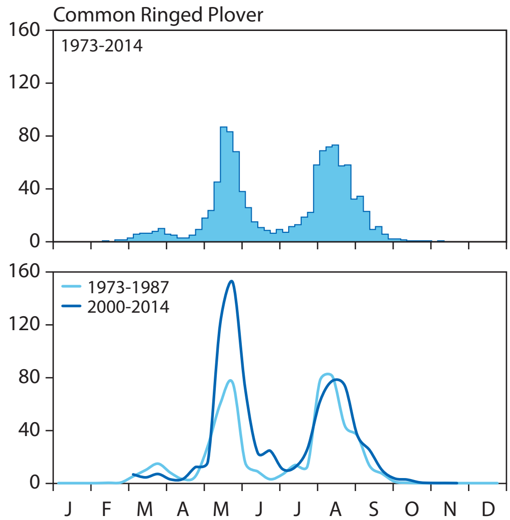


Fig. 21. The phenology of Common Ringed Plover on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Store Præstekraver på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

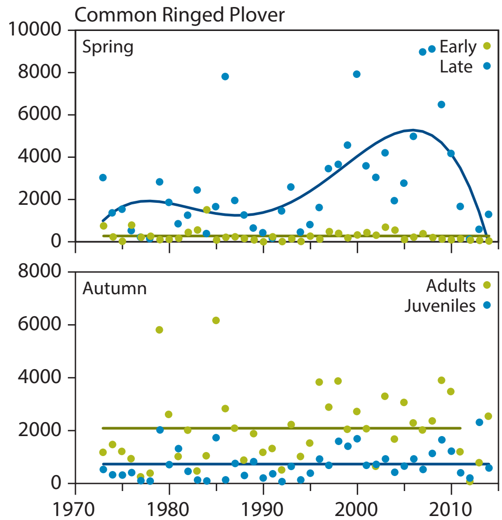


Fig. 22. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Common Ringed Plover with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Stor Præstekrave pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

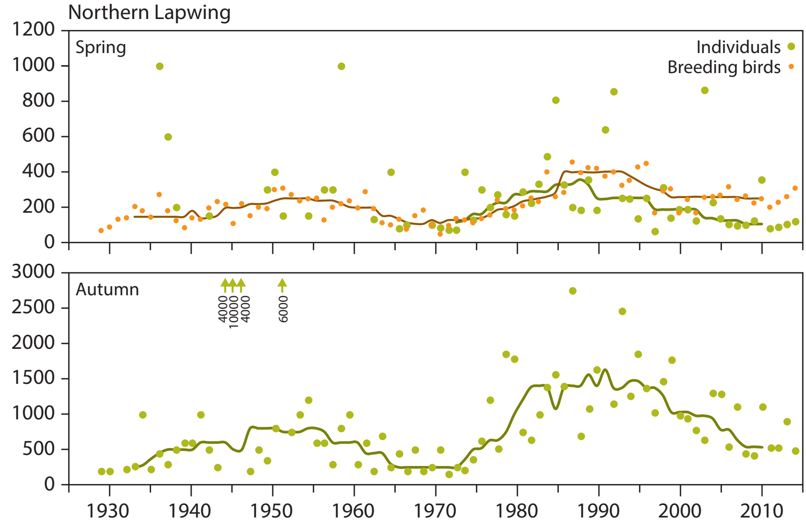


Fig. 23. Annual maximum numbers of Northern Lapwing per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Vibe pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

68 600 individuals, respectively (Nagy *et al.* 2015), and they constitute an important proportion of the birds of the Wadden Sea (Meltofte *et al.* 1994, Engelmoer 2008).

The size and trends of the *tundrae* population is unknown, but it is considered to number several hundred thousand birds and be the dominant component of the staging birds in late spring and early autumn in the Wadden Sea (Meltofte *et al.* 1994, Engelmoer 2008, Nagy *et al.* 2015). Numbers of Common Ringed Plovers passing Blåvandshuk south of Tipperne – probably primarily involving *tundrae* and secondarily *psammodroma* (Meltofte 1993) – on active autumn migration remained stable during the 40 year period 1964-2003 (Meltofte *et al.* 2006). The *psammodroma* population is given as fluctuating or decreasing 1979-2014 by van Roomen *et al.* (2015; see also Nagy *et al.* 2015), and the most recent analysis of the midwinter counts 1989-2012 shows an initial moderate increase in the *hiaticula* population, followed by fluctuating numbers (Nagy *et al.* 2014). Trend information on *tundrae* in the European Arctic of Russia is ambiguous (Lappo *et al.* 2012), but the breeding populations of mixed subspecies in Norway, Finland and northern Sweden increased during 2002-2013 (Lindström *et al.* 2015). The Danish breeding population was estimated to number 1900-2500 pairs at the turn of the century and to be in serious decline (Thorup 2006a, Nyegaard *et al.* 2015).

Conclusion: Numbers of Arctic and sub-Arctic *tundrae/psammodroma* Common Ringed Plovers staging on Tipperne Reserve in May have increased considerably during the study years, probably mainly because of lower water levels in spring, particularly in the 2000s. Numbers of autumn staging adults and juveniles of unknown subspecies have fluctuated less systematically.

Northern Lapwing *Vanellus vanellus*

Numbers of breeders on the Reserve – which varied from 50 to 200 pairs in most years between 1928 and 2014 – were outnumbered by staging migrants in autumn, when maximum numbers most often reached 500-1500 individuals, with absolute peak numbers of up to a few thousand (Figs 23 and 24). The breeding population peaked during the late 1980s and early 1990s, when fewer predators appear to have suppressed the wader populations, but has fallen again to 100-150 pairs since then (Fig. 23; Thorup 1998 and *in litt.*). Concomitantly, numbers on Værnengene on the southern part of the Tipperne Peninsula decreased from around 200 pairs in the late 1980s to generally around or less than 100 since 1999 (O. Amstrup *in litt.*). The absolute maximum number of staging individuals during recent decades was 5400 in November 1990, which is comparable to the peak numbers of up to 4000-6000 (or even 10 000) in the 1940s and early 1950s (Fig. 23; Meltofte 1987).

In spring, numbers recorded were generally lower than the number of breeders on the Reserve, but with a few records of more than 800 in March (Fig. 23). Numbers recorded tended to decrease during spring, when birds are increasingly incubating and hard to see, until the young hatch and early post-breeders begin to form flocks from late May (Fig. 24; Meltofte 1987). Autumn migrants tend to occur in two peaks; an early peak in July-September and a late peak between late October and early December, the latter probably representing birds that have moulted north and east of Denmark (Fig. 24; Meltofte 1993).

Many of the Lapwings staging on the Reserve between June and November primarily use the area as a

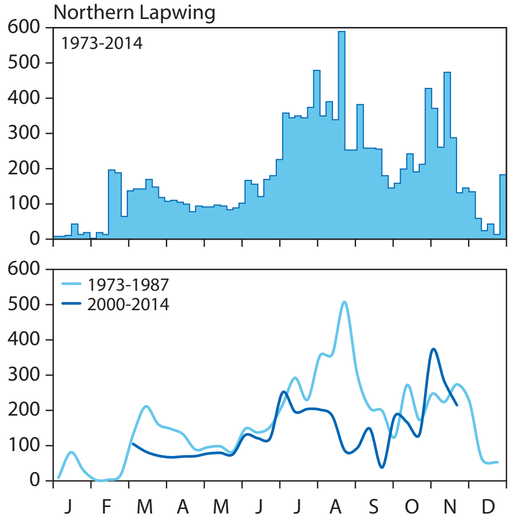


Fig. 24. The phenology of Northern Lapwing on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Viber på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

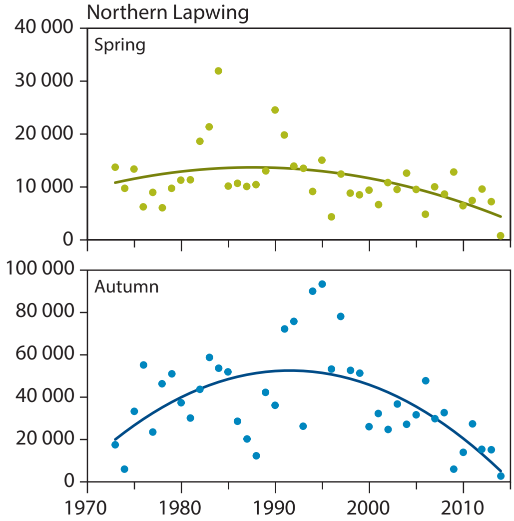


Fig. 25. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Northern Lapwing with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Vibe pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

The breeding populations of Northern Lapwings in parts of the Central- og West European agricultural landscape have declined considerable, while northern populations appear to be thriving. Photo: Jan Tandrup Petersen.

daytime roost during the moult, and their numbers are highly variable e.g. in relation to moon phase and exposure of the mudflats (Meltofte 1987; see also Eurasian Golden Plover). Moreover, the height of the vegetation on the meadows is important, which is reflected in the very low numbers recorded between the mid-1950s and the mid-1970s when there was little or no hay cutting and cattle grazing (Fig. 25; Meltofte 1987). The reason for the low numbers in the 1930s may be low densities of invertebrates on the meadows due to repeated salt water flooding in winter until 1937 (Fig. 2). The exceptionally high numbers recorded between the mid-1940s and early 1950s coincided with mass occurrences of crane fly *Tipula* larvae in west Jutland (Meltofte 1987). It is possible that the decreasing numbers of Lapwings during the last 15 years (Figs 23 and 25) are due to disturbances from increasing numbers of Peregrine Falcons *Falco peregrinus* hunting in the Reserve (Meltofte & Amstrup 2013). In the Danish reserve network as a whole, numbers of staging Lapwings were stable during the period 1994-2010 (Clausen *et al.* 2013).

Initiation of egg laying by Lapwings on Tipperne has advanced by nine days since 1928 (Petersen 2010). Arrival in spring has also advanced, apparently by about a week (cf. Tåning 1941 and Fig. 24). The Danish breeding population was estimated to number between 30 000 and 45 000 pairs at the turn of the century (Thorup 2006a) and has declined by about 80% between 1976 and 2014 – at least in the agricultural landscape (Nyegaard *et al.* 2015).

The European/West Asian flyway population of Northern Lapwings is estimated to number in the order of 5.5-9.5 million individuals and to be decreasing (Delany *et al.* 2009, Nagy *et al.* 2015). Like in Denmark, there is no doubt that the farmland breeding segment of Lapwings in central and western Europe has declined (Thorup 2006a), but a general decline in the flyway population has not been reflected in numbers staging in the Wadden Sea over the last 25 years (Blew *et al.* 2015). Similarly, with the exception of low numbers recorded twice in recent years, midwinter numbers on the African-Eurasian flyway have been rather stable during the last 25 years (Nagy *et al.* 2014). Furthermore, monitoring of breeding Lapwings in Norway, Finland and northern Sweden shows stable numbers, with even a statistically significant increase in Finland (Lindström *et al.* 2015). Finally, AEWA (no year) states that the flyway population increased between 1983 and 2007.

Recently, BirdLife International (2015b) has listed the Northern Lapwing as Near Threatened on IUCN's Red List of Threatened Species. Although the situation in the European farmland is certainly deeply worrying, based on the above data we find this listing hard to follow for the West Palearctic flyway population as such (see also under Eurasian Curlew *Numenius arquata*).

Rather than being linked to overall flyway population changes, any influence from outside areas on Lapwing numbers staging on Tipperne is likely to be primarily due to breeding and feeding conditions on nearby areas from where birds are recruited in summer and autumn (moulting birds using the Reserve as a daytime roost; Meltofte 1987, 1993).

Conclusion: Numbers of staging Lapwings on Tipperne Reserve are not only dependent on feeding possibilities on exposed mudflats and the meadows. Many birds also use the Reserve as a daytime roost, particularly around full moon. Their numbers are thus not determined only by feeding conditions on the Reserve but also by feeding opportunities in neighbouring recruitment areas.

Whimbrel *Numenius phaeopus*

Few Whimbrels occur on Tipperne Reserve in spring, while several hundred may use the Reserve in late summer, particularly as a night roost.

During the spring peak in the first half of May (Fig. 26), maximum numbers of Whimbrels on the Reserve rarely exceeded 50 (Fig. 27), and only twice (1991 and 1996) were more than 100 individuals recorded. During the most recent decades, maximum numbers actually increased (Fig. 27), while numbers of bird-days reached a

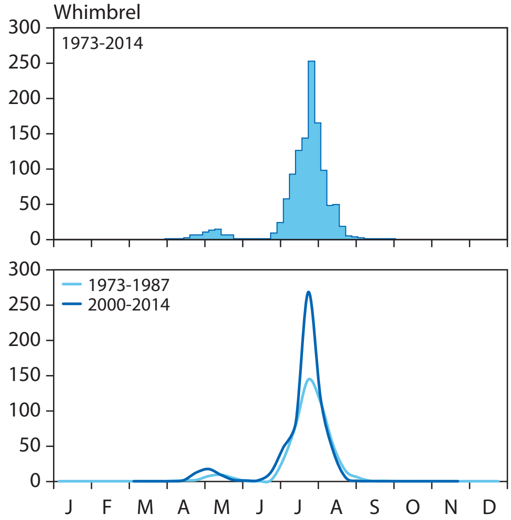


Fig. 26. The phenology of Whimbrel on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Småspover på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

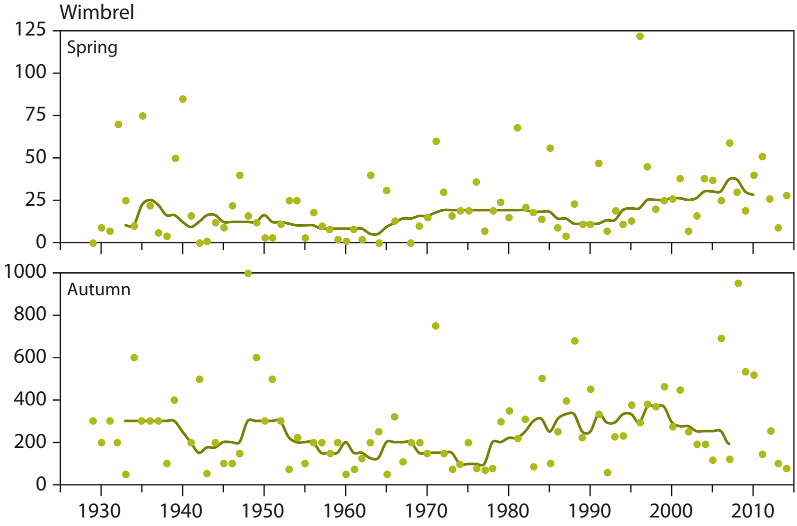


Fig. 27. Annual maximum numbers of Whimbrel per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Småspove pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

peak in the early 1990s (Fig. 28). Arrival of the first individuals on Tipperne advanced by 11 days during the 80 year period 1929-2008 (Petersen *et al.* 2012).

During the autumn migration in July-August (Fig. 26), maximum numbers generally were in the low hundreds, but up to around 1000 were recorded a few times (Fig. 27). Recording such numbers depended entirely on the special efforts of the observers to keep a look out for Whimbrels flying into the Reserve at night from feeding areas on adjacent heathlands and meadows – often so late that they are barely visible. Like in spring, numbers may have gone up from maximum numbers most often around 100-300 during the 1940s-1970s to often between 200 and 500 from the 1980s onwards (Fig. 27). Similarly, autumn bird-days have increased (Fig. 28).

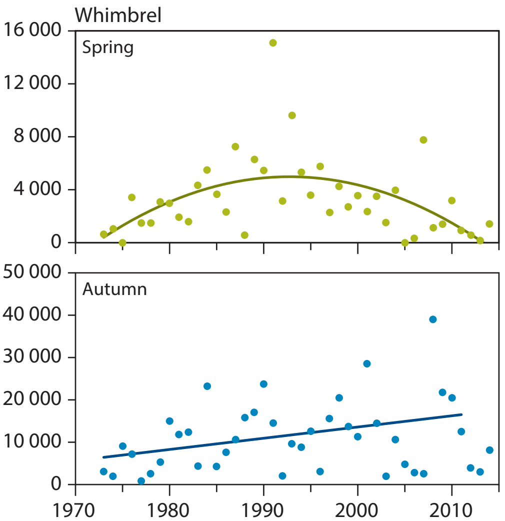


Fig. 28. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Whimbrel with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Småspove pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

In late summer, birds staging on the Reserve at daytime primarily feed on shortgrass meadows (Meltofte 1987, 1993). The fact that the meadows on Tipperne were not extensively grazed or mown between the mid-1950s and the mid-1970s may thus have contributed to the particularly low numbers in late summer in most years during this period. In spring, most birds stay on the mudflats (O. Amstrup *in litt.*).

The 'northeast' European breeding population of Whimbrels, to which the Danish migrants belong, is estimated to number 273 000-450 000 individuals after the breeding season (Nagy *et al.* 2015). The population was estimated by Delany *et al.* (2009) to be "possibly stable", which is in accordance with stable numbers of birds having passed Blåvandshuk south of Tipperne on autumn migration between 1964 and 2003 (Meltofte *et al.* 2006), and with the occurrence on Tipperne. This was changed to "stable/increasing" by Nagy *et al.* (2015) based on the most recent analysis of the midwinter counts 1988-2012 that shows substantially increasing numbers (Nagy *et al.* 2014). Still, breeding Whimbrels monitored in Norway, Sweden and Finland were overall stable during 2002-2013, though decreasing in Norway (Lindström *et al.* 2015). Numbers in the Wadden Sea have been decreasing since 1987, but the species has been poorly covered during counts in that area (Blew *et al.* 2015).

Conclusion: Relatively stable or even increasing numbers of Whimbrels have been recorded on Tipperne since 1929, but numbers recorded are highly sensitive to efforts by the observers to record immigration into the Reserve for night roosting. In spring, Whimbrels now arrive at least 11 days earlier than 80 years ago.

Eurasian Curlew *Numenius arquata*

Nowadays, maximum numbers of Curlews staging on the Tipperne Reserve are generally in the hundreds both during spring and autumn migration. Since 2000, a small breeding population has established itself on the Reserve, with 2-3 pairs breeding in recent years (O. Thorup *in litt.*), probably as an extension of the 3-7 pairs breeding on Værnengene just south of Tipperne (O. Amstrup *in litt.*), where the species has been known to breed since the 1960s (Ferdinand 1971).

During spring migration in March-April (Fig. 29), Curlews staging in Denmark in the early decades of the study period probably mainly fed on fields spread across the country, so that numbers on the Reserve were only moderate. During recent decades, spring numbers have been increasing quite markedly (Fig. 30). This is probably mainly as a result of birds remaining in northern areas

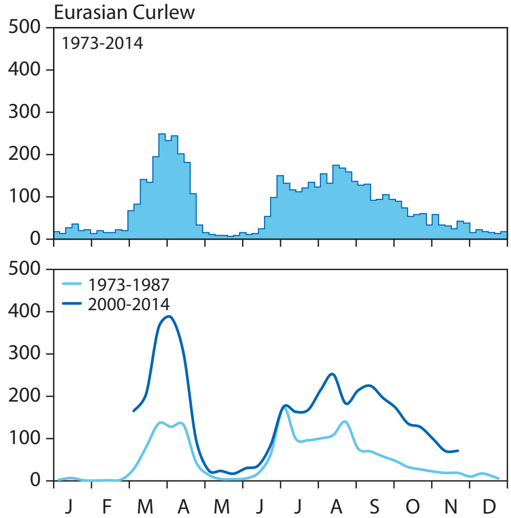


Fig. 29. The phenology of Eurasian Curlew on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Storspover på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

During the last 2-3 decades, the flyway population of Curlews has apparently been recovering after having been depleted by hunting during the 19th and 20th centuries. Photo: Erik Thomsen.

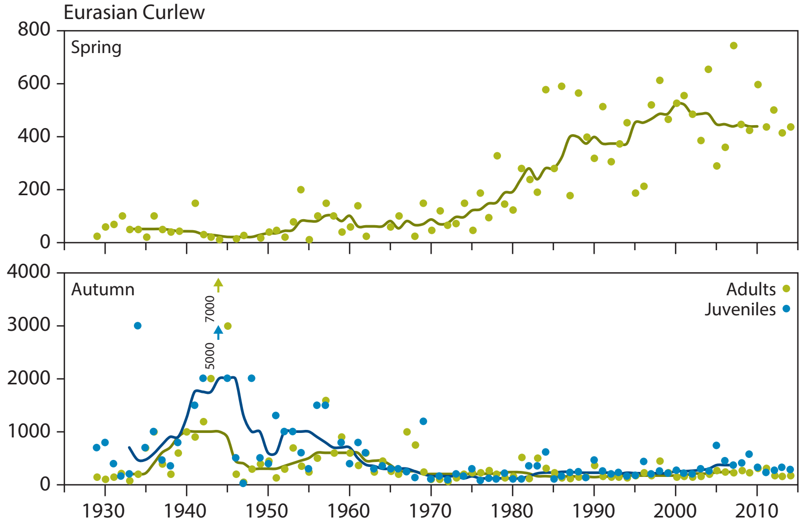


Fig. 30. Annual maximum numbers of Eurasian Curlew per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Storspove pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

during winter and staying into spring. This development has been made possible by milder winters and reduced shooting disturbance in autumn and early winter (see further below).

During autumn migration between June and November (Fig. 29), much higher numbers were recorded on the Reserve during the first decades of monitoring, when 1000 or more Curlews were recorded in 14 years between the 1930s and late 1950s (max. 7000 in 1944 and 5000 in 1943; Fig. 30). The reason for these high numbers was considered by Meltofte (1986, 1987) to be a combination of high densities of crane fly larvae and reduced hunting during World War II. Judging from the numbers of Common Starlings *Sturnus vulgaris* recorded on the Reserve, particularly high densities of crane fly larvae occurred on the meadows during the 1940s and partly also during the 1930s, levelling off during the 1950s (Meltofte 1987). Many birds fed on neighbouring areas and spent the night on the Reserve, a behaviour made possible by very limited hunting during the war – in spite of an open season beginning on 1 July.

Following this reasoning, the population of Curlews passing Denmark on autumn migration seems to have diminished during the second half of the 19th century and much of the 20th century because of hunting mortality along the flyway and also due to overflying Denmark because of intensive shooting disturbance (Meltofte 1986, 1993, Laursen 2005, Meltofte *et al.* 2009). From the late 1950s onwards, overgrown meadows on Tipperne together with drainage and conversion of adjacent meadows may also have contributed to the local decrease, but numbers did not increase again when management of the meadows was reintroduced on the reserve during the 1970s (Meltofte 1987). The negative flyway population trend was reversed when a number of protective measures were introduced from 1983 onwards, followed by full protection of the species in Denmark in 1994 (Laursen 2005). Numbers of individuals staging (and wintering) in Denmark outside of the Wadden Sea increased by an order of magnitude, while inside the Wadden Sea numbers increased by a factor of 2-3 (Meltofte *et al.* 2009, Clausen *et al.* 2013, Laursen & Frikke 2013; see also Taylor & Dodd 2013 for the marked effect of hunting on longevity in a Curlew population).

These changes are also reflected in the numbers staging on the Reserve, where Curlews have increased considerably in more recent decades both during spring and autumn migrations, so that maximum numbers for the last 15 years were generally more than twice as high as during 1973-1987 (Figs 29 and 30). This increase becomes even more pronounced in terms of bird-days spent on the Reserve (Fig. 31), implying that the birds stay on the Reserve (and moult?) longer into the autumn or even winter (Fig. 29). These rather extreme fluctuations in Curlew numbers on the Reserve since 1929 are accentuated by the much more modest changes in numbers of the much less hunted Whimbrel (Meltofte 1986, 1987, this study).

During recent decades, the Curlews on the Reserve have predominantly fed on the shallow flats, where they are less sensitive to water levels because of their long legs. Most likely, increasing densities of ragworms and other benthic invertebrates on the shallow flats up until the 1990s (Fig. 6) have contributed to the increase in bird numbers during recent decades.

In spite of the apparently declining flyway popula-

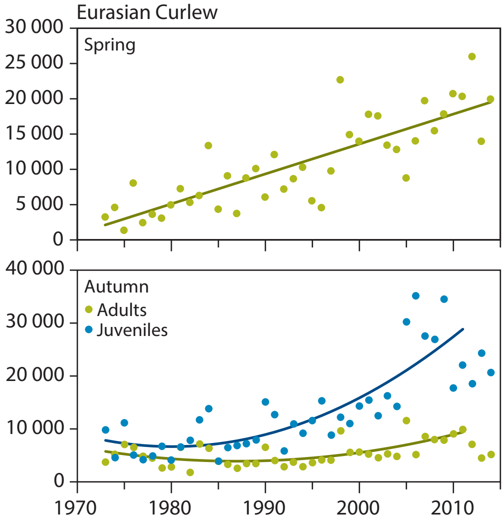


Fig. 31. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Eurasian Curlew with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Storspove pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

tion, the European *Numenius a. arquata* population probably increased in the northernmost part of the breeding range and spread northwards during the latter half of the 19th century and much of the 20th century (Cramp *et al.* 1983, Lappo *et al.* 2012). The population is now estimated to number 640 000-920 000 individuals in the non-breeding season and to be stable or decreasing (Nagy *et al.* 2015). This dubious decrease (see above) has placed the Curlew as Near Threatened on IUCN's Red List of Threatened Species since 2008 (Birdlife International 2015c), but this status is not supported by midwinter count data either from the Wadden Sea, where numbers have been stable since 1987 (Blew *et al.* 2015), or from West Europe in general, where the population has shown an upward trend since the 1980s (Nagy *et al.* 2014, van Roomen *et al.* 2015, AEWA no year). Numbers of birds passing Blåvandshuk south of Tipperne also increased between the 1970s and the 2000s (Meltofte *et al.* 2006), while extensive monitoring during 2002-2013 showed stable numbers in Norway and northern Sweden and significantly increasing numbers in Finland (Lindström *et al.* 2015). The discrepancy may be the result of the availability of far more breeding population data from the southern and western part of the breeding range, where the species has declined in several areas, while the bulk of the population in northern Europe may be stable or even increasing (Thorup 2006a, Meltofte *et al.* 2009, Lindström *et al.* 2015; see also Northern Lapwing).

The Danish breeding population has increased more or less continuously to 450-500 pairs since its recolonization of the country after a ban on spring hunting from 1931 (Nyegaard *et al.* 2014). In recent decades, the population has disappeared from many inland sites, while it is thriving on a number of well managed coastal sites.

Conclusion: The North European population of Curlews passing Tipperne and the rest of Denmark probably underwent significant declines induced by hunting from the middle of the 19th century onwards, followed by a partial recovery in recent decades as a result of improved protection. Thus, trends in numbers on the Reserve to a large extent reflect overall flyway population trends.

Bar-tailed Godwit *Limosa lapponica*

With maximum numbers of Bar-tailed Godwits in the thousands, Tipperne Reserve is – or used to be – one of the most important Danish spring staging areas for this species outside the Wadden Sea (Meltofte 1993). Numbers in autumn, when most Bar-tailed Godwits fly directly to the Wadden Sea, are much smaller both during the adult migration in July-August and the juvenile migration from late August onwards (Fig. 32).

During the first decades of monitoring on the Reserve, numbers of staging Bar-tailed Godwits in May

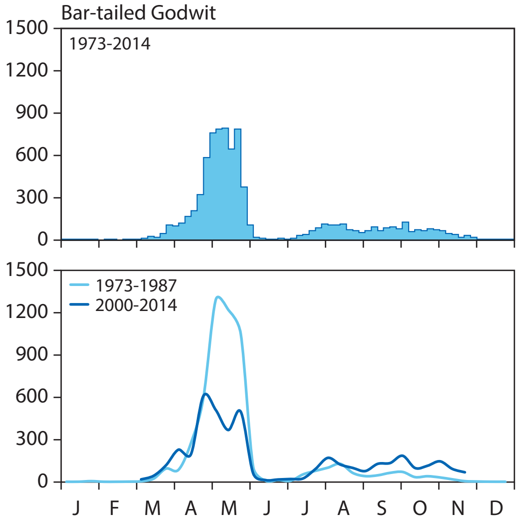


Fig. 32. The phenology of Bar-tailed Godwit on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Små Kobbersnepper på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

Numbers of both spring and autumn staging Bar-tailed Godwits on Tipperne have declined since 1970s and 1980s. Photo: Ole Amstrup.

generally peaked at around 2000-5000 individuals. In a number of years up to the late 1950s maximum numbers even reached 7000-10 000 (Fig. 33). Exceptions were springs following severe winters when the benthic fauna on the shallow flats had been exterminated by frost and ice (Meltofte 1987; see also Dunlin). Maximum numbers of around 4000-5000 continued to occur until 1969, and numbers of around 2000-3500 occurred even until the early 1980s, but since then numbers have decreased to around 1000 or even less in most years (Fig. 33).

Numbers of staging adult and juvenile Bar-tailed Godwits in autumn have also decreased from peak numbers generally of around 400-500 in most years up to the mid-1970s, to only half those numbers or even less since then (Fig. 33). Exceptions were numbers of up to 1000-5000 in the 1940s and early 1950s, and up to 400-600 in

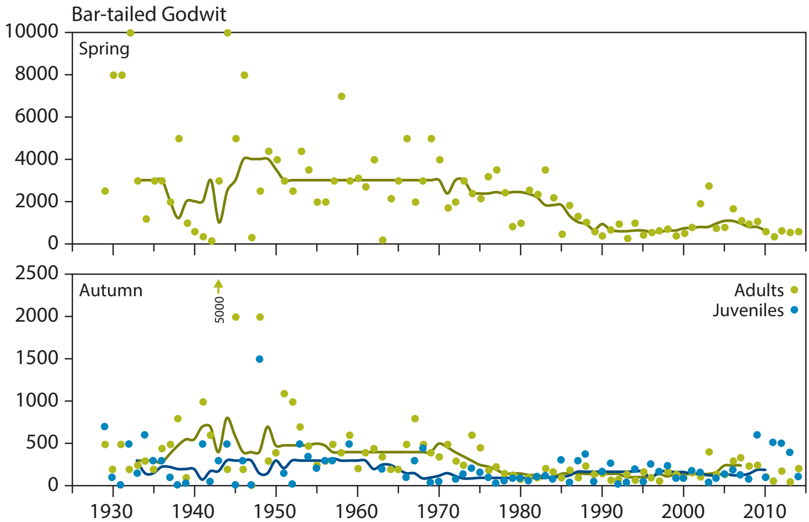


Fig. 33. Annual maximum numbers of Bar-tailed Godwit per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Lille Kobbersneppe pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

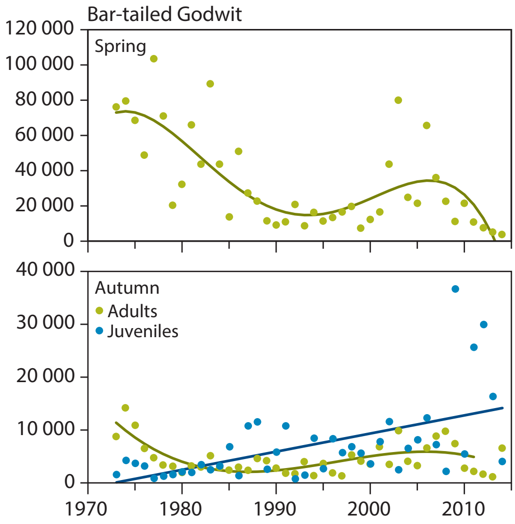


Fig. 34. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Bar-tailed Godwit with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Lille Kobbersneppe pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

the juvenile period in most years 2009-2013. This latter 'recovery' is even more pronounced when looking at the trends in bird-days (Fig. 34) which indicates that numbers at least in some years were relatively high both in spring and autumn during the 2000s and 2010s (Fig. 33).

Since records on Tipperne began in 1929, the first Bar-tailed Godwits have appeared progressively earlier in spring, so that the arrival of the 'pioneers' has advanced by no less than six weeks (Meltofte 1987, Petersen *et al.* 2012). Furthermore, the spring peak was almost entirely in late May in the first decade, progressively shifting towards early May and even late April since then (see Fig. 33 in Meltofte 1987 and Fig. 32 here). This is most likely the result of shifting populations dominating the numbers staging on the Reserve. Hence, in the first decade of the study period late arriving Siberian *taymyrensis* birds (having wintered in West Africa) predominated, while North European breeding *lapponica* progressively made up a larger and larger part of the spring staging Bar-tailed Godwits on Tipperne. These West European wintering birds now move north and east much earlier in spring than they did 80 years ago (see also Meltofte *et al.* 1994 and Laursen & Frikke 2013 for early arrival in the Danish and Schleswig-Holstein parts of the Wadden Sea). However, the migration to the (Siberian) breeding grounds also appears to have advanced, because numbers of birds remaining on Tipperne Reserve into June have decreased quite markedly since the 1930s (Meltofte 1987). Nowadays, significant numbers do not remain into June in the Wadden Sea either (Laursen *et al.* 2010).

The vast majority of the Bar-tailed Godwits feed in large flocks on the shallow flats, but flocks of a few hundred may also feed on the meadows in periods of high water in the fjord (Meltofte 1987). This was especially so during the early decades of the study period, when some of the very high numbers of birds in summer and autumn mentioned above fed on the newly mown meadows where they most likely took advantage of mass occurrence of crane fly larvae (Meltofte 1987).

The large changes in numbers of staging Bar-tailed Godwits on Tipperne may at least partly be explained by changes on the Reserve. In the 1930s, 1940s and 1950s, the water level in the fjord was lower during May-August (Fig. 2), so that the Bar-tailed Godwits had much larger areas of shallow flats to feed on – and also because Opgrøden did not cover large parts of the flats until the 1950s and 1960s (Meltofte 1987; see further below). Furthermore, small bivalves were present on the sandy flats in the first decades of the study period, and ragworms, which are a much desired food for Bar-tailed Godwits, were not utilized very much during March-April by Bar-tailed Godwits and other waders, simply because most did not arrive until May. During the more recent study period, water levels in late spring were particularly high during the late 1980s and early 1990s and dropped markedly up to around 2010 (Fig. 3). Similarly, water levels in July-early August were relatively low during most of the 2000s, when the adult 'recovery' took place, until water levels rose again in the very last years.

Not only has the water level in late spring and summer increased since the mid-1930s, also one third of the higher lying flats have been overgrown with reeds and bulrush which further reduced the available feeding ground. In contrast to this, the densities of ragworms have gone up. Ragworms are the staple food of the Godwits on the Reserve, and the birds 'harvest' ragworms to such an extent that the densities of large and much desired individuals have been significantly reduced (Petersen 1981, Meltofte 1987). Since Bar-tailed Godwits now utilize the food resources from as early as mid-March (Fig. 32), this in combination with the much reduced feeding area may mean that the availability of ragworms is already much reduced when the peak migration of Godwits occurs in May (Meltofte 1987).

The North European breeding and West European wintering *lapponica* flyway population is estimated to number 120 000 individuals (Nagy *et al.* 2015). Delany *et al.* (2009) stated it as stable, but this was changed to "stable/increasing" by Nagy *et al.* (2015). The latter is in accordance with the latest midwinter count data showing increase since the 1990s (Nagy *et al.* 2014, van Roomen *et* *al.* 2015). The birds passing Blåvandshuk south of Tipperne on active migration – probably primarily involving European breeders (Meltofte 1993) – also showed increasing numbers during 1964-2003 (Meltofte *et al.* 2006).

The North Siberian breeding *taymyrensis* population that winter in West Africa is estimated to number 500 000 individuals and to be decreasing (Delany *et al.* 2009, Nagy *et al.* 2015). Numbers of Bar-tailed Godwits have decreased in the Danish and Schleswig-Holstein parts of the Wadden Sea since the 1980s, have been stable in Niedersachsen, and have increased so much in the Dutch part that totals for the entire Wadden Sea are stable (Blew *et al.* 2015). These birds include both populations, but Siberian breeders predominate among the birds utilizing the Dutch and Niedersachsen parts of the Wadden Sea in May (Meltofte *et al.* 1994, Engelmoer 2008), and when analysed separately, *taymyrensis* numbers have also been stable in the Wadden Sea since monitoring began in 1987 (M. van Roomen *in litt.*). On the other hand, on the West African wintering grounds of the *taymyrensis* population, numbers have gone down since counts began in 1979, with the 'challenging' exception of a very high count in 2006 (van Roomen *et al.* 2015).

Over a longer time frame, the wintering area of Bar-tailed Godwits in West Europe has expanded north during the 20th century (Prater 1981: 111; MacLean *et al.* 2008). This lends further support to the notion above that these birds have shifted their winter and early spring distribution much in the same way as Grey Plovers (see above). These shifts are probably mainly the result of milder winters and earlier springs, but also reduced disturbance from shooting may be involved (Prater 1981, Tubbs 1996).

Conclusion: The changing numbers of Bar-tailed Godwits staging on Tipperne Reserve seem largely to be related to conditions on the Reserve, primarily water level during critical periods of the year, overgrowth of the flats with tall vegetation, and invertebrate densities on the flats together with food availability on the meadows. Furthermore, Bar-tailed Godwits now utilize the resources of the Reserve much earlier in spring than they did 80 years ago.

Black-tailed Godwit *Limosa limosa*

Tipperne Peninsula has been one of the most important breeding sites for Black-tailed Godwits in Denmark since the 1920s (Thorup 1998). On the Reserve alone, the population of this species fluctuated around 30-50 pairs up to the mid-1970s, after which an increase culminated in about 150-200 pairs during 1988-1995 (Fig. 35; Thorup 1998 and *in litt.*). At the same time, 25-90 pairs were found on Værnengene on the southern part of Tipperne Peninsula (O. Amstrup *in litt.*). Since then, numbers have declined to around 60-80 pairs on Tipperne and 10-30 pairs on Værnengene. These changes are related to varying management of the meadows of the Reserve and to the occurrence of avian and mammalian predators in the area (Thorup 1998).

Changes in numbers of individuals staging on the Reserve have been no less prominent than changes in numbers of breeders. Until the early 1990s, the Reserve functioned as a post-breeding staging and moulting area during June-August for adults as well as juveniles, including those from adjacent breeding sites (Fig. 36; Meltofte 1987). This means that up to about 200-700

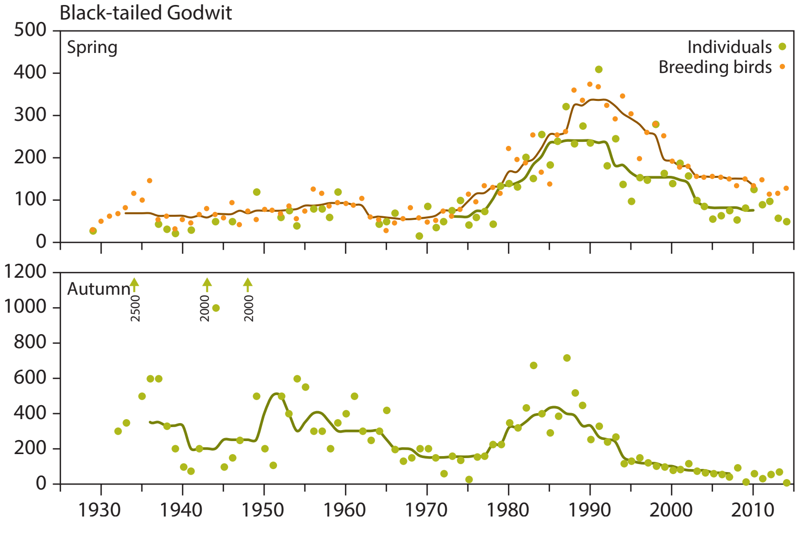


Fig. 35. Annual maximum numbers of Black-tailed Godwit per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Stor Kobbersneppe pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

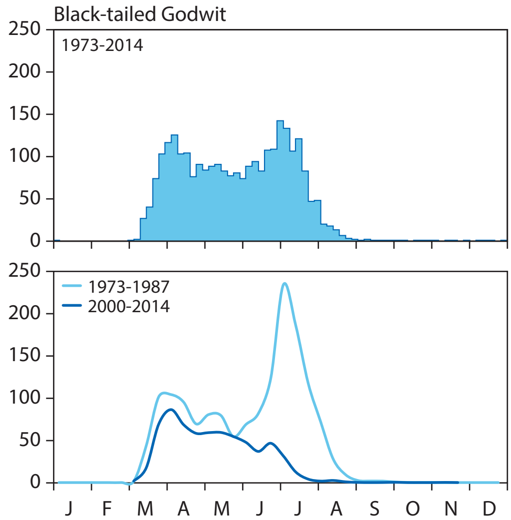


Fig. 36. The phenology of Black-tailed Godwit on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods in the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Store Kobbersnepper på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

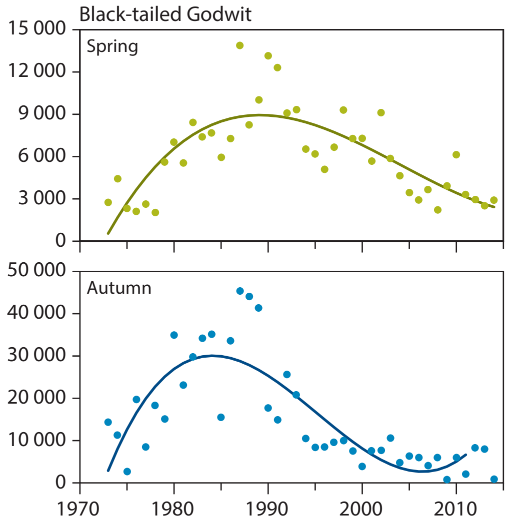


Fig. 37. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Black-tailed Godwit with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Stor Kobbersneppe pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

During the last two decades, Black-tailed Godwits have no longer used Tipperne as a post-breeding staging area. Photo: Jan Tandrup Petersen.

Black-tailed Godwits were recorded regularly on the Reserve up to around 1990, and record numbers of up to between 1000 and 2500 were registered a few times during the first two decades (Fig. 35; see Meltofte 1987 for reliability of these data and possible East European origin of the birds if trustworthy). The lower numbers from 1966 and the next 10 years or so may have been related to drainage and thereby destruction of the breeding area in the nearby river Skjern Å delta, followed by the increase in the breeding population of the Tipperne Peninsula a decade later (Meltofte 1987). Since the early 1990s, the Black-tailed Godwits have gone elsewhere to moult and build up body stores for the fall migration (Figs 35, 36 and 37). We have no explanation for this or any idea as to where they have gone.

Both prior to the breeding season and during post-breeding, Black-tailed Godwits primarily feed on the shallow flats of the Reserve, where flocks may number hundreds in June-August. During breeding in April-May, often only between half and two thirds of the population is recorded during counts (Fig. 36). Few breed north of Denmark, and the birds on Tipperne are probably almost entirely from local populations (Meltofte 1993).

In spring, Black-tailed Godwits now arrive on average two weeks earlier on Tipperne than they did around 1930 (Meltofte 1987, Petersen *et al.* 2012), whereas egg laying has not advanced significantly (Petersen 2010).

The West European flyway population of the *limosa* subspecies is estimated to number 86 000-141 000 individuals (Nagy *et al.* 2015). This population increased considerably during much of the 20th century, but has declined significantly during the last few decades, particularly on the very important Dutch breeding grounds (Thorup 2006a, Delany *et al.* 2009). Therefore, the species has been listed as Near Threatened by IUCN since 2006 (Birdlife International 2015d). In Denmark also, the Black-tailed Godwit has disappeared from many breeding sites, while it has increased on a number of key sites with targeted management, resulting in the population becoming stabilized around 500-600 pairs, which is not much below the peak numbers of a few decades ago (Nyegaard *et al.* 2014).

Conclusion: Numbers of Black-tailed Godwits occurring on Tipperne Reserve are closely related to the numbers of breeders in the nearby region and their breeding success. Up to around 1990 the Reserve functioned as a post-breeding staging and moulting area in June-August, when up to between 300 and 700 individuals were recorded regularly. Arrival at least of the pioneers now takes place two weeks earlier than 80 years ago.

Ruddy Turnstone *Arenaria interpres*

In most years the maximum numbers of Ruddy Turnstones on the Tipperne Reserve remained below 20-30 individuals both during spring migration in May and autumn migration in July – early September (Figs 38 and 39), and no significant changes have taken place during the study period (Figs 38, 39 and 40). Absolute peak numbers were around 60 in early May 1981 and mid May 1983 together with 110 in mid-August 1942.

In autumn, the Danish migrants are thought to originate from both the Greenland/NE Canadian and the Fennoscandian/NW Russian populations, while 'only' North European breeders are thought to pass Denmark in spring (Meltofte 1993). In winter, these populations

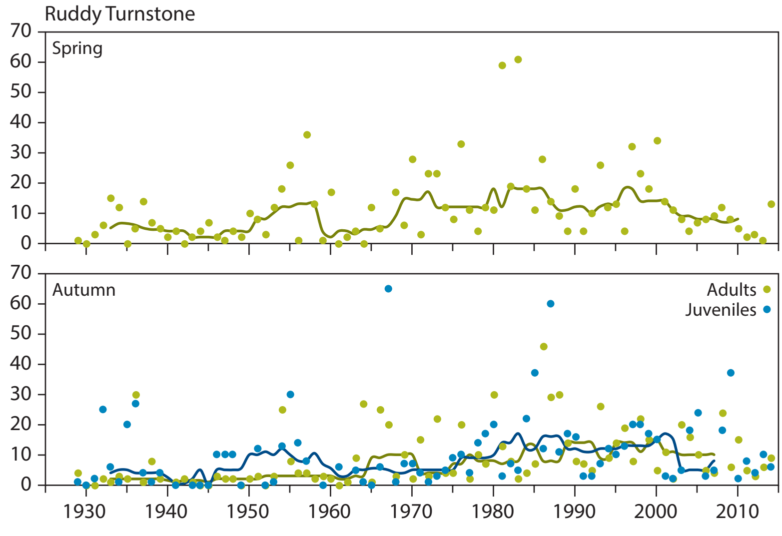


Fig. 38. Annual maximum numbers of Ruddy Turnstone per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Stenvender pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

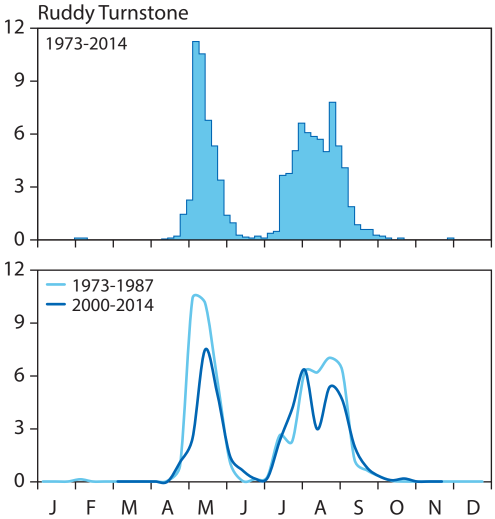


Fig. 39. The phenology in Ruddy Turnstone on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Stenvendere på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

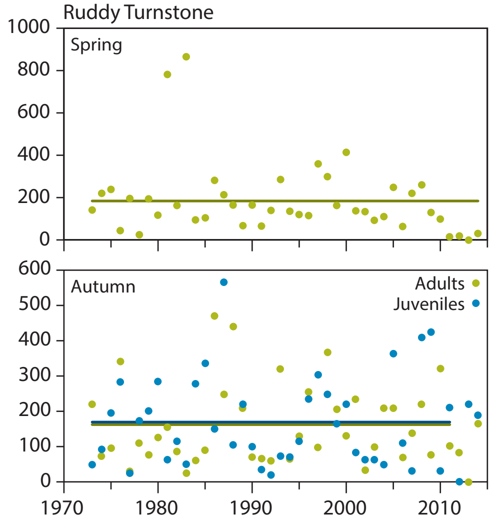


Fig. 40. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Ruddy Turnstone with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Stenvender pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

are estimated to number 100 000-200 000 in West Europe and 62 700-111 000 individuals in West Africa, respectively (Nagy *et al.* 2015). According to Delany *et al.* (2009), both populations have been "possibly decreasing" after an increase up to the mid-1990s in the European wintering population. In the most recent status from

Small but stable numbers of Ruddy Turnstones occur on Tipperne. Photo: Erik Thomsen.

Wetlands International (Nagy *et al.* 2015) this is changed to increasing in the Greenland/NE Canadian population. Indeed, according to the latest analyses of the midwinter counts on the East Atlantic flyway in 1988-2012 and 1993-2014, respectively (Nagy *et al.* 2014, van Roomen *et al.* 2015), the West European wintering population has increased quite markedly since 2003. Numbers in the Wadden Sea have fluctuated since the late 1980s (Blew *et al.* 2015).

Among Ruddy Turnstones passing Blåvandshuk south of Tipperne on active migration, numbers were stable during 1964-2003. However, when adjusted for effects of shifting wind direction, numbers increased up to the 1980s, after which they declined (Meltofte *et al.* 2006). This is in accordance with the status for the African wintering population given as stable or fluctuating by Nagy *et al.* (2015). This is again based on the latest analyses of the midwinter counts on the East Atlantic flyway (Nagy *et al.* 2014, van Roomen *et al.* 2015), according to which the West African wintering population has decreased since 1998 – though with a 'disturbingly' high count in 2009.

The Danish breeding population is stable at around 50 pairs found on only three sites (Nyegaard *et al.* 2014). However, Ruddy Turnstones may have attempted to breed on Tipperne a few times (Thorup 1998).

Conclusion: Low but stable numbers of Ruddy Turnstones were recorded on Tipperne Reserve during the study period. In spring 'only' North European breeders occur, while in autumn both North European and Nearctic birds are present. The Nearctic population has increased to a higher level during the last 10 years, while the North European population is probably fluctuating.

Red Knot *Calidris canutus*

In most years, Red Knots have occurred on Tipperne Reserve in numbers well below 100 in spring and 200 during autumn migration, while in some years several hundred or even thousands have been recorded

In 1985, large numbers of Red Knots were found on Tipperne, followed by highly elevated numbers the following year. Photo: Jan Tandrup Petersen.

(Fig. 41). The most extreme records were in 1985, when numbers built up from mid-August to a maximum of 7430 predominantly juveniles (98%) on 29 August, after which numbers dropped and most birds had left by 4 September. These high numbers were made possible by extremely low water levels persisting in the fjord, exposing large flats. The following year, a maximum of 258 adults was recorded in mid-May and 1920 adults in late July. This was the highest May number until then (though later, 310 were recorded in May 2007) and the highest adult number on autumn migration. Another special occurrence was of immature birds in 2003-2006, when annual maximum numbers of 186-682 were recorded around mid-June (Figs 41 and 42). Besides these 'outliers', numbers on the Reserve have remained rather stable during the entire study period (Figs 41 and 43).

Two Red Knot populations pass Denmark during spring and autumn migration: the North Siberian breeding *C. c. canutus* that numbers around 250 000 individuals and primarily winters in West Africa, and the likewise high Arctic Greenland/NE Canadian breeding *C. c. islandica* that numbers between 500 000 and 565 000 individuals and primarily spends the winter in north-western Europe (Meltofte 1993, Nagy *et al.* 2015). There is much uncertainty about the trends in population size in these two sub-species, with the best known *C. c. islandica* stated as "decreasing" (1980s-2007) by Delany *et al.* (2009), having long-term "increase" (1983-2007) by AEWA (no year), and as "stable/decreasing?" (2003-2014) by Nagy *et al.* (2015). It should be noted, however, that these varying statements followed a serious reduction in this population between the early 1970s and the early 1980s (Delany *et al.* 2009).

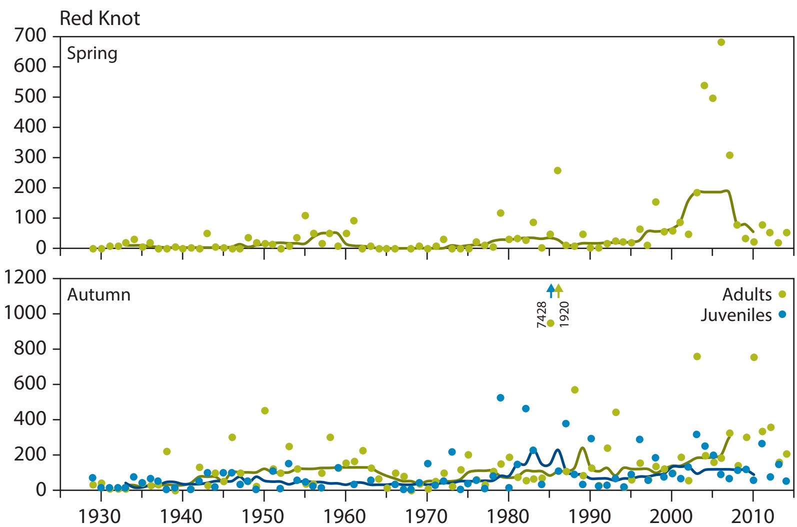


Fig. 41. Annual maximum numbers of Red Knot per season (cf. Tab. 1) with nine-year running medians.
Årlige maksimumtal for Islandsk Ryle pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

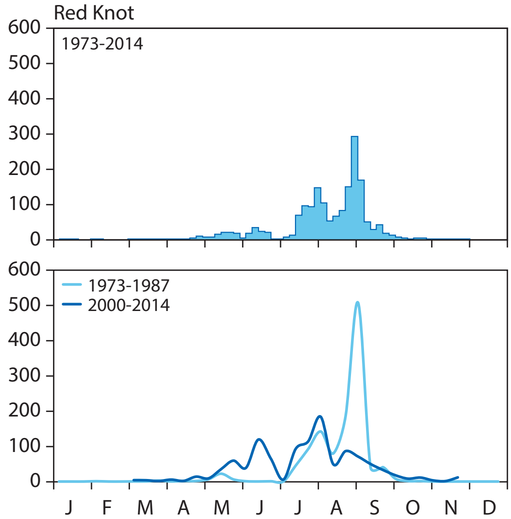


Fig. 42. The phenology of Red Knot on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.
Fænologien i forekomsterne af Islandske Ryler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

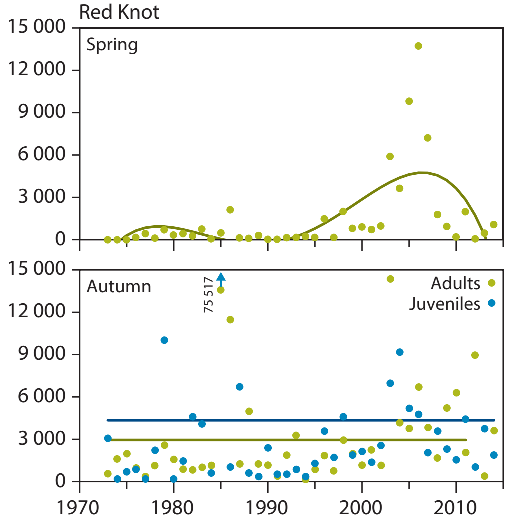


Fig. 43. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Red Knot with polynomial regression lines (see text).
Årlige antal akkumulerede fugledage for Islandsk Ryle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Due to highly varying numbers recorded on the West African wintering grounds, trends in *C. c. canutus* are even more uncertain but are given as "possibly decreasing" by Delany *et al.* (2009) and "decreasing" by Nagy *et al.* (2015) and van Roomen *et al.* (2015). This is based on census data from West Africa, where only half the numbers present around 1980 were recorded in 2014.

The large numbers of Red Knots from both populations passing Blåvandshuk south of Tipperne were stable during 1964-2003 (Meltofte *et al.* 2006), while numbers in the Wadden Sea fluctuated with a downward tendency during 1987-2011 (Blew *et al.* 2015).

Conclusion: Numbers of Red Knots staging on Tipperne Reserve have fluctuated widely without clear trends. This is more or less in accordance with the fluctuating trends in the two populations involved.

Ruff *Calidris pugnax*

During the period around 1980-2000 in particular, hundreds of Ruff could be present on Tipperne Reserve – especially in spring (Figs 44 and 45). During peak occurrence in the late 1980s and early 1990s, up to between 1000 and 2000 were recorded several times in spring, with 1950 birds in 1989 as the highest count (note that several of these peak counts were recorded during supplementary surveys and are not shown in the graphs). This is the same year as an all-time record count of 23 000 was recorded in Tøndermarsken just north of the Danish-German border at the Wadden Sea (Laursen & Frikke 2013). During the adult as well as the juvenile autumn migration, peak numbers of between 200 and 600 were recorded regularly on Tipperne during the same period (Fig. 44). Before and after this time period, more than 100 Ruff were recorded much less regularly.

A large proportion of the Ruffs in spring belonged to the local breeding population. This population numbered around 25-60 breeding females during the first several decades of the study period and reached a peak of more than 300 breeding females in the late 1980s and early 1990s, plus more than 50 on Værnengene on the southern part of the peninsula (Thorup 1998 and *in litt.*). Since then, numbers have declined to most often between 20 and 50 breeding females on the Reserve and a few on Værnengene (O. Amstrup and O. Thorup *in litt.*), which is still by far the biggest breeding population in Denmark (Nyegaard *et al.* 2014).

Besides breeders, large numbers of staging migrants during both the spring (May) and autumn (July-September) migration began to appear more regularly during the 1960s and 1970s (Fig. 46). This was apparently partly related to the increasing breeding population and partly to habitat management that from the 1970s re-established wet areas with low vegetation on the meadows in spring (Figs 44 and 46; Meltofte 1987). However, on other sites also in western and north-western Jutland, numbers of spring staging Ruffs appear to have increased up to around 1990 (Meltofte 1993). During autumn migration, when Ruffs switch to feeding to a larger extent on Tipperne's shallow flats, increased sedimentation and eutrophication from the mid-1960s onwards may have benefitted this species (Meltofte 1987).

The time of arrival of Ruffs on the Reserve has changed no less dramatically in that the pioneers now appear six weeks earlier than they did at the beginning of the study period (Petersen *et al.* 2012). This change can also be seen for the entire first part of the spring migration in March-April, which had already advanced by 2-4 weeks as early as in the 1980s (Meltofte 1987), whereas egg laying has not advanced significantly (Petersen 2010).

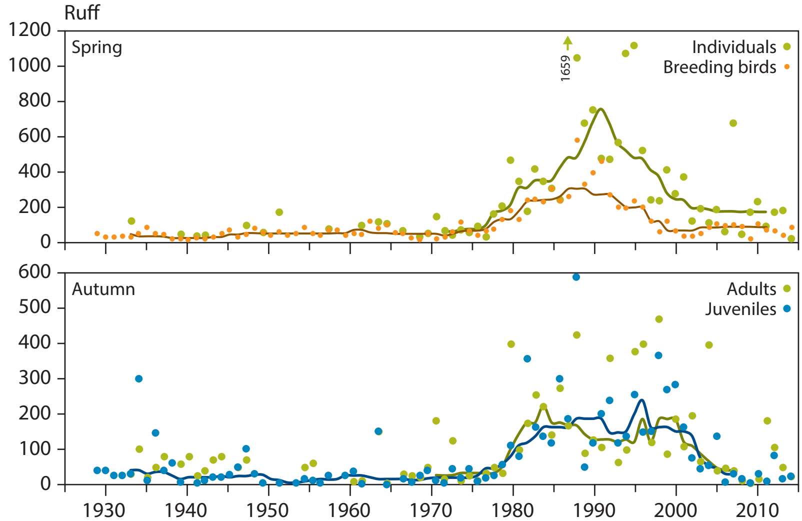


Fig. 44. Annual maximum numbers of Ruff per season (cf. Tab. 1) with nine-year running medians. Breeding birds given here are breeding females multiplied by two, but without taking into account the many non-successful females of the boom years of the 1980s and 1990s.

Årlige maksimumtal for Brushane pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

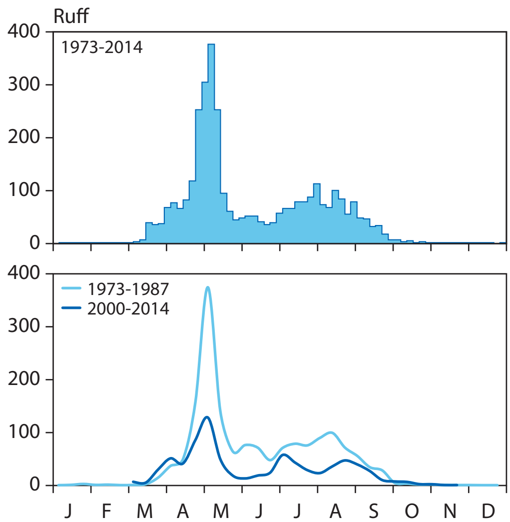


Fig. 45. The phenology of Ruff on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Brushøns på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

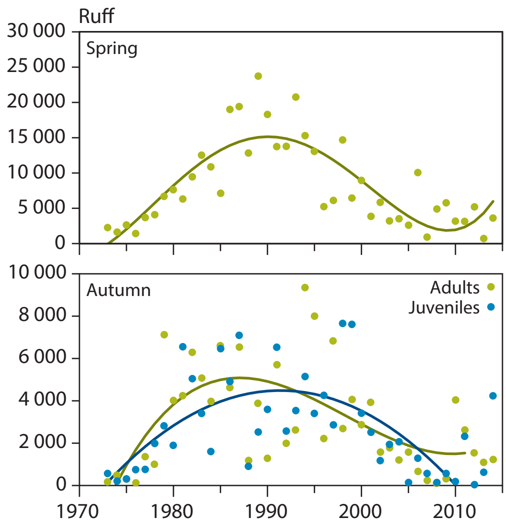


Fig. 46. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Ruff with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Brushane pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

The Ruffs passing Denmark on migration belong to the West African wintering population, which is crudely estimated to number between one million and 1.5 million individuals in winter (Nagy *et al.* 2015). The breeding populations in the southern parts of the range of these birds have shown a marked decrease for decades, while northern populations have expanded and increased during the 20th century (Cramp *et al.* 1983, Thorup 2006a). There is some uncertainty about more recent trends in numbers of this population because heavy decreases on the western fringe of the flyway may be the result of a massive redistribution of the spring migration towards the east, probably as a consequence of agricultural intensification in north-west Europe (Verkuil *et al*. 2012). Delany *et al.* (2009) stated the population to be decreasing, while this status was added a question mark by Nagy *et al.* (2015). A marked decrease is certainly true for the West European breeders (Thorup 2006a, Lindström *et al.* 2015), but within the entire Russian Arctic only one population at the White Sea is known to have decreased (Lappo *et al.* 2012). Indeed, the strong decline in numbers of both breeding and spring staging Ruffs on Tipperne Reserve is related to displacement of the spring migration 'corridor' from north-western towards eastern Europe and the related reduction in West European breeding populations.

Conclusion: After a 'boom' in the breeding population of Ruffs on Tipperne Reserve around 1990, the population of breeding as well as staging Ruffs declined strongly in parallel with a redistribution of spring staging Ruffs from north-western to eastern Europe in recent decades. This is probably a result of agricultural intensification in West Europe. Arrival of the pioneers in spring has advanced by six weeks during the study period.

Curlew Sandpiper *Calidris ferruginea*

Like in Denmark in general, numbers of Curlew Sandpipers on Tipperne Reserve increased quite markedly during the study period (Figs 47, 48 and 49; Thorup 2006b). In spring, the species was rare on the Reserve prior to the 1980s when an increase set in, culminating in maximum numbers of about 50 or more during the late 1990s and most of the 2000s (Figs 47, 48 and 49). Similarly, during the adult autumn migration, since the 1990s maximum numbers increased from generally below 25 (max. 60) to a number of records of between 300 and 450 (Figs 47, 48 and 49). Juveniles have most often occurred at maximum numbers of below 100, but with several records of up to between 100 and 525 in recent decades (Fig. 47). High juvenile numbers were often also recorded in years together with many Little Stints *Calidris minuta* and are related to breeding success in the Arctic (Meltofte 1993). The increase in adults on the Reserve during the last half century may at least partly be related to increased sedimentation and eutrophica-

Numbers of Curlew Sandpipers staging on Tipperne increased markedly during the study period. Photo: Ole Amstrup.

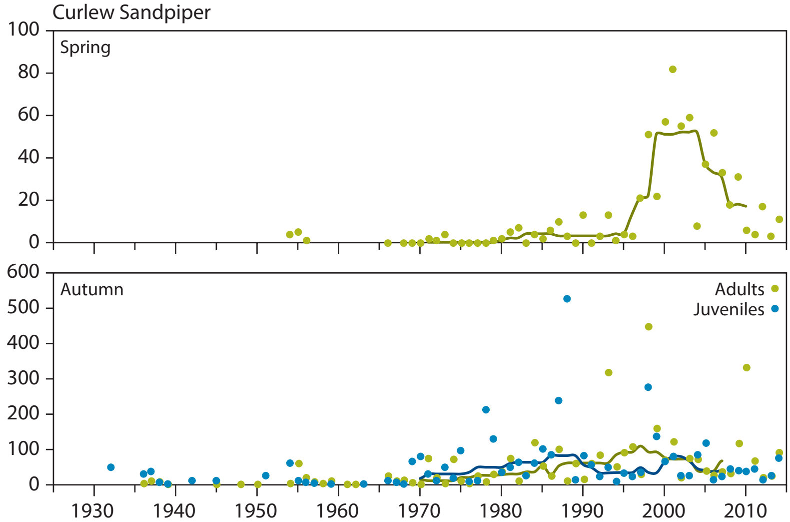


Fig. 47. Annual maximum numbers of Curlew Sandpiper per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Krumnæbbet Ryle pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

tion since the canalization of the river Skjern Å delta in the mid-1960s (Meltofte 1987). It remains to be seen, however, whether the lower numbers in spring during recent years are indicative of a decline or are the result of decreasing counting frequency (Figs 47 and 49).

The population of Curlew Sandpipers wintering in West Africa is estimated to number between 350 000 and 450 000 individuals (Nagy *et al.* 2015), a reduction from about one million given by Delany *et al.* (2009). The population was further estimated by Delany *et al.* to have been in "strong increase" since the 1980s. This was confirmed by 40 years (1964-2003) of monitoring of birds on active migration past Blåvandshuk south of Tipperne, where numbers peaked in the late 1980s and

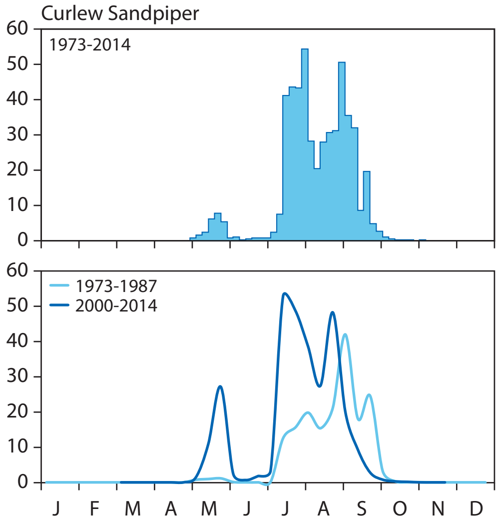


Fig. 48. The phenology of Curlew Sandpiper on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Krumnæbbede Ryler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

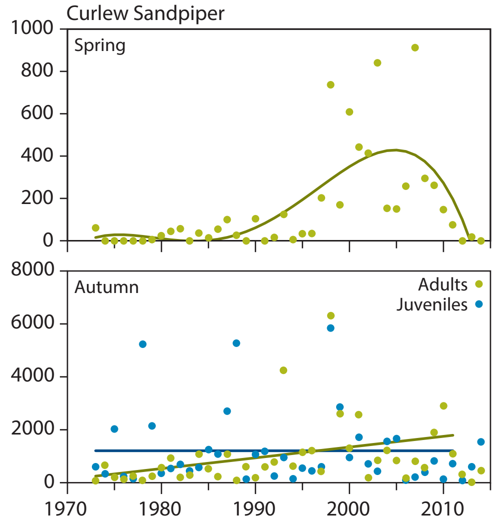


Fig. 49. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Curlew Sandpiper with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Krumnæbbet Ryle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

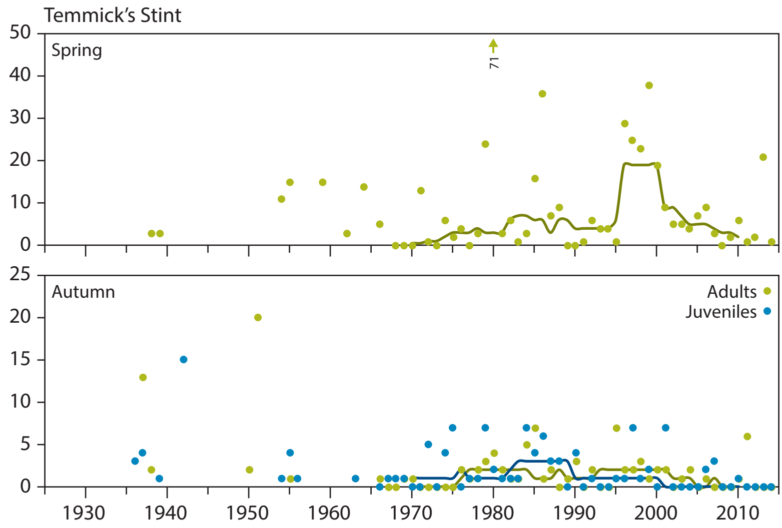


Fig. 50. Annual maximum numbers of Temminck's Stint per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Temmincksryle pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

early 1990s (Meltofte *et al.* 2006). According to the flyway counts, numbers peaked somewhat later followed by lower numbers during the last decade (Nagy *et al.* 2014, van Roomen *et al.* 2015). The lower numbers in recent years have led to a new status as "decreasing" given by Nagy *et al.* (2015). According to the Wadden Sea monitoring, however, numbers of Curlew Sandpipers have been increasing during the entire period since the 1980s (Blew *et al.* 2015).

Conclusion: Curlew Sandpipers on the east Atlantic flyway have shown a quite marked long-term increase, as they have on Tipperne Reserve. In the Reserve, adults were scarce before the 1960s, after which 'muddifying' of the shallow flats may have contributed to an increase there over and above the general flyway population increase. However, during the last decade or so, the flyway population may have declined.

Temminck's Stint *Calidris temminckii*

While maximum numbers of Temminck's Stints in spring usually remained below 10, up to between around 20 to 70 were recorded in some years (Fig. 50). In autumn, numbers were much smaller (Fig. 51) with 0-2 as the most common annual maximum and 20 as the top record. Numbers have fluctuated in both spring and autumn throughout the study period, possibly with less regular records of juveniles in autumn during recent years (Figs 50 and 52). This may be the result of less regular counts, however.

The Fennoscandian post-breeding population is estimated to number between 24 000 and 50 000 individuals and to be stable (Nagy *et al.* 2009). The population apparently decreased during the 20th century, and Delany *et al.* (2009) considered it still to be "possibly decreasing." In the White Sea region the populations may also be declining (Lappo *et al.* 2012), and in the Bothnian Bay the population is endangered by low adult survival (Koivula *et al.* 2008).

Conclusion: No clear trends are discernible from the limited numbers of Temminck's Stints that occurred on Tipperne Reserve.

Many more Temminck's Stints occur on Tipperne in spring than in autumn. Photo: Jan Tandrup Petersen.

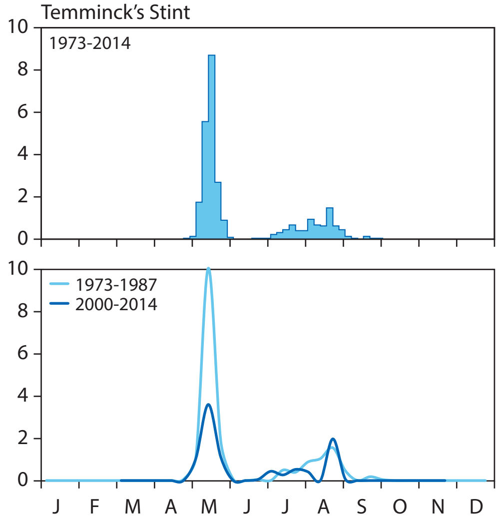


Fig. 51. The phenology of Temminck's Stints on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Temminckstyler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

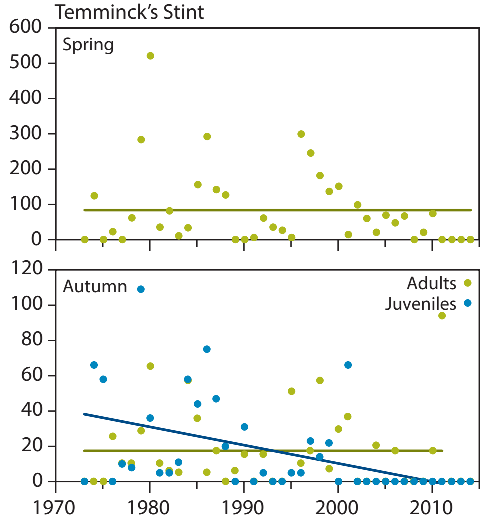


Fig. 52. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Temminck's Stint with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Temminckstyle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Dunlin *Calidris alpina*

The Dunlin is the most numerous wader on Tipperne Reserve and maximum numbers regularly reach 5000-10 000 or more individuals in both spring and autumn (Figs 53 and 54). Numbers have fluctuated over the entire study period without any clear long-term trends. Although some of the recorded figures from the early decades probably must be treated with some caution, numbers were particularly high in the 1930s and most of the 1940s, in the late 1960s and early 1970s (in spring), and again in the late 1980s and throughout the 1990s and early 2000s (Figs 53 and 55). However, the lower numbers since the mid-2000s may be related to reduced counting frequency from 2002 onwards together with higher water levels during the adult autumn migra-

Thousands of Dunlins and other *Calidris* sandpipers occur on Tipperne in both spring and autumn. Drawing by Jens Gregersen.

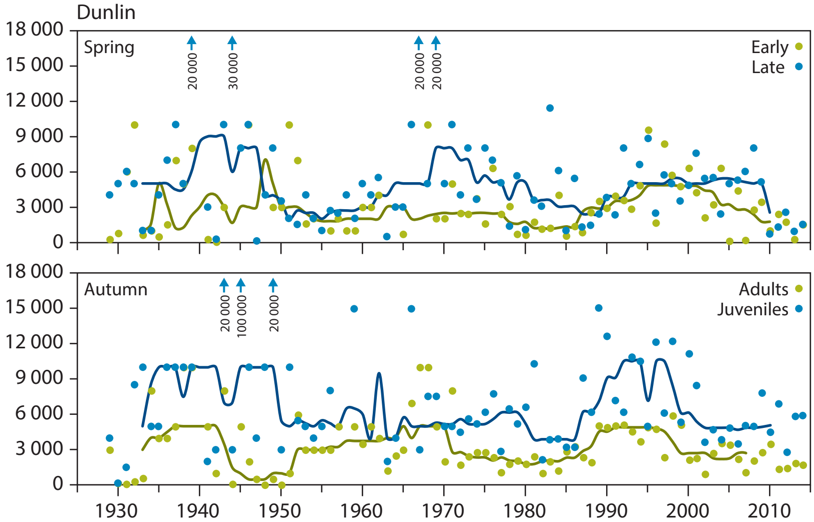


Fig. 53. Annual maximum numbers of Dunlin per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Almindelig Ryle pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

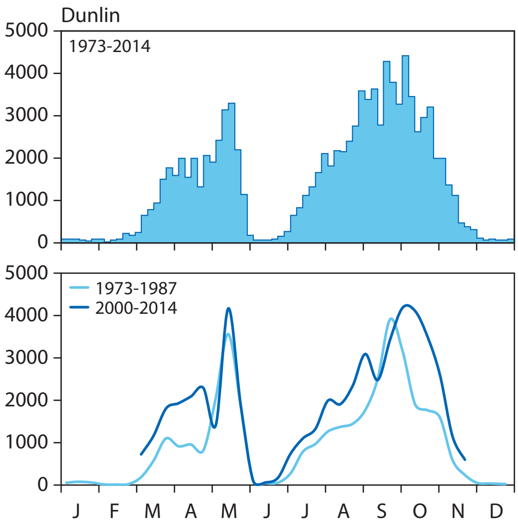


Fig. 54. The phenology of Dunlin on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Almindelige Ryler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

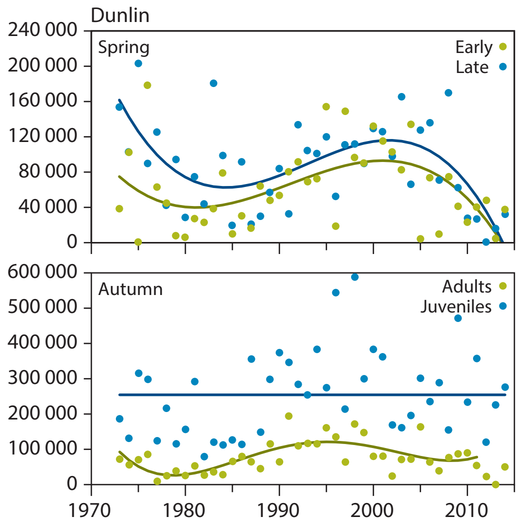


Fig. 55. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Dunlin with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Almindelig Ryle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

tion in the last four years (Fig. 3). Similar or even lower numbers of adults on autumn migration were recorded during the late 1940s, 1970s and 1980s, and similarly on early spring migration from 1979 and throughout the 1980s (Fig. 53). As a result, the phenological graph from the last 15 years shows higher numbers in both early spring and throughout most of the autumn migration compared to the period 1973-1987 (Fig. 54).

The two spring peaks of migrant Dunlins on Tipperne Reserve (Fig. 54) and elsewhere in Denmark are most likely the result of two waves of immigration of birds that have wintered along the West European seaboard (Meltofte 1987, 1993). The two waves may consist of different subpopulations with European sub-Arctic breeders dominating the first wave and Siberian breeders the second (Goede *et al.* 1990, Meltofte 1993).

Numbers of Dunlins on the Reserve are highly dependent on the water levels on the shallow flats (Meltofte 1987), but if no immigration takes place, low water level does not result in high numbers of Dunlins and other waders. Furthermore, under prolonged exposure of the flats invertebrate prey will burrow deeper and become less accessible to waders. This is the likely reason why there is no statistically significant correlation between water levels and numbers of Dunlins. Neither increased sedimentation nor eutrophication since the channelization of the river Skjern Å delta in the mid-1960s seem to have had any significant positive effect on Dunlin numbers (Meltofte 1987). Maybe these changes were counterbalanced by extensive overgrowth of the highest parts of the flats on the eastern part of the Reserve by reeds and other tall vegetation since the 1950s? Furthermore, numbers of staging Dunlins on the Reserve are often markedly reduced in the springs after severe winters with solid ice cover that kills the invertebrates on the flats (Meltofte 1987, Desholm 2000; see also Bar-tailed Godwit). This factor may have played a role in the low numbers recorded in springs between 1978/79 and 1986/87 and again in 1996, 2010 and 2011, when winters were cold and the flats covered in ice.

Besides the large numbers of staging migrants of the subspecies *C. a. alpina*, the Reserve is one of Denmark's most important breeding areas for *C. a. schinzii* Dunlins (Nyegaard *et al.* 2014). The population peaked at 130-145 pairs around 1990, but has fallen to about 20-25 pairs since then. This is, however, still more than during most of the period before 1980 (Thorup 1998 and *in litt.*). A further 50-60 pairs bred on the southern part of the peninsula in the late 1980s, declining to about 10 pairs during recent years (O. Amstrup *in litt.*).

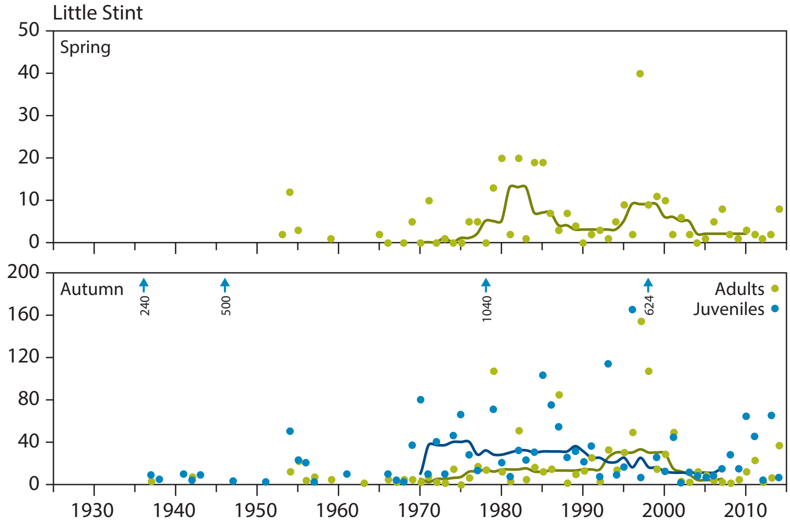
The *alpina* subspecies breeding in the north-western Palearctic and wintering in western Europe is estimated to number 1.33 million individuals in winter and to be decreasing (Nagy *et al.* 2015). According to the results of 40 years of monitoring of the active migration between 1964 and 2003 at Blåvandshuk south of Tipperne (Meltofte *et al.* 2006) and with numbers in the Danish reserve network 1994-2010 (Clausen *et al.* 2013), however, numbers are stable. This interpretation is further supported by breeding bird monitoring results from Norway, Finland and northern Sweden 2002-2013 (Lindström *et al.* 2015). However, a "slight decrease" has been found in the flyway population since 2003 (van Roomen *et al.* 2015) and a "moderate decrease" was recorded particularly in the northern part of the Wadden Sea during 1987-2012, whereas an increase was found in the Netherlands (Laursen & Frikke 2013, Blew *et al.* 2015).

Conclusion: No long-term trends were seen among the large numbers of Dunlin staging on Tipperne Reserve during spring and autumn. This agrees with the predominantly stable numbers recorded on the East Atlantic flyway during autumn and winter.

Little Stint *Calidris minuta*

The occurrence of Little Stints particularly in autumn on the Tipperne Reserve was quite erratic. In most years, maximum numbers were below 20 in spring and 120 in autumn, but more than 1000 have been recorded during the juvenile migration including 1200 on the entire peninsula in 1978 (Figs 56 and 57). Occurrence on the

Fig. 56. Annual maximum numbers of Little Stint per season (cf. Tab. 1) with nine-year running medians.
Årlige maksimumtal for Dværgryle pr. sæson (jvf. Tab. 1) med niårs glidende medianer.



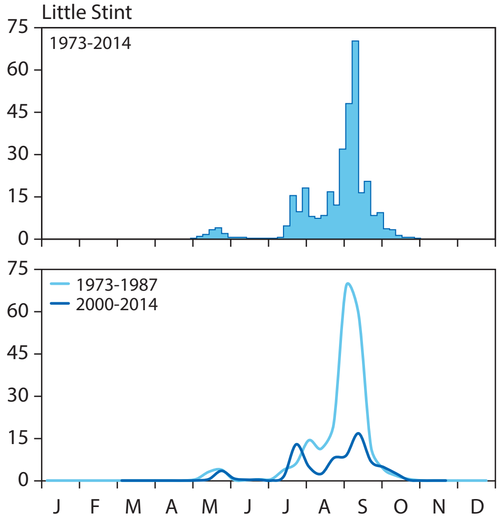


Fig. 57. The phenology of Little Stint on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Dværgryler på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

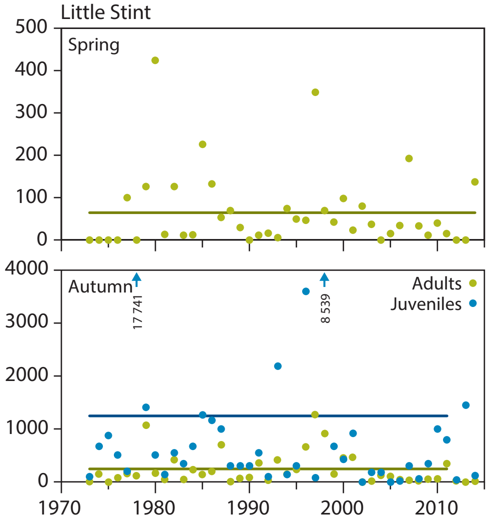


Fig. 58. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Little Stint with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Dværgryle pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Reserve increased following the increased sedimentation and eutrophication due to canalization of the river Skjern Å delta in the mid-1960s (Fig. 56; Meltofte 1987). Besides a period with low autumn numbers between 2002 and 2006, no clear short-term trends are seen (Figs 56 and 58).

The population of Little Stints frequenting the east Atlantic flyway is estimated to number about 300 000 individuals (Nagy *et al.* 2015). The population was given as "possibly stable" by Delany *et al.* (2009), but this was changed to possibly increasing by Wetlands International (2012), whereas Nagy *et al.* (2015) reversed it to decreasing. Long-term records of birds on active migration at Blåvandshuk south of Tipperne over 1964-2003 indicate increasing numbers, but this may be the result of better optics becoming available to the observers during the study period (Meltofte *et al.* 2006). While van Roomen *et al.* (2015) states the flyway population to be in marked decline – but expresses concern about the validity of that status – Nagy *et al.* (2014) show highly fluctuating numbers with no clear trend based on midwinter censuses on the east Atlantic flyway since 1991.

Conclusion: Numbers of Little Stints on Tipperne Reserve have been higher over the last 50 years than during the first decades of the study period because of increased muddiness of the shallow flats. The flyway population appears to be highly fluctuating with no clear trend.

Common Snipe *Gallinago gallinago*

In most years between 1970 and 1991 about 15-20 pairs of Common Snipes bred on Tipperne Reserve. Very few bred before then (and for the first time in 1944), and only 5-10 or even fewer pairs have bred since then (Fig. 59; Thorup 1998 and *in litt.*). Exceptions were about 35 pairs in 1986 and 15 in 2003. Numbers of snipes staging on the Reserve followed this trend to a large extent. Spring maximum numbers were around or exceeding 100 between the 1970s and 2003 and autumn maximum numbers were several hundred during much the same period (Fig. 59). A further 8-10 pairs breed on Værnengene on the southern part of the Tipperne Peninsula (Nielsen 1997).

Due to the cryptic behaviour of the species, numbers of Common Snipes recorded must be considered minimum numbers present on the Reserve. Even so, autumn maximum numbers of less than 100 were recorded up to 1965, which was the year when the delta of the main river emptying into the fjord was drained and sediments and nutrients loads reaching the fjord increased considerably. At the same time, vegetation on the Reserve increased from very low grazed meadow to much more varied patterns including common reeds and bulrush. From that time on, numbers of snipes recorded on the Reserve increased many fold to reach autumn peak numbers of almost 1700 recorded both in 1988 and

Maximum numbers of Common Snipes staging on Tipperne increased to often more than 1000 after reclamation in the mid-1960s of the delta of the main river flowing into the fjord, leading to considerable increase in sediments and nutrient loads reaching the fjord. Photo: Erik Thomsen.

1990 (Fig. 59; see also Meltofte 1987). When incomplete counts are included, a maximum of even 2225 was recorded in 1988.

Since the 1990s, both maximum numbers and bird-days have declined, so that maximum numbers have not exceeded 100 in spring since 2003 and 430 in autumn since 2001 (Fig. 59). Generally, numbers were halved between the periods 1973-1987 and 2000-2014 (Fig. 60), whereas numbers of bird-days declined by more than a factor five between the late 1980s and the late 2010s (Fig. 61). The reason for the decline is unknown, but summer cutting of extensive wet areas with reeds etc.

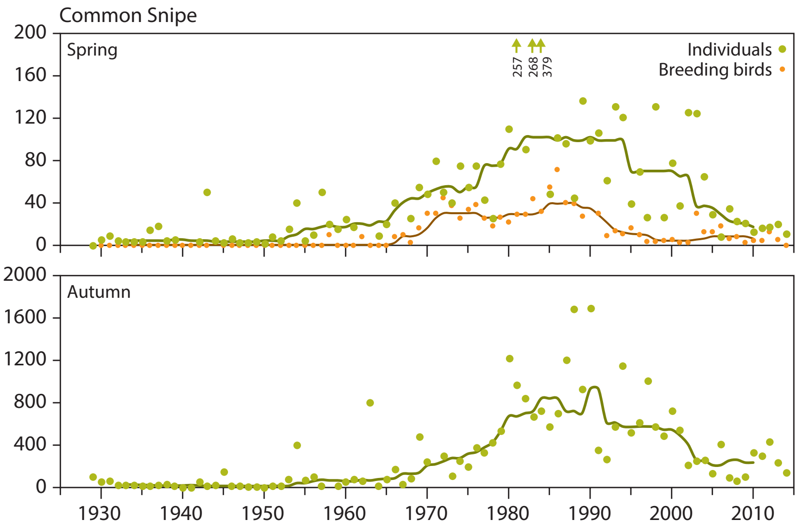


Fig. 59. Annual maximum numbers of Common Snipe per season (cf. Tab. 1) with nine-year running medians.

Årlige maksimumtal for Dobbeltbekkasin pr. sæson (jvf. Tab. 1) med niårs glidende medianer.

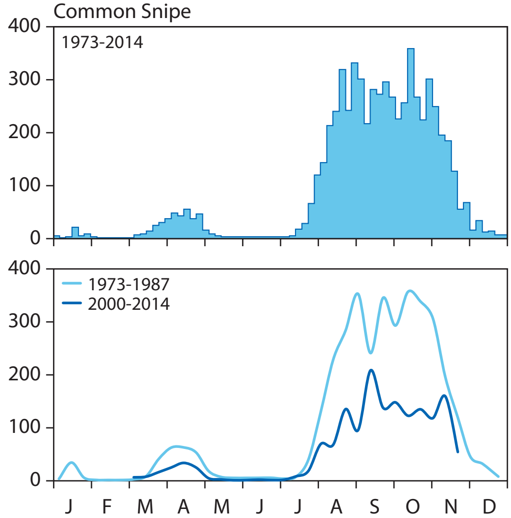


Fig. 60. The phenology of Common Snipe on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Dobbeltbekkasiner på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

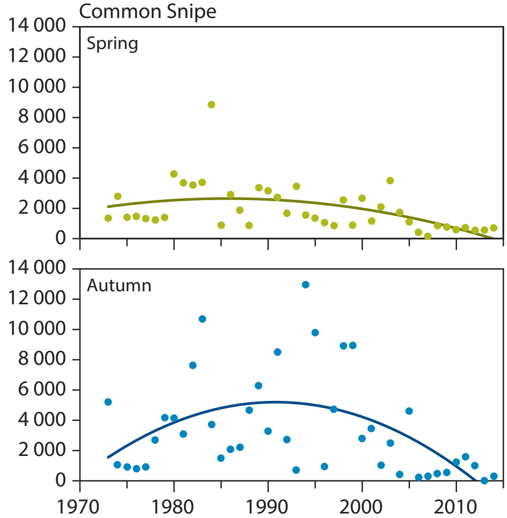


Fig. 61. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Common Snipe with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Dobbeltbekkasin pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

created optimal habitats for the autumn staging snipes during the 1970s and 1980s. Such cutting rarely takes place anymore. Furthermore, the frequency of counts was considerably reduced from 2002 onwards. Similarly, line transects on the meadows (see Meltofte 1987) were highly reduced as early as from 1996 onwards and ceased completely in 2003. It is even possible that more salt water from the mid-1990s and lower water levels from around 2000 onwards may have diluted the feeding opportunities for snipes in the swamps along the edges of the meadows.

The Common Snipes on Tipperne may, however, include large numbers of birds that roost during daytime and make night time feeding excursions to neighbouring sites (Meltofte 1987). In autumn, this most likely involves moulting adults that utilize the no-hunting Reserve for extended periods as a safe roosting and feeding area (cf. Meltofte 1993). The need for Tipperne as a safe daytime roost may have diminished after the reestablishment of the Skjern Å delta around year 2000.

The size of the European flyway Common Snipe population is uncertain, but it probably numbers in the order of 7.4-14.5 million individuals (Nagy *et al.* 2015). The population trend is no less uncertain. It is given as stable by Nagy *et al.*, but overall the breeding population has declined in the southern part of the range, whereas it may have been stable further north (Delany *et al.* 2009). Yet, monitoring in Norway, Finland and northern Sweden show fluctuating but slightly decreasing numbers (Lindström *et al.* 2015). The Danish breeding population was estimated to have numbered 2500-3000 pairs in 1993-1996 (Thorup 2006a), and to have been reduced by about 50% between 1983 and 2014 (Nyegaard *et al.* 2015).

Conclusion: Huge annual quantities of sediments and nutrients pouring from the main river into the fjord from the 1960s onwards resulted in significant increases in numbers of both breeding and staging Common Snipes on the Reserve. Since then, recorded numbers have decreased, possibly as a result of altered habitat management, reduced frequency of counts and higher salinity of the fjord water.

Jack Snipe *Lymnocryptes minimus*

The cryptic coloration and behaviour of Jack Snipes mean that only a fraction of the birds present on the Reserve during spring and autumn migrations are recorded. In spring, only a few are seen in most years, while in autumn maximum numbers are generally a little higher (Figs 62 and 63). Fewer were recorded in the 1930s than in the 1970s and 1980s, while the passage

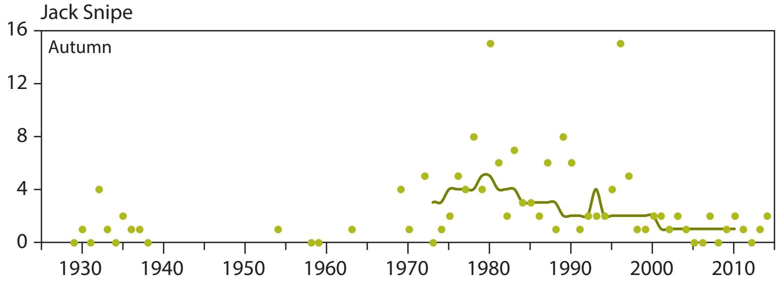


Fig. 62. Annual autumn maximum numbers of Jack Snipe (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Enkeltbekkasin (jvf. Tab. 1) med niårs glidende medianer.

periods were only fragmentarily covered by observations during the 1940s-1960s (Meltofte 1987). Autumn numbers in particular have gone down over the last few decades (Figs 62, 63 and 64), which probably can be attributed to less intensive fieldwork on the meadows and in their swampy margins, where this hard-to-find species primarily roosts. Not only was the frequency of counts considerably reduced from 2002 onwards, but line transects on the meadows (see Meltofte 1987) were greatly reduced as early as from 1996 onwards, ceasing completely in 2003.

The possibility cannot be excluded, however, that more salty fjord water from the mid-1990s and lower water levels from around 2000 onwards may have diminished the feeding opportunities for snipes in the swamps along the edges of the meadows.

The total European flyway population is estimated to number in the order of one million (Nagy *et al.* 2015). Trend data are inconclusive but are given as stable by Nagy *et al.*, and the species has expanded its range to the north during the 20th century (Lappo *et al.* 2012). In Norway, Finland and northern Sweden, the population has fluctuated without any clear trend (Lindström *et al.* 2015).

Conclusion: Numbers of Jack Snipes recorded on the Reserve have decreased most likely as a result of highly reduced frequency of the counts, but a local decline cannot be ruled out.

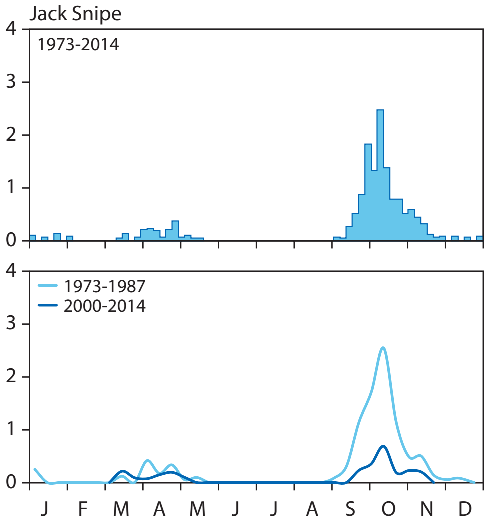


Fig. 63. The phenology of Jack Snipe on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Enkeltbekkasiner på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

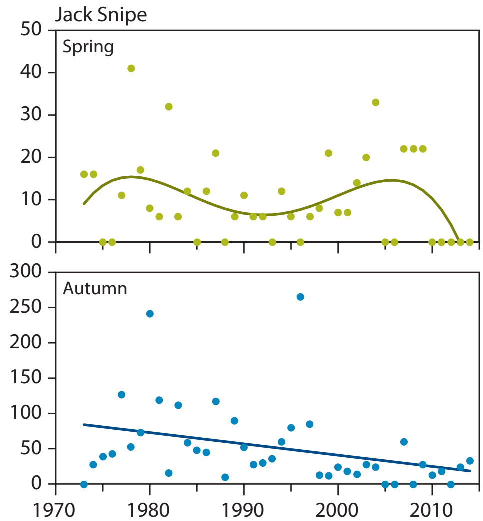


Fig. 64. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Jack Snipe with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Enkeltbekkasin pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

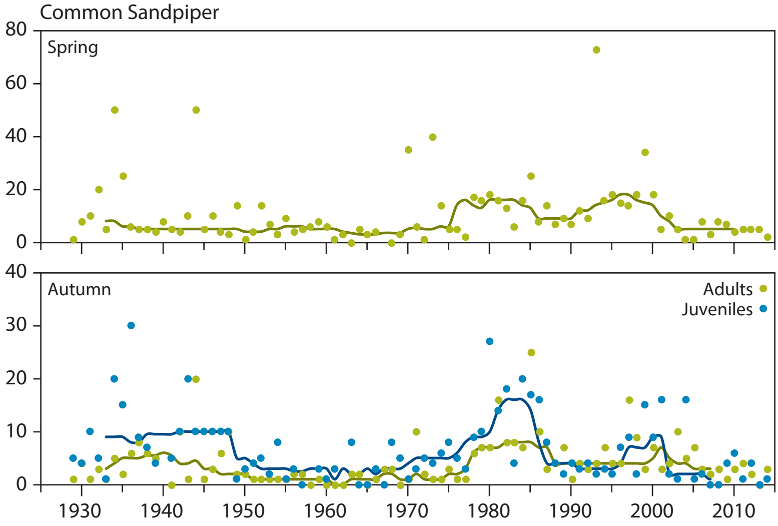


Fig. 65. Annual autumn maximum numbers of Common Sandpiper (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Mudderklire (jvf. Tab. 1) med niårs glidende medianer.

Common Sandpiper *Actitis hypoleucos*

Up to 73 Common Sandpipers have been recorded on the Tipperne Reserve in spring and 30 in autumn, but in most years numbers were much lower (Figs 65 and 66). Larger numbers seem to appear in a series of years alternating with periods of low numbers. But numbers recorded are very sensitive to the intensity of activities in the terrain, so that relatively few have been recorded since 2002, when count frequency was much reduced. This is also reflected in the bird-days (Fig. 67).

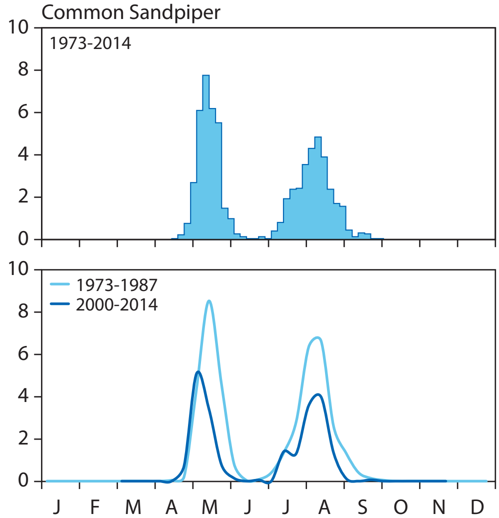


Fig. 66. The phenology of Common Sandpiper on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Mudderklirer på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

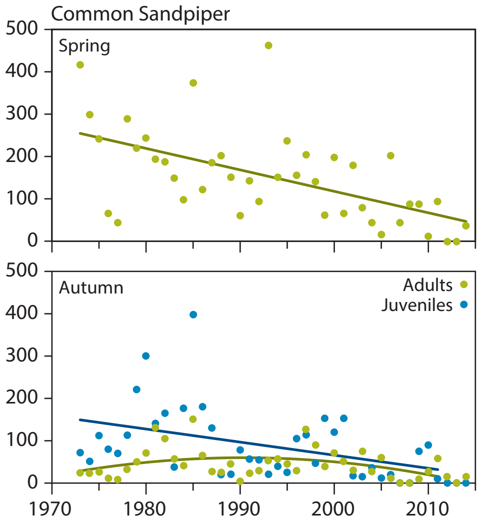


Fig. 67. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Common Sandpiper with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Mudderklire pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

The breeding population in North, West and Central Europe is estimated to number in the order of 1-3 million individuals in winter (Nagy *et al.* 2015). This population appears to have been relatively stable during the second half of the 20th century, but with decreases in some of the southern ranges, and now even some northern populations appear to be declining (Delany *et al.* 2009, Nagy *et al.* 2015). Extensive monitoring in Norway, Finland and northern Sweden, however, show stable populations (Lindström *et al.* 2015). The stability during the latter half of the 20th century is confirmed by data on actively migrating birds at Blåvandshuk south of Tipperne in 1964-2003 (Meltofte *et al.* 2006), and it is possible that the species even expanded to the north in Europe during the 20th century (Lappo *et al.* 2012). In Denmark, the species is only an irregular breeder (Nyegaard *et al.* 2014).

Conclusion: Varying numbers of Common Sandpipers occur on Tipperne during spring and autumn migration. Lower numbers were recorded during recent years, most likely as a result of lower counting intensity.

Green Sandpiper *Tringa ochropus*

Only small numbers of Green Sandpipers occur on Tipperne Reserve. Most records are from ditches and ponds on the meadows, and the numbers recorded are highly dependent on observer activity on the meadows. Since this activity has gone down since 2002, it is noteworthy that numbers of Green Sandpipers recorded in spring have been increasing in recent decades (Figs 68 and 69). Until 1983, no records of more than five individuals were obtained, while up to 15 have been recorded in recent years, a development that is also reflected in the bird-days (Fig. 70). On the other hand, we cannot rule out the possible influence of more intensive breeding bird census work in the meadows on records of increasing numbers.

Few adults were seen on autumn migration, while numbers during the juvenile migration from mid-July onwards also appear to have increased since around 1980 (Fig. 68). Evaluation of this finding is hampered by great uncertainty over the early records of large numbers, particularly in the 1930s, which must be considered unreliable (miss-identified Wood Sandpipers; Meltofte 1987). The recent increase is also not reflected in the bird-days (Fig. 70), and since around 2000 maximum numbers have decreased again.

The European breeding population is estimated to number between 1.8 and 3.3 million individuals in winter and to be stable (Nagy *et al.* 2015). However, the breeding range has expanded to the west and north during the 20th century – including the start of breeding in Denmark in the 1950s – and the populations of Norway and Finland have increased (Delany *et al.* 2009, Nyegaard *et al.* 2014). More recently, the large population of northern Sweden has also increased (Lindström *et al.* 2015). The Danish population now numbers about 50 pairs (Nyegaard *et al.* 2014).

Conclusion: Low but increasing numbers of Green Sandpipers were recorded on Tipperne Reserve during the study period. Whether this is related to local conditions or a general flyway population increase is unknown.

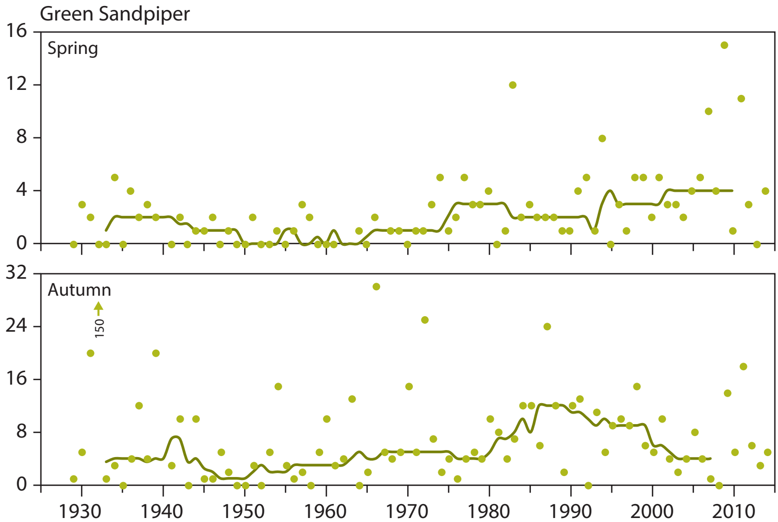


Fig. 68. Annual autumn maximum numbers of Green Sandpiper (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Svaleklire (jvf. Tab. 1) med niårs glidende medianer.

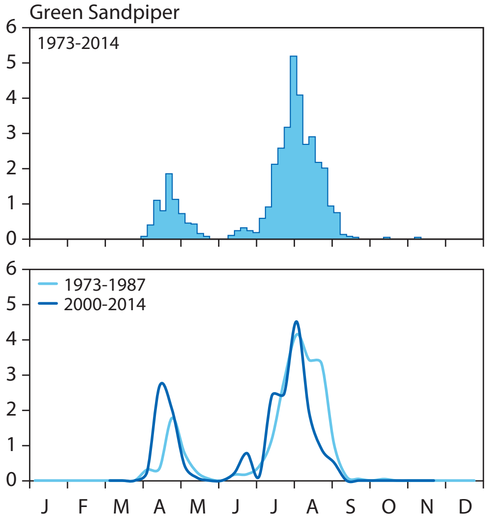


Fig. 69. The phenology of Green Sandpiper on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Svaleklirer på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

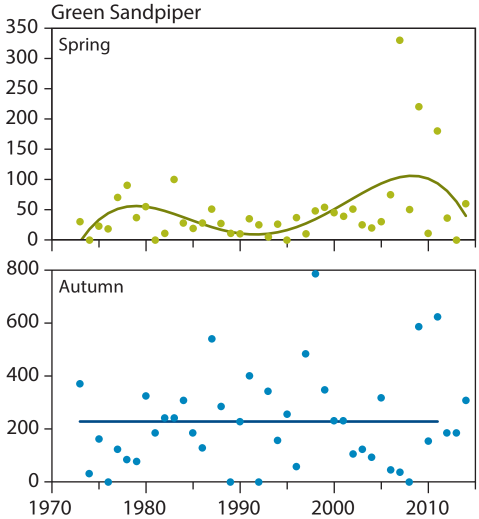


Fig. 70. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Green Sandpiper with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Svaleklire pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Spotted Redshank *Tringa erythropus*

Since the regulation and canalization of river Skjern Å in the mid-1960s, the amounts of sediments and nutrients flowing into Ringkøbing Fjord have grown considerably (see the chapter on Habitats and food resources). Consequently, numbers of Spotted Redshanks staging on the Tipperne Reserve increased to around a hundred or more during both spring and autumn migration (Fig. 71; Meltofte 1987).

During spring migration, annual maximum numbers increased from generally 0-5 to between 25 and 100, with peak numbers of up to around 140 for a few years in the 1980s and 1990s (Fig. 71). Furthermore, up to 350 have been recorded in early May in Værnsande on the southern part of the Tipperne Peninsula (DOFbasen). Lower numbers on Tipperne in most years since around 2000 may be the result of the reduction of the frequency of counts from 2002 onwards, which may have meant that the relatively narrow peak of spring migration was often missed (Fig. 72). Numbers of bird-days have seen similar trends (Fig. 73).

The Spotted Redshank is also one of the species on Tipperne which has advanced its arrival the most, i.e. by about 50 days in the 80 year period 1929-2008 (Petersen *et al.* 2012). This is probably the result of the increase in numbers itself combined with earlier movement of spring staging birds from West European wintering areas to spring staging areas further towards the northeast.

During autumn migration of both adults and juveniles, maximum numbers grew similarly from generally below five in the 1930s and generally below 20 in the 1940s and 1950s to between 50 and 150 in most years since the late 1970s (Fig. 71). The absolute peak number was 301 on Tipperne in June 1992.

There seems to be a difference between the trends in development in adult and juvenile numbers: first, very few adults occurred prior to the mid-1960s – either during spring or autumn migrations; second, adults in June-July have maintained higher numbers than juveniles in autumn (Figs 71 and 72). The most obvious explanation is that adults are more selective in their choice of habitat (soft sediments with high densities of invertebrates) than juveniles (Meltofte 1987). Measured in bird-days, adults have maintained the high level, while numbers of juveniles have decreased slightly since peak numbers in the 1980s (Fig. 73). Because of the reduced census frequency, this is hardly indicative of changes – especially not in the last few years with many missing counts.

The breeding population of northern Europe is estimated to number between 61 500 and 162 000 individuals in winter and probably to be stable (van Roomen *et al.* 2015, Nagy *et al.* 2015). A southwards

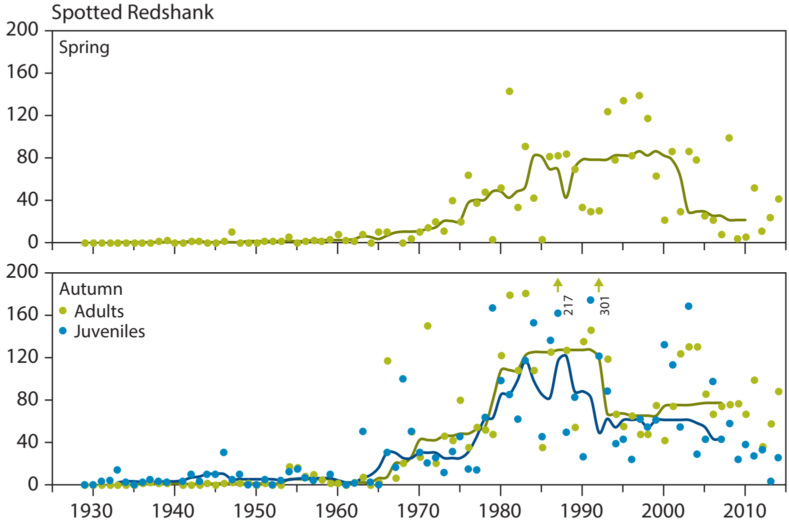


Fig. 71. Annual autumn maximum numbers of Spotted Redshank (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Sortklire (jvf. Tab. 1) med niårs glidende medianer.

expansion of the breeding range in Sweden has taken place during the 20th century (Delany *et al.* 2009). Numbers of staging birds in the Wadden Sea have declined during the last 25 years (Blew *et al.* 2015), and so have the breeding populations of Norway, Finland and northern Sweden (Lindström *et al.* 2015).

Conclusion: Spotted Redshanks have been favoured by the highly increased loads of sediments and nutrients flowing into the fjord following the drainage of the river Skjern Å delta in the mid-1960s. As a result, Tipperne Peninsula is the most important staging area in Denmark outside the Wadden Sea. Furthermore, the pioneers arrive at least 50 days earlier now in spring than they did 86 years ago.

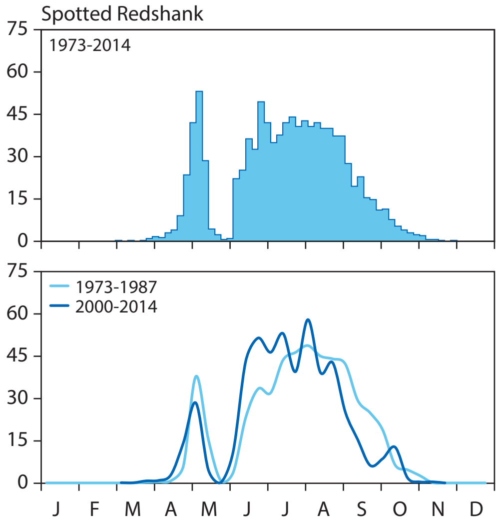


Fig. 72. The phenology of Spotted Redshank on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Sortklirer på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

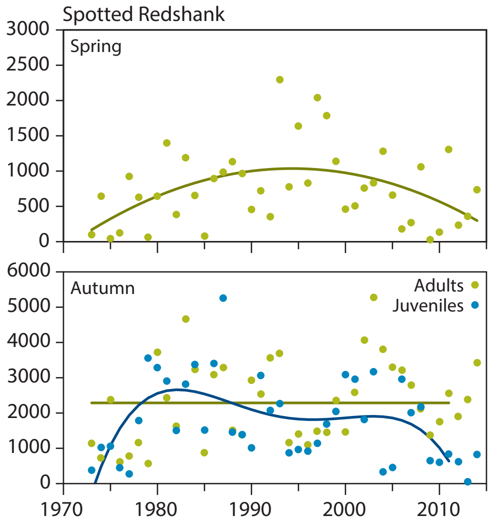


Fig. 73. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Spotted Redshank with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Sortklire pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Common Greenshank *Tringa nebularia*

The Common Greenshank is one of the species that reacted most dramatically to the increasing sedimentation and eutrophication of the fjord following the canalization and drainage of the river Skjern Å delta in the mid-1960s (Meltofte 1987). Spring and autumn adults in particular increased from maximum numbers of less than 50 to several hundred (Fig. 74), while juveniles increased from generally less than 100 to between 100 and 500 in most years since the early 1980s. The increase has continued even up to the last decade, so that recently up to 1000-1300 individuals have been recorded in spring and 1500-1800 in autumn. That it is adults which have responded most noticeably is also evident from comparison of the phenological characteristics of the periods 1973-1987 and 2000-2014, where the autumn increase was especially dramatic during the adult migration in July and early August (Fig. 75). This is also reflected in the bird-days (Fig. 76).

With 'regularly' more than 800-1000 Common Greenshanks on Tipperne Reserve both in spring and autumn, the area has developed into one of the most important staging areas in Europe outside of the Wadden Sea (Delany *et al.* 2009). To this must be added many records of several hundred (spring max. 625, autumn max. 570) in Værnsande on the southern part of Tipperne Peninsula (DOFbasen).

The Common Greenshank is also one of the species whose arrival has advanced most since 1929 so that the pioneers now arrive more than five weeks earlier than 80 years ago (Petersen *et al.* 2012; see also Meltofte 1987).

The size of the breeding population of Common Greenshanks in the western Palearctic is poorly known. In the westernmost part of the range, however, in northwestern Europe east to Finland and the Baltic states, the total population is estimated to number between 230 000 and 470 000 individuals in winter (Nagy *et al.* 2015). The population was thought to be stable (Delany *et al.* 2009), but is now considered increasing (Nagy *et al.* 2015). The latter is in accordance with data on birds passing Blåvandshuk on active migration, which increased during the 40 year period 1964-2003 (Meltofte *et al.* 2006) as did the numbers of birds on the East Atlantic flyway during 1993-2014 (van Roomen *et al.* 2015). The big Finnish breeding population has also increased together with the smaller populations of Estonia and Belarus (Delany *et al.* 2009). Even the breeding range in Sweden expanded (Thorup 2006a), but more recently the populations of Norway, Finland and northern Sweden were reported as stable during 2002-2013 (Lindström *et al*. 2015). On the contrary, the population in the Laplandsky Reserve on the Kola Peninsula has declined after an initial 2-3 fold increase in the 1960s (Lappo *et al.* 2012). Twenty-five years of monitoring in the Wadden

Numbers of Common Greenshanks staging on Tipperne have increased more dramatically than any other wader species following greatly increased sedimentation and eutrophication of the fjord after reclamation of the river Skjern Å delta in the mid-1960s. Photo: Erik Thomsen.

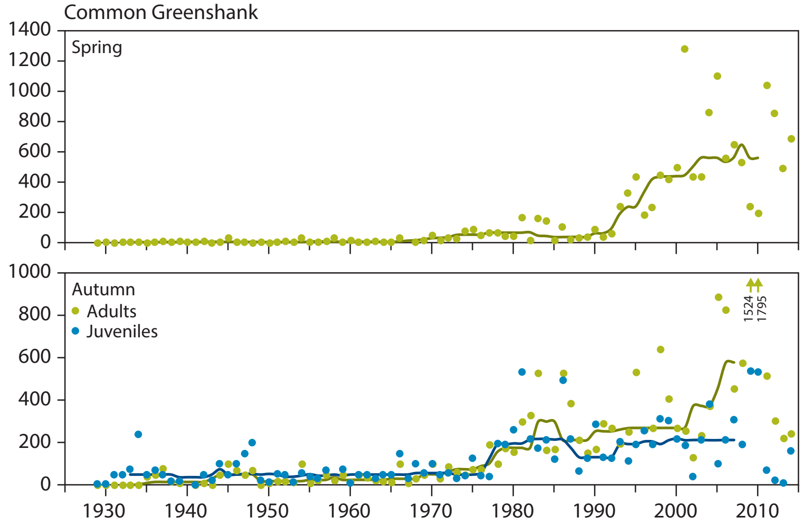


Fig. 74. Annual autumn maximum numbers of Common Greenshank (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Hvidklire (jvf. Tab. 1) med niårs glidende medianer.

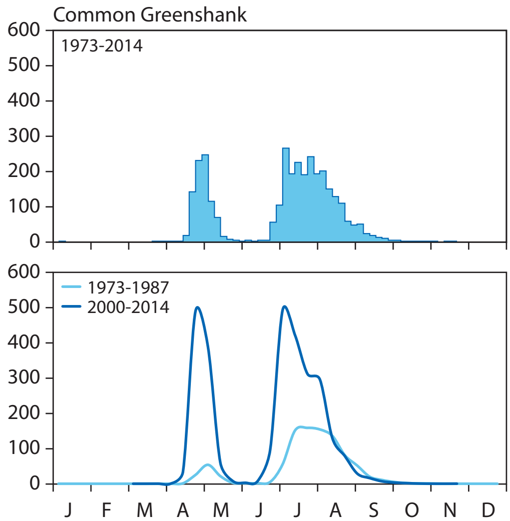


Fig. 75. The phenology of Common Greenshank on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Hvidklirer på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

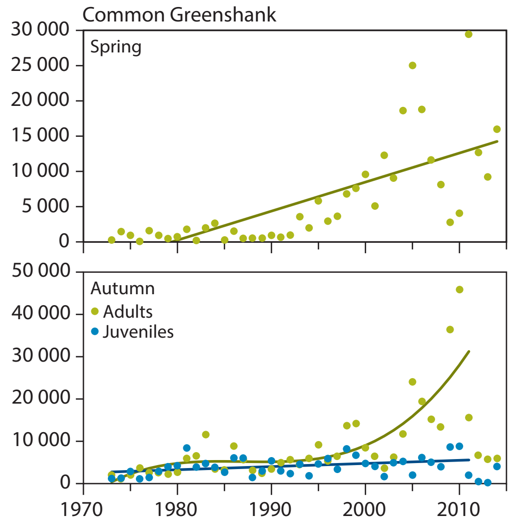


Fig. 76. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Common Greenshank with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Hvidklire pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

Sea show stable numbers (Blew *et al.* 2015).

Conclusion: Common Greenshanks on Tipperne Reserve have undergone one of the most dramatic population increases among all wader species, from maximum numbers of less than 50 adults in most years to more than 1000 in several recent years. Adults have increased more markedly than juveniles, and overall the increase is probably the result of increased sedimentation and eutrophication of the fjord following reclamation of the river Skjern Å delta. The flyway population has also increased – at least until recently. Furthermore, the first Common Greenshanks now arrive more than five weeks earlier than they did at the beginning of the study period.

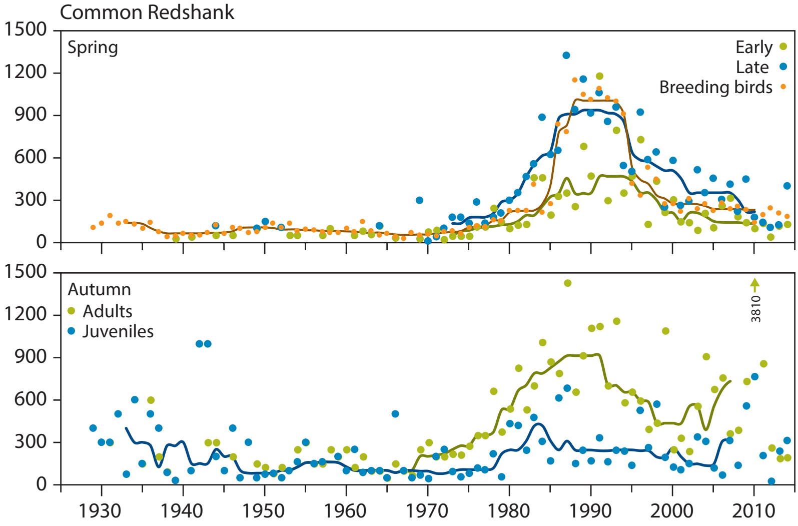


Fig. 77. Annual autumn maximum numbers of Common Redshank (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Rødben (jvf. Tab. 1) med niårs glidende medianer.

Common Redshank *Tringa totanus*

During the first 50 years of the study period around 25-75 pairs of Common Redshanks bred on Tipperne Reserve (Fig. 77; Thorup 1998 and *in litt.*). Since then the population increased to culminate in a peak of 400-575 pairs during the late 1980s and early 1990s, after which it decreased to a level of 100-140 pairs in the last 15 years. A further 225-550 unsuccessful pairs of Common Redshanks were estimated to belong to the population on the Reserve during 1990-2004 (O. Thorup *in litt.*) when 150-220 pairs also bred on Værnengene on the southern part of Tipperne Peninsula (O. Amstrup *in litt.*). The total breeding population in Denmark was estimated to number between 12 000 and 15 000 pairs in the late part of the 20th century (Thorup 2006a) and to be stable (Thorup & Ottvall 2014).

The increase among staging Common Redshanks on the Reserve began with the increase in sediments and nutrients led into the fjord as a result of the canalization of river Skjern Å and the intensification of agriculture from the mid-1960s onwards (Fig. 77; Meltofte 1987). This was followed by a further increase in spring numbers paralleling the breeding population and peaking at maximum numbers of around 1000 individuals in the late 1980s and early 1990s (Fig. 77). Since then, numbers of spring staging Common Redshanks have declined in line with the breeding population.

Compared to the early years of the study period, arrival of Common Redshanks on Tipperne Reserve has advanced by two weeks (Meltofte 1987, Petersen *et al.* 2012), while egg laying starts six days earlier (Petersen 2010).

Besides rich feeding possibilities on the shallow flats, the 'extreme' numbers of Common Redshanks breeding on the Reserve during much of the 1980s and 1990s were mainly the result of low predation pressure and optimal management of the meadows (Thorup 1998). The high numbers of breeders probably mask the occurrence of migrants staging on the Reserve, so that there is no clear difference between the early period in March-April with predominantly local ('southern') breeders and the occurrence during May of northern migrants from the Scandinavian Peninsula (cf. Meltofte 1993). The phenological graphs similarly do not show any pronounced separate peak in May (Fig. 78), and in the records from the first decades there are almost no indications of spring staging flocks in May, until records of up to 150-250 appear in 1955, 1958, 1959 and 1969 and probably continued to increase thereafter (Meltofte 1987).

During June-July, Common Redshanks on the Reserve are a mixture of local adults and juveniles probably from much of western Jutland, with increasing numbers of staging adult northern migrants during July (Meltofte 1987, 1993). During this time of the year, numbers grew from 1970-1975 onwards, when the Reserve began to function as a post-breeding feeding area for western Jutland Common Redshanks. Furthermore, maximum numbers from this period have seen a more modest decrease than in the spring numbers during recent years, so that several records in the 2000s have been almost as high as during the 1980-1990s boom in the breeding population (Fig. 77). In 2010, even an all-time high of 3810 Common Redshanks was counted on

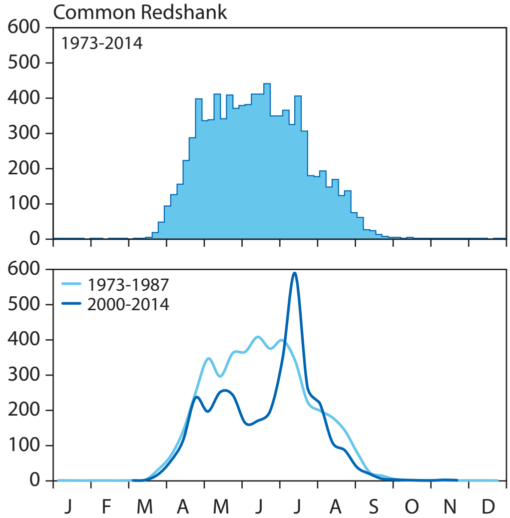


Fig. 78. The phenology of Common Redshank on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Rødben på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

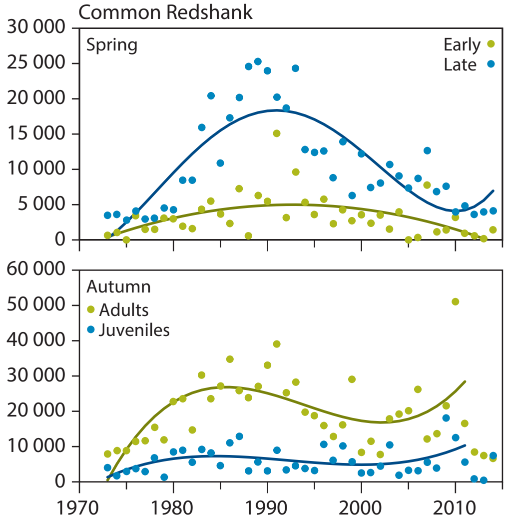


Fig. 79. Annual numbers of accumulated bird-days per season (cf. Tab. 1) for Common Redshank with polynomial regression lines (see text).

Årlige antal akkumulerede fugledage for Rødben pr. sæson (jvf. Tab. 1) med polynomiale regressionslinjer (se teksten).

16 July when 'huge' flats were exposed by strong southerly winds. It is this extraordinary number that accounts for the summer peak in Fig. 78. Aside from this extreme figure, maximum numbers in July during the most recent 15 years were almost as high as during 1973-87, a fact that is also reflected in the bird-days during the adult migration period (Fig. 79).

During the migration period of juveniles in August-September, long-term trends in numbers of staging Common Redshanks on the Reserve were somewhat different from trends in adults. Except for some very high records in the first 15 years of the study period, juvenile numbers were generally much lower than during the adult migration period, and the increase following greater sedimentation came later than in adults (Fig. 77). And as with the adult autumn migration period, maximum numbers in August-September did not decrease in parallel with the breeding population. Most likely, this is because local juveniles leave as early as during July,

An all-time high of 3810 Common Redshanks was recorded on Tipperne in July 2010, when a combination of post-breeders and their young from western Jutland and migrants from the Scandinavian Peninsula may have been present. Water colour by Jens Gregersen.

while the birds in August-September are northern migrants (Meltofte 1987, 1993). The result is that numbers have been as high during the last three decades as during the 1930s (Fig. 77), when low summer water levels may have facilitated the staging birds.

The Common Redshanks on Tipperne Reserve are *T. t. totanus* 'mudflat'-Redshanks (Meltofte 1987). We do not know whether the insular North Atlantic subspecies *T. t. robusta* occurs on Tipperne, but we find it unlikely that it occurs in any numbers. This is because of the differing habitat preferences of *robusta* Redshanks (sandy and stony coasts) and the fact that hardly any Common Redshanks are seen on Tipperne during winter, i.e. when *robusta* is found in Denmark (Meltofte 1993).

The Danish breeders are part of the continental population of Common Redshanks (*T. t. totanus* breeding from France and Iberia north to the southern Baltic and east to the Urals and Black Sea region) that are estimated to number between 372 000 and 664 000 individuals in winter and to be decreasing (Delany *et al.* 2009) or possibly decreasing (Nagy *et al.* 2015). The total winter population of Common Redshanks from northern Europe (*T. t. totanus* from Fennoscandia and NW European Russia and wintering mainly in West Africa) is estimated to number between 154 000 and 205 000 individuals and to be stable or fluctuating (Nagy *et al.* 2015). It is these birds that make up the May peak of migrants in Denmark and pass south again during July (adults) and August-September (juveniles; Meltofte 1993). Since the North European population probably makes up the vast majority of birds in August-September, our data support the notion that this population is stable, as does data from the East Atlantic flyway (van Roomen *et al.* 2015). However, 40 years of monitoring of Common Redshanks passing Blåvandshuk south of Tipperne show increasing numbers of this population (Meltofte *et al.* 2006), and so do data from the breeding grounds in Norway, Finland and northern Sweden (Lindström *et al.* 2015). Even the breeding range of this population in north-western Russia has expanded and numbers seem to have increased (Lappo *et al.* 2012; see further below).

In the Wadden Sea, where all three populations occur in large numbers, numbers have been stable (fluctuating) for the last 25 years (Blew *et al.* 2015). However, numbers have been increasing during the periods where Fennoscandian Common Redshanks are present (Laursen & Frikke 2013).

Conclusion: Numbers of Common Redshanks on Tipperne Reserve are dominated by local breeders (and their offspring) during March to July/early August, when

numbers have fluctuated very much in parallel with the breeding population. However, staging Fennoscandian migrants have made up an increasing proportion in July following the decline in the local breeding population; these migrants are probably numerically predominant among the juvenile Common Redshanks in August-September, when numbers in recent decades have been as high as in the 1930s.

Wood Sandpiper *Tringa glareola*

Numbers of adult Wood Sandpipers recorded on Tipperne Reserve have increased quite a lot in recent decades both during spring and autumn migration (Figs 80, 81 and 82). Up to around year 2000, maximum numbers were generally around or below 10 in spring, while even lower numbers were recorded during the adult autumn migration. Since then, maximum numbers of adults in several years reached 20-40 in spring and 10-30 in autumn. This is also reflected in the numbers of bird-days of adults in both spring and autumn, but here the often much higher and fluctuating numbers of juveniles dominate the overall pattern (Fig. 82). Hence, juveniles regularly reached maximum numbers of between 20 and 40 (Fig. 80) and in 1987, 98 were recorded on 9 August when the entire Reserve was not even covered. Such high numbers of juveniles have been recorded from the very beginning of the study period (three years with max. 100-150 in the 1960s – and in the 1930s when up to 150 were misidentified as Green Sandpipers e.g. in 1932; Fig. 69 and Meltofte 1987).

The increase in adult numbers has occurred in spite of lower observer activity since 2002, which in spring was possibly offset by increased breeding bird census activity in the same period. Increased prey density resulting from increased sedimentation and eutrophication is not likely to be the reason for this increase. This is because the increase in adult Wood Sandpiper records occurred more than 30 years after the canalization of the river Skjern Å delta and the increase in other soft sediment sandpipers like Spotted Redshank and Common Greenshank. Hence, we have no explanation for the increase, but obviously the adult migration has expanded to include western and northern Denmark to a greater extent than previously (cf. Meltofte 1993).

The Northwest European breeding population of Wood Sandpipers is estimated to number between 1.5 and 2.7 million individuals in winter and to be stable (Nagy *et al.* 2015). Several small populations in the southern part of the European breeding range – including Denmark with about 100 pairs at present – are known to have declined during the 20th century, whereas the species has increased in Norway, Finland and northern Sweden and expanded to the north in Siberia (Delany *et al.* 2009, Lappo *et al.* 2012, Lindström *et al.* 2015).

Conclusion: Increasing numbers of adult Wood Sandpipers have been recorded on Tipperne Reserve in recent decades, while numbers of juveniles have fluctuated widely during the entire study period. We have no unambiguous explanation for the increase among adults, but the possibility exists that it is the result of an increasing flyway population.

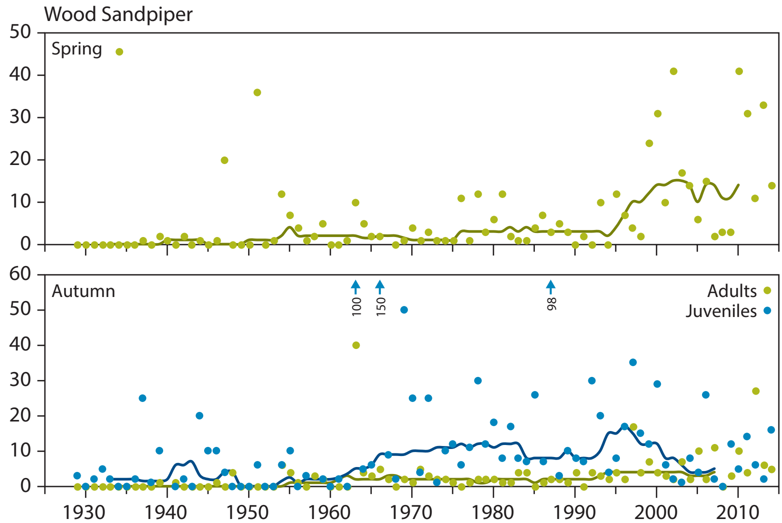


Fig. 80. Annual autumn maximum numbers of Wood Sandpiper (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Tinksmed (jvf. Tab. 1) med niårs glidende medianer.

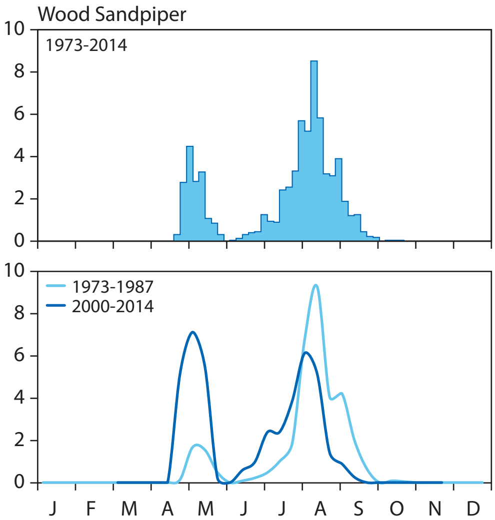


Fig. 81. The phenology of Wood Sandpiper on Tipperne shown in five-day periods for 1973-2014 and in smoothed 10-day periods for the first and the last 15 years of the same period, respectively.

Fænologien i forekomsterne af Tinksmede på Tipperne vist i femdagesperioder for 1973-2014 og i udjævnede tidagesperioder for hhv. de første og de sidste 15 år i samme periode.

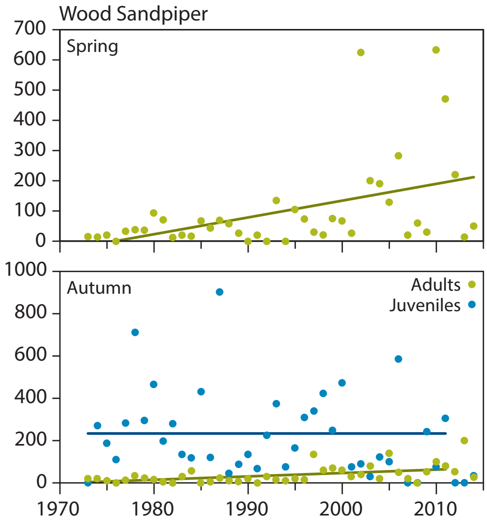


Fig. 82. Annual autumn maximum numbers of Wood Sandpiper (cf. Tab. 1) with nine-year running medians.

Årlige efterårs-maksimumtal for Tinksmed (jvf. Tab. 1) med niårs glidende medianer.

Numbers of staging adult Wood Sandpipers have gone up on Tipperne during the study period, possibly as a result of increasing breeding populations in Scandinavia. Photo: Erik Thomsen.

Discussion

An almost unbroken time series of 86 years of 'daily' records of waders is rather unique, but varying intensity – and skills – in conducting the censuses hampers our opportunity to draw firm conclusions. How good were the counts in the 1930s and 1940s when only hand-held binoculars were available, and how accurately did they count or estimate bird numbers in those days? No formalized census standards existed before 1973, and methods and accuracy probably varied quite considerably between observers. This was evident from a meeting in 1973, when all the 'old' observers were invited to Tipperne to discuss and evaluate the quality of the old data (see Meltofte 1987 and Meltofte & Clausen 2011 for staging birds). Possible shortcomings associated with the early censuses mean that not too much weight should be given to records like 100 000 Dunlins in 1945, 50 000 Eurasian Golden Plovers in 1959 or 10 000 Northern Lapwings in 1945, nor maybe even to a number of other very high records of Eurasian Curlew, Bar-tailed Godwit, Black-tailed Godwit and Common Redshank in the 1940s (see however, Meltofte & Clausen 2011, for reduced hunting during the war as a possible explanation for high numbers in the 1940s).

Having said this, it is clear that pronounced changes in the occurrence of the different wader species have taken place not only since the censuses were systematized in 1973 but also before that. We have therefore attempted to provide an overview of trends in wader numbers on Tipperne over the long-term (the entire study period) as well as the short-term (2000-2014) in Tab. 2, and we compare these evaluations with similar trends in the relevant flyways in the 20th century (long-term) and in recent decades (short-term). In making this evaluation, we have tried to balance data from a number of key sources referred to in the individual species accounts, i.e. we have critically reviewed each species on their respective flyways primarily in relation to the most recent authoritative accounts given by Delany *et al.* (2009), Nagy *et al.* (2014, 2015) and van Roomen *et al.* (2015).

Effects of varying local conditions

The long-term pattern in wader numbers on Tipperne Reserve is one of increase in many species following the canalisation and reclamation of the delta of the main river running into the fjord, Skjern Å, in the mid-1960s. These changes – in combination with increased use of fertilizers in agriculture in the catchment area – led to heavily increased sedimentation and eutrophication of the fjord, resulting in increasing densities of benthic invertebrates as food for the waders (see the chapter on habitats and food resources together with Meltofte 1987 and Meltofte & Clausen 2011). This led to many-fold increases in soft bottom feeders like Common Snipe, Spotted Redshank, Common Greenshank and Common Redshank, in particular, but also in a few other species. These increases are in stark contrast to marked decreases in a number of herbivorous waterfowl after a collapse in submerged vegetation in the fjord in 1979. That was when eutrophication had reached such a level that the macrophytes died due to shading by pelagic and epiphytic algae – a situation from which the ecosystem of the fjord is only now about to fully recover (Meltofte & Clausen 2011, unpubl.).

The increased sedimentation and eutrophication led not only to increased densities of benthos but also accelerated the spread of reeds etc. over the shallow flats, so that 1.35 km^2 of the highest and thereby most accessible flats have been lost to the waders up to the present time – a process that is continuing. In total, about one third of the mudflats have been lost in this way. We see little indication that the overgrowing has affected the numbers of staging waders, however. This is probably because loss of feeding areas took place at the same time as the many-fold increase in densities of prey on the remaining flats, and maybe because these high lying flats often dried out and therefore may have had little benthic life for extended periods.

An exception to this may be the decreasing numbers of Bar-tailed Godwits that utilized the Reserve in spring and autumn up to around 1990. This decrease may be the combined result of increased water levels, disappearance of small bivalves, overgrowth of the highest flats with reeds etc., and depletion of the much sought after ragworm stocks due to many more Bar-tailed Godwits 'grazing' them in early spring, i.e. before the arrival of peak numbers in May.

Greatly reduced spring numbers of Bar-tailed Godwits and Dunlins after severe winters where ice cover has killed the benthos on the shallow flats (Meltofte 1987, Desholm 2000) illustrates the relation of these species to food quantities. Studies in the 1970s also showed that waders depleted the stocks of larger individuals of ragworms in spring (Petersen 1981). On an annual basis, Petersen found that the three most numerous benthos-feeders, Dunlin, Bar-tailed Godwit and Avocet, removed about 65% of the standing stock of benthic prey on the accessible parts of the shallow flats, which was equal to about 30% of the annual production.

The increasing densities of benthos mainly involved ragworms, mud shrimps, mud snails and sludge worms,

Tab. 2. Orders of magnitude in regular maximum numbers of waders on Tipperne Reserve during recent decades together with trends in local numbers and in the relevant flyway populations. Orders of magnitude are given as ones (1-9) = +, tens = ++, hundreds = +++ and thousands = ++++. Long-term trends (> 30 years) are based on nine-year running medians in maximum numbers 1929-2014; short-term trends (15 years) are based on the polynomial regression analyses of trends in bird-days 2000-2014 presented for each species. In case of differing trends between seasons, the season with most birds has been used. Increase ▲ is defined as > 25% increase, decrease ▼ as > 20% decrease, strong increase ▲▲ as > 100% increase, strong decrease ▼▼ as > 50% decrease. Less change is considered stable/fluctuating = ●. Signs in brackets are uncertain, e.g. biased by changes in observer effort or simply insufficient data.

Størrelsesordenerne af regelmæssige maksimumforekomster af vadefugle på Tipperne i de senere årtier sammen med trends i de lokale antal rastende vadefugle samt i de relevante flyway-bestande. Størrelsesordenerne er angivet som encifrede antal (1-9) = +, tocifrede antal = ++, hundreder = +++ og tusinder = ++++. Langtids-trends (> 30 år) er baserede på niårs glidende medianer i maksimumtallene 1929-2014; korttids-trends (15 år) er baserede på polynomial-analyserne af trends i fugledagene 2000-2014. I tilfælde af afvigende trens mellem sæsonerne, er perioden med flest fugle brugt. Øgede antal ▲ er defineret som > 25 % øgning, nedgang ▼ som > 20 % nedgang, stærk stigning ▲▲ er defineret som > 100 % stigning, stor nedgang ▼▼ som > 50 % reduktion. Mindre ændringer er angivet som stabile/fluktuerende = ●. Signaturer i parenteser er usikre, fx biased af ændringer i tællefrekvens eller andre data-mangler.

Species	Max. numbers	Tipperne long-term	Tipperne short-term	Flyway long-term	Flyway short-term
Eurasian Oystercatcher *Haematopus ostralegus*[1]	++	▼▼	▲	▲▲	▼
Pied Avocet *Recurvirostra avosetta*[1]	+++	▲▲	●	▲▲	▲
Grey Plover *Pluvialis squatarola*	+++	▲▲	●	▲▲	(●)
Eurasian Golden Plover *Pluvialis apricaria*	++++	▲▲	▲▲	(▲)	(●)
Common Ringed Plover *Charadrius hiaticula*	+++	▲▲	●	(●)	(●)
Northern Lapwing *Vanellus vanellus*	++++	▲▲	▼▼	(●)	●
Whimbrel *Numenius phaeopus*	+++	●	●	(●)	●
Eurasian Curlew *Numenius arquata*	+++	▼▼	▲▲	(▼)	(▲)
Bar-tailed Godwit *Limosa lapponica*	+++	▼▼	●	(●)	▲[2]
Black-tailed Godwit *Limosa limosa*	+++	▲	▼▼	▲▲	▼▼
Ruddy Turnstone *Arenaria interpres*	++	●	●	▲	▲[3]
Red Knot *Calidris canutus*	+++	●	●	▼	●[4]
Ruff *Calidris pugnax*	+++	▲▲	▼▼	(●)	▼▼[5]
Curlew Sandpiper *Calidris ferruginea*	++	▲▲	●	▲▲	(▼)
Temminck's Stint *Calidris temminckii*	+	●	(●)	▼	(▼)
Dunlin *Calidris alpina*	++++	●	(▼)	●	●
Little Stint *Calidris minuta*	++	▲	●	(●)	(●)
Common Snipe *Gallinago gallinago*	(+++)	▲▲	▼▼	(●)	(▼)
Jack Snipe *Lymnocryptes minimus*	(+)	(▲)	(▼▼)	(●)	(●)
Common Sandpiper *Actitis hypoleucos*	++	●	(▼)	●	(●)
Green Sandpiper *Tringa ochropus*	++	▲	●	▲	(▲)
Spotted Redshank *Tringa erythropus*	+++	▲▲	●	(●)	(▼)
Common Greenshank *Tringa nebularia*	+++	▲▲	▲▲	▲	(●)
Common Redshank *Tringa totanus*[1]	+++	▲▲	●	▲	▲[6]
Wood Sandpiper *Tringa glareola*	++	▲	▲	(▲)	▲

1) On Tipperne, post-breeding birds only
2) Subspecies *lapponica* only; subspecies *taymyrensis* is stable or possibly decreasing
3) Nearctic birds only; North European breeders are possibly decreasing
4) Subspecies *islandica* only; subspecies *canutus* is probably decreasing
5) West European breeders and staging migrangts only
6) North European subspecies *totanus* only; continental subspecies *totanus* is decreasing

whereas bivalves (sand gaper) had already disappeared from the shallow flats by the 1960s due to reduced salinity of the fjord water. The decreasing numbers of Oystercatchers may at least partly be related to this latter factor.

To facilitate the breeding meadow birds on the Reserve, Thorup (1998) recommends that the water level in the fjord should be kept between about 15 cm and 30 cm DNN during the birds' breeding season in spring and early summer (see Figs 2, 3 and 4). For the majority of the waders dealt with here, this period overlaps with much of the spring and adult autumn passage. This makes the shallow flats accessible to waders and other birds and at the same time prevents the upper flats and lower meadows from drying out.

Another major development in the conditions for waders on the Reserve has been the changing management of the meadows during the study period. After a period with rather uniform short-grass swards due to extensive hay cutting and after-grazing by cattle until the 1950s, the area became overgrown with tall grasses, reeds etc. This resulted in decreasing numbers of grassland waders such as Eurasian Golden Plover and Northern Lapwing. The decline lasted only until more diversified management was introduced in the 1970s, and numbers of waders feeding and roosting on the meadows recovered. Common Snipes were also favoured by extensive cutting of reeds etc. from the 1970s onwards but this practice has now been reduced, which is possibly part of the reason for the decreasing numbers of snipes recorded during recent decades (see further below). During the study period, breeding and feeding possibilities on adjacent meadows around Ringkøbing Fjord also deteriorated as the meadows were increasingly converted to arable fields. This deterioration may have resulted in fewer Golden Plovers, Lapwings, Black-tailed Godwits and Curlews from these areas using the Reserve as a safe day-time, night-time or post-breeding roost (Meltofte 1987).

Turning to the short-term trends, the general picture becomes a bit gloomier in that trends in numbers of several species have turned from positive to negative. Some of these declines relate to declining breeding populations in the Reserve due to increased predation etc. (Black-tailed Godwit, Ruff, Common Redshank, in

particular, but also Eurasian Oystercatcher and Pied Avocet in spring), whereas other declines may be related to decreasing counting frequency. The significance of this was illustrated by Clausen & Holm (2011) using a data dilution exercise, where the full dataset from Tipperne (one count each five-day period, i.e. six per month), and reduced subsamples of the dataset were compared. They found that observed peak numbers of Eurasian Wigeon *Anas penelope* and Common Merganser *Mergus merganser* on average were reduced to about 86% of that observed in the full dataset when halving the count-frequency (three counts per months), and to 73% for Wigeon and 55% for Common Merganser when reducing to one count per month. Assuming the same orders of magnitude would be the case for waders on Tipperne, any decline in numbers by 25-50% that seem to have occurred from 1998 onwards may thus simply be a matter of reduced counting frequency, rather than true declines. In contrast, any observed increase in numbers reflects a true increase, because this most likely would have been even more prominent, if the protocol with a count each five-day period had continued until today.

Habitat management of the meadows on Tipperne is important not only for the breeding meadow birds but also for some of the staging species such as Eurasian Golden Plover and Common Snipe. Water colour by Jens Gregersen.

For the waders in particular, it is evident that the reduced census frequency since 2002 has reduced the chances of counting taking place during the often short time periods during which maximum numbers of birds are present in the area. Even though the observers have tried to time the counts during the peak occurrences of species such as (spring staging) Spotted Redshank and Common Greenshank, and we have reduced the effect of lowered census frequency by using bird-days as a measure for our short-term trends instead of maximum numbers, it is likely that reduced observer activity on the Reserve is at least part of the reason for decreasing numbers recorded of species such as Jack Snipe, Common Sandpiper and juvenile Temminck's Stint. It is clear here that heavily reduced census frequency has seriously hampered our ability to reaching firm conclusions about the trends in quite a number of species in addition to the fact that we have had to exclude autumn numbers from the last three years in the time series for most species.

Where these factors appear not to be the sole explanation for the declines, the apparent decline in densities of benthos after the culmination in the 1990s may also be the reason for shrinking numbers (see e.g. van de Pol *et al.* 2014). If this is so, it should be seen as a positive effect of serious efforts to reduce the losses of nutrients from cities, industry and agriculture ever since the late 1980s (EEA 2015, Riemann *et al.* 2015). Among the remaining species, numbers for several of them have simply stabilized at a higher level.

Relations to varying flyway populations

There is even more uncertainty when we compare trends on Tipperne with flyway trends in the same species (Tab. 2). Numerical flyway trends based on midwinter counts are only available for a limited number of species/populations and only from 1988 onwards (Nagy *et al.* 2014, van Roomen *et al.* 2015). For the remaining species, we have only more general statements like those by Delany *et al.* (2009) and Nagy *et al.* (2015). Moreover, these statements are often based on conflicting results from different sources or are contradicted by other data. Trustworthy long-term trends from the 20th century, are even rarer (Tab. 2; Cramp *et al.* 1983). Unfortunately, the result is often a plethora of conflicting statements based on partly different time periods (see the individual species accounts), and here we have not even included references to all the Waterbird Population Estimate reports (see Wetlands International 2015).

According to our critical review of long-term as well as short-term trends in the flyway populations (predominantly on the East Atlantic flyway), most populations are stable, fluctuating or increasing (Tab. 2[1]). Focusing on short-term trends at flyway level in our subset of 25 species with 29 distinct populations in Tab. 2, only four species/populations are decreasing and a further 6-7 species are possibly decreasing, while five species are increasing and a further two are possibly increasing. The remaining 11-12 species/populations are either fluctuating, stable or possibly stable. These ratios shift even further towards more stable populations when the remainder of the species listen in the footnote are included.

Similar results were obtained when analysing monitoring results of 24 breeding wader species along 1263 line transects in Norway, Finland and northern Sweden during 2002-2013 (Lindstöm *et al.* 2015). And among 17 waders species on active migration (i.e. independent of local habitat changes) monitored during 40 years from 1964-2003 at Blåvand Bird Observatory south of Tipperne, only one species, the Eurasian Oystercatcher was found to have substantially decreased (Meltofte *et al.* 2006).

Moreover, the extensive Fennoscandian study, which probably provides more reliable trends in boreal waders in northernmost Europe than mid-winter counts particularly in Africa, does not confirm the finding e.g. by Zöckler *et al.* (2003) that more long-distance than short-distance migrant waders are in decline (Lindström *et al.* 2015).

The positive long-term trends are probably related to two very prominent factors. These are reduced hunting pressure that may have kept several populations below natural levels for much of the 20th century (Prater 1981, Cramp *et al.* 1983, Tubbs 1996; see also Piersma *et al.* 2005, Meltofte *et al.* 2009, Péron *et al.* 2012 and Taylor & Dodd 2013), and nutrient 'enrichment' (eutrophication) of our environment from a multitude of human sources

1 Besides the species and subspecies presented in Tab. 2, a number of other wader populations regularly occur in western Europe where the Icelandic subpopulation of 'northern' Eurasian Golden Plover *Pluvialis apricaria altifrons* is possibly decreasing, Eurasian Dotterel *Eudromias morinellus* has unknown status, the *psammodroma* subspecies of Common Ringed Plover is considered stable, Little Ringed Plover *Charadrius dubius* is considered stable, Kentish Plover *Charadrius alexandrinus* is increasing, the *islandicus* subspecies of Whimbrel has unknown status, the *islandica* subspecies of Black-tailed Godwit is increasing, Broad-billed Sandpiper *Limicola falcinellus* has unknown status, Sanderling *Calidris alba* is increasing, two populations of Purple Sandpiper *Calidris maritima* are considered stable and possibly decreasing, respectively, the Icelandic subpopulation of Short-billed Dunlin *Calidris alpina schinzii* is considered stable, Eurasian Woodcock *Scolopax rusticola* is considered stable, Great Snipe *Gallinago media* is considered stable, the *faroeensis* subspecies of Common Snipe has unknown status, the *robusta* subspecies of Common Redshank is possibly decreasing, Red-necked Phalarope *Phalaropus lobatus* is considered stable, while the status of Red Phalarope *Phalaropus fulicarius* is unknown (Nagy *et al.* 2015).

with agricultural runoff as the most widespread (EEA 2015; see also Meltofte *et al.* 1994 and van Roomen *et al.* 2005).

Climate change may also have benefitted some populations in the form both of milder winters with enhanced survival particularly of juveniles (see e.g. Watkinson *et al.* 2004 and Ryan *et al.* 2015) and earlier snow-melt and increased invertebrate food availability for adults as well as chicks on tundra and other northern breeding habitats (Boyd & Madsen 1997, Lindström & Agrell 1999, Meltofte *et al.* 2007). In this regard the recent development among high latitude waders in Norway, Sweden and Finland is interesting because this geographically extensive study found more increasing than decreasing populations among 24 species (Lindström *et al.* 2015). Equally notable are the results of a long-term study 1972-2011 in Swedish Lapland that found statistically significant increases in half of 12 commonly breeding waders and no significant decreases (Svensson & Andersson 2013). This does not imply that human-induced climate change necessarily is good for northern waders; in the longer term many breeding habitats may be lost because of overgrowth with shrubs and trees together with desiccation of the tundra during summer (Meltofte 2013). On top of this come rising sea levels that will threaten important coastal staging and wintering habitats (Watkinson *et al.* 2004).

Finally, the EU Birds Directive and the network of Natura 2000 sites have reduced physical encroachment on most key wader staging and wintering areas on the East Atlantic flyway quite considerably (see e.g. Donald *et al.* 2007). In Africa, the Ramsar Convention has provided protection to a large number of important sites (Ramsar 2014), although protection is probably not as efficiently enforced. Indeed, among the flyways within the African-Eurasian Migratory Waterbirds Agreement area, the East Atlantic flyway has the highest proportion (40%) of increasing waterbird populations (Nagy *et al.* 2015), even without taking the reservations mentioned in the present paper into consideration.

Exceptions to the predominantly positive picture are species breeding and/or staging on semi-natural grasslands in temperate Europe (Black-tailed Godwit and Ruff, in particular) that have seen precipitous declines in recent decades due to agricultural intensification (Thorup 2006a). Overharvest of common cockles *Cerastoderma edule* in the Dutch Wadden Sea and the Wash in England have probably also caused the sharp decline in Eurasian Oystercatcher in recent decades (Delany *et al.* 2009).

Reduced nutritional 'enrichment' of staging and wintering wetland habitats may be a contributory reason for the degree to which the status of some species has changed from increasing to stable or even decreasing during recent decades, because many countries have implemented action plans to reduce the eutrophication of their waters (EEA 2015). If this is so, this should – like for the local environment on Tipperne – be seen as a positive effect of serious efforts to reduce the losses of nutrients from cities, industry and agriculture during recent decades.

The extraordinary advance in date of arrival at northern staging areas in spring is a further indication of improved conditions for waders in our part of the world. On Tipperne we have seen an advance of 2-9 weeks in arrival of the pioneers during 1929-2008 (Petersen *et al.* 2012), and the bulk of immigration already took place 2-4 weeks earlier during the 1980s (Meltofte 1987, Laursen & Frikke 2013; see also Thorup 1998). These changes may be related to milder winters and springs (Petersen *et al.* 2012) that make invertebrate food available to the waders earlier in the season. It is important to note, however, that these changes do not imply that migration to Arctic and sub-Arctic breeding areas also takes place that much earlier (but see Bar-tailed Godwit). The changes primarily mean that many waders move up to the northern parts of their temperate spring staging areas much earlier than they did before, i.e. in March-April instead of late April/early May (Meltofte 1987, Laursen & Frikke 2013), and that they also winter progressively farther to the northeast (MacLean *et al.* 2008, Visser *et al.* 2009). For local breeders this change actually involves a genuine advance in arrival on the breeding areas, and for some of them even an advance in initiation of egg laying (Petersen 2010). Some species (Pied Avocet, Northern Lapwing, Eurasian Curlew, Ruff and Common Redshank) apparently also remain on the Reserve longer into the autumn than they did before (Meltofte 1987).

When all the data are considered, few changes in wader numbers on Tipperne Reserve can be related to overall flyway population trends, with Pied Avocet, Grey Plover, Eurasian Curlew and Curlew Sandpiper as the most noteworthy examples (see also Meltofte & Clausen 2011 for the waterfowl). However, even in Pied Avocet and Eurasian Curlew local effects may have had at least as much influence as overall population trends. This is probably the case for most staging and wintering areas in the world – and even for the individual parts of a mega site like the Wadden Sea. Paying more attention to local factors may lessen the risk of 'alarmist' reports being publicized, such as *Seriously Declining Trends in Migratory Waterbirds: Causes – Concerns – Consequences* (Common Wadden Sea Secretariat 2007) and *Wader populations are declining – how will we elucidate the reasons?* (Zöckler *et al.* 2003). In highlighting this, we wish to warn against statements about large-scale decline that may turn out to be erroneous and therefore counterproductive in a world where prioritization of the most important conservation issues is the key to success. One such example of miss-prioritization is the dubious listing of the Eurasian Curlew as Near Threatened on IUCN's Red List of Threatened species (see species accounts).

Indeed, a little more patience might be needed to see whether more populations e.g. in West Africa are actually decreasing – and find out why. Or as expressed by van Roomen *et al.* (2013): "Because population counts are subject to short-term fluctuations, caused by both environmental effects and by counting errors, it necessarily takes time to distinguish systematic changes from such stochastic variation, in order to not 'raise the alert' unnecessarily."

We will end by citing Lappo *et al.* (2012) in their great *Atlas of breeding waders in the Russian Arctic*: "Surprisingly few wader species had negative trends …" It is our hope that we can focus on the well documented conservation problems like those facing the 'meadow birds' of temperate European grasslands and several waterbird populations on the East Asian/Austral-Pacific flyway (Thorup 2006a, van de Kam 2008, MacKinnon *et al.* 2012, Wetlands International 2012, Piersma *et al.* 2015) where serious efforts are obviously needed.

Postscript

When monitoring of bird numbers and habitat use began on Tipperne in 1928 (breeding birds; followed by staging birds in 1929), the Reserve was so remote from ministries, scientific institutions etc. in Copenhagen that it took a day or two to travel there. It was therefore necessary for the observers/guards to live in the Reserve during most of the year also in order to enforce the ban on public access and hunting in the Reserve. The same applied to researchers performing more specific studies in the Reserve. This had the positive side effect that both observers and researchers lived in the very center of the Reserve and came to 'live and breathe' the Reserve. Indeed, the research station on Tipperne was a power house of wetland and waterbird research in Denmark up to the 1980s.

Since then, monitoring and research have expanded to take place at many other sites. Combined with much easier transport this has meant that the authorities' motivation to run increasingly expensive research stations has diminished. Unfortunately, this has had a negative side effect in that the benefits are increasingly being lost of having a number of sites representative of important Danish nature types where intensive monitoring supports analyses of causal relationships between environmental conditions and numbers of staging and wintering birds (see Fredningsstyrelsen 1986). Such causal relationships make it possible to evaluate the reasons for similar changes documented in many less intensively monitored areas in other parts of Denmark and abroad. While monitoring of breeding birds on Tipperne continues unreduced, the research and monitoring there will no longer be able to provide comparable results for staging and wintering birds to the same extent as previously.

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2 Note that several of the photos presented were not taken on Tipperne Reserve.

Resumé

Udviklingen i forekomsterne af rastende vadefugle på Tipperne 1929-2014 med en kritisk gennemgang af udviklingen i flyway-bestandene

Antallene af rastende såvel som ynglende vandfugle i Tipperreservatet i Vestjylland (Fig. 1) er blevet talt lige siden 1928. Det har resulteret i en af verdens længste tidsserier til belysning af fuglenes forekomst og trivsel i relation til ændringer i områdets miljøforhold og flyway-bestandene som sådan (Madsen 1985, Meltofte 1987, 2010, Thorup 1998, Meltofte & Clausen 2011, Petersen *et al.* 2012, Meltofte & Amstrup 2013). Forekomsterne af rastende vadefugle er tidligere bearbejdet t.o.m. 1982 (Meltofte 1987), og i nærværende afhandling opdaterer vi disse resultater med fokus på forekomstændringer i reservatet og i flyway-bestandene generelt.

De første mange år var tællingerne af rastende fugle på reservatet ikke særlig systematiske. Man noterende antallene dagligt eller med nogle dages mellemrum, men fra 1973 blev tællingerne organiseret i faste rutiner, som fra 1978 fokuserede på en totaloptælling af fuglene i reservatet i hver løbende femdagesperiode året rundt (Fig. 7). Siden er der dog sket en række nedskæringer i aktiviteterne, således at tællingerne i vintermånederne december-februar ophørte fra 1998, fra 2002 blev tællingerne reduceret til tre pr. måned om foråret og to pr. måned om efteråret og fra 2012 yderligere til to pr. måned også om foråret.

Data præsenteres på tre måder. Ændringer i løbet af hele undersøgelsesperioden 1929-2014 vises som løbende niårs medianer af maksimumtal registreret indenfor afgrænsede perioder under forårs- og efterårstrækket (Tab. 1). For den mere systematisk dækkede periode 1973-2014 har vi beregnet fugledage for de samme perioder under forårs- og efterårstrækket og analyseret trends i disse ved hjælp af polynomial regression (R Core Team 2013). Endelig har vi opstillet fænologiske grafer, hvor vi sammenligner forekomsterne for de første 15 år i den intensivt dækkede periode med de sidste 15 år og med hele perioden samlet.

Betingelserne for fuglene i reservatet bestemmes i høj grad af reguleringen af vandstanden i fjorden ved slusen i Hvide Sande og af udledningen af sediment og næringsstoffer fra oplandet, dvs. i alt væsentlig grad fra Skjern Å. Hertil kommer forvaltningen af reservatets landområder ved græsning og slåning af enge og rørskove.

De mest markante ændringer i forekomsterne af vadefugle på reservatet skete således i forbindelse med afvandingen af Skjern Å-deltaet midt i 1960erne, hvor tilførslen af sediment steg voldsomt, og hvor udledningen af næringsstoffer samtidig øgedes kraftigt som følge af øget brug af kunstgødning i landbruget. Dette førte til stærkt øgede tætheder af bunddyr på vadefladerne og dermed til stærkt stigende forekomster af muddervadearter som Dobbeltbekkasin, Sortklire, Hvidklire og Rødben m.fl. (Tab. 2). Dette er i stærk kontrast til det 'sammenbrud' i forekomsterne af planteædende svømmefugle, der skete i fjorden omkring 1979 som følge af plantedød forårsaget af algeopblomstringer (Meltofte & Clausen 2011).

Den væsentligste undtagelse fra generelt positive langtidstendenser blandt de rastende vadefugle på Tipperne er de aftagende mængder af Små Kobbersnepper frem til omkring 1990. Årsagen hertil skal formentlig søges i en kombination af højere vandstande, overgroning af de højestliggende vader med 'opgrøde' og 'overudnyttelse' af kobbersneppernes foretrukne føde, børsteorme, pga. ankomsten af større mængder kobbersnepper og andre vadere tidligt på sæsonen.

Aftagende slåning og græsning af Tippernes enge fra 1950erne til 1970erne medførte tilgroning af engene med reducerede antal rastende Hjejler og Viber til følge. Med den genoptagne pleje af engene kom disse arter tilbage, ligesom Dobbeltbekkasinerne nød godt af en omfattende slåning af rørskovsområderne i 1970erne og 1980erne.

Ser vi i stedet på de mere kortsigtede ændringer (15 år) bliver billedet mindre positivt for en række arter (Tab. 2). Dette gælder først og fremmest en række ynglefuglearter, som toppede med ekstremt store ynglebestande først i 1990erne, men som siden er aftaget til væsentligt lavere niveauer primært som følge af øget prædation (Thorup 1998 og *in litt.*). For en række arter har den stærkt reducerede tællefrekvens siden 2002 givetvis reduceret mulighederne for at 'fange' de ofte kortvarige toppe i forekomsterne, som er typiske for vadefugletrækket. Men også aftagende bunddyrstætheder siden 1990erne som følge af vandmiljøplaner mv. kan have gjort sig gældende.

Sammenlignet med udviklingstendenserne i de samlede flyway-bestande er der kun få ændringer på Tipperne, der kan relateres hertil. Den væsentligste årsag er givetvis, at forekomsterne i et så relativt lille område som Tipperne i langt højere grad bestemmes af lokale forhold end af ændringer i de overordnede bestandsstørrelser, som i øvrigt også ofte er meget usikre. Vi har således foretaget en kritisk revision af angivelserne i de seneste autoritative oversigter, nemlig Delany *et al.* (2009), Nagy *et al.* (2014, 2015) og van Roomen *et al.* (2015). Denne kritiske revision tyder på, at mens en række af de ynglende 'engfugle' i det tempererede Europa er gået ganske markant tilbage (især Stor Kobbersneppe og Brushane) pga. intensiveringen af landbruget, så trives de fleste arktiske og boreale vadefuglebestande udmærket.

En indikation på forbedrede forhold for vadefuglene er, at en række arter nu ankommer flere uger tidligere om foråret, end de gjorde for 80 år siden, og at flere også påbegynder æglægningen tidligere (Petersen 2010, Petersen *et al.* 2012). Dette er formentlig et resultat af mildere vintre og forår og dermed tidligere tilgængelighed af bunddyr på vadefladerne, som gør det muligt for fuglene at flytte herop væsentligt tidligere end før i tiden.

Der er dog nogle arter, hvis antalsmæssige forekomstændringer på Tipperne i det mindste delvis kan relateres

til ændringer på flyway-niveau med Klyde, Strandhjejle og Storspove som de mest markante. Bestandene af alle tre arter blev formentlig stærkt reducerede i det 19.-20. århundrede pga. jagt og fangst, men er efterfølgende gået frem som følge af forbedret beskyttelse.

Sammenfattende kan man sige, at vores kritiske analyse af de involverede vadefuglebestandes trivsel tyder på, at de ofte påståede udbredte nedgange blandt vadefuglene (fx Common Wadden Sea Secretariat 2007, Zöckler *et al.* 2003) næppe er så generelle som fremført. Hvis vi fokuserer på kortsigtede tendenser på flyway-niveau blandt vores 25 arter med 29 forskellige populationer, er kun fire arter/bestande aftagende og yderligere 6-7 muligvis aftagende, mens fem er tiltagende, og yderligere to muligvis tiltagende. De resterende 11-12 arter/bestande er enten svingende, stabile eller måske stabile (Tab. 2). Disse fordelinger ændres endnu længere i retning af mere stabile bestande, hvis 18 europæiske arter/bestande, der ikke er omfattet i denne analyse, medtages. Vores anbefaling er derfor, at man fokuserer beskyttelsesarbejdet på de arter og bestande, som har veldokumenterede problemer såsom 'engfuglene' i vores del af verden (Thorup 2006a) og især vandfuglene i Østasien, hvoraf mange har alvorlige problemer pga. landvinding, jagt og fangst (Thorup 2006a, van de Kam 2008, MacKinnon *et al.* 2012, Wetlands International 2012, Piersma *et al.* 2015).

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