

East Atlantic Flyway Assessment 2023



The status of coastal waterbird populations and their sites



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Hans Schellekens

Common Ringed Plovers, Dunlins and Sanderling

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Roseate tern | Sterne de Dougall (*Sterna dougallii*) (Josh Jones / Agami)

Foreword

Migratory birds on the East-Atlantic Flyway need suitable habitat conditions each day in their life. A successful protection of those bird species with their unbelievable capacities to move from breeding sites far in the north to wintering areas along the (West-)African coasts requires a view on the entire range. Results from ecological research have shown that carry-over-effects may affect each stage of the whole flyway. This means, to achieve a favourable conservation status for the various migratory bird populations, a whole-flyway approach for conservation is necessary; even more, it is indispensable.

The Wadden Sea Flyway Initiative (WSFI) was founded in 2012 in response to a request from UNESCO's World Heritage Committee. What might sound as a 'formal task consisting mainly of paper work' actually means a structured and practical effort to implement international activities for Wadden Sea migratory bird conservation in the field. This was necessary because migratory birds are of utmost importance and a crucial key element of the Wadden Sea's World Heritage status.

The monitoring of bird populations is the first important step in successful conservation. Without comprehensive information about bird numbers and trends on the flyway and site level, no one is able to prioritise conservation action, identify research needs or pinpoint important political issues in bird conservation. In addition, monitoring the threats at each site can give a first indication of processes affecting bird numbers. And: without a common understanding of this relationship, joint action by colleagues and friends, institutions and governments, will not take place in an internationally cooperative manner. Therefore, the first activities of the Wadden Sea Flyway Initiative focus on monitoring, management and capacity building.

This report of the 2023 total January count of migratory and resident waterbirds along the whole flyway is – after the already impressive counts in 2014, 2017 and 2020 – another milestone in East-Atlantic Flyway monitoring. More than 13,000 observers were involved in more than 35 countries – a really impressive example of international cooperation, citizens' engagement and coordinated scientific action in this part of the world. Besides the value of this report for international nature conservation policy, this means a tremendous strengthening of ties between people all along the flyway. "Migratory birds inspire and connect people for future generations"; we can see and feel the vision of the WSFI.

The data presented in this report allow a clear view on the conservation status of waterbirds along the flyway of which many are dependent on the Wadden Sea as well: millions of birds have been counted, their habitats described and conservation issues raised. What is worrying is that more than a third of all populations are declining, particularly in recent years. Especially those species breeding in the high arctic and wintering in western Africa seem to experience problems. These are among the most iconic species, connecting the Wadden Sea with a substantial part of the world. This report throws light on those problems, discusses the pressures involved and gives wise advice for further actions. This report is a successful example of cooperation, continuity and scientific reliability.

I would like to express my recognition and admiration to all participants in all those countries involved in this effort throughout the years. Counting birds in often remote and hard-to-reach areas is more than a weekend outing into the countryside. The establishment of a community and a network around the migratory bird monitoring constitutes a success in itself. I would like to thank all coordinating, analysing and writing colleagues who did not only do the paper work but also managed to organise this enormous task. Special thanks to BirdLife International and Wetlands International with whom we jointly organize this monitoring and all other contributing partner institutions that together bring the Wadden Sea Flyway Initiative to life. Special thanks go to Sovon, particularly Marc van Roomen, whose hands held the ends of a multitude of strings and knitted them into a viable joint effort.

I not only wish for a wide dissemination of this striking publication, but also that it could trigger next steps in international migratory bird conservation beyond continents – because this is compelling: to widen the focus of the initiative to bring measures on the ground, to take actions for declining populations, to foster long-term research projects to understand underlying processes and to influence international policy and cooperation.

The Wadden Sea Flyway Initiative has been developed to a vital body for future living of biodiversity and international cooperation.

Peter Südbeck

Chair of the WSFI steering group

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General

The monitoring of waterbirds and their sites in the East Atlantic Flyway (EAF), consisting of January counts, breeding bird surveys, and assessment of environmental conditions, is carried out by a large number of observers, often volunteers, and staff of nature reserves, research institutes and conservation and management agencies, both governmental and non-governmental. Both for the counts of non-breeding waterbirds in January and for the coordination of breeding monitoring, activities in each country are streamlined by a national coordinator, employed by national research institutes, non-governmental organisations or government agencies. For the integral count of the EAF in 2023, covering 36 countries, it was estimated that 13,700 observers were involved in total. We thank all observers and national coordinators for their huge efforts. The use of breeding bird data from the Pan European Common Bird Monitoring Scheme as coordinated by The European Bird Census Council was organised by Eva Silarova and Anna Gamero from BirdLife Czechia.

More detailed information about the involvement of people, institutions and results in Western Europe and Western Africa can be found further down in this acknowledgements section under 'Principal sources of information per country', and in the country specific chapters in Agbblon et al. 2023.

The flyway coordination and support of the integral count in 2023 was funded by the Dutch Ministry of Agriculture, Fisheries, Food Security and Nature (to Sovon / Wadden Sea Flyway Initiative) and the MAVA Foundation (to Wetlands International and BirdLife International). Additional co-funding was received from governments in Denmark and Germany, National Park Wattenmeer Niedersachsen, National Park Schleswig-Holsteinisches Wattenmeer, National Park Wadden Sea Hamburg, Vogelbescherming Nederland, British Trust for Ornithology, United Kingdom and Tour du Valat, France. Also important in this respect is the funding for the coordination and database management of the International Waterbird Census, by the Member states of Wetlands International and the Swiss Federal Office for the Environment, and the funding of the breeding bird monitoring project PECBMS of the European Bird Census Council by the European Union. Further additional support for this report was received from Vogelbescherming Nederland.

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ing group of the Wadden Sea Flyway Initiative, particularly Peter Südbeck (National Park Niedersachsen), Thomas Borchers (Federal Ministry for the Environment, Germany) and Henrik G. Pind Jørgensen (Agency for Green Transition and Aquatic Environment, Denmark).

Principal sources of data by country

Norway Data for January 2023 as reported to the International Waterbird Census (IWC). National Coordinator (NC) is Svein-Hakon Lorentsen (Norwegian Institute for Nature Research, NINA). He organised also the registration of environmental factors; see Lorentsen 2023 for more details. Data about breeding birds is from BirdLife Norway and NINA.

Sweden Data for January 2023 as reported to the IWC. NC is Frederik Haas (University of Lund), no data about environmental factors were provided, see Haas & Nilsson 2023 for more details. Breeding bird data is collected through coordination of the University of Lund (Ake Lindstrom & Martin Green).

Finland Data for January 2023 as reported to the IWC. NC is Aleksii Lehtikainen (Finnish Museum of Natural History). No data about environmental factors were provided. Breeding bird data is also collected through coordination of Finnish Museum of Natural History.

Estonia Data for January 2023 as reported to the IWC. NC is Leho Luigujõe (Estonian Ornithological Society & Estonian University of Life Sciences). No data about environmental factors were provided, see Luigujõe 2023 for more details. Breeding bird data is provided through coordination of the Estonian Environment Agency (Meelis Leivits).

Latvia Data for January 2023 as reported to the IWC. NC is Antra Stipniece (Latvian Ornithological Society). No data about environmental factors were provided, see Stipniece 2023 for more details. Breeding bird data is also collected through coordination of the Latvian Ornithological Society (Ainars Aunins).

Lithuania Data for January 2023 as reported to the IWC. NC is Laimonas Sniauksta (Lithuanian Ornithological Society). No data about environmental factors were provided, see Sniauksta 2023 for more details. Breeding bird data is also collected through coordination of the Lithuanian Ornithological Society.

Poland Data for January 2023 as reported to the IWC. NC is Włodzimierz Meissner (University of Gdansk & OTOP/ BirdLife Poland). He provided also data about environmental factors. Details can be found in Meissner 2023. Data about breeding birds is collected under responsibility of Chief Inspectorate for Environmental Protection (coordination by Tomasz Chodkiewicz).

Ireland Data for January 2023 as reported to the IWC. NC

was Lesley Lewis (BirdWatch Ireland, coordination is currently taken care of by John Kennedy); the relevant project is the Irish Wetland Bird Survey which is under responsibility of the National Parks & Wildlife Service of the Department of Culture, Heritage & The Gaeltacht and BirdWatch Ireland. Data about environmental factors was also provided, see Fitzgerald et al. 2023 for more details. Breeding bird data were also provided through coordination of BirdWatch Ireland.

United Kingdom Data for January 2023 as reported to the IWC. NC is Teresa Frost (British Trust for Ornithology (BTO), on behalf of the Wetland Bird Survey, organised and funded by BTO, Royal Society for the Protection of Birds and Joint Nature Conservation Committee. No data about environmental factors were provided, see Frost 2023 for more details. In addition, data on wintering Barnacle Goose (Greenland population), Greylag Goose (Iceland population) and Brent Goose (NE Canada & Greenland population) are from the Goose and Swan Monitoring Programme as also coordinated under WEBS. Data about breeding birds were also provided by the BTO (James Heywood).

Denmark Data for January 2023 as reported to the IWC. NC is Preben Clausen (University of Aarhus). Data about environmental factors was provided by Thomas Bregnballe. Data about breeding birds is collected through DOF/BirdLife Denmark (coordination Thomas Vikstroem).

Germany Data for January 2023 as reported to the IWC. NC is Kim Lindner (Dachverband Deutscher Avifaunisten (DDA)), see further details in Lindner et al. 2023. Data about environmental factors was provided by Gregor Scheiffarth and Klaus Gunther. Breeding bird data was provided through coordination of DDA as well (Malte Busch).

The Netherlands Data for January 2023 as reported to the IWC. NC is Menno Hornman (Sovon Dutch Centre for Field Ornithology). He provided also data about environmental factors, see further details in Hornman et al. 2023. Breeding bird data is collected through coordination of Sovon as well (contact Chiel Boom). Both waterbird and breeding bird monitoring are funded through the government project 'Network Ecological Monitoring'.

Belgium Data for January 2023 as reported to the IWC. For this coastal review data from Flanders is used. NC in Flanders is Koen Devos (Research Institute for Nature and Forest (INBO)). He also provided environmental data, see further details in Devos 2023. Breeding bird data in Flanders is also collected through coordination of INBO.

France Data for January 2023 as reported to the IWC. NC was Caroline Moussy and is now Gwenaël Quaintenne (Ligue de Protection des Oiseaux (LPO)), Gwenaël organised also the collection of environmental data, see Moussy & Quaintenne 2023 for more details. Data about breeding birds is collected through coordination of Cen-

tre d'Écologie et des Sciences de la Conservation from the Museum national d'Histoire naturelle (contact Benoit Fontaine).

Spain Data for January 2023 as reported to the IWC. NC is Blas Molina (Sociedad Española de Ornitología/BirdLife Spain (SEO)) who provided also data about environmental factors. In 2023/2024 a major update of Spanish waterbird data for several recent years was provided giving a much completer overview of the importance of Spain for waterbirds. Breeding bird data is also collected through coordination of SEO (contact Virginia Escandell González).

Portugal Data for January 2023 as reported to the IWC. NC is Filipe Moniz (Instituto da Conservação da Natureza e das Florestas). He provided also environmental data. Breeding bird data is collected through coordination of Sociedade Portuguesa para o Estudo das Aves (SPEA), (contact Hany Alonso).

Morocco Data for January 2023 as reported to the IWC. Data provided by Mohamed Dakki (Mohamed V University & GREPOM), who also provided environmental data. The new IWC coordinator for Morocco is Rhimou El Hamoumi (GREPOM)

Mauritania Data for January 2023 as reported to the IWC. Counts were organised at the Parc National du Banc d'Arguin (PNBA) by Amadou Kidé (PNBA), in the Mauritanian part of the trans-boundary Biosphere Reserve of the Senegal River by Zeine El Abidine Sidaty (Parc National Diawling) and in Lac Alec and Lac Mal by Nature Mauritanie. Information about environmental conditions was also provided. Details in Kidé, Sidaty, El Abass & Ndiaye 2023. PNBA participated also in the West African survey of breeding terns and gulls in May/June 2023 (see Veen et al., chapter 9).

Senegal Data for January 2023 as reported to the IWC. NC in Senegal is Abdoulaye Ndiaye (Direction des Parcs Nationaux (DPN)). Data about environmental factors were provided, more details in Ndiaye 2023. DPN participated also in the West African survey of breeding terns and gulls in May/June 2023 (see Veen et al., chapter 9).

The Gambia Data for January 2023 as reported to the IWC. NC of The Gambia is Abdoulie Sawo (Department of Parks and Wildlife Management). He provided also data about environmental factors. For more details see Jallow, Jammeh & Sawo 2023.

Guinea-Bissau Data for January 2023 as reported to the IWC. NC for Guinea Bissau was Jãozinho Sá (Gabinete de Planificacao Costeira) who is now retired. National coordination is now provided by l'Institut de la Biodiversité et des Aires Protegees (IBAP, contact Aissa Regalla de Barros). Details about environmental conditions were also provided. More details of the count can be found in Sá & Regalla de Barros 2023. IBAP participated also in the West African survey of breeding terns and gulls in May/June 2023 (see Veen et al., chapter 9).

Guinea Data for January 2023 as reported to the IWC. NC is Kadiatou Soumay (Direction Parc Nationaux et Reserve

de Faune (OGPNRF)). Counts were carried out in cooperation with Guinea Ecology (Roger Doré). Details about environmental factors were also provided. For details of the count in 2023 see Soumay 2023.

Sierra Leone Data for January 2023 as reported to the IWC. NC is Papanie Bai-Sesay (Conservation Society of Sierra Leone (CSSL)). They provided also details about environmental factors. For details of the count in 2023 see Dauda 2023.

Liberia Data for January 2023 as reported to the IWC. NC is Emmanuel Loqueh (Society for the Conservation of Nature in Liberia (SCNL)). They provided also details about environmental factors. Details of the count in January 2023 are in Loqueh 2023.

Côte d'Ivoire Data for January 2023 as reported to the IWC. NC is Firmin Kouassi Kouame (Ministry of Water and Forests, direction de la Faune et des Ressources Cynegetiques). Details on environmental factors were provided, see details in Kouame & Uassa 2023.

Ghana Data for January 2023 as reported to the IWC. NC is Dickson Agyeman (Wildlife Division of the Forest Commission) in cooperation with Emmanuel Taye and Yaa Ntiama-Baidu (Centre for Biodiversity Conservation Research, University of Accra). They provided also details about environmental factors. For details of the count in 2023 see Taye et al. 2023.

Togo Data for January 2023 as reported to the IWC. NC is Piwelon Bakai (Ministère de l'Environnement et des Ressources Forestières). For details of the count in 2023 see Atutonu et al. 2023.

Benin Data for January 2023 as reported to the IWC. NC is Rémi Hefounme (Direction Générale des Eaux, Forêts et Chasse) in cooperation with Benin Environment & Education Society. They provided also details about environmental factors, see details in Djondo et al. 2023.

Nigeria Data for January 2023 as reported to the IWC. NC is Stella Egbe (Nigerian Conservation Foundation). She

organized also the collection of environmental data. See details of the count of 2023 in Onoja et al. 2023.

Cameroon Data for January 2023 as reported to the IWC. Long standing NC was Gordon Ajonina (Cameroon Wildlife Conservation Society, Postscript: Gordon Ajonina passed away in February 2025). Data about environmental factors were also provided. The counts of 2023 are reported in Ajonina et al. 2023.

Sao Tomé & Príncipe In the framework of the January count of January 2023 data about waterbirds were collected by the Gulf of Guinea Biodiversity Centre, details in Faustino de Lima et al. 2023.

Gabon Data for January 2023 as reported to the IWC. NC is Gabin Nzamba (Ministère des Eaux et Forêts). He provided also data about environmental factors. For details of the count in 2023 see Nzamba 2023.

Democratic Republic of Congo The count on the coast in January 2023 was coordinated by Pierre Mavuemba Tuvi (Institut Supérieur de Navigation et de Pêche). Details in Tuvi et al. 2023.

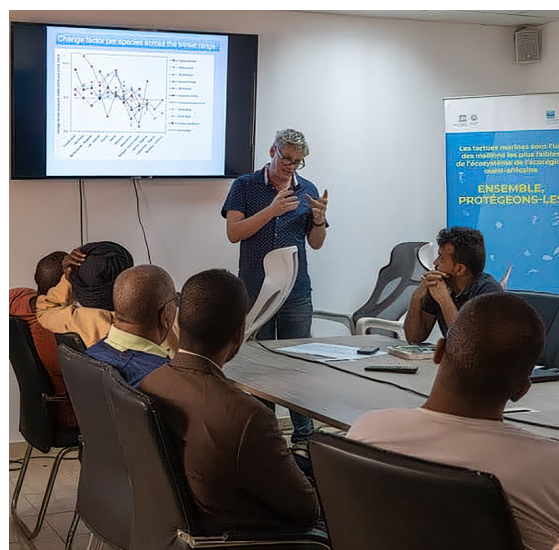
Angola Data for January 2023 as reported to the IWC. NC is Marta Zumbo (Instituto Nacional da Biodiversidade e Conservação (INBAC)), with help from Miquel Xavier. Details of the 2023 count are in Dala, Kodo, Dianquessa, Sá & Poot 2023.

Namibia Data for January 2023 as reported to the IWC. Long standing NC was Holger Kolberg (Ministry of Environment, Forestry and Tourism) who is now retired. He provided also data about environmental factors.

South Africa Data for January 2023 as reported to the IWC. NC is Michael Brooks (Fitzpatrick Institute of African Ornithology, University of Cape Town). Coordination in East Atlantic South Africa by Giselle Murison from BirdLife SA. Data about environmental factors were also provided. Details of the 2023 count are in Murison & Maree 2023.



Preparations for fieldwork, Banc d'Arguin, Mauritania



Nathalie Gibon



Eurasian Spoonbill | Spațule blanche (*Platålea leucorodia*) (Arnold Meijer / Blue Robin)

Summary

The East Atlantic Flyway Assessment 2023 raises concerns about the development of migratory waterbird populations along this globally important flyway. Although 42 percent of the monitored populations show long-term increases, short-term trends are deteriorating, with 35 percent now in decline. Arctic-breeding shorebirds wintering in Africa are among the most affected, indicating broader ecological changes across the flyway.

Overfishing, habitat loss, infrastructure developments, recreation and tourism, and climate change have been the most frequently reported pressures from key sites for migratory waterbirds. The specific mechanisms by which these and other pressures impact populations trends are not fully known and require additional research. Expanding the current monitoring programme to include additional counts, monitoring of vital rates (breeding success and survival), and a more robust quantitative assessment of environmental factors - combined with improved technical capacity for monitoring at the national level - are essential to better understand and address the threats faced by migratory birds in different parts of the flyway.

The results underline the urgent need for more effective management of coastal wetlands. Despite the ecological importance of many flyway sites, a lack of legal protection or weak enforcement of existing protection measures continue to undermine conservation efforts. Scientific information on the drivers of population changes is critical to develop tailored management strategies and strengthen marine and coastal policies along the flyway.

Finally, this East Atlantic Flyway assessment based on joint monitoring effort across the flyway underscores once more the need for stronger international collaboration and political recognition of the flyway approach. Migratory birds connect communities and countries across Africa and Europe, and their protection requires a coordinated, cross-sectoral approach. Long-term commitment from all flyway countries is required to reverse negative trends and to ensure that recent short-term declines do not lead to the local, flyway level or global extinction of species.

Résumé

L'évaluation 2023 de la voie de migration de l'Atlantique Est soulève des inquiétudes quant au développement des populations d'oiseaux d'eau migrants le long de cette voie de migration d'importance mondiale. Bien que 42 % des populations surveillées affichent des augmentations à long terme, les tendances à court terme se détériorent, 35 % d'entre elles étant désormais en déclin. Les oiseaux de rivage se reproduisant dans l'Arctique et hivernant en Afrique sont parmi les plus touchés, ce qui indique des changements écologiques plus vastes sur l'ensemble de la voie de migration.

La surpêche, la perte d'habitat, le développement d'infrastructures, les loisirs et le tourisme, ainsi que le changement climatique sont les pressions les plus fréquemment signalées sur les sites clés pour les oiseaux d'eau migrants. Les mécanismes spécifiques par lesquels ces pressions et d'autres ont un impact sur les tendances des populations ne sont pas entièrement connus et nécessitent des recherches supplémentaires. L'élargissement du programme de surveillance actuel pour inclure des comptages supplémentaires, la surveillance des paramètres vitaux (succès de la reproduction et survie) et une évaluation quantitative plus solide des facteurs environnementaux - combinés à une meilleure capacité technique de surveillance au niveau national - sont essentiels pour mieux comprendre et traiter les menaces auxquelles sont

confrontés les oiseaux migrants dans les différentes parties de l'itinéraire de migration.

Les résultats soulignent le besoin urgent d'une gestion plus efficace des zones humides côtières. Malgré l'importance écologique de nombreux sites de la voie de migration, l'absence de protection juridique ou la faible application des mesures de protection existantes continuent de freiner les efforts de conservation. Les informations scientifiques sur les facteurs de changement des populations sont essentielles pour élaborer des stratégies de gestion adaptées et renforcer les politiques marines et côtières le long de la voie de migration.

Enfin, cette évaluation de la voie de migration de l'Atlantique Est basée sur un effort de surveillance commun à travers la voie de migration souligne une fois de plus la nécessité d'une collaboration internationale plus forte et d'une reconnaissance politique de l'approche de la voie de migration. Les oiseaux migrants relient les communautés et les pays d'Afrique et d'Europe, et leur protection nécessite une approche coordonnée et intersectorielle. Un engagement à long terme de tous les pays de la voie de migration est nécessaire pour inverser les tendances négatives et pour garantir que les récents déclin à court terme ne conduisent pas à l'extinction locale, au niveau de la voie de migration ou au niveau mondial des espèces.

1. Introduction

Migratory waterbirds are famous for the connection that their travels establish between breeding habitats, stop-over sites and wintering ranges. However, this makes them also vulnerable to many different pressures operation in areas far apart from each other. Conservation and management of such migratory species is challenging as this requires global (flyway) perspective that encompasses on all sites used during their annual cycle. The East Atlantic Flyway (EAF) connects bird habitats in Africa with those in Europe and Asia, together with the Central Europe-Mediterranean-African Sahel Flyway and the West Asia - East African Flyway. The EAF stretches from the Arctic (NW Canada to Central Siberia) through W Europe to the entire western coastline of Africa (Fig 1.1). The quantity and quality of natural habitats and sites for breeding, migration and wintering for waterbirds along this flyway form the crucial basis for their sustainable future. However, the area covered by this flyway is also used by a substantial human population, with numerous cities, industries and activities distributed throughout the area. The natural habitats and sites in this flyway provides important ecosystem services in the form of food, prevention of flooding, building material, renewable energy and recreational opportunities. For proper co-existence between human needs and healthy ecosystems, conservation and management measures need to be applied. This requires careful decision making and adaptive management based on, and informed by, knowledge about the state and development of the eco-

systems. This information will help to signal problems, define priorities, identify measures to be taken and to evaluate the success or failure and improve management approaches. In addition to representing intrinsic natural values, migratory waterbirds are good indicators of the state and development of ecosystems, because they are relatively easy to quantify and react quickly to changes in habitats and pressures.

From 2013 onwards the Wadden Sea Flyway Initiative (coordinated by the Common Wadden Sea Secretariat), Wetlands International and BirdLife International combined efforts to coordinate and improve the monitoring of (coastal) waterbirds along the EAF for better conservation and management. The aim is to monitor waterbird abundance (flyway population sizes, flyway population trends and numbers and trends at sites) and to monitor environmental conditions (both pressures and conservation measures) at the sites they use (van Roomen *et al.* 2013). Consideration was also given to the need to include the monitoring of vital rates (reproduction and survival) as the link between environmental pressures and bird abundance, thereby increasing our understanding of the processes causing changes in numbers and distributions. The implementation of that last aim has however not started yet. The monitoring programme collects data annually in at least a selection of sites, focusing on waterbird counts in January. In most European countries nearly all sites

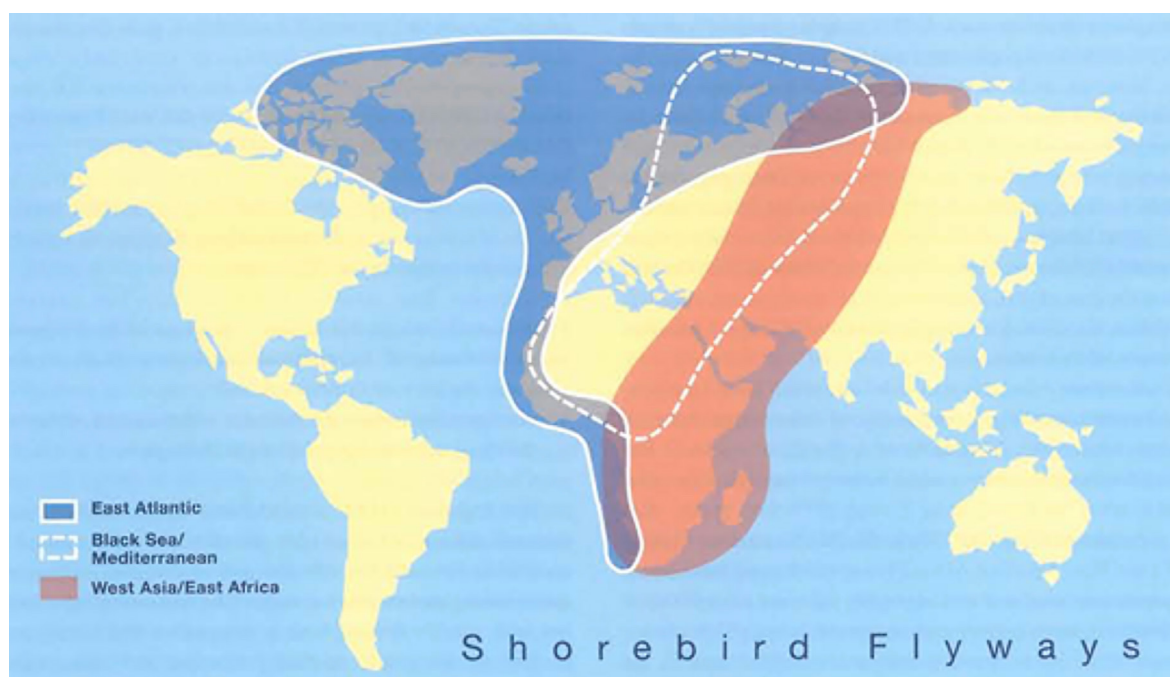


Figure 1.1. The three flyways in the African-Eurasian region as based on migratory shorebirds (Delany *et al.* 2009, after International Wader Study Group) with the East Atlantic Flyway in blue.

important for waterbirds are monitored on a yearly basis, in Africa often a selection of sites only. Therefore a more comprehensive survey is organised every three years ('total count' or integral count) with the aim to assess numbers of waterbirds at all relevant sites in January, and including an assessment of environmental conditions. More and more also breeding bird counts are organised, aiming to collect flyway- wide data for certain species whose populations are more easily assessed at this stage. Besides these simultaneous counts providing flyway-wide results, many of the sites involved have more frequent (monthly) counts of non-breeding waterbirds for local and national purposes, but these are not brought together

internationally. After each 'total count', a flyway assessment report is produced (van Roomen *et al.* 2014, 2018, 2022). The present report is the fourth assessment following the integral count of 2023. It has the same structure as the report of 2022, with two summary chapters reporting on the survey results regarding bird abundance and environmental conditions, several regional or thematic accounts and a final chapter with conclusions and recommendations. In an extensive annex, details about the numbers, trends and distribution of each species / population under study are given. The present Flyway Assessment report is based on data from 36 countries, brought together by more than 13,700 observers.



Hans Gernerad / Agami

Common Redshank | Chevalier gambette (*Tringa totanus*)



West African Crested Tern | Sterne d'Afrique (*Thalasseus albididorsalis*) (Markku Rantala /Agami)

2. An exploration of variation in short- and long-term population trends of coastal waterbirds along the East Atlantic Flyway

Jeroen Reneerkens, Hans Schekkerman, Tom Langendoen & Marc van Roomen

Summary

This chapter examines population trends of coastal waterbirds along the East Atlantic Flyway (EAF) using data from long- and short-term monitoring. Compared to previous assessments, trends remain relatively stable overall, but increasing short-term declines raise concerns. While 42% of populations showed moderate or strong long-term increases, recent short-term trends indicate worsening conditions, with a higher proportion of declining populations.

Shorebirds exhibit the most pronounced declines, particularly in Africa, while geese, flamingos, pelicans, and cormorants show more positive trends. Shorebirds breeding in the Arctic and boreal regions continue to experience the most unfavourable trends. We can only speculate about the potential causes of these unfavourable trends but there will likely be negative influence by the changing climate, habitat degradation, and changes in food availability.

Site-based analysis of nine shorebird species confirms that populations wintering in Africa generally decline, while those in Europe tend to increase. This suggests potential habitat-related issues in Africa or challenges during migration. Differences in breeding locations may also play a role in the observed trend variations.

Continued monitoring, particularly in Africa—where survey coverage is limited—is essential. Future research should focus on identifying causes of decline and improving conservation strategies across the entire flyway.

Résumé

Ce chapitre examine les tendances des populations d'oiseaux d'eau côtiers le long de la voie de migration de l'Atlantique Est (EAF), en s'appuyant sur des données de suivi à court et long termes. Comparées aux évaluations précédentes, les tendances restent globalement relativement stables, mais les déclin à court terme de plus en plus marqués suscitent des inquiétudes. Si 42 % des populations présentent des augmentations modérées ou importantes à long terme, les tendances récentes à court terme montrent une détérioration des conditions, avec une proportion plus élevée de populations en déclin.

Les limicoles présentent les déclin les plus marqués, en particulier en Afrique, tandis que les oies, flamants, péli-

cans et cormorans montrent des tendances plus positives. Les limicoles nichant dans les régions arctiques et boréales continuent d'afficher les tendances les plus défavorables. Les causes de ces déclin restent spéculatives, mais il est probable qu'elles soient liées aux effets négatifs du changement climatique, à la dégradation de l'habitat et aux variations de disponibilité alimentaire.

Une analyse ciblée de neuf espèces de limicoles confirme que les populations hivernant en Afrique déclinent généralement, tandis que celles hivernant en Europe tendent à augmenter. Cela suggère des problèmes potentiels liés aux habitats en Afrique ou des défis rencontrés durant la migration. Des différences dans les sites de reproduction pourraient également expliquer ces variations.

2.1 Introduction

This chapter provides an overview of the general trends in waterbird populations along the coastal East Atlantic Flyway (EAF), based on the population-specific trends as presented in Annex 1. We focus on summarising trends across populations and comparing these results with the findings of the earlier flyway assessment reports (van Roomen *et al.* 2015, 2018, 2022). In those reports, it was examined whether populations with similar ecological characteristics exhibited common patterns of increase or decrease, potentially highlighting shared factors influencing multiple bird populations across the EAF or specific regions within it (Schekkerman *et al.* 2018, 2022). Here, we revisit those patterns to determine if they persist with the inclusion of new monitoring data. One of the striking findings by Schekkerman *et al.* (2022) was that migratory shorebirds showed more negative regional changes in abundance between 1980s-1990s and 2010s-2020 in the southern (African) part of the flyway and more positive changes towards the north. Here we investigate this pattern in more detail by using site-specific trends in abundance. Identifying such patterns can help prioritise conservation efforts and offer initial insights into the causes of observed changes.

By comparing January counts, we investigate spatial patterns in changes to non-breeding populations between the 1980s-1990s and the present. These changes could reflect redistribution of individuals or varying rates of

increase or decrease among sub-populations in different regions of the flyway.

2.2 Methods

2.2.1 General patterns in trends of populations

The analysis of trends is based on long- and short-term waterbird population abundance data running up to 2023, as detailed in Annex 1. This dataset includes trends for 88 populations across 9 groups of populations or species with shared ecological characteristics: grebes, pelicans, herons, flamingos, ducks, geese, shorebirds, gulls and terns. This offers a representative cross-section of waterbirds in the flyway concerning taxonomy and breeding and non-breeding regions. While the populations analysed differ partly from those in Schekkerman *et al.* (2018, 2022), there is sufficient overlap to allow for comparisons of broad-scale patterns.

Most trends are based on counts conducted through the International Waterbird Census (IWC), although data from breeding bird monitoring programs, such as the Pan-European Common Bird Monitoring Scheme (PECBMS) and European Red List assessments, were also used for some populations (see Annex 1). Trends are expressed as the average percentage change per year over the respective trend periods, with long-term trends averaging 40 years and short-term trends covering 10 years.

To summarise waterbird trends across the coastal EAF, the number of populations within each of six formal trend categories (strong increase, moderate increase, stable, uncertain, moderate decline, and strong decline) was recorded. The threshold between moderate and strong

increases or declines is defined as a 5% annual change. Populations are classified as stable if the 95% confidence interval around the trend includes 0% change but does not encompass a 5% annual change in either direction. If the confidence interval includes 5% change, the trend is categorised as 'uncertain.' Additionally, the mean annual rate of change for all populations was calculated to provide a comprehensive metric for both long- and short-term trends. The short-term-trends up to 2023, in comparison with the long-term trends, provide an indication of recent changes in population trends, and are also compared with the short-term trends in the two previous assessment reports (10 years up until 2017 and 2020) to get an indication whether the population development is improving or worsening. Trait-based patterns were explored by calculating and comparing mean annual rates of change for groups of populations (or species) with shared ecological characteristics. Details on the traits used in this analysis can be found in Schekkerman *et al.* (2018, 2022).

2.2.2 Site-based population trends of shorebirds in Africa and Europe

Schekkerman *et al.* (2022) found that shorebird populations generally declined in the southern half of 12 regions along the flyway (along the African coastlines southward from Mauritania) between the 1980s–1990s and 2010s–2020, while they generally increased in the north. Schekkerman *et al.* (2022) derived this pattern from winter totals by country in four different time periods, subsequently grouped into two periods. Using site-specific data (232 sites of which 187 in Europe and 45 in Africa), we analysed whether the long-term population trends of 16 shorebird species which occur both in Europe and Africa in January, differed between both continents. The dataset was first divided into two subsets with sites in either Europe or



Greater Flamingo | Flamant rose (*Phoenicopterus roseus*)

Hans Schekkerman

Africa, where the boundary between both was at 21.33°N because Schekkerman *et al.* (2022) identified the boundary to approximately lie between Morocco and Mauritania. This implies that 'Europe' includes northern Africa. Only species that occurred in both continents were retained for analysis to allow direct comparisons.

Because the time series on average started at different times in either continent (on average in 1985 in Europe and in 1992 in Africa), we determined the earliest year for which each continent included counts at minimally five sites and took the later of those years (1990) as the common start year. For each species and continent, we fitted generalised linear mixed models (GLMMs). The log-transformed count of individuals was the response variable modelled as a function of time (year), while including site as a random intercept to account for heterogeneity between sites. Because changes in bird numbers at a site have a larger impact on the overall flyway population if that site contains more individuals, we applied a weight to each site representing the median count relative to the highest median imputed count for that species (i.e. ranging from 0 to 1 across sites within each species). To account for overdispersion in the count data, we used a negative binomial type 2 distribution in the GLMMs. Model fitting was performed within a grouped data structure, iterating separately for each species-continent combination.

The coefficient for time (slope) was extracted from the models, representing the annual proportional rate of change of the population. Species for which trends could not be estimated in one of the continents due to insufficient data were excluded. If model convergence errors occurred, the species was also excluded from further analysis. This resulted in comparisons of population trends in Europe and Africa for nine species. All these species were also part of the analyses by Schekkerman *et al.* (2022), but model convergence issues and/or insufficient data lead to excluding Eurasian Oystercatcher, Whimbrel, Eurasian Curlew, Common Greenshank, Pied Avocet, Kentish Plover and Curlew Sandpiper from our analyses. For some species included in the analysis, different populations or subspecies are involved that occupy distinct non-breeding locations in either Africa or Europe, and distinct breeding ranges (e.g. Red Knot where the *islandica* subspecies breeds in Greenland and Canada and winters in W Europe, and the *canutus* subspecies breeds in Siberian Russia and winters in W Africa). This implies that different trends between the two continents may thus not only be the result of different conditions in the non-breeding area but may also stem from factors that differ between the breeding areas and/or (staging sites along) the migration routes. However, also for species regarded as a single flyway population, connectivity between breeding and wintering regions (especially when gradual, or clinal) may lead to differences in trends similar to those seen in populations with distinct breeding or wintering areas, but this may be less obvious.

For each continent, trends were categorised as positive if the lower confidence bound was greater than zero, negative if the upper confidence bound was below zero, and stable/unclear otherwise. To globally assess whether trends differed between continents, we compared models with the explanatory variables time and continent only with models additionally including the interaction term between time and continent, based on the Akaike's Information Criterion (AIC; Burnham & Anderson 2004). To assess whether trends were different from 0 (i.e. a positive or negative population trend) and whether slopes differed between continents within species, we determined whether GLMMs (as described above) that included time (i.e. modelling changes in abundance over time) had an AIC at least 2 points lower than the intercept-only model. Within each species the slopes were considered to differ between continents if a model that included an interaction between time and continent had an AIC at least 2 points lower than a model with only additive effects of time and continent. All statistical analyses were performed in R (version 4.4.1).

2.3 Results

2.3.1 General summary of trends

Over the long term, most of the 88 waterbird populations analysed exhibited fairly favourable trends, with 42% showing moderate or strong increases and 25% remaining stable or uncertain (Fig. 2.1). However, declining populations accounted for 33%, luckily only including only one population in the unfavourable category of 'strong decline' (the *schinzii* population of Dunlin, Fig. 2.2). The overall mean annual rate of long-term change was +0.6% per year (95% CI: -0.009 to +1.25% per year), and can thus be considered stable, in contrast to the long-term trend in the previous report (Schekkerman *et al.* 2022) when the overall mean was significantly positive.

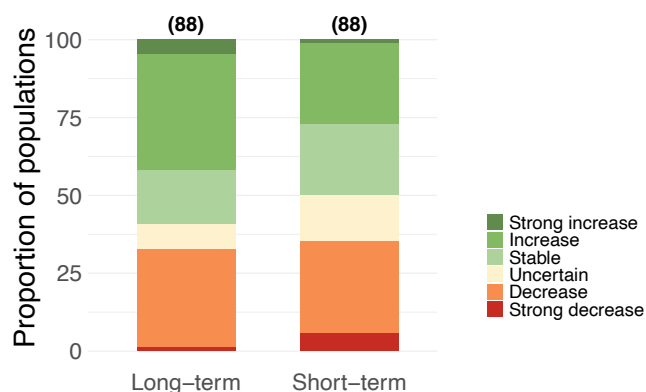


Figure 2.1 The proportion of waterbird populations that decreased, increased or remained stable in the long-term (leftmost bar) or in the short-term (i.e. a period of 10 years up to 2023). The sample size (number of populations) is shown between brackets above each bar.

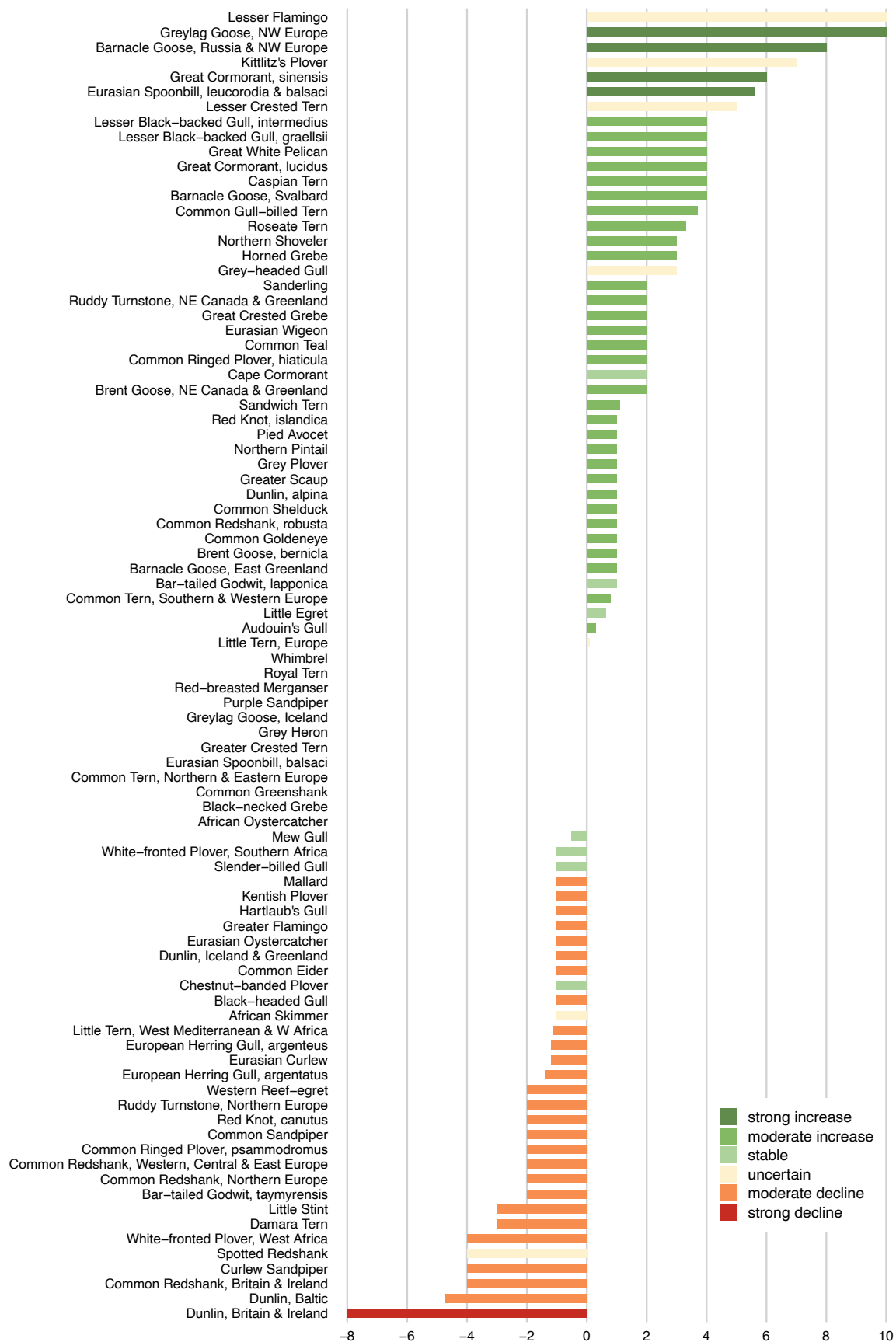


Figure 2.2. The long-term annual population growth rate (%) for each waterbird population considered up to 2023

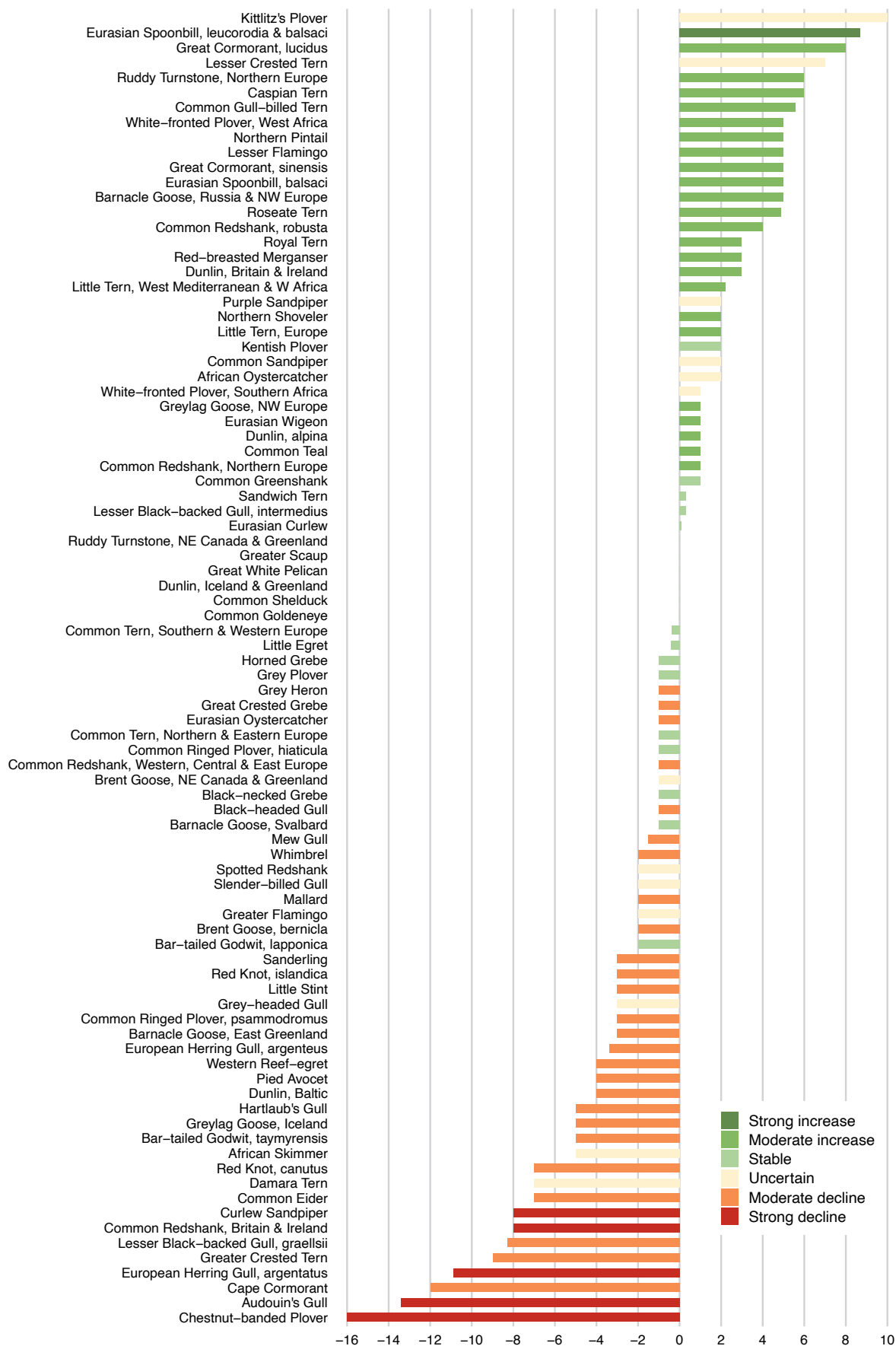


Figure 2.3. The short-term (2014-2023) annual population growth rate (%) for each waterbird population considered.

In the short term, during the last ten years up to 2023, 27% of populations increased, 37% remained stable or unclear, and 35% declined, including 6% in strong decline (Fig. 2.1). The overall mean short-term trend up to 2023 was -0.76% per year (95% CI: -1.76 to 0.23% per year), which is worse than the long-term trend but can still be considered stable. The mean short-term trend in the previous report was with -0.35% per year slightly more beneficial than the current short-term trend. The gradual shift in short-term trends could indicate that a change has taken place and population trends have on average started to decrease.

2.3.2 Associations between population trends and taxonomy and breeding area

In previous flyway assessment reports, Schekkerman *et al.* (2018, 2022) identified taxonomic signals in population trends among waterbird populations in the coastal EAF. Waders (Charadrii), especially those breeding in the Siberian Arctic, exhibited the most negative trends on average, with more pronounced declines in the short term. In contrast, geese, flamingos, pelicans, and cormorants showed the most favourable trends. Waterbirds that rely on intertidal mudflats and benthic food outside the breeding season, many of which are shorebird species, had less favourable trends compared to those feeding on plant material or fish. Regarding breeding climate regions, populations breeding in the Arctic and boreal regions, as well

as those in Southern Africa, exhibited particularly unfavourable short-term trends.

With the trend data updated to 2023, these patterns remain largely unchanged. Shorebirds continue to show among the least favourable trends, while gulls (Laridae) have seen a further decline in the short term (Fig. 2.4). In contrast, all other taxonomic groups show (slightly) positive long-term trends, with geese, pelicans, cormorants and flamingos still showing the most positive trends. Except for terns and shorebirds, the median short-term population trends are more negative than the median long-term trends (Fig. 2.4), again indicating that the situation is becoming less favourable for waterbird populations along the coastal EAF. If these trends persist, it is likely that also the long-term trends will become negative for a growing number of populations. It is unclear why gulls are showing such unfavourable trends over the last 10 years.

The overall pattern with respect to breeding region remains consistent with previous findings, with the most unfavorable short-term trends observed in populations breeding in the Arctic and Southern Africa (Fig. 2.5). In the Arctic, shorebird populations have shown the strongest declines among waterbird groups. It is difficult to draw definite conclusions about the cause of such declines, because it is not impossible that the cause of declining Arctic shorebird populations lies outside the arctic breeding area, but it is certainly not unthinkable that the rapid



Red Knot | Bécasseau maubèche (*Calidris canutus*)

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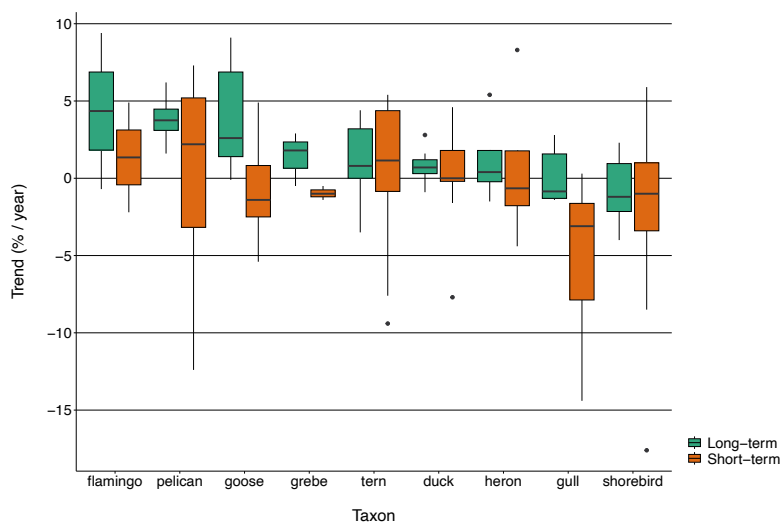


Figure 2.4. Long-term (green) and short-term (red) trends, presented as average annual change, across populations within each of nine taxonomic groups (bird families or orders) in the coastal EAF. The boxes indicate the central 50% of the data (interquartile range, IQR) with the median as a horizontal line, whiskers extend to 1.5×IQR, and dots denote observations outside that range. Groups are ordered by decreasing long-term trend values.

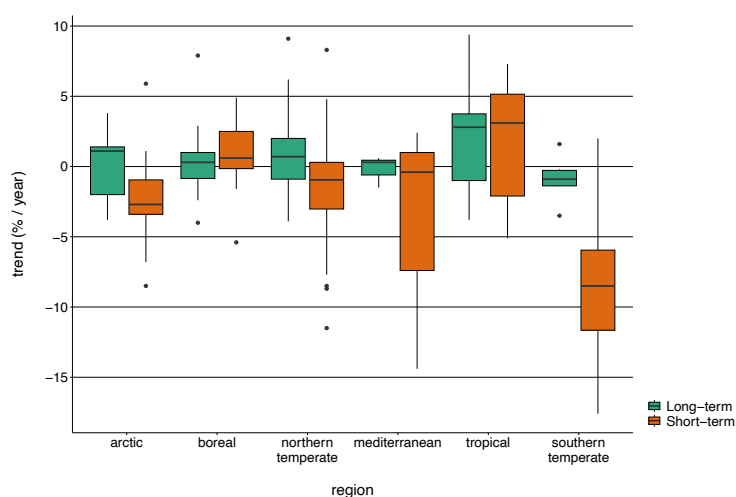


Figure 2.5. Average long-term (green) and short-term (red) trends of populations in the coastal EAF characterised by their breeding climate zone: arctic, boreal, north-temperate, Mediterranean, tropical (West and equatorial Africa) and south-temperate (southern Africa). Conventions as in Figure 2.4.

warming of the arctic region (Box *et al.* 2019) plays a role for example by negatively influencing reproduction, e.g. through disruption of lemming cycles (Nolet *et al.* 2013) which has reduced the frequency of years with high reproductive success when predators focus on lemmings rather than bird eggs and chicks. Climate warming may also reduce food availability for chicks due to phenological mismatches with invertebrate food sources (e.g. Lameris *et al.* 2018, 2022, Versluijs *et al.* 2024). Still, the evidence for both the disruption of lemming cycles (Ehrich *et al.* 2020, Gauthier *et al.* 2024) and phenological mismatches (Reneerkens *et al.* 2016, Schmidt *et al.* 2023, Zhemchuzhnikov *et al.* 2021) is not unequivocal, and their actual effect on flyway population trends are unknown.

2.3.3 Do wader populations decline in Africa and increase in Europe?

An analysis by Schekkerman *et al.* (2022) of changes in January numbers of shorebirds by region, comparing the 1980s–1990s with the 2010s–2020s, revealed a latitudinal gradient along the coastal EAF, with more negative trends

observed in the south, especially in Africa, and more positive trends in the north. This latitudinal dichotomy was evident in both shorebird species with a single flyway population with a non-breeding range across a large range of the EAF, and in species with multiple recognized flyway populations where populations spending the non-breeding season in Africa mostly showed declines, whereas those wintering primarily in Europe tend to show positive changes.

We tested for 9 shorebird species whether trends were more positive in Europe (including N Africa) compared with Africa, using site-specific data from 1990–2023. The model with interaction terms between time and continent as fixed effects and species and site as random intercept received more support than the model with time and continent as additive fixed effects ($\Delta\text{AIC} = 192.351$), suggesting that allowing slopes to differ between continents provides a much better model fit overall. This means that across all species, there is evidence that population trends differ between continents, even though their numbers (intercept) can still differ between continents. In agreement with the

Table 2.1. Trend slope estimates and 95% confidence intervals extracted from a generalised linear mixed model (GLMM) with negative binomial distribution of changes in abundance over time with site as a random intercept, of seven shorebird species in Africa and Europe. For each species–continent combination, a slope is considered significant if the GLMM that includes time has an AIC at least 2 points lower than the intercept-only model. Significant slopes are displayed in green when they indicate an increase in abundance and in red when they indicate a decline. In addition, within each species the slopes between continents are considered to differ if a model that includes an interaction between time and continent has an AIC at least 2 points lower than a model with only additive effects of time and continent. Due to the log-link function used by the negative binomial distribution in the GLMM, the slope estimate from the model represents a proportional change in the expected number of birds for each year.

species	Africa slope (95% CI)	Europe slope (95% CI)	different slopes
Grey Plover	-0.0226 (-0.0229 – -0.0222)	0.0102 (0.0099 – 0.0104)	Yes
Common Ringed Plover	-0.0074 (-0.0077 – 0.0071)	-0.0073 (-0.0082 – -0.0065)	No
Bar-tailed Godwit	-0.0277 (-0.0278 – -0.0276)	0.0345 (0.0341 – 0.0350)	Yes
Red Knot	-0.0270 (-0.0272 – -0.0269)	-0.0012 (-0.0014 – -0.0010)	Yes
Ruddy Turnstone	0.0028 (0.0021 – 0.0035)	0.0342 (0.0335 – 0.0350)	Yes
Sanderling	0.0177 (0.0174 – 0.0181)	0.0255 (0.0247 – 0.0263)	Yes
Dunlin	-0.0092 (-0.0093 – -0.0091)	0.0145 (-0.0144 – 0.0147)	Yes
Common Redshank	-0.025 (-0.026 – -0.025)	0.0055 (0.0046 – 0.0063)	Yes

finding by Schekkerman *et al.* (2022), four of the nine species showed negative trends in Africa and positive trends in Europe. For the two species (Sanderling and Ruddy Turnstone) with positive trends in both continents, the trends were still considerably more positive in Europe, while Red Knots decreased in both continents but much more so in Africa. Only Common Ringed Plover showed an equally

negative trend in both continents (Table 2.1, Fig. 2.6). These findings are in agreement with the finding by Schekkerman *et al.* (2022) that long-term population trends of shorebirds were more favourable towards the north.

We note that by taking site-level variation in trends into account, weighing site-effects by scaling their relative

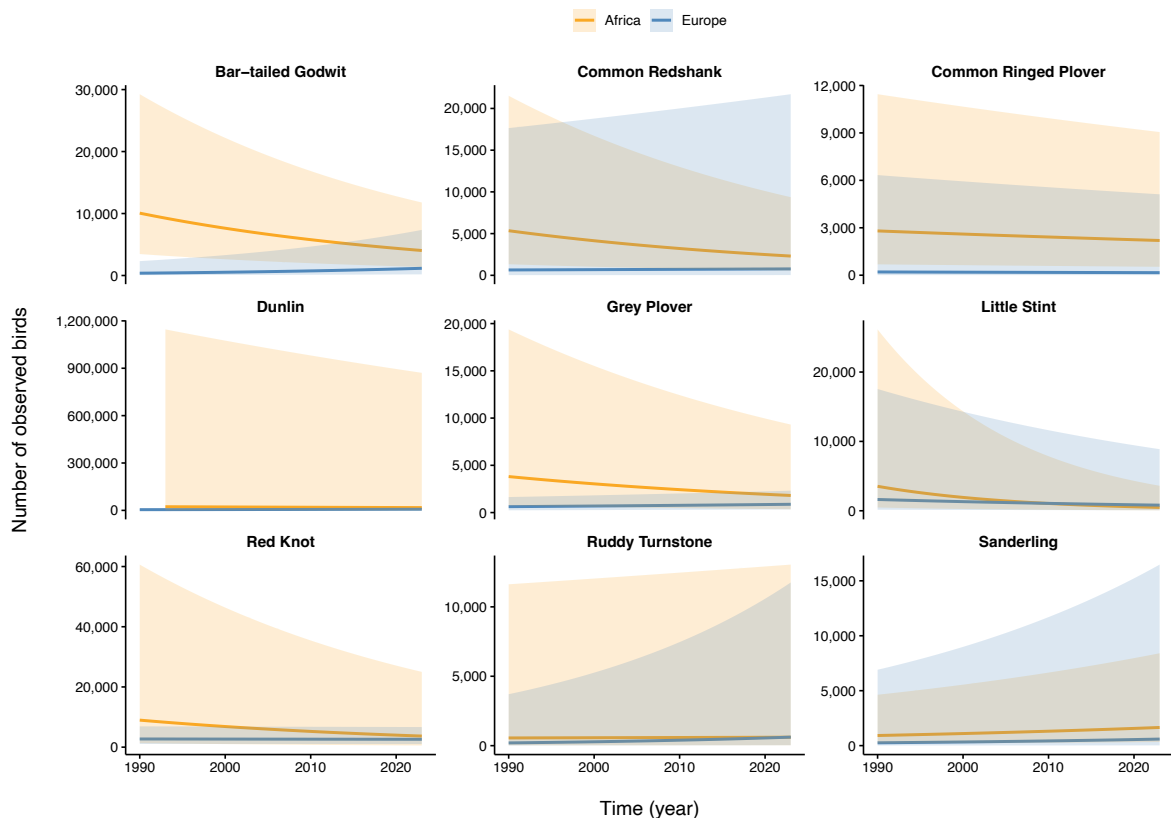


Figure 2.6. Model predictions from the GLMM for temporal trends in the numbers of nine shorebird species across sites in Africa (orange) and Europe (blue), with the temporal trends for each species at a large number of sites. Site was included in the models as a random effect. 95% confidence intervals are indicated by the shaded areas.

contribution to the trends and by correcting for overdispersion in the models, trends (1) may differ from the species-specific trends presented elsewhere in this report, (2) could only be estimated for both continents for nine shorebird species and (3) come with very large confidence intervals. Even though the time periods for analyses were adjusted to ensure counts at sufficient sites in either continent, the number of observations and sites in the analysed data for the nine shorebird species are considerably lower in Africa compared with Europe, except for Little Stint (Table 2.2). Especially for Dunlin, and to a lesser extent for Red Knot, the number of sites in Africa were much lower than in Europe (Table 2.2) and will have contributed to the large confidence intervals for both species in Africa (Table 2.1, Fig. 2.6).

Grey Plover, Ruddy Turnstone, Little Stint and Sanderling concern single flyway populations and the different trends in Africa and Europe may thus be the result of a lowered suitability of habitat in Africa or increased risk of migration of birds wintering in Africa compared to Europe. However, we can not rule out that different conditions in breeding areas play a role if birds spending the non-breeding season in Africa tend to breed elsewhere (e.g. more to the East) than those wintering in Europe, even though the whole area is considered one single flyway. Insights in population boundaries and migratory connectivity will be essential to better interpret geographical variation in population trends (e.g. see chapter 11). At present, we can only speculate what explains the dichotomy in population trends between the northern and southern part of the flyway. For Sanderlings there are indications that the survival probabilities in W Africa, but not in southern Africa, were considerably lower than in Europe (Reneerkens *et al.* 2020) and it was suggested that this may be due to suboptimal conditions to fuel up for northward migration.

Table 2.2. The number of observations (i.e. years with counts at a site) and sites for each of the nine shorebird species included in the analyses.

species	observations Africa	sites Africa	observations Europe	sites Europe
Bar-tailed Godwit	102	14	2094	68
Common Redshank	102	14	1218	41
Common Ringed Plover	236	20	3507	124
Dunlin	44	3	3113	100
Grey Plover	72	13	2381	76
Little Stint	200	17	144	5
Red Knot	34	7	1667	52
Ruddy Turnstone	75	11	1618	55
Sanderling	126	16	1986	68

2.4 Conclusions

Our analyses suggests that while population trends of coastal waterbirds along the coastal East Atlantic Flyway remain relatively stable, the increasing short-term declines are a cause for concern. Compared to other global migratory waterbird flyways, trends in this region are still more favourable, with a higher proportion of increasing populations (e.g. Nagy & Langendoen 2020). However, if short-term declines continue, long-term trends may also turn negative, raising conservation concerns. Shorebirds have shown the most significant declines, particularly in Africa, though the causes remain unclear and may vary by species. Continued monitoring, especially in Africa where survey coverage is far lower than in Europe, is essential. This should be combined with detailed ecological research across the entire flyway.



Hans Schekkerman

Sanderling, Common Ringed Plover & Dunlin | Bécasseau sanderling, Pluvier grand-gravelot & Bécasseau variable

(*Calidris alba*, *Charadrius hiaticula* & *Calidris alpina*)



Red Knot | Bécasseau maubèche (*Calidris canutus*) (Ralph Martin /Agami)

3. Pressures and conservation measures at coastal sites used by waterbirds along the East Atlantic Flyway, update 2023

Aïssatou Yvette Diallo, Tim Dodman, Fred Barasa, Geoffroy Citegetse, Kristine Meise, Jaime Garcia-Moreno, Olivia Crowe & Marc van Roomen

Summary

Waterbirds and their habitats face numerous threats along the East Atlantic Flyway, including at the large network of coastal wetlands. This chapter attempts to assess the pressures exerted on them and the effectiveness of conservation measures. This monitoring is based on data from 119 sites along the flyway, 30 in Europe and 89 in Africa. Fishing was assessed as a significant pressure at most sites across the flyway, and represents a major threat to seabirds in particular. Buildings, recreation/tourism, roads and hunting were also assessed as significant pressures. Climate change can disrupt or modify the migration cycles of waterbirds and cause the loss of critical habitats. Avian influenza impacted several waterbird species in Europe and West Africa between 2021 and 2023. Concerted efforts by scientists, governments, site managers and civil society can mitigate negative impacts at sites along the flyway, and help to conserve birds and their habitats. Importantly, implementing protection measures at already designated sites needs to improve, and effective people engagement is vital for achieving success. Conserving birds and their habitats is an ecological, economic and cultural investment for future generations, and depends on everyone.

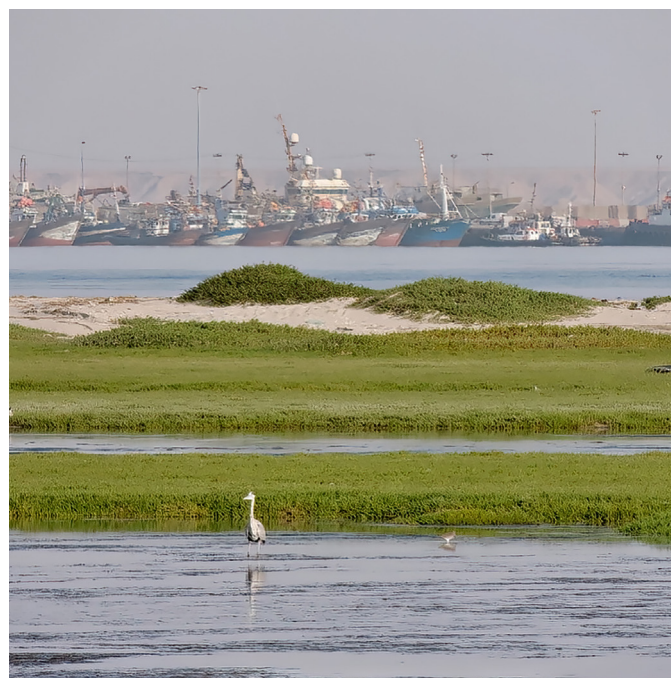
Résumé

Les oiseaux d'eau et leurs habitats font face à plusieurs menaces le long de la voie de migration de l'Atlantique Est ; y compris dans le vaste réseau de zones humides côtières. Ce chapitre tente d'évaluer les pressions exercées sur eux et l'efficacité des mesures de conservation. Ce suivi est basé sur des données provenant de 119 sites le long de la voie de migration, 30 en Europe et 89 en Afrique. La pêche a été évaluée comme une pression importante dans la plupart des sites de la voie de migration et représente une menace majeure pour les oiseaux de mer en particulier. Les bâtiments, les loisirs/tourisme, les routes et la chasse ont également été considérés comme des pressions importantes. Le changement climatique peut perturber ou modifier les cycles de migration des oiseaux d'eau et entraîner la perte d'habitats essentiels. La grippe aviaire a affecté plusieurs espèces d'oiseaux d'eau en Europe et en Afrique de

l'Ouest entre 2021 et 2023. Les efforts concertés des scientifiques, des gouvernements, des gestionnaires de sites et de la société civile peuvent atténuer les impacts négatifs sur les sites situés le long de la voie de migration et contribuer à la conservation des oiseaux et de leurs habitats. Il est important d'améliorer la mise en œuvre des mesures de protection sur les sites déjà désignés, et l'engagement efficace de la population est essentiel pour réussir. La conservation des oiseaux et de leurs habitats est un investissement écologique, économique et culturel pour les générations futures, et cela dépend de chacun.

3.1 Introduction

Human pressures such as the loss and degradation of habitats, pollution (plastic, light, sound, wastewater, pesticides, etc.), hunting and infrastructure impact bird populations, which are also susceptible to climate change.



Tim Dodman

Large fishing boats at the Baie de Dakhla, southern Morocco, a key passage site for migratory waterbirds

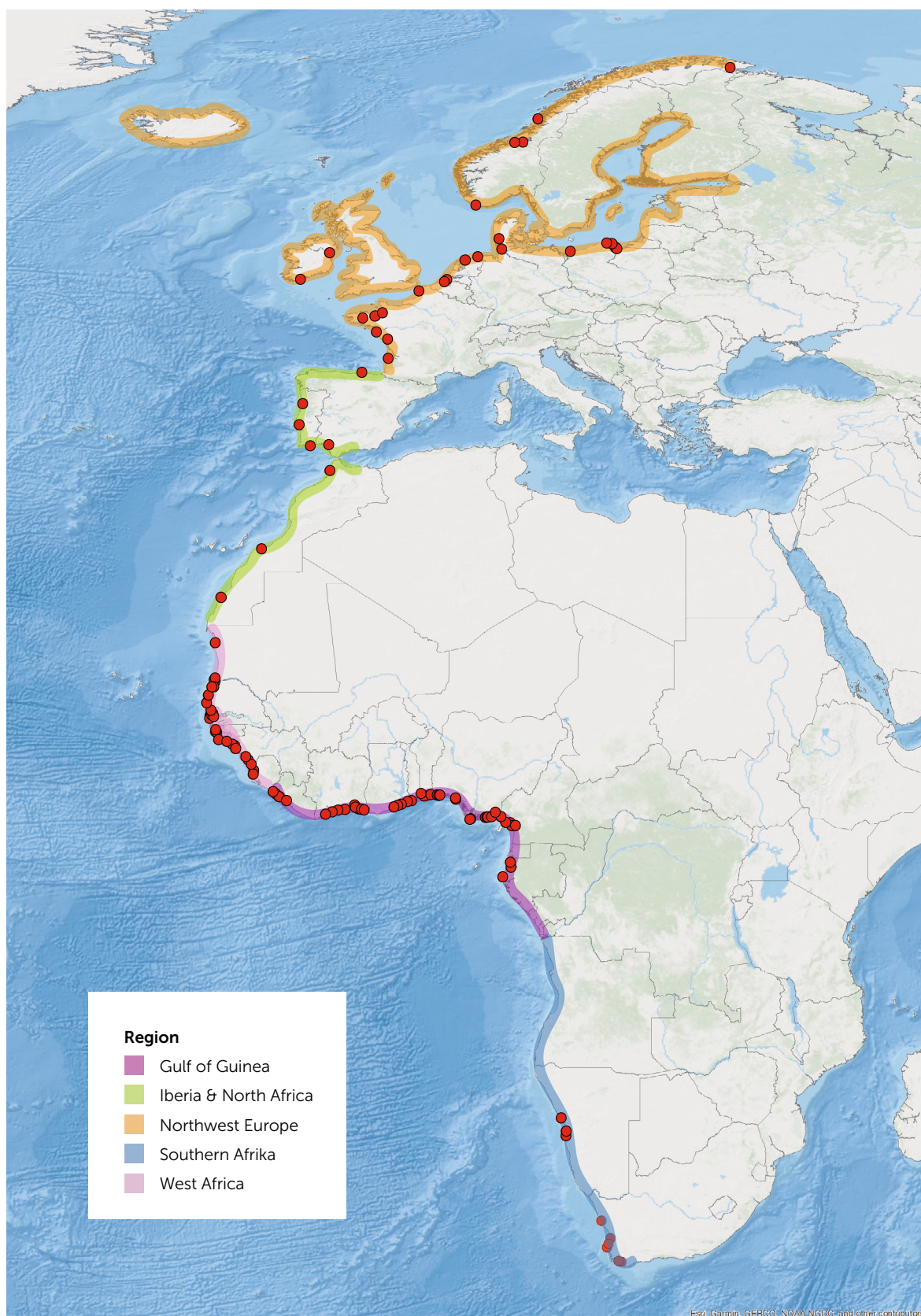


Figure 3.1. The coastal East Atlantic Flyway showing the 119 sites for which environmental monitoring assessments were included in the analyses.

Along the East Atlantic Flyway (EAF), these pressures need to be addressed through appropriate conservation measures to protect habitats for waterbirds, including improved management of sites, especially coastal wetlands, taking measures to minimise key threats such as pollution, raising awareness and environmental education. Monitoring sites and the pressures they face are important steps in the management cycle, underpinned by national and international cooperation.

This chapter summarizes the pressures considered by a large network of contributors to be impacting coastal sites used by migratory waterbirds and the conservation measures adopted to address them along the EAF. It builds upon the principles of Important Bird and Biodiversity Area (IBA) monitoring by scoring pressures and effectiveness of conservation measures at sites (BirdLife International 2006). The assessment presents the results from 2023, building on previous assessments of 2017 (Dodman *et al.* 2018) and 2020 (Crowe *et al.* 2022). Results presented are based primarily on questionnaires completed along the flyway by International Waterbird Census (IWC) national coordinators, site managers and a network of observers.

3.2 Materials and Methods

Environmental monitoring data with information on pressures and conservation measures relating to waterbirds and their sites were obtained using an evaluation questionnaire in English and French (created using Jotform, an online application for creating and publishing forms and collecting responses) sent to specific users, notably IWC national coordinators, for completion. The Jotform content was based on conventional questionnaires as introduced in van Roomen *et al.* (2014) and subsequently used and adapted by van Roomen *et al.* (2015), Dodman *et al.* (2018) and Crowe *et al.* (2022) to record information for each site: basic data of the site and persons providing information, natural characteristics, human uses and pressures and the extent of their impacts, and conservation measures taken.

Data were obtained from 162 sites (34 in Europe, 128 in Africa) along the flyway, but the environmental monitoring assessment is based on data from 119 sites (30 sites in Europe and 89 in Africa). Inland sites were omitted and for some countries with many small sites, the number of sites used were reduced. The distribution of these 119 sites by region and country is shown in Fig. 3.1 and summarised below, with number of sites assessed per country in brackets:

- Northwest Europe: Denmark (1), France (7), Germany (2), Ireland (2), Netherlands (3), Norway (5), Poland (5).
- Iberia - North Africa: Morocco (3), Portugal (3), Spain (2).
- West Africa: Guinea (6), Guinea-Bissau (5), Mauritania (4), Senegal (5), Sierra Leone (2), The Gambia (7).
- Gulf of Guinea: Benin (6), Cameroon (5), Côte d'Ivoire

(10), Gabon (3), Ghana (5), Liberia (5), Nigeria (14).

- Southern Africa: Namibia (3), South Africa (6).

For these analyses, data were extracted at two spatial scales: across the whole flyway, and at five regional levels. Analyses are mostly presented as frequencies (percentages) of the answered questions and ordered from most to less frequent (Figs. 3.2, 3.5, 3.6). A description of environmental monitoring assessments, and how data were treated for the analyses, is provided in Dodman *et al.* (2018) and Crowe *et al.* (2022).

In addition, trends of waterbird populations with at least four years of count data in January, were reviewed for 132 sites along the flyway that supported >0.1% of the flyway population of at least eight species during January counts. Linear trends of either increasing, stable/fluctuating or decreasing trend were calculated for each qualifying population at each site. The results were combined for all species per site to show the absolute numbers and the percentage of populations that were increasing, stable/fluctuating or decreasing. The number of populations decreasing or increasing at greater rates than the flyway trends were also calculated. The aim was to use waterbird count data to assess how sites were faring in terms of quality for birds, with the expectation that decreases in bird populations might highlight sites where conditions were not positive, and vice versa. Of these 132 sites, 36 were included in the environmental monitoring analysis, for which the sum of the different threat scores following IBA methodology was also compared with the percentage of decreasing trends.

3.3 Principal pressures and conservation measures along the EAF

3.3.1 Principal pressures to waterbirds and their sites along the EAF

As birds generally occupy higher trophic levels in food webs and are sensitive to environmental change, their population trends often reflect those of other taxa, and can act as early warning systems of threats to nature (BirdLife International 2022). Unsustainable consumption and economic development are the underlying drivers of most threats to the natural world (Marques *et al.* 2019). The growing demand for food and timber drives expansion of agriculture, fisheries and forestry, whilst urbanisation drives development of residential and commercial areas and related infrastructure. The combination of these and other pressures along the EAF present significant threats to waterbirds.

In 2023, fishing, buildings and recreation/tourism were assessed as the most frequent pressures along the EAF (Fig. 3.2). Fishing, especially bycatch, is a major pressure on sea-

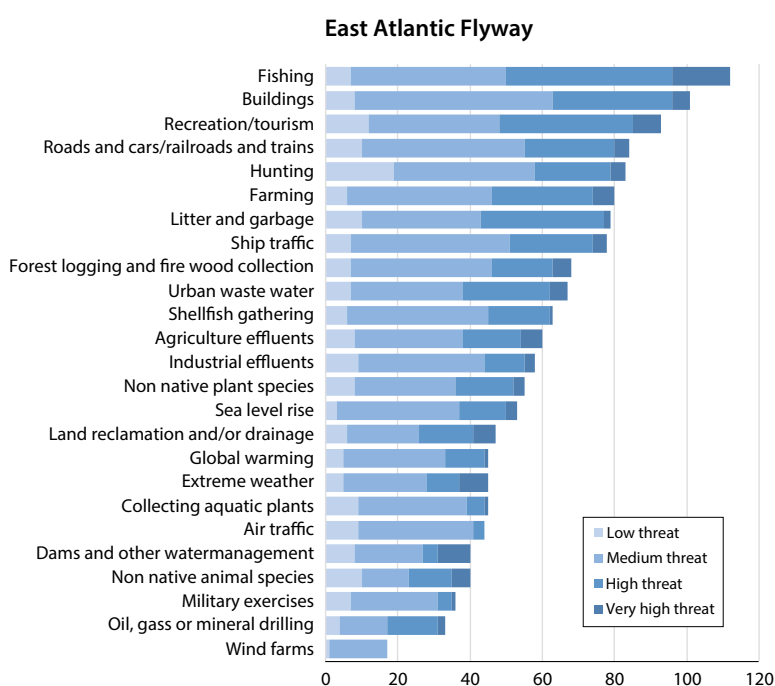
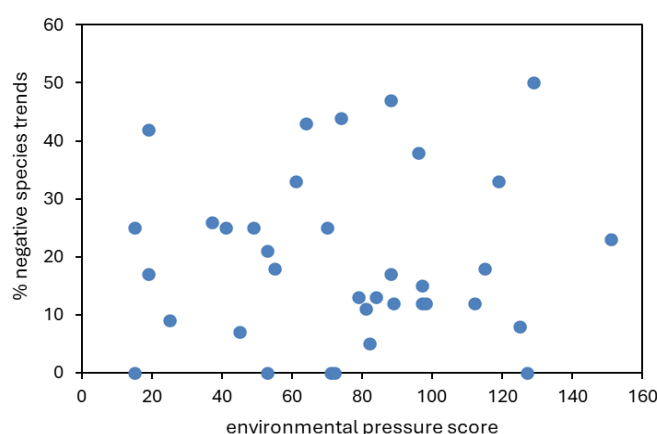


Figure 3.2. Frequencies of pressures reported from assessed sites along the EAF as having a low, medium, high or very high threat (ordered on the frequency of very high threat).



birds, one of the most threatened groups of birds in the world (BirdLife International 2022). Roads and cars were also ranked as high pressure. On land, the majority of Key Biodiversity Areas (KBAs) contain infrastructure and roads, increasing their accessibility, invariably facilitating urban development associated with pollution and disturbance. Further pressures are discussed at the regional level.

Based on January counts, 132 sites along the flyway were found to support the 0.1% threshold for a minimum of eight species/populations and for at least four years with count data; 36 of these sites were included in the environmental monitoring analysis. The transboundary Senegal Delta supported the highest number: 39 qualifying species. There were wide-ranging rates of increasing species trends, from 0-100% in northern Europe and Iberia - Morocco, from 0-72% in West Africa and from 0-75% in

Southern Africa. There were only two qualifying sites in the Gulf of Guinea. Declines along the flyway ranged from 0-55%. Highest declines were noted at Lindisfarne (55%) and Langstone Harbour (50%) in the UK, Olifants River Mouth, South Africa (50%), Keta Lagoon complex, Ghana (47%), Rio Kompony, Guinea (44%) and Walvis Bay, Namibia (43%). However, when comparing the sum of environmental pressures and percent population decreases for 36 sites, no correlation was found between the extent of site pressures and decreases in bird populations (Fig. 3.3). More specific information needs to be taken into account, such as the species of birds in decline and their habitat use at each site, and further analyses conducted before reasons for declines may be postulated.



Tim Dodman

Local fishing boats in Saint-Louis, Senegal, showing nets of low mesh size, which can be a source of bird bycatch

3.3.2 Conservation measures along the EAF

Many key sites along the EAF are protected areas, forming a fundamental pillar of flyway conservation. Protection and effective management of key sites for biodiversity (IBAs/KBAs) enable the conservation of species and the ecosystems of which they are a part and the services they provide (BirdLife International 2022). In the environmental monitoring assessment forms, 31 conservation measures were listed, mostly focussed on site protection, management and regulations (Fig. 3.4). However, their implementation was found not to be widely effective. Along the EAF, only 30% of sites were assessed as being effectively protected at international or national level, and management plans exist for only 25% of sites.

Regulations of most human activities recorded along the EAF were assessed at less than 25% effective (Fig. 3.4). Strengthening environmental regulations and advocating for climate change mitigation and adaptive management are crucial for long-term conservation success. The average score for effective habitat management and conservation measures of sites along the EAF was less than 10%, indicating a strong need for improvement. Rehabilitating impacted wetlands and limiting development near key sites are important steps to help ensure that coastal wetlands meet the needs for migratory birds. Effective efforts to engage people at sites were reported at 17%, with an additional 32% of sites recording some effectiveness of people engagement (Fig. 3.4).

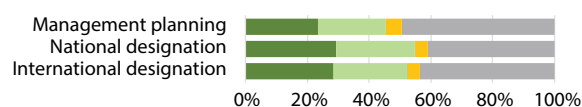
Overall, conservation measures across the EAF were assessed at lower levels of effectiveness than in 2020. It is unclear if this is due to genuine reductions in effectiveness or a change in assessment scales of site threat recorders, or a combination of both. It will be important to reflect on such variations in the future, and to consider why some measures are deemed not to apply for sites, especially for aspects such as protection and people management, which are likely to apply to the majority of sites.



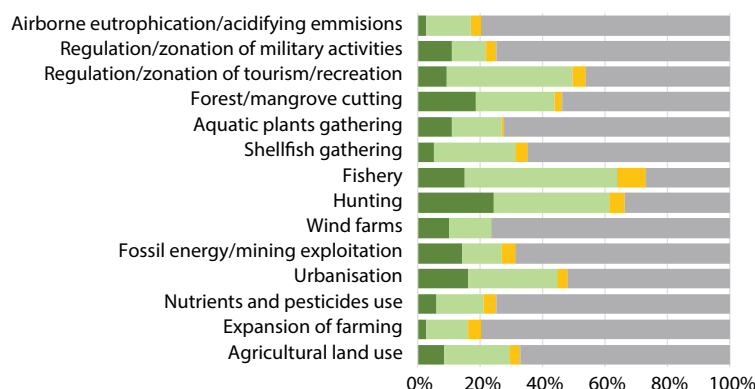
Tim Dodman

Field training at Cap Blanc, Mauritania

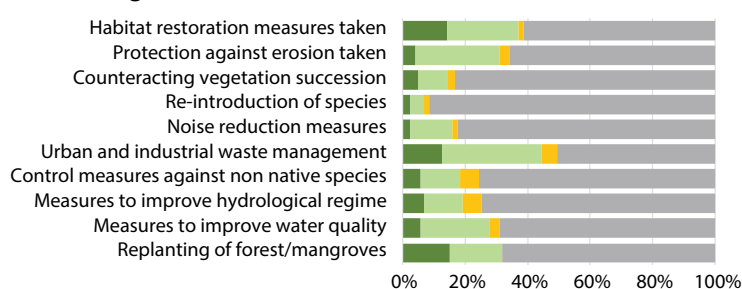
Protection and management



Regulations



Habitat management and conservation



People engagement

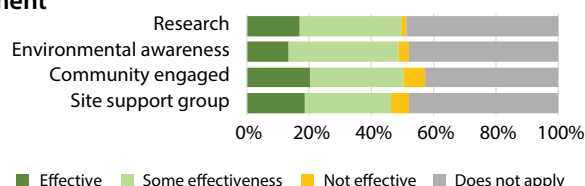


Figure 3.4. Conservation measures along the EAF, illustrating the proportion of sites where they are effective, have some effect, are not effective, and where they do not apply.

3.4 Principal pressures and conservation measures per region

The principal pressures and conservation measures resulting from the assessment are presented per region below. These are supported by regional graphs showing the top 25 pressures reported for participating sites (Fig. 3.5) and the conservation measures, illustrating the proportion of sites where measures are considered effective, of some effect or not effective, and where they do not apply (Fig. 3.6).

3.4.1 Northwest Europe: pressures

Fishing was reported from all 25 coastal sites assessed between Norway and France (Fig. 3.5). Its impacts could be direct, as a result of bycatch or disturbance, or indirect through prey depletion and habitat degradation. The scale of direct impacts has often been described as species-spe-

cific and influenced by habitat and the type of fishing gear. For example, bycatch in gillnets or trawl nets leads to injury or death of seabirds and diving waterbirds (Žydelis *et al.* 2009). Disturbance from fishing vessels may displace birds from important feeding or roosting areas (Fliessbach *et al.* 2019). Additionally, fishing practices like bottom trawling degrade critical feeding habitats, while overfishing depletes prey species. Sandeel fishing in the North Sea has raised concerns as it leads to reduced food availability for seabirds (Furness & Tasker 2000). The UK closed sandeel fisheries in specific areas to protect breeding seabird populations. However, there is an ongoing debate over the economic impact of such closures on the fishing industry and local livelihoods, highlighting the need to define effective conservation measures alongside sustainable fishing practices.

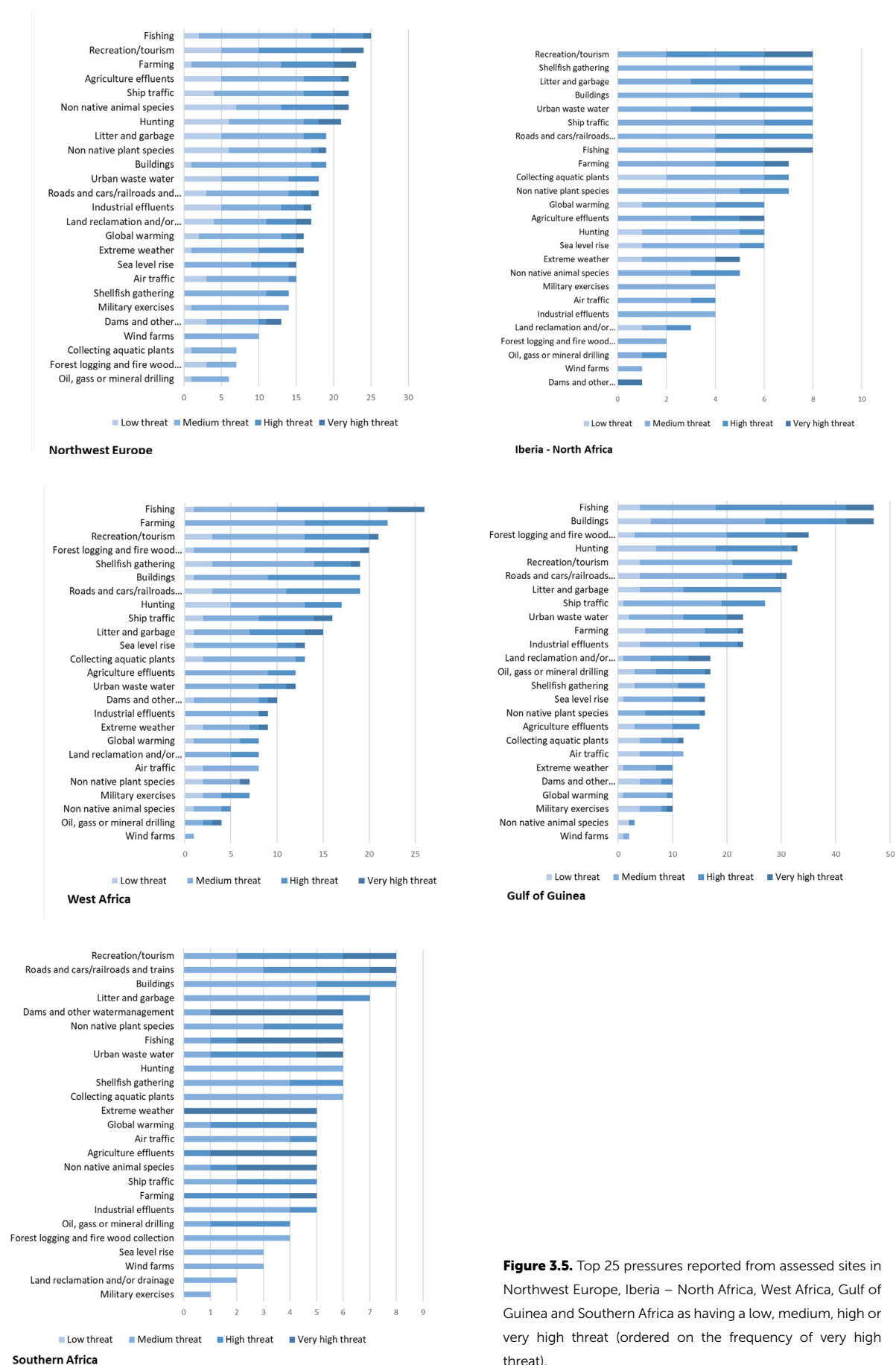
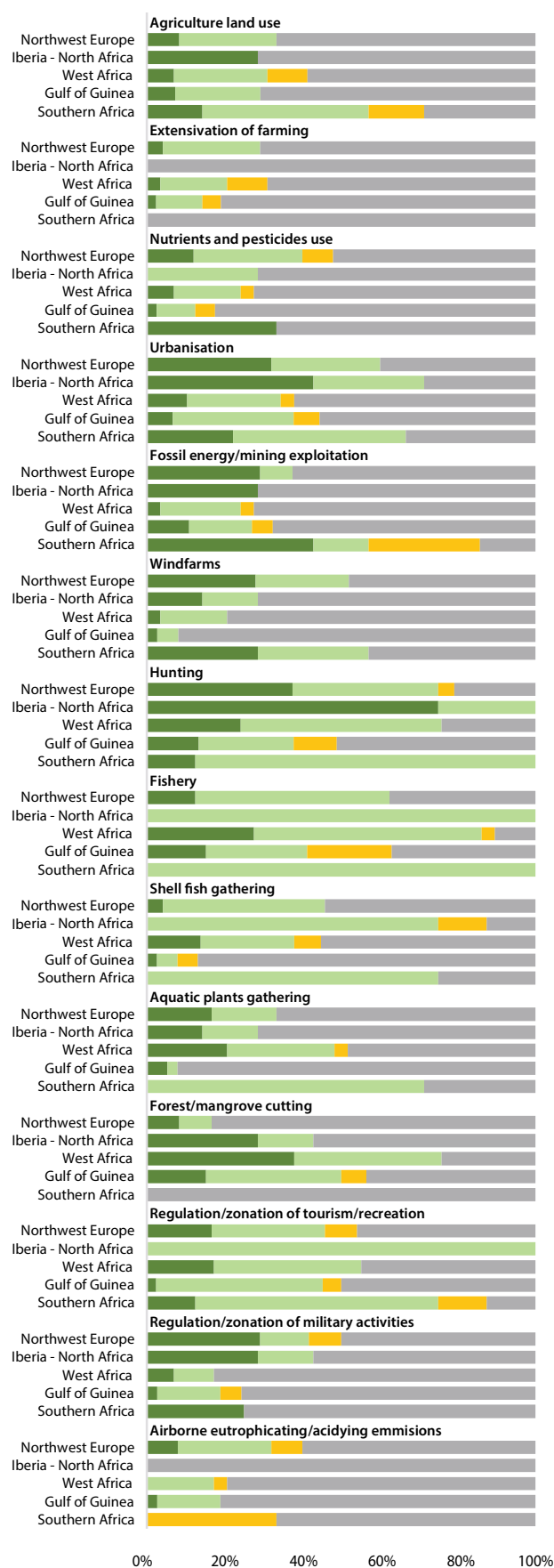
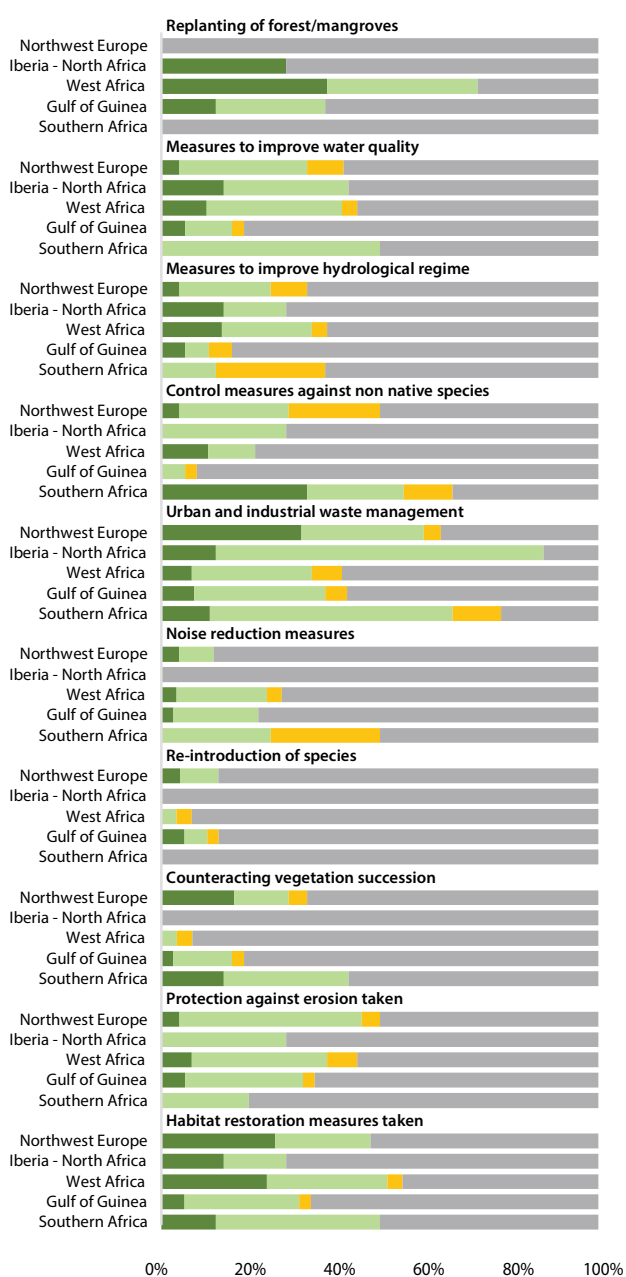
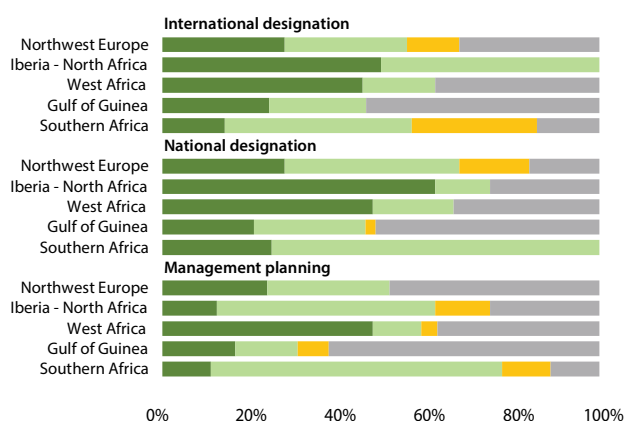


Figure 3.5. Top 25 pressures reported from assessed sites in Northwest Europe, Iberia – North Africa, West Africa, Gulf of Guinea and Southern Africa as having a low, medium, high or very high threat (ordered on the frequency of very high threat).



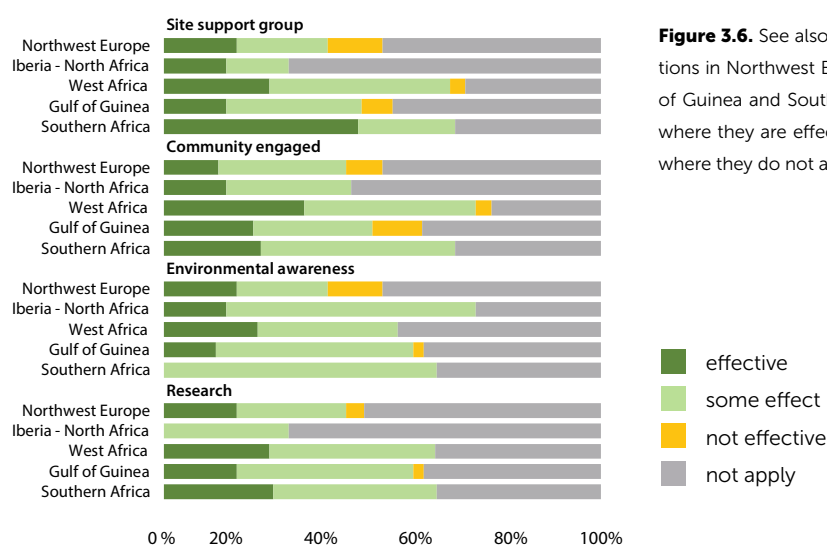


Figure 3.6. See also page 34. Conservation measures and regulations in Northwest Europe, Iberia – North Africa, West Africa, Gulf of Guinea and Southern Africa, illustrating the proportion of sites where they are effective, have some effect, are not effective, and where they do not apply.

Recreation/tourism was the second most prevalent pressure in NW Europe, named for all but one site (Fig. 3.5), and the extent of impact was higher than other pressures; high or very high impacts were reported for nearly 60% of sites. Tourism is a key economic factor in this region, and many coastal wetlands in proximity to coastal conurbations are used for recreational activities. Breeding and roosting birds, sensitive to human disturbance, are frequently flushed by unaware visitors, leading to increased energy demand or lower reproductive success (Steven *et al.* 2011).

Coastal areas close to large settlements are subject to a range of other pressures. Ship traffic to major harbours was among the five key pressures reported (22 sites), leading to disturbance of birds at feeding and roosting sites. Buildings (19 sites), roads and large infrastructure developments (18 sites) can lead to loss of habitat. Pressures from litter and garbage (19 sites), urban wastewater (18 sites), and industrial effluents (17 sites) were also reported. If these human activities are not managed appropriately, they can have considerable impact on migratory waterbirds.

Pressures linked to climate change (global warming, sea level rise and extreme weather events) were individually reported from 15-16 sites. However, the prevalence of these pressures and the extent of their impacts are expected to increase. Storm surges that flood breeding or roosting sites are already frequently reported, and loss of breeding habitat due to sea level rise is expected in many coastal wetlands, leading to lower reproductive success (Koffijberg *et al.* 2022; van de Pol *et al.* 2024). Rising sea levels will also impact the spatial and temporal availability of essential food sources, especially for migratory waders (Laursen *et al.* 2023). The development of hard infrastructure for coastal protection can further lead to the loss of important habitats. The effect of rising temperatures is

expected to be more indirect, affecting migration patterns, reproduction, distribution and abundance of prey species, with invasive species often replacing endemic species.

Wind farms were reported as a pressure at 10 sites. However, due to increased demand for green energy many large-scale infrastructure projects are planned, e.g. in the North Sea, and this pressure will likely become more frequent in the future. A strategic approach is required to ensure that the needs of migratory birds are considered in future development plans.



The industrial port of Esbjerg, Denmark north of the Wadden Sea World Heritage Site

3.4.2 Northwest Europe: conservation measures

More than 60% of the 25 assessed sites have a designated protection status considered to have at least some effectiveness (Fig. 3.6). This is likely linked to the fact that many sites have management plans in place as a result of specific EU and national regulations. However, the effective implementation of protection measures needs to be further improved.

Regulations exist in NW Europe for the most prominent threats. Implementation of regulations linked to fisheries and urbanisation were considered to be at least somewhat effective (>60%). Many steps are being taken for urban and industrial waste management and were mostly deemed to have some effectiveness (>60%). Strict regulations are also implemented for projects linked to offshore developments such as windfarms, mining and fossil energy extraction; 50% of sites with developments reported effective regulations.

Regulations linked to recreational and military activities were considered effective at 60% and ineffective at 10% of relevant sites. Measures relating to airborne eutrophication and the use of nutrients and pesticides often lacked effectiveness (effective in 23% of sites). Measures regarding agricultural effluents and the resulting eutrophication have been implemented in many areas and were deemed effective, but changes in the food web of intertidal areas have been observed (Singer *et al.* 2023); this should be considered in future management plans.

Other key issues that require more attention are the management of invasive species and measures to improve hydrological regimes and water quality, as current measures were deemed ineffective (10–20%). Activities to restore coastal wetlands to improve their natural functionality were assessed as somewhat effective (50%). Within the Danish Land Reclamation and Nature Restoration Project local farmers exchanged farmland close to the dykes for typically more fertile land elsewhere or for other incentives, and the previously drained farmland is being reflooded. Allowing natural tidal flows is key to restoring coastal habitats like saltmarshes and mudflats for wildlife, particularly migratory birds. With the new EU Nature Restoration Law an increase in restoration efforts for coastal wetlands is expected. If such efforts are to be effective for biodiversity conservation, site management plans need to consider the requirements of migratory waterbirds.

Involvement of local communities, support groups and researchers is key for effective implementation of policies and management measures, and more than 40% of sites reported people engagement leading to at least some positive effects. However, considering the many threats to coastal wetlands and migratory birds, environmental awareness programmes should be extended.

3.4.3 Iberia – North Africa: pressures

The coastal zone of Morocco, Portugal and Spain supports at least 60% of the human population and a large proportion of industry – as high as 90% in Morocco (European



Jaime Garcia-Moreno

Tourism development, San Sebastian, northern coastline of Spain

Union 2011; World Bank Group 2023). Fourteen pressures were recorded for at least six out of eight assessed sites. Recreation and tourism were recorded as representing the highest threat in parts of the region. This industry represents between 7% (Morocco) and 18% (Portugal) of employment, and is likely to continue to grow; Spain's tourism sector broke all records in 2024 (Heymann 2024). Unfortunately, this can negatively affect birds through habitat disturbance and loss, increased human pressure, pollution and other impacts. The most obvious is the development of tourism resorts that can alter the original nature of a site and render it unsuitable for many species. High levels of beach use can lead to disturbance, whilst construction of tourist facilities and urban expansion can degrade habitats and disrupt bird behaviours, affecting feeding and preventing resting. Terns or plovers can be affected by human disturbance during their nesting season (Burger 1994), and increased human presence can in the worst case lead to trampling of well disguised nests or abandonment of nesting sites (Medeiros *et al.* 2007).

Pollution was assessed as another problem in the region. Birds can eat plastics or become entangled in litter, whilst wastewater through runoff or sewage can affect prey species that birds rely on. Fishing was considered a high pressure, although the industry no longer contributes as much to the countries' economies as it once did – between 0.3% of GDP for Spain and 1.5% for Morocco (World Bank Group 2025). Nevertheless, the sector remains important in coastal communities and represents a relatively high source of employment. As in NW Europe, the impact of fisheries on waterbirds can be direct (bycatch or disturbance) or indirect (habitat degradation, prey depletion).

3.4.4 Iberia – North Africa: conservation measures

Many sites assessed in Iberia and Morocco have a national or international designation for protection, which seemed to have a positive effect for all internationally designated areas (50% effective and 50% some effectiveness) and for many nationally designated ones (>60% effective and 10% some effectiveness).

No measures were assessed as being taken for five types of activity, all of which are known to have potentially serious negative impacts (Fig. 3.6). Hunting regulations were considered to be broadly applied and helpful (75% effective, 25% some effectiveness). Other largely effective measures related to tourism and recreation (100% some effectiveness), fisheries (100% some effectiveness), urban and industrial waste management (15% effective, 75% some effectiveness), and shellfish gathering (75% some effectiveness, 15% not effective). Urbanisation measures were ranked as effective at three sites and partly effective at two sites. Several other measures seemed to be effective or show some effectiveness, but were not applied in



A duck is released after banding at Evoa Wetland Centre in the Tagus Estuary Nature Reserve, an important protected area for migratory birds

most cases (Fig. 3.6). Efforts are needed to apply measures more broadly in the region. People engagement was considered to be only 30% effective, but seemed to have some effectiveness, and environmental awareness was considerable (70%), with some positive effect.

The four qualifying sites in Spain that supported >0.1% of the flyway population of eight or more species during January counts saw increases in 23–30 species, many with trends higher than the flyway's average, and no population decreases. At Ria de Aveiro and Estuario do Tejo, two of five qualifying sites in Portugal that were also assessed through environmental monitoring, the majority of populations were stable or increasing. At Bacia do Guadiana, Portugal, one third of the qualifying populations were declining, whereas in the Tejo Estuary most populations were stable or increasing. In Morocco a majority of populations in the four qualifying sites were stable, with eight out of 58 having a more positive trend than the flyway average, and only a single population decreasing (in Merja Zerga).

3.4.5 West Africa: pressures

West African coastal sites from Mauritania to Sierra Leone face environmental pressures that threaten their ecological integrity, from resource exploitation to climate change, creating a complex web of threats to both marine and coastal ecosystems. W African coastal waters, historically among the world's richest fishing grounds, face unprecedented pressure from unsustainable fishing practices (FAO 2024). Industrial fishing operations have devastated local communities, with estimates suggesting the loss of over 300,000 artisanal fishing jobs (Gutierrez *et al.* 2024). Illegal fishing alone costs the region approximately \$10 billion annually (Geall *et al.* 2023). Beyond the socioeconomic impacts, these practices severely affect marine ecosystems through bycatch – the second leading cause of seabird deaths globally (Richard *et al.* 2024). Overfishing disrupts entire food webs, particularly impacting water-

birds that depend on West African waters on migration or during their non-breeding season.

Coastal development, rapid urban expansion and infrastructure development pose significant threats to coastal habitats. The Sierra Leone River Estuary demonstrates how urban growth can directly threaten sites (Kamara *et al.* 2023); the construction of roads, buildings and tourist facilities fragments natural habitats and disrupts breeding grounds. In Mauritania, Ndiago port is a threat to Diawling National Park, while the construction of the road connecting it to Nouakchott crosses the critical sites of Chott Boul and Aftout-es-Saheli, threatening migratory birds and breeding Lesser Flamingos *Phoeniconaias minor*. A new port constructed at Popenguine, Senegal is adjacent to important coastal wetlands, including a community reserve, with potential impacts that should be monitored closely.

Coastal ecosystems in the region face multiple pollution sources, from industrial effluents to plastic waste. In the Sierra Leone River Estuary, accumulated garbage and plastics pose dual threats to bird populations through direct harm and habitat degradation (Williams & Johnson 2023), compounded by water quality deterioration from pollutants, rendering many areas increasingly hostile to wildlife. The conversion of wetlands for agriculture continues to shrink available waterbird habitat across West Africa. This transformation is particularly damaging when it involves mangrove deforestation, which eliminates crucial nesting and feeding areas for numerous bird species (Diop & Barry 2022). Wetland loss not only impacts wildlife but also reduces natural coastal protection against storms and erosion (Leal & Spalding 2022).

Climate change, reported from most sites, acts as a threat multiplier, exacerbating existing pressures through sea level rise and extreme weather events. Coastal wetlands face increasing inundation risks, while breeding sites become more vulnerable to flooding and erosion. In Guinea-Bissau, parts of low-lying Bantambur islet can be unin-

dated, forcing Grey-headed Gulls *Chroicocephalus cirrocephalus*, Slender-billed Gulls *Chroicocephalus genei*, Caspian Terns *Hydroprogne caspia* and West African Crested Terns *Thalasseus albidorsalis* to move to other areas for breeding.

The interplay between these various pressures creates a challenging situation for conservation efforts in West Africa. The combination of direct human impacts with broader environmental changes requires comprehensive, multi-faceted solutions that address both immediate threats and long-term sustainability challenges.

3.4.6 West Africa: conservation measures

The coastal wetlands of West Africa serve as critical habitats for migratory waterbirds. Recent assessment of conservation measures reveals a complex picture of extensive protection designations facing significant implementation challenges. The foundation of conservation efforts lies in site protection through official designations, with 68% of assessed sites receiving national protected status and an additional 20% achieving partial designation. International recognition further strengthens this protection status, with 62% of sites receiving international designation, though only 45% were assessed as demonstrating effective implementation of international protection measures. In Mauritania, although Chott Boul and Aftout-es-Saheli are part of the Senegal River Transboundary Biosphere Reserve, neither are nationally protected. Niimi National Park in The Gambia and Cacheu Landscape in Guinea-Bissau have applied for Biosphere Reserve status. 64% of sites were assessed as having established management plans, although only half demonstrated effective implementation, and 5% showed no effectiveness at all, highlighting a crucial gap between planning and execution that persists even in protected areas.

Regulatory measures across different sectors revealed a striking pattern of high coverage but limited effectiveness. Fisheries regulations had the highest coverage at 90% of assessed sites, although only 30% demonstrated effective



Construction of a new port at Popenguine on Senegal's Petite Côte

Jaime Garcia-Moreno



Jaime Garcia-Moreno

A community meeting for women at Tocc Tocc Community Reserve, Senegal

enforcement. This pattern was repeated across other sectors: forest and mangrove protection measures were assessed as present in 75% of sites but achieved only 35% effectiveness, while hunting regulations were recorded in 80% of sites but effective in only 25%. Tourism management was considered effective in only 20% of sites. Agricultural land use regulations were present in 30% but considered effective in only 10% of sites, highlighting challenges in managing agricultural expansion, especially in unprotected sites.

Habitat management and restoration efforts demonstrated varying degrees of success across different interventions. 70% of sites were assessed as implementing mangrove restoration activities, with 38% showing effective results and 5% reporting no effectiveness at all. General habitat restoration measures, implemented at 55% of sites, achieved effectiveness in only 25% of sites. Erosion protection measures were assessed as present in half the sites, but effective in only 8%. The regional EU-supported PAPBio programme '*Sustainable management of biodiversity and protected areas in West Africa*' contributed to mangrove restoration in the Saloum Delta Biosphere Reserve (Senegal) and Niimi National Park (The Gambia), and from Senegal's Casamance through Guinea-Bissau to Iles Tristao (Guinea), and Yawri Bay and Sherbro –Turtle Islands (Sierra Leone).

Community engagement and research activities, crucial elements for sustainable conservation, showed similar

patterns of high initiation but limited effectiveness. Community involvement was reported in 75% of sites, with 37% assessed to have effective engagement. Research activities, present in 63% of sites, achieved effectiveness in 24%, suggesting significant challenges in maintaining research programmes. Local communities are represented on Marine Protected Area (MPA) management committees in Senegal, whilst in Yawri Bay (Sierra Leone) and Bantambour (Guinea-Bissau) they play key roles in conservation. Community engagement also takes place through policy, advocacy and communication during World Migratory Bird Day (WMBD) and World Wetlands Day (WWD) celebrations.

This assessment reveals a consistent pattern of protection measures and initiatives showing impressive coverage, but effective implementation lagging far behind. The average implementation gap of 47% between the existence of measures and their effectiveness points to systemic challenges in converting designations and regulations into practical conservation outcomes.

3.4.7 Gulf of Guinea: pressures

Assessments from 48 coastal wetlands between Liberia and Gabon ranked fishing and buildings as the principal threats, scored as being a high to very high threat at 62% and 43% of sites respectively. Large parts of the coastal zone are densely populated, with major cities such as Lagos, Accra and Abidjan. Many sites included in the survey are within or close to urban centres, where develop-

ment is a constant and ongoing pressure on wetlands. In Ghana, expansion of buildings (encroachment), growth of aquatic plants and intense farming activities were considered the principal pressures at Sakumo Lagoon, whilst expansion of buildings for settlements were also noted for Muni-Pomadze, along with resource overexploitation, use of unapproved fishing gear, cutting of trees and shrubs for firewood, and habitat destruction by annual bush fires. Other pressures across the region can also be partially linked to urban development, including logging, roads, litter and garbage, land reclamation and recreation.

The Gulf of Guinea is naturally rich in coastal lagoons formed behind a coastal sandy strip, which are widely used by people for fishing, hunting, recreation and farming. Surrounded by a growing urban population, the 208 km² Lagos Lagoon in southwest Nigeria is subject to numerous anthropogenic threats, including sand dredging, waste disposal and other forms of pollution (Nosazeogie *et al.* 2020). Land reclamation, dredging, beach erosion and deforestation are ongoing threats, whilst fishing and aquaculture are widespread, with related pressures including disposal of nets, lines and traps (Nosazeogie 2022).

Fishing is a key activity at sea and within coastal wetlands and estuaries. In lagoons, a range of traditional fisheries methods exist, including the *acadja* system. Over-exploitation of fish stocks has been recorded in coastal lagoons of Côte d'Ivoire, leading in the past to government bans on fishing in Aby and Ebrié Lagoons (Nicole *et al.* 1994). Overfishing was considered the main pressure for several sites in Nigeria, although the intensity was not necessarily severe.

The Gulf of Guinea supports extensive mangroves and other coastal forests, which serve key ecological functions and support biodiversity. They also provide important resources for people, harvested at both local and industrial levels. Mangrove cutting was recorded as a principal threat



Black Crane *Zapornia flavirostra*, Allen's Gallinule *Porphyrio alleni* and Lesser Moorhen *Gallinula angulata* for sale, Porto Novo, Benin

at several wetlands, recorded as a high to very high threat at 44% of sites. At Monrovia coastal Lagoon, mangroves are cut in order to dry the swamp for construction. Coastal forests are widely cleared for agriculture.

Urban centres exert a high demand for natural resources, including wildlife, and hunting is a constant pressure across much of the region, recorded as a medium to very high threat at 79% of sites, with a wide range of mammals, birds, reptiles and other fauna sold in local markets. In the wetlands of south Benin, Tchankpan *et al.* (2023) recorded widespread hunting of waterbirds during the rainy season, the most numerous species caught being Rallidae, especially Black Crane *Zapornia flavirostris* and Allen's Gallinule *Porphyrio alleni*. Larger birds, such as African Openbill *Anastomus lamelligerus*, White-faced Whistling-duck *Dendrocygna viduata* and egrets and herons were also captured. Hunting waterbirds was also recorded during the dry season, with African Jacana *Actophilornis africanus* caught in large numbers (Chaffra 2022). Most birds were captured using traps, nets or nooses, as well as catapults and guns; main uses were for food and medicinal purposes.



Tim Dodman

Wood for sale at a coastal wetland in Cameroon

Some countries noted that erosion was a pressure at sites, including in Liberia, Ghana, Benin and Nigeria, caused by a combination of high rainfall leading to a rise in water levels and with possible links to climate change and sea level rise, with some sea encroachment. Some coastal lagoons in Nigeria are prone to the spread of Water Hyacinth *Eichhornia crassipes*, and mangrove habitats to rapid colonisation by Nypa palms *Nypa fruticans*. Other threats noted for coastal lagoons of Nigeria were effluent spillage from petroleum mining activities, mining of crude oil, disturbance on beaches from motorbikes, garbage and solid and household waste, and pollution from large ships. Sand dredging was noted at Badagry Creek, which affected the sand banks used by some waterbirds for foraging and roosting. Continuous dredging activities were predicted to alter depth.

3.4.8 Gulf of Guinea: conservation measures

The percentage level of sites recorded as being under effective or partially effective management or under national or international designation was lower than for other regions, for instance with over 65% of sites recorded as having no management planning in place (Fig. 3.6). Cameroon's Douala-Edea is one of the newest nationally protected areas, but does not yet have a management plan in place, and management was deemed to be ineffective. Only two out of ten sites evaluated in Côte d'Ivoire were considered to have effective management in place, the Îles Ehotilé and d'Azagny national parks. The Lagos state government in Nigeria has some strong environmental regulations but enforcement is mostly weak and ineffective.

Although some regulations on managing pressures are in place across the region, the general level of effectiveness in their implementation was considered low, with regulations relating to forest/mangrove cutting, fisheries and hunting recorded as having greater effect than others, ranked together as being effective at 26% of sites where measures were applied. The effectiveness of mitigating measures likewise received low scores, with replanting of mangroves/forests considered the most effective, scored as has having at least some level of effectiveness at all sites where this took place.

People engagement scored somewhat higher, with at least some level of effectiveness in 50% of sites. Environmental awareness activities fairly regularly take place within the region, including WMBD and WWD events, often engaging local schools, with events in 2023 in Ghana, Benin and Nigeria. These included activities at environmental centres in Nigeria, whilst a new wetland centre is under consideration for Ghana's Densu Delta. The waterbird census itself provided an opportunity to raise conservation awareness among local people living around Nigeria's coastal lagoons. Local communities at Benin's Lac Nokoué were closely involved in implement-



Emmanuel Taye

Awareness-raising activity in Densu Delta



Destruction of vegetation at wetland site

ing protection and ecosystem restoration measures. At Lake Piso, Liberia, youth eco-brigade members act as environmental stewards for managing the site through the City Mayor of Robertsport, and environmental groups provided training and awareness.

A photographic exhibition and awareness raising event in 2022 at the National Theatre of Nigeria aimed at reversing plans to convert natural wetlands in the theatre grounds for various developments. After destruction of the mangroves at the site, efforts were made to preserve remaining wetlands, with the photo exhibition showcasing the value of coastal wetlands, especially for birds. It seems that these efforts have paid off, with planned sand-filling of the wetland arrested, at least for the time being, and other positive measures being considered.

3.4.9 Southern Africa: pressures

Nine sites from Namibia and South Africa were included in the analyses. Recreation and tourism present an increasing pressure along the coastline, and can lead to habitat

degradation and vegetation loss due to heavy foot traffic from tourists and off-road vehicle damage, destroying wetlands and grasslands critical for many bird species (Steven *et al.* 2011). The threat of recreation and tourism was ranked as high to very high in six of the nine sites (67%). Pollution arising from improper disposal of waste, from boating leading to oil spills and from chemical runoff in wetlands is also a major threat, often linked to recreational activities (Lemley *et al.* 2017). Disturbance from tourists also causes birds to flee repeatedly leading to increased energy expenditure, reducing their fitness and survival chances (Steven *et al.* 2011).

Another major threat recorded was roads and cars, ranked as high to very high for five of the nine sites (55.5%). Noise and light pollution disrupt bird behaviour, including foraging, nesting and migration, whilst roads facilitate activities like illegal hunting and land conversion, impacting bird populations (Begg 1978). Vehicles also contribute to air, soil and water pollution, negatively affecting habitats and food resources.

Buildings were the third highest threat recorded, ranked as medium to high threat in eight of nine sites, contributing to habitat loss and fragmentation, disturbance (especially for sensitive species like flamingos and pelicans), pollution and eutrophication. Urbanisation amplifies climate impacts like flooding or water scarcity in wetland habitats, indirectly affecting waterbirds. Dams were recorded as a major threat to migratory species in wetlands or along rivers altered or submerged by dam construction. Dams invariably disrupt predictable seasonal flooding, reducing food availability for birds, and can also

lead to eutrophication, sedimentation and pollution in downstream areas, reducing the quality of habitats for migratory species (Taylor *et al.* 1999). Dams may further fragment river systems reducing the availability of connected habitats for birds, and in the Verlorenvlei system contribute to reduced freshwater inflow.

Extreme weather and global warming, ranked as threats in five of nine sites, have wide-ranging implications for conservation efforts (Church *et al.* 2013), and pose significant pressures on waterbirds and their habitats in Namibia and South Africa. The pressures arising from sea level rise, recorded for three sites, include the loss of critical feeding areas for waterbirds. Pollutants from agricultural and industrial effluents degrade wetlands, reducing their suitability for waterbirds. Other threats recorded included litter and garbage, fishing, poor urban wastewater management, non-native plant species, hunting, shellfish gathering and collection of aquatic plants (Fig. 3.5).

3.4.10 Southern Africa: conservation measures

Coastal wetlands of Namibia and South Africa serve as vital stopover sites and foraging habitats for numerous migratory bird species, and their protection and management is critical for conservation. Based on the 2023 survey, 10% of assessed sites had effective management planning, whilst 78% recorded some effectiveness. Management planning requires a structured, site-specific approach that balances biodiversity conservation with local community needs and sustainable development, backed by clear ecological assessments and threat analyses. Regular bird surveys were implemented at all assessed sites.



Litter being collected at Catumbula wetland, Lobito, Angola during a community workshop



Miguel Xavier

Angolan team on an exchange visit in Namibia at an important breeding site for Damara Terns *Sternula balaenarum*

In relation to site protection, of the nine sites assessed, four were under national designation and six under international designation. However, site designation was only deemed effective at Langebaan Lagoon (South Africa). Conservation measures in relation to zonation of tourism/recreation, hunting, wind farms, mining exploitation, urbanisation, and agricultural land use were considered at least partially effective. Effective regulations should balance conservation priorities with sustainable development and socioeconomic needs. Habitat management and conservation are vital to maintain biodiversity, support ecosystem services and protect unique habitats, particularly in ecologically sensitive areas. However, most measures, such as habitat restoration, counteracting vegetation succession, and urban and industrial waste management, recorded only limited effectiveness; control measures against non-native species scored highest, at 50% of sites where this applied (Fig. 3.6).

People engagement is critical for conservation and habitat management. Involving local communities, researchers and Site Support Groups (SSGs) fosters stewardship, ensures sustainable practices, and strengthens the long-term resilience of ecosystems. SSGs often act as site custodians, playing a key role in grassroots conservation (e.g. monitoring of sites for illegal activities, advocacy and awareness and collaboration with authorities). In South Africa and Namibia, SSGs were recorded as being effective or of some effectiveness at all six sites where they applied. By prioritizing research, empowering communities, and supporting site groups, Namibia and South Africa can foster robust, inclusive, and sustainable conservation initiatives that protect habitats and biodiversity for future generations.

3.5 Discussion and recommendations

Waterbirds and wetlands are subject to adverse pressures along the EAF. Pressures identified in 2023 were similar to those of 2020 and largely linked to human activities, leading to a substantial reduction and degradation of natural habitats. An IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services) global assessment report estimated that 75% of land surface, 65% of oceans and 85% of wetlands were significantly modified or had completely disappeared (IPBES 2019), including through the damming of most major rivers (Chao *et al.* 2008, McCartney 2009) and the degradation of freshwater habitats (García-Moreno *et al.* 2014). Impacts at key sites for waterbirds directly affect population sizes they can support, leading to declines, local losses and in extreme cases extinction; Slender-billed Curlew *Numenius tenuirostris* was recently declared extinct (Buchanan *et al.* 2024).

In 2023, fishing was ranked as the most serious pressure across the flyway, similar to its ranking in 2020. Fishing, ranked highest in Northwest Europe, West Africa and the Gulf of Guinea, can lead to habitat degradation, diminished food sources and bycatch, contributing to a decline in seabirds. 30% of seabirds are globally threatened, 11% are Near Threatened and 57% are in decline (BirdLife International 2022).

Sea level rise was reported as the most serious pressure in 2020, but in 2023 was ranked much lower. This demonstrates the difficulty in quantitatively assessing qualitative information collated from questionnaires, which can be rather subjective, and in assessing the direct threats that different pressures pose to waterbirds. For instance, litter and garbage (ranked fifth at the flyway level) is often

Box 3.1: Avian Influenza outbreaks along the flyway, 2021-2023

Highly Pathogenic Avian Influenza (HPAI) is a zoonotic viral disease that primarily affects poultry and wild birds. Wild waterbirds are natural reservoirs for the virus, which spreads through inhalation, consumption and direct contact with infected birds, their eggs or contaminated equipment. Since late 2021, HPAI, primarily H5N1, has impacted poultry and wild birds globally, while cases in mammals, such as foxes, underscore the virus's zoonotic potential. Between 2 December 2023 and 15 March 2024, 227 domestic bird HPAI A(H5) outbreaks and 414 wild bird outbreaks were reported across 26 countries in Europe (EFSA *et al.* 2024), resulting in the culling of millions of birds.

In NW Europe, breeding tern colonies suffered severely, with the UK's only Roseate Tern *Sterna dougallii* colony on Coquet Island losing 15% of its population (RSPB 2024). Over 20,000 Sandwich Terns *Thalasseus sandvicensis* (>17% of the total NW European breeding population) died in breeding colonies in the Wadden Sea and other sites (Knief *et al.* 2024). Even greater losses were estimated in Great Skua *Chatharacta skua* and Northern Gannet *Morus bassanus* (Lane *et al.* 2023, Tremlett *et al.* 2024). H5N1 was also reported in waterbirds at many non-breeding sites, such as in wild geese on the Solway Firth in the UK (Ross *et al.* 2024). The continued spread of H5N1 poses a dire threat to waterbird and seabird colonies worldwide.

HPAI caused significant mortality in West Africa. An outbreak in January 2021 killed 750 Great White Pelicans *Pelecanus onocrotalus* in Djoudj National Park (Senegal) and 409 W African Crested Terns and 1,642 pelican chicks in Diawling National Park (Mauritania). An outbreak in early 2023 caused wild bird deaths in Senegal, The Gambia and Guinea-Bissau, prompting field investigations and containment efforts (BirdLife International 2023). It resulted in >25,000 wild bird deaths of 32 species in The Gambia and Senegal, including >10,000 W African Crested Terns and >4,000 Caspian Terns (Jatta *et al.* in press).

Joint risk assessments conducted in NW Europe identified likely risk pathways and led to the development of management guidelines (Bregnballe *et al.* 2022). Responses along the flyway included surveillance and monitoring in the Wadden Sea, The Gambia and Senegal. An expert group on One Health supported the Gambian government to prepare for potential future outbreaks, with a workshop on risk factors and possible management measures for future outbreaks, ranger training on adequate protection and preventing virus spread, and laboratory training to enable rapid detection (Common Wadden Sea Secretariat 2023).

Recommended measures such as disposal of infected birds, import restrictions and awareness campaigns have contained outbreaks, particularly in wild birds. Notable achievements include enhanced laboratory capacity and improved One Health collaboration. However, limited resources and logistical difficulties persist, and an integrated approach focusing on strengthened surveillance and biosecurity is vital for mitigating HPAI impact. In West Africa, Jatta *et al.* (in press) consider that enhancing surveillance, fostering collaboration and securing adequate resources are imperative for effective response and the control of future outbreaks.



Dead terns impacted by HPAI, Bantambour, Guinea-Bissau

noticeable at sites, although it can be hard to quantify its direct impact on birds. Ship traffic (ranked fourth) can be a visible feature through ships observed from coastal sites, but the direct threats of ship traffic to waterbirds at most sites are unclear.

Recreation/tourism was ranked as the highest pressure in Iberia - North Africa and in Southern Africa. The development of hotel complexes, roads, camping sites and other facilities can destroy or fragment habitats. Some bird resting or stopover places (beaches, wetlands, forests, etc.) are polluted by liquid and solid waste, noise, light and affected by other pressures. Tourism activities threaten 27% of the world's birds (BirdLife International 2022).

The expansion and intensification of agriculture leads to the degradation and loss of habitats, whilst threats increase significantly when agrochemicals enter water bodies. Habitat management and restoration within the agricultural matrix must be considered in the context of local community needs and of migratory passerines that use early successional habitats in traditionally managed farmland (Agger *et al.* 2024). Similarly, the needs of local communities and migratory waterbirds should be reflected in coastal wetland management.

Hunting has contributed to the extinction of at least 50 bird species over the last 500 years, and currently threatens at least 529 (37%) globally threatened species (BirdLife

International 2022). Although many countries have introduced legislation to prevent the over-exploitation of birds, illegal hunting and trapping remain widespread due to weak law enforcement, culturally entrenched practices, continued demand and a lack of alternative livelihoods.

The impacts of climate change (sea level rise, extreme heat, droughts, fires, false springs and heavy rainfall) on birds are multiplying and intensifying. Distribution models predicting the future impacts of climate change applied to 197 migratory waterbirds in Africa-Eurasia reveal that dispersive species in the Afro-tropical biome are expected to suffer the greatest net loss of range by 2050 (Nagy *et al.* 2022). Species already facing multiple pressures may be more exposed to climate change impacts (Bateman *et al.* 2020). Within a climate adaptation strategy, reducing pressures on species is a key element in helping species to adapt to inevitable climate change (Heller & Zavaleta 2009; Mawdsley *et al.* 2009).

Avian influenza outbreaks between 2021 and 2023 (Box 1) affected waterbirds and seabirds, which are natural reservoirs for the virus. The rapid spread of the virus often limits the ability to react quickly in affected areas. Additional and reinforced surveillance methods in potential propagation hot spots should improve early detection and enable a more rapid response to emerging epidemics (García-Moreno 2023). Experiences from Northern Europe have shown that collection of infected carcasses at early stages of an outbreak have the potential to curb virus spread (Knief *et al.* 2024). Securing resources for response and control and employing sound management options and precautions are essential when dealing with suspected disease cases in Africa (Jatta *et al.* in press; Dodman *et al.* 2023).

The pressures on waterbirds along the EAF are multiple and often interconnected. An integrated and collaborative approach is needed to reduce threats and preserve habitats. Success depends on the commitment of governments, environmental organisations and local communities. It is also essential to instil the importance of conserving, respecting and valuing nature, and an appreciation of the strong heritage value and ecological roles of migratory birds. Natural habitats can be preserved through sustainable management practices established by local communities; there is growing recognition of the importance of involving communities to ensure the fair and effective management of KBAs. Preserving biodiversity requires conserving remaining habitats and restoring and reconnecting those that have been lost or degraded. Restoring natural carbon-rich ecosystems such as coastal wetlands is also essential for climate change mitigation.

The importance of critical sites for long-distance migratory shorebirds was demonstrated by Catry *et al.* (2024), who found that, regardless of their wintering origin, Grey

Plovers *Pluvialis squatarola* spent more time in the Wadden Sea than at any other stopover site. Ongoing habitat deterioration in the Dutch Wadden Sea has led to declines of other shorebirds, and such impacts may affect the whole Eastern Atlantic population of Grey Plover. Such findings demonstrate the combined importance of monitoring both species and pressures at sites.

Although no correlation was found between site pressures and decreases in bird populations at qualifying sites that supported >0.1% of the flyway population (Fig. 3.3), this information could become a useful comparative tool, especially to highlight sites witnessing changes in bird trends and/or conservation status and to prioritise future management actions. It could provide a useful benchmark for sites where significant changes (positive or negative) take place, such as major developments or impacts from climate change.

Overall, it is evident that a number of pressures negatively impact coastal wetlands of the EAF, with fishing, building and tourism/recreation, roads and hunting ranked as the highest threats in 2023. However, conservation measures appear to be widely inadequate to address these and other pressures, with only 32% of measures rated as being effective at sites to which the measures applied. Improved conservation measures are needed, from site management to the effective regulation of fishing and hunting and to improving people engagement. Some pressures are likely to increase over time, especially those related to climate change, although it is a misnomer that climate-related pressures were ranked lower in 2023 than in 2020.

However, although taking effective conservation action invariably seems like an uphill battle, there are positive examples from across the flyway of people working together to achieve conservation successes. The fact that many key coastal sites of the flyway are under some form of national or international designation is very positive, and presents clear justification for protecting them through improved management and, inevitably, raising funds for positive actions. Much can also be achieved through maintaining a constant pressure on governments and policy makers and through the determination of key individuals. A good example is provided by the decision of the Portuguese government not to build a new commercial airport for Lisbon in the heart of the Tagus estuary, the most important wetland in Portugal for waders and other waterbirds (Alves 2020; chapter 6). This decision was no doubt influenced by NGOs launching a lawsuit to prevent this development and by sustained determination for their campaign, which received broad international support. Raising awareness and maintaining pressures on impactful developments really can make a difference, and effective conservation measures are not out of reach.



Pink-footed Goose | Oie à bec court (*Anser brachyrhynchus*) (Saverio Gatto / Agami)

4. Greenland-Canadian waterbirds on the East Atlantic flyway: status and trends

Hans Meltofte, Jannik Hansen, David Boertmann, Olivier Gilg & Benoît Sittler

Summary

The waterbirds breeding in Greenland and wintering on the East Atlantic Flyway are generally thriving. This is especially so for the Barnacle *Branta leucopsis* and Pink-footed Geese *Anser brachyrhynchus* breeding in high-arctic NE Greenland, which have both increased in numbers and expanded their range considerably during the last 100 years or so. Greenland White-fronted Geese *Anser albifrons flavirostris* and Pale-bellied Brent Geese *Branta bernicla hrota* have seen both ups and downs. Ducks and loons appear to do well. Waders also appear only to show minor changes at the single location where they are monitored systematically, Zackenberg Research Station in central high-arctic NE Greenland. Here, Common Ringed Plover *Charadrius hiaticula*, Red Knot *Calidris canutus islandica* and Sanderling *Calidris alba* have shown next to no deviations from zero in long-term trends in territory numbers since monitoring began in 1996, while Ruddy Turnstone *Arenaria interpres* have declined somewhat and Dunlin *Calidris alpina arctica* increased. Due to snow and vegetation cover gradients spanning a factor 20 in N-S and E-W directions in the Greenland high Arctic, we do not know to what extent the data from Zackenberg are representative for larger areas or high Arctic Greenland in general. A major change is evident, however, in that the year-to-year variability in critical environmental conditions and wader populations has increased during the study period, with 2018 showing the latest snowmelt on record with exceedingly few birds breeding. In parallel with increasing summer temperatures, a number of species have expanded their breeding range into or within Greenland: Golden Plover *Pluvialis apricaria*, Eurasian Whimbrel *Numenius phaeopus* and Lesser Black-backed Gull *Larus fuscus graellsii*, and this may include Northern Pintail *Anas acuta* and Eurasian Teal *Anas crecca* as well.

Résumé

Les oiseaux d'eau qui se reproduisent au Groenland et hivernent sur la voie de migration de l'Atlantique Est sont globalement en bonne santé. C'est particulièrement vrai pour la Bernache nonnette (*Branta leucopsis*) et l'Oies à bec court (*Anser brachyrhynchus*) qui nichent dans le nord-est arctique du Groenland. Leurs effectifs ont augmenté et leur aire de répartition s'est considérablement étendue au cours des cent dernières années. L'Oies rieuse

du Groenland (*Anser albifrons flavirostris*) et la Bernache cravant (*Branta bernicla hrota*) ont connu des phases d'augmentation et de déclin. Les canards et plongeurs semblent bien se porter. Les limicoles montrent peu de changements dans la seule station de suivi systématique de Zackenberg (centre nord-est arctique du Groenland). Le Grand-gravelot (*Charadrius hiaticula*), le Bécasseau maubèche (*Calidris canutus islandica*) et le Bécasseau sanderling (*Calidris alba*) ne présentent presque aucune tendance nette dans le nombre de territoires depuis le début du suivi en 1996. Le Tournepierre à collier (*Arenaria interpres*) montre un léger déclin, tandis que le Bécasseau variable (*Calidris alpina arctica*) est en augmentation.

4.1 Introduction

In the huge Greenland and NE-Canadian breeding area of the East Atlantic Flyway (Fig. 4.1), there is only one research station where long-term monitoring of populations and breeding performance of waders and other waterbirds is carried out: Zackenberg Research Station in central NE Greenland (74.47°N, 20.57°W), established in 1995 (Meltofte 1997, 2022). Supplementary data are available from two other high Arctic study sites with long-term monitoring of terrestrial vertebrates in NE Greenland; Karupelv Valley (72.50°N, 24°W) since 1988 (Meltofte *et al.* 2007a; Meyer *et al.* 2021), and Hochstetter Forland (75.15°N; 19.70°W) since 2010 (see Etchart *et al.* 2024).

Contrary to the much less mountainous and relatively homogeneous Siberian Arctic, high-arctic Greenland and Canadian islands, where most of the waders breed, are mountainous with pronounced geographical gradients both altitudinally and between regions, making extrapolations from one or a few sites problematic. Furthermore, very different conditions prevail between the Arctic deserts of northernmost Greenland and northern Ellesmere Island and the 20 times more snow- and vegetation rich southern parts of high Arctic Greenland (Meltofte 1985; and a factor 100 difference between little precipitation in the northernmost and much on southernmost tips of Greenland). As if this was not enough, a similar factor 20 gradient exists between the cool and humid outer coasts of NE Greenland and the desert-like interior close to the Inland Ice Sheet 100–300 km to the west (Pedersen *et al.* 2018).

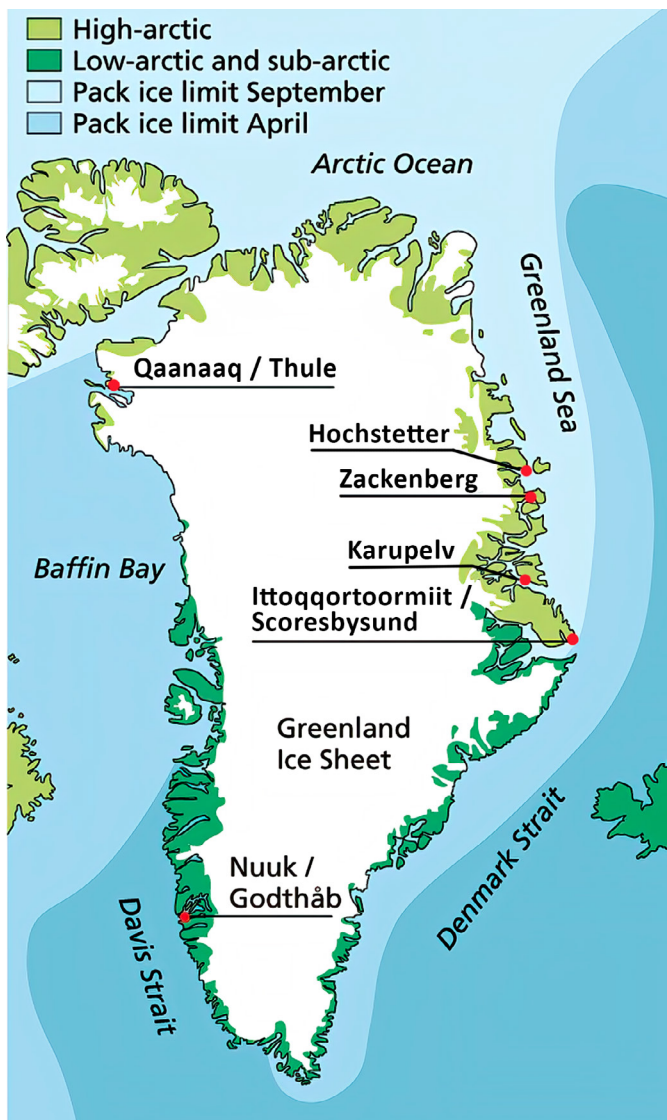


Figure 4.1. Arctic breeding habitats in Greenland and northeasternmost Canada. Study sites in Greenland are given.

In relation to these gradients, Zackenberg Research Station is situated right in the middle of both, where breeding bird populations are much denser than in the mentioned extremes. So, although it was never the intention that monitoring at Zackenberg should be representative of the entire region, we expected that the ecological relations between timing of snowmelt, invertebrate food availability, lemming fluctuations etc. and bird breeding performance would be (Møltøfte 1997, Petersen 1997).

4.2 Waterfowl and divers

By far the most pronounced changes in the bird fauna of NE and N Greenland is the marked increases in population size and distribution of Barnacle *Branta leucopsis* and Pink-footed Geese *Anser brachyrhynchus* during the 120 years of ornithological records from the area (Møltøfte 2006, Boertmann in print). From few and widely dispersed

breeding sites these two species have now expanded widely over 'all' lowlands, the Pink-feet also with large numbers of Icelandic non-breeders, having expanded their moulting area even into the northernmost land in the world, Peary Land (Boertmann in print). These changes comply with strong increases in the wintering areas (van Roomen *et al.* 2022), where the Barnacle Goose population was estimated at more than 70,000 individuals in recent years (Heldbjerg *et al.* 2021). The Greenland share of the half million strong Iceland-Greenland breeding population of Pink-footed Geese is unknown.

The marked ups and downs of the Greenland White-fronted Goose *Anser albifrons flavirostris* breeding in low-arctic W Greenland and wintering mainly in Ireland (a halving since a culmination of more than 35,000 in 1999; Fox *et al.* 2022) remain undetected on the breeding grounds, while the Pale-bellied Brent Goose *Branta bernicla hrota* has withdrawn from central NE Greenland since the first decades of the 20th century and now only breeds in the extreme North and Northeast. Here, around 1000 birds including non-breeders and perhaps 50-100 breeding pairs are found and may now be increasing (Boertmann *et al.* 2015). This population belongs to the Svalbard sub-population wintering in Denmark and NE England (Madsen *et al.* 1999), while less than 50 pairs breeding in western N Greenland belong to the Irish wintering population (Boertmann in print).

The Greenland breeding population of c. 25,000 pairs of Red-throated Loon *Gavia stellata*, wintering in NW European waters, appears to be thriving (Boertmann in print). So do Common Loons *Gavia immer*, of which a minor part most likely migrate to UK and Spain (Weir *et al.* 1996). Six waterfowl species are also found breeding in the region that migrate to open waters of southern Greenland and Iceland: King Eider *Somateria spectabilis*, Common Eider *Somateria mollissima borealis*, Harlequin Duck *Histrionicus histrionicus*, Long-tailed Duck *Clangula hyemalis*, Red-breasted Merganser *Mergus serrator* and Mallard *Anas platyrhynchos*. Of these, Common Eider has undergone pronounced decreases during the 20th century in the human-inhabited parts of Greenland but has largely recovered after better protection against overharvest (Merkel 2010). We have no data to show whether the abundance of the five other species has changed. As opposed to the extensive hunting in W Greenland, there are very few places in E Greenland where waterbirds are hunted.

In the 18 km² study area on Hochstetter Forland, only the Red-throated Loon seems to have significantly increased during the last 50 years, from a single census of three pairs in the mid-seventies (Møltøfte *et al.* 1981) to a current breeding population of more than 10 pairs (and up to 2-4 pairs/km² in the most favourable habitats; OG *et al.* unpublished).

As a recent (possibly climate induced) development, Northern Pintail *Anas acuta* and Eurasian Teal *Anas crecca* may now breed sporadically in E Greenland (Boertmann in print.).

4.2. Waders etc.

The majority of waders breeding in Greenland migrate to the Old World for the winter. Only Purple *Calidris maritima* and Baird's Sandpipers *Calidris bairdii* remain in the New World, while Red-necked *Phalaropus lobatus* and Red Phalaropes *Phalaropus fulicarius* winter at sea in the tropics and sub-tropics (Boertmann in print).

Out of 11 waders breeding in Greenland, nine breed exclusively or predominantly in the high-arctic part. The common species are Common Ringed Plover *Charadrius hiaticula*, Ruddy Turnstone *Arenaria interpres*, Red Knot *Calidris canutus islandica*, Sanderling *Calidris alba* and Dunlin *Calidris alpina arctica*¹. Of these, the East Atlantic Flyway populations of Common Ringed Plover, Ruddy Turnstone, Red Knot and Sanderling include breeders in Ellesmere and adjacent islands in the north-easternmost Canadian Arctic (Meltotte 1985, Boertmann in print).

During the 28 years of monitoring of the breeding wader populations at Zackenberg (Fig. 4.2), supplemented by more irregular observations in other areas, we have seen few dramatic changes during the last hundred years. Common Ringed Plover, Red Knot and Sanderling have shown next to no deviations from zero in long-term trends in territory numbers since monitoring began in 1996, while Ruddy Turnstone has declined somewhat and Dunlin increased. Also in other areas, Ruddy Turnstone together with Red Knot have shown both positive and negative changes of more than a factor two, and these two species appear to be the most sensitive to varying snow cover in spring (Meltotte 1985; see also Meltotte *et al.* 2008 and Schmidt *et al.* 2023). This includes mass mortality of Red Knots at Thule in N Greenland in 1972 and 1974 during severe spring weather, probably contributing to the following marked decline recorded on the wintering grounds in Northwest Europe during the following decades (Boyd & Piersma 2001; Meltotte *et al.* 2007b).

In the 15 km² study area at Karupelv, a slight decline was found in the small population of Ruddy Turnstones from five or more pairs during the period 1988-2014, to never more than three pairs in the last ten years. Based on numbers of families recorded after hatching, an apparent decline in the numbers of Red Knots also occurred at this site. For Dunlins, Common Ringed Plovers and Sander-

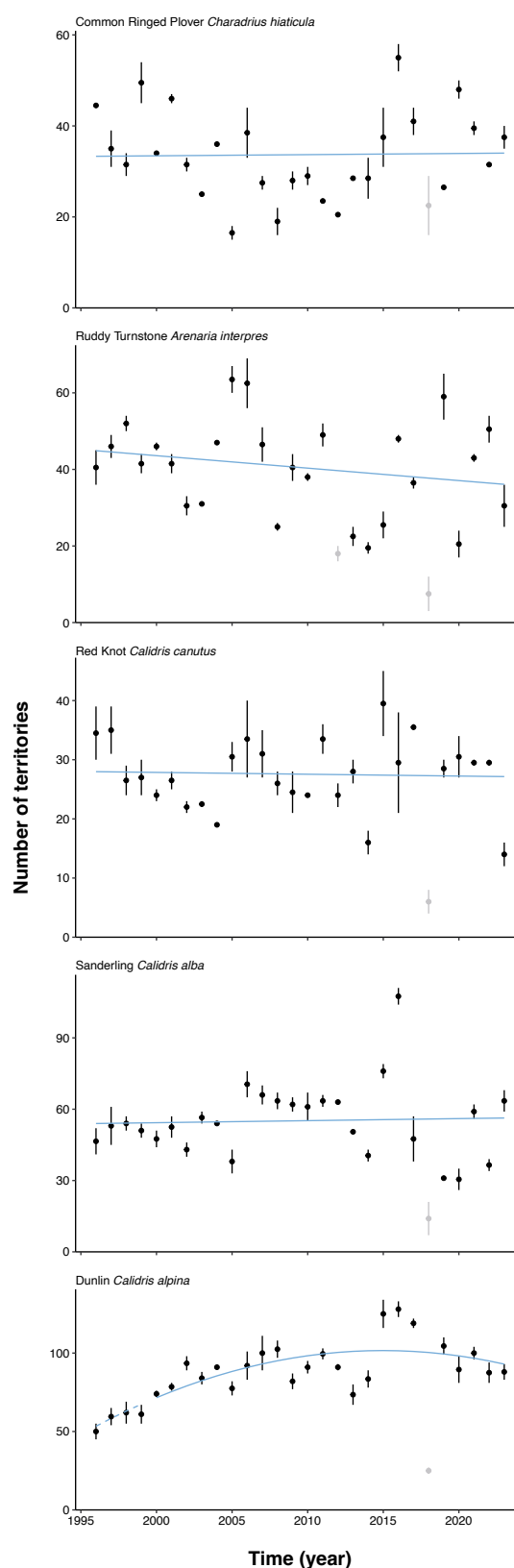


Figure 4.2. Annual minimum and maximum numbers of territories/site holding individuals of the five common wader species in the 15.8 km² census area at Zackenberg 1996-2023. These wader species are largely monogamous and territorial.

1) It is repeated again and again in the literature that (schinzii) Dunlins breed in Southeast Greenland, but this is based on a single confirmed breeding in 1933 and has never been recorded since then.



Hans Meltofte

Arctic waders can cope with quite harsh conditions. This Dunlin nest at Danmarkshavn 17 June 1975 hatched successfully (see Meltofte 1979).

lings, no clear trends have been detected so far. Lastly, breeding of Red-necked Phalarope was repeatedly found in recent years.

At Hochstetter, we found no indication of population changes for Common Ringed Plovers, Sanderlings and Red Phalaropes, but Ruddy Turnstones have apparently also declined over the past 50 years, and at a much faster speed than at the two other Greenlandic sites (i.e., from 36-39 pairs in the mid-seventies to less than 10 breeding pairs in all years during the period 2010-2024). The trend is unclear for Dunlins at this site, where 20-24 pairs were reported in 1976, but breeding pairs fluctuated from two to 31 in 2010-2024 (median = 12).

At all three Greenland sites, due to extreme late snow melt, 2018 was a non-breeding year for most tundra breeding birds.

Contrary to the situation in Barnacle and Pink-footed Geese, our data from the breeding grounds do not allow confirmation of the strong increases since around 1980 in the midwinter census indexes from the East Atlantic Flyway in Ruddy Turnstone, Sanderling (yet increasing until the local crash at Zackenberg in 2018; Schmidt *et al.* 2019, Meltofte *et al.* 2021) and Red Knot (mid-winter numbers moderately decreasing now; see Appendix 1). The migration of Greenlandic Sanderlings past the westernmost point of Denmark at the approach to the Wadden Sea also showed a strong increase in 1964-2017, while Red Knot and Turnstone (also including an unknown share of Palearctic birds) showed small increases (Meltofte *et al.* 2019). A possible explanation for the lack of response at our sites, at least for Common Ringed Plover, Sanderling and Red Knot, could be that in our existing three Greenlandic study areas (whose locations were historically chosen due to their higher than average productivity), shorebirds already breed at densities close to the sites'

carrying capacities, while important population processes may take place in marginal areas with so few birds that no research is conducted (Meltofte *et al.* 2020).

From densities recorded in the late 1900s in 34 census areas covering 425 km² and widely distributed in high-arctic Greenland and Ellesmere Island, Meltofte (2001) estimated the breeding population of Common Ringed Plover at 30,000-60,000 pairs, Ruddy Turnstone at 35,000-70,000 pairs, Red Knot at 55,000-70,000 pairs, Sanderling at 25,000-50,000 pairs and Dunlin at 7,000-15,000 pairs. These figures are (were) in reasonable agreement with the mid-winter census results for Common Ringed Plover, Sanderling and Dunlin, whereas they are far too low for Red Knot and too high for Ruddy Turnstone (Meltofte 2001). HM's interpretation was that there are more European-wintering Red Knots breeding in Canada than we know of, and that Ruddy Turnstones are underestimated at their wintering grounds because of their scattered occurrence on several types of often uncovered, non-estuarine coasts.

Two other wader species, European Golden Plover *Pluvialis apricaria* and Eurasian Whimbrel *Numenius phaeopus*, appear to have expanded their breeding area from



Lars Holst Hansen

A number of parameters at Zackenberg has shown increasing variability during the study years. 2018 is the most extreme example so far, and exceedingly few birds attempted to breed (24 June 2018).



Erik Thomsen

Ruddy Turnstone and Red Knot appear to be the two wader species most sensitive to varying snow melt and early spring food availability in high-arctic Greenland.

Iceland to include small populations in southernmost high-arctic Greenland (Boertmann in print). The same applies to Lesser Black-backed Gull *Larus fuscus graellsii*, one out of the 11 Laridae species regularly breeding in Greenland, which now breeds with several hundred pairs if not thousands in Greenland, and the population is increasing and expanding its range.

Lastly, it is worth mentioning that the Pectoral Sandpiper *Calidris melanotos*, whose closest breeding populations are in Canada and Siberia, has attempted to breed (i.e., incubated clutches of unfertile eggs) once at Zackenberg and twice at Hochstetter in recent years, possibly indicating a range extension from its current breeding grounds (Pagnon *et al.* submitted).

4.3. Increasing variability with more extreme events

While waterfowl are 'capital breeders', high Arctic waders are mainly 'income breeders' dependent on food resources acquired on the tundra upon arrival in spring (Klaassen *et al.* 2001) for producing and incubating their clutch. This major difference explains the high dependence of the latter group to the timing of snowmelt and the emergence of invertebrate food during their first one or two weeks on the tundra (Meltofte *et al.* 2008, 2021). Note that the high-arctic coasts are not available to waders for feeding

until July because of heavy snow and ice cover. This means that egg-laying in high Arctic Greenland varies by up to 2-3 weeks between years and areas (Meltofte 1985, Meltofte *et al.* 2007b). Here, NE Greenland differs markedly from other Arctic study areas in that the 'window' of invertebrate availability to the chicks is much wider and the risk of mismatch therefore reduced (Meltofte *et al.* 2021; see also Versluijs *et al.* 2024). Furthermore, if breeding conditions are unfavourable in one area in one year, there will most often be other regions on the N-S and E-W gradients where waders will have better breeding success. Hence, even in the year of latest snowmelt until then at Zackenberg, 1999, numbers of juvenile waders feeding in the coastal deltas of the Zackenberg Valley were at the same level as 1995-2005 numbers (Hansen & Meltofte 2006). Only in the extreme summer of 2018, when the extremely late snowmelt was regional, there were indications that poor breeding success was more widespread (Meltofte *et al.* 2021, B. Sittler in litt.).

Not only have the populations of most wader species varied from year to year around a relatively stable level during the last decades in our study areas, so have spring temperatures, timing of snowmelt and egg-laying, timing of invertebrate food availability and spring and summer precipitation (Meltofte *et al.* 2021, Schmidt *et al.* 2023). Conversely, the mean temperatures in July have steadily increased during the same period (Schmidt *et al.* 2023).

Although populations of Arctic foxes *Vulpes lagopus* are known to show important inter-annual fluctuations in Northeast Greenland, as a response to lemming cycles (Gilg *et al.* 2003, 2006, Schmidt *et al.* 2008), the prime predator on bird eggs and young has shown moderate (if any) long-term population trends in our region, and we have not seen the same regular ups and downs in predation pressure at Zackenberg as known from Siberia and other Arctic areas (Meltofte *et al.* 2007a, Schmidt *et al.* 2008, Meltofte *et al.* 2021; see also Bulla *et al.* 2019). Numbers of collared lemmings *Dicrostonyx groenlandicus* have fluctuated very irregularly since 2000, with lower densities and amplitude than before (Gilg *et al.* 2009, Schmidt *et al.* 2012, Schmidt *et al.* 2021; with the exception of a record peak density in Zackenberg in 2017), but without pronounced effects on wader breeding success at Zackenberg (Meltofte *et al.* 2020, 2021). In our three Greenlandic study sites, foxes appear to have access to other rich food resources besides lemmings, such as dead muskoxen *Ovibos moschatus*, Arctic char *Salvelinus alpinus*, goslings and moulting (flightless) geese (Schmidt *et al.* 2022).

This 'moderate' message is not the full story, however. A row of both biotic and abiotic parameters has shown markedly increased variability through the years, including more extreme events leading to less predictable conditions

(Meltofte *et al.* 2020, 2021, Meltofte 2022, Schmidt *et al.* 2023). Among the 14 bird species breeding at Zackenberg, six showed increasing year-to-year variability in population size, and spring snow cover and early season invertebrate availability were the most important drivers of year-to-year variability in several bird performance parameters.

The results of this increased variability may very well include that recurring extremely late springs and similar events will prevent or at least delay the 'colonisation' and growth of low-arctic species in the high Arctic and hence keep the high Arctic high Arctic (Schmidt *et al.* 2019). Arctic squeeze, where expansion of low-arctic ecosystems into the high Arctic (and sub-arctic ecosystems into the low Arctic) is considered one of the most severe threats to high-arctic biodiversity following climate amelioration and thereby to its rich wader communities (Meltofte *et al.* 2013).

In summary, most of Greenland and NE Canada's water-bird populations joining the East Atlantic Flyway appear to be thriving. This is in line with the finding that the wader populations in the East Atlantic Flyway are doing significantly better than many populations on the more eastern flyways in Europe and Asia or in the New World flyways (Smith *et al.* 2020, van Roomen *et al.* 2022).



Danielle Occhiato / Agami

Sanderling | Bécasseau sanderling (*Calidris alba*)



Glenn Barley / Agami

Ruddy Turnstone | Tournepierre à collier (*Arenaria interpres*)



Bar-tailed Godwit | Barge rousse (*Limosa lapponica*) (Jacob Garvelink / Agami)

5. Change in migrating and wintering waterbirds in the Wadden Sea

Romke Kleefstra, Katja Behm, Thomas Bregnballe, John Frikke, Klaus Günther, Bernd Hälterlein, Morten B. Hansen, Menno Hornman, Julia Meyer & Gregor Scheiffarth

Summary

The Wadden Sea is of great importance for many of the waterbird populations that occur in the East Atlantic flyway. The area is a crucial stopover and wintering site and also hosts relevant moulting concentrations of specific species. As in the previous assessment, many waterbirds are doing better in the short term than in the long term compared to the flyway trend. The Dutch and Danish parts of the Wadden Sea in particular contribute to a more positive picture. Species feeding mainly on vegetable matter or fish generally do well, unlike shellfish eating species that are generally in decline. Compared to the 1980s, when it was calculated that 10-12 million birds use the Wadden Sea annually, total numbers counted have not changed much. However, there has been a major shift among species groups and between species, with waders in particular becoming scarcer and herbivores, such as geese and some ducks, more numerous. Given a higher turnover rate among waders, it is likely that the total number of birds that use the Wadden Sea annually is lower at present than calculated 40 years ago.

Résumé

La mer des Wadden joue un rôle fondamental pour de nombreuses populations d'oiseaux d'eau de la voie de migration de l'Atlantique Est. Cette zone constitue un site crucial pour la halte migratoire, l'hivernage et la mue de certaines espèces. Comme lors de la précédente évaluation, de nombreuses espèces se portent mieux à court terme qu'à long terme comparé à la tendance générale le long de la voie de migration.

Les parties néerlandaise et danoise de la mer des Wadden contribuent particulièrement à cette dynamique plus positive. Les espèces se nourrissant principalement de végétaux ou de poissons montrent des tendances favorables, contrairement à celles qui consomment des coquillages, dont les populations sont en déclin.

Par rapport aux années 1980, où l'on estimait que 10 à 12 millions d'oiseaux utilisaient la mer des Wadden chaque année, les effectifs totaux comptabilisés n'ont pas beaucoup changé. Cependant, une réorganisation majeure s'est produite entre les groupes et les espèces : les limicoles sont devenus plus rares tandis que les herbivores, comme les oies et canards, sont plus nombreux.

Étant donné un taux de renouvellement plus élevé chez les limicoles, il est probable que le nombre total d'oiseaux utilisant annuellement la mer des Wadden soit aujourd'hui inférieur à celui estimé il y a 40 ans.

5.1 Introduction

Within the East Atlantic Flyway (EAF), the Wadden Sea is of outstanding international importance as a staging, moulting and wintering area for waterbird populations, whose breeding grounds extend as far as northern Siberia and northeastern Canada, and wintering grounds extend as far as NW-Europe and South Africa. The World Heritage site with 4,500 km² of intertidal flats and adjacent saltmarshes and grassland polders along the North Sea coast of Denmark, Germany and the Netherlands is known to hold up to over 6 million waterbirds at one time (Blew & Südbeck 2005) and an estimated 10 million birds use the area annually (Meltøfte *et al.* 1994). For over 20 species/subspecies, more than 20% to almost 100% of the flyway population use the Wadden Sea at some time of the year. Since the early 1990s, numbers and trends of migratory birds have been systematically monitored within the framework of the Trilateral Monitoring and Assessment Program (TMAP), using standardised routines for fieldwork and data processing. Details of this joint monitoring of migratory birds in the international Wadden Sea were presented through the years in Rösner *et al.* 1994, Blew & Südbeck 2005, Laursen *et al.* 2010 and Kleefstra *et al.* 2022b.

In this chapter we look at migratory bird trends within the Wadden Sea region, how these trends relate to trends within the entire EAF and how important the Wadden Sea is for migratory birds within the EAF. It is relevant to compare Wadden Sea trends with the flyway trends to find explanations for observed changes in numbers. If Wadden Sea the trends match, reasons for changes in bird numbers may lie outside the Wadden Sea, whereas if Wadden Sea trends differ significantly from those at flyway level, the reason for this will probably lie within the Wadden Sea. By looking at the proportion of migratory birds within the EAF that use the Wadden Sea, we can assess the importance of the Wadden Sea at an international level. By looking at how much bird numbers have changed since the 1980s, we can see if the estimated number of about 10-12 million

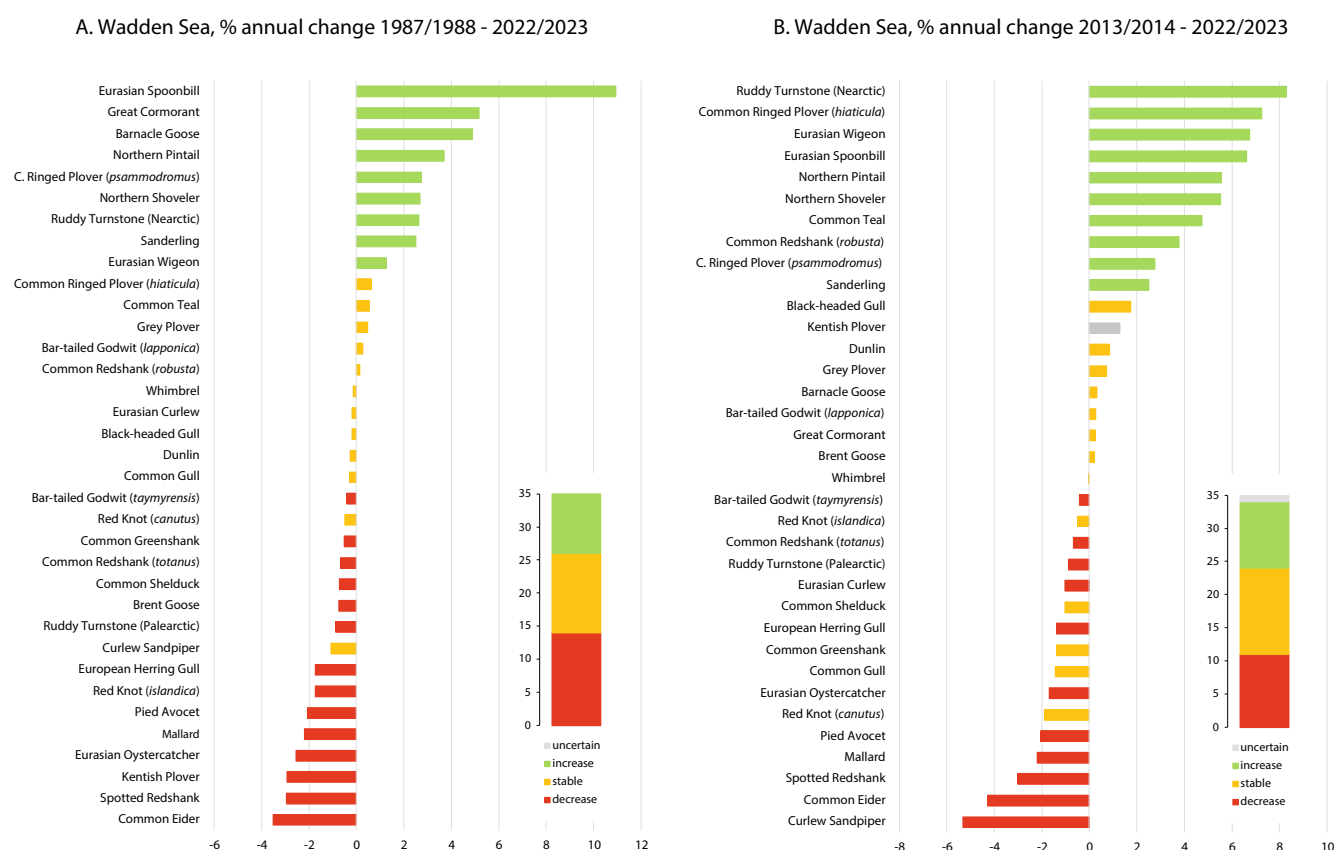


Figure 5.1. Trends for 35 species and sub-species of waterbirds in the Wadden Sea over the long-term (1987/88-2022/23; a) and short-term (2013/14-2022/23; b). Inserted graphs show percentage increasing, stable, decreasing or uncertain trends across all species and subspecies.

birds that use the Wadden Sea on an annual basis is still accurate.

Changes in breeding bird populations in the Wadden Sea are not considered in this chapter, because there has been no trend update since the East Atlantic Flyway Assessment 2020 (Kleefstra *et al.* 2022a).

5.2 Methods

Although numbers of migratory birds have been monitored within the framework of TMAP since the early 1990s, reliable trends are available for a slightly longer period, since the season of 1987/88. This allows us to provide a 36-year trend for 35 species, as well as a short-term trend over the last ten years (2013/14-2022/23).

The monitoring conducted by the Expert Group Migratory Birds (EGMB, formally JMMB), consists of (a) at least five synchronous, integral counts per year: in January (mid-winter count), May, September and November, and a fifth count in an additional month that shifts each year, (b) frequent (bi-monthly to monthly) spring tide counts at 60 monitoring sites, (c) aerial counts of Eider *Somateria mollissima* in winter, and of Shelduck *Tadorna tadorna* during wing moult (July/August) in Germany, and a ship-based

count in the Netherlands. At present, a total of 594 spatial counting units within the Wadden Sea are included in the analyses. These surveys allow statistically sound estimates of numbers, phenology and population trends. For a more detailed description, see Kleefstra *et al.* (2022b).

Despite a large dataset with long-term monitoring data, coverage is not always complete. To analyse the migratory bird count data, UINDEX (Bell 1995) was used to account for missing counts in the dataset, by estimating bird numbers for missing counts (imputing) taking into account effects of site, year and month (Underhill & Prŷs-Jones 1994). Sites were classified into four different Wadden Sea regions: Denmark, Schleswig-Holstein, Lower Saxony (including Hamburg) and the Netherlands. Yearly averages were calculated from the counted and imputed values for each month for the respective 'bird-years', covering the period from July to June of the following year (Kleefstra *et al.* 2022b). Subsequently, the program TrendSpotter was used to calculate flexible trends with 95% confidence intervals (Visser 2004, Soldaat *et al.* 2007). We report trend estimates for the long term 1987/88-2022/23 and a short term period spanning the last ten years (2013/14-2022/23). We compare the long term trend with the flyway trend in

the period 1980–2023, and the short term trend with the flyway trend in the period 2014–2023.

To assess changes in the total number of migratory waterbirds using the Wadden Sea, we compare recent monitoring data with data from the 1980s (Meltofte *et al.* 1994). In the 12 years 1980–1991, 36 counts were conducted in eight different months. From these counts the maximum number for each species was obtained and these maximum numbers were summed over species. To enable an objective comparison with two more recent periods (1996–2007 and 2012–2023) for which more (up to 60) counts are available, the same procedure of summing maximum numbers was applied to subsets of the recent data containing the same number and monthly timing of counts as in the 1980s.

To see if species per dietary guild in- or decreased in each spatial unit (units introduced by Meltofte *et al.* 1994) seasonal sums of the species within a dietary guild were geometrically averaged per season per unit and averaged again over the last 10 seasons. The abundance index is a percentage compared to the maximum of the food group in each unit.

The trend classification is assessed using the average annual rate of change of the trend and its confidence interval, where a strong increase or decrease is a significant >5% change/year, a moderate increase or decrease is a significant increase, but not certain whether it is >5%/year and stable is no significant change in numbers (i.e. confidence interval overlaps with 0, a stable trend).

5.3 Results

5.3.1 Numbers and trends in the Wadden Sea

Long-term trends of 35 species and sub-species of migratory birds in the Wadden Sea over the last 36 years show increasing numbers for 9 species, stable trends for 12 species and a decline in numbers for 14 species (Fig. 5.1a). The

short term trends show a slightly more positive picture with 10 species increasing, stable (or uncertain) trends for 14 species and a decline for 11 species (Fig. 5.1b). Species that increased in both the long and short term are Eurasian Spoonbill *Platalea leucorodia*, Eurasian Wigeon *Mareca penelope*, Northern Pintail *Anas acuta*, Northern Shoveler *Spatula clypeata*, Common Ringed Plover *Charadrius hiaticula* (*psammodroma/tundrae*-subspecies), Northern Lapwing *Vanellus vanellus* and Sanderling *Calidris alba*. Species that have shown a positive trend in the short term but not in the long term are Common Teal *Anas crecca*, Common Ringed Plover (*hiaticula*-subspecies), European Golden Plover *Pluvialis apricaria* and Ruff *Calidris pugnax*. Species that continue to decline in both the long and short term are Mallard *Anas platyrhynchos*, Common Eider (trend starts 1992/93), Eurasian Oystercatcher *Haematopus ostralegus*, Pied Avocet *Recurvirostra avosetta*, Bartailed Godwit *Limosa lapponica* (*taymyrensis*-subspecies), Common Redshank *Tringa totanus* (*totanus*-subspecies), Ruddy Turnstone *Arenaria interpres* (Scandinavian-Western Russian population), European Herring Gull *Larus argentatus* and Great Black-backed Gull *L. marinus*. The numbers of Curlew Sandpipers *Calidris ferruginea* and Eurasian Curlews *Numenius arquata* have decreased in the short but not in the long term.

In the long term, the Danish and Dutch parts of the Wadden Sea show the highest numbers of positive trends (for 16 and 17 species respectively), whereas in the German parts of the Wadden Sea many species decrease in number (20 in Schleswig-Holstein and 22 in Lower Saxony; Fig. 2). In the short term, this picture hardly differs for the Danish Wadden Sea, but the number of species that still show an increase is much smaller in the Dutch part (8), where many species show a stable and/or uncertain recent trend (20 species). The latter also applies to the German parts of the Wadden Sea (22 and 17 species stable or uncertain in Schleswig-Holstein and Lower Saxony respectively) and fewer species show a negative trend (6 and 16 respectively).

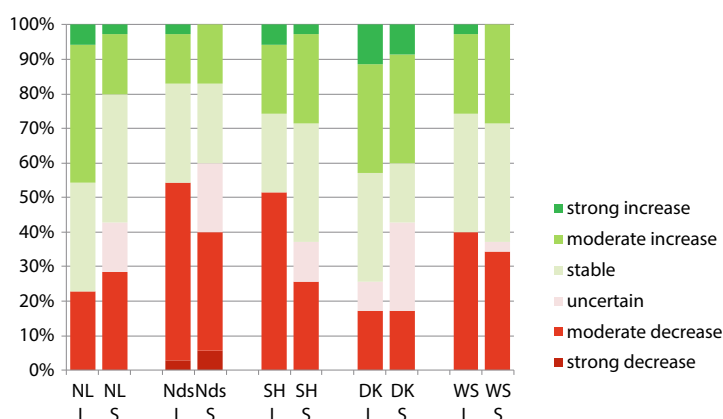


Figure 5.2. Proportions of 35 species increasing, stable or decreasing per country on the long (L) and short (S) terms. NL = Netherlands, Nds = Lower Saxony, SH = Schleswig Holstein, DK = Denmark.

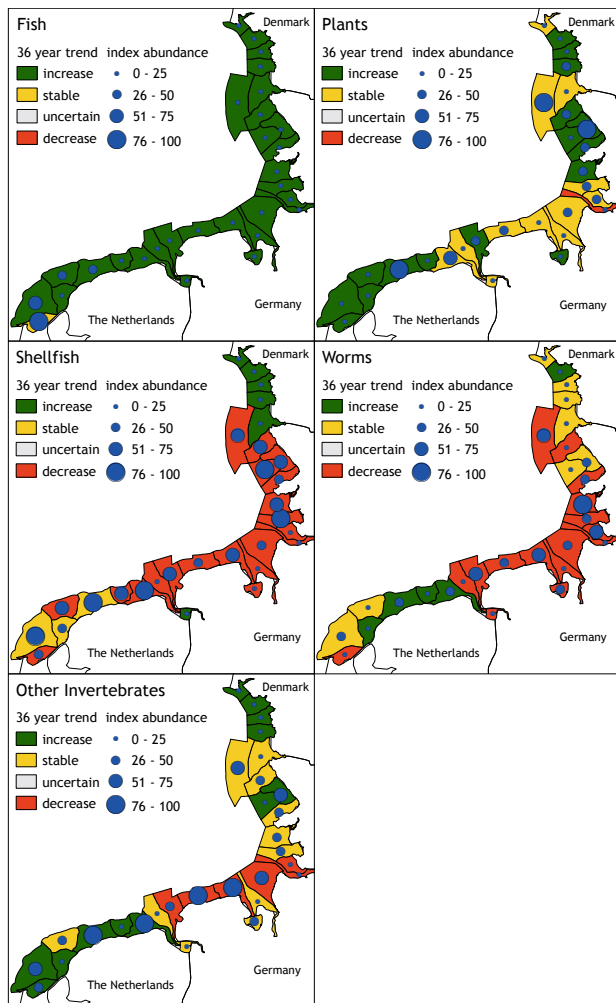
5.3.2 Trends in relation to diet

To help identify possible relationships between species trends and their diet, both for the entire Wadden Sea and at a regional level, trends of individual species were combined into 'dietary groups' (cf. Kleefstra *et al.* 2022b). Fish-eating bird species showed a long-term increase, but a stabilisation over the last ten years (Fig. 5.3a). This trend is somewhat distorted, because it is driven by the trends of the Great Cormorant *Phalacrocorax carbo* and Eurasian Spoonbill. Both species have increased strongly in the

long term, but the numbers of Great Cormorant have stabilised in almost the entire Wadden Sea over the last 10 years and have even decreased in some areas.

The trend of herbivores shows a moderate increase in both the long and short terms (Fig. 5.3), mainly due to increases in the Barnacle Goose, Northern Pintail, Eurasian Wigeon and Common Teal, and mainly in the Dutch and Schleswig-Holstein parts of the Wadden Sea.

A - Long term



B - Short term

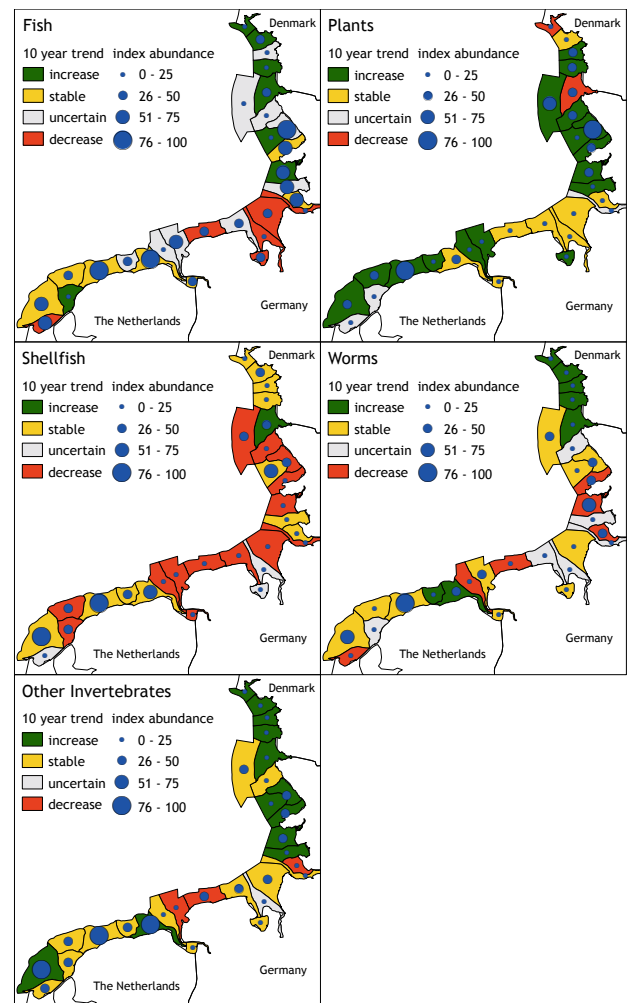
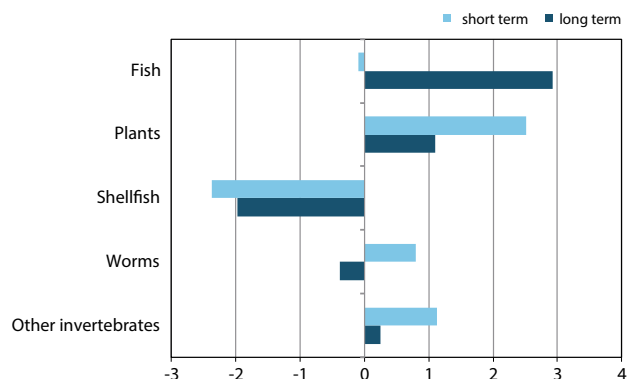


Figure 5.3. Combined long term (a) and short term trends (b) of Wadden Sea birds grouped into dietary guilds (feeding mainly on fish, plants, shellfish, worms, or other invertebrates), and divided by region in maps with fish, plants, shellfish, worms and other invertebrates. Index of abundance is % of the highest number per unit (highest number is the weighted average of all species per guild; see Methods).



While both the long-term and short-term trend of worm-eating species are stable for the entire Wadden Sea, the trends vary greatly per country/federal state (Fig. 5.3). In the long term, the German parts of the Wadden Sea show huge declines, while worm-eating birds increased in the Dutch part. In the western part of the Dutch Wadden Sea, worm-eating species stabilised in the last decade, as they did in parts of Lower Saxony and the northern part of the Schleswig-Holstein Wadden Sea. In the Danish Wadden Sea, worm-eating species still increased over the past decade. Common Ringed Plover (*psammmodroma/tundrae*-subspecies), Sanderling and Northern Lapwing are worm-eating species that did well over the whole period, while Ruff and Common Ringed Plover (*hiaticula*-subspecies) are increasing in the short run. For Northern Lapwing, Ruff and *hiaticula*-Common Ringed Plovers, an increase is seen, particularly in the northern part of the Wadden Sea. Worm-eaters that have become scarcer are mainly Pied Avocet, Curlew Sandpiper, Kentish Plover *Charadrius alexandrinus* and Bar-tailed Godwit (*taymyrensis*-subspecies). Although this decline occurs almost everywhere in the Wadden Sea, it appears to be strongest in the German parts.

The group of shellfish-eating birds shows a moderate decline over the whole monitoring period (Fig. 5.3a). In the long term, this is observed in the entire Wadden Sea, except the Danish part. In the last ten years, trends have been mainly negative in the German parts of the Wadden Sea and the western Dutch Wadden Sea (Fig. 5.3b). This negative trend applies to practically all species that are included in this group: Common Eider, Eurasian Oystercatcher, European Herring Gull, and both subspecies of Red Knot *Calidris canutus* (*islandica* and *canutus*). The numbers of *canutus* however seem to be stabilising in the northern part of the Wadden Sea, while the trend of *islandica* is uncertain in several regions. It is worth mentioning that *islandica* Knots are poorly covered by counts when roosting on the outer sands in the Schleswig-Holstein Wadden Sea in winter, and only partly covered there in late spring (late April / early May; K. Günther, Kempf *et al.* 2015).

Species that eat 'other invertebrates' (often with a broad diet including shellfish, snails, shrimps, crustaceans and worms) show a moderate increase in the long term (Fig. 5.3a). This increase is especially notable in the northern part of the Wadden Sea.

5.3.3 Trends in relation to breeding and wintering grounds

To further clarify trends it is interesting to look at migratory strategies of species. With respect to breeding range there is no large difference in trends between arctic and non-arctic breeding birds (Fig. 5.4). In both cases the overall long-term trend is stable. When looking at the wintering range, a moderate increase is shown by (sub)species wintering in Africa, while the trend for birds wintering in Europe is stable (Fig. 5.5).

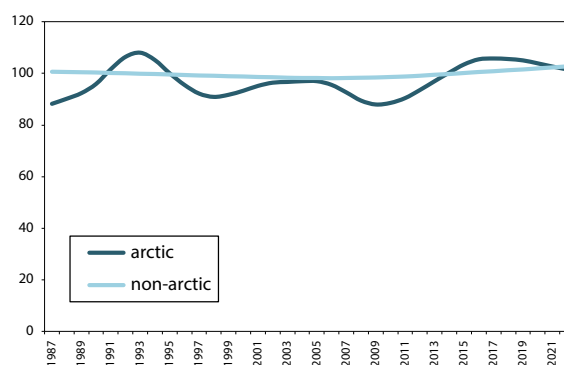


Figure 5.4. Combined long-term trends in the Wadden Sea of waterbirds grouped by breeding range.

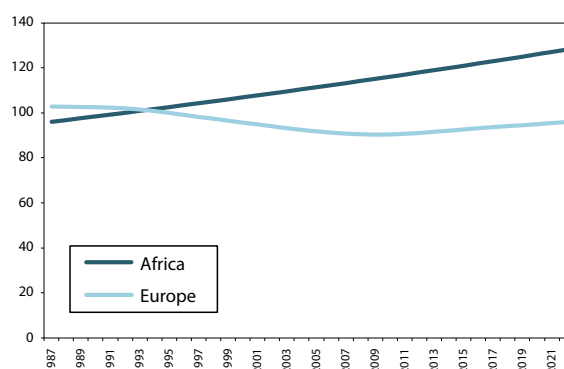


Figure 5.5. Combined long-term trends in the Wadden Sea of waterbirds grouped by wintering range.

5.3.4 Comparing recent monitoring data with the 1980s

We investigated whether total numbers of birds using the Wadden Sea changed between the time of one of the first Wadden Sea-wide publications covering data from the 1980s (Meltofte *et al.* 1994) and more recent periods. In Meltofte *et al.* (1994) an total number of 10-12 million waterbirds using the Wadden Sea was estimated, and this estimate has been widely used as the number of waterbirds depending on the Wadden Sea ever since. A direct recalculation of current numbers using the same methods as used by Meltofte *et al.* is hampered by the fact that that estimate was primarily based on a literature review and expert knowledge of the proportions of flyway populations which were likely to use the Wadden Sea. In doing so, both the maximum numbers counted at one moment and the likely turnover rate were taken into account.

Presently we have no clear and/or updated information about the total proportions of flyway populations, including turnover, using the Wadden Sea, but only estimates of maximum numbers present at one moment in time. To work around this, we analysed the total numbers present at one moment in 1980-1991 and in recent times. We used

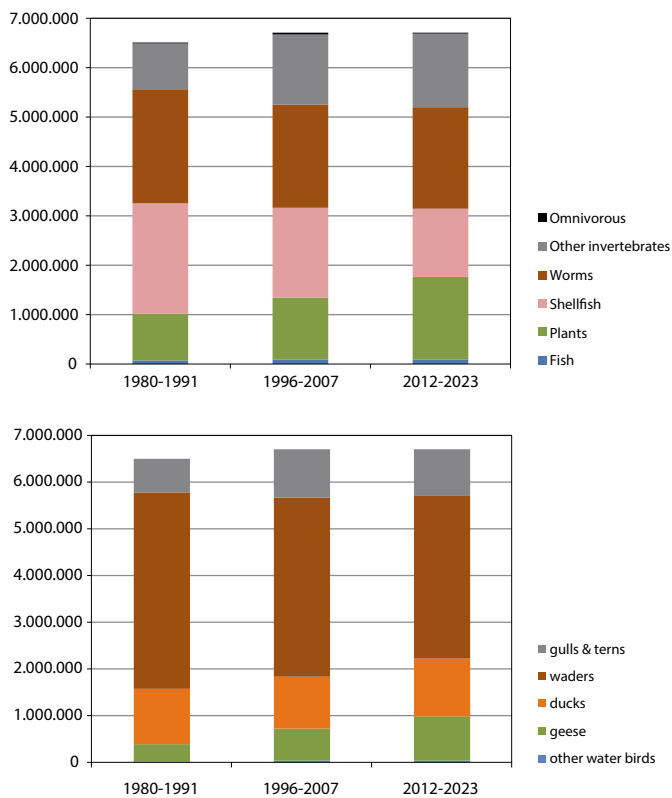


Figure 5.6. Comparison between maximum numbers present at one moment summed over species groups for the periods 1980-1991, 1996-2007 and 2012-2023, based on 36 counts spread over 8 different months in each 12-year period (see Methods).

the same selection of species/populations and also copied the number and timing of counts from the 1980s by sub-sampling the count data from two more recent periods. Figure 5.6 shows that the summed species/population maxima are not that different between the 1980s and recent periods. This suggests that there has been no significant change in total numbers of waterbirds using the international Wadden Sea; apparently increases and decreases among species are in balance (see below).

Although the total numbers of birds appear to have remained similar, there have been large shifts between species groups (Fig. 5.6). Today, herbivores, fish-eaters and species feeding on 'other invertebrates' are far more numerous than in the 1980s. On the other hand, the shellfish-eating birds have decreased and worm-eating species have remained stable overall. Among taxonomic groups it is mainly waders that have become scarcer, in particular the shellfish-eating wader species. As waders (of which relatively many winter further south and stage within the Wadden Sea during spring and autumn migrations) generally show more turnover of individuals in the course of the year than herbivores (of which more species occur in a more or less stable wintering population) and maybe some fish-eating species too, it could be that the total number of waterbirds using the Wadden Sea at some point in their annual cycle is currently lower than in the 1980s. Better knowledge of turn-over rates across many species may help substantiate this.



Common Shelduck | Tadorne de Belon (*Tadorna tadorna*)

Harvey van Diek

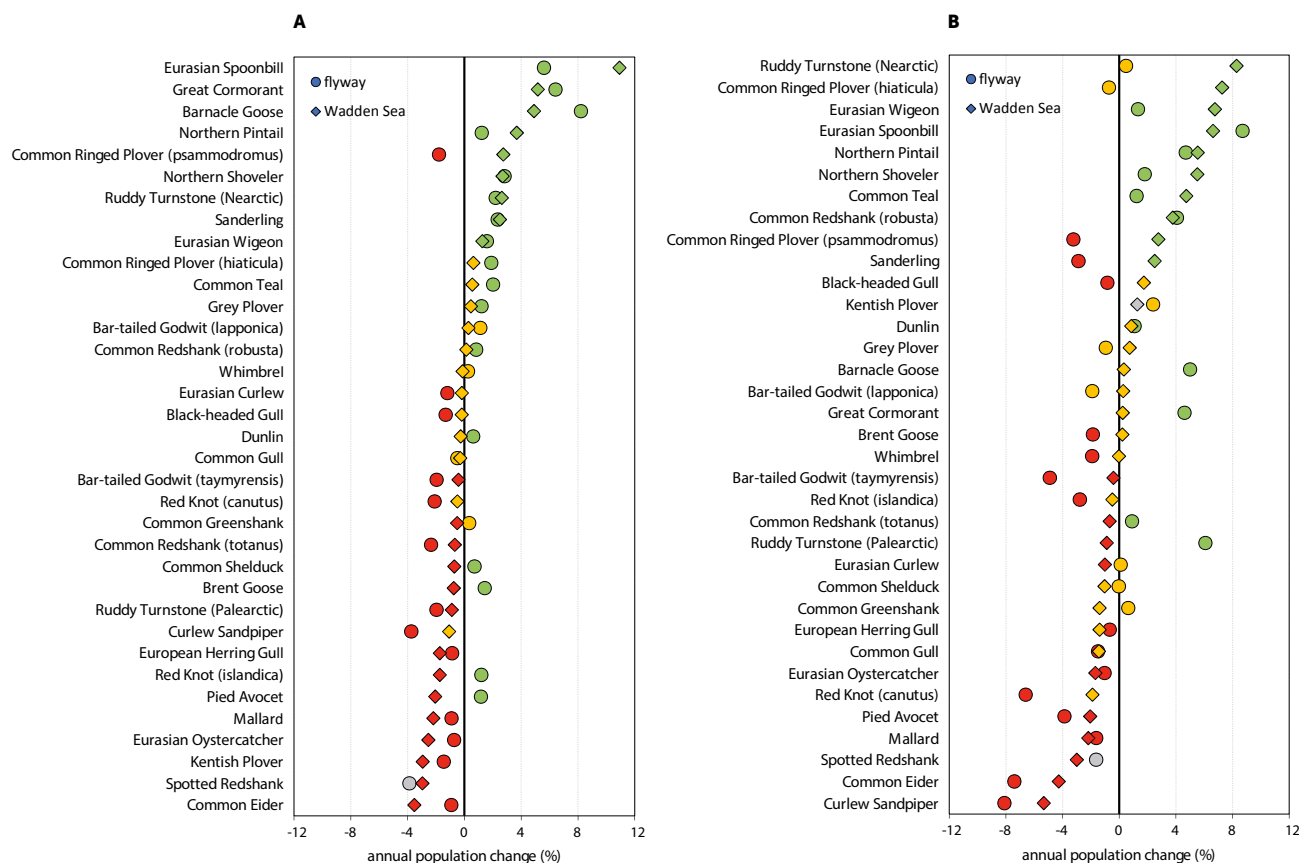


Figure 5.7. Comparison of trends in abundance of migratory birds in the Wadden Sea and at flyway level. Shown is the average rate of annual population change (in %) for the past 36 years (a) and (b) for the short term of 10 years. Species are ranked according to the rate of change in the Wadden Sea. Values above 0 denote increases, values below 0 decreases. Colours denote trend classification (red = decrease, yellow = stable, green = increase, grey = fluctuating).

5.3.5. Trends in the Wadden Sea compared with flyway trends

Previously, trends of migratory and wintering birds in the Wadden Sea until 2011 suggested that numbers declined faster within the Wadden Sea than at the flyway level, especially among species feeding on macrobenthos (e.g. van Roomen *et al.* 2015). Bregnballe *et al.* (2018) and Kleefstra *et al.* (2022a) showed that this pattern changed since then, with more of the benthic-feeding species showing stable or positive trends in the Wadden Sea. This observation still applies. The long-term trends show that 21 species are doing better at the flyway level than within the Wadden Sea (Fig. 5.7a). This is particularly true for Pied Avocet, Red Knot (*islandica*-subspecies), Brent Goose *Branta bernicla*, Common Shelduck, Common Greenshank *Tringa nebularia*, Dunlin *Calidris alpina*, Common Redshank (*robusta*-subspecies), Grey Plover *Pluvialis squatarola*, Common Teal and Common Ringed Plover (*hiaticula*-subspecies). Fourteen species do better in the Wadden Sea in the long-term than at flyway level (in particular Curlew Sandpiper, Red Knot (*canutus*-subspecies), Black-headed Gull *Chroicocephalus ridibundus* and Eurasian Curlew). In the short term, this picture is slightly different, with 19 species showing a more positive trend in the

Wadden Sea than at flyway level, and 15 that appear to be doing better at flyway level (Fig. 5.7b). Birds showing a particularly more favourable recent trend in the Wadden Sea are Red Knot (both subspecies), Whimbrel *Numenius phaeopus*, Brent Goose, Black-headed Gull, Sanderling, Common Ringed Plover (all subspecies) and Ruddy Turnstone (Greenland & NE-Canada). Species with a more favourable trend at flyway level include European Herring Gull, Eurasian Curlew, Ruddy Turnstone (Scandinavia-Western Russia), Common Redshank (*totanus* subspecies), Barnacle Goose and Dunlin.

When looking at dietary guilds again (Fig. 5.8), differences between trends in the Wadden Sea and at flyway level are not significant. In the case of fish-eating species, this is uncertain because this guild is a very limited set of species. Only the trend of the worm-eating species shows a significant difference; these do better in the Wadden Sea than at flyway level.

5.3.6 The importance of the Wadden Sea for migratory birds

Figure 5.9 shows the proportion of the flyway population of each species/subspecies that uses the Wadden Sea as a



Lars Soerink

Dunlin | Bécasseau variable (*Calidris alpina*)

staging, moulting and/or wintering area. Compared to a similar overview from the period 2010/11-2019/20 (Kleefstra *et al.* 2022b), it is noteworthy that Red Knot (two subspecies) and Bar-tailed Godwit (*lapponica* subspecies) now lead the list. The rest is largely the same. Of nine species, more than 50% of the flyway population occurs in the Wadden Sea. Furthermore, there are 18 species of which 10-50% uses the Wadden Sea. For the other 12 species in the list, this concerns 1-9% of the flyway population.

5.4 Discussion

When we look at the trend development of migratory birds in the Wadden Sea, we see that the number of species showing a positive trend is higher in the short term than in the long term (Fig. 5.1). This observation is consistent with previous reports on the trends of migratory birds in the international Wadden Sea (Bregnballe *et al.* 2018, Kleefstra *et al.* 2022a,c). In the long term, the Danish and Dutch parts of the Wadden Sea show the most positive trends, while in the German parts many species have decreased in numbers over the last 36 years (Figs. 5.2, 5.3). In the short term, fewer species show a negative trend also in the German Wadden Sea. We also see similar developments when comparing trends in the Wadden Sea to flyway trends. The long-term trends show that 21 species are doing better at the flyway level than within the Wadden Sea, while in the short term 19 species show a more positive trend in the Wadden Sea. This picture is consistent with that from the previous EAF assessment (Kleefstra *et al.* 2022a).

If we look over a longer period and consider the numbers of migratory birds since the 1980s, we see that total numbers counted have been consistent for almost 45 years (Fig. 5.6). Still, there are large changes between species and species groups. Herbivores, fish-eaters and species feeding on 'other invertebrates' are more numerous now than in the 1980s, but shell-fish eating birds have decreased and worm-eating species remained stable overall (but with strongly varying trends per species). It is mainly waders that have become scarcer. When taking into account that waders generally show more turnover than herbivores in the Wadden Sea, it is likely that the total number of waterbirds using the Wadden Sea was larger in the 1980s since there were more waders then.

If we look at which species are recently doing better (Fig. 5.7), it is worm-eaters whose trends in particular deviate positively from the flyway trend (Fig. 5.8). This dietary guild of birds feeding on worms (and often also other benthic invertebrates) consists of Pied Avocet, Common Ringed Plover, Kentish Plover, European Golden Plover, Grey Plover, Northern Lapwing, Sanderling, Curlew Sandpiper, Dunlin, Ruff and Bar-tailed Godwit. Within the Wadden Sea we continue to see contrasting trends in both the long and short term trends in different subregions (*cf.* Van Roomen *et al.* 2012, Ens *et al.* 2009). In Kleefstra *et al.* (2022b) the trends of various worm-eaters in the Danish and Dutch Wadden Sea deviated positively from those in the German parts. For Denmark this includes Pied Avocet, Kentish Plover and Eurasian Golden Plover, while for the Dutch Wadden Sea this applies to Grey Plover, Curlew

Sandpiper, Dunlin and Bar-tailed Godwit. Laursen *et al.* (2023) found significant relationships between the trend slopes of wader species and physical features such as sediment morphology, sea level rise and tidal amplitude, for the German parts of the Wadden Sea.

The group of fish-eating birds shows a more positive trend at flyway level than in the Wadden Sea, although this difference is not significant (Fig. 5.8), and the group of fish-eaters is limited. Eurasian Spoonbill and Great Cormorant dominate the picture, with the Great Cormorant showing a stable trend in the short term and a recent decrease in some parts of the Wadden Sea, related to the development of the breeding population (Koffijberg *et al.* 2020).

The guild of herbivores (Barnacle Goose *Branta leucopsis*, Dark-bellied Brent Goose, Eurasian Wigeon, Common Teal, Mallard and Northern Pintail) shows a positive trend in both the long and short term, in line with the flyway trend (Fig. 5.3). While Common Teal and Northern Pintail increase in all parts of the Wadden Sea in the short term, it is striking that Barnacle Goose, Dark-bellied Brent Goose and Eurasian Wigeon do so mainly in Schleswig-Holstein. As for the Barnacle Geese, this is partly due to the expansion of their distribution to the Halligen and islands in that area (Günther *et al.* 2024). It is also likely that the German (and Danish) Wadden Sea have become more important as numbers in the Netherlands have stabilised.

The guild of shellfish-eating birds contains four species (Common Eider, Eurasian Oystercatcher, Red Knot and Herring Gull) and shows a moderate decline in both the long and the short run (Fig. 5.3). Numbers of Common Eider, Red Knot (*canutus*) and Herring Gull seem to stabilise, while Eurasian Oystercatcher and *islandica*-Red Knot

continue their decrease. The long-term decline of all shellfish-feeding species is closely related to reduction of food stocks. The quantity and quality of bivalves like Blue Mussel *Mytilus edulis* and Edible Cockle *Cerastoderma edule* was seriously affected by shellfish fishery in various Wadden Sea regions (Piersma & Koolhaas 1997, Kraan *et al.* 2004, Scheiffarth & Frank 2005; Ens *et al.* 2009; Laursen *et al.* 2009). The decline of shellfish stocks is likely to have multiple causes. There are for instance indications for de-eutrophication effects (Singer *et al.* 2023), low recruitment due to mild winters (Meyer *et al.* 2021) and a reduced availability due to Pacific Oysters taking over mussel beds in the Wadden Sea (Scheiffarth *et al.* 2007). With depleted food stocks and no alternative food available or of low quality in the early 1990s, this had an immediate effect on the number of Common Eider (Ens & Kats 2004), and the condition and survival of Eurasian Oystercatchers (Verhulst *et al.* 2004). This also made Oystercatchers vulnerable to severe winters (Schwemmer *et al.* 2014). Even though the food stock is recovering (Baltic Tellin *Macoma balthica*, Cockles, Blue Mussels), the recovery of the shellfish-eating bird populations is lagging behind and numbers of Common Eiders, Red Knots (*canutus*) and Herring Gulls only started to stabilise in the last decade or so. For Common Eider also declines are reported for the Baltic breeding population, that will have an impact on wintering numbers in the Wadden Sea (Morelli *et al.* 2021, Lehtikoinen *et al.* 2022).

However, the supply of shellfish is still vulnerable, especially now that more frequent and intense heat waves can lead to mass mortality of shellfish (Beukema & Dekker 2020, Reneerkens 2020, Suyderbuyk *et al.* 2021). Further restrictive conditions that hinder population recovery might be found in the breeding grounds (e.g. suggested by Van Gils *et al.* 2016) as well as the wintering grounds

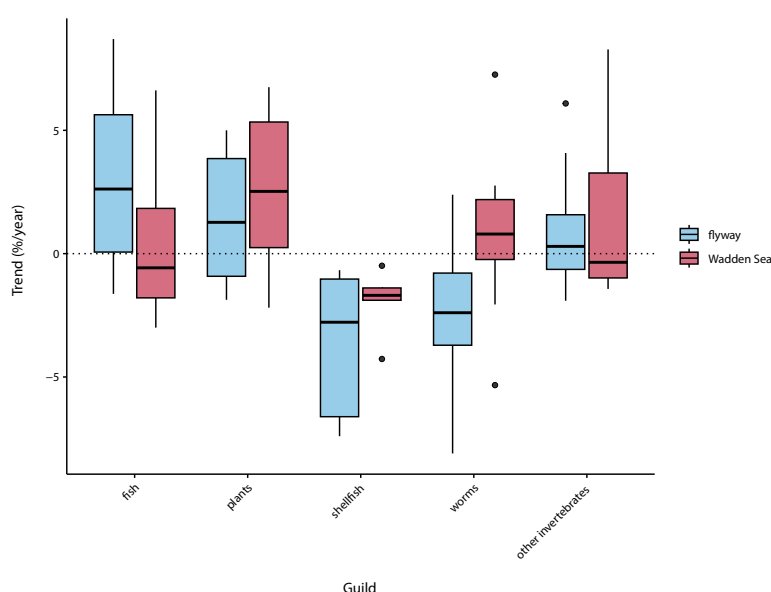


Figure 5.8. Comparison between short-term trends of food guilds (%/year) along the flyway and within the Wadden Sea.

proportion of flyway population (2013/2014 - 2022/2023) in %

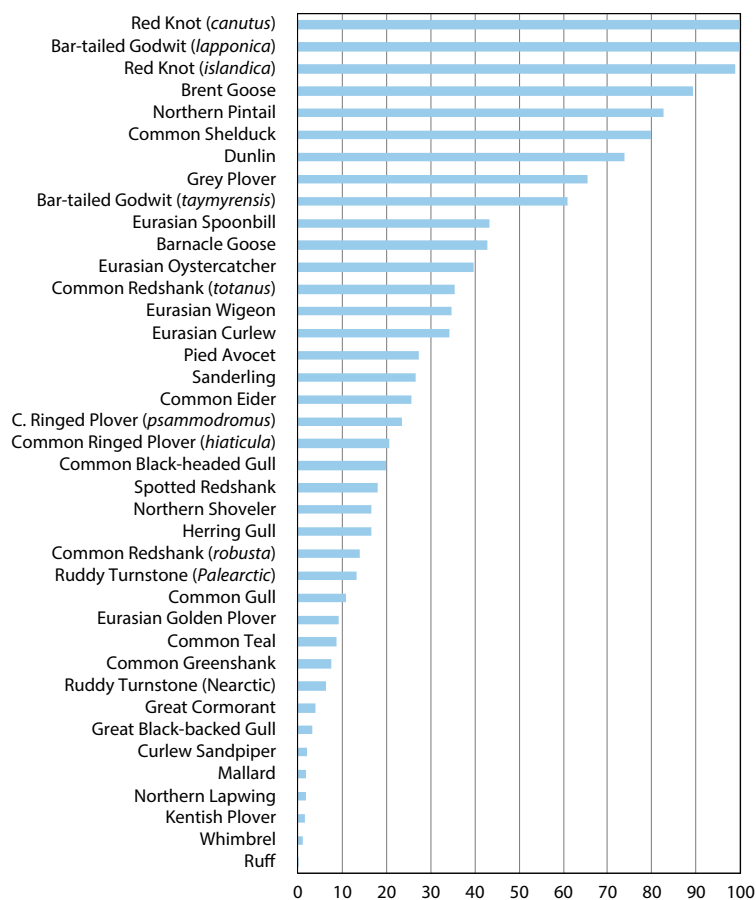


Figure 5.9. Proportion of the East Atlantic Flyway population that uses the Wadden Sea as a staging, moulting and/or wintering area.

(Leyrer *et al.* 2016). For Herring Gull, the reduction of Mussel stocks may provide only part of the explanation for the lower numbers at high tide roosts, as this omnivorous species also make use of other food sources.

The guild of species feeding on 'other invertebrates' consists of eight species: Common Shelduck, Northern Shoveler, Whimbrel, Eurasian Curlew, Common Redshank, Ruddy Turnstone, Black-headed Gull and Common Gull *Larus canus*. The long-term trend of this group shows a moderate increase, which stabilised in the short term (Fig. 5.3). The group seems to be doing particularly well in the northern part of the Wadden Sea, with trends of Northern Shoveler, Whimbrel, Common Redshank, Ruddy Turnstone (Greenland-NE Canada) deviating positively from trends in the southern part of the Wadden Sea. The latter also applies to Eurasian Curlew and Black-headed Gull in the Danish Wadden Sea. Blew *et al.* (2017) already noted that the food (a wider variety of 'other invertebrates') separates this group from the group of worm-eating birds, but it includes species that use different prey as well as a variety of bird taxa, such as ducks, waders and gulls. This variety makes an assessment of this group difficult, although variation in trends between the species also seems to be related to food availability, like the contrasting

trends that worm-eating species show within the Wadden Sea (Kleefstra *et al.* 2022c).

Based on the counts in the Wadden Sea no difference is found between the long-term trends of species breeding in the Arctic and elsewhere respectively (Fig. 5.4). When looking at wintering range, bird populations wintering in Africa overall seem to do better in the Wadden Sea (moderate increase) than populations wintering mainly in Europe (stable; Fig. 5.5). Such comparisons always have caveats, since both composite species groups contain species that are increasing in number and those that are decreasing. However, this could fit in with the observation that within the EAF as a whole, populations wintering along the African coasts south of Morocco tend to have decreased since the 1980s-1990s while those wintering further north more often show increases (Schekkerman *et al.* 2022, Chapter 2). If this pattern is partly caused by a northward shift of birds formerly wintering in Africa, we may be seeing this in the Wadden Sea at present. Species that winter in Africa and are increasing are Spoonbill (increasingly wintering in Europe), Pintail, Common Ringed Plover and Sanderling, while Red Knot, Curlew Sandpiper and Spotted Redshank show the opposite. That such comparisons have caveats also applies to species

that winter in Europe, although several of the species that are doing poorly also show shrinking breeding populations, e.g. Mallard, Common Eider, Eurasian Oystercatcher, Pied Avocet, Eurasian Curlew and European Herring Gull (Koffijberg *et al.* 2020).

Van Roomen *et al.* (2020) observed that flyway populations for which the Wadden Sea is an important staging site in part of their annual cycle are not doing worse than populations which are less dependent on the Wadden Sea. Where the assessment following the first total EAF count in 2014 raised concern that the Wadden Sea formed a

'weak link' in the flyway (van Roomen *et al.* 2015), the 2020 assessment found this 'weak link' effect to be no longer apparent among migratory birds. Although the newest trend indications still support this, numerous developments in the Wadden Sea may affect migratory birds negatively. Kleefstra *et al.* (2022c) outlined several pressure factors and unfavourable developments, including climate change, military exercises and increasing recreation. All this gives reason to keep a close eye on the situation with optimal monitoring and adequate research, also because negative developments still dominate among trends of waterbird breeding in the Wadden Sea.



Harvey van Diek

Pied Avocet | Avocette élégante (*Recurvirostra avosetta*)



Dunlin & Curlew Sandpiper | Bécasseau variable & Bécasseau cocorli (*Calidris alpina* & *Calidris ferruginea*) (Jacob Garvelink / Agami)

6. From local impacts to flyway conservation: exploring potential effects of Montijo airport using GPS tracked waterbirds in the Tagus estuary protected area

José A. Alves, Afonso D. Rocha, João R. Belo, Manuela Rodrigues, Joshua Nightingale & Camilo Carneiro

Summary

Achieving effective conservation of migratory waterbirds requires international cooperation between signatories of international treaties and law, but countries are currently failing to protect this highly mobile biodiversity. Promoting waterbird productivity does not translate to positive population growth (or even stability) if non-breeding survival is completely disregarded and vice-versa. A recent threat to waterbirds wintering in the most important wetland for these species in Portugal, the Tagus estuary, originated from a positive environmental impact declaration issued in 2020. This allowed the establishment of a commercial airport in the heart of this wetland, the "Montijo airport" with aircrafts overflying the protected area designated for birds. Here we provide an update on the state of affairs regarding this development and present further evidence of the negative effects expected to arise from it, using recent data from GPS tracking of wintering and breeding waterbirds. We give particular attention to the compliance, or lack of it, with the European Union's Habitats (Article 6) and Birds Directives (Article 4), as this site is listed as Natura 2000, as are many other wetlands in the EU territory of the East Atlantic Flyway. We show that for all four species tracked the level of noise disturbance produced by aircrafts overlying the protected area will: (1) have a "significant effect on the conservation objective of migratory birds" that occur in the protected area, and (2) "affect the integrity of the site". We further discuss the importance for countries whose species may be under pressure elsewhere in their distribution range to frequently revise potential threats and stressors arising beyond their territory, and enforce the shared international responsibility for the conservation of those species by holding other countries accountable at appropriate forums.

Résumé

La conservation efficace des oiseaux d'eau migrateurs nécessite une coopération internationale entre les signataires des traités et lois internationaux, mais les pays échouent actuellement à protéger cette biodiversité hautement mobile. Promouvoir la productivité des oiseaux

ne se traduit pas nécessairement par une croissance positive des populations (ni même une stabilité) si la survie hors période de reproduction est ignorée, et vice-versa. Une menace récente pèse toujours sur les oiseaux d'eau hivernants dans la zone humide la plus importante du Portugal pour ces espèces : l'estuaire du Tage. Cela malgré plusieurs tentatives pour annuler la licence environnementale délivrée en 2020, qui a autorisé la construction de l'aéroport commercial de Montijo, situé en plein cœur de cette zone humide, survolant une aire protégée. Nous faisons ici le point sur la situation actuelle et apportons de nouvelles preuves des effets négatifs attendus, en utilisant des données récentes de suivi GPS d'oiseaux d'eau en période de reproduction et d'hivernage. Nous mettons l'accent sur le respect (ou le non-respect) des directives européennes « Habitats » (article 6) et « Oiseaux » (article 4), car ce site fait partie du réseau Natura 2000, comme de nombreuses autres zones humides de la voie de migration de l'Atlantique Est. Pour toutes les quatre espèces suivies, les niveaux de perturbation sonore causés par les avions auront : (1) un « effet significatif sur l'objectif de conservation des oiseaux migrateurs » présents sur le site protégé, et (2) « affecteront l'intégrité du site ». Nous soulignons également l'importance pour les pays, dont les espèces subissent des pressions au-delà de leur territoire, de réévaluer fréquemment les menaces externes et de faire respecter la responsabilité internationale partagée en tenant d'autres pays responsables dans les forums appropriés.

6.1 Introduction

The conservation of migratory shorebirds is extremely challenging. These species have disjunct breeding and non-breeding areas, typically separated by thousands of kilometres, and frequently use stop-over sites between them to rest and refuel during their migrations (van de Kam 2004). This creates wide distribution ranges often spanning more than one continent, across which species establish and rely upon an intricate network of sites that is of fundamental importance for individuals to complete their annual cycle and survive and breed successfully. International cooperation between multiple nations is

therefore of paramount importance for the conservation of migratory shorebirds (Stroud 2022, Prokosch 2024). For example, if a given country invests heavily on improving and managing breeding sites for a target shorebird, but other countries that host this same species during the non-breeding season allow the degradation or promote the disappearance of its sites (by habitat conversion), then this species of concern will very likely decline, and conservation efforts and funds have been wasted.

Although several international treaties, which are termed conventions or agreements, are in place for the conservation of migratory species (CMS), waterbirds in Africa and Eurasia (AEWA), and their wetland habitats (RAMSAR), only among shorebird, 16 species were recently reclassified to higher threat categories on the IUCN Red List (IUCN 2024). Several of these species that occur in the East Atlantic Flyway have declined by more than 30% worldwide (over three bird generations), such as the Grey Plover *Pluvialis squatarola* and Curlew Sandpiper *Calidris ferruginea*, which have been updated to globally *Vulnerable* (from *Least Concern* and *Near Threatened*, respectively; IUCN 2024). For these species with populations occurring in Europe, as well as for Turnstone *Arenaria interpres* and Dunlin *Calidris alpina*, such treaties, including the Bern Convention, are currently failing to halt declining trends and thus the ongoing biodiversity crises. It is nonetheless the responsibility of signatory nations of those agreements

to uphold site and habitat quality for the listed species and promote their conservation (Stroud 2022). This is particularly relevant for sites in the European Union that are part of the Natura 2000 Network, as funding for the conservation of such sites is chiefly provided by the LIFE Program. However, even in what is claimed to be “the largest coordinated network of protected areas in the world” (EEA 2024), international cooperation is often lacking. Local, i.e. national, developments and actions that might negatively affect important sites for a given species during the non-breeding season, can occur without any knowledge of such change in countries elsewhere in the flyway, including those benefiting from LIFE funding for the conservation of breeding sites of the same species. Furthermore, the degradation of only one of these sites may have consequences that extend far beyond any local depression in numbers (Beal *et al.* 2025). For instance, if site quality is reduced, individual capacity to fuel for migration might be constrained (Anderson *et al.* 2021) which can result in lower survival (Piersma *et al.* 2016) and this may further propagate across the flyway, for example by decreasing productivity due to fewer breeding pairs or by mate exchange following partner loss. It is therefore paramount that countries uphold their statutory commitments if we, as a society, are serious about the conservation of migratory shorebirds.



Ernst Schade

Black-tailed Godwit | Barge à queue noire (*Limosa limosa*)

The Tagus estuary is the largest and most important wetland for waterbirds in Portugal, including migratory shorebirds. It typically hosts 200,000 individuals in the non-breeding stationary period but during migration this figure is estimated to increase to 300,000 (Encarnação pers. comm.). Owing to its location in the outskirts of Lisbon, the country's capital, it is affected by multiple threats such as urban development, tourism (and associated disturbance) and habitat conversion, among others. In January 2020 the Portuguese Environmental Agency issued a positive environmental impact declaration allowing to move forward with the plans for the construction of an international airport by converting and expanding an air force base located along the banks of this wetland, on the Montijo peninsula (Alves 2020, Alves & Dias 2020). Despite this infrastructure being projected to be mostly located outside of the protected area of the Tagus estuary Natura 2000 site on the ground, the proposed flight paths mean that commercial aircrafts would overfly the protected area at low altitudes (c. 200-500m) with a minimum frequency of approximately one per 5 minutes (24 movements/hour; EIA 2019). Although this environmental impact declaration expired in 2024 and its renewal, as requested by the private developer, was not granted by the authorities given new evidence that has meanwhile emerged (Nightingale *et al.* 2023), there is currently no ruling on the court action requesting the annulment of that declaration.

This study investigates potential effects resulting from the implementation of a commercial airport at Montijo peninsula, on the estuarine avifauna that occurs in the Tagus estuary protected area (Special Protection Area for Birds - SPA No. PTZPE0010 and Special Area of Conservation - SAC No. PTCON0009). To do so we use recently collected data (2020-2023) amassed using state-of-the-art technology (GPS), on the usage of this protected area and its habitats by waterbirds that occur locally, and to confront this with the levels of noise disturbance caused by aircrafts overflying the protected area, as described in the respective Environmental Impact Study (EIS) of the Environmental Impact Assessment procedure (EIA) concerning the "Montijo Airport". We aimed to obtain information that allows assessing compliance, or lack thereof, with the Birds and Habitats Directives that regulate the conservation and management of species and habitats in sites designated as Natura 2000, such as the Tagus estuary. Special attention was given to specific stipulations of these directives, in particular whether the disturbances resulting from Montijo Airport operation (i) have a significant effect on the conservation objectives of migratory birds that occur in the protected area (Birds Directive, Article 4 nr 4 - and more broadly also indicated in the Habitats Directive, Article 6 nr 2) and whether (ii) they affect the integrity of the site (Habitats Directive Article 6, nr 3). It is understood that a disturbance will have a significant effect on the conservation of a particular species, for example, when it will affect the natural utilisation of the protected

area and the different habitats therein by the species in question, namely by reducing the species' natural range within the site (EU 2019). That is, when the behaviour of a given species in terms of habitat use (in a spatially explicit way), as recorded before the future disturbance, is altered by it. Similarly, the integrity of a site (i.e. ecological integrity) is understood to be affected when it ceases to be whole or complete, for example by the degradation of habitats (reduction in their representativeness) or the alteration of the composition (reduction in surface area) of those habitats (EU 2019). The same applies to species habitats, for example wetlands for birds (EU 2019).

The disturbance in this study concerns the noise caused by aircraft overflights, as presented in the EIA. In order to investigate the effects of such noise disturbance on the avifauna that occurs in the protected area, particularly waterbirds and its habitats, the following specific objectives were established for this study: (1) quantify the main estuarine habitats for several waterbird species based on its proportional use (establishing a pre-disturbance benchmark); (2) determine the variation in the use of these estuarine habitats throughout the tidal and circadian cycles (knowing from previous studies that these cycles can influence their use); (3) quantify the frequency of locations of these birds in the areas affected by the 55 and 65 dB noise cones (see below), and what proportion of these occur inside the protected area designated by the Special Protection Area (SPA).

6.2 Materials and Methods

6.2.1 Disturbance levels and map layers

The area of noise on the ground that will be produced at different levels by a commercial aircraft given is termed "noise cone" and here we use the information for the proposed runways for Montijo airport as estimated in the EIA. As in the original study considered in the EIA (Wright *et al.* 2010), the noise level that presents no disturbance (i.e. above which birds showed abnormal behaviour) is 50 decibel (dB). Above this noise value, the probability of a behavioural change occurring is higher than zero. Hence, disturbance on birds, as defined above, is considered to occur above this value. Since the EIA presents the noise cones at intervals of 5 dB, the noise levels considered here are 55 dB (the first value presented in the EIA in which there is disturbance) and 65 dB, the value indicated in the environmental impact declaration (DIA 2020) as that for which impact compensation measures should be imposed. The 65dB and 55dB noise cones were therefore taken from the EIA. The habitat map for the Tagus estuary was drawn up from the CORINE Land Cover land use map (EU 2018), and the layer corresponding to the intertidal mudflat habitat was obtained from www.intertidal.app (Murray *et al.* 2019), with the boundaries of the Tagus estuary SPA and the Tagus Estuary Nature Reserve (RNET)

obtained from <https://ec.europa.eu/environment/nature/natura2000>.

6.2.2 Capturing and tagging birds with GPS transmitters

In order to deploy GPS transmitters on wintering shorebirds we carried out several mist-netting sessions across the Tagus estuary SPA (N 38.741; W -8.982) between 2020 and 2022, starting in October of each year. Sixty-two Pathtrack GPS/UHF transmitters and eleven HQSX GPS/GSM transmitters were placed on individuals of three species: Black-tailed Godwit *Limosa limosa* (henceforward godwit), Grey Plover *Pluvialis squatarola* and Common Redshank *Tringa totanus* (henceforward redshank).

Despite the limited number this selection encompasses a relevant representation of the wintering shorebird community, with species occupying different ecological niches

in this ecosystem given their feeding techniques and dietary variation. The godwit is predominantly tactile in its search for food, consuming mainly polychaetes, bivalves and rice seeds (left in the fields after harvest), while the grey plover is mostly a visual predator of polychaetes and crustaceans, and the redshank is a 'mixed' predator, using sight and touch to obtain its food, which is mostly composed of bivalves, gastropods, polychaetes and small crustaceans (Moreira 1995, Martin & Piersma 2009, Alves *et al.* 2013a).

Pathtrack transmitters weighed 4.0 g (for redshank) or 5.0 g (for the other two species), while the HQSX transmitters weighed 4.5 g. In all cases, the GPS transmitter and its harness weighed less than 4% of the weight of each bird (redshank 3.9%, grey plover 2.8%; godwit 2.1% in all cases ± 0.5 SE), as recommended (Barron *et al.* 2010, Weiser *et al.* 2016). Three receiving stations were placed at high tide

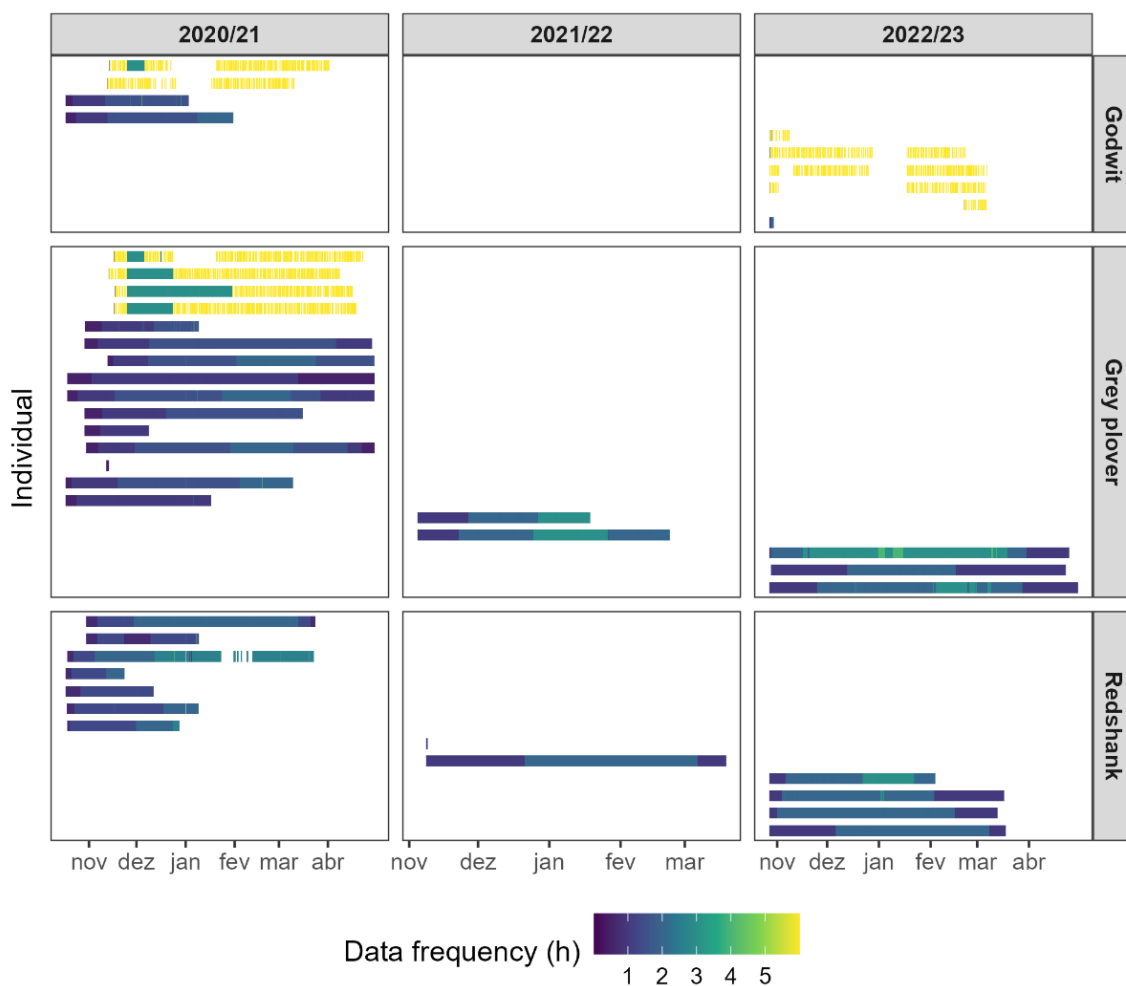


Figure 6.1. Variation in the frequency of location data acquisition over the tracking period for individuals of three species of wintering shorebirds, captured and tagged with GPS transmitters in the Tagus estuary SPA. Each location obtained (vertical line) is coloured according to the time elapsed since the previous location. Periods with no data are shown in white (no locations), due to temporary low voltage in the battery of the transmitter (hence no data acquisition). Only locations with data intervals up to 6 hours are considered.

roosts to retrieve the data from the GPS/UHF transmitters, taking advantage of the high fidelity of wintering shorebirds to their resting places at high tide (Alves *et al.* 2012, Catry *et al.* 2012, Lourenço *et al.* 2014). Even so, it was not possible to obtain data from several of these transmitters. The reason for the lack of transmission remained unclear and may include birds remaining out of reach of the stations or even migrating to other areas (if on active migration at time of capture), their premature death, or technical failure of the transmitters.

Pathtrack transmitters were programmed to record a GPS position every 30 minutes (in godwits and grey plovers) and every 45 minutes (in redshank), and the HQSX transmitters every 4 hours (all species). However, as these devices depend on exposure to sunlight to (re)charge batteries, the frequency of data acquisition was not constant. Pathtrack transmitters automatically decreased the data acquisition frequency when the battery level dropped below a voltage threshold and returned to the initial frequency when the battery was recharged. In the HQSX transmitters, this management was done manually via the GSM network, changing the frequency to 6-hour intervals when the battery was not maintaining the voltage level at the initially programmed data acquisition frequency (Fig. 6.1).

GPS transmitters were also used to mark a breeding waterbird species in the Tagus estuary SPA, the Eurasian

spoonbill *Platalea leucorodia* (henceforward spoonbill). Nineteen Movetech-Telemetry GPS/GSM transmitters were deployed in pre-fledging juveniles in July 2020 (n=3), June 2021 (n=7), June 2022 (n=4) and June 2023 (n=5) at the breeding colony located at Mouchão da Póvoa (N 38.86; W -9.05). These transmitters weigh 33 g, corresponding on average to 2.1% ($\pm$ 0.1 SE) of the birds' mass, and were programmed to acquire a GPS location every 30 minutes between 5am and 9pm and: one additional location at 1am (2020 & 2021); or a GPS locations every two hours outside that period (2022 & 2023). Like with the other transmitters sun exposure sometimes limited the frequency of data acquisition (Fig. 6.2). Two transmitters deployed in 2021 and another two in 2022 did not yield relevant information (stopped working in less than three weeks post deployment).

6.2.3 Data analysis

As one of the aims of this work was to determine the use of habitats throughout the tidal cycle by wintering shorebirds, and the tidal cycle in the Tagus estuary has 12 hours and 25 minutes (Freire 1999), the data was first filtered by removing locations obtained more than 6 hours after the previous location. This ensured that no location was more than 6 hours away from the closest tidal peak (low or high tide). Applying this filter reduced the total number of locations recorded by 3.4%. This resulted in information on 43 individuals (10 godwits; 20 grey plovers and 13 redshanks)

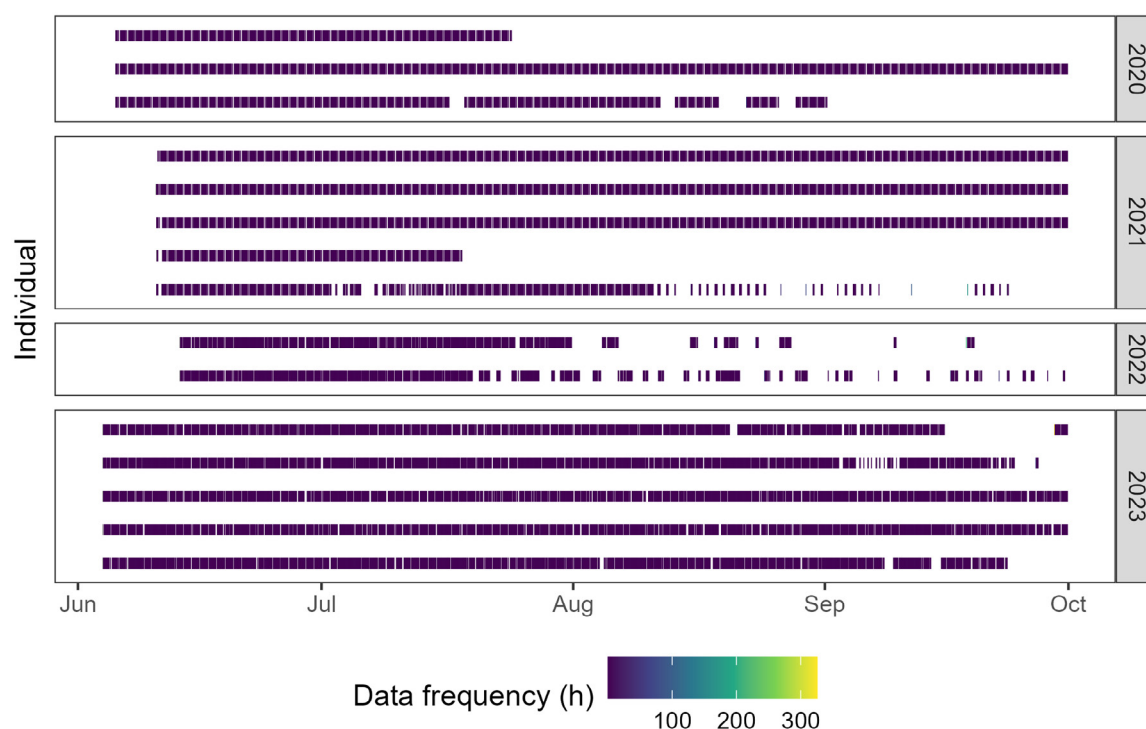


Figure 6.2. Variation in the daily frequency of location data acquisition over the tracking period of individuals of a breeding waterbird species (*P. leucorodia*), tagged in the Tagus estuary SPA with GPS tracking devices. Each location obtained (vertical line) is coloured according to the time elapsed since the previous location. Periods with no data are shown in white (no locations), due to a temporary low voltage in the battery of the transmitter.

Table 1. Summary information collected from GPS transmitters deployed on shorebirds wintering in the Tagus estuary SPA and tracked during three non-breeding seasons; and on juvenile spoonbills that hatched in the SPA between 2020 and 2023 breeding seasons. (Lim lim = Black-tailed Godwit; Plu squ = Grey Plover; Tri tot = Common Redshank; Pla leu = Eurasian Spoonbill).

Species	Transmitter	First transmission (date & time)	Last transmission (date & time)	Tracking period (days)	N of locations	Average interval ($\pm$ SE) between locations (hours)
Lim lim	hqsx	21/02/2023 06:00	06/03/2023 06:00	13	22	6.0 ($\pm$ 0.000)
Lim lim	hqsx	27/10/2022 10:00	05/03/2023 22:00	130	92	6.0 ($\pm$ 0.000)
Lim lim	hqsx	27/10/2022 10:00	06/03/2023 16:00	130	179	6.0 ($\pm$ 0.000)
Lim lim	hqsx	27/10/2022 10:00	21/02/2023 18:00	117	166	6.0 ($\pm$ 0.023)
Lim lim	hqsx	27/10/2022 10:00	08/11/2022 06:00	12	20	5.8 ($\pm$ 0.205)
Lim lim	hqsx	12/11/2020 16:00	10/03/2021 16:00	118	157	6.0 ($\pm$ 0.000)
Lim lim	hqsx	13/11/2020 22:00	01/04/2021 16:00	139	275	5.1 ($\pm$ 0.084)
Lim lim	pathtrack	27/10/2022 11:18	29/10/2022 16:18	2	41	1.3 ($\pm$ 0.074)
Lim lim	pathtrack	17/10/2020 09:55	30/01/2021 20:20	105	1961	1.3 ($\pm$ 0.011)
Lim lim	pathtrack	17/10/2020 09:20	02/01/2021 16:26	77	1508	1.2 ($\pm$ 0.011)
Plu squ	hqsx	16/11/2020 22:00	18/04/2021 16:00	153	466	4.5 ($\pm$ 0.069)
Plu squ	hqsx	17/11/2020 16:00	16/04/2021 16:00	150	690	3.7 ($\pm$ 0.048)
Plu squ	hqsx	13/11/2020 16:00	08/04/2021 04:00	145	452	4.5 ($\pm$ 0.071)
Plu squ	hqsx	16/11/2020 22:00	22/04/2021 22:00	157	310	5.2 ($\pm$ 0.077)
Plu squ	pathtrack	27/10/2022 09:00	29/04/2023 23:40	185	2847	1.6 ($\pm$ 0.012)
Plu squ	pathtrack	28/10/2022 09:40	22/04/2023 16:14	176	3441	1.2 ($\pm$ 0.007)
Plu squ	pathtrack	27/10/2022 10:30	24/04/2023 18:16	179	1952	2.2 ($\pm$ 0.021)
Plu squ	pathtrack	04/11/2021 16:42	22/02/2022 10:28	110	1404	1.9 ($\pm$ 0.018)
Plu squ	pathtrack	04/11/2021 16:30	18/01/2022 15:30	75	1079	1.7 ($\pm$ 0.023)
Plu squ	pathtrack	17/10/2020 10:05	16/01/2021 21:26	91	2351	0.9 ($\pm$ 0.004)
Plu squ	pathtrack	17/10/2020 09:45	09/03/2021 15:19	143	2500	1.4 ($\pm$ 0.008)
Plu squ	pathtrack	12/11/2020 08:05	13/11/2020 08:34	1	50	0.5 ($\pm$ 0.000)
Plu squ	pathtrack	30/10/2020 09:40	29/04/2021 23:32	182	3462	1.3 ($\pm$ 0.008)
Plu squ	pathtrack	29/10/2020 09:20	08/12/2020 12:35	40	1203	0.8 ($\pm$ 0.007)
Plu squ	pathtrack	29/10/2020 09:35	15/03/2021 20:06	137	2880	1.1 ($\pm$ 0.007)
Plu squ	pathtrack	18/10/2020 10:35	29/04/2021 23:03	194	3571	1.3 ($\pm$ 0.007)
Plu squ	pathtrack	18/10/2020 10:25	29/04/2021 23:43	194	6133	0.8 ($\pm$ 0.003)
Plu squ	pathtrack	13/11/2020 01:59	29/04/2021 23:13	168	2764	1.5 ($\pm$ 0.008)
Plu squ	pathtrack	29/10/2020 09:30	28/04/2021 15:09	181	3610	1.2 ($\pm$ 0.006)
Plu squ	pathtrack	29/10/2020 20:28	09/01/2021 01:15	71	1652	1.0 ($\pm$ 0.012)
Tri tot	pathtrack	27/10/2022 09:12	17/03/2023 19:06	141	2283	1.5 ($\pm$ 0.010)
Tri tot	pathtrack	27/10/2022 09:11	12/03/2023 19:09	136	1992	1.6 ($\pm$ 0.011)
Tri tot	pathtrack	27/10/2022 09:14	16/03/2023 19:09	140	2263	1.5 ($\pm$ 0.011)
Tri tot	pathtrack	27/10/2022 09:21	03/02/2023 13:13	99	1183	2.0 ($\pm$ 0.019)
Tri tot	pathtrack	08/11/2021 10:06	18/03/2022 19:08	130	2225	1.4 ($\pm$ 0.010)
Tri tot	pathtrack	08/11/2021 10:56	08/11/2021 19:56	9	10	1.0 ($\pm$ 0.000)
Tri tot	pathtrack	18/10/2020 10:00	27/12/2020 23:35	71	1123	1.5 ($\pm$ 0.012)
Tri tot	pathtrack	18/10/2020 07:56	08/01/2021 22:26	83	1427	1.4 ($\pm$ 0.010)
Tri tot	pathtrack	17/10/2020 09:05	11/12/2020 12:53	55	1163	1.1 ($\pm$ 0.009)
Tri tot	pathtrack	17/10/2020 09:10	23/11/2020 03:52	37	656	1.3 ($\pm$ 0.016)
Tri tot	pathtrack	18/10/2020 10:05	22/03/2021 19:03	155	1557	2.1 ($\pm$ 0.018)
Tri tot	pathtrack	30/10/2020 09:50	09/01/2021 01:39	71	1684	1.0 ($\pm$ 0.009)
Tri tot	pathtrack	30/10/2020 09:55	23/03/2021 19:50	144	2139	1.6 ($\pm$ 0.011)
Pla leu	movetech	05/06/2020 15:25	01/09/2020 06:00	88	2553	0.8 ($\pm$ 0.048)
Pla leu	movetech	05/06/2020 15:32	30/09/2020 20:48	117	3744	0.7 ($\pm$ 0.013)
Pla leu	movetech	05/06/2020 15:30	24/07/2020 09:22	49	1593	0.7 ($\pm$ 0.021)
Pla leu	movetech	10/06/2021 16:47	23/09/2021 14:10	105	1986	1.2 ($\pm$ 0.123)
Pla leu	movetech	10/06/2021 17:19	18/07/2021 06:34	38	1231	0.7 ($\pm$ 0.025)
Pla leu	movetech	10/06/2021 16:12	30/09/2021 20:52	112	3747	0.7 ($\pm$ 0.013)
Pla leu	movetech	10/06/2021 15:40	30/09/2021 20:51	112	3723	0.7 ($\pm$ 0.013)
Pla leu	movetech	10/06/2021 19:35	30/09/2021 20:58	112	3745	0.7 ($\pm$ 0.013)
Plaleu	movetech	13/06/2022 13:12	30/09/2022 13:16	109	1797	1.5 ($\pm$ 0.131)
Plaleu	movetech	13/06/2022 13:21	19/09/2022 08:52	98	1654	1.4 ($\pm$ 0.270)
Plaleu	movetech	04/06/2023 01:19	23/09/2023 10:44	111	3174	0.8 ($\pm$ 0.018)
Plaleu	movetech	04/06/2023 01:21	30/09/2023 21:18	119	3160	0.9 ($\pm$ 0.014)
Plaleu	movetech	04/06/2023 01:16	30/09/2023 20:53	119	3251	0.9 ($\pm$ 0.013)
Plaleu	movetech	04/06/2023 01:25	27/09/2023 06:28	115	3217	0.9 ($\pm$ 0.025)
Plaleu	movetech	04/06/2023 01:09	30/09/2023 21:03	119	2926	1.0 ($\pm$ 0.112)

between 17 October 2020 and 29 April 2023 (Table 1). As spoonbill can feed during both low and high tide (Triplet *et al.* 2008), all locations attained for the 15 individuals of this species were included in the analyses. The monitoring period considered was between June and September, since in October migrant spoonbills from populations further north arrive in the Tagus estuary (Lourenço *et al.* 2018). Furthermore, only locations obtained in the Tagus estuary area were considered, as some individuals dispersed elsewhere before the 30th of September.

After processing the GPS locations, these were annotated with the habitat in which they were recorded (typical GPS error 3-5 m). This made it possible to determine the most important estuarine habitats for these species, assuming that the percentage of locations obtained in a given habitat (or area) corresponds to the frequency of use by each individual during the period considered (Obj 1). Locations were also classified as lying inside or outside the SPA and in the areas under the influence of noise at 55 and 65 dB. The percentage of locations within each habitat was also determined (Obj 3). For the wintering shorebirds, locations were also categorised according to: (i) the standard time at which they were obtained (night or day); and (ii) one-hour intervals in relation to the nearest tidal peak (tidal timetable obtained from www.worldtides.info for Vila Franca de Xira). This made it possible to determine the percentage utilisation of each habitat in the diurnal and nocturnal periods in relation to the state of the tide (Obj 2). Finally, for shorebirds, the minimum distance travelled between the low- and high-tide was calculated using locations that were temporally closest to, and at most 1.5 h before or after, each of those two consecutive tidal peaks.

6.3 Results

6.3.1 Wintering shorebirds

Considering all the locations obtained for the three species of shorebirds tracked with GPS transmitters ($n = 69467$), the habitat where most locations were obtained was intertidal flats (54.0%), followed by salt pans (15.4%), rice paddies (14.6%) and saltmarsh (5.1%), with all other habitats being used in a percentage lower than 5% (Table 2).

However, the use of the various habitats varied considerably between species, according to the tidal stage and diel cycle (Fig. 6.3). In general, in the three hours before and after peak low tide (c. up to 43% of tide coverage), shorebirds use almost exclusively the intertidal flats, with the percentage of locations in this habitat consistently above 50% (with the exception of godwits at night). As the intertidal flats become submerged and thus unavailable due to the rising tide, shorebirds reduce their use of this habitat, moving into higher elevations, particularly salt pans (redshank and grey plover) and saltmarsh (godwits) during

Table 2. Locations obtained in each estuarine habitat for all individuals of the shorebird tracked with GPS transmitters during the three stationary non-breeding periods in the Tagus Estuary SPA.

Habitat	Locations	%
Intertidal flats	37537	54,0
Salt pans	10718	15,4
Rice paddies	10150	14,6
Saltmarsh	3543	5,1
Coastal lagoons	3384	4,9
Non irrigated crops	1641	2,4
Natural meadows	872	1,3
Pastures	827	1,2
Other	795	1,1

the day. This pattern is very similar at night for redshank and grey plover. But, in godwits the pattern of habitat use at night is quite different, with >50% of all locations in rice paddies, regardless of the state of the tide (Fig. 6.3). It should be noted that both godwits and redshank consistently make use of rice paddies to a certain extent (c. 25% at least), at any stage of tide and diel cycles.

The spatial distribution of the locations obtained falls almost entirely within the Tagus estuary SPA: redshank $99.5\% \pm 0.5$ SE; grey plover $94.0\% \pm 3.4$ SE; godwit $83.6\% \pm 6.6$ SE; (Fig. 6.4). The three species show considerable variation in the distance travelled between the locations obtained in the intertidal and supratidal zones, with the smallest being for redshank ($1.28 \text{ km} \pm 0.02$ SE; range: 0-12.3 km; $n = 4392$), being larger for grey plover ($2.67 \text{ km} \pm 0.03$ SE; range: 0-15.7 km; $n = 8581$) and even larger for godwits ($4.75 \text{ km} \pm 0.14$ SE; range: 0-14.6 km; $n = 889$). In any case, the overlap between bird locations and aircraft noise cones is inevitable, both at 55 dB and 65 dB (Fig. 6.4). It is also noticeable that in the areas within the noise cones (the disturbed area), the habitat most affected by the noise according to the locations obtained is intertidal flats (Fig. 6.5). This is particularly the case at 65dB, which will almost exclusively affect this habitat. But also very relevant in the 55 dB noise cone, which will additionally affect other important habitats: salt pans (and to a lesser extent) saltmarsh used by redshank and grey plover; and rice paddies for godwits and redshank (Fig. 6.5).

Due to the extensive overlap of the noise cones with the Tagus estuary SPA and given that almost all the GPS locations obtained are within the SPA, the percentage of locations of these birds in the 65 dB noise cone that are within the SPA is 100% for all individuals of the three species. And it is also 100% in the 55 dB cone for all individual godwits, all but one redshank (one individual with 88.1%), and all but two grey plovers (99.7% and 83.1%; Table S1).

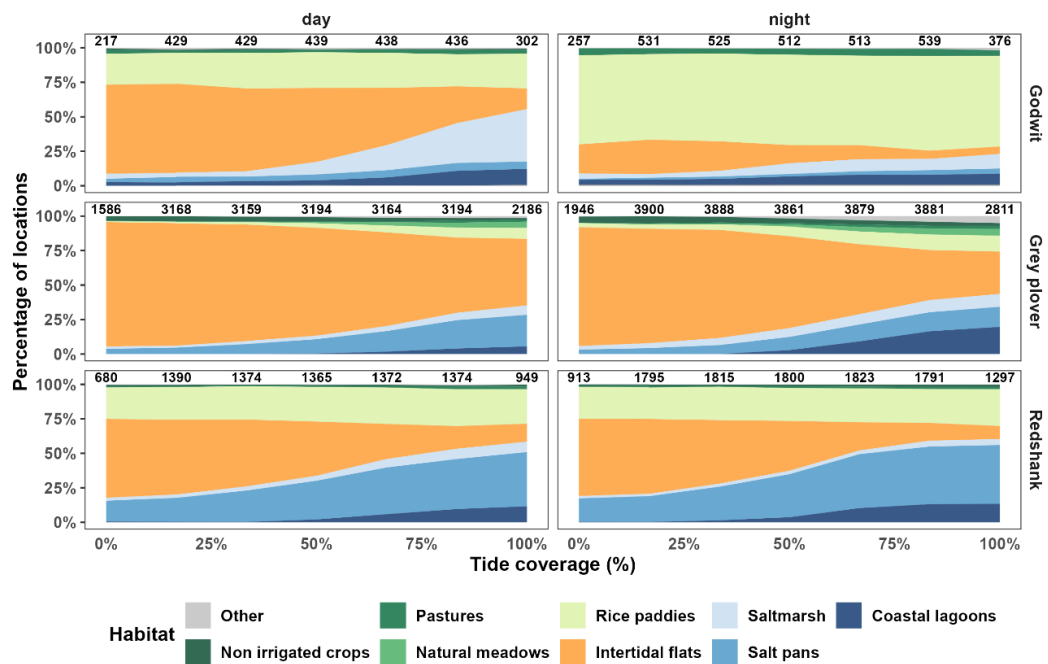


Figure 6.3. Use of different habitats in the Tagus estuary by three shorebird species during day (left) and night (right) in relation to the state of the tide between October and April. The locations obtained were grouped into one-hour intervals in relation to the low tide peak. The numbers on top above each plot indicate the sample size (n locations) obtained in each hour interval of the tidal cycle (0% tide coverage = tide code 0, i.e. low tide peak; 100% tide coverage = tide code 6, i.e. high tide peak).

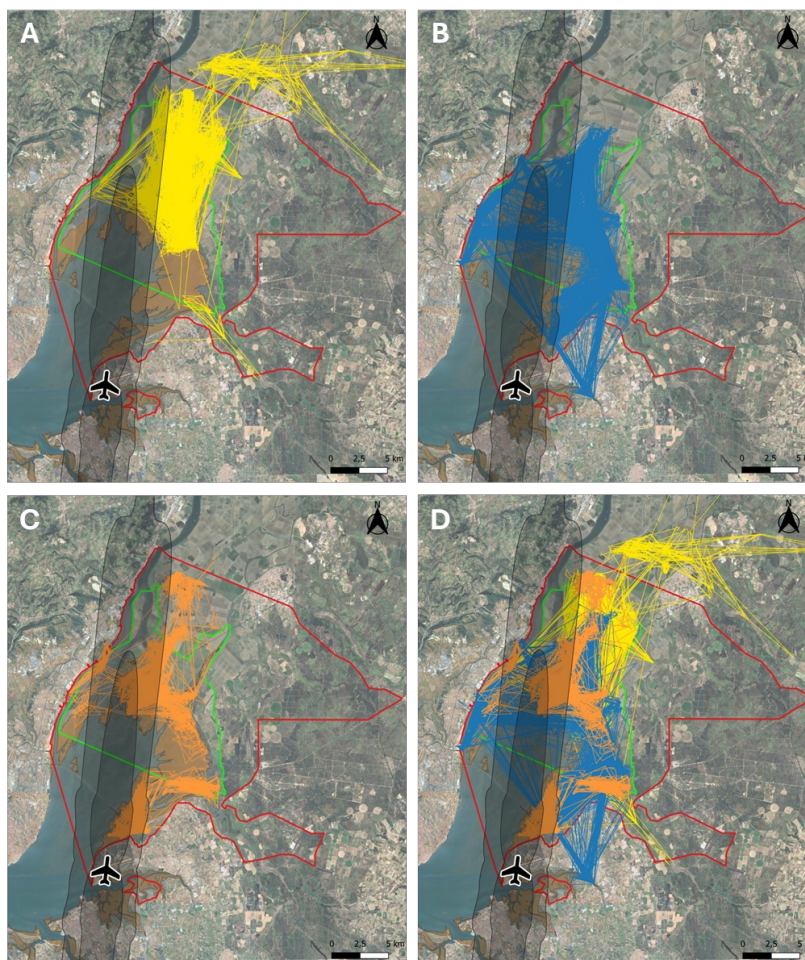


Figure 6.4. Space use recorded by GPS transmitters deployed on shorebirds wintering in the SPA (red line) and Tagus Estuary Nature Reserve (green line): godwit (yellow lines, n = 10; A); grey plover (blue lines, n = 20; B); Redshank (orange lines, n = 13; C); and all combined (D). Each GPS location of the same individual was joined to the next by straight line. The most frequently used habitat, 'intertidal flats' (Table 2) was noted in brown and the proposed location of Montijo Airport (aircraft symbol), as well as the noise cones at 65 and 55dB (smaller and larger shaded areas, respectively) were taken from the Environmental Impact Assessment.

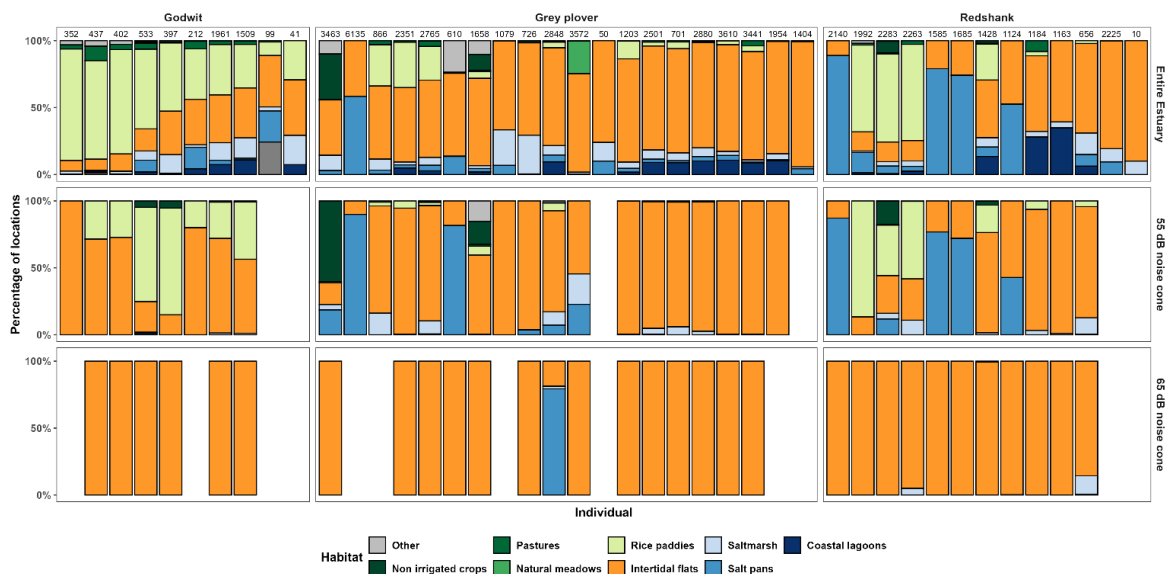


Figure 6.5. Variation in the percentage of locations of each individual of the three species shorebirds tracked with GPS transmitters obtained in different habitats, considering: the entire estuary (top); the area affected by the 55 dB noise cone (middle); and the area affected by the 65 dB noise cone (bottom). Individuals with no information (blank columns) did not register any location in the area considered by the respective plot.

6.3.2 Breeding species: spoonbill

Of the 41,604 locations obtained for the 15 juvenile spoonbills the habitat where these occurred in the highest percentage is saltmarsh (32.4%), followed by rice paddies (20.8%) and salt pans (16.1%). As expected, locations on the intertidal flats were relatively few (< 5%), as this species does not rely on this habitat for feeding. Locations in coastal lagoons (11.7%) complete the group of habitats with locations in excess of 5% for this species. On average, the vast majority of the locations obtained were located within the SPA of the Tagus estuary ($86.4\% \pm 5.3$ SE; Fig. 6).

Once again, due to the large overlap between the noise cones (disturbed area) and the SPA, all the locations within the 65 and 55 dB cones occur within the SPA. Similar to the wintering shorebirds sampled, the used habitats most affected by noise are intertidal habitats. In particular, the saltmarsh, with many locations in this habitat within the 55 dB noise cone, and the intertidal flats (as well as salt pans), with many locations within the 65 dB noise cone (Fig. 6.7).

6.4 Discussion

The information obtained from waterbirds with GPS transmitters indicates that implementing a commercial airport operation at the Montijo military base will have significant adverse effects on these species. In other words, this project will negatively affect the birds that naturally occur in this protected area, simply because the disturbance caused by aircraft noise will affect the current use of various habitats by these species, therefore inevitably com-

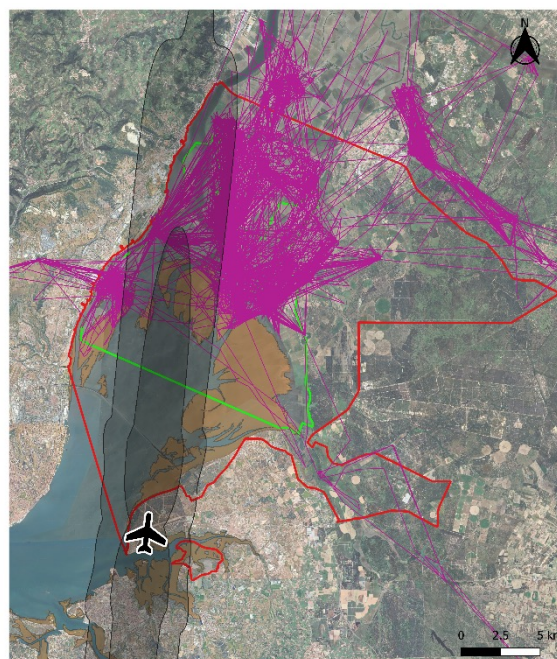


Figure 6.6. Space use recorded by GPS transmitters deployed on juvenile Eurasian Spoonbills born in Mouchão da Póvoa marsh, inside the protected area - SPA (red line) and Tagus Estuary Nature Reserve (green line). Each GPS location of the same individual was joined to the next by a straight line (purple). The intertidal mudflat habitat was marked in brown and the location of the Montijo Airport (aircraft symbol), as well as the noise cones at 65 and 55dB (smaller and larger shaded area, respectively) were extracted from the Environmental Impact Assessment.

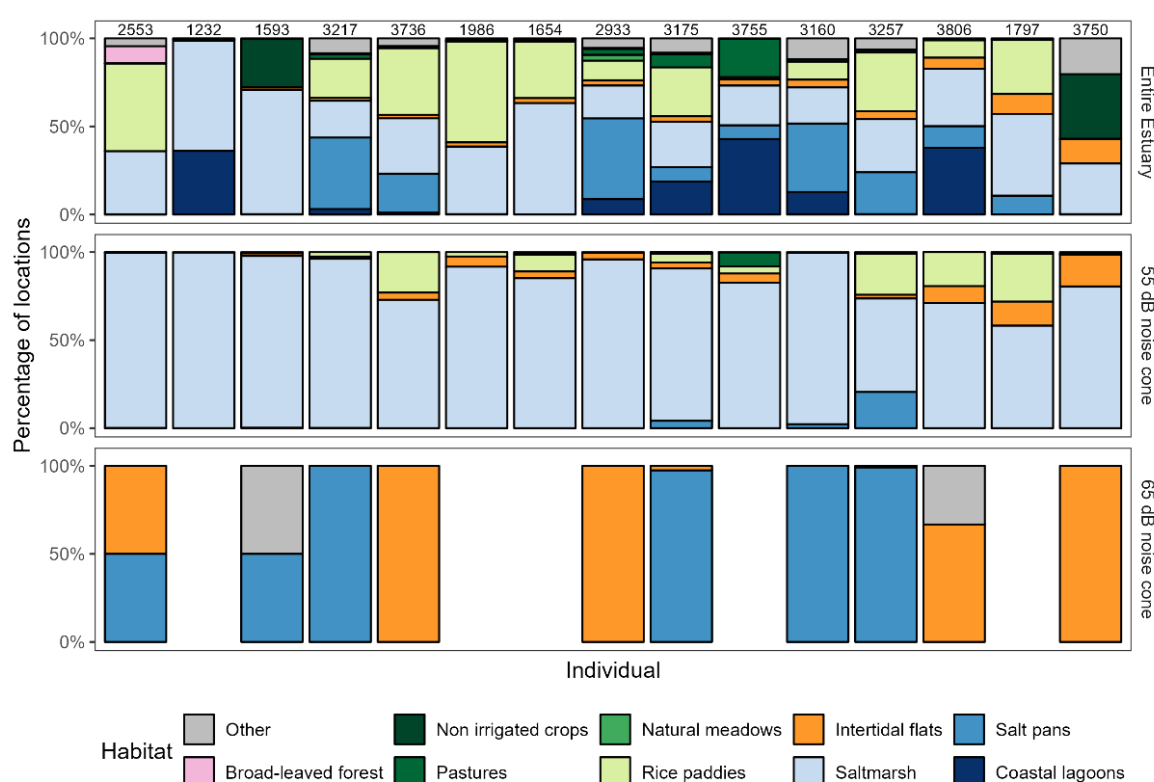


Figure 6.7. Variation in the percentage of locations of each juvenile spoonbill tracked with GPS devices obtained in different habitats, considering: the entire estuary (top); the area affected by the 55 dB noise cone (middle); and the area affected by the 65 dB noise cone (bottom). Individuals with no information (blank columns) did not register any location in the area considered by the respective plot.

promising its integrity. Given the frequency of flights (24 movements per hour; DIA 2020) which results in aircrafts flying over the protected area approximately every five minutes at least, and the fact that at 55 dB the probability of response is greater than zero (include fleeing from the disturbed area, to which birds may or not return at all, or not do so before the next aircraft flies over), it is reasonable to assume that such disturbance will result in a direct loss of habitat for species within the protected area.

It is noteworthy that for the wintering shorebirds the percentage of GPS locations obtained within the SPA varies between 83% and 99% across species. This indicates that shorebirds using the Tagus estuary's high tide roosts, where these GPS tracked individuals were captured, are heavily dependent on the SPA. The aforementioned overlap of the noise cones with the SPA results in a very high superposition between the locations of the birds in the SPA and the areas under the influence of the noise cones, i.e. the disturbed areas.

6.5.1 Conservative estimation

It should be noted that for the noise value of 65 dB, the study by Wright *et al.* (2010), as presented in the opinion of the EIA Evaluation Commission (CA 2019), indicates that 50% of the birds show abnormal behaviour, including flight due to the disturbance (by 25% of the birds). In addition,

that original study is based on the behaviour of birds quantified in response to a noise disturbance emitted by a *blow horn* activated for 3 seconds (Wright *et al.* 2010). This causes an almost instantaneous and short noise (*impulse noise*), which is not similar to the noise emitted by an aircraft on approach or take-off, specifically in its length (*intermittent noise*). Although the noise level quantified in decibels may be the same, the noise caused by a jet aircraft at approach or take-off speed (when it is over the protected area) will last much longer than three seconds from any point on the ground. It will also be emitted by a source that is in motion and not static and will therefore have what can be considered a 'sweeping' effect along its path. Given all these aspects regarding the disturbance in question, its impacts should therefore be considered as a conservative estimation. Thus, if the disturbance considered here is shown to have a significant effect on the conservation objective of migratory shorebirds or to affect the integrity of the site, then it is logical and reasonable to assume that the real situation will certainly lead to larger negative effects.

6.5.2 Pervasive effects of impacts on intertidal flats

As previously described for shorebirds, the different estuarine habitats they use fulfil different ecological functions (Fig. 6.3). The use of the intertidal flats takes place during

the period when this habitat is exposed (low tide) and it is here that wintering shorebirds feed on their most profitable prey. As the tide rises, birds move to other habitats such as salt pans, salt marshes, coastal lagoons or rice paddies (Fig. 6.8), where although feeding can continue, the quantity/quality of the food is generally lower (Masero *et al.* 2000). In the salt pans or in habitats where feeding cannot continue, such as saltmarshes, the rest of the time is used for other activities such as preening and resting.

For example, throughout the tidal cycle, 74% of the time that godwits occupy the intertidal flats (7.6 hours) is spent feeding, while this is only 26% in salt marshes (4.4 hours; Alves *et al.* 2013b). Also, feeding on the intertidal flats provides an average of 2.4 times more energy than feeding in the salt pans (J. Alves pers. obs.). This example demonstrates not only that one habitat cannot simply be replaced by another, but also the complexity of using various estuarine habitats, which fulfil different ecological functions for these birds and whose simultaneous availability is fundamental for sustaining these populations. It is therefore difficult to understand why the EIS, *de facto* accepting that the site's integrity will indeed be compromised, proposes

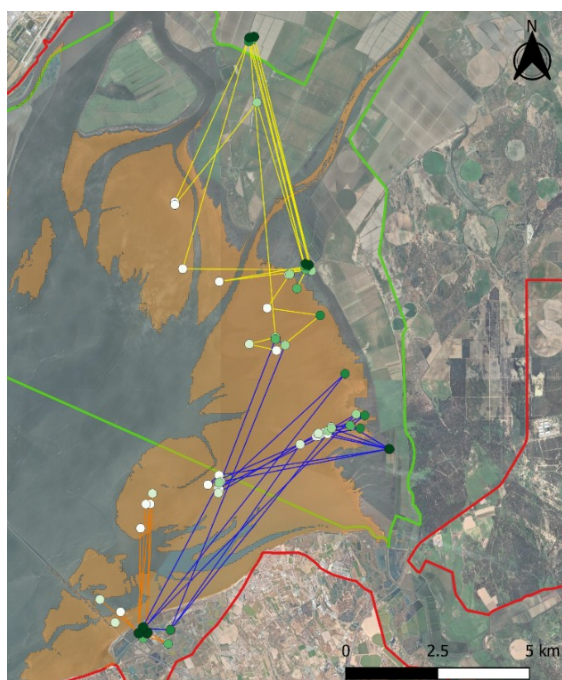


Figure 6.8. Detail of GPS locations recorded on 15 and 16 November 2020 for one individual of each wintering shorebird species sampled: godwit (yellow lines); grey plover (blue lines) and redshank (orange lines). Each location (represented by a circle) was coloured according to the tidal cycle, being dark green at high tide peak and following a colouration gradient until the low tide peak when it is white. SPA (red line) and Tagus Estuary Nature Reserve (green line), with the habitat 'intertidal mudflats' marked in brown. Each GPS location of the same individual was joined to the next by a straight line.

to compensate "1600 ha, corresponding to 20 per cent of the total feeding biotope in the SPA [i.e. intertidal flats], from 65db (strictly speaking below that value)" with the acquisition and requalification of salt pans. This measure is doubly lacking in rigour, firstly because the strong disturbance (as described in the EIS) occurs over an area of more than 1600 ha, and secondly because the disturbance occurs above 50 dB and not 65 dB. Furthermore, the assumption is flawed from the onset, given that the ecological function by the intertidal flats, particularly as prime feeding areas cannot be fulfilled in the same way by salt pans habitat, or indeed by any other estuarine habitat.

6.5.3 Compliance with Habitats & Birds directives

As recognised in the EIA and EIS, "...the project will induce negative, significant, moderate/high magnitude, permanent and irreversible impacts on the estuary's avifauna, resulting from the disturbance caused by the movement of aircraft and which will result in a reduction in the refuge, feeding and nesting areas of the aquatic birds that frequent it." Despite the outdated dataset used in the EIS (in some case going back to almost 15 years) and based mostly on count data, the conclusion that negative impacts originating from aircraft overlying the protected area is therefore not new. But our analysis further demonstrates that:

(i) the disturbance will have a **significant effect on the conservation objective of the various species of migratory birds** studied, since it will affect the natural usage of the protected area and of the different habitats that occur therein, specifically of intertidal flats, by reducing the natural distribution area of the species within the site (EU 2019). By displacing birds from the protected area (which will move to other sites outside the disturbed area) this disturbance will clearly reduce the area previously occupied by the species within the site. This raises further questions related to intraspecific competition for food that will likely arise due to the increase in numbers in the undisturbed areas that the displaced birds will move to. It is not straightforward that once displaced from the disturbed area, birds will continue to use the same habitat type and may in fact be relegated to lower quality habitats in terms of quantity and quality of food (e.g. salt marshes), with potential implications on their survival.

(ii) the **integrity of this site will be affected** since it will no longer be whole or complete due to the degradation of species habitats, i.e. the reduction in their representability, since birds will be excluded from them, specifically on intertidal flats, salt pans, rice paddies and saltmarsh habitats. In addition, due to the loss of usability of these habitats as birds will be displaced from them, there will be a change in the composition of these species habitats due to a clear reduction in their usable surface area (EU 2019).

The precautionary principle states that the *absence* of negative effects on the conservation of species and the maintenance of the site integrity must be demonstrated in

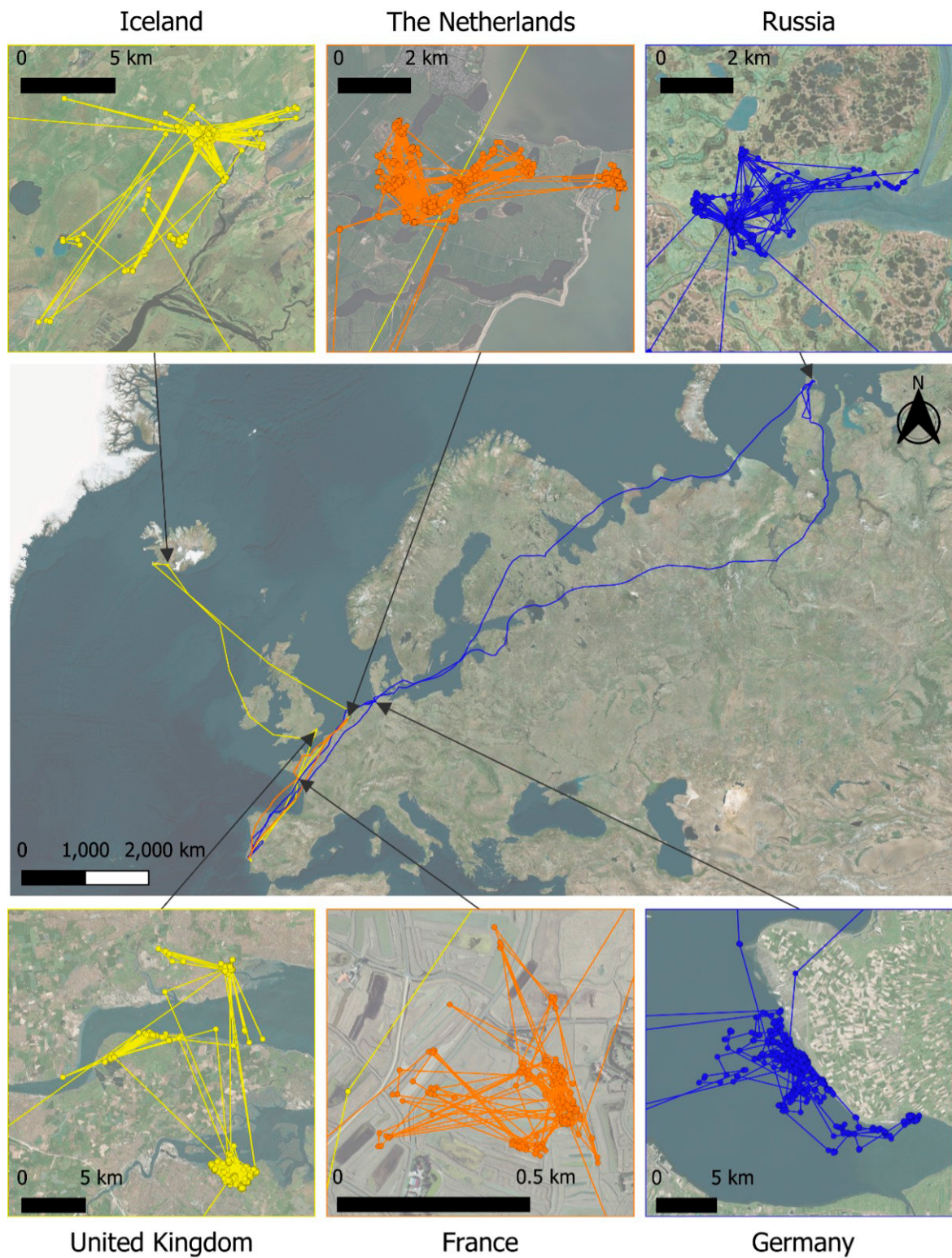


Figure 6.9. Migratory journeys of three individuals of each of the wintering shorebird species tracked in this study (godwit - yellow; redshank - orange; grey plover - blue). The migrations tracked between the Tagus estuary and each individual's breeding area are indicated on the central map. The locations in the breeding areas are shown at a smaller scale at the top and in the migratory stopover areas at the bottom of the figure.



Veronica Méndez Aragón

Black-tailed Godwit | Barge à queue noire (*Limosa limosa*)

the context of an impact assessment (EU 2019). However, using the best technical and scientific knowledge available on the topic (EU 2019), we further demonstrate here that on the contrary, such negative effects *will* occur and highlight that this principle was not applied in the EIA nor in the environmental impact declaration. Furthermore, although only four species were studied here, this includes species using different feeding techniques and consuming prey types present in the diet of many other waterbirds, thus encompassing a wide range of ecological traits of this community. This suggests that for other species also dependent on these habitats, particularly on intertidal flats, the effects of this disturbance will also be negative. In addition, the disturbance effects will certainly be larger than estimated here, since the actual noise will not just be instantaneous (lasting 3 seconds, as in the original study quantifying its effects, Wright *et al.* 2010), but repetitive, last considerably longer and sweep over the entire length of the SPA (Figs. 6.4 and 6.6).

A larger sample, particularly in terms of the number of bird species but also the number of individuals, would be highly desirable. The information presented here is particularly limited with regard to breeding species, and additional information would allow to better assess impacts. Tracking a larger number of individuals could indicate the use of other sites in the estuary, particularly for birds

tagged in high-tide roosts located outside of the SPA. The strong association between these roost sites and the feeding areas on the intertidal flats was previously established for Dunlin, with their use of those flats decreasing with distance from high-tide roosts, but with some individuals travelling up to 10-11 km (Dias *et al.* 2006). For some species this distance can even be around 16 km, as recorded in the present study. Even so, the negative effects caused inside the SPA go beyond its boundaries, since individuals naturally move between sites (and habitats) inside and outside of it. For example, in this study, one Grey Plover moved to sites in the Tagus estuary outside the SPA (Fig. 4B), as did several spoonbills (Fig. 6.6). Such connectivity between sites inside and outside the SPA should also be considered in a rigorous impact assessment. A recent study looking at 10 years of sightings of individually colour-ringed godwits in the Tagus estuary showed that 61% of the individuals sighted inside the SPA were also observed outside it (Nightingale *et al.* 2023). Using a network analysis, which considers the connectivity of the various sites in the Tagus estuary used by the species, it is estimated that by avoiding sites under the 55 dB noise cone, this disturbance will have an effect on 68.3% of the godwit population in the first half of winter and 37.8% in the second half (Nightingale *et al.* 2023). These proportions are much higher than those indicated in the EIA for this species (0.46-5.5% affected, EIA 2019),

which only considers the 65 dB noise cone. Regardless of the arbitrary decibel value that may be considered, the real impact on estuarine avifauna inside (and outside) the SPA will begin as soon as it exceeds 50 dB.

Finally, it should be noted that due to the connectivity along the Eastern Atlantic Flyway established by waterbirds that occur in the Tagus estuary SPA with other protected areas and particularly with other Natura 2000 sites, these negative impacts will not only have a local effect but also internationally and even intercontinentally (Beal *et al.* 2025). This is exemplified here by the migrations of one individual from each of the three wintering shorebird species tracked in this study (Fig. 6.9): with migratory stopover sites in the United Kingdom, France and Germany on their journeys to breeding sites in Iceland, the Netherlands and Russia. The impacts that these species will experience can certainly have international repercussions via carry-over effects (Harrison *et al.* 2011). These effects, which have been described in several migratory species (including Black-tailed Godwit), indicate that the conditions encountered in one location and stage of the annual cycle can affect individual performance in a following stage, for example, individuals that face adverse conditions in the non-breeding season can be penalised in the breeding season (Alves *et al.* 2013b). We call for a more streamlined process in which countries signatory of international treaties like RAMSAR or AEWA, and particularly within Natura 2000 network of sites, may be able to determine ongoing threats to species they have committed to protect that may occur elsewhere in the species distribution range. For

example, in the case of Montijo Airport, despite required by supranational obligations (e.g. Espoo Convention), it was only when local scientists and NGOs alerted about the ongoing public consultation of the EIA, that the international community, including countries where species that spend the non-breeding season in the Tagus breed, realised the potential of this threat. This was rather late, as despite contributions presented by those local actors during public consultation, the environmental impact declaration was issued anyway. Complaints to the various treaties were thus only formalised and considered afterwards (according to the meeting schedules of each treaty). It would be advantageous for countries to leap into action as soon as possible when a relevant threat is detected, and hold their counterparts accountable for the conservation of species that they may be investing in protecting.

If we are indeed serious about the conservation of biodiversity and on meeting UN's Sustainable Development Goals, specifically SDG 15.5 "*Take urgent and significant action to reduce the degradation of natural habitats, halt the loss of biodiversity...*" it is therefore paramount that Montijo Airport does not come into existence.

Acknowledgements

This work would not have been possible without the contribution of several people who substantially supported the fieldwork and/or made various suggestions to this document, which improved its quality, Pedro Miguel Araújo, Triin Kaasiku, and many volunteer ringers.



Common Ringed Plover | Pluvier grand-gravelot (*Charadrius hiaticula*)

Harvey van Diek

Supplementary materials

Table S1. Percentage of locations recorded for each individual within the Tagus estuary SPA and, for each noise cone (65 dB and 55 dB), the percentage of those locations occurring within the Tagus estuary SPA.

Species	SPA	65 dB in SPA	55 dB in SPA
Lim lim	100.0%	100.0%	100.0%
Lim lim	100.0%	100.0%	100.0%
Lim lim	100.0%	NA	NA
Lim lim	74.7%	NA	NA
Lim lim	43.8%	NA	100.0%
Lim lim	59.3%	100.0%	100.0%
Lim lim	66.7%	100.0%	100.0%
Lim lim	96.2%	NA	100.0%
Lim lim	99.2%	100.0%	100.0%
Lim lim	96.2%	100.0%	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	NA	NA
Plu squ	89.6%	100.0%	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	59.2%	100.0%	100.0%
Plu squ	100.0%	NA	100.0%
Plu squ	99.9%	100.0%	99.7%
Plu squ	100.0%	100.0%	100.0%
Plu squ	88.6%	100.0%	83.1%
Plu squ	100.0%	NA	NA
Plu squ	100.0%	NA	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	NA	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	NA	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	100.0%	100.0%	100.0%
Plu squ	43.6%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	99.9%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	NA	NA
Tri tot	100.0%	NA	NA
Tri tot	94.1%	100.0%	88.1%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Tri tot	100.0%	100.0%	100.0%
Pla leu	36.4%	100.0%	100.0%
Pla leu	99.8%	NA	100.0%
Pla leu	100.0%	100.0%	100.0%
Pla leu	42.5%	NA	100.0%
Pla leu	99.9%	NA	100.0%
Pla leu	79.6%	100.0%	100.0%
Pla leu	85.7%	100.0%	100.0%
Pla leu	99.8%	100.0%	100.0%
Pla leu	99.8%	NA	99.9%
Pla leu	99.3%	NA	99.9%
Pla leu	96.6%	100.0%	99.1%
Pla leu	84.0%	100.0%	100.0%
Pla leu	90.8%	100.0%	99.9%
Pla leu	83.8%	100.0%	100.0%
Pla leu	98.5%	100.0%	99.8%



Black Heron | **Aigrette ardoisée** (*Egretta ardesiaca*) (Daniele-Occhiato / Agami)

7. Long-term trends in the populations of waterbirds on the Ghanaian coast

Yaa Ntiama-Baidu, Emmanuel N. A. Taye & Alfred Ali Nuoh

Summary

Globally, waterbirds and the wetlands that they depend on face threats from human activities. Monitoring their populations is crucial for tracking changes and guiding conservation. We analyse four decades (since 1986) of waterbird counts from nine Ghanaian coastal wetlands, including five Ramsar sites. A total of 109 species were recorded. Curlew Sandpiper, Black-winged Stilt, Greenshank, Spotted Redshank, Little Stint, Ringed Plover, and Sanderling made up 84% of shorebird sightings. Common, Black, Sandwich, Royal, and Little Terns accounted for over 97% of terns. Other abundant species included Western Reef Heron, Little Egret, and White-faced Whistling Duck.

The total waterbird populations increased in the first two decades, then declined rapidly in the third, stabilising in the fourth. Most shorebirds followed this pattern, except Greenshank, Ringed Plover, Eurasian Curlew and Collared Pratincole, which remained stable throughout the four decades, while Grey Plover, Spotted Redshank, Pied Avocet, Red Knot and Turnstone declined. Only Whimbrel, Long-tailed Cormorant, and Black Heron showed continuous population increases. Overall, waterbird numbers and numbers of species occurring in internationally important Ramsar sites declined.

We attribute these losses to wetland destruction from urbanisation, industrial expansion, coastal erosion, pollution, and siltation, that contribute to reducing feeding, breeding, and roosting habitats. Climate change will likely worsen these impacts. We call for stronger site protection, continued monitoring, and in-depth studies to improve understanding of waterbird population dynamics.

Résumé

À l'échelle mondiale, les oiseaux d'eau et les zones humides dont ils dépendent sont menacés par les activités humaines. Leur suivi est essentiel pour observer les changements et orienter les actions de conservation. Nous analysons quatre décennies (depuis 1986) de dénombrements d'oiseaux d'eau dans neuf zones humides côtières du Ghana, dont cinq sites Ramsar. Un total de 109 espèces a été recensé. Le Bécasseau cocorli, l'Échasse blanche, le Chevalier aboyeur, le Chevalier arlequin, le Bécasseau minute, le Pluvier grand-gravelot et le Bécasseau sanderling représentaient 84 % des limicoles

comptés. Les Sternes pierregarin, hansel, caugek, royale et naine représentaient plus de 97 % des sternes. Parmi les autres espèces abondantes : l'Aigrette des récifs, l'Aigrette garzette et le Dendrocygne veuf.

La population totale a d'abord augmenté pendant deux décennies, avant de chuter rapidement durant la troisième, puis de se stabiliser. La majorité des limicoles ont suivi ce schéma, sauf le Chevalier aboyeur, le Grand-gravelot, le Courlis et le Glaréole à collier, dont les populations sont restées stables. Le Pluvier argenté, le Chevalier arlequin, l'Avocette élégante, le Bécasseau maubèche et le Tourne-pierre ont décliné. Seuls le Courlis corlieu, le Cormoran africain et l'Aigrette ardoisée ont montré une augmentation continue.

Globalement, les effectifs et la richesse spécifique ont diminué dans les sites Ramsar d'importance internationale. Nous attribuons ces pertes à la destruction des zones humides par l'urbanisation, l'expansion industrielle, l'érosion côtière, la pollution et l'envasement, qui réduisent les habitats d'alimentation, de reproduction et de repos. Le changement climatique risque d'aggraver ces impacts. Nous appelons à une meilleure protection des sites, à la poursuite du suivi, et à des recherches approfondies sur la dynamique des populations d'oiseaux d'eau.

7.1 Introduction

The importance of Ghana's coastal wetlands and coastal communities for the conservation of migratory waterbirds was recognised internationally in the 1970s (Grimes 1974, Macdonald 1978). Of particular significance was the work of the British Trust for Ornithology, which showed that 80% of ringing returns of the Roseate Tern¹ originated from Ghana. Concerns over declining British and Irish populations led to investigations that highlighted the intensive tern trapping by coastal children and the threat that this posed to the rare Roseate Tern, prompting calls for conservation action (Hepburn 1986, Everett *et al.* 1987, Avery *et al.* 1995). A 1986 survey of the Ghana coast identified key wetlands for migratory waterbirds and the findings led to the designation of five Ramsar sites and establishment of the long-term waterbird monitoring (Ntiama-Baidu 1991, Ntiama-Baidu & Hepburn 1988, Ntiama-Baidu & Gordon 1991).

¹ For scientific names of species mentioned in the text, see Table 1.

The ecological and conservation value of long-term monitoring of biodiversity cannot be over-emphasised. Biodiversity data are essential for conservation policy and management, and for defining relevant interventions at protected sites (Stephenson *et al.* 2015, 2020, 2022). Migratory waterbirds are among the faunal groups that have been widely monitored, with well-established schemes to track species populations (Wetlands International 2002, 2010, 2021). However, wetland habitats, that are so crucial to the survival and sustainability of waterbirds, are undergoing very rapid changes which are mostly detrimental. Hence, there is a critical and urgent need to establish and implement systematic and repeatable methodologies for waterbird counts for better assessment and understanding of population changes (Smit & Piersma 1989, van Roomen *et al.* 2015, Reneerkens *et al.* 2024, Chapter 11).

One of the earliest site-specific systematic waterbird population monitoring along the East Atlantic Flyway (EAF) is the counts at the Parc National du Banc d'Arguin in Mauritania, which were started in 1980 (Altenburg *et al.* 1983, Oudman *et al.* 2020, El-Hacen *et al.* 2024). A more encompassing and comprehensive waterbird monitoring programme along the EAF is the International Waterbird Census (IWC) organised by Wetlands International and its partners (van Roomen *et al.* 2022).

In this chapter, we present waterbird population trends

based on four decades of waterbird population monitoring along coastal Ghana, arguably the longest running, systematic, waterbird monitoring programme within the West African region. We discuss changes in the international importance of the Ghanaian coastal wetlands and the threats contributing to the declining ecological integrity of the wetlands designated as Ramsar sites.

7.2 Methods

7.2.1 Waterbird counts

The Ghana waterbird population monitoring started in 1986 with a survey of the entire coast. This was achieved by two teams of three persons, one moving from east to west and the second from the west to east. All wetland formations along the coast, as well as stretches of sandy beaches were mapped out and described, and all waterbirds encountered were identified and counted. This survey identified 20 coastal sites that held appreciable numbers of waterbirds. All 20 sites were counted monthly during the first two years of the monitoring programme. Subsequently, nine key sites were selected for long-term waterbird monitoring: a sandy beach on the western coast (Esiama), two salt pan complexes (Densu delta and Lalui), two small estuarine/lagoon formations (Amisa and Narkwa), two small brackish water lagoons (Muni and Sakumo) and the two extensive lagoon complexes associ-



Figure 7.1. Map of showing lagoon formations along the Ghana coast

ated with the Volta River (Songor and Keta) (Fig. 7.1). Together, these nine sites hold ca. 90% of the total waterbird population along the Ghana coast (Ntiamo-Baidu & Gordon 1991).

The nine sites were counted every month for nearly three decades, except for the four years 2011–2014, when some sites were not counted due to funding constraints. From 2016 onwards the counts were reduced to four months each year, in September, October (peak months), June (lean month) and January (coinciding with the IWC), resulting in a monitoring that has been ongoing for four decades now. The counts were carried out by 2–4 persons through a combination of transect and point counts on foot, except at the sandy beach between Esiam and Ankobra Rivers, where it was possible to count from vehicles along some parts of the beach. Birds were counted with the aid of binoculars and telescopes, and hand-held tally counters. Counts were normally done between 06:00–11:00 h. On each visit, the date, location, species, numbers and other general observations at the site were entered into field notebooks and later extracted into Microsoft Excel spreadsheets. One author (AAN) who was involved in the 1986 total coastal counts as a field assistant, has since led the counts, thereby ensuring consistency.

7.2.2 Analysis of trends in abundance

To describe the changes in waterbird populations over time (years), we employed Generalized Additive Models (GAMs) with a log link function under a Poisson distribution (Wood 2017). GAMs were suitable for this study because they allowed the identification of non-linear trends and periods of change as well as the magnitude of change (Simpson 2018). All statistical analyses and graphics were done using R v4.1.1 (R Core Team 2024) in the RStudio environment (RStudio Team 2024).

The GAM models were fitted using the `gam()` function from the `mgcv` package, with restricted maximum likelihood (REML) as the method for smoothing parameter selection (Wood 2017). The function `'summary()'` was used to access the model parameters to check for significance of non-linear trends, the effective degrees of freedom (edf), the adjusted r^2 , and the explained deviance. A p -value < 0.05 indicates that the predictor value (time) has a significant non-linear effect on the count, however, $p > 0.05$ does not indicate the absence of significance but rather that the predictor might either not have a strong effect (no change with time) or that the change over time is more or less linear.

Model diagnostics were performed using the `gam.check()` function, where residuals were examined to assess model fit. To address issues of oversmoothing or undersmoothing, comparisons were made between the basal dimension k and the effective degrees of freedom (edf),

and the value of p (k -index). Adjustments to k were applied when $\text{res } p$ -value was low or k -index was below 1, while the default values were retained when no evidence of oversmoothing or undersmoothing was observed.

Because of differences in the numbers of months in which counts were conducted annually in different periods, we used the annual peak counts for the analysis. The data for 2012–2014 were excluded from the general species trend analysis as some sites were not counted in those years. The analysis presented here covers populations trends of all waterbirds at all sites split into the major groups (shorebirds, terns/gulls, herons/egrets/ducks) as well as trends in the populations of the most abundant species in each group.

7.3 Results

7.3.1 Waterbird species recorded on the Ghana coast

A total of 109 waterbird species were recorded on the coastal sites over nearly four decades of monitoring. These included species belonging to four groups: shorebirds, terns and gulls, herons and egrets, and ducks (Table 7.1). Shorebirds were the most abundant group, accounting for c. 52% of the numbers, followed by terns (33%), with ducks and herons/egrets contributing 8% and 6% respectively.

Curlew sandpiper was the most numerous shorebird species, with 27% of all shorebird numbers. Together with Black-winged Stilt, Greenshank, Spotted Redshank, Little Stint, Ringed Plover and Sanderling, the seven species contributed 84% of total shorebird numbers on the Ghana coastal sites. Common Tern and Black Tern were the most abundant tern species accounting for over 65% of all terns encountered; together with Sandwich, Royal and Little Terns accounting for over 97% of all terns recorded. Roseate Tern was recorded regularly but in smaller numbers. The most common gull species recorded was Lesser Black-backed Gull, while Western Reef Heron and Little Egret were the most abundant herons and egrets. The most abundant ducks encountered were White-faced Whistling Duck and Fulvous Whistling Duck.

7.3.2 Trends in populations of waterbirds

The trends in the populations of waterbirds on the Ghana coast are presented in Figs. 7.2–7.5. The combined population of waterbirds showed a continuous steep increase in the first two decades of monitoring, peaking at over 160,000 birds in 2001, followed by a rapid decline during the third decade and stabilisation in the final decade (Fig. 7.2a); a significant non-linear pattern (GAM smooth terms $p < 0.01$, Table 7.2). Shorebirds, which accounted for over 50% of the waterbirds recorded, show the same pattern, with annual increases up to the early 2000 and then

Table 7.1. Peak counts of waterbird species recorded at the monitored sites along the Ghana coast. Only species for which peak counts of more than 10 individuals were recorded at any one time are presented. A further 30 waterbird species recorded in fewer numbers are excluded from this table. Note that the peak counts are the peaks recorded at any one time for the year blocks and are not total counts recorded during the period.

Species name	1986-1990	1991-1995	1996-2000	2001-2005	2006-2010	2011-2015	2016-2020	2020-2024
White-faced Whistling Duck <i>Dendrocygna viduata</i>	5860	24263	25910	33531	2950	2800	2011	4705
Fulvous Whistling Duck <i>Dendrocygna bicolor</i>	40	7125	2387	4602	17	61	21	1230
Egyptian Goose <i>Alopochen aegyptiacus</i>	9	13						
Pigmy Goose <i>Nettapus auritus</i>	45	53	19	38		4		
Common Pochard <i>Netta rufina</i>	25							
Garganey <i>Spatula querquedula</i>	4590	7267	84	88	12	7	1	
Northern Pintail <i>Anas acuta</i>	13		6		2		20	
Common Teal <i>Anas crecca</i>	92							
Little Grebe <i>Tachybaptus ruficollis</i>	12	24	20	14	60	23	14	22
Allen's Gallinule <i>Porphyrio alleni</i>				13				
Black Crake <i>Zapornia flavirostra</i>				49				32
Common Moorhen <i>Gallinula chloropus</i>				24			38	170
African Openbill <i>Anastomus lamelligerus</i>								1127
Black Stork <i>Ciconia nigra</i>								103
Eurasian Spoonbill <i>Platalea leucorodia</i>	16	24	26	10	17	6		1
Glossy Ibis <i>Plegadis falcinellus</i>	119	94	35	17	24	22	9	6
Dwarf Bittern <i>Ixobrychus sturmii</i>	12							
Common Little Bittern <i>Ixobrychus minutus</i>	15	29	71	83	23	4	3	9
Green-backed Heron <i>Butorides striata</i>	10	500	26	40	21	16	14	14
Squacco Heron <i>Ardeola ralloides</i>	225	544	321	296	120	150	82	97
Grey Heron <i>Ardea cinerea</i>	553	754	990	554	709	157	649	533
Black-crowned Night Heron <i>Nycticorax nycticorax</i>	1		217	199	200	16	7	11
Purple Heron <i>Ardea purpurea</i>	48	70	40	45	28	12	14	22
Great White Egret <i>Ardea alba</i>	1612	687	545	745	1021	283	598	979
Yellow-billed Egret <i>Ardea brachyrhyncha</i>	33	20	80	143	91	116	60	46
Black Heron <i>Egretta ardesiaca</i>	158	99	60	460	120	354	365	710
Little Egret <i>Egretta garzetta</i>	3166	3793	6865	10403	2977	2258	2477	3611
Western Reef Egret <i>Egretta gularis</i>	1047	611	1235	1167	1016	7778	618	860
Pink-backed Pelican <i>Pelecanus rufescens</i>	830	602	278	687	311	462	146	89
Great White Pelican <i>Pelecanus onocrotalus</i>	213							
Long-tailed Cormorant <i>Microcarbo africanus</i>	246	1773	1350	5322	3145	3794	3270	5765
Senegal Thick-knee <i>Burhinus senegalensis</i>	23	10	14	21	16	24	19	50
Eurasian Oystercatcher <i>Haematopus ostralegus</i>	28	45	39	70	31	28	15	8
Pied Avocet <i>Recurvirostra avosetta</i>	3642	782	1144	416	216	62	73	11
Black-winged Stilt <i>Himantopus himantopus</i>	3047	11973	15046	11518	7254	3964	2947	2662
Grey Plover <i>Pluvialis squatarola</i>	716	684	696	624	465	629	377	232
Common Ringed Plover <i>Charadrius hiaticula</i>	2880	4480	6802	3412	2752	1894	3775	5449
Kittlitz's Plover <i>Charadrius pecuarius</i>	324	1620	458	508	182	164	336	168
White-fronted Plover <i>Charadrius marginatus</i>	68	76	6470	1704	57	27	28	15
Spur-winged Lapwing <i>Vanellus spinosus</i>	9	8	28	252	299	167	233	396
Wattled Lapwing <i>Vanellus senegallus</i>	10	56	23	85	316	114	75	103
African Jacana <i>Actophilornis africanus</i>	250	587	130	102	100	390	570	346
Whimbrel <i>Numenius phaeopus</i>	115	73	183	228	218	152	278	372
Eurasian Curlew <i>Numenius arquata</i>	182	76	80	649	60	27	174	108
Black-tailed Godwit <i>Limosa limosa</i>	1460	3117	2000	1543	814	392	71	256
Bar-tailed Godwit <i>Limosa lapponica</i>	204	271	129	111	58	241	23	81
Ruddy Turnstone <i>Arenaria interpres</i>	262	195	198	282	172	127	45	172
Ruff <i>Calidris pugnax</i>	273	299	449	606	78	105	20	27
Broad-billed Sandpiper <i>Limicola falcinellus</i>	15							
Curlew Sandpiper <i>Calidris ferruginea</i>	14002	16989	22319	31250	12900	10029	8668	7112
Sanderling <i>Calidris alba</i>	4516	3520	4246	4238	4514	2750	2223	3436
Red Knot <i>Calidris canutus</i>	1591	1701	1167	435	112	168	194	199
Little Stint <i>Calidris minuta</i>	4295	14469	8512	5992	2232	3964	1706	2832

Species name	1986-1990	1991-1995	1996-2000	2001-2005	2006-2010	2011-2015	2016-2020	2020-2024
Common Sandpiper <i>Actitis hypoleucos</i>	111	202	330	1043	79	180	92	132
Green Sandpiper <i>Tringa ochruros</i>	3			195				
Spotted Redshank <i>Tringa erythropus</i>	6350	8129	11500	12835	2822	1736	844	398
Common Greenshank <i>Tringa nebularia</i>	4557	2729	8451	16748	6407	4838	2771	3932
Marsh Sandpiper <i>Tringa stagnatilis</i>	243	1499	1452	623	145	204	99	207
Common Redshank <i>Tringa totanus</i>	321	584	123	179	69	74	45	47
Wood Sandpiper <i>Tringa glareola</i>	172	601	385	158	209	323	394	397
Collared Pratincole <i>Glareola pratincola</i>	1423	890	2004	6000	704	557	644	1036
African Skimmer <i>Rynchops flavirostris</i>	92	82	232	251	148	36	278	84
Little Gull <i>Chroicocephalus minutus</i>	33	20	82	24	12			
Black-headed Gull <i>Larus ridibundus</i>	188	138	60	36	54	28	27	
Grey-headed Gull <i>Larus cirrocephalus</i>			66					
Kelp Gull <i>Larus dominicanus</i>			4	33				
Lesser Black-backed Gull <i>Larus fuscus</i>	462	415	652	657	203	72	215	168
Sooty Tern <i>Onychoprion fuscatus</i>	4	6	68	4	1	1		
Little Tern <i>Sternula albifrons</i>	1666	3091	2193	2200	2300	3030	4373	1762
Gull-billed Tern <i>Gelochelidon nilotica</i>	57	251	643	266	164	64	156	86
Caspian Tern <i>Hydroprogne caspia</i>	442	113	188	574	96	85	222	163
Whiskered Tern <i>Chlidonias hybrida</i>	100	513	111	2	8			
White-winged Black Tern <i>Chlidonias leucopterus</i>	11	155	56	23	25			
Black Tern <i>Chlidonias niger</i>	17710	17345	22300	27743	8279	11788	8846	967
Roseate Tern <i>Sterna dougallii</i>	197	521	656	1214	541	170	232	210
Common Tern <i>Sterna hirundo</i>	11900	15293	11475	21513	31190	19295	5561	10300
Sandwich Tern <i>Thalasseus sandwicensis</i>	5119	3342	6793	10114	6006	11368	2174	2285
W African Crested Tern <i>Thalasseus albididorsalis</i>	3198	4311	7714	8912	8951	8703	2860	2458
Pied Kingfisher <i>Ceryle rudis</i>							149	365

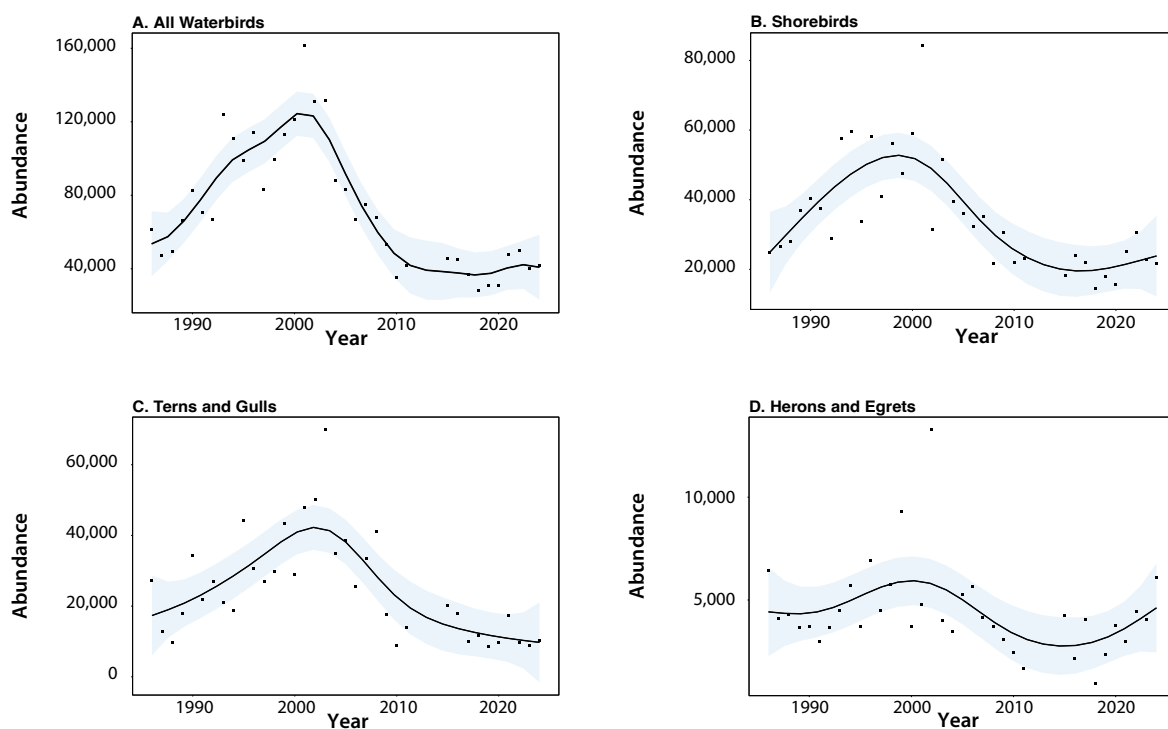


Figure 7.2. Changes in the populations of waterbirds on the Ghana coastal sites monitored over the past four decades (1986-2024); (A) All Waterbirds combined; (B) All Shorebirds; (C) Terns and Gulls; (D) Herons and Egrets. Note: the black dots in Figs 2-5 denote the year peak, and should not be equated to population sizes. Black line is the smoothed trend line.

decreasing rapidly over the next decade and stabilising from around 2015 onwards (Table 7.2, Fig. 7.2b). A similar pattern is also shown by the terns and gulls in the first two decades of the counts, albeit with a gentler increase, peaking around 2003. A steep decline followed in the third decade, which continues at a slower rate over the fourth decade (Table 7.2, Fig. 7.2c). The populations of herons and egrets on the Ghana coast have remained more or less stable, showing a non-linear pattern that was not significant (Table 7.2, Fig. 7.2d).

The individual shorebird species show interesting patterns, with year-to-year variations. The trend presented by Curlew Sandpiper, the most abundant shorebird species on the Ghana coast, reflects the general pattern of total

Table 7.2. Summary of smooth terms of the Generalised Additive Models (GAM) for changes in waterbird populations on the monitored Ghana coastal sites over four decades based on annual peak counts. P-values below 0.05 were considered statistically significant (shown in bold).

group / species	degrees of freedom	R ² (adj)	% deviance explained	p-value	p-res
All waterbirds	6.2	0.83	86	<0.01	1
Shorebirds	4.5	0.62	67	<0.01	1
Terns and Gulls	4.7	0.59	65	<0.01	0.4
Heron and Egrets	3.9	0.25	33	0.05	0.9
Black-winged Stilt	3.7	0.37	44	<0.01	0.9
Collared Pratincole	4.0	0.23	32	0.08	0.6
Curlew Sandpiper	6.1	0.84	87	<0.01	0.9
Greenshank	2.7	0.19	26	0.06	0.9
Grey Plover	1.1	0.31	33	<0.01	0.3
Little Stint	3.8	0.38	45	<0.01	0.5
Long-tailed Cormorant	1.0	0.62	64	<0.01	0.0
Common Ringed Plover	3.4	0.19	27	0.11	0.8
Sanderling	4.1	0.47	54	<0.01	0.7
Spotted Redshank	1.1	0.26	29	<0.01	0.1
Black-tailed Godwit	3.5	0.48	53	<0.01	0.2
Kittlitz's Sandplover	3.7	0.22	29	0.07	0.1
Red Knot	1	0.009	8	0.31	0.7
Pied Avocet	1.5	0.29	33	<0.01	0.01
Whimbrel	2.0	0.46	49	<0.01	0.3
Eurasian Curlew	1.5	-0.007	3	0.7	0.02
Ruddy Turnstone	1	0.26	28	<0.01	0.2
Black Tern	2.8	0.38	43	<0.01	0.7
Little Tern	2.3	0.18	24	<0.05	0.4
Common Tern	3.1	0.24	30	<0.05	0.3
W African Crested Tern	3.6	0.45	51	<0.01	0.8
Sandwich Tern	6.5	0.72	77	<0.01	0.9
Black Heron	2.7	0.43	48	<0.01	0.5
Great White Egret	1.9	0.02	75	0.54	0.2
Little Egret	2.3	0.13	20	0.14	0.9
Western Reef Egret	4.9	0.32	42	<0.05	0.8
Grey Heron	3.7	0.39	46	<0.01	0.7
White-faced Whistling Duck	4.5	0.51	57	<0.01	0.3
Fulvous Whistling Duck	1.2	-0.01	32	0.55	0.2



Black-winged Stilt | Échasse blanche (*Himantopus himantopus*)

shorebirds, with a strong increase during the initial two decades, peaking around 2000, then declining over the third decade, stabilising in the first half on the fourth decade and showing signs of increase in the final years (Table 7.2, Fig. 7.3a). Black-winged Stilt, Little Stint, Kittlitz's Sandpiper and Black-tailed Godwit showed significant non-linear trends with varying degrees of increases in the initial two decades, followed by gradual declines in the third decade, stabilizing in the final decade (e.g. Fig. 7.3b). The Sanderling is the only shorebird species that showed a decrease in the first few years of monitoring, but this was followed by a pattern that is consistent with the combined shorebird trend (Fig. 7.3c). Four other shorebird species occurring commonly on the Ghana coast, Greenshank, Ringed Plover, Curlew and Collared Pratincole, showed more or less stable populations throughout the four decades with non-linear trends that were not significant (Table 7.2, Figs. 7.3d,e). Another five species, Grey Plover, Pied Avocet, Spotted Redshank, Ruddy Turnstone and Red Knot, showed consistently decreasing numbers throughout the four decades, with statistically significant trends (Table 7.2, Figs. 7.3f-i). The Whimbrel was the only shorebird species that showed a consistent population increase throughout the four decades (Fig. 7.3j).

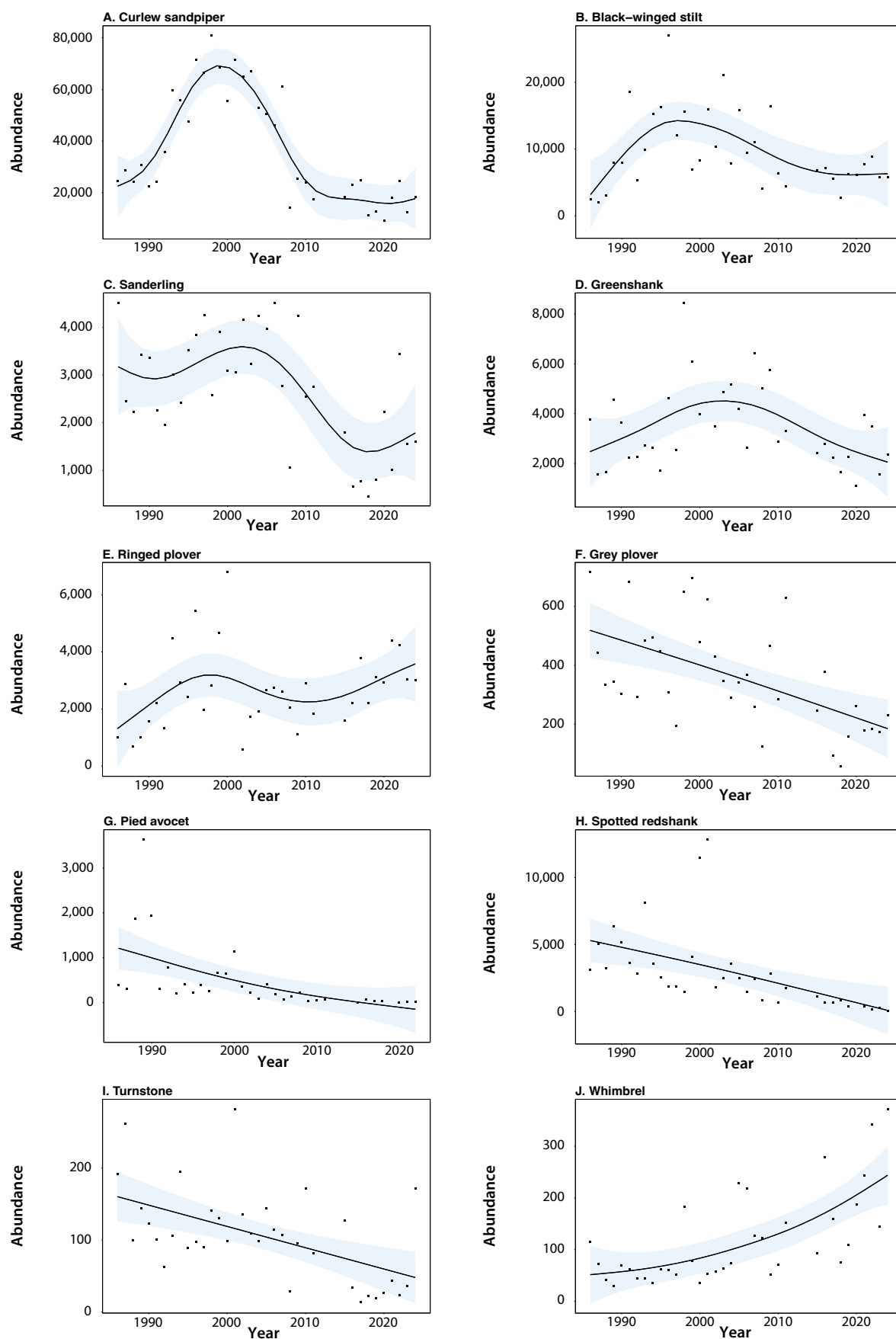


Figure 7.3. Changes in populations of shorebird species on the Ghana Coast: A: Curlew Sandpiper; B: Black-winged Stilt; C: Sanderling; D: Greenshank; E: Ringed Plover; F: Grey Plover; G: Pied Avocet; H: Spotted Redshank; I: Turnstone J: Whimbrel.

The five most common tern species found on the Ghana coast present three patterns of population trends that are all statistically significant (Table 7.2). Black Tern, Common Tern and W African Crested Tern showed slight population increases in the first two decades, followed by gradual decreases that continued into the fourth decade (Figs. 7.4a-c), while the Little Tern showed more or less stable numbers throughout (Fig. 7.4d). Sandwich Tern decreased in the first few years of the counts, followed by strong increases up to the early 2000s, a steep decline through the third decade and stabilisation in the final decade (Fig. 7.4e).

Of the larger waterbird species, Black Heron, a resident, showed a statistically significant trend, characterized by a stable population in the first three decades, increasing slightly through the fourth decade (Table 7.2, Fig. 7.5a). Great White Egret and Little Egret showed a stable population throughout the four decades, with non-linear trends that were not statistically significant (Table 7.2, Figs. 7.5b-c).

Western Reef decreased in the first few years of the monitoring, followed by strong increase up to around 2005 and a gradual decrease thereafter (Fig. 7.5d), while Grey Heron increased in the first decade and a half, followed by a very gradual decrease that continued for another 15 years (Fig. 7.5e). Long-tailed Cormorant showed a consistent upward trend throughout the four decades, which was statistically significant (Table 7.2, Fig. 7.5f). Of the two most common species of ducks found on the Ghana coast, White-faced Whistling Duck showed an increase during the first two decades up to around 2000, followed by gentle decline during the third decade, with the numbers stabilising in the fourth decade (Fig. 7.5g), while the numbers of the second species, Fulvous Whistling Duck, remained stable throughout the four decades (Fig. 7.5h).

7.4 Discussion and conclusions

We present data on waterbird species and population trends based on four decades of waterbird counts at

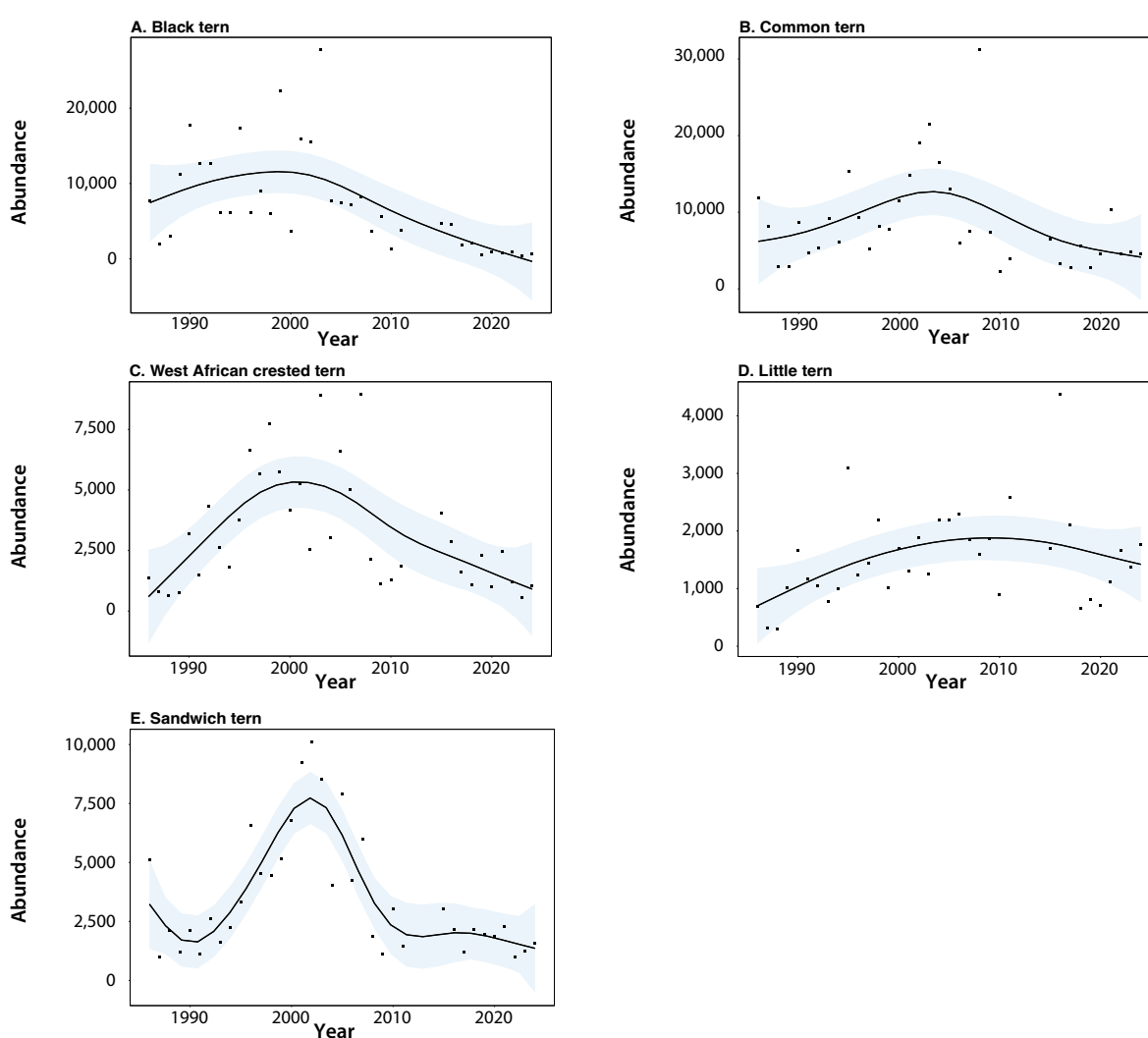


Figure 7.4. Changes in the populations of the five most common Tern species occurring on the Ghana coast A: Black Tern; B: Common Tern; C: W African Crested Tern; D: Little Tern; E: Sandwich Tern

selected sites along the Ghana coast. Together, the nine sites involved in the counts hold over 90% of the total waterbird numbers on the Ghana coast and therefore provide a good reflection of the general situation of waterbirds in coastal Ghana. A total of 109 species of waterbirds were recorded during the period, the most abundant groups being shorebirds, followed by terns and gulls, ducks and herons in that order. Waterbirds are most abundant on the Ghana coast during September and October.

The trends of waterbird populations on the Ghana coast in the first two decades of the monitoring is characterized by increases in total numbers of waterbirds, shorebirds, terns and gulls and several individual species, including Curlew Sandpiper, Black Tern, Grey Heron and White-faced Whistling Duck. The third decade presents declining total numbers of waterbirds and populations of shorebirds, terns and gulls and some individual species. The picture in the final decade shows stabilisation in the

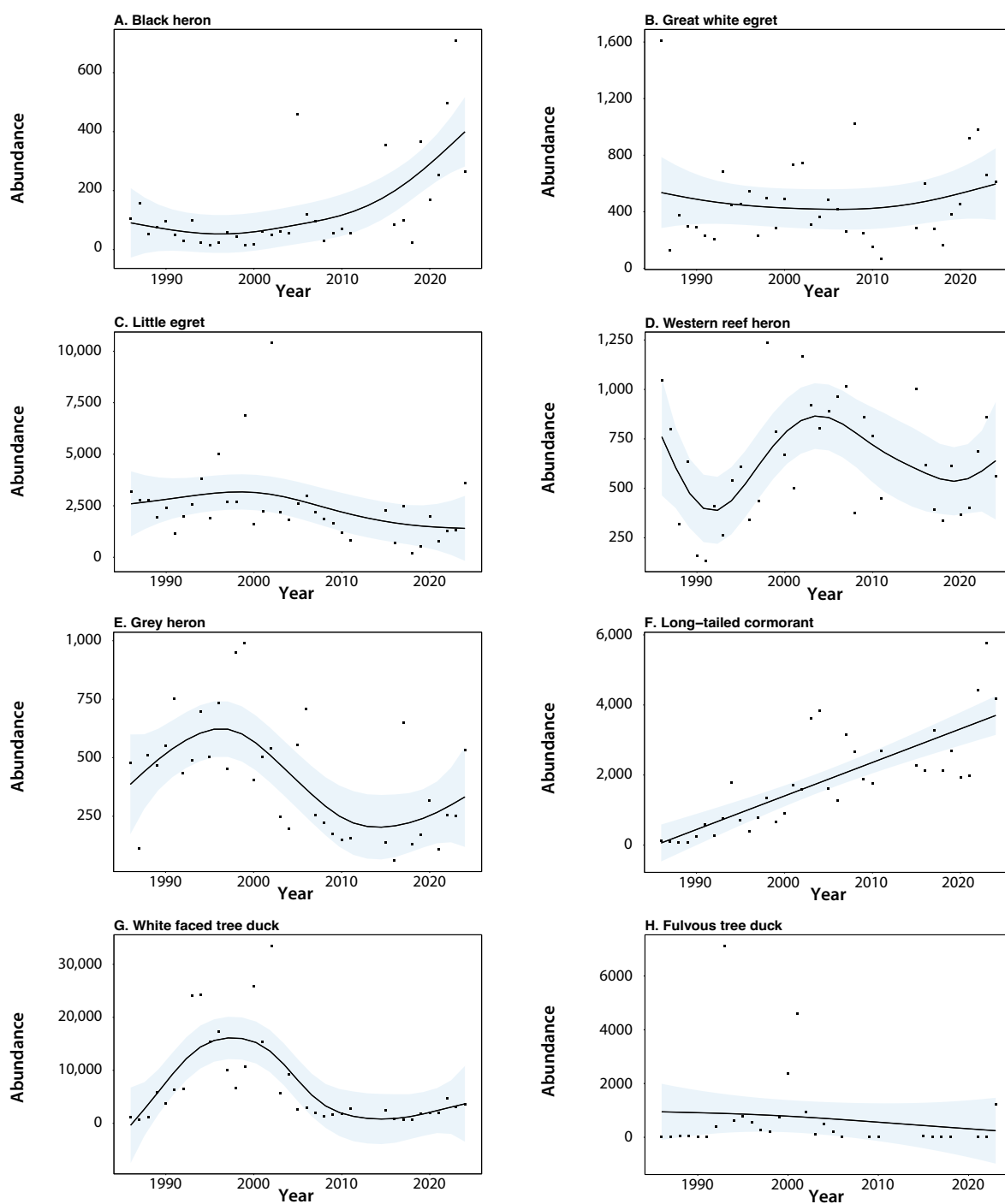


Figure 7.5. Changes in the populations of the common larger waterbirds occurring on the Ghana coast. A: Black Heron; B: Great White Egret; C: Little Egret; D: Western Reef Heron; E: Grey Heron; F: Long-tailed Cormorant; G: White-faced Whistling Duck; H: Fulvous Whistling Duck.

populations of most species, and in a few cases, signs of increasing numbers. These observations are consistent with the population trends analysis presented by van Roomen *et al.* (2022) and Schekkerman *et al.* (2022) for waterbirds using the EAF. A comparison of the 1980s-1990s with 2010s-2020s waterbird populations showed declines in coastal Africa southward from Mauritania and Senegal (Schekkerman *et al.* 2022). The periodic variations have been observed also in the Iwik area of the Parc National du Banc d'Arguin in Mauritania, where Oudman *et al.* (2020) reported increases in numbers of waterbirds from 2003-2017, in contrast to declines reported in the 2020s by El-Hacen *et al.* (2024). These are consistent with the trends observed for the populations on the Ghanaian coast, where several species exhibited increases from the beginning of the monitoring in 1986, continuing for about two decades and then showing declines in later years.

Three waterbird species, Whimbrel, Black Heron, and Long-tailed Cormorant, exhibited consistent increases in their numbers throughout the four decades of counts. El-Hacen *et al.* (2024) reported that Whimbrel was the only species that showed steady local population increase in the Iwik area of the Banc d'Arguin in Mauritania. van Roomen *et al.* 2022 indicate an increasing Whimbrel population along the East Atlantic Flyway and assess the population status as 'stable'. The EAF population of Long-tailed Cormorant also shows increasing numbers and is assessed as 'moderately increasing'. Black Heron, a local resident, is a waterbird species that is regarded as a sacred animal by some communi-

ties living around coastal lagoons in Ghana and is traditionally protected by taboos. Ntiama-Baidu (1991b) reported that the species was most abundant at the lagoons where they were protected by these traditional beliefs.

Five shorebird species, Grey Plover, Spotted Redshank, Pied Avocet, Red Knot and Ruddy Turnstone, showed persistent decreases throughout the four decades. The Spotted Redshank and the Pied Avocet occurred in internationally important numbers at several sites during the early stages of the monitoring programme in the late 1980s, with roosts of over 1000 Pied Avocets and several thousand Spotted Redshanks frequently observed (Ntiama-Baidu 1991). The trends observed for Grey Plover, Spotted Redshank and Red Knot are consistent with the assessment by van Roomen *et al.* (2022) where the three are assessed as 'moderate decline' or 'declining'. The decrease observed on the Ghana coast in Pied Avocet and Ruddy Turnstone is not consistent with the flyway assessment, where both species are assessed as 'stable'. This would suggest that these species may be spending the non-breeding season further north, taking advantage of increasing availability of prey items due to changing climate and global warming, as well as the advantage of shorter flight distance to the breeding grounds.

Declines in waterbird populations have been attributed to a number of factors including food availability and declines in preferred prey densities, e.g. for Red Knot and the bivalve *Dosinia isocardia* in Mauritania (Oudman *et al.* 2014;



Long-tailed Cormorant and Caspian Tern

Hans Schekkerman



Memo Horman

Observers group

Oortwijn *et al.* 2022), decreasing reproductive success on the breeding grounds and anthropogenic changes compounded by climate change, which affect arctic breeding birds in particular (Lameris *et al.* 2022). On the other hand, increases in some waterbird species populations have been attributed to increases in abundance of preferred prey species (Zwarts & Blomert 1990, Carneiro *et al.* 2017).

Ghana's five coastal Ramsar sites (Keta, Songor, Sakumo, Densu Delta and Muni) were designated on the basis of their international importance for waterbirds (total numbers as well as individual species numbers). The five sites supported waterbird populations that were above the 20,000 birds threshold at the time, as well as internationally important populations (1% of the estimated EAF populations) of several shorebird species (Ntiemoa-Baidu 1991, Ntiemoa-Baidu & Gordon 1991, Piersma & Ntiemoa-Baidu 1998). The sites also held significant populations of terns. Over the years, the degree of international importance of all sites has declined. The number of shorebird species occurring in internationally important numbers has decreased from eight to two for Keta; seven to two for Songor; and four to three for Densu Delta. The three sites continue to support important populations of Common, Roseate, W African Crested and Sandwich Terns. The Sakumo and Muni sites no longer support more than 1% of any shorebird population. These changes may be partly a reflection of adverse changes in the ecological character of Ghana's coastal wetlands.

The greatest threats to wetlands along the Ghanaian coast are habitat loss (Peters 2019, Ntiemoa-Baidu & Taye

2025), pollution (Nartey *et al.* 2011, Ekumah *et al.* 2020) and coastal erosion. We attribute the declining ecological character of the wetlands and decreases in the populations of waterbirds supported by the sites to (i) losses in wetland areas resulting from urbanisation, encroachment by settlements and industrial developments and coastal erosion, and the associated loss in waterbird feeding habitats, (ii) heavy pollution from both domestic and industrial waste and siltation of open water bodies, leading to declines in invertebrate prey available for waterbirds. The projected impacts of climate change are expected to further affect the ecological integrity of Ghanaian coastal wetlands adversely, with losses in feeding, breeding and roosting areas (Appeaning-Addo & Adeyemi 2013, Avornyo *et al.* 2023). There is already evidence of the adverse impact of flooding on wader nests at the Densu Delta Ramsar site (Taye *et al.* 2022).

Maintaining the long-term waterbird counts entails major challenges, mainly with respect to financing and human capital. However, the work also presents major opportunities in terms of valuable, unique, long-term dataset, development of local ornithological expertise through training, and enhanced conservation awareness among coastal communities. We advocate for increased protection of the designated sites to prevent further encroachment, and continued investment of resources to keep the counts running, and also support relevant ecological studies on individual waterbird species to provide better understanding of the population dynamics observed through these counts.



Kittlitz's Plover | Pluvier pâtre (*Anarhynchus pecuarius*) (Eduard Sangster / Agami)

8. Safeguarding South Africa's Berg River estuary: protecting and restoring vital blue carbon habitats

Giselle Murison

Summary

Estuaries are among South Africa's most productive yet threatened ecosystems. The Berg River estuary on South Africa's west coast is known for its biodiversity, productive fish and invertebrate fisheries, and critical ecological functions, including serving as a nursery for marine fish, facilitating water filtration, and mitigating floods. Designated as a Ramsar site in 2022, the estuary provides essential habitat for numerous waterbird species and serves as a crucial feeding and staging area for large numbers of migratory waders along the East Atlantic Flyway. Additionally, the estuary constitutes a vital blue carbon ecosystem, supporting the largest remaining salt marshes in South Africa, which play a significant role in carbon sequestration and climate resilience. However, its ecological integrity is increasingly threatened by reduced freshwater inflows, declining water quality, expanding development, human disturbance, habitat degradation, and the impacts of climate change. In response, BirdLife South Africa is implementing biodiversity stewardship and estuarine habitat restoration initiatives to help safeguard this system. A 7,000 hectare protected environment has been established along the northern bank, with an additional 11,000 ha identified for future protected and conservation area expansion. By-laws have been introduced and enacted to restrict harmful recreational activities and establish core conservation zones within the estuary. Restoration efforts include rehabilitating salt marsh and reed marsh habitats, stabilising riverbanks, and removing invasive plant species, supported by comprehensive ecological monitoring. Furthermore, BirdLife South Africa engages in community training and awareness initiatives while integrating estuarine (blue and teal carbon) habitat restoration activities into management plans, working towards the development of a dedicated maintenance management plan to guide the long-term maintenance of the estuary's ecological infrastructure. These efforts will contribute to safeguarding the Berg River estuary: supporting its role in climate change mitigation, maintaining its biodiversity, and sustaining local livelihoods for the estuary's long-term ecological and economic viability.

8.1 Introduction

South Africa is home to 290 estuaries and 42 micro-estuaries, representing a high diversity of estuary types stemming from the country's diverse climatic, oceanographic, and geological drivers (Van Niekerk *et al.* 2019a, Van Niekerk *et al.* 2020). Estuaries are among South Africa's most productive habitats, crucial for biodiversity and the provision of key ecosystem services. They support significant resident and migrant waterbird populations and important fish and invertebrate fisheries and perform vital functions, such as water filtration and flood attenuation (Turpie *et al.* 2002, Van Niekerk *et al.* 2019a). Less well-known is their essential role in climate regulation and carbon storage.

The concept of blue and teal carbon has gained significant traction in global climate discussions as key elements in mitigating climate change and enhancing biodiversity. Blue carbon refers to the carbon captured and stored by aquatic 'blue' habitats, specifically coastal and marine ecosystems, such as mangroves, salt marshes, and seagrasses (Nellemann *et al.* 2009, Raw *et al.* 2023). Teal carbon, an emerging term, extends this concept to include freshwater elements and wetlands, recognising their crucial role in carbon sequestration (Zinke 2020). Coastal systems are known to capture and store carbon much faster than land-based forests; they are so effective because the carbon is stored in anoxic (oxygen-depleted) sediments below the waterline (Human *et al.* 2022). Conversely, if these habitats are degraded or destroyed, they release disproportionately higher volumes of carbon.

A high-wave-energy coastline means South Africa's blue carbon ecosystems are confined mainly to its estuaries. Unfortunately, the estuarine biome is South Africa's most threatened (Van Niekerk *et al.* 2019a), with minimal formal protection and significant ongoing challenges. South Africa has already lost about half of this blue carbon habitat (Adams *et al.* 2019, Carnie 2020). Challenges include pollution, habitat destruction due to urban and agricultural development, and reduced freshwater inflow and quality from water extraction for agriculture and other human activities (Van Niekerk *et al.* 2019b). This has led to considerable concern about the sustainability of these ecosystems and the services they provide.

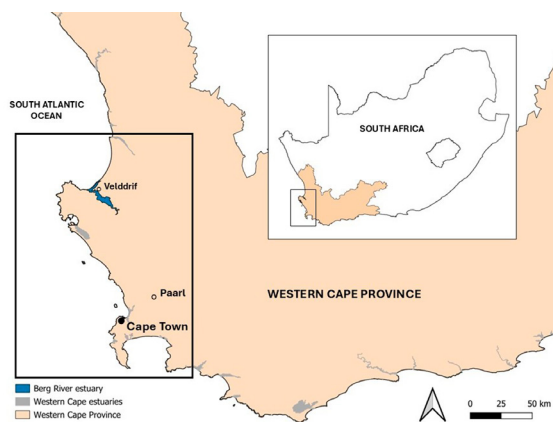


Figure 8.1. Situation of the Berg River estuary within South Africa.

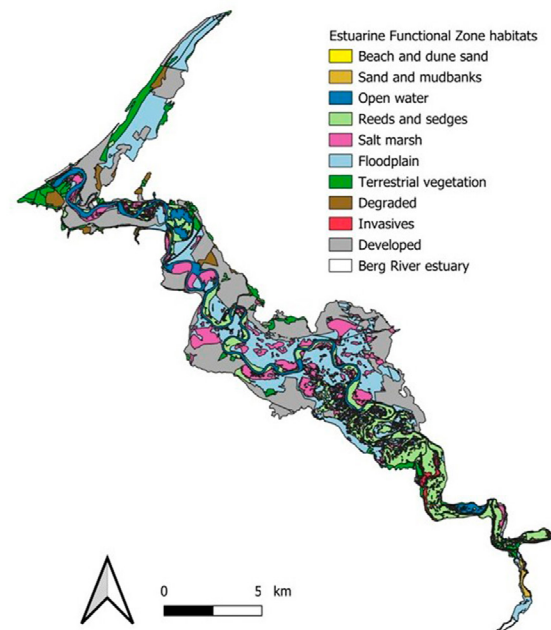


Figure 8.2. Map of the Berg River estuary showing the main habitat and vegetation types.

8.2 The Berg River Estuary

The Berg River estuary, located on South Africa's west coast in the Western Cape Province (Fig. 1), is one of the region's most valuable estuarine systems (Turpie *et al.* 2002, Turpie *et al.* 2007). Declared a Ramsar site in 2022, the Berg is also one of South Africa's largest estuaries, covering almost 8,000 ha and incorporates a wide variety of habitats, including intertidal mudflats, intertidal and supratidal salt marsh, sedge and reed marsh, submerged macrophytes, and open water (Anchor 2008, Van Niekerk *et al.* 2019a; Fig. 8.2). The estuary is fed by the Berg River, which flows from its source in the mountains of the Boland region into the Atlantic Ocean. Richly vegetated and sheltered from the open sea, the Berg River estuary provides ideal nursery grounds for fish and invertebrates, acts as a haven for bird life and other animal communities, and plays a vital role in water filtration, nutrient cycling, and flood attenuation (DEA&DP 2020). The estuary's ecological health directly impacts local livelihoods and the wider economy, such as the West Coast's fisheries, as it serves as a nursery for several commercially important fish species (DEA&DP 2020).

8.2.1 The Berg's Birdlife

The Berg is well-known for supporting abundant and diverse birdlife and is part of a vital network of estuarine systems that play a crucial role within the East Atlantic (African Eurasian) Flyway. More than 270 bird species have been recorded from the Berg River Estuary and Berg River Strandveld Key Biodiversity Areas (KBAs), c. 130 of which are waterbirds. The estuary and floodplain regularly support more than 20,000 non-passerine waterbirds in sum-

mer (Marnewick *et al.* 2015). As South Africa marks the southernmost point of global migration routes, the estuary provides essential habitat for numerous migratory birds and is a critical feeding and staging area for large populations of migratory waders within the East Atlantic Flyway (Velásquez *et al.* 1991, Harebottle *et al.* 2015). Numbers in summer are strongly influenced by the influx of Palearctic migrants, particularly Curlew Sandpiper *Calidris ferruginea* and Little Stint *C. minuta*, which join resident waders like Kittlitz's Plover *Anarhynchus pecuarius*, the regionally, near-threatened Chestnut-banded Plover *A. pallidus* and Pied Avocet *Recurvirostra avosetta*. In recent years, the Berg has seen declines in waterbird numbers, most significantly those of migrant waders like the Curlew Sandpiper, echoing global decreases of around 30–49% over the last three bird generations (15 years), necessitating uplisting to Vulnerable on the IUCN Red List of Threatened Species (BirdLife International 2024). The exact causes of these declines, as well as other migrant wader declines, are unknown. Still, they are likely to include habitat loss and degradation (particularly at stopover and wintering grounds), climate change impacts (affecting breeding productivity), and disturbance and hunting (BirdLife International 2024).

8.2.2 Threats and pressures

Despite its ecological and economic significance, the Berg River estuary faces several pressures. Much of the estuarine habitat at the Berg River estuary is in a poor state of conservation because of broad-scale modification, including permanent opening and canalisation of the river mouth, construction of bridges, a marina, and agricultural,

industrial, and urban development within the floodplain (Marnewick *et al.* 2015). Reduced freshwater inflow and water quality, as well as changes in flow regime, impact habitat health and ecosystem functioning (Marnewick *et al.* 2015; DEA&DP 2020), as does ongoing disturbance through inappropriate recreational activities, like kitesurfing, jet-skiing, walking with dogs off-leash in sensitive areas, etc., over-fishing, and mismanagement of sensitive habitats, such as salt marsh (Marnewick *et al.* 2015). Recently, hypersaline conditions, a result of the prolonged drought along the West Coast, caused massive reed die-off and estuarine habitat degradation, further destabilizing banks and accelerating riparian habitat loss. Climate change exacerbates these challenges by causing sea-level

rise and altering hydrological patterns (DEA&DP 2020). Such changes can disrupt the delicate balance of saline and freshwater inflows necessary for the estuary's ecological functions. Addressing these threats is crucial to maintaining the estuary's role as a carbon sink and area of particular importance for biodiversity. Targeted restorative interventions are required to re-establish these systems' natural functionality and resilience (Raw *et al.* 2023).

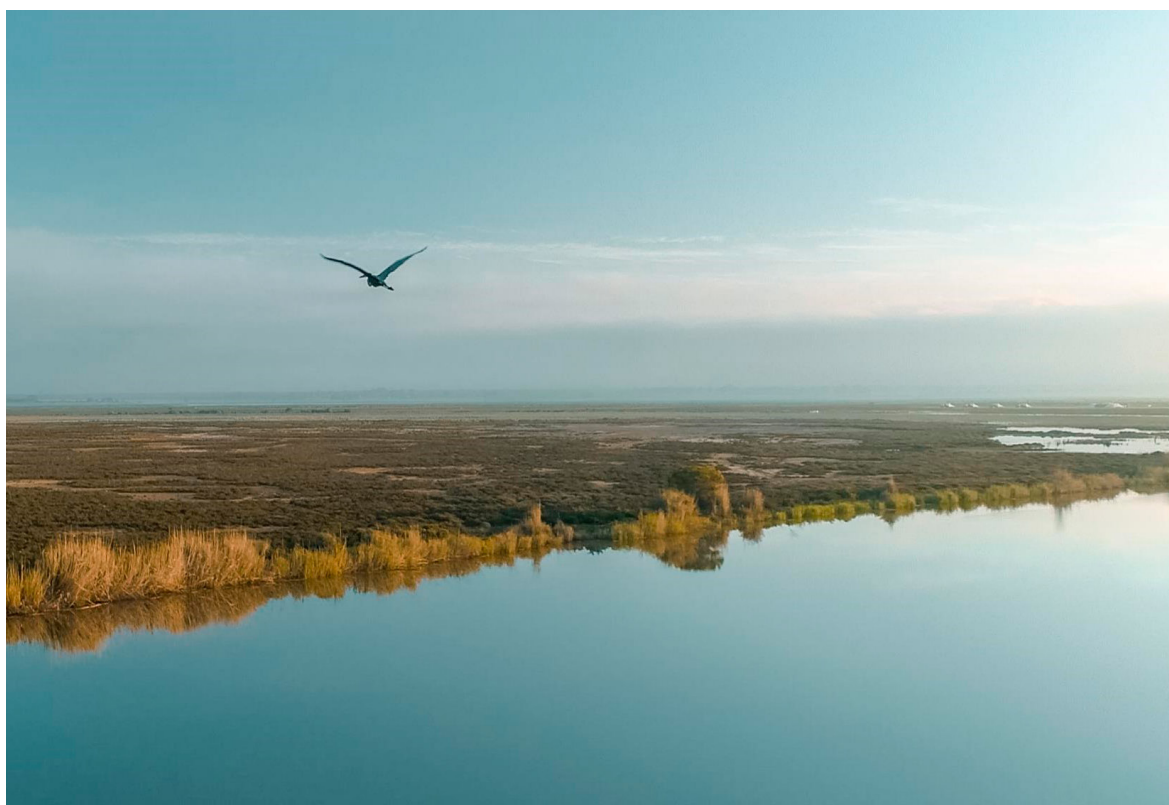
8.2.3 The Berg's blue and teal carbon value

The Berg River estuary is a vital blue carbon ecosystem, supporting extensive salt marshes and other threatened habitats that play a crucial role in carbon sequestration and climate resilience (Raw *et al.* 2023). It is home to the largest



Hugh Harrop / Agami

Curlew Sandpiper | Bécasseau cocorli (*Calidris ferruginea*)



Overview Berg River habitat

existing salt marsh areas in South Africa, with approximately 1,310 ha of intertidal salt marsh and 2,178 ha of supratidal salt marsh (Adams *et al.* 2021). It also supports blue and teal carbon habitats, Cape eelgrass *Zostera capensis*, and significant areas (around 2,000 ha) of sedge and reed marsh, respectively, including large stands of *Phragmites australis* (van Niekerk *et al.* 2019). Given their ecological importance, these habitats have been identified as priorities for restoration and rehabilitation (Adams *et al.* 2021).

Key interventions to protect and restore these carbon-rich ecosystems include restoring salt marsh areas converted for agriculture, ensuring sufficient freshwater flow, improving water quality, restoring commercial salt pans, removing invasive alien vegetation, and stabilising eroding riverbanks (Adams *et al.* 2021). These efforts are essential for enhancing the estuary's role in carbon storage and maintaining the bird life, wider biodiversity and ecosystem services it supports.

8.3 Conservation Actions

BirdLife South Africa is actively engaged in multiple conservation initiatives at the Berg River estuary, working with government agencies and local community groups. These efforts aim to enhance the estuary's formal protection and improve its management through habitat restoration and the rehabilitation of degraded areas. A key focus is also raising awareness of the estuary's ecological, social, and

economic significance, contributing to its sustainable use and long-term viability for both its rich biodiversity and the communities that rely on it.

8.3.1 Improved Protection and Management

Using the mechanism of biodiversity stewardship, BirdLife South Africa is working with the provincial environmental agency, CapeNature, to formally protect privately- and communally-owned estuarine floodplain areas. In South Africa, biodiversity stewardship is an approach to securing land in biodiversity priority areas through entering into agreements with private landowners, Community Property Associations, and the occupiers of communal land, led by conservation authorities and supported by conservation non-governmental organisations (SANBI 2018). Biodiversity stewardship aims to conserve and manage important biodiversity areas through voluntary agreements with landowners and communities. This approach includes provisions outlined in South Africa's National Environmental Management: Protected Areas Act (NEM: PA Act 57 of 2003) and enables conservation efforts to extend beyond state-owned protected areas, allowing landowners to receive recognition for the biodiversity value of their land. It also supports them in enhancing land management practices that promote both ecological sustainability and sustainable livelihoods.

This approach was used at the Berg River estuary to declare a 7,000-ha protected environment on the north-



BirdLife South Africa

Overview Berg River habitat

ern bank of the estuary. A protected environment is one of four main categories of protected area defined in NEM: PA Act. It can be declared over multiple properties and offers more flexible land use than nature reserves and national parks; landowners can continue their existing activities within the protected environment while integrating designated conservation areas, provided these activities do not harm the site's biodiversity. The conservation areas within the protected environment at the Berg comprise more than 1,000 ha of estuarine floodplain habitat and approximately 5,000 ha of intact, threatened fynbos and strandveld vegetation types. This will be the first formal protection for the Berg River estuary and the first protected environment in South Africa dedicated to the long-term protection of estuarine habitats. Additionally, a further 11,000 ha are targeted for future expansion of protected and conservation areas at this site.

Stewardship activities directly address a key objective outlined in the Groot Berg River Estuarine Management Plan (EMP), which prioritises the increased formal protection of estuarine coastal public property and privately-owned estuarine areas at the Berg. The National Estuarine Management Protocol, promulgated in 2013 under the National Environmental Management: Integrated Coastal Management Act (Act No. 24 of 2008, as amended by Act No. 36 of 2014), sets out the minimum requirements for individual estuarine management plans, which aim to guide the sustainable use, conservation, and

management of estuaries in South Africa. The Groot Berg River EMP is supported by an active Estuary Advisory Forum, which advises on its implementation.

BirdLife South Africa, in collaboration with Forum members, has successfully incorporated provisions into the estuary's recreational use by-laws to protect threatened intertidal estuarine habitats and sensitive bankside erosion areas. This has facilitated the designation of core conservation zones within the estuary, where recreational activities are strictly prohibited. Additionally, instructional signage has been developed to minimize disturbances to sensitive bankside erosion areas and salt marsh habitats, forming the foundation for enforcing compliance through fines for by-law violations.

8.3.2 Habitat Restoration

In 2023, funded by the Ecological Restoration Fund and the Royal Society for the Protection of Birds, BirdLife South Africa launched a dedicated project focused on actively protecting and restoring key, threatened estuarine habitats, specifically salt marsh and reed marsh habitats. This initiative builds upon several small-scale pilot projects conducted by BirdLife South Africa at the Berg River estuary and seeks to trial estuarine habitat restoration techniques and soft-engineering interventions for bankside erosion control. The primary objectives are to evaluate the effectiveness of these approaches and establish a framework for their broader implementation, specifically



BirdLife South Africa

Signs inform the public



BirdLife South Africa

Berg River overview

through the development of an estuarine Maintenance Management Plan (MMP). MMPs guide ongoing infrastructure maintenance, including ecological infrastructure, within environmentally sensitive areas, such as estuaries, ensuring that maintenance activities do not degrade ecosystems, water quality, or biodiversity. A critical component of the project is the integration and mainstreaming of estuarine habitat restoration and blue carbon opportunities within estuary management plans and their reporting frameworks. It also aims to generate knowledge products—such as reports and media materials—to facilitate the scalability of these restoration efforts across the Western Cape’s priority estuaries and, more broadly, within South Africa’s estuarine systems.

The initiative has laid a strong foundation for piloting estuarine habitat restoration, rehabilitation, and safeguarding measures. Comprehensive monitoring protocols, including water, soil, bathymetric, and vegetation condition assessments, have been implemented. Additionally, spatial databases have been developed to document estuarine habitat extents (including blue and teal carbon habitats) and identify areas needing rehabilitation or restoration. These databases are continually ground-truthed and updated, aligned with the latest research protocols. Restoration activities are underway, including re-seeding and re-vegetation efforts in degraded reed and salt marsh areas. These efforts are supported by propagating key indigenous plant species in partnership with a local nursery. Furthermore, the project addresses habitat degradation, particularly within supratidal salt marsh ecosystems, by mitigating the impacts of overgrazing and disturbance using measures such as fencing, establishing exclusion areas, and re-vegetation, working with landowners to ensure the sustainable use of the floodplain.

8.3.3 Monitoring, training, and environmental-awareness raising

BirdLife South Africa is a key Western Cape partner in the Coordinated Waterbird Counts (CWAC), a national citizen science programme conducting bi-annual waterbird counts at registered waterbodies across South Africa. BirdLife South Africa provides government and non-government stakeholders at estuaries across the Western Cape Province training in CWAC, waterbird identification, and estuarine habitat restoration techniques. The Berg River projects also contribute to local capacity building by employing and upskilling community members to assist with monitoring and restoration activities. Additionally, BirdLife South Africa engages with local schools and businesses to raise awareness about the estuary’s ecological importance, its role in supporting biodiversity, and the ecosystem services it provides. Public awareness initiatives emphasise the need for the conservation of vulnerable estuarine habitats along the East Atlantic Flyway. These efforts include the development of informational signage, educational materials, and community outreach programs



Hans Scheekerman

Little Stint & Curlew Sandpiper | Bécasseau minute & Bécasseau cocorli (*Calidris minuta* & *Calidris ferruginea*)

to foster greater protection and restoration of these critical ecosystems.

8.4 Going Forward

Developing a comprehensive maintenance management plan for the Berg River estuary is essential for expanding restoration efforts at the Berg. Similarly, extending these initiatives to include other estuarine KBAs along the West Coast, such as the Olifants River estuary to the north, will facilitate the exploration of alternative mechanisms for safeguarding critical blue and teal carbon habitats. Further research is also needed to enhance our understanding of how waterbird species, particularly migratory waders, utilize the network of estuarine sites along the West Coast. This will be pursued through a tagging and colour-ringing programme to track species movements and habitat use.

South Africa’s Berg River estuary exemplifies a vital estuarine ecosystem that serves as an essential blue and teal carbon sink. Recognising its value and investing in its protection and better management, including habitat restoration, are crucial steps toward promoting its long-term sustainability and resilience in the face of environmental change. By safeguarding this estuary, BirdLife South Africa can help secure its role as a haven for diverse and abundant waterbird populations while sustaining the livelihoods of communities that depend on it.

Acknowledgements

I am grateful to BirdLife South Africa staff members, Dr Charlotte Boyd and Jessica Wilmot for their input on this chapter. The work at the Berg River estuary is supported by the Royal Society for the Protection of Birds, the Ecological Restoration Fund, WWF South Africa’s Elizabeth Harding Bequest, and the Rupert Nature Foundation.



Caspian tern | Sterne caspienne (*Hydroprogne caspia*) (Daniele-Occhiato / Agami)

9. Breeding populations of West African gulls and terns: an overview of counts, compared with data obtained in 2023

Jan Veen, Eelke Folmer, Hanneke Dallmeijer, Thor Veen, Marc van Roomen, Khady Gueye Fall & Aïssatou Yvette Diallo

Summary

Between 1998 and 2023 four gull and tern species have been studied in breeding colonies along the coasts of Mauritania, Senegal, The Gambia, Guinea Bissau and Guinea: the Grey-headed Gull *Chroicocephalus cirrocephalus*, Slender-billed Gull *Chroicocephalus genei*, Caspian Tern *Sterna caspia* and West African Crested Tern *Thalasseus albididorsalis*. Here we present an overview of all data on breeding population sizes. Historical counts covering all breeding sites indicate stable populations of approximately 14,000 pairs of Grey-headed Gulls, 13,000 pairs of Slender-billed Gulls, 28,000 pairs of Caspian Terns and 100,000 pairs of W African Crested Terns between 1998 and 2019. However, a census carried out in 2023 revealed significantly lower numbers for most species across the majority of sites. Firm conclusions about the totals of the 2023 census are impossible as major methodological problems in survey methods occurred between the regions. The declines can partly be explained by (1) counts conducted at inappropriate dates, (2) colonies being not or partly included in a count and (3) mortality from an avian influenza epidemic that particularly affected W African Crested Tern populations in 2023.

We recommend implementing a more standardised and complete monitoring program with region-wide censuses at 3-year intervals. This should include: (1) comprehensive training of local conservation staff in census methodology, (2) development of a standardised protocol with specific survey timing aligned with breeding periods, and (3) systematic collection of environmental data to better understand factors affecting breeding success and population dynamics. These measures will provide more reliable population estimates and improve our ability to identify and address conservation threats to these important colonial waterbird species.

Résumé

Entre 1998 et 2023, quatre espèces de goélands et de sternes ont été étudiées dans des colonies de reproduction le long des côtes de Mauritanie, du Sénégal, de la Gambie, de la Guinée-Bissau et de la Guinée : la Mouette à tête grise (*Chroicocephalus cirrocephalus*), le Goéland railleur (*Chroicocephalus genei*), la Sterne caspienne

(*Sterna caspia*) et la Sterne royale africaine (*Thalasseus albididorsalis*). Les dénombrements historiques couvrant tous les sites indiquent des populations stables entre 1998 et 2019 : environ 14 000 couples de Mouettes à tête grise, 13 000 de Goélands railleurs, 28 000 de Sternes caspiennes et 100 000 de Sternes royales africaines. Cependant, un recensement de 2023 a révélé des chiffres nettement inférieurs pour la plupart des espèces dans la majorité des sites. Il est difficile de tirer des conclusions fermes en raison de problèmes méthodologiques : dates inadaptées pour les comptages, colonies partiellement incluses, et mortalité liée à une épidémie de grippe aviaire en 2023, ayant touché surtout la Sterne royale africaine. Nous recommandons la mise en place d'un programme de suivi standardisé avec des recensements régionaux tous les 3 ans, incluant : (1) formation des personnels de conservation aux méthodes de recensement, (2) protocole standardisé avec calendrier précis de comptage selon les périodes de reproduction, et (3) collecte systématique de données environnementales pour mieux comprendre les dynamiques de population.

9.1 Introduction

The Atlantic coast of West Africa is strongly influenced by the Sahelian Upwelling Marine Ecoregion. The upwelling of nutrient-rich water fuels the biological productivity of the system which results in abundant food for a variety of waterbirds and seabirds. Among the latter are four gull and tern species which nest in colonies along the coasts of Mauritania, Senegal, The Gambia, Guinea Bissau and Guinea: the Grey-headed Gull *Chroicocephalus cirrocephalus*, Slender-billed Gull *Chroicocephalus genei*, Caspian Tern *Sterna caspia* and West African Crested Tern *Thalasseus albididorsalis*.

Knowledge about population size and trends is essential for determining a species' conservation status and implementing effective protection measures. The Wadden Sea Flyway Initiative, Wetlands International and BirdLife International jointly coordinate annual January censuses in the West Africa region (van Roomen et al 2022), which primarily target waders and other waterbirds at high tide roosts. Gulls and terns, however, often disperse at sea rather than

Table 9.1. Sites and islands where breeding colonies of gulls and terns were found in the period 1998–2023. Sizes of islands are approximations and depend on e.g. the stage of the tide and erosion. Many islands have changed considerably in shape and size over the years.

site name, country	island name	coordinates	size (ha)	habitat characteristics
Banc d'Arguin, Mauritania	Îlot aux Pélicans	20043°N 160 41°W	2	oceanic, sandbank on rocky sub-strate
	Île Marguerite	20016°N 16019°W	170	sheltered and low vegetation
	Île Ardent	20053°N 16027°W	1150	sandy, partly low vegetation
	Île Kiaone	20000°N 06018°W	170	barrier island, rocky
	Île Arel	19054°N 16030°W	9	barrier island, rocky
	Île Zira	19052°N 16017°W	4	sheltered, sandy
	Île Nair	19051°N 16052°W	25	sandy
Langue de Barba-rie NP, Senegal	Îlot aux Oiseaux	15050°N 16031°W	70	in estuary of Senegal River, sandy
Saloum Delta NP, Senegal	Jakonsa	13047°N 16044°W	50	barrier island, sandy
	Ansoukala	13042°N 16040°W	7	barrier island, sandy
	Île aux Oiseaux	13037°N 16037°W	160	barrier island, sandy
Tanji Bird Reser-ve, The Gambia	Bijol Islands	13023°N 16049°W	13	oceanic, sandy on rocky substrate
Kalissaye Orn. Reserve, Senegal	Kalissaye	12041°N 16047°W	15	oceanic, sandbank
Guinea Bissau	Jêta-Bantambour	11058°N 16018°W	8	sheltered, sandy
	Acapa Imbone	11001°N 16004°W	1	barrier island, sandy
	Cavalos	10059°N 15040°W	400	barrier island, sandy
Guinea	Île du Naufrage	10037°N 15022°W	1	oceanic, sand on rocky substrate

Table 9.2. Breeding characteristics of four species of gull and tern nesting along the W African seaboard. The vegetation mentioned almost exclusively consisted of Beach Morning Glory *Ipomoea pes-caprae* and Sea Purslane *Sesuvium portulacastrum*. Colony size range refers to birds nesting together and not to all birds of a particular species nesting on an island. Incubation and fledging periods after Del Hoyo (1997) and Peeters (1999).

species	nesting habitat	internest distance (m)	colony size range	incubation period (days)	fledging period (days)
Grey-headed Gull	sparse to dense vegetation	1-5	10 - >1000	24 (22-26)	c. 35
Slender-billed Gull	sparse vegetation	0.3-0.5	10 – 200	25 (22-28)	c. 35
Caspian Tern	bare sand to sparse vegetation	1-3	10 - >1000	27 (26-28)	35-45
W African Crested Tern	bare sand	0.22	100 - >10000	28 (25-31)	c. 30

Table 9.3. Calculated proportion of breeding birds missed during a single count in May, based on data collected in the Saloum Delta and the Bijol Islands. For explanation see text.

species	early breeders	late breeders	total missed
Grey-headed Gull	16%	35%	51%
Slender-billed Gull	6%	41%	47%
WA Crested Tern	19%	9%	28%
Caspian Tern	72%	10%	82%

Table 9.4. Calculation of the size of the total W African (WA) breeding population of Slender-billed Gull, Grey-headed Gull, Caspian Tern and W African Crested Tern based on counts covering the whole region in single years.

Species	year	single counts		monthly counts	total WA population (pairs)
		counted	corrected		
Slender-billed Gull	2004	3845	7254	7642	14896
	2005	2734	5158	7428	12586
	2015	2248	4241	7527	11768
Grey-headed Gull	2004	2322	4739	10069	14808
	2005	19	39	13355	13394
	2015	800	1633	12201	13834
Caspian Tern	2015	3735	20750	7636	28386
W African Crested Tern	2015	45837	63663	27732	91395
	2019	75135	104354	3769	108123

concentrate at these roosts, resulting in unreliable (incomplete) counts for these species. For these colonial breeders a census of the number of breeding pairs is a far better method to collect information about population changes. An additional advantage of censusing the breeding population is that information can be collected about breeding habitat suitability, reproductive output and other environmental factors affecting the species.

From 1998 onwards a research and monitoring program was carried out along the W African coast with attention to breeding parameters, diet composition and environmental variables of the above mentioned gull and tern species. Field activities started in Senegal and were expanded to include breeding colonies in The Gambia, Mauritania, Guinea Bissau and Guinea, although Senegal remained the focal point of the program. In the course of the years local conservation staff was trained, initially in conventional counting methods, and more recently also in drone photography and AI. Here we present an overview of the results obtained on breeding population size in the past and compare with a recent census carried out in 2023, by the authors (EF, YD) in Senegal together with local conservation staff and by local conservation staff in Mauritania and Guinea Bissau.

9.2 Methods

9.2.1 Study area and species

Observations were made on 17 breeding islands along the Atlantic Coast of Mauritania, Senegal, The Gambia, Guinea Bissau and Guinea (Table 9.1 and Figure 9.1, see also Veen *et al.* 2009). In many cases several species nested in close proximity to each other. However, there were marked differences in habitat choice, colony structure, incubation and fledging periods (Table 9.2).

9.2.2 Timing of counts

Breeding colonies were either counted during one visit (single counts), or by means of successive visits at monthly intervals (monthly counts). Single counts were usually done in the middle of May and included breeding pairs with both eggs and chicks. When a monthly census program was carried out only nests with eggs were counted. Counts were done on the 15th of each month to the extent possible. Breeding population size was then calculated by summing numbers counted during successive months. Note that the incubation period of all species is close to, but somewhat shorter, than a month (see Table 2). It was therefore considered that eggs seen in any month were new, additional nests to the ones counted in earlier months.

It should be stressed that a single count in May will result in an incomplete estimate of the breeding population size. May has been selected because it corresponds to



Figure 9.1. Sites with breeding colonies of Grey-headed Gulls, Slender-billed Gulls, Caspian Terns and W African Crested Tern included in the monitoring program.

the peak of laying for most of the species (Figure 2). However, the species vary in laying synchrony – those with asynchronous laying will tend to be underestimated by using single counts. Because single counts tend to include both breeding pairs with eggs as well as chicks, such counts will include part of the birds that started laying in April. Assuming that half of the birds laying in April are included in such a count, birds breeding before and after the period covered by the count will still be missed (the latter are defined as “early” and “late” breeders, respectively, in Table 3). The missed part of the population thus calculated varies between 28% in the W African Crested Tern and 82% in the Caspian Tern, with intermediate figures for Slender-billed Gull (47%) and Grey-headed Gull (51%) (Table 3).

9.2.3 Census methods and counting errors

The following methods were used for counting or estimating breeding numbers. (1) Binocular scans: estimating

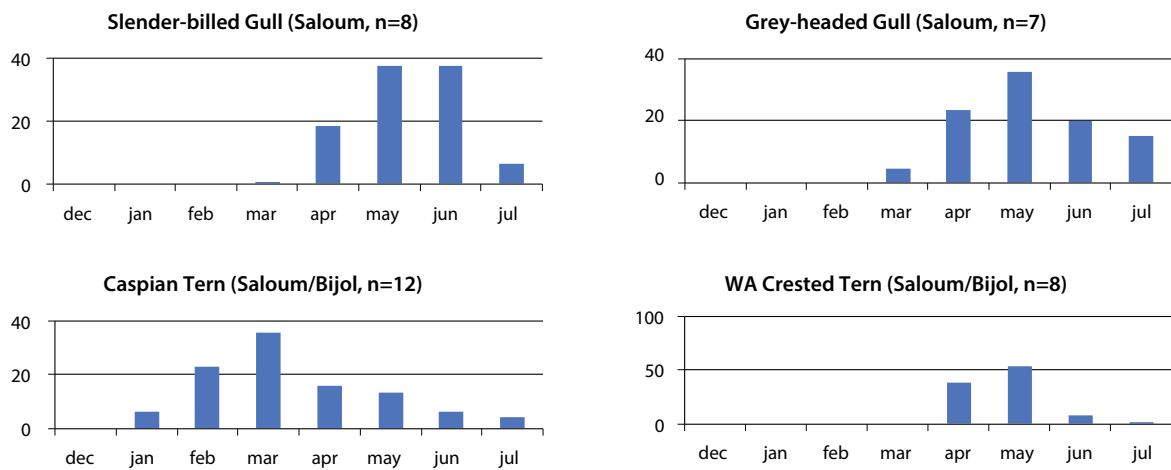


Figure 9.2 Monthly egg-laying patterns (mean % of eggs laid) of Slender-billed Gull, Grey-headed Gull, Caspian Tern and W African Crested Tern based on monthly counts of nests with eggs in the Saloum Delta and the Bijol Islands. N-values refer to number of years from which data were available.

the number of birds (usually sitting on a nest) from a distance using binoculars. (2) GPS counts: calculating the number of nests in a colony by multiplying colony surface area measured by GPS and mean nest density measured in at least 3 randomly chosen 4 m² plots. (3) Drone counts: counting the number of nests using drone photography. Photos were collated and the number of nests could then be counted manually, or automatically using computer vision technology (Folmer *et al.* 2019, Folmer 2023). (4)

Manual counts: manually counting the number of nests with eggs by walking through the colony. An analysis of counting methods showed major differences in the errors involved, drone counts being far more accurate and causing less disturbance than any of the other methods (Veen *et al.* 2022)

9.2.4 2023 survey

The results of the 2023 census are exclusively based on



Grey-headed Gull | Mouette à tête grise (*Chroicocephalus cirrocephalus*)

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single counts carried out between 12 May and 14 June. In Mauritania (Banc d'Arguin), data were collected during a series of ground counts. In Senegal (Saloum Delta and Langue de Barbarie) colonies were mapped using drones and in Guinea-Bissau (Bantambour and Acapa-Imbone) a combination of drones and ground counts were used (Monteiro & Ledo Pontes 2023). Guinea was not included in the census because of the inaccessibility of the site (Île Alcatraz). No surveys were undertaken in The Gambia because the Bijol Islands have become unsuitable for breeding. A complete census of Slender-billed Gulls, Caspian Terns and W African Crested Terns was carried out, but incomplete data were collected on Grey-headed Gulls, the latter being caused by the dispersed distribution pattern of this species.

9.3 Results

An overview of all numbers of breeding pairs at the site level collected between 1998 and 2023 is given in Figure 3 (data per breeding island are given in Veen *et al.* 2023). Data are based on single counts (light bars) and monthly counts (dark bars). Single counts have been corrected for birds missed as described above (table 3). It should be stressed that single counts were carried out using different methods, some of which (e.g. binocular scans) may involve substantial counting errors (Veen *et al.* 2022).

Over the years, it appears that there are major differences in the numbers of the different species in the sites

included. W African Crested Terns were present in large numbers at the Banc d'Arguin, Delta du Saloum, Bijol Islands and Guinea Bissau, Caspian Terns only occurred in large numbers at the Banc d'Arguin and in the Saloum Delta, Slender-billed Gulls were especially numerous at the Banc d'Arguin, the Langue de Barbarie and the Saloum Delta, whereas large numbers of Grey-headed Gulls were only found in the Langue de Barbarie and the Saloum Delta. There were also major differences between years within a particular site, which might be related to (1) counting errors, especially in case of single counts, and (2) the tendency of birds to move between sites with worsening breeding conditions such as increased predation or habitat deterioration. For instance, the disappearance of large numbers of W African Crested Terns and other species from the Bijol Islands was related to severe habitat erosion which made these islands unsuitable for breeding.

Table 4 gives the results for years in which counts are available for all breeding islands, providing estimates of the total population of the species monitored (2004, 2005 and 2015 depending on the species). Single counts are corrected based on the proportion of birds missed as presented in Table 3. The results of 2023 are not included as these cannot be considered a total count (see discussion). Considering the small number of years for which data are available no conclusions can be drawn with respect to a trend in the number of breeding pairs for any of the species studied.

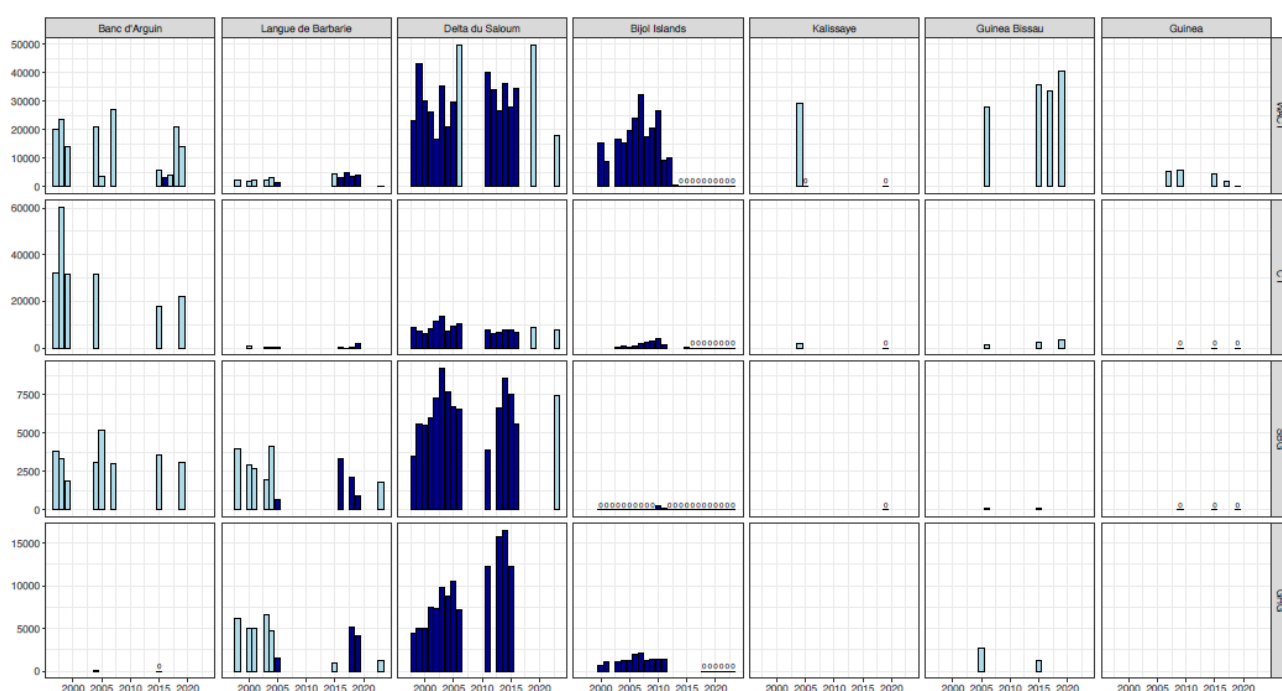


Figure 9.3. Overview of the number of breeding pairs of W African Crested Tern, Caspian Tern, Slender-billed Gull and Grey-headed Gull counted between 1998 and 2023 in seven sites along the Atlantic coast of Mauritania, Senegal, The Gambia, Guinea-Bissau and Guinea. Light blue bars denote estimates based on single counts, dark bars on monthly counts. The absence of breeding birds has been indicated by 0, empty spaces refer to 'no data'.

9.4 Discussion

9.4.1 Breeding distribution and total population size

All species dealt with in this study nest on isolated islands which are situated within formally protected areas. Over a timespan of 26 years, 17 breeding islands have been identified, most of which were used by the gulls and terns during many years in succession. Numbers can fluctuate strongly between years because of the ability to move between islands. Such changes could be related to predation, human disturbance and nest flooding. The total W African breeding population could only be calculated for a few years for all species. Because of the many gaps in our data set it is not possible to detect changes in population size over the past two decades.

9.4.2 Counts on high tide roosts vs. breeding colonies

Counts on high tide roosts (van Roomen *et al.* 2022) can be expected to result in complete counts of wader populations and other species using tidal flats, but will underestimate numbers of species like gulls and terns that use the estuarine open water and outer sea as feeding grounds also during high tide. As a consequence population estimates of gulls and terns are usually based on a census of the number of pairs in breeding colonies. However, our results also show that data on breeding numbers should be considered with caution, as the accuracy of the meth-

ods used may vary considerably and strongly depends on the experience of the observer. Moreover, it should be stressed that breeding numbers ignore year-classes of non-breeding birds which may be substantial for species starting breeding at the age of 3 years or older, like Caspian Terns (Cramp & Simmons 1984). Yet, regular counts of breeding populations should be regarded of high value as they may easily reveal population changes. Moreover, the fieldwork involved in a census can be combined with the collection of information on conditions affecting such changes. Over the years, it has been shown for instance, that human disturbance, predation and erosion of suitable breeding habitat were important factors determining the size and distribution of the various species in most sites (Veen *et al.* 2023).

9.4.3 Lessons learnt from the 2023 census

In 2023 partner organizations in the region were trained in using drone photography for counting the breeding birds. All teams had drones to be used during fieldwork. In all cases the data obtained resulted in far smaller numbers of breeding birds as compared to censuses in 2019 and previous years (Table 4), which raises the question whether a recent decline occurred in the size of the populations.

Information on the circumstances under which the 2023 count was carried out shows the following: (i) On the Banc d'Arguin numbers of W African Crested Terns were considerably lower than in 2019 (89 vs 10,000). However,



Hans Schekkerman

Slender-billed Gull | Goéland railleur (*Chroicocephalus genei*)



Nathalie Gibon

Taking notes by well known observer

counts were made on the 14th of June instead of in mid-May. Most terns must have finished breeding by then. (ii) At the Langue de Barbarie, counts were carried out correctly, but the relatively small number of W African Crested Terns might be explained by the impact of highly pathogenic avian influenza (HPAI) on this species earlier in the season. (iii) In the Saloum Delta only 13,000 pairs of W African Crested Terns were counted as compared to 35,000 in 2019. Also at this site large numbers of HPAI casualties were recorded by park staff. (iv) In Guinea-Bissau only 2400 W African Crested Terns were counted as compared to 29,000 in 2019, which might again be explained by the date on which the census was made (13 June instead of mid-May). No information on Slender-billed Gulls and Grey-headed Gulls was provided. This information leads to conclude that: (1) it is of vital importance to carry out the census at the appropriate date, (2) all species should be included in the counts and (3) avian influenza may have led to considerable mortality among W African Crested Terns in 2023.

Considering the above, conducting a follow-up survey becomes imperative. A fresh assessment would not only provide clarity regarding the current population figures but also offer insights into the factors contributing to a possible decline, such as the recent avian influenza epidemic. Some additional training of local teams might be needed. Preferably, a future census should also be used to systematically collect information required for informing conservation efforts and implementing targeted interventions aimed at mitigating any threats to the population.

Acknowledgements

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Common Tern | Sterne pierregarin (*Sterna hirundo*) (Marc Guyt / Agami)

10. Terns in trouble: lessons from a long-term, individual-based study

Sandra Bouwhuis

Summary

Many migratory animals are declining. Understanding when and where which threats are affecting which vital rates is crucial if we are to develop and implement effective conservation strategies. As such, demographic monitoring is more important than ever. Here I present a detailed long-term individual-based study on Common Terns *Sterna hirundo* from a breeding colony located on the German North Sea coast. This colony provides a rare example of a population that, until recently, was thriving, at least when considering the number of breeding pairs. Annual rates of reproductive performance and survival were variable, but not showing a trend over time. Food availability explained a substantial amount of variation in annual reproductive success, but not adult survival, a pattern expected when breeding birds prioritise their own survival over that of their chicks. During the breeding seasons of 2022 and 2023, the colony was hit by an outbreak of highly pathogenic avian influenza. The population decline that followed was estimated to be in the range of 61–69%, and reproductive output was at an all-time low. Further monitoring will be crucial to assess whether recovery is possible, especially given the fact that newly implemented tracking and physiological studies hint at pollution problems in the main wintering grounds used by the birds. Continuing these new studies to deepen our understanding of potential threats and demographic processes at the scale of the flyway will be key.

Résumé

*De nombreuses espèces migratrices sont en déclin. Il est crucial de comprendre quand et où les menaces affectent quels paramètres démographiques pour élaborer des stratégies de conservation efficaces. Le suivi démographique devient ainsi indispensable. Ici nous présentons une étude de recherche individuelle de long terme sur la Sterne pierregarin (*Sterna hirundo*) dans une colonie nichant sur la côte de la mer du Nord en Allemagne. Jusqu'à récemment, cette population semblait prospère en termes de nombre de couples nicheurs. Les taux annuels de reproduction et de survie étaient variables, sans tendance claire. La disponibilité en nourriture expliquait une grande part des variations du succès reproducteur, mais pas de la survie adulte – les oiseaux donnant priorité à leur propre survie. Durant la saison de reproduc-*

tion de 2022 et 2023, une épidémie de grippe aviaire hautement pathogène a frappé la colonie, entraînant une chute estimée à 61–69 % de la population et un succès reproducteur historiquement bas. Un suivi continu est essentiel pour évaluer les perspectives de récupération, d'autant que des études récentes montrent des signes de pollution dans les zones d'hivernage. Approfondir ces recherches à l'échelle de la voie de migration est indispensable pour mieux comprendre les menaces et dynamiques démographiques.

10.1 Introduction

As part of the human-induced sixth mass extinction (Ceballos et al. 2015), large numbers of populations of migratory animals have been found to decline (UNEP-WCMC 2024). If we are to develop and implement effective policies and measures to protect migratory species, we need to identify the causal pathways underpinning these declines. This process will require multiple steps. We will need to (i) estimate population size, and (ii) quantify how it emerges from birth and death rates, as well as from rates of immigration and emigration. We will (iii) need to pinpoint when in the annual cycle threats to vital rates occur, and, specific to migratory species, where this happens. Only then can we (iv) make an inventory of potential factors responsible, test and identify these, and (v) find ways to mitigate negative effects.

This is no easy task, because each step, even the seemingly simplest one, comes with its own challenges. To not yet speak of the complications coming with migratory animals per se, the size of populations can, for example, be surprisingly hard to establish when, among others, individuals are not easy to see or distinguish, move in and out of the population for foraging, or when breeding occurs asynchronously. Similarly, although it is clear that changes in population size will be the result of variation in the relative rates of birth and immigration versus death and emigration, estimation of each of these rates requires large-scale monitoring of a representative set of individuals across sufficient time. To provide an example, fledglings can vary substantially in the age at which they recruit back into their breeding population (e.g. Zhang et al. 2015a), which needs to be taken into account when estimating the contribution of reproductive performance to

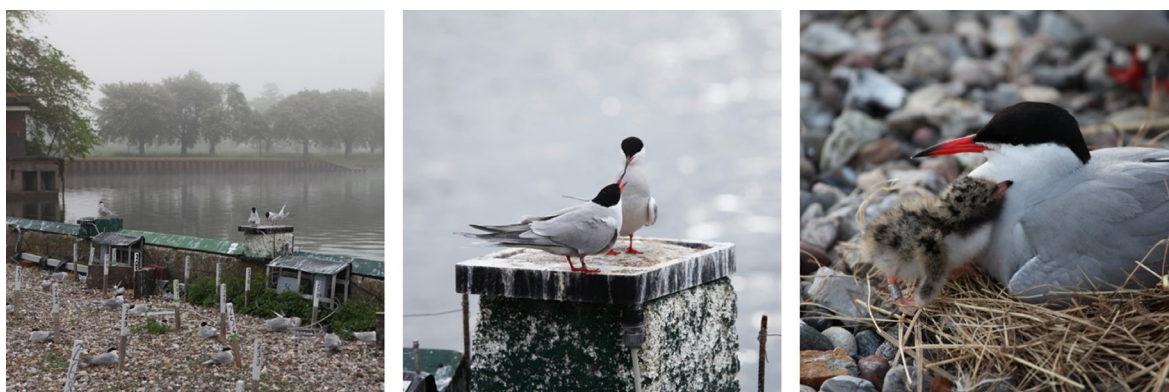


Figure 10.1. The Banter See Common Tern colony on a foggy May morning (a), a Common Tern couple sitting on one of the 44 platforms that is equipped with an antenna that reads transponder codes to register the presence of breeding as well as non-breeding transponder-marked birds (b), and a ringed chick with its parent (c).

population size. To complicate that matter even more, the age of recruitment can have effects on population size even further downstream too, in the case of it correlating with survival and reproductive performance in adulthood (Zhang *et al.* 2015a). Identifying such connections among events at different stages of the life history of individuals, and incorporating them in calculations of changes in population size, is only possible in long-term individual-based studies, which themselves are not always easy to maintain (Clutton-Brock & Sheldon 2010).

Here, I will describe a long-term individual-based study of Common Terns *Sterna hirundo*, relatively long-lived, colony-breeding and piscivorous seabirds, and use data collected since 1992 to present annual estimates of population size, local reproductive performance and survival to tackle parts of steps (i) and (ii) mentioned above. Moving towards steps (iii) and (iv), I will provide case studies to show how reproductive performance and adult survival at the breeding grounds are affected by fish availability and a catastrophic two-year outbreak of avian influenza, before introducing the tracking study I added to the long-term study in 2016 in the hope of in the next decade elucidating further environmental factors underpinning variation in these main fitness components, from a full annual cycle perspective.

10.2 Banter See Common Terns - a long-term, individual-based study

My team and I study Common Terns at a breeding colony located at the Banter See in Wilhelmshaven (Fig. 10.1a). Here, the presence and reproductive performance of individual Common Terns has been monitored following standard protocol since 1992, a study initiated by the visionary Peter H. Becker. In 1992, 101 adult birds were caught and marked not only with rings, but also with individually numbered subcutaneously injected transponders. Since 1992 all locally hatched chicks have similarly been marked with such a transponder shortly prior to fledging.

The colony site consists of six concrete islands, each of which measures 10.7×4.6 m and is surrounded by a 60 cm wall. The walls support 44 platforms for terns to land on, and each platform is equipped with an antenna that reads transponder codes at a distance of <11 cm every 5–10 sec to automatically record the presence of transponder-marked individuals, independent of whether they are breeding or 'just visiting' (Fig. 10.1b). All transponder-marked breeders are additionally identified by placing an antenna around each clutch for at least 24 hours during incubation, which is shared between partners. As such, for all locally hatched birds, we know who is present at the breeding colony in which year, and who is breeding with whom.

Three times weekly checks of the colony provide information on reproductive performance of breeders: laying date, clutch size, and brood size are recorded and chicks are individually marked with metal rings upon hatching (Fig. 10.1c). These checks also allow us to assign a hatching order to each chick, to monitor the fate of each chick and to assess the number and weight of fledglings.

After fledging, birds usually do not return to the Banter See for 2–4 years, after which they often 'prospect' for a year, before settling as breeders. Once established as local breeders, i.e. once they have recruited, the birds are highly philopatric: skipped observation years are relatively rare (e.g. 65 bird-years out of a maximum number of 1670 bird-years between recruitment and age at last observation for the years 1992–2011, i.e., 3.9%; Bouwhuis *et al.* 2015). Skipped observation two years in a row is even more scarce (3.0% over the same time period; Bouwhuis *et al.* 2015), such that we can fairly safely assume birds to be dead if they have not been registered at the Banter See for two years or more.

Because immigrants have not been caught and marked, rates of immigration cannot be calculated directly, and only inferred from the number of unringed breeders, by

assuming that annual survival, philopatry and breeding probability are identical between locally hatched birds and immigrants. Emigration rates are even harder to estimate, but in the period 1992–2021, in which 9653 chicks fledged, 60 of these were reported breeding elsewhere, on average 105 km away (range 6–735 km). This is a rate of 0.6%, whereas 2092 chicks recruited locally, i.e. 21.7%.

10.3 Population size

Population size can be estimated using various approaches. One would be to count the number of breeding pairs, although this leaves the question when such a count should best be made, given that breeding is often asynchronous, failed breeders produce replacement clutches and successful breeders can initiate second clutches. At the Banter See, where every nest gets marked and checked for parental identity, I first use the nests with at least one transponder-tagged breeding bird to calculate which proportion of these nests are first clutches, and which are second or replacement clutches. Assuming these proportions to be the same for nests of two non-locally hatched breeders, I estimate the number of first clutches for this category of nests too, and add it to the number of first clutches for breeding pairs consisting of at least 1 locally hatched transponder-marked bird. Doing so, the number of breeding pairs was found to range between 90 and 740 between 1992 and 2024 (Fig. 10.2a).

When using the number of breeding pairs to estimate population size, we need to be aware that non-breeders are not taken into account. Reasons for non-breeding may be prospecting prior to recruiting, having divorced or having become a widow, but often they also simply are unknown, and potentially linked to individual condition. At the Banter See, between 1994 and 2024, i.e. omitting the first two years of the study, in which adult birds were first caught at the nest to be marked with transponders, I have

used the subset of transponder-marked birds registered at the colony to estimate the annual proportion of non-breeding birds. In total, the annual number of transponder-marked birds registered by the system ranged between 92 and 1257, of which the number of breeders ranged between 72 and 788. As such, the proportion of non-breeders in these years ranged from 0.22 to an astonishing 0.53, with an average of 0.38 (Fig. 2b). In future studies, I aim to split these data into categories and elucidate what factors underlie annual variation in the proportions for each of these categories.

10.4 Annual reproductive performance and adult survival

I will now reduce the dataset a little, to the period 1992–2021. I do so, because these are the years in which the population was not yet hit by a major outbreak of avian influenza (see below), and for which I have data available that can function as a proxy for food availability at the breeding grounds: the so-called ‘ICES North Sea herring recruitment index’. This index quantifies the total abundance of juvenile herring hatched in the previous winter in a standardised survey area, measured with a fine-meshed pelagic trawl net (ICES 2024). It was previously shown to be the measure of food availability best able to predict fledging success at the Banter See Common Tern colony (Dänhardt & Becker 2011), due to explaining a substantial amount of variation in offspring mortality (Vedder *et al.* 2019), which makes sense given that juvenile herring makes up the main share of the chick diet (38%, Cansse *et al.* 2024).

For estimating annual reproductive performance, I have three measures available: the total number of eggs laid, the number of chicks hatched from these eggs, and the number of these chicks that fledged, per breeding pair. The coefficient of variation in these traits, which describes

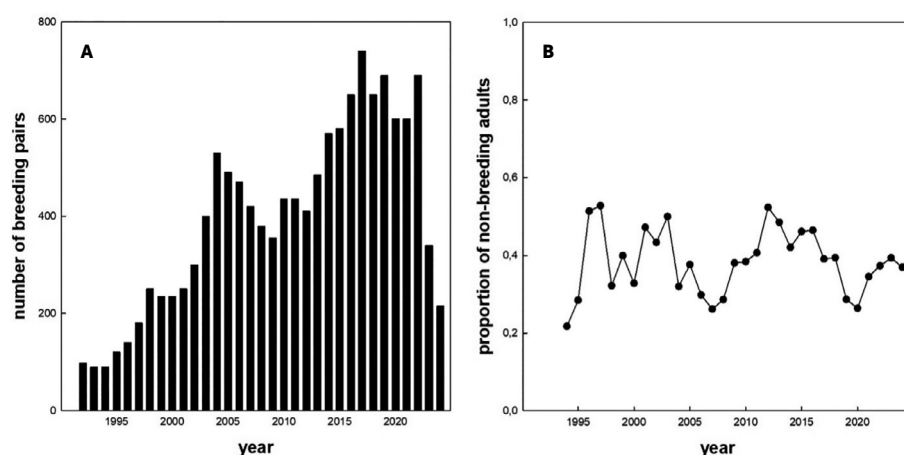
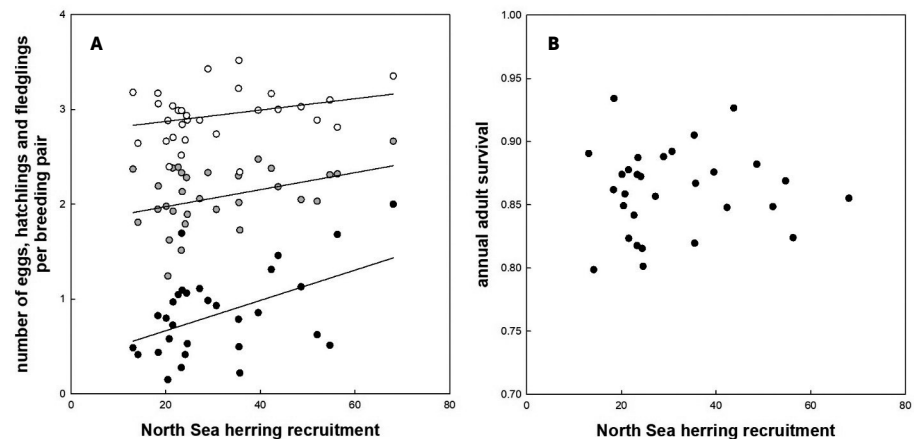


Figure 10.2. The number of breeding pairs (a) and proportion of non-breeding transponder-marked Banter See Common Terns registered by the automated antenna system (b) between 1992/1994 and 2024.

Figure 3. Annual reproductive performance (a) and survival (b) of Banter See Common Terns for the period 1992–2021 in relation to a proxy for food availability: the ICES North Sea herring recruitment index. This index quantifies the total abundance of juvenile herring hatched in the previous winter (here divided by 1,000,000) in a standardised survey area (ICES 2024). In (a), reproductive performance is quantified as the number of eggs, hatchlings and fledglings per breeding pair in white, grey and black circles, respectively.



the extent of variability in relation to the mean, increases as the breeding season progresses, being 10% for annual mean clutch size, 15% for brood size and 53% for the number of fledglings. In a first attempt to explain the annual variability for each trait, I used three simple linear models to test for trends over time by adding 'year' as a linear covariate. No trend emerged for any of the traits. When replacing 'year' for the 'ICES North Sea herring recruitment index', however, a correlation with food availability emerged (Fig. 10.3a), confirming previous findings using a smaller subset of the data (Vedder *et al.* 2019). In years in which there was more herring in the North Sea, there was a trend towards clutch size being slightly larger (0.006 ± 0.004 per unit of herring recruitment, $X^2_1 = 2.751$, $p = 0.097$). Looking at the number of chicks hatched from

these eggs, the correlation became stronger and statistically significant (0.009 ± 0.004 , $X^2_1 = 6.301$, $p = 0.012$), and at the level of the number of fledglings produced it was strongest and most significant (0.016 ± 0.005 , $X^2_1 = 9.821$, $p = 0.002$). Looking at the explanatory power of food availability, expressed as the R^2 -value, this ranged from 8% for clutch size, 17% for brood size and 25% for the number of fledglings. That means that large amounts of additional variation still need explaining. We know parental age is one factor doing so (Zhang *et al.* 2015), but other factors still need identification.

Between 1992 and 2021, survival of transponder-marked birds to the next year ranged between 80% and 93%. This annual variation did not show a trend over time, and was



Common Tern | Sterna pierregarin (*Sterna hirundo*)

Arnold Meijer / Blue Robin

not correlated with the level of food availability in the breeding season either (figure 3b). This makes sense, given that adult Common Terns are known to strategically reduce their broods when food is scarce, thereby minimizing the energy spent on reproduction and prioritizing their own survival (Vedder *et al.* 2017). It also means that variation in annual survival must be explained by other factors. Given that survival is known to linearly decrease with age in the Banter See Common Terns (Zhang *et al.* 2015; Vedder *et al.* 2021), the age structure of the population may be one factor, but others, such as environmental conditions at the wintering grounds are likely to play a role too.

10.5 Avian influenza

In 1961, a disease outbreak made Common Terns in South Africa delay their spring migration along the East Atlantic Flyway, and killed at least 1300 of them (Rowan 1962). Once isolated, the causal agent of this disease turned out to be an avian influenza virus, the first isolated in wild birds, and potentially coming from an outbreak in Scottish chickens 17 months earlier (Becker 1966). After that, the virus, however, was only very seldom found in terns (Olsen *et al.* 2006).

From 2016, infections by highly pathogenic avian influenza viruses of clade 2.3.4.4b of the H5 goose/Guangdong (gs/GD) lineage emerged globally and led to high levels of mortality in both poultry and wild birds, including nearly 300 Common Terns in South Africa in January 2018 (Roberts *et al.* 2023). From May 2022, an unprecedented situation then arose when several colony-breeding seabird species, including Common Terns, started showing mass mortality in their breeding areas (Pohlmann *et al.* 2024).

During the Banter See breeding season of 2022, the antenna system registered the first Common Tern to have

returned from the wintering grounds on the 11th of April and the first nest was found on the 4th of May. On the 29th of May, the team found a dead adult bird. It turned out to be a 3-year old male that had arrived only two weeks before, on the 15th of May. From then on, the number of dead birds found increased exponentially, and tests confirmed the cause of death to be H5N1 (Pohlmann *et al.* 2024). On the 20th of July, the team found the last dead bird, which brought the total number of dead birds found to 511 (Fig. 10.4a). During the breeding season of 2023, first arrival occurred on the 14th of April, and the first nest was again found on the 14th of May. On the 21st of June, the team found the first dead bird, a 7-year old male that had arrived on the 27th of April and had been breeding since the 17th of May. Tests confirmed that avian influenza had returned to the colony. As in 2022, the mortality rate increased exponentially before tapering off after nearly a month. On the 28th of July, the team found the last dead bird, bringing the total number of dead birds found to 111 (Fig. 10.4b).

The demographic consequences of these two catastrophic events can be estimated in different ways. During the 2022 breeding season, 26% of the registered transponder-marked birds was found dead. In 2023, however, the number of arrived individuals, visiting the colony at least once, was down by 42%, and the number of breeding pairs by 51%, suggesting that more mortality had taken place outside of the breeding area (such that we did not retrieve the birds), during migration, or at the wintering grounds. Similarly, during the 2023 breeding season, 9% of registered transponder-marked birds was found dead, whereas the number of registered individuals was down by another 33% in 2024, and the number of breeding pairs by another 37%. Overall, the two seasons of high adult mortality led to a 69% decline in breeding pairs (Fig. 10.2a) and a 61% decline in the number of registered birds (Fig. 10.4c).

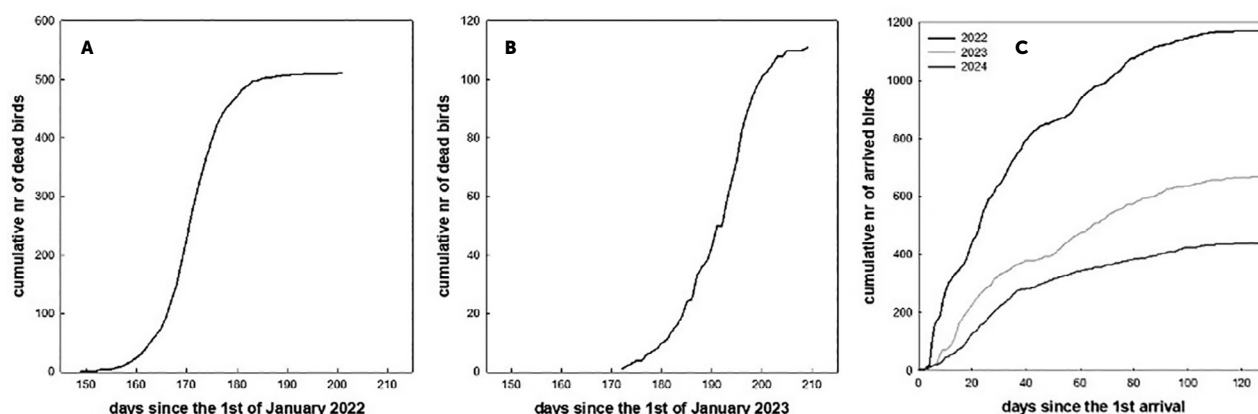


Figure 4. The number of Common Terns found to have died from avian influenza at the Banter See colony site increased exponentially in the breeding seasons of both 2022 (a) and 2023 (b). The effect on the total number of transponder-marked birds registered to have arrived (c) shows the impact on the population (breeders and non-breeders included). This impact is similar to that observed in the number of breeding pairs presented in figure 2a.

In terms of reproductive performance, the fledging success values of 2022 and 2023 were the lowest across the entire long-term study, with 9 and 12% of hatchlings fledging, respectively. Luckily, 2024 came without further outbreaks of avian influenza, and, at 87%, we observed the highest fledging success since the onset of the long-term study. As such, some hope for population recovery may remain.

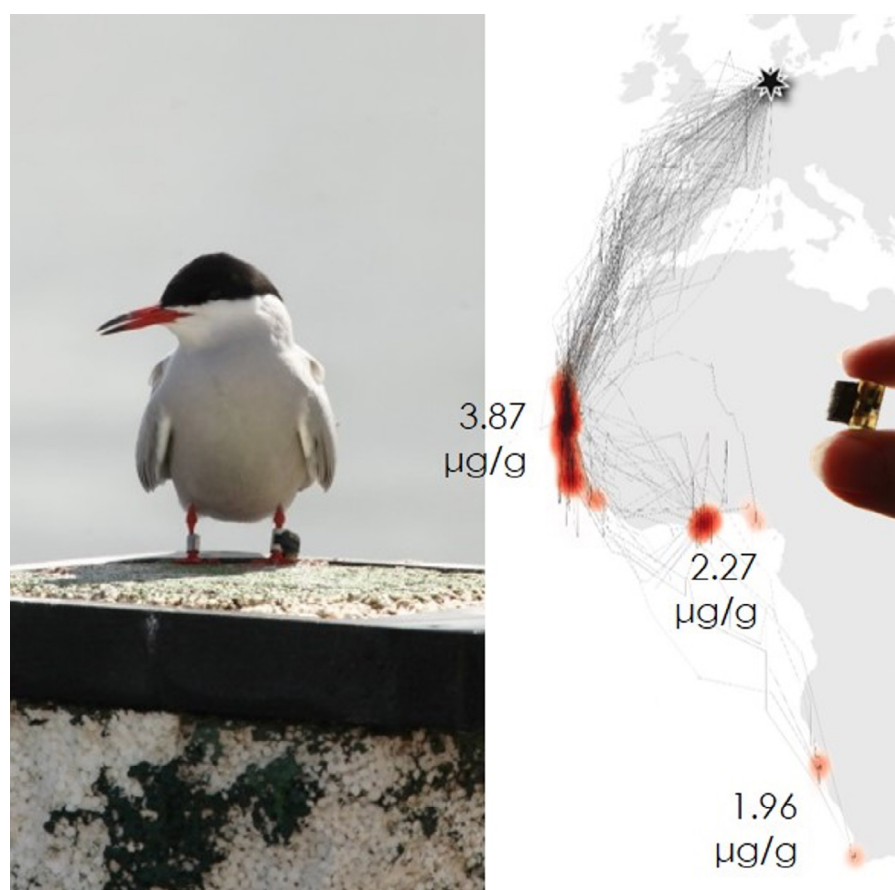
10.6 Extending our view beyond the breeding season

Detailed work at the colony site of the Banter See Common Terns has enabled me to present annual estimates of population size, local reproductive performance and survival, as well as to understand how reproductive performance and adult survival at the breeding grounds are affected by fish availability and a catastrophic two-year outbreak of avian influenza. Further variation in these vital rates, beyond that underpinned by the age structure of the population, has, however, proven difficult to explain. Extending our view beyond the breeding season might prove crucial to do so, as it may enable us to assess carry-over effects of conditions encountered on migratory journeys and at the wintering grounds used by the birds.

Hereto, we have made use of technological advances miniaturising tracking devices and, after a successful pilot experiment (Kürten *et al.* 2019), started tracking part of the Common Terns breeding at the Banter See with light-level geolocators (Fig. 10.5). These devices record the intensity of ambient light at regular intervals, logging the light levels together with information on the date and the time. After retrieving the devices from the birds upon their return to the breeding colony after migration, the extracted data can be used to determine the time of sunrise and sunset and to compare these to those at known geographic locations to estimate the birds' latitude and longitude. Light-level geolocators are lightweight and power-efficient, such that they can be used on small birds, but have as a disadvantage that the location estimates obtained are not as precise as those obtained by GPS tracking, and require recapture of the bird to retrieve the data, such that only successful migratory journeys can be tracked.

Since 2016, my team has tracked 284 migratory journeys performed by 140 individual Common Terns of known sex and age (see Fig. 10.5 for a subset of these journeys). So far, we have learned that all birds use the East Atlantic Flyway, and that 76% of birds wintered along the coast of West Africa, but that 19% wintered further south,

Figure 5. A Common Tern carrying a light-level geocator, and a map showing the migration routes (black lines) and main wintering areas (heat-maps of twilight positions) of Common Terns breeding at the Banter See (black star) and reporting the average mercury concentrations in feathers grown at the three main wintering grounds.



or even as far south as Namibia and South Africa (5%; Kürten *et al.* 2022). In addition, we have learned that birds are highly faithful to their individual wintering areas, returning there year after year (Kürten *et al.* 2022). Given that the timing of migration is rather consistent within individuals too (Kürten *et al.* 2022), this means that, at least in the case of the Banter See Common Terns, once a bird has been tracked once, we can assign it to a wintering area and study the conditions in this wintering area during the approximate time of its presence there, and assess whether these conditions explain variation in its survival or reproductive performance. To do so, in the light of variability, as well as in the light of ongoing climate change, is a main aim for the coming years.

An additional extension of the long-term individual-based study at the Banter See is that we collect feathers of tracked birds. When feathers grow, they are connected to the birds' blood vessels, and circulating pollutants, such as mercury, can bind to the keratin, thereby not only allowing the birds a form of detoxification, but also providing researchers with a tissue that can be collected without the need to sacrifice the bird and used to obtain insight in the levels of pollution the birds experienced in the area in which they moulted (e.g. Whitney & Cristol 2017). For the Banter See Common Terns, we collect feathers from the back of the neck, because these were moulted in the wintering area, to study levels of mercury pollution, because mercury is increasing in the environment, especially toxic in marine environments where it is methylated by bacteria, and magnifying across the food chain, therefore being likely to leave top predators, such as the Common Tern, especially vulnerable to its negative effects (Bertram *et al.* 2022). For 2017 – 2021, our dataset comprised 3 repeated measurements of 213 feather samples from 80 birds, of which 54, 23 and 3 individuals overwintered at the Canary, Guinea and Benguela Current, respectively. Mercury levels in the analysed feathers ranged between 0.77 and 9.87 $\mu\text{g g}^{-1}$, with a mean ($\pm$ standard error) of 3.4 ($\pm$ 0.11) $\mu\text{g g}^{-1}$. These concentrations varied with the wintering area of the birds: individuals wintering in the Canary Current had significantly higher mercury concentrations than those wintering in the Guinea Current, which in turn had significantly higher mercury concentrations than those wintering in the Benguela Current (figure 5). Notably, 17% of feather samples showed mercury levels that exceeded the admitted toxicity threshold of 5 $\mu\text{g g}^{-1}$, above which reproductive disorders have been reported for birds (Eisler, 1987). As such, further studies should determine whether mercury uptake during the non-breeding period affects survival, and/or interacts with mercury uptake at the breeding site and has carry-over effects on reproductive performance.

10.7 Outlook

Seabirds, as a group, have been found to suffer from many threats, both fisheries-related (bycatch and overfishing)

and coming from predation by invasive species, such as cats and rats (Phillips *et al.* 2023). Climate change, fastest in our northern hemisphere, leads to reduced reproductive performance across seabird species (Sydemann *et al.* 2021) and pollution levels were, and are, increasing and within seabirds, such as the Banter See Common Terns, reaching concentrations above the levels deemed to be safe (e.g. Bertram *et al.* 2022; Bertram *et al.* 2024). With catastrophic events, such as the recent outbreaks of avian influenza along the entire East Atlantic Flyway, a key question is whether seabird populations can recover in such a setting. Investing further in long-term demographic studies will be key to answer it.

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I am indebted to Peter H. Becker for setting up the long-term individual-based study at the Banter See, to Götz Wagenknecht for tirelessly providing the precision and continuity that a long-term study requires and to Timo Ubben for brilliant technical support. Special thanks also goes to the former and current members of my wonderful Banter See tern team, in alphabetical order: Justine Bertram, Coraline Bichet, Nathalie Kürten, Maria Moiron, Benno Rodemann and Oscar Vedder – working with you is a joy, even in times when the terns are in trouble. Last but not least, I want to thank the many postdocs, PhD students, volunteers, students and field assistants that have made up or supplemented the team since 1992 – your contributions still grow in importance with every season that gets added to the books.



Monitoring crew at Guinea Bissau (Peter de Boer)

11. The future of monitoring of coastal waterbird populations along the East Atlantic Flyway: an exploration of possibilities for improvements and innovation

Jeroen Reneerkens, Sjoerd Duijns, Allert Bijleveld & Marc van Roopen

Summary

This chapter evaluates the integrated monitoring program for waterbird populations along the East Atlantic Flyway. It reviews current methods—primarily annual and triennial surveys that estimate bird abundance and distribution—and identifies limitations such as uncertainties in population boundaries, observer capacity issues, and challenges in accurately assessing vital rates and environmental pressures. We advocate for enhanced approaches including remote sensing, drone-based counts, individual tracking, and integrated population models to improve data accuracy and better link environmental factors with population trends. A roadmap of implementation scenarios is presented, prioritising cost-effective innovations to future-proof the monitoring program and support targeted conservation actions in the face of global biodiversity challenges.

Résumé

Ce chapitre évalue le programme intégré de suivi des populations d'oiseaux d'eau le long de la voie de migration de l'Atlantique Est. Il examine les méthodes actuelles – principalement des relevés annuels et triennaux – visant à estimer l'abondance et la distribution, tout en identifiant des limites : incertitudes sur les frontières des populations, capacités des observateurs, et difficultés à estimer précisément les taux vitaux et pressions environnementales. Nous plaçons pour des approches améliorées, incluant la télédétection, les comptages par drone, le suivi individuel et les modèles de population intégrés, afin d'augmenter la précision des données et de mieux relier les tendances aux facteurs environnementaux. Une feuille de route propose des scénarios de mise en œuvre, avec des innovations à coût maîtrisé pour rendre le suivi plus efficace et orienter les actions de conservation face aux défis globaux de la biodiversité.

11.1 Introduction

Assessment reports of waterbird populations along the East Atlantic Flyway (EAF) typically include estimates of population sizes and trends. These estimates are based on the result of hard work and sustained efforts by many field workers that engage in waterbird counts along the entire coast of the East Atlantic Flyway. In addition to estimating the size and distribution of flyway populations and to detecting changes herein ('abundance monitoring'), the goals of the integrated monitoring program along the East Atlantic Flyway (Fig. 11.1, van Roopen *et al.* 2013) are to gain insight into the causes that explain these changes through monitoring of reproduction and survival ('vital rate monitoring') and to assess pressures at sites along the flyway ('environmental monitoring'; van Roopen *et al.* 2013). At present mainly the abundance monitoring (chapter 2) and the environmental monitoring (chapter 3) have been implemented during the cooperation between the Wadden Sea Flyway Initiative, Wetlands International and BirdLife International to improve the monitoring of coastal waterbirds along the East Atlantic Flyway. The results inform stakeholders about flyway populations at risk, the presence of human pressures at sites and the potential effect of management measures. The results can also be used to target important sites for increased conservation. In addition to addressing the unimplemented aspects of the monitoring program, assessing other innovations that could enhance abundance, and environmental monitoring will help keep the program future-proof. It is therefore important to ensure that all three aspects of the integrated monitoring program are conducted as reliably and comprehensively as possible. In this chapter we describe current limitations and possible solutions to improve the international waterbird monitoring programme, based on a EU funded project with that purpose (Reneerkens *et al.* 2024, Hofmann *et al.* 2024).

11.2 Current methodology

Abundance monitoring is implemented during annual surveys and extra triennial simultaneous survey campaigns at

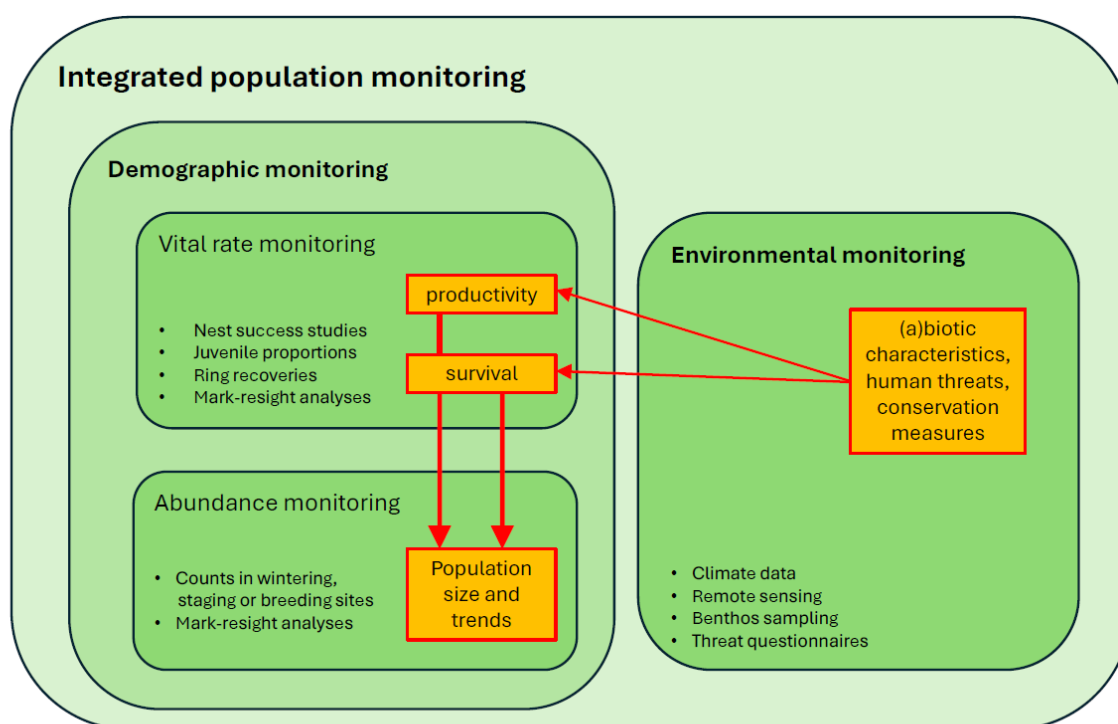


Figure 11.1. An integrated approach to monitoring populations is the goal of the ongoing monitoring of waterbirds along the coasts of the East Atlantic Flyway (van Roomen *et al.* 2013). This involves environmental monitoring and demographic monitoring (i.e. the monitoring of bird numbers and of their vital rates). This approach is considered to be the best set of tools to inform conservationists, policy makers and site managers on how to conserve bird populations. All aspects of this approach can be improved and/or extended in the current program.

(almost) all relevant wetlands along the flyway. All waterbirds are being counted from the ground or from boats, and in NW Europe some large wetlands are also counted using low-flying airplanes). The counts are mostly performed by experienced observers (volunteers and professionals) and usually organised in January, when there is generally little migratory movement of birds. Large colonial-breeding waterbirds are (also) counted during the breeding season (e.g. Veen *et al.*, chapter 9, Koffijberg *et al.* 2022). These efforts yield reliable estimates of the total number of birds per population and changes over time therein, as well as (changes in) their geographical distribution. This is the core of the current coastal waterbird monitoring along the East Atlantic Flyway and the updated results are published in East Atlantic Flyway assessment reports (e.g., van Roomen *et al.* 2022, current report).

The estimation of *vital rates* (reproduction and survival) is currently hardly implemented in the monitoring programme, while the relative impacts of vital rates and/or spatiotemporal variation therein can provide essential information about possible causes of changes in population size. Several research groups have however set up studies in which vital rates are estimated for several species (e.g. Rakhimberdiev *et al.* 2015, Lok *et al.* 2015, 2017, Reneerkens

et al. 2020). Annual survival is usually estimated using mark-recapture analyses based on observational data of individually colour-marked birds (Sandercock 2003, Reneerkens 2022). Reproductive success can be measured at different stages (incubation, chick rearing, post-fledging), through detailed observations at the breeding grounds or by assessing proportions of juveniles at migration and wintering sites (Reneerkens 2022). These different metrics of reproductive success provide different information and ideally, they should be all measured in concert, leading to essential insights into the number of offspring produced per female that initiates a breeding attempt.

Environmental variables, including human pressures, are currently being monitored by questionnaires filled in by local waterbird counters and site managers, about the state of the habitat and the presence or absence of human pressures at the sites that they know (Crowe *et al.* 2022; chapter 3). Knowledge of the impacts of pressures on vital rates should help to explain the (changes in) numbers of coastal waterbirds (Fig. 11.1). Such insights into which environmental pressures are primarily responsible for changes in waterbird populations would greatly help policy makers and conservationists to act correspondingly.

11.3 Limitations of the current methodology

Counting waterbirds along the entire flyway and for a long period is a true challenge and, inevitably, the current abundance monitoring can still be improved. To start with, there is often uncertainty about the geographical boundaries of flyway populations and the degree of co-occurrence of subspecies that cannot be distinguished in the field. This has obvious consequences for the determination of population sizes and trends. This problem may possibly increase if changes in the distribution of birds lead to mixing of populations previously assumed distinct populations (Schekkerman *et al.* 2022, Bom *et al.* 2024, Conklin *et al.* 2024).

The abundance monitoring by field observers counting the number of birds at sites is also subject to inaccuracy, due to several natural and logistical factors. Incomplete coverage is one such cause of inaccuracy. Particularly for the African part of the flyway, observer capacity and availability of equipment as well as continuity of experienced observers continues to be a challenge. Locally, reliability of the counts is an issue. Counting waterbirds is particularly challenging at sites where they are spread across vast areas with numerous small and large islands, requiring multi-day planning, and at sites with mangrove where birds must be counted on their feeding grounds, sometimes necessitating sampling and extrapolation methods, and when birds occur in large mixed-species flocks.

Estimation of vital rates estimates is currently only done for a few species, and usually not systematically embedded in the flyway monitoring scheme. Systematic monitoring of population-level reproductive output is currently

only performed for geese and swans, based on observations of the number of juveniles within families (e.g. Jensen *et al.* 2023). The current method of monitoring environmental pressures via questionnaires is suboptimal due to their qualitative and subjective nature as well as limits and biases in the detectability of various pressures.

11.4 Possible solutions

The rapid development of monitoring techniques, satellite imagery, statistical analyses, computational power, and machine learning could significantly improve current monitoring accuracy (Lahoz-Monfort and Magrath 2021). Individual tracking, colour-ringing, and genetic and isotopic methods can help define flyway boundaries and connectivity. Abundance monitoring could benefit from power analyses to determine the optimal frequency and locations of surveys for detecting statistically significant trends (Godeau *et al.* 2024). Remote sensing, particularly using drones, has proven effective for counting colonial waterbirds and birds at high tide roosts (Francis *et al.* 2020, Kellenberger *et al.* 2021, Marchowski 2021, Rasmussen *et al.* 2023), though challenges like disturbance, species recognition, and limited coverage remain. Potentially, drones can be used to count birds during low tide, which will improve accuracy of counts at sites where high tide roosts are inaccessible or invisible. Observations of colour-marked birds can also aid in estimating numbers at the local scale or even that of the flyway (Spaans *et al.* 2011, Kentie *et al.* 2016).

Survival probabilities can be measured best through colour-ringing projects. They are preferably long-term (>10-15 yrs) and large-scale to estimate survival at a repre-

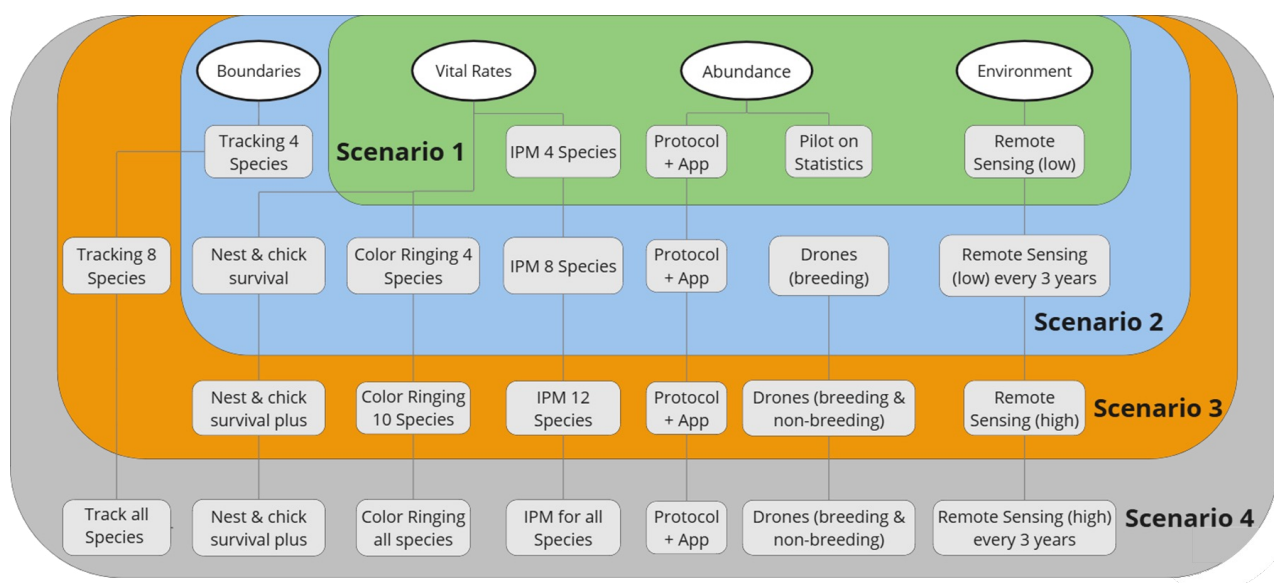


Figure 11.2. Four scenarios to improve waterbird monitoring along the East Atlantic Flyway by implementing innovative and integrative techniques (from Hofmann *et al.* 2024).



Memo Horrmann

Counts at low tide are difficult



Hans Schiekerman

White-breasted Cormorant | Cormoran à poitrine blanche (*Phalacrocorax lucidus*)

sentative number of sites in concert with abundance monitoring. Monitoring the percentage of juvenile birds in populations based on counts and catches offers insights into reproductive rates but should be performed at sufficient spatiotemporal scale to correct for heterogeneity in the data. Integrative population models can be used to estimate vital rates not directly measured in the field, but the accuracy depends on that of other variables such as bird abundance. Molecular methods can enhance age classification and allow for more detailed demographic analysis, improving population dynamics estimates.

Environmental monitoring using satellite imagery can collect quantitative data across the flyway, measuring variables such as mudflat extent, water bodies, human constructions, and snowmelt timing in Arctic breeding areas (Kersten *et al.* 2023). Tracking individual birds with transmitters or loggers helps assess habitat quality and behavioural changes but requires substantial investment to integrate into flyway monitoring. All metrics could provide early warnings (Stolze and Lisovski 2023) of serious population declines and the potential causes thereof. Linking environmental variables to bird abundance remains however complex due to time lags, carry-over effects, and compensatory mechanisms.

11.5 Recommended innovations

Developing management or conservation actions requires precise knowledge of how many individuals exist in each waterbird population, where they occur along the flyway, where, when and why they survive and reproduce, which are the goals of the integrated waterbird monitoring along the East Atlantic Flyway. Several improvements that are identified with a high impact and urgency by Reneerkens *et al.* (2024) are:

(1) Knowledge of birds' biogeographic origins forms the basis for estimating population sizes and trends. Currently, the designation of birds to certain populations often relies on (reasonable) assumptions and potentially biased ringing data, rather than precise data. A combination of **tracking** (Henriques *et al.* 2024) and **molecular methods** (e.g., Conklin *et al.* 2022) could considerably improve this information. Monitoring individuals at representative sites across breeding and non-breeding ranges would clarify which biogeographic populations exist, their seasonal locations, and their levels of mixing and distinctiveness in the field.

The current abundance monitoring can benefit from **power analyses, re-sampling techniques and/or simulations** (Godeau *et al.* 2024) to explore how the current monitoring scheme can become more reliable with adjustments in frequency and site selection. Additionally, the incorporation of new methods to correct for missing values in the dataset will make the outcomes of waterbird abundance monitoring more efficient, powerful and reliable. Accessibility challenges and counting errors in large flocks could be minimised using **high-flying drones with high-definition**

cameras (Rasmussen *et al.* 2023), reducing disturbance. Machine learning could help identify and count birds in recorded footage efficiently. Ground-based counts and drone results can be cross-validated for reliability. Better **instructions, training of observers** and the development of simple and robust monitoring protocols could increase the long-term commitment and reliability of counts in areas where there is no long-standing history in counting birds. Data could be stored and quickly shared via phone apps, checked for accuracy by admins and by automated tools spotting anomalies. Additionally, **densities of individually marked birds** could be used to estimate numbers for some bird populations.

(2) There is an urgent need to integrate into the current monitoring scheme the long-term **monitoring of vital rates** of at least a selection of key species that represent specific habitats and life histories. Estimates of **seasonal or annual survival** and **population-level annual reproductive output** can be measured for these species. Annual and seasonal survival rates could be estimated for a representative sample of populations using mark-resighting analyses of individually color-ringed birds (Reneerkens 2022; Sandercock 2006) or tracking and used in **population models** to indicate where along the flyway, and when in a life cycle, population growth is limited. This requires capturing and monitoring birds across the entire flyway over time. Productivity could be measured through frequent counts of age groups or family sizes at representative sites or estimated using integrated population models combining survival and abundance data (Besbeas *et al.* 2002). **Integrated populations models** combine count data, such as those collected in the waterbird monitoring along the EAF, with demographic data into a statistical analysis. A large advantage is that it allows the estimation of more demographic variables, results in more precise estimates of parameters enhancing the statistical power compared to separate analyses and adequately includes all sources of uncertainty (Abadi *et al.* 2010, Schaub & Abadi 2011, Schaub & Kéry 2012).

(3) The final step would involve uncovering the ecological factors driving changes in vital rates that lead to population change. For declining populations, this also implies assessing whether conservation measures effectively address human and environmental stressors. Insights into the spatial occurrence of various (potential) stressors could come from **remote sensing using satellite imagery** (Kersten *et al.*, 2023) and targeted ecological studies could quantify the effect of the stressors on birds' vital rates.

It is highly recommended to use old and new methods simultaneously for some period to enable evaluation and validation of both methods and the extent to which they affect the outcomes. Some techniques can already be implemented easily (e.g. remote sensing of habitat, estimates of vital rates, counts of colony-breeding birds by use of drones), while other techniques require pilot studies to obtain the necessary knowledge and gain experience of how to reliably adopt and implement them into the current monitoring scheme. Pilot studies on the suitability of using



Harvey van Diek

Caspian Tern | Sterne caspienne (*Hydroprogne caspia*)

drones to count waterbirds at high tide roosts and to estimate coastal waterbird densities at low tide will be useful. The suitability of individual tracking of birds in combination with measurements of habitat quality as a monitoring tool on the flyway scale needs to be further explored. Ultimately, to understand population trends and define options for management measures, a full integration of methods is essential. Hence, the combined use of bird counts, estimates of vital rates, individual tracking of birds and site and habitat use and quality, resulting in species-specific integrated population models will be very powerful.

11.6 A roadmap towards implementation

A full implementation of all of the proposed techniques is limited by financial constraints. Many areas along the flyway struggle to fund even basic annual counts or maintain skilled teams. The techniques that can help to improve aspects of this integrated monitoring differ in their costs and benefits and their potential impact on improving the current waterbird monitoring. Ultimately, choices about whether, and which, techniques should be developed and/or adopted are dependent on information needs and resources available. However, all the methods described above are feasible and have been successfully applied to specific populations or sites, and many of the methods are still being developed and are becoming cheaper too.

To work towards implementation of solutions and innovations several scenarios have been developed to realistically enhance migratory waterbird monitoring along the EAF (Hofmann *et al.* 2024), prioritising innovative techniques based on their potential impact on abundance monitoring, vital rates, environmental monitoring, and flyway boundary identification. Techniques were ranked by implementation speed, cost, impact, and urgency (Reneerkens *et al.* 2024).

The roadmap outlines four implementation scenarios (Fig. 11.2). Scenario 1 focuses on simple enhancements to the current monitoring, significantly improving data quality in abundance and environmental monitoring, supported by integrated population models to better understand population changes. Scenario 2 introduces advanced techniques like tracking, drone-based counts, productivity measurements, and color-marking. These methods build on ongoing research and require integration into existing monitoring programs, with key decisions on cooperation, financing, data usage, and reporting. Scenarios 3 and 4 expand activities from Scenario 2 to include more species and sites, necessitating higher financial investments.

Waterbird populations were prioritised based on the associated potential to improve abundance monitoring and understand population dynamics. Only species present in significant numbers in the Wadden Sea at some time

in the year were considered. Assessments included reliability of flyway boundaries and the feasibility of tracking, color-ringing, or measuring annual reproduction at the population level. Additionally, it was evaluated whether integrated population models (IPMs) are currently viable or could be developed within five years with targeted investments in vital rate measurements.

The current global crisis in the state of nature, coupled with (inter)national obligations to conserve natural values, highlights the critical importance of reliable waterbird monitoring schemes. Diagnosing the causes of troubling population trends has become increasingly relevant. The financial investments required to significantly enhance integrated monitoring are relatively small by comparison.



Harvey van Diek

Dunlin | Bécasseau variable (*Calidris alpina*)



Mixed group of terns at the Banc d' Arguin, Mauritania (Hans Schekkerman)

12. East Atlantic Flyway Assessment 2023: conclusions, analytical considerations and recommendations

The present study provides updated flyway trends for 88 waterbird populations representing the majority of species and populations for which coastal sites of the Atlantic seaboard of Europe and Africa are of international importance during some part of their annual cycle. The coastal East Atlantic Flyway (EAF) is also important for other waterbird populations, representing species that make more use of inland freshwater or grassland habitats or the high seas or belonging to other flyways but for which the coastal EAF is not their core range and have therefore not been considered in this study. Together, the 88 populations represent a large variety of arctic, temperate and tropical breeding birds, resident, short distance and long distance migrants, different habitat and food specialists and taxonomic waterbird families. This variety makes these populations good indicators of the extent and quality of habitats and sites of this coastal EAF. They also represent important intrinsic natural values, targeted by local, national and international conservation policies. Besides the monitoring of waterbird abundance, this study provides also data about the presence of pressures and conservation measures at site level along the coastal EAF, based on filled in questionnaires at 119 sites.

12.1 Conclusions

The conclusions given below are based on the waterbird monitoring (Chapter 2 and Annex 1) and environmental monitoring (chapter 3) as well as on the other chapters with information on specific sites and regions (chapter 4 – 8) or thematic subjects (chapter 9 – 11).

East Atlantic Flyway

- The overall pattern of population trends is comparable to that in previous assessments, but the higher number of species with short-term declines raises concern. While 42% of all populations considered showed moderate or strong long-term increases and 25% stable or uncertain long-term trends, recent short-term trends indicated worsening conditions, with a higher proportion of declining populations, from 29% showing a decline and 4% a strong decline in 2020 to 35% showing a decline in 2023 and 6% a strong decline now (chapter 2).
- Shorebirds exhibited the most pronounced declines, particularly migratory Palearctic-breeding populations wintering in Africa, while geese, flamingos, pelicans, and cormorants showed more positive trends. The

declines in these migratory shorebirds in Africa mirror declines of shorebirds breeding in the Arctic and boreal regions, as they are largely the same populations. We can only speculate about the potential causes of these unfavourable trends which can be related to breeding grounds, stop-over sites or wintering region or combinations of these (chapter 2).

- In addition to abundance monitoring, we present the results of monitoring assessments of pressures and conservation measures at sites used by the study populations. Fishing was determined as a significant pressure at most sites across the flyway. Buildings, recreation/tourism, roads and hunting were also recorded as significant pressures. Climate change was also named frequently as it can disrupt or modify the migration cycles of waterbirds and cause the loss of critical habitats. Highly Pathogenic Avian Influenza (HPAI) impacted several waterbird species in Europe and West Africa between 2021 and 2023. From the assessment of conservation measures it can be concluded that more designated sites and better implementation of protective measures at already designated sites are needed. Effective people engagement is vital for achieving success (chapter 3).

Regions and sites

- **Greenland Arctic:** In contrast to reported decreases in the Russian Arctic (Soloviev *et al.* 2022), most waterbird species breeding in Greenland and wintering in the EAF seem to be in good health. This is especially true for geese, but also ducks and loons appear to be doing well. Waders appear to show only minor changes at the single location where they are monitored, at Zackenberg in NE Greenland. Here, Common Ringed Plover *Charadrius hiaticula*, Red Knot *Calidris canutus* and Sanderling *C. alba* have shown stable trends in territory numbers since 1996, while Dunlin *C. alpina* has increased and Ruddy Turnstone *Arenaria interpres* declined. We do however not yet know to what extent the data from Zackenberg are representative for larger areas or high Arctic Greenland in general. Despite stable or increasing numbers of birds, major changes in environmental conditions have become evident, with 2018 having the latest snowmelt on record and increasing summer temperatures (chapter 4).
- **Wadden Sea:** Many non-breeding waterbirds showed more favourable short-term trends in the Wadden Sea

than in the flyway. Compared to the 1980s, total numbers of non-breeding waterbirds estimated to be present at one moment in the Wadden Sea at some point during their annual cycle have not changed much. However, there has been a shift among species groups and between species, with waders in particular becoming less numerous, especially in the German Wadden Sea, and herbivores, such as geese and ducks, more abundant. Given that the amount of migratory turnover of individuals is generally larger among waders than among waterfowl, it is possible that the total number of birds that use this area in the course of the year is lower nowadays than the 10-12 million estimated 40 years ago (chapter 5).

- **Portugal, Tagus Estuary:** Achieving effective conservation of migratory waterbirds requires international cooperation between countries with respect to their protected sites. A recent example is the threat to the Tagus Estuary, the most important wetland for wintering waterbirds in Portugal. A license for the establishment of a commercial airport in the heart of this wetland raised concern among the national and international ornithological communities. The plans were not in compliance with the European Union's Habitats (Article 6) and Birds Directives (Article 4), as this site is listed as a Natura 2000 site and the level of noise disturbance produced by aircrafts overflying the protected area were expected to have a significant effect on the conservation objective for migratory birds that

use the area, and affect the integrity of the site. International advocacy efforts resulted in the plans to be put on hold, suggesting that this case could be an important example for other sites where birds come under pressure (chapter 6).

- **Coastal Ghana:** The coastal region of Ghana is of international importance for many waterbirds, and large-scale monitoring has been conducted since 1986. Overall, waterbird numbers and numbers of species occurring in internationally important Ramsar sites have declined. These losses have largely been attributed to wetland destruction from urbanisation, industrial expansion, coastal erosion, pollution, and siltation, reducing the extent and quality of feeding, breeding, and roosting habitats. Climate change will likely worsen these impacts. A stronger protection, continued monitoring, and in-depth studies to improve understanding of these dynamics are all needed (chapter 7).
- **Berg River, South Africa:** The Berg River estuary on South Africa's west coast is known for its biodiversity, productive fish and invertebrate fisheries and large numbers of migratory waders, and was designated as a Ramsar site in 2022. The estuary supports the largest remaining salt marshes in South Africa, which play a significant role in carbon sequestration and climate resilience. However, its ecological integrity is threatened by reduced freshwater inflows, declining water quality, expanding development, human disturbance,



Dunlin | Bécasseau variable (*Calidris alpina*)

Hans Schekkerman



Hans Schekkerman

mixed group of *Calidris* wader species

habitat degradation and the impacts of climate change. In response, BirdLife South Africa is implementing biodiversity stewardship and estuarine habitat restoration initiatives to help safeguard this system, including restoration of salt marsh and reed marsh habitats, stabilising of riverbanks, removal of invasive plant species, restriction of harmful recreational activities, and community training and awareness initiatives (chapter 8).

Themes

- **Using data from breeding bird counts alongside January counts:** Between 1998 and 2023 breeding numbers of four gull and tern species have been studied in colonies along the coasts of West Africa: Grey-headed Gull *Chroicocephalus cirrocephalus*, Slender-billed Gull *C. genei*, Caspian Tern *Sterna caspia* and West African Crested Tern *Thalasseus albidorsalis*. Historical counts covering all breeding sites indicated stable populations. The census carried out in 2023 revealed significantly lower numbers for most species across most sites, but methodological issues prevent firm conclusions on population changes. The apparent declines could be explained by counts conducted at inappropriate dates, colonies being not or partly included in a count, and/or mortality from an HPAI epidemic that particularly affected the W African Crested Tern populations in 2023. Implementing a standardised monitoring programme with region-

wide censuses at triannual intervals will be key to understanding developments of breeding populations in West Africa.

- **In depth studies of species along the flyway:** Understanding how specific threats influence vital rates is crucial to developing and implementing effective conservation strategies for waterbirds. A long-term individual-based study on Common Terns *Sterna hirundo* from a breeding colony located on the German Wadden Sea coast demonstrated the applicability of demographic monitoring in understanding potential threats and demographic processes at the scale of the flyway. The study colony has until recently been thriving, with an increase in the number of breeding pairs, although the rates of reproductive performance and survival were variable and showed no clear trend pattern. Food availability explained a substantial amount of variation in annual reproductive success, but not adult survival, a pattern expected when breeding birds prioritise their own survival over that of their chicks. In the breeding seasons of 2022 and 2023, the colony was severely hit by an outbreak of HPAI, resulting in a 61-69% population decline. If and how the population will recover needs to be closely monitored, especially given levels of mercury pollution at their main wintering grounds (situated predominantly off West Africa) that in some individuals exceed levels reported to lead to reproductive disorders in birds.

- **Overview of possibilities for innovating the flyway monitoring:** This chapter advocates the integration of new approaches into the current flyway monitoring, including remote sensing of environmental conditions, drone-based counts, individual tracking, and measuring vital rates to be used in integrated population models to improve data accuracy and better link population trends to environmental factors.

12.2 Analytical considerations

Depending on the species' distribution pattern, flyway population trends can best be calculated on the basis of simultaneous representative counts during the non-breeding period (in January for Palearctic species) or during the breeding period (often between March and May depending on the population). Recommendations have been made to standardise and harmonise these efforts (Hearn *et al.* 2018, Nagy *et al.* 2021). Decisions in the current report on whether flyway population trends are based on January or breeding bird data follow as much as possible these sources and are consistent with those for the previous EAF assessment (van Roomen *et al.* 2022).

Since the previous EAF trend assessment, no update of breeding abundance trends for European breeding populations has become available, because this is reliant on data from the European Union Article 12 Birds Directive reporting which takes place every six years with the most

recent one including data up to 2018. The consequence is that for several populations whose trends are based on breeding bird data, no update could be provided and the trends reported by Van Roomen *et al.* (2022) have been retained in the current report. By the next flyway assessment in 2026, new breeding bird data are expected to be available; in the future this suboptimal situation could be improved if such data, especially for colonial breeding species, could be updated on a triannual or even yearly basis.

Because of the absence of updated data from the Article 12 Birds Directive reporting, which also provides estimates of the size of breeding as well as wintering populations, this current report does not give updates of population sizes. Such updates are key for selecting sites of international importance using the 1% criterion. To ensure that they are as accurate as possible, it was decided to update population sizes only once every six years when all data are available, so the next update will be in the 2026 assessment report.

On a positive note, in this 2023 Assessment report, most European countries updated their January data until 2023 in time for these analyses, keeping data from Europe in line with the data from Africa. Substantial improvements in the completeness of the January data was achieved in the Iberia-North Africa region through submission of missing data from Spain (Molina 2023), while there were improve-



Counting Spoonbills

Lars Soerink



Arnold Meijer / Blue Robin

Slender-billed Gull | Goéland railleur (*Chroicocephalus genei*)

ments in North Africa through support of the Mediterranean Waterbird Network (Dami *et al.* 2023).

Another important factor required for robust flyway trends (and flyway population sizes) is knowledge of the flyway boundaries of populations, especially their distribution during January or their main breeding period. Current flyway boundaries are mostly based on older information including ringing recoveries and data on biometrics. However, many uncertainties exist, leading to large hypothetical overlaps between populations (see Critical Sites Network tool 2.0). In addition, non-breeding distributions may shift over time, in response to climate change or other factors (Chapter 2, Schekkerman *et al.* 2022). Genetic studies signal more mixing of wader populations than previously expected (e.g. Conklin *et al.* 2022, 2024). Whether this has been the case for a long time or is a recent development is as yet unclear but could perhaps be elucidated by genetic analyses of historic museum specimens. As shifts in the spatial distribution of populations affect estimates of flyway trends and population sizes, which in turn are used for prioritising site protection and species conservation under Ramsar, AEWA and the EU Birds Directive, changes in flyway boundaries also have significant policy implications. It is therefore of importance to extend research into migratory connectivity and flyway boundaries in the EAF, using modern methodology including individual tracking and molecular methods.

The choice of analytical framework for the analyses of flyway trends remains a challenge. We continued using the same methods as in the 2020 Assessment report and the AEWA Conservation Status Reports analyses by Wetlands International, in which (r)TRIM is the main tool used for trend calculation. Choices about site selection and different steps in the imputing process have also remained the same. Recent analyses based on datasets from the EAF show however that methodological choices are not trivial, and include sampling design as well as the statistical analysis (method of imputing, data distribution, spatial autocorrelation) (Godeau *et al.* 2024).

A new addition in the present report is the analysis of site trends using time series of January counts (see Annex 1 and Chapters 2 and 3). It shows how the overall flyway trend is composed of a combination of varying trends across the sites. Investigating this site variation, and what factors are influencing it, may offer us valuable insights in the broader flyway trends and in the question as to whether trends in specific areas are driven more by global (e.g. climate) or local factors. The current analysis brings about many questions, but it seems a useful tool to signal possible site-level impacts affecting flyway trends. Also, tallying across species the frequency of decreasing and increasing trends at sites could form a first attempt to signal sites suffering from significant local pressures or enjoying positive effects of management (see chapter 3). This could

become even more powerful if site trends based on multiple counts per year instead of January data could be used.

On the basis of the environmental monitoring assessments we can signal a broad range of pressures potentially impacting local wetlands (and thereby waterbirds), and list management measures taken or planned in the various countries (see Chapter 3). However, many important wetland sites are not yet covered by the questionnaires, especially many European sites, making it difficult to develop a full picture of the threats and their extent of impact across the EAF. National information from different years sometimes refers to different sites, and the data is often qualitative in nature and may be affected by differences in evaluation and interpretation by persons contributing the questionnaires. Besides improving the uptake of the IBA approach by observers at the counted sites, making more use of remote sensing is an essential step to improve environmental monitoring along the flyway (see chapter 11).

12.3 Recommendations

- **Continue and enhance the flyway monitoring programme.** With the current monitoring effort in the EAF we are able to regularly update flyway trends and distribution of waterbird populations, and contribute to the development of population estimates, as well as signal changes in pressures and conservation measures at sites. To maintain a strong level of cooperation

and information it is important to continue and increase the current level of coordination, further improve capacity building and enhance the financial sustainability of monitoring efforts.

- **Further enhance the capacity of countries to implement site and bird monitoring.** Despite increasing quality of the data collected, we are still not in a situation where these data are collected routinely in sufficient quality along the entire flyway. Nationally limited human capacity and resources substantially affect the potential to carry out the monitoring. Increased national capacity building efforts based on local and national gaps and priorities are needed.
- **Widen the scope from January counts at sites to counts during the autumn, spring and summer period as well.** For conservation and management it is important to identify important sites during migration, moult as well as dry/rainy seasons to better understand the importance of different sites during different stages of their annual cycle. As the optimal timing for these counts changes across the flyway this task needs to be addressed at national level. The scope for flyway wide or regional coordination and simultaneous efforts needs to be investigated.
- **Improve the collection and utility of breeding bird data besides non-breeding counts:** In addition to January counts of non-breeding birds, breeding bird monitoring is providing important information for flyway level assessments. Within Europe more species



Arnold Meijer / Blue Robin

Greater Flamingo | Flamant rose (*Phoenicopterus roseus*)



Arnold Meijer / Blue Robin

Common Shelduck | Tadorne de Belon (*Tadorna tadorna*)

should be monitored under the umbrella of the European Bird Census Council cooperation, especially colonial breeding and rare waterbirds, ensuring all member states harbouring these species contribute to overarching efforts. For the assessment it is important that trends are reported and can be analysed separately for different flyway populations. In Western Africa the triannual survey of colonial terns and gulls needs to continue and improve in quality.

- **Extend research into migratory connectivity and flyway boundaries.** For several waterbird populations our knowledge of their geographic flyway boundaries and the connectivity between breeding and non-breeding sites is limited, hampering the assessment of their status and thereby their conservation and management on a flyway scale. Moreover, as climate change and other pressures may induce distributional shifts, the geographical flyway delimitations themselves may be in need of monitoring. Nowadays, a suite of new technologies is available to aid in this, including tracking technology.
- **Improve the monitoring of environmental pressures and responses.** Further analyses are needed to separate local drivers of population changes from flyway wide developments and to link them to environmental conditions on the ground, ultimately leading to advice about local management measures. Through the current questionnaires on environmental conditions at

sites we are able to collect important information on local pressures and conservation measures. However, the information collected is not quantitative, often open to interpretation and sometimes difficult to assess by observers, which hampers the identification of drivers of population declines. It is recommended that quantitative data, e.g. from satellite remote sensing, will be considered in future assessments.

- **The flyway approach.** Finally, this East Atlantic Flyway assessment based on joint monitoring effort across the flyway underscores once more the need for stronger international collaboration and political recognition of the flyway approach. Migratory birds connect communities and countries across Africa and Europe, and their protection requires a coordinated, cross-sectoral approach. Long-term commitment from all flyway countries is required to reverse negative trends.



Audouin's Gull | Goéland d'Audouin (*Ichthyaeus audouinii*) (Arnold Meijer / Blue Robin)

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Harvey van Diek

Bar-tailed Godwit | Barge rousse (*Limosa lapponica*)



Bar-tailed Godwit | Barge rousse (*Limosa lapponica*) (Menno van Duijn /Agami)

Annex 1. Trends and distribution of waterbird populations using the coastal East Atlantic Flyway, update 2023

Marc van Roomen, Jeroen Reneerkens, Tom Langendoen, Khady Gueye Fall, & Erik van Winden

A1.1 Introduction

In this Annex the results of ongoing abundance monitoring of waterbirds using the coastal East Atlantic Flyway (EAF) are reported. It presents flyway population trends up to 2023 based on January (IWC) or breeding season counts, distribution in the non-breeding season (January) in 2019-23 and - new this time - site trends, based on January counts, where appropriate. It updates the results reported previously in Van Roomen *et al.* 2022. No formal updates of flyway population sizes have been carried out for this annex; this will be done again after the 'total' count of 2026, when the required updates on European population sizes are available as well. The population sizes reported in van Roomen *et al.* 2022 are therefore maintained.

A1.2 Study area, study populations, data sources, analyses and presentation

A1.2.1 Study area

The coastal EAF covers a large area spanning parts of the Arctic, western Europe and western and southwestern Africa. The arctic part of the EAF includes NE Canada, Greenland, Iceland, Svalbard, NW Russia and Siberia east to and including the Taimyr Peninsula. The countries bordering the Baltic Sea include Sweden, Finland, Estonia, Latvia, Lithuania, Poland and parts of Germany and Denmark. Southward from there the coastal EAF includes Ireland and the countries bordering the North Sea, and then follows the eastern shore of the Atlantic Ocean from France all the way to South Africa.

Within this report we focus on coastal sites of the EAF and 88 waterbird populations (sites depicted in red in Fig. A1.1). These are called the 'coastal EAF Flyway sites' (or main sites). These (main) sites can be large or small and are defined as representing areas functioning more or less as an ecological unit (whole estuary, lake or lagoon, mostly also including whole Ramsar, Natura 2000, or Important Bird Areas). Sites may encompass subsites and multiple counting units. In addition, for flyway population trends (and population sizes) we use also data from sites located further inland if they are within the January range of a fly-

way population. Fig. A1.1. shows these in blue as the sum of all sites across the study populations. Which of these sites are actually included in the flyway trend analyses varies between populations; selection criteria include the availability of at least five counts of the species of which at least one in the last 10 years and one before that period.

In analyses based on breeding bird data, the data used is defined by the overlap of the flyway population boundary, the breeding range of that particular flyway population and the availability of data in that range.

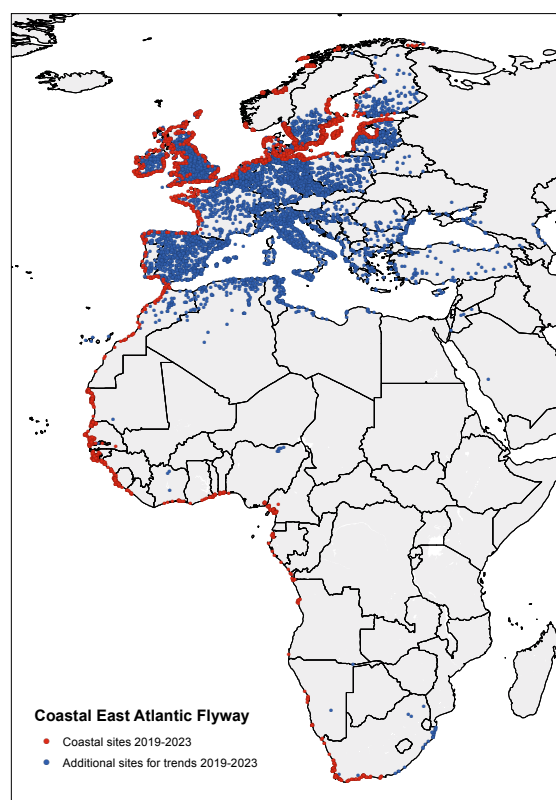


Figure A1.1. Sites from which January counts are included in the trend analysis. Sites within the coastal EAF are indicated as red dots, inland sites as blue dots. Sites having less than five counts or none in the most recent 10 years are not shown.

A1.2.2 Study populations

This study focuses on waterbird species from bird families defined as 'waterbirds' under the Ramsar convention. In some waterbird species the complete world population occurs within the boundaries of the coastal EAF. Often, and especially in migratory waterbird species, the global occurrence of a species is subdivided in different subspecies, flyway populations and/or biogeographic populations because of biological, ecological and practical management reasons (Scott & Rose 1996). These subdivisions are the focus of our study and are all called 'flyway populations' in this report. For species' names and subdivisions in flyway populations we follow Wetlands International (2024).

Based on the following criteria we selected waterbird species and flyway populations for coverage in this report:

- Flyway populations which occur predominantly or with important numbers within coastal EAF sites during January.
- Flyway populations which depend predominantly or for a significant part on coastal and estuarine food resources.
- Some flyway populations have been added to the initial selection because of their interest from a Wadden Sea context, despite a greater occurrence in inland sites as well.
- Only populations for which enough data (from January or breeding season counts) are available to calculate flyway trends were included.

In comparison to the 2020 update (van Roomen *et al.* 2022), Gadwall *Mareca strepera* was omitted in this report as this species is more dependent on non-coastal than coastal sites. Populations newly included in this annex are the *hrota* population of Brent Goose, *balsaci* population of Eurasian Spoonbill, two *schinzii* populations of Dunlin, and the Kittlitz's Plover and Lesser Crested Tern. The flyway populations for which trends were calculated are listed in Table A1.1.



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Audouin's Gull | **Goéland d'Audouin** (*Ichthyophaga audouinii*)

Table A1.1 Waterbird species and flyway populations for which flyway trends are analysed in this report. In a few cases, multiple flyway populations have been taken together as count data for the separate populations are not available or not reliable (indicated with **). Given are the species and flyway population names in English (Wetlands International 2024). The recommended method for flyway trend monitoring is also given (b=based on data during breeding, nb=based on data during non-breeding, mostly January) see Hearn *et al.* (2018) and Nagy *et al.* (2021). The data sources and methods used for the trends presented in this report are given as well (see text for further explanation). Also indicated are the extent to which populations use coastal EAF sites (% of their January totals in 2019-2023 in coastal EAF sites being the red ones in Fig. A1.1, * assessed for the combination of populations).

Species and population	Recommended project	Data origin	Trend method 2023	% in Coastal EAF sites
Brent Goose , Siberia/W Europe	nb-geese	IWC	Trim	97
Brent Goose , Canada & Greenland/Ireland	nb-geese	WEBS	TrendSpotter	95
Brent Goose , Svalbard/Denmark & UK	nb-geese	Aarhus	no data	95
Barnacle Goose , Siberia & NW Europe/NW Europe	nb-geese	EGMP	TrendSpotter	45
Barnacle Goose , East Greenland/NW Europe	nb-geese	WEBS	TrendSpotter	82
Barnacle Goose , Svalbard/NW Europe	nb-geese	WEBS	TrendSpotter	99
Greylag Goose , NW Europe/NW & SW Europe	nb-geese	EGMP	TrendSpotter	35
Greylag Goose , Iceland/NW Europe	nb-geese	WEBS	TrendSpotter	30
Common Eider , Baltic Sea, North Sea & Celtic Sea	nb-special	IWC	Trim	99
Common Goldeneye , NW & C Europe (winter)	nb-January	IWC	Trim	68
Red-breasted Merganser , NW & C Europe (winter)	nb-January	IWC	Trim	94
Common Shelduck , NW Europe (winter)	nb-January	IWC	Trim	95
Greater Scaup , Northern Europe/Western Europe	nb-January	IWC	Trim	88
Northern Shoveler , NW & C Europe (winter)	nb-January	IWC	Trim	46
Eurasian Wigeon , W Siberia & NE Europe/NW Europe	nb-January	IWC	Trim	61
Mallard , NW & E Europe & Siberia/NW Europe	nb-January	IWC	Trim	35
Northern Pintail , N Europe & Siberia/NW Europe	nb-January	IWC	Trim	87
Common Teal , NW & E Europe & Siberia/NW Europe	nb-January	IWC	Trim	51
Great Crested Grebe , <i>cristatus</i> , NW & W Europe	nb-January	IWC	Trim	30
**Horned Grebe , NW Europe (winter) combi of pop.	nb-special	IWC	Trim	80
Black-necked Grebe , Europe/S & W Europe & N Africa	nb-January	IWC	Trim	15
Greater Flamingo , W Africa	nb-January	IWC	Trim	99
Lesser Flamingo , W Africa	nb-January	IWC	Trim	100
Eurasian Spoonbill , W Europe/W Europe & W Med & W Africa	b-colony	Erlob	IUCN	81
Eurasian Spoonbill , <i>balsaci</i> , coastal W Africa (Mauritania)	b-colony	PNBA	TrendSpotter	100
Grey Heron , N & W Europe	nb-January	IWC	Trim	30
Little Egret , SW Europe	b-colony	PECBMS	Trim	52
Western Reef-egret , W Africa	nb-January	IWC	Trim	91
Great White Pelican , W Africa	nb-January	IWC	Trim	99
Great Cormorant , <i>sinensis</i> , N & C Europe	nb-January	IWC	Trim	38
Great Cormorant , <i>lucidus</i> , coastal W Africa	nb-January	IWC	Trim	100

Species and population	Recommended project	Data origin	Trend method 2023	% in Coastal EAF sites
Cape Cormorant , coastal S Africa	b-colony	IWC	Trim	100
African Oystercatcher , coastal S Africa	nb-January	IWC	Trim	98
Eurasian Oystercatcher , Europe/W & S Europe & NW Africa	nb-January	IWC	Trim	98
Pied Avocet , W Europe & NW Africa (breeding)	nb-January	IWC	Trim	97
Grey Plover , W Siberia/W Europe & W Africa	nb-January	IWC	Trim	88
Common Ringed Plover , <i>hiaticula</i> , NW Europe/ SW Europe & N-Africa	nb-January	IWC	Trim	89
Common Ringed Plover , <i>psammmodromus</i> , NE Canada to Iceland/W & S Africa	nb-January	IWC	Trim	90
Kittlitz's Plover , W Africa	nb-January	IWC	Trim	70
White-fronted Plover , <i>hesperius</i> , W Africa	nb-special	IWC	Trim	86
** White-fronted Plover , Gabon - South Africa, combi of pop.	nb-special	IWC	Trim	93
Kentish Plover , W Europe & W Mediterranean/W Africa	nb-January	IWC	Trim	54
Chestnut-banded Plover , Southern Africa	nb-January	IWC	Trim	95
** Whimbrel , East Atlantic (wintering), combi of pop.	b-dispersed	IWC	Trim	82
Eurasian Curlew , Europe/NW Europe, N & W Africa	b-special	Erlob	IUCN	88
Bar-tailed Godwit , <i>lapponica</i> , N Europe /W Europe	nb-January	IWC	Trim	99
Bar-tailed Godwit , <i>taymyrensis</i> , W Siberia /W & S Africa	nb-January	IWC	Trim	94
Ruddy Turnstone , NE Canada & Greenland/W Europe & NW Africa	nb-January	IWC	Trim	96
Ruddy Turnstone , N Europe/W Africa	nb-January	IWC	Trim	90
Red Knot , <i>islandica</i> , NE Canada & Greenland/W Europe	nb-January	IWC	Trim	98
Red Knot , <i>canutus</i> , W Siberia/W & S Africa	nb-January	IWC	Trim	98
Curlew Sandpiper , W Siberia /W Africa	nb-January	IWC	Trim	84
Sanderling , W Europe & W & S Africa (winter)	nb-January	IWC	Trim	91
Dunlin , <i>alpina</i> , NE Europe & NW Siberia /W Europe & NW Africa	nb-January	IWC	Trim	81
Dunlin , <i>schinzii</i> , Iceland /NW & W Africa	nb-January	IWC	Trim	100
Dunlin , <i>schinzii</i> , Baltic/SW Europe & NW Africa	b-special	PECBMS	Trim	95
Dunlin , <i>schinzii</i> , Brittain & Ireland/SW Europe & NW Africa	b-special	PECBMS	Trim	95
** Purple Sandpiper , NW Europe (winter), combi of populations	nb-January	IWC	Trim	96
Little Stint , N Europe & NW Siberia/N & W Africa	nb-January	IWC	Trim	44
Common Sandpiper , W & C Europe/W Africa	nb-January	IWC	Trim	54
Spotted Redshank , N Europe /SW Europe, N & W Africa	nb-January	IWC	Trim	35
Common Greenshank , N Europe/ W & SW Europe, NW & W Africa	nb-January	IWC	Trim	76
Common Redshank , <i>robusta</i> , Iceland & Faroes /W Europe	b-dispersed	IWC	Trim	94
Common Redshank , <i>totanus</i> , Britain, Ireland/Britain, Ireland & France	b-dispersed	PECBMS	Trim	95
Common Redshank , <i>totanus</i> , N Europe (breeding)	b-dispersed	PECBMS	Trim	99
Common Redshank , <i>totanus</i> , Western, Central & East Europe (breeding)	b-dispersed	PECBMS	Trim	85
African Skimmer , W & C Africa	b-colony	IWC	Trim	100
Slender-billed Gull , W Africa	nb-January	IWC	Trim	96

Species and population	Recommended project	Data origin	Trend method 2023	% in Coastal EAF sites
Black-headed Gull , W Europe/W Europe, W Med - W Africa	nb-January	IWC	Trim	24
Hartlaub's Gull , coastal SW Africa	nb-January	IWC	Trim	81
Grey-headed Gull , W Africa	nb-January	IWC	Trim	92
Audouin's Gull , Mediterranean/N & W Africa	b-colony	Erlob	IUCN	81
Mew Gull , NW & C Europe /NW Europe & W Med.	b-colony	Erlob	IUCN	54
Lesser Black-backed Gull , <i>graellsii</i> , NW Europe /East Atlantic	b-colony	Erlob	IUCN	48*
Lesser Black-backed Gull , <i>intermedius</i> , W Europe /East Atlantic	b-colony	Erlob	IUCN	48*
European Herring Gull , <i>argenteus</i> , NW Europe/East Atlantic	b-colony	Erlob	IUCN	75*
European Herring Gull , <i>argentatus</i> , W Europe /East Atlantic	b-colony	Erlob	IUCN	75*
Little Tern , Europe north of Mediterranean /East Atlantic	b-colony	Erlob	IUCN	93*
Little Tern , West Mediterranean/ East Atlantic	b-colony	Erlob	IUCN	93*
Damara Tern , Namibia & South Africa	b-special	IWC	Trim	100
Gull-billed Tern , W Europe/W Africa	b-colony	Erlob	IUCN	68
Caspian Tern , coastal W Africa	nb-January	IWC	Trim	81
Roseate Tern , W Europe/East Atlantic	b-colony	Erlob	IUCN	100
Common Tern , N & E Europe /East Atlantic	b-colony	Erlob	IUCN	96*
Common Tern , S & W Europe/East Atlantic	b-colony	Erlob	IUCN	96*
Lesser Crested Tern , <i>emigratus</i> , S Mediterranean/NW & W Africa	b-colony	IWC	Trim	99
Sandwich Tern , W Europe /W Africa	b-colony	Erlob	IUCN	88
West African Crested Tern , W Africa	b-colony	IWC	Trim	94
Greater Crested Tern , S Africa	b-colony	IWC	Trim	95



Hans Schekkerman

Little Stint | Bécasseau minute (*Calidris minuta*)

A1.2.3 Data sources

For the current assessment of trends, a mixture of January count data and breeding bird data is used (see table A1.1). The majority of the analyses are based on January data, mostly from the IWC but in the case of goose populations also from the European Goose Management Platform (EGMP) and the WEBS project as coordinated by the British Trust for Ornithology (BTO). Within the monitoring program in the EAF reported in this study, a substantial number of additional counts have been collected, particularly in W Africa, and added to the IWC database. From 2013 onwards, sample counts have been organised annually, and 'total' counts aiming for a comprehensive coverage in 2014, 2017, 2020 and 2023. Efforts were made to obtain data complete up to January 2023 from as many countries as possible (see Acknowledgments and the count reports in Agblonon *et al.* 2023). This more recent data collected in the flyway could be combined with older counts from

the IWC and data from several expeditions in W Africa which are stored in the IWC database as well. Fig. A1.2 shows the extent of IWC site coverage available for the trend analyses in this report for the period 1975-2023. Compared with the 2020 report, substantial updates were received from the Iberia - N Africa region. In southern Africa, a temporary reduction in coverage has occurred between the integral counts of 2020 and 2023.

The current dominant source of breeding bird data in Europe for species of interest in this study is the Article 12 reporting for the EU Birds Directive, supplemented with similar data collected for non-EU countries in the framework of the European Red List of Birds (Erlob). These data are collated once every six years and were not collected internationally for the intervening period until 2023. To maintain stability in the method of trend calculation (which can strongly influence the outcome; van Turnhout *et al.*

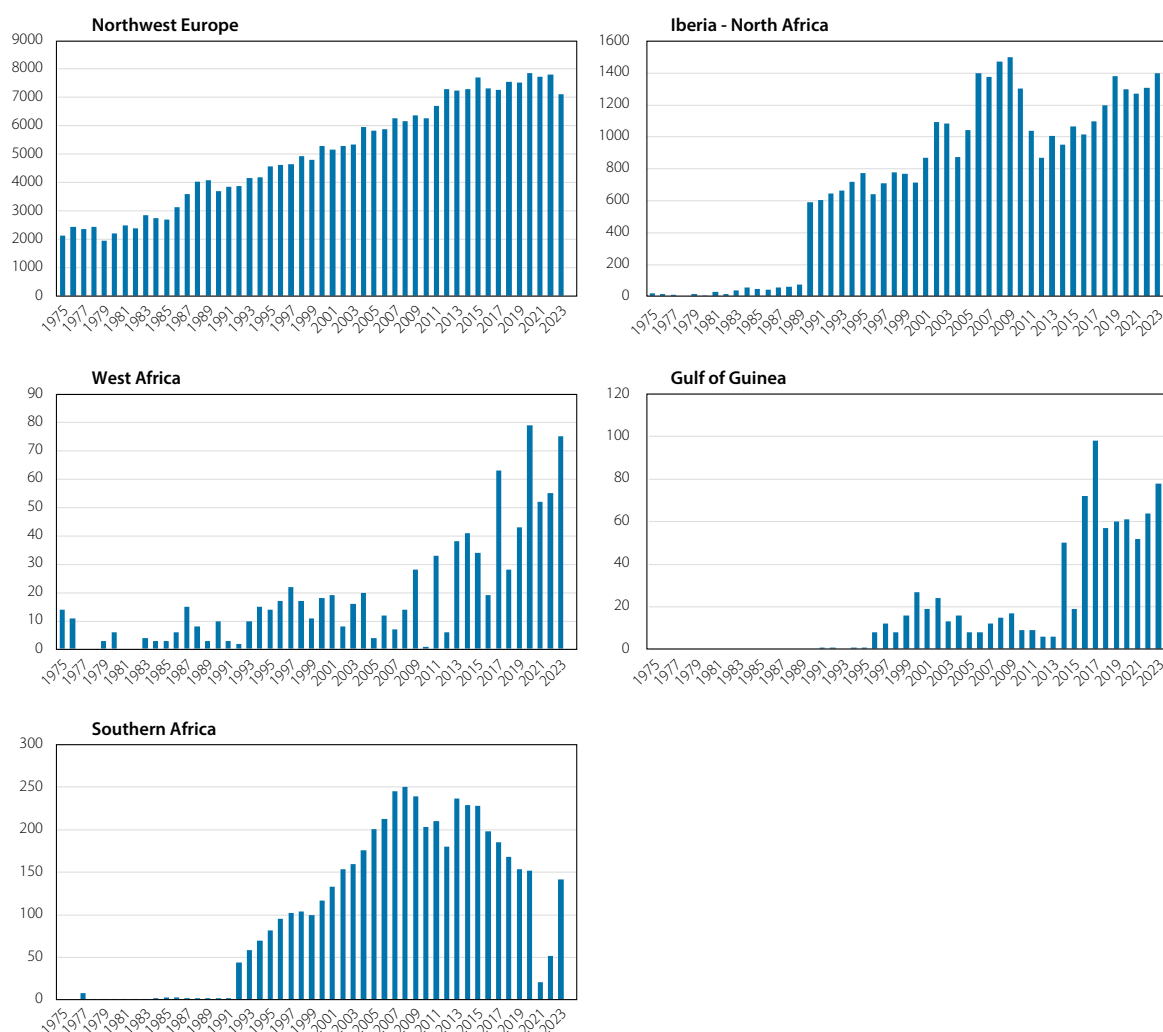


Figure A1.2. Numbers of sites per region included in the trend analyses 1975-2023. Shown is the availability of data in July 2024, when trend analyses for this report started.

2022), in the current report the population trends based on breeding bird data up to 2018 (BirdLife International 2021) are maintained. This dataset is supplemented with more recent data for 2019–2023 from the Pan-European Common Bird Monitoring Scheme (PECBMS), coordinated by the European Bird Census Council (<http://pecbms.info/>). It is hoped that in the future more frequent updates of breeding bird data can be provided for waterbird populations of interest for this EAF, e.g. through a cooperation between EBCC and its national partners.

The population size estimates given in Van Roomen *et al.* 2022 have been maintained in this report. Updating these population sizes partly requires updates on breeding bird numbers and these are not yet available for the 2020–2023 period, but will be in 2026. Likewise, population size estimates based on January data profit from national reporting under Art 12 and these will be available also in 2026, making an overall update of population sizes at that time more comprehensive and complete. As population sizes are very much connected to the 1% threshold for identifying sites of international importance, and thus may have policy implications when they change, having a less regular update of population sizes based on data as complete and up-to-date as possible is preferred.

Distribution maps in this Annex are for January and based on 2019–2023 IWC results. New in this report is the presentation of site trends based on the IWC January data. These time series are presented for sites where at least 0.1% of the flyway population occurs and for which at least four complete January counts are present in the IWC database. The 0.1% threshold prevents that trends are calculated for sites where the species or populations is scarce or rare, which can lead to spurious trend outcomes. The threshold of four counts allows inclusion of some extra sites in western Africa where data is scarce.

A1.3 Analyses and presentation of data

A1.3.1 Flyway population-trends

For the calculation of flyway population trends based on January data, the same methods were applied as used for the AEWA Conservation Status Report 9 (Nagy & Langendoen 2025). This involves selection of sites with a minimum of five counts of which at least one in the last 10 year and at least one in an earlier period. Imputing of missing counts at sites is applied at the level of countries and later on at a higher geographic level again if the first run results into too high levels of imputing. Trend calculation is done with the *rTrim* package (Bogaart *et al.* 2020). The trends calculated with this method are indicated with 'IWC - Trim' in table A1.1. For western Africa some extra rules were applied to include extra sites at one hand and exclude some unreliable data at the other. Other exceptions are a few trends that are calculated with *Trendspotter* (Soldaat *et al.* 2007) when the January data did not originate directly from the IWC (indicated by 'TrendSpotter' in table A1.1).

Trends based on breeding bird data from PECBMS were calculated with *rTrim* following the method described in Nagy *et al.* 2020 ('PECBMS - Trim' in table A1.1). Breeding bird trends based on data from the EU Art. 12 Birds Directive and European Red List monitoring are calculated by using a tool supplied by IUCN to estimate combined trends (BirdLife International 2021, 'IUCN' in table A1.1).

For all flyway populations the trends are expressed as mean yearly proportional change during the complete time series (most often 1975–2023 or 1980–2023, called 'long-term') and for the most recent 10 years (2014–2023, 'short-term'). These trend values, the length of the time series and the trend indication are given for each population in a table in the species accounts. For flyway trends with (near-)annual abundance estimates (all, except those calculated from Erlob - IUCN methodology) trend graphs are given. Trends based on PECBMS have number of territories/breeding pairs as the unit. Trends based on January data are given as number of individual birds (including imputing), with years with more than 70% imputing removed. These estimates (breeding pairs/number of individual birds) should not be used as the total population size but show the magnitude of data included in the trend graph. To visualise the pattern in the yearly estimates a smoothed trend line is shown, calculated with *TrendSpotter* (Soldaat *et al.* 2007). This can be a flexible line or a straight line depending on the number of year estimates available.

A1.3.2 Flyway population distribution and coastal EAF distribution in January

The distribution maps show the breeding and non-breeding ranges of the species (based on BirdLife International & Handbook of Birds of the World 2017). The flyway population boundaries are from the Critical Site Network Tool (Wetlands International & BirdLife International 2018) with a few modifications of aesthetic nature. Flyway boundaries are given only for populations for which flyway trends are included in this report. Two different selections of distribution are shown on the maps. One is the January distribution based on available IWC data, for all flyway populations with flyway trends in this report, including those for which the trend is based on breeding bird data. These are the combination of red and blue dots within the boundary of a flyway population. The other selection shows numbers for the same species, but other biogeographical populations, in other coastal EAF sites (recognisable as red dots outside the flyway boundary if present). All dots denote the average numbers of birds per site in January 2019–2023. January distribution data from Iceland is not available. In Sweden, gulls are not counted during the IWC.

A1.3.3 Site trends in coastal East Atlantic flyway based on January data

In total, we analysed site-level trends for 64 species and 454 sites. Because not each species was observed in suffi-

cient number (at least 0.1 % of flyway population size needed to be present in January) at each site, we ended up with 2,393 combinations of species x site for which trends were calculated. For each species x site combination, a generalized linear model (GLM) with a quasi-poisson distribution was fitted to the time series of (imputed) abundances. The quasi-poisson family was used because it does not assume a fixed relationship between the mean and variance; absence of such a relationship is common in ecological count data.

The slope of the time effect in this model represents the rate of change in abundance over time at that site. Confidence intervals (95% in this case) were calculated around the slope estimates to assess the uncertainty in the trend. Trends at each site were categorised as an increase (green dot in the graphs) if the lower bound of the confidence interval was greater than zero. If the upper bound of the confidence interval was less than zero, the species was considered to be decreasing in abundance at that site (red dots). If the confidence interval included zero, the trend was categorised as stable (light grey dots).

A1.4 Results

The species texts in this section provide some background on the flyway populations considered in this report and their distribution and ecology. Just a few references are included to sources of particular species information; otherwise information provided in these accounts is based mostly on the *Handbook of the Birds of the World* (del Hoyo et al. 1992, 1994, 1996), the *Atlas of Anatidae popu-*

lations in Africa and Western Eurasia (Scott & Rose 1996), the *Atlas of Wader populations of Africa and Western Eurasia* (Delany et al. 2009), van de Kam et al. (2004) and van Roomen et al. (2015, 2018 and 2022).

The main results per species consist of a table which updated trends (long-term and short term) per population until 2023. Population size estimates are the same as in the report following the total count in 2020 (van Roomen et al. 2022). A map illustrates the distribution in January 2019-2023 of the flyway populations covered by this report and the additional distribution of other flyway populations along the coastal EAF. Flyway trend graph(s) are shown if available (not for populations based on Erlob breeding bird data). Trends are based on the source which is considered at present as the most suitable and available for the monitoring of the flyway populations (see table A1.1). New in this report are graphs depicting slope of trends at sites based on IWC January counts arranged along a latitudinal y-axis which corresponds to the distribution map. Within these graphs also the slope of the flyway population trends are given as a vertical line (length corresponds with the latitudinal January range of the flyway population). In this way, slopes of site trends can be compared with the slope of the overall flyway trend. They show comparable or contrasting trends and also geographic patterns in trends (for instance decreases in the South, increases in the North). Both observations can help in understanding the origin of trends at sites.

For literature references mentioned in this Annex, see the main reference list.



Standing in the water to prevent disturbance of high tide roost on small island

Albert de Jong

Brent Goose | *Branta bernicla* | Bernache cravant

Within the EAF three flyway populations occur, belonging to two different subspecies (*B. b. bernicla* and *B. b. hrota*). The largest population is formed by the dark-bellied subspecies *B. b. bernicla*, which breeds on arctic tundra in W and C Siberia and winters mainly in coastal NW Europe, from the Wadden Sea to W France, with smaller numbers migrating further south. Birds of the pale-bellied subspecies *B. b. hrota* breeding on Svalbard winter mainly in Denmark, while those breeding in Greenland and NE Canada winter in Ireland. The species is fully migratory, arriving on the breeding grounds in early June. Breeding occurs in small, loose colonies or dispersed in single pairs. The preferred breeding sites are grassy coastal meadows or islands where large raptors, snowy owls or gulls are present that can deter mammalian predators. Non-breeding birds inhabit estuaries and bays with seagrass *Zostera*, coastal saltmarshes and cultivated grasslands.



Arnold Meier / Blue Robin

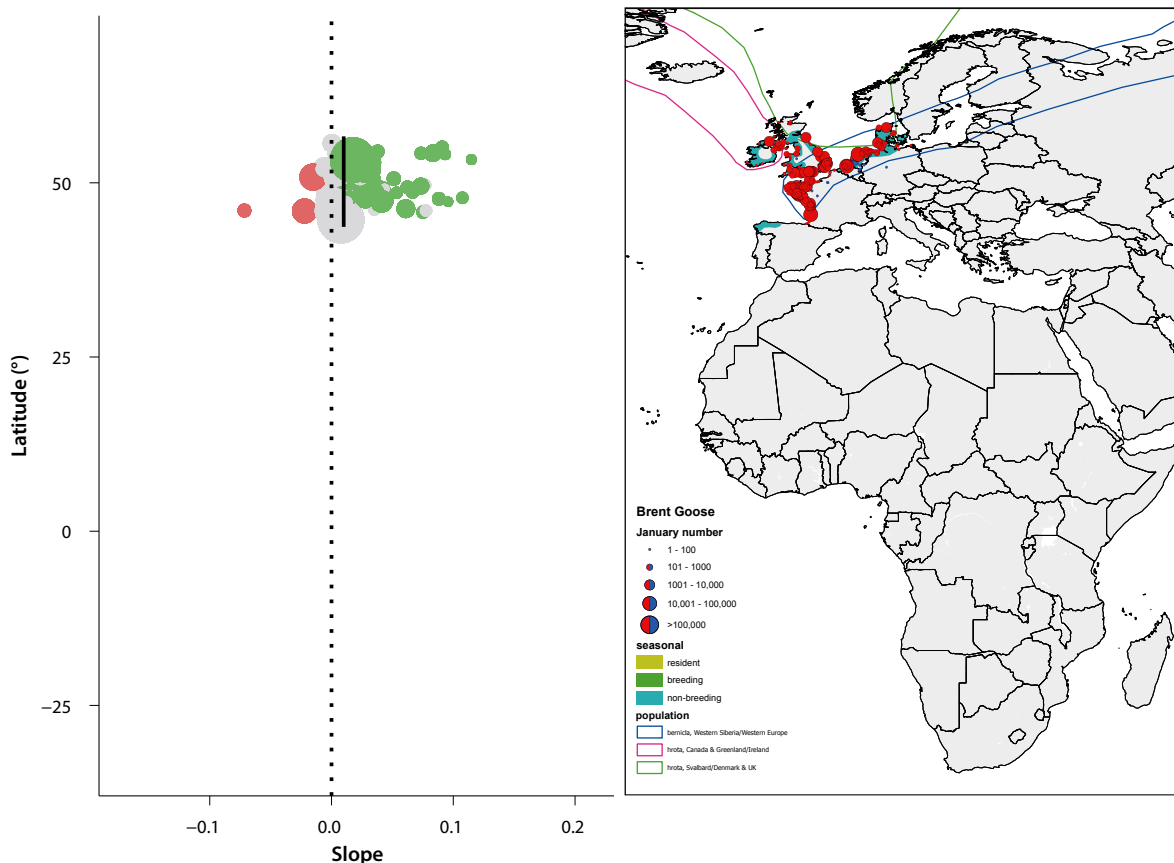


Figure A1.3. (Left) Plot of Brent Goose site January trends, shown as the slopes of generalized linear models (GLM; e.g. a slope of +0.1 denotes that the population grows with 10% per year). Latitude on the y-axis is aligned with the range of latitude in the distribution map. The slope of the flyway trend is also given as a thick vertical line (position on the x-axis indicates slope; the vertical height corresponds with the latitudinal range of the flyway population in January). Red dots indicate decreasing trends, green dots increasing and grey dots 'stable' trends not significantly deviating from zero. The size of the dots is scaled to the median number of individuals at each site in January 2019-2023. The symbols (dots, triangles, squares) represents different flyway populations if present. (Right) Distribution of Brent Goose in the East Atlantic Flyway in January 2019-2023. Lines indicate the flyway boundaries of the populations for which the flyway trend is presented. Red dots denote numbers in coastal EAF sites; inland sites also included in the calculation of the flyway trend but located outside the coastal EAF boundaries are indicated with blue dots.

population Brent Goose	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
<i>bernicla</i> , Siberia/W Europe	w	1976-2023	1,014	moderate increase	2014-2023	0,981	moderate decrease	2011	211000	211000
<i>hrota</i> , Svalbard/Denmark & UK	w							2020	13800	13800
<i>hrota</i> , Canada & Greenland/Ireland	w	1976-2021	1,022	moderate increase	2014-2021	0,987	uncertain	2021	35000	35000

Table A1.2. Summary of flyway trends and population size estimates for Brent Goose. Mentioned are the flyway populations under investigation, data type used for the trend (w = January counts, b = breeding bird counts), the time period of the long-term trend (period-L), the trend slope (trend-L; proportional change per year) and the trend category, the same for the short-term trend (-S), and finally the time period for the population size estimate (period pop. size) and the minimum and maximum population estimates (no. of individuals) which are repeated from van Roomen *et al.* 2022.

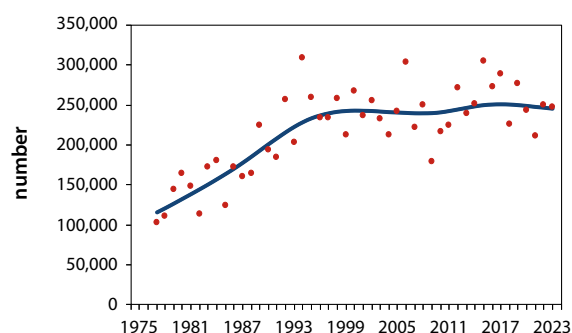


Figure A1.4. Flyway population trend for Brent Goose population *Branta, b. bernicla*. Red dots denote the yearly abundance results (not to be equated to total population sizes, see main text), the blue line is the smoothed trend line.

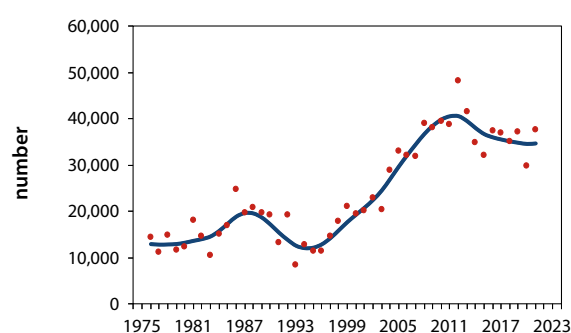


Figure A1.5. Flyway population trend for Brent Goose population *Branta, b. hrota* Canada & Greenland. For explanation see fig A1.4.



Harvey van Diek

Barnacle Goose | *Branta leucopsis* | Bernache nonnette

Three flyway populations are distinguished within the EAF. The largest is the population breeding on coastal tundra in arctic Russia and the Baltic, wintering mainly in The Netherlands and Germany, where also a rapidly increasing resident breeding population has become established in recent decades (van der Jeugd *et al.* 2009). The Svalbard breeding population migrates via Norway to southern

Scotland and England, while the Greenland breeding population winters in Ireland and Britain. Arctic breeding sites are typically rocky outcrops, slopes, crags, cliffs or coastal islands near wetlands or coastlines. Non-breeding birds inhabit coastal meadows, saltmarshes and tidal mudflats, with increasing use of cultivated grasslands for feeding in recent decades.



Harvey van Diek

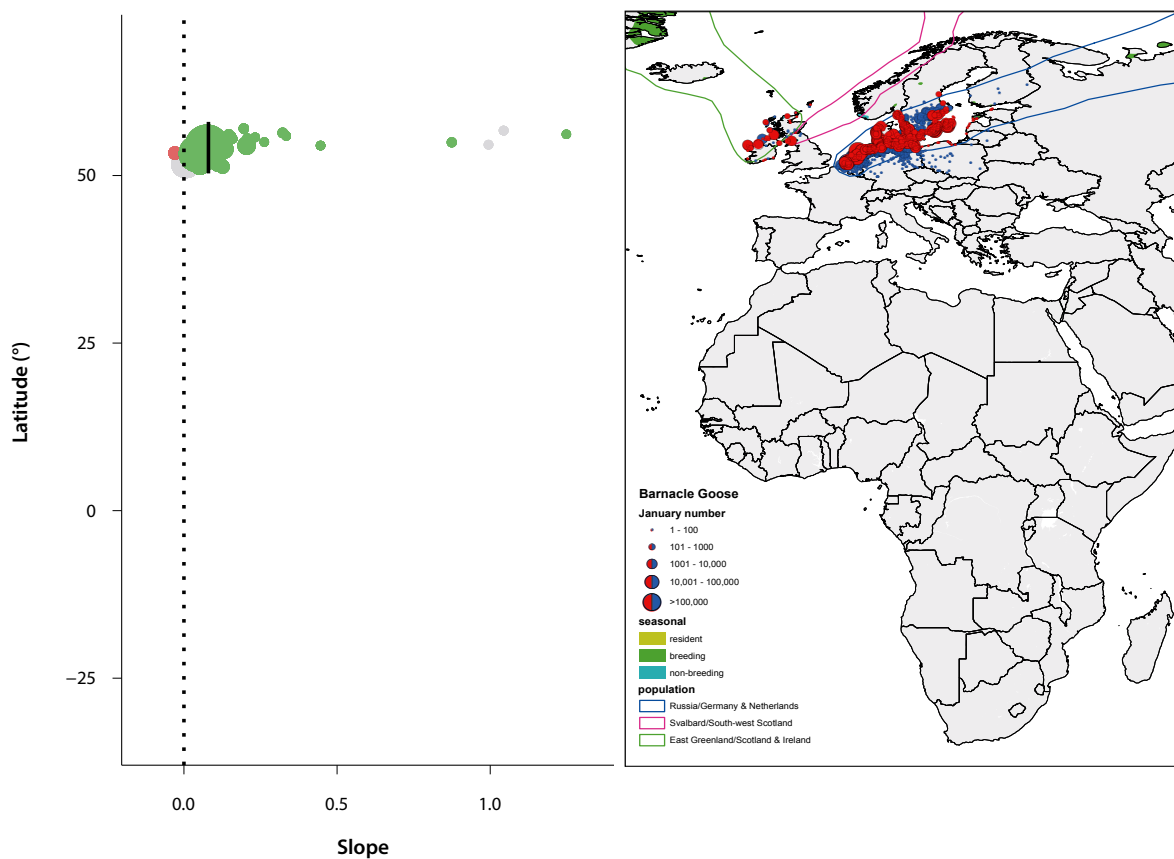


Figure A1.6. Plot (left) of site January trends and distribution (right) of Barnacle Goose in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Barnacle Goose	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Siberia & NW Europe/NW Europe	w	1976-2023	1,082	strong increase	2014-2023	1,050	moderate increase	2018	1400000	1400000
East Greenland/Scotland & Ireland	w	1978-2023	1,014	moderate increase	2014-2023	0,974	moderate decrease	2018	72000	72000
Svalbard/South-west Scotland	w	1975-2023	1,039	moderate increase	2014-2023	0,991	stable	2015- 2020	40000	40000

Table A1.3. Summary of flyway trend and population size estimates for Barnacle Goose. For explanation see table A1.2.



Harvey van Diek

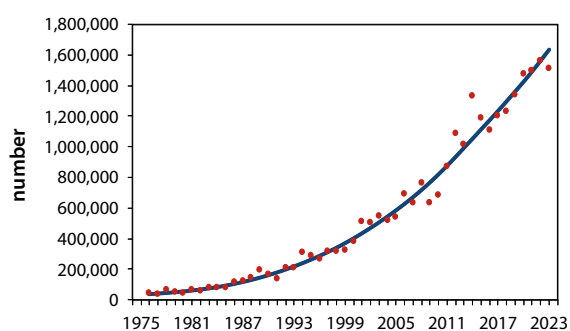


Figure A1.7. Flyway population trend for Barnacle Goose population Siberia & NW Europe/NW Europe. For explanation see fig. A1.4.

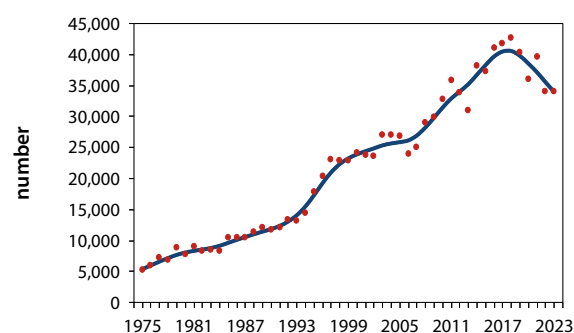


Figure A1.8. Flyway population trend for Barnacle Goose population E Greenland / Scotland & Ireland. For explanation see fig. A1.4.

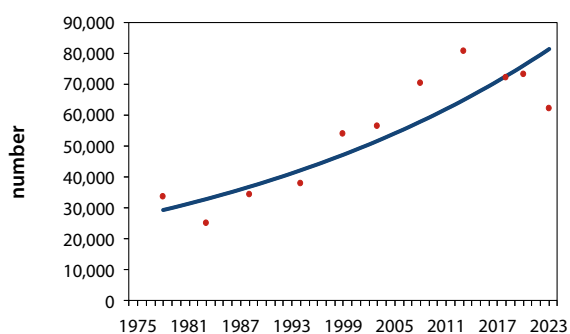


Figure A1.9. Flyway population trend for Barnacle Goose population Svalbard / SW Scotland. For explanation see fig. A1.4.



Arnold Meijer / Blue Robin

Greylag Goose | *Anser anser* | Oie cendrée

At least three distinct populations occur in the EAF. The largest occurs from northern Norway across continental W Europe to Morocco. The Nordic birds traditionally wintered in Spain, but nowadays an increasing part of the population stays closer to the breeding areas. In temperate Europe, mainly resident breeding populations have increased strongly. The Icelandic breeding population winters in the United Kingdom and Ireland. A smaller population breeds and winters in N Scotland (not included here). Greylag Geese breed in a wide variety of wetlands, close to potential feeding sites such as meadows, grasslands or agricultural fields, often in loose colonies. During the non-breeding season, the species is highly gregarious and flocks can be found on lowland farmland or in swamps, lakes, saltmarshes and coastal lagoons. Greylag Geese are herbivorous, feeding on grass, on roots and above-ground parts of herbaceous marsh vegetation, aquatic plants and on cereals and potatoes.



Ran Schols / Agami

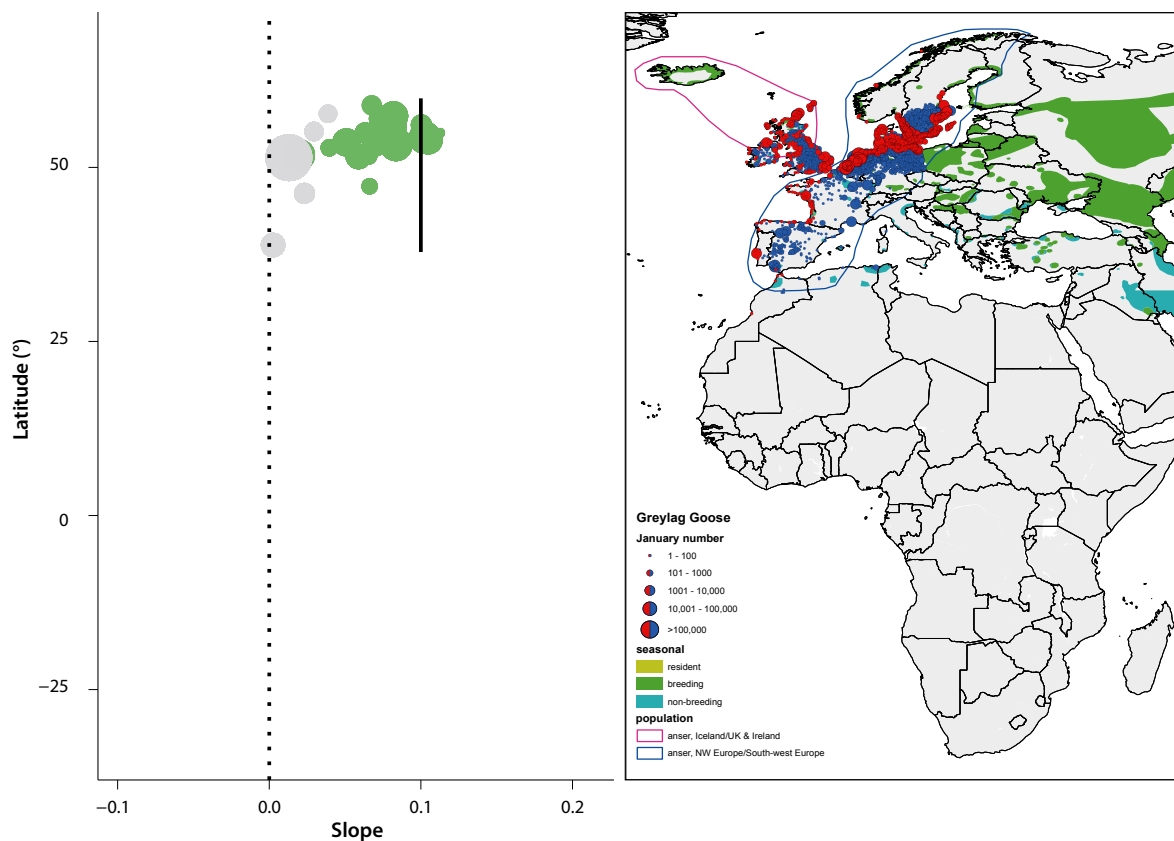


Figure A1.10. Plot (left) of site January trends and distribution (right) of Greylag Goose in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Greylag Goose	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
anser, NW Europe/NW & SW Europe	w	1981-2023	1,096	strong increase	2014-2023	1,014	moderate increase	2016-2018	710000	780000
anser, Iceland/NW Europe	w	1975-2023	0,999	stable	2014-2023	0,948	moderate decrease	2015-2019	76000	76000

Table A1.4. Summary of flyway trend and population size estimates for Greylag Goose. For explanation see table A1.2.

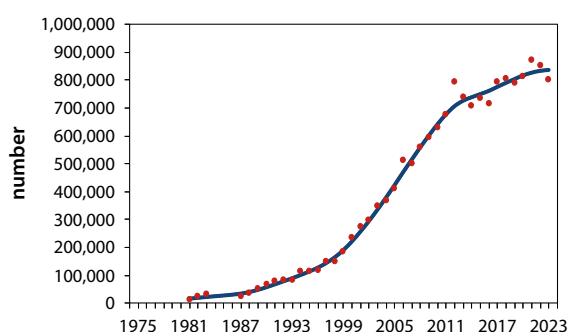


Figure A1.11. Flyway population trend for Greylag Goose population NW Europe / SW Europe. For explanation see fig. A1.4.

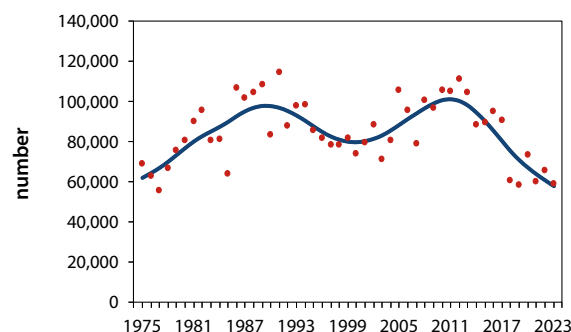


Figure A1.12. Flyway population trend for Greylag Goose population Iceland / UK & Ireland. For explanation see fig. A1.4.

Common Eider | *Somateria mollissima* | Eider à duvet

The Common Eider has a Holarctic breeding distribution with several sub-populations in Europe. Recently the Baltic-Wadden Sea population and Britain-Ireland population have been merged, and this constitutes the population considered here. Breeding occurs in coastal areas of the Baltic Sea and North Sea and in Scotland and Ireland. It is a partial or short-distance migrant, and wintering areas are mainly within the breeding range and south to Atlantic France. Breeding habitats include offshore islands and

islets with grassy or dense, low vegetation (shrubs and bushes) or rocks, but also mainland coasts. Breeding occurs in loose colonies of up to a few thousand pairs. Outside the breeding season, the species is highly gregarious and concentrates in shallow coastal seas and estuaries. Its diet in the Wadden Sea consists predominantly of large benthic molluscs, predominantly mussels (*Mytilus sp.*) and to lesser extent cockles (*Cerastoderma sp.*). American Razor Shell (*Ensis sp.*) has also been recorded in the diet.



Harvey van Diek

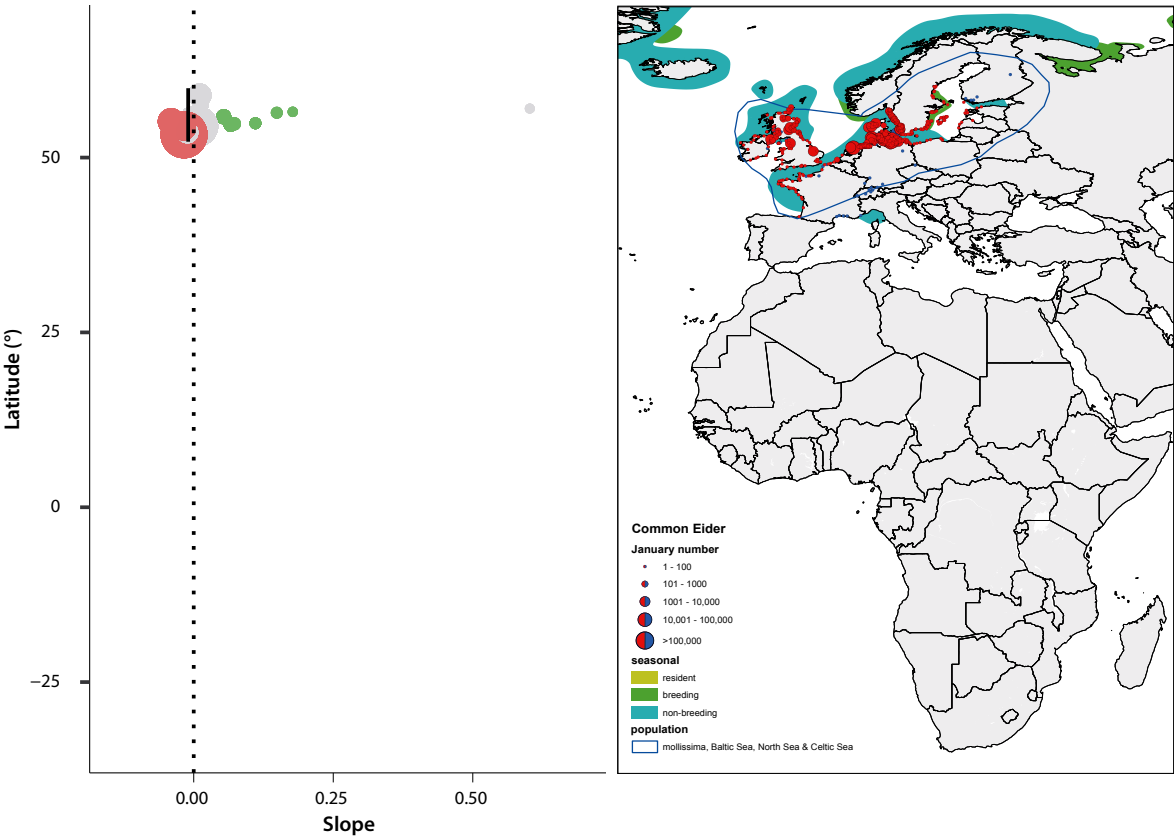


Figure A1.13. Plot (left) of site January trends and distribution (right) of Common Eider in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Common Eider	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
mollissima, Baltic Sea, North Sea & Celtic Sea	w	1980-2023	0,991	moderate decrease	2014-2023	0,926	moderate decrease	2012-2018	560000	920000

Table A1.5. Summary of flyway trend and population size estimates for Common Eider. For explanation see table A1.2.

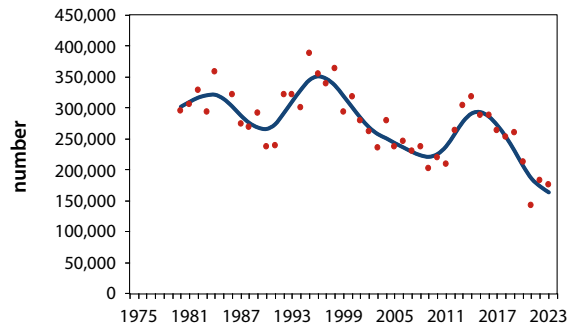


Figure A1.14. Flyway population trend for Common Eider population mollissima Baltic, North & Celtic Seas. For explanation see fig. A1.4.



Harvey van Diek

Common Goldeneye | *Bucephala clangula* | Garrot à oeil d'or

In the EAF, the Common Goldeneye's breeding areas range across the boreal forests of Scandinavia and E Europe. Its wintering range is very broad, encompassing the coast of N Europe and coastal and inland waters in NW and C Europe. Most individuals of this species are migratory, although they may only travel short distances. The species is mainly restricted to inshore waters and it requires tree-holes (or artificial nestboxes) for nesting. Suitable breeding habitats include freshwater lakes, pools, rivers and deep marshes surrounded by coniferous forest. The diet is rather broad and consists of aquatic invertebrates such as molluscs, worms, crustaceans, aquatic insects and insect larvae, as well as amphibians, small fish and some plant material.



Markus Varesvuo / Agami

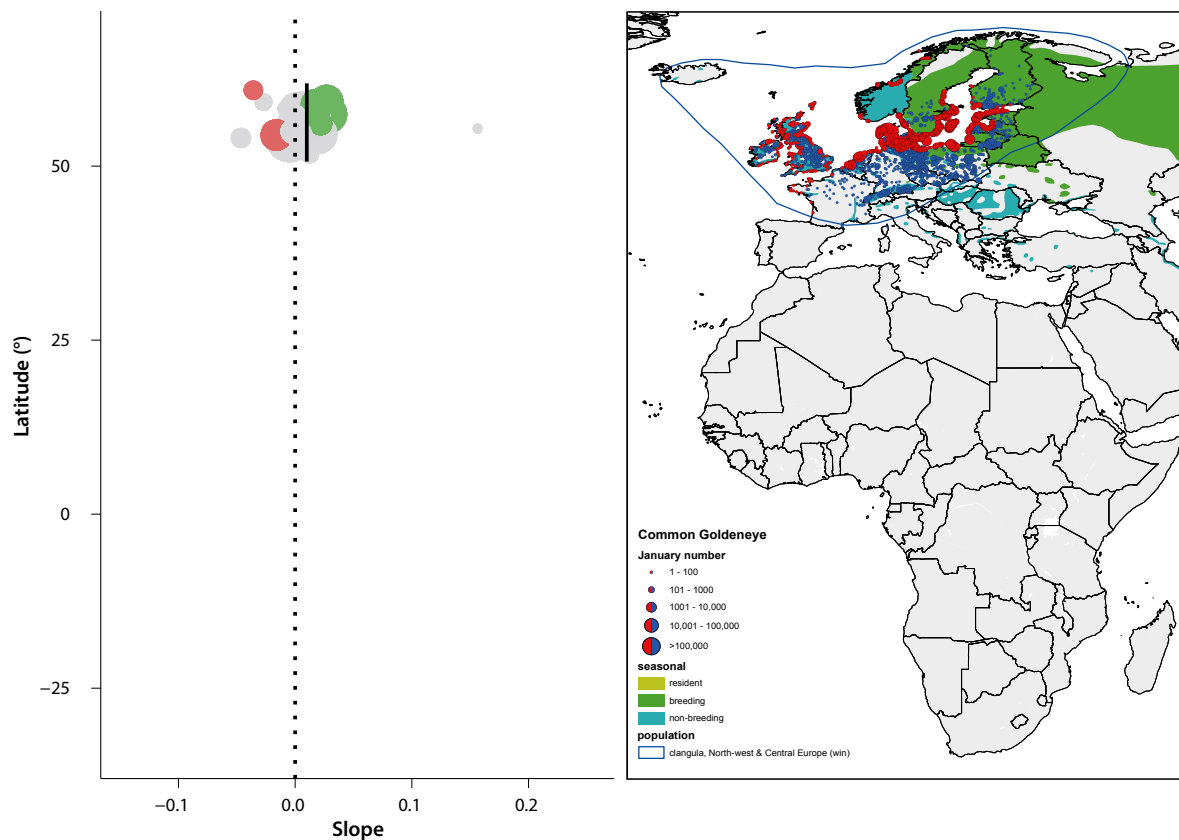


Figure A1.15. Plot (left) of site January trends and distribution (right) of Common Goldeneye in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Common Goldeneye	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW and C Europe (winter)	w	1975-2023	1,006	moderate increase	2014-2023	0,999	stable	2011- 2018	750000	1500000

Table A1.6. Summary of flyway trend and population size estimates for Common Goldeneye. For explanation see table A1.2.

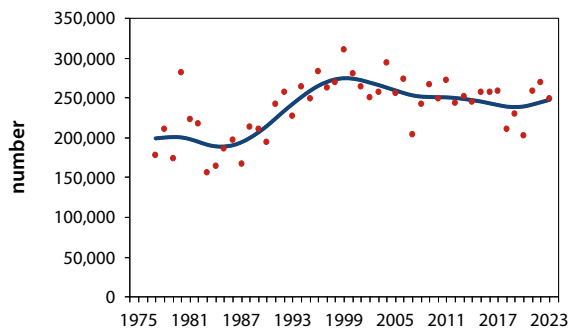


Figure A1.16. Flyway population trend for Common Goldeneye population NW & C Europe (win). For explanation see fig. A1.4.



Ralph Martin / Agami

Red-breasted Merganser | *Mergus serrator* | Harle huppé

This species has a large distribution and breeds in Greenland, Denmark, Iceland and much of N Eurasia south to the United Kingdom and parts of E Europe. The population considered here winters in NW and C Europe, while birds breeding further east migrate to the Mediterranean and Black and Caspian Sea basins. This species is fully migratory, although in temperate regions it only undertakes

short-distance movements to nearby coasts, or remains close to its breeding waters throughout the year. Most birds winter at sea, frequenting both inshore and offshore waters, estuaries, bays and brackish lagoons. The diet consists of small fish, as well as small amounts of plant material and aquatic invertebrates, worms and insects.

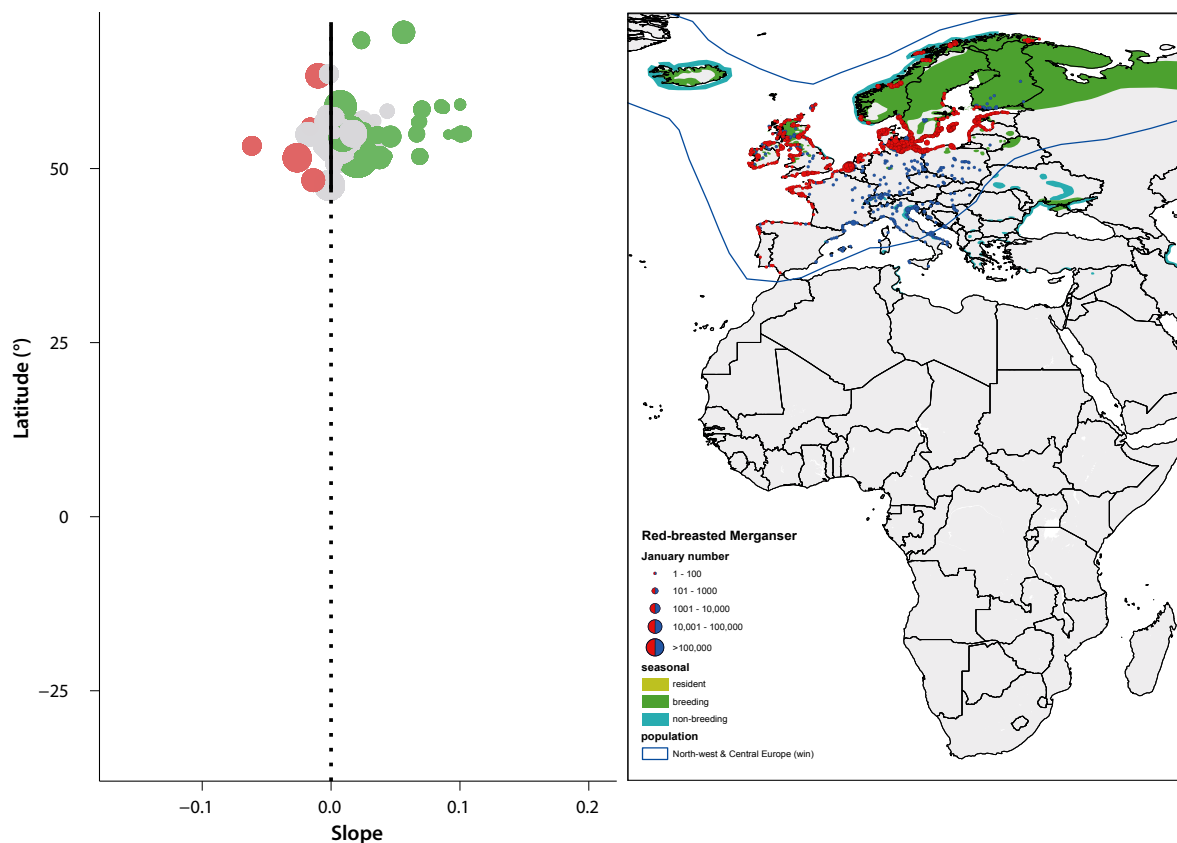


Figure A1.17. Plot (left) of site January trends and distribution (right) of Red-breasted Merganser in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Red-breasted Merganser	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
NW Europe and C Europe (winter)	w	1975-2023	1,003	stable	2014-2023	1,033	moderate increase	1992- 2019	100000	160000

Table A1.7. Summary of flyway trend and population size estimates for Red-breasted Merganser. For explanation see table A1.2.

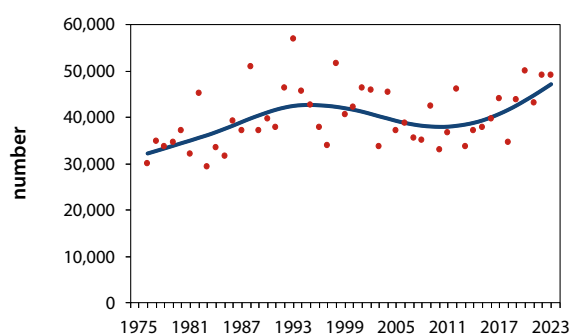


Figure A1.18. Flyway population trend for Red-breasted Merganser population NW & C Europe (win). For explanation see fig. A1.4.



Ralph Martin / Agami

Common Shelduck | *Tadorna tadorna* | Tadorne de Belon

The Common Shelduck has two distinct populations in Europe: one in NW Europe and one in the Mediterranean and Black Sea basins. The NW European population is the one considered here. These Shelduck breed in countries around the North Sea and the Baltic, Norway and Iceland, and south to France and Spain. The largest numbers breed in the UK, The Netherlands, Germany, Denmark and Sweden. After the breeding season in which the species is mostly dispersed, it congregates in huge flocks to moult at

specific sites, sometimes after traveling hundreds of kilometres. Breeding occurs in coastal dune areas where it uses burrows, but also inland along rivers and lakes. Common Shelduck are partially migratory and wintering occurs in the same range as breeding. The moulting and wintering habitats are saline lagoons, estuaries and mudflats where it feeds mainly on small molluscs and other aquatic invertebrates, including mud snails *Peringia ulvae* and small crustaceans *Corophium volutator*.



Harvey van Diek

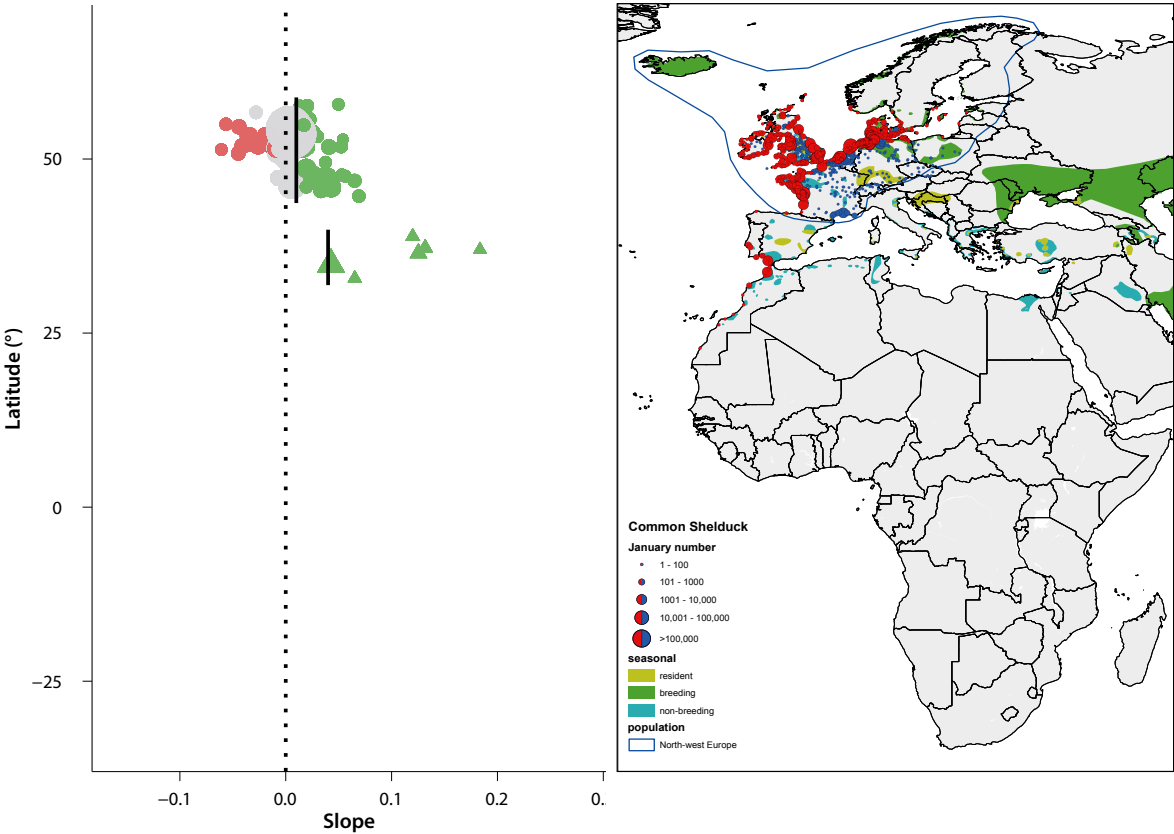


Figure A1.19. Plot (left) of site January trends and distribution (right) of Common Shelduck in the East Atlantic Flyway in January . For explanation see fig. A1.3.

population Common Shelduck	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
NW Europe	w	1976-2023	1,007	moderate increase	2014-2023	1,000	stable	2010- 2019	310000	310000

Table A1.8. Summary of flyway trend and population size estimates for Common Shelduck. For explanation see table A1.2.

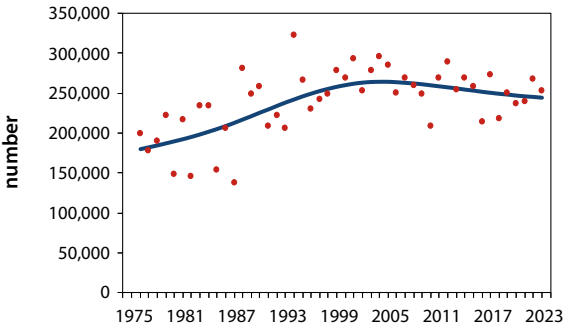
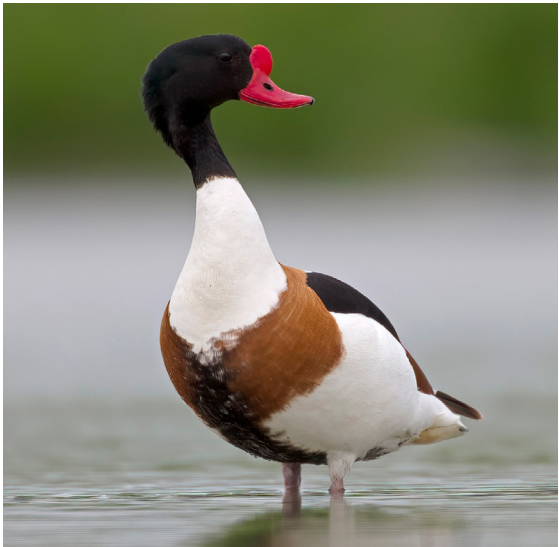


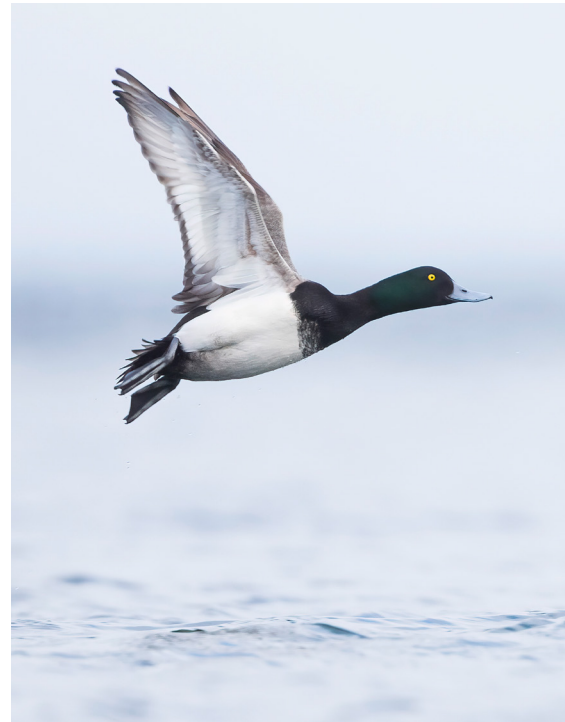
Figure A1.20. Flyway population trend for Common Shelduck population NW Europe. For explanation see fig. A1.4.



Harvey van Diek

Greater Scaup | *Aythya marila* | *Fuligule milouinan*

One flyway population of the Greater Scaup occurs within the EAF, the nominate subspecies *A. m. marila*. The breeding grounds range across the northern limits of Europe, including Iceland. It winters mainly along the northern coastlines of continental Europe. This species is fully migratory, with males tending to remain further north than females or immatures. It winters on shallow coastal waters as well as sheltered bays, estuaries and brackish coastal lagoons, but is also found inland on large lakes and reservoirs. During the northern winter, it feeds on mussels, cockles, clams and *Hydrobia* snails. Other food sources include insects, aquatic insect larvae, crustaceans such as amphipods, worms, small fish, and the roots, seeds and vegetative parts of aquatic plants.



Ralph Martin / Agami

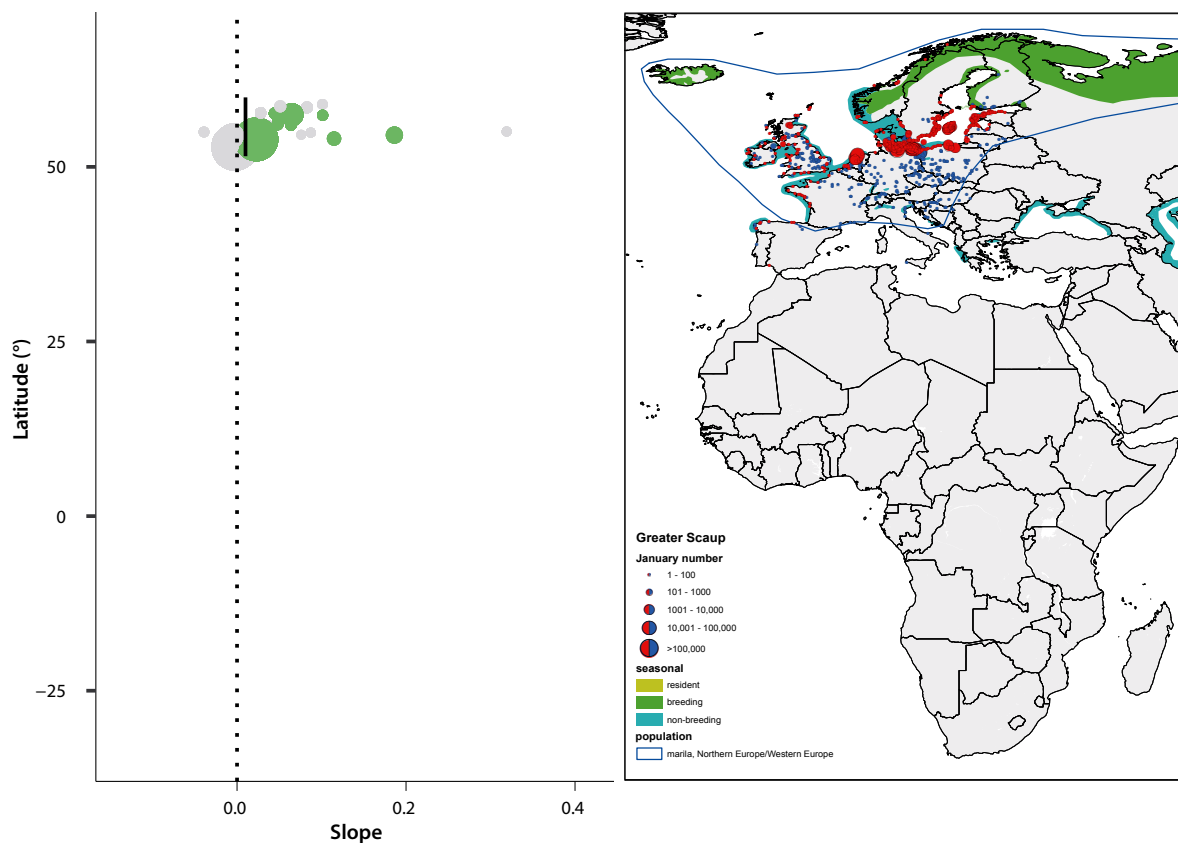


Figure A1.21. Plot (left) of site January trends and distribution (right) of Greater Scaup in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Greater Scaup	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
Northern Europe/Western Europe	w	1976-2023	1,008	moderate increase	2014-2023	0,998	stable	2014- 2018	240000	280000

Table A1.9. Summary of flyway trend and population size estimates for Greater Scaup. For explanation see table A1.2.

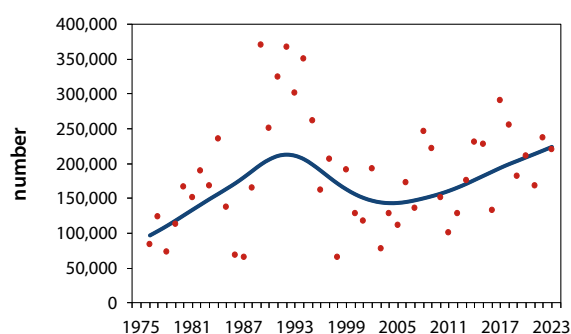


Figure A1.22. Flyway population trend for Greater Scaup population N Europe / W Europe. For explanation see fig. A1.4.

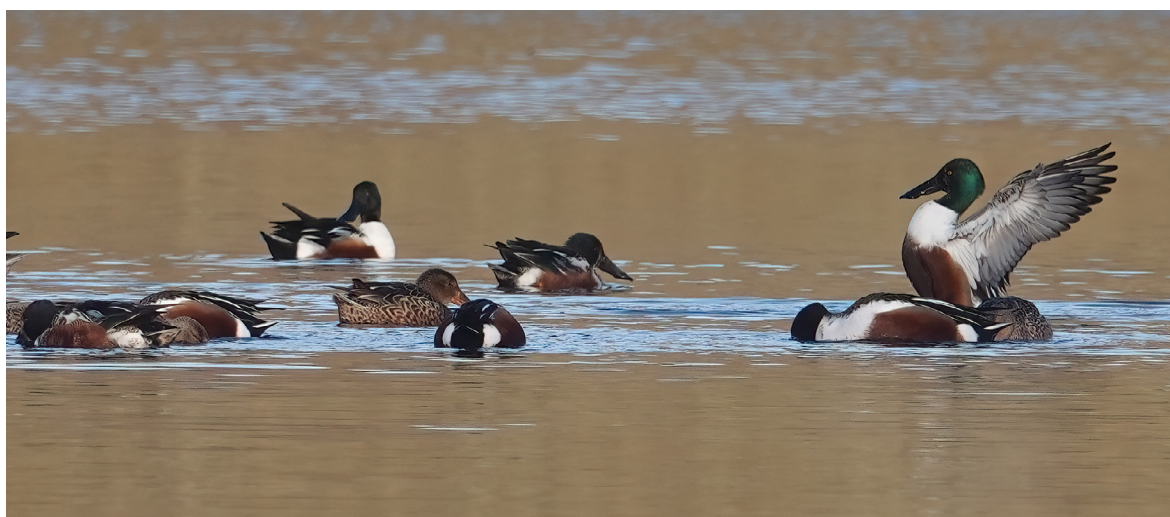


Harvey van Diek

Northern Shoveler | *Spatula clypeata* | Canard souchet

This Holarctic species is highly migratory with a wide breeding distribution. There seems to be considerable overlap in the breeding areas of populations wintering in Europe and W Africa. The flyway population considered in this report is defined as those wintering in NW and C Europe (including the Wadden Sea), while birds breeding in NE and E Europe and W Siberia are thought to winter in S Europe and N and W Africa. The species breeds in shallow freshwater marshes, lakes and along rivers in open habitats with a dense (semi)aquatic vegetation layer. Large

numbers mainly breed in the boreal zones of Fenno-Scandinavia, with the highest numbers in Finland and probably Russia. After the breeding season, large numbers congregate to moult, some in W Europe. The NW European population winters as far south as southern France. Wintering and moulting habitats include coastal lagoons, saline marshes, estuaries and tidal flats but also freshwater wetlands. The species is omnivorous with seeds, algae, grasses, and benthic invertebrates in its diet.



Hans Schekkerman

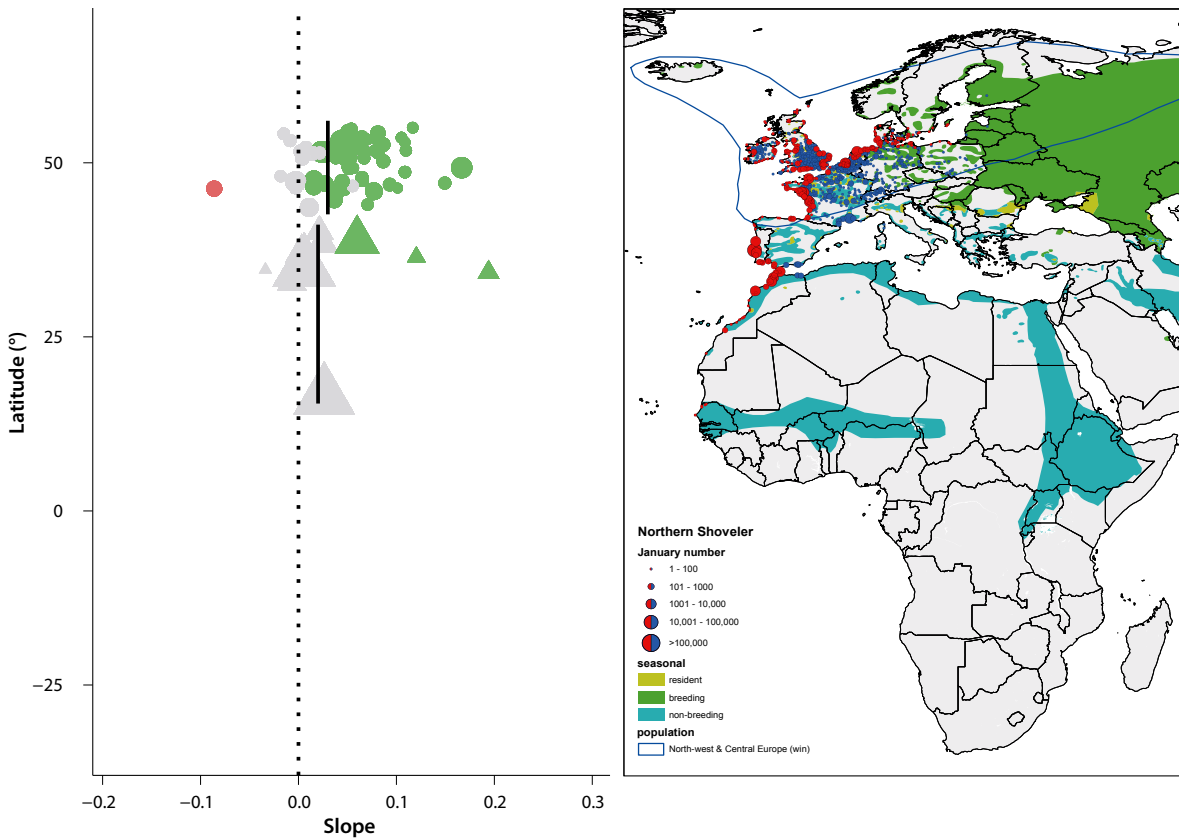


Figure A1.23. Plot (left) of site January trends and distribution (right) of Northern Shoveler in the East Atlantic Flyway in January . For explanation see fig. A1.3.

population Northern Shoveler	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
NW & C Europe (winter)	w	1975-2023	1,028	moderate increase	2014-2023	1,018	moderate increase	2014- 2018	70000	80000

Table A1.10. Summary of flyway trend and population size estimates for Northern Shoveler. For explanation see table A1.2.

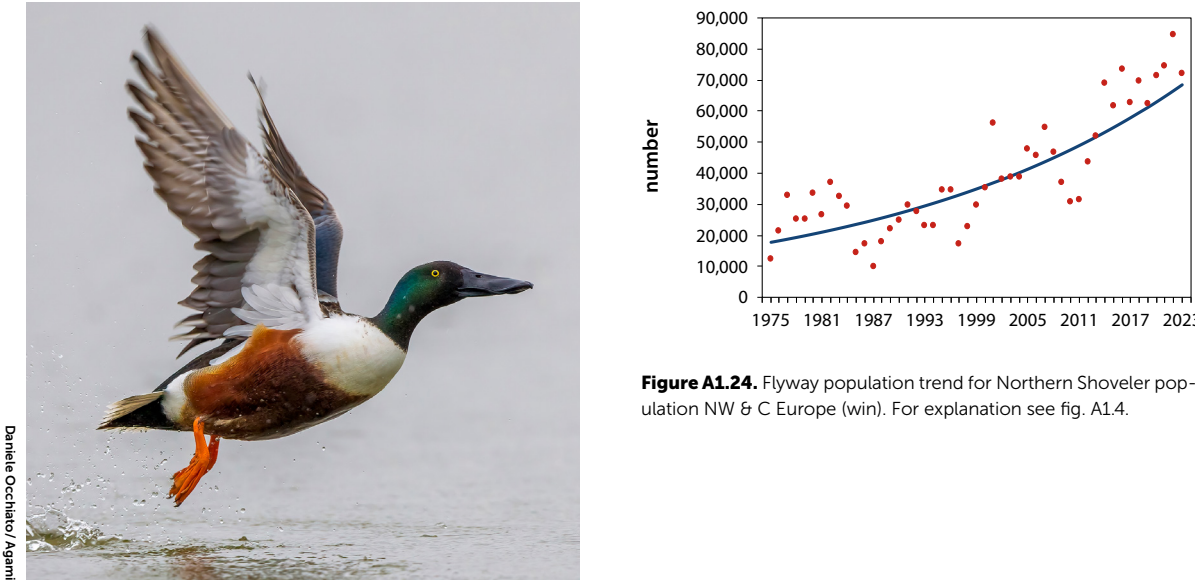


Figure A1.24. Flyway population trend for Northern Shoveler pop-
ulation NW & C Europe (win). For explanation see fig. A1.4.

Eurasian Wigeon | *Mareca penelope* | Canard siffleur

There are two populations of Eurasian Wigeon in Europe: a NW European wintering population and a Black Sea-Mediterranean wintering population. The breeding origins of these two populations largely overlap in large areas in N Russia. Here the NW European wintering population is considered, of which more than 75% winters in The Netherlands, UK and France. These birds breed mainly in the boreal zone of Fennoscandia, with large numbers in Finland, Sweden and Russia east to the Yenisei river, and much lower numbers in countries further south to the North Sea. A decline in abundance in the west and south of the wintering range (Spain and Ireland) may result from short-stopping consistent with milder winters further north, although the core of the wintering distribution does not seem to have shifted much (Fox *et al.* 2016). Breeding habitat consists of freshwater wetlands such as marshes, small lakes, mires in sparsely forested areas, avoiding tundra. Wintering occurs in marine habitats such as salt-marshes, saline lagoons and estuaries and also extensively on agricultural grasslands. The species is largely herbivorous, but in the breeding season relies also on invertebrates. During winter it mainly feeds on grasses.



Harvey van Diek

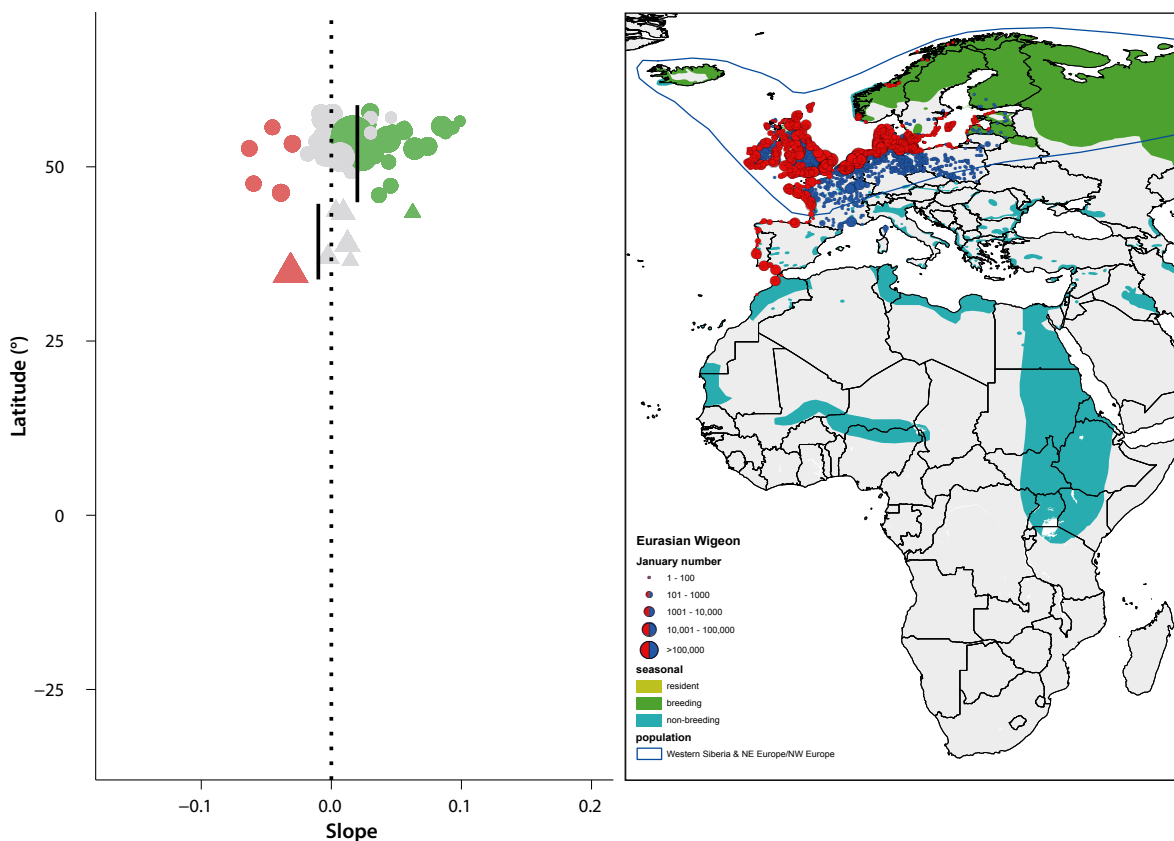


Figure A1.25. Plot (left) of site January trends and distribution (right) of Eurasian Wigeon in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Eurasian Wigeon	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Siberia and NE Europe/NW Europe	w	1976-2023	1,016	moderate increase	2014-2023	1,013	moderate increase	2014- 2018	1300000	1600000

Table A1.11. Summary of flyway trend and population size estimates for Eurasian Wigeon. For explanation see table A1.2.

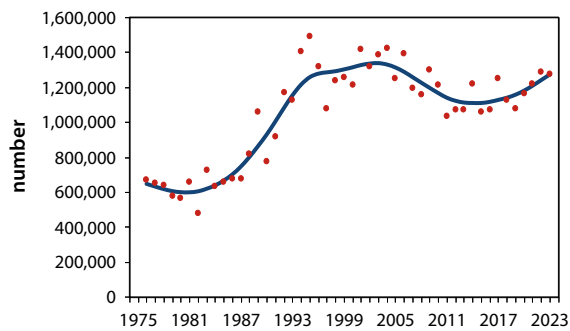


Figure A1.26. Flyway population trend for Eurasian Wigeon population W Siberia & NE Europe / NW Europe. For explanation see fig. A1.4.



Hans Schekkerman

Mallard | *Anas platyrhynchos* | Canard colvert

In the EAF area two flyway populations occur based on their wintering distribution, but likely with overlapping breeding ranges. The NW European population considered here includes Mallards wintering south to N France, with largest concentrations along the Baltic coast, in The Netherlands and the United Kingdom, and along the French Atlantic coast. Two other populations, breeding in N and E Europe respectively, winter in the Mediterranean and Black Sea regions. The species is partially migratory, with many

northern-breeding birds migrating south in winter to mix with resident birds in temperate regions. The species occurs on nearly all wetland types with shallow water and some cover, but avoids fast-flowing or oligotrophic waters. Mallards are omnivorous and opportunistic, adjusting their diet to the seasonally variable availability of animal and plant matter. This means that their summer diet consists mainly of invertebrates and their winter diet of seeds and vegetative parts of aquatic and terrestrial plants.



Harvey van Diek

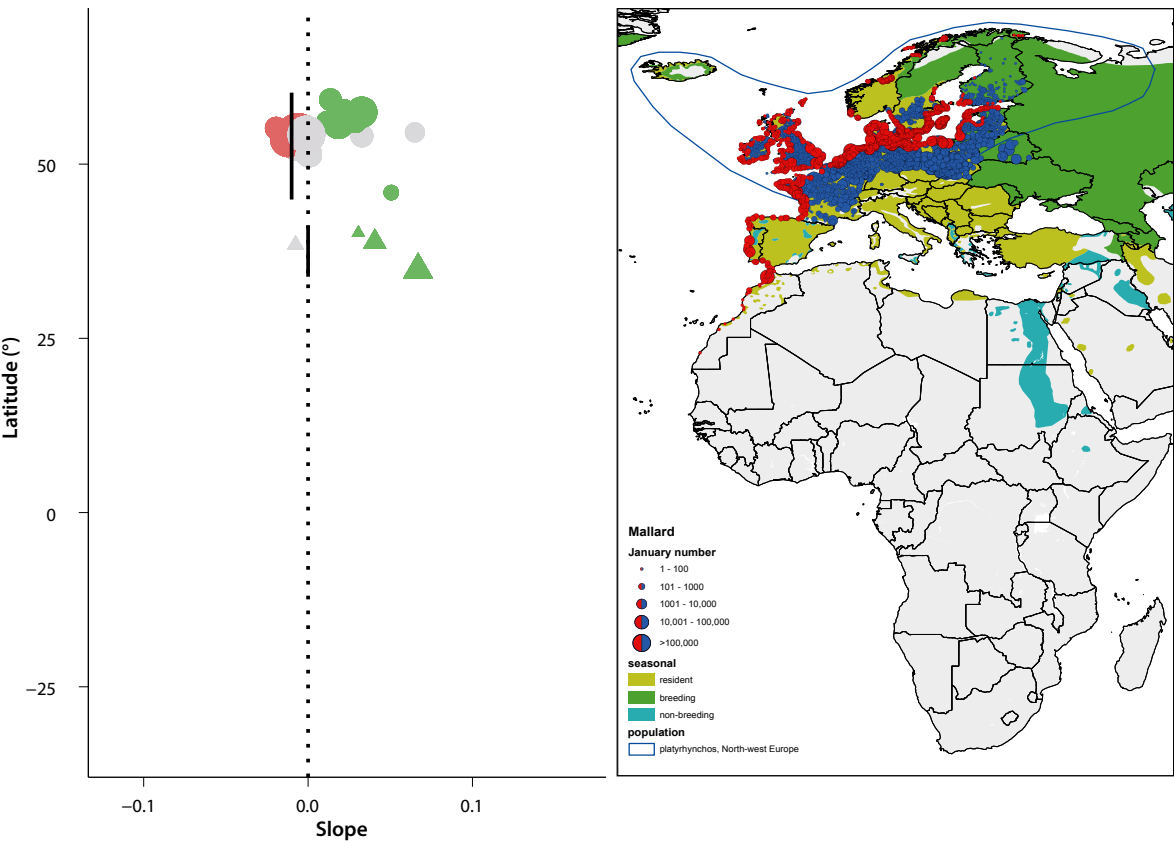


Figure A1.27. Plot (left) of site January trends and distribution (right) of Mallard in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Mallard	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
NW & E Europe & Siberia/NW Europe	w	1976-2023	0,991	moderate decrease	2014-2023	0,984	moderate decrease	2013- 2018	4500000	7100000

Table A1.12. Summary of flyway trend and population size estimates for Mallard. For explanation see table A1.2.

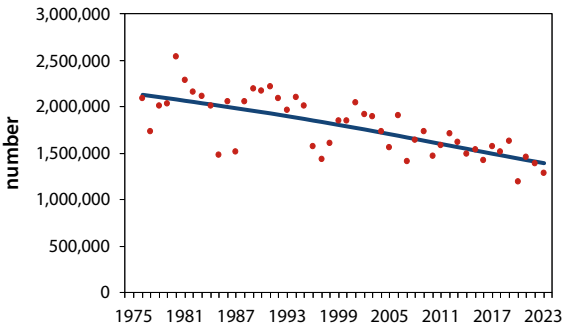
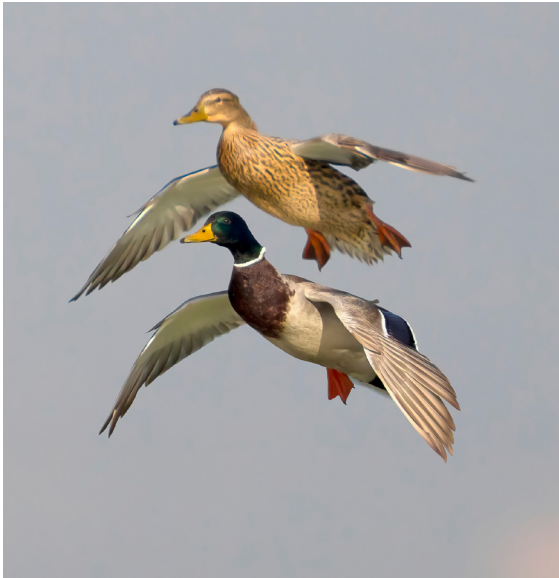


Figure A1.28. Flyway population trend for Mallard population NW Europe. For explanation see fig. A1.4.



Daniele Occhialio/Agami

Northern Pintail | *Anas acuta* | Canard pilet

Within the EAF, two populations are distinguished based on the wintering distribution, although their breeding areas likely overlap to a large extent. The NW European population considered here includes birds wintering in the Baltic and North Sea regions, the UK and Ireland and the Atlantic coast of France. The W Siberia - NE Europe breeding population winters mainly in the Mediterranean region and across Sahelian W Africa. The species is strongly migratory and breeds in shallow freshwater marshes, small lakes and rivers, preferably with dense vegetation in open country, from temperate regions in E Europe north to the Russian Arctic. In winter, the species congregates in large flocks on brackish coastal lagoons, estuaries and deltas, and on large inland lakes. Northern Pintails are omnivorous and opportunistic feeders, and their diet includes algae, seeds, tubers, vegetative parts of aquatic plants and grasses, aquatic invertebrates, amphibians and small fish.



Markus Varesvuo / Agami

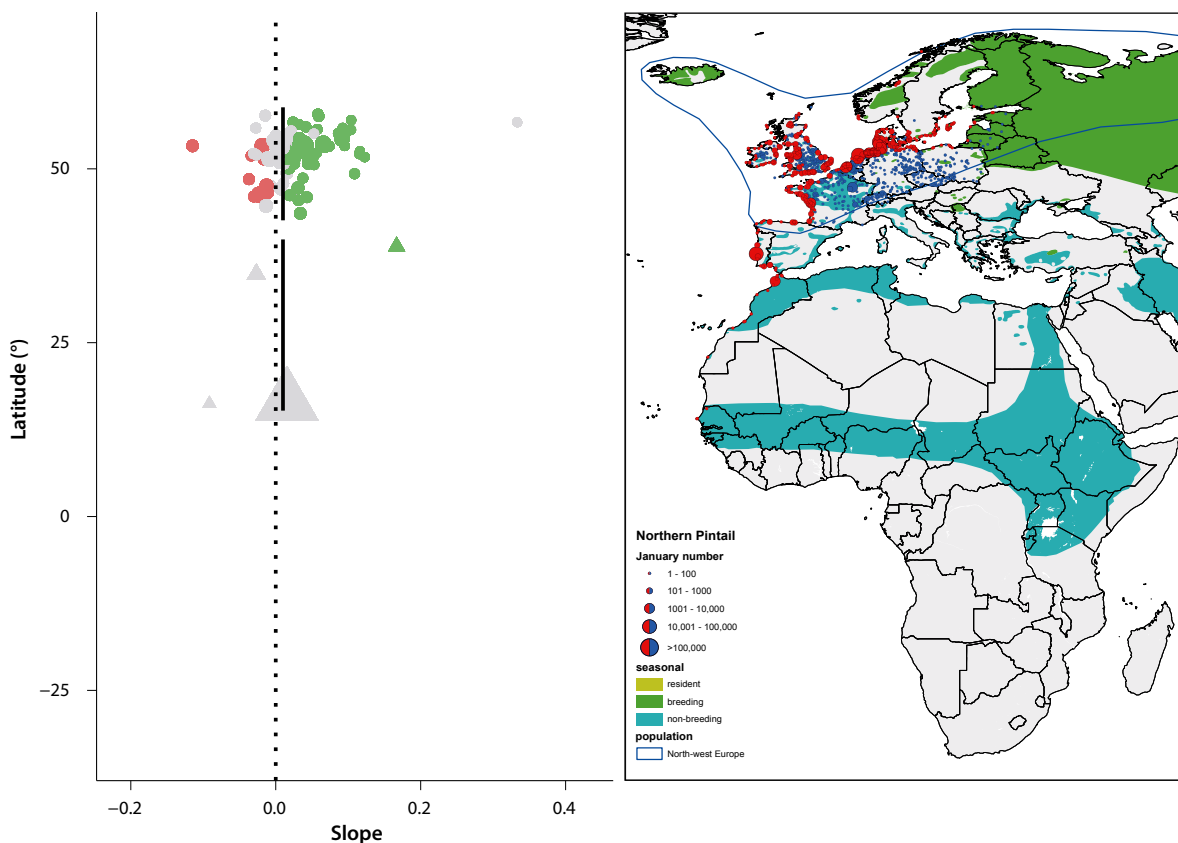


Figure A1.29. Plot (left) of site January trends and distribution (right) of Northern Pintail in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Northern Pintail	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
N Europe & Siberia/NW Europe	w	1975-2023	1,012	moderate increase	2014-2023	1,047	moderate increase	2014- 2018	74000	74000

Table A1.13. Summary of flyway trend and population size estimates for Northern Pintail. For explanation see table A1.2.

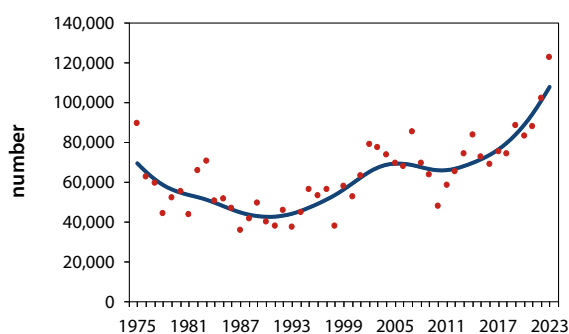


Figure A1.30. Flyway population trend for Northern Pintail population NW Europe. For explanation see fig. A1.4.



Harvey van Diek

Common Teal | *Anas crecca* | Sarcelle d'hiver

Two flyway populations are distinguished within the EAF (Scott & Rose 1996), but mainly for practical reasons. It is doubtful whether they are truly distinct as ring recoveries indicate that birds can change between flyways (Fiedler *et al.* 2005). The NW European population includes breeding birds from N Europe east to W Russia with wintering grounds in W Europe. The W Siberia - NE European breeding population includes birds breeding east to the Ural mountains and wintering in the Mediterranean region and N Africa (Fiedler *et al.* 2005). Breeding birds from northern Europe are highly migratory, while those from more temperate regions are largely sedentary. In the breeding sea-

son the species has a preference for shallow, permanent water, especially in woodland with dense herbaceous cover and with abundant emergent vegetation. In the non-breeding period, Common Teal are found in marshes, lakes and other sheltered waters with high productivity and abundant vegetation, but also along the coast in saline and brackish lagoons, deltas and saltmarshes. For foraging, marshes with mudflats are preferred over open-water habitat. In spring and summer the species feeds mainly on animal matter, such as molluscs, worms, insects and crustaceans. In winter it switches to aquatic plant seeds, grasses, sedges and agricultural seeds.



Harvey van Diek

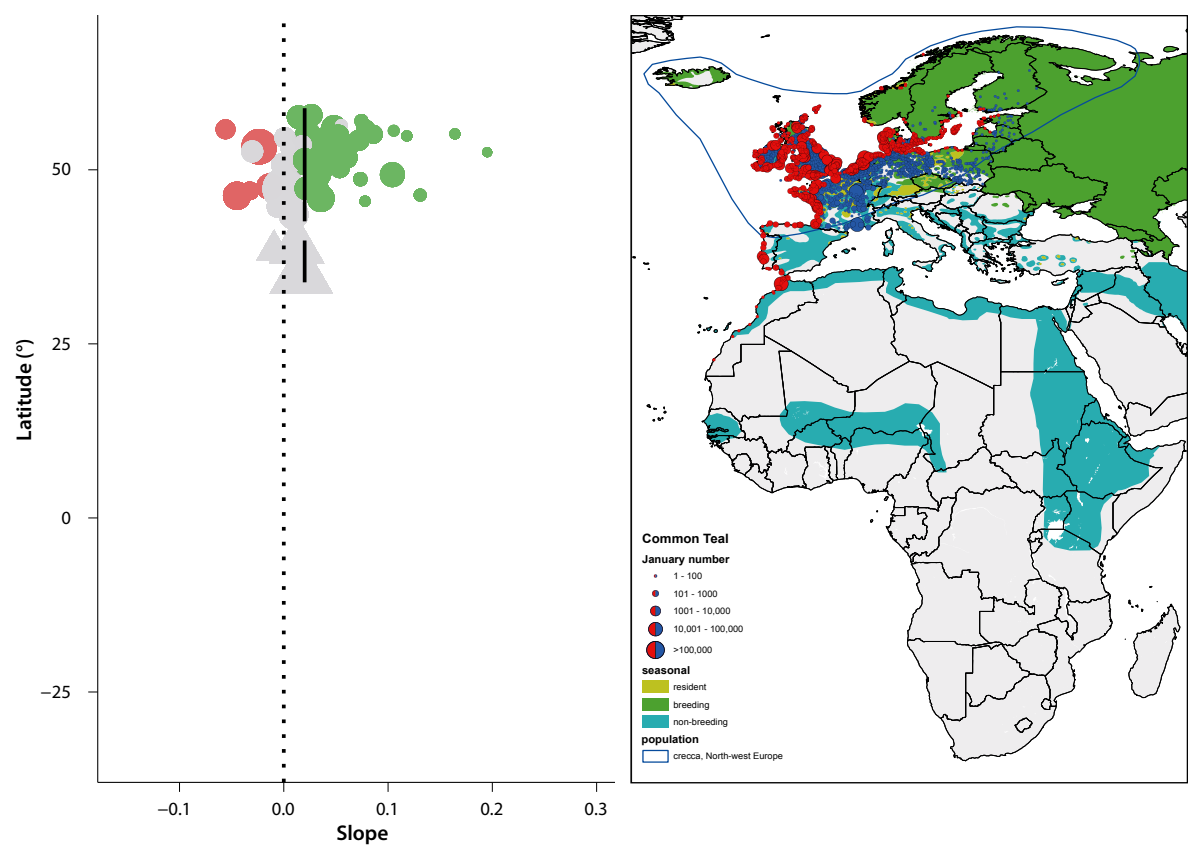


Figure A1.31. Plot (left) of site January trends and distribution (right) of Common Teal in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Common Teal	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
NW & E Europe & Siberia/NW Europe	w	1975-2023	1,020	moderate increase	2014-2023	1,012	moderate increase	2011- 2018	670000	670000

Table A1.14. Summary of flyway trend and population size estimates for Common Teal. For explanation see table A1.2.

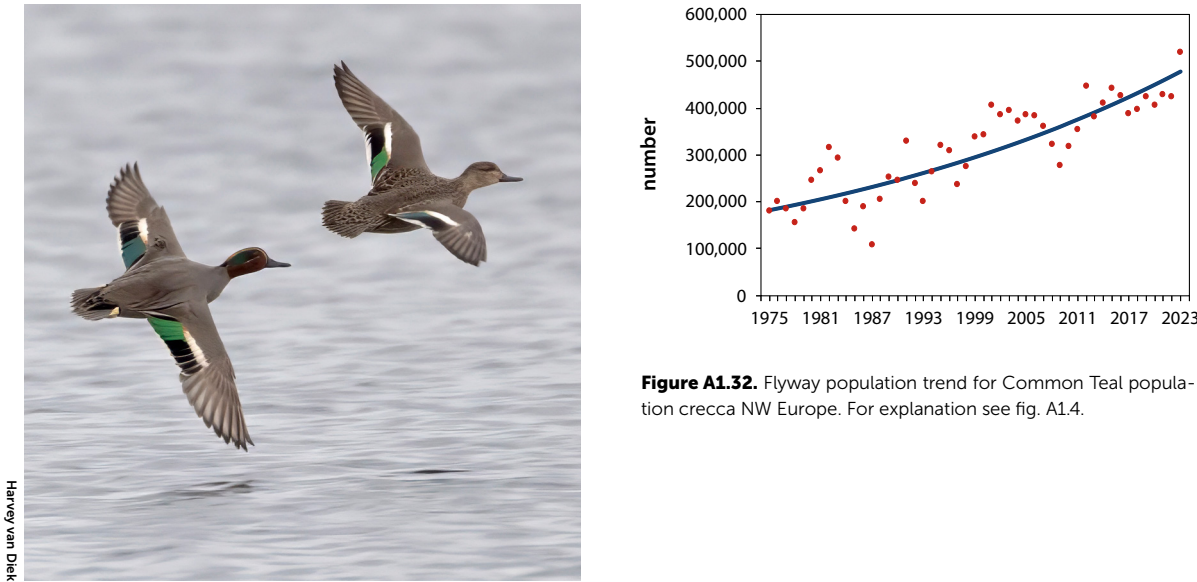


Figure A1.32. Flyway population trend for Common Teal popula-
tion crecca NW Europe. For explanation see fig. A1.4.

Harvey van Diek

Great Crested Grebe | *Podiceps cristatus* | Grèbe huppé

Two biogeographical populations of Great Crested Grebe occur in the study area - the NW and W European population of the Eurasian subspecies *P. c. cristatus* and the southern African population of the subspecies *P. c. infuscatus*. The latter occurs in scattered breeding colonies in southern Africa, as well as in E Africa. In Europe, the species breeds from W Russia and the southern half of Scandinavia south to N Africa and is migratory in the north-eastern parts of its range. In C and W Europe it is mostly sedentary, although a large part of the population moves to large open waters, including inshore coastal waters, for moulting and wintering. Congregations up to several thousand individuals can occur during the non-breeding season, although many birds remain solitary. Breeding occurs in a variety of freshwaters and brackish waters, such as pools and lakes, backwaters of slow-flowing rivers and artificial waterbodies. The diet consists mainly of small and medium-sized fish.



Harvey van Diek

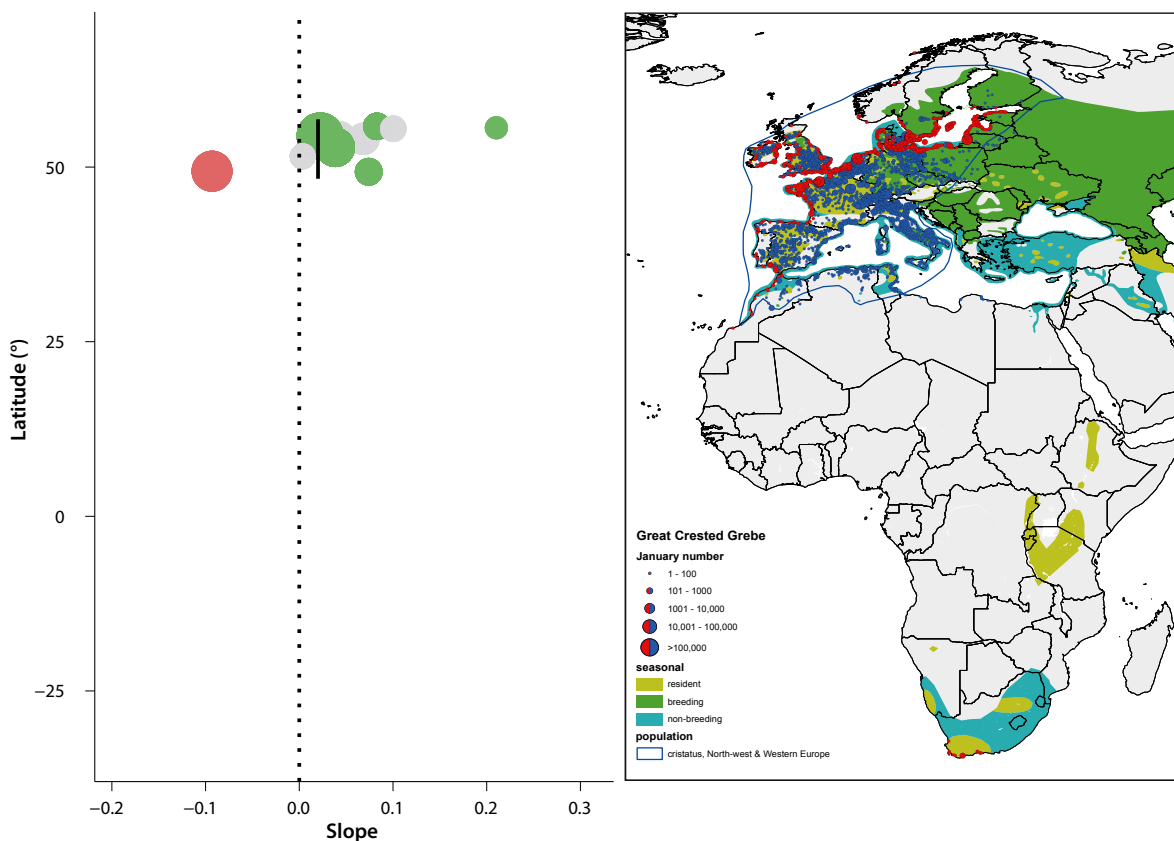


Figure A1.33. Plot (left) of site January trends and distribution (right) of Great Crested Grebe in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Great Crested Grebe	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
<i>cristatus</i> , NW Europe & W Europe	w	1976-2023	1,018	moderate increase	2014-2023	0,986	moderate decrease	2007- 2018	500000	690000

Table A1.15. Summary of flyway trend and population size estimates for Great Crested Grebe. For explanation see table A1.2.

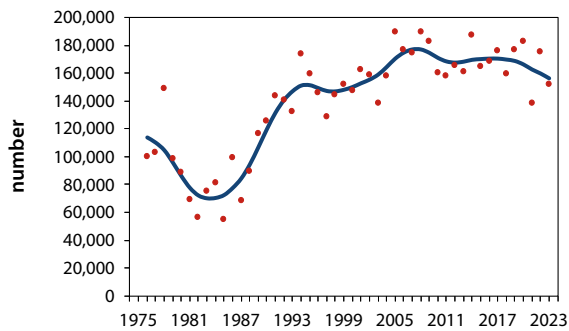


Figure A1.34. Flyway population trend for Great Crested Grebe population *cristatus* NW & W Europe. For explanation see fig. A1.4.



Harvey van Diek

Horned Grebe | *Podiceps auritus* | Grèbe esclavon

Two populations are distinguished in the EAF: one breeding in the northern Atlantic region and wintering along the Atlantic coasts of Norway, Scotland and Ireland, and another breeding from Sweden east into the boreal zone of Russia and wintering in the Baltic, Black and Mediterranean Seas and W Europe. As considerable mixing occurs likely in winter and breeding trends are lacking we present here the IWC trend for both populations combined.

Breeding occurs on small, shallow, well-vegetated fresh or brackish waters, such as pools, marshes and secluded sections of rivers and lakes in forested areas. In winter, the species is mainly coastal, visiting sheltered bays, lagoons and estuaries, but may also occur on large lakes or river systems. The diet consists of fish and a wide range of aquatic invertebrates, with fish and crustaceans forming a larger part of the diet for birds wintering at sea.



Harvey van Diek

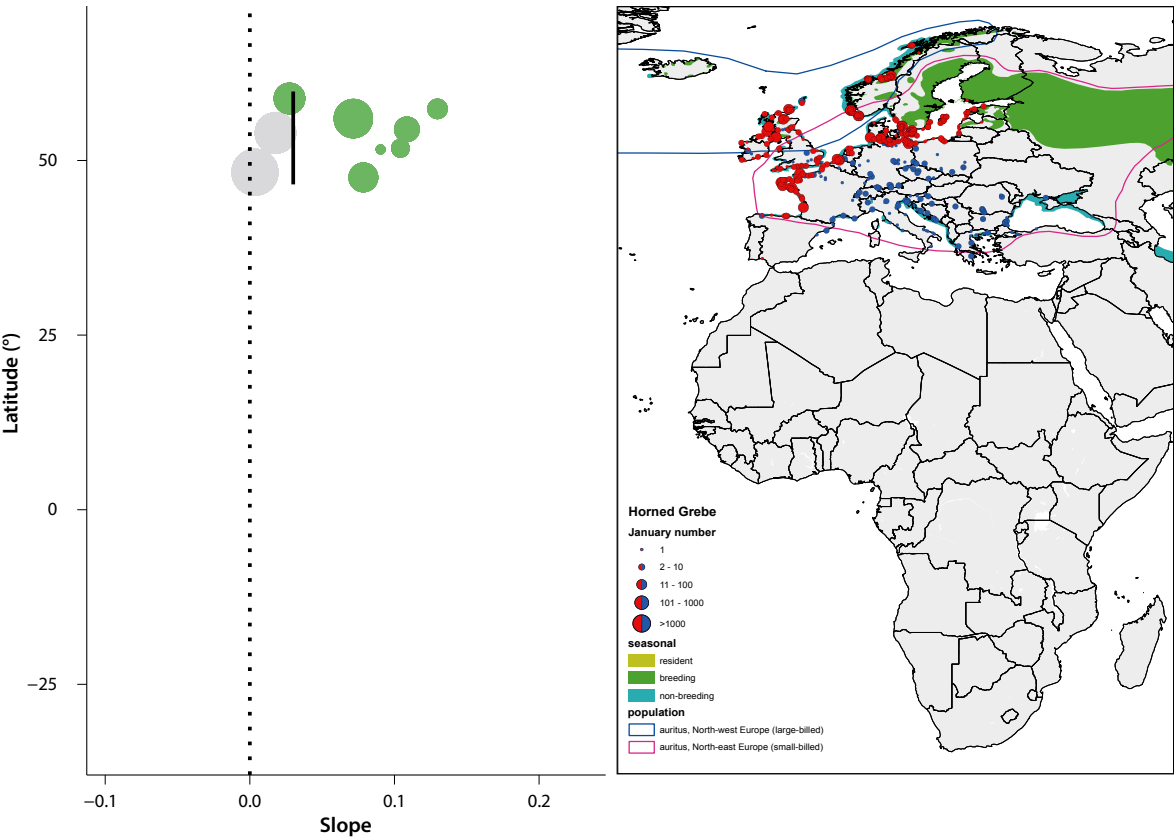


Figure A1.35. Plot (left) of site January trends and distribution (right) of Horned Grebe in the East Atlantic Flyway in January . For explanation see fig. A1.3.

population Horned Grebe	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW Europe (winter) combi of pop.	w	1980-2023	1,029	moderate increase	2014-2023	0,995	stable			
NW Europe (winter large billed)								1994- 2028	4600	5300
NW Europe (winter small billed)								2011- 2018	22000	31000

Table A1.16. Summary of flyway trend and population size estimates for Horned Grebe. For explanation see table A1.2.

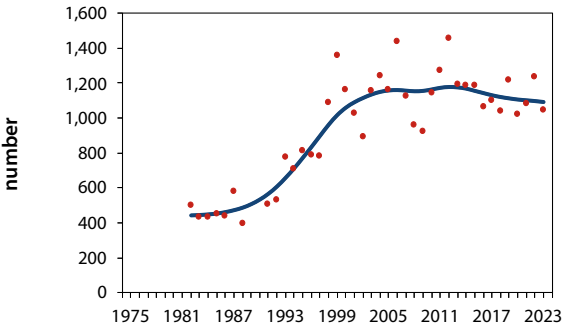


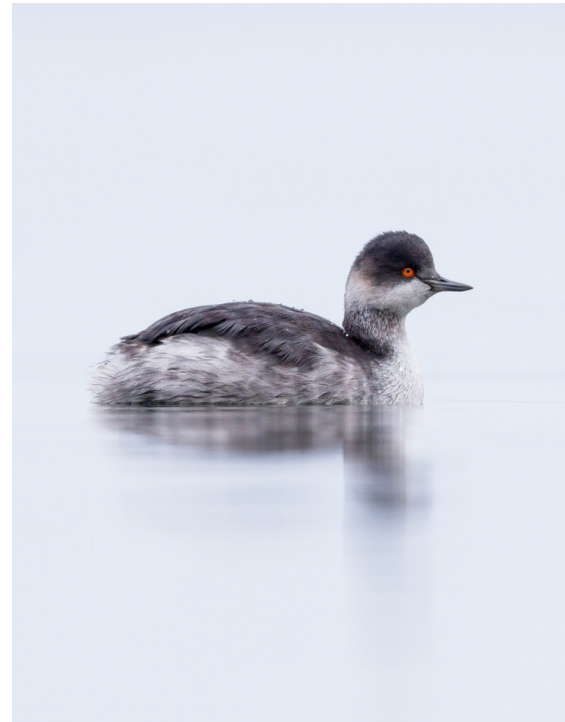
Figure A1.36. Flyway population trend for Horned Grebe popula-
tion auritus (combined). For explanation see fig. A1.4.



Harvey van Diek

Black-necked Grebe | *Podiceps nigricollis* | Grèbe à cou noir

In the northern part of the flyway, Black-necked Grebes are considered to form a single population, breeding in small or large colonies throughout Europe, but almost absent in Scandinavia. Except for some populations breeding in the far southwestern part of its range, the species is fully migratory, spending the northern winter mainly in the coastal regions of the Mediterranean Basin and W Europe. The breeding habitat consists of eutrophic, well-vegetated freshwater marshes and lakes, ponds, sewage farms, river backwaters and floodplains. In winter, the species moves to saline ponds and lakes, coastal estuaries, inshore bays and channels, where it is highly gregarious. The diet consists of aquatic insects, midges, brine-flies, molluscs, crustaceans, amphibians, worms, snails and small fish. In Southern Africa another population occurs, of the subspecies *P. n. gurneyi*, for which coastal waters in Namibia are important in the non-breeding season.



Ralph Martin / Agami

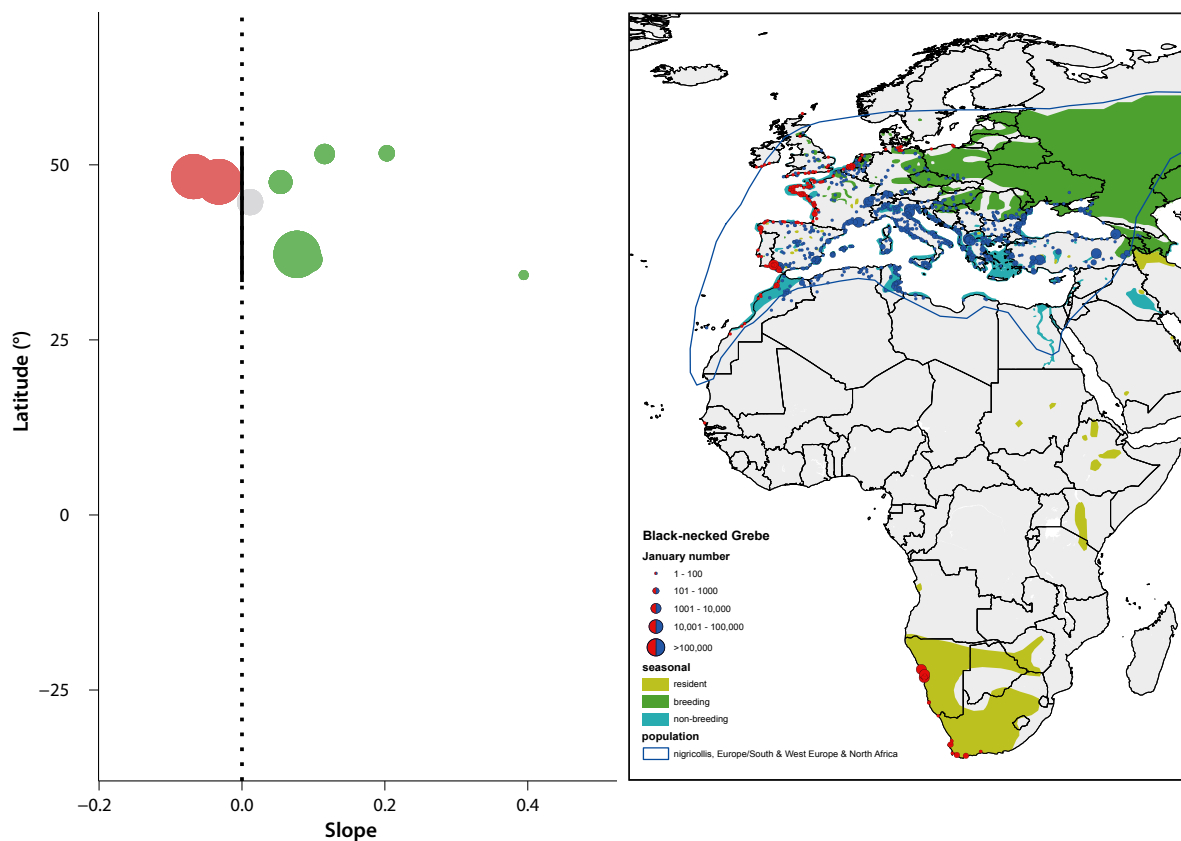


Figure A1.37. Plot (left) of site January trends and distribution (right) of Black-necked Grebe in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Black-necked Grebe	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
Europe/S & W Europe & N Africa	w	2003-2023	0,995	moderate decrease	2014-2023	0,990	stable	2007- 2018	110000	170000

Table A1.17. Summary of flyway trend and population size estimates for Black-necked Grebe. For explanation see table A1.2.

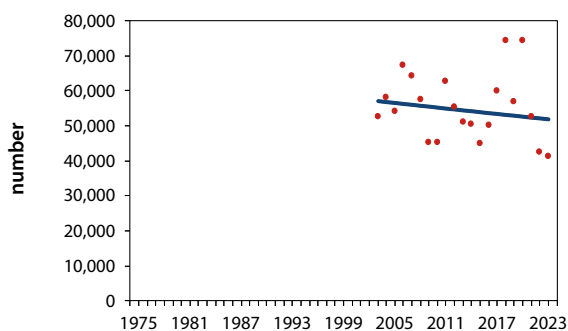


Figure A1.38. Flyway population trend for Black-necked Grebe population nigricollis Europe / S & W Europe & N Africa. For explanation see fig. A1.4.



Harvey van Diek

Greater Flamingo | *Phoenicopterus roseus* | Flamant rose

Greater Flamingo has an extensive range in southern Europe, Africa and Asia. In the coastal EAF one population is present year-round; the local breeding population of W Africa occurring from Mauritania to Sierra Leone. Limited overlap exists during the non-breeding season with the population of the W Mediterranean (Iberian Peninsula, Italy, France and parts of N Africa). Another population in

southern Africa partially makes use of coastal sites. Breeding in W Africa occurs mainly in Mauritania. Foraging occurs in shallow saline or alkaline water bodies such as lagoons, salt pans, and lakes, but also intertidal mudflat areas. It feeds on crustaceans, diatoms and other small food items, especially brine shrimp *Artemia*.



Hans Schekkerman

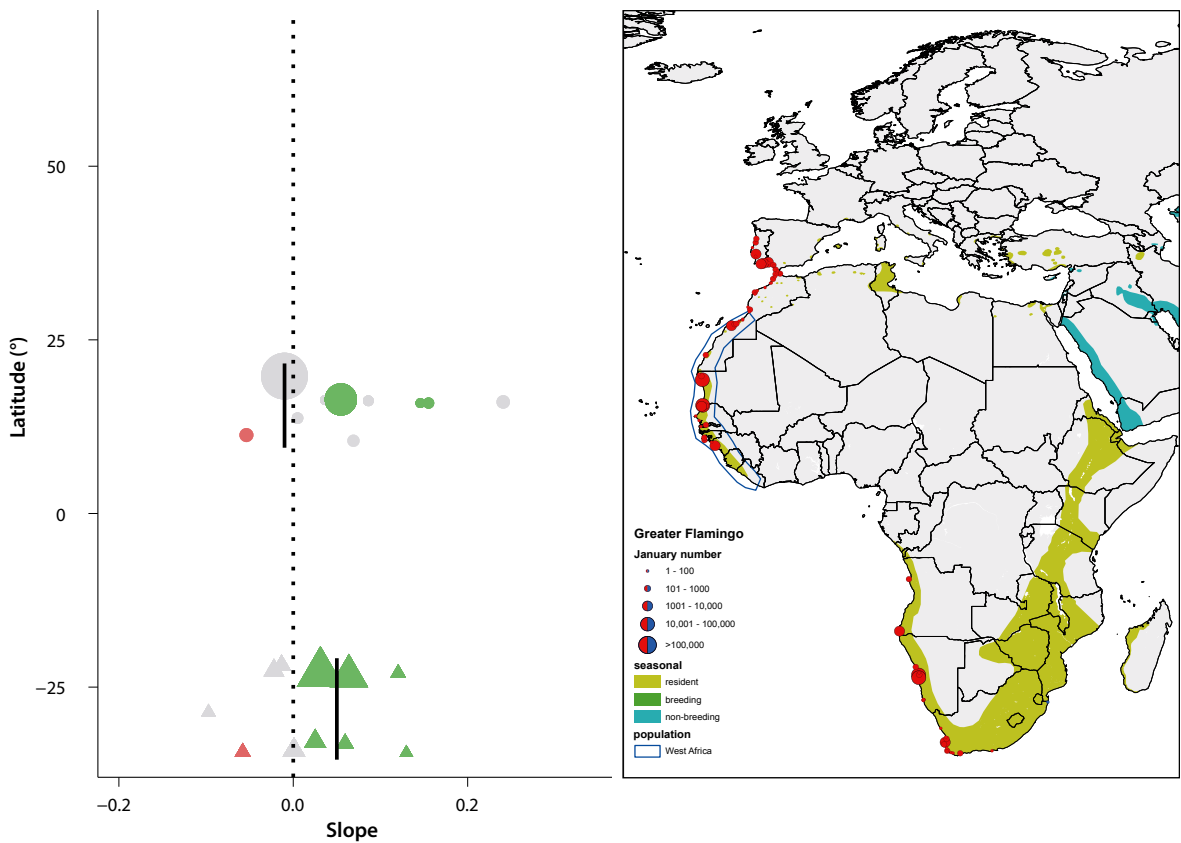


Figure A1.39. Plot (left) of site January trends and distribution (right) of Greater Flamingo in the East Atlantic Flyway in January . For explanation see fig. A1.3.

population Greater Flamingo	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Africa	w	1979-2023	0,993	moderate decrease	2014-2023	0,978	uncertain	2016- 2020	90000	110000

Table A1.18. Summary of flyway trend and population size estimates for Greater Flamingo. For explanation see table A1.2.

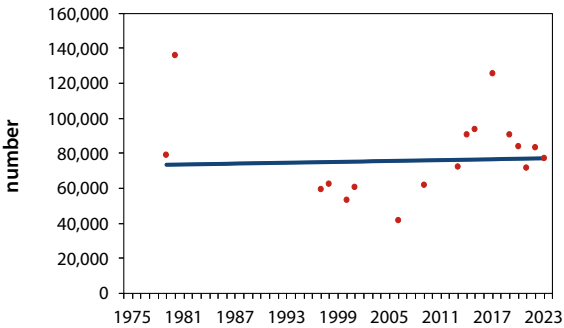


Figure A1.40. Flyway population trend for Greater Flamingo pop-
ulation W Africa. For explanation see fig. A1.4.



Harvey van Dijk

Lesser Flamingo | *Phoeniconaias minor* | Flamant nain

The Lesser Flamingo is patchily distribution throughout Africa and W Asia. In W Africa it breeds rather erratically only at one site in Mauritania. In the non-breeding season it may occur along the entire coast from Mauritania to Guinea. Further south another population ranges from the coasts of Angola and S Africa to inland areas including Botswana. The species is highly gregarious, often occurring together with the Greater Flamingo. Nesting occurs on large saline or alkaline lakes, lagoons and salt pans, and the same habitats are visited outside the breeding season.

In W Africa, Lesser Flamingos also visit estuarine waters. The species is a specialist foraging mainly on blue-green algae and diatoms in saline or alkaline waters.



Pete Morris / Agami

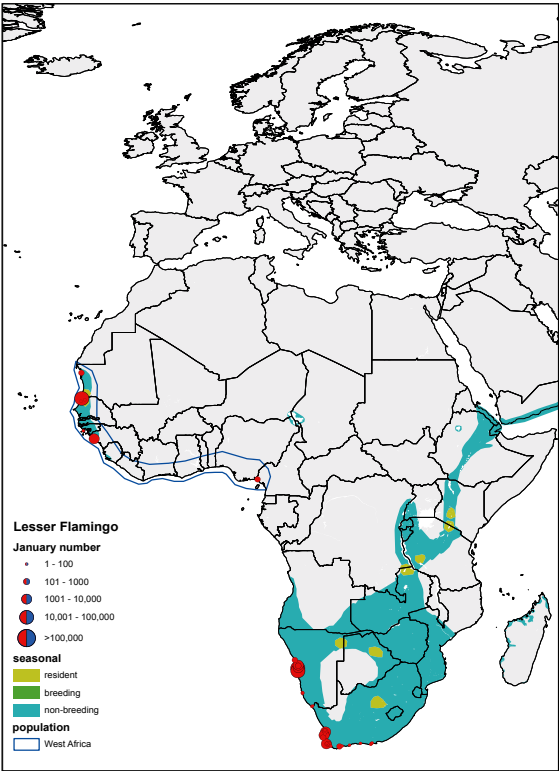


Figure A1.41. Distribution of Lesser Flamingo in the East Atlantic Flyway in January. Not enough data exits at present for site trends. For explanation see fig. A1.3.

population Lesser Flamingo	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa	w	1991-2023	1,099	uncertain	2014-2023	1,051	moderate increase	2016-2020	25000	30000

Table A1.19. Summary of flyway trend and population size estimates for Lesser Flamingo. For explanation see table A1.2.

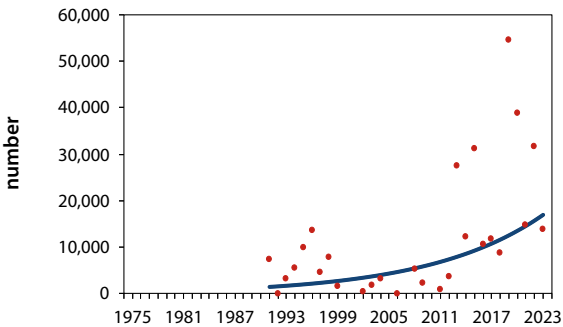


Figure A1.42. Flyway population trend for Lesser Flamingo population W Africa. For explanation see fig. A1.4.

Eurasian Spoonbill | *Platalea leucorodia* | Spatule blanche

The Eurasian Spoonbill has two populations in the EAF: a migratory population of the nominate subspecies *P. l. leucorodia* breeding in W and SW Europe and wintering in W Africa and increasingly in SW Europe (Lok *et al.* 2011), and a resident population of the subspecies *P. l. balsaci* on the Banc d'Arguin in Mauritania. The species is gregarious

all year round and breeds in colonies on the ground or in emergent vegetation (reedbeds) or in trees/shrubs. Foraging occurs mainly in shallow fresh and saltwater, usually with a mud, clay or sandy substrate, floodplains, lakes, lagoons and mudflats. Preferred food items are fish and crustaceans.



Hans Gernerad / Agami

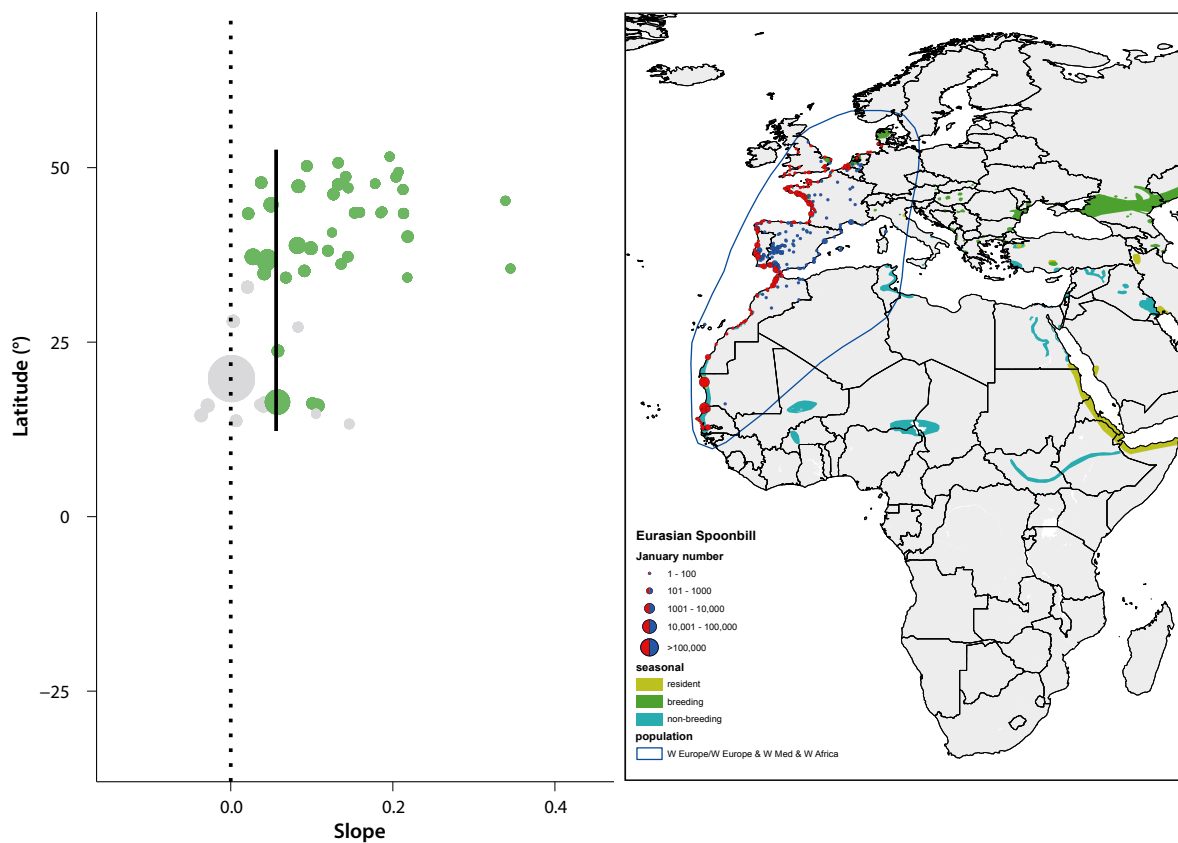


Figure A1.43. Plot (left) of January site trends and distribution (right) of Eurasian Spoonbill in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Eurasian Spoonbill	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Europe/W Europe & W Med & W Africa	b	1980-2018	1,056	strong increase	2009-2018	1,087	strong increase	2013-2018	19000	24000
<i>balsaci</i> , Coastal West Africa (Mauritania)	b	1996-2023	0,998	moderate decrease	2014-2023	1,046	moderate increase	2020	2250	2250

Table A1.20. Summary of flyway trend and population size estimates for Eurasian Spoonbill. For explanation see table A1.2.

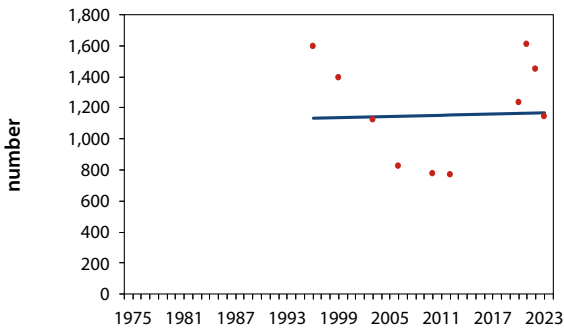


Figure A1.44. Flyway population trend for Eurasian Spoonbill *balsaci* population W Africa (Mauritania). For explanation see fig. A1.4. Number are breeding pairs



Harvey van Diek

Grey Heron | *Ardea cinerea* | Héron cendré

Palaearctic populations of this species are fully or partly migratory and disperse widely after the breeding season. Grey Herons breed in mixed colonies of up to hundreds or occasionally thousands of pairs, although it may also nest solitarily or in small groups. The species is a habitat generalist. Its diet consists predominantly of fish, as well as

amphibians, crabs, molluscs, crustaceans, aquatic insects, snakes, small rodents, small birds and plant matter. In West-Africa, predominantly in coastal Mauritania, the resident subspecies *A. c. monicae* occurs. In the African region also mixing with other African populations will occur.



Harvey van Diek

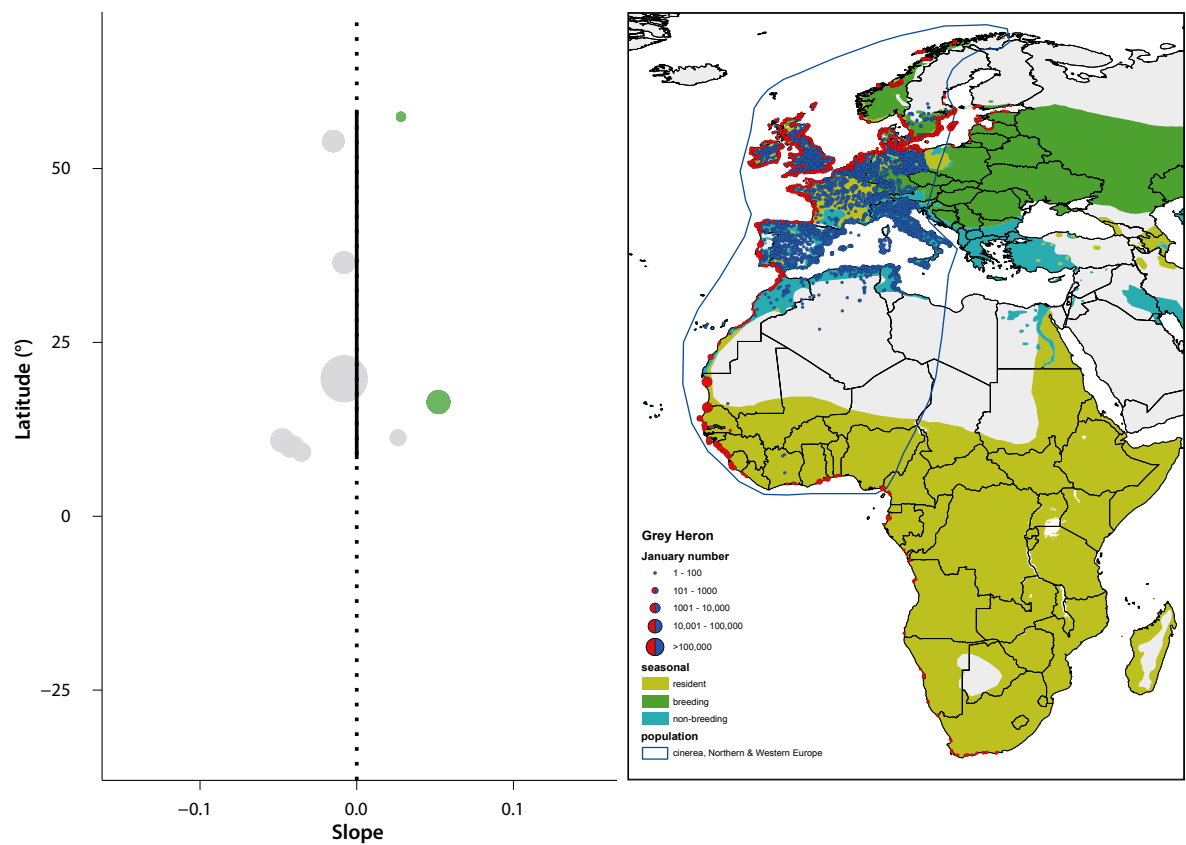


Figure A1.45. Plot (left) of January site trends and distribution (right) of Grey Heron in the East Atlantic Flyway in January . For explanation see fig. A1.3.



Harvey van Diek

population Grey Heron	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
N & W Europe	w	1990-2023	1,002	moderate increase	2014-2023	0,991	moderate decrease	2008- 2018	320000	380000

Table A1.21. Summary of flyway trend and population size estimates for Grey Heron. For explanation see table A1.2.

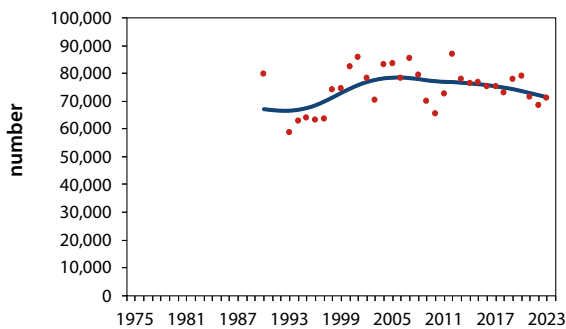


Figure A1.46. Flyway population trend for Grey Heron population *cinerea* N & W Europe (breeding). For explanation see fig. A1.4.



Harvey van Diek

Little Egret | *Egretta garzetta* | Aigrette garzette

Little Egrets breeding in the Palearctic are highly migratory. In the African region mixing with other African populations will occur. The species usually nests in colonies, sometimes of thousands of pairs and often with other species. Some populations also breed solitarily or in small single-species groups of under 100 pairs. This species commonly feeds solitarily or in loose flocks during the day and

can be found in a range of habitats, such as fresh, brackish or saline wetlands and shows a preference for shallow waters. Some populations are almost entirely coastal, inhabiting rocky or sandy shores, reefs, estuaries, mudflats, saltmarshes, mangroves and tidal creeks. Little Egrets have a broad diet and forage on small fish, aquatic and terrestrial insects, crustaceans, as well as amphibians, molluscs.



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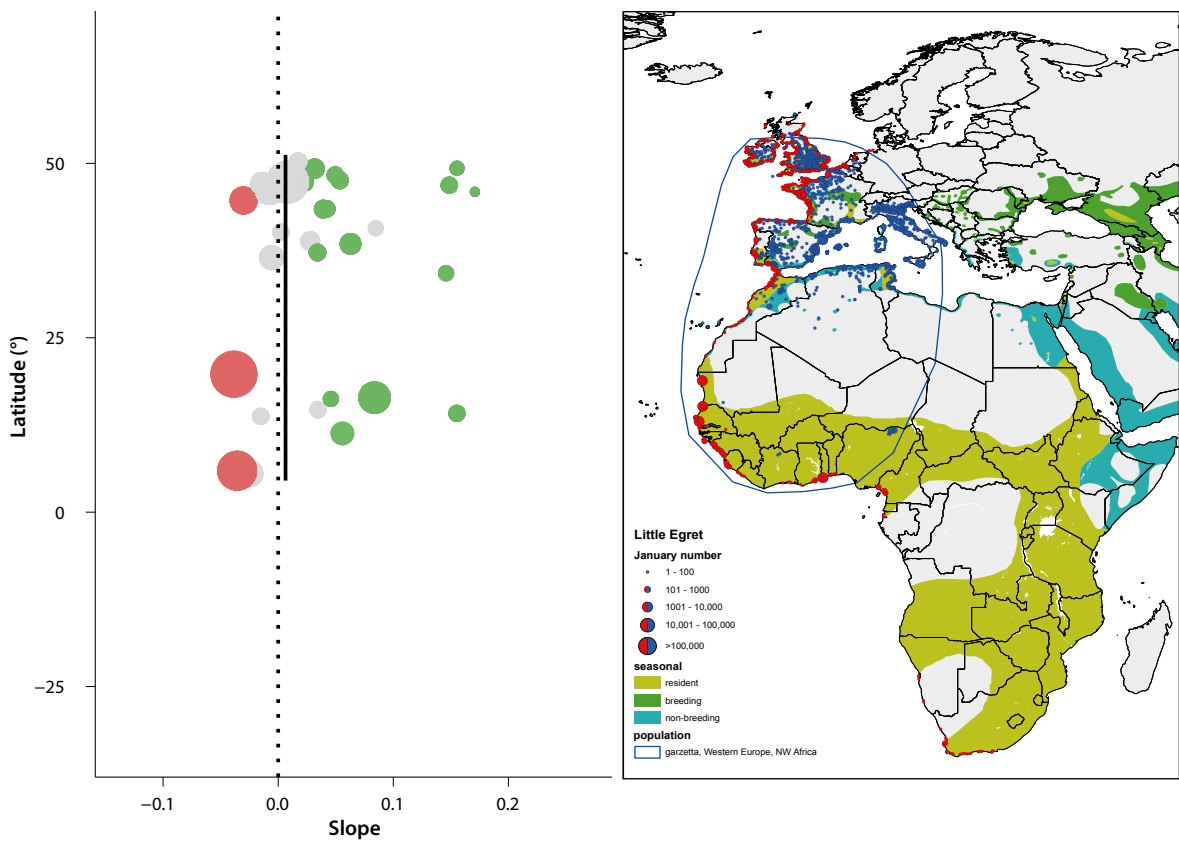


Figure A1.47. Plot (left) of January site trends and distribution (right) of Little Egret in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Little Egret	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
SW Europe	b	2005-2023	1,006	stable	2014-2023	0,996	stable	2003-2018	95000	105000

Table A1.22. Summary of flyway trend and population size estimates for Little Egret. For explanation see table A1.2.

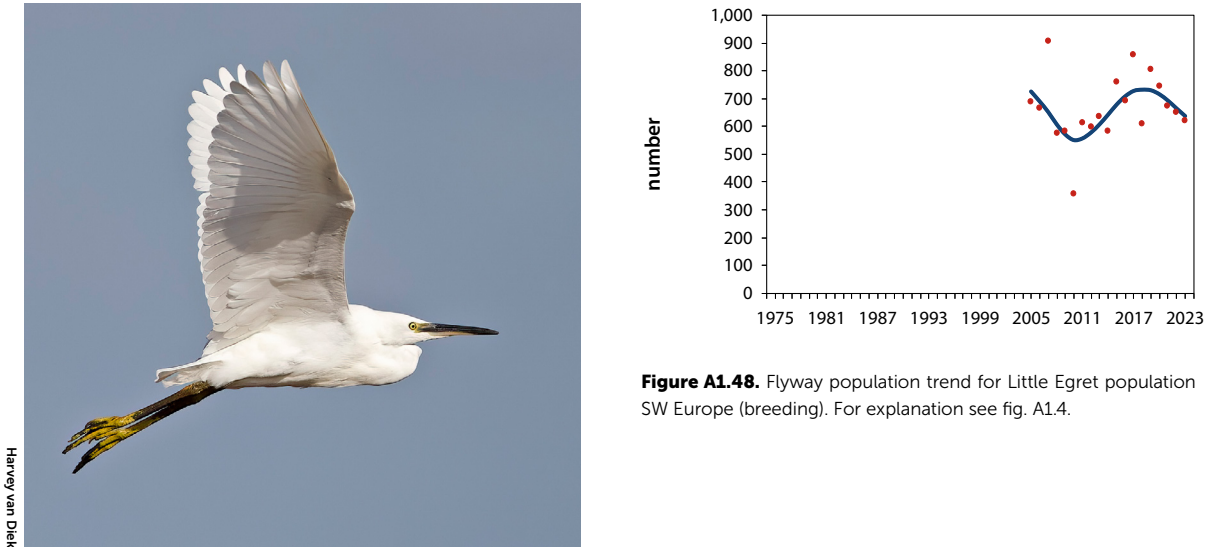


Figure A1.48. Flyway population trend for Little Egret population SW Europe (breeding). For explanation see fig. A1.4.

Western Reef-egret | *Egretta gularis* | Aigrette à gorge blanche

The Western Reef-egret population of the subspecies *E. g. gularis* is confined to W Africa, where it occurs along the entire coastline and at some inland sites from Morocco to Gabon. The preferred foraging sites are small pools in

mudflat areas, sandy or rocky shores and reefs. It nests on the ground, in mangrove trees or in reedbeds, either solitarily or in small colonies. The food is variable: fish, crustaceans, worms and other invertebrates.



Harvey van Diek

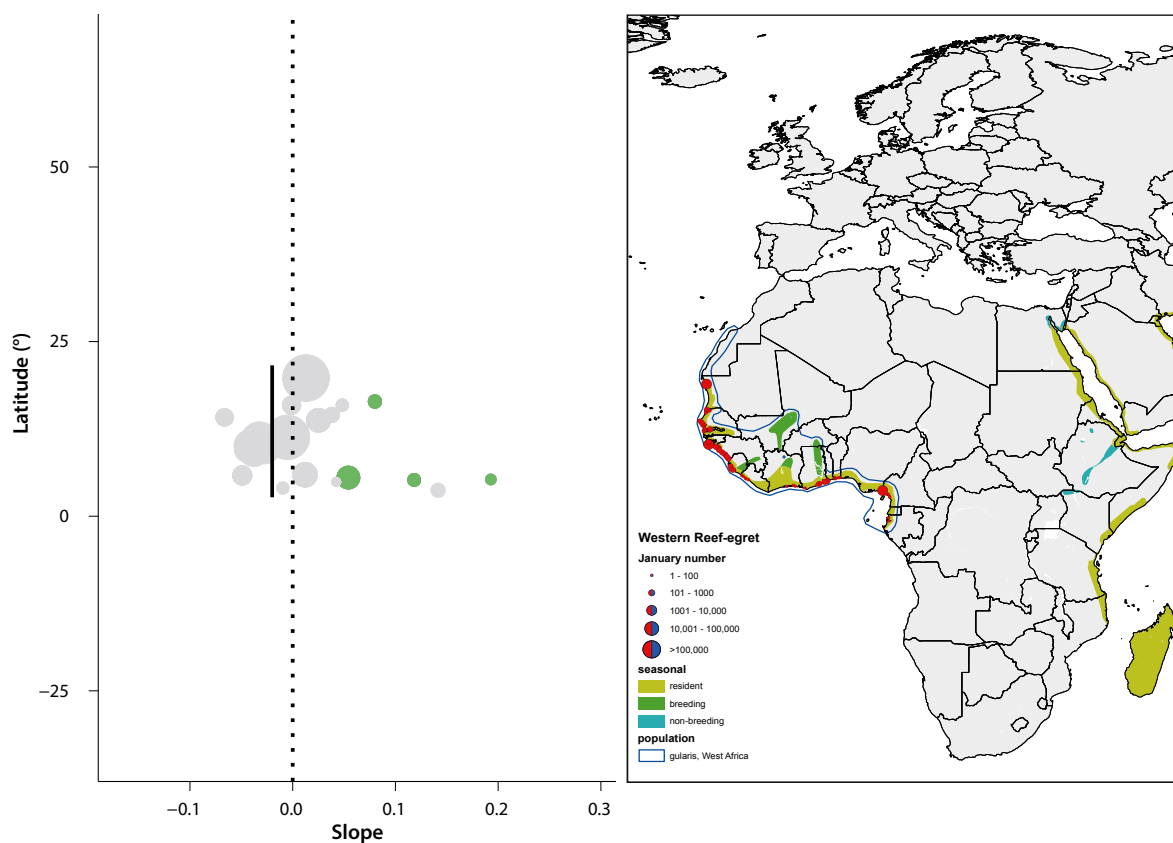


Figure A1.49. Plot (left) of January site trends and distribution (right) of Western Reef-egret in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Western Reef-Egret	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
<i>gularis</i> , W Africa	w	1990-2023	0,985	uncertain	2014-2023	0,957	moderate decrease	2016- 2020	25000	35000

Table A1.23. Summary of flyway trend and population size estimates for Western Reef-egret. For explanation see table A1.2.

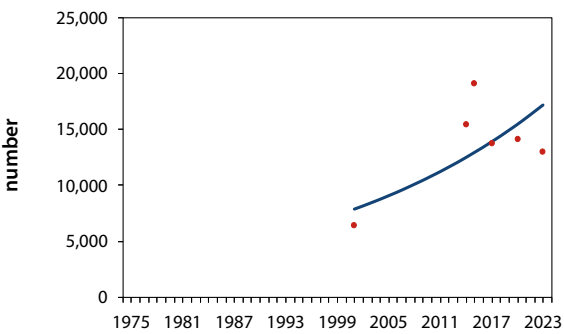


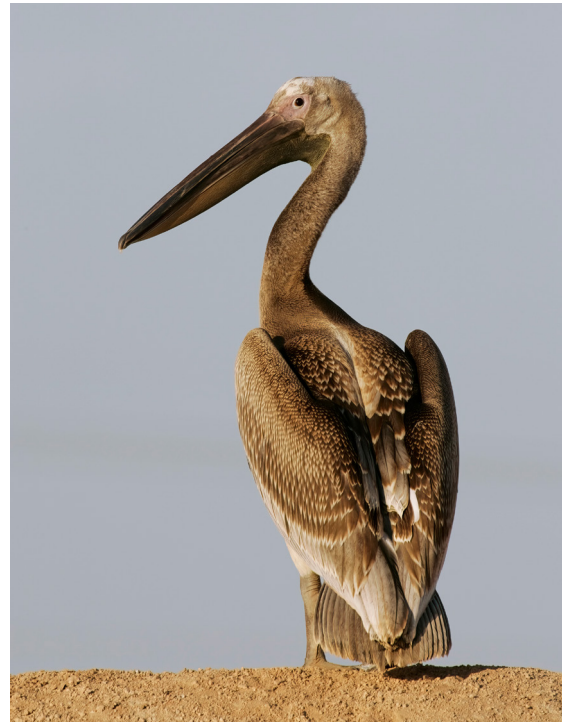
Figure A1.50. Flyway population trend for Western Reef-egret population *gularis* W Africa. For explanation see fig. A1.4.



Harvey van Diek

Great White Pelican | *Pelecanus onocrotalus* | Pélican blanc

Within the study region two biogeographical populations occur: one in coastal W Africa and the Sahelian floodplains east to Chad, and one in southern Africa. Great White Pelicans are large fish-eating colonial breeding birds of which the populations within the study area are largely resident or partly migratory and nomadic. The limits of the ranges of different populations are not well known. The birds in Guinea and Sierra Leone, and perhaps also those of Nigeria, most likely belong to the West African population which is considered here. The pelicans in coastal Cameroon and Gabon can also be of W African or southern African origin. The exact limits towards the east are even less clear. The species feeds in coastal creeks, estuaries, floodplain and other inland shallow lakes. The preferred breeding sites are swamps and sandbanks that are secure from disturbance by humans and natural predators. The largest breeding site within the EAF is at Parc National des Oiseaux du Djoudj in Senegal.



Daniele Occhiato / Agami

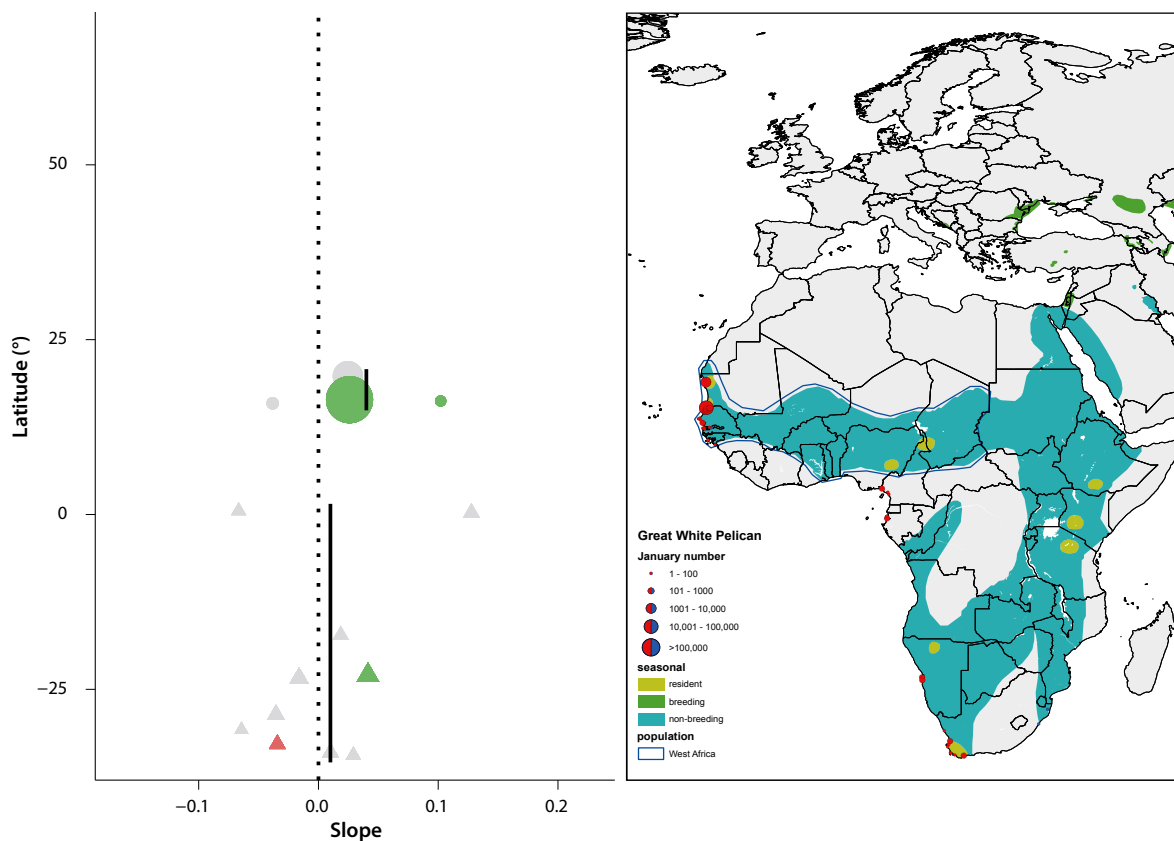


Figure A1.51. Plot (left) of January site trends and distribution (right) of Great White Pelican in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Great White Pelican	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa	w	1991-2023	1,036	moderate increase	2014-2023	0,999	stable	2016- 2020	35000	45000

Table A1.24. Summary of flyway trend and population size estimates for Great White Pelican. For explanation see table A1.2.



Hans Schelkerman

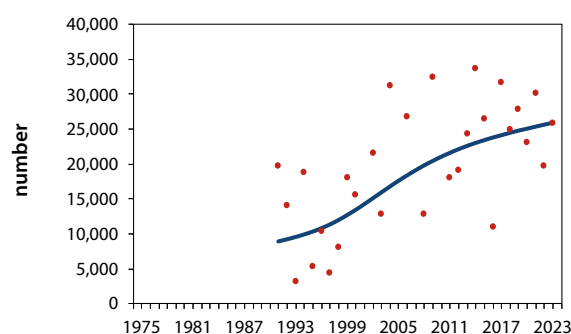


Figure A1.52. Flyway population trend for Great White Pelican population W Africa. For explanation see fig. A1.4.

Great Cormorant | *Phalacrocorax carbo* | Grand Cormoran

The Great Cormorant is found in many parts of Eurasia and Africa. In the EAF several flyway populations occur. The sub-species *P. c. carbo* occurs mainly along rocky shores of NW Europe, and is not considered in this report. The *P. c. sinensis* subspecies occurs mainly in continental Europe, breeding in N and W Europe and wintering in W Europe and the SW Mediterranean. In the African sub-species *P. c. lucidus* two populations are distinguished in the study region: one in W Africa from Mauritania to Sierra Leone, and one in southern Africa. The endemic subspecies *P. c. maroccanus* is confined to rocky coasts in Morocco and not considered in this report. The species occurs in freshwater and marine habitats, is gregarious year-round (in both colonies and feeding flocks) and is a piscivore foraging in shallow coastal waters or freshwater lakes. Its diet consists of benthic and benthic-pelagic fish species varying in size from 10 to 20 cm. Breeding sites vary from trees to bare ground in (mixed) colonies. It is capable of performing foraging flights up to 25 km or more from a nesting colony.



Arnold Meijer / Blue Robin

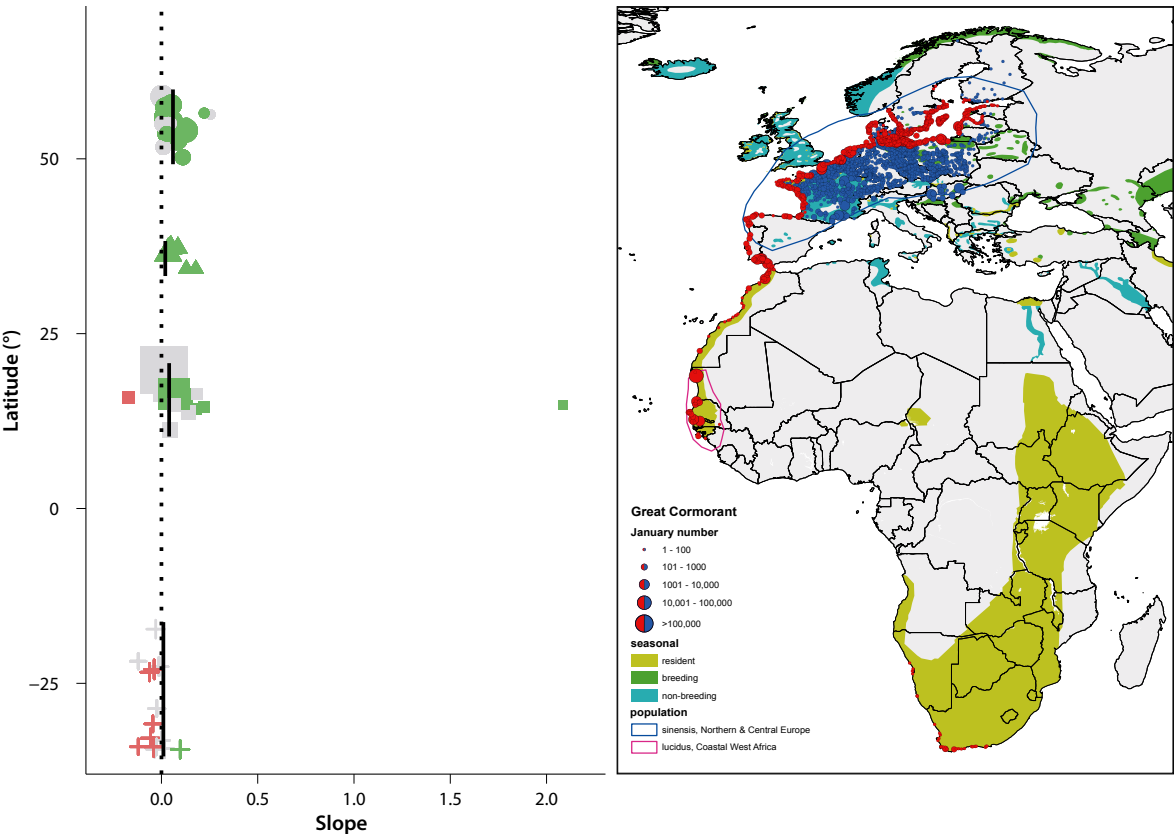


Figure A1.53. Plot (left) of January site trends and distribution (right) of Great Cormorant in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Great Cormorant	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
<i>sinensis</i> , N & C Europe	w	1976-2023	1,064	strong increase	2014-2023	1,046	moderate increase	2005- 2018	610000	740000
<i>lucidus</i> , coastal W Africa	w	2000-2023	1,039	moderate increase	2014-2023	1,075	moderate increase	2016- 2020	35000	45000

Table A1.25. Summary of flyway trend and population size estimates for Great Cormorant. For explanation see table A1.2.

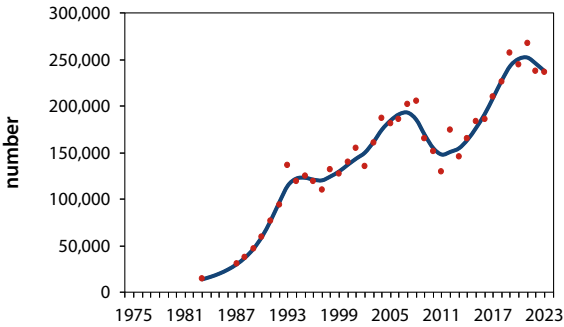


Figure A1.54. Flyway population trend for Great Cormorant population *sinensis* N & C Europe. For explanation see fig. A1.4.

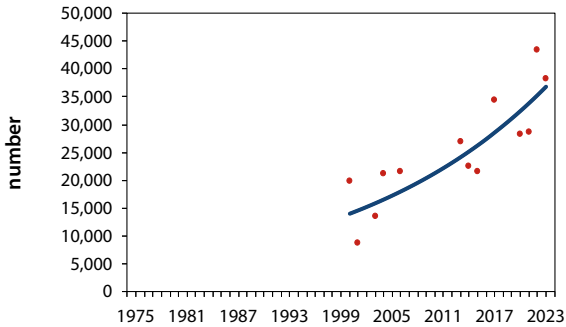


Figure A1.55. Flyway population trend for Great Cormorant population *lucidus* coastal W Africa. For explanation see fig. A1.4.

Cape Cormorant | *Phalacrocorax capensis* | Cormoran du Cap

The distribution of this species is limited to the coasts of Angola, Namibia and South Africa, with few breeding colonies but extensive post-breeding dispersive movements along the coast (Crawford *et al.* 2007). Breeding occurs in large colonies of up to 120,000 individuals on cliffs and ledges on the mainland and on offshore islands. In the

non-breeding season the species can also be found in coastal lagoons, estuaries and harbours. Its distribution and breeding activity is highly dependent on food resources, which consist almost entirely of pelagic schooling fish, including mainly pilchard *Sardinops ocellata* and anchovy *Engraulis capensis*.



WIL Leurs / Agami

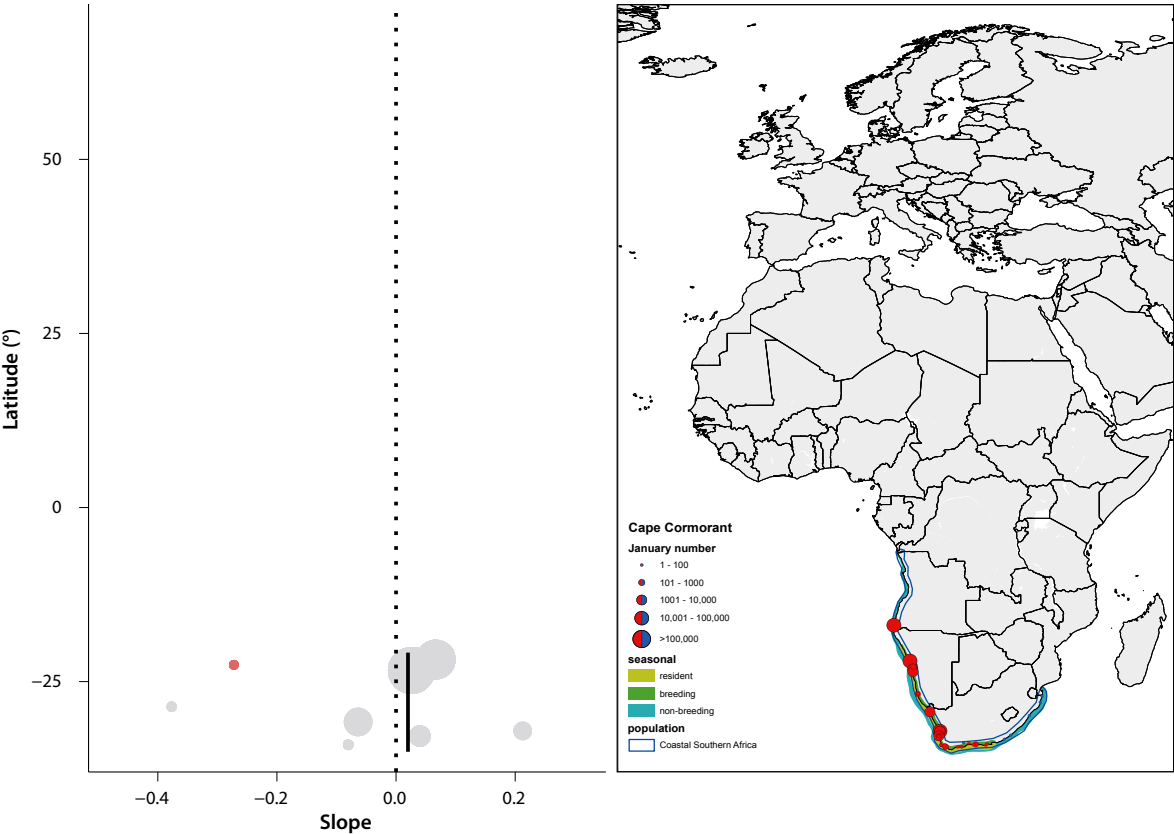


Figure A1.56. Plot (left) of January site trends and distribution (right) of Cape Cormorant in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Cape Cormorant	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
coastal S Africa	w	1992-2023	1,016	stable	2014-2023	0,883	moderate decrease	2005- 2014	351000	351000

Table A1.26. Summary of flyway trend and population size estimates for Cape Cormorant. For explanation see table A1.2.

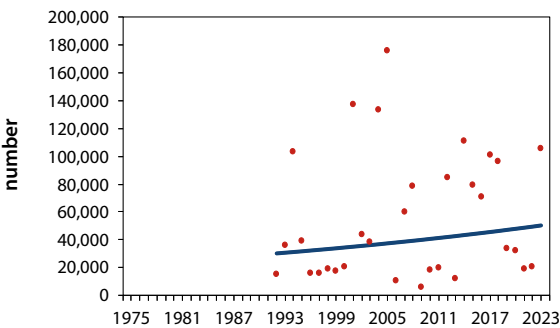


Figure A1.57. Flyway population trend for Cape Cormorant population coastal Southern Africa. For explanation see fig. A1.4.



Marc Guyt / Agami

African Oystercatcher | *Haematopus moquini* | Huitrier de Moquin

The African Oystercatcher is a species with a limited range, occurring only on the coasts of Namibia and South Africa (Leseberg *et al.* 2000). It occurs along rocky and sandy coasts, either along the shoreline or in estuaries. The adults are largely sedentary with only limited movements outside the breeding season but young birds move rela-

tively long distances. Preferred breeding sites are rocky islands and sandy beaches. The species forages year-round in the intertidal zone and feeds primarily on bivalves. Within the breeding season, the species is solitary, and outside the breeding season small groups of up to a few hundred individuals can be found.

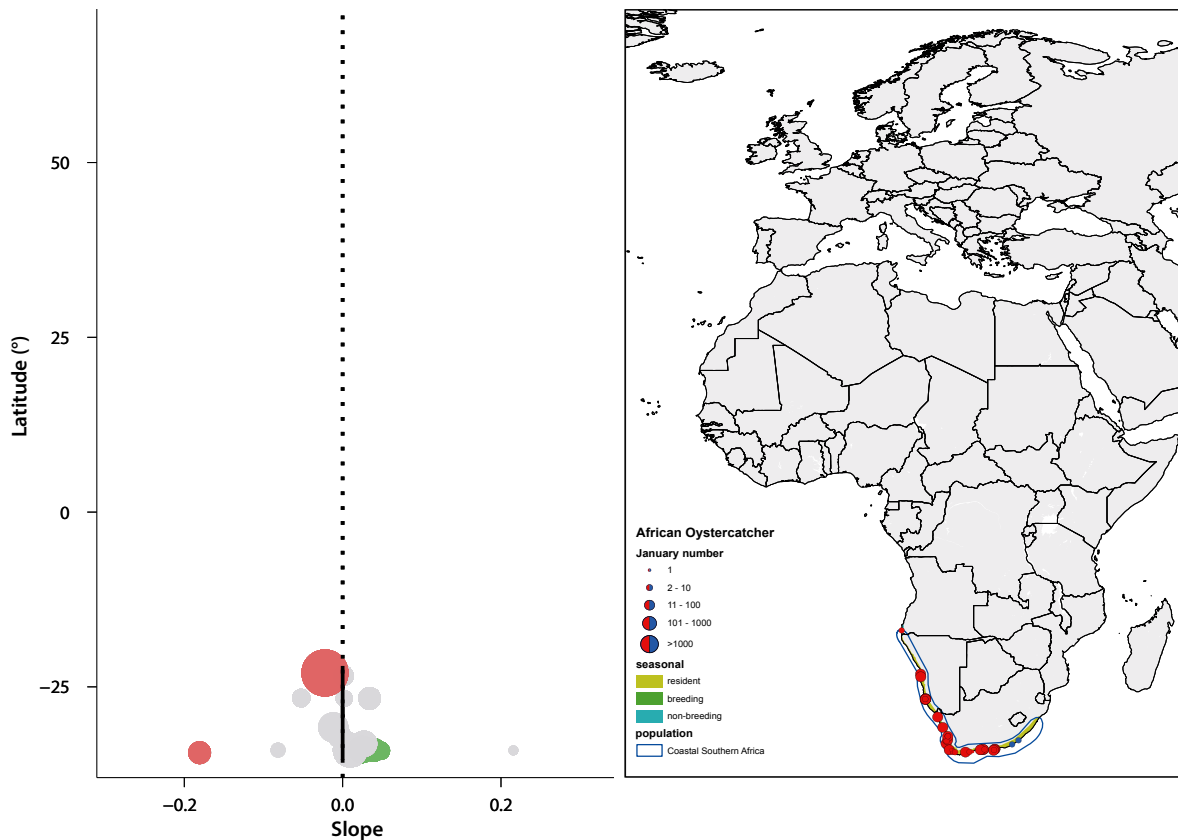


Figure A1.58. Plot (left) of January site trends and distribution (right) of African Oystercatcher in the East Atlantic Flyway in January. For explanation see fig. A1.3.



population African Oystercatcher	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
coastal S Africa	w	1993-2023	0,995	stable	2014-2023	1,020	uncertain	1997-2003	6600	6700

Table A1.27. Summary of flyway trend and population size estimates for African Oystercatcher. For explanation see table A1.2.

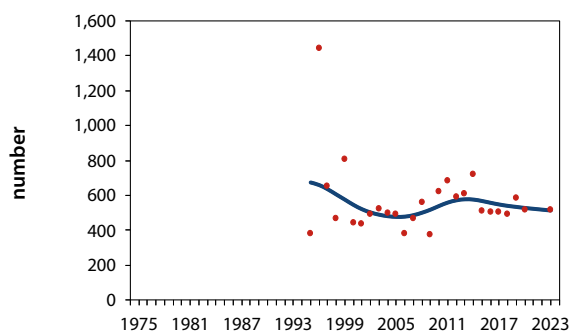


Figure A1.59. Flyway population trend for Africa Oystercatcher population coastal Southern Africa. For explanation see fig. A1.4.



Wim Leurs / Agami

Eurasian Oystercatcher | *Haematopus ostralegus* | Huïtrier pie

One flyway population of the Eurasian Oystercatcher occurs within the EAF, the nominate subspecies *H. o. ostralegus*. The largest breeding numbers occur in the countries around the North Sea (UK, The Netherlands and Germany) and in Scandinavia and Iceland. Further south in Europe, breeding populations are small and dispersed. Most birds are migratory, some over small distances, others over much larger distances (N Europe to NW Africa). The Eurasian Oystercatcher typically breeds in coastal habitats (dunes, saltmarshes, rocky shores, sand beaches), but also occurs inland along lakes and rivers and in farmland, both arable and grassland. Small numbers even occur in urban habitats, breeding on flat roofs. Breeding occurs solitarily but densities in suitable habitats can be quite high. Outside the breeding season the species is highly gregarious and roosts and forages in large flocks, congregating mainly on estuarine mudflats and saltmarshes. The preferred food is either bivalves and intertidal worms in estuarine situations or earthworms and insect larvae in farmland areas (van de Pol *et al.* 2009).



Harvey van Diek

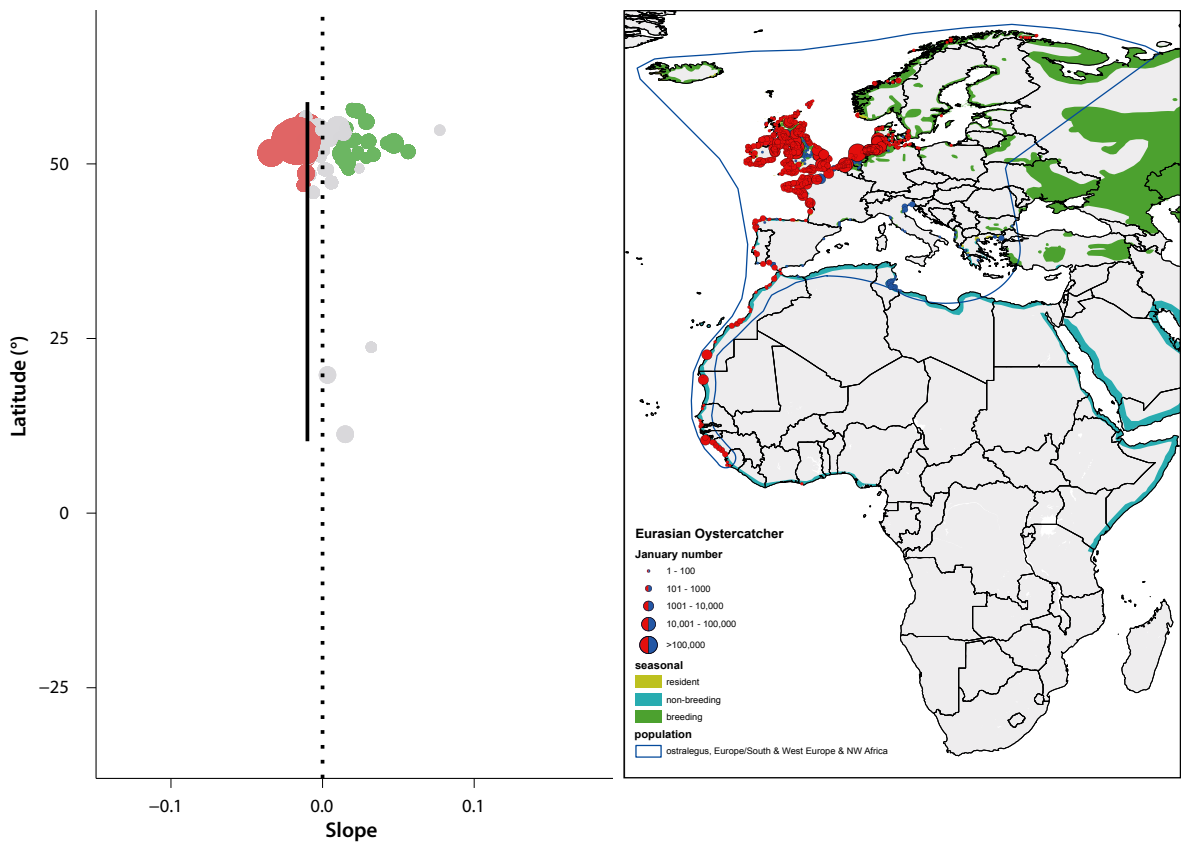


Figure A1.60. Plot (left) of January site trends and distribution (right) of Eurasian Oystercatcher in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Eurasian Oystercatcher	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
Europe/S. & W Europe & NW Africa	w	1976-2023	0,993	moderate decrease	2014-2023	0,990	moderate decrease	2007- 2018	750000	970000

Table A1.28. Summary of flyway trend and population size estimates for Eurasian Oystercatcher. For explanation see table A1.2.

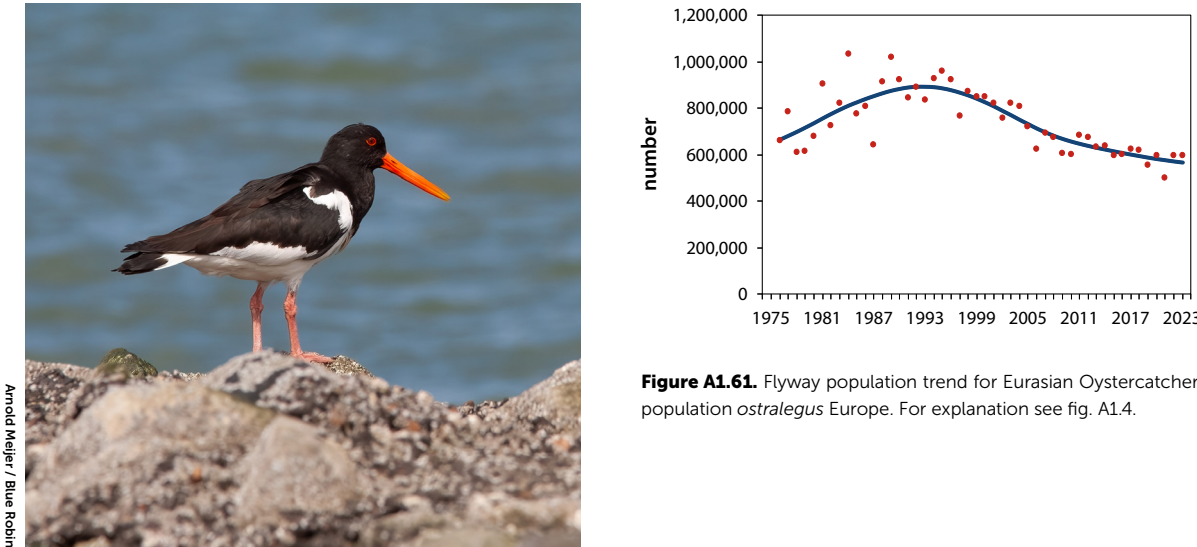


Figure A1.61. Flyway population trend for Eurasian Oystercatcher population *ostralegus* Europe. For explanation see fig. A1.4.

Pied Avocet | *Recurvirostra avosetta* | Avocette élégante

Pied Avocet breeds in many parts of W and S Europe and Southern Africa. It is a highly migratory species. Within the EAF, three populations occur of which two are considered here. The breeding birds of W Europe and NW Africa (mainly Denmark, Germany, The Netherlands, France and Spain) migrate approximately as far south as Sierra Leone in W Africa. A second population in Southern Africa uses many inland sites as well as some sites on the Atlantic coast. In between, mainly in the Gulf of Guinea, Avocets spending the non-breeding season mainly originate from breeding populations in the Mediterranean and SE Europe and the Mediterranean and Black Seas; a larger part of this flyway population winters outside the EAF. The species is gregarious year-round and it breeds in loose colonies and usually migrates and winters in large flocks. Breeding occurs in sparsely vegetated sites in saline and brackish wetlands. Outside the breeding season, the species occurs on coastal mudflats, lagoons and estuaries. Pied Avocets feed on a wide variety of items such as oligochaete and polychaete worms, crustaceans, small fish and aquatic insects, which they find in shallow water.



Arnold Meijer / Agami

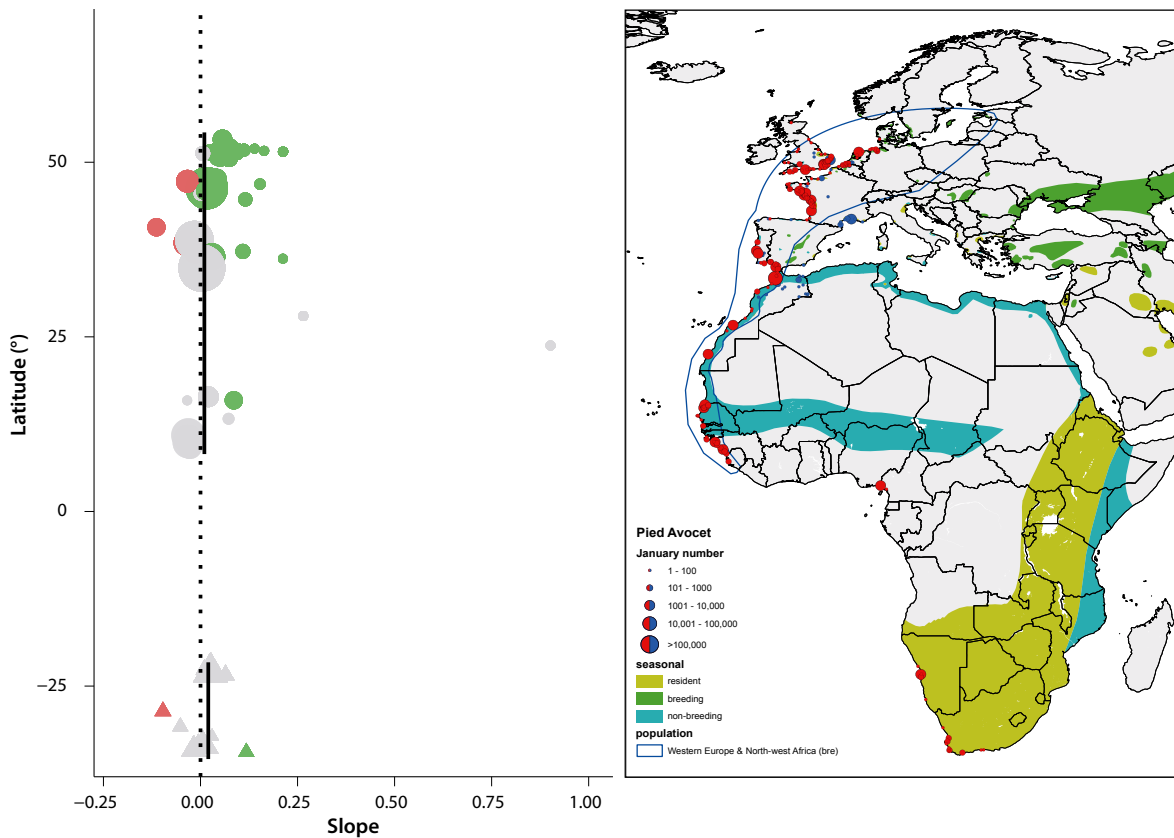
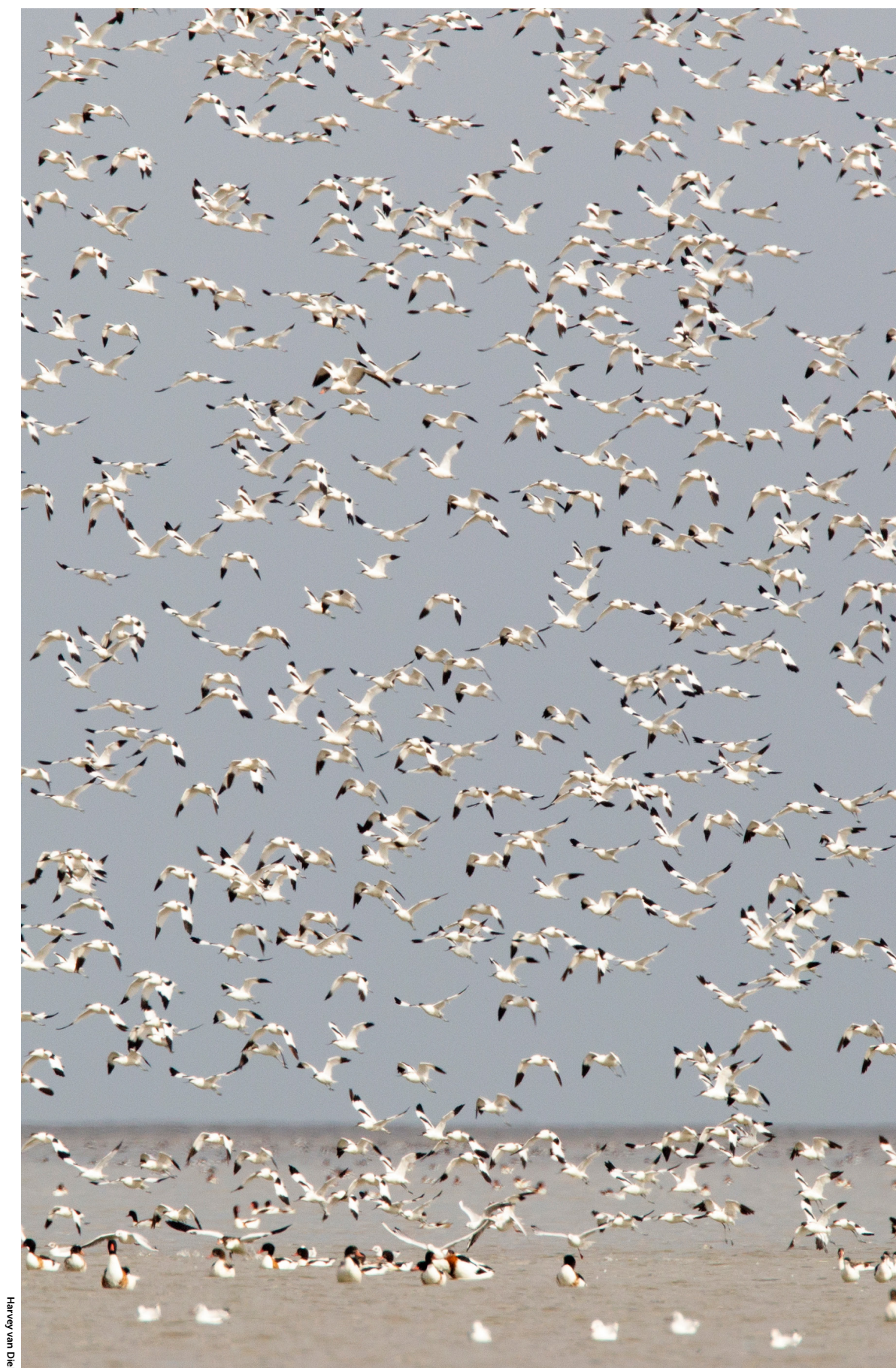


Figure A1.62. Plot (left) of January site trends and distribution (right) of Pied Avocet in the East Atlantic Flyway in January. For explanation see fig. A1.3.



Harvey van Diek

population Pied Avocet	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Europe & NW Africa (breeding)	w	1978-2023	1,012	moderate increase	2014-2023	0,961	moderate decrease	2007- 2018	100000	110000

Table A1.29. Summary of flyway trend and population size estimates for Pied Avocet. For explanation see table A1.2.

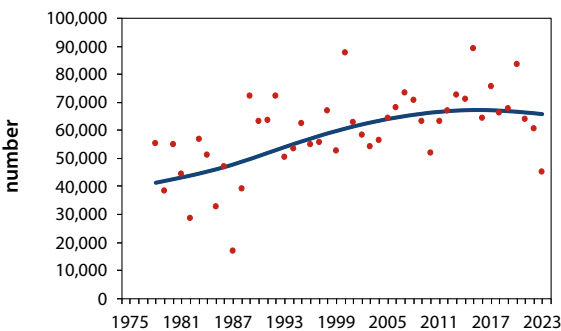


Figure A1.63. Flyway population trend for Pied Avocet population W Europe & NW Africa (breeding). For explanation see fig. A1.4.



Harvey van Diek

Grey Plover | *Pluvialis squatarola* | Pluvier argenté

The nominate subspecies of the Grey Plover *P. s. squatarola* breeds in the tundra zone of Siberia east of the Kanin peninsula. This subspecies has two recognised flyway populations, an eastern one, where birds winter in SW Asia, E Africa and southern Africa, and a western population wintering from NW Europe south to the Gulf of Guinea. Here we consider the latter, as the former occurs for a large part

outside the EAF. During migration, the species frequents coastal areas in large parts of W and S Europe and W Africa. In the high Arctic Grey Plover breed dispersed in various types of open tundra. At other times of the year they are gregarious, occurring mainly on intertidal mudflats and salt marshes. The main food sources outside the breeding season are polychaete worms, molluscs and crustaceans.



Hans Schekkerman

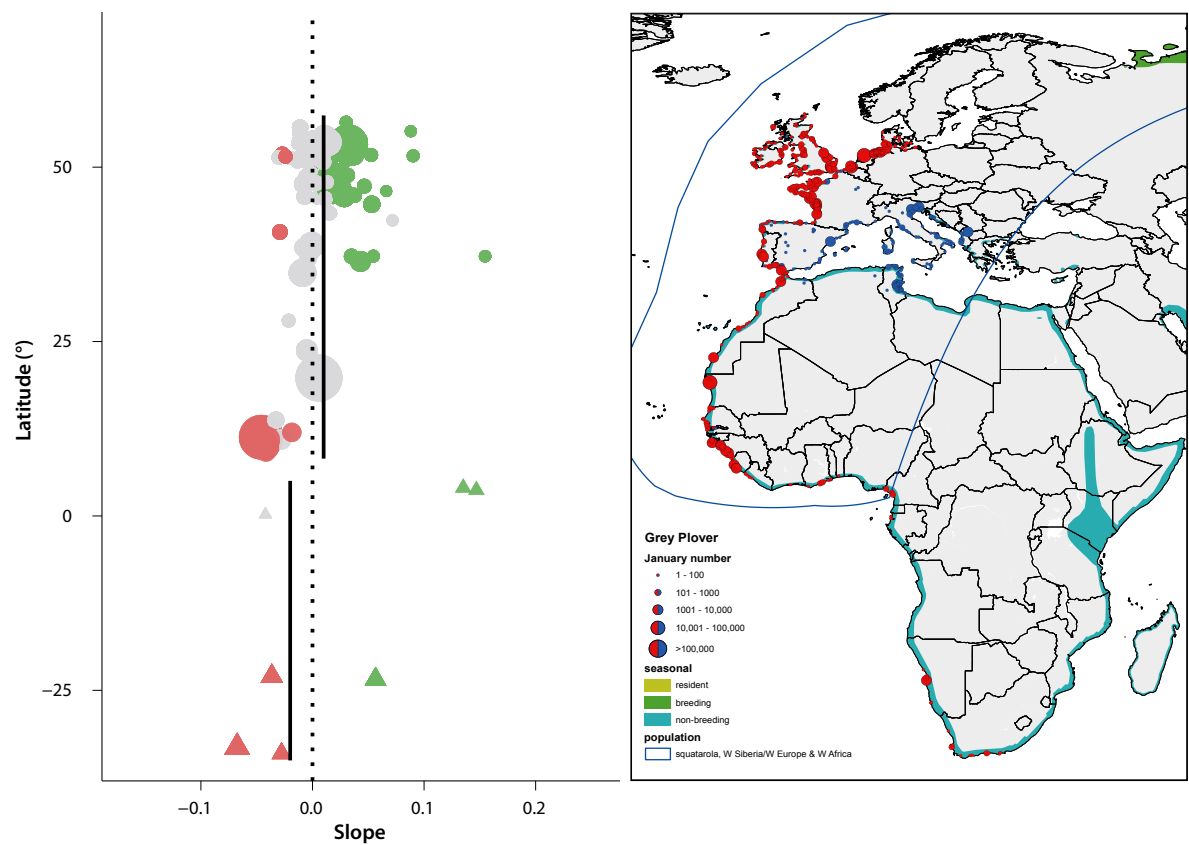


Figure A1.64. Plot (left) of January site trends and distribution (right) of Grey Plover in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Grey Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsiz-min	popsiz-max
W Siberia/W Europe & W Africa	w	1978-2023	1,012	moderate increase	2014-2023	0,990	stable	2010- 2018	200000	200000

Table A1.30. Summary of flyway trend and population size estimates for Grey Plover. For explanation see table A1.2.

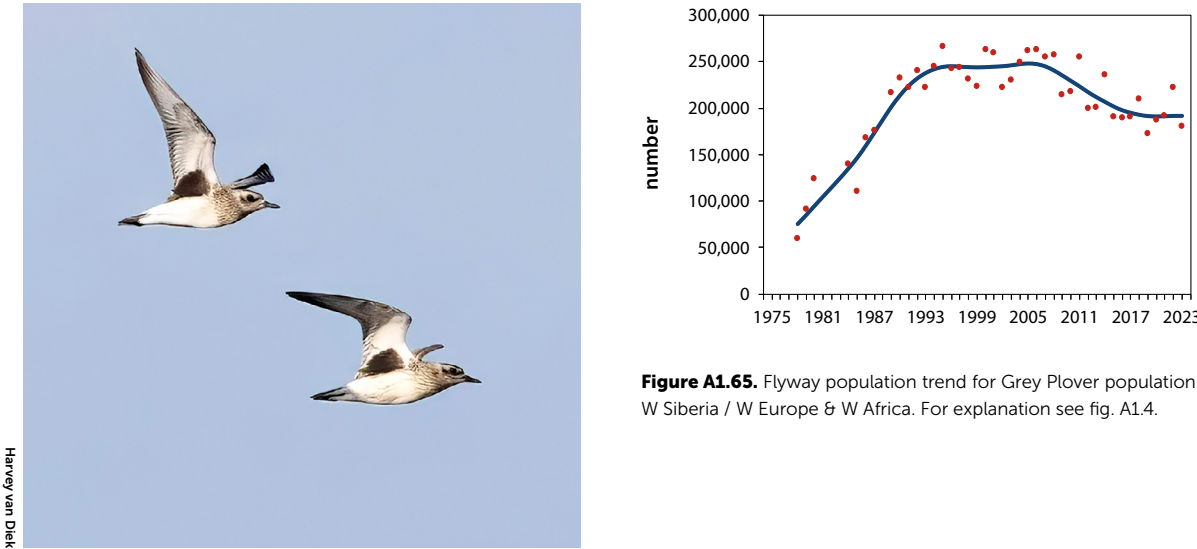


Figure A1.65. Flyway population trend for Grey Plover population W Siberia / W Europe & W Africa. For explanation see fig. A1.4.

Harvey van Diek

Common Ringed Plover | *Charadrius hiaticula* | Pluvier grand-gravelot

Three subspecies are recognised. The nominate form *C. h. hiaticula* breeding in NW Europe is partly sedentary and a short distance migrant, and mainly remains in Europe during winter. *C. h. psammодromus* breeds in NE Canada, Greenland and Iceland, and migrates to winter along the coasts of Africa south to about Namibia. *C. h. tundrae* is thought to breed from NE Europe through N Russia as far as the Bering Straits, and to winter mainly in SW Asia and E and Southern Africa, where it just reaches the southern part of the EAF. Birds from N Scandinavia migrate to

W Africa (Lislevand *et al.* 2017). Breeding occurs mostly dispersed. Preferred breeding habitat is sand or shingle beaches along the Atlantic coast, sometimes also inland on sand and gravel along large rivers, lakes and reservoirs. Further north the species breeds on tundra. Outside the breeding season it is gregarious, preferring muddy and sandy coasts, e.g. estuaries, tidal mudflats and lagoons. Its diet consists of small invertebrates such as crustaceans and insects, worms and small molluscs.

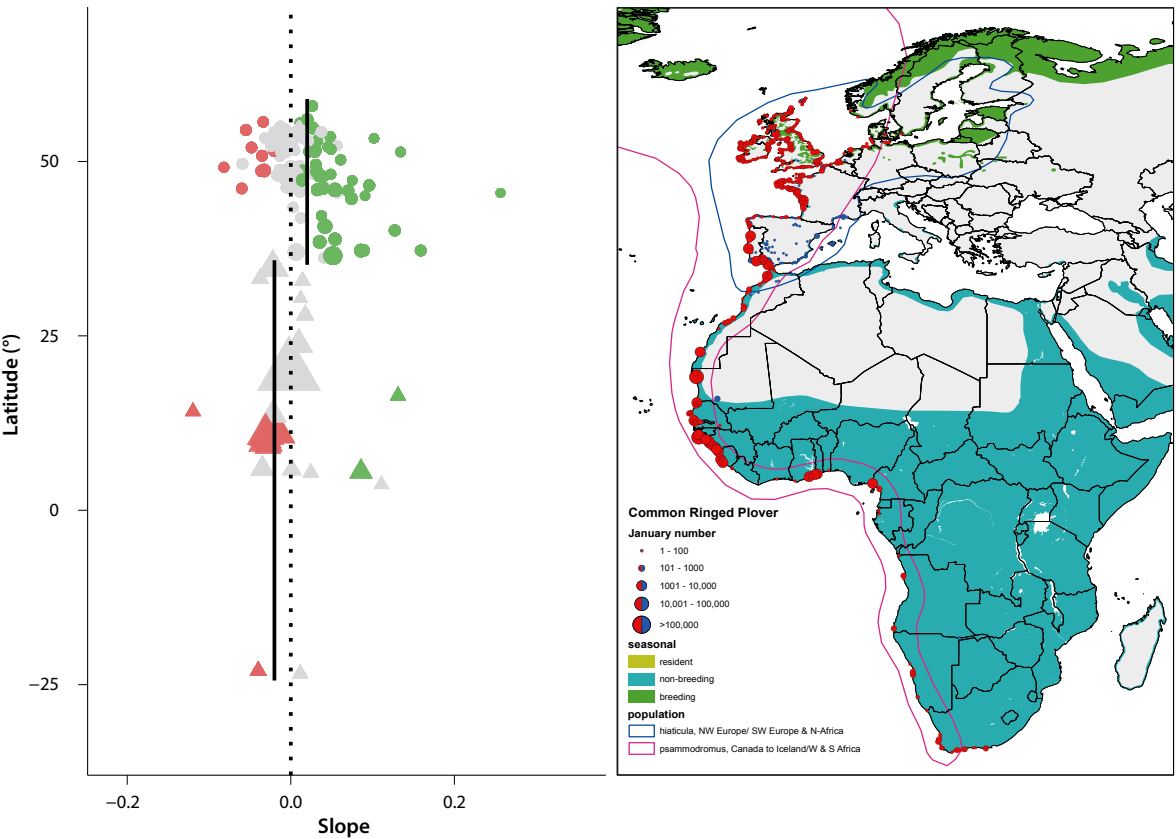


Figure A1.66. Plot (left) of January site trends and distribution (right) of Common Ringed Plover in the East Atlantic Flyway in January . For explanation see fig. A1.3.

population Common Ringed Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsiz	popsiz-min	popsiz-max
<i>hiaticula</i> , NW Europe/ SW Europe & N-Africa	w	1978-2023	1,019	moderate increase	2014-2023	0,993	stable	2007-2019	50000	68000
<i>psammодromus</i> , Canada to Iceland/ W & S Africa	w	1979-2023	0,982	moderate decrease	2014-2023	0,968	moderate decrease	2010-2014	240000	240000

Table A1.31. Summary of flyway trend and population size estimates for Common Ringed Plover. For explanation see table A1.2.

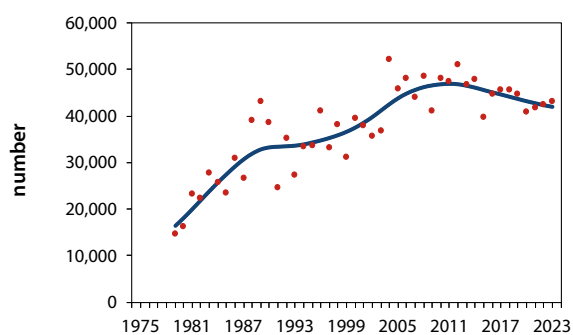


Figure A1.67. Flyway population trend for Common Ringed Plover population *hiaticula* N Europe / Europe & N Africa. For explanation see fig. A1.4.

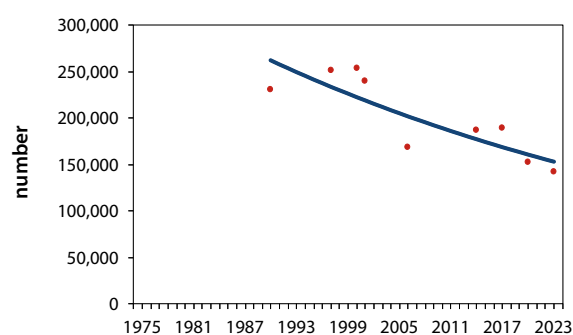


Figure A1.68. Flyway population trend for Common Ringed Plover population *psammmodromus* Nearctic & Iceland / W & S Africa. For explanation see fig. A1.4.



Harvey van Diek

Kittlitz's Plover | *Charadrius pecuarius* | Pluvier pâtre

The Kittlitz's Plover is an African species with four geographical populations recognised. It is largely sedentary but some are migratory or partial migratory. Along the East Atlantic coast, the population occurring in West Africa is the focus of this report. Further along the East Atlantic Flyway the geographical population of East, Central and Southern Africa is thought to occur also at the coasts of Congo, Namibia and Southern Africa. Two smaller distributed populations occur in the Nile Delta and at Madagascar. The population of West Africa is found in good numbers in coastal southern Mauritania, Senegal and Ghana and also more inland (Inner Niger Delta). Kittlitz's Plovers are not really mudflat waders but are more found at short grass and bare ground. They are typically inhabiting the margins of non-tidal wetlands and lagoons and coastal savanna. Outside the breeding seasons they are found in small flocks. They are dependent on terrestrial invertebrates, mainly insects.



Danielle Occhialio / Agami

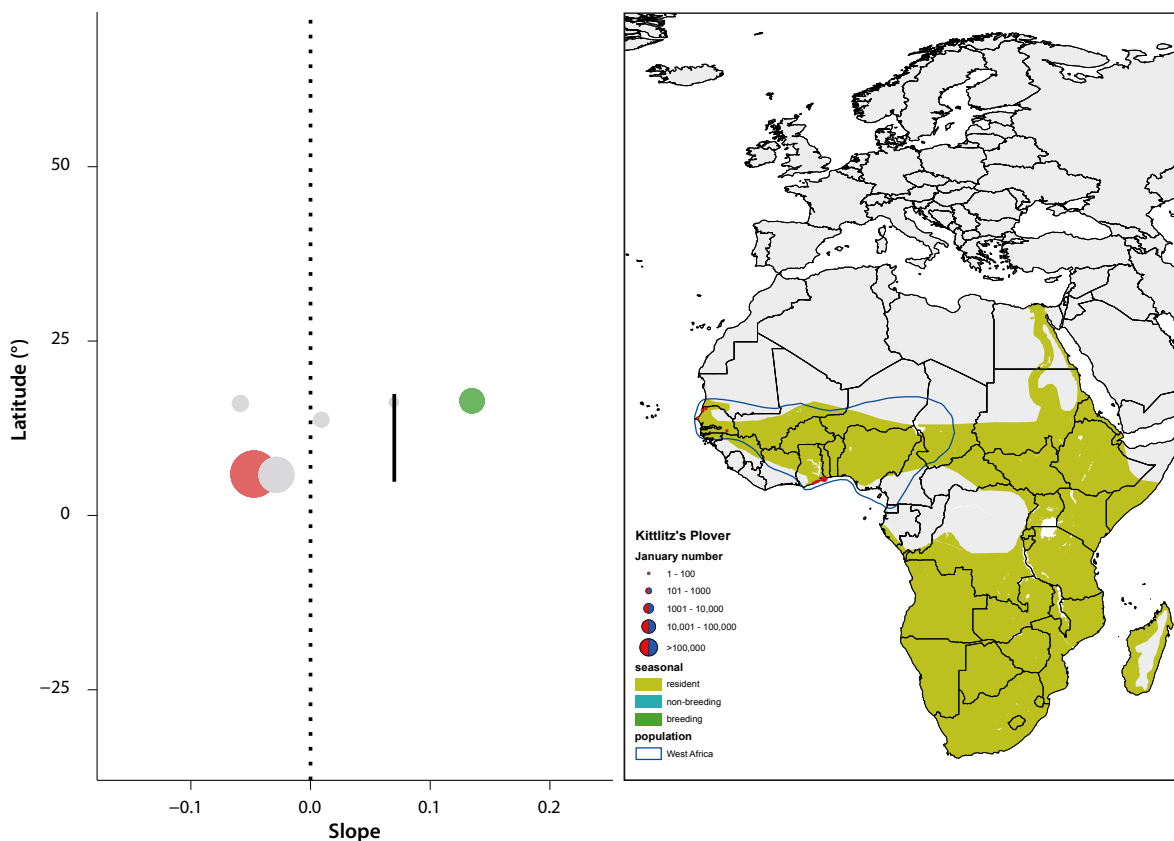


Figure A1.69. Plot (left) of January site trends and distribution (right) of Kittlitz's Plover in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Kittlitz's Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa		1991-2023	1,071	uncertain	2014-2023	1,098	uncertain	2000	20000	50000

Table A1.32. Summary of flyway trend and population size estimates for Kittlitz's Plover. For explanation see table A1.2.

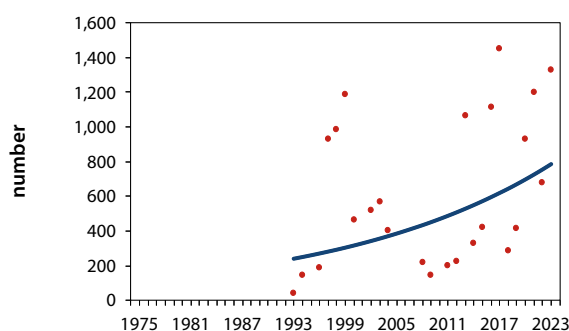


Figure A1.70. Flyway population trend for Kittlitz's Plover population W Africa. For explanation see fig. A1.4.



Bas Hasnoot / Agami

White-fronted Plover | *Charadrius marginatus* | Pluvier à front blanc

The White-fronted Plover is an African species occurring in most of sub-Saharan Africa. It is a sedentary and partially migratory species that breeds along the coasts and large rivers. Along the East Atlantic African coast, four populations occur from Senegal to South Africa. These have been combined here, although for the trend analyses we were able to present the trend separately for the *hesperius* population, which occurs along the coast from Senegal to Cameroon and in inland areas mainly in the Niger Basin. During the breeding season the species is solitary, in the

non-breeding periods larger groups can occur up to a few hundred individuals. Its breeding habitat in W Africa consists of sandy beaches and dunes, but it can also be found on a wide variety of other coastal habitats such as estuaries, lagoons and salt-pans. Inland, the species breeds on the sandy shores of large rivers, and it occurs in the same habitats outside the breeding season. Its diet consists of a wide variety of small invertebrate food items like insects, gastropods, molluscs, bivalves, crustaceans, isopods and worms.



Marc Guyt / Agami

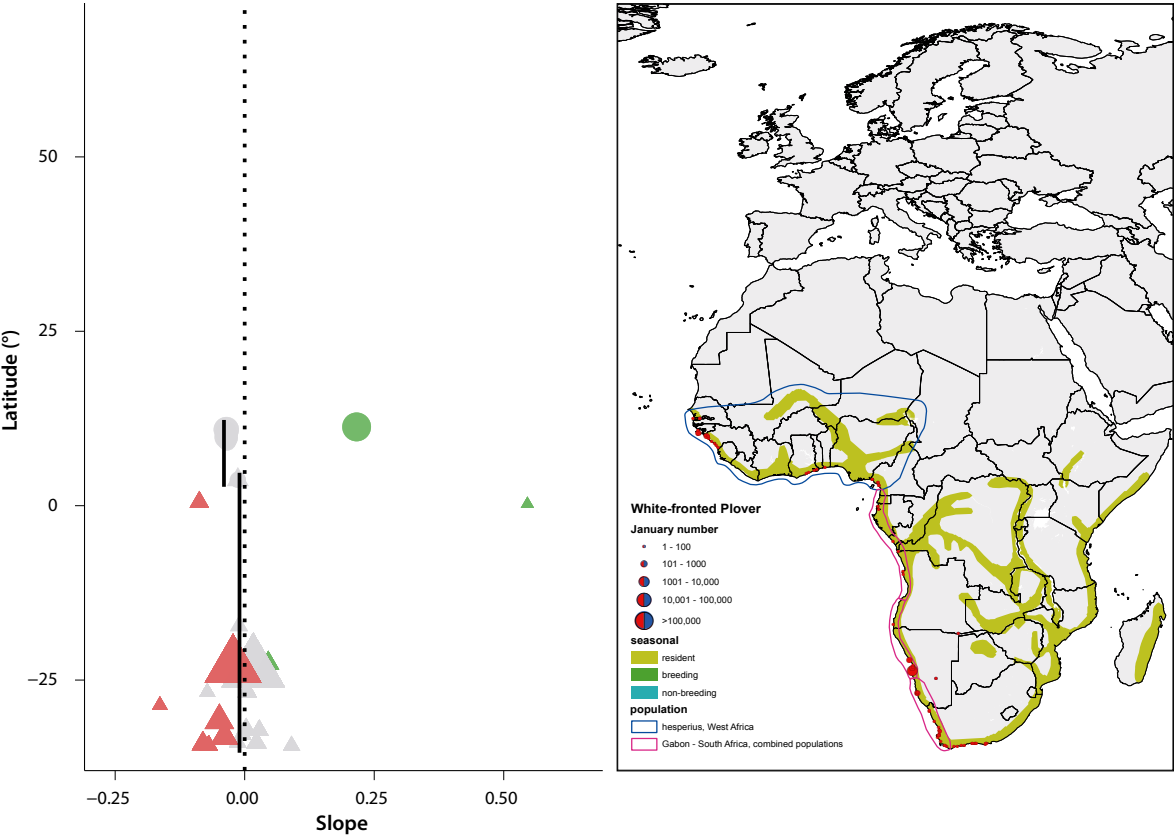


Figure A1.71. Plot (left) of January site trends and distribution (right) of White-fronted Plover in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population White-fronted Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
hesperius, W Africa	w	1990-2023	0,963	moderate decrease	2014-2023	1,046	moderate increase	1998-2007	10000	15000
Gabon - South Africa, combi of pop.	w	1992-2023	0,994	stable	2014-2023	1,011	uncertain			

Table A1.33. Summary of flyway trend and population size estimates for White-fronted Plover. For explanation see table A1.2.

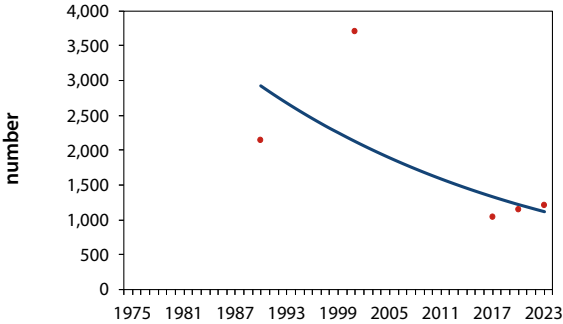


Figure A1.72. Flyway population trend for White-fronted Plover population hesperius W Africa. For explanation see fig. A1.4.

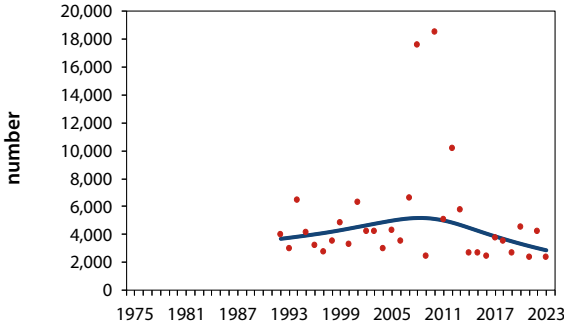


Figure A1.73. Flyway population trend for White-fronted Plover populations Gabon - S Africa combined. For explanation see fig. A1.4.

Kentish Plover | *Charadrius alexandrinus* | Pluvier à collier interrompu

Two flyway populations of the nominate form of Kentish Plover are distinguished in Europe: one in W Europe and W Mediterranean and one in SE Europe and the E Mediterranean. The EAF population range covers breeding areas in W Europe and the W Mediterranean, and in Africa along the north and west coasts south to Senegal. The range of this population also covers wintering areas of the migratory northern birds, in S Europe, N Africa, coastal W Africa and the Sahel. The majority of the European breeding population occurs in France, the Iberian Peninsula and

NW Africa. In the breeding season, the Kentish Plover is mostly a coastal species in this part of its range, breeding in solitary pairs or loose colonies. They mainly forage on sand and silt mudflats and breed on sandy and sparsely vegetated places in e.g. lagoons, dunes, beaches, estuaries and salt pans. Outside the breeding season, the species is more gregarious and is usually seen in small flocks. The diet consists mainly of insects, crustaceans (e.g. gammarids), small molluscs and polychaete worms.

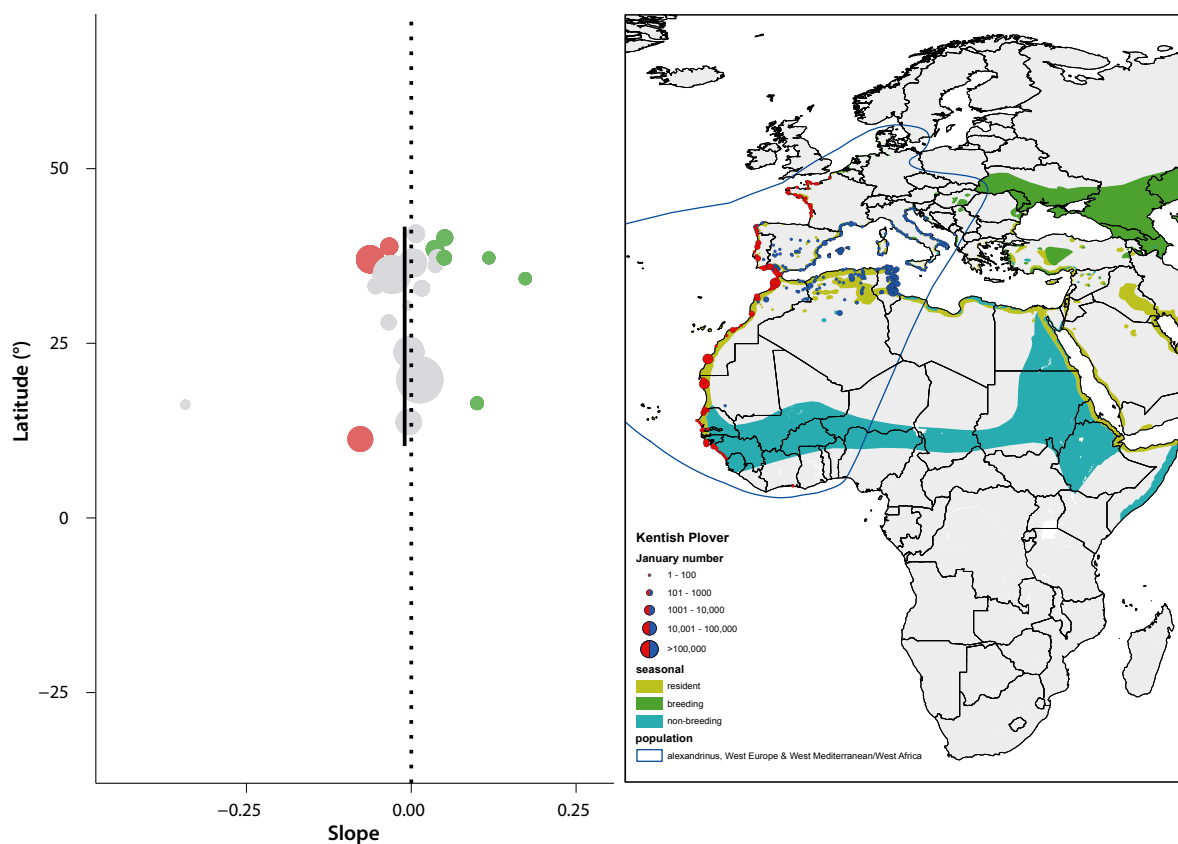


Figure A1.74. Plot (left) of January site trends and distribution (right) of Kentish Plover in the East Atlantic Flyway in January. For explanation see fig. A1.3.



Harvey van Diek



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population Kentish Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Europe & W Mediterranean/W Africa	w	1993-2023	0,986	moderate decrease	2014-2023	1,024	stable	2007- 2018	40000	65000

Table A1.34. Summary of flyway trend and population size estimates for Kentish Plover. For explanation see table A1.2.

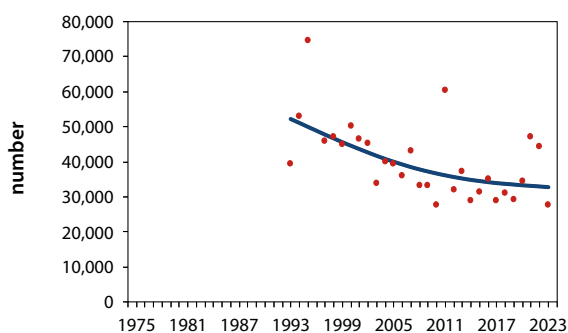


Figure A1.75. Flyway population trend for Kentish Plover population *alexandrinus* W Europe & W Mediterranean / W Africa. For explanation see fig. A1.4.



Sylvain Rey / Agami

Chestnut-banded Plover | *Charadrius pallidus* | Pluvier élégant

The EAF includes one biogeographical population of Chestnut-banded Plover, representing the nominate subspecies *C. p. pallidus*. It has a patchy distribution and little is known about its movements, but coastal birds in South Africa are probably sedentary, while some of the coastal birds in Namibia probably migrate inland for breeding. The species is typically found in pairs or small groups, but

aggregations of several hundred individuals are occasionally observed in the non-breeding season. Breeding takes place in alkaline and saline wetlands, including natural and man-made salt pans. During the non-breeding period the species is usually found in coastal habitats including intertidal mudflats. The diet consists of insect larvae and small crustaceans.



Pete Morris / Agami

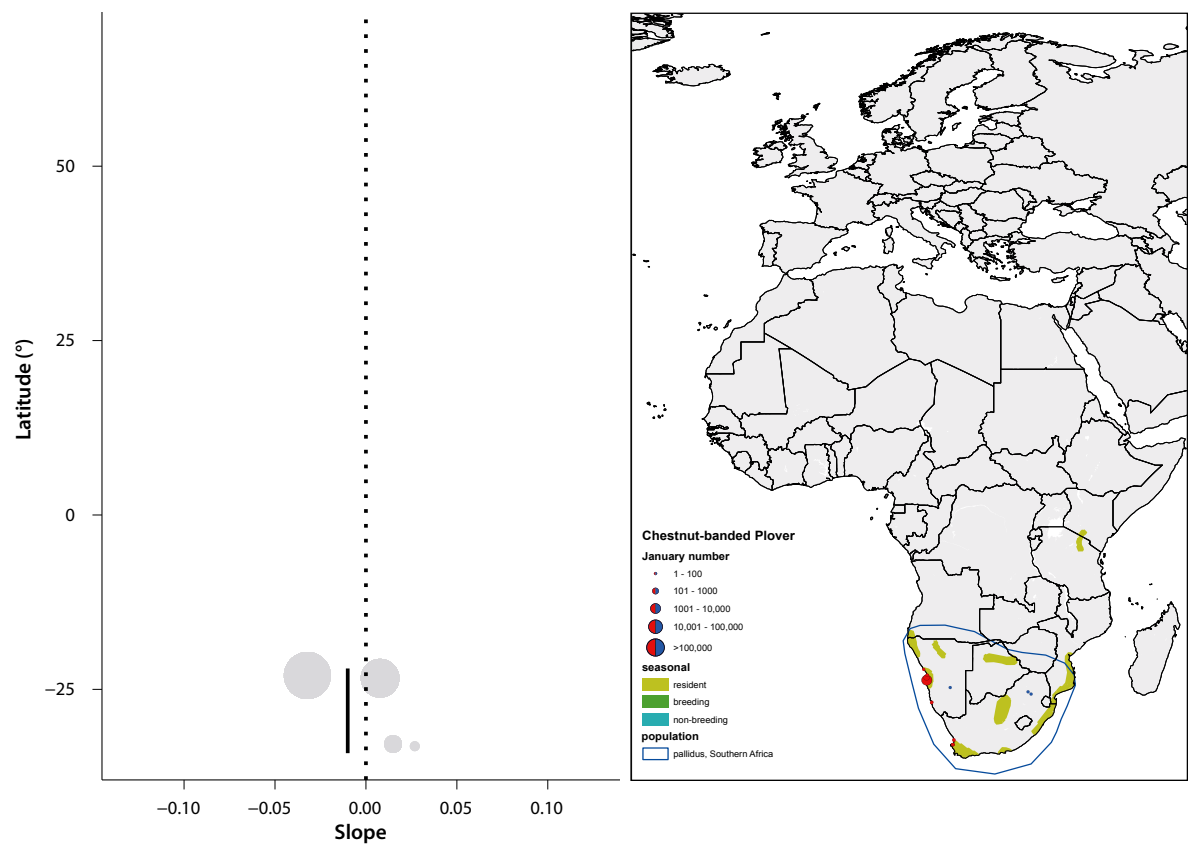


Figure A1.76. Plot (left) of January site trends and distribution (right) of Chestnut-banded Plover in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Chestnut-banded Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsiz	popsiz-min	popsiz-max
S Africa	w	1992-2023	0,986	stable	2014-2023	0,839	strong decrease	2000-2007	11000	16000

Table A1.35. Summary of flyway trend and population size estimates for Chestnut-banded Plover. For explanation see table A1.2.



Pete Morris / Agami

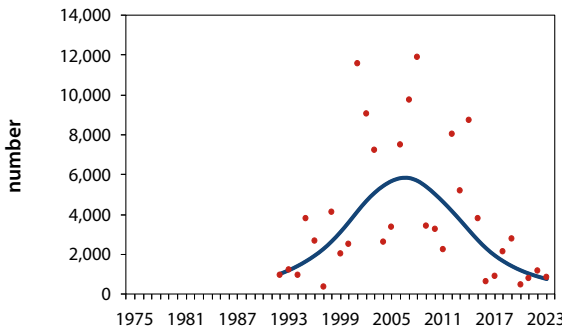


Figure A.77. Flyway population trend for Chestnut-banded Plover population *pallidus* Southern Africa. For explanation see fig. A1.4.

Whimbrel | *Numenius phaeopus* | Courlis corlieu

Two subspecies of Whimbrel use the EAF - the nominate *N. p. phaeopus*, breeding in Fennoscandia, the Baltic states and northern Russia and wintering all along the coast of W Africa south to Gabon and maybe even Namibia, and *N. p. islandicus* breeding in Iceland and a small part of Greenland and wintering in the same African region probably not further than Gabon (Alves *et al.* 2016; Carneiro *et al.* 2021). Both populations are treated together here, as they cannot be separated in the overlapping non-breeding range. A third population of *N. p. phaeopus* from Siberia reaches the EAF in Southern Africa. Large breeding populations of Whimbrel occur in Iceland, Finland and northern Russia. The species breeds in solitary pairs on wet and dry heathlands and wetlands, moors and bogs in Boreal and Arctic regions. Sometimes breeding in open forested areas occurs. During migration and wintering the species prefers sandy and rocky coasts, tidal mudflats and mangroves. During migration it congregates in flocks and besides the mentioned habitats also uses heathland and short grasslands more inland. Important food items during breeding are invertebrates e.g. insects and worms. In coastal habitats during the non-breeding season, the species specialises in feeding on crustaceans such as crabs, but foraging on berries (*Empetrum* sp.) is also not uncommon.



Harvey van Diek

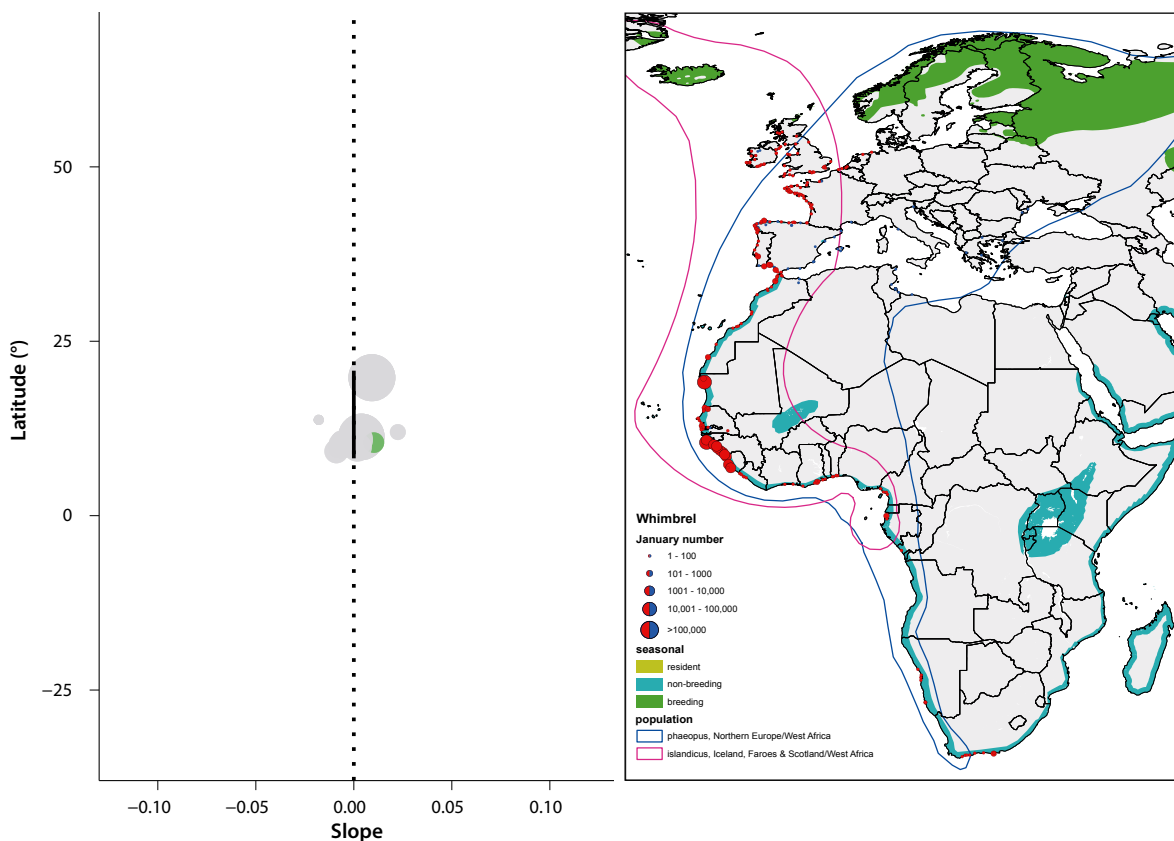


Figure A1.78. Plot (left) of January site trends and distribution (right) of Whimbrel in the East Atlantic Flyway in January. For explanation see fig. A1.3.



Arnold Meijer / Blue Robin

population Whimbrel	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
East Atlantic (wintering) combi of pop.	w	1979-2023	1,003	stable	2014-2023	0,981	moderate decrease			
<i>phaeopus</i> , N Europe /W Africa								2010-2018	240000	390000
<i>islandica</i> , Iceland, Faroer, Scotland / W Africa								2009-2018	770000	780000

Table A1.36. Summary of flyway trend and population size estimates for Whimbrel. For explanation see table A1.2.

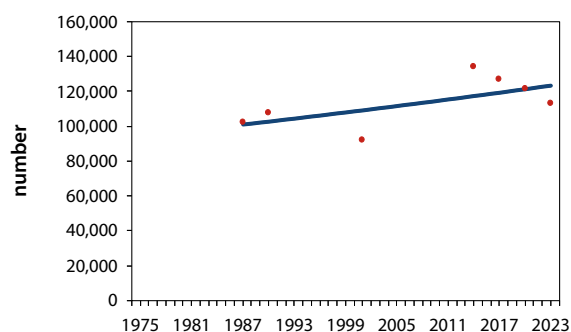


Figure A1.79. Flyway population trend for the EAF wintering population of Whimbrel (populations combined). For explanation see fig. A1.4.



Harvey van Diek

Eurasian Curlew | *Numenius arquata* | Courlis cendré

The population of Eurasian Curlew considered here is the nominate form which breeds across large parts of Europe, with the largest numbers in Sweden, Finland and Russia. Wintering occurs in W and S Europe, and partly also on the coast of W Africa south to about Guinea-Bissau. Curlew wintering further south are considered to belong to the subspecies *N. a. orientalis*, which is not considered here as its non-breeding distribution lies mainly outside the EAF, in E Africa and SW Asia. Curlew breed solitarily on heathland, upland moors, peat bogs, coastal marshlands but also farmland areas (both grasslands and arable fields). During migration and in the northern winter, they frequent in coastal habitats such as estuaries, tidal mudflats, mangroves and saltmarshes, but also agricultural grasslands. The breeding season diet consists of a variety of invertebrate food items like annelid worms and insects and their larvae. On the coast during the northern winter the species feeds mostly on polychaete worms, crustaceans (e.g. crabs), and bivalves.



Arnold Meijer / Blue Robin

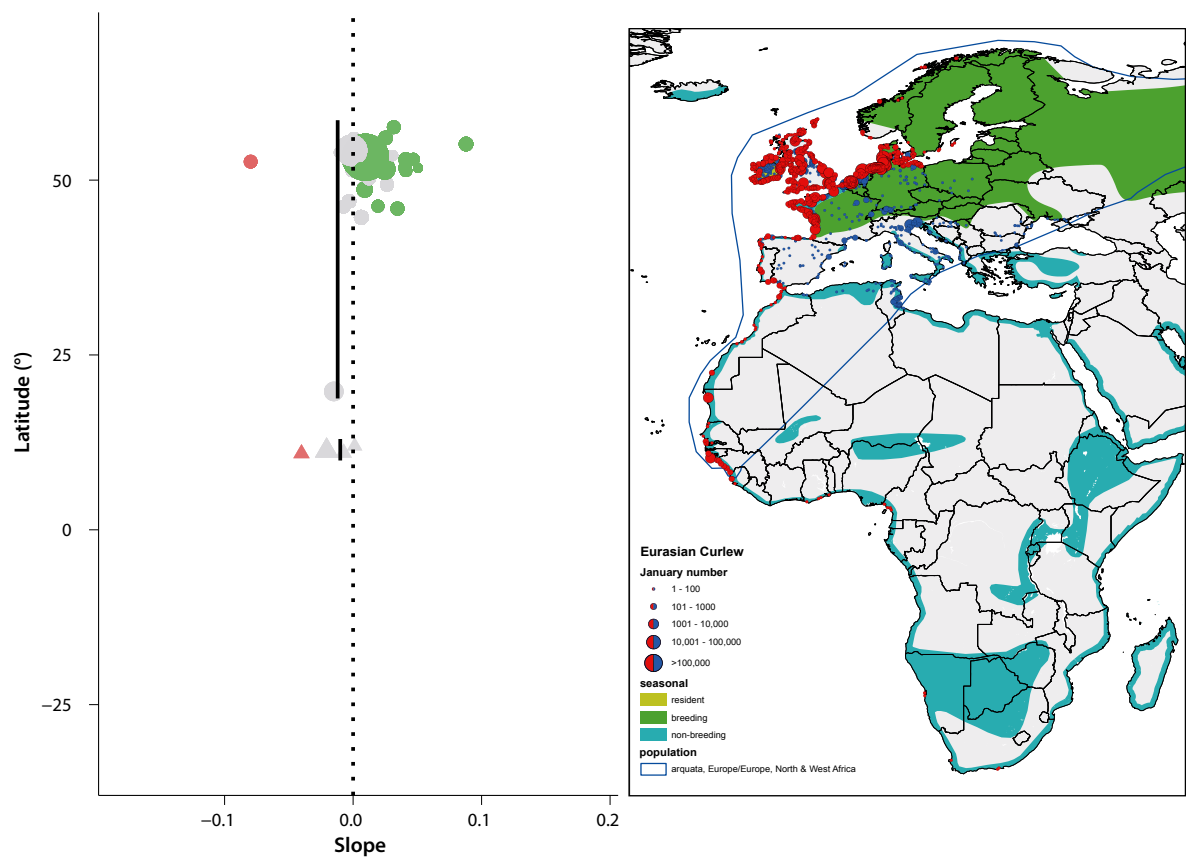


Figure A1.80. Plot (left) of January site trends and distribution (right) of Eurasian Curlew in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Eurasian Curlew	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsiz	popsiz-min	popsiz-max
Europe/NW Europe, N & W Africa	b	1980-2018	0,988	moderate decrease	2009-2018	1,001	stable	1990- 2019	610000	830000

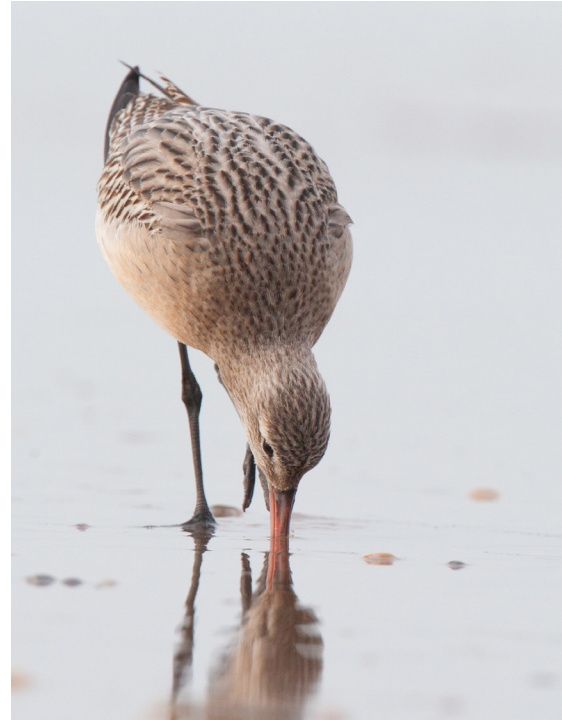
Table A1.37. Summary of flyway trend and population size estimates for Eurasian Curlew. For explanation see table A1.2.



Arnold Meijer / Blue Robin

Bar-tailed Godwit | *Limosa lapponica* | Barge rousse

Two subspecies of Bar-tailed Godwit use the EAF, showing a classic leapfrog migration pattern, with breeders from the Siberian high Arctic (*L. l. taymyrensis*) migrating further south than the population of *L. l. lapponica* breeding in Fennoscandia (Duijns *et al.* 2012). The nominate *lapponica* breed in N Fennoscandia east to the Kanin Peninsula and winter in W Europe south to Portugal and Spain. The *taymyrensis* subspecies migrates through W Europe (mainly the Wadden Sea) to winter in W and SW Africa. However, recent findings indicate that more *taymyrensis* winter in the Wadden Sea than previously thought (Conklin *et al.* 2024). If this pattern of more northern wintering will continue it will make our monitoring of these two populations based on the IWC rather difficult (see also chapter 2). Breeding habitats are swampy tundra, heathlands, and open bogs in the far north. Nests are dispersed. During migration and wintering it is highly gregarious and occurs in huge flocks of up to tens of thousands of individuals. Its preferred foraging habitats are intertidal flats, lagoons and estuaries. Foraging can however also occur on short-grass meadows, mainly in spring (Duijns *et al.* 2009). The diet of Bar-tailed Godwit consists mainly of worms.



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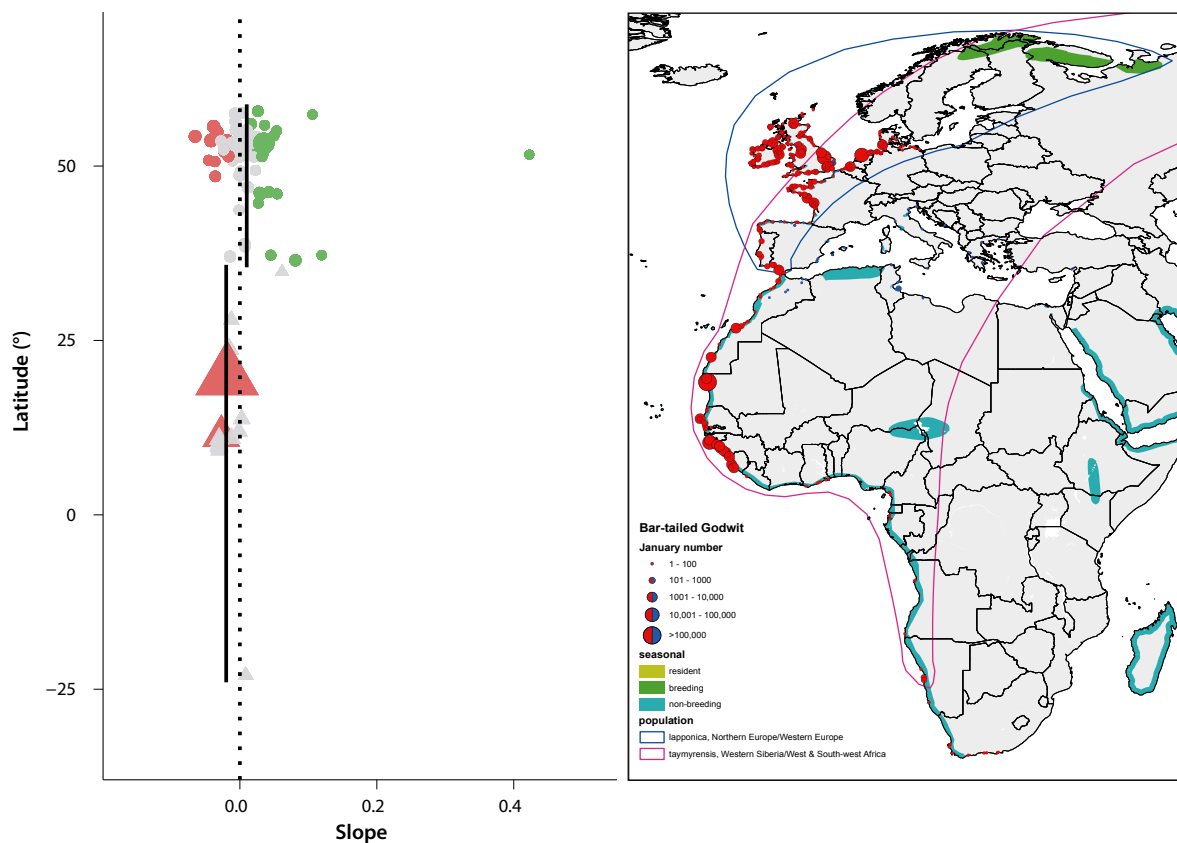


Figure A1.81. Plot (left) of January site trends and distribution (right) of Bar-tailed Godwit in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Bar-tailed Godwit	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
<i>lapponica</i> , N Europe /W Europe	w	1976-2023	1,011	stable	2014-2023	0,981	stable	2011-2018	150000	180000
<i>taymyrensis</i> , W Siberia /W & S Africa	w	1979-2023	0,980	moderate decrease	2014-2023	0,951	moderate decrease	2016-2020	380000	420000

Table A1.38. Summary of flyway trend and population size estimates for Bar-tailed Godwit. For explanation see table A1.2.

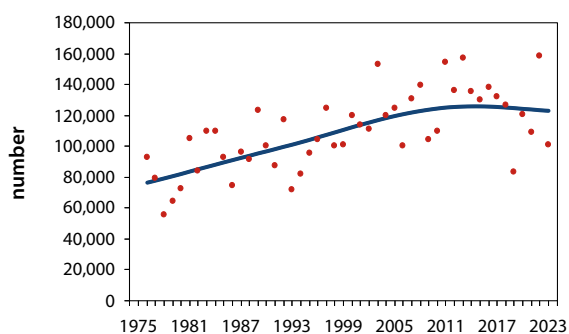


Figure A1.82. Flyway population trend for Bar-tailed Godwit population *lapponica*. For explanation see fig. A1.4.

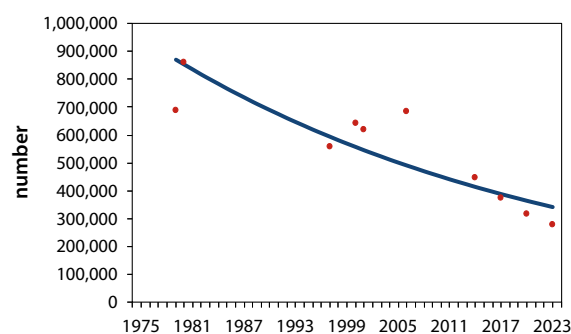


Figure A1.83. Flyway population trend for Bar-tailed Godwit population *taymyrensis*. For explanation see fig. A1.4.



Ruddy Turnstone | *Arenaria interpres* | Tourneepierre à collier

The Ruddy Turnstone is a high arctic breeding species with a cosmopolitan range. Two sub-populations of the nominate subspecies are treated here as EAF populations: a Nearctic population breeding in NE Canada and Greenland that winters mainly in W Europe, and a Palearctic population breeding in northern Scandinavia and W Russia, including Svalbard, that winters in W Africa (Helseth *et al.* 2005). Turnstones breeding in W and C Siberia also reach the southern parts of the EAF (SW Africa) in the

non-breeding season, but also winter extensively in E Africa and SW Asia. Turnstones breed dispersed in tundra and coastal habitats in the (high) Arctic. Outside the breeding season they are mainly coastal and frequent rocky or shingle shores, also sandy beaches with seaweed, reefs and mudflats. It is mainly insectivorous during the breeding season. Outside the breeding season it mainly feeds on crustaceans, molluscs, annelids, echinoderms and fish, and even takes carrion.

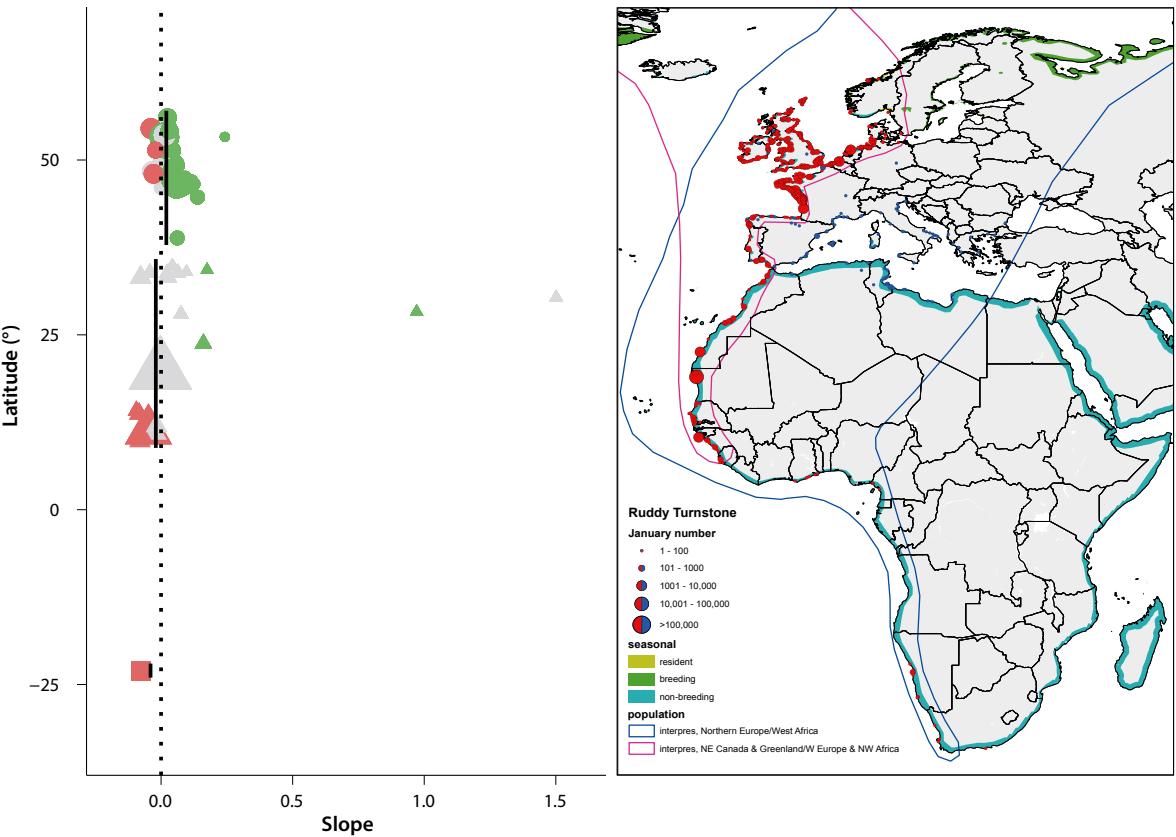


Figure A1.84. Plot (left) of January site trends and distribution (right) of Ruddy Turnstone in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Ruddy Turnstone	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
NE Canada & Greenland /W Europe & NW Africa	w	1976-2023	1,022	moderate increase	2014-2023	1,005	stable	2000	200000	260000
Ruddy Turnstone, N Europe/W Africa	w	1979-2023	0,980	moderate decrease	2014-2023	1,061	moderate increase	2008- 2018	44000	87000

Table A1.39. Summary of flyway trend and population size estimates for Ruddy Turnstone. For explanation see table A1.2.

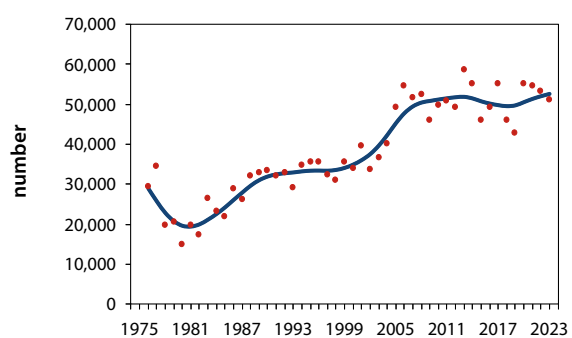


Figure A1.85. Flyway population trend for Ruddy Turnstone population *interpres* Nearctic / W Europe. For explanation see fig. A1.4.

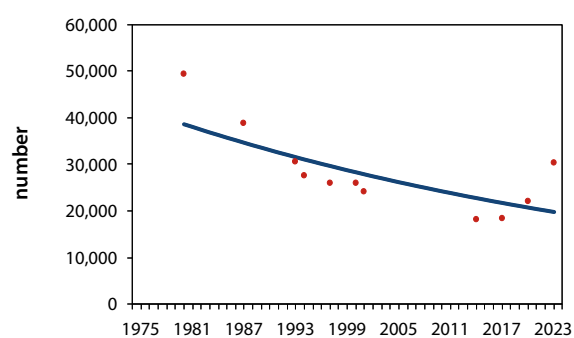


Figure A1.86. Flyway population trend for Ruddy Turnstone population *interpres* N Europe / W Africa. For explanation see fig. A1.4.



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Red Knot | *Calidris canutus* | Bécasseau maubèche

Two subspecies of Red Knot use the EAF. The Palearctic nominate *C. c. canutus* breeds in the Arctic zones of Russia (Taymyr Peninsula) and migrates through Europe to the coast of W Africa, whilst the Nearctic breeding population of Greenland and E Canada *C. c. islandica* winters in W Europe. However like in Bar-tailed Godwit also in this species genetic evidence suggests more mixing of these populations than originally thought (Conklin *et al.* 2022). Breeding occurs dispersed on high-arctic tundra, mostly in dry upland tundra and gravel. Migrating and wintering birds concentrate in large flocks in coastal areas, with a preference for tidal mud- or sand-flats. Insects are the main food items during the breeding season, but early in the season leftover berries, seeds and grass shoots are also eaten. The non-breeding diet is specialised towards small to medium-sized bivalves which are ingested whole and crushed in the muscular gizzard, but knots also take small gastropods and shrimps when available and recently also *Zostera* (van Gils *et al.* 2003).



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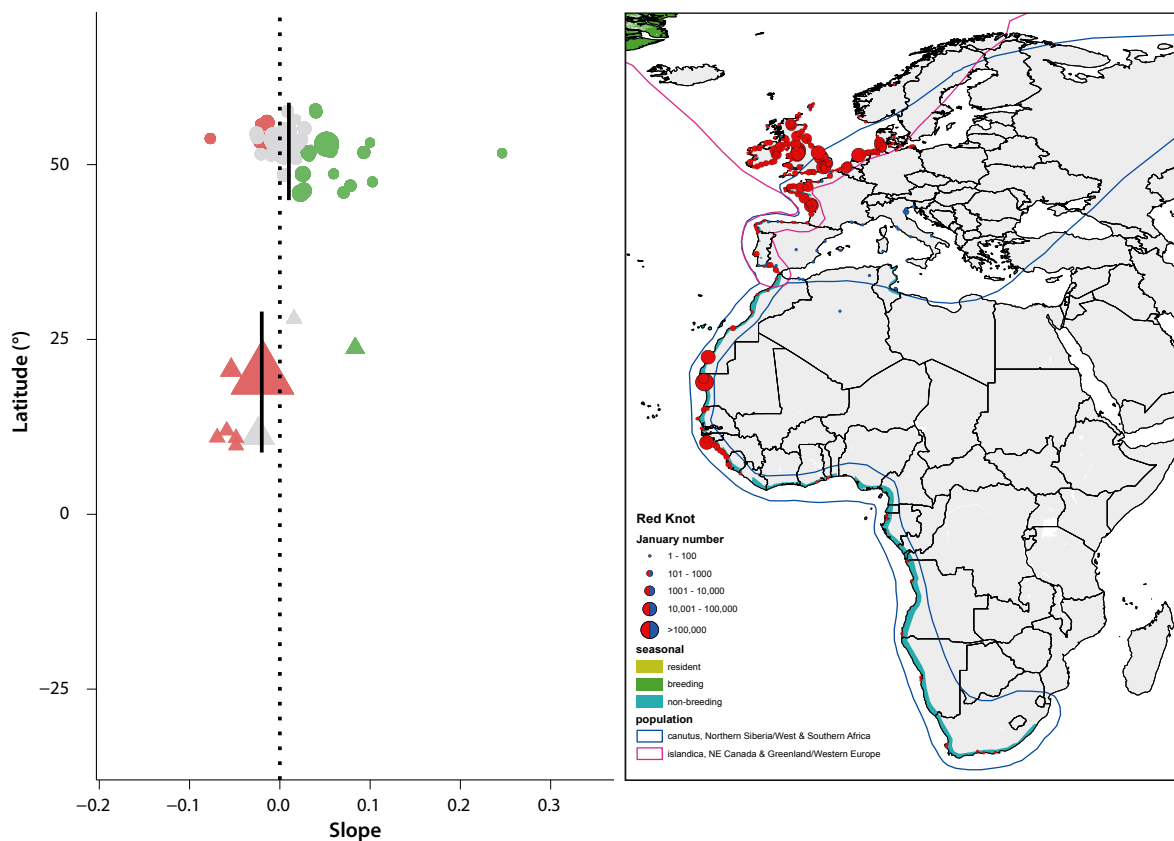


Figure A1.87. Plot (left) of January trends and distribution (right) of Red Knot in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Red Knot	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
<i>islandica</i> , NE Canada & Greenland /W Europe	w	1975-2023	1,012	moderate increase	2014-2023	0,972	moderate decrease	2013- 2017	310000	360000
<i>canutus</i> , N Siberia/W & S Africa	w	1979-2023	0,979	moderate decrease	2014-2023	0,934	moderate decrease	2016- 2020	260000	275000

Table A1.40. Summary of flyway trend and population size estimates for Red Knot. For explanation see table A1.2.

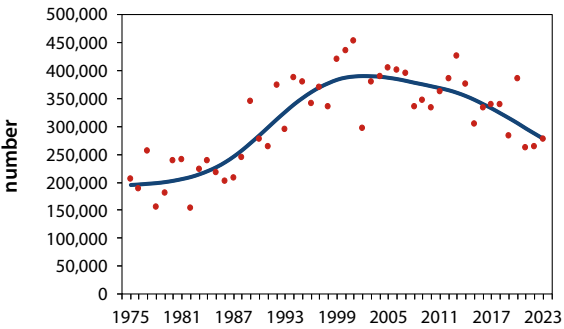


Figure A1.88. Flyway population trend for Red Knot population *islandica*. For explanation see fig. A1.4.

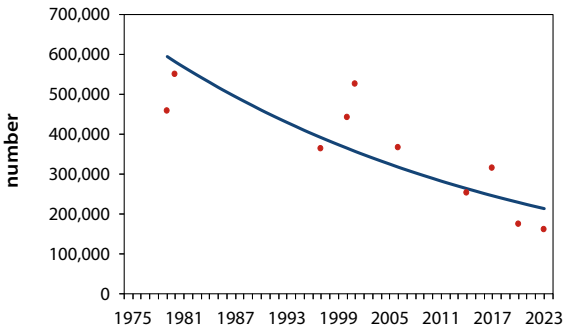


Figure A1.89. Flyway population trend for Red Knot population *canutus*. For explanation see fig. A1.4.



Harvey van Diek

Curlew Sandpiper | *Calidris ferruginea* | Bécasseau cocorli

Curlew Sandpipers breed in N Russia between the Yamal Peninsula and E Siberia. Part of the birds from W and C Siberia migrate through W and E Europe to W Africa, and another part migrate to SW, S and E Africa through the eastern Mediterranean, Black and Caspian Seas. For this report, the boundary between their non-breeding ranges is placed between Nigeria and Gabon. Breeding occurs dispersed in lowlands of the high Arctic, with a preference

for open tundra with wet marshy areas. In winter, the species is mainly coastal and occurs on brackish lagoons, tidal mud- and sand-flats, estuaries and saltmarshes. Inland habitats such as muddy edges of freshwater wetlands are also used. The species is mainly insectivorous during the breeding season and forages on polychaete worms, molluscs and crustaceans in more saline habitats on passage and in winter.



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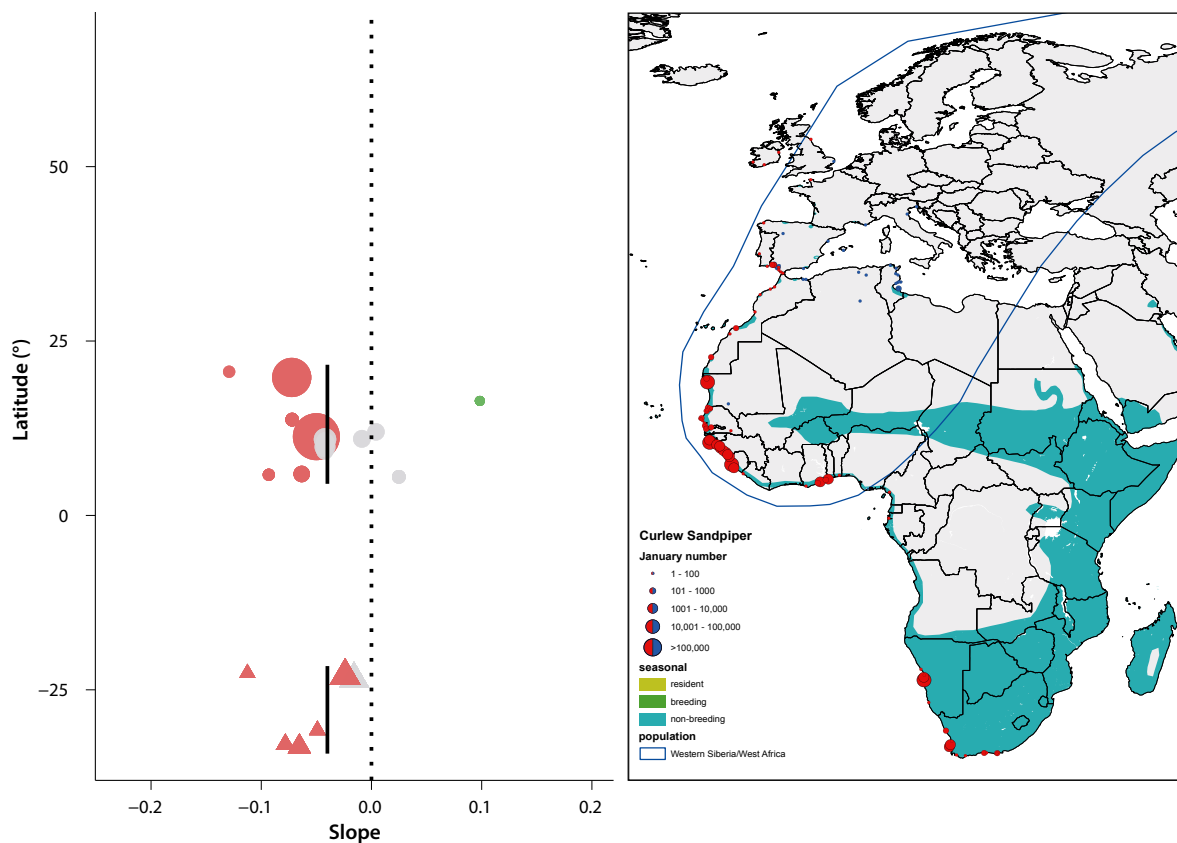


Figure A1.90. Plot (left) of site January trends and distribution (right) of Curlew Sandpiper in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Curlew Sandpiper	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Siberia / W Africa	w	1979-2023	0,963	moderate decrease	2014-2023	0,919	strong decrease	2016- 2020	300000	400000

Table A1.41. Summary of flyway trend and population size estimates for Curlew Sandpiper. For explanation see table A1.2.

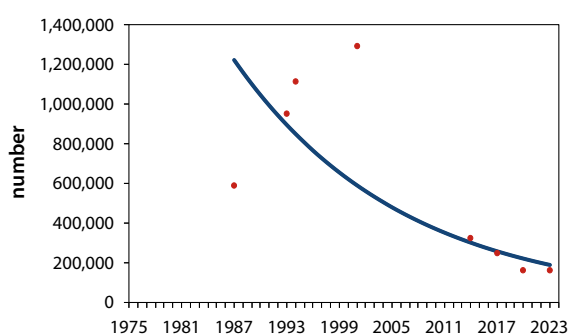


Figure A1.91. Flyway population trend for Curlew Sandpiper population W Siberia / W Africa. For explanation see fig. A1.4.



Harvey van diek

Sanderling | *Calidris alba* | Bécasseau sanderling

Two populations of Sanderling occur in the EAF, one wintering in E Atlantic Europe, W and Southern Africa ('E Atlantic population') and one wintering in SW Asia, E and Southern Africa ('W Asia – S Africa population'). Breeding of the E Atlantic population occurs in the high Arctic tundra of Greenland and northeast Canada. Whether a part of the breeding birds from the Taymyr peninsula in Siberia also belong to this flyway is still unclear (Reneerkens *et al.* 2009). Birds of the W Asia – S Africa population probably reach the coastal East Atlantic from Cameroon south-

wards for wintering, but Greenlandic birds also occur south to Namibia but do probably not reach South Africa (Reneerkens *et al.* 2020). The species is strictly coastal outside the breeding season. It breeds dispersed in well-drained barren or stony tundra. The breeding diet consists mainly of insects and spiders, and plant material when insects are too scarce in spring. On passage and in winter, its diet consists of polychaete worms, small molluscs and crustaceans (Grond *et al.* 2015).



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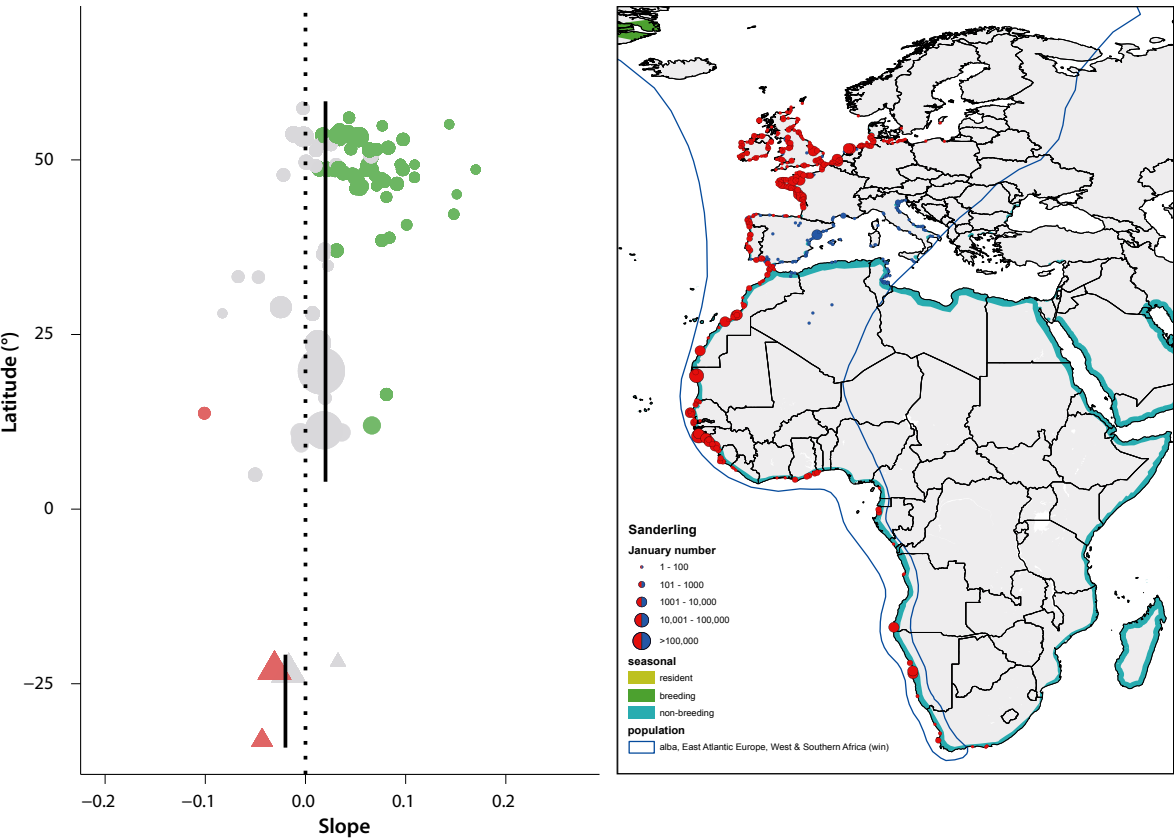


Figure A1.92. Plot (left) of January site trends and distribution (right) of Sanderling in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Sanderling	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
alba, W Europe & W & S Africa (winter)	w	1979-2023	1,023	moderate increase	2014-2023	0,971	moderate decrease	2016- 2020	200000	250000

Table A1.42. Summary of flyway trend and population size estimates for Sanderling. For explanation see table A1.2.

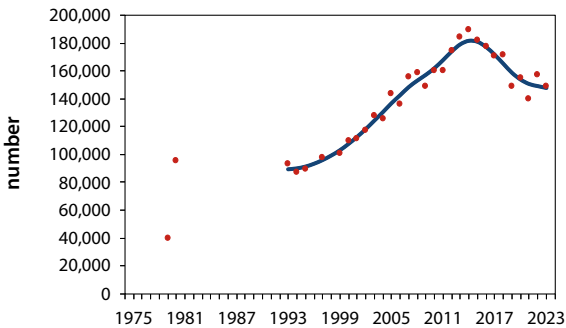


Figure A1.93. Flyway population trend for Sanderling population alba E Atlantic Europe, W & S Africa (win). For explanation see fig. A1.4.



Harvey van diek

Dunlin | *Calidris alpina* | Bécasseau variable

The Dunlin has a flyway population/subspecies structure with large overlap spatiotemporal distribution outside the breeding season. Five populations can be distinguished which use (part of) the EAF. The first is the nominate subspecies *C. a. alpina* breeding in N Scandinavia, N Russia east to Taymyr and wintering mainly in W Europe, but also in Morocco and the W Mediterranean. A second subspecies *C. a. arctica* comprises a relatively small population breeding in NE Greenland and winters in W Africa (see chapter 4). A third, *C. a. schinzii*, is divided into three flyway populations, one breeding in Iceland and wintering in W Africa, one relatively small population breeding in Britain and Ireland and wintering in NW Africa and SW Europe, and one threatened and very small population breeding in the Baltic region and formerly the Wadden Sea and wintering

mainly in NW Africa (Thorup *et al.* 2009, Pakanen *et al.* 2018). Birds counted during the northern winter in W Africa probably belong mainly to the large Icelandic *schinzii* population, while those in W Europe and Morocco are mainly *alpina*. Dunlins breed in a dispersed manner but aggregate in huge flocks at other times of the year. Breeding habitats vary according to latitude, but Dunlins seem to prefer moist ground near open water, ranging from tussock or peat tundra in the Arctic to wet coastal grasslands and wet upland moorland further south. In the non-breeding season the species mainly prefers estuarine mudflats. It feeds on insects, spiders, mites, earthworms, snails, slugs and seeds in the breeding season and mainly on worms, small gastropods, crustaceans in the non-breeding season.



Harvey van diek

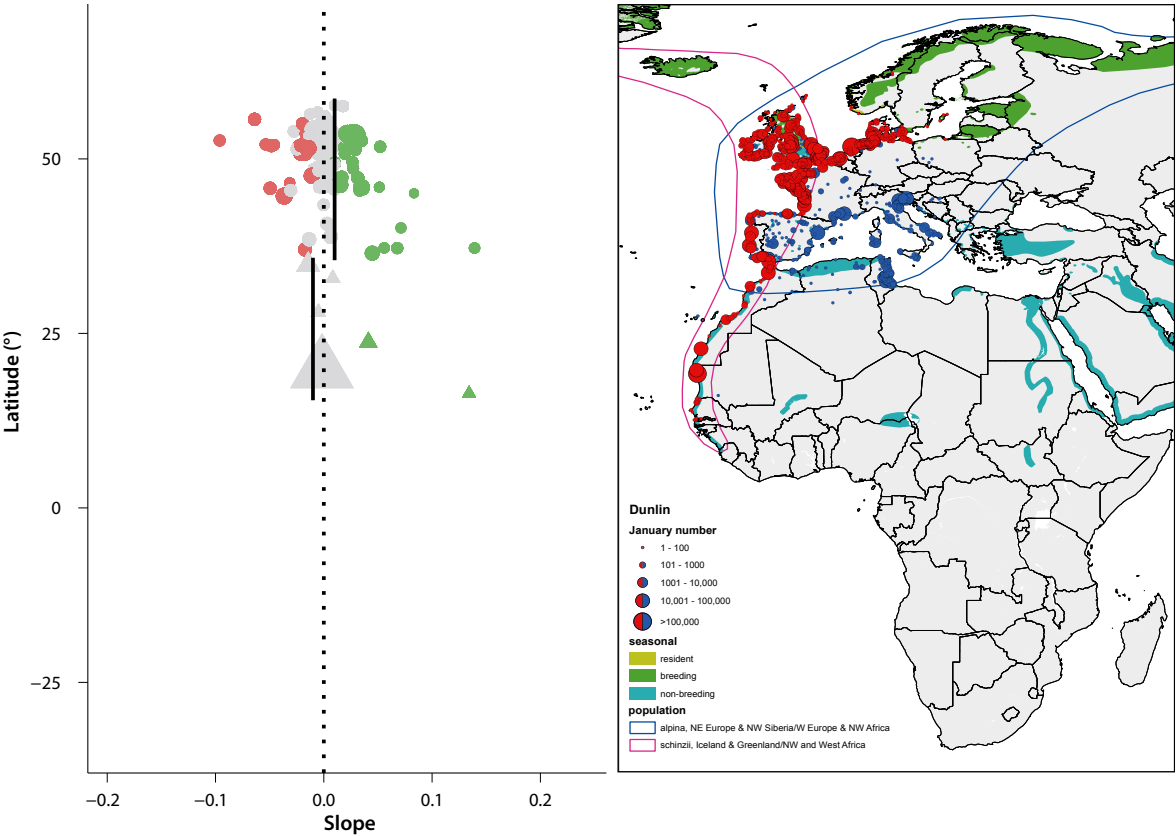


Figure A1.94. Plot (left) of January site trends and distribution (right) of Dunlin in the East Atlantic Flyway in January. For explanation see fig. A1.3.



population Dunlin	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
alpina, NE Europe & NW Siberia / W Europe & NW Africa	w	1976-2023	1,006	moderate increase	2014-2023	1,011	moderate increase	2000- 2018	1300000	1400000
schinzii, Iceland /NW & W Africa	w	1979-2023	0,995	moderate decrease	2014-2023	1,002	stable	2016- 2020	800000	1000000
schinzii, Baltic/SW Europe & NW Africa		1992-2023	0,953	moderate decrease	2014-2023	0,956	moderate decrease	2000	880	1600
schinzii, Britain & Ireland/SW Europe & NW Africa		2001-2023	0,920	strong decrease	2014-2023	1,026	moderate increase	2000	26000	32000

Table A1.43. Summary of flyway trend and population size estimates for Dunlin. For explanation see table A1.2.

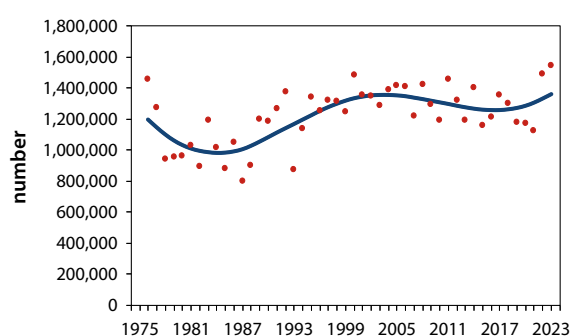


Figure A1.95. Flyway population trend for Dunlin population *alpina* NE Europe & NW Siberia / Europe & NW Africa. For explanation see fig. A1.4.

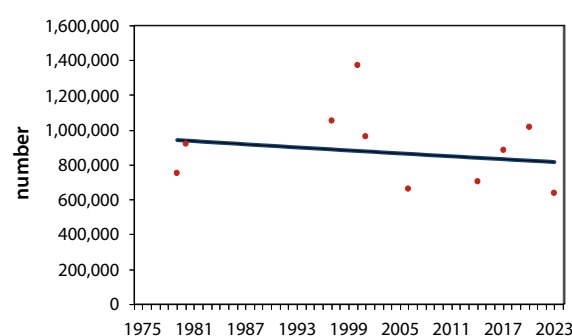


Figure A1.96. Flyway population trend for Dunlin population *schinzii* Iceland / NW & W Africa. For explanation see fig. A1.4.

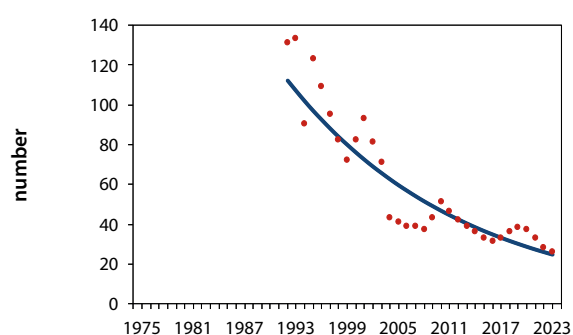


Figure A1.97. Flyway population trend for Dunlin population *schinzii* Baltic/SW Europe & NW Africa. For explanation see fig. A1.4.

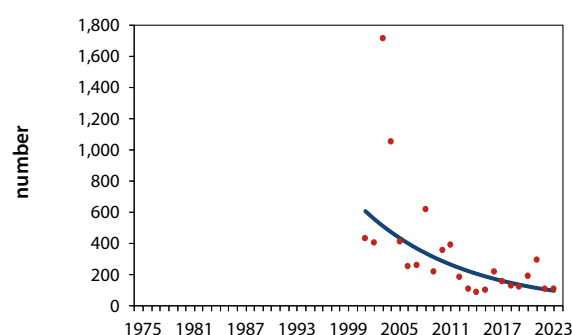


Figure A1.98. Flyway population trend for Dunlin population *schinzii* Britain & Ireland/SW Europe & NW Africa. For explanation see fig. A1.4.

Purple Sandpiper | *Calidris maritima* | Bécasseau violet

The East Atlantic coast is used predominantly for wintering by two flyway populations of Purple Sandpiper: one breeding in NE Canada and Greenland, the other breeding in northern Scandinavia and the Russian Arctic. The majority of the birds that winter in Britain and Ireland originate from Canada (Summers *et al.* 2014). The majority in Norway, Sweden and Denmark belong to the N Scandinavia/Russia population. Both populations mix along the North Sea and English Channel and a combined flyway trend is calculated. Breeding occurs mainly in the Arctic along the coast and in upland areas close to the fringes of snow and ice on wet moss or barren tundra, rocky islands or shingle beaches. During the non-breeding season the species gathers in small flocks along the coast with a preference for rocky shores with strong wave action, and artificial structures such as sea defences and breakwaters. The diet in the breeding season consists mostly of insects and springtails *Collembola*. During the non-breeding season the species feeds mainly on molluscs, small crustaceans, insects, worms, small fish and algae (e.g. Summers *et al.* 1990).



Harvey van diek

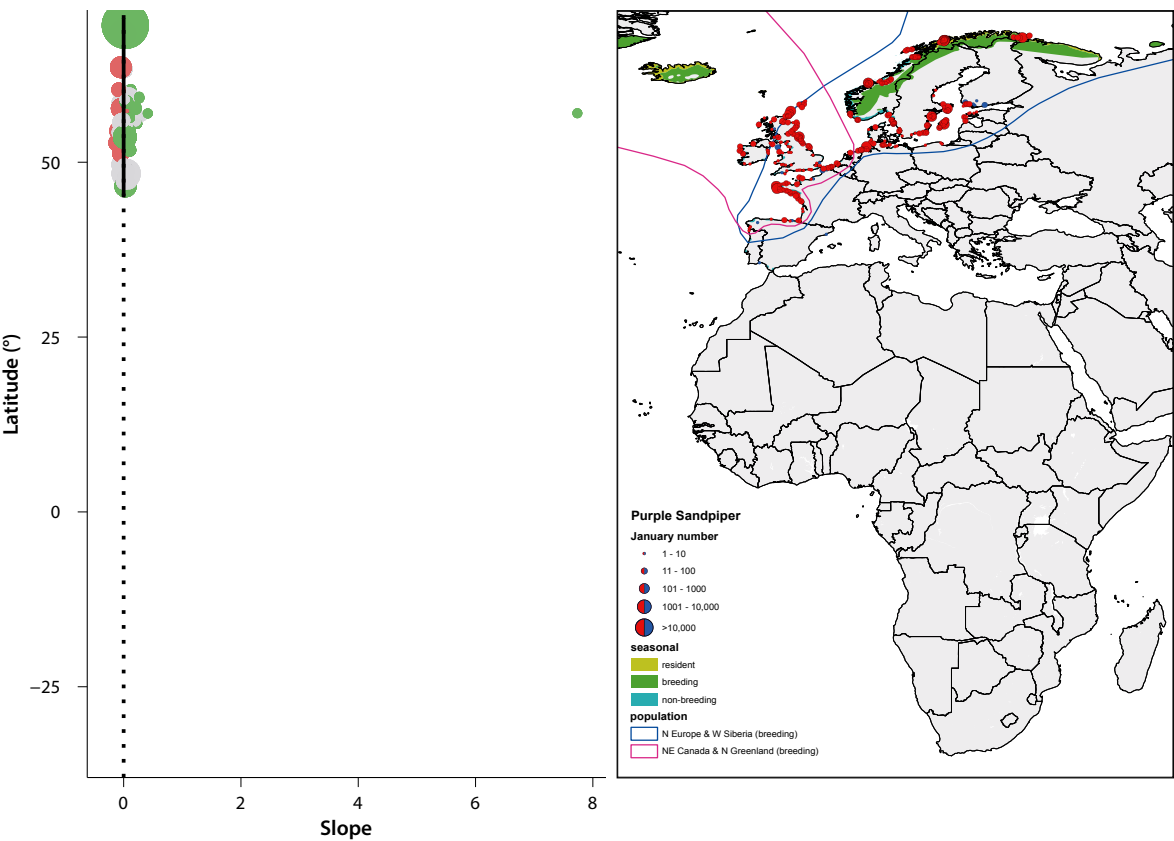


Figure A1.99. Plot (left) of January site trends and distribution (right) of Purple Sandpiper in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Purple Sandpiper	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW Europe (winter), combi of pop.	w	1982-2023	1,002	stable	2014-2023	1,020	uncertain			
N Europe - W Siberia (breeding)								1986-2018	58000	110000
NE Canada - N Greenland (breeding)								1992-2016	11000	12000

Table A1.44. Summary of flyway trend and population size estimates for Purple Sandpiper. For explanation see table A1.2.

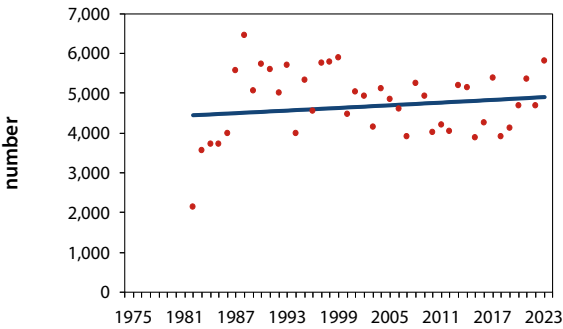


Figure A1.100. Flyway population trend for Purple Sandpiper (populations combined) wintering in NW Europe. For explanation see fig. A1.4.



Harvey van diek

Little Stint | *Calidris minuta* | Bécasseau minute

Two populations of Little Stint are distinguished with relevance to the EAF. Birds of the N Europe population breed in northern Fennoscandia and (mainly) parts of Russia, although the exact borders with the more easterly population breeding in W Siberia is unclear. The N European birds migrate through Europe to winter in S Europe and N and W Africa. W Siberian birds are supposed to migrate to SW Asia and E, C and Southern Africa, where they also reach the East Atlantic coast (but are not considered here as the majority occurs outside the EAF). The species breeds in a dispersed manner, though often in high densities on tundra vegetation at low altitudes. It prefers moist open tundra with dwarf willows or crowberries *Empetrum*. Outside the breeding season it is found in a wide range of freshwater wetlands and on coastal mudflats and seashores. In its African non-breeding range both coastal and inland wetlands are used. The diet in the breeding areas consists primarily of insects. A much wider group of invertebrates, depending on the habitat, is taken outside the breeding season including crustaceans and small molluscs.



Daniele Occhiano / Agami

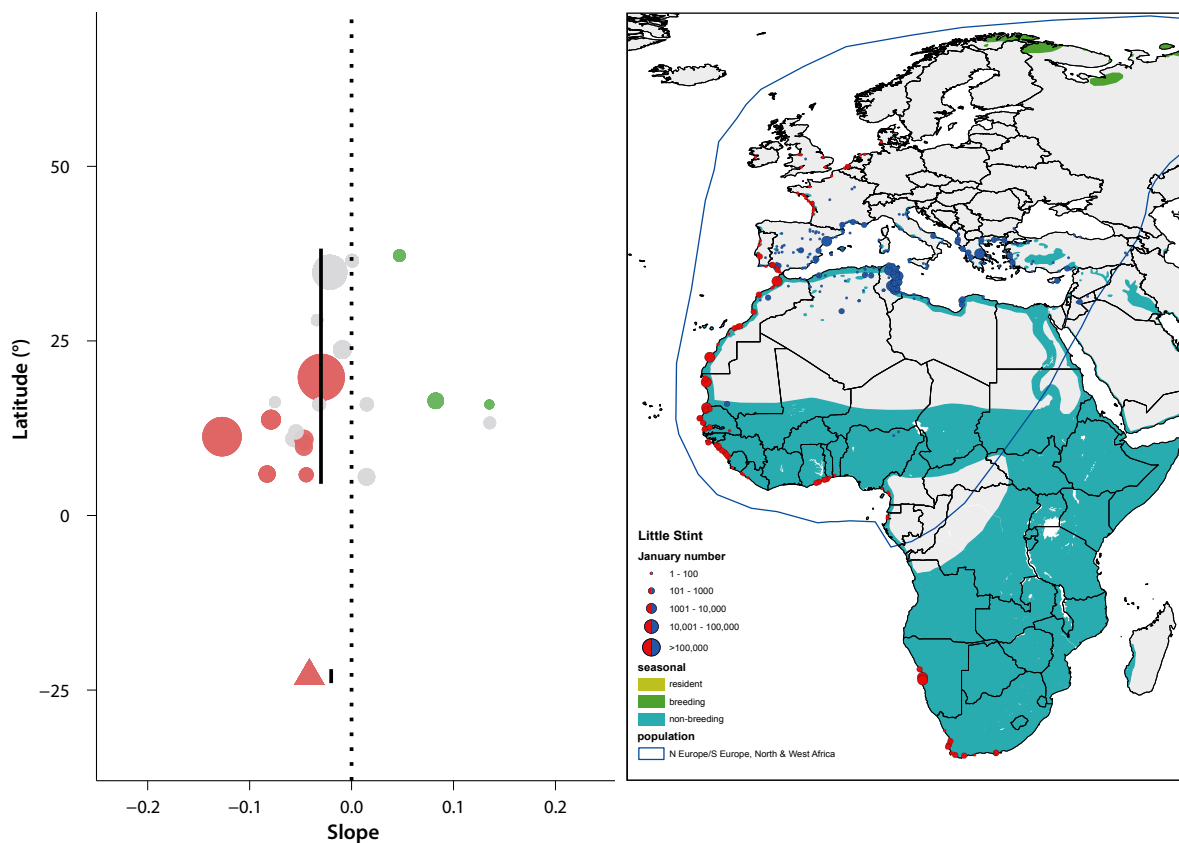


Figure A1.101. Plot (left) of January site trends and distribution (right) of Little Stint in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Little Stint	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
N Europe & W Siberia/N & W Africa	w	1980-2023	0,969	moderate decrease	2014-2023	0,965	moderate decrease	2010- 2018	270000	360000

Table A1.45. Summary of flyway trend and population size estimates for Little Stint. For explanation see table A1.2.

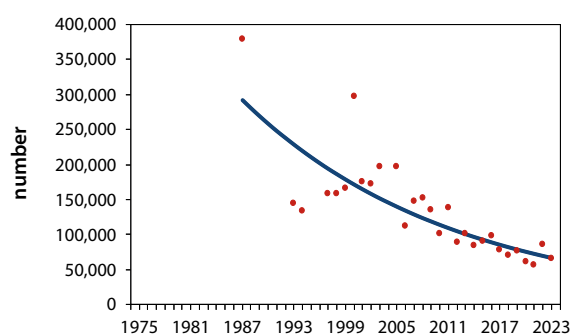


Figure A1.102. Flyway population trend for Little Stint population N Europe / S Europe, N & W Africa. For explanation see fig. A1.4.



Sylvain Rey / Agami

Common Sandpiper | *Actitis hypoleucos* | Chevalier guignette

Common Sandpipers have a large breeding range all over Europe with the largest numbers in Scandinavia and E Europe – W Russia, whilst there is more scattered breeding in W and C Europe. Two flyway populations are distinguished in the Europe-Africa region, with one population (considered here) wintering in S and W Europe and W Africa, and the other using the Middle East and E, C and southern Africa, and also partly the Atlantic coastal zone from Cam-

eroon to South Africa. During breeding Common Sandpipers prefer margins of water bodies, such as rivers, small ponds and lake shores. Outside the breeding season they occur in a wide variety of habitats, both inland and coastal. Besides inland wetlands and riverbanks they favour tidal lagoons and mangrove areas. They forage on a large variety of insects, small fish and crustaceans, often by actively running behind prey, much less by probing mudflats.



Harvey van Diek

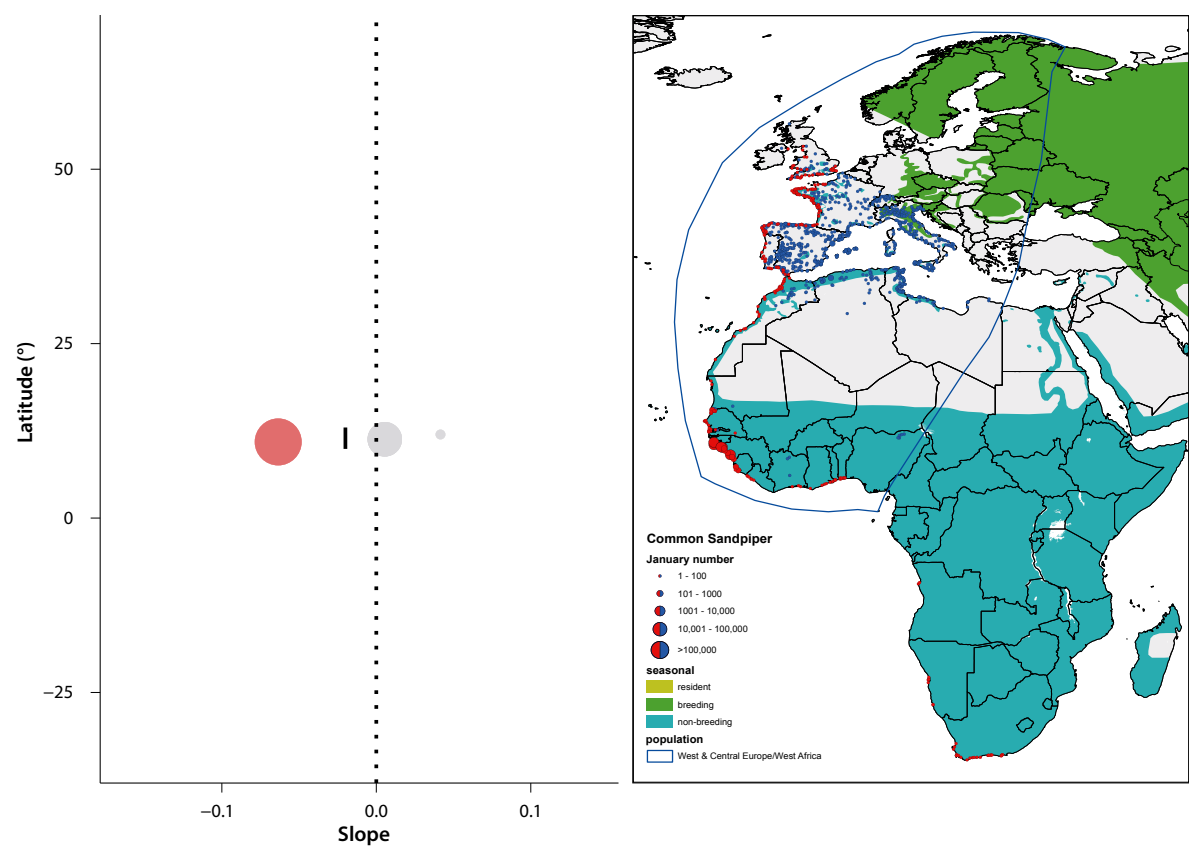


Figure A1.103. Plot (left) of January site trends and distribution (right) of Common Sandpiper in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Common Sandpiper	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W & C Europe/W Africa	w	1987-2023	0,978	moderate decrease	2014-2023	1,017	uncertain	2000- 2019	1100000	1700000

Table A1.46. Summary of flyway trend and population size estimates for Common Sandpiper. For explanation see table A1.2.

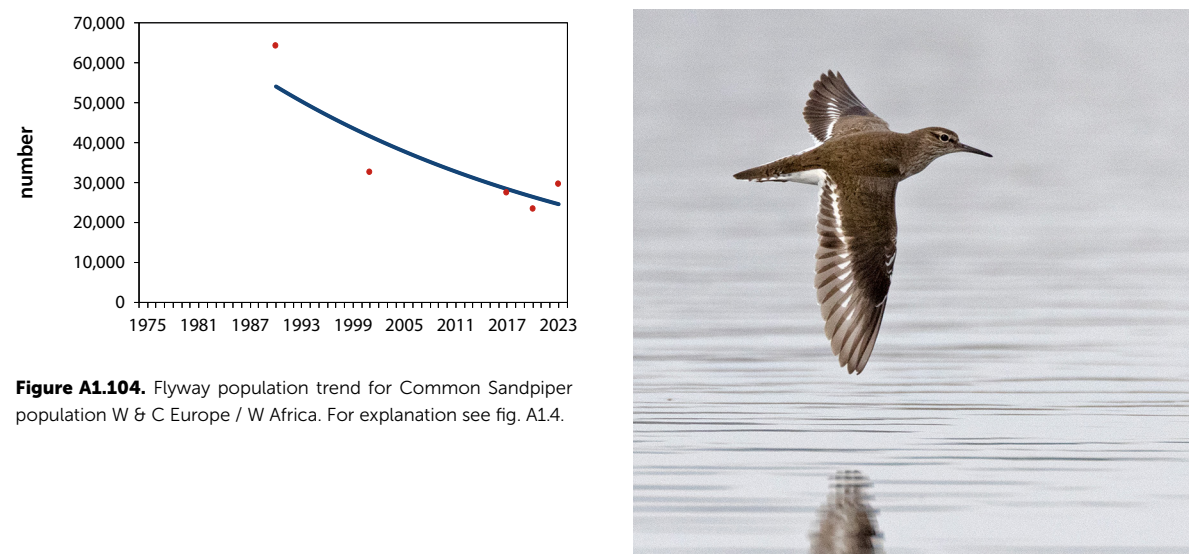


Figure A1.104. Flyway population trend for Common Sandpiper population W & C Europe / W Africa. For explanation see fig. A1.4.

Spotted Redshank | *Tringa erythropus* | Chevalier arlequin

The Spotted Redshank is breeds in northern Fennoscandia and further east in Russia. The entire N European breeding population forms one flyway; the border with the W Siberian population to the east is uncertain. The European breeding population winters around the Mediterranean Sea and in W Africa along the coast as well as in inland wetlands (Senegal, Mali, Nigeria, Chad). The Spotted Redshank breeds dispersed in shrub and open tundra and in marshes south of the arctic treeline. On migration, flocks use specific but widely dispersed staging areas in both fresh, brackish and saltwater wetlands such as lagoons, salt marshes, tidal mudflats, sewage farms and rice fields.

The species forages on invertebrates such as aquatic insects, crustaceans, polychaete worms, and regularly also small fish. The coverage of this population with the IWC is relatively small.



Harvey van Diek

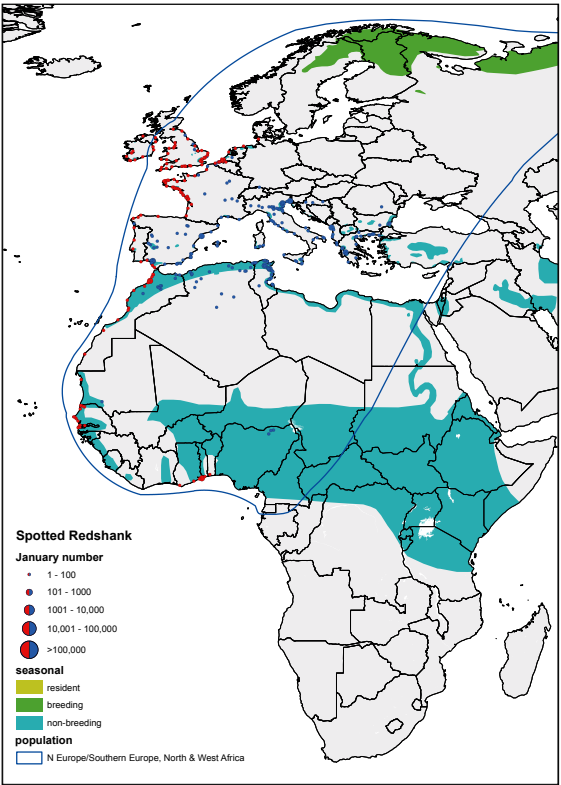


Figure A1.105. Distribution of Spotted Redshank in the East Atlantic Flyway in January. Not enough data exits at present for site trends. For explanation see fig. A1.3.

population Spotted Redshank	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
N Europe /S Europe, N & W Africa	w	1997-2023	0,961	uncertain	2014-2023	0,984	uncertain	2008-2018	57000	120000

Table A1.47. Summary of flyway trend and population size estimates for Spotted Redshank. For explanation see table A1.2.

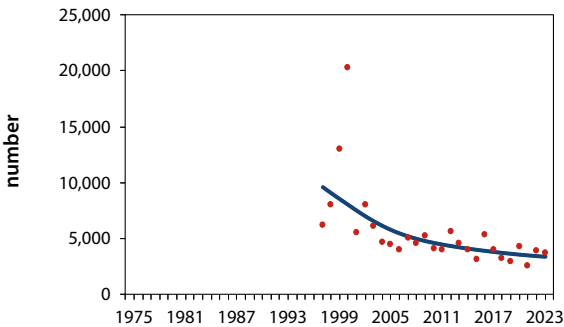


Figure A1.106. Flyway population trend for Spotted Redshank population N Europe / S Europe, N & W Africa. For explanation see fig. A1.4.

Common Greenshank | *Tringa nebularia* | Chevalier aboyeur

The Common Greenshank breeds in boreal and arctic habitats in the north of Europe and Russia. The N Europe breeding population shows a broad-front migration through Atlantic, continental and Mediterranean Europe and mainly winters in W Africa. During this period, birds are found in coastal areas, but also inland in sub-Saharan wetland areas. Greenshanks breeding further east ('W Siberia') form a second flyway population winter in the Middle East and Africa, mainly outside the EAF but including the Atlantic coast from Cameroon to South Africa. Breeding occurs

solitarily in the boreal forest zone in swampy clearings, bogs, marshes and moorlands and at small lakes. During migration and wintering, the species congregates in small flocks, usually of less than 100 individuals. In the wintering areas in Africa, the species occurs in a variety of freshwater, marine and artificial wetlands. On migration it occurs on tidal mudflats and estuaries, but also frequents inland shallow water wetlands. The diet consists of insects, crustaceans, worms, molluscs, amphibians and small fish.

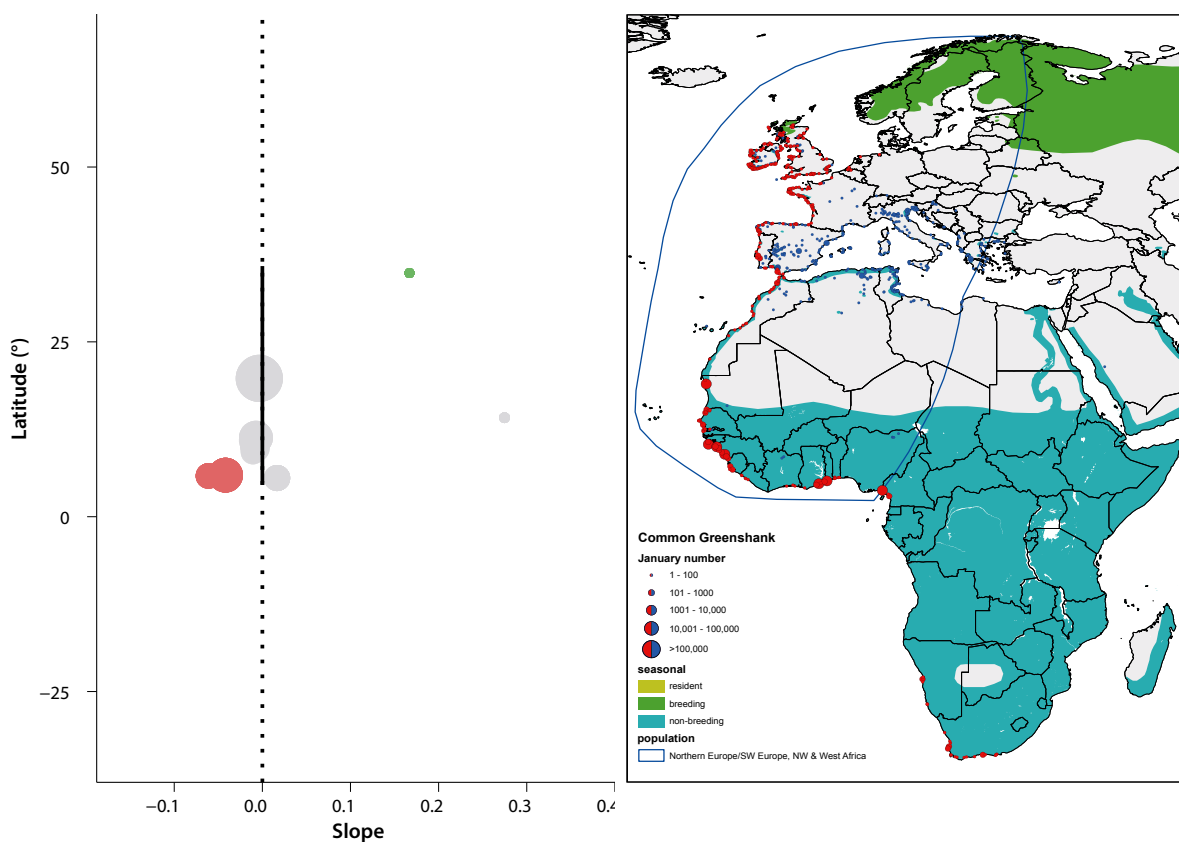


Figure A1.107. Plot (left) of January site trends and distribution (right) of Common Greenshank in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Common Greenshank	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
N Europe/ SW Europe, NW & W Africa	w	1990-2023	1,003	stable	2014-2023	1,006	stable	1995-2018	230000	360000

Table A1.48. Summary of flyway trend and population size estimates for Common Greenshank. For explanation see table A1.2.

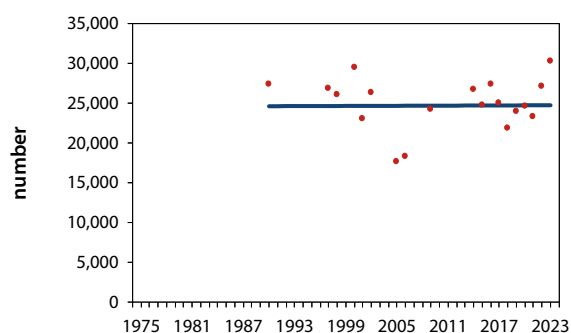


Figure A1.108. Flyway population trend for Common Greenshank population N Europe / SW Europe, NW & W Africa. For explanation see fig. A1.4.



Saverio Gatto / Agami



Marius Varesuo / Agami

Common Redshank | *Tringa totanus* | Chevalier gambette

The Common Redshank breeds in large parts of western, northern and eastern Europe. A complex system of flyway populations has been identified, involving four populations assigned to the EAF: (1) *T. t. robusta* breeding in Iceland and the Faroes and wintering in the North Sea countries and France. (2) *T. t. totanus* breeding in the UK and Ireland being short distance migrants, (3) a northern *T. t. totanus* population breeding in Fennoscandia and mainly wintering on the Atlantic coasts of W Africa, and (4) a mainland Europe *T. t. totanus* population, breeding in con-

tinental W, C and NE Europe and wintering in Iberia, the Mediterranean, and NW Africa. Breeding occurs in a wide variety of habitats: coastal saltmarshes, inland wet grasslands, swampy heathlands and moors and river or lake borders. During the northern winter, however, the species is largely coastal, frequenting a variety of habitats such as beaches, saltmarshes, tidal mudflats, lagoons and estuaries. The diet consists of insects, spiders and annelid worms in the breeding season and mainly worms, crustaceans and molluscs in other seasons.



Arnold Meijer / Blue Robin

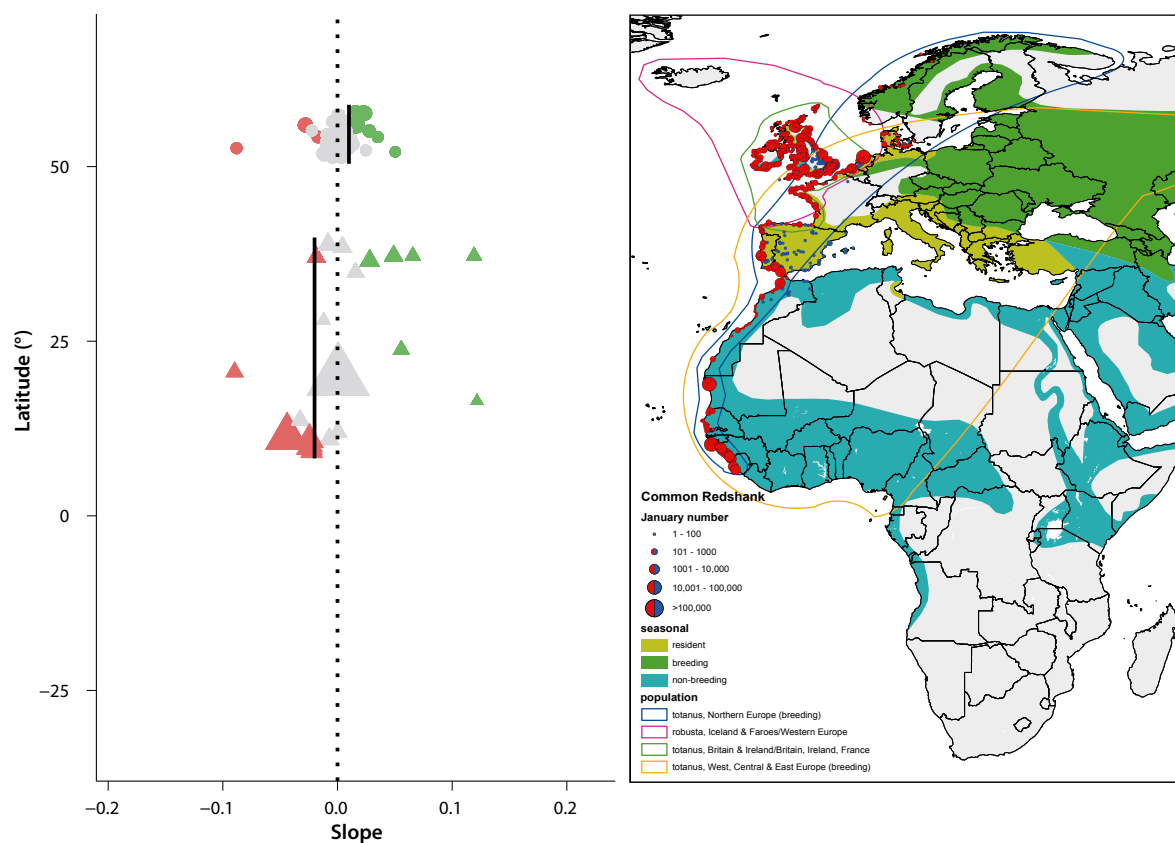


Figure A1.109. Plot (left) of January site trends and distribution (right) of Common Redshank in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Common Redshank	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
<i>robusta</i> , Iceland & Faroes /W. Europe	w	1975-2023	1,008	moderate increase	2014-2023	1,041	moderate increase	2014- 2016	230000	230000
<i>totanus</i> , W, C & E Europe/SW. Europe	b	1994-2023	0,982	moderate decrease	2014-2023	0,985	moderate decrease	1981- 2019	310000	450000
<i>totanus</i> , Britain, Ireland/Britain, Ireland, France	b	1984-2023	0,961	moderate decrease	2014-2023	0,918	strong decrease	2013- 2018	66000	67000
<i>totanus</i> , N Europe /W Africa	b	1991-2023	0,977	moderate decrease	2014-2023	1,009	moderate increase	2008- 2018	160000	240000

Table A1.49. Summary of flyway trend and population size estimates for Common Redshank. For explanation see table A1.2.

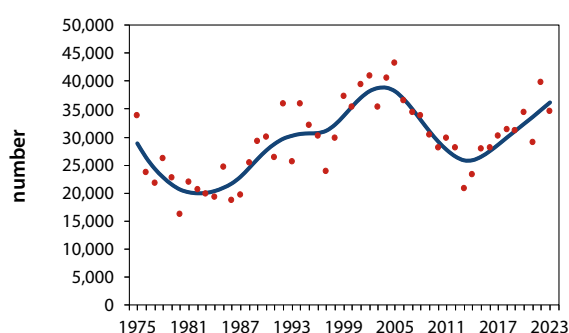


Figure A1.110. Flyway population trend for Common Redshank population *robusta*. For explanation see fig. A1.4.

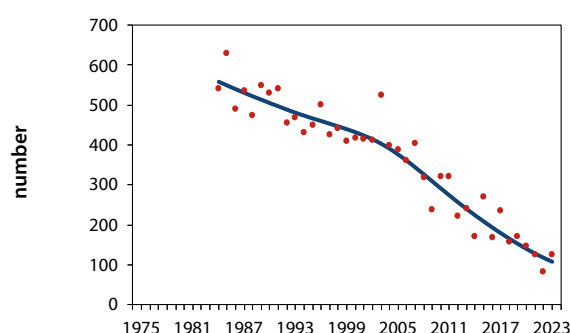


Figure A1.111. Flyway population trend for Common Redshank population *totanus* Britain & Ireland / Britain, Ireland, France. For explanation see fig. A1.4.

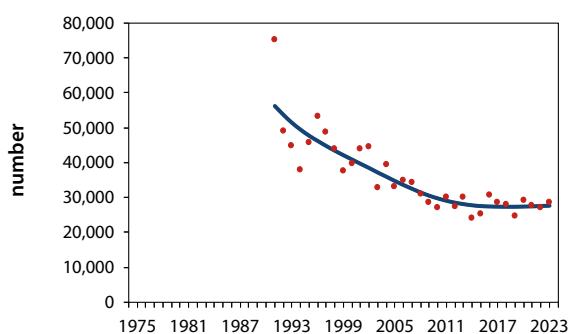


Figure A1.112. Flyway population trend for Common Redshank population *totanus* N Europe / W Africa. For explanation see fig. A1.4.

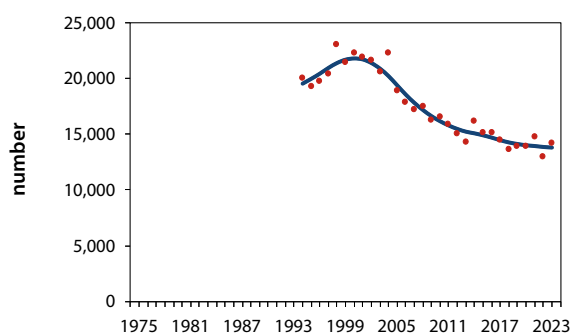


Figure A1.113. Flyway population trend for Common Redshank population *totanus* W, C & E Europe /SW Europe. For explanation see fig. A1.4.



Markus Varesvuo / Agami

African Skimmer | *Rynchops flavirostris* | Bec-en-ciseaux d'Afrique

One biogeographical population of African Skimmer is present in our study area, occurring in both coastal and inland sites. Along the coast it ranges between the Senegal River and Angola, but also occurs inland in the W Sahelian region and in Chad, DRC and Angola. Another biogeographical population of African Skimmer occurs in E and S Africa. African Skimmers are birds from large rivers, coastal lagoons and open marshes. They breed on sandbars. They feed on fish while skimming the water in flight, both during the day and night. They are often encountered in groups resting on sandbars and beaches.

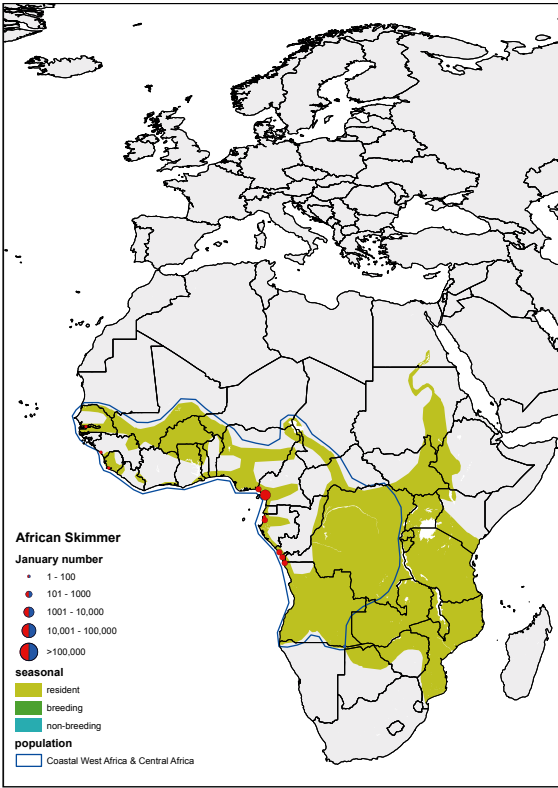


Figure A1.114. Distribution of African Skimmer in the East Atlantic Flyway in January. Not enough data exits at present for site trends. For explanation see fig. A1.3.

population African Skimmer	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
Coastal W & C Africa	w	2007-2023	0,987	uncertain	2014-2023	0,951	uncertain	2016-2020	8000	12000

Table A1.50. Summary of flyway trend and population size estimates for African Skimmer. For explanation see table A1.2.

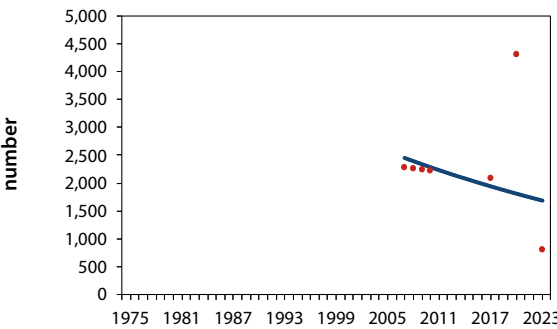


Figure A1.115. Flyway population trend for African Skimmer pop-ulation W & C Africa. For explanation see fig. A1.4.



Slender-billed Gull | *Larus genei* | Goéland railleur

Two biogeographical populations are distinguished within the study area: (1) birds breeding and wintering in the Mediterranean and Black Sea region and wintering in the Mediterranean and along the Atlantic coast of NW Africa, and (2) the resident population of coastal W Africa. The species is gregarious year-round and breeds in monospecific or mixed colonies on beaches, sand spits, islands and coastal marshes near shallow tidal waters and in inland saline seas or lakes. In the non-breeding season it is almost entirely coastal, visiting shallow inshore waters and salt-pans. The diet consists mainly of fish, but also marine invertebrates and insects. Chapter 9 provides more details on breeding of Slender-billed Gull in West Africa.



Hans Schekkerman

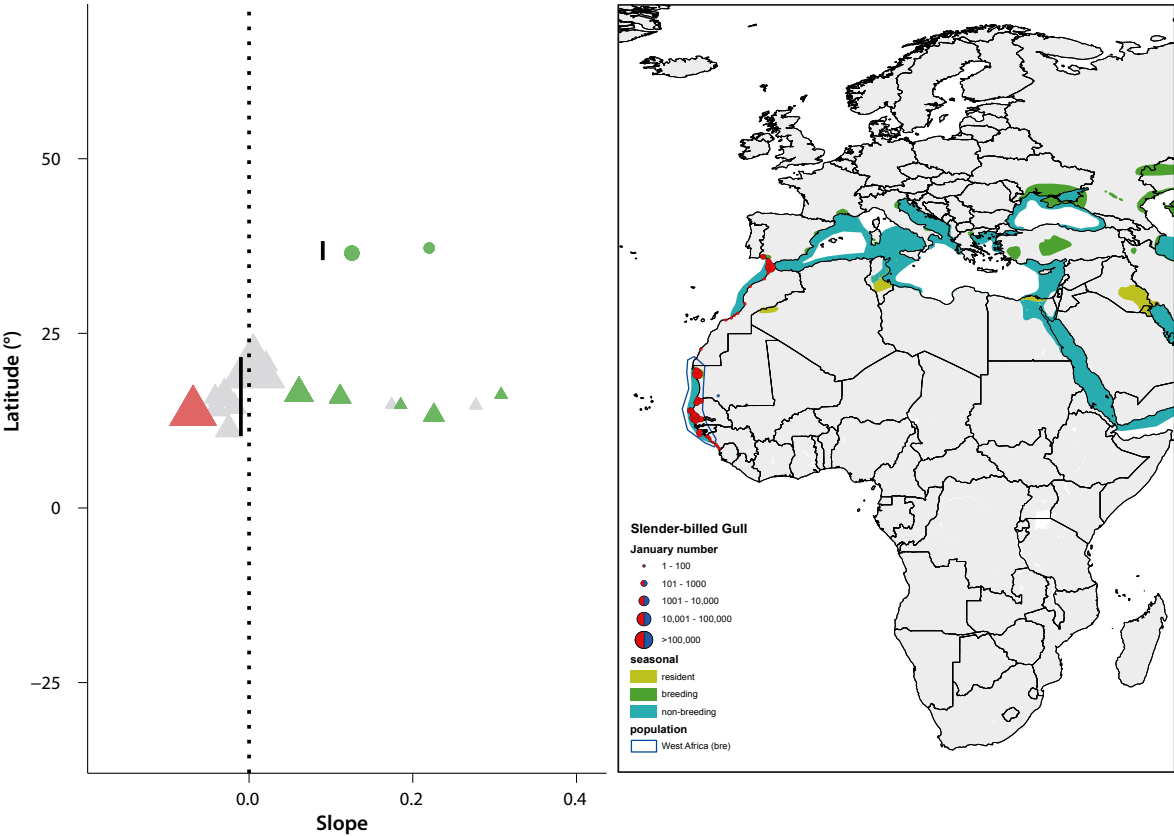


Figure A1.116. Plot (left) of January site trends and distribution (right) of Slender-billed Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Slender-billed Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa	w	1997-2023	0,987	stable	2014-2023	0,980	uncertain	2016-2020	24000	30000

Table A1.51. Summary of flyway trend and population size estimates for Slender-billed Gull. For explanation see table A1.2.

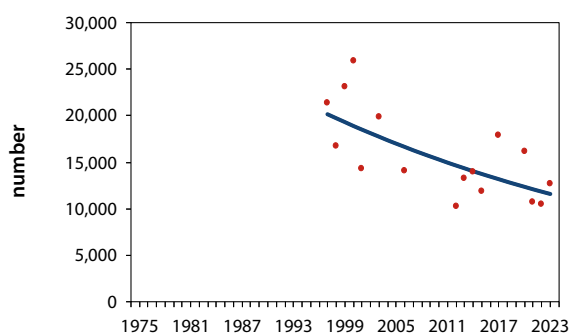


Figure A1.117. Flyway population trend for Slender-billed Gull population W Africa. For explanation see fig. A1.4.



Arnold Meijer / Blue Robin

Black-headed Gull | *Larus ridibundus* | Mouette rieuse

The Black-headed Gull is a common breeding bird in most European countries. The range of the W European flyway population which uses the EAF covers most of Europe including Iceland and the southern tip of Greenland. Large breeding numbers occur in Belarus, the Czech Republic, Denmark, Estonia, Finland, Germany, Lithuania, The Netherlands, Norway, Poland, Sweden and the UK. Northern breeding birds are highly migratory, wintering mainly around the North Sea and in France. Colonial breeding occurs chiefly in inland habitats, though e.g. in The Netherlands a shift from inland to coastal sites has occurred. Breeding habitats range from freshwater wetlands with lush vegetation such as lakes, rivers, marshes with tus-

socks, lowland peat marshes to marine habitats such as estuaries, lagoons, saltmarshes, dunes and offshore islands. In the non-breeding season its distribution is more coastal, including for example estuaries and other tidal waters, but large flocks also occur on farmland (wet grasslands) and in urban areas (city parks, rubbish dumps). The diet is diverse and the species is quite opportunistic. During the breeding season, the birds at inland sites take earthworms, insects and fish, while those occurring along the coast feed on molluscs, crustaceans, worms and fish. In the non-breeding season some populations rely heavily on anthropogenic food sources, for example in urban areas, or scavenging for fish waste while following fishing vessels.



Harvey van Diek

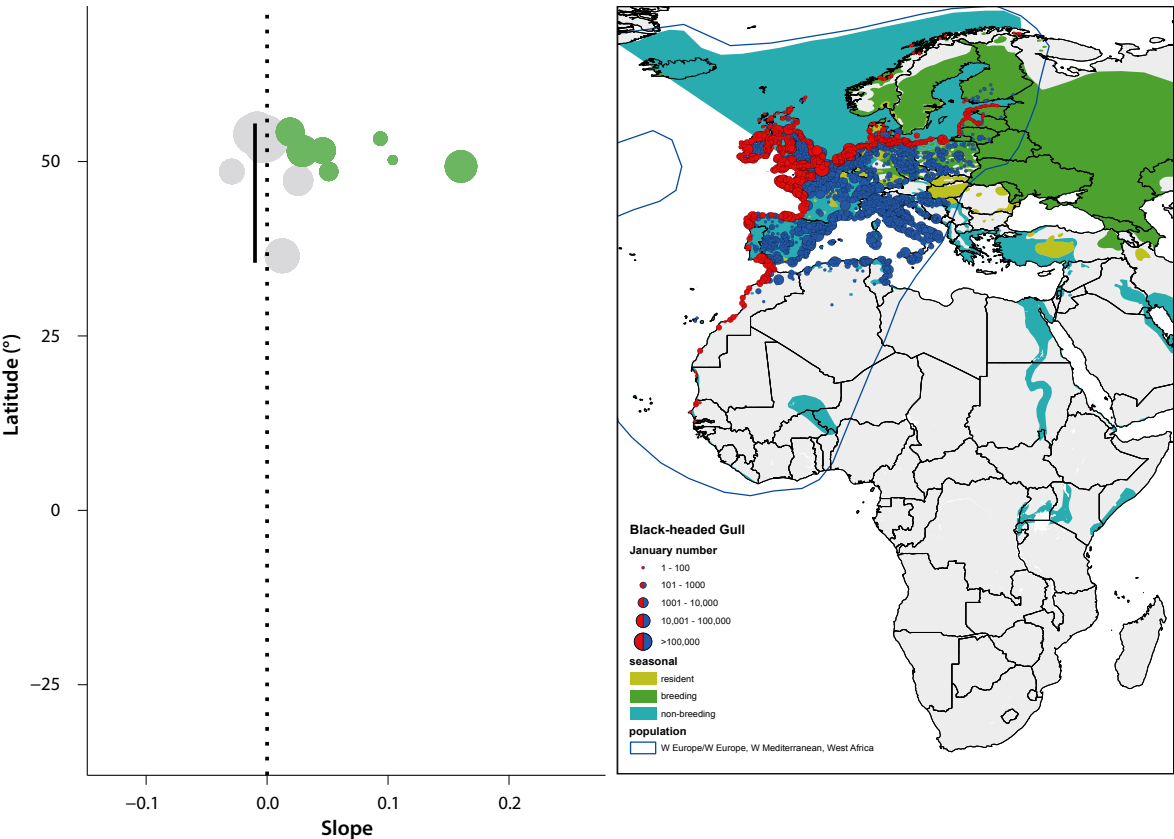


Figure A1.118. Plot (left) of January site trends and distribution (right) of Black-headed Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Black-headed Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Europe/W Europe, W Med - W Africa	w	1993-2023	0,987	moderate decrease	2014-2023	0,992	moderate decrease	1981- 2018	2500000	3400000

Table A1.52. Summary of flyway trend and population size estimates for Black-headed Gull. For explanation see table A1.2.

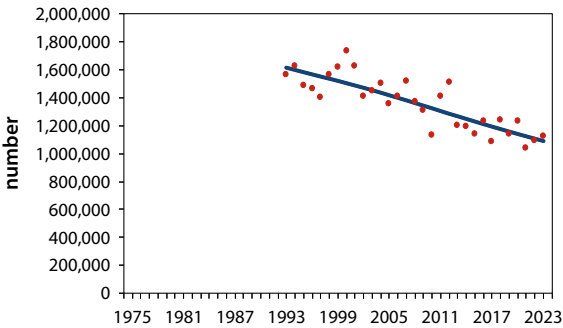


Figure A1.119. Flyway population trend for Black-headed Gull population W Europe/W Europe, W Med. – W Africa. For explanation see fig. A1.4.



Harvey van Diek

Hartlaub's Gull | *Larus hartlaubii* | Mouette de Hartlaub

The entire world population of this species breeds along the coast of Namibia and South Africa, where it is mostly sedentary. The species is gregarious year-round, breeding in colonies of up to 1,000 pairs, occasionally with Greater Crested Terns or other colonial species; it also forages and roosts in groups during the non-breeding season. The

species is strictly coastal and breeds on offshore flat rocky islands near kelp beds in shallow waters, frequenting estuaries, lagoons, beaches and occasionally rubbish dumps and sewage and salt works. It feeds mainly on invertebrates associated with stranded kelp, but also terrestrial insects, fish, earthworms, fruits and garbage.



Wil Leurs / Agami

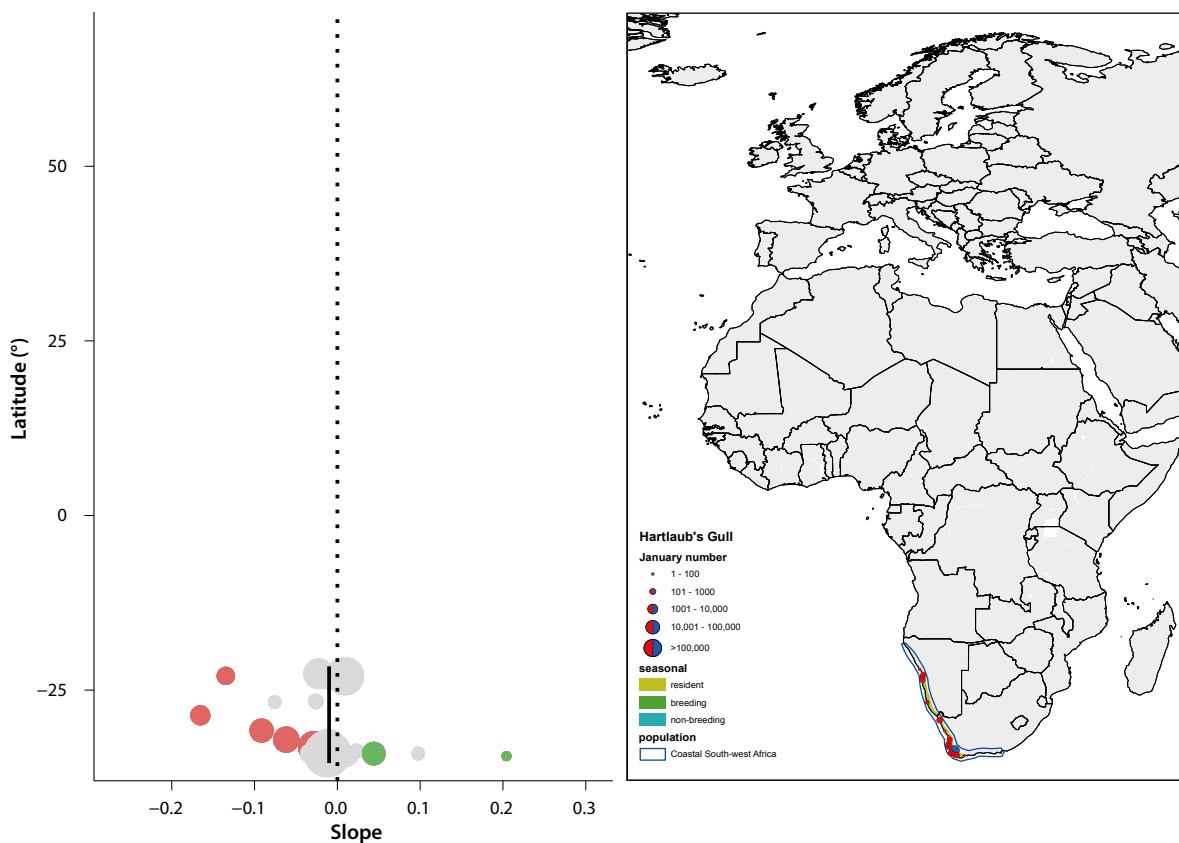


Figure A1.120. Plot (left) of January site trends and distribution (right) of Hartlaub's Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Hartlaub's Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
coastal SW Africa	w	1992-2023	0,987	moderate decrease	2014-2023	0,947	moderate decrease	2018	17000	24000

Table A1.53. Summary of flyway trend and population size estimates for Hartlaub's Gull. For explanation see table A1.2.

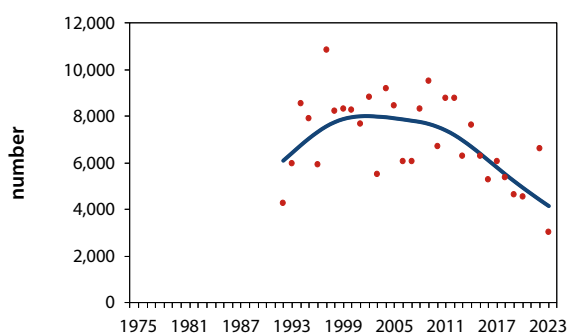


Figure A1.121. Flyway population trend for Hartlaub's Gull population SW Africa. For explanation see fig. A1.4.



WIL Leurs / Agami

Grey-headed Gull | *Larus cirrocephalus* | Mouette à tête grise

The Grey-Headed Gull breeds in sub-Saharan Africa and South America. Three flyway populations are distinguished within the African continent, of which two are relevant to the EAF: those breeding in W Africa and in coastal southern Africa. In W Africa this largely resident species is a coastal colonial breeder, but also occurs on large inland lakes (e.g. in Chad and Mali). Important breeding sites are

in Senegal, The Gambia and Guinea-Bissau, where it breeds on offshore islands and in estuaries (see chapter 9). The same habitats are frequented outside the breeding season. Its diet consists predominantly of fish and invertebrates. Grey-headed Gulls further south along the EAF belong mainly to the coastal Southern African population.



Jacob Garvelink / Agami

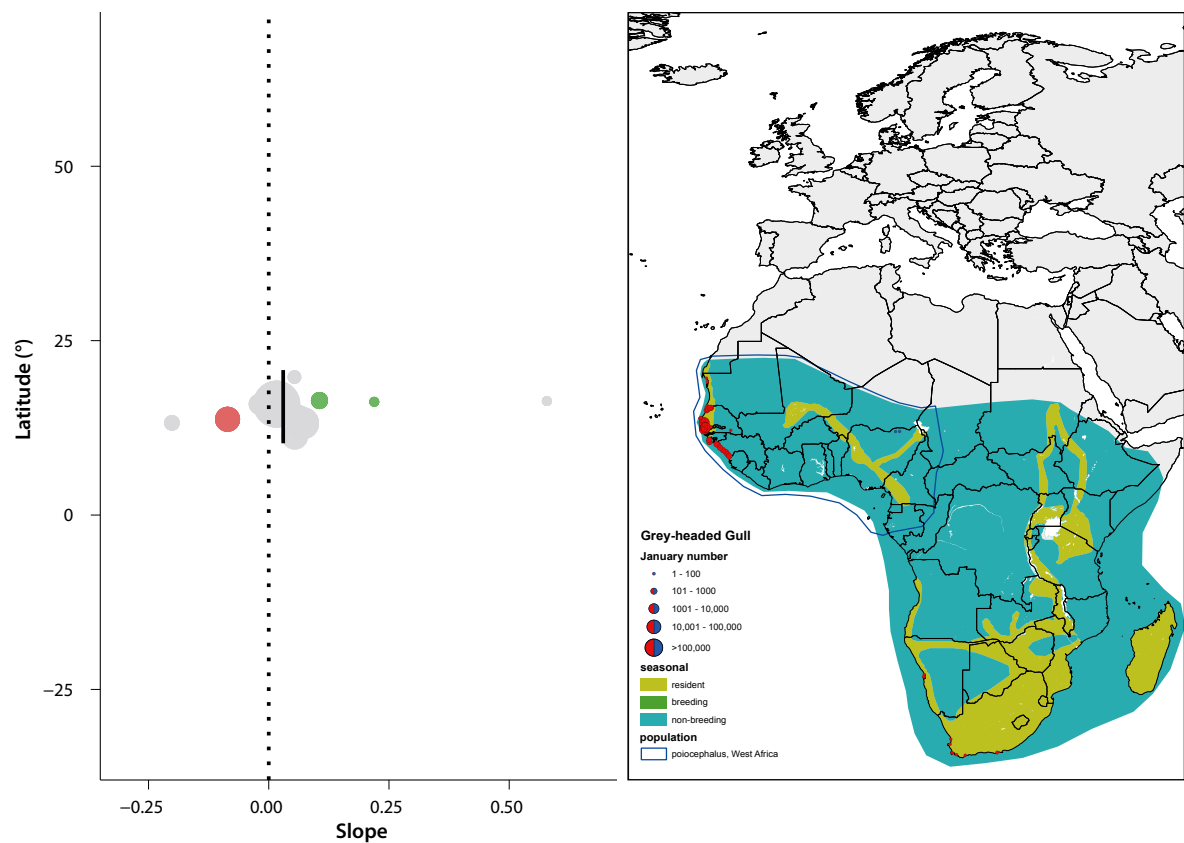


Figure A1.122. Plot (left) of January site trends and distribution (right) of Grey-headed Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Grey-headed Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Africa	w	1997-2023	1,029	uncertain	2014-2023	0,973	uncertain	2016-2020	25000	35000

Table A1.54. Summary of flyway trend and population size estimates for Grey-headed Gull. For explanation see table A1.2.



Josh Jones / Agami

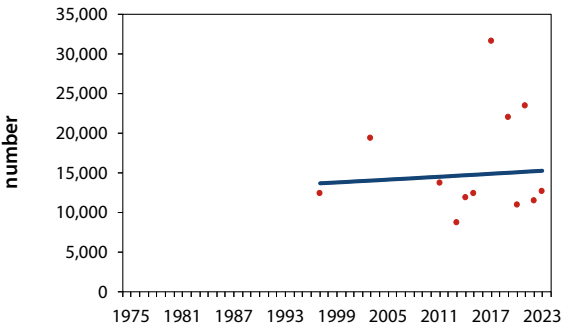


Figure A1.123. Flyway population trend for Grey-headed Gull population poiocephalus W Africa. For explanation see fig. A1.4.

Audouin's Gull — *Larus audouinii* - Goéland d'Audouin

One biogeographical population exists of this species in the EAF, which equates to the entire world population. The main breeding colonies are found in the western Mediterranean region, and the vast majority breed in Spain. The species spends the winter along the coast of the W Mediterranean, S Iberia and (particularly) NW Africa, east to Libya and south to The Gambia. Breeding colonies are in variable habitats on rocky cliffs, offshore islands, salt-

marshes or sandy peninsulas. During the non-breeding season the species prefers sheltered bays and beaches with freshwater outlets. The diet consists mainly of epipelagic fish, although the large colony of the Ebro Delta has adopted more terrestrial foraging habits, including feeding on invasive crayfish in rice fields, food discards and fish waste dumped from boats.

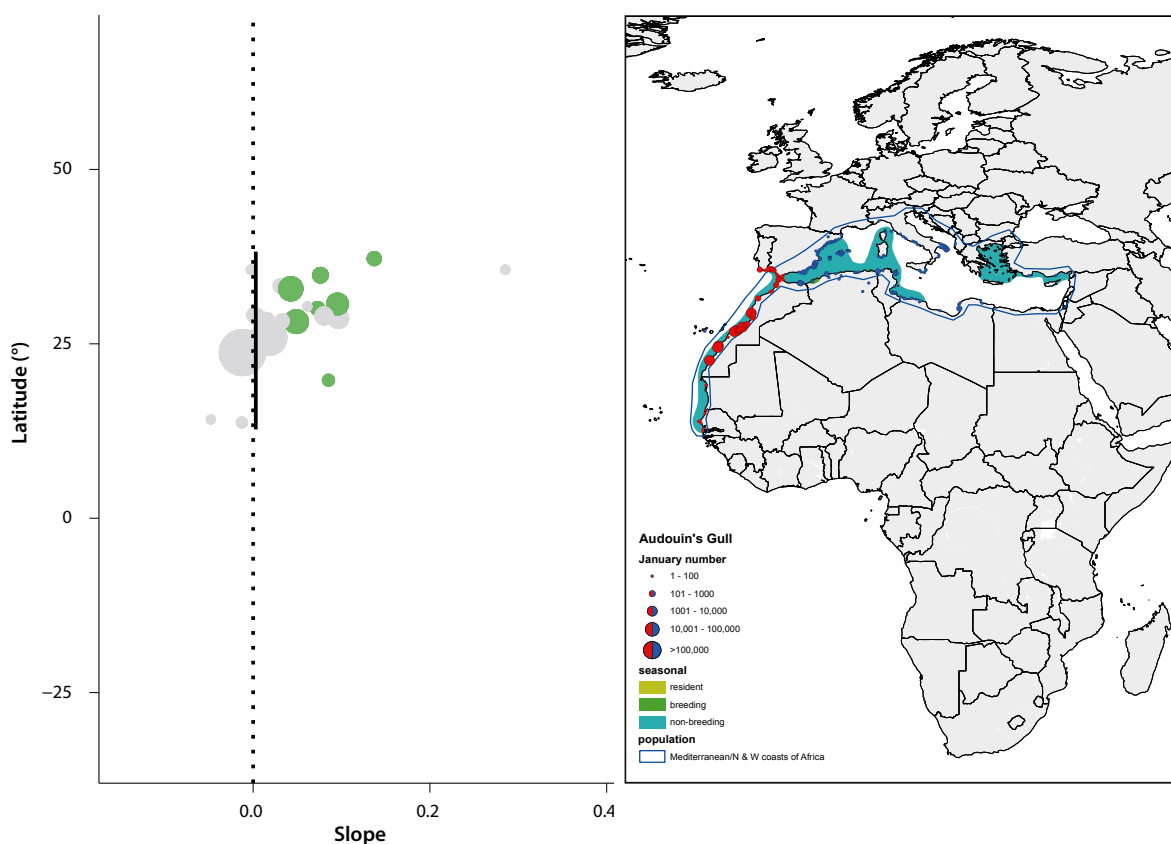


Figure A1.124. Plot (left) of January site trends and distribution (right) of Audouin's Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.



Harvey van Diek



Arnold Meijer / Blue Robin

population Audouin's Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
Mediterranean/N & W Africa	b	1980-2018	1,003	moderate increase	2009-2018	0,866	strong decrease	1998- 2019	48000	66000

Table A1.55. Summary of flyway trend and population size estimates for Audouin's Gull. For explanation see table A1.2.



Arnold Meijer / Blue Robin

Mew Gull | *Larus canus* | Goéland cendré

Two subspecies of this gull species are distinguished in Europe, the nominate *L. c. canus* and Russian *L. c. heinei*. The nominate subspecies occurs in the EAF, breeding in large areas of N and E Europe and wintering in W and C Europe, including offshore areas. Breeding numbers are high in some Nordic countries such as Denmark, Norway, Sweden, Finland, Estonia and European Russia, but also in Germany and the UK. Breeding occurs in smaller numbers in many other countries in W and C Europe. Countries around the North Sea support the majority of wintering birds. This gull breeds in solitary pairs and (mixed) colonies in a variety of coastal and inland habitats, including dune areas, beaches, grassy islands and rocky or grassy cliff ledges along the coast and small islands or shores of inland waterbodies or in bogs. It occupies similar habitats outside the breeding season, and is often found foraging in agricultural grasslands and on intertidal mudflats, but also in urban habitats and at sea, usually in flocks. The diet consists of earthworms and insects in terrestrial habitats and crustaceans, molluscs and fish in marine habitats.



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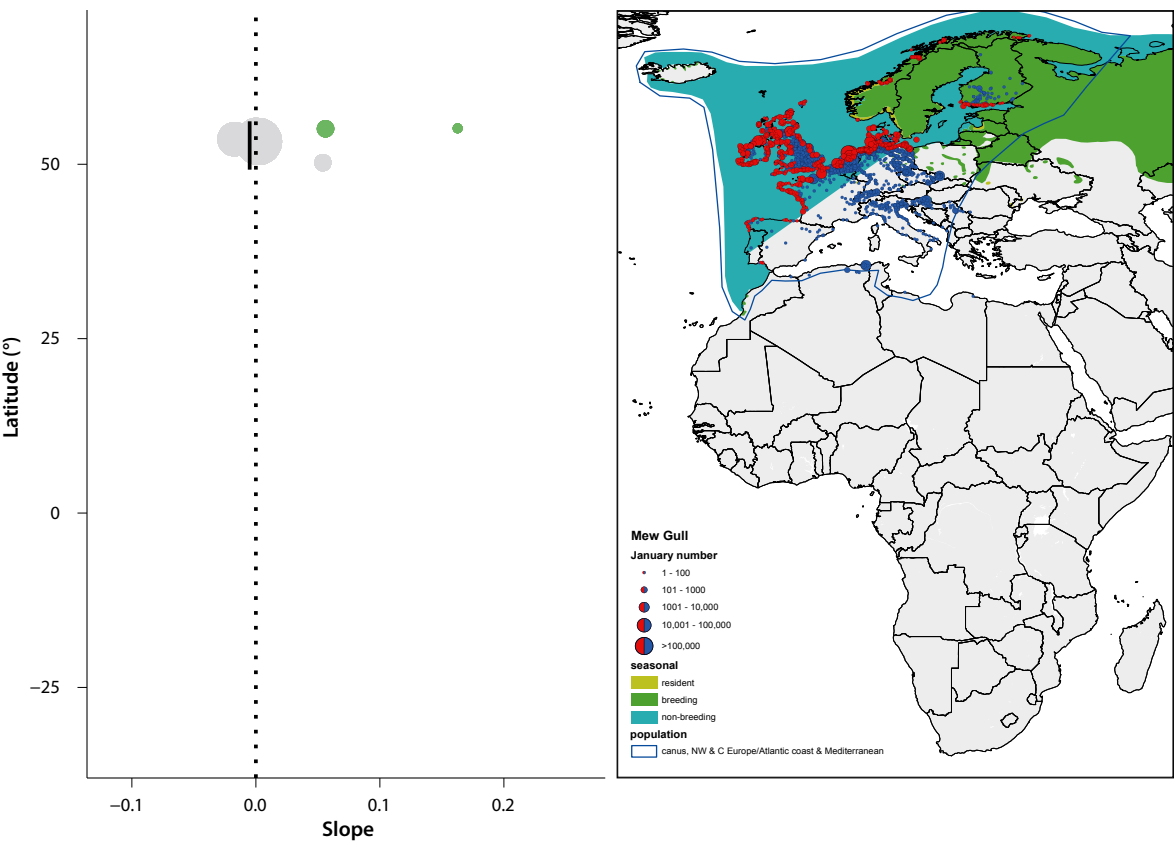


Figure A1.125. Plot (left) of January site trends and distribution (right) of Mew Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.



Harvey van Diek

population Mew Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW & C Europe /NW Europe & W Med.	b	1980-2018	0,995	stable	2009-2018	0,985	moderate decrease	1981- 2018	1400000	2000000

Table A1.56. Summary of flyway trend and population size estimates for Mew Gull. For explanation see table A1.2.

Lesser Black-backed Gull | *Larus fuscus* | Goéland brun

Within the EAF study area two populations are distinguished, *L. f. graellsii* and *L. f. intermedius* with overlapping wintering ranges. *Graellsii* breeds mainly in Iceland, the British Isles, The Netherlands, France, Spain and Portugal and winters from SW Europe to W Africa (Hallgrímsson et al. 2012, Baert et al. 2018). *Intermedius* breeds in coastal Norway and southern Sweden, Denmark, Germany and The Netherlands, wintering largely in the same areas as *graellsii* (Helberg et al. 2009). Breeding occurs in colonies, often mixed with Herring Gulls, on coastal grassy slopes, saltmarshes, sand dunes, cliffs, offshore and inland islands, lake margins and increasingly on flat rooftops. During the non-breeding season the species remains gregarious, with flocks on beaches, in harbours, estuaries, lagoons and occasionally inland close to lakes or rivers. The species forages opportunistically year-round in marine habitats, also following fishing vessels, and inland on agricultural fields, rubbish dumps and in cities. The diet includes fish, aquatic and terrestrial invertebrates, eggs, seeds and carrion.



Harvey van Diek

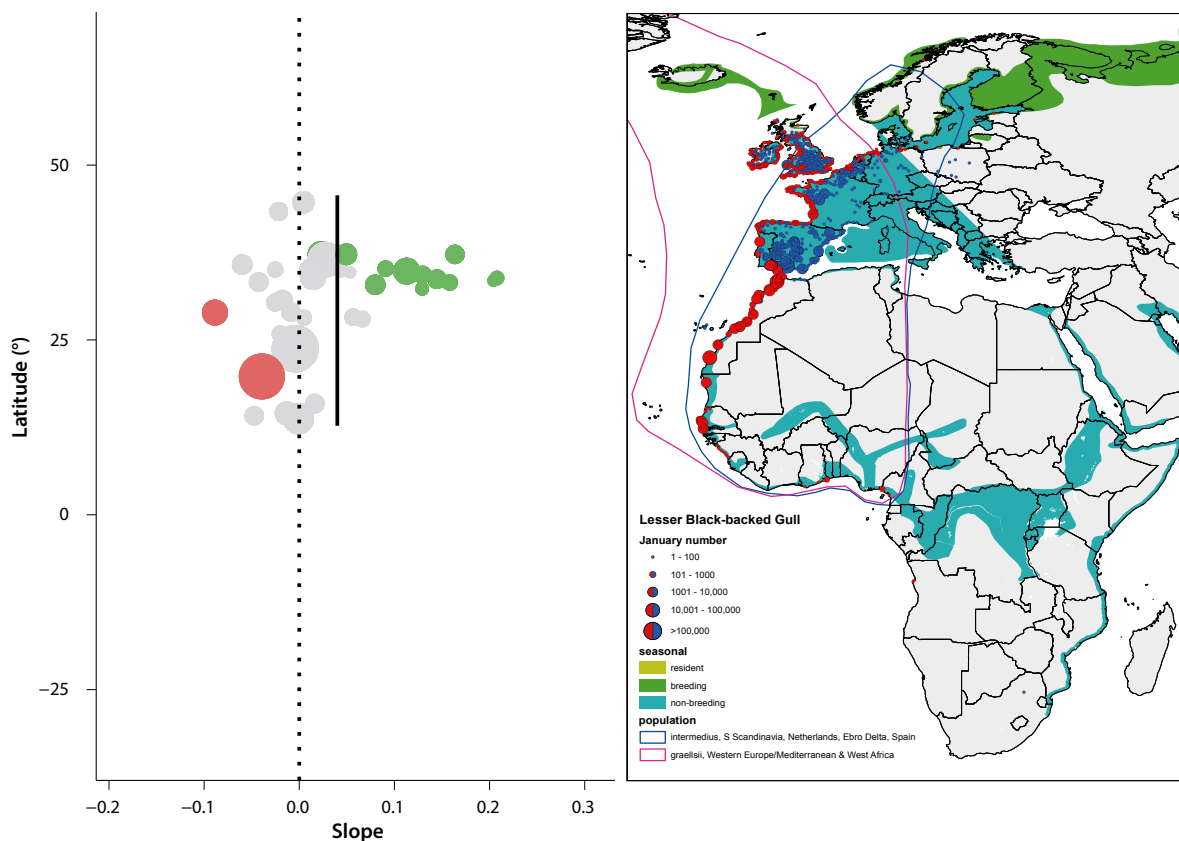


Figure A1.126. Plot (left) of January site trends and distribution (right) of Lesser Black-backed Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Lesser Black-backed Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
<i>graellsii</i> , NW Europe /East Atlantic	b	1980-2018	1,045	moderate increase	2009-2018	0,917	moderate decrease	1981- 2018	480000	500000
<i>intermedius</i> , W Europe /East Atlantic	b	1980-2018	1,045	moderate increase	2009-2018	1,003	stable	2013- 2018	560000	610000

Table A1.57. Summary of flyway trend and population size estimates for Lesser Black-backed Gull. For explanation see table A1.2.

European Herring Gull | *Larus argentatus* | Goéland argenté

The European Herring Gull occurs in two subspecies in the EAF: the nominate *L. a. argentatus* breeding in Fennoscandia and European Russia and *L. a. argenteus* breeding in countries around the North Sea and NW Europe including Iceland. Large numbers of the nominate form breed in Denmark, Norway, Sweden, Finland, Estonia and Russia. They are partial migrants, with some birds wintering further south e.g. along the North Sea coasts. Large *argenteus* populations occur mainly in the UK, France and The Netherlands, and are mainly short-distance migrants (Camphuysen 2013). Breeding occurs in colonies mostly in

or near coastal areas, in a wide variety of habitats, for example islands with grassy vegetation, dune areas, sandy beaches, rocky outcrops and roofs in urban areas. In the non-breeding season a wide variety of habitats is also used, but populations in W Europe seem to prefer intertidal habitats including tidal mudflats beaches and along seawalls. The species is opportunistic, certainly in the breeding season, and will take almost any food available. Outside the breeding season it has a preference for bivalves (mussels, cockles) in tidal habitats and along beaches which is more marked than in other gull species.



Arnold Meijer / Blue Robin

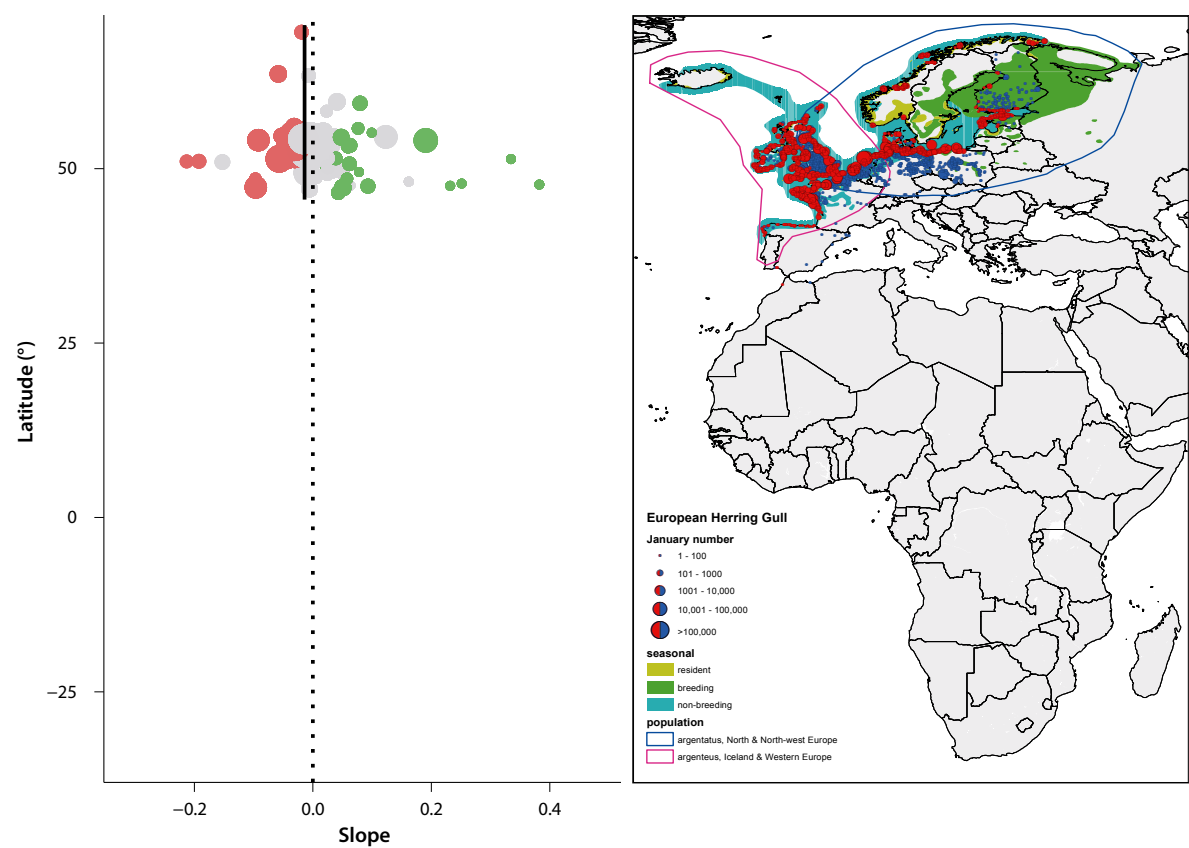


Figure A1.127. Plot (left) of January site trends and distribution (right) of European Herring Gull in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population European Herring Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
argenteus, NW Europe/East Atlantic	b	1980-2018	0,988	moderate decrease	2009-2018	0,966	moderate decrease	1981- 2018	740000	780000
argentatus, W Europe /East Atlantic	b	1980-2018	0,986	moderate decrease	2009-2018	0,891	strong decrease	2008- 2018	860000	1000000

Table A1.58. Summary of flyway trend and population size estimates for European Herring Gull. For explanation see table A1.2.



Harvey van Diek

Little Tern | *Sternula albifrons* | Sterne naine

The Little Tern is a widely, though generally sparsely, distributed tern breeding in Europe and Africa as well as Asia and Oceania. In Europe, relatively small breeding numbers occur in most countries, both coastal and inland. Along the EAF, three populations have been identified covering two subspecies: the nominate *S. a. albifrons* with a population breeding in NW Europe and another breeding in the W Mediterranean, and a resident rather unknown subspecies (*S. a. guineae*) in W Africa. Important breeding populations in (S)W Europe occur in Spain, Italy, France and the

UK. The *guineae* population breeds in widely dispersed small colonies along the coast from Mauritania to Cameroon and in inland river basins (Niger, Chad). European birds migrate to W Africa outside the breeding season. The breeding habitats are coastal (beaches, sandy islands, salt-marshes etc.) but also include shores and islands of large rivers and lakes. Outside the breeding season, coastal waters are preferred and foraging occurs in tidal creeks, lagoons and saltpans. Its diet consists mainly of small fish and crustaceans.

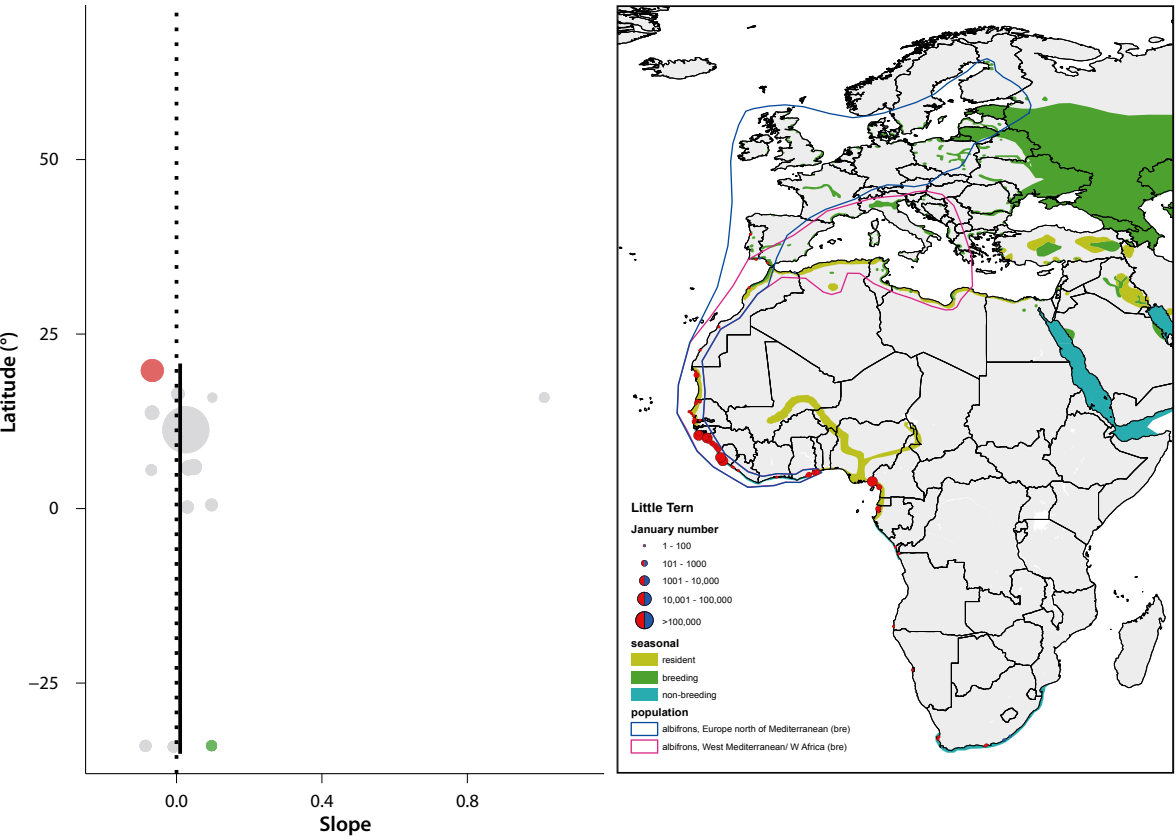


Figure A1.128. Plot (left) of January site trends and distribution (right) of Little Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Little Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
Europe north of Mediterranean / East Atlantic	b	1980-2018	1,001	uncertain	2009-2018	1,020	moderate increase	2012- 2018	21000	26000
West Mediterranean/ East Atlantic	b	1980-2018	0,989	moderate decrease	2009-2018	1,022	moderate increase	2006- 2018	16300	26000

Table A1.59. Summary of flyway trend and population size estimates for Little Tern. For explanation see table A1.2.



Marc Guyt / Agami

Damara Tern | *Sternula balaenarum* | Sterne des baleiniers

Damara Terns breed in coastal areas of Namibia and South Africa and winter further north, maybe as far as Ghana and Côte d'Ivoire but with the majority from Cameroon to South Africa. The species breeds in colonies on gravel and

in stony places, often some distance inland, and also in salt pans and on deserted beaches. Outside the breeding season it occurs on exposed coasts where it forages in shallow water and feeds on small fish.



Pete Morris / Agami

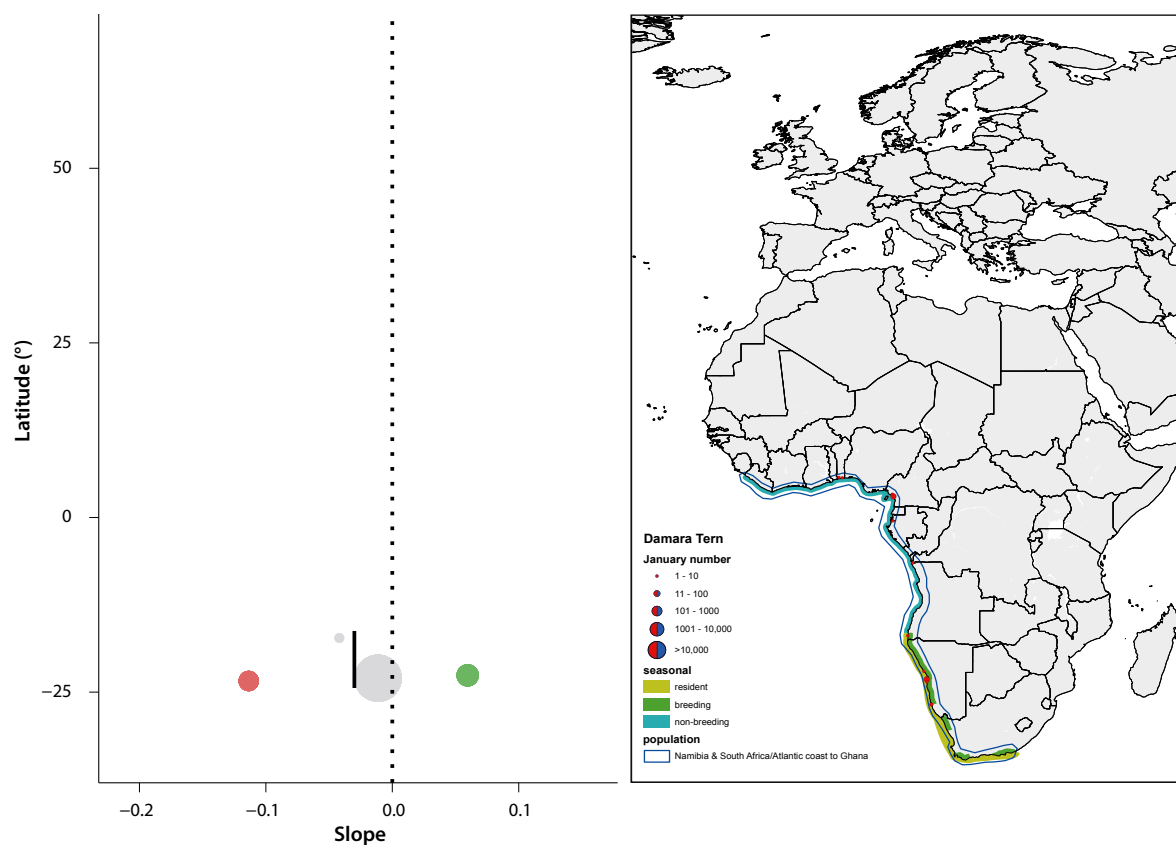


Figure A1.129. Plot (left) of January site trends and distribution (right) of Damara Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Damara Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Namibia & South Africa	w	1992-2023	0,966	moderate decrease	2014-2023	0,927	uncertain	2010- 2011	3400	8500

Table A1.60. Summary of flyway trend and population size estimates for Damara Tern. For explanation see table A1.2.

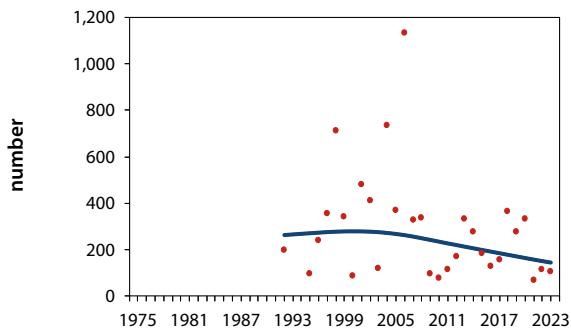


Figure A1.130. Flyway population trend for Damara Tern population Namibia & S Africa (bre). For explanation see fig. A1.4.

Common Gull-billed Tern | *Gelochelidon nilotica* | Sterne hansel

The Common Gull-billed Tern breeds across a wide range in Europe and Africa. The few breeders of NW Europe and those in the W Mediterranean area and NW and W Africa are considered to belong to the same flyway population. European breeders are strictly migratory, to W Africa where they mix with the largely resident African sub-population. In W Africa, important breeding colonies are known from Mauritania, Senegal and Guinea-Bissau. The breeding habitat is highly variable and includes bare or

sparsely vegetated places such as islands, banks, dunes, saltmarshes and salt pans, along the coast and in freshwater lagoons and inland lakes. Migrating birds are often seen over salt pans, coastal lagoons and various other coastal wetland types, but it also forages over large rivers, lakes and rice fields. It is largely insectivorous but quite opportunistic, taking a wide variety of food items including reptiles, amphibians and fish. In coastal W Africa crabs are taken frequently.



Dani Lopez Velasco / Agami



Vincent Legrand / Agami

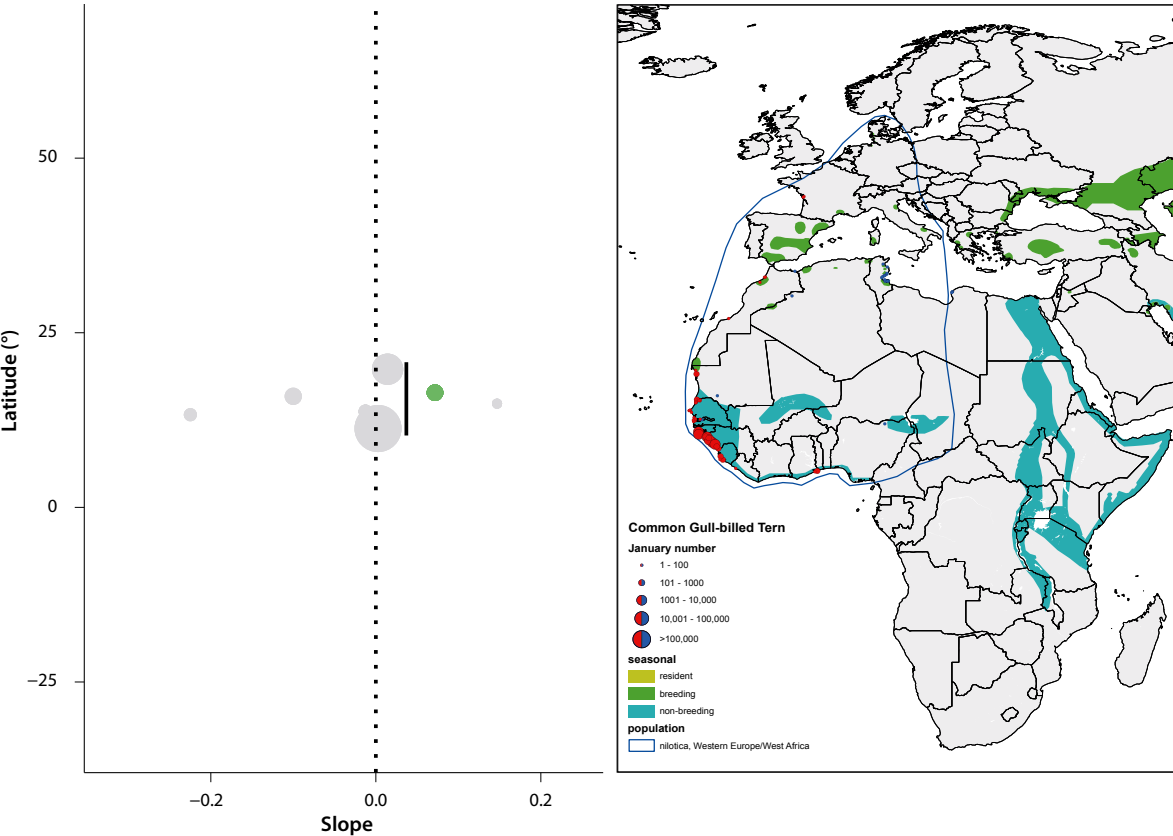


Figure A1.131. Plot (left) of January site trends and distribution (right) of Common Gull-billed Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Gull-billed Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Europe/W Africa	b	1980-2018	1,037	moderate increase	2009-2018	1,056	moderate increase	1972- 2018	28000	38000

Table A1.61. Summary of flyway trend and population size estimates for Common Gull-billed Tern. For explanation see table A1.2.

Caspian Tern | *Hydroprogne caspia* | Sterne caspienne

The Caspian Tern is a cosmopolitan species of which several regional subpopulations occur in the EAF. The Baltic breeding population mainly winters in the W Sahel (Mali) but also in S Spain, Chad and the Nile basin (Rueda-Urbe *et al.* 2021). A Southern African breeding population occurs both inland and at coastal sites. The W African population along the coast ranges from Mauritania south to Guinea during breeding (see chapter 9) and is present in W Africa as well during non-breeding. There are also small breeding colonies in Equatorial Guinea and Gabon. Habitat requirements are quite similar year-round: it prefers sheltered coastal waters and estuaries including salt pans, lagoons, inlets, bays, harbours, freshwater lakes and saline inland wetlands. It often nests on shell and shingle beaches and islands. Roosting occurs on sandbars or shell banks. The diet consists mainly of fish.



Arnold Meijer / Blue Robin

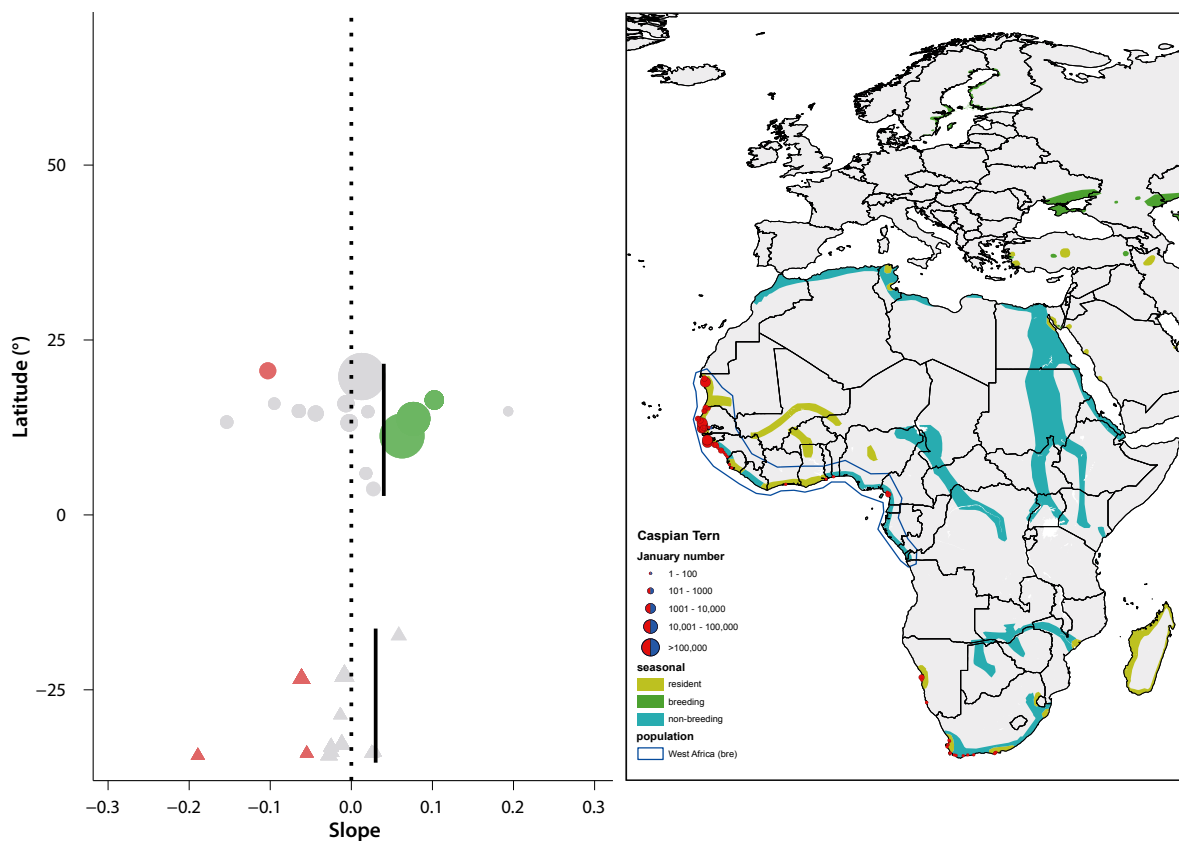


Figure A1.132. Plot (left) of January site trends and distribution (right) of Caspian Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Caspian Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
coastal W Africa	w	1997-2023	1,045	moderate increase	2014-2023	1,056	moderate increase	2019	21000	22000

Table A1.62. Summary of flyway trend and population size estimates for Caspian Tern. For explanation see table A1.2.

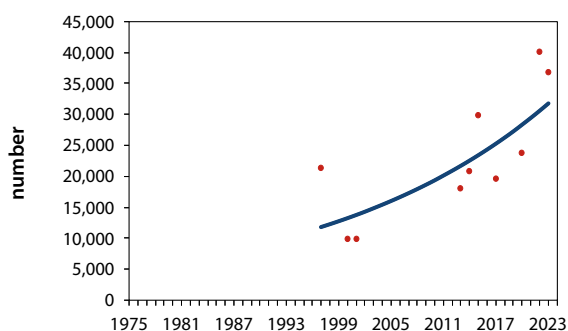


Figure A1.133. Flyway population trend for Caspian Tern population W Africa (bre). For explanation see fig. A1.4.



Harvey van Diek

Roseate Tern | *Sterna dougallii* | Sterne de Dougall

The Roseate Tern is a globally widespread species of mostly tropical and subtropical regions. The most northerly population breeds in W Europe and is considered in this report. Its breeding distribution is scattered on off-shore islands in the Atlantic region, with the largest numbers on the Azores and in Ireland and the UK, and smaller numbers in mainland Portugal, France, Madeira and the Canary Islands. W European birds winter in the Gulf of Guinea, mainly in cold-water upwelling systems off the coasts of Ghana and Sierra Leone-Liberia (Redfern *et al.* 2020). Breeding occurs in colonies on islands and islets with rocky coasts, but also on shingle and sandy beaches, often mixed with other tern species such as the Common Tern. It remains gregarious all year round, roosting in flocks and also congregating with other terns and gulls. Outside the breeding season, the species is largely pelagic but also occurs inshore. The diet is rather specialised compared to other terns and consists of small pelagic fish such as sandeel and sprat.

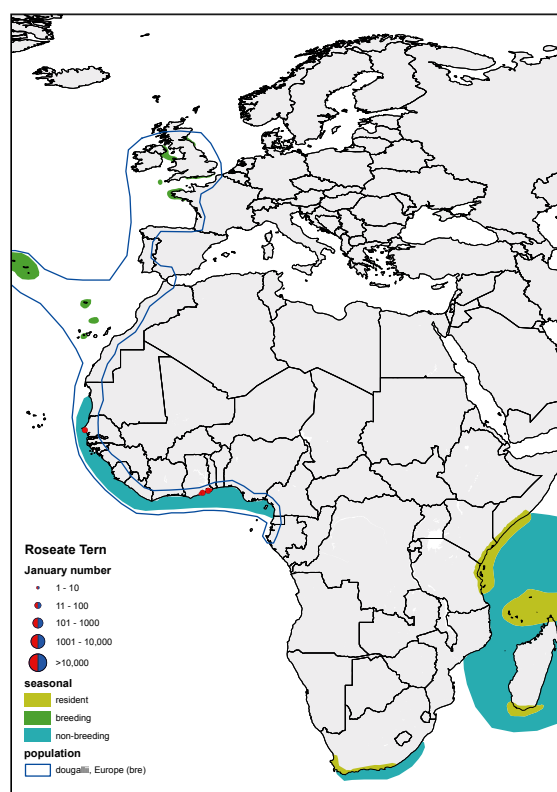


Figure A1.134. Distribution of Roseate Tern in the East Atlantic Flyway in January. Not enough data exists at present for site trends. For explanation see fig. A1.3.

Harvey van Diek



population Roseate Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Europe/East Atlantic	b	1980-2018	1,033	moderate increase	2009-2018	1,049	moderate increase	2012- 2018	7500	9200

Table A1.63. Summary of flyway trend and population size estimates for Roseate Tern. For explanation see table A1.2.

Common Tern | *Sterna hirundo* | *Sterna pierregarin*

The Common Tern is one of the most globally numerous and widespread tern species. Three populations occur in the EAF. The two largest, one breeding in W and S Europe and also including breeding birds from NW-Africa and the other breeding in N and E Europe, winter in Africa south to S Africa, mainly off W Africa (Becker & Ludwigs 2004, Becker *et al.* 2016). Here they meet birds from a smaller relatively unknown W African population breeding scattered along the coast between SW Morocco and Gabon. Migration shows a leapfrog pattern with the northernmost breeders wintering furthest south (Becker & Ludwigs 2004). Breeding in Europe is quite scattered, occurring both in coastal and inland situations. Along the coast it prefers rocky surfaces on inshore islands, shingle and sand beaches, dunes and islands in estuaries, lagoons and salt-marshes. Inland it occurs on sand or shingle lake shores and gravel banks on river or lake islands, sand and gravel pits. Its diet is mainly fish and small crustaceans. See chapter 10 for an indepth study on Common Tern breeding in the Wadden Sea region and wintering along the EAF.



Harvey van Diek

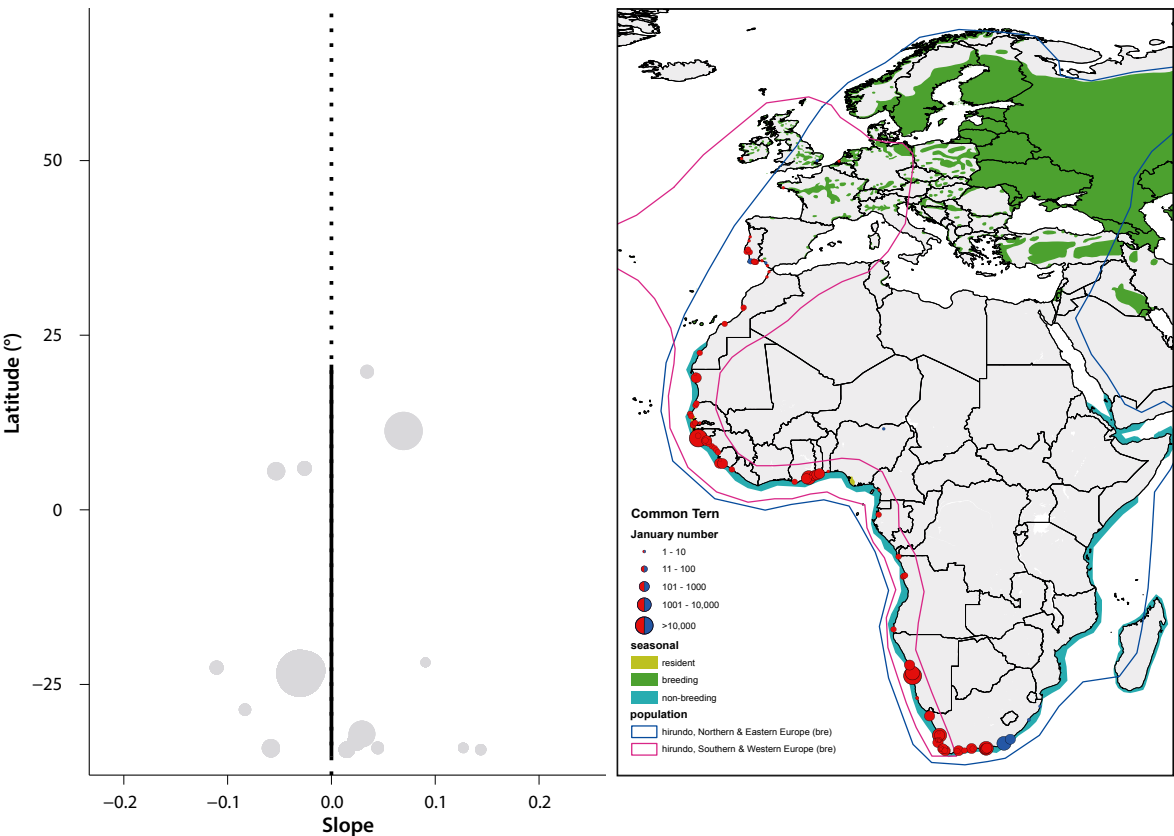


Figure A1.135. Plot (left) of January site trends and distribution (right) of Common Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.



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population Common Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
N & E Europe /East Atlantic	b	1980-2018	1,000	stable	2009-2018	0,990	stable	1991-2019	1100000	1800000
S & W Europe/East Atlantic	b	1980-2018	1,008	moderate increase	2009-2018	0,996	stable	2006-2018	170000	220000

Table A1.64. Summary of flyway trend and population size estimates for Common Tern. For explanation see table A1.2.

Lesser Crested Tern | *Thalasseus bengalensis* | Sterne voyageuse

The Lesser Crested Tern has a large distribution in Middle East, Eastern Africa and Australasia. However there is a rather isolated breeding population in Lybia in the Mediterranean which winters along the coasts of NW Africa and West Africa along the East Atlantic Flyway (subspecies *emigrata*). During wintering they occur in small numbers from Banc d'arguin to coastal Sierra Leone within groups of West African Crested Terns. Probably many are missed because they are overlooked among other terns and also occur out at sea. Their diet is mainly fish and small crustaceans.



Aurélien-Audvard / Agami

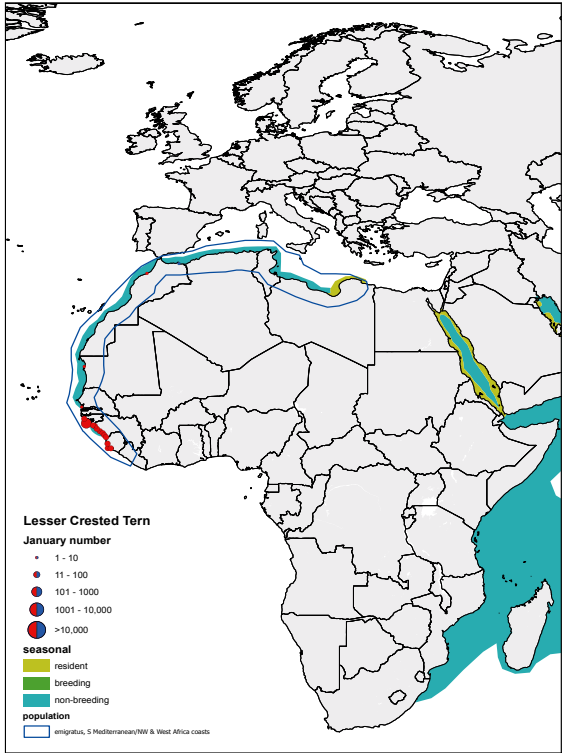


Figure A1.136. Distribution of Lesser Crested Tern in the East Atlantic Flyway in January. Not enough data exists at present for site trends. For explanation see fig. A1.3.

population Lesser Crested Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
emigratus, S Mediterranean/NW & W Africa		1993-2023	1,049	uncertain	2014-2023	1,070	uncertain	2000	6000	7000

Table A1.65. Summary of flyway trend and population size estimates for Lesser Crested Tern. For explanation see table A1.2.

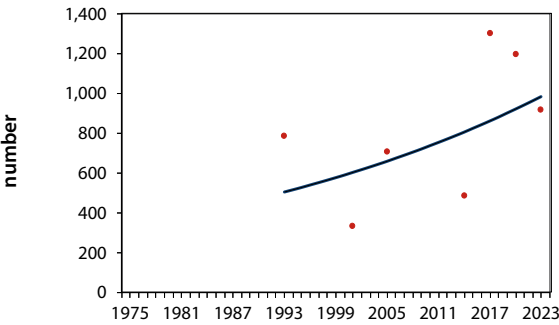


Figure A1.137. Flyway population trend for Lesser Crested Tern population S Med. / NW & W Africa. For explanation see fig. A1.4.



Arie Ouwerkerk / Agami

Sandwich Tern | *Thalasseus sandvicensis* | Sterne caugek

The Sandwich Tern is a coastal species occurring in many parts of Europe and Africa as well as the Americas. The EAF is used by the W European breeding population, which winters along the western seaboard of Africa and in the W Mediterranean. Large breeding colonies (>5,000 pairs) in the EAF occur in The Netherlands, the UK, Germany, Denmark and France. The species is gregarious throughout the year. It is likely that part of the population that winters in the Mediterranean Sea mixes with individuals from eastern European origin (particularly Ukraine). Ring recov-

eries of birds from The Netherlands show a strict coastal non-breeding distribution, with birds found along the entire European and African Atlantic coast as far south as South Africa. Colonies occur on sandy islands, sand dunes and rocky islets near suitable foraging grounds. Outside the breeding season the species is found on the open sea, but also frequents sandy or rocky beaches. The diet consists of fish of up to 15 cm in length. The population of Sandwich Tern was severely impacted by outbreaks of avian influenza in 2022 and 2023 (see chapter 4).

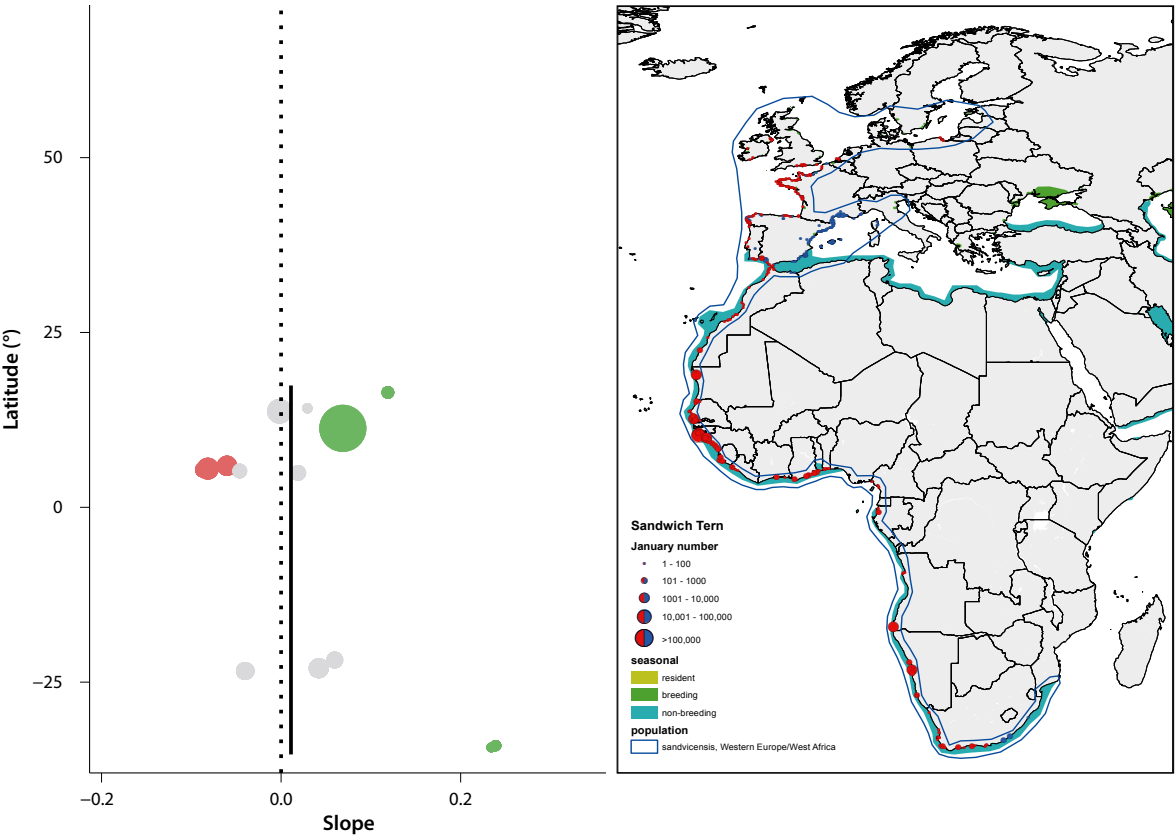


Figure A1.138. Plot (left) of January site trends and distribution (right) of Sandwich Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Sandwich Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Europe /W Africa	w	1980-2018	1,011	moderate increase	2009-2018	1,003	stable	2006- 2018	170000	200000

Table A1.66. Summary of flyway trend and population size estimates for Sandwich Tern. For explanation see table A1.2.



Arnold Meijer / Blue Robin

West African Crested Tern | *Thalasseus albididorsalis* | d Afrique

‘Royal Terns’ occurring in the EAF have been considered a subspecies (*T. maxima albididorsalis*) of a species also occurring in the Americas. However recently it was shown that they are more closely related to Lesser Crested Tern *T. bengalensis* than to American *T. m. maxima*, and hence should be considered a separate species, West African Crested Tern *T. albididorsalis* (Collinson *et al.* 2017). This form breeds on the W African coast from Mauritania to

Guinea (see chapter 9) and winters from Morocco to Angola. It is gregarious year-round. It shows a preference for inaccessible breeding sites such as sandy or coral islands, lacking vegetation and offering a good vantage point. Foraging occurs in coastal waters including estuaries, lagoons and mangroves. The diet consists mainly of small fish, but also squid, shrimps and crabs.

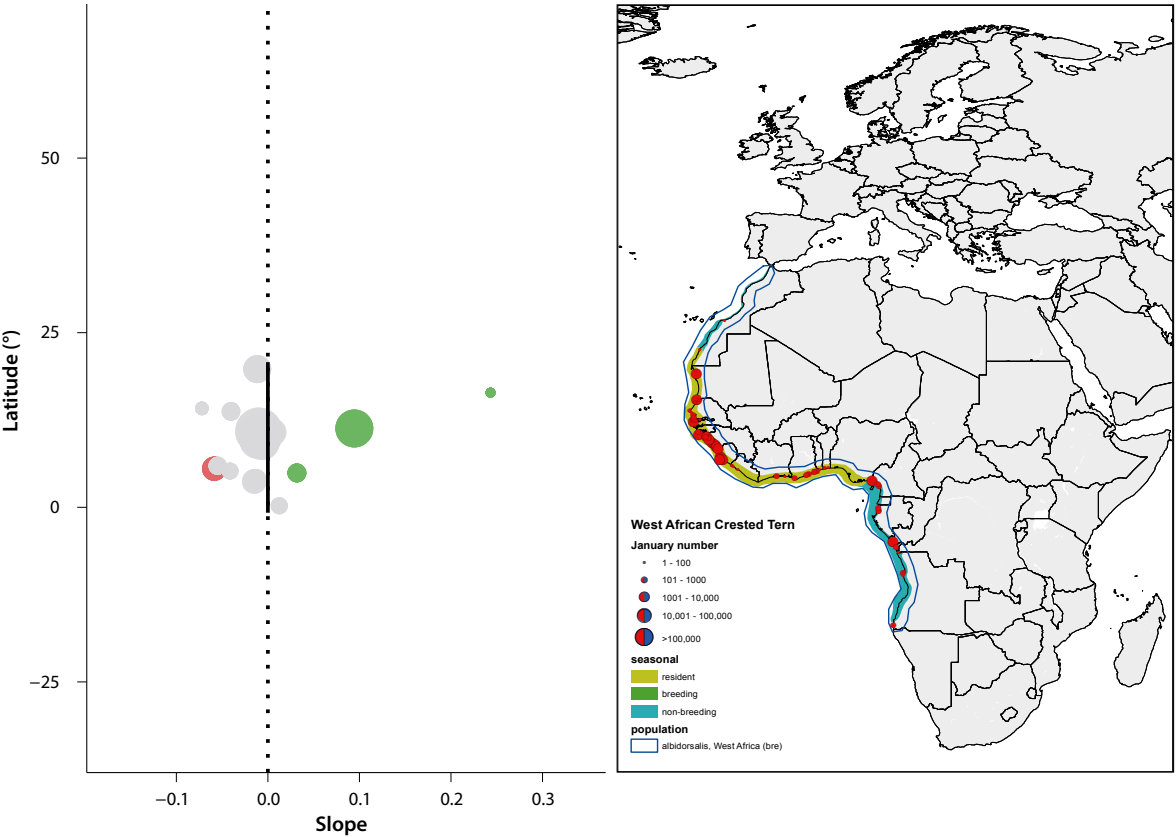


Figure A1.139. Plot (left) of January site trends and distribution (right) of West African Crested Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population West African Crested Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
W Africa	w	1990-2023	1,005	moderate increase	2014-2023	1,031	moderate increase	2019	220000	230000

Table A1.67. Summary of flyway trend and population size estimates for West African Crested Tern. For explanation see table A1.2.

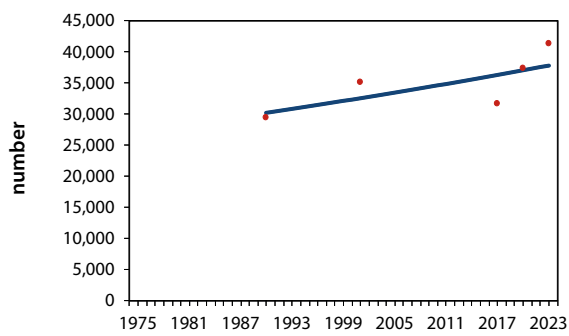


Figure A1.140. Flyway population trend for West African Crested Tern. For explanation see fig. A1.4.



Dani Lopez Velasco / Agami



Pete Morris / Agami

Greater Crested Tern | *Thalasseus bergii* | Sterne huppée

Greater Crested Terns of the nominate subspecies breed on the coast from Namibia to South Africa and use a wider coastal range outside the breeding season. The species is highly gregarious in the breeding season and roosts in flocks during the non-breeding season, although individuals usually forage alone or in small groups. Nesting occurs

mostly on offshore islands, on bare sand, rock or coral. The species forages mainly in shallow coastal waters including estuaries, lagoons and mangroves, but may also venture far out to open sea. The diet consists predominantly of pelagic fish of 10-50 cm length, but also includes squid, shrimps and crabs.

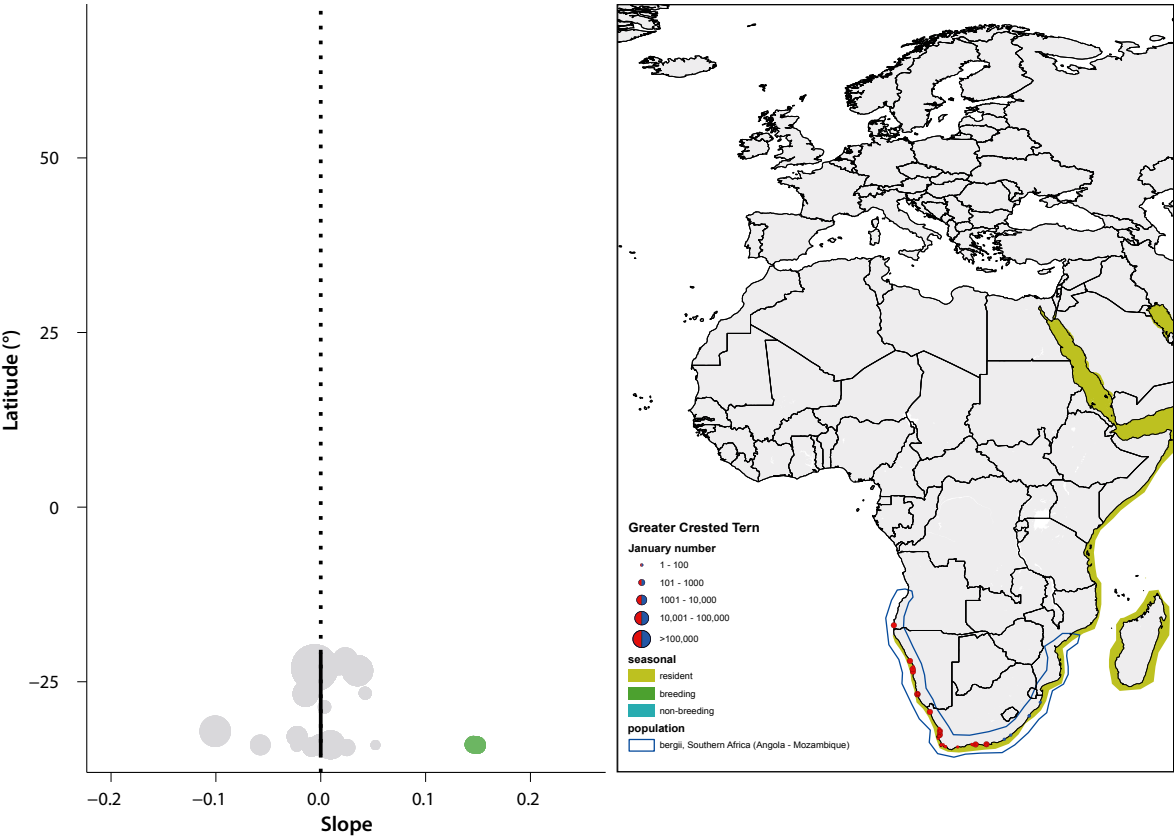


Figure A1.141. Plot (left) of January site trends and distribution (right) of Greater Crested Tern in the East Atlantic Flyway in January. For explanation see fig. A1.3.

population Greater Crested Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsze	popsze-min	popsze-max
S Africa	w	1996-2023	0,998	stable	2014-2023	0,910	moderate decrease	1994- 1996	15000	25000

Table A1.68. Summary of flyway trend and population size estimates for Greater Crested Tern. For explanation see table A1.2.

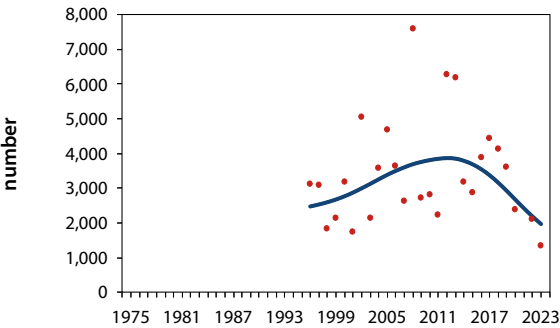


Figure A1.142. Flyway population trend for Greater Crested Tern population Southern Africa. For explanation see fig. A1.4.



Marc Guyt / Agami



Little Tern | Sterne naine (*Sternula albifrons*) (Harvey van Diek)