East Atlantic Flyway assessment 2020

The status of coastal waterbird populations and their sites

THE WADDEN SEA FLYWAY INITIATIVE

Wetlands INTERNATIONAL

BirdLife INTERNATIONAL

East Atlantic Flyway Assessment 2020

The status of coastal waterbird populations and their sites

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Foreword

This report is the culmination of the work of many individuals and institutions who have dedicated much of their time, resources, and energy to monitoring migratory waterbirds along the East Atlantic Flyway – an important migration route used by millions of waterbirds covered by the African-Eurasian Waterbird Agreement (AEWA), a legally-binding UN Treaty for migratory waterbirds signed by 81 countries and the European Union.

As one of the major waterbird flyways in the world found within the geographic area of AEWA, the East Atlantic Flyway (EAF) stretches from Northeast Canada to Northern Siberia in Russia and southward along the coasts of Europe and Africa all the way to South Africa.

Now in its third edition, the *East Atlantic Flyway Assessment* documents the flyway trends for more than 80 populations of 66 coastal waterbird species, based on the counts conducted at thousands of sites along the East Atlantic Flyway. In addition to presenting an assessment of the trends in bird abundance, the report also provides a unique assessment of the pressures occurring at different key sites along the Flyway, which include unsustainable fishing, disturbance by tourism, agricultural use, and encroachment by buildings amongst others.

The collective work which led to this unique report is in many ways an excellent example of international flyway cooperation. Not only does the report capture the waterbird monitoring work carried out by around 12,000 observers in 36 countries, 16 in Europe and 20 in Africa, it also captures the sophisticated work of a group of very experienced scientists and monitoring experts in analysing and compiling the international monitoring data.

This report is tangible proof for the notable boost in monitoring work conducted along the EAF, which can in many ways be attributed to the designation of the Wadden Sea as a UNESCO World Heritage Site in 2009. At the time, the Wadden Sea was internationally recognised as a World Heritage Site largely because of its status as one of the most important breeding, stopover, and wintering site for millions of migratory waterbirds on the EAF.

In this historic decision, the World Heritage Committee explicitly requested a strengthening of cooperation amongst the State Parties along the various African-Eurasian Flyways on management and research activities for conserving migratory birds. The WHC decision ultimately led to the creation of the Wadden Sea Flyway Initiative (WSFI), which in turn initiated the development of an integrated waterbird monitoring programme together with Wetlands International and BirdLife International, which has become the foundation for this report. Collaborative efforts to increase local capacity for monitoring and management along the flyway ensure that the results and recommendations from the assessment report are taken up to improve the conditions at key sites for migratory birds.

As Executive Secretary of the African-Eurasian Migratory Waterbird Agreement (AEWA), I would like to compliment all those who have contributed to this report and sincerely thank them for their continued support to international waterbird conservation. The knowledge generated through your efforts is helping to guide policy decisions and is also being used by AEWA to improve the conservation status of waterbirds through a range of concerted actions and international species action plans being carried out by AEWA Parties.

I would like to therefore encourage the governments of the Netherlands, Germany and Denmark - as the custodians of the Wadden Sea World Heritage site and as AEWA Parties - to continue to actively support the important monitoring and capacity building work being carried out under the Wadden Sea Flyway Initiative (WSFI) through the Common Wadden Sea Secretariat (CWSS) in Wilhelmshaven, Germany.

From the northern breeding areas, to the key Wadden Sea stopover and along the entire African coastline, let us continue to work together to improve the monitoring and conservation of migratory waterbirds along this important flyway.

Dr. Jacques Trouvilliez
Executive Secretary
African Eurasian Waterbird Agreement (AEWA)

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General

Counts of waterbirds in the flyway, during non-breeding and breeding, are carried out by a large number of birdwatchers, rangers of nature reserves, field staff of research institutes and conservation and management agencies, both governmental and non-governmental. Both for the counts of non-breeding waterbirds in January during the International Waterbird Census and for the coordination of breeding monitoring, each country is coordinated by a national coordinator, coming from national research institutes, non-governmental organisations or government agencies. For the January survey of the total count 2020 of the East Atlantic Flyway it was estimated that 13,000 observers were involved. We thank both observers, national and regional coordinators for their huge efforts. More detailed information about the involvement of people, institutions and results in Europe and Africa can be found further on in this acknowledgements under Sources of information per country and in the country specific chapters in van Roomen *et al.* 2020.

The flyway coordination and support of the total count 2020 was funded by the Dutch Ministry of Agriculture, Nature and Food Quality (to Sovon) 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, Vogelbescherming Nederland, Webs United Kingdom, Wildfowl & Wetlands Trust United Kingdom and Tour du Valat, France. Further support for this report was received from the Dutch Ministry of Agriculture, Nature and Food Quality and from the Life IP Deltanatuur project of the European Union (to the Ministry of Agriculture, Nature and Food Quality The Netherlands and Vogelbescherming Nederland). Also important in this respect is the funding for the coordination and database management of the International Waterbird Census, by the Member countries of Wetlands International and the Swiss Federal Office for the Environment and the funding

of the breeding bird monitoring project of the European Bird Census Council and the Birds in Europe Project of BirdLife International by the European Union.

Guidance of the work by Sovon was provided by Karst Jaarsma (Ministry of Agriculture, Nature and Food Quality) and by Kristine Meise (Common Wadden Sea Secretariat). Further guidance and support were provided by other members of the Steering group of the Wadden Sea Flyway Initiative, particularly Peter Südbeck (National Park Niedersachsen), Thomas Borchers (Federal Ministry for the Environment, Germany), Henrik G. Pind Jørgensen (Agency for Water and Nature Management, Denmark) and Courtney Price (AMBI-Caff).

Principal sources of data by country

Norway Data for January used as reported to the International Waterbird Census (IWC). National Coordinator (NC)is Svein-Hakon Lorentsen (Norwegian Institute for Nature Research NINA), see Lorentson 2020 for more details. Data about breeding birds is from BirdLife Norway and NINA.

Sweden Data for January used as reported to the IWC. NC is Frederik Haas (University of Lund), see Haas & Nilsson 2020 for more details. Breeding bird data is collected through coordination of University of Lund.

Finland Data for January used as reported to the IWC. NC is Aleksi Lehikoinen (Finnish Museum of Natural History), see Lehikoinen, Toivanen & Mikkola-Roos 2020 for more details. Breeding bird data is also collected through coordination of Finnish Museum of Natural History.

Estonia Data for January used as reported to the IWC. NC is Leho Luigujoe (Estonian Ornithological Society & Estonian University of Life Sciences), see Luigujoe 2020 for more details.

Latvia Data for January used as reported to the IWC. NC is Antra Stipniece (Latvian Ornithological Society), see Stipniece 2020 for more details. Breeding bird data is also collected through coordination of the Latvian Ornithological Society.

Lithuania Data for January used as reported to the IWC. NC is Laimonas Sniauksta (Lithuanian Ornithological Society), see Sniauksta 2020 for more details. Breeding bird data is also collected through coordination of the Lithuanian Ornithological Society.

Poland Data for January used as reported to the IWC. NC is Wlodzimierz Meissner (University of Gdansk). Data about breeding birds is from OTOP/BirdLife Poland.

Ireland Data for January used as reported to the IWC. NC is Lesley Lewis (BirdWatch Ireland), on behalf of the Irish Wetland Bird Survey, which is a joint project of the National Parks & Wildlife Service of the Department of Culture, Heritage & the Gaeltacht and BirdWatch Ireland. Breeding bird data is also collected through coordination of BirdWatch Ireland and National Parks and Wildlife Service.

United Kingdom Data for January used 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, Wildlife and Wetlands Trust, Royal Society for the Protection of Birds and Joint Nature Conservation Committee, see Frost 2020 for more details. In addition, data on wintering Barnacle Goose and Greylag Goose are from the Goose and Swan Monitoring Programme as coordinated by the Wildfowl & Wetlands Trust. Data about breeding birds is collected through BTO. Environmental data for important coastal sites were coordinated through JNCC in cooperation with Natural England and Nature Scotland.

Denmark Data for January used as reported to the IWC. NC is Preben Clausen (University of Aarhus), see Clausen, Nielsen, Bregnballe, Sterup & Krag Petersen 2020 for more details. Data about breeding birds is collected through DOF/BirdLife Denmark.

Germany Data for January used as reported to the IWC. NC in Germany is Johannes Wahl (Dachverband Deutscher Avifaunisten (DDA)), see further details in Wahl, Günther, Ludwig & Prior 2020. Breeding bird data is also collected through coordination of DDA.

The Netherlands Data for January used as reported to the IWC. NC is Menno Hornman (Sovon Dutch Centre for Field Ornithology), see further details in Hornman 2020. Breeding bird data is collected through coordination of Sovon. Both waterbird and breeding bird monitoring are funded through the government project "Network Ecological Monitoring".

Belgium Data for January used 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), see further details in Devos 2020. Breeding bird data is collected through coordination of Aves Natagora.

France Data for January used as reported to the IWC. NC is Caroline Moussy (Ligue de Protection des Oiseaux (LPO)), see Moussy & Quaintenne 2020 for more details. Data about breeding birds is collected through coordination of Centre d'Écologie et des Sciences de la Conservation from the Museum national d'Histoire naturelle.

Spain Data for January used as reported to the IWC. NC is Blas Molina (Sociedad Española de Ornitología/BirdLife Spain). Breeding bird data is also collected through coordination of SEO.

Portugal Data for January used as reported to the IWC. NC at that time was Vitor Encarnação (Instituto da Conservação da Natureza e das Florestas), now taken over by Filipe Moniz, see further details in Encarnação 2020. Breeding bird data is collected through coordination of Sociedade Portuguesa para o Estudo das Aves (SPEA).

Morocco Data for January used as reported to the IWC. NC at that time was Mohamed Dakki (Scientific Institute, Mohamed V University & Grepom), now taken over by Asmaâ Ouassou, see further details in Dakki, Ouassou, El Hamoumi, El Agbani & Qninba 2020.

Mauritania Data for January used as reported to the IWC. Counts were organized at the Parc National du Banc d'Arguin (PNBA) by Amadou Kidé (PNBA), see details in Kidé 2020. In the Mauritanian part of the trans-boundary Biosphere Reserve of the Senegal River, the counts were provided by Zeine El Abidine Sidaty (Parc National Diawling), see Daf, El Abidine Sidaty, El Abass & Aveloitt 2020 for details. Counts in Lac Alec were provided by Nature Mauritanie.

Senegal Data for January used as reported to the IWC. NC of the IWC is Samuel Dieme, data management by Aminita Sall Diop (Direction des Parcs Nationaux), more details in Gueye, Diop & Niass 2020.

The Gambia Data for January used as reported to the IWC. NC of the IWC is Abdoulie Sawo (Department of Parks and Wildlife Management). For more details see Sawo, Jammeh & Jammeh 2020.

Guinea-Bissau Data for January used as reported to the IWC. NC of the IWC is Jãozinho Sá (Gabinete de Planificacao Costeira . More details of the count can be found in Sá, Monteiro & Regalla de Barros 2020.

Guinea Data for January used as reported to the IWC. NC is Namory Keita (Division Faune et Protection de la Nature). For details of the count in 2020 see Conde 2020.

Sierra Leone Data for January used as reported to the IWC. NC is Papanie Bai-Sesay (Conservation Society of Sierra Leone). For details of the count in 2020 see Bai-Sesay, Haffner & Showers 2020.

Liberia Data for January used as reported to the IWC. NC is Jerry Garteh (Society for the Conservation of Nature in Liberia). Details of the count in January 2020 are in Garteh 2020.

Côte d'Ivoire Data for January used as reported to the IWC. NC is Firmin Kouassi Kouame (Ministry of Water and Forests, direction de la Faune et des Ressources Cynegetiques), see details in Kouame 2020.

Ghana Data for January used as reported to the IWC. NC is Dickson Agyeman (Wildlife Division of the Forest Commission) in cooperation with Jones Quartey and Yaa Ntiamoa-Baidu (Centre for African Wetlands, University of Accra). For details of the count in 2020 see Quartey, Agyeman & Ntiamoa-Baidu 2020.

Benin Data for January used as reported to the IWC. NC is Rémi Hefounme (Direction Générale des Eaux, Forets et Chasse) in cooperation with NGO BEES, see details in Aliou, Adikpeto, Chaffra & Sossoukpe 2020.

Nigeria Data for January used as reported to the IWC. NC is Joseph Onoja (Nigerian Conservation Foundation). Details in Onoja 2020.

Cameroon Data for January used as reported to the IWC. NC is Gordon Ajonina (Cameroon Wildlife Conservation Society). The counts of 2020 are reported in Ajonina, Timba & Francis 2020.

Sao Tomé & Principe In the framework of the January count of January 2020 data about waterbirds were collected by the Gulf of Guinea Biodiversity Centre, details in Faustino de Lima & Martins 2020.

Gabon Data for January used as reported to the IWC. NC at that time was Alphonsine Koumba Mfoubou, now taken over by Gabin NZAMBA (Ministère des Eaux et Fôrets). For details of the count in 2020 see Koumba Mfoubou 2020.

Congo (Brazzaville) Data for January used as reported to the IWC. NC is Jérôme Mokoko Ikonga (Wildlife Conservation Society of Congo). For details see Mokoko Ikonga 2020.

Democratic Republic of Congo The count on the coast in January 2020 was coordinated by Pierre Mavuemba Tuvi (Institut Supérieur de Navigation et de Pèche). Details in Tuvi, Yalusila & Sambiandi 2020.

Angola Data for January used as reported to the IWC. NC is Marta Zumbo (Instituto Nacional da Biodiversidade e Conservação (INBAC)) in cooperation with the NGO Bioconserv (Filipe Kodo and others), details of the 2020 count are in Kodo, Dala, Xavier & Eugenia Lopes 2020.

Namibia Data for January used as reported to the IWC. NC is Holger Kolberg (Ministry of Environment, Forestry and Tourism). Details of the 2020 count are in Kolberg 2020.

South Africa Data for January used as reported to the IWC. NCis Michael Brooks (Fitzpatrick Institute of African Ornithology, University of Cape Town).

Menno Hornman

Bird counting in Buinea Bissau

Bergeend | **Tadorne de Belon** *(Tadorna tadorna),* Nijdam Wommels, Netherlands (Harvey van Diek)

Summary

The East Atlantic Flyway stretches from Northeast Canada to North Siberia in Russia, southward along the coastlines of the Baltic Sea, North Sea and Eastern Atlantic Ocean all the way to South Africa. Many waterbird populations use this flyway during their breeding and non-breeding seasons. The monitoring along this flyway received a new impulse when the Wadden Sea was granted World Heritage Status. The Wadden Sea Flyway Initiative started an integrated monitoring programme together with Wetlands International and BirdLife International. Within this programme censuses of selections of sites have been conducted annually in Western Africa from 2013 onwards and total January counts along the entire flyway have taken place with an interval of three years, in 2014, 2017 and 2020. Increasingly also monitoring data from breeding bird programmes are used. In addition to the assessment of bird abundance, an assessment of environmental circumstances occurring in sites along the flyway was included. Thanks to this programme we can document flyway trends for more than 80 waterbird populations of 66 species, based on bird counts at thousands of sites. Environmental data, including the presence of pressures and also the extent of conservation measures taken, have been collated from 115 important coastal sites in both Africa and Europe. Besides results from the ongoing general monitoring, this report brings together specific accounts about regions and sites (Russian Arctic, Wadden Sea, North Africa, Banc d' Arguin, Bijagos archipelago, urban wetlands in Dakar and Lagos and coastal wetlands in Angola) and thematic chapters on the importance of monitoring vital rates of waterbirds and the differences in flyway trends based on non-breeding January data or breeding bird data.

Generally, the status of flyway populations using the coastal EAF appears relatively favourable, but with notable exceptions. In the long term, almost twice as many populations show an increasing or stable trend than a declining one. However in particular, arctic-breeding waders migrating over long distances show on average more negative trends than other taxonomic and functional groups. At the sites within the EAF used by waterbirds many anthropogenic pressures occur. The extent to which these pressures directly influence the conservation status of individual populations along the flyway, cannot be assessed from the current data, but fishing, agriculture, disturbance from humans, waste pollution and urbanisation have all large influences. Also, the flyway is already under significant impact from climatic change, and this is bound to intensify in the coming decades. 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 pressures and conservation measures at the sites they use on a regular basis. To maintain a strong level of cooperation and information it will be important to continue and expand the current level of activities and coordination. Despite increasing quality of the data, we are still far from a situation where these data are collected routinely and in ongoing good quality along the flyway. Continued and increased cooperation, innovation and capacity building remains very important. Besides this, it is recommended to Improve the collection and utility of breeding bird data for flyway monitoring, to extend and update research into migratory connectivity and flyway boundaries, To invest in remote sensing of natural conditions and pressures, to reinforce and expand the monitoring of conditions in the Arctic and to Invest in research into causation of observed trends and relevant management responses.

Résumé

La voie de migration de l'Atlantique Est (EAF) s'étend du nord-est du Canada au nord de la Sibérie en Russie, puis vers le sud le long des côtes de la mer Baltique, de la mer du Nord et de l'Est de l'océan Atlantique jusqu'à l'Afrique du Sud. De nombreuses populations d'oiseaux d'eau utilisent cette voie de migration pendant leurs saisons de reproduction et de non-reproduction. Le suivi de cette voie de migration a reçu un nouvel élan lorsque la mer des Wadden a obtenu le statut de patrimoine mondial. L'initiative relative à la voie de migration de la mer des Wadden (WSFI) a lancé un programme de suivi intégré en collaboration avec Wetlands International et BirdLife International. Dans le cadre de ce programme, des recensements de sites sélectionnés ont été réalisés annuellement en Afrique de l'Ouest depuis 2013, et des comptages complets ont été effectués en janvier sur l'ensemble de la voie de migration à des intervalles de trois ans, en 2014, 2017 et 2020. De plus en plus, les données de suivi des programmes de reproduction des oiseaux sont également utilisées. En plus de l'évaluation sur l'abondance des oiseaux, une évalua-

Erik van Winden

Bird counting in Guinea

tion sur les circonstances environnementales survenant dans les sites le long de la voie de migration a été ajoutée. Grâce à ce programme, nous pouvons documenter les tendances de la voie de migration pour plus de 80 populations d'oiseaux d'eau de 66 espèces, à partir de comptages d'oiseaux sur des milliers de sites. Les données environnementales, y compris la présence de menaces et de l'ensemble des mesures de conservation prises, ont été rassemblées sur 115 sites côtiers importants en Afrique et en Europe. En plus des résultats du suivi général en cours, ce rapport rassemble des comptes rendus spécifiques sur des régions et des sites (Arctique Russe, mer des Wadden, Afrique du Nord, Banc d'Arguin, archipel des Bijagos, zones humides urbaines à Dakar et Lagos et zones humides côtières en Angola) et des chapitres thématiques sur l'importance du suivi des taux de survie des oiseaux d'eau et les différences dans les tendances des voies de migration basées sur les données des oiseaux non reproducteurs en janvier ou sur celles des oiseaux reproducteurs.

En général, l'état des populations utilisant la voie de migration côtière Atlantique (EAF) semble relativement favorable, mais avec des exceptions notables. À long terme, près de deux fois plus de populations présentent une tendance à la hausse ou stable qu'une tendance au déclin. Cependant, les échassiers se reproduisant dans l'Arctique et migrant sur de longues distances présentent en moyenne des tendances plus négatives que les autres groupes taxonomiques et fonctionnels. De nombreuses pressions anthropiques s'exercent sur les sites de l'EAF utilisés par les oiseaux d'eau. Les données actuelles ne permettent pas d'évaluer dans quelle mesure ces pressions influencent directement l'état de conservation des populations individuelles le long de la voie de migration, mais la pêche, l'agriculture, les perturbations causées par l'homme, la pollution par les déchets et l'urbanisation ont tous une grande influence. En outre, l'itinéraire de migration subit déjà l'impact significatif du changement climatique, qui devrait s'intensifier dans les décennies à venir. Grâce à l'effort de suivi actuel de l'EAF, nous sommes en mesure de mettre régulièrement à jour les tendances de la voie de migration et la distribution des populations d'oiseaux d'eau, et de contribuer au développement des estimations de population, ainsi que de signaler les pressions et les mesures de conservation sur les sites qu'ils utilisent régulièrement. Pour maintenir un niveau élevé de coopération et d'information, il sera important de poursuivre et d'étendre le niveau actuel des activités et de la coordination. Malgré l'amélioration de la qualité des données, nous sommes encore loin d'une situation où ces données sont collectées de manière routinière et de bonne qualité tout au long de la voie de migration. La poursuite et le renforcement de la coopération, de l'innovation et du renforcement des capacités restent très importants. En outre, il est recommandé d'améliorer la collecte et l'utilité des données sur les oiseaux nicheurs pour la surveillance de la voie de migration, d'étendre et d'actualiser la recherche sur la connectivité migratoire et les limites de la voie de migration, d'investir dans la télédétection des conditions et pressions naturelles, de renforcer et d'étendre la surveillance des conditions dans l'Arctique et d'investir dans la recherche sur les causes des tendances observées et les réponses de gestion pertinentes.

1. Introduction

The East Atlantic Flyway (fig. 1.1) is one of the major flyways for waterbirds connecting breeding areas with stopover sites and non-breeding grounds during their annual cycle. It stretches from the Arctic (Northwestern Canada to Central Siberia) through Western Europe (mainly Atlantic and North Sea areas) to the entire western coastline of Africa. 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. The flyway is also used by a substantial human population, with numerous cities, industries and activities distributed throughout the coastal zone. The region provides important ecosystem services in the form of food, prevention of flooding, renewable energy and leisure opportunities. In some areas, people and wildlife, including migratory birds, co-exist in reasonable harmony, but in other areas human activities exert a strong pressure on wildlife and their habitats. For proper co-existence between humans and biodiversity, of which birds are important indicators, conservation and management measures need to be applied. This requires careful decision making and adaptive management. To ensure their sustainability, these processes need to be based on, and informed by, knowledge about the state and trends of the bird populations themselves and the environments they use. This information will help to signal problems, define priorities and evaluate what measures need to be taken. For migrating populations it is crucial to use not only information from individual sites, but to take a flyway perspective, as individual birds move across a chain of habitats and sites and across different countries, and the combination of conditions at all these sites contributes to influence their a conservation status.

The Wadden Sea is a major coastal wetland forming an important breeding, stop-over and wintering site for waterbird populations along the East Atlantic Flyway. With the designation of the Wadden Sea as a World Heritage site in 2009, the World Heritage Committee requested a strengthening of cooperation with state parties along the flyway on management and research activities for conserving migratory species. As a result the Wadden Sea Flyway Initiative was established and among many other activities, a proposal for integrated monitoring along the East Atlantic Flyway was formulated (van Roomen *et al.* 2013). The aim was to monitor waterbird abundance (population sizes, trends and distribution) and to monitor environmental conditions (both pressures and conservation measures) at the sites that they use. Consideration was also given to 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 these ambitious aims started with improvements to abundance monitoring and environmental monitoring through a cooperation

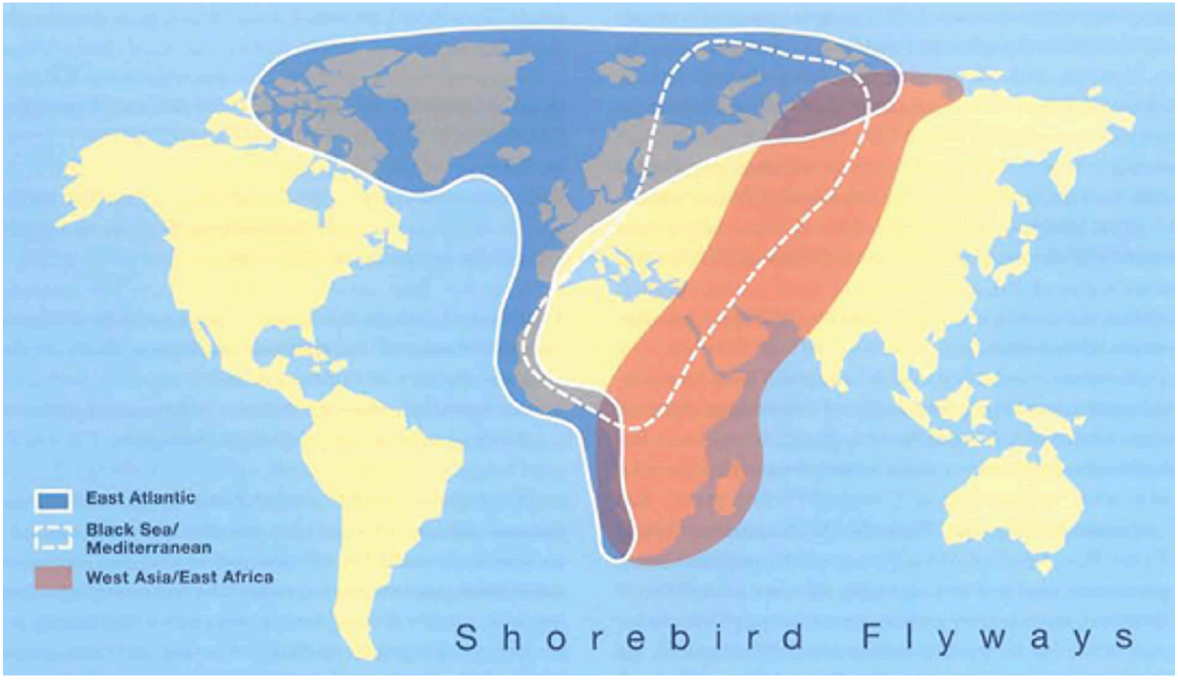

Figure 1.1. The three flyways in the African-Eurasian region as based on migratory shorebirds (Delany *et al.* 2009, after International Wader Study Group) with the East Atlantic Flyway in blue.

Kim Fischer

Embarking the boat again after bird count at Bijagos, Guinea Bissau

between the Wadden Sea Flyway Initiative, Wetlands International and BirdLife International, with national coordinators involved in each country. The monitoring aims to collect data annually in at least a selection of sites. In most European countries nearly all important sites are monitored on a yearly basis, but this is not the case along the Atlantic coast of Africa where resources are limited. Therefore a more comprehensive survey is organised every three years which aims to collect data from all relevant sites, which forms the basis of an updated flyway assessment. Besides the effort on a simultaneous flyway census, countries are encouraged to increase the coverage and frequency of monitoring visits to individual sites also between the triannual surveys.

This monitoring scheme started in 2013 with a 'total count' in 2014 and a first flyway assessment report being published in 2015 (van Roomen *et al.* 2015). After continuation of yearly data collection in 2015 and 2016, another 'total count' was organized in 2017 (van Roomen *et al.* 2018). After further data collection at selected sites in 2018 and 2019, the third 'total' count was organized in 2020. The present report provides the third flyway assessment. This report starts with two chapters giving an update on the current trends, distribution and population sizes of coastal waterbird species using the East Atlantic Flyway (chapter 2, Schekkerman *et al.* 2022) and the current pressures and conservation measures at sites used during the non-breeding season along the whole flyway (chapter 3, Crowe *et al.*2022).

A series of regional accounts provide more details and issues of important regions and sites. It starts with the Russian Arctic (chapter 4, Soloviev *et al.*2022), the Wadden Sea (chapter 5, Kleefstra *et al* 2022) and North Africa (chapter 6, Dakki *et al.*2022). These chapters are followed by accounts of the Banc d' Arguin, Mauritania (chapter 7, El-Hacen & Kidé 2022), the urban Technopôle wetland in Dakar, Senegal (chapter 8, Diallo & Manga 2022et al), the Bijagós Archipelago in Guinea-Bissau (chapter 9, Henriques et al 2022), , wetlands in Lagos, Nigeria (chapter 10, Nosazeogie 2021) and waterbirds along the Angola coastline and their key pressures (chapter 11, Xavier2022). These national and site accounts are not aimed at giving a complete picture, but they do provide a good understanding of local issues and patterns showing data and knowledge available but also gaps in information and conservation and management issues. These accounts are followed by two thematic chapters showing the value and importance of data about vital rates, using Sanderling *Calidris alba* as an example (chapter 12, Reneerkens 2022) and a chapter investigating similarities and differences between breeding bird trends and trends based on non-breeding January data for the same flyway populations (chapter 13, van Turnhout *et al.* 2022). In a final chapter, the results are discussed and conclusions and recommendations formulated (chapter 15). Annexes provide more details and basic information on the abundance monitoring of the bird species (Annex 1) and the environmental monitoring of sites (Annex 2). A third annex gives information about pilot results on remote sensing of natural conditions and environmental pressures at sites along the flyway (Annex 3).

2. Patterns in trends of waterbird populations using the coastal East Atlantic Flyway, update 2020

Hans Schekkerman, Szabolcs Nagy, Khady Gueye Fall, Tom Langendoen & Marc van Roomen

Summary

In this chapter, long-term (2-4 decades) and short-term (10 years) trends of 83 waterbird populations in the coastal East Atlantic Flyway (EAF) are summarised, based on the trend analyses detailed in Annex 1 of this report. In the long term, almost twice as many populations of the coastal EAF show an increasing or stable trend than a declining one. This overall status compares favourably with some other global waterbird flyways. However, the distribution of short-term trends is less favourable, and includes several strong declines. The overall mean of the short-term trends has become slightly negative and has declined since the previous assessment in 2017.

Exploration of patterns in the population development of groups of waterbirds sharing common characteristics confirms patterns identified in a previous assessment up to 2017. The most notable pattern is that waders (shorebirds) show considerably less favourable trends than other taxonomic waterbird groups, with particularly strong declines occurring in several species breeding in the Siberian Arctic. Flamingos, pelicans and cormorants – all large birds – show the most positive population development.

Flyway-scale changes in the distribution of non-breeding waders within the EAF were explored by comparing regional population estimates based on count data from the 1980s-1990s with those from the 2010s-2020. A strong common pattern emerged, with a predominance of regional increases in Europe (particularly in the SW and also for some species in northern sites), and declines along the African coasts southward from Mauritania and Senegal. Potential (partial) explanations for this pattern include migratory connectivity between specific wintering and breeding regions in the EAF that experienced different conditions, a buffer effect where populations that are in decline retreat into northern sites which are preferred for their shorter distance to the breeding grounds, and a distributional shift enabled by a warming winter climate in (NW) Europe. The latter process may have consequences for our ability to assess the status of different biogeographical populations within species on the basis of non-breeding counts. This highlights the importance of maintaining and strengthening monitoring along the entire EAF, and of extending and updating research into migratory connectivity and delimitation of flyway populations.

Resumé

Dans ce chapitre, les tendances à long terme (2-4 décennies) et à court terme (10 ans) de 83 populations d'oiseaux d'eau de la voie de migration côtière Est Atlantique (EAF) sont résumées, sur la base des analyses de tendances détaillées à l'annexe 1 de ce rapport. A long terme, près de deux fois plus de populations côtières de l'EAF présentent une tendance à la hausse ou stable qu'une tendance au déclin. Ce statut général se compare favorablement à celui de certaines autres voies de migration mondiales des oiseaux d'eau. Cependant, la distribution des tendances à court terme est moins favorable, et comprend plusieurs déclins importants. La moyenne générale des tendances à court terme est devenue légèrement négative et a diminué depuis la précédente évaluation en 2017.

L'exploration des schémas de développement des populations de groupes d'oiseaux d'eau partageant des caractéristiques communes confirme les schémas identifiés dans une précédente évaluation jusqu'en 2017. Le schéma le plus notable est que les oiseaux de rivages (limicoles) présentent des tendances nettement moins favorables que les autres groupes taxonomiques d'oiseaux d'eau, avec des déclins particulièrement forts chez plusieurs espèces se reproduisant dans l'Arctique sibérien. Les flamands, les pélicans et les cormorans - tous de grands oiseaux - présentent l'évolution la plus positive des populations.

Les changements à l'échelle de la voie de migration dans la distribution des échassiers non nicheurs au sein de l'EAF ont été explorés en comparant les estimations de la population régionale basées sur les données de comptage des années 1980-1990 avec celles des années 2010-2020. Un schéma commun très marqué est apparu, avec une prédominance des augmentations régionales en Europe (en particulier dans le Sud-Ouest et également pour certaines espèces dans les sites du Nord), et des déclins le long des côtes africaines vers le sud, depuis la Mauritanie et le Sénégal. Les explications potentielles (partielles) de ce schéma comprennent la connectivité migratoire entre des régions spécifiques d'hivernage et de reproduction dans l'EAF qui ont connu des conditions différentes, un effet tampon où les populations qui sont en déclin se retirent dans des sites nordiques qui sont préférés pour leur distance plus courte aux zones de reproduction, et un changement de distribution rendu possible par un

réchauffement du climat hivernal en Europe (NW). Ce dernier processus peut entraîner des conséquences sur notre capacité à évaluer le statut des différentes populations biogéographiques d'une même espèce sur la base de comptages en dehors de la période de reproduction. Cela souligne l'importance de maintenir et de renforcer le suivi le long de l'ensemble de l'EAF, et d'étendre et d'actualiser la recherche sur la connectivité migratoire et la délimitation des populations de la voie de migration.

2.1 Introduction

This chapter summarises general patterns in the trends of waterbird populations occurring along the coastal East Atlantic Flyway (EAF). The trends for each species are presented and discussed in Annex 1. Here we present a global summary of trends across populations, and compare these results with a previous summary in the 2017 report (Schekkerman *et al.* 2018). In that report, we also explored whether common patterns in increase and decrease existed across populations with similar ecological characteristics, which may point to factors affecting multiple bird populations in similar ways across the coastal EAF or in specific parts of it. Such patterns may help in identifying priority areas for conservation and provide a first clue to possible causes of observable changes. Here, we repeat this analysis to see whether the main patterns then identified are still visible with new monitoring data added.

New in this assessment is an exploration of changes in the non-breeding distribution across the coastal EAF in waders (shorebirds, Charadrii). Waders form a characteristic and numerically important clade of waterbirds in this flyway, but also show the least favourable population changes of all taxonomic groups inhabiting it. Based on a comparison of January counts, we explore whether any spatial patterns are apparent in the changes of non-breeding populations between the 1980s-1990s and the present. Such changes may come about by redistribution of individuals or by differential decrease or increase of sub-populations in different parts of the flyway.

2.2 Data and analysis

The raw data used in the analysis of trends consists of the long- and short-term trends in waterbird population abundance up to 2020 as presented in Annex 1. In total, trends for 83 populations of 66 species are included. These form a cross-section of all waterbirds occurring within the flyway with respect to taxonomy, breeding and non-breeding regions, diets and migration strategies. Although the set of populations included is not exactly the same as that in Schekkerman *et al.* (2018), the overlap is sufficiently large to compare broad-scale patterns.

Most of the trends used are based on counts undertaken as part of the International Waterbird Census (IWC), but for some populations they were based on data from breeding bird monitoring programmes (the Pan-European Common Bird Monitoring Scheme PECBMS, and Red List assessments in Europe) (see Annex 1). In this chapter, all trends are expressed as the average % change per year over the trend period (mean 37 years for long-term trends, 10 years for all short-term trends).

For an overall summary of waterbird trends in the coastal EAF, the number of populations falling in each of six formal trend classes were tabulated (strong increase, moderate increase, stable, uncertain, moderate decline, strong decline; see Fig. 2.1 for class definitions). Also, we calculated the mean of all population-specific annual rates of change as a summarising metric of long and short-term trends. Trait-based patterns in population trends were explored by calculating and comparing mean annual rates of change for groups of populations (species) that share similar ecological characteristics. For a complete account of traits used in this exploration, see Schekkerman *et al.* (2018).

Figure 2.1. Trend classifications of long- and short-term trends of 83 waterbird populations considered in this report. The boundary between moderate and strong increases or declines is a change of 5% per year. Populations are considered stable if the 95% confidence interval around the trend includes 0% change and does not include 5% change per year in either direction. If it does include 5% change, the trend is 'uncertain'.

For populations of waders (Charadrii), we explored changes in non-breeding distribution at the scale of the entire coastal EAF. To do this the flyway was divided in 12 regional sections consisting of one or several countries, from north to south: (1) Scandinavia with Denmark, the Faroes and Iceland, (2) Germany, (3) Netherlands and Belgium, (4) United Kingdom and Ireland, (5) France, (6) Spain and Portugal, (7) Morocco (with the W Mediterranean if the populations' flyway includes Algeria, Tunisia, Italy), (8) Mauritania, (9) Senegal to Sierra Leone, (10) Liberia to Nigeria, (11) Cameroon to Angola, (12) Namibia and South Africa.

More or less complete counts of non-breeding waders are made annually in western Europe at most estuarine sites, while complete coverage of rocky and sandy coastlines is generally less frequent . In most African countries a sample of sites is counted annually; more complete coverage has been achieved irregularly in the past and from 2014 onwards every third year. We tabulated published and unpublished estimates of non-breeding (January) wader numbers by country in four time periods: (1) around 1980 (1980s, Smit & Piersma 1989), 1995 (1990s, Stroud *et al.* 2006), 2014 (van Roomen *et al.* 2015 and 2020 (this report). Empty cells in this table were filled by either linear interpolation (if estimates were available for both adjoining periods) or by duplicating the estimate for a single adjoining period. Because this creates dependency between periods, particularly between 1980 and 1995 (as 1980 had the most empty cells), we simplified the comparison to two main periods '1980s - 1990s' versus '2010s -2020', using for each region the sum over countries of the mean cell entries for 1980 and 1995, and for 2014 and 2020 respectively. Changes in regional numbers between these main periods were calculated by dividing the 2010s-2020 mean by that for 1980s-1990s, so the resulting change factor denotes increase when greater than 1, and decrease when less than 1. To look for spatial patterns we graphed the change ratios by region aligned from north to south, for each species or population within a species separately and for all species combined.

2.3 Waterbird trends in the East Atlantic Flyway

2.3.1 General summary of trends

On the long-term timescale, the majority of all 83 waterbird populations considered fell into a favourable trend category, with 50% showing either a moderate or strong increase, whilst a further 16% were stable (Fig. 2.1). Declining populations made up 30% of the total, with none in the most unfavourable category of 'strong decline'. Trends were uncertain in just three populations (4%). The mean annual rate of long-term change across all populations was +0.8 %/year, with 95% confidence interval (C.I.) +0.2 to +1.5 %/year, i.e. significantly different from a stable situation (0 %/year).

On the short-term timescale, 26% of all populations showed an increase, 26% were stable, and 29% were in decline, of which 4% strongly in decline (Fig. 2.1). In addition, a larger share of short-term trends (19%) fell in the 'uncertain' category than among the long-term trends. This is to be expected as over a shorter time period fluctuations in bird numbers and random errors in the counts exert greater influence. Partly for the same reason, short-term trends also showed more variation between populations in the rate of change than long-term trends, although those that increased on the long term also tended to increase on the short term and *vice versa* (correlation r= 0.56, P<0.01). The overall mean of the short-term trends was -0.35 %/year (95% C.I. 0.7 to +1.0 %/year). This is not significantly different from stable, but the sign of the overall mean is now negative while it was still slightly positive (+0.4 %/yr) up to 2017. Also, short-term trends up to 2020 were significantly less favourable than the long-term trends of the same populations (paired t-test, t= 2.08, d.f.= 80, P=0.04). This indicates a change to less favourable recent population trajectories.

Nevertheless, compared to other global flyways of migratory waterbirds, population trends in the EAF seem to be relatively favourable in general, with a greater share of increasing populations and less (strong) declines (e.g. Wetlands International 2010). For instance, among flyways recognised within the area covered by the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA), waterbird populations in the eastern part of the agreement area are faring worse than those in the west, where the EAF is located (Nagy & Langendoen 2021).

2.3.2 Trend patterns among groupings of populations

In a previous exploration of common patterns in trends among waterbird populations in the EAF, Schekkerman *et al.* (2018) found a strong taxonomic signal. Waders (Charadrii), particularly those breeding in the Siberian Arctic, showed particularly negative trends on average (more severely so in the short term), while geese, flamingos, and pelicans and cormorants showed the most favourable trends. Probably related to this finding, waterbirds using intertidal mudflats and depending on benthic food outside the breeding season (both groups comprising many wader species) did less well compared to those using other habitats and feeding on plant material or fish. With respect to breeding climate region, unfavourable short-term trends were found in populations breeding in (particularly Siberian) arctic and boreal regions (again including many waders), and in those breeding or spending the non-breeding season in Southern Africa.

With the trend data updated to 2020, the taxonomic pattern remains broadly the same. Waders still show among the least favourable trends, although on the short term the mean trend of gulls (Laridae) has become even more negative (Fig. 2.2). The mean annual rate of change

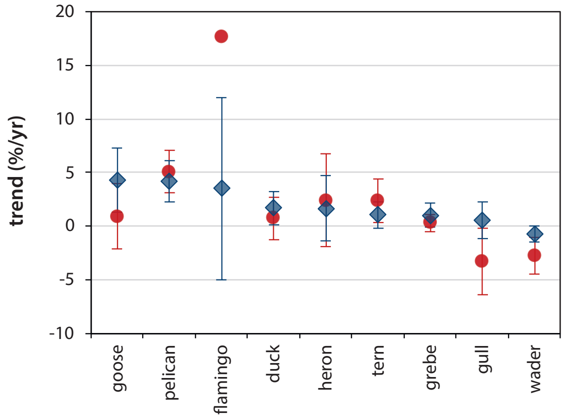

Figure 2.2. Average long-term (blue symbols) and short-term (red symbols) trends presented as mean annual change across the populations within each of the taxonomic groups (bird families or orders) in the coastal EAF. Bars denote 95% confidence interval of the mean. Groups are ordered by decreasing long-term trend values.

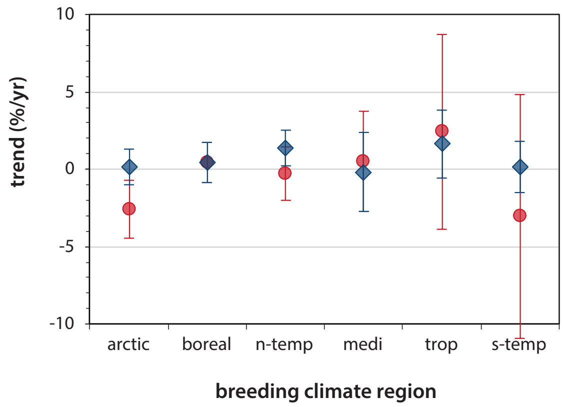

Figure 2.3. Mean long-term (blue symbols) and short-term (red symbols) trends of populations in the coastal EAF characterised by their breeding climate zone: arctic, boreal, north-temperate, Mediterranean, tropical (West and equatorial Africa) and south-temperate (Southern Africa).

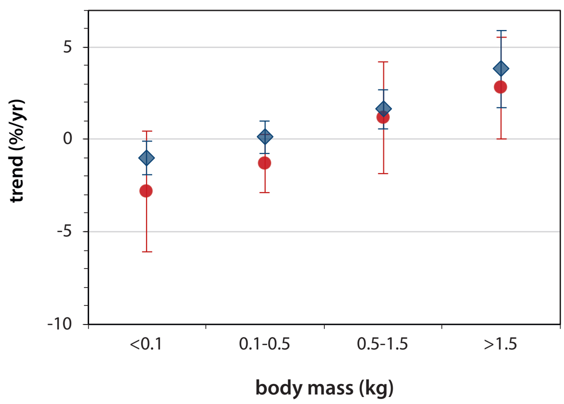

Figure 2.4. Mean long-term (blue symbols) and short-term (red symbols) trends of populations in the coastal EAF characterised by the species' body mass.

per population in all other taxonomic groups remains (slightly) positive, in both the long and short time scales, with flamingos and pelicans and cormorants still showing the most positive trends. A deceleration of the growth of goose populations, already noted in the previous assessment, has become clearer and the mean annual change in this group now approaches 0%. This is mirrored in a relatively unfavourable mean short-term trend for waterbirds with a herbivorous diet outside the breeding season. In some goose populations, signs of negative density dependence of reproduction have shown up after a decades-long period of population growth (e.g. Nolet *et al.* 2013, Layton-Matthews *et al.* 2019), while management (shooting) aimed at reducing damage to agricultural crops also impacts some populations, e.g. of Greylag Goose *Anser anser* and Barnacle Goose *Branta leucopsis* (Jensen *et al.* 2018, Powolny *et al.* 2018).

The overall pattern with respect to breeding region also remains similar to the previous analysis, with unfavourable mean short-term trends standing out in two groups of populations: those breeding in the Arctic and in Southern Africa (Fig. 2.3). In the arctic region, waterbirds breeding in the Siberian tundras show the strongest declines. Here, the faltering of lemming cycles which has occurred since the late 1980s is likely to have reduced the frequency of years with high reproductive success when predators concentrate on superabundant lemmings instead of birds' eggs and young (Underhill *et al.* 1993, Gilg *et al.* 2009, Nolet *et al.* 2013). Although Soloviev & Tomkovich (2014, and Chapter 4) found indications for an increase rather than a decline in nest success of waders in the western Russian Arctic in the period since then, the occurrence of such 'boost years' may be particularly important to population development, as shown for Brent Geese *Branta bernicla* by Nolet et al. (2013). Other problems may be associated with climate warming which strongly influences the entire arctic region (e.g. Box *et al.* 2019) and which may lead to reduced food availability for chicks due to a mismatch with the phenology of invertebrate food sources (Tulp & Schekkerman 2008, van Gils *et al.* 2016, Rakhimberdiev *et al.* 2018, Lameris *et al.* 2021, but see Meltofte *et al.* 2021).

The unfavourable trajectory of wader populations is also reflected in several other trait-based patterns, e.g. relatively negative (short-term) trends in waterbirds with a breeding season diet consisting of invertebrates, a non-breeding diet dominated by benthic fauna of intertidal habitats, and a small body size. These traits are shared by many wader species. The pattern of increasingly favourable trends in larger-sized birds extends beyond the size range of waders however, with the most strongly positive mean trends found in waterbirds weighing more than 1.5 kg (Fig. 2.4), reflecting the increases among flamingo, cormorants and pelicans.

2.4 Spatial patterns of change in non-breeding populations of waders

2.4.1 Declines in the south, increases in the north

When looking at changes in January numbers of waders by region between the 1980s-1990s and the 2010s-2020 for all wader species considered (Fig. 2.5), there is a strong tendency towards more negative changes from north to south along the coastal EAF, particularly in Africa. There seems to be an approximate latitudinal turning point between Morocco and Mauritania, with declines predominating further south along the African coastline, and increases more common to the north. The most positive changes seem to have occurred in southern Europe (Iberia). In NW Europe (north of France), the variability in change ratios becomes larger, with many populations showing relatively small changes while others show notable increases, particularly in the Wadden Sea (Netherlands to Denmark), but generally not still further north in Scandinavia. In Ireland and the United Kingdom, declines have been noted in several species within the recent period (e.g. Burke *et al.* 2018, Frost *et al.* 2021).

This large-scale latitudinal dichotomy is visible both in wader species with a single flyway population occurring throughout the entire EAF or large parts of it (Fig. 2.5a) and

Hans Glader

Wintering Wigeons

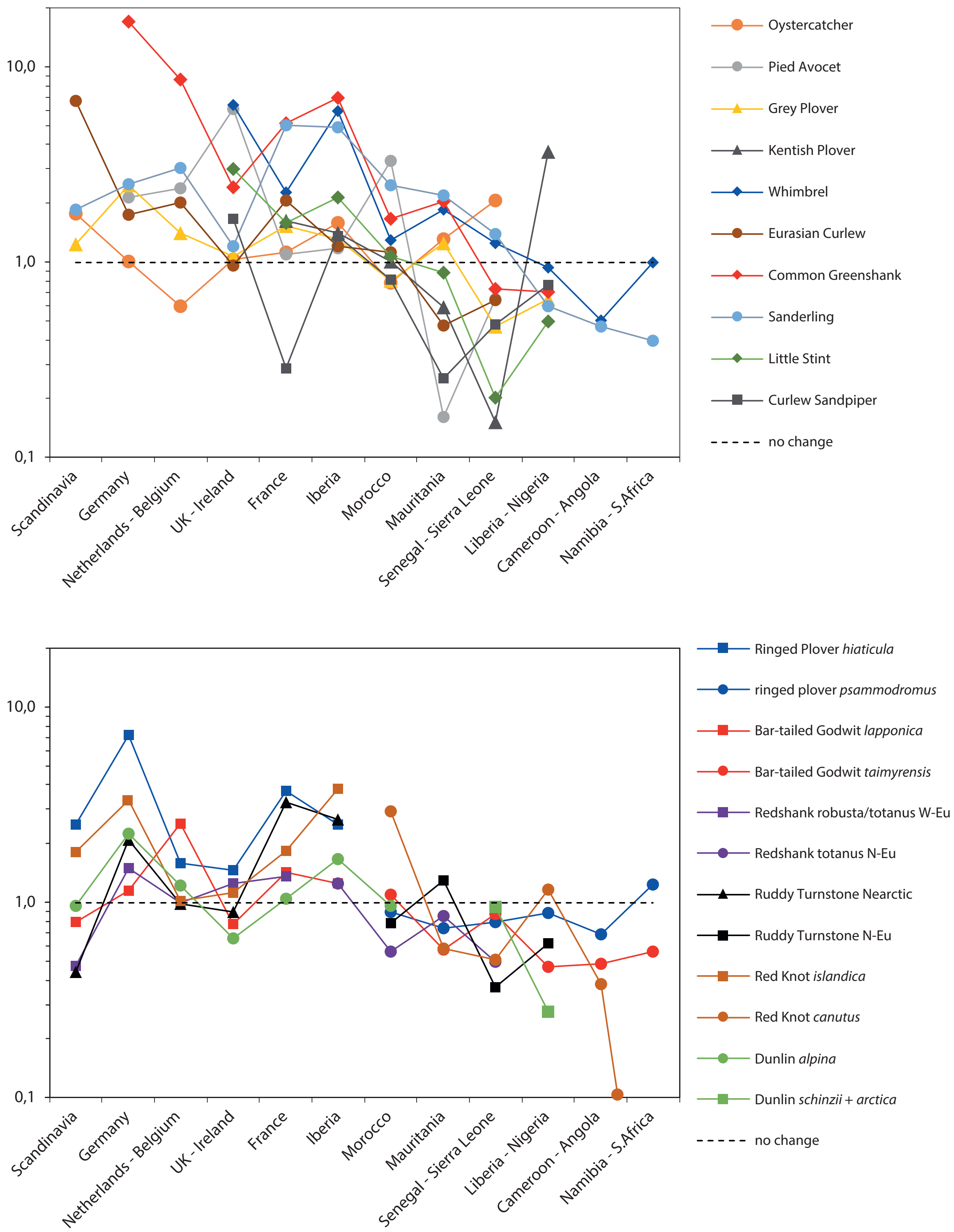

Figure 2.5. Spatial patterns in changes of non-breeding wader populations in 12 regions in the coastal EAF between the 1980s-1990s on the one hand and the 2010s-2020 on the other. The metric on the vertical axis is the proportional change in number between the two periods (values >1 denote increase, <1 decrease), here on a logarithmic scale for visual clarity. Symbols and lines of the same colour identify populations. The top panel (a) shows wader species of which a single flyway population occurs in the EAF (or for which a combined trend was calculated because the data did not allow separation of defined populations). The bottom panel (b) shows species in which multiple flyway populations are recognised. (In both panels, species are denoted by colour, populations by differing symbols connected by lines). In Red Knot, the change factor for Namibia – S Africa is close to 0.

in species in which multiple flyway populations are recognised (Fig. 2.5b). In the latter group as well, those populations spending the non-breeding season on the shores of Africa have mainly shown declines while in populations wintering mainly in Europe, positive changes predominate. Moreover, within several species declines also tend to become be stronger towards the south along the sub-Saharan part of the coastal EAF, although there are exceptions, e.g. Common Ringed Plover *Charadrius hiaticula* (Fig. 2.5).

A spatial pattern also exists in the completeness of count data, with the coastal regions of Africa east and south of Guinea-Bissau generally less well covered than those to the north. However, good coverage has been achieved in both periods in several major sites in Ghana, Namibia and South Africa, and counts there also indicate that declines prevail (e.g. Simmons *et al.* 2015, Barshep *et al.* 2017). It therefore seems unlikely that the pattern visible in Fig. 2.5 is an artefact resulting solely from diminished coverage of sites along the more southerly part of the African coastline after the turn of the century. Also, there is good agreement between overall changes in population size (%/year) per species/population as derived from our tabulation of country totals and those obtained by the formal trend analyses in Annex 1 (correlation r_{20}= 0.80, P<0.001).

2.4.2 Possible explanations

One possible explanation for the stronger decline of migratory waders in the southern parts of the coastal EAF than in the north is that the suitability of coastal habitats has deteriorated particularly in southern regions, e.g. through disturbance, degradation or destruction by human activity. However, at present it seems unclear why such developments would predominate along the more southerly coasts of Africa (see also Chapter 3).

A second hypothesis is that waders spending the non-breeding season in various parts of the EAF may originate from different breeding areas. Changes occurring in specific breeding areas may then cause changes in non-breeding distributions via spatially differential population trends. Particularly, as mentioned above, wader populations breeding in the Siberian Arctic have shown stronger than average declines, and these winter mainly in the African parts of the EAF (e.g. Little Stint *Calidris minuta*, Curlew Sandpiper *C. ferruginea* and Red Knot *C. c. canutus)*. However, stronger declines in the southernmost parts of the EAF are not entirely restricted to waders from Siberian breeding grounds (e.g. Sanderling *Calidris alba* and Common Greenshank *Tringa nebularia*), hence changes in the Siberian Arctic cannot explain the entire pattern in Fig. 2.5.

Menno Hornman

Discussing best counting strategy with maps of the area

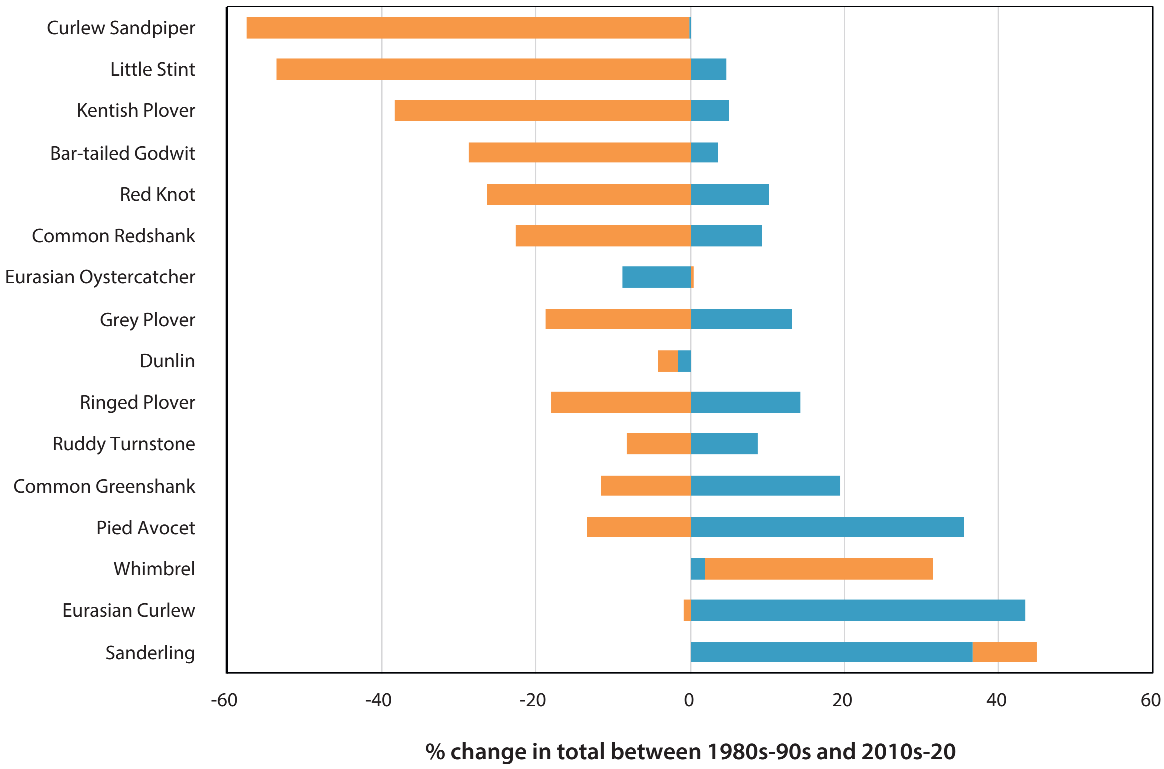

Figure 2.6. Proportional changes between the 1980s-1990s and 2010s-2020 in total wader numbers in January in the northern part (N Europe south to Morocco; blue bars) and in the southern part (Mauritania to S Africa; orange bars) of the East Atlantic Flyway, both expressed as % of the 1980s-90s species total. Species are shown in the order of net change in abundance throughout the EAF (species with the most negative sum of increases and decreases on top).

A more general way in which changes in population size caused elsewhere may differentially affect numbers of birds in different non-breeding sites is a 'buffer effect' (Brown 1969, Gunnarsson *et al.* 2005). Such an effect may become apparent in populations that occur in habitats of different quality where the 'best' sites are occupied preferentially. When the population increases, birds are forced or choose to settle in less preferred sites as the best sites become saturated. Conversely, the lower-quality sites will be vacated first when the population declines. For migratory waders, one aspect of site quality may be the distance to the breeding grounds. Longer migrations are likely to be more costly than shorter ones in terms of overall energy requirements, duration, risk of encountering fatal hazards or conditions leading to delays, or options to match arrival to variable conditions in the breeding area. It may therefore be expected that when waders breeding at northern latitudes and migrating to non-breeding sites spread along the EAF decline, numbers in the southernmost areas will decrease more strongly than those further north. Indeed, those wader species that decline most strongly as a whole also show the largest declines along the African coasts, and Little Stint and *canutus* Red Knot particularly so in the southernmost regions (Figs. 2.5 & 2.6). However, under a buffer effect we may also expect a relationship between the steepness with which the change ratios (1980s-1990s to 2010s-2020) of non-breeding wader numbers decline towards the south and the overall population trends: strongly declining populations should show a steeper slope than stable or increasing ones. However, no such pattern is apparent in this dataset, neither along the EAF as a whole nor along the African coastline in particular (analysis not shown here).

A fourth hypothesis which may explain the spatial pattern of change is a general northward shift of non-breeding waders in response to warming winter climate in Europe. Increasing temperatures and a declining frequency of cold spells render it more feasible, and profitable, for waders to overwinter in areas such as the Wadden Sea or even in Scandinavia or Iceland. A north-eastward shift in the non-breeding distribution has been documented in several wader species both within the UK and Ireland (Austin & Rehfish 2005) and on a larger NW European scale (MacLean *et al.* 2008), as well as among wildfowl within Europe (Lehikoinen *et al.* 2013; Pavón-Jordán *et al.* 2019) and North America (Meehan *et al.* 2021). In Fig. 2.5, such a shift into more northern sites seems apparent in species like Eurasian Curlew *Numenius arquata*, Bar-tailed Godwit *Limosa lapponica*, Grey Plover *Pluvialis* *squatarola* and Dunlin *Calidris alpina* which already wintered in NW Europe in the 1980s, but also to some extent in species that migrated mainly to southern Europe and Africa, like Common Ringed Plover, Pied Avocet *Recurvirostra avosetta*, Whimbrel *Numenius phaeopus*, Common Greenshank *Tringa nebularia* and even Little Stint *Calidris minuta*. It is then conceivable that a northward shift in European wintering populations opens up space in estuaries in SW Europe, that can be occupied by wader populations formerly migrating to Africa. Some of these did so via stopover sites in W Europe, and these birds might thus reduce their migration distance by short-stopping. Shifts in non-breeding distribution from Africa towards Europe have been observed in some other waterbird groups, for instance in Eurasian Spoonbill *Platalea leucorodia* (Lok *et al.* 2011). In several dabbling duck species as well, decreases in Africa and the Mediterranean region seem to coincide with increasing trends in numbers in NW Europe (e.g. Chapter 6).

In this winter-climate scenario, increases in the north can be expected to balance or outweigh declines further south, the latter resulting in a net population increase. While a breakdown of losses and gains (Fig. 2.6) suggests that this expectation is met in some wader species in the EAF (near-balance in Ruddy Turnstone *Arenaria interpres*, Ringed Plover and Grey Plover, increase in Pied Avocet), others do not conform, particularly those showing the strongest declines along the African coasts (Little Stint, Curlew Sandpiper, Bar-tailed Godwit and Kentish Plover *Charadrius alexandrinus)*. This suggests that other driving factors are involved as well. The four explanations discussed here are however not mutually exclusive and several or all might apply, and affect different species and populations the most.

2.4.3 Consequences of distributional shifts for population monitoring and assessment

Clearly, more analyses and research than the exploration presented here are needed to evaluate the explanations discussed above and the possibility that waders formerly migrating to Africa now increasingly remain in Europe. If such a shift does occur however, this will 'muddle up' the distinction between intraspecific flyway populations (formerly) migrating to different parts of the EAF. For instance, distinguishing between birds spending the non-breeding season in Europe (sometimes including Morocco, as in Dunlin of the subspecies *alpina*) and those in Africa is used as a proxy for assignment to biogeographical populations in several species. Though not perfect (see e.g. Engelmoer & Roselaar 1998; Delaney *et al.* 2009) this is useful as waders from different populations usually cannot be distinguished in the field in non-breeding plumage. Our current knowledge of distribution and boundaries of biogeographical populations is based on data from ring recoveries, colour-ring resightings and morphometrics of birds, assembled to a considerable extent in the 1980s and 1990s (e.g. Dick *et al.* 1976, Piersma *et al.* 1987, Summers *et al.* 1989, Wymenga *et al.* 1990, Engelmoer & Roselaar 1998). A risk therefore exists that in species with multiple biogeographical populations, distributional shifts resulting in overlap and mixing remain undetected (see also Tománková *et al.* 2013). They may then be misinterpreted as differential changes in the size of these populations, seriously compromising our ability to monitor them. It is therefore advisable to extend and refresh research into the current (and future) migratory connectivity and flyway population boundaries of waders and other waterbirds. Nowadays, a suite of new technologies is available to aid such a research programme, including analysis of DNA and isotopes and sophisticated tracking techniques.

Distributional shifts may also affect the reliability of using IWC counts for population monitoring in other ways (e.g. Fox *et al.* 2019). If a population shifts into a part of its range that is monitored better than its former haunts, this can cause an apparent increase even if the population remained stable as a whole, and *vice versa*. The Eurasian Curlew may be a case in point, as the IWC trend clearly differs from the decline indicated by all breeding data (see Chapter 13). This emphasises the importance of maintaining and strengthening monitoring networks along entire flyways such as the EAF.

Harvey van Diek

Whimbrel | **Courlis corlieu** *(Numenius phaeopus)*

Eurasian Oystercatcher | **Huîtrier pie** *(Haematopus ostralegus),* Schiermonnikoog, Netherlands (Harvey van Diek)

3. Pressures and conservation measures for waterbirds along the East Atlantic Flyway, update 2020

Olivia Crowe, Agyemang Opoku, Geoffroy Citegetse, Tim Dodman & André van Kleunen

Summary

An assessment of the pressures that waterbirds and their sites along the EAF are facing, as well as of some conservation measures underway, was based on the environmental monitoring assessment from 115, mostly major, coastal sites spread over 31 countries. This selection of sites is estimated to support a large proportion, around 60%, of all waterbirds occurring along the whole coastal flyway during the non-breeding season. It was supported by a pilot study on the use of high-resolution environmental datasets that are available online. Direct anthropogenic pressures continue to dominate the profile of pressures affecting the flyway, and the direct effects of climate change, such as shifting migration patterns and impacts on habitat suitability and availability, are becoming increasingly evident. Many countries are considering management needs for waterbirds, and are implementing a range of mechanisms including site protection, appropriate regulations, waste management, as well as a range of targeted actions towards species and habitat management. These actions have had limited effectiveness to date, and it may be timely to unpack these broad conservation measure categories towards identifying specific measures that are most effective in a variety of scenarios, perhaps supported by case studies. Ongoing engagement with local communities is hugely valuable, and future initiatives and efforts to increase awareness-raising on the value of protecting ecosystem service of coastal wetlands should be encouraged and supported.

Résumé

Une évaluation des pressions auxquelles sont confrontés les oiseaux d'eau et leurs sites le long de l'EAF, ainsi que de certaines mesures de conservation en cours, s'est basée sur l'évaluation du suivi environnemental de 115 sites côtiers, pour la plupart majeurs, répartis dans 31 pays. On estime que cette sélection de sites abrite une grande proportion, environ 60 %, de tous les oiseaux d'eau présents le long de l'ensemble de la voie de migration côtière pendant la saison de non-reproduction. Elle a été soutenue par une étude pilote sur l'utilisation d'ensembles de données environnementales à haute résolution qui sont disponibles en ligne. Les pressions anthropiques directes continuent de dominer le profil des pressions affectant la voie de migration, et les effets directs du changement climatique, tels que le déplacement des schémas de migration et les impacts sur la qualité et la disponibilité des habitats, deviennent de plus en plus évidents. De nombreux pays prennent en considération les besoins de gestion des oiseaux d'eau et mettent en œuvre une série de mécanismes comprenant la protection des sites, des réglementations appropriées, la gestion des déchets, ainsi qu'une série d'actions ciblées sur la gestion des espèces et des habitats. Ces actions ont eu une efficacité limitée jusqu'à présent, et il serait peut-être opportun de décortiquer ces grandes catégories de mesures de conservation afin d'identifier les mesures spécifiques qui sont les plus efficaces dans une variété de scénarios, peut-être en s'appuyant sur des études de cas. L'engagement continu avec les communautés locales est extrêmement précieux, et les initiatives et les efforts futurs pour accroître la sensibilisation à la valeur de la protection des services écosystémiques des zones humides côtières devraient être encouragés et soutenus.

3.1 Introduction

This chapter provides an overview of the pressures that waterbirds and their sites along the East Atlantic Flyway (EAF) are facing, as well as of some conservation measures underway. It includes a comprehensive assessment that has built upon the programme of environmental monitoring initiated in 2013 (van Roomen *et al.* 2013). It is based on a questionnaire completed by observers, national coordinators of the International Waterbird Census (IWC), site managers and other authorities from the non-breeding sites within the flyway (see details in Annex 2).

The assessment is an update of the overview of human activities and conservation measures along the EAF until 2017 (Dodman *et al.* 2018). It is based on the "Driving Force - Pressures - State - Impacts – Responses" framework (DPSIR, Gabrielsen & Bosch 2003), Oesterwind *et al.* 2016). Driving forces (e.g. the need for food by humans) can result in pressures (e.g. overfishing), which change the state of the ecosystem (e.g. reduced fish biomass as food for waterbirds), which then has consequences or impacts that require a management response. Effective responses in turn should over time lessen the impacts of the driving forces.

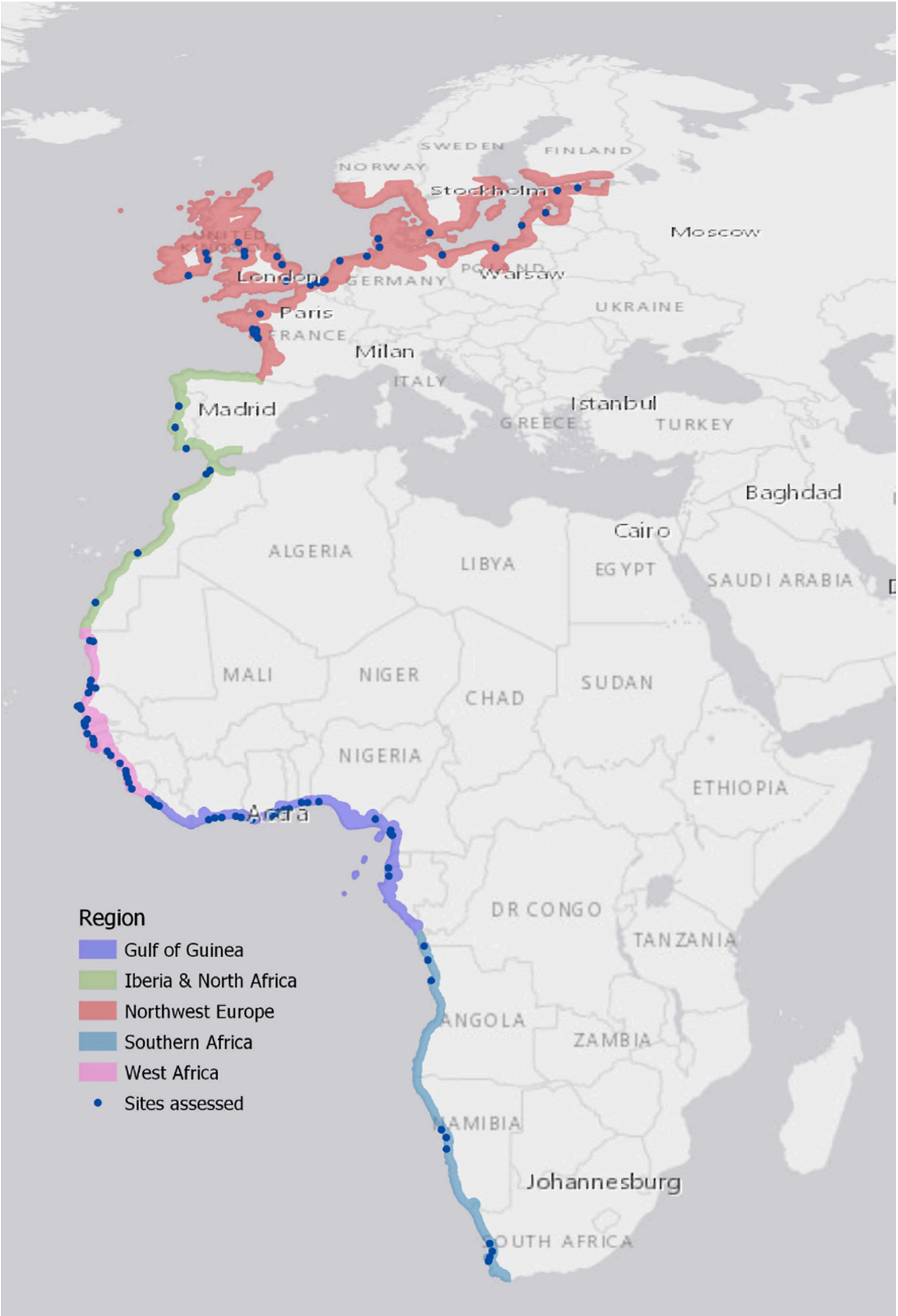

Figure 3.1. Coastal EAF regions as used in this study, and the 115 sites for which environmental monitoring assessments were submitted and included in these analyses.

Litter and garbage on beaches and in estuaries is an increasing problem

The results presented in this chapter are based on completed updated questionnaires, and are supplemented by a pilot study on the use of high-resolution environmental datasets that are available online (Annex 3).

3.2 Materials and Methods

The data used to inform pressures and conservation measures as relevant to waterbirds and their sites are based mostly on environmental monitoring forms (questionnaires) which were completed by observers, site managers and IWC National Coordinators in 2020. Supplementary data were extracted from other global datasets available online (Annex 3).

The environmental monitoring assessment is based on the data from 115, mostly major, coastal sites spread over 31 countries. This selection of sites is estimated to support around 60% of all waterbirds occurring during the non-breeding season along the whole coastal flyway. These sites and their allocation to geographical regions are illustrated in figure 3.1. The regions are (number of sites per region in brackets):

- Northwest Europe: South-Sweden, Baltic Countries - Atlantic France (37 sites)
- Iberia - North Africa: Iberia – Morocco (8 sites)
- West Africa: Mauritania - Sierra Leone (37 sites)
- Gulf of Guinea: Liberia - DR Congo (22 sites)
- Southern Africa: Angola - South Africa (11 sites)

A full description of the environmental monitoring assessments, and how the data were treated for the analyses, is given in Dodman *et al.* (2018) and van Kleunen *et al.* (2018). The full results of the environmental monitoring assessment for 2020 are summarised in Annex 2.

For these analyses, data were extracted at two spatial scales: (a) across the whole flyway and (b) at regional level. Analyses are mostly presented as frequencies (percentages) of the answered questions and ordered from most frequent to less frequent. As not all questionnaires were filled in completely the assessments for separate questions often deviate from the total number of sites which questionnaires from the flyway or region. The question specific sample size are given in the graphs presenting the results.

3.3 Principal pressures to waterbirds along the EAF

While direct anthropogenic pressures continue to dominate the profile of pressures affecting the flyway, direct effects of climate change are becoming increasingly evident, acting alone or in some cases in combination with other factors. Warming temperatures are resulting in shifting migration routes and distributions (Pavón-Jordán *et al.* 2019), and/ or are affecting waterbirds on their breeding grounds (e.g. Fox *et al.* 2016). Sea-level rise and increasing extreme weather events are affecting habitat availability (Clausen & Clausen 2014, Breiner *et al.* 2021) at many key sites along the flyway. Breiner *et al.* (2021) projected that climate change will reduce habitat suitability for waterbirds at 58% of the existing Critical Sites within Africa-Eurasia, with a majority of the African sites being affected. These effects add to a multitude of others, some of which

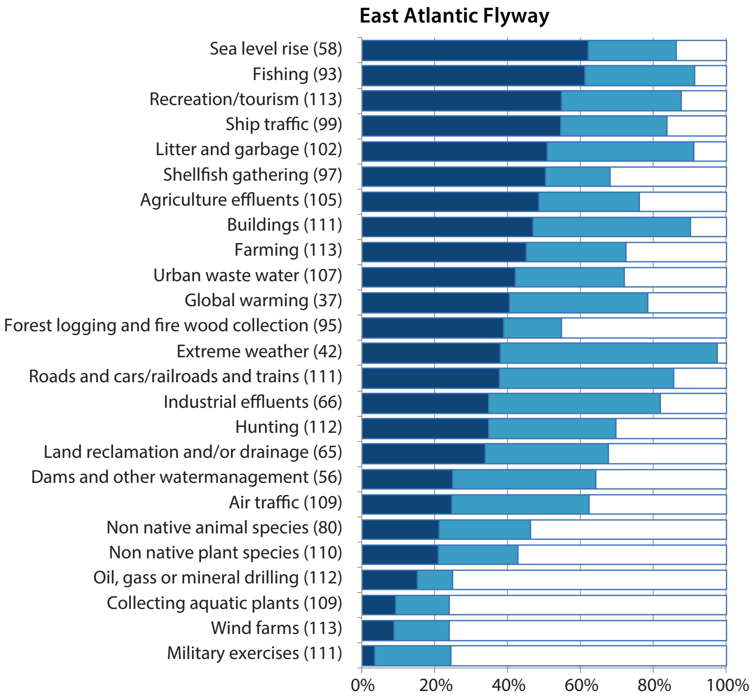

Figure 3.2. Frequencies of pressures reported from the 115 participating sites along the EAF having much, little or no effect (ordered on the frequency of pressures with much effect). The number of monitoring assessments that answered the specific question on the subject are given between brackets.

seem to be expanding and/or increasing in intensity compared to the last assessment in 2017 (Dodman *et al.* 2018).

Recreation continues to feature highly in the environmental monitoring reports and was among the most prevalent of the pressures reported in 2020 (98% of assessments), with a relatively high proportion reportedly affecting waterbirds (figure 3.2). No doubt, levels are increasing with general increases in human densities and urbanisation at sites across the flyway (Annex 3), and the latter has doubled in the past 25 years across the flyway. Litter and garbage also featured highly, and are probably also increasing with human population levels; it was reported in almost 90% of the assessments and at many sites litter was reported to affect waterbirds.

Levels of fishing and shipping remain high in terms of prevalence throughout the network of sites, and in their effects on waterbirds. This is unfortunately a very difficult pressure to monitor effectively, with relatively high levels of unregulated and artisanal fishing across many parts of the flyway. However, technological improvements (Annex 3) in monitoring traffic and activities in the future should help to assess the effective implementation of related regulations.

The full range of pressures reported in the assessments are illustrated in figure 3.2. Given the cumulative impacts of these pressures on waterbirds, it is becoming increasingly important, and urgent, to implement effective measures that will reduce the effects of those pressures that we have the ability to manage, given there are many others that we cannot mediate.

3.4 Principal pressures per region

3.4.1 Northwest Europe

Rising sea levels, increasing frequency of storms and extreme weather events were reported among the greatest pressures facing waterbirds in NW Europe, reducing the availability of, and displacing waterbirds from suitable roosting and feeding habitats. Most countries in Europe have estimated that there have been significant losses of coastal wetlands (Airoldi & Beck 2007), with expanding coastal development and defence reported as having the greatest known impacts on soft-sediment habitats. Through 'coastal squeeze', the process in which rising sea levels and other factors, such as hard infrastructure, causes loss of space in both directions (land and sea), coastal ecosystems no longer have the ability to maintain their essential functions (Silva *et al.* 2020). Other climate change effects like global warming which can lead to notable shifts in distributions, are increasingly reported in the region (Austin & Rehfisch 2005, Maclean *et al.* 2007, Pavon-Jordán *et al.* 2020, see also chapter 2), and while rising temperatures were reported for relatively few sites, it is important to point out that this factor was considered to impact on waterbirds where it was reported.

Recreation/tourism was also among the most prevalent pressures causing greatest impacts in NW Europe (Fig. 3.3) and was reported at proportionally more sites in 2020 than in 2017. With many key sites in the region centred near highly developed coastal towns and cities, coastal wetlands tend to be located near highly popular recreation areas, which if not managed appropriately, causes disturbance to waterbirds.

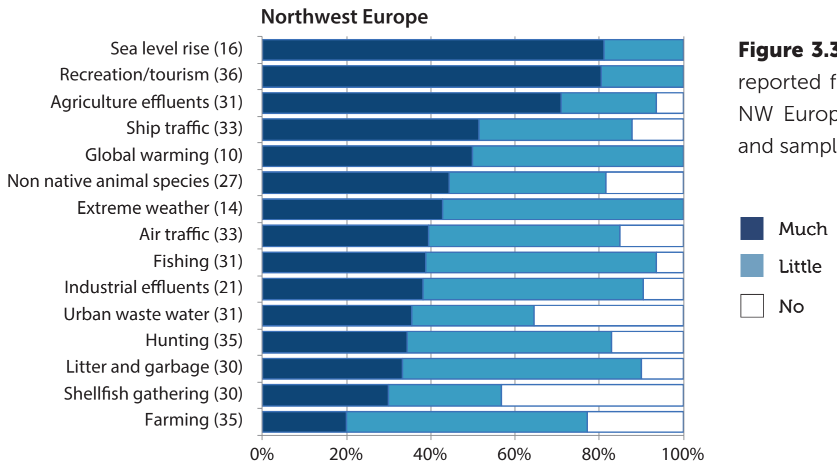

Figure 3.3. Top 15 of the listed pressures reported from the 37 participating sites in NW Europe. Explanation of effect classes and sample sizes as in Fig. 3.3.

Farming is a major economic practice throughout much of the region, and damaging side-effects of runoff and pollution continue to present major threats to the sites. Other pressures identified as posing at least some effect on waterbirds at a relatively high proportion of sites were fishing, industrial effluents, litter and garbage and roads and rails. However, the proportion of sites where these pressures were considered to strongly affect the birds was proportionally lower relative to the effects of other threats listed above.

Compared to 2017, the most notable changes were an increase in the reported effects of recreation and fishing and a decrease in the effects of introduced species, waste-water (domestic and urban) and collection of shellfish.

Many countries in the region reported increasingly milder winters, generally resulting in increasing numbers of waterbirds in north and eastern European countries and declining numbers further south and west (van Roomen *et al.* 2020). In some cases, the warmer winters were accompanied by other issues, e.g. increased recreation activities causing disturbance. Pressure from offshore windfarm development was also mentioned. Other pressures mentioned included bycatch, increased shipping and potential impacts of oil-spills.

3.4.2 Iberia – North Africa

The environmental monitoring information for Iberia – N Africa came from three sites in Portugal and five in Morocco. Recreation and tourism, shellfish gathering, litter and garbage, buildings and farming were identified among the most prevalent pressures in this region (Fig. 3.4).

At a country level (Encarnação 2020), disturbance by human activities linked to fishing and recreation, as well as the destruction of habitats (e.g. through pollution, conversion to fish-farms etc.) were reported to be the greatest pressures in Portugal. Specifically in some estuaries, the transformation of salt complexes into intensive fish farms, together with rising sea levels, is causing a reduction in the availability of places of refuge and intertidal rest for a wide range of waders. In Morocco, poaching, high levels of dis-

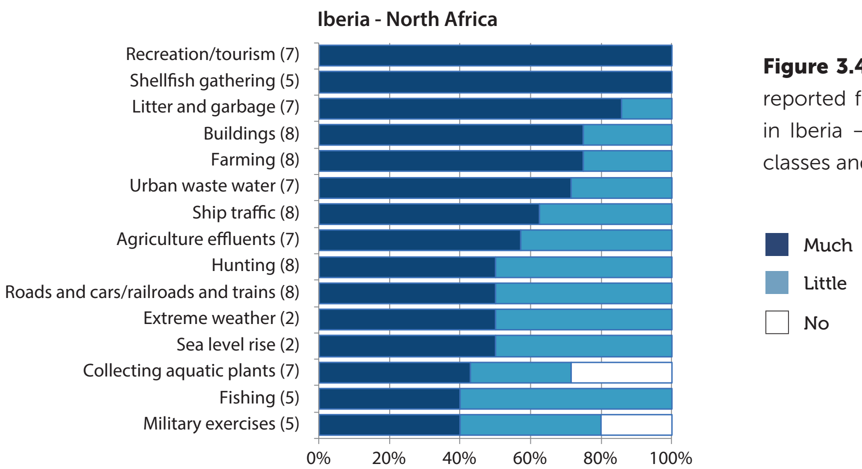

Figure 3.4. Top 15 of the listed pressures reported from the eight participating sites in Iberia – N Africa. Explanation of effect classes and sample sizes as in Fig. 3.3.

turbances to birds, and increasing numbers of stray dogs were also reported (Dakki et al 2020).

A case study that featured one of the sites included in the environmental monitoring assessment, namely Ria Formosa in southern Portugal, provided further insights on some of these widespread pressures (Newton *et al.* 2020). The site is a popular destination for sand and beach recreation, as well as ecotourism. Artisanal fishing and shellfish harvesting are important activities; shellfish concessions also exist in the wetland, presumably causing disturbance to waterbirds as well as loss of the resource. Other pressures reported included considerable development (port, marinas) and associated activities (dredging) as well as increased encroachment from nearby towns. Although much debated and maybe not going ahead, the plans for enlarging the airport of Lisbon in the most important wetland of the country, the Tagus estuary, is a major threat to waterbirds of the EAF (Alves & Dias 2020).

The threat from sea level rise does not feature strongly in the environmental monitoring assessments from the region. However, more than two-thirds of the Moroccan coastline is known to be retreating, and its preservation is a question of increasingly growing concern (Kasmi *et al.* 2020).

The small sample of sites monitored in 2017 and/ or 2020 limits our ability to interpret change since 2017, but there does seem to have been an increase in the prevalence and effects of hunting and trapping and introduced species.

3.4.3 West Africa

In W Africa, several pressures are affecting migratory and resident waterbirds and wetlands. The key threats reported were overfishing, shipping, mangrove cutting and firewood collection, shellfish harvesting, farming and global warming (Fig. 3.5).

W Africa is home to one of the most diverse fisheries in the world. It has a social and economic importance for millions of people living along the coast from Mauritania to Sierra Leone. Most fish populations in W Africa are now overexploited, putting the livelihoods of populations at risk as well as affecting populations of waterbirds. Artisanal and industrial fishing has increased, resulting in the depletion of fish stocks. Cutting of mangroves for construction and firewood destroys the habitat and deprives some species of their roosting sites. Mangroves are also used to smoke fish and as a source of energy to produce salt in many sites in W Africa. Collection of shellfish is a common activity and is permitted in marine protected areas. It is done specifically by women who spend most of the time collecting them in mudflats and in the mangroves, often

Tim Dodman

Salt collection is a common activity in coastal EAF

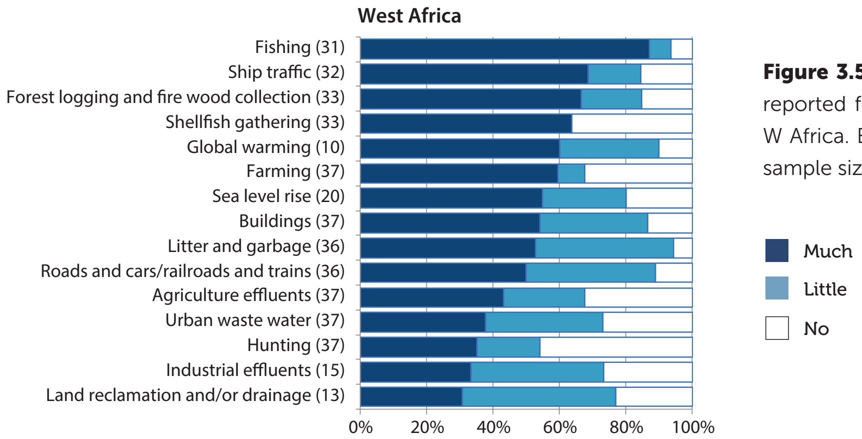

Figure 3.5. Top 15 of the listed pressures reported from the 37 participating sites in W Africa. Explanation of effect classes and sample sizes as in Fig. 3.3.

causing disturbance to birds, especially during periods when the number of collectors is considerable.

Farming is another main threat to wetlands in W Africa. Some of the sites that were assessed have human populations living within and around the sites, who continue to expand their cultivated fields, thereby destroying suitable waterbird habitats. Additionally, increases in pollution of waterbird habitats is caused by increased pesticides caused mostly by the development of agribusiness (i.e. large scale rice, cashew, palm oil etc.). In some cases, farmers are shifting their former rice fields to other areas causing further destruction to mangroves and other waterbird habitats.

Climate change is having impact in W Africa, accelerating coastal erosion, changing the dynamics of sandbanks, mudflats and islands and in many cases reducing the availability of critical habitats for colonial breeding birds and waders.

Compared with the 2017 assessment, the most notable changes were increases in the reported effects of urban waste, and industrial and agricultural effluents, while most other threats remained constant or recorded marginal declines.

3.4.4 Gulf of Guinea

The principal pressures reported for the Gulf of Guinea in January 2020 relate to fishing, forest logging and firewood collection, litter and garbage, and buildings, with over 70% of sites recording a high pressure from these four categories (Fig. 3.6). Although fishing is prevalent throughout much of the coastal zone, it does not always present a direct pressure to coastal habitats or to waterbirds. Urbanisation and destruction of coastal habitats have more significant long-term impacts in terms of waterbird habitat. Sea level rise was the principal pressure noted relating to climate change.

Encroachment of sites, pollution, plastic waste and the conversion of sites for dwellings were the most significant threats noted in Liberia, where there is only limited formal protection of coastal sites (Garteh 2020). In Côte d'Ivoire, urbanization and destruction of habitats particularly impact breeding birds around Abidjan, where pollution is also an issue, especially of plastics; waterbirds are also hunted at some coastal sites (Kouame 2020).

The principal pressures noted in Benin were uncontrolled development of settlements and dumping of waste, whilst fishing occurs in most areas surveyed (Daouda *et al.* 2020). In Nigeria, specific threats recorded at sites were dredging, logging and cutting of mangroves for fuel wood

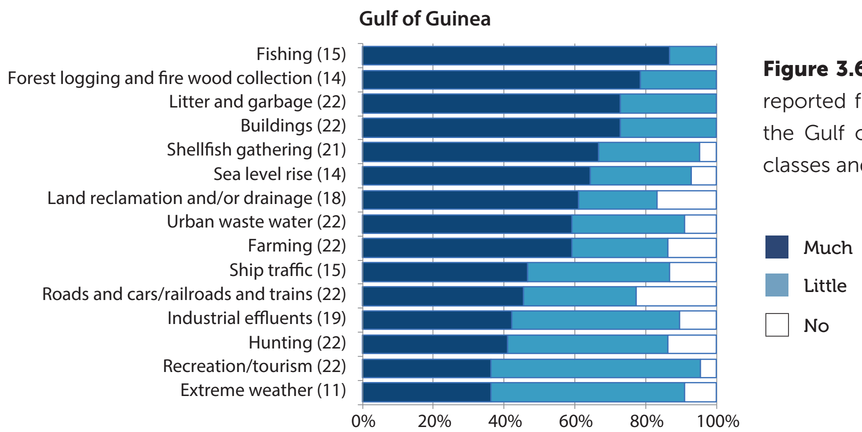

Figure 3.6. Top 15 of the listed pressures reported from the 22 participating sites in the Gulf of Guinea. Explanation of effect classes and sample sizes as in Fig. 3.3.

Grazing by life stock at wetland edges can lower habitat quality for waterbirds

and commercial purposes, industrial impacts, fisheries, waste and pollution and general disturbance (Onoja 2020). In Cameroon, habitat destruction remains a high risk for the Wouri Estuary due to industrial development, hydropower dam construction, heavy sand extraction, clearing of mangroves and overfishing, notably in the densely populated Douala region; habitat destruction also threatens the Lower Sanaga River (Ajonina *et al.* 2020).

Offshore and onshore oil exploitation represents the most significant pressure in the coastal zone of The Congo, with oil deposits noted along the coastal zone (Mokoko Ikonga 2020). Specific pressures in the narrow coastal belt of the Democratic Republic of Congo include disturbance, overfishing, deforestation, pollution and urbanisation, resulting in degradation of some wetland habitats (Mavuemba Tuvi *et al.* 2020).

Parts of the Gulf of Guinea coastal zone are densely populated, including major cities such as Abidjan, Accra, Lagos and Douala. Thus, disturbance of sites important for waterbirds in various forms inevitably presents pressures, including from activities such as fishing, shellfish gathering, agriculture and collection of natural resources, notably mangrove wood. Pollution is also a widespread by-product of human habitations, with plastic recorded as a particular problem in the region. However, the most significant impacts of urbanisation, land reclamation and various industrial and agricultural developments relate to habitat destruction, which may often be irreversible.

Although oil, gas and mineral drilling were not reported widely as a significant threat at many sites (Fig. 3.6), the oil industry exerts a high environmental impact in several coastal areas of the Gulf of Guinea, especially in the coastal zone between Nigeria and The Congo. Other pressures noted in 2017 included coastal erosion, deforestation, conversion to agriculture, dredging, sand mining, pollution, introduction of exotic species and depletion of coastal resources and deforestation (Dodman *et al.* 2018). Most likely these pressures remain relevant in 2020.

3.4.5 Southern Africa

The principal pressures reported for Southern Africa in January 2020 relate to recreation/ tourism, land transport,

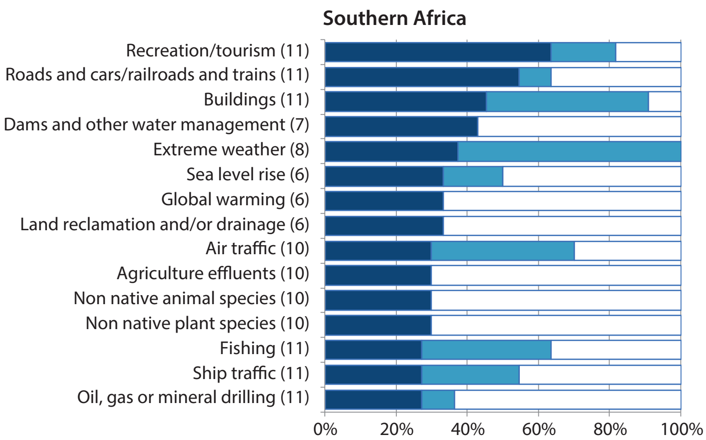

Figure 3.7. Top 15 of the listed pressures reported from the 11 participating sites in Southern Africa. Explanation of effect classes and sample sizes as in Fig. 3.3.

buildings and water management (Fig. 3.7). These pressures are not likely to represent priority threats to sites at a regional level, rather reflecting the location of the 11 sites that were assessed, notably in central Namibia and north of Cape Town in South Africa, pockets of the long coastline where coastal development, recreation and tourism are prevalent. Dams and water management was only recorded as a threat in three coastal sites of close proximity to each other in South Africa, but rank highly as only seven sites were assessed for this pressure. By comparison, the most relevant threats to waterbirds from sites assessed in 2017 were overfishing, including shellfish gathering, and urbanisation (Dodman *et al.* 2018).

Recreation and tourism are popular pastimes in the coastal zone of South Africa's Western Cape Province and at Namibia's Walvis and Sandwich Bays. The Walvis Bay Wetlands constitute the most important coastal wetland for migratory birds in Southern Africa, yet just north is Swakopmund, Namibia's main seaside holiday resort (Demasius & Marais 1999). At Walvis and Sandwich Harbour, low-flying aircraft operated by tour companies pose a direct threat to birds, such as flamingos (BirdLife International 2021).

Three of the four sites assessed in Angola are all within the Mussulo Lagoon close to the capital city Luanda. These sites and the wider lagoon are under constant pressure from various human activities, including illegal settlement on islands and threats of land reclamation both for housing and for the development of large touristic resorts (Kodo *et al.* 2020). The settlements on Ilhéu dos Pássaros in Mussulo Lagoon by fishing communities who had been displaced by urban developments elsewhere on the lagoon reported by Dodman *et al.* (2018) still remain. Further north, 50 ha of mangroves have recently been lost at the site Mangais de Nzeto due to infrastructural developments.

Extreme weather, sea level rise and global warming were noted as exerting high pressure in 2-3 sites assessed. Kolberg (2020) reported an evident effect of the worst drought in recorded history in Namibia with many sites being completely dry at the time of the counts, though this principally refers to inland sites.

The four sites assessed in South Africa were the Berg River, Langebaan, Olifants River Estuary and Verlorenvlei, all on the southwest coast. Langebaan Lagoon Nature Reserve is a popular tourism destination. Metal pollution and oiling incidents from urbanization and shipping pose a threat to the lagoon's future, whilst a planned steel smelter may cause water abstraction on the Lower Berg River and indirectly at Langebaan Lagoon. Commercial fisheries may also negatively impact piscivorous birds. The principal threat at Verlorenvlei is the disruption of water flow due to a series of man-made obstructions that disrupt hydrological fluctuations, causing flooding upstream, extensive siltation and reduction of freshwater load into the estuary, whilst intensive farming practices also persist around the main lake. Further north, threats to the Olifants River Estuary include damage to vegetation from vehicles and overgrazing (BirdLife International 2021).

Menno Hornman

Shell fish collection in Bijagos, Guinea Bissau

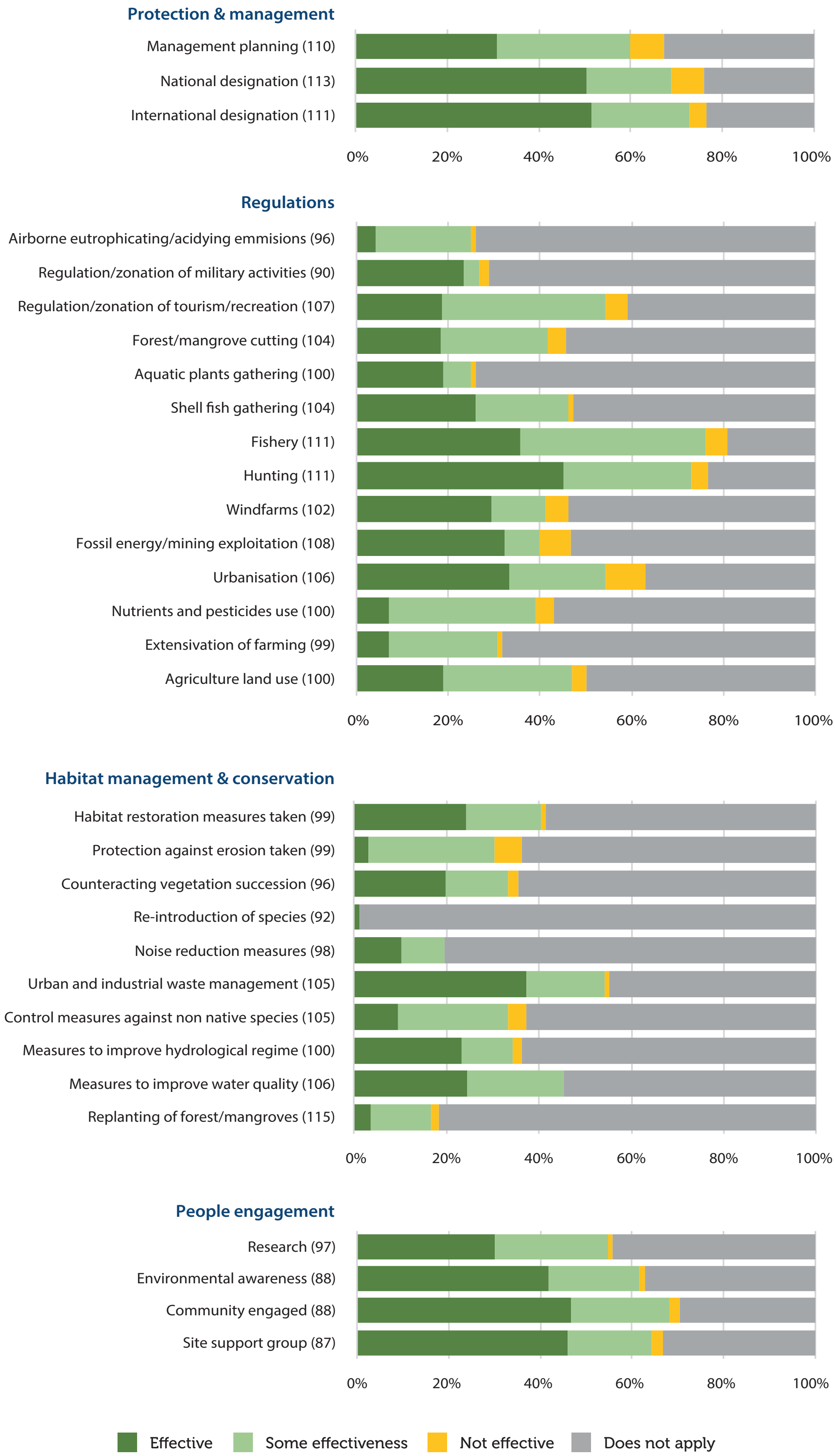

Figure 3.8. Conservation measures across the EAF, illustrating the proportion of sites where they are effective, have some effect, are not effective and where they do/are not applied. The number of sites for which the question about the occurrence of the pressure was answered is shown in brackets.

3.5 Conservation measures along the EAF

3.5.1 EAF overview

The conservation of coastal habitats along the EAF is key to ensuring that a healthy and resilient network of sites and habitats is available to migratory waterbirds. The list of measures included in the environmental monitoring assessment forms includes 31 potential measures, with most focussed on legal protection, site policy and regulations. Other measures inform active management of habitats, species management, communications (awareness-raising) and engaging communities, and research. Dodman *et al.* (2018) and van Kleunen *et al.* (2018) present a detailed overview of the merits of each of these conservation measures in the context of the EAF.

Throughout all regions, the need for conservation measures was identified for almost all assessed sites. In more than 80% of the sites, at least some measures were reportedly implemented. Quite high proportions of assessed sites are protected by national and/ or international designations and have management plans in place (Fig. 3.8). In European countries this is mostly due to the requirements of the Birds and Habitats Directives of the

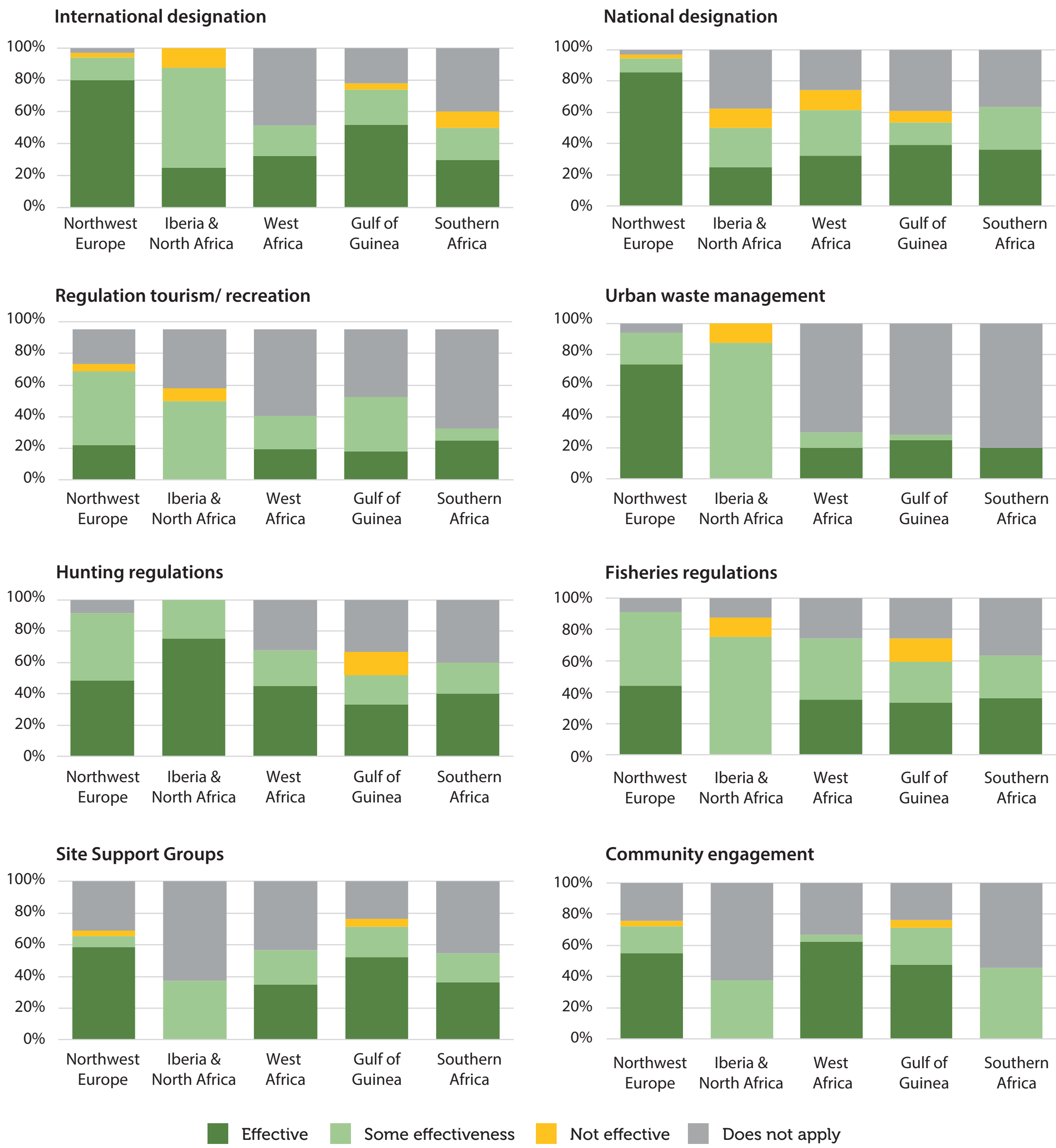

Figure 3.9. Conservation measures across the EAF regions, illustrating the proportion of sites where they are effective, have some effect, are not effective and where they do/are not applied.


Miguel Xavier

Community collection of rubbish, washed ashore in Angola

European Union. In Africa, many assessed sites have been protected at an international level, especially as Ramsar Sites and often also under national legislation.

Regulations are in place addressing many of the key pressures identified, with hunting and fishing regulations existing in the majority of sites. In many cases these regulations are considered to be effective only to a certain degree. Some regulations do not apply to all sites, e.g. exceptions exist for windfarms, aquatic plant gathering and others.

Waste management, water quality improvements and habitat restoration were identified as the most widely implemented habitat management and conservation measures. It is important not to lose sight of those measures implemented at relatively few sites, some of which are very specific and very important (e.g. mangrove management and replanting, habitat restoration and control of non-native species). It is interesting to see that most of the measures implemented to protect sites against erosion have been effective at a very small proportion of the sites where they have been implemented. Thus, replanting of mangroves was considered effective only at a few sites.

The impact of conservation activities is enhanced where local people are involved, given their proximity to (in many cases) and presence at the sites, their interest in their local wetlands, and their ability to report where damaging activities are taking place. Indeed, many of the sites throughout the flyway benefit from the interests and actions of local conservation groups and/or community engagement, where their inputs are considered to be effective. Many sites are also likely to benefit from research activities aimed at targeting the specific conservation needs of the sites.

3.5.2 Northwest Europe

The extent of designation of assessed sites is very high relative to the flyway overall and to most other regions (Fig. 3.9). Also, a higher proportion of sites (relative to the flyway overall) have management plans in place, mostly due to the requirements of the EU Birds and Habitats Directives. Most of the measures assessed are more effective in this region when compared with the overall flyway.

In terms of addressing the pressures in the region, regulation of recreation is taking place at almost 80% of sites, but is highly effective at only a small proportion of those. Urban and industrial waste management is high, and regulation of agricultural land use is slightly higher than overall, and the effectiveness of these measures is also considered to be relatively high, perhaps resulting in the slight improvements noted between 2017 and 2020 in relation to pollution. However, litter and garbage remains a relatively large and influential pressure that does not seem to be improving, despite existing regulations. Community engagement in policy and management and the involvement of site support groups is also high relative to other regions and considered to be effective measures.

At a country level, Belgium reported large nature restoration and development projects (with support of European LIFE programs) in some of the major wintering areas for waterbirds along the Belgian coast. The most recent project involved the enlargement of the Zwin nature reserve by another 120 ha of tidal habitats, replacing adjacent agricultural land, with positive effects on the number of waders already observed in this area (Devos *et al.* 2020). Denmark reported a significant proportion of the Danish Natura 2000 network as being of importance for staging, moulting or wintering waterbirds, and designated 90 shooting-free reserves to protect waterbirds from hunting, whilst many sites also restrict other more disturbing recreational activities (e.g. through speed limits for motorboats and zonation of wind- and kite-surfing activities) (Clausen *et al.* 2020).

3.5.3 Iberia – North Africa

International designation was reported for all assessed sites in Iberia – N Africa (100% of sites at least partially designated), while national designations existed for only 60% of the sites. However, there are many more sites of international importance in the region that have not yet been formally designated (Popoff *et al.* 2021). Management and protection of sites in Portugal and Spain are subject to the requirements of the EU Directives, and the Atlantic coast of Europe is generally better covered by protected areas (see also Annex 2), e.g. 0.68% of marine and coastal areas in Morocco are covered, compared with 16.8% in Portugal and 12.8% in Spain (UNEP-WCMC 2021).

Measures are being implemented at some sites to manage the key pressures impacting waterbird sites in the region, e.g. waste management and regulation of tourism and recreation, although for most sites these measures have limited effect (Fig. 3.9).

Kasmi *et al.* (2020) highlighted the urgent need for improved and integrated coastal management to reduce local pressures on the shorelines of Morocco (at least those still undeveloped) and protect them from erosion. It does not seem (from the sample of sites included in these analyses) that protection from erosion is considered a priority.

3.5.4 West Africa

Countries in W Africa have put in place regulations and national and regional policies to manage fisheries, notably the Convention on the Determination of Minimum Conditions for Access to and Exploitation of Fishery Resources within Maritime Areas (SRFC). Member states have created Inshore Exclusion Zones, areas along the coast reserved by law for small-scale fishing, using specific fishing methods and with the aim of conserving fish stocks on which local communities depend. However, poor governance and illegal, undeclared, and unregulated fishing have affected seriously the small-scale fishermen and fishing communities, whilst overfishing also impacts marine biodiversity including fish-eating waterbirds.

Some 60% of sites assessed in W Africa are mostly or partially protected under national law as protected areas or marine protected areas. However, there are sites in Guinea and Guinea-Bissau that are not yet officially protected; further advocacy is needed for their protection. In Senegal, the Grand Niayes of Pikine was designated as an urban community nature reserve in 2019 (see chapter 8). Most of the assessed sites that are protected areas in W Africa have a management plan. Unfortunately most of them are not implemented and are not updated due to a lack of financial and technical resources, with the exception of some sites in Senegal, Mauritania and Sierra Leone whose plans have been updated.

There is no clear law in W African countries on the involvement of communities in policy and management of wetlands, but communities are becoming more and more involved, especially in the conservation of marine protected areas and community nature reserves, and through management committees that support site management on a voluntary basis. Local communities participate in a range of site activities, such as mangrove restoration or shellfish harvesting in marine protected areas, as well as production of salt and surveillance.

Some 61% of assessed sites in W Africa contain mangroves, which cover 11% of the assessed sites by area (Annex 3). While there are several major protected areas designated that protect large areas of mangroves, there is no regulation specific to the protection of mangroves, and provision is done through management plans, but there is a protocol on mangroves which is additional to the Abidjan Convention that all countries have adopted. There have been huge efforts in W Africa to restore mangroves in recent years. These involve the local communities and help to improve their livelihoods while also improving the resilience of sites to climate change.

There are a number of assessed sites in W Africa that are designated as Ramsar Sites, Biosphere Reserves and World Heritage Sites (WHS). Despite the encouragement of Ramsar, AEWA and WHC to identify more sites for protection, few new sites have applied recently for these designations.

3.5.5. Gulf of Guinea

Conservation measures are needed at all sites assessed in the Gulf of Guinea, though some measures were being taken in 86% of sites. The most frequent regulation measures taken were in relation to forest/mangrove cutting and fisheries, as well as regulation of hunting, urbanisation and tourism/recreation. However, the regulations on urbanisation, forest cutting and fisheries appear to be largely ineffective in Benin and Cameroon, with low effectiveness of forest cutting regulations also noted in Gabon and west Ghana. Other frequent measures reported were involving local communities in policy and management, habitat restoration and communication. Most sites assessed were under some form of designation, with 71% designated at the international level and 64% at the national level (Fig. 3.9).

In Liberia, only Lake Piso receives formal protection of the assessed sites. Other Ramsar Sites benefit from signposting but there is essentially no enforcement (Garteh 2020). The coastline of Côte d'Ivoire comprises several coastal lagoons, some of which are protected, such as d'Azagny National Park. Ghana likewise is rich in coastal lagoons, mostly designated under national legislation and/or as Ramsar Sites, and under some form of management. Benin also has coastal lagoons similarly designated, including the extensive Lac Nokoué. There is only limited protection of biodiversity along Nigeria's extensive and in part densely populated or industrialised coastal zone (see also chapter 10).

In Cameroon, the Lower Sanaga River is now partly under formal protection status through the creation of Douala-Edea National Park in 2018. The new park includes a significant marine component. However, it is recommended to designate important coastal sites as Ramsar Sites and afford them greater protection (Ajonina *et al.* 2020). The Wouri Estuary is unprotected, although it contains a number of important wetlands. Along the northern coastline of Gabon, Akanda and Pongara are designated national parks, situated respectively in the Bay of Corsico and the Gabon Estuary near Libreville. The Banio Lagoon in southern Gabon is not protected, though it lies adjacent to Mayumba National Park.

There is one designated national park in the coastal zone of The Congo – Conkouati-Douli, and one Ramsar Site - the Bas Kouilou-Yombo (or Lower Kouilou Basin). Parc Marin des Mangroves is a designated protected area at the coast of the Democratic Republic of Congo, though its protection measures are not effective.

3.5.6 Southern Africa

Conservation measures are underway in 10 out of 11 sites assessed in Angola, Namibia and South Africa, although further measures still need to be taken. The majority of sites assessed have measures in place to carry out research needed for conservation and to regulate fisheries and hunting, and have national and/or international designations in place. Management plans were present in five sites. Several regulatory measures were only recorded in half or less of the sites, including regulations for fossil fuel exploitation, urbanisation, control of invasive species, agricultural land use and wind farms, whilst less than 40% of sites have tourism/recreation zonation in place (Fig. 3.9).

Of the four sites assessed in Angola, the three sites in Mussulo Lagoon have some measure of protection, although the effectiveness of protection measures is limited. Undoubtedly there is high pressure on these sites due to the proximity to Luanda, although there should be scope for improved management and protection, noting the proximity of offices of governmental environmental institutions and NGOs. However, governmental action has been limited, and NGOs and volunteers have led most conservation measures. The only site in the region without any formal protection is Mangais de Nzeto. The recent loss of mangroves at this site has raised local and national concern, and some measures have been taken to identify the causes, noting that mangrove habitats are protected by law.

In Namibia, Walvis Bay and Sandwich Harbour are both Ramsar Sites and fall partly within national parks, whilst Cape Cross Lagoon is a private Nature Reserve. The coastal sites in central Namibia fall within the National West Coast Tourist Recreation Area, which has been the focus of an Integrated Coastal Zone Management Project. This has promoted some controlled tourism, noting that some activities may be damaging to nature, such as driving quad bikes through sensitive areas. Sandwich Harbour is part of the extensive Namib Naukluft Park, requiring a permit to visit.

Of the assessed sites in South Africa, Langebaan Lagoon is a Marine Protected Area and Nature Reserve and a part of the West Coast National Park. Both it and Verlorenvlei are Ramsar Sites, although Verlorenvlei receives no national protection. The Olifants River Estuary is also unprotected, though it receives some local form of management. The Lower Berg River also has no legislative protection, although it falls partly within a Biosphere Reserve. Some measures have been taken to regulate urbanisation, although this is only partially effective in most of the sites assessed. Overall, improved conservation status and management are needed within this coastal belt of South Africa, which supports high numbers of waterbirds and seabirds.

3.6 Discussion and recommendations

In this chapter, we have demonstrated a broad range of pressures impacting waterbirds and wetlands across the EAF. Given the short time interval, it is not surprising that many of the pressures identified in 2017 remained high on the list in 2020. Efforts to alleviate the impacts of fishing, recreation and tourism, shipping, litter, agriculture and others have had mixed outcomes, many serving to partially address the issue.

Other factors that are much less controllable are becoming increasingly dominant, such as the effects of climate change, which is affecting the spatial and temporal availability of coastal wetland habitats, and movement patterns of many migratory species. For some regions (e.g. Northern Europe), milder winters are serving to increase the availability of wetlands for waterbirds, resulting in increasing numbers wintering there and consequent decreases in bird numbers further south (*Pavón-Jordán et al.* 2019, see chapter 2). Breiner *et al.* (2021) measured the geographical variation in projected changes in waterbird distributions across the Africa–Eurasian waterbird flyways by 2050, inferring that sites across sub-saharan Africa, N Africa and the Middle East will suffer greatest deterioration in suitability, while the greatest improvement in suitability will be at sites in Eastern Europe. Future challenges for countries in Northern and Eastern Europe may lie in making sure these 'new' sites are identified and their importance justified, so that they are adequately protected and managed for future use by migratory waterbirds. This includes their consideration in relation to other potentially increasing pressures such as offshore renewable energy development. Meanwhile, adequate resourcing, governance mechanisms and institutional capacity in the most affected regions such as in Africa will be essential for managing climate adaptation.

Given the extent of change in waterbird populations and their movements in recent decades, in the quality of their sites and habitats and in the pressures that these sites and species face, it is important to consider and manage these sites at a network or flyway scale. In doing so this maximises the possibility for a robust network of sites to be protected and adequately managed while allowing for ongoing and projected change. Sites should continue to be regularly monitored, in order to track ongoing changes in waterbirds and just as importantly the pressures that are affecting them, so that timely actions can be implemented as necessary.

The assessments of the environmental pressures and conservation measures have shown that many countries are considering management needs for waterbirds, implementing a range of mechanisms including site protection, appropriate regulations, waste management etc., as well as a range of targeted actions towards species and habitat management. Given the limited effectiveness of actions to date, it may be timely to consider unpacking the impact, for example of the many types of regulations that have been identified, and developing and promoting case studies that demonstrate specific actions that have worked well, and in what conditions.

There already exist networks of people who take an interest in their local wetland sites, including through the network of IWC Coordinators and their counters. Some countries have successfully developed networks of Species Support Groups. Ongoing engagement with local communities is hugely valuable to the management and conservation of wetland habitats, and future initiatives and efforts to increase awareness-raising on the value of protecting ecosystem service of coastal wetlands should be encouraged and supported.

Arnold Meijer/ Blue Robin

Eurasian Spoonbills, and farmer collecting water edge vegetation in the back ground

Bonte Strandloper | **Bécasseau variable** *(Calidris alpina)*
(Marcus Varesvuo / AGAMI)

4. Potential impacts of climate warming and changing predator-prey dynamics on breeding shorebird populations of the western Russian Arctic

Mikhael Soloviev, Evgeny Syroechkovskiy†, Aleksander E. Dmitriev, Victor V. Golovnyuk, Vladimir V. Morozov, Pavel Tomkovich & Elena Lappo

† During the editing phase of this chapter, we lost Evgeny Syroechkovskiy, great friend, husband, and leader in research and conservation of birds in Russia and the circumpolar Arctic. Among his many achievements were leading contributions to the Arctic Migratory Bird Initiative under the Arctic Council's working group on the Conservation of Arctic Flora and Fauna, the Atlas of breeding waders in the Russian Arctic, and safeguarding the globally threatened Spoon-billed Sandpiper *Calidris pygmaea*. Evgeny will be sorely missed.

Summary
This chapter summarises results of research in the Russian arctic breeding grounds of waterbirds migrating in the African-Eurasian flyway systems in the past decades. It focuses on a selection of arctic-breeding shorebird species studied during surveys on breeding grounds in the western Russian Arctic, especially important to populations using the East Atlantic Flyway. In this region, a pronounced warming of the climate during the breeding season is ongoing, which has led to observable changes the distribution, abundance, phenology of reproduction, and sometimes even the morphology of shorebirds. Significant changes in the pressure of predators and the abundance of their alternative prey have not been observed. While certain other impacts of climate warming on shorebirds are to be expected (e.g. effects on reproduction mediated by availability of arthropod food, habitat change related to permafrost retreat), these are difficult to evaluate without the collection of additional data on the breeding grounds. However, research activity in the Russian Arctic has declined rather than intensified due to various reasons. We urge to restore and expand intensive shorebird monitoring at key sites of the tundra area of the East Atlantic Flyway.

Résumé
Ce chapitre résume les résultats des recherches menées dans les zones de reproduction de l'Arctique Russe des oiseaux d'eau empruntant les voies de migration d'Afrique-Eurasie au cours des dernières décennies. Il se concentre sur une sélection d'espèces d'oiseaux de rivage se reproduisant dans l'Arctique, étudiées lors d'enquêtes sur les sites de reproduction de l'Arctique Russe occidental, particulièrement importants pour les populations empruntant la voie de migration de l'Atlantique Est. Dans cette région, un réchauffement prononcé du climat pendant la saison de reproduction est en vigueur, ce qui a entraîné des changements visibles dans la distribution, l'abondance, la phénologie de la reproduction et parfois même la morphologie des oiseaux de rivage. Des changements significatifs de la pression des prédateurs et de l'abondance de leurs proies alternatives n'ont pas été observés. Bien que certains autres impacts du réchauffement climatique sur les oiseaux de rivage soient à prévoir (par exemple, les effets sur la reproduction liés à la disponibilité de la nourriture composée d'arthropodes, les changements d'habitat liés au retrait du permafrost), ils sont difficiles à évaluer sans la collecte de données supplémentaires sur les sites de reproduction. Cependant, l'activité de recherche dans l'Arctique Russe a diminué plutôt que de s'intensifier pour diverses raisons. Nous demandons avec instance de restaurer et d'étendre le suivi intensif des oiseaux de rivage sur les sites clés de la zone de toundra de la voie de migration de l'Atlantique Est.

4.1 Introduction
The wintering ranges of some Arctic bird species' populations overlap, which reduces the reliability of population estimates and trend assessments based on winter counts. Moreover, flyway delineations of many biogeographic populations of Arctic migratory birds are still largely unknown. Since population estimates for waterbirds are mainly based on counts on the wintering grounds, these are likely to be inaccurate where there is insufficient data to support flyway delineations. This is the case for at least some populations of arctic-breeding shorebirds.

Data from the arctic breeding grounds on breeding densities and reproductive success can supplement wintering data and help to understand trends and changes in distribution in the flyway. Monitoring in the Arctic can also provide valuable insights into other threats and cumulative

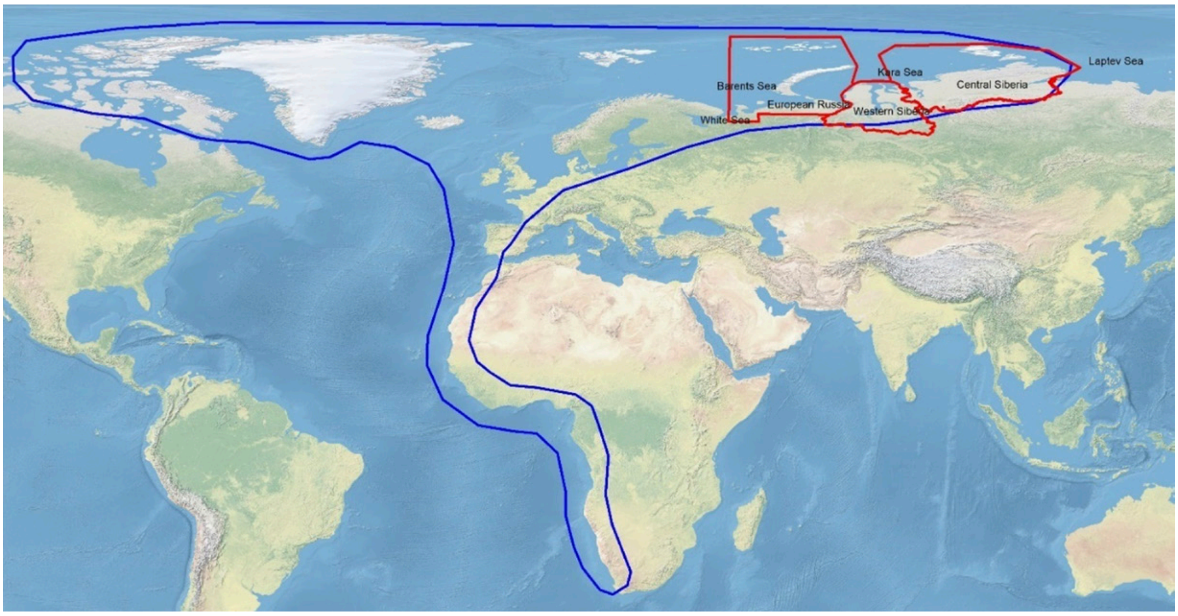

Figure 4.1. The East Atlantic Flyway (EAF) (blue line) and its principal arctic breeding regions in the Western Russian Arctic (red lines).

impacts specific to breeding areas, including harvest, pollution, climate change, and habitat degradation. In the African-Eurasian flyways system (AEF) including the East Atlantic Flyway (EAF), breeding grounds of arctic-breeding shorebirds are often poorly known compared to the wintering grounds and stopover sites. Information currently available in English and even in Russian is scarce, and systematic data synthesis has not been updated since the publication of the Atlas of the Breeding Waders in the Russian Arctic (Lappo *et al.* 2012), which includes data up to 2008.

This chapter is a follow up of the summary report of the research on shorebirds in the Russian breeding grounds of the AEF conducted since 2008 (Soloviev *et al.* 2021). That summary report is an Arctic Migratory Birds Initiative (AMBI) CAFF report, developed with support of the government of the Netherlands as Arctic council observer contribution to the work of CAFF. The current chapter summaries knowledge of a selection of arctic-breeding shorebird species gathered during surveys on breeding-grounds in the Western Russian Arctic, especially important to populations using the East Atlantic Flyway (Fig. 4.1).

4.2 Methods of monitoring breeding shorebirds in the Russian Arctic

The method of choice to monitor shorebirds on their arctic breeding grounds depends on available resources and biological characteristics of surveyed species. Monogamous territorial species can be monitored using territory mapping, which requires relatively little labour and thus resources. This method is being used in Greenland for long-term monitoring at the Zackenberg research station (Schmidt *et al.* 2019). Non-monogamous species and species with poorly developed territorial behaviour can be surveyed using nest searches on plots, which is labour-intensive and becomes effective only at moderate and high nesting densities. This approach was documented in detail in field protocols of the Arctic Shorebird Demographics Network (ASDN, Brown *et al.* 2014). Nest searches on plots has a long-term history of use in Arctic Russia, particularly in Yamal, Taimyr and Chukotka.

In Arctic Russia, the most widely used method of abundance assessments are bird counts using line transects (Hayne 1949). However, this method provides results which are difficult to interpret for most shorebird species as they are often non-monogamous and can be very cryptic during incubation. The studies that have used this method are often not taken into account in the current assessment. Experts of the terrestrial monitoring group of the Circumpolar Biodiversity Monitoring Programme (CBMP) have recognized challenges associated with regional aspects of shorebird monitoring in the Arctic. Hence, the importance of harmonization rather than standardization of survey methods was emphasized under the CBMP. In some regions with a low abundance of shorebirds (e.g. Alaska), point counts were also used to count birds, but we are not aware of such attempts in the Russian Arctic.

A list of sites in the Russian Arctic where long-term monitoring of shorebirds has been conducted is provided in Table 4.1, with comments on features of different sites. The following regional conclusions can be drawn based on this overview.

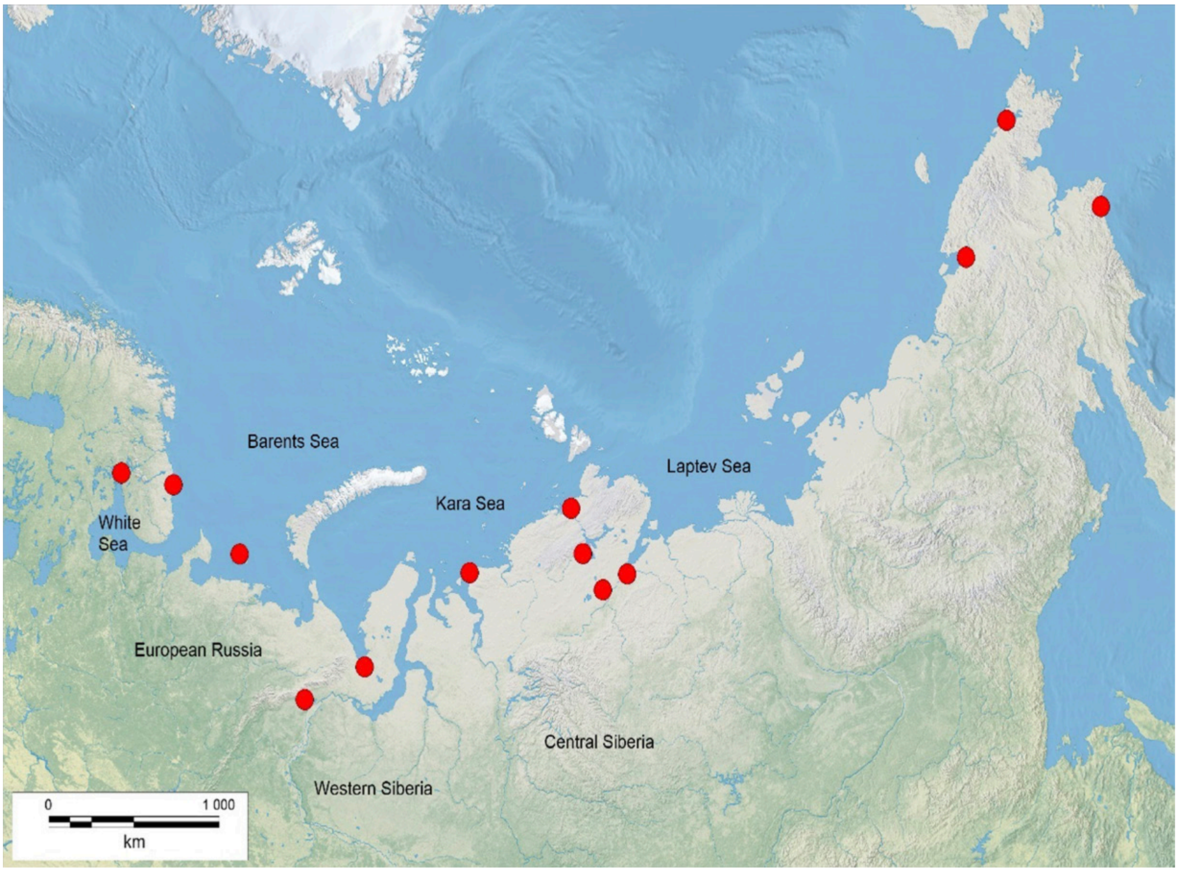

Figure 4.2. Sites of intensive shorebird monitoring in the Russian Artic.

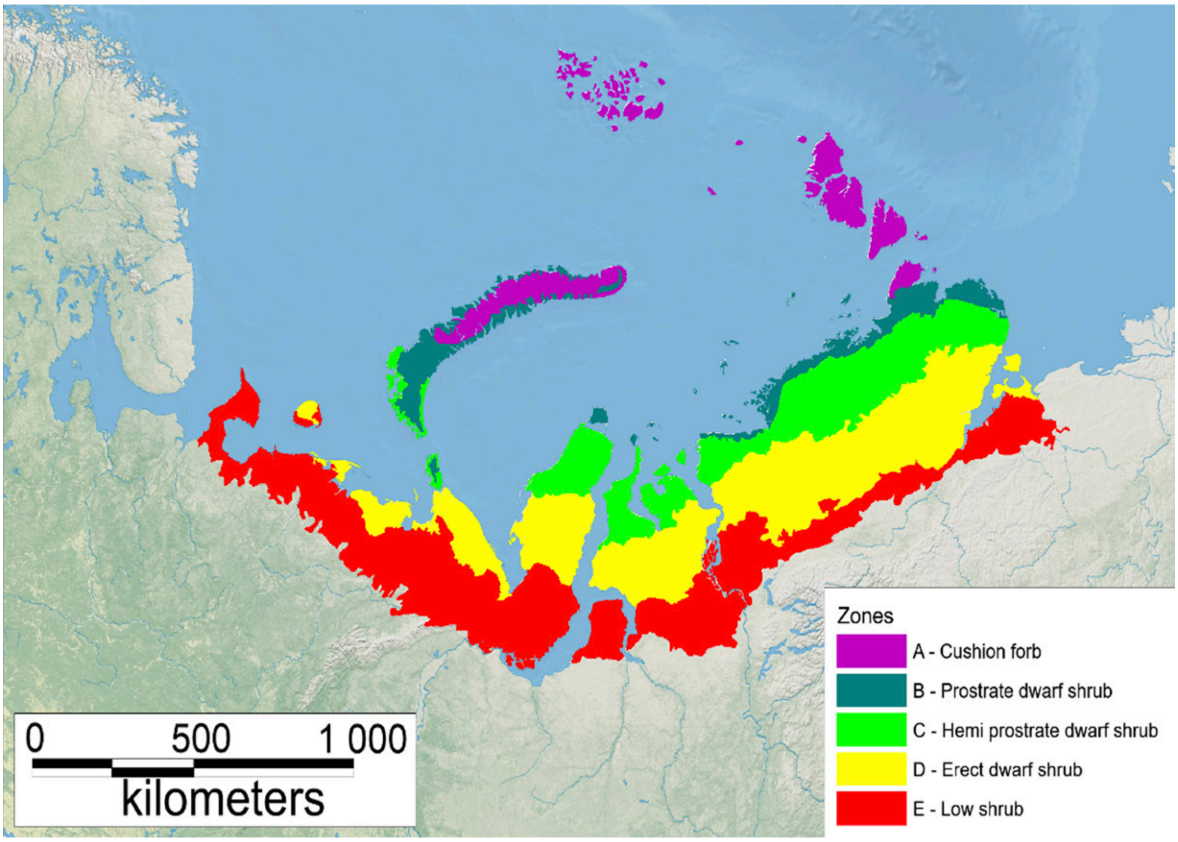

Figure 4.3. Bioclimatic zones of the Western Russian Arctic (based on CAVM team 2003).

European Russia: Tundra habitats are almost absent in the westernmost part of the Russian Arctic, on the Barents and White Sea coasts. Shorebird monitoring conducted there in reserves is mostly restricted to coastal species, such as Eurasian Oystercatcher *Haematopus ostralegus* or migrating shorebirds. Kolguev Island is a very important tundra site further to the east, although there is some inconsistency of shorebird monitoring at that location.

West-Siberian Arctic: the lower reaches of the Erkatayakha and Payutayakha Rivers on the Yamal Peninsula is the most important long-term shorebird monitoring site in Western Siberia. This site has been monitored for decades making use of nest searches on plots. Prospects for monitoring at other points on Yamal, where research has been carried out in recent years (Sabetta, Bely Island), are currently unclear.

Central-Siberian Arctic: two shorebird monitoring projects are running on the Taimyr Peninsula, one of which (conducted by the Willem Barentsz Biological Station) has been mostly involved in territory mapping, while another project (at different sites across the peninsula) focuses on nest searches on plots.

Eastern Russian Arctic: Intensive shorebird monitoring in Yakutia has not been conducted for several decades. Shorebird monitoring sites in western Chukotka were established in the framework of the ASDN and employed nest searches on plots for some limited time. Belyaka Spit in northern Chukotka is an important site of long-term monitoring of the Spoon-billed Sandpiper *Calidris pygmaea*, a critically endangered species. But this monitoring is no longer continued. Vicinities of Meinypylgyno village are an important hub of monitoring and research activities on shorebirds, including nest searches on plots, transect counts and variety of ecological studies.

Shorebird monitoring has mostly been conducted using nest searches on plots in Yamal, Taimyr and Chukotka. Consistent intensive monitoring of shorebirds is generally missing in the westernmost Russian Arctic and Yakutia. Table 4.1 shows a list of locations in the Russian Arctic where the number of breeding shorebirds was estimated for a certain period of time. The points where the abundance was estimated by counts on transects are, despite their imperfection, given in some cases. The locations of intensive monitoring points across the territory of the Russian Arctic are shown in Fig. 4.2.

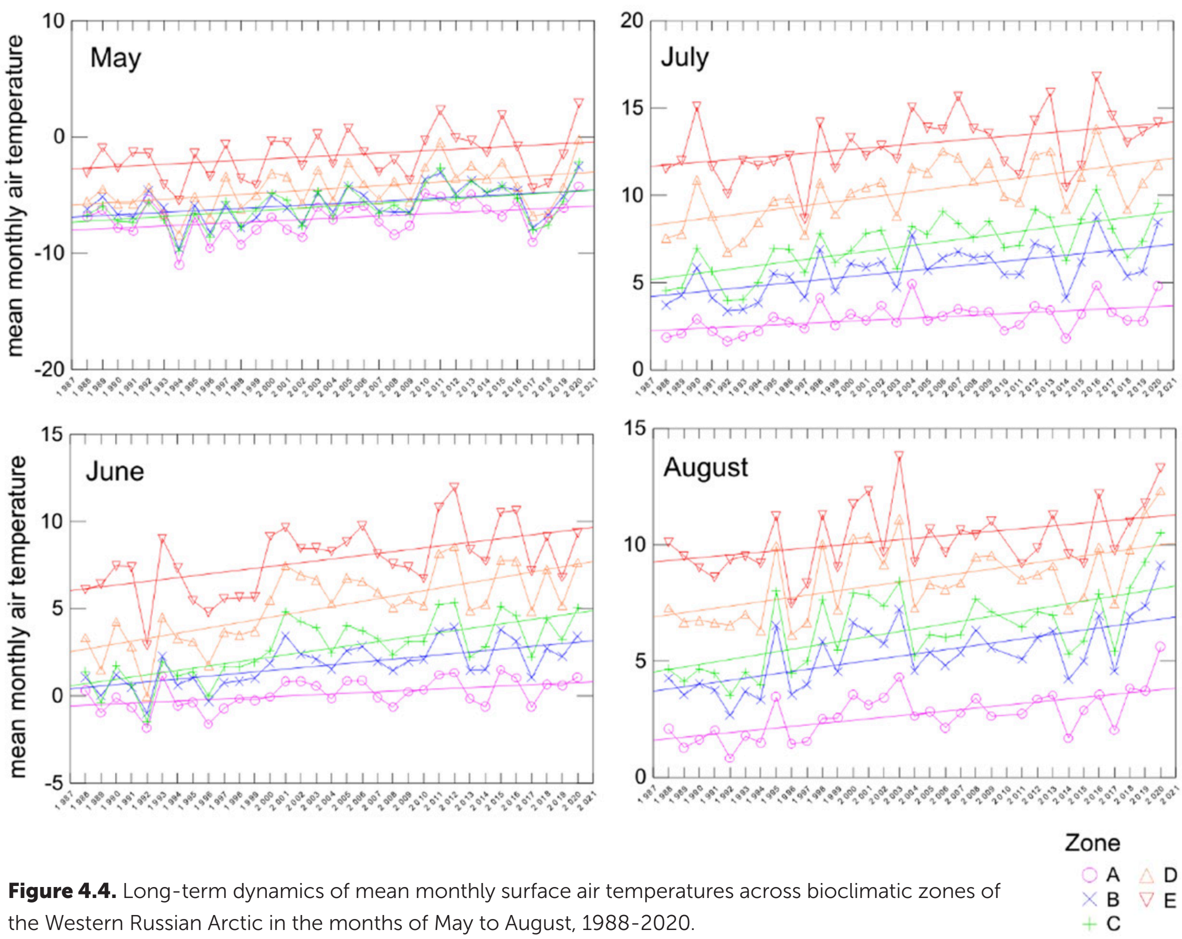

Figure 4.4. Long-term dynamics of mean monthly surface air temperatures across bioclimatic zones of the Western Russian Arctic in the months of May to August, 1988-2020.

4.3 Potential impacts of climate warming and prey-predator interactions on Arctic-breeding shorebird populations

Trends in temperatures during the breeding season of arctic shorebirds were assessed for the arctic portion of the AEF in the five bioclimatic zones of the Russian Arctic, as defined by the Circumpolar Arctic Vegetation Map (CAVM team 2003; Fig. 4.3). Daily average air temperature data from May to August over the years 1988–2020 from all available weather stations to the north of 50°N (Global Surface Summary of Day project at the National Climatic Data Centre, ftp://ftp.ncdc.noaa.gov/pub/data/gsod) were used to create daily surface air temperature maps. These maps were overlaid with portions of the bioclimatic zones, and temperature values averaged resulting in a daily temperature value for each zone. Daily values were subsequently averaged to obtain monthly means for May, June, July and

Markus Varesvuo / Agami

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

Name	Description	Latitude	Longitude	Monitoring years *
Northern arch., Kandalaksha Bay, White Sea, Kandalaksha Nature Reserve	Long-term surveys of seabirds including coastal wader species	67°01′N	32°36′E	2003-2020
Arch. Sem Ostrovov, Barents Sea, Kandalaksha Nature Reserve	Long-term surveys of seabirds including coastal wader species	68°48′N	37°20′E	2004-2019
Kolguev Island	Unique ecosystems due to the absence of rodents on the island	69°10′N	48°54′E	2006-2008, 2011-2019
Voikar River, lower Ob′ River region	Subarctic point of long-term counts of birds and mammals	65°48′N	63°57′E	2000-2011
Rivers Erkutayakha and Payutayakha, Yamal Peninsula	Point of long-term research of birds and mammals	68°13′N.	69°09′E	2002-2010, 2016-2020
Meduza Bay, Willem Barents station, north-western Taimyr, Great Arctic Reserve	Point of long-term monitoring of shorebirds by the method of territory mapping, and on plots in 2015-2017	73°21′N	80°32′E	1993-1994, 1996-2007, 2012, 2014-2020
Environs of Knipovich Bay, northern Taimyr, Great Arctic Reserve, Nizhnyaya Taimyra area	Shorebird surveys on plots repeated after an interval of two decades	76°05 ′N	98°32′E	1990-1992, 2018-2019
The mouth of the river Upper Taimyra, central Taimyr, Taimyr reserve	Taimyr reserve, main area, shorebird surveys on plots	74°08 ′N	99°34′E	2004-2007
Sparse larch forest patch "Ary-Mas", southeastern Taimyr, Taimyr reserve	Shorebirds were counted on linear transects; nevertheless, these data span several decades and can be used to assess long-term trends	72°29′N	101°50′E	1993, 1997, 1999, 2002, 2004, 2005, 2009-2011
The mouth of the Bludnaya river lower Khatanga area, southeastern Taimyr	Intensive monitoring of shorebirds was carried out for 17 years	72°51′N	106°02′E	1994-2003, 2008-2014
Delta of the Chaun-Palyavaam rivers, western Chukotka	Station of long-term monitoring of birds and mammals, shorebirds were monitored according to the ASDN protocol	68°50′N	170°30′E	2001-2002, 2004-2005, 2007-2020
Meinypylgyno settlement area, Koryak Upland, Chukotka	Station for long-term monitoring of birds and mammals. Key monitoring point for the endangered spoon-billed sandpiper	62°32 ′N	177°04′E	2003-2005, 2007-2020
Spit Belyaka and Yuzhny Island, Chukotka, National Park "Beringia"	Point of long-term monitoring of birds and mammals. Key monitoring point for the endangered spoon-billed sandpiper	67°03′N	174°36′W	1986-1988, 1994, 1996-1997, 2002, 2005, 2007, 2009-2019

* according to the Arctic Birds Breeding Conditions Survey (http://www.arcticbirds.net/)

Table 4.1. Sites of intensive shorebird monitoring in the Russian Arctic. Intensive monitoring is the one allowing to obtain reliable data about trends in shorebird abundance and/or breeding success.

August. The Arctic portion of the flyway was restricted to the tundra zone extending from the Kanin Peninsula in the west to the Anabar River in the east (Fig. 3).

In 1988-2020 significant warming occurred across all five bioclimatic zones during the breeding season (all months May-August $P < 0.05$). In June, when Arctic shorebirds undertake the most important decisions regarding reproduction, warming was faster in zones C and D compared with zones A, B and E (Fig. 4.4). It therefore may be expected that high-arctic species (such as Purple Sandpiper, Red Knot and Sanderling) and southern tundra species (such as Bar-tailed Godwit and Spotted Redshank) will be less affected by this warming compared to species typical for the central belt of tundra, such as Dunlin, Red Phalarope and Pacific Golden Plover. However, currently clear evidence of adverse effects of warming on species in zones C and D is missing.

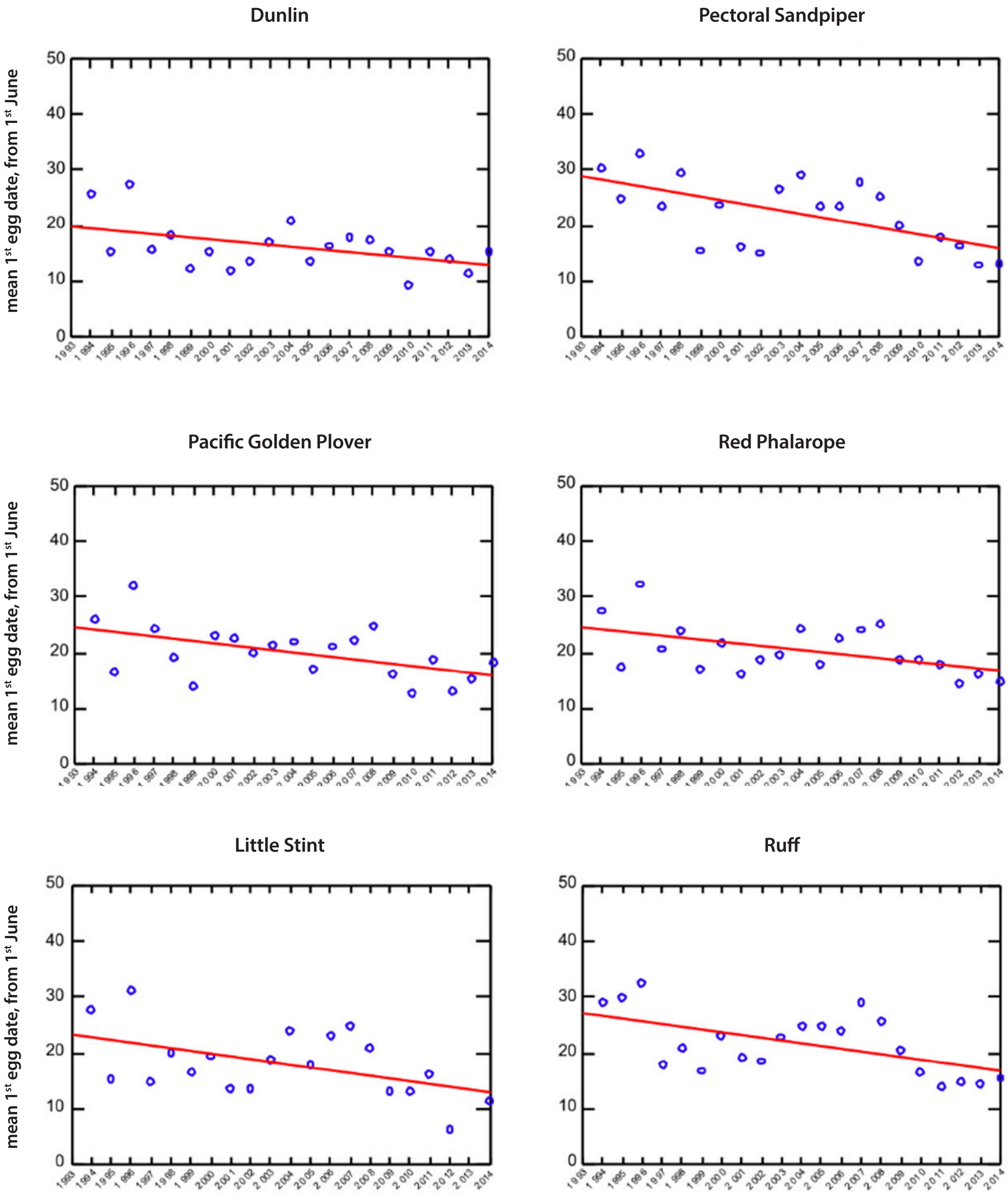

Figure 4.5. Changes in breeding phenology of birds in southeastern and central Taimyr.

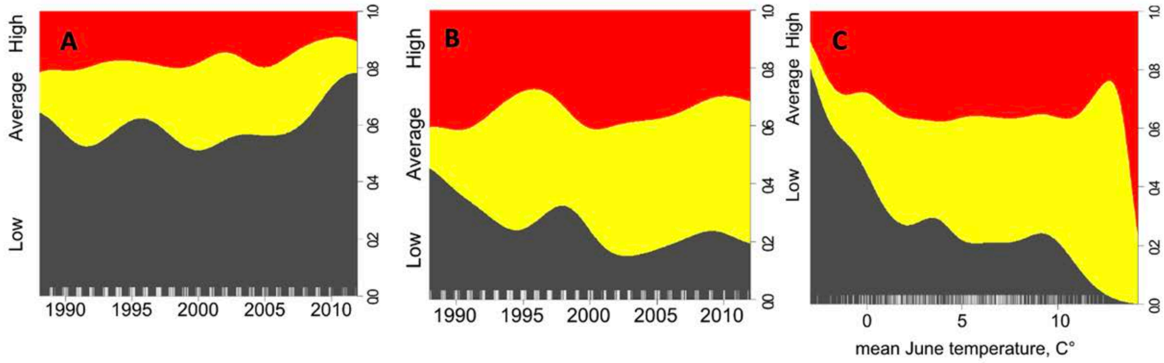

Figure 4.6. Changes between 1988 and 2012 in (A) rodent abundance and (B) shorebird nest success in the Russian Arctic over time and of nesting success with varying mean temperatures in June (C). Proportions of sites per year with high (red), average (yellow) and low (black) rodent abundance or nest success are shown; the distributions of data points are shown on the horizontal axes (from Soloviev & Tomkovich 2014).

In all zones, the warming trend was least pronounced in May. In May, the average monthly air temperature crossed zero, i.e. most of the territory at this time was covered with snow. This apparently prevented such a rapid heating of the air compared with the summer months. Among the zones, the warming was least expressed in zone A – cushion forb or the polar deserts. Perhaps this is due to the stabilizing effect of the ocean that surrounds the Arctic islands with polar deserts.

4.4 Changes in breeding phenology and abundance of shorebirds

The timing of breeding is one of the most obviously changing parameters for Arctic birds in relation to climate warming. Breeding dates of shorebirds have advanced during the past 17 years in southeastern and central Taimyr (Fig. 4.5), and shorebirds increased their use of floodplain habitats instead of river terrace habitats. Although some bird nesting dates changed with weather conditions at nesting sites in a close to linear pattern, there is evidence that warming in the Arctic as a result of 'arctic amplification' may render it impossible to adequately adjust the timing of nesting to changing conditions. This is especially likely for long-distance migrants who, on their southern wintering grounds, must cue the start of migration solely on the basis of astronomical criteria. For Bar-tailed Godwits (*Limosa lapponica*) however, it was shown that in some years when food conditions at staging sites are good, they are able to compensate for the impact of climate warming on arctic spring phenology (Rakhimberdiev *et al.* 2018). This confirms the importance of protecting the stopover sites of migratory birds along with nesting sites in the Arctic and wintering sites.

Long-term studies on the Russian breeding grounds revealed a variety of impacts of climate warming on abundance of shorebirds. Numbers of relatively southern species (Dunlin, Pectoral Sandpiper, Red Phalarope, Pacific Golden Plover) increased in northern Taimyr from 1990-1992 to 2018-2019, while numbers of High Arctic species (Red Knot, Sanderling, Turnstone) decreased (Golovnyuk *et al.* 2019).

4.5 Dynamics in small mammal abundance, breeding success and survival

Microtine rodents in the Arctic include lemmings and voles, which are characterized by periodic fluctuations in the abundance of local populations by orders of magnitude. In years of high abundance, arctic microtine rodents form the food base for birds of prey and mammals. The most recent analysis (Ehrich *et al.* 2019) did not find a general decrease in the number of lemmings in Russia, with the exception of a few southern points where lemmings live together with voles. In some areas of the Russian Arctic, voles were observed spreading to the north, which is probably associated with climate warming (Golovnyuk *et al.* 2017).

Data on rodent abundance and bird breeding success have been collated since 1988 in the framework of the Arctic Birds Breeding Conditions Survey (Soloviev & Tomkovich 2021). Most of these data were converted to rank scale (low/average/high for abundance/success) for consistency and analyzed using ordinal regression (Soloviev & Tomkovich 2014). In the Russian Arctic there was no significant change in the abundance of rodents in the period 1988-2012 (Fig. 4.6a), which is consistent with later circumpolar data (Ehrich *et al.* 2019). In the same period the proportion of sites with average nest success increased and the number of sites with low nest success decreased (Fig. 4.6b), which resulted in significant overall increase in nest success. Nest success increased both with an increase in the abundance of rodents and with an increase in mean June temperatures (Fig. 4.7c). Given that mean June air

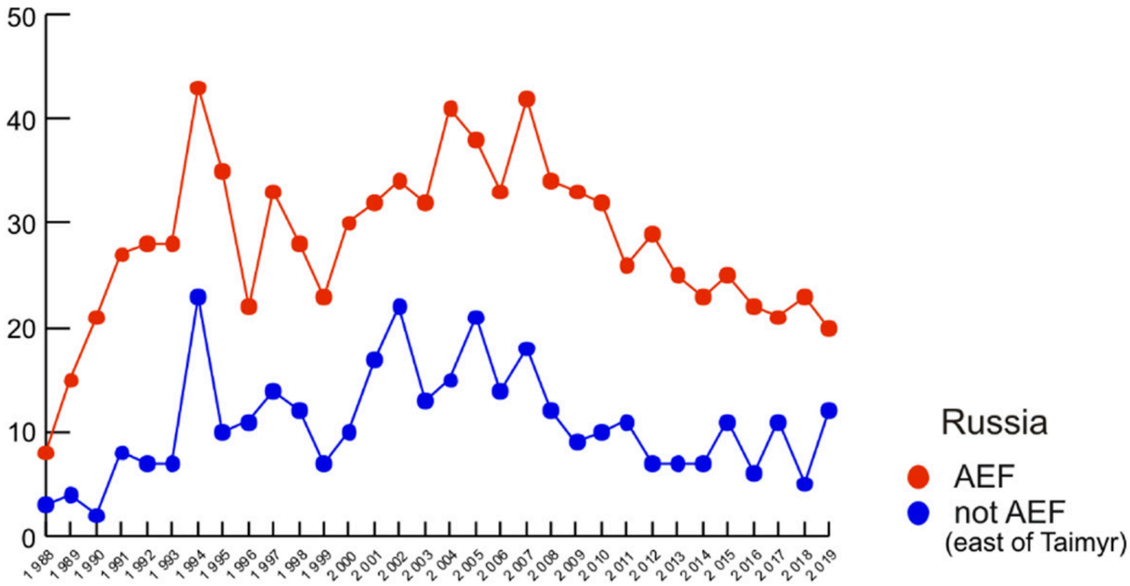

Figure 4.7. Numbers of active bird study sites in the Russian Arctic, 1988-2019, based on data from the Arctic Bird Breeding Conditions Survey (from Soloviev & Tomkovich 2021).

temperatures increased in 1988-2012 and the abundance of rodents did not change significantly, we suggest that the long-term trend in nest success of shorebirds is explained by increasing temperatures during incubation. Hence, according to the study by Soloviev & Tomkovich (2014), shorebirds in the Russian Arctic seemed to benefit from climate warming on the breeding grounds during the nesting phase.

However, climate warming may affect prospects for chicks negatively due to mismatch with an advancing peak in availability of insect food (e.g. Lameris *et al.* 2021), or increases in summer droughts. Survival of juvenile Red Knots on their African wintering grounds was lower after breeding seasons with earlier snowmelt (van Gils *et al.* 2016). Juveniles with lower survival had also shorter bills. The relationship between bill length and timing of snowmelt are currently not completely understood however.

4.6 Changes in breeding range of shorebirds in northern European Russia, 2008-2020

An analysis of published and unpublished data from 74 shorebird observation sites revealed that the most intensive field work was carried out in Kolokolkova Bay (Malozemelskaya Tundra), Kolguev Island, eastern Bolshezemelskaya Tundra (eight sites), the Polar Urals (four sites) and Franz Josef Land (10 sites). In the same period, one or two seasons' surveys were conducted on other tundra and forest-tundra territories of European Russia (Soloviev *et al.* 2021). This work has indicated changes in the breeding distribution of several shorebird species. Some boreal species are shifting their breeding range northwards. The Common Greenshank *Tringa nebularia* has started breeding at the northern edge of the forest-tundra zone, and Common Sandpiper *Actitis hypoleucos* and Terek Sandpiper *Xenus cinereus* at the northern limit of the southern tundra subzone (zone E in Fig. 4.3). Jack Snipe *Lymnocryptes minimus* have bred on Kolguev Island for several years and have also reached the Barents and Kara Seas' coasts in the Malozemelskaya and Bolshezemelskaya Tundra. At the same time, some shorebird species characteristic of the typical tundra subzone (zones C and D), such as Little Stint *Calidris minuta*, have also shifted northwards, from the north of the southern tundra subzone (zone E) to breed only at the sea coast in


Jeroen Reneerkens

Sanderling | **Bécasseau sanderling** *(Calidris alba)*

Eurasian Dotterel | **Pluvier guignard** *(Charadrius morinellus)*

the typical tundra subzone (zones C and D in Fig. 4.3).

Preliminary analysis identified that the distribution range has contracted and numbers have declined for Eurasian Dotterel *Eudromias morinellus* and Great Snipe *Gallinago media*. Declining numbers were found in Ruff *Philomachus pugnax* and Red-necked Phalarope *Phalaropus lobatus* in the forest-tundra and southern tundra subzones. These two species currently do not breed any more in forest-tundra in warm seasons with an early spring, and in such years numbers in the southern tundra subzone (E in Fig. 4.3) are now very low whereas these species are still common in the typical tundra subzone. An evaluation of the changes in distribution of other shorebird species breeding in the western Russian Arctic, as well as completion of the geographical coverage (adding the Kola peninsula and the White Sea area) are needed and are planned for the second stage of the AMBI project of CAFF in 2022.

4.7 Conclusions and recommendations

In the western Russian Arctic during the breeding season of shorebirds, a pronounced warming of the climate is ongoing, which changes the distribution, abundance, phenology of reproduction, and sometimes even the morphology of shorebirds. Significant changes in the pressure of predators and the abundance of their alternative prey has not been observed.

While certain impacts of climate warming on shorebirds are to be expected (e.g. shifts in distribution, breeding phenology, etc.), other effects are difficult to evaluate without the collection of additional data on the breeding grounds. Some impacts are hypothesized to emerge from variations in biomass or phenology of invertebrate prey for shorebirds and their chicks, while this parameter has rarely been assessed in the Russian Arctic in recent years. Accordingly, programs of invertebrate sampling should be initiated at monitoring sites, in collaboration with researchers of Arctic invertebrates. For a better understanding climate change effects, arctic ornithologists could also benefit from the professional expertise of climatologists and landscape ecologists (e.g. with respect to permafrost change) and from the local knowledge of indigenous peoples in the Arctic.

If we aim to better understand the mechanisms of climate change impacts on shorebirds, we need to collect more data associated with this topic. This would require an increase of the total research effort on birds in the Russian Arctic, but this is not yet the case. An assessment of declining intensity of bird research activities in the Russian Arctic is illustrated in Fig. 4.9. There are over 20 active sites in the arctic part of the African-Eurasian flyway system, even in the last years. However, at only a few of these intensive shorebird monitoring programs are executed, and in most years the number of sites running such programs is below 5 in the entire western Russian Arctic, and similary low in the eastern Russian Arctic. This is insufficient to evaluate breeding conditions and nest success of shorebird species with distributions spreading across several regions of the Arctic (e.g. Little Stint and Grey Plover). The decline in the number of active bird study sites since 2008 makes it challenging to assess parameters of breeding conditions for shorebirds, such as rodent abundance. COVID-19 restrictions in 2020-2021 resulted in the cancellation of research, and intensive shorebird monitoring was discontinued at some sites in the western Russian Arctic.

There are various reasons for this declining trend in Arctic research on birds. Financial resources for Arctic bird research are declining, related to increasing costs of research activities, overall economy declines and marginalisation of an environmental agenda. There is also a lack of interest by decision makers in wildlife conservation in general and birds in particular, and a decrease in the overall number of professional ornithologists in Russia. This general trend is worrisome, as the Arctic is warming much faster than the rest of the world and the effects on migratory shorebirds need to be monitored. All possible efforts need to be made to restore intensive shorebird monitoring at key sites of the tundra AEF area (Yamal and Taimyr peninsula), Kolguev Island and some other locations. In the long-term future these sites may establish a network, which will aim to address some currently challenging issues of shorebirds ecology, like juvenile survival on the breeding grounds.

For this, a variety of collaborative projects and alliances in research, conservation and fundraising should be built within Russia and internationally. The potential of involvement of regional governments (particularly Yamal and may be some others) should be explored. BirdsRussia, a Russian bird conservation NGO which has good links within Russian research, conservation communities, as well as connections within governmental structures including Arctic regions, may play a pivotal role in developing this concept further.

Acknowledgements

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Hans Schekkerman

Little Stint | **Bécasseau minute** *(Calidris minuta)*

Eurasian Oystercatcher | **Huîtrier pie** *(Haematopus ostralegus)* (Arnold Meijer / Blue Robin)

5. Trends of waterbird populations in the Wadden Sea in comparison with flyway trends

Romke Kleefstra, Thomas Bregnballe , John Frikke, Benjamin Gnep, Klaus Günther, Bernd Hälterlein, Morten Bentzon Hansen, Menno Hornman, Kees Koffijberg, Julia Meyer, Gundolf Reichert, Gregor Scheiffarth & Jens Umland

Summary

The Wadden Sea is of considerable importance for many of the waterbird populations occurring in the East Atlantic Flyway. The area is an indispensable stopover and wintering site and additionally harbours relevant populations of coastal breeding birds and moulting concentrations of specific species. The latest results of the breeding bird monitoring in the Wadden Sea show that many birds breeding in the Wadden Sea experience negative trends, some even more so than on the flyway level. This indicates that local factors – to a large extent – drive these Wadden Sea declines. Demographic data suggests that several species fail in producing enough offspring to maintain a stable population, due to (increased) predation and flooding which reduce nest success and/or chick survival. Migratory birds staging in the Wadden Sea in the non-breeding season seem to do better than was observed in earlier studies. The majority of species currently show trends that are more favourable in the Wadden Sea than on the flyway-level, an improvement in comparison with 10 years ago. Only three species still do less well in the Wadden Sea than in the flyway as a whole: Great Cormorant, Pied Avocet and Redshank.

Resumé

La mer des Wadden a une importance considérable pour de nombreuses populations d'oiseaux d'eau présentes sur la voie de migration de l'Atlantique Est. Cette zone est un site d'escale et d'hivernage indispensable et abrite en outre des populations importantes d'oiseaux nicheurs côtiers et des concentrations d'espèces spécifiques en période de mue. Les derniers résultats du suivi des oiseaux nicheurs dans la mer des Wadden montrent que de nombreux oiseaux qui s'y reproduisent connaissent des tendances négatives, dont certaines sont encore plus marquées qu'au niveau de la voie de migration. Cela indique que des facteurs locaux sont - dans une large mesure - à l'origine de ces déclins dans la mer des Wadden. Les données démographiques suggèrent que plusieurs espèces ne parviennent pas à produire suffisamment de descendants pour maintenir une population stable, en raison de la prédation (accrue) et des inondations qui réduisent le succès des nids et/ou la survie des poussins. Les oiseaux migrateurs qui font étape dans la mer des Wadden en dehors de la saison de reproduction semblent mieux se porter que ce qui avait été observé dans des études antérieures. La majorité des espèces présentent actuellement des tendances plus favorables dans la mer des Wadden qu'au niveau de la voie de migration, une amélioration par rapport à il y a 10 ans. Seules trois espèces sont encore moins bien loties dans la mer des Wadden que dans l'ensemble de la voie de migration : le grand cormoran, l'avocette élégante et le chevalier gambette.

5.1 Introduction

For over 30 of the waterbird populations using the East Atlantic Flyway (EAF), the Wadden Sea, situated along the coasts of Denmark, Germany and the Netherlands, is of considerable importance as a stopover, wintering or moulting site. Up to 6.1 million birds can be present in the Wadden Sea at the same time, and an average of up to 10 million birds pass through each year. The Wadden Sea is also an important breeding area for several waterbird species. For instance, for Common Shelduck *Tadorna tadorna*, Eurasian Spoonbill *Platalea leucorodia*, Eurasian Oystercatcher *Haematopus ostralegus*, Pied Avocet *Recurvirostra avosetta*, Common Redshank *Tringa totanus*, Lesser Black-backed Gull *Larus fuscus*, Sandwich Tern *Thalasseus sandvicensis* and Common Tern *Sterna hirundo*, the Wadden Sea supports a significant proportion of the breeding population in Northwest-Europe.

Changes in the breeding populations as well as changes in the numbers and distribution of birds staging in the Wadden Sea outside the breeding season have been followed systematically as part of the Trilateral Monitoring and Assessment Program (TMAP), using standardized routines for fieldwork and data processing (Koffijberg *et al.* 2020, Kleefstra *et al.* 2021). In order to find explanations for the observed trends, it is relevant to compare the 'local' Wadden Sea trends with international trends at the flyway scale. For instance, if Wadden Sea trends and flyway trends are in line with each other, the drivers of changes in numbers are more likely to be global in nature, whereas if local trends differ from flyway trends, processes within the Wadden Sea are more likely to be of importance. In this chapter, population trends of breeding and migratory waterbirds in the Wadden Sea are reviewed against the trends of the same populations in the entire flyway.

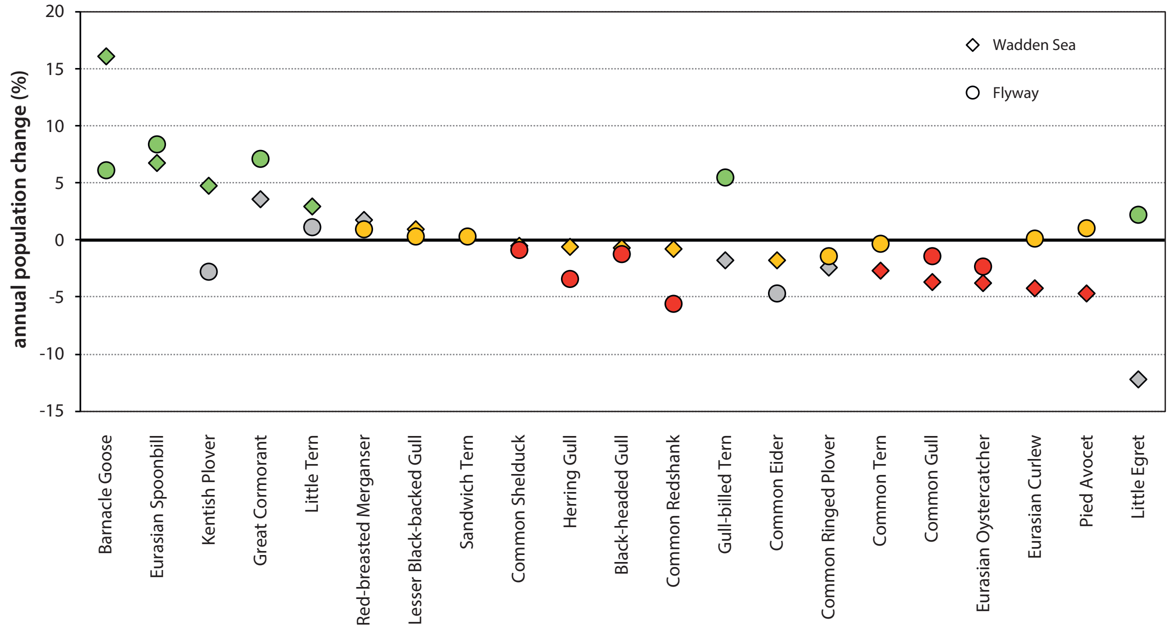


Figure 5.1. Comparison of trends in abundance of breeding birds in the Wadden Sea and at flyway level. Shown is the average rate of annual population change for the past 12 years (in %), with species ranked according to the rate of change in the Wadden Sea. Values above 0 denote increases, values below 0 decreases. Colours denote trend classification (red = decrease, green = increase , yellow = stable, grey = fluctuating).

5.2. Methods

5.2.1 Monitoring of breeding populations

Monitoring of breeding birds within TMAP has been carried out for a selection of 35 coastal breeding birds since 1991 and is coordinated by the Expert Group Breeding Birds in the Wadden Sea (EGBB). The survey strategy aims to carry out complete surveys annually of all rare and colonial breeding birds (e.g. Great Cormorant *Phalacrocorax carbo*, Eurasian Spoonbill, Pied Avocet, gulls and terns) while common breeding birds (e.g. Common Shelduck, Eurasian Oystercatcher, Common Redshank) are counted annually in a set of census areas spread over the Wadden Sea, aimed to provide a representative sample of annual abundance. Once every 6 years a total survey of all 35 breeding bird species is carried out, in order to update total population estimates for common breeding birds (last total count in 2018, next scheduled for 2024). Missing counts are accounted for by using imputing (carried out with rTRIM by Statistics Netherlands). With the imputed results, trends are calculated with the same rTRIM routine. Trend classifications are standardized (based on the rate of change and its standard error). They identify strong or moderate changes (either increase or decrease), stable trends and fluctuating (i.e. non-significant) trends.

We report trend estimates for the last 27 years (1991-2017) and the last 10 years (2008-2017). The most recent update of trends of breeding birds was published in the latest Quality Status Report and covered the period 1991-2017 (Koffijberg *et al.* 2021). Details on methods used and regional results can be found in the last annual progress report (Koffijberg *et al.* 2020).

5.2.2 Monitoring of migratory birds

Monitoring of 45 migratory waterbird species in the international Wadden Sea has taken place for 33 years now. The monitoring conducted by the Expert Group Migratory Birds (EGMB), consists of (a) at least five synchronous, integral counts per year: in January (mid-winter count), May, September and November, and a fifth in a different month each year, (b) frequent (bi-monthly to monthly) spring tide counts at 60 sites, (c) aerial counts of Eider *Somateria mollissima* in winter and of Shelduck during wing moult (July/August; by boat in the Netherlands). At present, a total of 594 count units from 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.* (2021).

Despite a large dataset with long-term monitoring data, coverage is not always complete. UINDEX (Bell 1995) is used to impute for missing counts in the dataset, taking into account effects of site, year and month (Underhill & Prys-Jones 1994). Sites were classified into four different Wadden Sea regions: Denmark, Schleswig-Holstein, Lower Saxony (including Hamburg) and the Netherlands. The counted and imputed values for each month were added to yearly averages for the respective "bird-years", covering the period from July to June of the following year (Kleefstra *et al.* 2021). Subsequently, TrendSpotter was used to calculate flexible trends with 95% confidence intervals (Visser 2004, Soldaat *et al.* 2007). We report trend estimates for 1987-2020 and for the last 10 years (2011-2020). The trend classes used are the same as in the breeding birds.

5.3. Results

5.3.1 Breeding populations

Of the 33 out of 35 breeding bird species for which trends could be assessed, 55% have experienced significant declines on the long term (i.e. 1991-2017; Koffijberg *et al.* 2021; Fig. 5.1). The highest rates of decline have been observed in Ruff, Hen Harrier *Circus cyaneus* and Common Snipe *Gallinago gallinago*, but negative trends are also found in a number of abundant and characteristic Wadden Sea species like Arctic Tern *Sterna paradisaea*, Common Eider, Kentish Plover, Pied Avocet and Eurasian Oystercatcher. Among the thriving species, relatively new breeding birds in the Wadden Sea show the highest rates of increase, like Barnacle Goose, Mediterranean Gull *Chroicocephalus melanocephalus*, and Great Black-backed Gull *Larus marinus*. Eurasian Spoonbill is another species showing sustained population growth.

For many species, the short-term trends for the last 10 years are similar to the long-term patterns recorded, apart from becoming non-significant due to annual variation in combination with the fewer years included. For a number of species, the trends (either decline or increase) tend to stabilise, as observed in Common Shelduck, Common Eider, Black-tailed Godwit *Limosa limosa*, Common Redshank, Black-headed Gull, Lesser Black-backed Gull and European Herring Gull *Larus argentatus*. The positive trend for Kentish Plover *Charadrius alexandrinus* is mainly the result of a recent increase (from a much lower level in the past) in coastal wetlands in the Schleswig-Holstein part of the Wadden Sea (Koffijberg *et al.* 2021).

In Bregnballe *et al.* (2018) the comparison of flyway trends and Wadden Sea breeding bird trends revealed that in some breeding birds, trends were more negative in the Wadden Sea than in the flyway. Currently still five species are doing less well in the Wadden Sea than on the flyway level (Fig. 5.1). Of these species Eurasian Oystercatcher, Common Ringed Plover *Charadrius hiaticula*, Eurasian Curlew *Numenius arquata*, and Pied Avocet show the highest rates of decline in breeding bird numbers in the Wadden Sea. Common Gull *Larus canus* is in this group of species as well. Its decrease started in the Dutch part of the Wadden Sea, followed by a more recent drop in numbers in the other parts.

Currently, the overall picture of trends of Wadden Sea

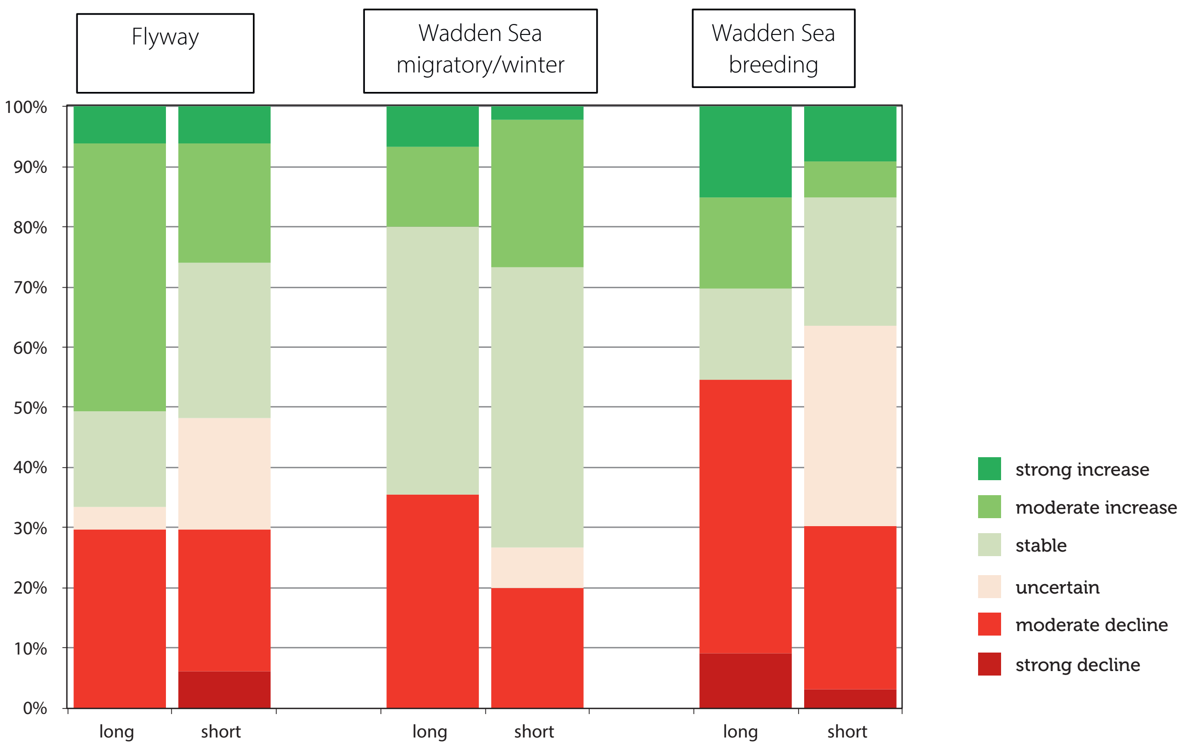


Figure 5.2. Summary of trend classifications of waterbird populations in the East Atlantic Flyway, compared with those for migratory birds and breeding birds in the Wadden Sea. Trends are shown for the long-term (long, i.e. since the start of the monitoring series) and short-term (short, last 10 years). Shown is the proportion of populations in each category.

breeding birds is still less favourable than that of trends of non-breeding birds in the Wadden Sea, and of waterbirds in the EAF as a whole flyway (Fig. 5.2).

5.3.2 Migratory birds

Trends in the Wadden Sea

Long-term trends of migratory birds in the Wadden Sea over the last 33 years (1987/1988-2019/2020) show increasing numbers for 20% of 45 populations, stable or uncertain trends for 44% and a decline in numbers for 35% (Fig 5.2). The short-term trends over the last 10 years of the monitoring period shows an increase for 27% species, stable/uncertain numbers for 53%, and a decline for 20% all populations (Fig 5.2). Comparison between the long- and short-term trends shows that six species increased in both periods (Eurasian Spoonbill, Barnacle Goose, Northern Pintail *Anas acuta*, Northern Shoveler *A. clypeata*, Common Ringed Plover and Sanderling *Calidris alba*). Great Cormorants increased in the long run but numbers stabilized in the last 10 years. Populations of six species decreased both in the long and the short term: Common Shelduck, Mallard *Anas platyrhynchos*, Eurasian Oystercatcher, Pied Avocet, Dunlin *Calidris alpina* and Spotted Redshank *Tringa erythropus*. The numbers of most species that showed negative long-term trends stabilised over the last 10 years, except for those of Ruff *Philomachus pugnax*, which increased slightly since 2010.

Of 14 species with stable long-term trends, eight also showed stable numbers over the last 10 years (Grey Plover, Northern Lapwing *Vanellus vanellus*, Bar-tailed Godwit *Limosa lapponica*, Eurasian Whimbrel *Numenius phaeopus*, Eurasian Curlew, Common Redshank, Black-headed Gull *Chroicocephalus ridibundus* and Common Gull), while three species showed an increase over the last 10 years (Eurasian Wigeon *Mareca penelope*, Common Teal *Anas crecca* and Ruddy Turnstone *Arenaria interpres*) and numbers of Common Greenshank started to decline. Eurasian Golden Plover *Pluvialis apricaria* and Curlew Sandpiper *Calidris ferruginea* show no significant trend over the short-term period.

Trends in the Wadden Sea compared with flyway trends

Previous analyses of the trends of migratory and wintering birds in the Wadden Sea until the season 2010/11 suggested that waterbird numbers declined faster within the Wadden Sea than at the flyway level, especially among

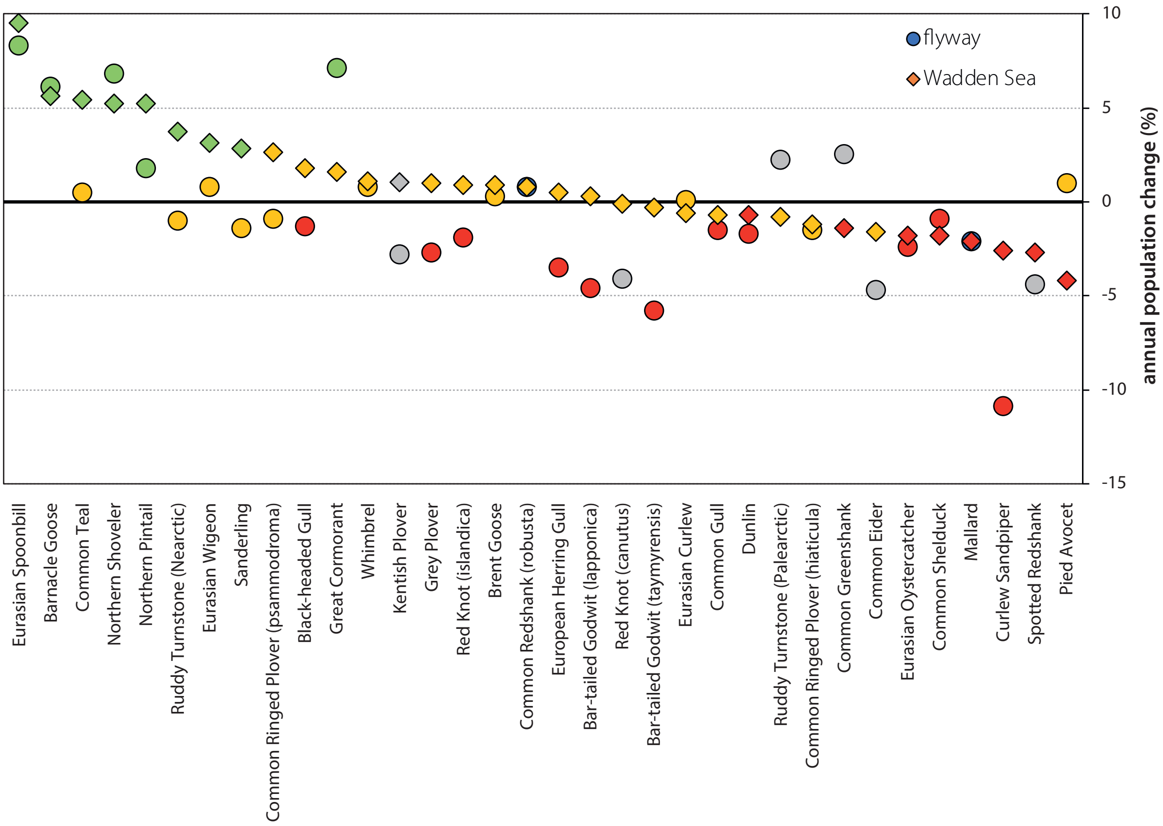


Figure 5.3. Comparison of trends in abundance of non-breeding birds in the Wadden Sea and at the flyway level. Shown is the average rate of annual population change for the past 10 years (in %), with species ranked according to the rate of change in the Wadden Sea. Values above 0 denote increases, values below 0 decreases. Colours denote trend classification (red = decrease, green = increase, yellow = stable, grey = fluctuating).

Harvey van Diek

Curlew Sandpiper | **Bécasseau cocorli** *(Calidris ferruginea)* & **Dunlins** | **Bécasseau variable** *(Calidris alpina), Westhoek, Netherlands*

species feeding on macrobenthos (e.g. van Roomen *et al.* 2015). More recently Bregnballe *et al.* (2018) showed that this pattern changed, based on Wadden Sea trends up to 2016/17, with more of the benthic-feeding species showing stable or positive trends in the Wadden Sea. At present quite a few populations of benthic-feeding species show less favourable trends at the scale of the EAF than in the Wadden Sea (Fig. 5.3). This particularly concerns Curlew Sandpiper, but also Common Ringed Plover (*psammodroma* subspecies), Grey Plover, Sanderling, Bar-tailed Godwit (both *taymyrensis* and *lapponica*), Eurasian Curlew and Nearctic Ruddy Turnstone. The gulls are also doing better in the Wadden Sea than in the flyway as a whole. Of the benthic feeders, only the two populations of Common Redshank show a more positive trend at the flyway level than in the Wadden Sea (*robusta* and *totanus* from Britain & Ireland/Britain, Ireland, France).

Trends of herbivorous waterbirds seemed to differ less between the Wadden Sea and the flyway, although those of Brent Goose *Branta bernicla*, Eurasian Wigeon and Common Teal tend to be slightly less favourable at the flyway level (Fig. 5.3). Trends of piscivorous species are generally positive for Great Cormorant and Eurasian Spoonbill, with the Cormorants doing a bit better at the flyway-level.

5.4 Discussion

5.4.1 Breeding populations

The Wadden Sea is an important breeding area for several waterbird species in the East Atlantic Flyway. For many species the Wadden Sea is an attractive breeding area due to the accessibility of suitable breeding habitats on islands, in salt marshes, coastal wetlands, dune areas and coastal grasslands, in combination with the proximity to rich food stocks in the intertidal and offshore areas. However, the overall picture emerging from the results of the breeding bird monitoring shows that many breeding birds in the Wadden Sea currently experience negative trends. More than half of the species considered have declined in the long-term, and 30% in the last decade with trends being uncertain in a further 30% of species (Fig. 5.2). In five species this decline has been stronger than at the flyway level: Eurasian Oystercatcher, Pied Avocet, Common Ringed Plover, Eurasian Curlew and Common Gull.

Bar-tailed Godwit | **Barge rousse** *(Limosa lapponica)* (Arnold Meijer / Blue Robin)

Already for quite some time several species fail in producing enough offspring to maintain a stable population. Particularly Eurasian Oystercatcher, Pied Avocet and Arctic Tern suffer from poor reproduction rates in nearly all parts of the Wadden Sea, both on the islands and on the mainland (van der Jeugd *et al.* 2014, Thorup & Koffijberg 2016). Recent data, until 2019, from the breeding success monitoring scheme in the Dutch Wadden Sea point at ongoing inadequacy of breeding output in Eurasian Oystercatcher, Pied Avocet, Black-headed Gull and Arctic Tern in nearly every year (Koffijberg *et al.* 2021). Among the mechanisms of reproductive failure, (increased) predation and flooding have been mentioned as the most important ones. However, it should be noted that breeding failures are often not caused by a single factor, and interactions occur with other issues such as food availability or human disturbance (JMBB 2016, Thorup & Koffijberg 2016, Koffijberg *et al.* 2021).

Several management measures have been proposed to stop or slow down the ongoing declines and conservation efforts have been initiated in many parts of the Wadden Sea. Wadden Sea countries have seen an increase in national and regional conservation activities, like the trilateral framework for an action plan (Koffijberg *et al.* 2016) and the Breeding Bird Action Plan for the Dutch Wadden Sea (PRW 2018) and various local or regional projects (incl. LIFE-programs) in the German and Danish Wadden Sea. Recently studies have started to investigate the way predators operate and interact with breeding birds and how this knowledge can be translated into measures to reduce losses caused by mammalian predation, beyond known opportunities to reduce predation risk through habitat measures like re-wetting (Leyrer *et al.* 2019).

5.4.2 Migratory birds

The trends presented here for non-breeding birds reflect the developments in each species' abundance in the Wadden Sea thoughout the annual cycle, as they are based on count data from all months of the year. Trends in the Wadden Sea are therefore not only affected by the maximum number of individuals recorded at a specific time of the year but also by the length of the period they make use of the Wadden Sea. For example, a year with a mild winter may for some species lead to a more positive overall value for that year because more individuals remain in the Wadden Sea instead of migrating further. Likewise, a declining trend can be recorded in a Wadden Sea because the number of individuals using the area declines and/or because these stage in the Wadden Sea for shorter periods.

Bregnballe *et al.* (2018) found that among the populations in decline in the Wadden Sea, seven were doing worse in the Wadden Sea than in the flyway as a whole. This was already a more positive picture than obtained by Van Roomen *et al.* (2015). The latest data show that currently just four populations of three species still do less well in the Wadden Sea than at the level of the flyway: Great Cormorant, Pied Avocet and the two populations of Redshank. Especially when it comes to short-term trends more species are now increasing in the Wadden Sea than in the flyway (Fig. 5.3): this is the case for Eurasian Wigeon, Common Teal, Common Ringed Plover, Grey Plover, Northern Lapwing, Red Knot (*Calidris canutus islandica*), Sanderling, Curlew Sandpiper, Bar-tailed Godwit, Common Redhank, Ruddy Turnstone, Black-headed Gull, Common Gull and European Herring Gull. Compared to Van Roomen *et al.* (2015) this means that for several of the above species the Wadden Sea is now doing better within the framework of the flyway.

Most of the benthivorous species, specialised in worms show a (much) more positive trend in the Dutch than in the German and Danish parts of the Wadden Sea, e.g. Common Ringed Plover, Grey Plover, Sanderling, Curlew Sandpiper, Dunlin and Bar-tailed Godwit. Species that also feed on 'other invertebrates', such as Common Shelduck, Whimbrel, Eurasian Curlew, Common Redshank, Ruddy Turnstone and Common Gull show similar contrasting trends between the different parts of the Wadden Sea (Kleefstra *et al.* 2021). The fact that *islandica*-Red Knots are currently doing better in the Wadden Sea overall seems to be related to the curtailment of mechanical shellfish fisheries. While food stocks are recovering (Baltic Tellin, cockles), the recovery of various shellfish-eating bird populations lags behind in most areas (Kleefstra *et al.* 2021).

January 2020 count in Tunisia (Hichem Azafzaf)

6. Wintering waterbirds in N Africa 1990-2017

Mohamed Dakki, Geneviève Robin, Marie Suet, Abdeljebbar Qninba, Mohammed A. El Agbani, Asmâa Ouassou, Rhimou El Hamoumi, Hichem Azafzaf, Wed A. L. Ibrahim, Khaled Etayeb, Mohamed S. Sayoud, Nadjiba Bendjedda, Laura Dami, Pierre Defos du Rau

Summary
In N Africa, waterbird counts have been carried out since the 1960s but only became regular in the 1980s. However, the lack of stable funding, of suitable optical equipment, of a stable and reliable national network of counters in the five countries, and of facilities to cover large areas and move from one site to another, meant that surveys never covered all the sites and were not really carried out regularly. It was with the setting up of the Mediterranean Waterbirds Network in 2012 that a certain regularity of counts was established, with financial and technical support, and with the development of the Medwaterbirds database. This database contains 36,436,391 records but suffers from over 60% missing data for 1990-2017. In order to accommodate this missing data issue, a new imputation method, LORI, was applied to the Medwaterbirds data set using multiple site- and/or year-specific covariates. N African trends for 16 species were assessed and discussed with respect to the corresponding trends at the international level (AEWA CSR 8).

Because the issue of missing data is an acute one in ecological monitoring, we argue that our modelling tool could be useful for many studies on wildlife. The joint work with other Mediterranean countries, the possible monitoring of breeding birds in the Mediterranean, and synchronous counts with the East Atlantic Flyway countries could all be considered as lines of action to improve these analyses, but also their interpretation in relation to the status of populations.

Résumé
En Afrique du Nord, les comptages d'oiseaux d'eau sont effectués depuis les années 1960 mais ne sont devenus réguliers que dans les années 1980. Cependant, faute de financements stables, d'équipements optiques adaptés, d'un réseau national de compteurs stables et fiables dans les cinq pays, et de facilités pour couvrir de grandes surfaces et se déplacer d'un site à l'autre, les relevés n'ont jamais couvert tous les sites et n'ont pas vraiment été réalisés régulièrement. C'est avec la mise en place du Réseau méditerranéen des oiseaux d'eau en 2012 qu'une certaine régularité des comptages a été établie, avec un soutien financier et technique, et avec le développement de la base de données Medwaterbirds. Cette base de données contient 36 436 391 enregistrements mais souffre de plus de 60% de données manquantes pour la période de 1990 à 2017. Afin de résoudre ce problème de données manquantes, une nouvelle méthode d'imputation, LORI, a été appliquée à la base de données Medwaterbirds en utilisant de multiples covariantes spécifiques aux sites et/ou aux années. Les tendances africaines pour 16 espèces ont été évaluées et discutées par rapport aux tendances correspondantes au niveau international (AEWA CSR 8).

Comme la question des données manquantes est un problème majeur dans le suivi écologique, nous soutenons que notre outil de modélisation pourrait être utile pour de nombreuses études sur la faune sauvage. Le travail conjoint avec d'autres pays méditerranéens, le suivi éventuel des oiseaux nicheurs en Méditerranée, et les comptages synchronisés avec les pays de la voie de migration de l'Atlantique Est pourraient tous être considérés comme des pistes d'action pour améliorer ces analyses, mais aussi leur interprétation par rapport à l'état des populations.

6.1 Introduction
Given its position on the southern shore of the Mediterranean, NW Africa (a region comprising Morocco, Algeria & Tunisia) is of major importance for waterbirds of both the East Atlantic Flyway (EAF) and the Black Sea - Mediterranean Flyway (El Agbani *et al.* 1996, Isenmann & Moali 2000, Dakki *et al.* 2001, Dakki *et al.* 2002, Isenmann *et al.* 2005). In these three countries some of the largest Mediterranean wetlands, of tremendous importance for passing or wintering waterbirds, are found, e.g. the Merja Zerga, the Sebkha of Oran and the Ichkeul lagoon. During both autumn and spring migrations, wetlands in N Africa provide a 'last chance saloon' for migratory waterbirds before they cross the Sahara or the Mediterranean Sea; both crossings are energetically demanding for waterbirds. These wetlands also shelter hundreds of thousands of waterbirds during the winter. Managing these waterbird populations and the N African wetlands hosting them requires filling in various knowledge gaps on species' population sizes and distribution patterns (Samraoui *et al.* 2011).

The first waterbird counts in N Africa took place in the

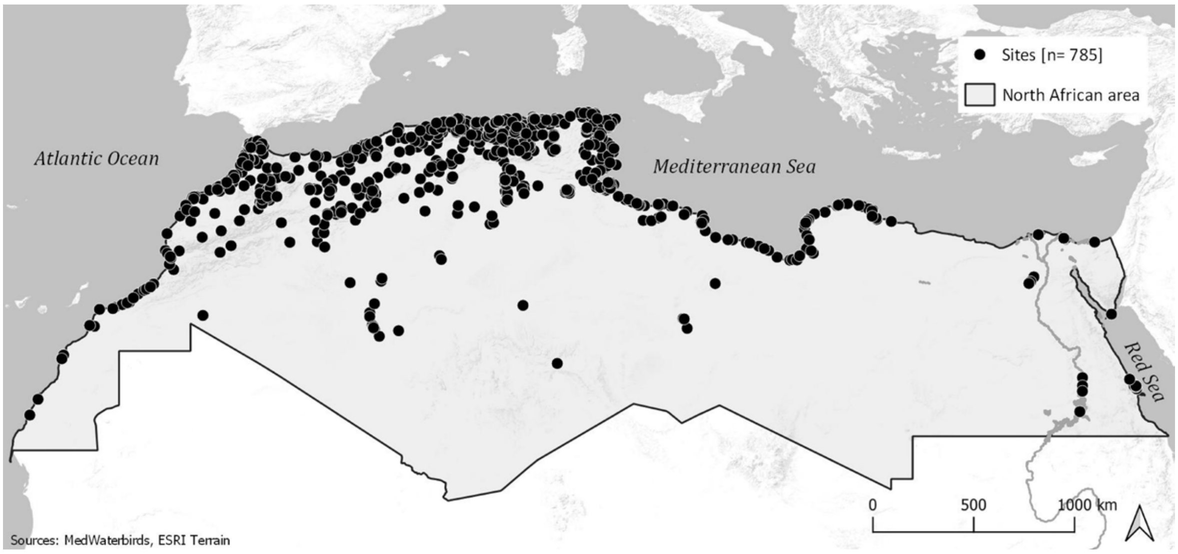


Figure 6.1. The 785 IWC monitoring sites surveyed for at least two years between 1990 and 2017.

1960s. Initial counts were carried out in 1964 in Morocco (Blondel & Blondel 1964), in 1967 in Tunisia (Isenmann *et al.* 2005) and in 1971 in Algeria (van Dijk & Ledant). The censuses became more regular from 1983 in Morocco, from 1985 in Algeria and from 2002 in Tunisia but for political and/or financial reasons the spatial coverage of wetlands has been variable. Depending on the country it is estimated that up to 60% of the wintering populations were not covered at that time.

The Mediterranean Waterbirds network (MWN) for waterbird monitoring started from 2012 (Dami & Gaget 2021), initially as a cooperation between all N African International Waterbird Census (IWC) coordinators, the French Agency for Biodiversity and Tour du Valat. The MWN developed different tools to strengthen the capacity of partners and organised annual international training sessions on waterbird censusing in Tunisia. Nowadays the network also includes cooperation with other countries from the northern coast of the Mediterranean. In 2014 the first coordinated N African winter census took place (Sayoud *et al.* 2017). To analyse the complex dataset, the MWN developed a new methodology to estimate trends for the region (Dakki *et al.* 2021), characterised by a large proportion of missing data.

This chapter presents trends of a selection of species with relevance for the EAF and with large populations in Morocco, Algeria and Tunisia.

6.2 Material and methods

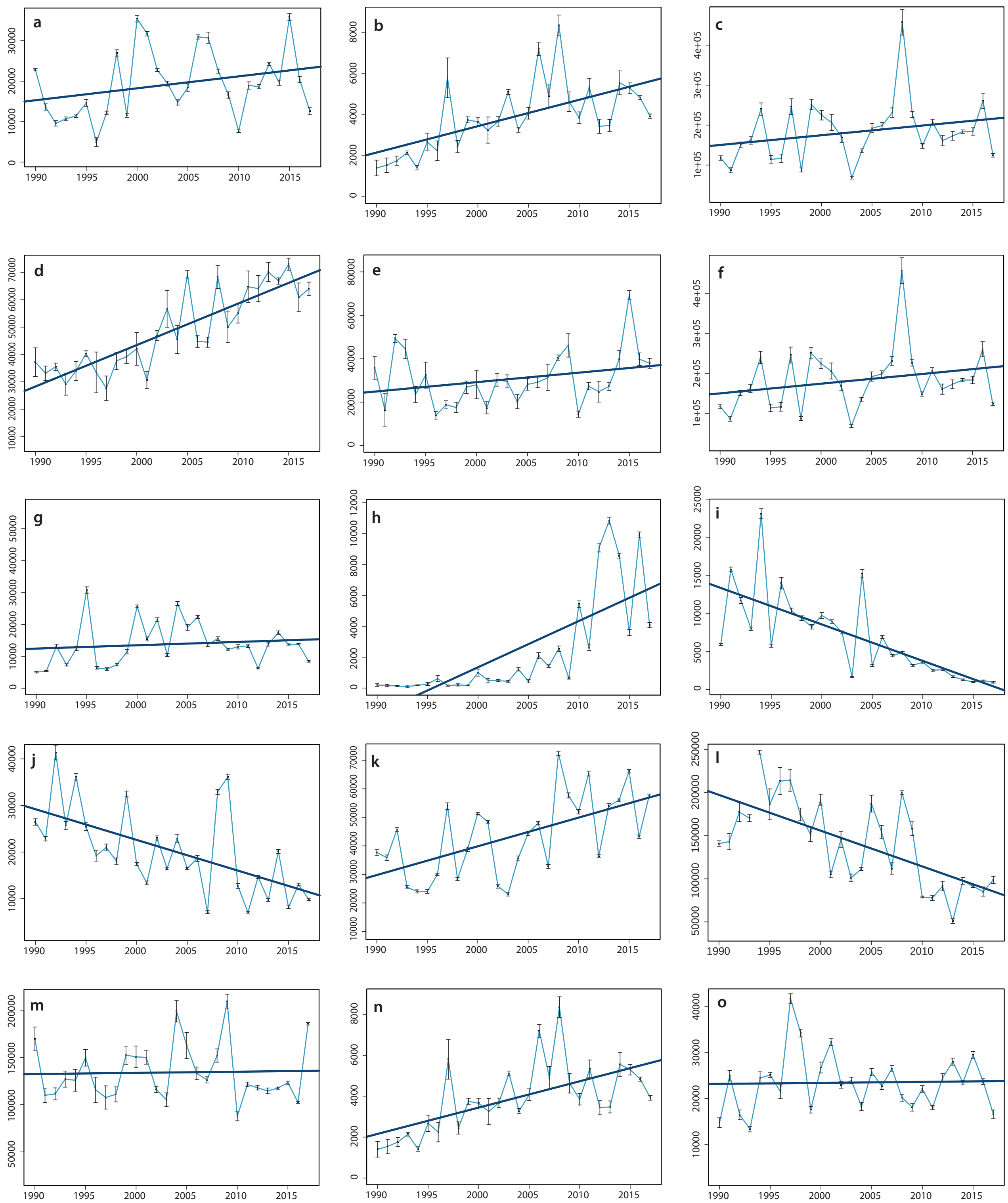
Biodiversity monitoring datasets, emerging in particular from citizen-science monitoring programmes, potentially allow answering many important ecological or conservation questions (Pereira *et al.* 2013, Stephenson *et al.* 2017). However, they are challenging to analyse when they display many missing values. This gap is particularly problematic in the monitoring of rare or threatened species, and in regions where data surveys are difficult to carry out for financial, political or logistic reasons.

The overall IWC dataset for N Africa includes up to 60% of missing values. Various statistical methods already exist to handle missing data or sparse datasets (van Strien *et al.* 2004). However, because of the high percentage of missing values we applied a new modelling tool, using a lasso approach and which allows both imputation and inference from predictors over sparse data (Robin *et al.* 2019). This LORI method is well suited to analyse multi-site and multi-year matrices of wildlife counts including many missing entries. We applied this method on the analysis of time series of count data for 16 waterbird species over 785 N African wetlands between 1990 and 2017. Twenty-one temporal, spatial and spatio-temporal covariates (qualitative or quantitative), extracted from GIS and taken from the web, were used to help imputing missing values in this dataset (Dakki *et al.* 2021).

6.3 Results

During the waterbird mid-winter counts in N Africa, from 1990 to 2017, data were collected at 785 wetland sites (Fig. 6.1). Of these, 21% were in Morocco, 47% in Algeria, 17% in Tunisia, 12% in Libya and 3% in Egypt (Dakki *et al.* 2021). This coverage reflects the relative abundance of wetlands in the region, although, especially in Egypt, coverage remains incomplete. Coverage varied over the years but has increased and became more consistent recently.

Using the LORI method to analyse the time series for 16 waterbird species, we estimated their trends at the N African scale over all 785 IWC sites for 1990 – 2017 (Fig. 6.2). Of the sixteen species analysed, only five are increasing strongly (Mallard *Anas platyrhynchos*, Glossy Ibis *Plegadis*



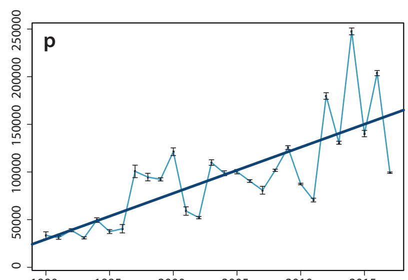


Figure 6.2. Yearly count totals and linear trends over all N African sites for the Pied Avocet (a), Dunlin (b), Gadwall (c), Mallard (d), Great Cormorant (e), Common Coot (f), Common Crane (g), Glossy Ibis (h), Greylag Goose (i), Northern Pintail (j), Common Teal (k), Eurasian Wigeon (l), Northern Shoveler (m), Eurasian Spoonbill (n), Common Ringed Plover (o) and Greater Flamingo (p) as modelled by LORI.

Hichem Azafzaf

Greater Flamingo | **Flamant rose** *(Phoenicopterus roseus)*

falcinellus, Common Teal *Anas crecca*, Eurasian Spoonbill *Platalea leucorodia* and Greater Flamingo *Phoenicopterus roseus*) in N Africa. Three are strongly declining (Greylag Goose *Anser anser*, Northern Pintail *Anas acuta* and Eurasian Wigeon *Mareca penelope*), while the others are more or less stable.

Among the species increasing in N Africa, the Mallard is on the contrary declining in the Mediterranean and Black Sea area (AEWA CSR8 results) and in the EAF, and the Common Teal is stable in the EAF after a long-term increase (Appendix 1). Among the species strongly decreasing in N Africa, Greylag Goose is increasing in the Mediterranean and Black Sea area, Eurasian Wigeon seems to be rather stable internationally, but also decreasing also at the EAF scale as well as Northern Pintail (AEWA CSR8 results). Among the species more or less stable in our study area, three (Pied Avocet *Recurvirostra avosetta*, Common Crane *Grus grus* and Northern Shoveler) are increasing in the Mediterranean and Black Sea area and Pied Avocet also in the EAF, as well as Great Cormorant *Phalacrocorax carbo* in this flyway. On the contrary, Common Coot *Fulica atra* is decreasing at the international scale, while Dunlin *Calidris alpina* and Ringed Plover *Charadrius hiaticula* decrease in the EAF.

Harvey van Diek

Ringed PLover | **Pluvier grand-gravelot** *(Charadrius hiaticula)*

6.4 Discussion

The three species displaying strong decline in N Africa may therefore appear of conservation concern for N African countries. However, their decrease may primarily be a response to climate change, notably to milder winter weather allowing populations to winter closer to their North European breeding sites (Pavón-Jordán *et al.* 2019, Fox *et al.* 2019). Greylag Geese have been shown to have shortened their migration (Podhrazsky *et al.* 2017; Ramo *et al.* 2015) through individual temperature-dependent decisions to remain closer to the breeding grounds, thus allowing birds to acquire breeding territories earlier. Northern Pintail may also respond to milder weather in its northernmost winter quarters as shown in North America (Meehan *et al.* 2021). This is not in contradiction with the South European/W African flyway trend which remains uncertain. Eurasian Wigeon are impacted by loss of good breeding habitats (Pöysä *et al.* 2017). Possibly decreasing numbers of breeding Wigeon in the north may directly impact the number of wintering birds in N Africa, although the flyway trend remains uncertain overall.

Ecological datasets, notably those concerning wildlife monitoring, often suffer from missing data, for instance because of the difficulties to monitor remote sites every year, to find sufficient funding, to recruit field observers for all the sites or to benefit from the adequate weather at survey time. In this study, we used a new multiple imputation method for count data with supplementary covariates. Given the observed differences in imputation accuracy between LORI and other existing methods like MICE, MISSFOREST, TRIM, CA, MEAN and GLM (Dakki *et al.* 2021), this new method seems to perform better than other methods currently used, but it also might potentially indicate different trends. If differences between these methods appear over such a long time span (28 years), they could potentially be even stronger at a shorter time span of for example 10 years, which is the usual timescale for short- term waterbird trend assessment. As trend estimation is a major diagnostic tool in the conservation and management of wild species, we argue that LORI is a tool well adapted to supporting conservation decisions as it provides good imputation performance, and hence trend estimation that is more likely to be reliable, particularly when predictor covariates are available.

The consolidation of cooperation with other Mediterranean countries and the development of a joint database could contribute to a more consistent monitoring of the species' conservation status, but also a more accurate interpretation of the resulting trend estimates. Monitoring of colonial breeding birds could provide crucial additional information especially for pelagic or largely dispersing species inadequately monitored by mid-winter counts. Another action that could be taken to consolidate inference on population size and trend estimates would be to better synchronise field activities of the MWN with those of the EAF count, by increasing field effort and coverage in the same years. We believe such synchronous work could also bring benefits in terms of spatial coverage through better funding, as well as in communication both with decision makers and the general public.

Acknowledgement

We would like to thank all the field observers who participated and/or participate today in the trans N African IWC, and who made this study possible.

Temporary wetland, North Africa

Long-tailed Cormorant | **Cormoran africain** (Microcarbo africanus) (Jacques van der Neut / Agami)

7. Status of coastal waterbirds at the Parc National du Banc d'Arguin, Mauritania 2020

El-Hacen Mohamed El-Hacen & Amadou Kidé

Summary

The Parc National du Banc d'Arguin (PNBA), Mauritania is one of the most important sites along the East Atlantic Flyway, not only for migratory waders but also for breeding waterbirds. The avifauna of the area, however, is increasingly under anthropogenic pressure and the numbers of many species are showing sharp changes. This short study presents an update on the long-term trends in the total number of birds at the Banc d'Arguin based on the total count that took place in January 2020. Further, we assessed the dynamics of six species that showed significant changes in numbers up to 2017 based on eight complete counts since 1980. We found that the total number of waders is still showing a significant decline and the total count of 2020 was similar to that of 2017. Of the six species assessed, only Great White Pelican *Pelecanus onocrotalus* showed a marginal increase in numbers, while Red Knot *Calidris canutus*, Bar-tailed Godwit *Limosa lapponica*, Eurasian Curlew *Numenius arquata* and Western Marsh-harrier *Circus aeruginosus* still show a significant decrease in numbers. Finally, Long-tailed Cormorant *Phalacrocorax africanus* showed no significant change in numbers since 1980, although the species seems to be increasing since 2007. Most of the declining species belong to migratory Western Palearctic – W Africa populations, and these local trends are similar to trends observed at the flyway level. The Afrotropical species seem to be generally stable or increasing in many cases, with a marked increase in the numbers of Lesser Flamingo *Phoenicopterus minor*. The causes of the observed changes in waterbird numbers are discussed.

Resumé

Le Parc National du Banc d'Arguin (PNBA), en Mauritanie, est l'un des sites les plus importants le long de la voie de migration Est-Atlantique, non seulement pour les limicoles migrateurs mais aussi pour les oiseaux d'eau nicheurs. Cependant, l'avifaune de la zone est de plus en plus soumise à des pressions anthropiques et le nombre de plusieurs espèces montre des changements brusques. Cette étude présente une mise à jour sur les tendances à long terme du nombre total d'oiseaux au Banc d'Arguin sur la base d'un comptage total qui a eu lieu en Janvier 2020. De plus, nous avons évalué la dynamique de six espèces qui ont montré des changements significatifs en nombre en 2017 sur la base de huit comptages complets depuis 1980. Nous avons constaté que le nombre total d'oiseaux est toujours en baisse, mais les chiffres du comptage de 2020 sont similaires au précédent (2017). Parmi les six espèces évaluées, seul le Pélican blanc Pelecanus onocrotalus *a montré une augmentation marginale du nombre, tandis que le Bécasseau maubèche* Calidris canutus*, la Barge rousse* Limosa lapponica*, le Courlis cendré* Numenius arquata *et le Busard des roseaux* Circus aeruginosus *montrent toujours une diminution significative de Nombres. Enfin, le Cormoran africain* Phalacrocorax africanus *n'a montré aucun changement significatif en nombre depuis 1980, bien que l'espèce semble augmenter depuis 2007. La plupart des espèces en déclin appartiennent à des populations migratrices du Paléarctique occidental – Afrique de l'Ouest, et ces tendances locales sont similaires aux tendances observées au niveau de la voie de migration. Les espèces Afrotropicales semblent, generalement, stables ou en augmentation dans de nombreux cas, avec une augmentation marquée du nombre de Flamants nains* Phoeniconaias minor*. Les causes des changements observés dans le nombre d'oiseaux sont discutées.*

7.1 Introduction

The Parc National du Banc d'Arguin along the coast of Mauritania is the largest marine protected area in W Africa (Fig. 7.1), and is known for its enormous numbers of wintering waterbirds (Altenburg *et al.* 1982). The area hosts the largest concentrations of wintering shorebirds in the world (Wolff & Smit 1990) as well as among the largest

El-Hacen Mohamed El-Hacen

Catching rays with traditional lanches by local Imraguen

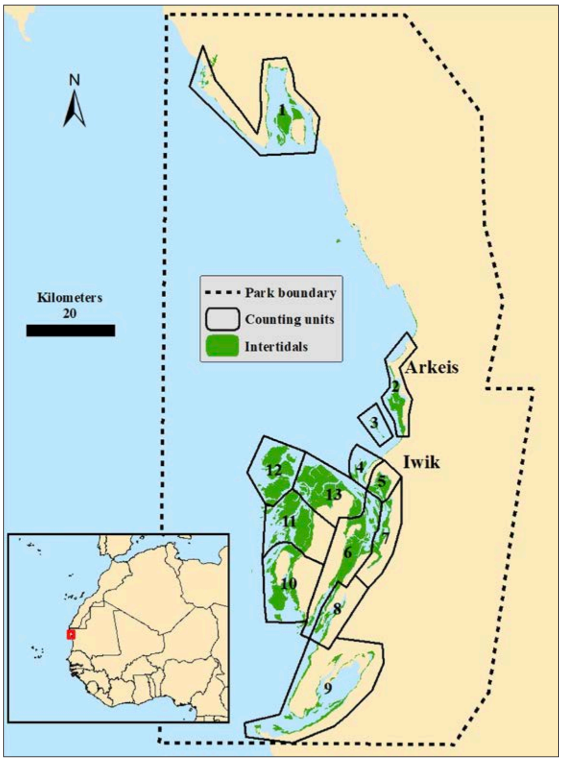


Figure 7.1. Map of the Parc National du Banc d'Arguin showing the 13 different functional units (of intertidal flats exploited by birds from largely different high-tide roost) distinguished by Zwarts et al. 1997.

breeding colonies of waterbirds in W Africa (Campredon 2000), including two endemic subspecies (El-Hacen *et al.* 2013). The park has been listed as a UNESCO World Heritage Site for its importance for migratory animals, especially birds. There are two major groups of waterbirds that use the area either for wintering or breeding (Oudman *et al.* 2020). The first group consists of arctic and sub-arctic breeding birds (Western Palearctic), dominated by intertidal feeding shorebirds (Charadrii) (Wymenga *et al.* 1990). The second group consists of Afrotropical breeding waterbirds, which are mostly piscivorous.

Despite its importance for migratory birds, the Banc d'Arguin lacks systematic long-term monitoring of the avifauna that inhabit the area (Oudman *et al.* 2020). Nonetheless, total counts of waterbirds have been made more frequently in recent years (van Roomen *et al.* 2015, Kidé & Diakhite 2017, Oudman *et al.* 2020). This is mainly due to the establishment of a monitoring programme along the East Atlantic Flyway led by the Wadden Sea Flyway Initiative, Wetlands International and BirdLife International (van Roomen *et al.* 2015).

Over the last few decades, the Banc d'Arguin ecosystems have been subjected to many human disturbances, particularly related to unsustainable fishing practices (Lemrabott *et al.* in revision). The establishment of the trans-Saharan road in early 2000s between the administrative capital (Nouakchott) and the economic capital Nouadhibou opened the Banc d'Arguin fishery to national and international markets (Lembrabott *et al.* in revision). Also, the establishment of the mining town of Chami just outside the park's eastern borders has put more pressure on its natural resources. Further, the intertidal systems of the Banc d'Arguin have experienced tremendous changes in seagrass cover and benthic communities over the last three decades (El-Hacen *et al.* 2020), which might have severe consequences for shorebirds that depend on these mudflats for feeding.

In this assessment, we aim to give a brief update on waterbird populations at the Banc d'Arguin based on the latest total count in January 2020, and discuss this in relation to ecological changes and human pressures. Further, we assess the dynamics of the six waterbird populations that showed significant changes in number up to 2017 (Oudman *et al.* 2020).

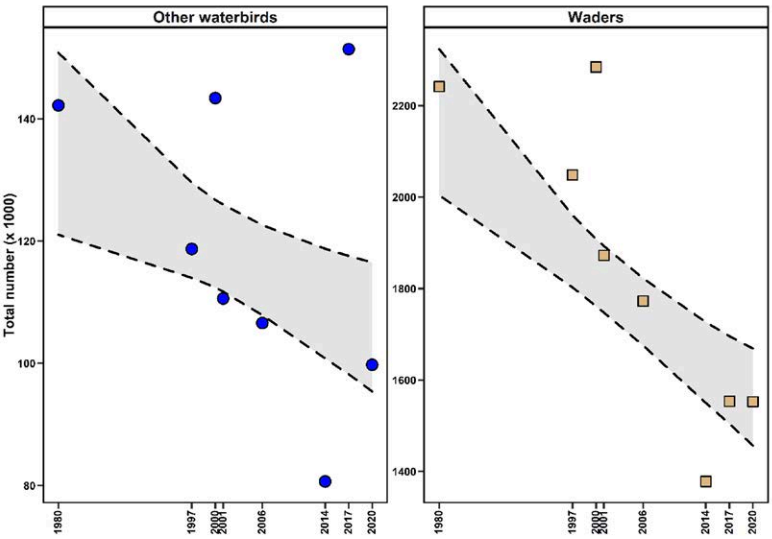


Figure 7.2. Long-term changes in the numbers of waders (right) and other waterbird species (left) in the Parc National du Banc d'Arguin, Mauritania. Coloured dots represent total counts and the grey shaded area depicts the 95% confidence limits of the linear trend.

7.2 Methodology

Long-term trends in the numbers of waterbird species were analysed using eight complete count data sets recorded in the Banc d'Arguin by various teams since 1980. All counts were carried out in the months of January/February around high-tide. Changes over time in the total numbers of waders, other waterbirds and six species that showed significant decrease in 2017 were assessed by linear regressions.

7.3 Results: Trends in waterbird populations up to 2020

The total number of waders counted is in significant decline ($F_{1,7}$ = 16.45, R^2 = 0.68, P = 0.006; Fig. 7.2). The total number of the rest of the waterbirds (mostly seabirds: gulls and terns) did show a significant change, although the numbers fluctuated a lot between counts ($F_{1,7}$ = 1.23, R^2 = 0.04, P = 0.3, Fig. 7.2). The last count of 2020 (Kidé 2020) was marked by an unprecedented presence of a large flock of Lesser Flamingo *Phoeniconaias minor* (around 1,119 individuals) and perhaps the first record of African Spoonbill *Platalea alba*.

Of the six assessed species, four showed significant declines, one showed a marginal increase and another one showed no significant change (Fig. 7.3). Red Knot *Calidris canutus* ($F_{1,7}$ = 31, R^2 = 0.81, P = 0.001), Bar-tailed Godwit *Limosa lapponica* ($F_{1,7}$ = 21, R^2 = 0.74, P = 0.003), Eurasian Curlew *Numenius arquata* ($F_{1,7}$ = 13.5, R^2 = 0.64, P = 0.01) and Western Marsh-harrier *Circus aeruginosus* ($F_{1,7}$ = 8.6, R^2 = 0.52, P = 0.02) all showed sharp declines. Great White Pelican showed a marginally significant increase in numbers ($F_{1,7}$ = 4.6, R^2 = 0.34, P = 0.07). Finally, Long-tailed Cormorant *Phalacrocorax africanus* numbers did not show a significant change over time ($F_{1,7}$ = 8.06, R^2 = 0.08, P = 0.4, Fig. 7.3).

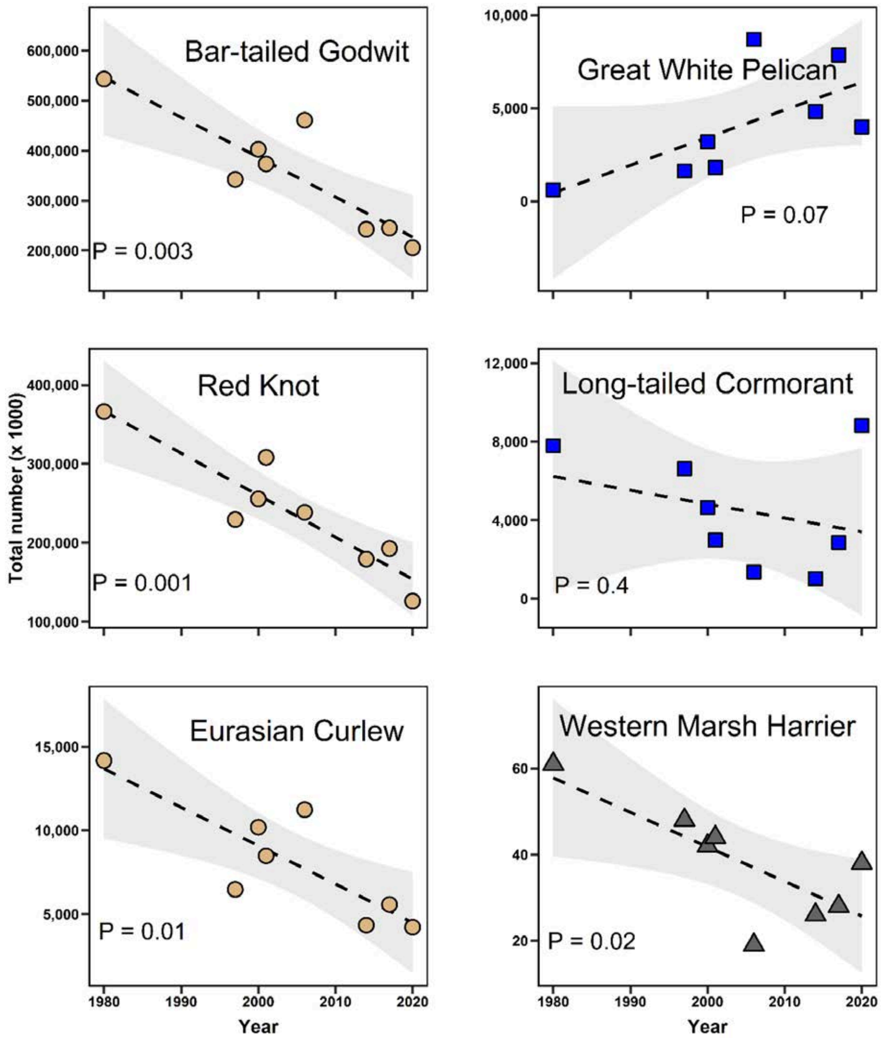


Figure 7.3. Long-term changes in species that showed a significant change at the Banc d'Arguin, Mauritania, based on eight counts from 1980 to 2017. Coloured dots represent total counts, grey shaded area depicts the 95% confidence limits around the regression lines and the P values are for this linear trend.

Red Knot | **Bécasseau maubèche** *(Calidris canutus)* & **Common Redshank** | **Chevalier gambette** *(Tringa totanus)*

7.4 Discussion

The total number of Western Palearctic breeding waders wintering at the Banc d'Arguin is declining in an alarming manner. Numbers of Red Knot, Bar-tailed Godwit and Eurasian Curlew in the 2014 - 2020 period were less than half of their usual numbers in the 1980s (Altenburg *et al.* 1982, Oudman *et al.* 2020). Similar declines of these species have been observed elsewhere along the East Atlantic Flyway (van Roomen *et al.* 2018, Appendix 1 of this report). The future of these long-distance migratory shorebirds is now of great concern as they face enormous loss of and rapid changes in habitats along their migratory routes.

Mounting evidence has shown that global warming might affect East Atlantic Flyway shorebirds differently in their three vital migratory regions: (1) increasing temperature and changes in food availability in the Arctic breeding ground (van Gils *et al.* 2016, Rakhimberdiev *et al.* 2018), (2) sea level rise, eutrophication, turbidity, and extreme weather events in the staging sites along the W European coast (Galbraith *et al.* 2005, Piersma 2006, 2007, van Roomen *et al.* 2012), and (3) sea level rise, extreme weather events, and unsustainable use of coastal resources in their non-breeding/ wintering grounds in W Africa (Lemrabott *et al.* in revision, de Fouw *et al.* 2016, El-Hacen *et al.* 2020).

Fishing baskets at Iwik, the Banc d'Arguin

Along the flyway, habitat deterioration is a known factor that is negatively affecting waterbird populations at many sites. In W Africa, Dodman *et al.* (2018, and Chapter 3) identified that overfishing and urbanization are among the most significant pressures to coastal biodiversity including waterbirds. Indeed, lately the Banc d'Arguin has experienced tremendous pressure from the national and international markets on its marine resources. Imraguen fishing in the Banc d'Arguin has shifted during the last 40 years from subsistence and seasonal to commercial, with increasing pressure towards the top of the food web (Boulay 2013, Lemrabott *et al.* in revision).

The ongoing unsustainable harvest of rays and sharks is expected to degrade the intertidal habitats of the PNBA which shorebirds depend on to feed. Removal of rays and sharks may release the W African Bloody Cockle *Senilia* *senilis* from predation, leading to an increase in their abundance, and likely outcompetion of important prey items for shorebirds such as *Dosinia spp.* (van Gils *et al.* 2013). Further, it is not clear how the changes in Imraguen fishing practice will affect the abundance and distribution of small pelagic fish that the local piscivorous breeding waterbird populations feed on (Veen *et al.* 2018). Removal of top predators from marine ecosystems may increase small pelagic fish for a while (Pauly *et al.* 1998, Jackson *et al.* 2001). This could explain the increase in colonial breeding seabirds in the Banc d'Arguin. Another factor that might have contributed to the observed dynamics of waterbirds in PNBA is the anthropogenic activities as well as conservation measures in the Lower Senegal Delta. The nature of these changes and its consequences on waterbirds merit further investigation.

The favourable prey items of many shorebirds in the Banc d'Arguin such as the bivalve *Dosinia* and polychaetes are losing ground to less favourable species like *Loripes* (El-Hacen *et al.* 2020). With climate change projections suggesting that the intensity and frequency of extreme weather events such as floods, drought and dust storms will increase in W Africa (Easterling *et al.* 2000, Jentsch *et al.* 2007, IPCC 2012), it is likely that seagrass beds of the Banc d'Arguin will experience severe die-offs as a consequence of desiccation (de Fouw *et al.* 2016), sediment deposition (Han *et al.* 2012, Hirst *et al.* 2017) and adverse conditions such as anoxia (Brodersen *et al.* 2017) and ammonium toxicity (El-Hacen *et al.* 2019). In view of the stark differences in the macrozoobenthic communities between seagrass and non-seagrass covered intertidal flats (Honkoop *et al.* 2008, Bouma *et al.* 2009), we expect cascading impacts on the food and populations of wintering shorebirds. Thus, the interplay between seagrass, sediment, benthos and shorebirds should be studied in more details at the Banc d'Arguin to establish an effective management plan for shorebirds and to contribute to reversing their decline in the area.

Hans Schekkerman

Fiddler Crabs are abundant at the upper intertidal zone

Black-winged Stilt | **Échasse blanche** (Himantopus himantopus) (Daniele Occhiato / Agami)

8. Conservation of an urban wetland: Grande Niaye de Pikine Urban Nature Reserve (Technopôle), Dakar, Senegal

Aïssatou Yvette Diallo and Pape Aibo Daniel Manga

Summary
The Grande Niaye de Pikine is the westernmost of the chain of 'niayes', coastal wetlands between Dakar and St-Louis in Senegal formed between coastal sand dunes. Technopôle is the key wetland area of the site, designated as an urban nature reserve in 2019. The Niayes of Dakar support a large diversity of birds, especially given their location within a major city. Of particular note, Technopôle is the key breeding site in Senegal for Black-winged Stilt *Himantopus himantopus*. The most numerous waterbird species recorded is Great Cormorant *Phalacrocorax carbo*. However, a range of anthropogenic pressures impact the site, including urbanisation and development, as well as conversion of wetland areas for agriculture. Waste is also a persistent probem. Since it received protected area status, various actions are underway to strengthen conservation, including a management plan, community engagement, monitoring and surveillance.

Résumé
La Grande Niaye de Pikine est la plus occidentale des Niayes, qui sont des zones humides constituées de dépressions inter-dunaires côtière allant de Dakar à St-Louis au Sénégal. Le Technopôle est l'une des zones humides clés du site, et est désigné comme réserve naturelle urbaine en 2019. Les Niayes de Dakar abritent une grande diversité d'oiseaux, surtout compte tenu de leurs emplacements au sein d'une grande ville. Le Technopôle en particulier est un site de reproduction clé au Sénégal pour l'Echasse blanche Himantopus himantopus*. L'espèce d'oiseau d'eau la plus décomptée est le Grand Cormoran* Phalacrocorax carbo*. Cependant, les pressions anthropiques telles que l'urbanisation, le développement des activités agricoles et les déchets sont également des problèmes persistants. Depuis son statut de zone protégée, diverses actions sont en cours pour renforcer la conservation, notamment un plan de gestion, l'engagement de la communauté, le suivi et la surveillance.*

8.1 Introduction
The Niayes of Dakar play a decisive role in the reproduction and survival of many migratory and resident bird species. However, knowledge about these species within the Niayes is limited due to inadequate studies in these habitats. The Niayes are made up of several wetlands, including the Grande Niaye de Pikine, as well as Lac Tanma, Lac Mbeubeuss, Lac Mbaouane and Lac Retba or Lac Rose (Figure 1). Recent studies have focused on some of the species that frequent Technopôle within the Grande Niaye de Pikine, and results are summarised here with an emphasis on pressures and conservation.

8.2 Site description
The Grande Niaye de Pikine is a wetland area located in an depression between coastal dunes, and part of it is the site commonly known as 'Technopôle'. It is surrounded by several outlying districts of the city of Dakar (capital of Senegal). Technopôle is a low-lying depression and the natural receptacle of large quantities of water originating from rainwater from the surrounding higher land. The Grande Niaye de Pikine was designated in March 2019 as an Urban Nature Reserve. The reserve has a total surface area of 650 ha, of which Technopôle comprises 313 ha, including 185 ha of water subdivided into five basins. The Grande Niaye de Pikine comprises three zones: a residential zone (zone 3), a farming zone (zone 2) and the water body itself (zone 1), where almost all the birds of the site are found (Fig. 8.2).

Technopôle is within the sub-canarian microclimate zone influenced by the maritime trade winds that are present year-round, with the cold Canary current cooling tropical air. These trade winds attenuate seasonal thermal contrasts and soften temperatures (Touré 2004). The short rainy season generally lasts 3 to 4 months. In 2017-2019 the average annual rainfall was 341 mm, with maximum temperatures recorded in August and September, whilst the average annual temperature was 25°C (Diallo 2021).

Technopôle's water table involves a saltwater-freshwater interaction or saltwater intrusion, which reflects the balance between fresh groundwater flowing into the ocean and salty oceanic water flowing towards the continent (Diouf 2005). The vegetation is dominated by typically Guinean species (for instance *Elaeis guineensis* and *Cocos nucifera*) and a herbaceous layer influenced by the topography of the environment (Touré 2004).

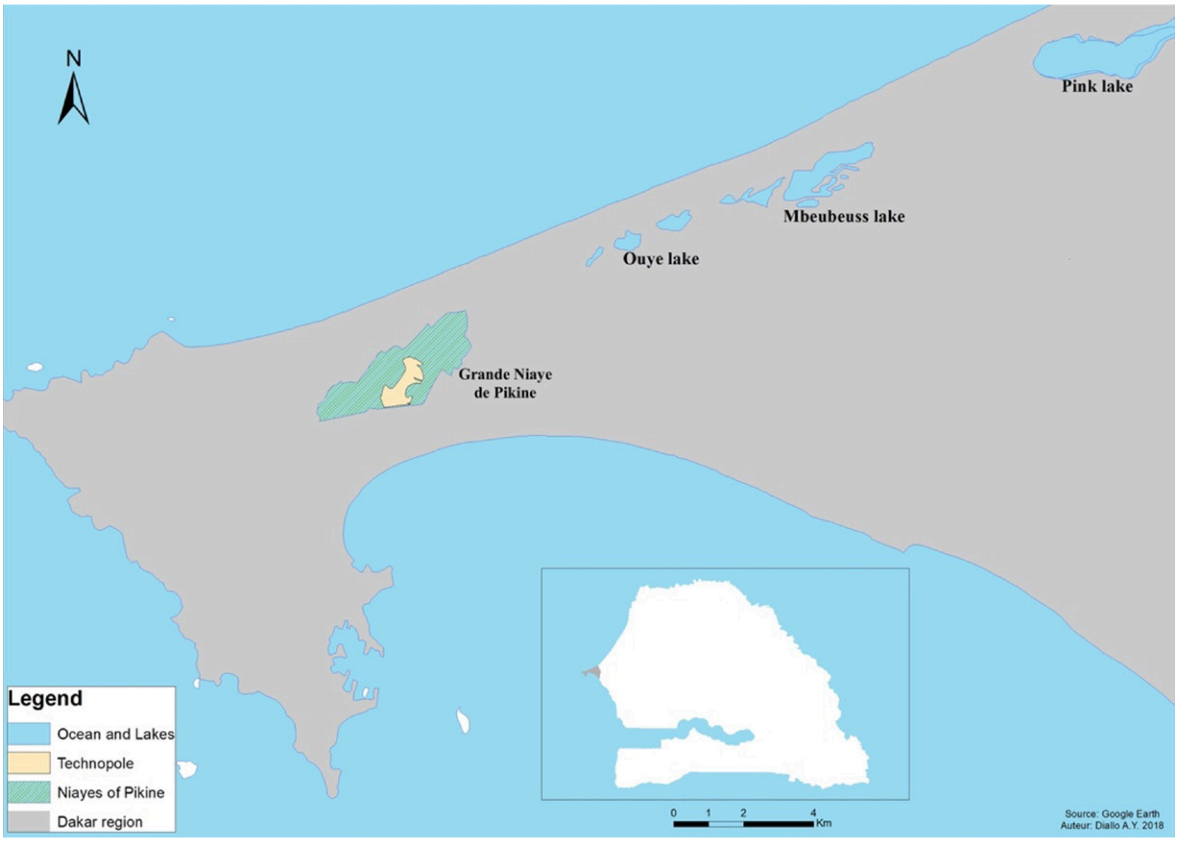


Figure 8.1. The Niayes of Dakar, showing the location of Technopôle (yellow) within Grande Niaye de Pikine (green).

English name	Scientific name	Number
Little Grebe	*Tachybaptus ruficollis*	3,601
Grey Heron	*Ardea cinerea*	1,256
Squacco Heron	*Ardeola ralloides*	1,077
Great White Egret	*Ardea alba*	3,906
Little Egret	*Egretta garzetta*	1,302
Great White Pelican	*Pelecanus onocrotalus*	1,591
Pink-backed Pelican	*Pelecanus rufescens*	2,171
Great Cormorant	*Phalacrocorax carbo*	11,543
Long-tailed Cormorant	*Phalacrocorax africanus*	7,520
White-faced Whistling-duck	*Dendrocygna viduata*	1,612
Black-winged Stilt	*Himantopus himantopus*	6,133
Spur-winged Lapwing	*Vanellus spinosus*	3,189
Common Ringed Plover	*Charadrius hiaticula*	1,793
Slender-billed Gull	*Larus genei*	4,442
Lesser Black-backed Gull	*Larus fuscus*	3,468
Black-headed Gull	*Larus ridibundus*	1,893
Black Tern	*Chlidonias niger*	1,338
Sandwich Tern	*Thalasseus sandvicensis*	1,835

Table 8.1. The more numerous species of birds recorded at Technopôle, with high counts in 2020-2021.

8.3 Fauna of Technopôle

The most noteworthy fauna are birds, with more than 200 species recorded (Hopkins & Diop 2011; Diallo *et al.* 2019); the total had reached 239 species by June 2019 (Piot 2019). Black-winged Stilt *Himantopus himantopus* is the focus of a monitoring programme, with 79 nests recorded in 2017 (Diallo *et al.* 2021); Technopôle supports around half of Senegal's breeding Black-winged Stilt population (Piot *et al.* 2021). Breeding has also been recorded for Spur-winged Lapwing *Vanellus spinosus*, African Jacana *Actophilornis africanus*, Black Crake *Zapornia flavirostra*, Common Moorhen *Gallinula chloropus* and White-faced Whistling-duck *Dendrocygna viduata*. Greater Flamingos *Phoenicopterus roseus* frequent the Niayes wetlands, including Technopôle at times. Table 1 illustrates some species with large cumulative numbers (over 1,000 individuals) at Technopôle - the result of regular monitoring of birds over 18 months from January 2020 to September 2021.

At Technopôle, there are a few reptiles (*Varanus panoptes*, *Varanus niloticus*, *Agama agama*). In some water bodies invaded by reeds, the presence of three species of fish (*Tilapia guineensis*, *Tilapia sp*., *Clarias anguillaris*) was noted (Diallo *et al.* 2019). Common Patas Monkeys (*Erythrocebus patas*) have been reported by several people in the area, although evidence of their existence at Technopôle has not yet been confirmed. The only mammals currently encountered at Technopôle are stray dogs, which are sometimes harmful to birds, as they may predate their nests (Diallo *et al.* 2019).

8.4 Anthropogenic pressures at Technopôle

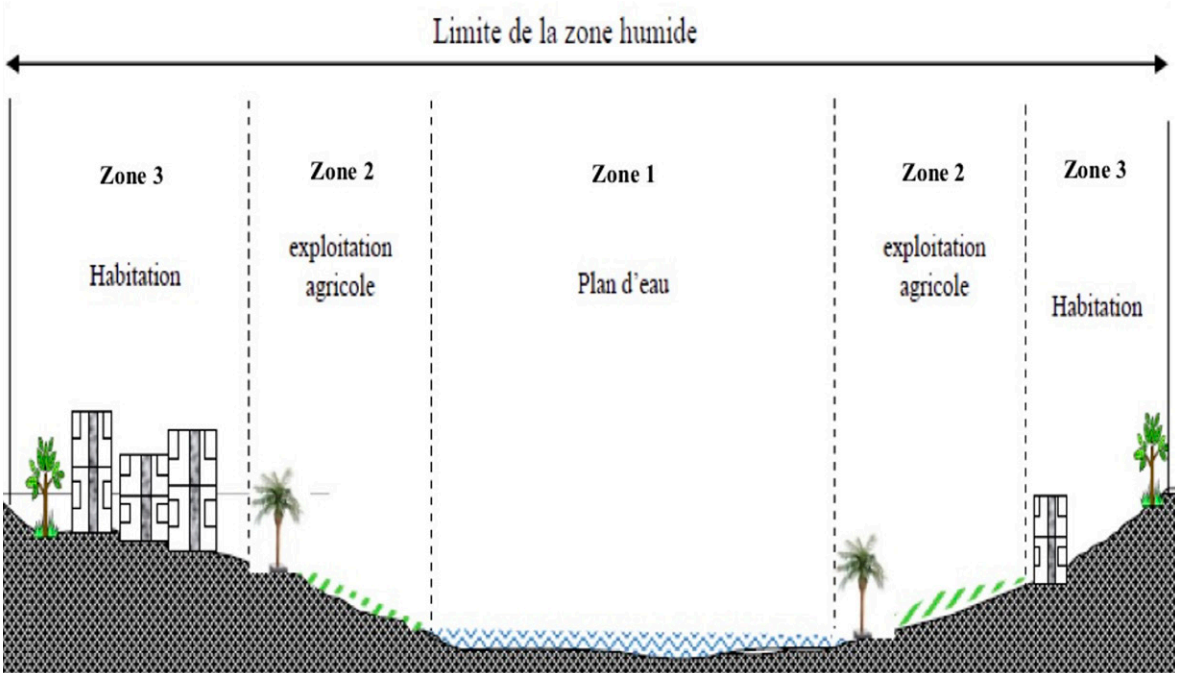
Wetlands provide many ecological services, most of which are vital to all living things. The Technopôle urban wetland serves as both a purification and a groundwater recharge station for the environment. However, anthropogenic pressures are having a negative impact on this environment, with the loss and degradation of habitats leading to a loss of biodiversity.

The pressures to this area are largely due to urbanisation and development. These pressures have been in place for many years; indeed, a large part of the area had previously been designated for development of a golf club. Technopôle is losing a lot of natural space by conversion of water bodies to land for work infrastructure or agriculture. In addition, there is a proliferation of aquatic plants and salinisation of the soil. Fishermen are in competition with birds for resources (Diallo 2021).

The dumping of household waste near water bodies is a persistent problem for the Niayes, including the clandestine dumping of wastewater, which has a huge impact on the environment, and chemical pollution due to the use of pesticides in market gardening areas (Diallo 2021).

8.5 Establishment of Technopôle site as an urban nature reserve

In the early 2010s, the association Nature - Communauté - Développement (NCD), the new BirdLife Partner in Sen-



Fi gure 8.2. Schematic cross-section of the Grande Niaye de Pikine (unpublished data from the Direction des aires marines communautaire protégées du Sénégal, 2014). Limite de la zone humide = wetland boundary; habitation = settlements; exploitation agricole = farmland; plan d'eau = water body].


Tim Dodman

View of the urban wetland Technopole

egal, became interested in Technopôle, linked to the Niayes IBA, and as the closest water body to Dakar with relative ease of access. Regular monitoring of the site since 2011 confirmed its importance for birds, and in 2016, NCD transferred its headquarters to the site. Furthermore, when Senegal hosted the 14th Pan African Ornithological Congress (PAOC) in 2016, things started to take shape, and NCD members took advantage of the site visit by PAOC participants to discover its wealth of birdlife. This enthusiasm continued into 2018, when Senegal organised the Ramsar PreCOP meeting, whose delegates visited Technopôle. Following this, members of the delegation and the Ramsar Senior Advisor for Africa made a resolution to the State of Senegal to confer protected area status to Technopôle.


Tim Dodman

Technopole is within the large city of Dakar, Senegal

Under Decree 2019-748, the site Grande Niaye de Pikine was declared an Urban Nature Reserve on 29 March 2019. This decision was reinforced and magnified by the President of the Republic during his inauguration speech on 3 April 2019. From that moment on, the management of the site was assigned to the Directorate of Community Marine Protected Areas (DAMCP). Thus, a conservator with about 30 staff commenced surveillance to control violations to the site and improve management. This has led to a dramatic increase in bird numbers (Table 1).

At the moment, various actions are underway at Technôpole to strengthen its conservation status. The development and implementation of a management plan is underway. The recommendations of Diallo (2021) contributed to the plan, which aims to improve management and valorisation of the site and its natural resources. Local populations are involved in management of the site, and there is collaboration with NCD, which is involved in training agents in the identification and monthly monitoring of birds.

Another urban wetland, Lac Mbeubeuss, is currently in the advocacy plans of NCD in collaboration with Senegal's Direction des Parcs Nationaux (National Parks Department). This site is similar to Technopôle and faces similar threats. Despite its location near the largest rubbish dump in Senegal, it still supports a reasonable number of birds.

Arnold Meijer / Blue Robin

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

Grey Plover | **Pluvier argenté** (Pluvialis squatarola), **Curlew Sandpipers** | **Bécasseau cocorli** (Calidris ferruginea), **Whimbrel** | **Courlis corlieu** (Numenius phaeopus), **Sanderling** | **Bécasseau sanderling** (Calidris alba) & **Bar-tailed Godwit** | **Barge rousse** (Limosa lapponica) Guinee-Bissau (Kim Fischer)

9. The Bijagós Archipelago: a key area for waterbirds of the East Atlantic Flyway

Mohamed Henriques, João R. Belo, Joãozinho Sá, Hamilton Monteiro, José A. Alves, Theunis Piersma, Tim Dodman & Marc van Roomen

Summary

The Bijagós Archipelago is a group of 88 islands and islets off the coast of Guinea-Bissau in W Africa. It is a site with an undisputable ecological value, recognized nationally by the implementation of three marine protected areas, and internationally by its classification as a Biosphere Reserve and Ramsar Site. Its relatively pristine ecosystem mostly arises from local community cultural traits that have limited the overexploitation of resources until recently. Among the diverse set of habitats, its extensive mangrove forests, totalling 524 km^2, cover c. 30% of the area of the archipelago, and provide crucial ecosystem services, including nursery for several fish species, safe roosting areas for waterbirds, and organic matter input to adjacent habitats. The Bijagós Archipelago also features ca. 450 km^2 of intertidal flats, among the largest in the world, which sustain highly diverse benthic communities. This site holds an important part of the regional populations of several waterbird species, especially migratory shorebirds. It is the third most important site on the East Atlantic Flyway for Palearctic migratory shorebirds during their non-breeding period, and second in Africa, after the Banc d'Arguin in Mauritania. Nevertheless, very steep declines in most shorebird species are being observed in the Bijagós Archipelago, in accordance with overall declines along the flyway, and in other important sites like the Banc d'Arguin. The reasons for these declines are not fully known. Conservation, research and monitoring efforts have been increasing in the area, in an attempt to gather baseline knowledge on different aspects of relevance to waterbirds, their habitats and the ecological processes they depend upon, ultimately aiming at protecting the extraordinary biodiversity value of the Bijagós Archipelago.

Resumé

L'archipel des Bijagós est un groupe de 88 îles et îlots au large de la Guinée-Bissau, en Afrique de l'Ouest. C'est un site dont la valeur écologique est indiscutable, reconnue au niveau national par la mise en place de trois aires marines protégées, et au niveau international par son classement en tant que réserve de biosphère et site Ramsar. Son écosystème relativement vierge résulte principalement des traits culturels des communautés locales qui ont limité la surexploitation des ressources jusqu'à récemment. Parmi les divers habitats, ses vastes forêts de mangroves, d'une superficie totale de 524 km2, couvrent environ 30 % de la superficie de l'archipel et fournissent des services écosystémiques cruciaux, notamment des nurseries pour plusieurs espèces de poissons, des aires de repos sûres pour les oiseaux aquatiques et un apport en matière organique aux habitats adjacents. L'archipel des Bijagós compte également environ 450 km2 de plaines intertidales, parmi les plus grandes du monde, qui abritent des communautés benthiques très diverses. Ce site abrite une part importante des populations régionales de plusieurs espèces d'oiseaux d'eau, notamment les limicoles migrateurs. C'est le troisième site le plus important sur la voie de migration de l'Atlantique Est pour les limicoles migrateurs du Paléarctique pendant la saison de non-reproduction, et le deuxième en Afrique, après le Banc d'Arguin en Mauritanie. Néanmoins, des déclins très marqués de la plupart des espèces d'oiseaux limicoles sont observés dans l'archipel des Bijagós, conformément aux déclins généraux le long de la voie de migration, et dans d'autres sites importants comme le Banc d'Arguin. Les raisons de ces déclins ne sont pas entièrement connues. Les efforts de conservation, de recherche et de suivi se sont multipliés dans la région, afin de rassembler des connaissances de base sur différents aspects concernant les oiseaux d'eau, leurs habitats et les processus écologiques dont ils dépendent, dans le but ultime de protéger l'extraordinaire valeur de biodiversité de l'archipel des Bijagós.

9.1 The Bijagós Archipelago

The Bijagós Archipelago, off the coast of Guinea-Bissau in W Africa, is composed of 88 islands and islets with very low human presence (Fig. 9.1). This archipelago has a triangular shape, typical of estuarine delta archipelagos (Pennober 1999). The tide regime is semi-diurnal, with spring tides reaching amplitudes of 4.5 m (Campredon & Catry 2017). The Bijagós Archipelago comprises a relatively diverse range of habitats, including a set of wetland ecosystems: channels and shallow reefs, intertidal flats, and mangroves, which are ubiquitous there and cover as much as 30% of its surface area (Cardoso 2017, Temudo & Cabral 2017). This mosaic of habitats support high levels of biodiversity, including internationally important populations of Green Turtle *Chelonia mydas* (Barbosa *et al.* 2018, Catry *et al.* 2010), Timneh Parrot *Psittacus timneh* (Lopes *et al.* 2018, 2019), African Manatee *Trichechus senegalensis* (Silva & Araujo 2001), and Atlantic Humpbacked Dolphin *Sousa teuszii* (van Waerebeek *et al.* 2004). There are

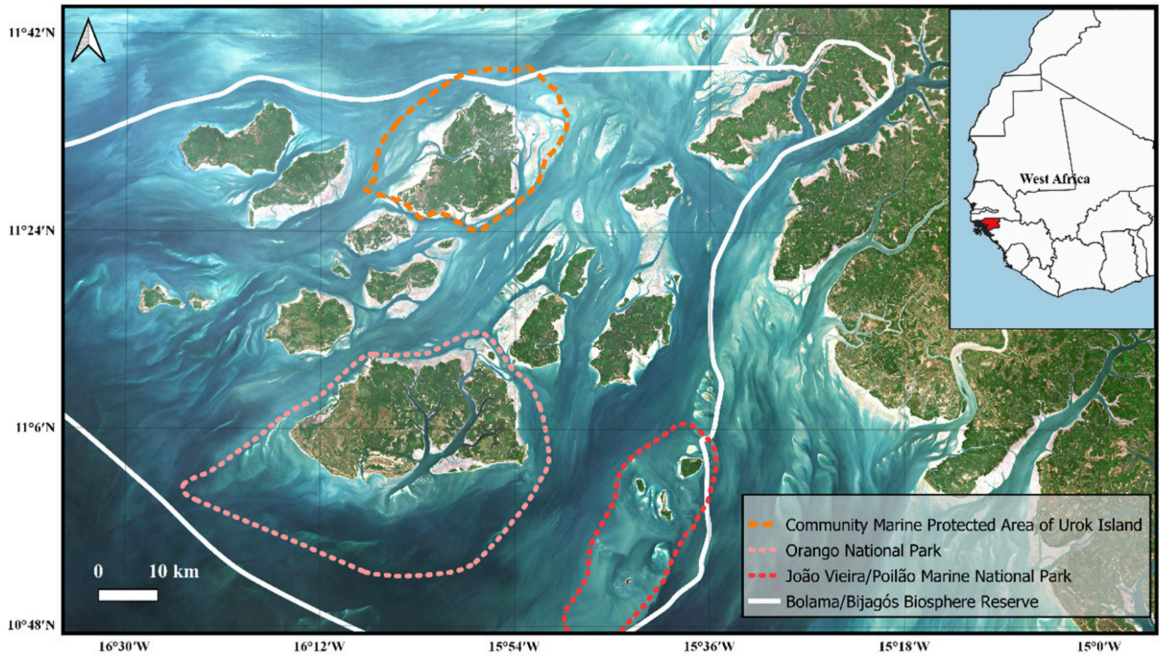


Figure 9.1. Satellite image of a low-tide moment (tide height: 1.04 m) in the Bijagós Archipelago, Guinea-Bissau. The limits of the Biosphere Reserve and of the three marine protected areas are represented in the map. Satellite image (taken 21 November 2019) from Copernicus' Sentinel-2B, courtesy of the European Space Agency.

also important sites for colonial breeding waterbirds (Birdlife International 2013, Zwarts *et al.* 2009) and the second most important assemblage of wintering shorebirds in W Africa (Delany *et al.* 2009, Dodman & Sá 2005, Salvig *et al.* 1997, 1994, Zwarts 1988). The undisputable ecological value of this site led to its recognition as a Biosphere Reserve by UNESCO in 1996 (Biai 2015), followed by classification as an Important Bird & Biodiversity Area (IBA) in 2001. Three marine protected areas (MPAs) of different management levels have been established within the Bijagós: Orango National Park, João Vieira and Poilão Marine National Park, and the Marine Community Protected Area of the Islands of Formosa, Nago & Chediã (Urok) (Biai *et al.* 2003, Daniel Suleimane Embalo *et al.* 2008, Embalo *et al.* 2008, INEP 2006, Fig. 9.1). In 2014, in recognition of the importance of the Bijagós Archipelago for numerous wetlands values, including waterbirds, it was also designated as a Ramsar Site (Campredon & Catry 2017). An application for an inscription of the archipelago as a World Heritage Site is currently under preparation.

9.2 The Importance of the Bijagós Archipelago for Waterbirds

Key ecological features

There are several ecological features that likely contribute to the large numbers of waterbirds found in the Bijagós Archipelago. The ecosystem in this archipelago is considered to still be relatively pristine, mostly as a result of the local community cultural traits, whose beliefs include animist-based religious regulations that have kept the resources and the environment from being overexploited until recently (Campredon *et al.* 2010, Campredon & Catry 2017, Maretti 2015, Rachid *et al.* 2011). Moreover, there is no large-scale industry and coastal development, whilst hardly any infrastructure has thus far been built. There are still low levels of pollutants in the environment (Catry *et al.* 2021, 2017, Coelho *et al.* 2016, Mullié 2017). Tourism is underdeveloped, with low long-term visible impacts on the environment (Polet 2011). Fishing areas around the MPAs are only used by the local communities, for local consumption. All these features combined result in quite small levels of human disturbance in this area.

The Bijagós Archipelago constitutes a relatively productive system, with high levels of organic matter, but presents low levels of chlorophyll due to relatively low nutrient content in the water (Campredon & Catry 2017). This is partly compensated by the fact that this archipelago is situated south of a large upwelling zone, benefiting also from the seasonal influence of upwellings linked to the Canary currents during the dry season, from small-scale coastal upwellings formed through trade winds and contributions of organic matter from continental run-off (Campredon & Catry 2017, Pennober 1999). The extensive mangrove forests occupy 524 km^2 (Temudo & Cabral 2017) and constitute a very productive ecosystem that may play an important role for intertidal invertebrates, on which several shorebird species rely for food.

A recent study by Henriques *et al.* (2021) assessed the role of mangrove forests as a direct carbon source to benthic macroinvertebrates living in adjacent intertidal flats of the Bijagós Archipelago. While no overall evidence of a direct contribution to several groups of benthic invertebrates was found, a significantly higher contribution of mangrove carbon to the sediment organic matter of intertidal flats adjacent to mangrove forest was detected, when compared to intertidal flats without mangrove. However, mangrove carbon was present only in benthic macroinvertebrates within the first 50 m from the mangrove edge, with this contribution fading away rapidly with increasing distance. The authors suggested that the contribution of mangrove forests to the productivity of intertidal flats could occur mostly indirectly, with mangrove organic matter being transformed in nutrients like inorganic carbon and processed nitrogen (through mineralization), which may then be assimilated by other primary producers (like algae) and fuel intertidal food webs.

Complementary to this, Meijer *et al.* (2021) also assessed the importance of mangrove forests to benthic macroinvertebrates in the Bijagós Archipelago, but taking a landscape-scale approach and comparing the benthic macroinvertebrate community composition and abundance between tidal basins with different levels of connectivity to mangroves. They found that the configuration and types of mangrove basin had a significant effect on the structure and composition of the benthic macroinvertebrate community. In fact, intertidal flats with higher influence of mangrove forests (larger mangrove areas) had higher levels of organic matter and suspended materials, and consequently also had richer and more abundant benthic macroinvertebrate communities. These ecological processes at the base of the food webs contribute to the productivity of this system, creating a set of specific features that are important for waterbirds.

The Bijagós Archipelago features one of the very large intertidal flats worldwide, estimated at over 450 km2 (Henriques *et al.* unpublished data; Hill *et al.* 2021), where shorebirds find vast foraging areas. Within these intertidal flats there is a very high diversity of benthic macroinvertebrate prey (contrasting with low abundances) when compared to temperate intertidal flats, which in turn results in competition avoidance due to low overlap between the trophic niches of the different shorebird species (Catry *et al.* 2016, Hickey *et al.* 2015, Lourenço *et al.* 2017, Lourenço *et al.* 2018, Piersma *et al.* 1993). This has a structuring effect on the community of shorebirds along the different areas of the intertidal flats of the Bijagós Archipelago, promoting habitat partitioning. An extreme example of this effect is the case of the widespread West African Fiddler Crab *Afruca tangeri*, a known ecosystem engineer in intertidal flats. Areas colonized by this crab constitute a very different sub-habitat for shorebirds and their prey (Fig. 9.2), presenting significantly lower biomass of all other

José Pedro Granadeiro

Figure 9.2. Intertidal flats in the bay of Adonga, Orango National Park, Bijagós Archipelago.

Species	1986/87 1992/93	2014 - 2020	
		mean	s.d.
Flamingos, Pelicans and Herons			
Greater Flamingo *Phoenicopterus roseus**	2,438	550	130
Pink-backed Pelican *Pelecanus rufescens***	N/A	1,300	110
Great Cormorant *Phalacrocorax carbo*	N/A	400	70
Western Reef Heron *Egretta gularis*	1,800	1,500	390
Western Cattle Heron *Bubulcus ibis**	50,000	130	50
Black Heron *Egretta ardesiaca**	2,200	20	10
Gulls and Terns			
Grey-headed Gull *Larus cirrocephalus***	170	600	370
Slender-billed Gull *Chroicocephalus genei*	N/A	400	110
Gull-billed Tern *Gelochelidon nilotica*	10,130	7,600	5,350
Caspian Tern *Hydroprogne caspia*	1,456	4,600	1,830
Sandwich Tern *Thalasseus sandvicensis*	2.952	10,000	3,650
Royal Tern *Thalasseus maximus*	2,078	4,200	1,630
Lesser Crested Tern *Thalasseus bengalensis*	384	500	310
Common Tern *Sterna hirundo*	5,988	12,000	8,150
Little Tern *Sternula albifrons*	6,348	5,700	2,000
Black Tern *Chlidonias niger*	4,295	3,200	4,300
Palearctic migratory shorebirds			
Common Ringed Plover *Charadrius hiaticula*	26,300	17,000	3,600
Kentish Plover *Charadrius alexandrinus*	5,000	600	270
Grey Plover *Pluvialis squatarola*	39,100	9,900	1,350
Red Knot *Calidris canutus*	31,300	38,000	22,000
Sanderling *Calidris alba*	24,300	20,000	6,000
Curlew Sandpiper *Calidris ferruginea*	326,500	59,000	27,900
Bar-Tailed Godwit *Limosa lapponica*	108,700	56,000	28,600
Eurasian Whimbrel *Numenius phaeopus*	22,000	23,000	3,280
Eurasian Curlew *Numenius arquata**	9,300	3,400	1,000
Common Redshank *Tringa totanus*	2,920	17,000	3,270
Common Greenshank *Tringa nebularia*	1,400	1,700	540
Ruddy Turnstone *Arenaria interpres*	7,900	2,300	610
Little Stint *Calidris minuta**	59,700	1,600	1,550

Table 9.1. List of waterbird species for which the estimated number of individuals based on several counts conducted during the non-breeding period meets the Ramsar Convention's 1% criterion. 1% thresholds were calculated from the Waterbird Population Estimates 5th Edition for the estimates of 1986/87-1992/93 (Wetlands International 2012), and from the Agreement on African-Eurasian Migratory Waterbirds Conservation Status Report (AEWA CSR8) for the 2020 estimates, both available at wpe.wetlands.org. Data for 1986/87-1992/93 were retrieved from Dodman & Sá 2005. Data for 2014-2020 are means and standard deviation of three estimates based on sample counts in January 2014, 2017 and 2020 (GPC/IBAP/WSFI unpublished data, see van Roomen *et al.* 2021 for details of the counts and extrapolation) . * Species for which estimates in 2020 would not qualify for the Ramsar 1% criterion. ** Species for which estimates in 1986/87-1992/93 did not qualify for the Ramsar 1% criterion, but that currently qualify based on the estimate for 2020.

Species	Year	N breeding pairs
African Sacred Ibis *Threskiornis aethiopicus*	1994	742
African Spoonbill *Platalea alba**	1992	1,000
Black-crowned Night-heron *Nycticorax nycticorax*	1994	168
Squacco Heron *Ardeola ralloides*	1994	318
Green-backed Heron *Butorides striata**	1994	513
Cattle Egret *Bubulcus ibis*	1994	270
Great White Egret *Ardea alba**	1994	925
Little Egret *Egretta garzetta*	1994	553
Western Reef-egret *Egretta gularis**	1994	870
Grey-headed Gull *Larus cirrocephalus**	1994	800
Caspian Tern *Hydroprogne caspia**	1987	300
	1997	594*
	2019	259
Royal Tern *Thalasseus maximus**	1994	1,867*
	2015	335
	2019	475

Table 9.2. List of notable waterbird species breeding in the Bijagós Archipelago with estimates of the number of breeding pairs. Data from Dodman & Sá 2005, Folmer *et al.* 2019 and Veen *et al.* 2004, 2015. * Species for which the number of breeding pairs has qualified for Ramsar 1% criterion. When estimates for more than one year are given, * marks the estimate that qualified for Ramsar 1% criterion.

benthic macroinvertebrates than areas without fiddler crabs (Paulino *et al.* 2021, Zwarts 1988). On the other hand, fiddler crabs themselves are key prey for many shorebird species in the Bijagós Archipelago (Carneiro *et al.* 2021, Lourenço *et al.* 2017, Zwarts 1985). This results in very different shorebird species assemblages between these two sub-habitats (Paulino *et al.* 2021, Zwarts 1988).

The availability of many undisturbed high-tide roost locations and resting places for waterbirds, within dense mangrove forests, on open and vast beaches or around elevated sand banks, is another important trait of the Bijagós Archipelago. Here, as elsewhere, shorebirds have been noted to display high fidelity to foraging areas (NIOZ & University of Aveiro, unpublished data; Bom *et al.* 2020; www.globalflywaynetwork.org), and therefore the availability of suitable roosting sites in the vicinity of their foraging grounds will be advantageous. Shorebird feeding patterns also depend on the balance between the availability of food resources and energetic requirements. During the non-breeding season, it is expected that species specific diet and the trophic network will vary across different periods (wintering and migration) accompanying shorebird's energetic demands (Carneiro *et al.* 2021). For instance, just before migrating northwards to their breeding grounds, shorebirds need to fuel up and either increase their prey intake rate and/or shift their diet to more energetically profitable prey, in order to store fat for their long migration. However, this is only possible if the wintering areas provide adequate conditions. In the Bijagós Archipelago, preliminary results of ongoing studies indicate that prey availability and size increase from the northern winter to the fuelling period, and the main consumed prey becomes increasingly important throughout the fuelling period (Coelho *et al.* unpublished data). Thus, the demanding energetic requirements of shorebirds in order to fuel up for migration seems to be met by the productivity of the intertidal flats of the Bijagós Archipelago.

The Bijagós Archipelago also has an abundant and diverse small pelagic fish community (Campredon & Cuq 2001; Correia *et al.* 2017). This is important for several piscivorous waterbirds, particularly terns, which are very efficient fish predators and represent the bulk of the seabird numbers along the W African coasts (Correia *et al.* 2018, Veen *et al.* 2004).

Waterbird populations

Waterbirds in the Bijagós Archipelago comprise about 105 species, totalling 37% of the 287 bird species reported for this IBA (Carneiro *et al.* 2017, Dodman *et al.* 2004, Dodman & Sá 2005). This includes 53 Afrotropical resident species (from 18 families) and 50 Palearctic and intra-African migrants (from 19 families). Because of their numerical importance at the Bijagós Archipelago, most attention is given to Palearctic (migratory) shorebirds, which may spend more than half of their year in the Bijagós (Dodman & Sá 2005, Frikke *et al.* 2002, van de Kam *et al.* 2004). Also important are the populations of terns and gulls, with over 33,000 terns estimated in the Bijagós Archipelago in 1992/1993 (Salvig *et al.* 1997 in Dodman & Sá, 2005).

From an international importance perspective, many sites in the Bijagós Archipelago counted during the northern winter between 1986/87 and 1992/93 have been found to meet the Ramsar Convention's 1% criterion (Ramsar Convention 2014, criterium A6). This was the case for 12 shorebird species, eight species of terns, two species of gulls, three species of herons and the Greater Flamingo *Phoenicopterus roseus* (Dodman *et al.* 2004; Table 9.1). Likewise, breeding estimates obtained in the 1990s revealed Ramsar designating numbers for several of these species (Dodman *et al.* 2004, Veen *et al.* 2015; Table 9.2).

More recent estimates of numbers of non-breeding birds conducted during the northern winter, in 2014, 2017 and 2020, highlighted contrasting numbers for some Ramsar-triggering species when compared to the 1986/87-1992/93 period, with those of species like Greater Flamingo, Little Stint *Calidris minuta* and Eurasian Curlew *Numenius arquata* no longer qualifying for the Ramsar 1% criterion (Table 9.1). On the other hand, new estimates for two additional species in 2020 are considered to meet the criterion, for Pink-backed Pelican *Pelecanus rufescens* and Grey-headed Gull *Larus cirrocephalus*. Despite these changes, recent data undoubtedly supports the claims of the Bijagós Archipelago holding one of the very important waterbird assemblages in the world (van de Kam *et al.* 2004).

9.3 Colonial breeding waterbirds

Several colonial breeding waterbird species occur in the Bijagós Archipelago, distributed across a large range among the islands. These include herons and egrets, African Sacred Ibis *Threskiornis aethiopicus*, African Spoonbill *Platalea alba* and several species of gulls and terns (Table 9.2).

Terns are among the most abundant in W Africa, and have been the focus of studies, especially Caspian Tern *Hydroprogne caspia* and Royal Tern *Thalasseus maximus*, which present quite large breeding populations in near shore islands of Guinea-Bissau and (at least in the past) in the Bijagós Archipelago (Fig. 9.3). Waterbird counts conducted in the 1990s by Altenburg *et al.* (1992), Quade (1994), Schmanns *et al.* (1997) and Brenninkmeier *et al.* (1998), summarized in Dodman & Sá 2005, reported internationally important breeding colonies of these two species. Outside

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Figure 9.3. Caspian Tern breeding colony on the sand islet of Acapa-Imbone, in Orango National Park, Bijagós Archipelago.

Year	Period of count	Areas covered	Area covered (%)	Count method	Estimation method	Global estimate	Literature source
1982-1983 1986-1987	December-February	Bubaque, Bolama, Bijagós	25	In situ low-tide counts	Density extrapolation	699,120	Poorter & Zwarts 1984 Zwarts 1988
1992-1993	October-May	16 areas in 14 islands	64	In situ low-tide counts	Density extrapolation	710,000	Salvig et al. 1994
1994	December	Full coverage	54	Aerial survey & ground counts	Detection Rate	750,000	Salvig et al. 1997
1994 1995 1997	February-November April-September January-March	Bubaque, Soga	N/A	In situ low-tide counts	Total count	15,000*	Salvig et al. 1997
2001	January-February	44 areas in 18 islands	64	In situ low-tide counts	Density extrapolation	871,750	Frikke et al. 2002 Dodman & Sá 2005
2009	May, July September, December	9 counts in 2 marine protected areas	N/A	In situ low-tide counts	Total count	23,241*	Monteiro 2011
2010	August, October, December	9 counts in 2 marine protected areas	N/A	In situ low-tide counts	Total count	31,092*	Monteiro 2011
2011	August, October, December	9 counts in 2 marine protected areas	N/A	In situ low-tide counts	Total count	17,979*	Monteiro 2011
2014	January	56 sample plots in all Bijagós	69	In situ low-tide counts	Density extrapolation & total count	352,000	van Roomen et al. 2015
2017	January	77 sample plots in all Bijagós	14	In situ low-tide counts	Density extrapolation	231,000	Sa & Regalla 2017
2020	January	62 sample plots in all Bijagós	14	In situ low tide-counts	Density extrapolation	193,000	Sá et al. 2020

Table 9.3. Summary of historical and recent shorebird counts conducted in the Bijagós Archipelago, with distinction between count method, areas covered, estimation method and the source of information. * Counts did not produce a global estimate.

the breeding season, terns, and specifically Royal Terns, are known to disperse along the W African coast from Morocco in the north to Namibia in the south (Veen *et al.* 2015). During the northern winter months, numbers of terns fluctuate markedly due to the arrival of non-breeding birds from other parts of W Africa and from the Northern hemisphere (Brenninkmeijer *et al.* 2002, van Roomen *et al.* 2015, Veen *et al.* 2015). Little is known on Caspian Tern movements within the Bijagós Archipelago or in the region.

Caspian and Royal Tern populations have been increasing in W Africa over the past 20 years (van Roomen *et al.* 2015). However, these trends are to be interpreted with caution, as data are scarce and few studies have been conducted on these species across the region, especially away from their main breeding sites.

9.4 Migratory shorebirds

The Bijagós Archipelago receives annually large numbers of shorebirds of Palearctic breeding origins during their non-breeding period (Alves *et al.* 2021, Dodman & Sá 2005, Zwarts 1988) and it is among the most important wintering sites along the East Atlantic Flyway (EAF) (Delany *et al.* 2009). Shorebird species assemblages are diverse in the Bijagós Archipelago, with the counts conducted during the non-breeding period typically yielding 16 to 19 migratory species. These birds spend the high tide essentially hidden inside the mangrove forests, and are only visible during the low tide, while foraging on the intertidal flats. The extensive nature of these flats makes it challenging to conduct counts and assess the total numbers of shorebirds using them (Fig. 9.4). After trying a number of different approaches, the best results appear to have been


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Figure 9.4. Palearctic migratory shorebirds feeding on intertidal flats partially covered by fiddler crab burrows, in the Bay of Adonga, Orango National Park, Bijagós Archipelago.

obtained by determining shorebird densities during low tide at a number of smaller sampling sites distributed across the entire area, and then extrapolating based on the total surface of available feeding habitat (Zwarts 1988; see Table 9.3). Results so far suggest a rather low density of foraging shorebirds, especially when compared to the numbers and densities in the Banc d'Arguin (Lourenço *et al.* 2016, van de Kam *et al.* 2004, Wolff *et al.* 1993, Zwarts 1988). This may be related to several factors, including a lower density of prey items, but this is currently unknown.

Some studies have unveiled a markedly low density of benthic invertebrates on which shorebirds feed on in the Bijagós Archipelago (Larénie & Anne-Laure 2009, Pedro M. Lourenço *et al.* 2018, Zwarts 1988), which may be structuring the distribution patterns observed in wintering shorebirds. As shorebird prey items are distributed in low densities across the mudflats of the archipelago, shorebirds may also follow these patterns, occurring in low densities during low tide (Lourenço *et al.* 2017). Nonetheless, previous shorebird counts have shown important low tide densities in the intertidal flats around the islands of Bubaque and Soga, as well as in the mudflats around Orango National Park and the Community Marine Protected Area of Urok - two of the three MPAs of the Biosphere Reserve (Dodman *et al.* 2004, Dodman & Sá 2005, Monteiro 2011). Recent ongoing studies have unveiled marked spatial variations in the density and biomass of benthic macroinvertebrates throughout the Bijagós Archipelago's intertidal flats (Coelho *et al.* unpublished data), which may be a process driving differential distributions of shorebird species according to their preferred prey type.

Overview of historical and current counts

Over the last four decades there have been several counts aiming at estimating the total shorebird numbers in the Bijagós Archipelago. The first population estimates were produced in 1982-83 (Zwarts & Poorter 1984 in Dodman & Sá 2005), but only in 1986-87 was the low-tide density extrapolation method established (Zwarts 1988). Since then, two other complete estimates were produced with this method, in 1992 (Salvig *et al.* 1994) and in 2001 (Frikke *et al.* (2002) in Dodman & Sá 2005), with counts using different methods or targeting smaller areas of the Bijagós Archipelago also taking place (Table 9.3). In 2014, 2017 and 2020, total complete estimates based on sample counts were again obtained in the framework of the Wadden Sea Flyway Initiative, Wetlands International and BirdLife International cooperation as a contribution to the International Waterbird Census (IWC; Agblonon *et al.* 2017, van Roomen *et al.* 2020, van Roomen *et al.* 2015). An overview of the data available from historical and recent counts is presented in Table 9.3.

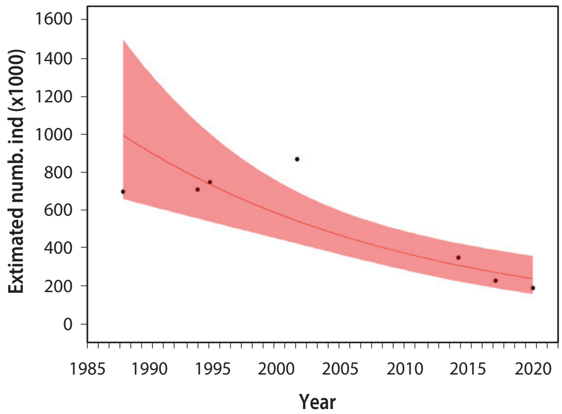

Figure 9.5. Estimates (represented by black dots) of the total number of migratory shorebirds spending the non-breeding period at the Bijagós Archipelago in 1987, 1993, 1994, 2001, 2014, 2017 and 2020 (estimated using different methods, see Table 9.3). Data retrieved from Zwarts *et al.* (1988), Salvig *et al.* (1994, 1997), Frikke *et al.* (2002) and GPC/IBAP/WSFI unpublished data (2014, 2017, 2020, see Van Roomen *et al.* 2021). The trend line was fitted using a generalized linear model with negative binomial distribution and log link. The coloured area around the line represents the 95% confidence interval. The analysis shows a significant decline over time (P <0.001).

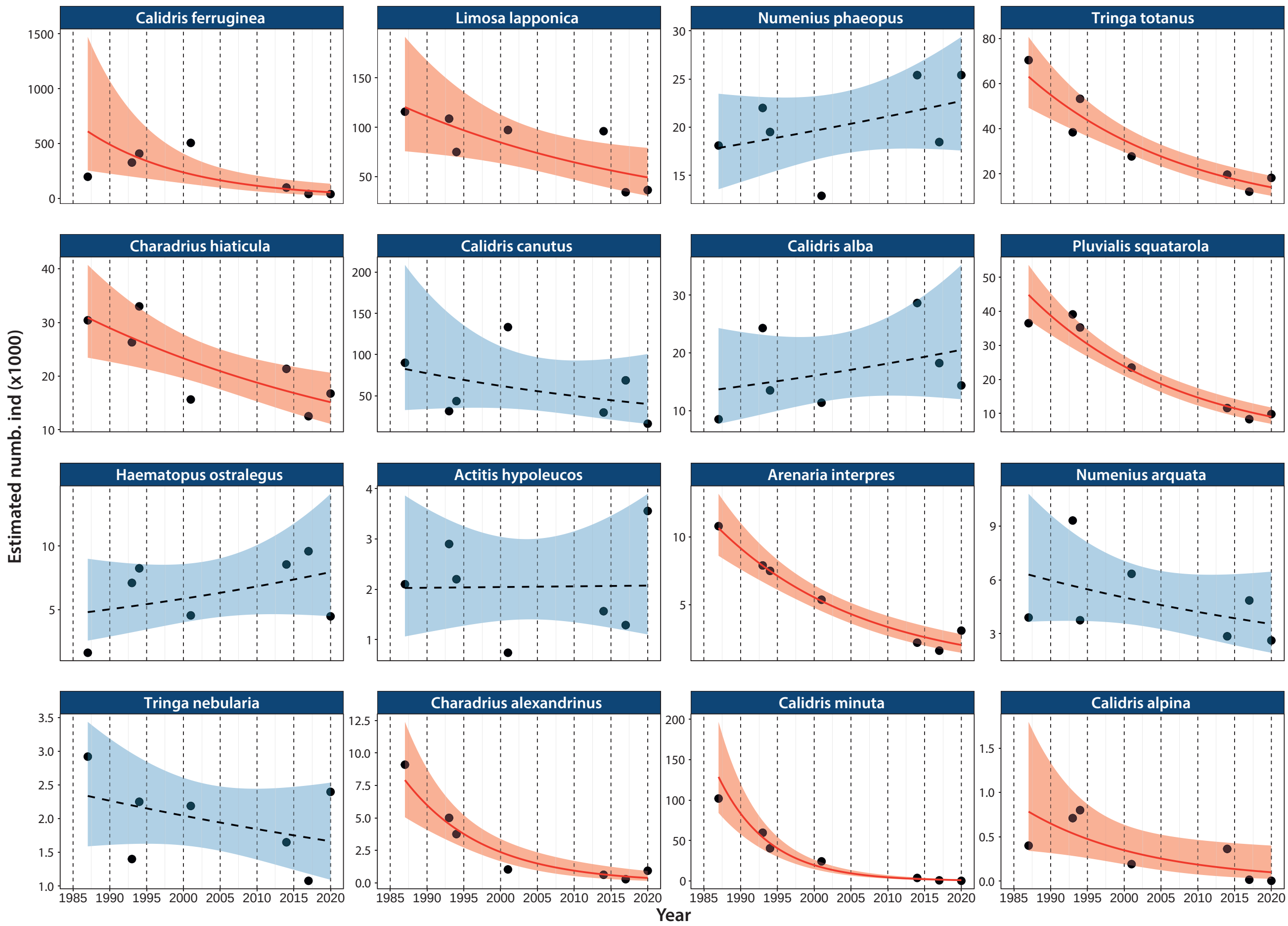

Figure 9.6. Estimates (represented by black circles) of the number of individuals for 16 migratory shorebird species spending the non-breeding period at the Bijagós Archipelago in 1987, 1993, 1994, 2001, 2014, 2017 and 2020 (estimated using different methods, see Table 9.3). Data retrieved from Zwarts *et al.* (1988), Salvig *et al.* (1994, 1997), Frikke *et al.* (2002) and GPC/IBAP/WSFI unpublished data (2014, 2017, 2020, see Van Roomen *et al.* 2021). Graphs are organized in decreasing order from the most abundant species in 2020 (left to right in each row). Trend lines were fitted using generalized linear models with negative binomial distribution and log link. Shaded areas around the line represent the 95% confidence interval. Solid lines and red shading represent species declining significantly; dashed lines and grey shading represent species with non-significant declines or increases (P ≥ 0.05).

Figure 9.7. Shellfish collecting in the intertidal flats of the island of Bubaque, the most populated of the Bijagós Archipelago, Guinea-Bissau. (photo:).

Current population estimates and trends

During the 2020 IWC for Guinea-Bissau, more than 31,000 shorebirds were counted in the sampling area of the Bijagós Archipelago (Sá *et al.* 2020). Estimates of the total number of wintering shorebirds were produced, indicating that about 193,000 migratory shorebirds were using the Bijagós Archipelago in 2020 (van Roomen *et al.* 2021). Previous IWC counts for this area in 2014 and 2017 yielded total estimates of 352,000 and 231,000 respectively. These numbers are strikingly low when compared with the estimates in the 1980s and 1990s for the same sampling area (800,000 – 900,000), highlighting the steep declines observed in this important area (Fig. 9.5). These trends partly agree with those established for the EAF populations, revealing decreases in some Arctic breeding shorebird species (van Roomen *et al.* 2018), and with the declines reported for Banc d'Arguin in Mauritania (Oudman *et al.* 2020). However, some trends appear to be more specific to the Bijagós Archipelago.

At the species level most of the 16 species assessed here presented notable declines, and half of them presented statistically significant negative trends (Fig. 9.6). There was a striking decrease in the number of Curlew Sandpipers *Calidris ferruginea* from 2001 through to 2020, by about 75%. A similar pattern was observed for Little Stint *Calidris minuta*, which decreased by more than half from the 2001 count, and by more than 85% from the first estimate of 1986-87. Other previously abundant species like Grey Plover *Pluvialis squatarola*, Bar-tailed Godwit *Limosa lapponica*, Common Redshank *Tringa totanus* and Ruddy Turnstone *Arenaria interpres* also presented very steep declines. Red Knot *Calidris canutus*, Common Greenshank *Tringa nebularia* and Eurasian Curlew *Numenius arquata* have also decreased, although not statistically significantly. By contrast, there was an apparent increase in Eurasian Whimbrel *Numenius phaeopus*, Sanderling *Calidris alba* and Eurasian Oystercatcher *Haematopus ostralegus* (Fig. 9.6). These data are to be interpreted with caution due however to the difficulties inherent to the survey methods, particularly related to potential errors in estimating the proportion of the area counted by each observer among years and uneven distribution of shorebirds across the mudflat areas of the archipelago. The analysis presented here should therefore be regarded as preliminary. More sophisticated analyses are needed to establish whether the trends are as worrying as we now think they are, and to describe and explore ecological correlates at local and flyway levels.

9.5. Anthropogenic pressures on waterbirds

There are currently very few studies that address the anthropogenic pressures to waterbirds in the Bijagós

Archipelago, and more information is urgently needed. Based upon the on-ground experience, it is fair to consider that anthropogenic pressures within the archipelago appear to be of lower magnitude when compared, for instance, to threats faced by Palearctic migratory waterbirds further north in their distribution range in Europe, due to relatively low levels of human disturbance in the Bijagós Archipelago. However, we highlight here some of the potential threats that must be considered.

Recently, the governments of Guinea-Bissau and Senegal signed a protocol aiming to start oil exploitation within the shared marine areas of both countries (Brownfield & Charpentier 2003). Oil exploitation impacts go beyond the area of effective extraction and may affect ecosystems and biodiversity around it (UNEP-WCMC *et al.* 2007). In this case, oil spills may affect marine and intertidal ecosystems within the Bijagós Archipelago, resulting in the accumulation of hydrocarbons in the sediment and in the marine and intertidal food web. As high-level consumers, waterbirds will be the final recipient of a chain of bioaccumulation, which ultimately may affect their survival and breeding success.

The ancient practices and culture of the Bijagós communities have been responsible for the regulation of access to natural resources, promoting the sustainable use of marine resources (Biai *et al.* 2003, Campredon *et al.* 2010, Rachid *et al.* 2011). However, increasing global interactions are promoting the growth of international trade and exchange of cultural ideas, and Guinea-Bissau and the Bijagós islands are no exception. People are prone to adjust their ways of living, particularly younger generations eager to experience the westernised culture globalised through the media and the internet, which can lead to the substitution of traditional and sustainable use of natural resources for more unstainable practices. For instance, the overexploitation of a shellfish, the Bloody Cockle *Senilia senilis*, by local communities in the intertidal flats of the Island of Bubaque has recently been reported (Fig. 9.7). The main driver behind this is the high demand of Senegalese markets for this type of resource (P. Campredon pers. comm).

Despite its slow growth rate and despite having the potential to be the most beneficial activity in promoting both the development of local communities and the conservation of their natural environment, poorly managed and unregulated tourism may also present a serious potential threat to waterbirds and their habitats in the Bijagós Archipelago. When inadequately planned, tourism may alter the land use among the islands, and ultimately result in habitat loss and degradation, in addition to increased pollution and disturbance levels (especially when considering that sport fishing with fast and noisy boats is among the most explored touristic interests), which may also negatively affect waterbird populations (Davenport & Davenport 2006, Polet 2011). Moreover, because the conservation and protection of the unique ecological traits of waterbird habitats in this archipelago has been promoted by the ancient rules of the local culture (Maretti 2015), unregulated tourism also has the potential to disturb the fragile balance of the cultural structure of the Bijagós communities, jeopardising many centuries of community-based conservation (Ozorio & Lima 2019).

Global warming and human induced climate change may constitute the major threats within the Bijagós. Global warming is accelerating the rate of sea level rise, which negatively affects the availability of foraging habitat for shorebirds (Lourenço *et al.* 2013, Piersma & Lindström 2004, van de Kam *et al.* 2004, Zwarts *et al.* 2009). The mudflats and sandbanks of the Bijagós Archipelago require particular attention as these areas have low elevation and almost no coastal slope (Granadeiro *et al.* 2021). The sea level is predicted to rise by up to 20 cm at the coastline of Guinea-Bissau by 2050 (Republic of Guinea-Bissau 2018), which may have significant impacts on the intertidal area extent and on the distribution of benthic macroinvertebrate prey (Beninger 2018).

The impact of fisheries on waterbirds and on marine and intertidal food webs is difficult to measure and little knowledge is currently available on this topic in the Bijagós Archipelago. Nevertheless, fishing practices are shifting towards more intensive activity, especially for export pur-

Left: **Afonso Rocha**
centre: **Pedro Lourenço**
right: **Mohamed Henriques**

Figure 9.8. Participatory research in the Bijagós Archipelago, Guinea-Bissau.

Caspian Terns | **Sterne caspienne** *(Hydroprogne caspia),* **Black-headed Gulls** | **Mouette rieuse** *(Chroicocephalus ridibundus)* & **Common Terns** | **Sterne pierregarin** *(Sterna hirundo)*

poses by migrant fishermen from neighbouring countries or through Asian and European large industrial fishing boats (Campredon & Catry 2017, Campredon & Cuq 2001, Diop & Dossa 2011). Such activities place increasing pressure on the Bijagós Archipelago marine and intertidal ecosystems (Leurs *et al.* 2021), having led to a clear reduction in the abundance of top predators like sharks and rays including the W African Sawfish *Pristis pristis*, so important to the Bijagós culture as an emblem animal, which had not been observed for a long time until recently (Leeney & Poncelet 2015). These changes can have top-down effects over entire food webs, affecting waterbirds and their prey. Moreover, the establishment of several temporary fishing camps by foreign fishermen has resulted in the cutting down of parts of mangrove forests to smoke fish, weakening coastal protection against erosion. More studies are required to assess the extent to which these apparently less relevant threats may affect waterbirds in the future.

9.6. Research, conservation and management in the Bijagós Archipelago

Conservation actions aimed at waterbirds in the Bijagós Archipelago have been going on since the 1980s. Several national and international organisations have worked together with local communities to create the MPA network in the Bijagós, comprising the Marine National Parks of João Vieira and Poilão, Orango National Park and the Community MPA of Urok. These have been the strongest conservation points in the archipelago through a long-term implementation of a set of conservation measures and regulation of access and use of marine resources, together with local communities. The success of these MPAs in conserving both natural resources, including areas used by waterbirds, and the local culture and identity, which is an important part of the conservation of the Bijagós Archipelago's ecological traits, is deserving of international recognition (Ramsar Convention 2012, UNDP 2019). The management of these protected areas and the implementation of conservation and monitoring activities have been maintained through the combined efforts of public institutions, like the Institute of Biodiversity and Protected Areas of Guinea-Bissau (IBAP), the National Institute for Research (INEP), and the Applied Fishing Research Centre (CIPA), with national and international NGOs, like Tiniguena - Esta Terra É Nossa!, Palmeirinha, ODZH, Noé Conservation and CBD Habitat Foundation, and several international universities.

More recently, IBAP, with the support of the MAVA Foundation, developed several projects in collaboration with European universities and research institutes, under a wide framework termed *Waders of the Bijagós: Securing* *the ecological integrity of the Bijagós Archipelago as a key site for waders along the EAF*. Shorebirds have been the focus of one of these large research projects, in which intertidal ecosystems of the Bijagós are being studied. This important international project has already produced multiple results such as the filling of knowledge gaps on shorebird ecology in the archipelago and on the way that shorebirds actively 'connect' the Bijagós with other parts of the world (Alves *et al.* 2021, Belo 2019, Bom *et al.* 2020, 2021, Carneiro *et al.* 2021, Catry *et al.* 2021, Henriques *et al.* 2021, Pedro Miguel Lourenço *et al.* 2018, Mathijssen 2020, Meijer *et al.* 2021, Parente 2020, Paulino *et al.* 2021) and on key factors regarding the hydrology and intertidal sediment (e.g. Granadeiro *et al.* 2021), as well as raising awareness and training of local community members.

Before this large project, other research initiatives, led by researchers from the University of Lisbon, had already started to address some of the key research questions, with important findings being published (Catry *et al.* 2017, 2016, Coelho *et al.* 2016, Lourenço *et al.* 2017, Lourenço *et al.* 2017, Lourenço *et al.* 2018). Another MAVA Foundation funded project entitled *La recherche participative au service de la conservation de la biodiversité du Parc National Marin de João Vieira-Poilão (Archipel des Bijagós)*, led by IBAP and ISPA from Portugal, had also achieved high success, with many key publications concerning terns, coastal fish and predator prey interactions (Carneiro *et al.* 2017, Correia *et al.* 2021, 2019, 2018, 2017, Correia 2018). It is of utmost importance that the ecological research results are used together with the monitoring efforts deployed during the IWC to strengthen and inform conservation, while it is also vital that efforts be made to secure the collection of relevant data on the long-term. The world is changing fast and the only way to try to cope is to know what is ecologically changing and why. Monitoring programmes for mangroves, intertidal flats, benthos, fish and birds need to be combined with focussed research that includes local participation (Fig. 9.7), so that knowledge, work, training and outreach go hand in hand. Research, monitoring and conservation are the three pillars on which simultaneous investments must be made to support the protection of sites of unique ecological value, like the Bijagós Archipelago.

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Harvey van Diek

Ruddy Turnstones | **Tournepierre à collier** *(Arenaria interpres) &* **Grey Plovers** | **Pluvier argenté** *(Pluvialis squatarola)*

Cattle Egret | **Héron garde-boeufs** (Bubulcus ibis) (Wil Leurs / Agami)

10. Pressures on coastal wetlands and waterbirds in Lagos State, Nigeria

Esther Nosazeogie

Summary

Whilst the coastal wetlands in Lagos State, Nigeria, support a range of habitats and a diversity of waterbirds, they are also heavily utilised by people and face a variety of threats. A number of activities exert direct pressure on wetlands, resulting in habitat loss, modification and pollution. These include indiscriminate plastic waste dumping, land reclamation, mangrove deforestation, sand dredging, fisheries and transport. Some major industries such as oil refineries are also present in the area – sources of industrial waste. Many of these activities exert pressures on the waterbirds of Lagos State. Conservation education is key to building awareness of wetland values and environmental stewardship. The Nigerian Bird Atlas project has helped to promote the development of bird clubs, which have important roles to play in citizen science and awareness raising. More regular monitoring of sites is also needed, both of birds and pressures on wetlands, which should guide future management. There are currently very limited conservation management measures in place; threats are likely to expand unless this situation is reversed.

Résumé

Alors que les zones humides côtières de l'État de Lagos, au Nigeria, abritent une variété d'habitats et une diversité d'oiseaux d'eau, elles sont également fortement exploitées par l'homme et font face à diverses menaces. Un certain nombre d'activités exercent une pression directe sur les zones humides, entraînant la perte, la modification et la pollution des habitats. Il s'agit notamment du déversement inconsidéré de déchets plastiques, de la mise en valeur des terres, de la déforestation des mangroves, du dragage du sable, de la pêche et des transports. Certaines grandes industries telles que les raffineries de pétrole sont également présentes dans la zone - sources de déchets industriels. Nombre de ces activités exercent des pressions sur les oiseaux d'eau de l'État de Lagos. L'éducation à la conservation est essentielle pour sensibiliser le public aux valeurs des zones humides et à la gestion de l'environnement. Le projet d'Atlas des oiseaux du Nigéria a contribué à promouvoir le développement de clubs d'oiseaux, qui ont un rôle important à jouer dans la science citoyenne et la sensibilisation. Un suivi plus régulier des sites est également nécessaire, tant pour les oiseaux que pour les pressions exercées sur les zones humides, ce qui devrait guider la gestion future. Les mesures de gestion de la conservation sont actuellement très limitées ; les menaces sont susceptibles de s'étendre si cette situation n'est pas redressée.

10.1 Introduction

The Nigerian coastal zone cuts across eight states: Akwa-Ibom, Bayelsa, Cross-River, Delta, Edo, Lagos, Ogun, Ondo and Rivers States. However, information about waterbirds from this coastal region is readily available only for Lagos in the southwest and Cross River and Akwa-Ibom in the southeast (e.g. Onoja 2020). The habitats such as mudflats, mangroves and sandy shores provided by this coastal zone are particularly important for shorebirds and other waterbirds. However, in Nigeria, the value of these habitats for birds is given little attention. Additionally, as is increasingly the case in many other parts of the world, these coastal wetlands are also hubs of urban concentration causing high pressures on the natural wetland habitats.

Key sites for waterbirds that have been identified in this zone include the Lagos Lagoon complex in Lagos, Ibeno beach and Mkpat Enin in Akwa-Ibom and Itu wetland between Cross River and Akwa-Ibom (Uwatt *et al.* 2018). The Lagos metropolis in particular is marked by the presence of several waterbodies: lagoons, creeks and other wetlands, including Lagos Lagoon, Lekki Lagoon, Badagry Creek, Yewa Creek and Ologe Lagoon. In Lagos, wetlands are continually being reclaimed for development without regard to their value to many resident waterbird species, as well as to non-breeding migratory shorebirds. The status of Lagos as the most populated state in Nigeria and the economic hub of the country (Obiefuna *et al.* 2012) is problematic for wetlands. Many of the natural wetlands in Lagos have been destroyed, and others continue to be destroyed in order to make space for human habitation and development.

Some surveys have been undertaken to collate information on the status of waterbirds and their wetland habitats in Lagos and Cross River States in Nigeria (e.g. Onoja 2020, Nosazeogie 2021). As part of the International Waterbird Census (IWC) in January 2020, bird surveys were carried out in Cross River and Lagos states (Onoja 2020) through line transects from a boat. 2,328 birds were recorded in Lagos state, the most numerous species being White-faced Whistling Duck *Dendrocygna viduata* and African Jacana *Actophilornis africana*, although numbers were

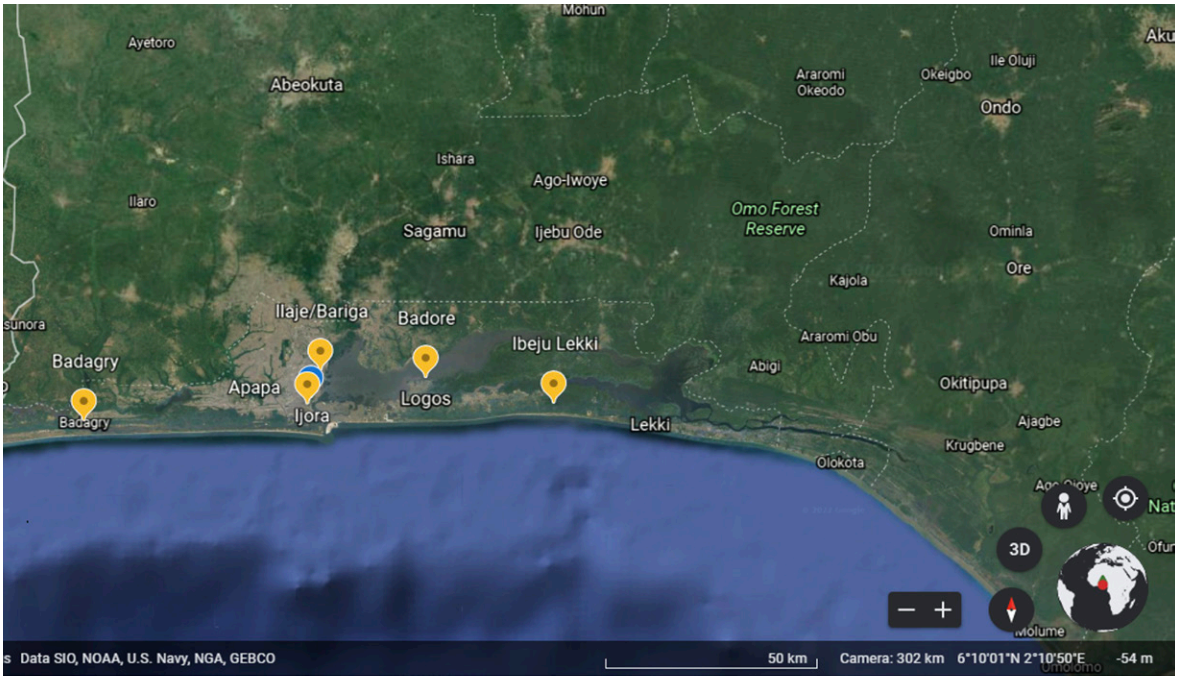

Figure 10.1. Sites surveyed around Lagos in February 2021.

much lower than in the January 2017 count (Onoja 2017). Onoja (2020) considered that frequent dredging could be contributing to waterbird declines. These form part of a monitoring scheme, which contributes to developing waterbird conservation recommendations.

10.2 Sites surveyed around Lagos, February 2021

In February 2021, five locations within the Lagos metropolis were surveyed for their habitats, human use and pressures, whilst birds were also recorded along each route from a boat.

Badagry Creek

This creek, though more than 50 km from the main Lagos metropolis, empties into Lagos Harbour. It is an important part of the drainage system (lagoons and creeks) that flows into the Atlantic Ocean through Lagos Harbour. It also extends to Porto Novo in Benin Republic, and receives tidal influences from both the Lagos and Cotonou Harbours (Akintola *et al.* 2011). Key habitats include open water, grass-covered swamps and vegetation including oil palm, raphia palm and coconut palm, and floating weeds such as *Eicchornia sp*., *Pistia sp*. and *Ceratophyllum sp*. The area supports a diversity of herons and storks (Okosodo *et al.* 2018).

Local communities of Badagry predominantly carry out artisanal fishing, transportation services (commercial boating) and aquaculture. Commercial sand dredging occurs in the creek, although there is some local regulation, such that people are prohibited by the *Baale* (traditional ruler) from dredging in some areas. The people of Badagry consider their creek to have special historical and cultural significance, which seems to shape their use of the waterbody.

More extensive surveys of the whole area are needed, including along the beaches. This knowledge would be useful for conservation education, which would enable locals to have a stronger sense of the value of their environment and sustainable use. Local understanding of the area's biodiversity and its value will also be useful when decisions need to be made by community leaders about economic/ development pressures encroaching from the nearby Lagos metropolis.

Bariga / Ilaje wetland

This wetland is directly under the Third Mainland Bridge. Here, both artisanal and commercial sand dredging activities occur, as well as artisanal fishing. In addition to open water, this area is marked by several tidal mudflats with scanty vegetation, mostly grass. It is managed by community chiefs led by the *Baale*. The greatest environmental pressure observed was plastic pollution. The area is like a huge dumpsite, full of massive piles of plastic and surrounded by slums. As well as posing a threat to health of the local population, there is a concern that the plastic waste may cover valuable feeding areas on the mudflats and reduce the amount of food available to birds. The most numerous birds recorded at Bariga/Ilaje in February 2021 were Little Egret *Egretta garzetta* and Long-tailed Cormorant *Phalacrocorax africanus*, whilst nine Black Herons *Egretta ardesiaca* were also found.

Lekki Lagoon

The Lekki Lagoon, accessed through Ibeju-Lekki, drains directly into Lagos Lagoon in the west. Key habitats at Lekki include open water, floating vegetation, swamp forest (containing mangrove trees), palm trees and grasses - similar to those at Badagry. Artisanal fishing seems to be the major activity in this area. Transportation services (commercial boating) are not developed here, as there is no commercial jetty (as in Badagry), and wooden (fishing) boats fitted with outboard engines were the only available means of transportation. Commercial sand dredging is a common feature in Lekki Lagoon, and does not appear to be regulated. There is a huge demand for sand due to the many building projects that surround the lagoon. Logging is also likely driven by the demand for wood for building projects and domestic use for firewood.

The Ibeju-Lekki area is experiencing intense development, including the creation of a sea port and a petroleum refinery, as well as housing developments due to economic opportunities. The area appears to be managed directly by local community leaders, and indirectly by the government, which has approved the development activities. Greater efforts are needed to encourage the regulation of sand-dredging and development activities.

Lagos Lagoon (between Badore/ Ikorodu)

This part of Lagos Lagoon was accessed through Badore in Ajah, but it is also connected with some parts of Ikorodu. The major habitats include open water and mudflats which are mostly vegetated by grasses. We encountered a flock of 148 White-faced Whistling-duck *Dendrocygna viduata* roosting on the bare part of one of the mudflats. The wetland is surrounded by swamp forests containing mangrove vegetation. This area is also controlled by local community leaders, although there do not appear to be any regulatory management practices in place. Activities in this area included fishing and dredging, as well as developed commercial boat transportation.

Ijora / Apapa

Ijora and Apapa are in close proximity to each other. Ijora Creek is a shallow tidal creek on the Lagos mainland carrying water directly into Lagos Harbour, from where it receives saltwater from the Atlantic Ocean at high tide. The Apapa area is located around the mouth of Lagos Harbour, where Lagos Lagoon empties into the sea, and consists of a collection of islands and creeks. Key habitats in both areas are tidal mudflats, which are rich in visible benthic shells and crabs, sandy vegetated shores (at Apapa), mangrove trees and shallow open water.

The Ijora wetland is surrounded by factories and oil depots, as well as homes, which are sources of indiscriminate disposal of both municipal and industrial waste (Ogungbile *et al.* 2017). Battery chargers and car mechanics dispose of their waste into the wetland. Also, it is quite common to see people defecate openly in these areas. Apapa is home to a container terminal, the Lagos Port Complex, as well as several factories and refineries. There

Esther Nosageozie

Bariga / Ilaje wetland

Esther Nosageozie

African Jacana flying over open water at Lekki Lagoon

are also a number of housing settlements (mostly slums) on its islands. In addition to the industrial activities going on in both areas, fishing is also commonly carried out. The most numerous waterbirds recorded were Cattle Egret *Bubulcus ibis* and Common Sandpiper *Actitis hypoleucos.*

10. 4 Threats and pressures

Based on observations during the surveys, the following threats to urban coastal wetlands and waterbirds which depend on them were observed:

Habitat loss

Several ongoing activities in Nigeria's coastal zone may threaten waterbird survival in the long run by removing the habitats that they need to survive. Activities include indiscriminate plastic waste disposal, land reclamation, dredging, beach erosion and mangrove deforestation. Plastic pollution may also cover available mudflats, reducing or totally eliminating the area available to shorebirds for feeding by burying wetland vegetation and preventing benthos from thriving. Additionally, plastic waste may release harmful chemicals during decomposition that may affect the suitability of the soil or mud for bird prey.

Land reclamation, dredging and beach erosion have serious implications for benthic-dwelling species, which many shorebirds (particularly waders) feed on. Similarly, mangrove trees provide important resources for waterbirds, as food sources and roosts. However, a huge amount of mangrove forests has been lost in Lagos (Obiefuna *et al.* 2012), and deforestation is ongoing as development drives the demand for resources. Vegetation and forest loss through logging can cause sedimentation and affect many species of birds.

Habitat modification and deterioration

Activities such as boat transportation and repair, aquaculture, untreated municipal and industrial waste, and grazing of domestic animals, which are commonplace in urban wetlands could reduce the value of the coastal wetland habitats for waterbirds. Boat transportation is an important alternative to the traffic-laden road transportation within Lagos in particular. This usually takes place via the Lagos Lagoon and the creeks that connect to it. The boat operators indiscriminately dump oils and grease into the water during travel or boat repair. This could contribute to pollution in the long run, and reduce the value of the wetlands as waterbird habitat.

Similarly, untreated or improperly treated waste disposal could compromise water and sediment quality in the long run. Open defecation was also commonplace in the areas visited, and domestic waste is often disposed of untreated into wetlands. At Ijora in Lagos, we observed waste water being discharged from a flour mill; we do not know how much treatment the discharged water received, but Nigeria is not known for strict enforcement of environmental laws. In some areas, Water Hyacinth *Eichhornia crassipes* flourishes; this invasive aquatic weed can choke waterways and impact biodiversity.

Fisheries and aquaculture

Fishing is widespread in the wetlands within and around Lagos, whilst there are several aquaculture developments. Although it is yet to spread fully to all areas, aquaculture was observed in Badagry. There are various potential environmental impacts related to fisheries and aquaculture, including the indiscriminate disposal of fishing nets, lines and traps. Aquaculture waste may contain antibiotics, which are emerging pollutants (Milic *et al.* 2012). Such issues, including bycatch, may well pose a threat to waterbirds in the wetlands around Lagos. However, further research is needed to assess the pressures from fisheries.

10. 5 Conservation actions through bird clubs and research

It is difficult to directly address the threats and pressures faced by coastal waterbirds in urban areas by conservation measures without fully understanding the problems first. However, through the work of the Nigerian Bird Atlas Project (Ivande *et al.* 2016), bird clubs are springing up all around Nigeria. In the coastal zone, there are bird clubs in Lagos, Ogun, Akwa-Ibom and Cross River States. These bird atlassing efforts are contributing bird data, with which we may be able to assess the problems more fully. In addition, a research project is underway to understand waterbird communities and their habitat associations in urban Lagos, as well as to raise awareness for the value of wetlands in this area (Nosazeogie 2021). However, there is much more that needs to be done towards waterbird and wetland conservation in the Nigerian coastal zone.

10.6 Recommendations

In all areas, more extensive and regular surveys are needed to improve knowledge about the extent of pressures and threats to waterbirds and to document the birdlife present throughout the year. Local community education is vital to promote the sustainable management of wetlands, including for the benefit of waterbirds. Key recommendations include:

Education: First of all, we need to understand the values of wetlands to the indigenous communities that depend on them, and then come up with ways to keep the natural values of urban wetlands alive in the minds of local people. The annual IWC events could be used as an opportunity for local awareness campaigns focused on coastal communities, to promote environmental stewardship.

Monitoring: The current monitoring efforts through the IWC need to be expanded and made more consistent in order for trends to be detected, which may inform conservation action. More key sites need to be identified, especially in parts of the coastal zone lacking information. In addition, a programme in which the demography and habitat characteristics of the birds (including plant, benthos and fish as food and habitat sources) could be monitored would be highly beneficial. Increased effort is also needed to monitor and assess wetland pressures.

Management: At present, there are only a handful of Important Bird and Biodiversity Areas (IBAs) in the Nigerian coastal zone. No areas visited in February 2021 were subject to any conservation management at the time of the surveys. There are many other more remote areas which have not been surveyed, and some may prove to be important for birds. Threats to waterbirds and their habitats in coastal Nigeria are likely to expand through the consequences of unsustainable development. Therefore, more surveys are necessary to identify key areas for monitoring and to guide future conservation management.

Esther Nosageozie

Artisanal sand dredging at Bariga

Cape Cormorants | **Cormoran du Cap** (Phalacrocorax capensis) (Harvey van Diek)

11. Waterbirds along the Angola Coastline and their key pressures

Miguel Xavier

Summary

Angola's long coastline of over 1,650 km supports a range of ecosystems from tropical mangroves in the north to an arid coastal bet in the south. Waterbird surveys in the coastal zone have been carried out erratically in the past but with more focused attention since 2016, extending to the southernmost stretch since 2017. The most numerous species recorded in 2017 and 2020 was Cape Cormorant *Phalacrocorax capensis*, especially at the Cunene River mouth, which borders Namibia. Several coastal wetlands are under significant pressure in Angola, especially from urbanisation and urban waste and illegal occupation of sites, including protected areas. Waterbirds are also prone to disturbance, including from fishing and collection of shellfish. It is important to build environmental awareness in the coastal zone, establish small-scale projects for removal of urban waste, initiate wetland or waterbird centres and undertake site restoration, including at the Mangal do Lobito.

Résumé

Le grand littoral angolais est long de plus de 1 650 km et abrite une variété d'écosystèmes allant des mangroves tropicales dans le nord à un tapis côtier aride dans le sud. Les enquêtes sur les oiseaux d'eau dans la zone côtière ont été menées de manière irrégulière dans le passé, mais avec une attention plus ciblée depuis 2016, s'étendant à la partie la plus au sud depuis 2017. L'espèce la plus nombreuse enregistrée en 2017 et 2020 était le cormoran du Cap Phalacrocorax capensis*, en particulier à l'embouchure de la rivière Cunene, qui fait frontière avec la Namibie. Plusieurs zones humides côtières subissent une pression importante en Angola, notamment en raison de l'urbanisation et des déchets urbains et de l'occupation illégale des sites, y compris des zones protégées. Les oiseaux d'eau sont également susceptibles d'être perturbés, notamment par la pêche et la collecte de coquillages. Il est important de sensibiliser sur l'environnement des zones côtières, d'établir des projets à petite échelle pour l'élimination des déchets urbains, d'initier des centres relatifs aux zones humides ou aux oiseaux d'eau et d'entreprendre la restauration des sites, notamment au Mangal do Lobito.*

11.1 Introduction

Since 2016 Angola has been making efforts to identify, monitor and conserve waterbirds and their sites along its coastline, leading to knowledge of the most important sites for waterbirds, as well as the realization of some projects to mitigate the impact of human activities, such as the removal of urban waste at the Ilhéu dos Pássaros Integral Reserve and at Mangal do Lobito.

The results obtained during periodic waterbird counts and the survey of data from each site has led to a better understanding of the existing habitats, pressures and threats, enabling the development of some environmental awareness programmes and conservation projects, mostly financed by the Wadden Sea Flyway Initiative. These have had a very positive impact, involving both the authorities and surrounding communities.

Although the January 2020 waterbird counts saw prospects of reaching new count sites in Zaire province, they were most difficult for the Angolan team, as tragically our colleague Maria Eugênia Lopes (Jeni) died after an accident suffered by the team in southern Angola (see Kodo *et al.* 2020).

11.2 Overview of the Angolan coastline and its importance for waterbirds

The Angolan coastline extends over 1,650 km, from the Kissi River in Cabinda in the north to the mouth of the Cunene River in Moçâmedes in the south (Fig. 11.1). Along this coastline there are several types of ecosystems, including the mangroves and lagoon ecosystems that offer important habitats for waterbirds, both resident and migratory. The Angolan coastline offers favourable conditions for different groups of waterbirds, including waders, cormorants, herons, terns, ducks and other groups, which utilise a range of wetland habitats. Important habitats for waterbirds include estuaries and coastal lagoons.

The work carried out has consisted of surveying waterbird sites along the entire Angolan coastline to identify the most important sites. During coastal surveys, 21 important waterbird sites have been identified, distributed in seven provinces of Angola, namely, Cabinda, Zaire, Bengo, Luanda, Cuanza Sul, Benguela and Moçâmedes (Fig. 11.1).

This network of sites was monitored for three years to better understand their functioning and the population dynamics of waterbirds, and to identify the most important

Figure 11.1. Waterbird count sites in coastal Angola.

sites. Based on the results obtained, notably of population density, occurrence of migratory waterbirds and ecological conditions, three critical sites were identified: the Integral Reserve of Ilhéu dos Pássaros (Luanda), the Saco dos Flamingos (Luanda) and the Baia dos Tigres (Moçâmedes). Mangal do Lobito (Benguela) also has great potential but is in need of further surveys.

11.3 Waterbird count in January 2020 compared to previous counts

During the 2020 waterbird census it was possible to cover seven sites, including two new ones in the province of Zaire. In terms of waterbird numbers, the 2020 records were significantly higher than in 2016 and 2017, whilst bird diversity at the different count sites was also higher (Xavier 2017; Kodo *et al.* 2020). Black-winged Pratincole *Glareola nordmanni* was recorded in 2020 for the first time during the January counts.

Count results from 2016, 2017 and 2020 for selected species are shown in Fig. 11.2. Results in 2017 and 2020 were higher than previous years due mainly to greater coverage, especially in southern Angola. Total numbers in both 2017 and 2020 were influenced significantly by the most numerous species, Cape Cormorant *Phalacrocorax capensis*: 109,275 were recorded in 2020 at the Cunene River mouth - a significant count for this globally endangered species endemic to coastal Southern Africa, whilst over 18,283 were counted in 2017.

2020 also saw the highest count of Greater Flamingo *Phoenicopterus roseus*, with about 2,600 individuals at four sites - a high number and good representation in relation to past counts. There was also a significant number of Kelp Gulls *Larus dominicanus* recorded in 2017, mainly from the south. Counts of most waders and other groups did not differ too much from past counts, although numbers of Sanderling *Calidris alba* were higher in 2020, with around 2,000 individuals counted. Different species showed different trends between counts (Fig. 11.2).

11.4 Pressures and threats to waterbird habitats

Despite the efforts made in recent years, there are still great pressures and threats to waterbird habitats in all sites along the Angolan coast. These differ from site to site depending on the types of human activity around the site. The principal pressures at four key sites for waterbirds are shown in Table 1. However, in general, all sites are under the following pressures and threats, with some intensity:

Urban Waste: This is one of the main threats to waterbirds all along the coast. The case of the Integral Reserve of Ilhéu dos Pássaros is the most illustrative of this situation. Most sites are located close to urban towns, which produce garbage that generally ends up invading the sites, endangering most species that frequent the natural environments.

Site	Type of pressure
Ilhéu dos Pássaros	Fishing, shellfish collection, illegal occupation by people, urban waste
Saco dos Flamingos	Fishing, illegal occupation by people
Baia dos Tigres	Fishing, illegal occupation by people
Mangal de Caponte	Urban waste, illegal occupation by people, waste water drainage

Table 11.1. The main pressures to waterbirds at four different sites.

Fishing and shellfish collection: Artisanal fishing practiced in the coastal zone and shellfish collection are two activities that have caused high pressure on natural waterbird sites and represent a direct threat to coastal wetland environments. In almost every place there are individuals practising both artisanal fishing and shellfish collection. Both activities involve frequent movements that disturb waterbirds.

Illegal occupation: In recent years, invasion of waterbird sites has been observed in most areas, including illegal occupation of key waterbird habitats. The greatest pressures and threats recorded exist in the Ilhéu dos Pássaros Reserve, Saco dos Flamingos and Baia dos Tigres. Communities that have illegally occupied these sites have tended to increase in recent years, posing a danger to waterbird habitats through site degradation and disturbance.

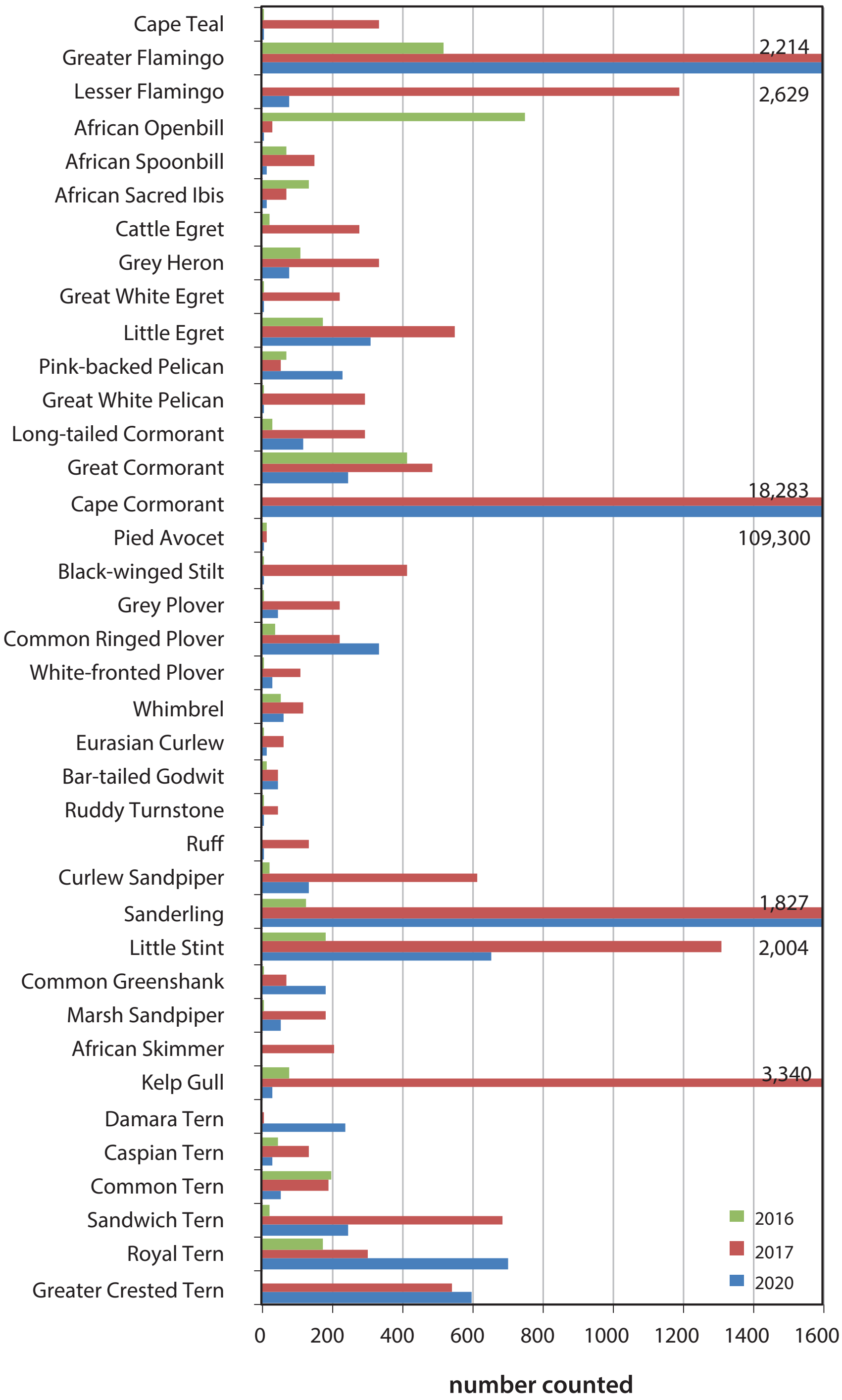

Figure 11.2. The most abundant species recorded during counts of 2016, 2017 and 2020.

Miguel Xavier

Collection of shellfish is one of the human pressures to waterbirds at Ilhéu dos Pássaros

Miguel Xavier

Mangal do Lobito is surrounded by urban development

Miguel Xavier

Black Egrets | **Aigrette ardoisée** *(Egretta ardesiaca)* **among urban waste at Mangal do Lobito**

11.5 Conservation measures and other recommendations

Since 2016, serious efforts have been made to identify waterbird sites along the Angolan coastline, and consequently develop mechanisms for their conservation. Selected sites were monitored for three years to identify the most important sites, for which efforts were made to reduce human impact through environmental awareness and small-scale projects, such as cleaning sites of and removing urban waste.

The efforts undertaken so far have significantly helped to improve the knowledge and conservation of the identified sites. However, it is necessary to continue to maintain the results obtained so far and, in a future perspective, to improve the conservation of these sites. As such, the following steps are important to capitalize on the preliminary results:

Ongoing financial support: This is essential for maintaining environmental awareness programmes and small-scale projects to remove urban waste from the most important sites.

Construction of a small waterbird centre (or centres): The idea is to build a small centre where interested parties can come to learn about waterbirds, their habitats and their migration routes.

Recovering the Mangal do Lobito site: The Mangal do Lobito has great potential for both resident and migratory waterbirds. However, considerable support is needed to lessen the impact of human activities to reverse the current state of degradation.

Miguel Xavier

Waders, Great Cormorant and Greater Flamingo feeding at Ilhéu dos Pássaros

Sanderling | **Bécasseau sanderling** (*Calidris alba*) (Arnold Meijer / Blue Robin)

12. Demographic monitoring along the East-Atlantic Flyway: a case study on Sanderlings using international citizen science

Jeroen Reneerkens

Summary

The size of waterbird populations continuously change. Counts of waterbirds describe these changes. If long-term and/or steep declines are detected, this should signal the need for conservation measures. However, conservation actions will only be effective if they tackle or mitigate threats that negatively impact population growth at the scale of the flyway. Identifying where and when in the annual cycle of a flyway population declines are caused could be a first step towards identifying which ecological factor is responsible for the decline and how this could be reversed. This entails demographic monitoring, i.e. the investigation of spatial and temporal variation in survival and reproduction. This is exemplified with a case study in which observations of individually colour-marked Sanderlings *Calidris alba* along the coasts of Europe and Africa were used to estimate temporal and spatial variation in the probabilities of annual adult survival and the age of first reproduction. Combined with estimates of clutch survival from the Greenlandic breeding grounds, it could be shown that the growth of the Sanderling flyway-population is currently limited by annual variation in clutch survival and adult survival in W Africa. Despite its potential to effectively target conservation action, demographic monitoring is not the standard practice and we are often in the dark about the causes of population change. Increased and continued long-term and flyway-wide efforts to monitor survival and reproduction of waterbird populations could considerably improve this situation.

Résumé

La taille des populations d'oiseaux d'eau change continuellement. Les comptages d'oiseaux d'eau décrivent ces changements. Si des déclins à long terme et/ou importants sont détectés, cela devrait signaler la nécessité de prendre des mesures de conservation. Toutefois, les mesures de conservation ne seront efficaces que si elles s'attaquent aux menaces qui ont un impact négatif sur la croissance des populations à l'échelle de la voie de migration ou si elles les atténuent. Identifier où et quand, dans le cycle annuel d'une voie de migration, les déclins de population sont causés pourrait être un premier pas vers l'identification du facteur écologique responsable du déclin et de la manière dont il pourrait être inversé. Cela implique un suivi démographique, c'est-à-dire l'étude des variations spatiales et temporelles de la survie et de la reproduction. Ceci est illustré par une étude de cas dans laquelle des observations de Bécasseaux sanderlings Calidris alba *marqués individuellement par une couleur le long des côtes d'Europe et d'Afrique ont été utilisées pour estimer la variation temporelle et spatiale des probabilités de survie annuelle des adultes et l'âge de la première reproduction. En combinaison avec les estimations de la survie des pontes dans les zones de reproduction du Groenland, il a été démontré que la croissance de la population de bécasseaux sanderling est actuellement limitée par la variation annuelle de la survie des pontes et des adultes en Afrique de l'Ouest. Malgré son potentiel pour cibler efficacement les actions de conservation, le suivi démographique n'est pas la pratique standard et nous sommes souvent dans l'ignorance des causes des changements de population. Des efforts accrus et continus à long terme et à l'échelle de la voie de migration pour suivre la survie et la reproduction des populations d'oiseaux d'eau pourraient considérablement améliorer cette situation.*

12.1 Introduction

The coordination and standardised implementation of international counts of waterbirds is a logistical challenge. Fortunately, along the East Atlantic Flyway (EAF) this coordination and implementation is well taken care of. The results of the most recent counts are presented in this report. Knowledge of the numbers and trends of bird populations is valuable to keep track of how these populations are faring, whether the numbers are stable, increasing or decreasing. However, only the realisation that populations change and the rate at which they do, does not inform us about the cause(s) of these changes. This is a problem, because only if these causes are known it becomes possible to counteract or mitigate them. In other words, a diagnosis of the ecological and demographic mechanisms of population changes is essential for an effective management strategy (Robinson *et al.* 2005). If the international counts indicate that the population size of a waterbird species is declining along the EAF, what could be done to halt this decline? To answer that question, we first need to identify the cause of the decline. Pinpointing where and when in

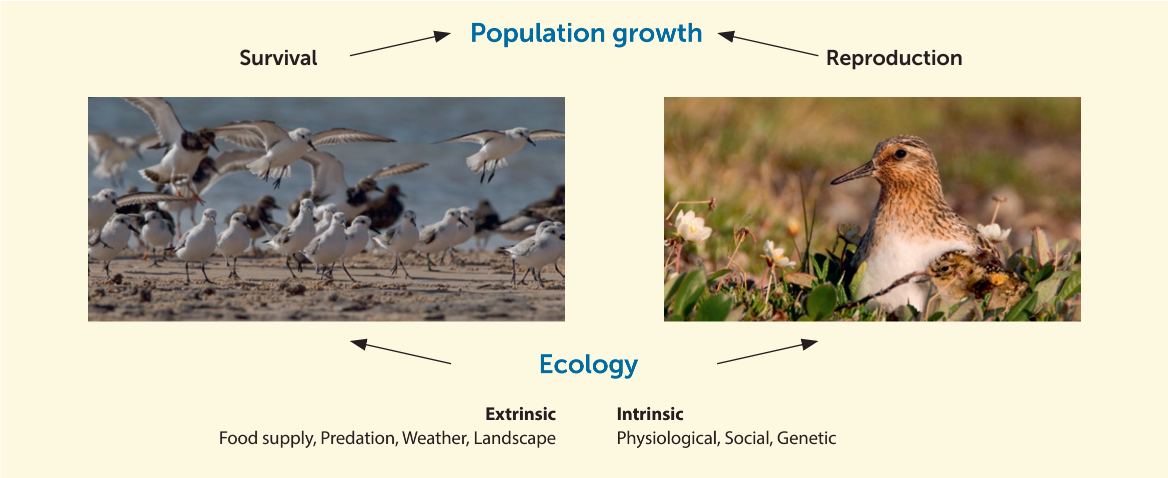

Figure 12.1. Flyway population change is mostly determined by the combined effect of the survival of individuals and the number of young they produce. Both survival and reproduction are influenced by extrinsic and intrinsic factors, of which a few examples are given.

the annual cycle these declines are being caused is a useful first step towards identifying the underlying ecological causes. To identify these, we need a basic understanding of three aspects:

(1) Knowledge of the biogeography; where and when does (a population of) a species breed and winter, and which staging sites are used during migration?
(2) Basic ecological understanding of the species, such as knowledge of their diet in different seasons and sites along the flyway, the quantity and quality of the available food and which anthropogenic pressures the birds suffer from (e.g., land reclamation, fisheries, climate change, hunting).
(3) An insight into the demographic factors affecting the population. This concerns details about the spatial and temporal variation in survival and reproduction of the population.

We generally have a good understanding of the biogeography of most waterbird species along the EAF (Scott & Rose 1996, Engelmoer & Roselaar 1998, Delany *et al.* 2009), although current developments in the technology of lightweight tracking devices and molecular techniques are rapidly leading to new insights (e.g. Bridge *et al.* 2011, Bom *et al.* 2021). The world is continuously changing, and bird populations are affected by, and respond to, such changes in their environment. Thus, the understanding of the ecology and distribution of waterbird species needs to be continuously updated.

To understand the changes in population sizes, we need to understand what demographic components are related to these changes. Long-term changes in survival or productivity may be evident before changes in population numbers occur and signal conservation need (Piersma & Lindström 2004). Unfortunately, it is not standard practice to estimate survival and/or reproductive success in waterbird populations. Hence, conservation actions are often not based on a demographic evaluation. Consequently, significant gaps remain in our understanding of bird population trends, as exemplified in this report.

There are three main demographic components that determine population trajectories: *births*, *deaths*, and *movements* (immigration and emigration). Although redistribution between flyways may (partially) explain changes in flyway populations (Rakhimberdiev *et al.* 2011, see also Chapter 2)thousands of northward migrating ruffs (Philomachus pugnax, flyway populations are usually considered 'closed' and thus mostly affected by the combination of reproduction ('births') and survival (the counterpart of 'deaths') (Fig. 12.1). Reproduction and survival may vary between years and seasons, migration trajectories and locations, and between individuals. A combination or interaction of these factors is also possible. In Eurasian Spoonbills *Platalea leucorodia* for example, particularly older males with longer migration distances breed later and consequently produce fewer chicks that survive until adulthood (Lok *et al.* 2017). Temporal and spatial variation in demographic components can be caused by intrinsic or extrinsic factors. Examples of intrinsic factors are an individual's sex, age, social status, physiological condition or genetic make-up. Examples of extrinsic factors are predation, food supply, disease, weather conditions and habitat (Fig. 12.1). Extrinsic factors are usually of interest to conservationists because they can most easily be managed.

Population changes are most effectively diagnosed by demographic studies that can identify whether survival or reproduction are limiting the population growth and moreover, when in the annual cycle and thus where along the flyway this occurs. This will generate hypotheses about the ecological factors that can be tested locally and preferably should lead to measurable conservation actions. The alternative – often followed – approach is to test at a local scale which ecological factors affect (components of) survival and/or reproduction. However, this approach may not always result in the desired impact on the flyway population growth since local effects may be buffered or counteracted by other ecological factors elsewhere along a flyway.

It is of great importance that governments, nature conservation organisations and other funding agencies invest in demographic knowledge and studies to ensure effective conservation. However, it is not an easy task to identify the factors that regulate changes in population sizes (Weiser *et al.* 2020). This is especially true for species with vast geographic distributions. Here, I will describe a citizen science project on Sanderlings *Calidris alba*, showing that with international collaboration it is possible to diagnose what regulates a population, even when its distribution spans the entire EAF. First, I will briefly describe what is currently known about the biogeography and ecology of Sanderlings in the EAF, followed by a description of what it takes to set up and maintain a large international colour-ringing project, and finish with showing how analyses based on observations of colour-ringed Sanderlings have identified limiting factors in their population growth.

12.2 Sanderlings along the East Atlantic Flyway: biogeography and ecology

Sanderlings are one of the few shorebird species using the EAF that breed entirely in the High Arctic. The breeding area of Sanderlings using the EAF spans from NE Canada to east and N Greenland and the Taimyr peninsula in N Russia (Reneerkens *et al.* 2008, Scott 2009, Lappo *et al.* 2012). The species has also been reported to breed on Jan Mayen and Svalbard, although in (very) low numbers. During the non-breeding period, Sanderlings inhabit sandy beaches and mudflats along the entire Atlantic coasts of Europe and Africa, and to a lesser extent the Mediterranean coast.

It is unclear whether birds from the Siberian breeding population co-occur with the Greenlandic and Canadian breeding population along the E Atlantic coast during the non-breeding season (Reneerkens *et al.* 2009, Conklin *et al.* 2016). Sanderlings occurring along the Atlantic coast of Africa south of the equator have been assumed – without evidence – to be of Siberian breeding origin (Scott 2009, van Roomen *et al.* 2018). Moreover, the Canadian and Greenlandic breeding population has been suggested not to migrate further south than Ghana and Benin. Therefore, Sanderlings that spend the non-breeding season from Cameroon south to South Africa were previously not included in the estimates of the size of the East-Atlantic population (van Roomen *et al.* 2015). Recently though, the importance of southern Africa (i.e. Walvis bay in Namibia) for the Greenlandic breeding population has been identified (Loonstra *et al.* 2016, Reneerkens *et al.* 2020) and this finding has been implemented in this report (see Annex 1). Greenlandic and Siberian Sanderlings cannot yet be distinguished using genetic markers (Conklin *et al.* 2016). Since the density of observers and ringing activity is very low in Siberia, the easiest approach to determine whether Siberian-breeding Sanderlings make use of the EAF would be to track a representative sample of individuals using solar geolocators.

Remarkably, the EAF population of Sanderlings has been increasing for four decades (van Roomen *et al.* 2015). The number of Europe-wintering birds has been increasing at a faster rate than those that spend the non-breeding period in Africa (see Chapter 2 and Annex 1 in this report). Only in the last few years this increase in population size has come to a halt, and even turned into a decrease (this report). Despite Sanderlings being common along the entire EAF, the mechanisms that have resulted in the population growth, nor the recent stabilisation, are unknown.

There are indications that the growing population in itself may have limited the population growth rate ('density-dependence') (Ntiamoa-Baidu *et al.* 2014). Anthropogenic threats to Sanderlings concern habitat destruction, climate change (Reneerkens *et al.* 2016, Schmidt *et al.* 2019, Reneerkens 2020), human disturbance (Burger & Gochfeld 1991), pollution (Bianchini & Morrissey 2018) and hunting. It is yet unknown to which extent any of these threats has affected Sanderlings' population growth. To learn whether certain anthropogenic threats affect population size and via which mechanism, flyway-wide schemes of individually marked birds can be very helpful (Box 12.1).

12.3 Starting an international colour-ringing project

Estimates of survival are best obtained by using observations of individually marked animals (Box 12.1), but such observations can be applied for many other purposes too. Based on my experiences with the international Sanderling project described here, I will indicate what it takes to establish, maintain, and coordinate a large scheme of colour-ringed birds that occur along the entire length of the EAF.

When initiating demographic monitoring studies by using individual colour-marked birds, it is important to consider the purpose of the study, to ensure that there are sufficient resources to make it sustainable, and that the set-up will produce reliable results. These are matters of bird behaviour, logistics and personnel. The potential power of a colour-ringing project is very large, but the efforts to coordinate, maintain and finance such a scheme are often overlooked. The (licensed) catching and colour-marking of many birds is useless if there is no system in place to collect and store the observations and without the statistical skills to eventually analyse the data. It is important to carefully consider when and how many individuals need to be caught and colour-marked and especially how to assure a

Box 12.1: Estimating survival probabilities

Survival is a demographic parameter that has been shown to have one of the greatest potential impacts on population growth (Crone 2001). Conservation actions will often have the best potential to effectively influence rates of population change when they influence survival rates (Sandercock 2006). However, the timing and cause of mortality of individual free-living animals is often unknown and survival rates of populations can only be estimated from long-term data. Another complication is that in most bird populations, the age classes with often different survival rates to which individuals belong cannot be distinguished based on their appearance.

The most common method to estimate survival rates is the analysis of live observations of individually marked birds within a population, which can be analysed using mark-recapture analyses (Sandercock 2003). Observations can be either physical (re) captures, resightings of colour-marked birds or a combination of both. The concept is rather straightforward; when you individually mark a sample of birds, you monitor how many and which of them are present in the future and thus have survived. The more individuals return to the site of marking, the more have survived. However, the probability of re-encountering a bird not only depends on its survival until the next period, but also on its site fidelity. Individuals that permanently emigrate to other sites are still alive but will not be re-encountered. Furthermore, individuals that are site faithful should be available to be detected by observers. Even if they are present, there still is a probability that they will not be detected. Mark-recapture analyses are able to disentangle the probability of 'true survival' - the variable of interest to ecologists and conservationists - from site fidelity, site propensity and detection probability (Sandercock 2006). Usually, models include sex, time, and age-class but ecological variables, such a predator densities or weather, can also be included as covariates.

A necessity to study (changes in) demographic variables within populations over the vast area of a flyway, is to make use of a network of citizen scientists. International colour-ringing projects with individually recognisable birds can be used to estimate various demographic variables. Along the EAF, there are numerous such projects (http://www.c-birding.org/), but only few of those are used to measure demographic variables that can inform nature conservationists and policy makers.

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balanced resighting effort of those individuals. Capture-mark-recapture analyses (Box 12.1) assume that marked individuals are representative of the population of interest. In addition, it assumes that each marked individual has the same probability of being resighted, and that birds distribute themselves randomly within populations. In practice, these assumptions will rarely be met, but advanced statistical techniques can fortunately deal with these issues.

Observations of individually colour-ringed birds can be used in numerous ways. They may help to unravel migration patterns and study the timing of migration or length of stay at certain staging sites. However, where and when birds are observed does not only depend on their migration trajectories, but to a large extent on whether observers are active in those areas. For example, it is unlikely to learn about the breeding location of arctic-breeding waterbirds given the very low density of observers on the arctic tundra. For such questions, the use of tracking devices may be more suited. Also, estimating the total size of a passage population accounting for turn-over at staging sites based on observations of colour-ringed individuals (e.g. Loonstra *et al.* 2016), requires that there are observers present at these sites during the (expected) total stay of all individuals.

An important consideration in starting colour-marking schemes to estimate (annual) survival rates is that they require long time series. A first estimate of apparent annual survival can only be made after three years, yielding two estimates of annual survival. Moreover, if the species of interest has a large non-breeding range, temporal variation in annual survival at one study site could be biased and may not capture existing spatial variation in annual survival. Thus, besides the duration of these studies, it also requires field work at a representative selection of sites along the flyway.

Given the longevity of many bird species, durable material for colour-rings that minimises the risk of loss or discolouration of the rings is needed. Both ring loss and discolouration would violate an important assumption in mark-recapture analyses that each marked individual has

Box 12.2: Estimating reproduction

Free-ranging populations remain stable if during a lifetime each pair of animals replaces themselves with two young that will start reproducing themselves, provided that the life span and age of first reproduction is similar for each generation. Clearly, we will not understand what causes population change if we do not measure both survival (Box 12.1) and reproduction simultaneously. Estimating individual lifetime reproductive success of wild animals is complicated: most individuals cannot be followed from birth to death, nor can their offspring. Fortunately, there are useful alternative metrics available that can be used in population models. Reproduction consists of several components:

(I) the probability to occupy a territory and find a partner;
(II) the number of eggs laid;
(III) the probability of eggs to hatch;
(IV) the probability of chicks to survive from hatching to fledging and;
(V) survival until first reproduction.

The boundary between what is considered part of reproduction and of survival is sometimes a bit vague and may vary between sources. While (V) above is often included in 'fecundity' in population models, it is usually described as (juvenile or fist year) 'survival' and estimated in the same way as adult survival. How it is treated may depend on how and when in the annual cycle reproductive output is quantified, e.g. as the number of fledged young or from an age ratio among birds reaching a wintering ground. The distinction between reproduction and survival does not really matter as long as all components of the life cycle are covered in population models, without overlap. The latter can be an issue when reproduction is quantified as the number of young fledged, but first-year survival is estimated from data of young ringed at an earlier stage in the pre-fledging period.

Ideally, we would have reliable estimates of all the probabilities associated with reproduction, but that is usually not possible. Many studies focus on daily clutch survival or daily chick survival, albeit both components have complications too, because most clutches cannot be followed from the day they have been laid until either failure or hatching (Weiser 2021). Similary, most chicks cannot be followed from hatch until death or fledging. There are however useful field methods and statistical methods to get reliable estimates of both clutch and chick survival (e.g. Dinsmore *et al*. 2002, Ruthrauff & McCaffery 2005). These metrics can be useful indicators of which ecological factors have a local impact on reproductive success.

To identify whether annual reproductive success at the population level is limiting, the number of juvenile birds that recruit into the non-breeding population can be estimated. This measure of productivity includes components of mortality prior to fledging and from the first southward migration. However, the critical recruitment parameter from a demographic point of view is that of birds recruiting into the breeding population (Robinson *et al*. 2005). In geese, families migrate and stay together until spring, and family sizes and the proportion of juveniles in the population can easily be determined using field observations (e.g. Nolet *et al*. 2013). In shorebirds, the number of recruits into the non-breeding population can be determined by counting the number of juveniles and adults during field observations or in catches (e.g. Blomqvist *et al*. 2002, Lemke *et al*. 2012).

an equal probability of being observed. International colour-mark studies entail individual recognition of many different birds. Thus, the number of unique combinations of rings that can be used with a given number of colours or inscriptions on rings for a given duration of the study is an important consideration. The choice of ring colours and/or inscriptions and positions on the birds' legs needs to be coordinated with other ongoing colour-ringing schemes that are active in the same flyway. Clearly, individually marked birds from different research projects should be distinguishable from each other. For shorebirds along the EAF, the International Wader Study Group (www.waderstudygroup.org) takes up this coordination. Collaboration among projects may be a good way to maintain a large international network of ringers and observers. The success of an international colour-ringing project depends particularly on the communication with observers. Observers often need to make considerable effort to learn where they should report their observations. Therefore, it is only fair if they receive a polite and speedy response about the whereabouts of the individual bird(s) they reported. Even when the observation was incomplete and the individual cannot be recognised, observers appreciate it when they learn that their effort to read the rings and to report the observation is valued. It can guide and motivate them to continue to look out for and report their observations of colour-ringed birds.

12.4 Tropical-wintering Sanderlings perform poorest

Since 2003, Sanderlings have been colour-ringed at numerous locations along the EAF. Starting in Mauritania in 2003,

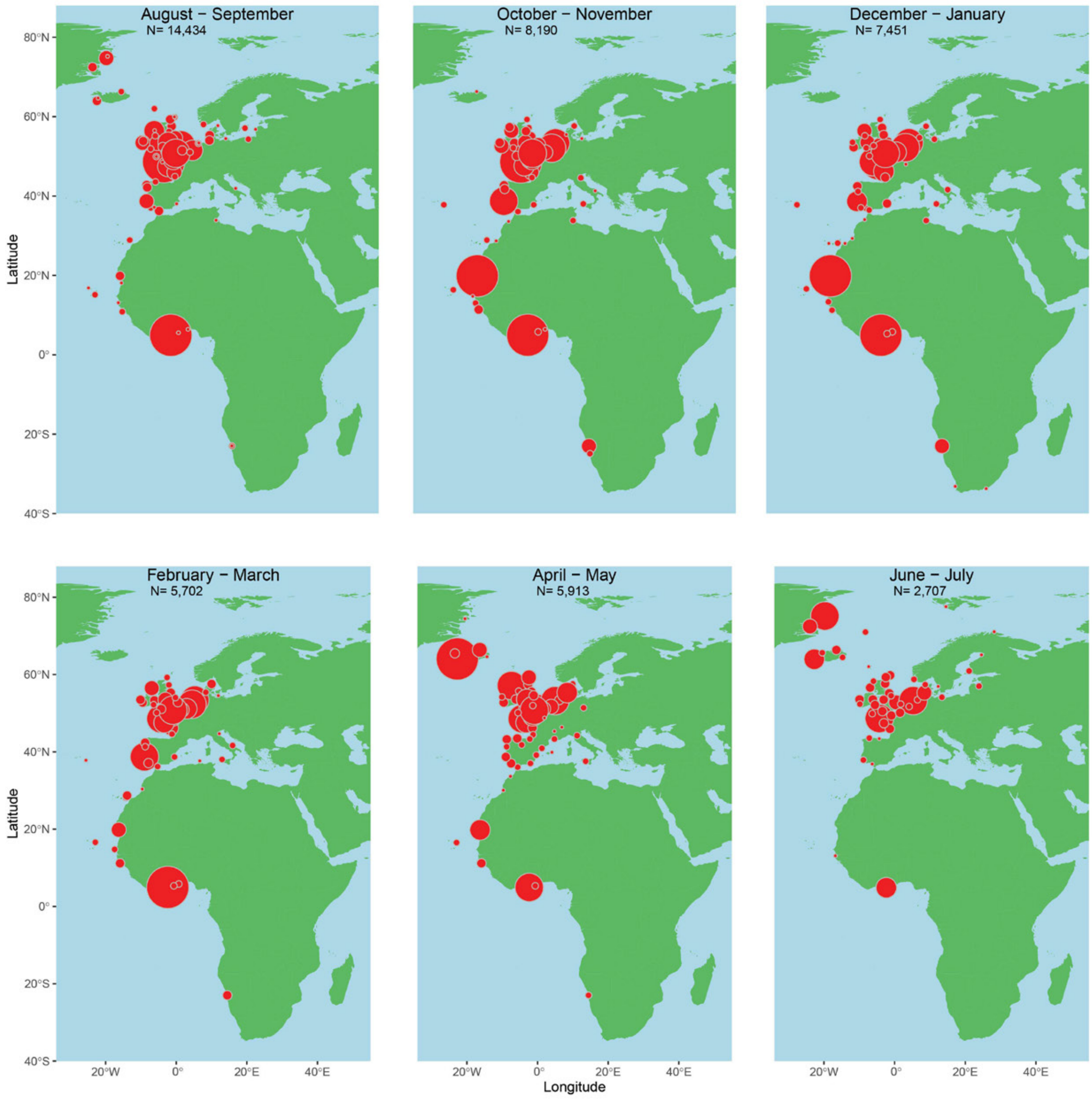

Figure 12.2. Locations along the EAF where individually colour-ringed Sanderlings were observed per two-month period in 2003-2021. Larger red dots represent more observations. For graphical clarity, locations within 200 km of each other were pooled. The sample sizes in the graph refer to the total number of unique observations. In total 100,670 observations were reported of 6,592 individual Sanderlings, allowing a detailed tracking of individuals throughout their lifetime, and analyses of e.g., migration phenology and survival probabilities.

the work was extended from 2007 onwards to Ghana, Portugal and England, as well as staging sites (Iceland and Dutch Wadden Sea) and the breeding area in Greenland. Between 2003 and September 2021, a total of 6,592 Sanderlings have been individually colour-ringed, which thus far resulted in 100,760 observations, allowing us to follow individuals from year to year and between seasons (Fig. 12.2). We have used these observations to estimate three important (demographic) variables: (1) annual adult survival, (2) the probability of northward migration of juvenile Sanderlings and (3) the timing of northward migration through Iceland, the last staging area before the flight to the breeding area in Greenland and Canada. The details of the methodology can be found in (Reneerkens *et al.* 2020).

One of the key findings of our study was that annual adult survival depended on winter location. Sanderlings spending the non-breeding season in W Africa (Mauritania and Ghana) had a lower annual survival probability than Sanderlings from three European wintering sites and Namibia (Fig. 12.3a). Also, the probability of juveniles to migrate northwards compared to that of birds older than one year was considerably lower in Ghana and Mauritania than in Portugal and England, where adult and juveniles were equally likely to migrate northwards (Fig 12.3b). Sanderlings from Mauritania and Ghana also migrated northward through Iceland on average 5-15 days later than birds wintering either further north or south (Fig. 12.3c). This suggests that the growth of the EAF population of Sanderling is currently limited by the conditions in W Africa (Reneerkens *et al.* 2020).

We suggest that relatively poor conditions in W Africa for fuelling up for northward migration may explain this pattern, perhaps due to a depleted food availability prior to the breeding season, when a higher food intake is needed to fuel the migratory flight (Reneerkens *et al.* 2020). A lower annual adult survival will result in a shorter lifespan and thus in fewer years of reproduction. Similarly, the tendency of first-year Sanderlings to spend the summer in W Africa also means that these Sanderling skip their first potential breeding season. The later spring migration through Iceland – the last possible staging site for the northbound birds – is likely to correlate with a late arrival in the breeding grounds. Reproductive success may be influenced by a seasonal decline in reproductive performance (Weiser *et al.* 2018), but this seems unlikely to also apply for Sanderlings arriving late in Greenland, because early clutches have a larger risk to fail due to depredation (Reneerkens *et al.* 2016).

On the basis of three published demographic parameters (Fig. 12.3, Reneerkens *et al.* 2020), in combination with published estimates of seasonal variation in clutch survival (Reneerkens *et al.* 2016) it could be shown that the growth of the Sanderling flyway-population is currently limited by annual variation in clutch survival and adult survival in W Africa (Sandercock 2020). A flyway-wide annual monitoring scheme of juveniles recruiting into the non-breeding population (Box 12.2), together with estimates of seasonal survival and annual survival estimates of first-year and older Sanderlings along the flyway (Box 12.1 & 12.2) will lead to an even better understanding of what drives changes of the flyway population size. Detailed studies of the ecological factors that are expected to cause the variation in the demographic parameters that have the largest impact on the population trajectory (Benton & Grant 1999, Caswell 2001) may then inform conservationists (Fig. 12.1). An increased and continued long-term and flyway-wide effort to monitor survival and reproduction of waterbird populations is essential to diagnose threats and the effects of conservation efforts.

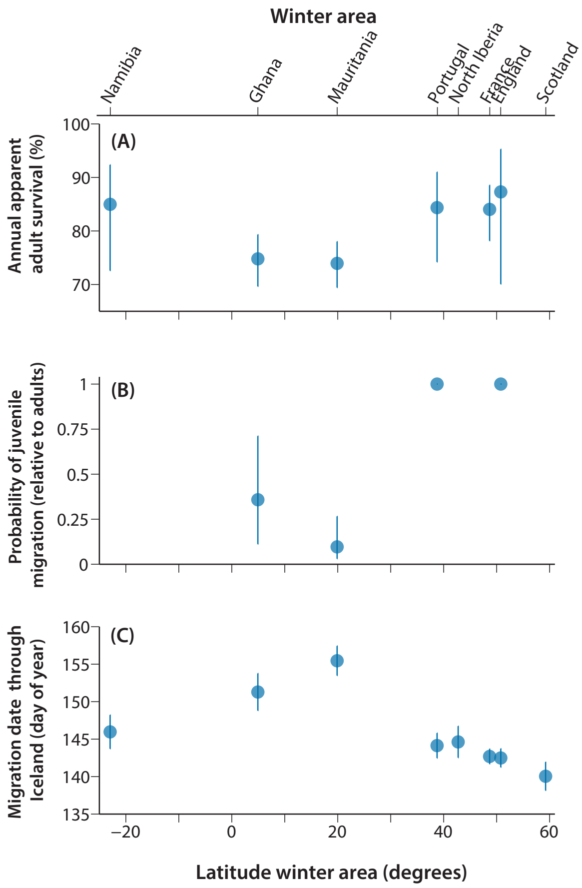

Figure 12. 3. Annual adult survival probabilities of Sanderlings wintering in six areas within the EAF (A), probability that juvenile Sanderlings from four winter areas migrated northwards in the summer following their first winter, relative to that in adults (B), and timing of northward migration through Iceland of Sanderlings wintering in eight wintering areas (C). Day of year 140 represents 20 May. Latitudes are those from the main study sites within winter areas. Dots are averages, and error bars indicate 95% confidence intervals. (From Reneerkens *et al.* 2020.)

Acknowledgements

Clearly, a long-term international research project like described here can only be successful if it is carried by many people. First, I want to thank the more than 2,500 enthusiastic and dedicated observers who reported colour-ring observations and age counts of Sanderlings. This work would not have existed without them or without the pleasant and fruitful collaboration with many colleagues and co-workers in the field, in the lab and behind computers. Theunis Piersma and Yaa Ntiamoa-Baidu have played key roles from the start. The work was financially supported through two grants from the Netherlands Polar Programme of the Netherlands Organisation for Scientific Research (NWO) and from the Metawad project awarded by Waddenfonds to Jeroen Reneerkens and Theunis Piersma, the Prins Bernhard Cultuurfondsprijs to Theunis Piersma and INTERACT grants for Transnational Access from the European Community's Seventh Framework Programme. Sjoerd Duijns, Marc van Roomen en Hans Schekkerman provided useful comments on an earlier draft of the text.

Eurasian Curlews | **Courlis cendré** (*Numenius arquata*) (Markus Varesvuo / Agami))

13. A comparison of flyway population trends as based on breeding versus winter counts

Chris van Turnhout, Szabolcs Nagy, Alena Klvaňová & Marc van Roomen

Summary

In this contribution, we assess to what extent counts during the breeding season (derived from the Pan-European Common Bird Monitoring Scheme PECBMS and the European Red List of Birds) and in winter (derived from the International Waterbird Census IWC) generate similar signals of changes in each flyway population. Furthermore, we identify the species and populations for which trend indications between the breeding and winter data sources strongly differ. We compared trends of in total 42 species/flyway populations, by correlating both long-term and short-term trend estimates (average annual change) and long-term trend directions (increase, decrease or stable/unknown). Our analyses show that population trend estimates from breeding counts are not very well aligned with trend estimates from winter counts, for flyway populations for which both data sources are available. For 15 paired comparisons, reflecting 12 flyway populations, breeding and winter counts even show contrasting trends, i.e. increase versus decrease or *vice versa*. 14 of these refer to populations that increase according to winter counts, but decrease according to breeding counts. IWC trends are thus generally more positive than breeding bird trends. We discuss possible reasons for these mismatches and suggest further research priorities and improvements of both breeding and winter monitoring programs.

Resumé

Dans cette contribution, nous évaluons dans quelle mesure les comptages effectués pendant la saison de reproduction (dérivés du Programme Pan-européen de suivi des oiseaux communs PECBMS et de la Liste rouge des oiseaux européens) et en hiver (dérivés du dénombrement international des oiseaux d'eau DIOE) génèrent des signaux similaires de changements dans chaque population de la voie de migration. De plus, nous identifions les espèces et les populations pour lesquelles les indications de tendances entre les sources de données de reproduction et d'hiver diffèrent fortement. Nous avons comparé les tendances de 42 espèces/populations de la voie de migration, en corrélant les estimations des tendances à long et court terme (changement annuel moyen) et les directions des tendances à long terme (augmentation, diminution ou stable/inconnu). Nos analyses montrent que les estimations des tendances des populations issues des comptages de reproduction ne sont pas très bien alignées avec les estimations des tendances issues des comptages hivernaux, pour les populations de la voie de migration pour lesquelles les deux sources de données sont disponibles. Pour 15 comparaisons appariées, reflétant 12 populations de la voie de migration, les comptages de reproduction et d'hiver montrent des tendances contrastées, c'est-à-dire une augmentation par rapport à une diminution ou vice versa. 14 d'entre elles concernent des populations qui augmentent selon les comptages d'hiver, mais diminuent selon les comptages de reproduction. Les tendances du DIOE sont donc généralement plus positives que celles des oiseaux reproducteurs. Nous discutons des raisons possibles de ces disparités et suggérons des priorités de recherche et des améliorations des programmes de surveillance hivernale et de reproduction.

13.1 Introduction

In order to evaluate the population trends of breeding, migrating and wintering birds in the international Wadden Sea, the trends along the entire flyway of each population are often used as a reference, to discriminate between local and global drivers of population changes (van Roomen *et al.* 2015, 2018, Chapter 5 of this report). These flyway population trends can be based on either counts during the breeding season, or counts in winter. For breeding population trends two data sources are available, with some overlap in species covered. The Pan-European Common Bird Monitoring Scheme (PECBMS) aggregates national monitoring programs of common breeding birds and updates trends annually, while the periodically updated European Red List of Birds (ERLoB) focusses on all breeding birds and is based on national assessments under article 12 EU Birds Directive reporting and additional assessments by non-EU states. Winter population trends are based on the International Waterbird Census (IWC). All three data sources have their own strengths and limitations in terms of spatial coverage, length of study periods, data quality and data quantity. In this contribution, we assess to what extent the breeding and winter counts generate similar signals of changes in each flyway population. Furthermore, we identify the species and populations for which trend indications between the breeding and winter data sources strongly differ, and try to shed some more light on the backgrounds of these mismatches.

13.2 Data and methods

PECBMS is run by the European Bird Census Council (EBCC) and compiles Pan-European trends from national time totals and covariance matrices submitted on an annual basis by national coordinators of breeding bird monitoring programs in 29 European countries (www.pecbms.info). Here, we use the population trends compiled and reported by Nagy *et al.* (2020), who calculated flyway population trends of selected waterbird species using PECBMS data for the 8th edition of the AEWA Conservation Status Report. Individual countries were allocated to flyway populations using the definitions of the AEWA Action Plan, as presented on the Waterbird Population Estimates Portal (Wetlands International 2021). Nagy *et al.* (2020) also present brief information on the participating countries per flyway population and the resulting completeness of coverage. Both long-term (period 1980-2017) and short-term (period 2008-2017) trends are used, with average annual change (%) and qualitative trend direction (increase, decrease, stable, unknown) as metrics for each time period.

The new edition of the European Red List of Birds was published recently by BirdLife International (2021). Here, we use the flyway population trends compiled from the national trend data by S. Nagy in the framework of AEWA's Conservation Status Report 8, as presented on the Waterbird Population Estimates Portal (Wetlands International 2021). The same definitions as for PECBMS were used to allocate individual countries to flyway populations. Wetlands International (2021) also gives brief information on participating countries per flyway population and completeness of coverage. Both long-term (period 1980-2018) and short-term (period 2009-2018) trends are used. Estimated minimum and maximum changes over each time period were converted to average annual changes (%), using geometric means of minimum and maximum estimates. Trend directions were used additionally.

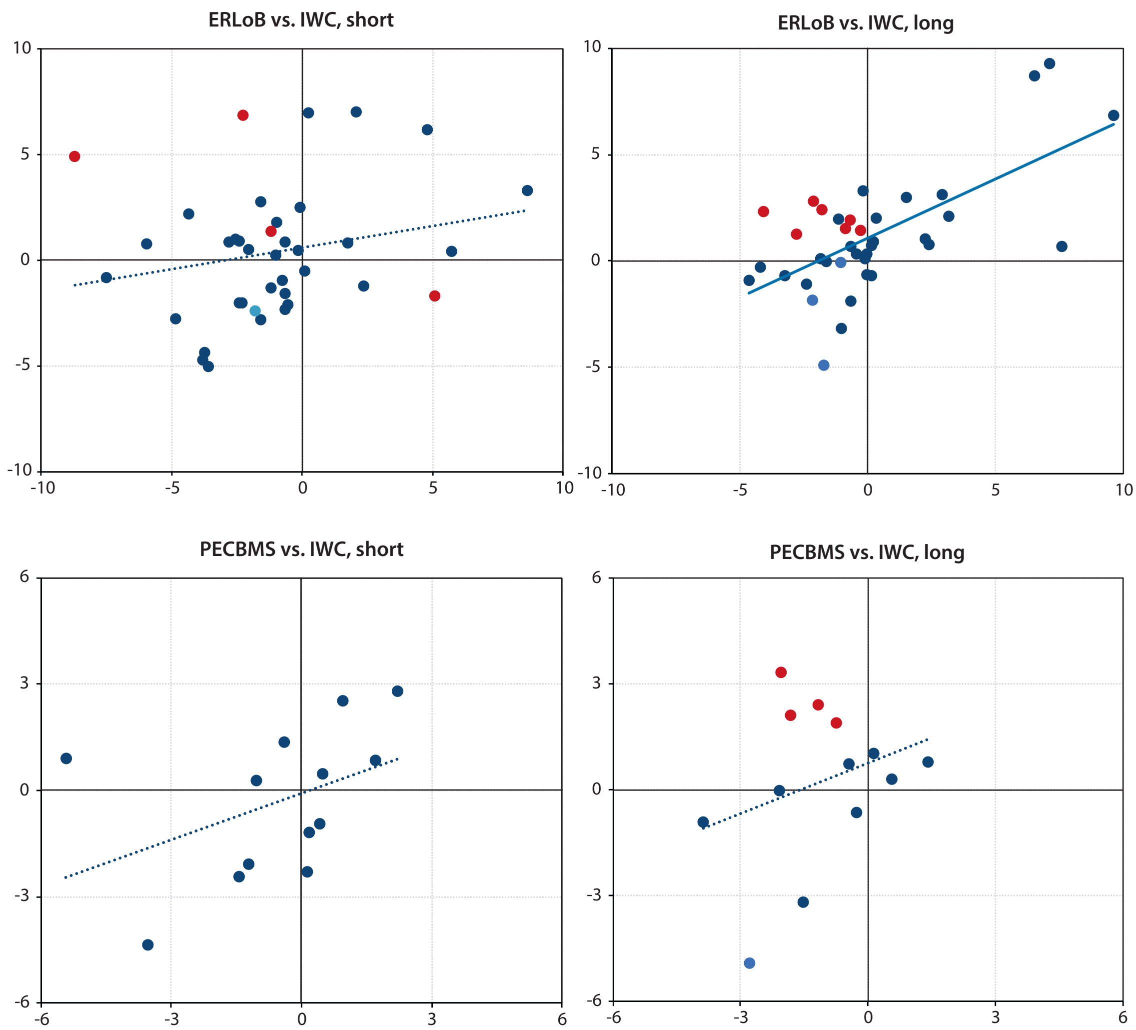

Figure 13.1. Correlations between trend estimates (% annual change) of IWC (y-axes) versus ERLoB (x-axes; upper panels) or PECBMS (x-axes; lower panels), on the short-term (left panels) and long-term (right panels). Note different scaling of axes between upper and lower panels. Species with contrasting trend directions (significant increase versus significant decrease, or vice versa) are depicted as red dots. For test-statistics of correlations, see text.

species	population	short-term						long-term					
		ERLoB 2009-18		PECBMS 2008-17		IWC 2011-20		ERLoB 1980-18		PECBMS var-2017		IWC var-2020	
Anser anser	*anser*, NW Eur/S-W Eur	5.72	+			0.42	=	7.15	+			9.26	+
Somateria mollissima	*mollissima*, Baltic, N & Celtic Seas	-3.82	-			-4.70	?	-2.37	-			-1.09	-
Bucephala clangula	*clangula*, N-W & C Eur (win)	-2.42	-			-2.01	-	-0.40	?			0.35	+
Mergus serrator	N-W & C Eur (win)	-2.43	?			0.94	=	0.23	?			0.93	+
Tadorna tadorna	N-W Eur	-0.76	-	0.43	=	-0.94	-	0.18	=	-0.45	=	0.74	+
Spatula clypeata	N-W & C Eur (win)	-2.26	-			6.84	+	-2.09	-			2.80	+
Mareca strepera	*strepera*, N-W Eur	4.79	+			6.19	+	6.54	+			8.70	+
Mareca penelope	W Siberia & NE Eur/NW Eur	-5.95	-			0.78	=	-4.06	-			2.31	+
Anas platyrhynchos	*platyrhynchos*, N-W Eur	-0.54	=	-1.21	-	-2.10	-	-0.01	=	-0.28	-	-0.65	-
Anas crecca	*crecca*, N-W Eur	-2.07	-			0.51	=	-1.11	-			1.96	+
Podiceps cristatus	*cristatus*, N-W & W Eur	-0.14	=	0.49	=	0.47	=	-0.66	-	-0.74	-	1.88	+
Podiceps auritus	combination of NE- and NW-Eur	0.11	=			-0.48	=	-2.77	-			1.24	+
Podiceps nigricollis	*nigricollis*, Eur/S & W Eur & N Afr	-0.65	-			0.89	=	-1.02	-			-0.07	=
Platalea leucorodia	*leucorodia*, W Eur/W Med. & W Afr	8.64	+			3.30	+	7.61	+			0.70	+
Ardea cinerea	*cinerea*, N & W Eur	-1.17	-	-0.40	=	1.34	+	2.26	+	0.12	=	1.02	+
Egretta garzetta	*garzetta*, W Eur, NW Afr	-1.60	-	2.21	?	2.80	?	3.21	+	-1.81	-	2.10	+
Phalacrocorax carbo	*sinensis*, N & C Eur	2.07	+			7.06	+	9.63	+			6.86	+
Haematopus ostralegus	*ostralegus*, Eur/S & W Eur & NW Afr	-1.80	-	-1.44	-	-2.44	-	-1.61	-	-2.09	-	-0.01	=
Recurvirostra avosetta	W Eur & N-W Afr (bre)	-2.55	-			1.02	=	-0.25	-			1.42	+
Pluvialis apricaria	*altifrons*, N Eur/W Eur & NW Afr	0.82	?	-1.12	-			0.61	?	1.19	+		
Charadrius hiaticula	*hiaticula*, N Eur/Eur & N Afr	-0.66	?			-1.54	=	-0.86	?			1.49	+
Charadrius alexandrinus	*alexandrinus*, W Eur & W Med./W Afr	-4.85	-			-2.76	?	-2.14	-			-1.86	-
Vanellus vanellus	Eur, W Asia/Eur, N Afr & SW Asia	-1.98	-	-1.94	-			-2.21	-	-2.02	-		
Numenius phaeopus	*phaeopus*, N Eur/W Afr	1.74	?	1.71	=	0.84	=	2.41	+	1.40	+	0.79	=
Numenius arquata	*arquata*, Eur/Eur, N & W Afr	-0.64	-	0.14	=	-2.30	-	-1.76	-	-1.16	-	2.40	+
Limosa limosa	*limosa*, W Eur/NW & W Afr	-3.45	-	-3.53	-			-4.53	-	-3.36	-		
Arenaria interpres	*interpres*, N Eur/W Afr	-4.36	-			2.21	?		-			-2.86	-
Calidris alpina	*alpina*, N Eur & NW Sib/W Eur & NW Afr	5.10	+			-1.69	-	-0.08	=			0.11	?
Calidris maritima	N Eur & W Siberia (breeding)	-1.59	?			-2.78	?	-4.19	-			-0.28	=
Actitis hypoleucos	W & C Eur/W Afr	-1.01	?	-1.05	=	0.28	?	-1.00	-	-1.54	-	-3.17	-
Tringa erythropus	N Eur/Sern Eur, N & W Afr	-3.75	?	-3.55	?	-4.36	?	-1.69	-	-2.77	?	-4.92	-
Tringa nebularia	N Eur/SW Eur, NW & W Afr	-0.08	=	0.95	=	2.52	?	-0.02	?	0.56	=	0.31	=
Tringa totanus	*totanus*, UK & IR/UK, IR, FR	-2.79	-	-5.43	-	0.90	?	-4.63	-	-3.88	-	-0.90	-
Tringa totanus	*totanus*, N Eur (breeding)	2.34	+	0.18		-1.20	=	-0.14	=	-2.03	-	3.30	+
Larus ridibundus	W Eur/W Eur, W Med., W Afr	-1.18	=			-1.27	-	-3.24	-			-0.67	-
Larus audouinii	Med/ N & W Afr	-8.72	-			4.90	+	0.36	+			2.00	=
Larus canus	*canus*, NW & C Eur/ W Eur & Med.	-2.31	-			-2.00	-	-0.65	?			0.70	+
Larus fuscus	combination *graellsii* + *intermedius*	-3.58	-			-5.00	-	2.94	+			3.10	+
Larus argentatus	combination of N- and W-Eur	-7.50	-			-0.80	=	-1.82	-			0.10	=
Sternula albifrons	combination of Eur and Med.					-1.90	-	-0.65	-			-1.90	-
Sterna hirundo	combination of SW- and NE-Eur	-0.97	=			1.80	?	0.16	=			-0.70	=
Thalasseus sandvicensis	*sandvicensis*, W Eur/W Afr	0.23	?			7.00	?	1.55	+			3.00	?

Table 13.1. Trends of flyway populations based on counts in breeding season (ERLoB and PECBMS) and winter (IWC), on short-term and long-term. For all trends both average annual change (%) and trend direction (increase, decrease, stable, unknown) are given.

short-term

ERLoB	IWC -	IWC =/?	IWC +
-	6	10	3
=/?	2	11	
+	1	2	3

PECBMS	IWC -	IWC =/?	IWC +
-	2	1	
=/?	2	7	1
+			

long-term

ERLoB	IWC -	IWC =/?	IWC +
-	8	4	7
=/?	3	3	4
+		3	7

PECBMS	IWC -	IWC =/?	IWC +
-	3	1	4
=/?	1	1	2
+		1	

Figure 13.2. Comparison of trend directions (increase, decrease or stable/unknown) of IWC versus ERLoB (upper panels) / PECBMS (lower panels) on the short-term (left panels) and long-term (right panels). The number of flyway populations is given in each cell; green cells refer to similar trend directions, orange cells refer to contrasting trend directions.

The International Waterbird Census is coordinated by Wetlands International, who compile flyway population trends from site-level count data submitted by coordinators of national waterbird monitoring programs (https://www.wetlands.org/international-waterbird-census, Nagy & Langendoen 2020). The counts during January as organized in the East Atlantic Flyway (EAF) in cooperation between the Wadden Sea Flyway Initiative, Wetlands International and Birdlife International, contribute to this IWC dataset. Flyway population trends as based on the IWC in the EAF are based on the analyses in this report (see Annex 1). Both long-term (up to 2020, with variable start year) and short-term (period 2011-2020) trends are used, with again average annual change (%) and trend direction (increase, decrease, stable, unknown) as metrics for each time period.

We compared both long-term and short-term trends of in total 42 species/flyway populations, by correlating trend estimates from winter counts (IWC) with those from breeding bird counts, derived from either PECBMS (N=13) and/or ERLoB (N=39). Apart from comparing trend estimates (average annual change), we also compared long-term trend directions (increase, decrease or stable/unknown), and paid special attention to flyway populations with contrasting long-term trend directions (increase vs. decrease or *vice versa*) between winter and breeding counts. The latter approach was not well applicable for short-term trends, because (1) short-term trends are much more sensitive to differences in the study periods (start and end years of trends) than long-term trends, and (2) a large proportion of short-trends is classified as unknown due to a lack of power resulting from large

ERLoB vs. IWC	Contrasting trends (n=7)	Identical trends (n=15)
Coverage ERLoB	2.6	2.1
Coverage IWC	1.9	2.4
Overlap breeding-winter	2.4	2.3
PECBMS vs. IWC	**Contrasting trends (n=4)**	**Identical trends (n=3)**
Coverage PECBMS	2.0	1.2
Coverage IWC	2.4	2.5
Overlap breeding-winter	2.3	2.0

Table 13.2. Semi-quantitative assessment of data quality/coverage of different monitoring programs, averaged for flyway populations with contrasting trend directions (left) and for species with identical trend directions (right). The lower the score for each metric, the better the coverage or fit (see text for further explanation).

annual fluctuations within a shorter time period, compared to long-term trends.

13.3 Results

Correlations between trend estimates of ERLoB/PECBMS and IWC are shown in Fig. 13.1, for the short term (left panels) and the long term (right panels). See Table 13.1 for trend estimates per species/flyway population.

Generally, population trend estimates from breeding counts do not correspond well with trend estimates from winter counts. Correlations are weak and non-significant (note that probabilities are one-tailed, as a positive correlation is expected):

- ERLoB vs. IWC, short-term: r= 0.23, P= 0.09
- PECBMS vs. IWC, short-term: r= 0.43, P= 0.07
- PECBMS vs. IWC, long-term: r= 0.30, P= 0.16

Only the long-term trend estimates of ERLoB and IWC show a stronger and significant correlation:

- ERLoB vs. IWC, long-term: r= 0.65, P<0.001 (upper right panel in Figure 13.1).

However, this latter correlation depends strongly on three populations with strongly increasing trends: Greylag Goose *Anser anser*, Gadwall *Mareca strepera* and Great Cormorant *Phalacrocorax carbo*. Without these, the correlation between ERLoB and IWC remains only weak: r= 0.27, P= 0.06.

Since part of the trends depicted in Fig. 13.1 refer to estimates that are rather imprecise and reflect non-significant trends (stable/unknown) rather than significant increases or decreases, we additionally compared trend *directions* of breeding versus winter counts (Fig. 13.2). In contrast to the results of the trend estimates comparison, both for ERLoB and PECBMS similarities with IWC are larger for short-term trends (53% resp. 69%) than for long-term trends (46% resp. 31%). Four (short-term) resp. seven (long-term) out of 39 populations show contrasting trends between IWC and ERLoB (orange cells). None (short-term) resp. four (long-term) out of 13 populations show contrasting trends between IWC and PECBMS. These contrasting species are also depicted as red dots in Fig. 13.1.

Fourteen of the 15 contrasting trends (either PECBMS vs. IWC or ERloB vs. IWC, short- and long-term combined) refer to populations that increased according to winter counts, but decreased according to breeding counts. IWC trends are thus generally more positive than ERLoB/PECBMS trends, which also appears from Figure 13.1 (more populations above than below line y=x, and concentrated in top left panel).

The following 12 populations are involved in the contrasting trends:

- **Northern Shoveler** *Spatula clypeata* (ERLoB vs. IWC, both short- and long-term)
- **Eurasian Wigeon** *Mareca penelope* (ERLoB vs. IWC, long-term)


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Greylag Goose | **Oie cendrée** *(Anser anser)*

Audouin's Gull | **Goéland d'Audouin** *(Larus Audouinii)*

- **Eurasian Teal** *Anas crecca* (ERLoB vs. IWC, long-term)
- **Great Crested Grebe** *Podiceps cristatus* (ERLoB and PECBMS vs. IWC, long-term)
- **Horned Grebe** *Podiceps auritus* (ERLoB vs. IWC, long-term)
- **Grey Heron** *Ardea cinerea* (ERLoB vs. IWC, short-term)
- **Little Egret** *Egretta garzetta* (PECBMS vs. IWC, long-term)
- **Pied Avocet** *Recurvirostra avosetta* (ERLoB vs. IWC, long-term)
- **Eurasian Curlew** *Numenius arquata* (ERLoB and PECBMS vs. IWC, long-term)
- **Dunlin** *Calidris alpina alpina* (ERLoB vs. IWC, short-term)
- **Audouin's Gull** *Larus audouinii* (ERLoB vs. IWC, short-term)
- **Common Redshank** *Tringa totanus*, North-Europe (PECBMS vs. IWC, long-term)
-

The match between PECBMS trends and ERLoB trends is much better for those breeding populations for which both are available, with r= 0.68 and r= 0.73 for short and long-term trends, respectively (N=16). However, both data sources cannot be regarded as 'independent'. ERLoB trend estimates would logically be derived from PECBMS trend data, whenever available.

13.4 Discussion

Generally, our analysis shows that population trend estimates from breeding bird counts are not very well aligned with trend estimates from non-breeding (January) counts in the EAF, for flyway populations for which both data sources are available. For 15 paired comparisons, reflecting 12 flyway populations, breeding and winter counts even show contrasting trends, i.e. increase versus decrease, or *vice versa*. This of course creates uncertainty about the true signal for these flyway populations: which one is most appropriate to use as a reference to compare flyway trends with site trends and describe the status of flyway populations? Differences in flyway trend assessments may result from various factors influencing the counts and monitoring programs, such as current and historical coverage, representativeness of sampling (e.g. IWC targets primarily aquatic habitats, whereas PECBMS covers aquatic habitats only partly through randomized plot selection), data quality (including precision of field work and analytical/trend methods), time period and, in case of incomplete coverage, the overlap between the two data sources: to what extent are the same individuals counted during breeding and in winter? For instance, breeding bird monitoring of the often large Russian populations is largely lacking. Moreover, flyway population delimitations do not follow country borders, so allocation at the country-level introduces additional noise (e.g. in Czechia, where several species have a migratory divide). Also, different flyway populations of the same species may partly mix outside the breeding season, making it impossible to separate them during winter counts (see Chapter 2).

As a first step to shed more light on the relative importance of each of the factors mentioned above, we indicatively quantified four additional metrics, based on the information on coverage in Wetlands International (2021) together with our expert judgement:

- Coverage (and quality) of ERLoB trends, based on countries with and without available data within the entire flyway delimitation, categorized as: 1) good, 2) reasonable, 3) moderate and 4) poor;
- Coverage (and quality) of PECBMS trends, idem;
- Coverage (and quality) of IWC trends, idem;
- Overlap between breeding and winter counts terms of subpopulations counted, categorized as: 1) good, 2) reasonable, 3) moderate and 4) poor.

Next, we averaged each of these four metrics for populations with (1) contrasting long-term trend directions between two data sources (increase versus decrease, or vice versa) versus (2) identical long-term trend directions between two data sources (both increase or both decrease) (Table 13.2). If differences in coverage or overlap would be important factors in causing overall differences in trend direction, we would expect that the coverage or overlap is better (i.e. the score is lower) for populations with identical trends than for populations with contrasting trends. This is true, at least to some extent, for coverage of breeding bird data, both for ERLoB and PECBMS. The match with trend directions based on IWC is better if the coverage of breeding bird count data is more complete (i.e. fewer countries which hold important numbers are missing). The average scores for coverage of IWC and overlap between breeding and winter counts do not clearly differ between contrasting trends and identical trends, or not in the expected direction.

Although differences in average scores are small and the scores are only indicative, this might suggest that particularly the improvement of trend assessments of breeding populations could be an important way forward. This can be effectuated by expanding the PECBMS scheme to cover more countries in Eastern (and Southeastern) Europe (including large populations in Russia), and by expanding the species selection of PECBMS to also cover scarce, rare and colonial breeding species. This will improve the trend assessments available for EU Birds Directive and ERLoB as well. As a matter of fact, both improvements are currently important targets for the PECBMS coordination team, and progress is being made. Expanding the PECBMS species selection can only be achieved by including data from species specific monitoring programs, since the methods of generic common bird monitoring schemes are not suitable for colonial breeding species, nocturnal species, breeding waterbirds, etcetera.

On the other hand, it is remarkable that in 14 out of 15 populations with contrasting trends, the winter counts show positive trends whereas the breeding counts indicate negative trends. We looked into four examples with contrasting long-term trends.

Eurasian Curlew *Numenius arquata* shows a long-term decrease in ERLoB and PECBMS, but an increase in IWC. However, seven out of nine countries holding the largest breeding populations within the flyway (>1,000 breeding pairs) report a negative trend in ERLoB, including Russia, United Kingdom and Sweden. Only Finland and France show a stable resp. unknown trend. So it is unclear what the origin is of the positive IWC-trend.

Great Crested Grebe *Podiceps cristatus* shows a long-term decrease in ERLoB and PECBMS, but an increase in IWC. In this species, only one out of eight countries that hold the largest breeding populations within the flyway (>10,000 breeding pairs) reports a positive trend in ERLoB: France. The other countries show a negative (e.g. the larger populations in Finland and Sweden) or stable/unknown trend.

Horned Grebe *Podiceps auritus* (combination of small-billed and large-billed population) shows a long-term decrease in ERLoB, but an increase in IWC. However, five out of six countries holding substantial breeding populations within the flyway (>200 breeding pairs) report a negative trend in ERLoB, including those with the largest populations (Finland, Sweden and Russia). Only Iceland shows a positive trend.

Common Teal *Anas crecca* shows a long-term decrease in ERLoB, but an increase in IWC. However, seven out of 10 countries holding substantial breeding populations within the flyway (>2,000 breeding pairs) report a negative trend in ERLoB, including the largest populations (Russia, Finland and Sweden). No country reports a positive trend.

Of course, issues related to both breeding and winter counts can simultaneously cause this disagreement between flyway population trends, but in these four examples the breeding bird assessments seem to be based on very homogeneous trends across the countries contributing most to the overall flyway trend. Although not all those countries can rely on extensive standardized monitoring data to assess their breeding bird trends (e.g. large populations in Russia), the coverage or quality of (historic) winter bird counts might need more attention as well. In particular we suggest to further evaluate IWC imputing methodologies, such as underlying stratifications (regions) and imputed values for sites in which many (historic) counts are missing, also in relation to distributional shifts between and within regions/strata that have occurred in response to climate change within this time period (e.g. Lehikoinen *et al.* 2013, Pavón-Jordán et al. 2019, see also Chapter 2).

Caspian Terns | **Sterne caspienne** (*Hydroprogne caspia*), **Royal Terns** | **Sterne royale** (*Thalasseus maximus*) **& Lesser Black-backed Gulls** | **Goéland brun** (*Larus fuscus*) (Jacques van der Neut / Agami)

14. East Atlantic Flyway Assessment 2020: discussion, conclusions and recommendations

15.1 Choice of species and populations to include

The choice of species and populations to be included in this report always proves difficult. While a large proportion of the populations has remained the same in each of the three assessment reports of 2014, 2017 and 2020 (number of populations covered in all three is 57), some species and populations have been dropped or added between them. Several criteria were considered in deciding which species and populations to include.

One criterion is the coverage of populations that are important in the Wadden Sea context. We define these as species and populations for which the Wadden Sea is designated as a Special Protected Area in the Natura 2000 framework in at least one of the three Wadden Sea countries. However, several of those species are not covered in the present report as they mainly occur in inland habitats and require a different type of site coverage for a comprehensive count in January than achieved in the coastal East Atlantic Flyway sites (e.g. Greater White-fronted Goose *Anser albifrons*, Northern Lapwing *Vanellus vanellus*), or predominantly use the open seas in the northern winter and hence are not well covered by January counts of estuarine and coastal areas either (e.g. Great Black-backed Gull *Larus marinus*, Black Tern *Chlidonias niger*). On the other hand, species such as Greater Scaup *Aythya marila*, Red-breasted Merganser *Mergus serrator* and Common Goldeneye *Bucephala clangula* have now been included, because of their importance in the Wadden Sea context and because they have strongholds in the Baltic region, which is gradually becoming a more important wintering region in the EAF.

The other criterion is coverage of populations which are largely present in the coastal EAF sites. For example, the Dunlin breeding in Iceland and wintering in Mauritania can be monitored rather easily within the coastal sites of the EAF. On the African continent, several populations are included which are largely confined to coastal sites, such as the those of Greater *Phoenicopterus roseus* and Lesser Flamingo *Phoeniconaias minor* in W Africa and African Oystercatcher and Damara Tern in Southern Africa.

In the 2017 report we also included several species and populations with a much more widespread and easterly distribution within Africa reaching far beyond our study area. As our results of these species and populations do not represent proper flyway assessments, these have been omitted from the present report (e.g. White-faced Whistling-duck *Dendrocygna viduata*, W African wintering populations of Northern Shoveler *Anas clypeata*). These species and populations are better analysed in a different context, to which the data collected in the coastal EAF will also contribute (RESSOURCE project in prep.).

The present study gives updated population trends for 83 populations from 66 species which represent a large selection of species and populations important in the Wadden Sea context and also reflect the quality of sites and habitats of the coastal EAF. For future flyway assessments we need to determine whether we can reach a permanent selection of populations to focus on. This will depend both on the preferences within the current cooperation under the Wadden Sea Flyway Initiative and the further development of similar cooperations and analyses in the Black Sea – Mediterranean -Sahel and West Asia-East Africa Flyways.

14.2 Monitoring methods and flyway boundaries

Based on accumulated knowledge of the geographical distribution (and mixing) of populations (see Scott & Rose 1996, Delany *et al.* 2009, and the Critical Sites Network Tool of Wetlands International & BirdLife International 2018) and knowledge about present and planned monitoring programs, choices have been made about the preferred timing and method of monitoring different flyway populations (Hearn *et al.* 2018 , Nagy *et al.* 2021). In the current EAF 2020 update, these recommendations have been followed as much as possible (see Annex 1). In contrast to the 2017 update, we had access to updated breeding bird data, and for several populations (notably gulls and terns), trends in this report are based on breeding bird counts instead of IWC data, which in 2017 was still often only possible for combinations of populations.

In this report we also investigated the similarities and differences in trends calculated on the basis of breeding and non-breeding (January) data for a selection of populations for which these were both available (Chapter 13). In addition to similar trend pairs, also a number of cases with contrasting breeding and winter trends were found. Probably the most frequent reasons for such mismatches were that the January and breeding bird counts do not refer to the same 'population' of birds, e.g. because they sample only part of a flyway population. However, for another

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Eurasian Oystercatchers | **Huîtrier pie** (*Haematopus ostralegus*) **and Red Knots** | **Bécasseau maubèche** (*Calidris canutus*) **at The Wash, United Kingdom**

group of populations, no obvious reason for the dissimilarity of trends was identified. It seems that the breeding bird trends tend to be less favourable than the non-breeding January trends for the same populations. These contrasting trend patterns show that we need to think and analyse further which signal we use to describe the status of the flyway populations in the future.

Despite increasing knowledge of flyway boundaries as a result of colour-ringing and tracking studies (see Chapter 12 for an example), for many populations much uncertainty still exists about migratory connectivity and flyway population boundaries. This is for instance the case in Common Redshank *Tringa totanus*, particularly with respect to the *totanus* breeding in N Europe and those of continental W Europe. Similarly the amount of mixing between *psammodromus* and *tundrae* Common Ringed Plovers wintering in W Africa, and between the Nearctic and Palearctic breeding populations of Ruddy Turnstone in the non-breeding season, are unclear to name a few. On top of these uncertainties, non-breeding distributions might shift over time, for instance in response to climate change. In Chapter 2, spatial patterns in changes in January numbers of waders in in different regions within the EAF are explored, and some possible indications for range shifts were found in species wintering both in Europe and Africa. For monitoring purposes this is not problematic as long as the whole January range remains within the study area. However, it can cause serious problems in the interpretation of flyway trends if some of the wintering birds move out of the study area completely or if different flyway populations overlap and mix increasingly. Apart from the waders mentioned above this may apply also to some ducks in the W Mediterranean and NW Europe (e.g. Northern Shoveler and Northern Pintail *Anas acuta*; see Chapter 6). In such cases, the interpretation of trends based on counts within the original wintering ranges may become problematic. It is therefore of importance to extend and refresh research into migratory connectivity of waterbirds in the EAF, using modern methodology

14.3 Methods of analyses

We tried to update trends and population sizes for the most recent time period, at least including the data from the 'total' EAF count of 2020. Fortunately, the time lag between the trends based on IWC data from Africa and from Europe (from where data were less recent in the assessments of 2014 and 2017) could be closed now. Most trends based on the IWC could be updated to include 2020. However, unfortunately the IWC data from N Africa and Spain were not yet available for 2018-2020, whilst data from South Africa shows a steady decline in coverage. For the trends based on breeding bird data, we could update most trends until 2018. It remains to be seen which strategy for updating the trends proves most suitable for the future. It seems that in general trends based on IWC counts can be updated faster than trends based on breeding bird counts. But also for the IWC counts problems arise if data from important countries are missing. Breeding bird data from the Pan-European Common Bird Monitoring Scheme can in principle also be updated on a yearly basis, but several species of interest are yet not covered by that scheme (among which many scarcer and colonial breeders). It is currently under investigation whether more waterbird species and populations can be monitored through the PECBMS mechanism.

In large-scale, partly volunteer-based, monitoring programmes such as the one discussed here and when working with populations with a clustered distribution as most migratory waterbirds are, missing values will be present in the dataset and the choice of the right analytical framework is a challenge. We decided to follow in principle the same methods as used for the Conservation Status Reports for AEWA (e.g. Wetlands International 2021). In this, TRIM is the analytical tool, and also choices about site selection, strata and different steps in the imputing process follow the methods for CSR 8. For W Africa, some more particular choices were made to select sites and counts to use in the trend analyses. However, TRIM is mostly designed for analysing data from many relatively small independent samples, while waterbird data often covers large proportions of the total populations, so site counts cannot be treated as independent samples. Analytical ways to overcome this problem are not so successful and can be very time consuming, making their application not very easy (Zuur 2020). Recently, a new method of imputing and trend analyses has been developed for waterbird data in the Mediterranean region (e.g. Dakki *et al.* 2020, see also Chapter 6) which seems promising to consider for future use.

The analytical methods for estimating population size are also under discussion. The present strategy based on IWC results is basically to count as much as possible, and to work from there towards national estimates and add the national or regional estimates together to come up with estimates for a flyway population. In this process a range of methods are applied, from simply using the total counted numbers to adding missed birds based on expert judgement to various analytical methods to estimate totals (Frost *et al.* 2019, BirdLife International 2021, Wetlands International 2021). For several W Africa countries, for a selection of populations, national estimates can be derived from counts carried out at the key coastal sites (for instance in Mauritania, Senegal, The Gambia and Ghana), or totals for the entire coast can be estimated on the basis of sample counts and extrapolations on the basis of knowledge about total availability of suitable habitat (especially important in countries rich in mangroves, like Guinea-Bissau (Chapter 9) and Guinea). On the other

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Whimbrels | **Courlis corlieu** (*Numenius phaeopus*) & **Bar-tailed Godwits** | **Barge rousse** (*Limosa lapponica*) hand, population sizes can also be estimated from breeding counts, in which case extrapolation is often used for dispersed species, or similar methods are applied as for IWC results for colony breeding birds. In the future more standardisation between the methods to reach national estimates for breeding and non-breeding results and from there to flyway estimates may improve the quality of the current estimates.

14.4 Monitoring of environmental conditions

On the basis of the currently applied environmental monitoring methods, we are able to describe the presence of various habitat types and natural features in wetland sites within the EAF. We can also signal a broad range of pressures impacting waterbirds and wetlands, and list management measures taken or planned in the various countries (see Chapter 3 and Annex 2). However, many sites are not yet covered, making it difficult to generalise the information received to the entire EAF. Also, some weaknesses render it difficult to use the information as a quantitative monitoring tool. Information from different years sometimes refers to different sites, and the data is often qualitative in nature which also contributes to differences in evaluation and interpretation by persons contributing the questionnaires. Besides improving the take up of the IBA approach by observers in the counted sites, making more use of remote sensing (see Annex 3) seems a promising step to improve environmental monitoring along the flyway.

14.5 Conclusions

East Atlantic Flyway overall

- Generally, the status of flyway populations using the coastal EAF appears relatively favourable, but with notable exceptions. In the long term, almost twice as many populations show an increasing or stable trend than a declining one, and this pattern is more positive than in other waterbird flyways in the world, e.g. the West-Asian – East African Flyway (Wetlands International 2021) and the East Asian – Australasian Flyway (MacKinnon *et al.* 2012). However, the short-term trends indicate a somewhat less positive pattern, and include several strong declines. The overall mean of the short-term trends has also become slightly less favourable since the previous assessment in 2017.
- In particular, arctic-breeding waders migrating over long distances show on average more negative trends than other taxonomic and functional groups. This applies especially to wader populations breeding in the Siberian Arctic (and migrating to sub-Saharan Africa), although recently Sanderling and the *lapponica* population of Bar-tailed godwit have also shown declining trends.
- At the sites within the EAF used by waterbirds many anthropogenic pressures occur. The extent to which

Arnold Meijer / Blue Robin)

Curlew Sandpipers | **Bécasseau cocorli** (*Calidris ferruginea*) & **Dunlins** | **Bécasseau variable** *(Calidris alpina)*, **Dutch Wadden Sea**

these pressures directly influence the conservation status of (which) populations along the flyway, cannot be assessed from the current data. Although it seems logical that the potential influence is larger and more urgent for those pressures mentioned more often in the 115 questionnaires collected, some pressures (e.g. fishing, disturbance, waste pollution, agriculture and urbanisation) are more obvious to *in situ* observers than others of which the impact may be just as severe.

- Particularly, the flyway is already under significant impact from climatic change, and this is bound to intensify in the coming decades. It is vital to address this through improved monitoring, research into solutions, awareness raising and conservation management.

Regional issues

- **Arctic**: in the Arctic, a pronounced warming of the climate during the breeding season is ongoing (e.g. Chapter 4), which changes the distribution, abundance, phenology of reproduction and sometimes even the morphology of birds that live there. While certain responses of shorebirds to climate warming are to be expected (e.g., northward distribution shifts during non-breeding, altered breeding phenology, northward breeding range extension of boreal species etc.), other effects are difficult to envisage or explain without the collection of additional data on the breeding grounds – not only about the birds themselves but also on changes in their food base and availability of habitats.
- **Wadden Sea**: On average, 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. The assessment following the first total EAF count in 2014 raised concern that the Wadden Sea formed a 'weak link' in the flyway, affecting the status of populations occurring there (van Roomen *et al.* 2015). After the current update of trends, this 'weak link' effect is no longer apparent among migratory birds outside the breeding season. Reasons underlying this can be manifold but this finding correlates with improved trends for migrants within the Wadden Sea. However, negative developments still dominate among trends of waterbirds breeding in the Wadden Sea.
- **N Africa**: In this study, three species (Greylag Goose *Anser anser*, Northern Pintail and Eurasian Wigeon *Mareca penelope*) display strong differences between trends within N Africa and at the EAF flyway scales (Chapter 6). These differences seem to be related to the response of species to climate change, notably short-stopping of migration closer to their breeding grounds made possible by milder winters.
- **W Africa**: although most of the sites have official conservation designations and have management plans in place, unfortunately their implementation is often still weak and therefore the treats continue to affect waterbirds and their habitats (e.g. Chapter 8, Annex 2). In several key sites like de Banc d'Arguin and the Bijagos Archipelago, quite substantial declines of non-breeding shorebirds have occurred over the past decades, although other species groups seem less affected generally (Chapters 7, 9). Advocacy and effective involvement of local communities could help improve the management of W African wetlands and mobilisation of funding for conservation.
- **Gulf of Guinea**: Principal pressures observed in 2020 relate to fishing, forest logging and firewood collection, littering and garbage dumping, and building/urbanisation (Chapter 3 &10). Heavy sand extraction was noted in Cameroon. Offshore and onshore oil exploitation remains a significant pressure, especially between Nigeria and The Congo. Conservation measures are widely needed in the Gulf of Guinea; although some regulatory measures are in place with respect to mangrove cutting, fisheries, hunting and urbanisation, effectiveness is low at many sites (Annex 2). Although most countries have some coastal protected areas, there is only limited effective protection of biodiversity along the coastal zone.
- **Southern Africa**: Principal pressures reported for Southern Africa in 2020 related to recreation and tourism, urbanisation and water management. Recreation and tourism are widespread along parts of the coastal belt of South Africa's Western Cape Province and at Namibia's Walvis and Sandwich Bays. Some wetlands are under constant pressure from various human activities, including illegal settlement on islands and threats of land reclamation, especially at important sites close to Luanda, Angola. Metal pollution and oiling incidents from urbanisation and shipping pose a threat to South Africa's Langebaan Lagoon. Conservation measures are underway at most sites assessed, although further measures are still needed. Some regulatory measures were in place at key sites, including for fossil fuel exploitation, urbanisation, control of invasive species, agricultural land use and wind farms.

14.6 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 pressures and conservation measures at the sites they use on a regular basis. To maintain a strong level of cooperation and information it will be important to continue the current level of activities and coordination.
- **Further enhance the position of countries to implement site and bird monitoring**. Despite increasing quality of the data collected, we are still far from a situation where these data are collected routinely and in ongoing good quality along the flyway. Particularly in Africa, but also in Spain, capacity and resources substantially limit the potential to carry out the monitoring. Continued and

increased capacity building is needed as well as options for a stable financial basis for these activities in these countries.

- **Improve the collection and utility of breeding bird data for monitoring**. In addition to January counts of non-breeding birds, breeding bird monitoring is also important for flyway level information. Current large scale international breeding bird monitoring programmes in Europe such as PECBMS need to be maintained. Their utility for flyway monitoring can be increased by including data on more species, including colonial waterbirds, and by increasing knowledge of migratory connectivity of the populations involved (see below). In Africa there is scope for increased availability of counts of colonial-breeding waterbirds (Gulls, Terns, Pelicans, Cormorants, Flamingo's, Herons). Besides field surveys also modern techniques like drones and automated image processing may contribute to this.
- **Extend and update research into migratory connectivity and flyway boundaries**. For many flyway populations we have fairly good information about their geographic boundaries and the connectivity between breeding, stop-over and wintering sites. For other populations this knowledge is still limited, which hampers their conservation and management on a flyway scale as well as assessment of their status. In addition, climate change and other pressures may induce distributional shifts, so that the geographical flyway delimitations themselves are also in need of monitoring. Nowadays, a suite of new technologies is available to aid in this, including analysis of DNA and isotopes and sophisticated tracking technology.

Georg Bangjord

Waterbird Count Norway

- **Improve the monitoring of environmental pressures and responses**. Through the current questionnaires on environmental conditions at sites we are able to collect important information on natural resources, pressures and conservation measures. However, the information collected is not quantitative, often open to interpretation and sometimes difficult to assess by observers. It is recommended that quantitative data will also be collected, and that information from remote sensing is incorporated in the monitoring.
- **Widen the focus from the northern winter situation to other parts of the year**. The collection and dissemination of information in the current series of flyway assessments focuses on the wintering phase of the annual cycle of northern breeding birds: January counts. For conservation and management however it is of important as well to collect flyway-wide information focused on the migration periods and other seasons, including periods relevant from the viewpoint of intra-African migrants (such as dry and rainy seasons).
- **Reinforce and expand the monitoring of conditions in the Arctic**. As many of the populations using the East Atlantic Flyway are breeding in the Arctic and often showing decreasing trends, the monitoring of conditions in the Arctic is a high priority. Because of the vast area involved, careful choices need to be made about which information can be feasibly collected on a regular basis.
- **Invest in research into causation of observed trends and relevant management responses**. The results from the monitoring of the EAF provide an important basis for management and conservation, but the monitoring of bird numbers is not sufficient to unravel the mechanisms behind observed declines. Neither will the monitoring be very specific in terms of pinpointing management actions for improving their conservation status. Therefore it is important that deeper-delving research, including the collection of data about vital rates, is carried out along the flyway to identify the mechanisms behind declines and collect conservation evidence for management actions.

Kim Fischer

Low water count in Guinea Bissau

Greater Flamingo | **Flamant rose** (*Phoenicopterus roseus*) (Arnold Meijer / Blue Robin)

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Arnold Meijer / Blue Robin

Sanderling | **Bécasseau sanderling** (*Calidris alba*)

Purple Sandpipers | **Bécasseau violet** (*Calidris maritima*) & **Ruddy Turnstones** | **Tournepierre à collier** (*Arenaria interpres*)
Netherlands (Arnold Meijer / Blue Robin)

Annex 1. Trends, distribution and abundance of waterbird populations using the coastal East Atlantic Flyway, update 2020

Marc van Roomen, Tom Langendoen, Szabolcs Nagy, Khady Gueye Fall, Sjoerd Duijns & 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, distribution in the non-breeding season (January) and flyway population sizes as an update on the results reported previously in Van Roomen *et al.* 2018.

For populations for which the flyway trends are based on non-breeding counts in January, the trends have been updated to include January 2020. All national coordinators of the International Waterbird Census (IWC) organized by Wetlands International took an extra effort to provide their most recent available count results in time for this (van Roomen *et al.* 2020). For populations for which trends are based on breeding bird monitoring, the update uses data collected through the Pan-European Common Bird Monitoring Scheme, and reporting for the EU Birds Directive and European Red List of Birds (BirdLife International 2021). Depending on the source, data could be updated until the breeding season of 2017 or 2018.

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

A1.2.1 Study area

The EAF as defined in this study is depicted in Fig. 1.1 in Chapter 1. It 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 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 EAF boundary, during January, we focus on coastal sites and the waterbird populations using these (sites depicted in red in Fig. A1.1). They were originally selected on the basis of the occurrence of a few estuarine species within IWC counts and a location not further inland than 40 km. In total, more than 2000 sites are included and denoted as the 'coastal EAF Flyway sites'. These sites can be large or small and are delimited to represent areas functioning more or less as an ecological unit (whole estuary, lake or lagoon); they may encompass multiple subsites or counting units. In addition to data from the coastal sites, also data from sites further inland are used. This is done flyway population-specifically for those inland sites considered part of the January range of a particular population. The inland sites additionally included are shown in the species maps. All inland sites included for at least one population mentioned in the Annex are shown in blue in Fig. A1.1.

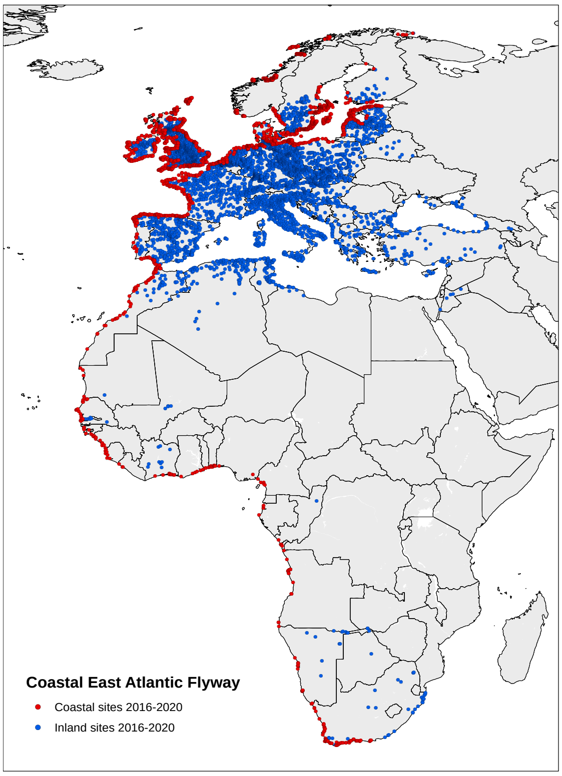


Figure A1.1. Sites from which January counts are included in the analysis. Sites of the coastal EAF are indicated as red dots, inland sites of the populations with flyway trends (see species accounts) as blue dots. The map covers the period 2016-2020.

In analyses based on breeding bird data, the study area is defined by the overlap of the flyway population boundary and breeding range of that particular flyway population, on a by country basis (see the species maps).

A1.2.2 Species and populations

This study focuses on waterbird species from bird families defined as 'waterbirds' under the Ramsar convention (Wetlands International 2002). In some waterbird species the complete world population occurs within the boundaries of the EAF. Mostly, 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 (including the rare cases of a species endemic to the EAF) 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 (2021).

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

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

In comparison to the 2017 update (van Roomen *et al.* 2018), all populations with a flyway trend included in that report are also included in this study, except (for various reasons) Greater Flamingo *Phoenicopterus roseus* (Mediterranean population), Mediterranean Gull *Choicocephalus melanocephalus*, Kelp Gull *Larus dominicanus*

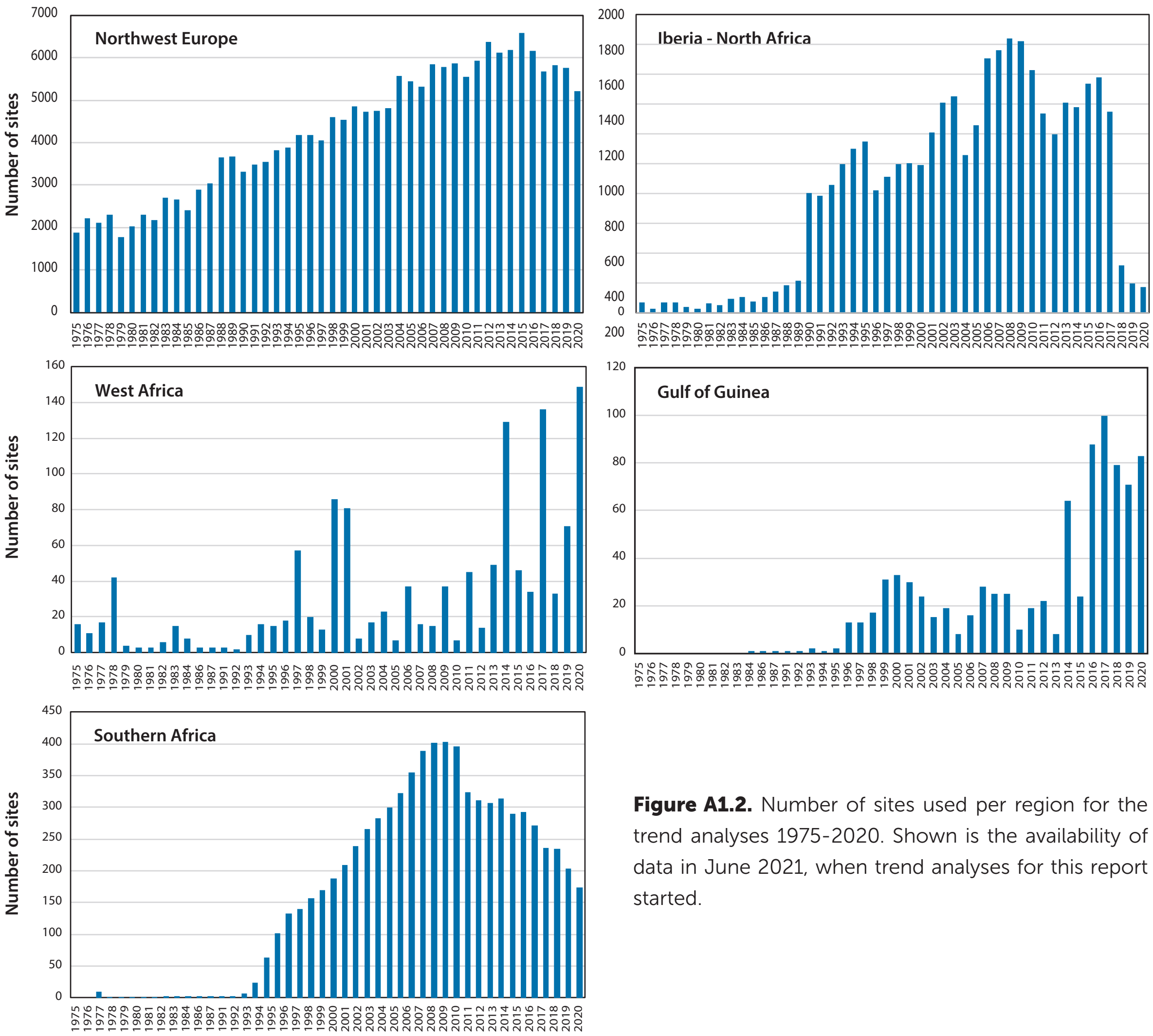


Figure A1.2. Number of sites used per region for the trend analyses 1975-2020. Shown is the availability of data in June 2021, when trend analyses for this report started.


Vincent Legrand / Agami

Kelp Gull | **Goéland dominicain** *(Larus dominicanus vetula)*

(Southern African population) and Great Black-backed Gull *L. marinus* (NW European population). The previous report also included regional trends (based on count data from sites within the coastal EAF) for several populations of which the distribution extends for a large part outside the EAF; these have now been omitted. On the other hand, because of their high presence in the coastal EAF the following species have now been included: Greater Scaup *Aythya marila*, Goldeneye *Bucephala clangula*, Red-breasted Merganser *Mergus serrator*, Little Egret *Egretta garzetta* and African Skimmer *Rhynchops flavirostris*.

The flyway populations for which trends are calculated are listed in Table A1.1. In a few cases, flyway populations have been taken together as count data for the separate flyway populations are not available or not reliable. In table A1.1 it is also indicated how much of the January distribution is within the coastal EAF sites and the extent to which the populations make use of the Wadden Sea at some time in their annual cycle.

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 other sources (European Goose Management Platform 2021, Wildfowl & Wetlands Trust 2021). Within the monitoring program in the EAF reported in this study, a substantial number of additional counts has been collected, particularly in W Africa, and added to the IWC database. From 2013 onwards sample counts have been organised annually, with 'total' counts aiming for a comprehensive coverage organised in 2014, 2017 and 2020. Efforts were made to obtain data complete up to January 2020 from as many countries as possible (see Acknowledgments and Van Roomen *et al.* 2020). This more recent data collected in the flyway could be combined with older counts from the IWC and from several expeditions in W Africa which are stored in the IWC database as well. Figure A1.2 shows the extent of IWC site coverage which could be used for the trend analyses in this report for the period 1975-2020. Unfortunately the data for the countries in Iberia and N Africa could not be updated in time but this will happen in the future (see Chapter 6).

The dominant source of breeding bird data 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. These data are collated once every six years. The current update provides breeding bird trends up to 2018 (BirdLife International

Table A1.1 Waterbird species and flyway populations for which flyway trends are analysed in this report. Given are the species and flyway population names in English (Wetlands International 2021). The recommended method for flyway trend monitoring (b=based on data on breeding birds, nb=based on data during non-breeding) based on Hearn *et al.* (2018) and Nagy *et al.* (2021) and the data sources and methods used for the trends in this report are also given (see text for further explanation). Also indicated are the extent to which populations use coastal EAF sites (% of their January totals in 2016-2020 in coastal EAF sites, * assessed for the combination of populations), and make use of the Wadden Sea (1 = more than 5% of flyway population uses Wadden Sea, 2 = using Wadden Sea but less than 5%, 3 = palearctic migrant not using Wadden Sea, 4 = Intra-Africa population) .

Species and population	Recom-mended method trend	Data source 2020	Trend method 2020	% in Coastal EAF sites	Occuring in the Wadden Sea
Brent Goose, Siberia/W Europe	nb-geese	IWC	Trim-WI	97	1
Barnacle Goose, Siberia & NW Europe/NW Europe	nb-geese	EGMP	TrendSpotter	43	1
Barnacle Goose, East Greenland/NW Europe	nb-geese	WWT	TrendSpotter	98	3
Barnacle Goose, Svalbard/NW Europe	nb-geese	WWT	TrendSpotter	95	3
Greylag Goose, NW Europe/NW & SW Europe	nb-geese	IWC	Trim-WI	28	1
Greylag Goose, Iceland/NW Europe	nb-geese	WWT	TrendSpotter	55	3
Common Eider, Baltic Sea, North Sea & Celtic Sea	nb-special	IWC	Trim-WI	100	1
Common Goldeneye, NW & C Europe (winter)	nb-January	IWC	Trim-WI	68	2
Red-breasted Merganser, NW & C Europe (winter)	nb-January	IWC	Trim-WI	94	2
Common Shelduck, NW Europe (winter)	nb-January	IWC	Trim-WI	95	1
Greater Scaup, Northern Europe/Western Europe	nb-January	IWC	Trim-WI	96	2
Northern Shoveler, NW & C Europe (winter)	nb-January	IWC	Trim-WI	42	2
Gadwall, NW Europe	nb-January	IWC	Trim-WI	19	2
Eurasian Wigeon, W Siberia & NE Europe/NW Europe	nb-January	IWC	Trim-WI	45	1
Mallard, NW & E Europe & Siberia/NW Europe	nb-January	IWC	Trim-WI	35	2
Northern Pintail, N Europe & Siberia/NW Europe	nb-January	IWC	Trim-WI	84	1
Common Teal, NW & E Europe & Siberia/NW Europe	nb-January	IWC	Trim-WI	49	1
Great Crested Grebe, *cristatus*, NW & W Europe	nb-January	IWC	Trim-WI	26	2
****Horned Grebe**, NW Europe (winter) combi of pop.	nb-special	IWC	TrendSpotter	82	2
Black-necked Grebe, Europe/S & W Europe & N Africa	nb-January	IWC	Trim-WI	14	2
Greater Flamingo, W Africa	nb-January	IWC	Trim-WI*	100	4
Lesser Flamingo, W Africa	nb-January	IWC	Trim-WI*	100	4
Eurasian Spoonbill, W Europe/W Europe & W Med & W Africa	b-colony	ERLoB	IUCN	85	1
Grey Heron, N & W Europe	nb-January	IWC	Trim-WI	33	2
Little Egret, SW Europe	b-colony	PECBMS	Trim-EBCC	70	2
Western Reef-egret, W Africa	nb-January	IWC	Trim-WI*	99	4
Great White Pelican, W Africa	nb-January	IWC	Trim-WI*	96	4
Great Cormorant, *sinensis*, N & C Europe	nb-January	IWC	Trim-WI	34	1
Great Cormorant, *lucidus*, coastal W Africa	nb-January	IWC	Trim-WI*	99	4
Cape Cormorant, coastal S Africa	b-colony	IWC	Trim-WI	99	4
African Oystercatcher, coastal S Africa	nb-January	IWC	Trim-WI	98	4

Species and population	Recom-mended method trend	Data source 2020	Trend method 2020	% in Coastal EAF sites	Occuring in the Wadden Sea
Eurasian Oystercatcher, Europe/W & S Europe & NW Africa	nb-January	IWC	Trim-WI	99	1
Pied Avocet, W Europe & NW Africa (breeding)	nb-January	IWC	Trim-WI*	98	1
Grey Plover, W Siberia/W Europe & W Africa	nb-January	IWC	Trim-WI*	92	1
Common Ringed Plover, *hiaticula*, NW Europe/ SW Europe & N-Africa	nb-January	IWC	Trim-WI	94	1
Common Ringed Plover, *psammodromus*, NE Canada to Iceland/W & S Africa	nb-January	IWC	Trim-WI*	99	1
White-fronted Plover, *hesperius*, W Africa	nb-special	IWC	Trim-WI	99	4
** **White-fronted Plover**, Gabon - South Africa, combi of pop.	nb-special	IWC	Trim-WI	88	4
Kentish Plover, W Europe & W Mediterranean/W Africa	nb-January	IWC	Trim-WI*	62	2
Chestnut-banded Plover, Southern Africa	nb-January	IWC	Trim-WI	97	4
** **Whimbre**l, East Atlantic (wintering), combi of pop.	b-dispersed	IWC	Trim-WI*	99	2
Eurasian Curlew, Europe/NW Europe, N & W Africa	b-special	ERLoB	IUCN	85	1
Bar-tailed Godwit, *lapponica*, N Europe /W Europe	nb-January	IWC	Trim-WI	100	1
Bar-tailed Godwit, *taymyrensis*, W Siberia /W & S Africa	nb-January	IWC	Trim-WI*	100	1
Ruddy Turnstone, NE Canada & Greenland/W Europe & NW Africa	nb-January	IWC	Trim-WI	97	1
Ruddy Turnstone, N Europe/W Africa	nb-January	IWC	Trim-WI*	96	1
Red Knot, *islandica*, NE Canada & Greenland/W Europe	nb-January	IWC	Trim-WI	100	1
Red Knot, *canutus*, W Siberia/W & S Africa	nb-January	IWC	Trim-WI*	100	1
Curlew Sandpiper, W Siberia /W Africa	nb-January	IWC	Trim-WI*	97	2
Sanderling, W Europe & W & S Africa (winter)	nb-January	IWC	Trim-WI*	96	1
Dunlin, *alpina*, NE Europe & NW Siberia /W Europe & NW Africa	nb-January	IWC	Trim-WI	81	1
Dunlin, *schinzii*, Iceland /NW & W Africa	nb-January	IWC	Trim-WI*	100	3
** **Purple Sandpiper**, NW Europe (winter), combi of populations	nb-January	IWC	Trim-WI	97	2
Little Stint, N Europe & NW Siberia/N & W Africa	nb-January	IWC	Trim-WI*	51	2
Common Sandpiper, W & C Europe/W Africa	nb-January	IWC	Trim-WI	90	2
Spotted Redshank, N Europe /SW Europe, N & W Africa	nb-January	IWC	Trim-WI	29	1
Common Greenshank, N Europe/ W & SW Europe, NW & W Africa	nb-January	IWC	Trim-WI	85	1
Common Redshank, *robusta*, Iceland & Faroes / W Europe	b-dispersed	IWC	Trim-WI	98	1
Common Redshank, totanus, Britain, Ireland/Britain, Ireland & France	b-special	PECBMS	Trim-EBBC	99	2
Common Redshank, totanus, Central & Eastern Europe (breeding)	b-dispersed	PECBMS	Trim-EBBC	65	1
Common Redshank, totanus, N Europe /W Africa	b-dispersed	PECBMS	Trim-EBBC	99	1
African Skimmer, W & C Africa	b-colony	IWC	Trim-WI*	95	4
Slender-billed Gull, W Africa	nb-January	IWC	Trim-WI*	100	4
Black-headed Gull, W Europe/W Europe, W Med - W Africa	nb-January	IWC	Trim-WI	24	1
Hartlaub's Gull, coastal SW Africa	nb-January	IWC	Trim-WI	81	4

Species and population	Recom-mended method trend	Data source 2020	Trend method 2020	% in Coastal EAF sites	Occuring in the Wadden Sea
Grey-headed Gull, W Africa	nb-January	IWC	Trim-WI*	99	4
Audouin's Gull, Mediterranean/N & W Africa	b-colony	ERLoB	IUCN	89	3
Mew Gull, NW & C Europe /NW Europe & W Med.	b-colony	ERLoB	IUCN	37	2
Lesser Black-backed Gull, *graellsii*, NW Europe / East Atlantic	b-colony	ERLoB	IUCN	48*	2
L**esser Black-backed Gull**, *intermedius*, W Europe / East Atlantic	b-colony	ERLoB	IUCN	48*	1
European Herring Gull, *argenteus*, NW Europe/East Atlantic	b-colony	ERLoB	IUCN	75*	2
European Herring Gull, *argentatus*, W Europe /East Atlantic	b-colony	ERLoB	IUCN	75*	1
Gull-billed Tern, W Europe/W Africa	b-colony	ERLoB	IUCN	98	2
Little Tern, Europe north of Mediterranean /East Atlantic	b-special	ERLoB	IUCN	99*	2
Little Tern, West Mediterranean/ East Atlantic	b-special	ERLoB	IUCN	99*	3
Damara Tern, Namibia & South Africa	b-special	IWC	Trim-WI	100	4
Caspian Tern, coastal W Africa	nb-January	IWC	Trim-WI*	99	4
Common Tern, N & E Europe /East Atlantic	b-colony	ERLoB	IUCN	99*	2
Common Tern, S & W Europe/East Atlantic	b-colony	ERLoB	IUCN	99*	1
Roseate Tern, W Europe/East Atlantic	b-colony	ERLoB	IUCN	100	3
Sandwich Tern, W Europe /W Africa	b-colony	ERLoB	IUCN	97	1
Royal Tern, W Africa	b-colony	IWC	Trim-WI*	100	4
Greater Crested Tern, S Africa	b-colony	IWC	Trim-WI	94	4

2021). This dataset is supplemented with data from the Pan-European Common Bird Monitoring Scheme (PECBMS) as coordinated by the European Bird Census Council (http://pecbms.info/).

For the update of population size estimates, mostly Wetlands International (2021) has been used, which includes updates up to 2018. This source uses mostly the new EU Art. 12 and European Red List of Birds data mentioned above, and IWC data. Details about these assessments can be found in the population accounts on the Waterbird Population Estimates portal (http://wpp.wetlands.org). For a selection of populations concentrated in coastal W Africa in January, updates of population size estimates including the January 2020 census are provided in this Annex (see below). These are updates of the last estimates published for these populations, which ran up to 2014 (van Roomen *et al.* 2015).

Distribution maps in this Annex are for January and based on 2016-2020 IWC results. In these maps, in addition to numbers at sites in 2016 -2020 which are included in the flyway population trends (both from the coastal EAF and more inland sites), numbers of birds from other biogeographical populations of the same species present at sites in the coastal EAF are shown as well.

A1.3 Analyses

A1.3.1 Trends

Trends based on non-breeding January data

For the calculation of trends, in principle, the same methods were applied (to data up to 2020) as used for the AEWA Conservation Status Report 8 (Nagy & Langendoen 2020; Wetlands International 2021) which includes trends up to 2018. Compared with van Roomen *et al.* (2018) this involves a stricter selection of sites; a minimum of five counts per site is required instead of two. Imputing of missing counts is first applied at the level of countries instead of groups of countries in the same region. Start and end years of trends are standardised more strictly and trend calculation is done with the rTrim package (Bogaart *et al.* 2020) instead of TrendSpotter (Soldaat *et al.* 2007). The trends calculated with this method are indicated with 'Trim-WI' in table A1.1. For the data from W Africa the rules of five counts for site inclusion and for start and end years were less strictly applied. In this way more flexibility remained in choices between reliable and less reliable data, imputing and start years, and better use could be made of scarce data (indicated by TRIM-WI* in table A1.1). Other exceptions are a few trends that were still calculated with Trendspotter as output from rTrim was not available for various reasons (indicated by 'TrendSpotter' in table A1.1).

Trends based on breeding bird data

Trends based on data from PECBMS were calculated with rTrim following Nagy *et al.* 2020 ('Trim-EBCC' in table A1.1). 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 for multiple populations (BirdLife International 2021, 'IUCN' in table A1.1), further following Wetlands International 2021).

For all flyway populations and data sources, the trends are expressed as mean yearly change during the complete time series (most often 1975-2020 or 1980-2020, called long-term) and for the most recent 10 years (2011-2020, short-term). These trend values, the length of the time series and the trend indication following Soldaat *et al.* 2007 (increase, decrease, stable, uncertain) are given for each population in a table in the species accounts. For flyway trends with (near-)annual abundance estimates (all, except those calculated with IUCN methodology) trend graphs are given. Trends based on PECBMS are indices, trends based on January counts are given as absolute yearly estimates, with years with more than 70% imputing removed. These estimates should not be used as the total population size (see below) 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 (Visser 2004). This can be a flexible line or a straight line depending on the number of year estimates available.

A1.3.2 Distribution

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 shown as the average numbers per site in January 2016-2020, represented by red dots when referring to coastal EAF sites and by blue dots for inland sites. Sometimes, blue dots are outside the flyway boundary as sites were allocated to fly-

Saverio Gatto / Agami

Kentish Plover | **Pluvier à collier interrompu** *(Charadrius alexandrinus)*

Lesser Black-backed Gull | **Goéland brun** *(Larus fuscus)*

way populations on a country or regional basis rather than per individual site. The other selection shows numbers from the same species, but other biogeographical populations, in other coastal EAF sites (recognisable as red dots outside the flyway boundary if present). For these populations the distribution at inland sites is not shown. The purpose of this additional selection is to show the importance of the coastal EAF for other flyway populations as well. In the maps, counts from count-units and sub-sites within the same site are summed. The yearly totals for 2016-2020 were averaged per main site, using only the more or less complete counts or estimates for a site.

A1.3.3 Population size estimates

For some populations with large concentrations in coastal EAF sites in W Africa, updates of population sizes up to 2020 are provided in this Annex. Priority has been given to populations with their last update of population size based on data from 2000 – 2012. The sum of the average January counts 2016-2020 at sites allocated to the flyway population form the backbone of this estimation. However, in most cases additional expert judgment was necessary to adjust this calculation to account for incomplete coverage at the coast and especially inland. Details can be found in the species accounts.

A1.4 Results

The species accounts in this section provide some background on 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 populations 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). The main results per species and population consist of a table which summarises trends and population size estimates, a map illustrating the distribution in January 2016-2020, and the flyway trend graph if available. Trends are based on the source which is considered at present as the most suitable for the monitoring of the flyway population. However, sometimes considerable uncertainty exists about which source is more reliable or representative. Chapter 13 of this report explores contrasts in trends based on IWC and breeding bird counts. Remarkable differences are found, which need further consideration in the future. Also, shifts in January distribution may be happening, possibly affecting flyway trends based on IWC results (see Chapter 2). For those population size estimates that update information in CSR 8 (Wetlands International 2021), some information is provided on the underlying numbers counted and further expert judgement.

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 (Cleasby *et al.* 2017). The species is fully migratory, arriving on the breeding grounds in early June (Spaans *et al.* 2007). Breeding occurs in small, loose colonies or dispersed in single pairs (Nolet *et al.* 2013). The preferred breeding sites are grassy coastal meadows or islands where large raptors, snowy owls or gulls are present that can deter mammalian predators (de Fouw *et al.* 2016). Non-breeding birds inhabit estuaries and bays with seagrass *Zostera*, coastal saltmarshes and cultivated grasslands.

Arnold Meijer / Blue Robin

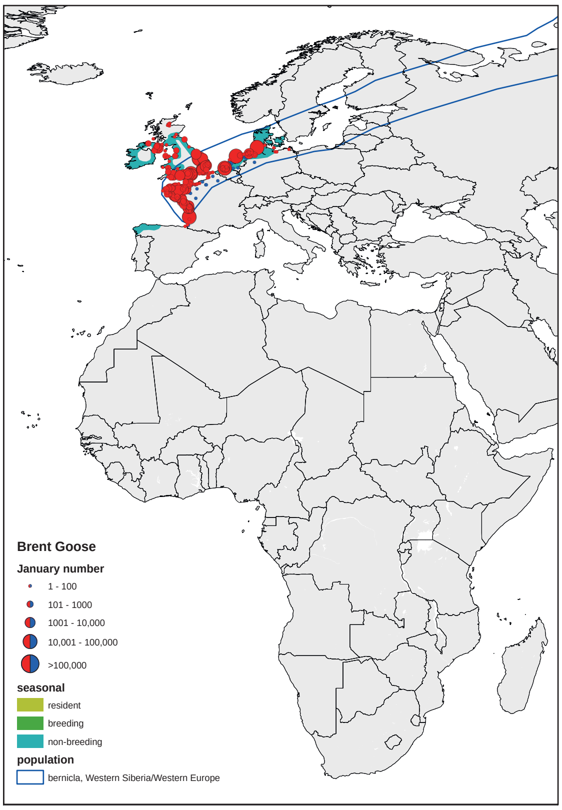


Figure A1.3. Distribution of Brent Goose in the East Atlantic Flyway. Red dots denote numbers in coastal EAF sites in January 2016-2020. Blue lines indicate the flyway boundaries of the population for which the flyway trend is presented. Inland sites used for the calculation of the flyway trend together with the coastal EAF sites within the flyway boundaries are indicated as blue dots.

population Brent Goose	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
bernicla, Siberia/W Europe	w	1975-2020	1,015	moderate increase	2011-2020	1,003	stable	2011	211000	211000

Table A1.2. Summary of flyway trend and population size estimates for Brent Goose. Mentioned are the flyway population under investigation (English name and description according to Wetlands International 2021, data type used for the trend (w = January counts, b = breeding bird counts), the time period of the long-term trend (period-L), the slope (trend-L) and the trend category (following Soldaat et al. 2007), 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).

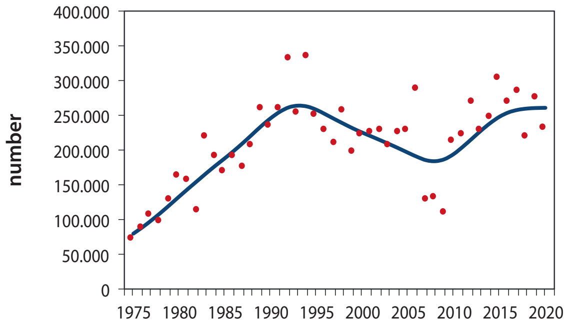


Figure A1.4. Flyway population trend for Brent Goose population Branta, b. benicla. Red dots denote the year results (not to be equated to total population size, see main text), the blue line is the smoothed trend line.

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 in 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 (Prop *et al.* 2015). Non-breeding birds inhabit coastal meadows, saltmarshes and tidal mudflats, with increasing use of cultivated grasslands for feeding in recent decades (Eichhorn *et al.* 2009).

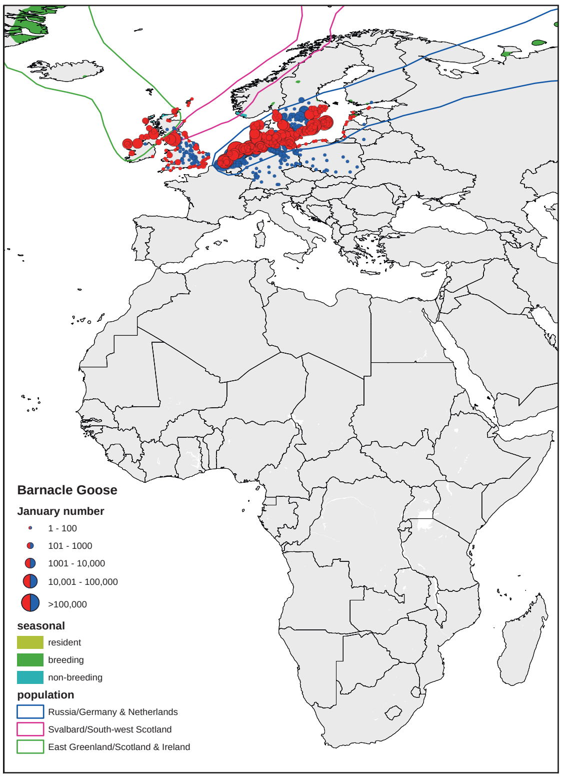


Figure A1.5. Distribution of Barnacle Goose in the East Atlantic Flyway. For explanation see fig. A1.3.

Harvey van Diek

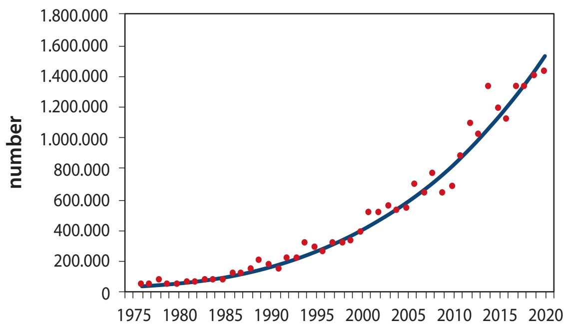


Figure A1.6. Flyway population trend for Barnacle Goose population Siberia & NW Europe/NW Europe. For explanation see fig. A1.4.

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	1978-2020	1,086	strong increase	2011-2020	1,063	moderate increase	2018	1400000	1400000
East Greenland/NW Europe	w	1976-2020	1,026	moderate increase	2011-2020	1,026	moderate increase	2018	72000	72000
Svalbard/NW Europe	w	1975-2020	1,044	moderate increase	2011-2020	1,014	moderate increase	2015-2020	40000	40000

Table A1.3. Summary of flyway trend and population size estimates for Barnacle Goose. For explanation see table A 1.2.

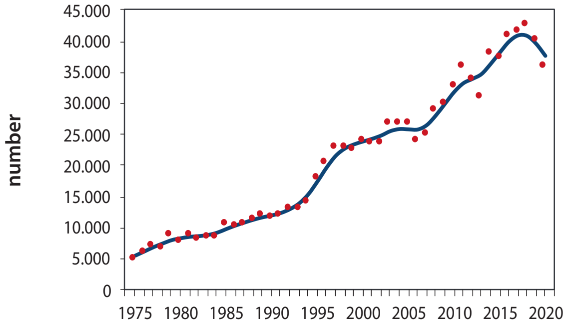


Figure A1.7. Flyway population trend for Barnacle Goose population E Greenland / Scotland & Ireland. For explanation see fig. A1.4.

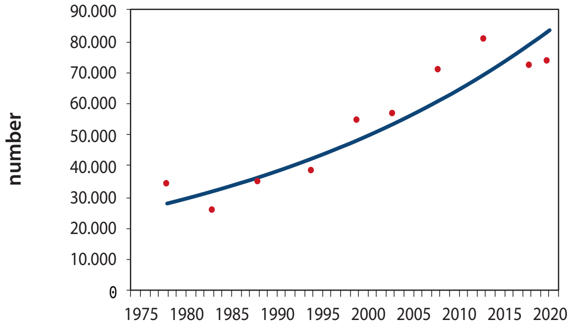


Figure A1.8. Flyway population trend for Barnacle Goose population Svalbard / SW Scotland. For explanation see fig. A1.4.

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.

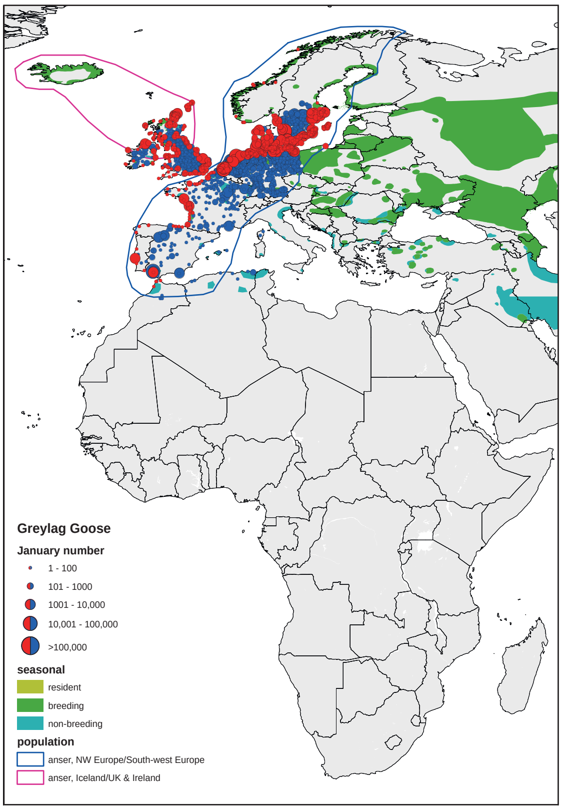


Figure A1.9. Distribution of Greylag Goose in the East Atlantic Flyway. For explanation see fig. A1.3.

Ran Schols / Agami

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	1975-2020	1,097	strong increase	2011-2020	1,004	stable	2016-2018	710000	780000
anser, Iceland/NW Europe	w	1977-2020	1,000	stable	2011-2020	0,946	moderate decline	2015-2019	76000	76000

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

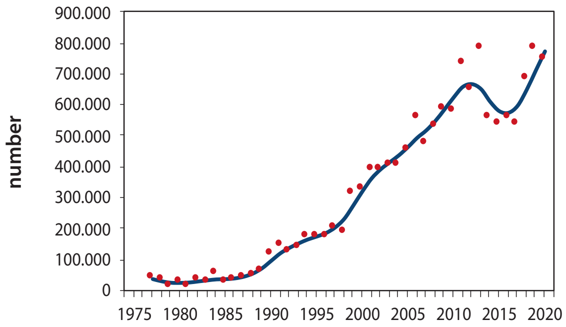


Figure A1.10. Flyway population trend for Greylag Goose population NW Europe / SW Europe. For explanation see fig. A1.4.

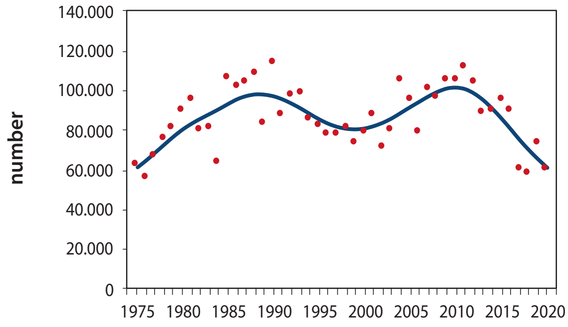


Figure A1.11. Flyway population trend for Greylag Goose population Iceland / UK & Ireland. For explanation see fig. A1.4.

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 (Swennen 1991). Breeding habitats often 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.*). More recently, American Razor Shell (*Ensis sp.*) has also been recorded in the diet (Kats 2007).

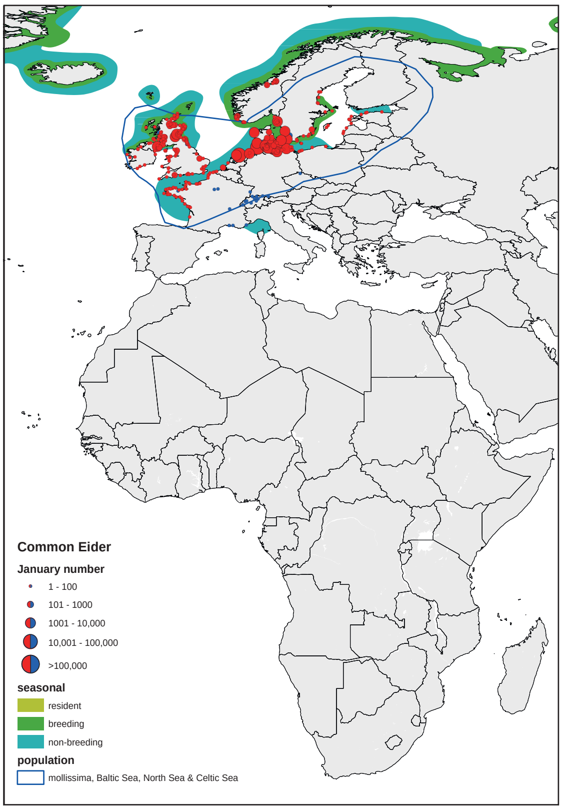
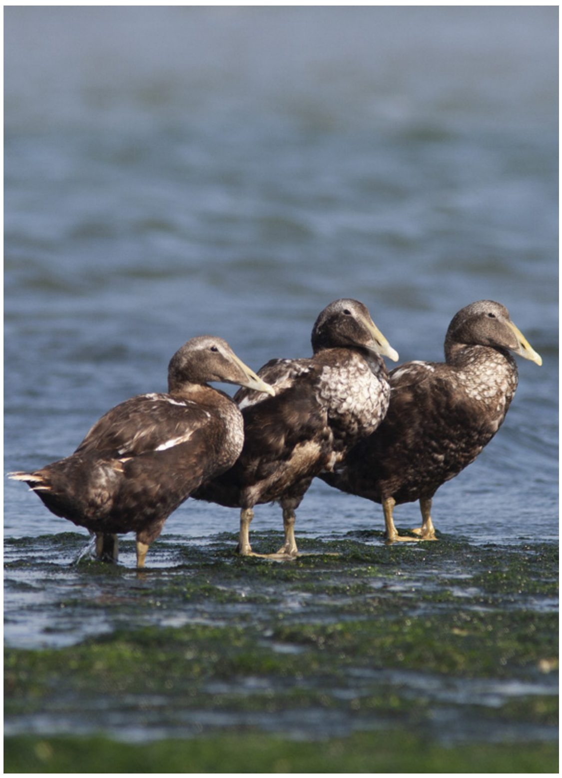


Figure A1.12. Distribution of Common Eider in the East Atlantic Flyway. For explanation see fig. A1.3.



Arnold Meijer / Blue Robin

population Common Eider	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
mollissima, Baltic Sea, North Sea & Celtic Sea	w	1980-2020	0,989	moderate decline	2011-2020	0,954	uncertain	2012-2018	560000	920000

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

Harvey van Diek

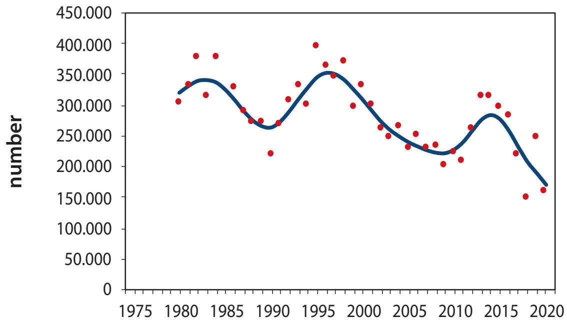


Figure A1.13. Flyway population trend for Common Eider population *mollissima Baltic*, North & Celtic Seas. For explanation see fig. A1.4.

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.

Harvey van Diek

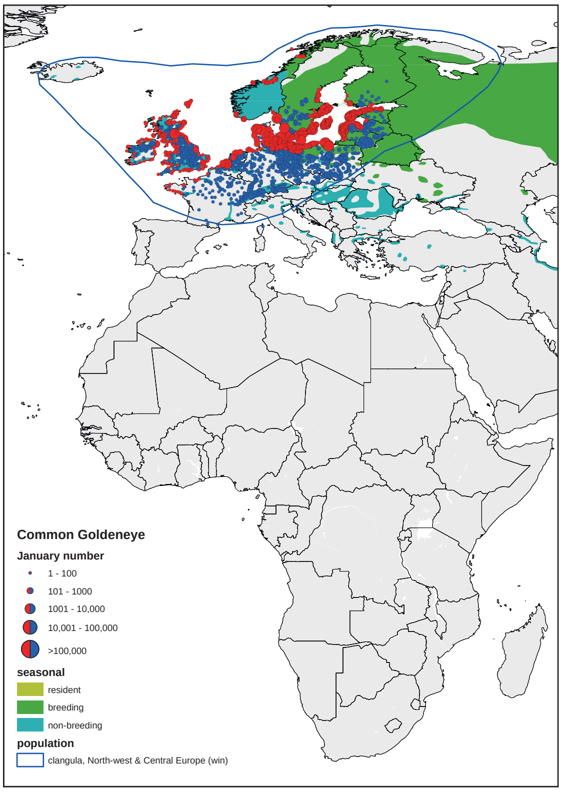


Figure A1.14. Distribution of Common Goldeneye in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1975-2020	1,004	moderate increase	2011-2020	0,980	moderate decline	2011-2018	750000	1500000

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

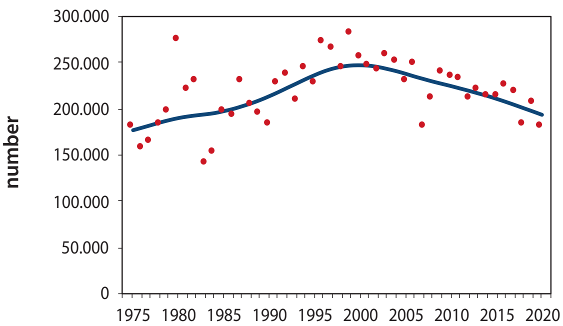


Figure A1.15. Flyway population trend for Common Goldeneye population NW & C Europe (win). For explanation see fig. A1.4.

Harvey van Diek

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.

Harvey van Diek

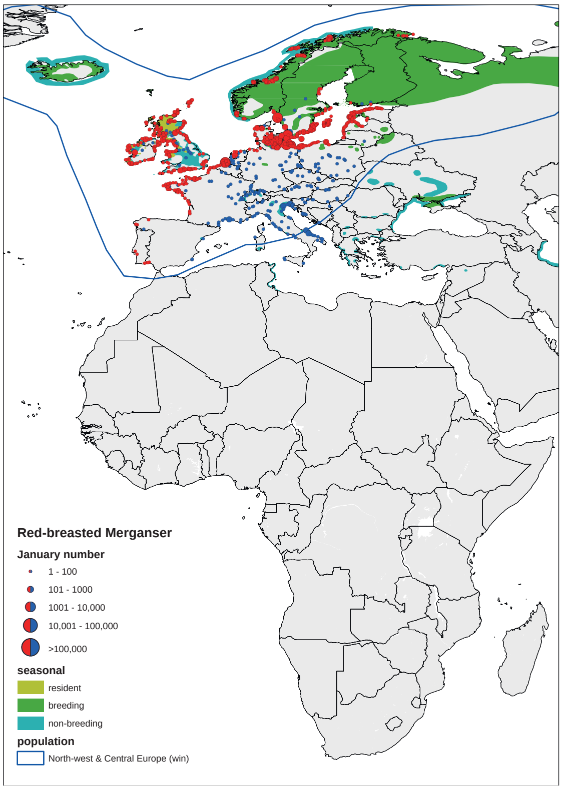


Figure A1.16. Distribution of Red-breasted Merganser in the East Atlantic Flyway. For explanation see fig. A1.3.

population Red-breasted Merganser	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW Europe and C Europe (winter)	w	1975-2020	1,009	moderate increase	2011-2020	1,009	stable	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.

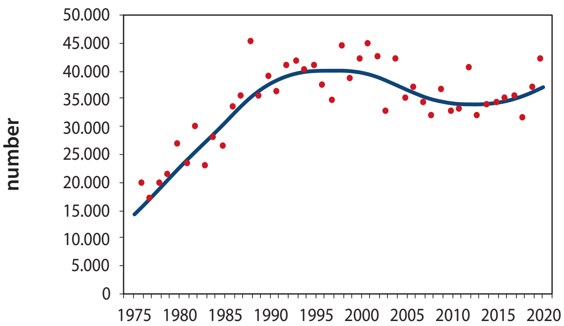
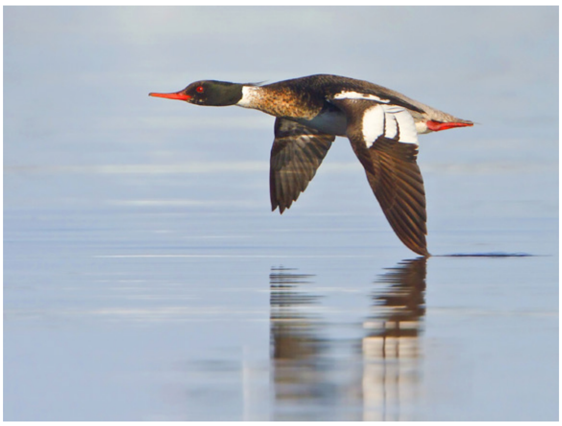


Figure A1.17. Flyway population trend for Red-breasted Merganser population NW & C Europe (win). For explanation see fig. A1.4.



Glenn Bartley / 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 relevant to the EAF. 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* (Kraan *et al.* 2006).

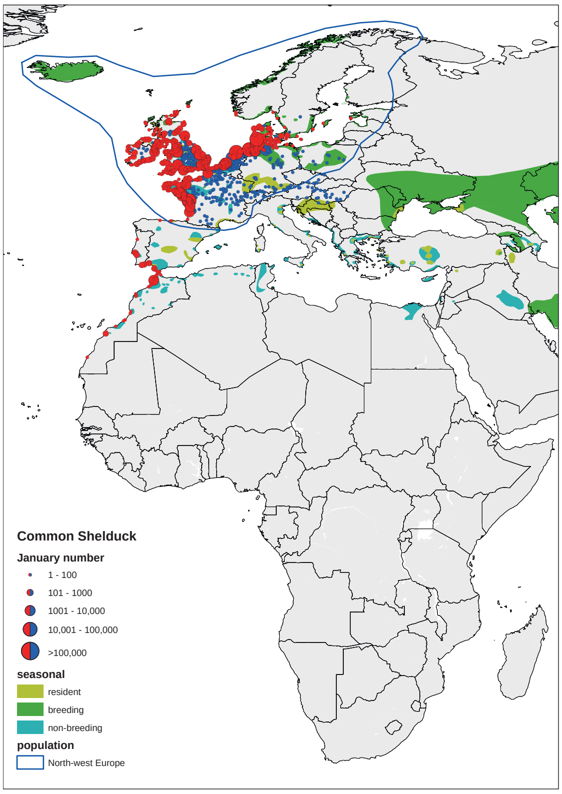


Figure A1.18. Distribution of Common Shelduck in the East Atlantic Flyway. For explanation see fig. A1.3.

Harvey van Diek

population Common Shelduck	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW Europe	w	1976-2020	1,007	moderate increase	2011-2020	0,991	moderate decline	2010-2019	310000	310000

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

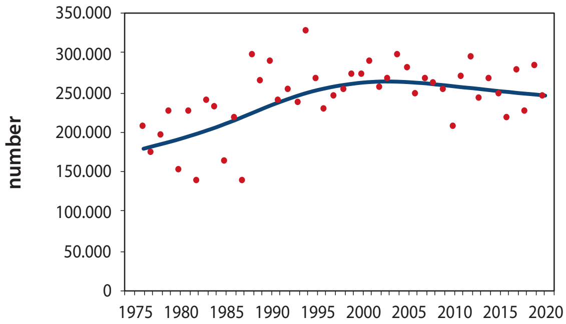


Figure A1.19. Flyway population trend for Common Shelduck population NW Europe. For explanation see fig. A1.4.

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.

Harvey van Diek

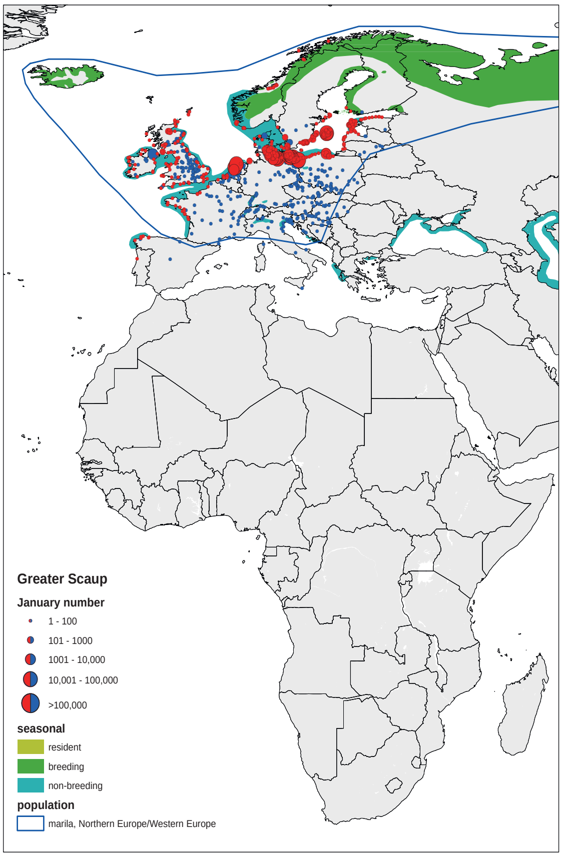


Figure A1.20. distribution map

population Greater Scaup	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Northern Europe/Western Europe	w	1976-2020	1,013	moderate increase	2011-2020	1,005	moderate increase	2014-2018	240000	280000

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

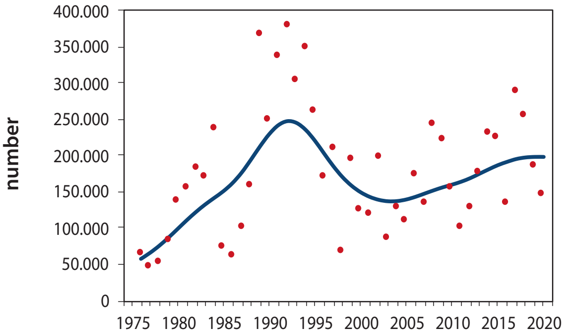


Figure A1.21. Flyway population trend for Greater Scaup population N Europe / W Europe. For explanation see fig. A1.4.

Markus Varesvuo / Agami

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.

Harvey van Diek

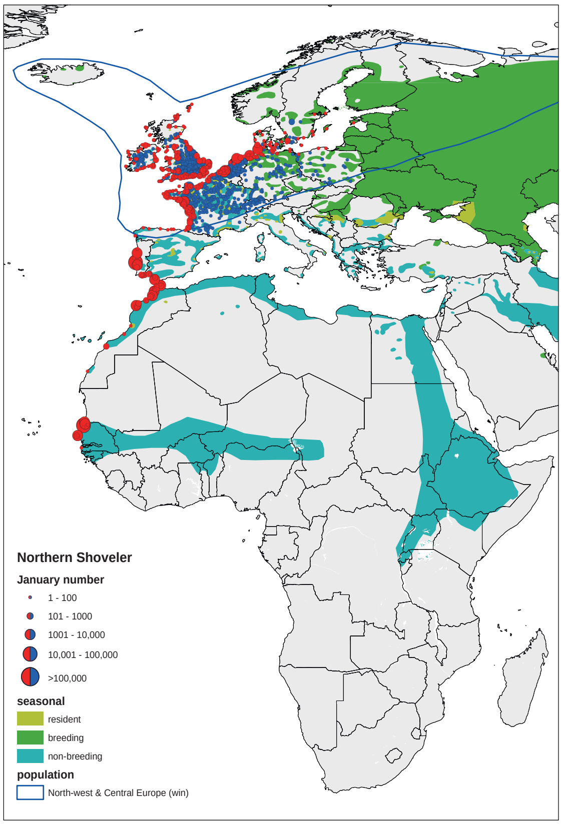


Figure A1.22. Distribution of Northern Shoveler in the East Atlantic Flyway. For explanation see fig. A1.3.

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.

population Northern Shoveler	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW & C Europe (winter)	w	1975-2020	1,028	moderate increase	2011-2020	1,071	strong 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.

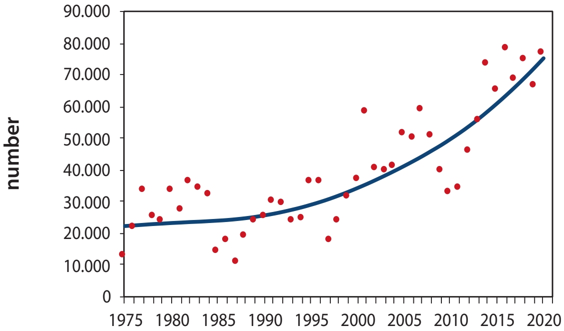


Figure A1.23. Flyway population trend for Northern Shoveler population NW & C Europe (win). For explanation see fig. A1.4.

Markus Varesvuo / Agami

Daniele Occhiato / Agami

Gadwall | *Mareca strepera* | Canard chipeau

This dabbling duck has an extremely wide distribution across the Palearctic and Nearctic regions. The Gadwall is strongly migratory in the north of its range, although birds breeding in temperate regions are largely sedentary. The population considered here is the one wintering in NW Europe; another population is distinguished as wintering in the Mediterranean and Black Sea regions. The species inhabits a range of different habitats, such as highly productive and eutrophic freshwater marshes or lakes habitats and in open lowland grassland. Gadwall are predominantly herbivorous and their diet consists of the seeds, leaves, roots and stems of aquatic plants.

Harvey van Diek

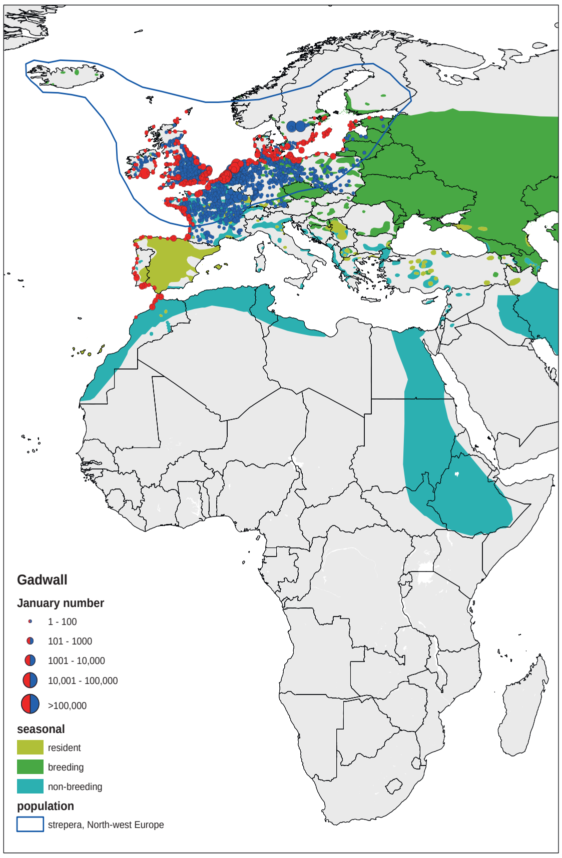


Figure A1.24. Distribution of Gadwall in the East Atlantic Flyway. For explanation see fig. A1.3.

population Gadwall	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW Europe	w	1975-2020	1,091	strong increase	2011-2020	1,064	strong increase	2013-2018	140000	140000

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

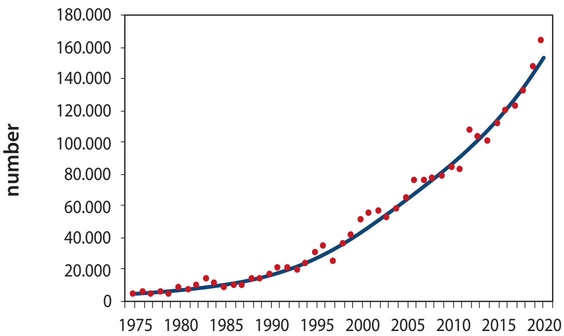


Figure A1.25. Flyway population trend for Gadwall population NW Europe. For explanation see fig. A1.4.

Harvey van Diek

Eurasian Wigeon | *Anas 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 that of the EAF, as >75% of this population winters in The Netherlands, UK and France (Fox *et al.* 2015). These birds breed mainly in the boreal zone of Fennoscandia, with large numbers in Finland, Sweden and Russia east to the Yenissei 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.* 2015). 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.

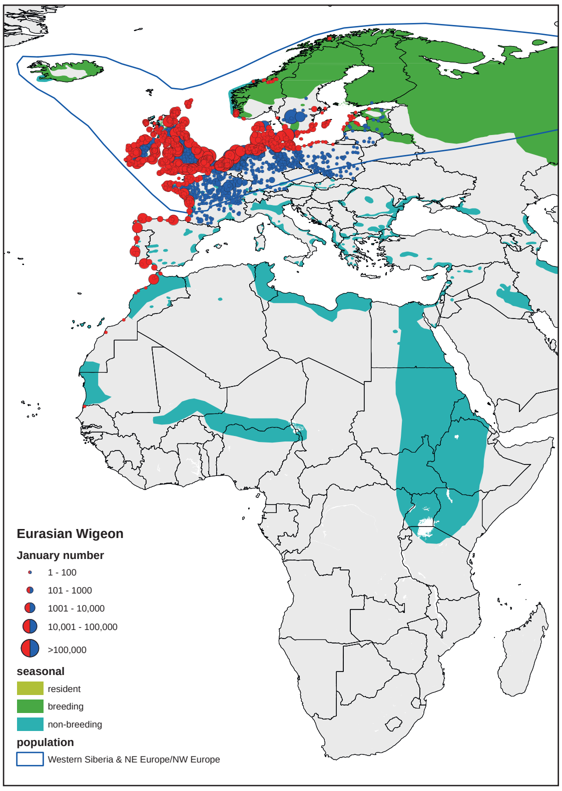


Figure A1.26. Distribution of Eurasian Wigeon in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1976-2020	1,023	moderate increase	2011-2020	1,008	stable	2014-2018	1300000	1600000

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

Markus Varesvuo / Agami

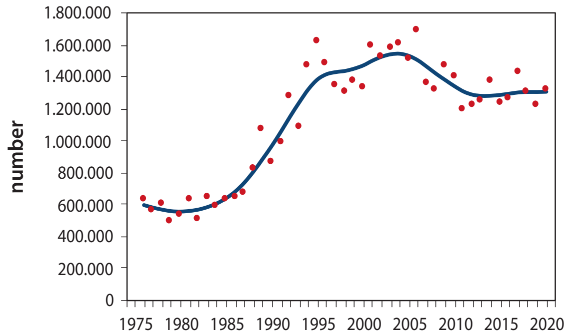
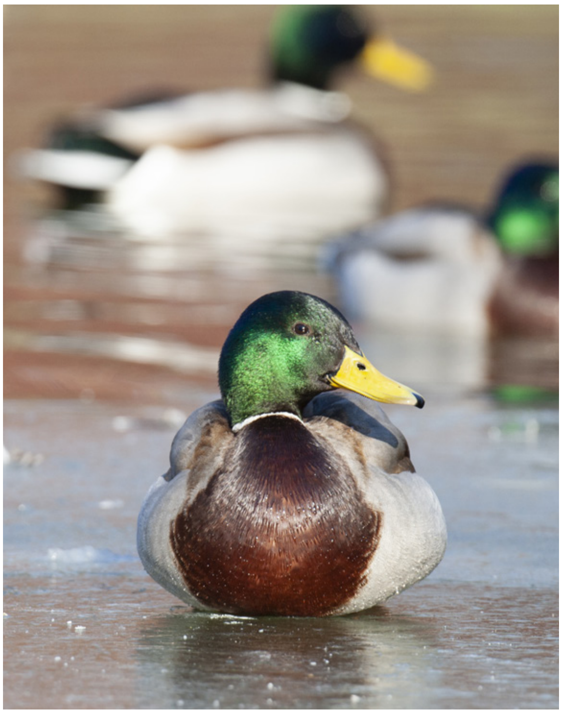


Figure A1.27. Flyway population trend for Eurasian Wigeon population W Siberia & NE Europe / NW Europe. For explanation see fig. A1.4.

Harvey van Diek

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 (Dessborn *et al.* 2011).



Arnold Meijer / Blue Robin

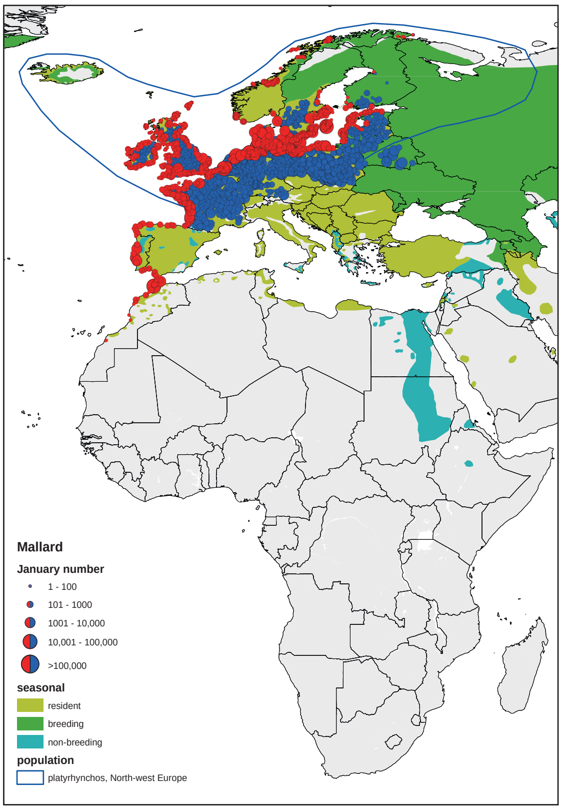


Figure A1.28. Distribution of Mallard in the East Atlantic Flyway. For explanation see fig. A1.3.

population Mallard	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW & E Europe & Siberia/NW Europe	w	1976-2020	0,994	moderate decline	2011-2020	0,979	moderate decline	2013-2018	4500000	7100000

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

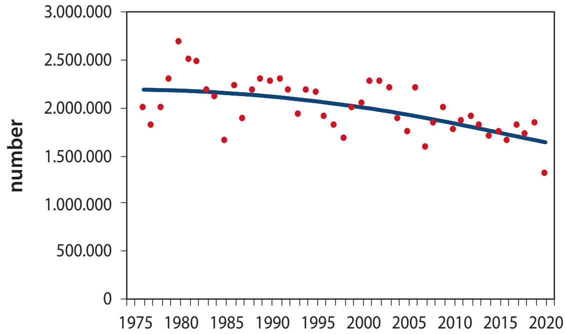


Figure A1.29. Flyway population trend for Mallard population NW Europe. For explanation see fig. A1.4.

Daniele Occhiato / 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.

Harvey van Diek

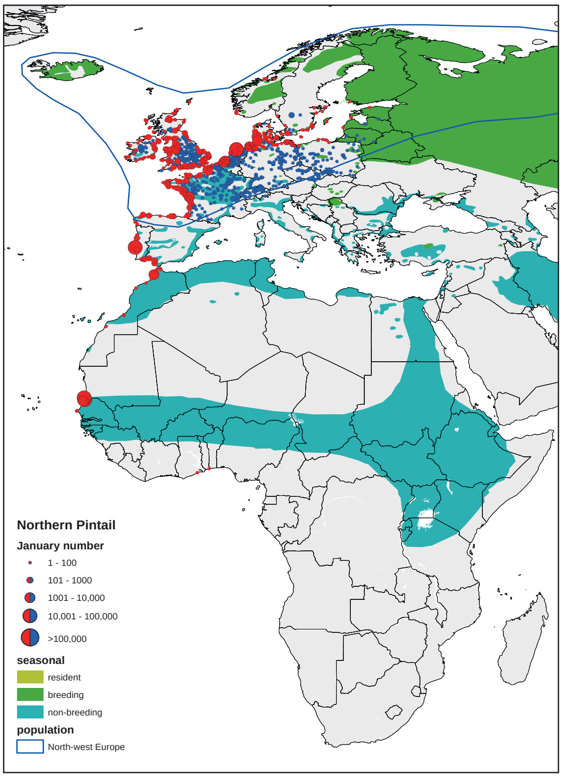


Figure A1.30. Distribution of Northern Pintail in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1975-2020	1,013	moderate increase	2011-2020	1,019	moderate increase	2014-2018	74000	74000

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

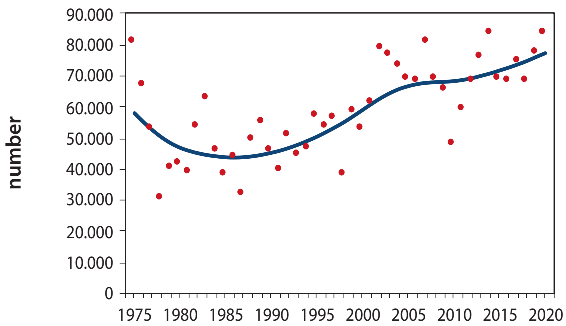


Figure A1.31. Flyway population trend for Northern Pintail population NW Europe. For explanation see fig. A1.4.

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 season 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 (Dessborn *et al.* 2011).

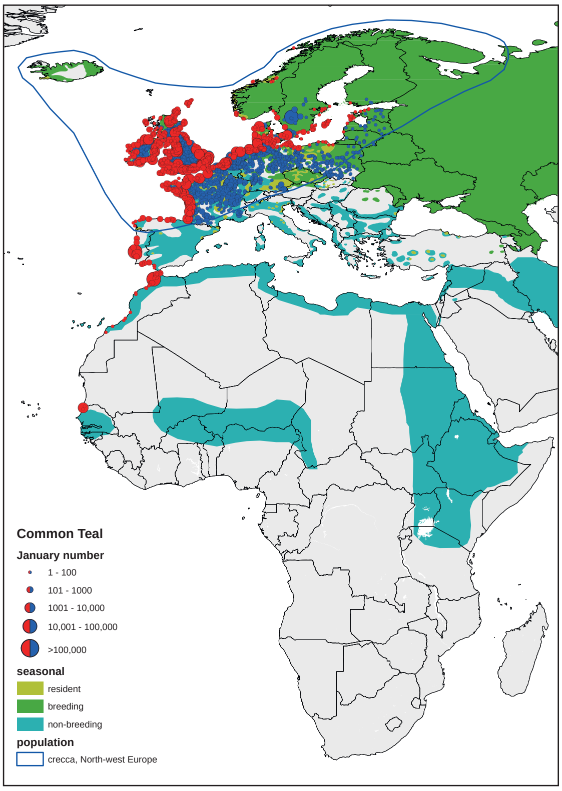


Figure A1.32. Distribution of Common Teal in the East Atlantic Flyway. For explanation see fig. A1.3.

population Common Teal	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW & E Europe & Siberia/NW Europe	w	1975-2020	1,020	moderate increase	2011-2020	1,005	stable	2011-2018	670000	670000

Table A1.15. Summary of flyway trend and population size estimates for Common Teal. For explanation see table A1.2.

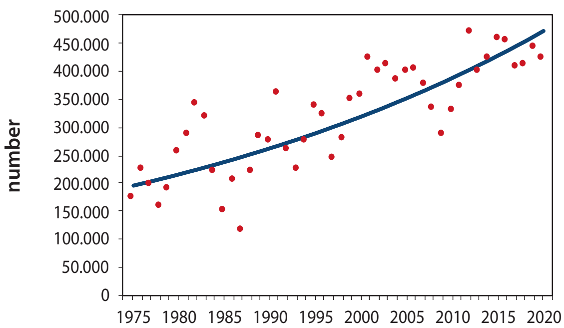


Figure A1.33. Flyway population trend for Common Teal population *crecca* NW Europe. For explanation see fig. A1.4.

Hans Germeraad / Agami

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 freshwater 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.

Ralph Martin / Agami

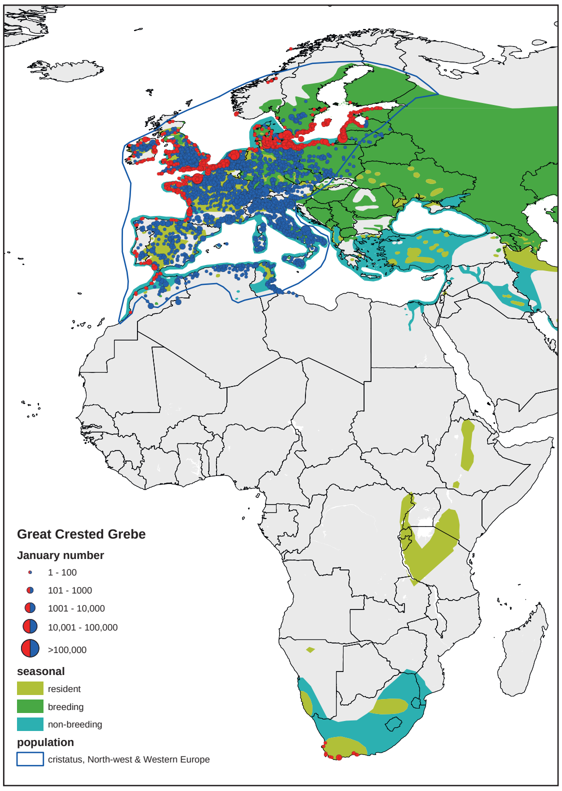


Figure A1.34. Distribution of Great Crested Grebe in the East Atlantic Flyway. For explanation see fig. A1.3.

population Great Crested Grebe	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
cristatus, NW Europe & W Europe	w	1977-2020	1,019	moderate increase	2011-2020	1,005	stable	2007-2018	500000	690000

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

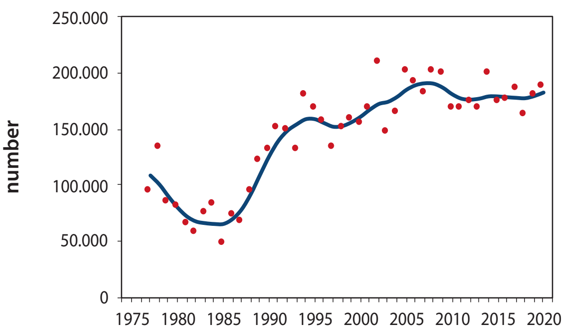


Figure A1.35. Flyway population trend for Great Crested Grebe population *cristatus* NW & W Europe. For explanation see fig. A1.4.

Markus Varesvuo / Agami

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. 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

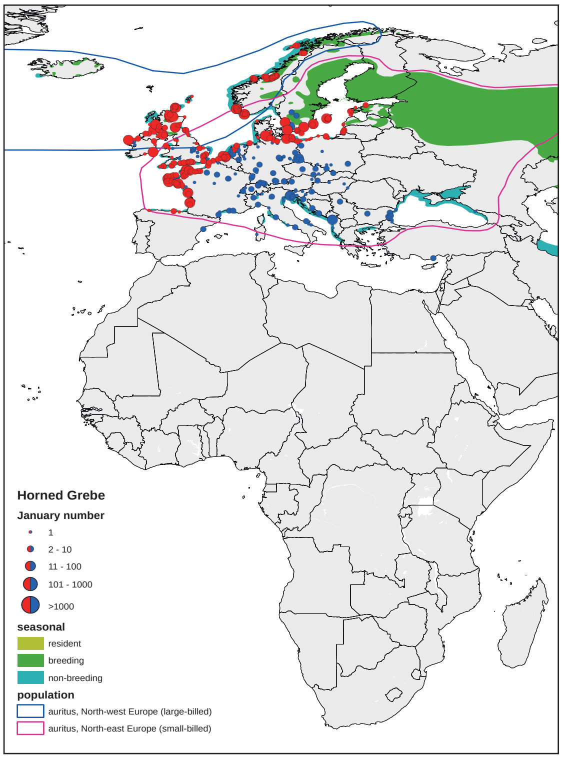


Figure A1.36. Distribution of Horned Grebe in the East Atlantic Flyway. For explanation see fig. A1.3.

population Horned Grebe	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NW & NE Europe (winter) combi of pop.	w	1993-2020	1,013	moderate increase	2011-2020	0,995	stable			
NW Europe (large billed, winter)								1994-2018	4600	5300
NE Europe (small billed, winter)								2011-2018	22000	31000

Table A1.17. Summary of flyway trend and population size estimates for Horned Grebe. For explanation see table A1.2.

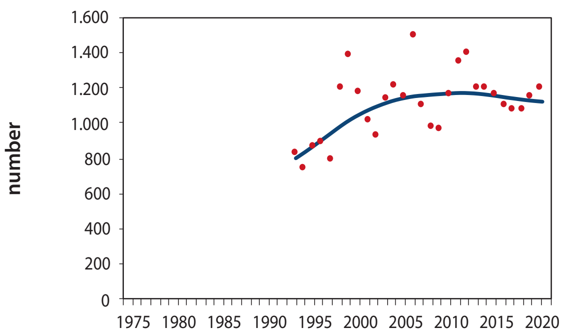


Figure A1.37. Flyway population trend for Horned Grebe population *auritus* (combined). For explanation see fig. A1.4.

Arie Ouwerkerk / Agami

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 in 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.

Harvey van Diek

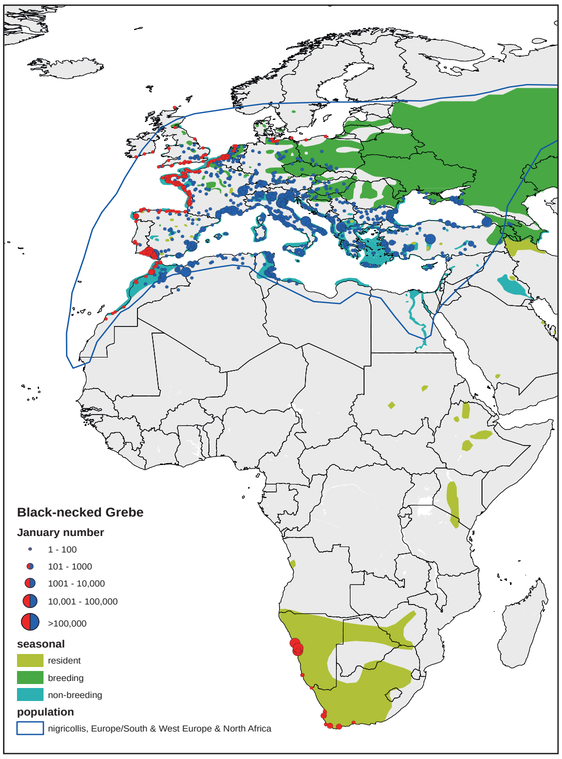


Figure A1.38. Distribution of Black-necked Grebe in the East Atlantic Flyway. For explanation see fig. A1.3.

population Black-necked Grebe	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Europe/S & W Europe & N Africa	w	2003-2020	0,999	stable	2011-2020	1,009	stable	2007-2018	110000	170000

Table A1.18. Summary of flyway trend and population size estimates for Black-necked Grebe. For explanation see table A1.2.

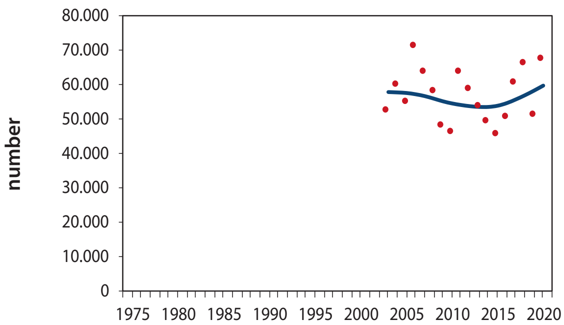


Figure A1.39. Flyway population trend for Black-necked Grebe population *nigricollis* Europe / S & W Europe & N Africa. For explanation see fig. A1.4.

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, saltpans, and lakes, but also intertidal mudflat areas. It feeds on crustaceans, diatoms and other small food items, especially brine shrimp *Artemia*.

The combined average sum of Greater Flamingos of the W African population counted in 2016 – 2020 is almost 94,000. Considering that most important sites have been included in this estimate, but that some mixing with the W Mediterranean population will occur, a population size of 90.000 – 110.000 is tentatively proposed for the 2016-2020 period.

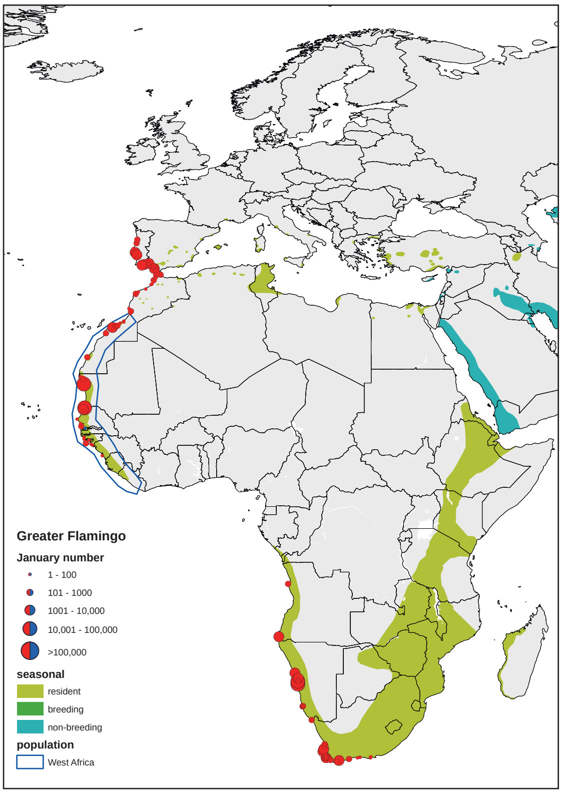


Figure A1.40. Distribution of Greater Flamingo the East Atlantic Flyway. For explanation see fig. A1.3.

population Greater Flamingo	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa	w	1979-2020	0,992	moderate decline	2011-2020	1,072	moderate increase	2016-2020	90000	110000

Table A1.19. Summary of flyway trend and population size estimates for Greater Flamingo. For explanation see table A1.2.

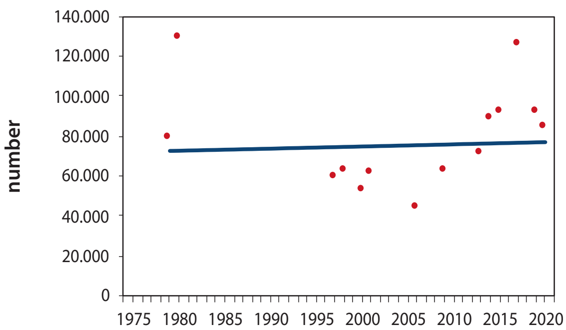


Figure A1.41. Flyway population trend for Greater Flaingo population W Africa. For explanation see fig. A1.4.

Arnold Meijer / Blue Robin

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.

The combined average sum of Lesser Flamingo of the W African population counted in 2016 – 2020 is more than 25,000. Considering that almost all important sites have been included in this estimate, a population size of 25,000 – 30,000 is proposed for the 2016-2020 period.

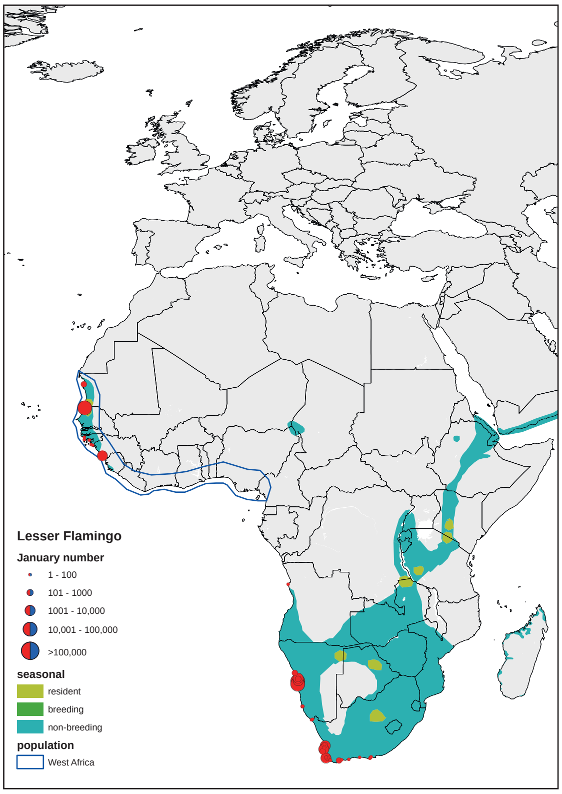


Figure A1.42. Distribution of Lesser Flamingo the East Atlantic Flyway. For explanation see fig. A1.3.

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	1991-2020	1,082	moderate increase	2011-2020	1,328	strong increase	2016-2020	25000	30000

Table A1.20. Summary of flyway trend and population size estimates for Lesser Flamingo. For explanation see table A1.2.

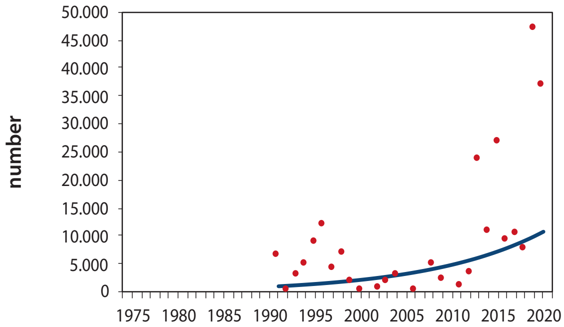


Figure A1.43. Flyway population trend for Lesser Flaingo population W Africa. For explanation see fig. A1.4.

Karel Mauer / Agami

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 (Piersma *et al.* 2012). 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.

Harvey van Diek

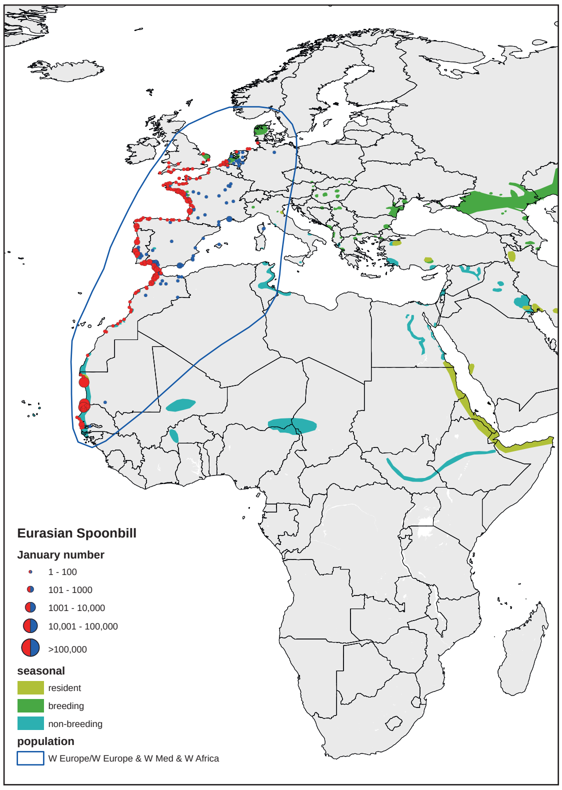


Figure A1.44. Distribution of Eurasian Spoonbill the East Atlantic Flyway. For explanation see fig. A1.3.

Arnold Meijer / Blue Robin

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

Table A1.21. Summary of flyway trend and population size estimates for Eurasian Spoonbill. For explanation see table A1.2.

Grey Heron | *Ardea cinerea* | Héron cendré

Palearctic 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 even 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 *A. c. monicae* occurs. In the African region also mixing with other African populations will occur.

Harvey van Diek

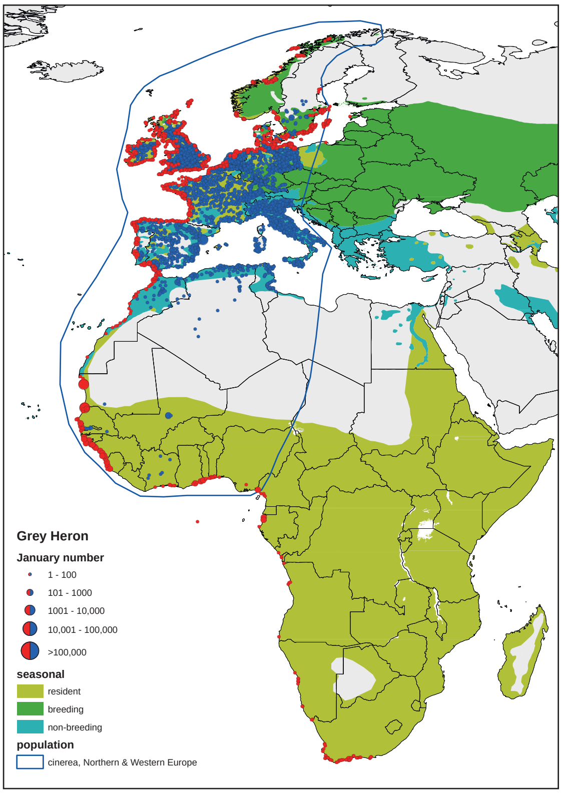


Figure A1.45. Distribution of Grey Heron the East Atlantic Flyway. For explanation see fig. A1.3.

population Grey Heron	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
N & W Europe	w	1990-2020	1,010	moderate increase	2011-2020	1,014	moderate increase	2008-2018	320000	380000

Table A1.22. Summary of flyway trend and population size estimates for Grey Heron. For explanation see table A1.2.

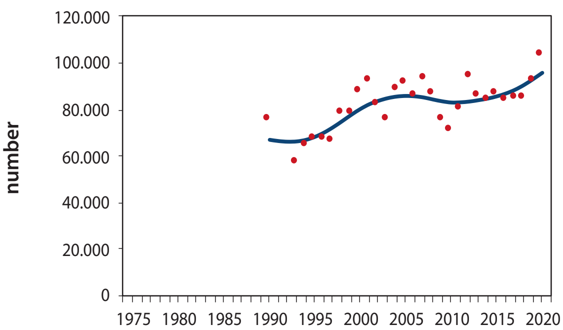


Figure A1.46. Flyway population trend for Grey Heron population *cinerea* N & W Europe. For explanation see fig. A1.4.

Arnold Meijer / Blue Robin

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.

Arnold Meijer / Blue Robin

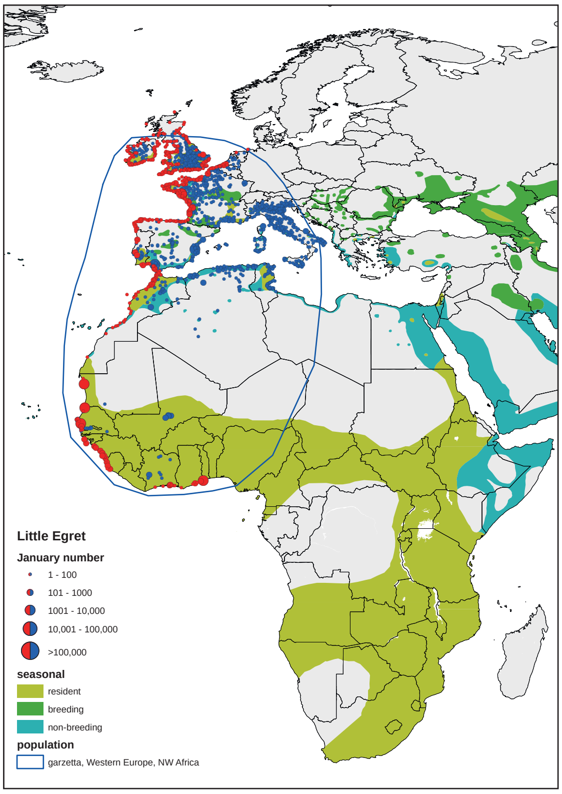


Figure A1.47. Distribution of Little Egret in the East Atlantic Flyway. For explanation see fig. A1.3.

population Little Egret	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
SW Europe	b	1980-2018	1,024	moderate increase	2008-2017	1,022	moderate increase	2003-2018	95000	105000

Table A1.23. Summary of flyway trend and population size estimates for Little Egret. For explanation see table A1.2.

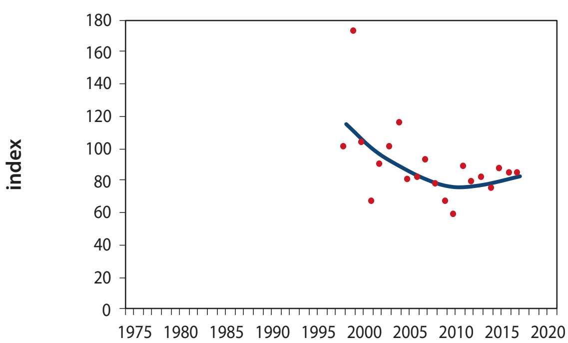


Figure A1.48. PECMBS population trend for Little Egret. For explanation see fig. A1.4.

Harvey van Diek

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.

The combined average sum of Western Reef-egrets counted in 2016 – 2020 is almost 16,000. Considering that considerable numbers will certainly have been missed or underestimated, a population size of 25,000 – 35,000 is tentatively proposed for the 2016-2020 time period.

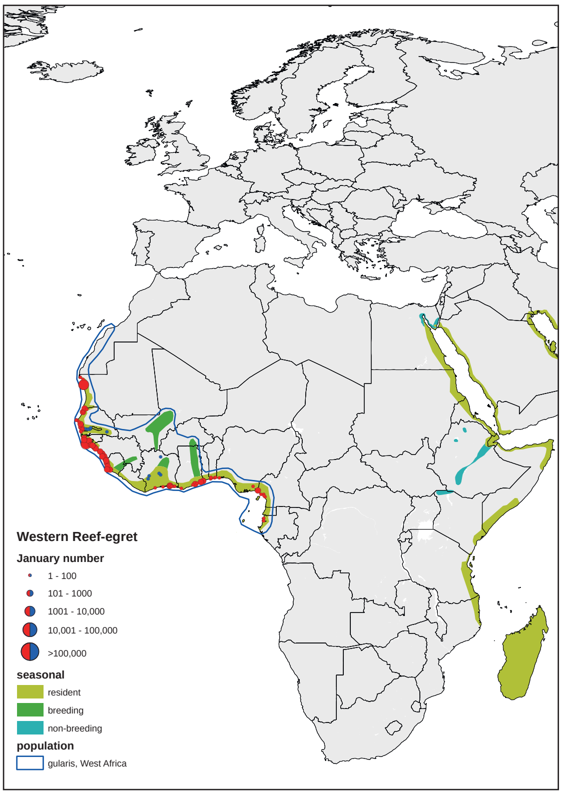


Figure A1.49. Distribution of Western Reef Egret in the East Atlantic Flyway. For explanation see fig. A1.3.

population Western Reef-Egret	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
gularis, W Africa	w	1990-2020	0,978	moderate decline	2011-2020	0,978	moderate decline	2016-2020	25000	35000

Table A1.24. Summary of flyway trend and population size estimates for Western Reef Egret. For explanation see table A1.2.

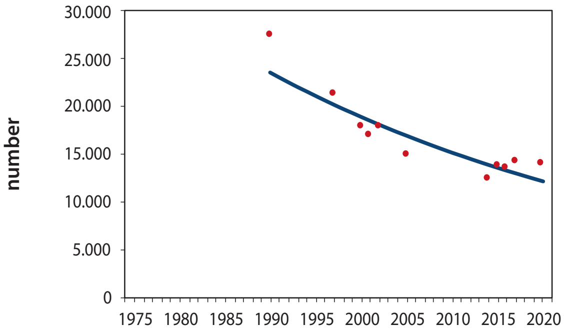


Figure A1.50. Flyway population trend for Western Reef Egret population *gularis* W Africa. For explanation see fig. A1.4.

Saverio Gatto / Agami

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 Guinee 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.

The combined average sum of Great White Pelicans counted in W Africa in 2016 – 2020 was more than 27,000. Despite the fact that birds will have been missed and will also be present at sites not covered during the total counts of 2017 and 2020, we think that the population size of 60,000 in 2014 (van Roomen *et al.* 2015) is an overestimate, especially as the flyway trend has been increasing since then. A new population estimate of roughly 35,000 – 45,000 is tentatively proposed for the 2016-2020

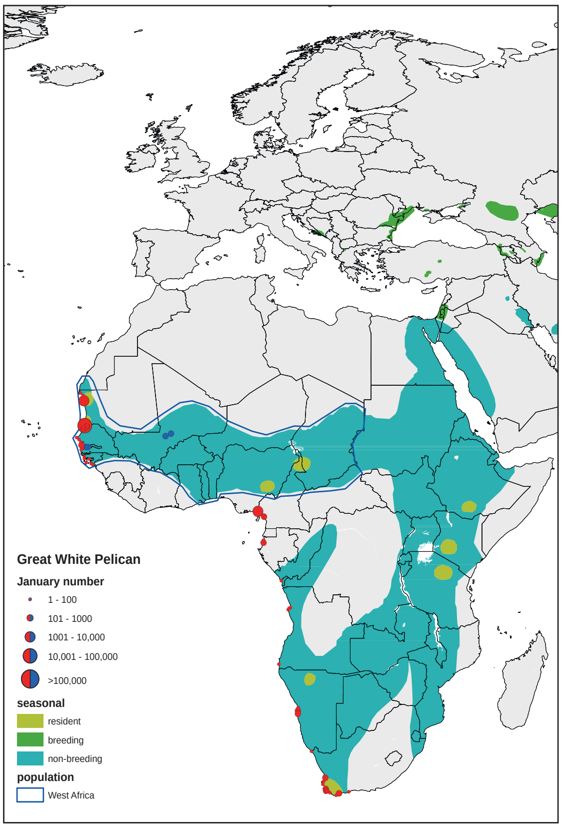


Figure A1.51. Distribution of Great White Pelican in the East Atlantic Flyway. For explanation see fig. A1.3.

period. Recently (2020-2021), many Great White Pelicans died from avian influenza, which impacted especially breeding areas in the Senegal Delta. A simultaneous count of the breeding colonies is recommended to gain more insight in the size of this population.

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	1991-2020	1,044	moderate increase	2011-2020	1,023	uncertain	2016-2020	35000	45000

Table A1.25. Summary of flyway trend and population size estimates for Great White Pelican. For explanation see table A1.2.

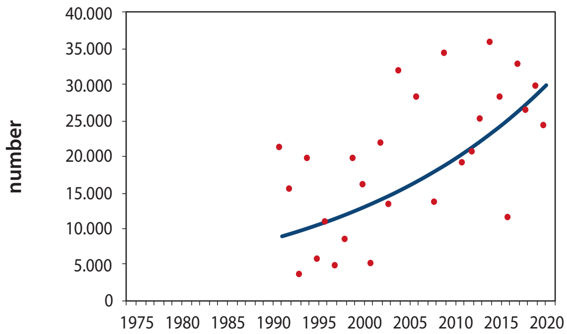


Figure A1.52. Flyway population trend for Great White Pelican population W Africa. For explanation see fig. A1.4.

Harvey van Diek

Hans Schekkerman

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 bentho-pelagic fish species varying in size from 10 to 20 cm (Veen *et al.* 2012). 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.

The combined average sum of Great Cormorant of the *lucidus* population counted in West Africa in 2016 – 2020 was almost 30,000. It is proposed that a population size estimate of about 35,000 – 45,000 birds is used for the 2016-2020 years.

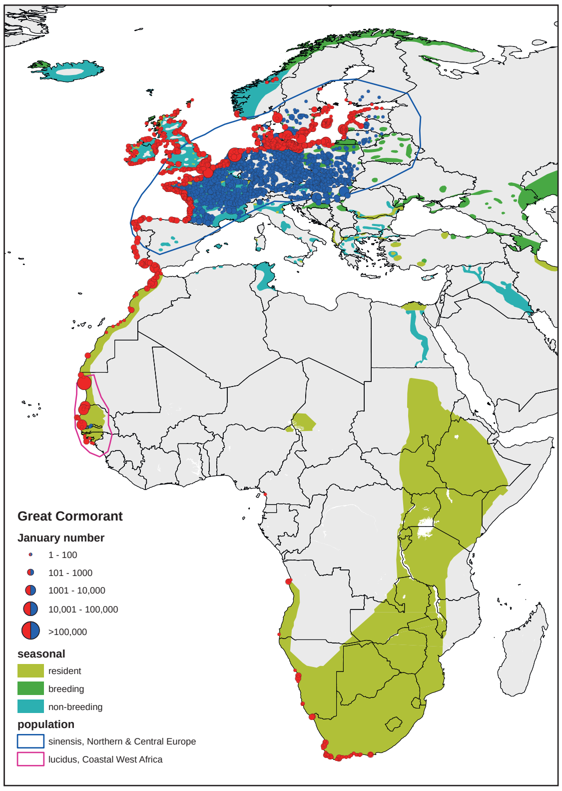


Figure A1.53. Distribution of Great Cormorant in the East Atlantic Flyway. For explanation see fig. A1.3.

Arnold Meijer / Blue Robin

population Great Cormorant	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
sinensis, N & C Europe	w	1978-2020	1,071	strong increase	2011-2020	1,073	strong increase	2005-2018	610000	740000
lucidus, coastal W Africa	w	2000-2020	1,028	moderate increase	2011-2020	1,059	moderate increase	2016-2020	35000	45000

Table A1.26. Summary of flyway trend and population size estimates for Great Cormorant. For explanation see table A1.2.

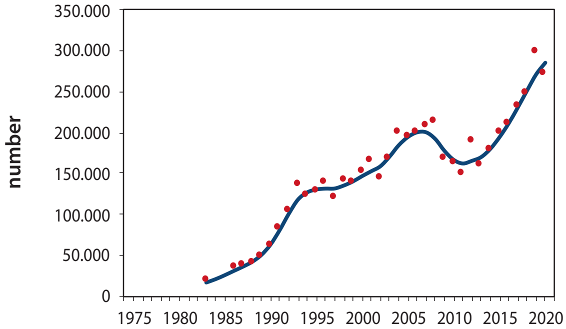


Figure A1.54. Flyway population trend for Great Cormorant population *sinensis* N & C Europe. For explanation see fig. A1.4.

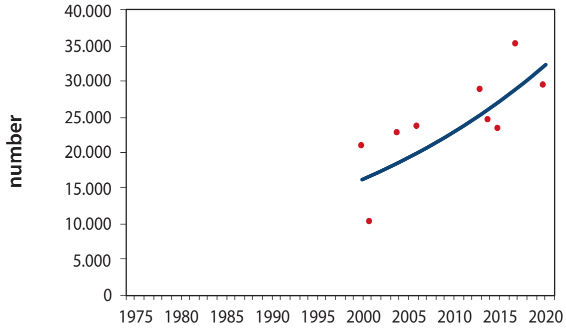


Figure A1.55. Flyway population trend for Great Cormorant population *lucidus* coastal W Africa. For explanation see fig. A1.4.

Vincent Legrand / Agami

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*.

A population estimate is difficult due to poor breeding seasonality, limited breeding site fidelity and the difficulty of obtaining accurate counts of large colonies, whilst numbers fluctuate considerably in relation to fish stocks (Kemper & Simmons 2015).

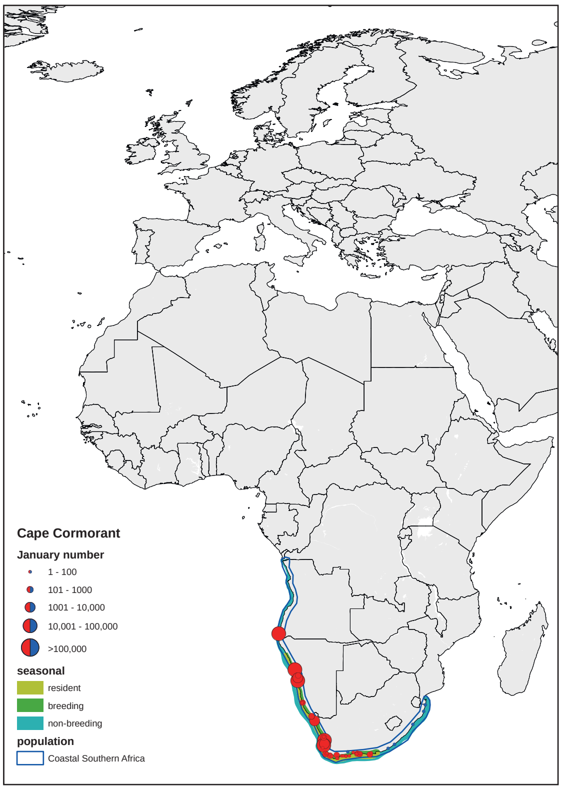


Figure A1.56. Distribution of Cape Cormorant in the East Atlantic Flyway. For explanation see fig. A1.3.

population Cape Cormorant	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
coastal S Africa	w	1992-2020	1,028	moderate increase	2011-2020	1,054	uncertain	2005-2014	351000	351000

Table A1.27. Summary of flyway trend and population size estimates for Cape Cormorant. For explanation see table A1.2.

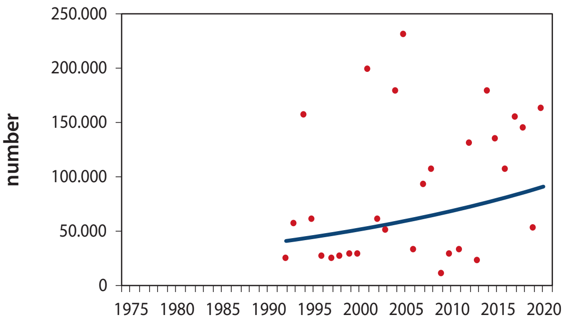


Figure A1.57. Flyway population trend for Cape Cormorant population coastal Southern Africa. For explanation see fig. A1.4.

Harvey van Diek

African Oystercatcher | *Haematopus moquini* | Huîtrier 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 relatively 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 (Hockey & Underhill 1984). 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.

Harvey van Diek

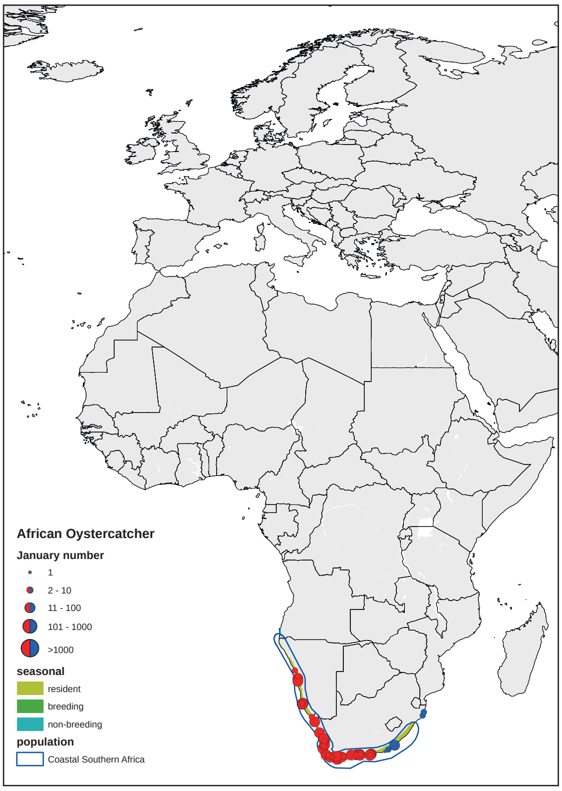


Figure A1.58. Distribution of African Oystercatcher in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1993-2020	0,992	stable	2011-2020	0,980	uncertain	1997-2003	6600	6700

Table A1.28. Summary of flyway trend and population size estimates for African Oystercatcher. For explanation see table A1.2.

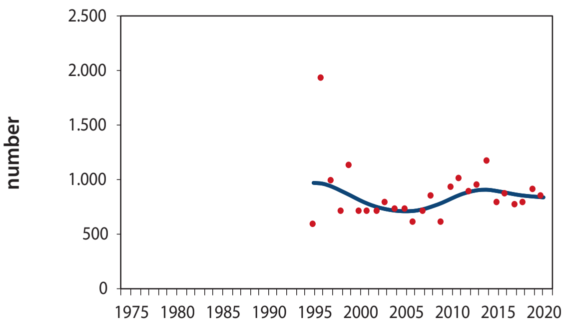


Figure A1.59. Flyway population trend for Africa Oystercatcher population coastal Southern Africa. For explanation see fig. A1.4.

Marc Guyt / Agami

Eurasian Oystercatcher | *Haematopus ostralegus* | Huîtrier 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; Méndez *et al.* 2020). 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).

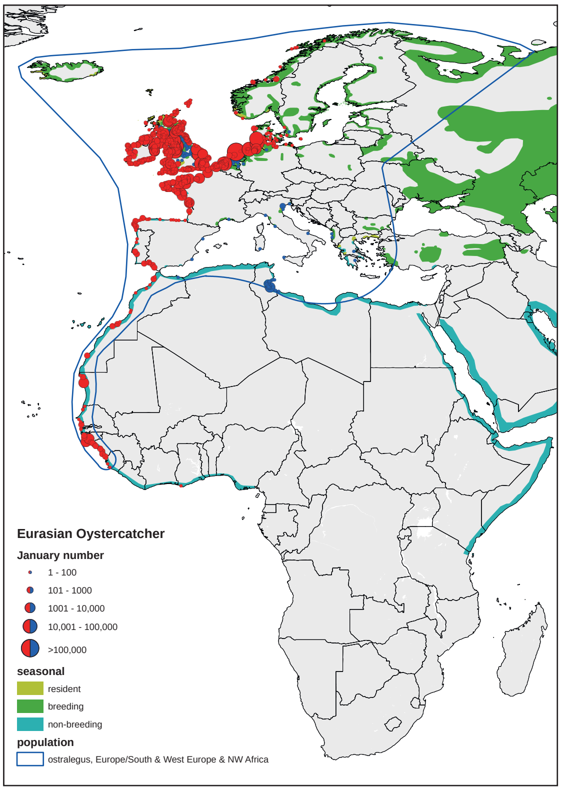


Figure A1.60. Distribution of Eurasian Oystercatcher in the East Atlantic Flyway. For explanation see fig. A1.3.

population Eurasian Oystercatcher	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Europe/S. & W Europe & NW Africa	w	1976-2020	1,000	stable	2011-2020	0.976	moderate decline	2007-2018	750000	970000

Table A1.29. Summary of flyway trend and population size estimates for Eurasian Oystercatcher. For explanation see table A1.2.

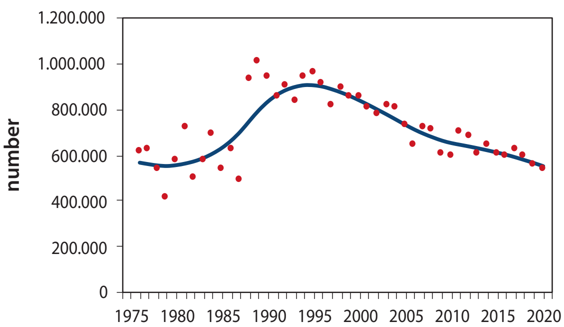


Figure A1.61. Flyway population trend for Eurasian Oystercatcher population *ostralegus* Europe. For explanation see fig. A1.4.

Harvey van Diek

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 poulation 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.

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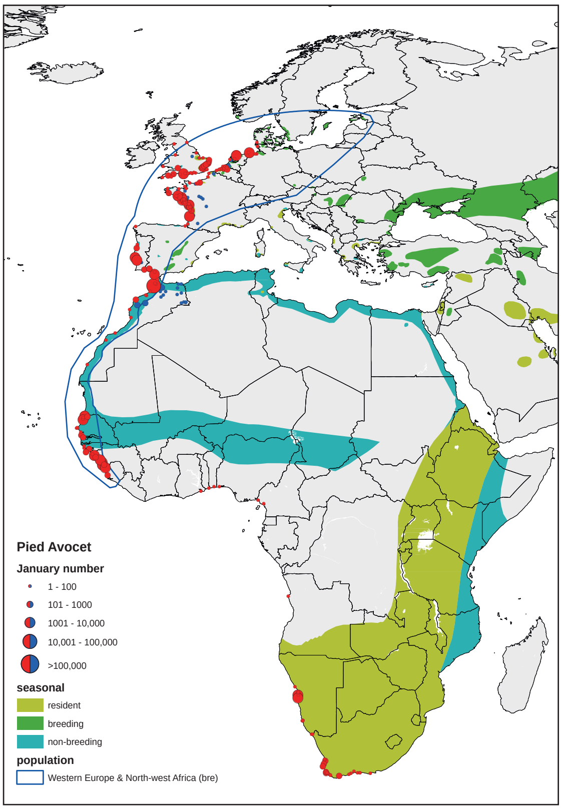


Figure A1.62. Distribution of Pied Avocet in the East Atlantic Flyway. For explanation see fig. A1.3.

population Pied Avocet	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Europe & NW Africa (breeding)	w	1978-2020	1,014	moderate increase	2011-2020	1,010	stable	2007-2018	100000	110000

Table A1.30. Summary of flyway trend and population size estimates for Pied Avocet. For explanation see table A1.2.

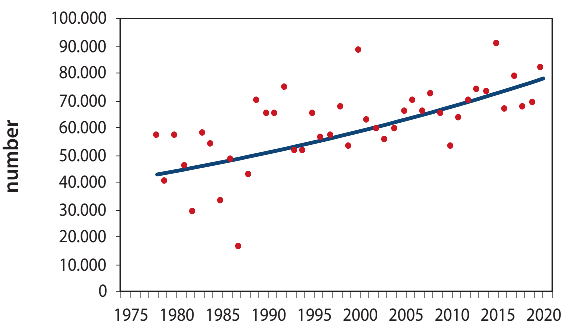
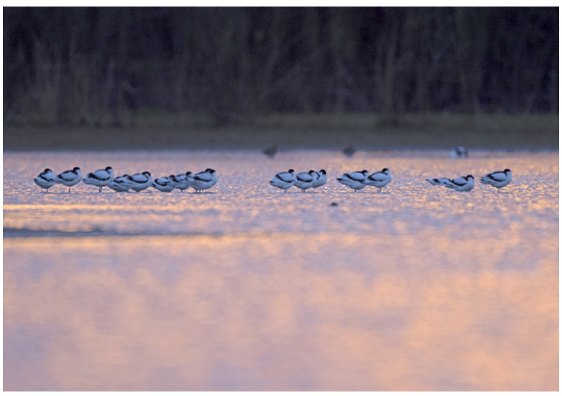


Figure A1.63. Flyway population trend for Pied Avocet population W Europe & NW Africa (bre). For explanation see fig. A1.4.



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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 recognized 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 only, 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 (Exo *et al.* 2019). 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 (Durell & Kelly 1990; Perez-Hurtado *et al.* 1997).

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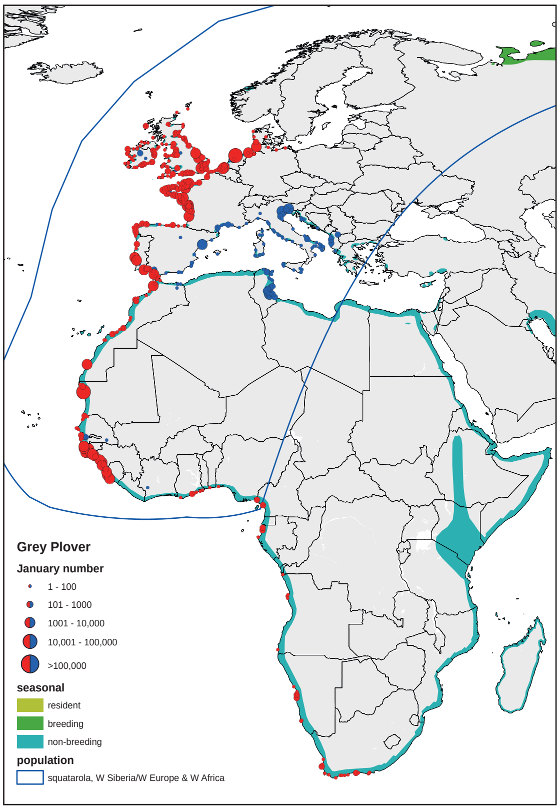

Figure A1.64. Distribution of Grey Plover in the East Atlantic Flyway. For explanation see fig. A1.3.

population Grey Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Siberia/W Europe & W Africa	w	1978-2020	1,015	moderate increase	2011-2020	0,973	moderate decline	2010-2018	200000	200000

Table A1.31. Summary of flyway trend and population size estimates for Grey Plover. For explanation see table A1.2.

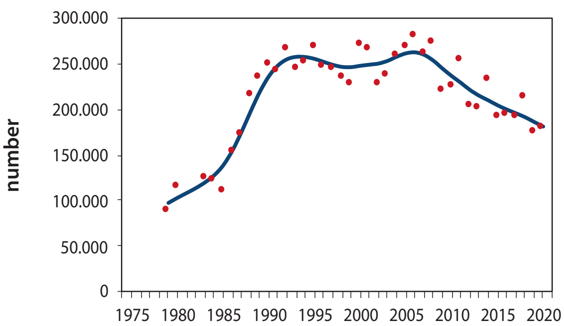

Figure A1.65. Flyway population trend for Grey Plover population W Siberia / W Europe & W Africa. For explanation see fig. A1.4.

Harvey van Diek

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

Three subspecies are recognized. 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. psammodromus* 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 (Pedro & Ramos 2009).

The combined average sum of Common Ringed Plovers of the *psammodromus* population counted in W Africa in 2016 – 2020 is more than 170,000. As considerable numbers will have been present in sites not surveyed, but probably also unknown numbers of the *tundrae* population will have been present, a population size estimate of 220,000 – 280,000 is proposed for the 2016-2020 timeframe.

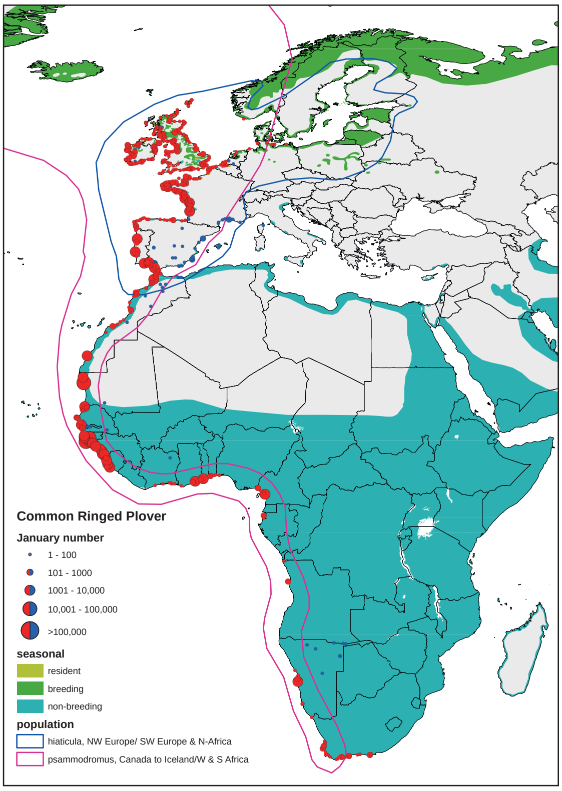

Figure A1.66. Distribution of Common Ringed Plover in the East Atlantic Flyway. For explanation see fig. A1.3.

Harvey van Diek

population Common Ringed Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
hiaticula, NW Europe/ SW Europe & N-Africa	w	1978-2016	1,015	moderate increase	2011-2020	0,985	stable	2007-2019	50000	68000
psammodromus, Canada to Iceland/ W & S Africa	w	1979-2020	0,982	moderate decline	2011-2020	0,991	stable	2010-2014	240000	240000

Table A1.32. Summary of flyway trend and population size estimates for Common Ringed Plover. For explanation see table A1.2.

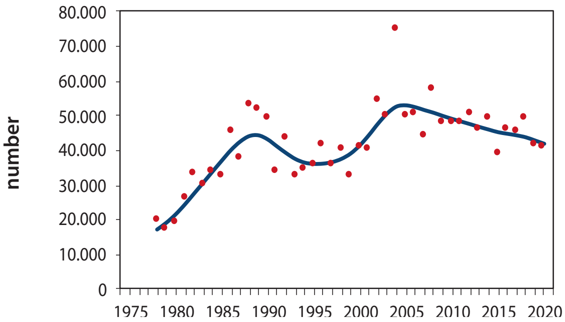

Figure A1.67. Flyway population trend for Common Ringed Plover population *hiaticula* N Europe / Europe & N Africa. For explanation see fig. A1.4.

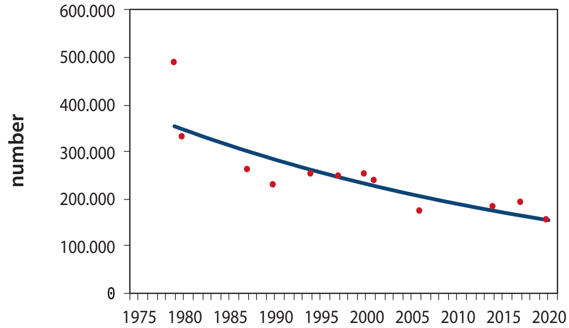

Figure A1.68. Flyway population trend for Common Ringed Plover population *psammodromus* Nearctic & Iceland / W & S Africa. For explanation see fig. A1.4.

Arnold Meijer / Blue Robin

Harvey van Diek

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 (Lloyd 2008). 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.

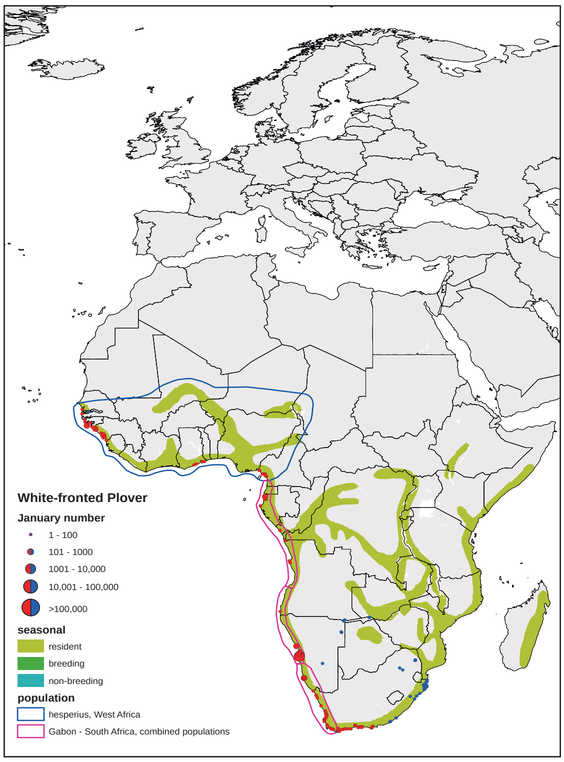

Figure A1. 69. Distribution of White-fronted Plover in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1990-2020	1,002	moderate decline	2011-2020	0,929	moderate decline	1998-2007	10000	15000
Gabon - South Africa, combi of pop.	w	1992-2020	0,967	stable	2011-2020	0,976	moderate decline			

Table A1.33. Summary of flyway trend and population size estimates for White-fronted Plover. For explanation see table A1.2.

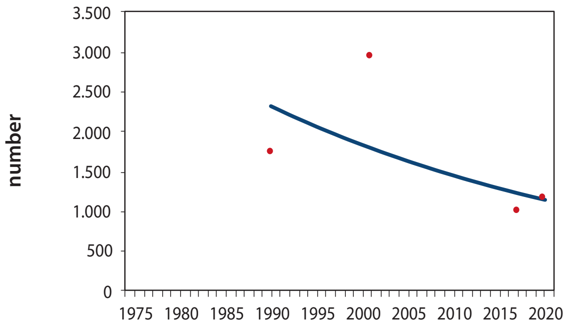

Figure A1.70. Flyway population trend for White-fronted Plover population *hesperius* W Africa. For explanation see fig. A1.4.

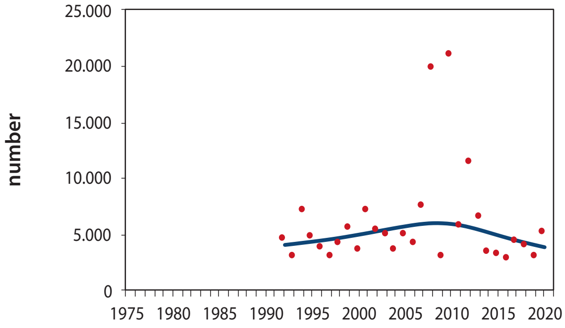

Figure A1.71. Flyway population trend for White-fronted Plover populations Gabon - S Africa combined. For explanation see fig. A1.4.

Kentish Plover | *Charadrius alexandrines* | 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.

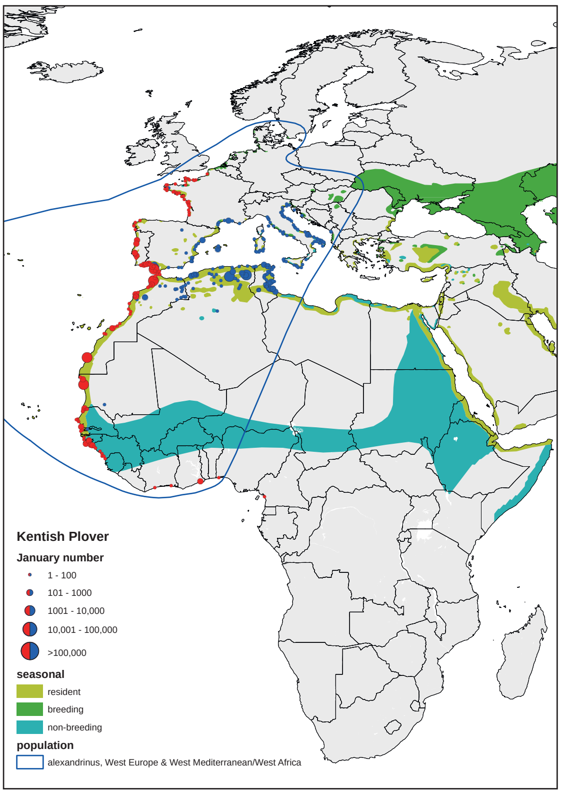

Figure A1.72. Distribution of Kentish Plover in the East Atlantic Flyway. For explanation see fig. A1.3.

population Kentish Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Europe & W Mediterranean/W Africa	w	1993-2020	0,982	moderate decline	2011-2020	0,973	uncertain	2007-2018	40000	65000

Table A1.34. Summary of flyway trend and population size estimates for Kentish Plover. For explanation see table A1.2.

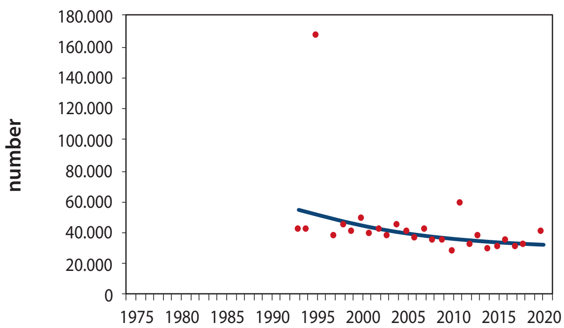

Figure A1.73. Flyway population trend for Kentish Plover population *alexandrinus* W Europe & W Mediterranean / W Africa. For explanation see fig. A1.4.

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

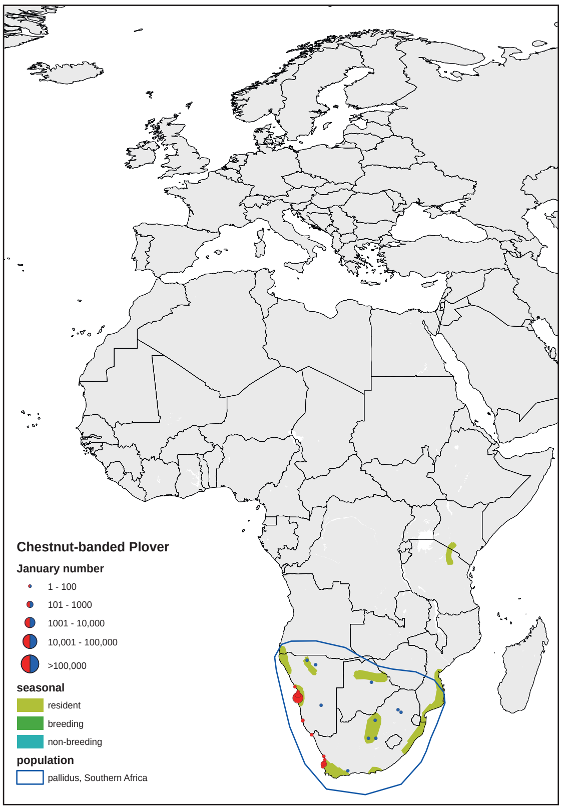

Figure A.74. Distribution of Chestnut-banded Plover in the East Atlantic Flyway. For explanation see fig. A1.3.

population Chestnut-banded Plover	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
S Africa	w	1992-2020	1,005	stable	2011-2020	0,818	strong decline	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.

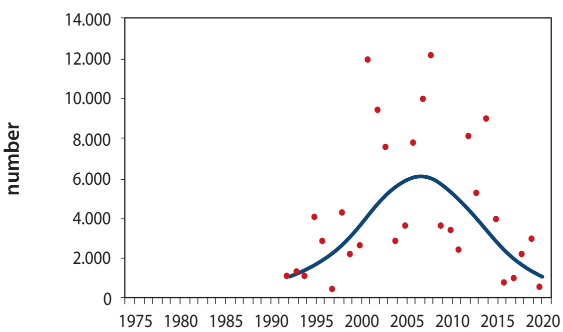

Figure A.75. Flyway population trend for Chestnut-banded Plover population *pallidus* Southern Africa. For explanation see fig. A1.4.

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.

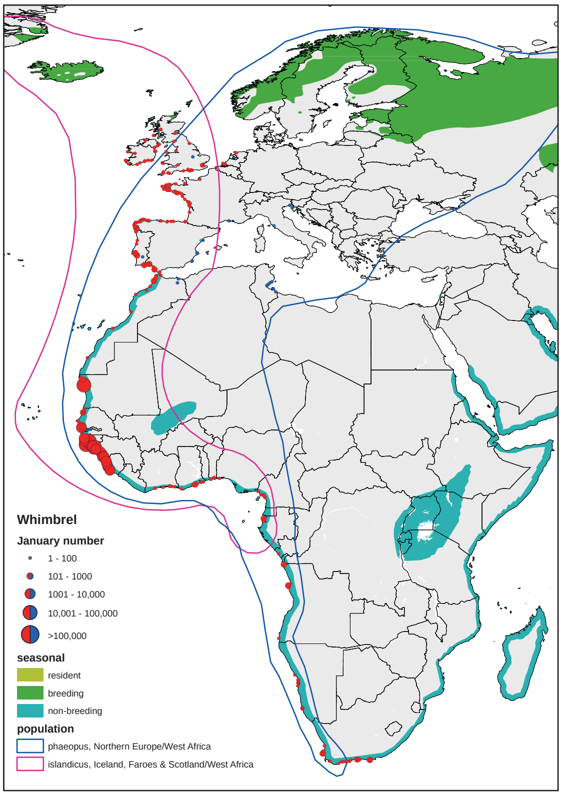

Figure A.76. Distribution of Whimbrel in the East Atlantic Flyway. For explanation see fig. A1.3.

Vincent Legrand / Agami

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	1979-2020	1,008	stable	2011-2020	1,008	stable			
phaeopus, N Europe/W Africa								2010-2018	240000	390000
islandica, Iceland,Faroer, Scaotland/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.

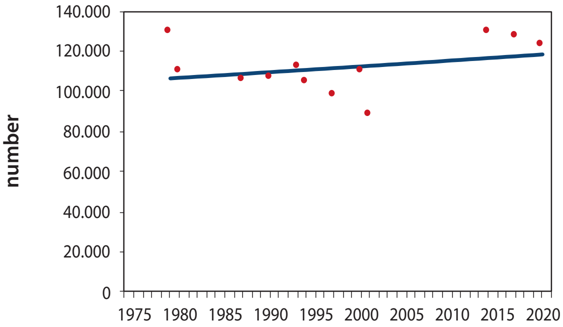

Figure A.1.77. Flyway population trend for the EAF wintering population of Whimbrel. For explanation see fig. A1.4.

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 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.

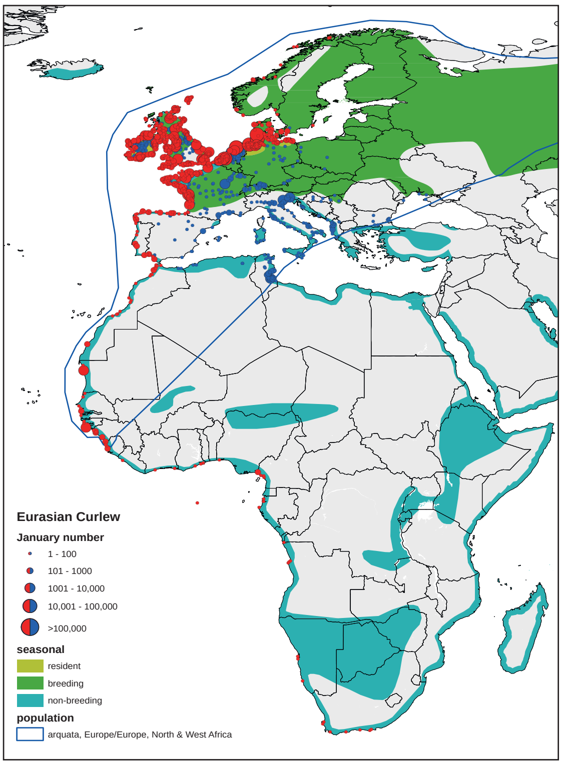

Figure A1.78. Distribution of Eurasian Curlew in the East Atlantic Flyway. For explanation see fig. A1.3.

population Eurasian Curlew	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Europe/NW Europe, N & W Africa	b	1980-2018	0,988	moderate decline	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.

Harvey van Diek

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 Fennoscandinavia (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. 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 (Duijns *et al.* 2013).

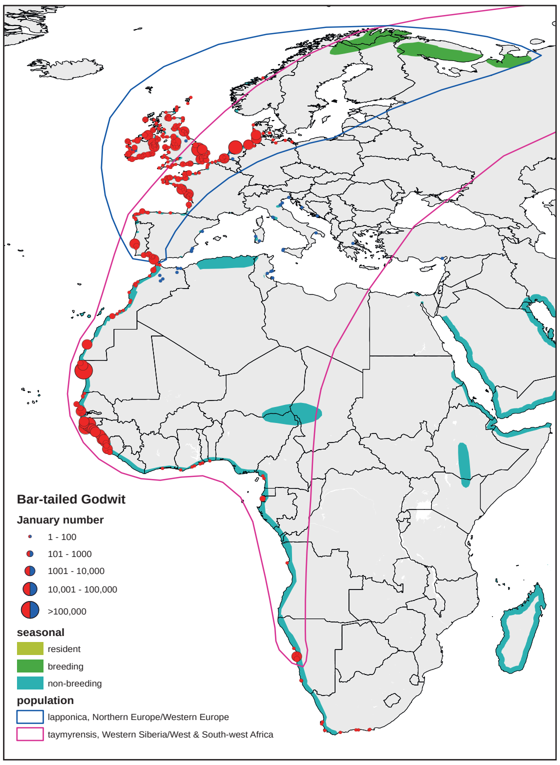

Figure A1.79. Distribution of Bar-tailed Godwit in the East Atlantic Flyway. For explanation see fig. A1.3.

population Bar-tailed Godwit	**data**	**period-L**	**trend-L**	**assessment-L**	**period-S**	**trend-S**	**assessment-S**	**period popsize**	**popsize-min**	**popsize-max**
lapponica, N Europe /W Europe	w	1975-2020	1,014	moderate increase	2011-2020	0,956	moderate decline	2011-2018	150000	180000
taymyrensis, W Siberia /W & S Africa	w	1979-2020	0,983	moderate decline	2011-2020	0,944	moderate decline	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.

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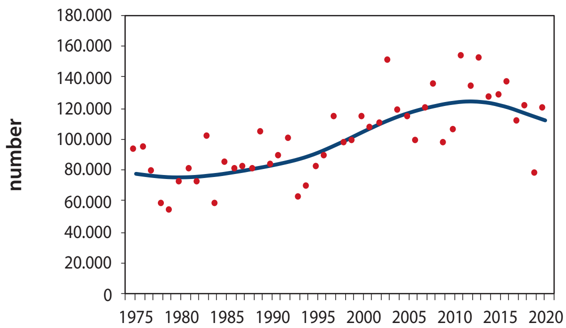

Figure A1.80. Flyway population trend for Bar-tailed Godwit population *lapponica*. For explanation see fig. A1.4.

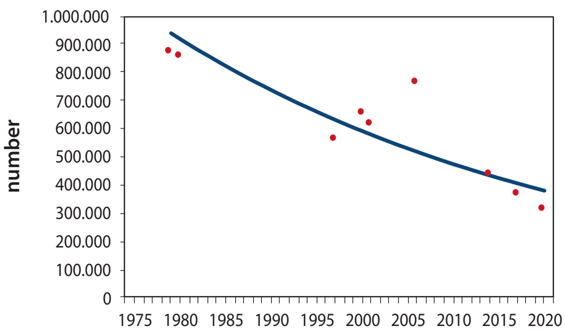

Figure A1.81. Flyway population trend for Bar-tailed Godwit population *taimyrensis*. For explanation see fig. A1.4.

The combined average sum of Bar-tailed Godwits of the *taymyrensis* population counted in West Africa in 2016–2020 was almost 340,000. Considering that most of the key sites have been counted and that the population is in decline, a new smaller population size estimate of roughly 380,000 – 420,000 is proposed for the 2016-2020 years.

Arnold Meijer / Blue Robin

Ruddy Turnstone | *Arenaria interpres* | Tournepierre 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.

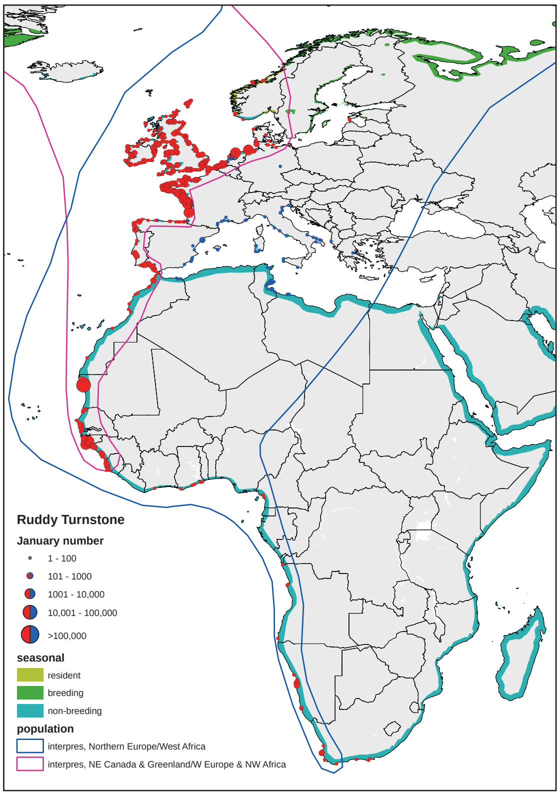

Figure A1.82. Distribution of Ruddy Turnstone in the East Atlantic Flyway. For explanation see fig. A1.3.

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population Ruddy Turnstone	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
NE Canada & Greenland /W Europe & NW Africa	w	1977-2020	1,023	moderate increase	2011-2020	0,990	stable	2000	200000	260000
N Europe/W Africa	w	1980-2020	0,972	moderate decline	2011-2020	1,022	uncertain	2008-2018	44000	87000

Table A1.39. Summary of flyway trend and population size estimates for Ruddy Turnstone. For explanation see table A1.2.

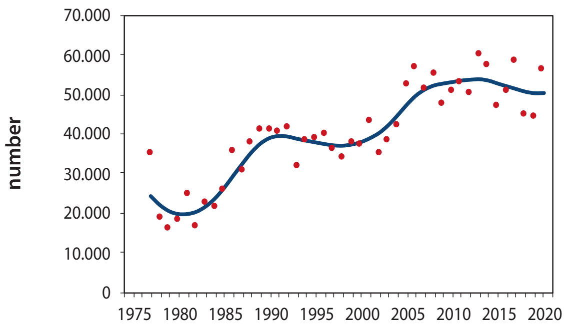

Figure A1.83. Flyway population trend for Ruddy Turnstone population *interpres* Nearctic / W Europe. For explanation see fig. A1.4.

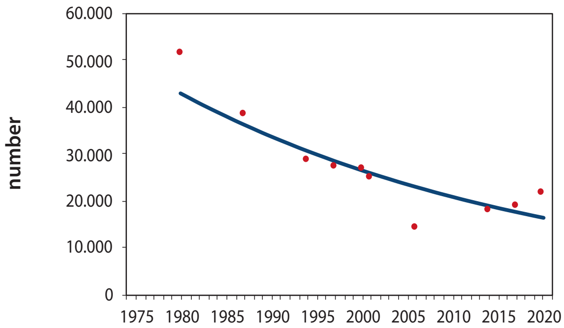

Figure A1.84. Flyway population trend for Ruddy Turnstone population *interpres* N Europe / W Africa. For explanation see fig. A1.4.

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 (Piersma 2007). 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 (van Gils *et al.* 2003).

The combined average sum of Red Knots of the *canutus* population counted in Western Africa in 2016 – 2020 is more than 250,000. Considering that all key sites have been covered during the total counts of 2017 and 2020, but that the flyway population is decline, the previous population size estimate of 250,000 will have been an underestimation (van Roomen *et al.* 2015). A population size of 260,000 – 275,000 is proposed for the 2016-2020 time period.

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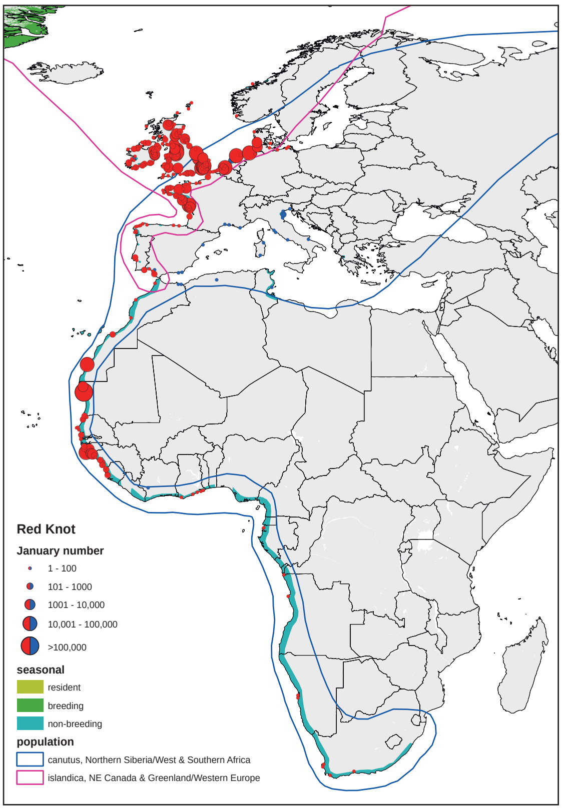

Figure A1.85. Distribution of Red Knot in the East Atlantic Flyway. For explanation see fig. A1.3.

population Red Knot	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
islandica, NE Canada & Greenland/ W Europe	w	1975-2020	1,009	moderate increase	2011-2020	0,981	moderate decline	2013-2017	310000	360000
canutus, NSiberia/W & S Africa	w	1979-2020	0,983	moderate decline	2011-2020	0,960	uncertain	2016-2020	260000	275000

Table A1.40. Summary of flyway trend and population size estimates for Red Knot. For explanation see table A1.2.

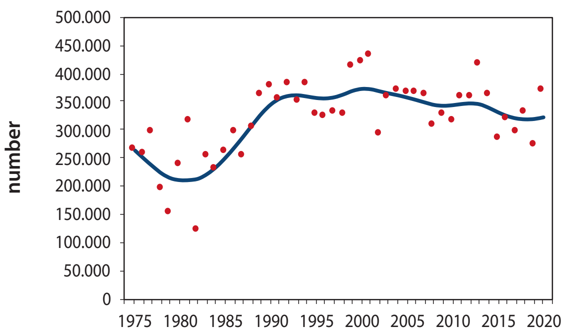

Figure A1.86. Flyway population trend for Red Knot population *islandica*. For explanation see fig. A1.4.

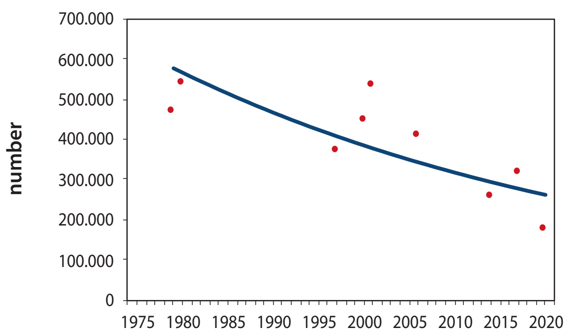

Figure A1.87. Flyway population trend for Red Knot population *canutus*. For explanation see fig. A1.4.

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 an other 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; no trend data are presented for the other population wintering mainly in Africa outside the EAF. Breeding occurs dispersed on 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 on passage and winter in more saline habitats.

The combined average sum of Curlew Sandpipers counted along the EAF as far south as Nigeria in 2016 – 2020 is almost 210,000. Considering that substantial numbers will have been present at sites not surveyed, but also that the population is still in considerable decline, a new smaller population size estimate of around 300,000 – 400,000 is tentatively proposed for 2016-2020.

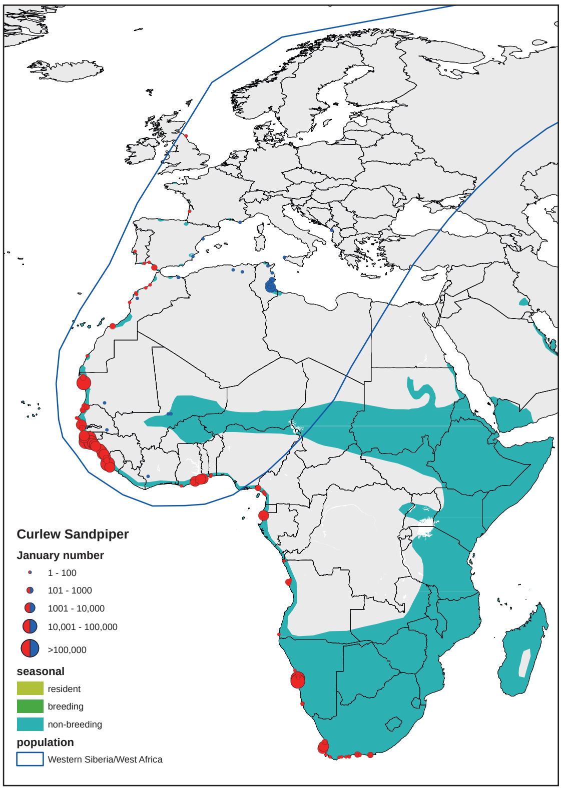

Figure A1.88. Distribution of Curlew Sandpiper in the East Atlantic Flyway. For explanation see fig. A1.3.

population Curlew Sandpiper	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Siberia /W Africa	w	1979-2020	0,969	moderate decline	2011-2020	0,897	strong decline	2016-2020	300000	400000

Table A1.41. Summary of flyway trend and population size estimates for Curlew Sandpiper. For explanation see table A1.2.

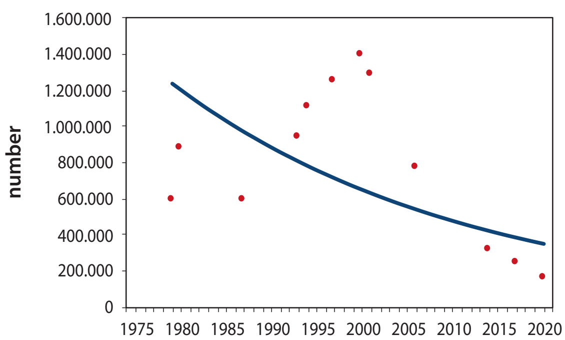

Figure A1.89. Flyway population trend for Curlew Sandpiper population W Siberia / W Africa. For explanation see fig. A1.4.

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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 southwards for wintering, but Greenlandic birds also occur south to Namibia but probably not until South Africa (Reneerkens *et al.* 2020, Chapter 12). 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).

The combined average sum of counted Sanderling along the EAF as far south as Namibia in 2016 – 2020 is more than 170,000. Considering that substantial numbers will have been present on sites not surveyed (mainly extensive sandy beaches) but also that mixing with the West Asia – Southern Africa population may occur and that the EAF population shows some recent decline, a population size of roughly 200,000 – 250,000 is tentatively proposed for the 2016-2020 years.

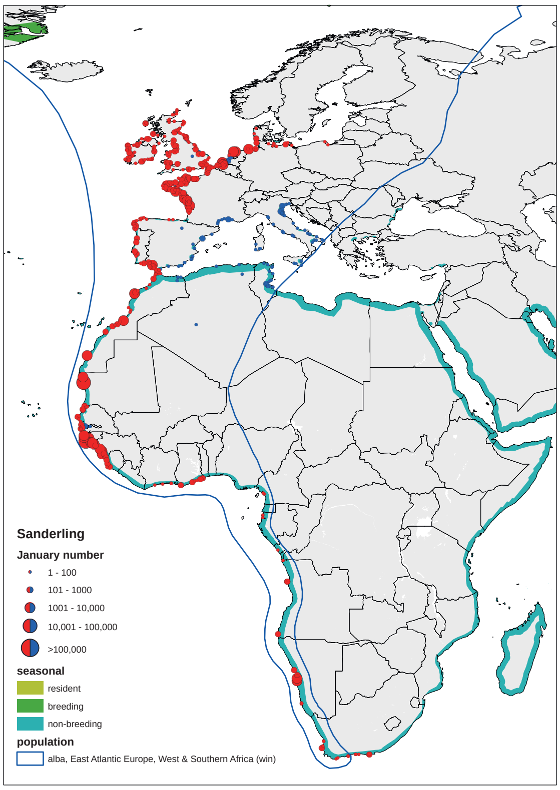

Figure A1.90. Distribution of Sanderling in the East Atlantic Flyway. For explanation see fig. A1.3.

population Sanderling	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
alba, W Europe & W & S Africa (winter)	w	1979-2020	1,026	moderate increase	2011-2020	0,986	stable	2016-2020	200000	250000

Table A1.42. Summary of flyway trend and population size estimates for Sanderling. For explanation see table A1.2.

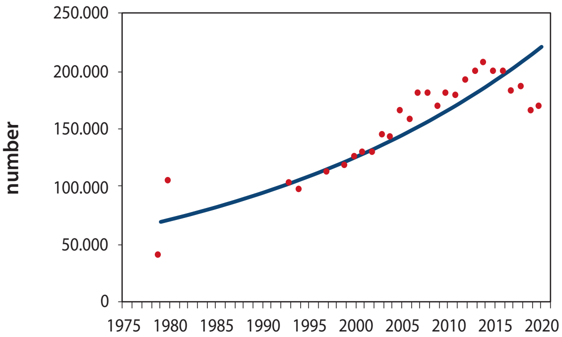

Figure A1.91. Flyway population trend for Sanderling population *alba* E Atlantic Europe, W & S Africa (win). For explanation see fig. A1.4.

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. A third, *C. a. schinzii*, is divided into three flyway populations, one breeding in Iceland and wintering in W Africa, one breeding in Britain and Ireland and wintering in NW Africa and SW Europe, and one breeding in the Baltic region 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 (though locally in high densities) 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, although it also occurs in a wide variety of freshwater and brackish wetlands, mainly on migration. It feeds on insects,

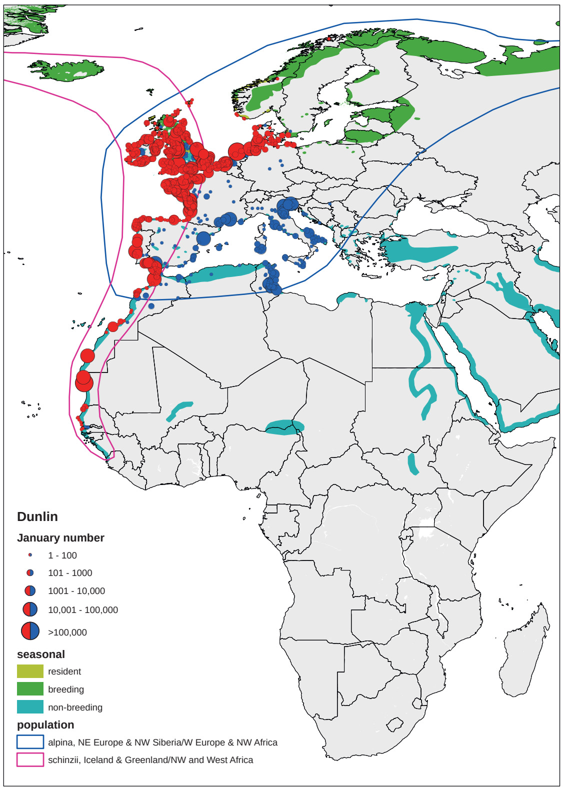

Figure A1.92. Distribution of Dunlin in the East Atlantic Flyway. For explanation see fig. A1.3.

population Dunlin	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
alpina, NE Europe & NW Siberia / W Europe & NW Africa	w	1976-2020	1,001	uncertain	2011-2020	0,983	moderate decline	2000-2018	1300000	1400000
schinzii, Iceland /NW & W Africa	w	1979-2020	0,996	stable	2011-2020	1,047	moderate increase	2016-2020	800000	1000000

Table A1.43. Summary of flyway trend and population size estimates for Dunlin. For explanation see table A1.2.

spiders, mites, earthworms, snails, slugs and seeds in the breeding season and mainly on worms, small gastropods, crustaceans and bivalves in the non-breeding season.

The combined average sum of Dunlin of the *schinzii* population from mainly Iceland counted in 2016 – 2020 is more than 900,000. Considering that all key sites for this *schinzii* population were covered in 2017 and 2020 (mainly Banc d'Arguin) but that over- and underestimates may easily occur because of the huge concentrations on some high tide roosts, that the breeding bird estimate for Iceland gives a population size of 730,000 birds (BirdLife International 2021) and that also the *arctica* population probably occurs at the same sites as *schinzii* from Iceland, a population size of 800,000 – 1,000,000 is proposed for the 2016-2020 time-period.

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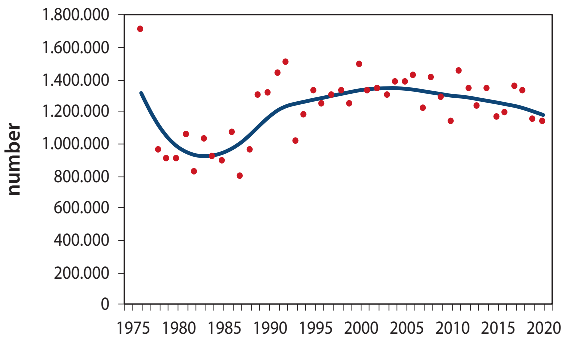

Figure A1.93. Flyway population trend for Dunlin population *alpina* NE Europe & NW Siberia / Europe & NW Africa. For explanation see fig. A1.4.

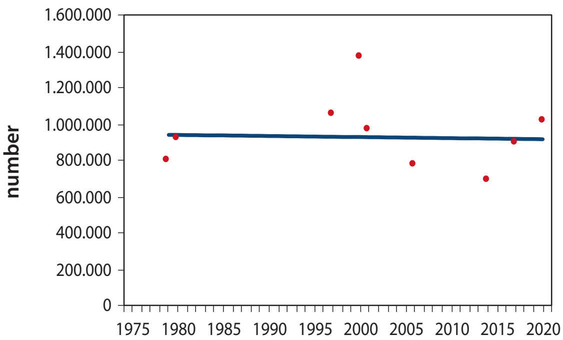

Figure A1.94. Flyway population trend for Dunlin population *schinzii* Iceland / NW & W Africa. For explanation see fig. A1.4.

Ralp Martin / Agami

Purple Sandpiper | *Calidris maritima* | Bécasseau violet

The East Atlantic coast is used predominantly for wintering by two flyway populations of Purple Sandpiper: one population 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. 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*, but also includes other invertebrates and some plant material. 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).

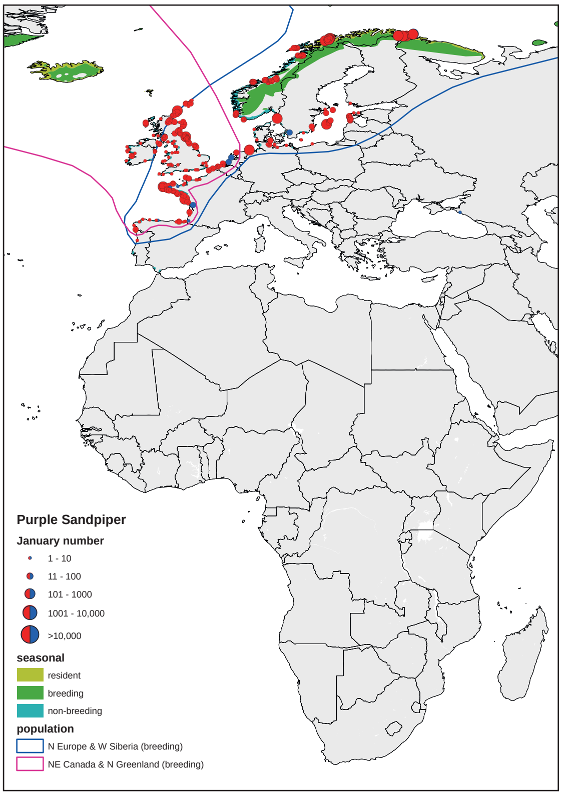

Figure A1.95. Distribution of Purple Sandpiper in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1982-2020	0,997	stable	2011-2020	0,973	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.

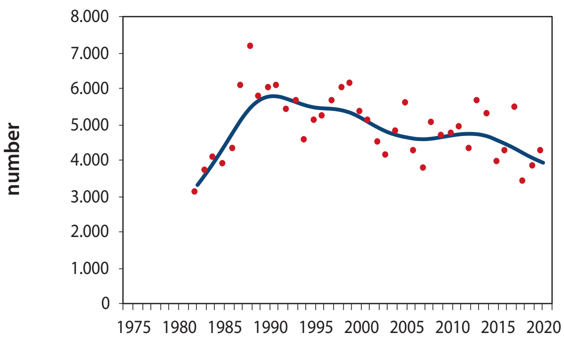

Figure A1.96. Flyway population trend for Purple Sandpiper populations wintering in NW Europe. For explanation see fig. A1.4.

Arnold Meijer / Blue Robin

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 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.

The combined average sum of Little Stints counted along the EAF as far south as Gabon in 2016 – 2020 is not more than 75,000. This is considerably less than the 130,000 in the same countries in 2014 (excluding the estimates for Mali and Niger, van Roomen *et al.* 2015). Considering the fact that many sites have not been counted but also that the flyway population is in decline, a tentative population size of 200,000 – 300,000 is proposed for the 2016-2020 years.

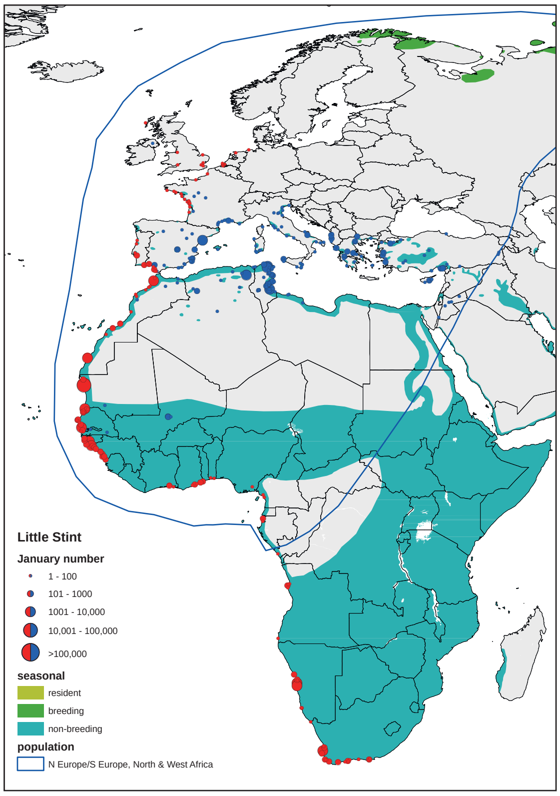

Figure A1.97. Distribution of Little Stint in the East Atlantic Flyway. For explanation see fig. A1.3.

population Little Stint	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popside-max
N Europe & W Siberia/N & W Africa	w	1979-2020	0,967	moderate decline	2011-2020	0,907	strong decline	2016-2020	200000	300000

Table A1.45. Summary of flyway trend and population size estimates for Little Stint. For explanation see table A1.2.

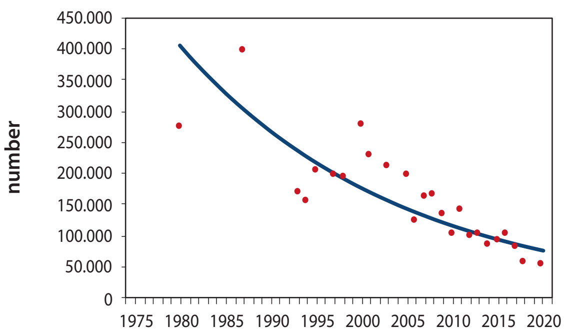

Figure A1.98. Flyway population trend for Little Stint population N Europe / S Europe, N & W Africa. For explanation see fig. A1.4.

Harvey van Diek

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 Cameroon 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.

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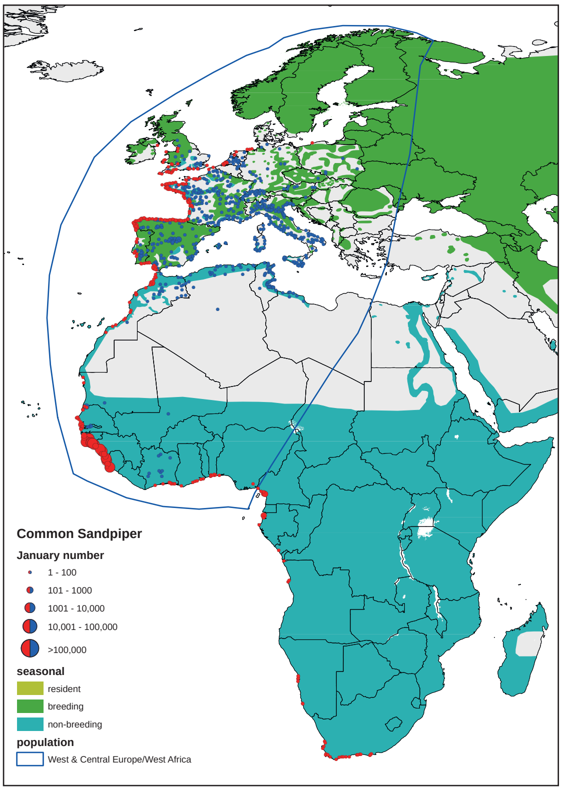


Figure A1.99. Distribution of Common Sandpiper in the East Atlantic Flyway. For explanation see fig. A1.3.

population Common Sandpiper	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W & C Europe/W Africa	w	1990-2020	0,969	moderate decline	2011-2020	1,003	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.

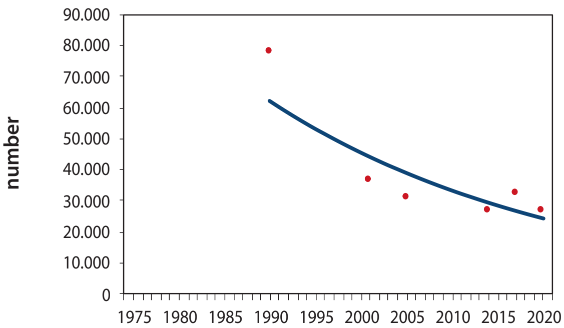


Figure A1.100. Flyway population trend for Common Sandpiper population W & C Europe / W Africa. For explanation see fig. A1.4.

Harvey van Diek

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.

Harvey van Diek

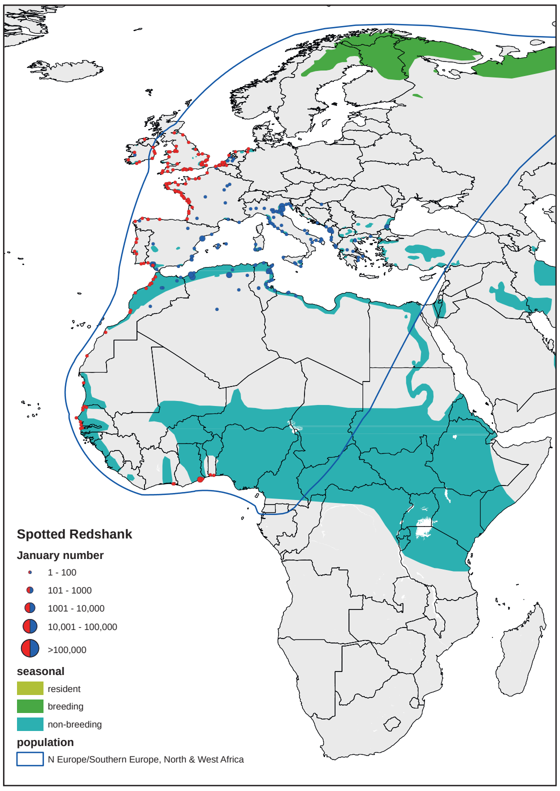


Figure A1.101. Distribution of Spotted Redshank in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1997-2020	0,952	moderate decline	2011-2020	0,957	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.

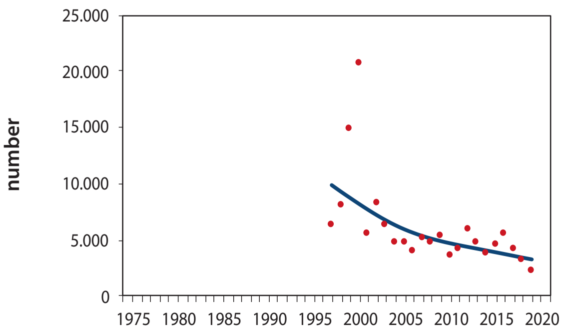


Figure A1.102. Flyway population trend for Spotted Redshank population N Europe / S Europe, N & W Africa. For explanation see fig. A1.4.

Harvey van Diek

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.

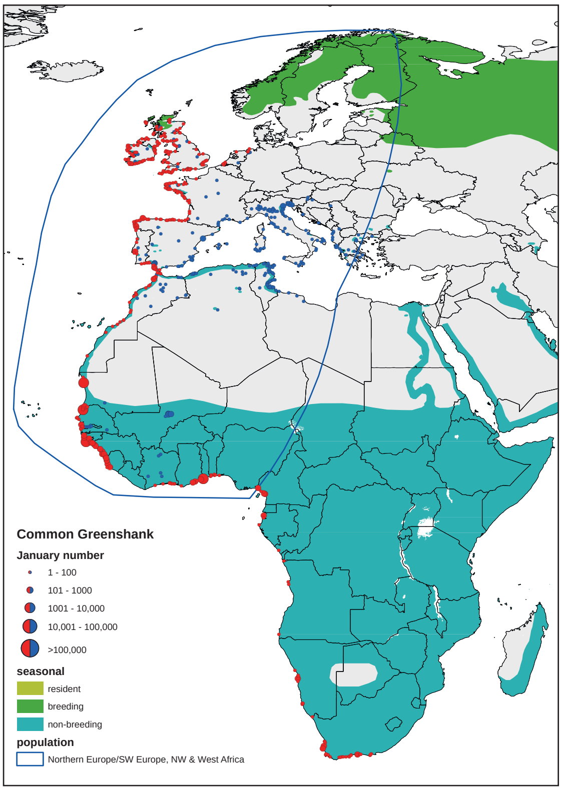


Figure A1.103. Distribution of Common Greenshank in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1990-2020	1,003	stable	2011-2020	1,026	uncertain	1995-2018	230000	360000

Table A1.48. Summary of flyway trend and population size estimates for Common Greenshank. For explanation see table A1.2.

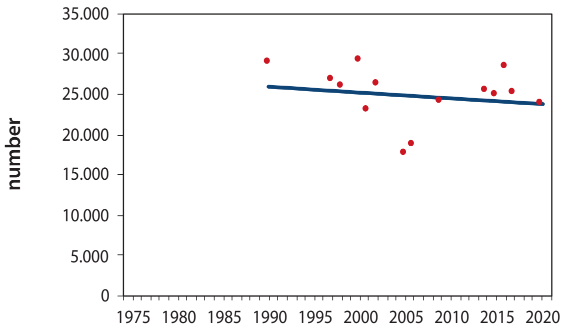


Figure A1.104. Flyway population trend for Common Greenshank population N Europe / SW Europe, NW & W Africa. For explanation see fig. A1.4.

Harvey van Diek

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 northwestern *T. t. totanus* population breeding in Fennoscandia and mainly wintering on the Atlantic coasts of W Africa, and (4) an eastern *T. t. totanus* population, breeding in continental W, C and NE Europe and wintering in Iberia, the Mediterranean, N and W Africa, reaching the East Atlantic coast from Ghana southward. 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.

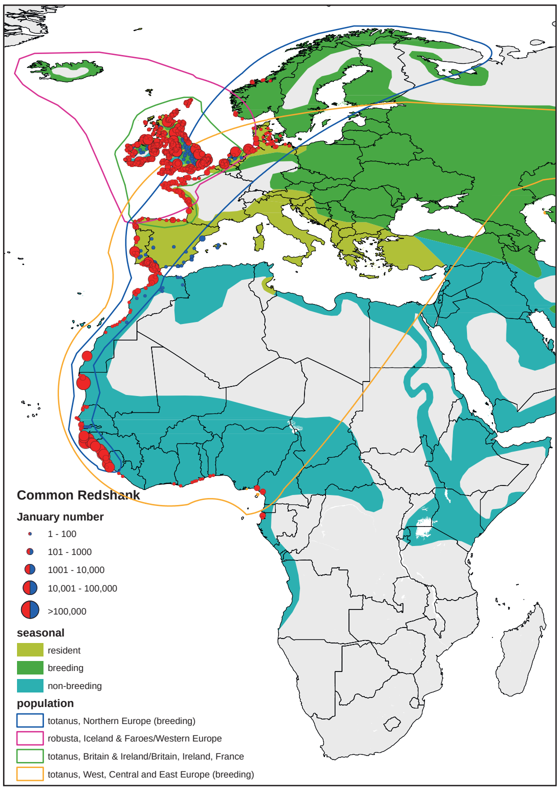


Figure A1.105. Distribution of Common Redshank in the East Atlantic Flyway. For explanation see fig. A1.3.

Harvey van Diek

population Common Redshank	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
robusta, Iceland & Faroes /W. Europe	w	1975-2020	1,006	moderate increase	2011-2020	1,009	moderate increase	2014-2016	230000	230000
totanus, Britain, Ireland/Britain, Ireland, France	b	1980-2017	0,961	moderate decline	2008-2017	0,946	moderate decline	2013-2018	66000	67000
totanus, N Europe /W Africa	b	1980-2017	0,980	moderate decline	2008-2017	1,002	stable	2008-2018	160000	240000
totanus, West, Central & East Europe (breeding)	b	1980-2017	0,990	moderate decline	2008-2017	0,997	stable	1981-2019	310000	450000

Table A1.49. Summary of flyway trend and population size estimates for Common Redshank. For explanation see table A1.2.

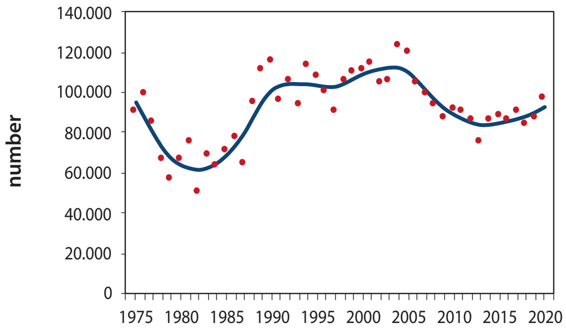


Figure A1.106. Flyway population trend for Common Redshank population *robusta*. For explanation see fig. A1.4.

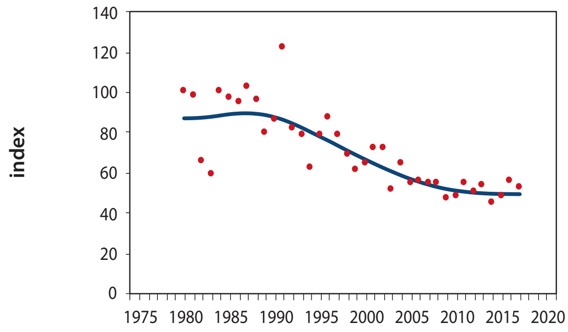


Figure A1.107. Flyway population trend for Common Redshank population *totanus* Britain & Ireland (bre) based on PECMBS. For explanation see fig. A1.4.

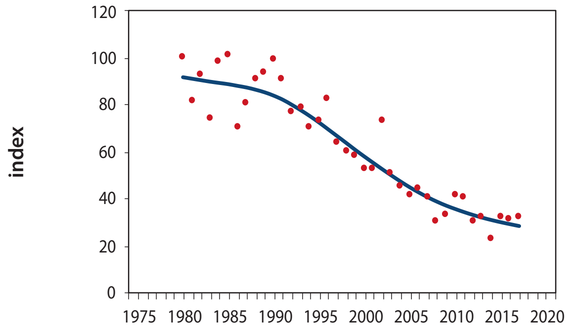


Figure A1.108. Flyway population trend for Common Redshank population *totanus* N Europe (bre) based on PECMBS. For explanation see fig. A1.4.

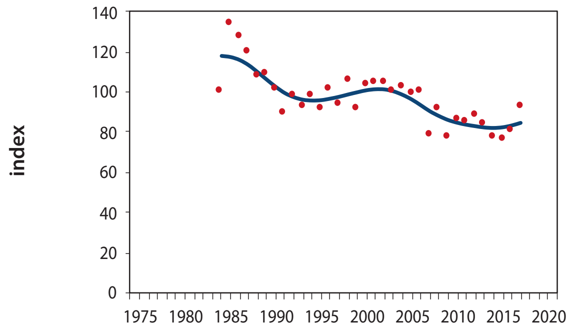


Figure A1.109. Flyway population trend for Common Redshank population *totanus* W, C & E Europe (bre) based on PECMBS. For explanation see fig. A1.4.

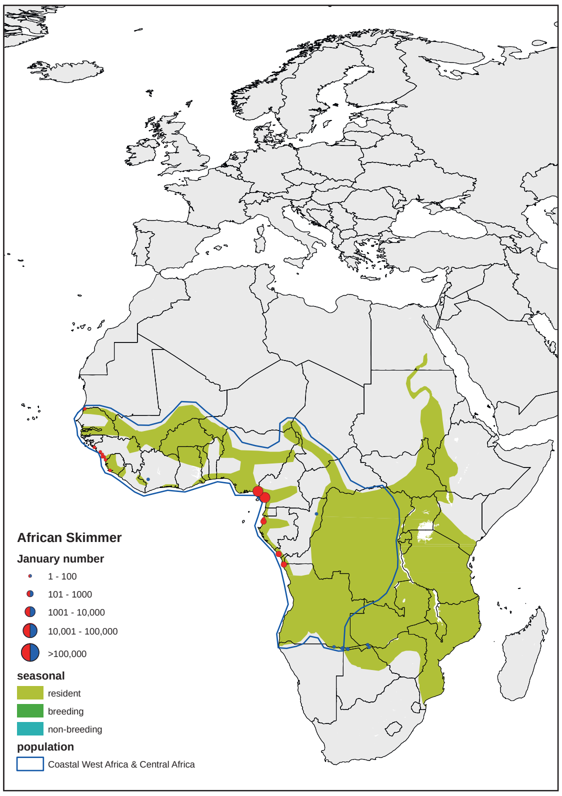
Harvey van Diek

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.

The combined average sum of African Skimmers counted along the EAF in 2016-2020 was almost 4,500. Considering that substantial numbers will have been present at sites not surveyed we tentatively propose a population size of 8,000 – 12,000 for the 2016-2020 years.

Harvey van Diek



A1.110. Distribution of African Skimmer in the East Atlantic Flyway. For explanation see fig. A1.3.

population African Skimmer	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Coastal W & C Africa	w	1998-2020	0,954	moderate decline	2011-2020	0,820	strong decline	2016-2020	8000	12000

Table A1.50. Summary of flyway trend and population size estimates for African Skimmer. For explanation see table A1.2.

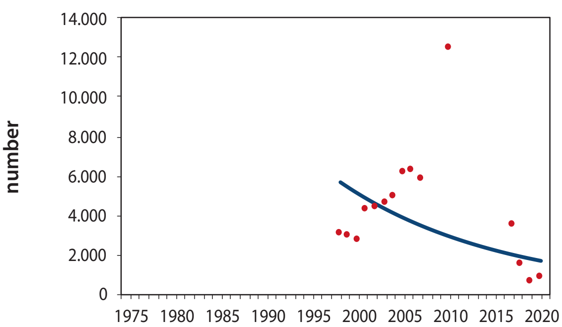


Figure A1.111. Flyway population trend for African Skimmer population W & C Africa. For explanation see fig. A1.4.

Hans Germeraad / Agami

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 saltpans. The diet consists mainly of fish, but also marine invertebrates and insects.

Harvey van Diek

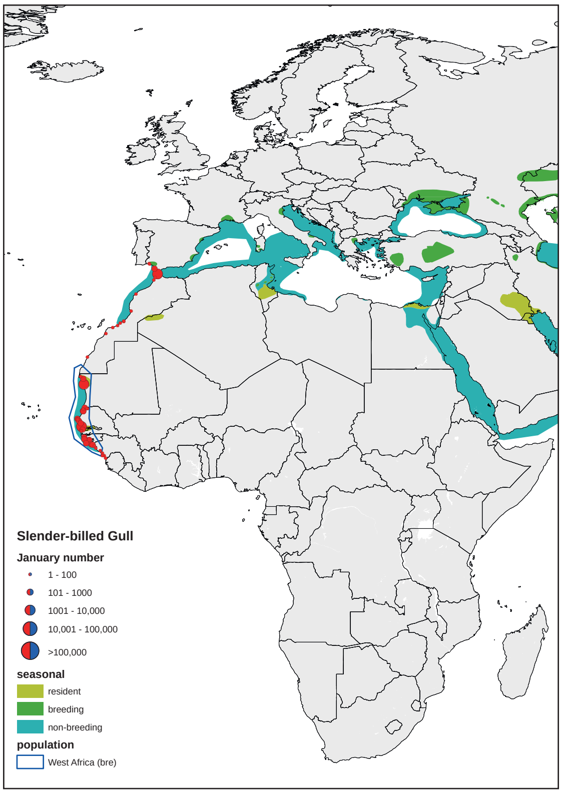


Figure A1.112. Distribution of Slender-billed Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1997-2020	0,991	stable	2011-2020	1,013	uncertain	2003-2019	24000	30000

Table A1.51. Summary of flyway trend and population size estimates for Slender-billed Gull. For explanation see table A1.2.

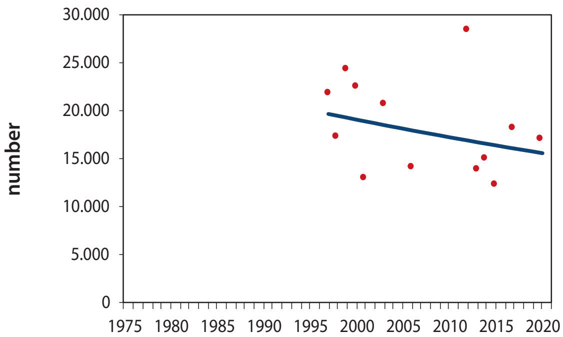


Figure A1.113. Flyway population trend for Slender-billed Gull population W Africa (bre). For explanation see fig. A1.4.

Arnold Meijer / Blue Robin

Black-headed Gull | *Chroicocephalus 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 (>50,000 pairs) 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 tussocks, 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.

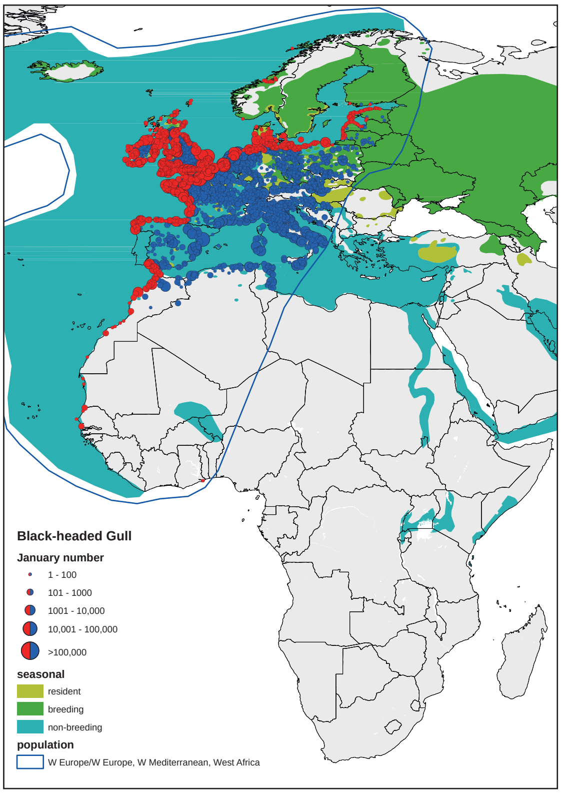


Figure A1.114. Distribution of Black-headed Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

population Black-headed Gull	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	w	1993-2020	0,993	moderate decline	2011-2020	0,987	moderate decline	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.

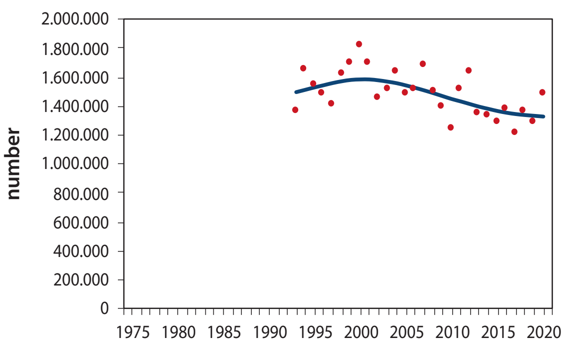


Figure A1.115. Flyway population trend for Black-headed Gull population W Europe (bre). For explanation see fig. A1.4.

Arnold Meijer / Blue Robin

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 rumps and sewage and salt works. It feeds mainly on invertebrates associated with stranded kelp, but also terrestrial insects, fish, earthworms, fruits and garbage.

Saverio Gatto / Agami

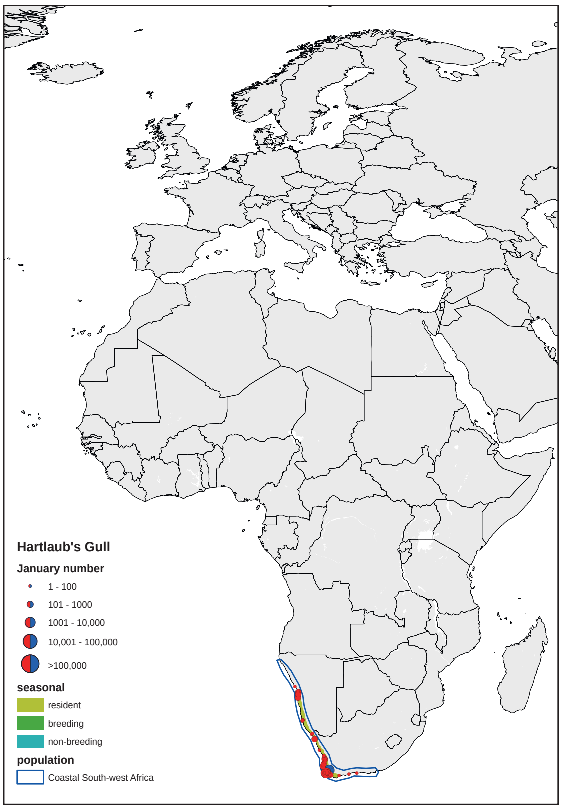


Figure A1.116. Distribution of Hartlaub's Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1992-2020	0,992	moderate decline	2011-2020	0,932	moderate decline	2018	17000	24000

Table A1.53. Summary of flyway trend and population size estimates for Hartlaub's Gull. For explanation see table A1.2.

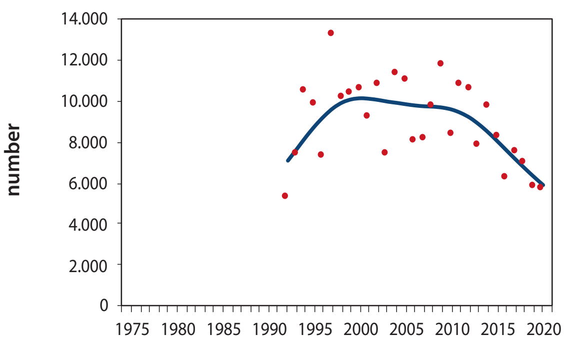


Figure A1.117. Flyway population trend for Hartlaub's Gull population SW Africa. For explanation see fig. A1.4.

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. 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.

The combined average sum of Grey-headed Gulls counted in West Africa in 2016 – 2020 is almost 12,000.

Considering that substantial numbers will have been present at sites not surveyed, and the increasing trend, a population size estimate of 25,000 – 35,000 is proposed for the 2016-2020 years.

Harvey van Diek

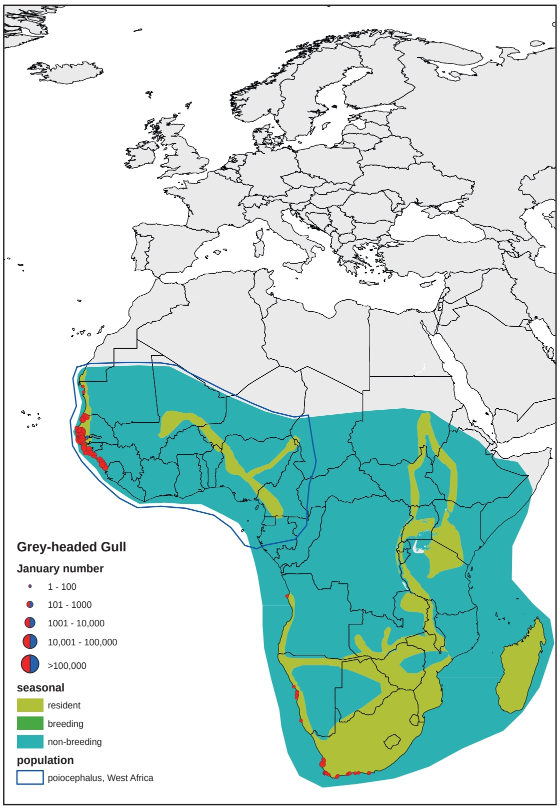


Figure A1.118. Distribution of Grey-headed Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

population Grey-headed Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa	w	1997-2020	1,067	moderate increase	2011-2020	1,020	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.

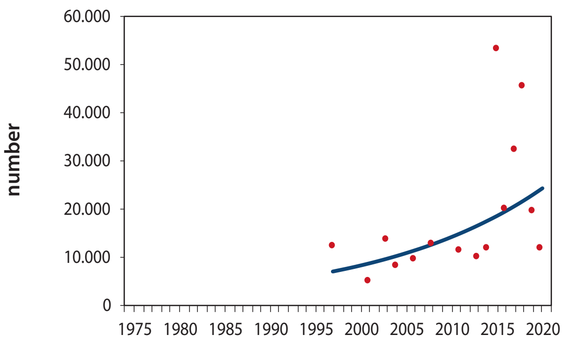


Figure A1.119. Flyway population trend for Grey-headed Gull population *poiocephalus* W Africa. For explanation see fig. A1.4.

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 (win) 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, saltmarshes 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.

Arnold Meijer / Blue Robin

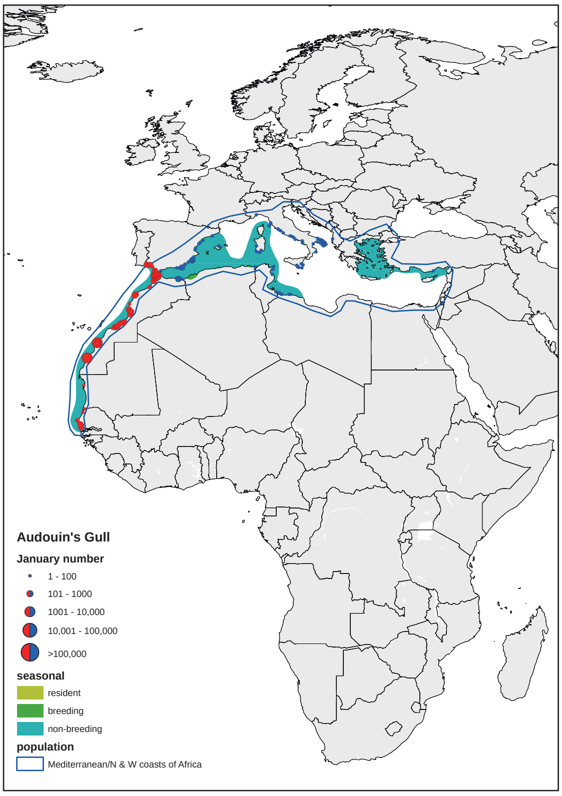


Figure A1.120. Distribution of Audouin's Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

population Audouin's Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
Mediterranean/N & W Africa	b	1980-2018	1,003	moderate increase	2009-2018	0,866	strong decline	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.

Harvey van Diek

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.

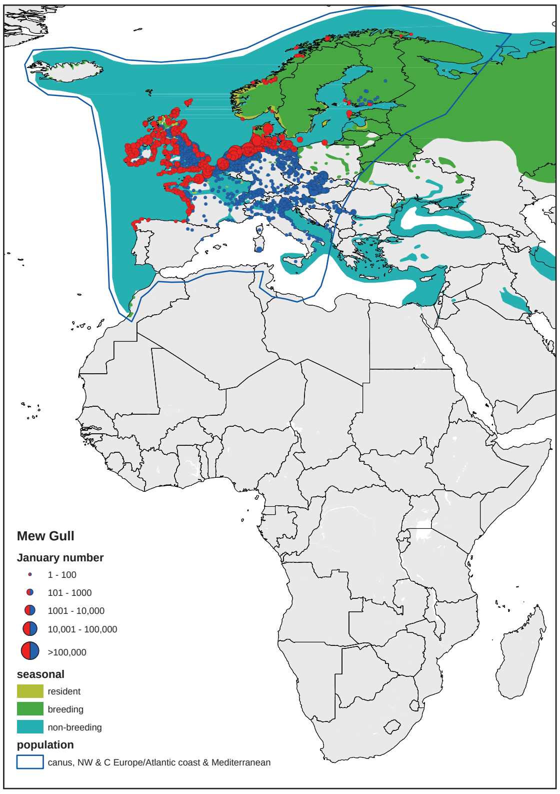


Figure A1.121. Distribution of Mew Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

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 decline	1981-2018	1400000	2000000

Table A1.56. Summary of flyway trend and population size estimates for Mew Gull. For explanation see table A1.2.

Arnold Meijer / Blue Robin

Arnold Meijer / Blue Robin

Lesser Black-backed Gull | *Larus fuscus* | Goéland brun

Within the EAFstudy area two populations are distinguished, representing the *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 (Hallgrimsson *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.

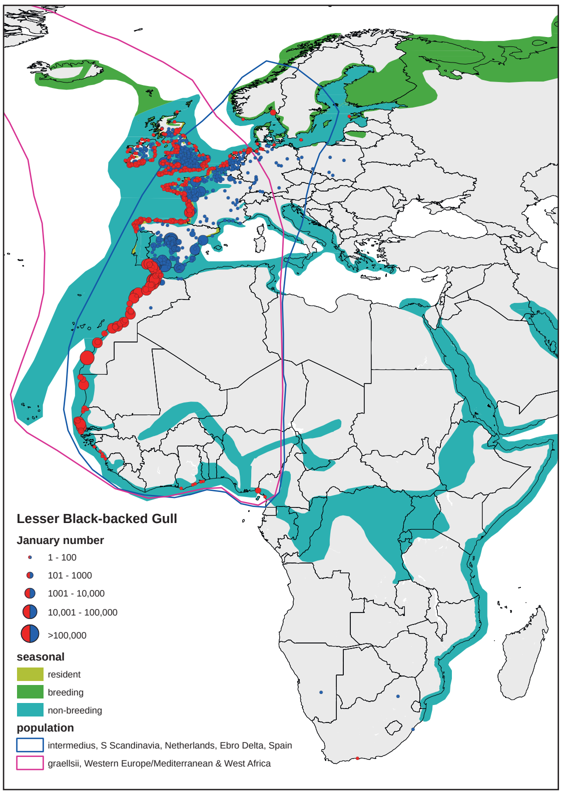


Figure A1.122. Distribution of Lesser Black-backed Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

Harvey van Diek

population Lesser Black-backed Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
graellsii, NW Europe /East Atlantic	b	1980-2018		moderate increase	2009-2018	0,917	moderate decline	1981-2018	480000	500000
intermedius, W Europe /East Atlantic	b	1980-2018		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 among other gull species.

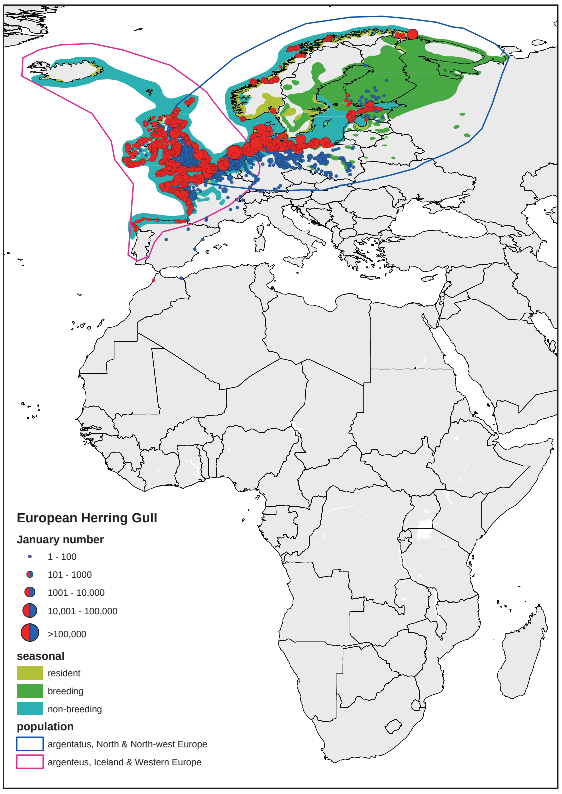


Figure A1.123. Distribution of European Herring Gull in the East Atlantic Flyway. For explanation see fig. A1.3.

Arnold Meijer / Blue Robin

population European Herring Gull	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
argenteus, NW Europe/East Atlantic	b	1980-2018	0,988	moderate decline	2009-2018	0,966	moderate decline	1981-2018	740000	780000
argentatus, W Europe /East Atlantic	b	1980-2018	0,986	moderate decline	2009-2018	0,891	strong decline	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.

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 saltpans, along the coast and in freshwater lagoons and inland lakes. Migrating birds are often seen over saltpans, 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.

Daniele Occhiato / Agami

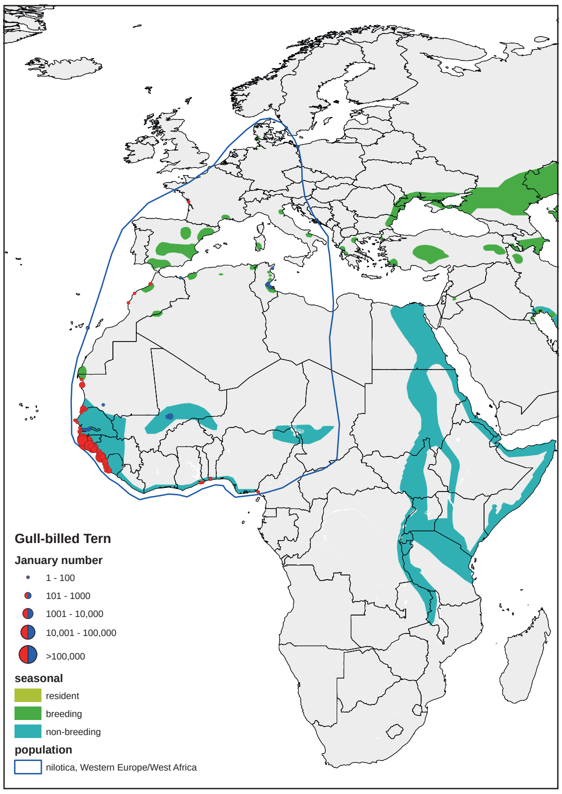


Figure A1.124. Distribution of Gull-billed Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

population Gull-billed Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Europe/W Africa	b	1980-2012	1,037	moderate increase	2009-2018	1,056	moderate increase	1972-2018	28000	38000

Table A1.59. Summary of flyway trend and population size estimates for Gull-billed Tern. For explanation see table A1.2.

Little Tern | *Sternula albifrons* | Sterne naine

The Little Tern is a widely, though generally sparsely, distributed species 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 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 colo-

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nies along the coast from Mauritania to Cameroon and in inland river basins (Niger, Chad, Ogooué). 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.

Marc Guyt / Agami

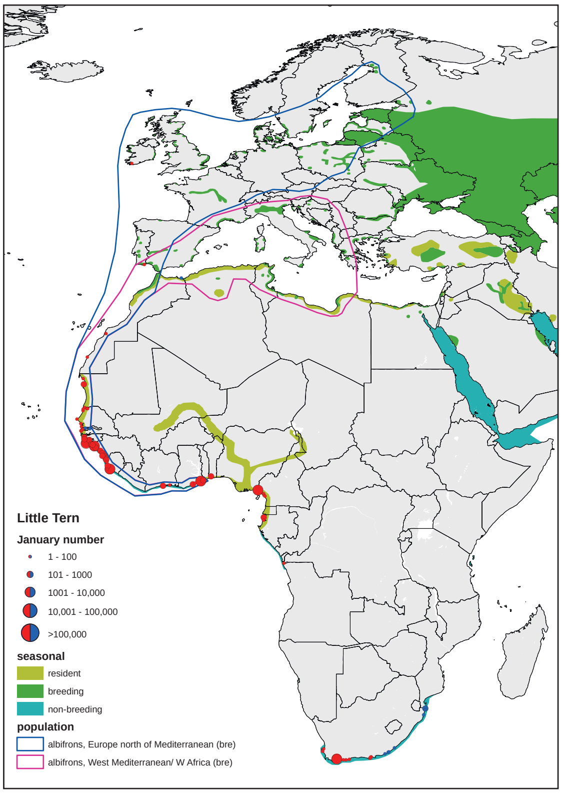

Figure A1.125. Distribution of Little Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

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population Little Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-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 decline	2009-2018	1,022	moderate increase	2006-2018	16300	26000

Table A1.60. Summary of flyway trend and population size estimates for Little Tern. For explanation see table A1.2.

Damara Tern | *Sternula balaenarum* | Sterne des baleiniers

Damara Terns breed in coastal areas of Namibia and South Africa and winter further north, probably 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

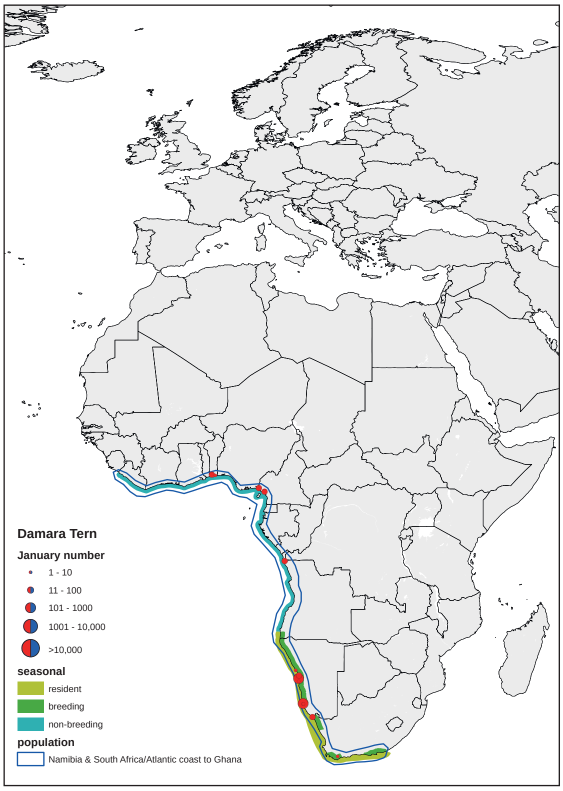

Figure A1.126. Distribution of Damara Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

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	1992-2020	0,972	moderate decline	2011-2020	1,077	uncertain	2010-2011	3400	8500

Table A1.61. Summary of flyway trend and population size estimates for Damara Tern. For explanation see table A1.2.

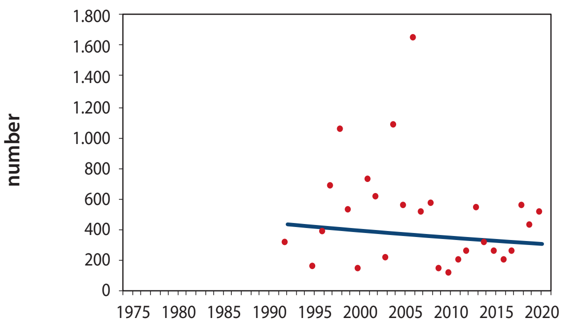

Figure A1.127. Flyway population trend for Damara Tern population Namibia & S Africa (bre). For explanation see fig. A1.4.

Pete Morris / Agami

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 W Africa along the coasts from Mauritania to Guinea and in the W Sahel (Mali) but also in S Spain, Chad and the Nile basin (Rueda-Uribe *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 and is more widely dispersed 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 saltpans, 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.

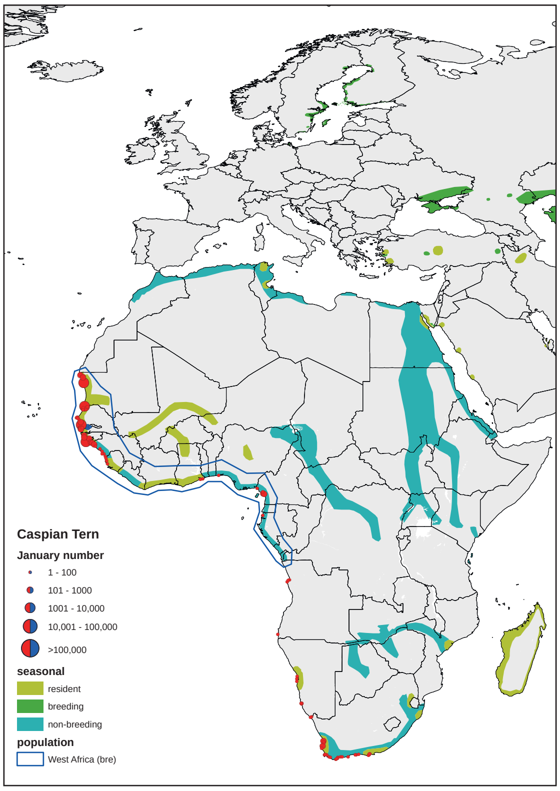

Figure A1.128. Distribution of Caspian Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

population Caspian Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
coastal W Africa	w	1997-2020	1,043	moderate increase	2011-2020	1,077	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.

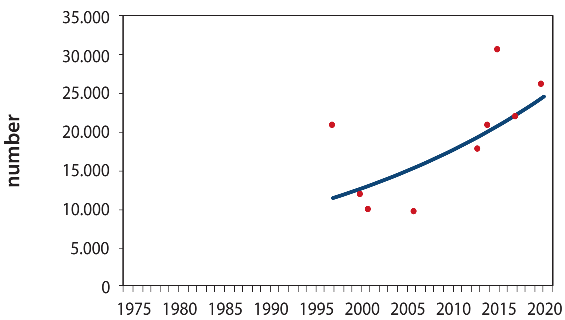

Figure A1.129. Flyway population trend for Caspian Tern population W Africa (bre). For explanation see fig. A1.4.

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Common Tern | *Sterna hirundo* | Sterne pierregarin

The Common Tern is one of the most globally numerous and widespread tern species. Three populations occur in the EAF. The two lagest, 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 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 saltmarshes. 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.

Daniele Occhiato / Agami

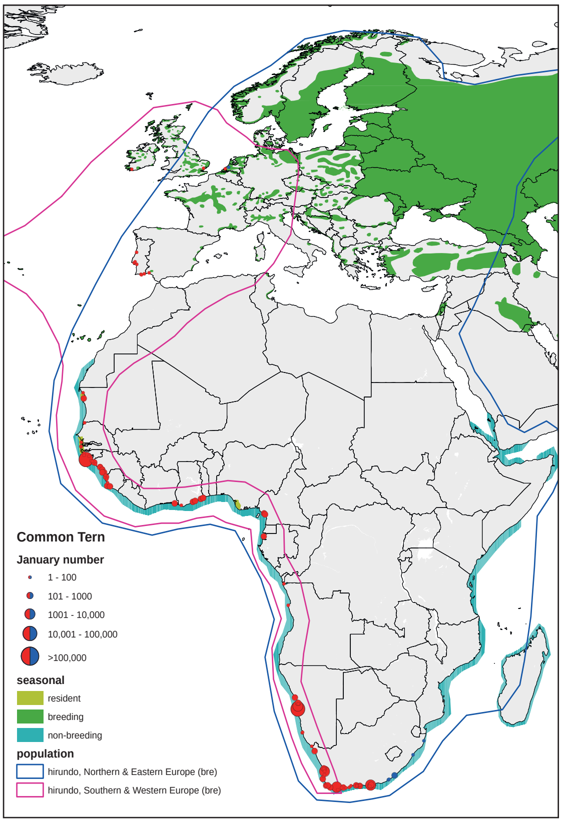

Figure A1.130. Distribution of Common Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

population Common Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-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 A1.63. Summary of flyway trend and population size estimates for Common Tern. For explanation see table A1.2.

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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 offshore 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 specialized compared to other terns and consists of small pelagic fish such as sandeel and sprat.

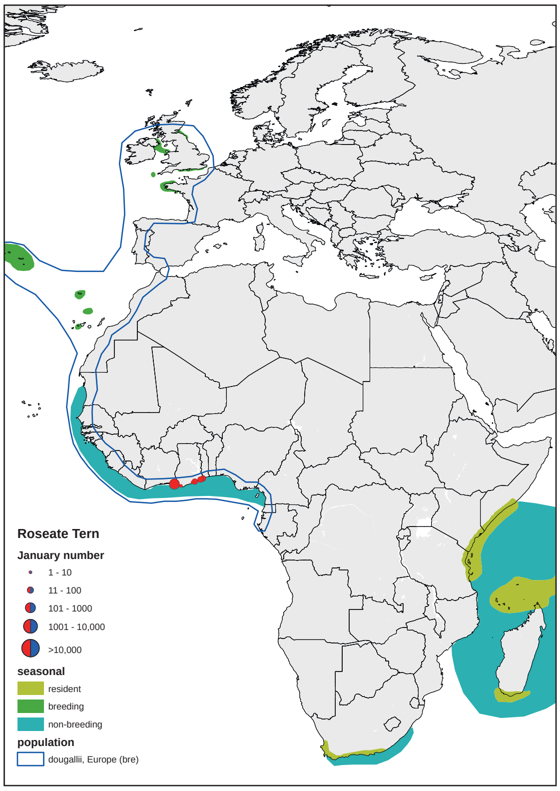

Figure A1. 131. Distribution of Roseate Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

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population Roseate Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Europe/East Atlantic	b	1980-2018	1,033	moderate increase	2009-2018	1,049	moderate increase	2012-2018	7500	9200

Table A1.64. Summary of flyway trend and population size estimates for Roseate Tern. For explanation see table A1.2.

Sandwich Tern | *Thalasseus sandvicensis* | Sterne caugek

The Sandwich Tern is a strictly 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 in the W Mediterranean and along the western seaboard of Africa. 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 recoveries 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.

Arnold Meijer / Blue Robin

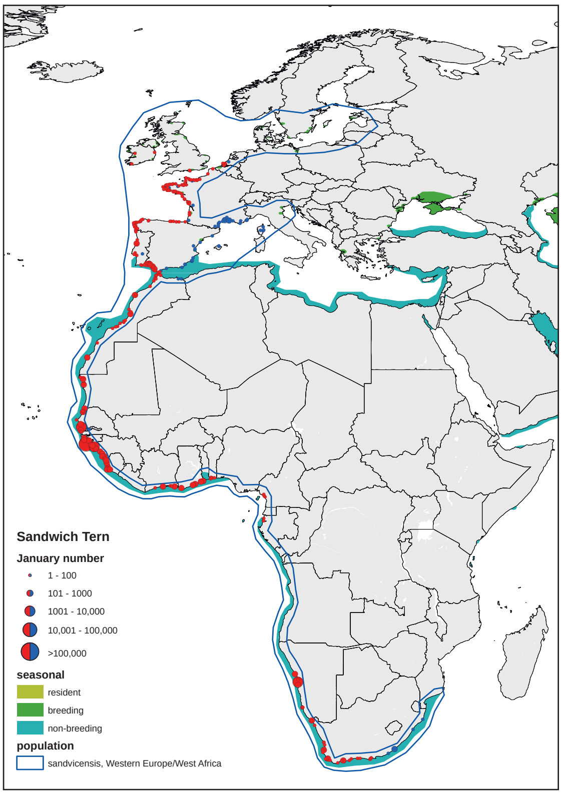

Figure A1.132. Distribution of Sandwich Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

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 A1.65. Summary of flyway trend and population size estimates for Sandwich Tern. For explanation see table A1.2.

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Royal Tern | *Thalasseus maximus* | Sterne royale

Royal Terns occurring in the EAF have been considered a subspecies (*T. m. 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. maxima maxima*, and hence should be considered a separate species, West African Crested Tern *T. albididorsalis* (Collinson *et al.* 2017). This form breeds on the West African coast from Mauritania to Guinea 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.

Vincent Legrand / Agami

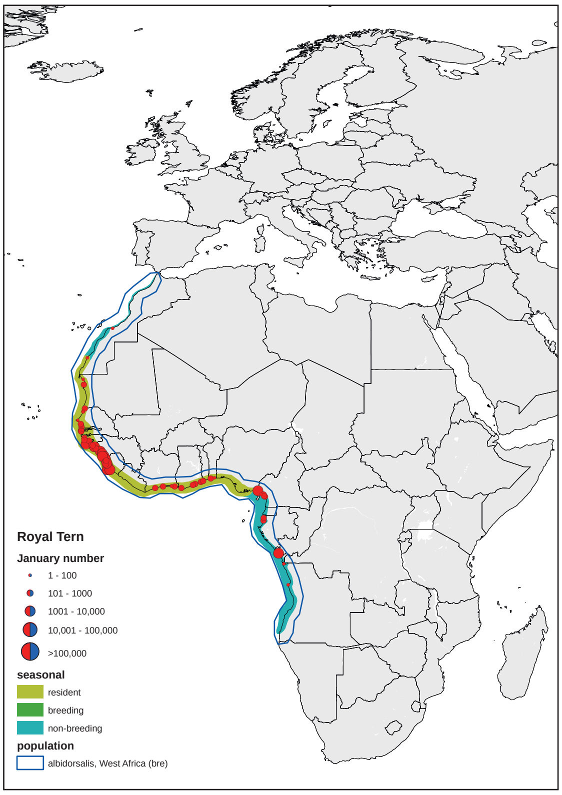


Figure A1.133. Distribution of Royal Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

population Royal Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
W Africa	w	1990-2020	1,002	uncertain	2011-2020	0,995	stable	2019	220000	230000

Table A1.66. Summary of flyway trend and population size estimates for Royal Tern. For explanation see table A1.2.

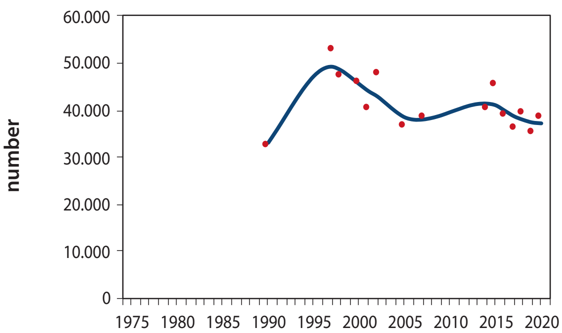


Figure A1.134. Flyway population trend for Royal Tern population *albididorsalis*. For explanation see fig. A1.4.

Vincent Legrand / 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.

James Eaton / Agami

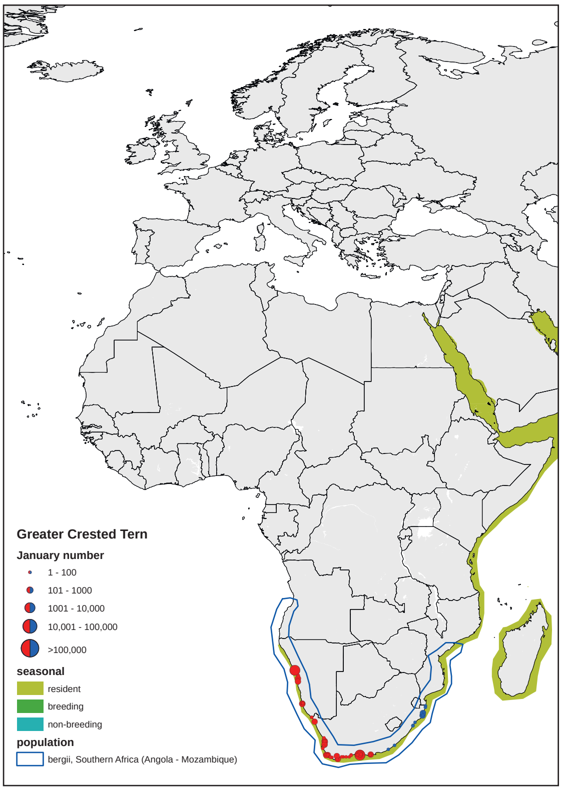


Figure A1.135. Distribution of Greater Crested Tern in the East Atlantic Flyway. For explanation see fig. A1.3.

population Greater Crested Tern	data	period-L	trend-L	assessment-L	period-S	trend-S	assessment-S	period popsize	popsize-min	popsize-max
S Africa	w	1992-2020	1,022	moderate increase	2011-2020	0,983	uncertain	1994-1996	15000	25000

Table A1.67. Summary of flyway trend and population size estimates for Greater Crested Tern. For explanation see table A1.2.

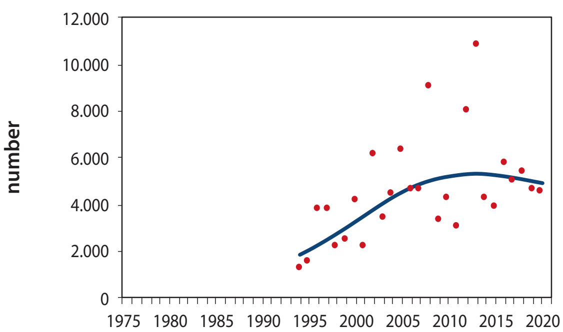


Figure A1.136. Flyway population trend for Greater Crested Tern population Southern Africa. For explanation see fig. A1.4.

Ralph Martin / Agami

For literature references mentioned in this Annex, see the main reference list in Chapter 15

Lars Soerink

Bar-tailed Godwit | **Barge rousse** (*Limosa lapponica*) (Han Bouwmeester / Agami)

Annex 2. Natural conditions, human pressures and conservation measures along the East Atlantic Flyway based on questionnaires, 2020

André van Kleunen, Geoffroy Citegetse, Olivia Crowe, Tim Dodman, Agyemang Opoku & Marc van Roomen

A2.1. Introduction

The East Atlantic Flyway is a recognized route for migratory birds, stretching from the Arctic through Western Europe to the entire western coastline of Africa. The flyway also supports a substantial human population, with numerous cities, industries and activities all along the coastal zone. In some areas people and wildlife co-exist in reasonable harmony, but in other areas human activities exert a constant pressure on birds and their habitats.

Systematic waterbird censuses are essential to assess the status of waterbird populations. However, proper information on the environmental status of critical sites for waterbirds is also needed to monitor the quality of these locations either as breeding, staging or wintering/non-breeding sites. Only through a combination of both species and site information is it possible to clarify causes of changes in waterbird numbers - information that is essential for policy makers and conservationists.

In recent years, considerable progress has been made in the environmental monitoring of sites along the East Atlantic Flyway. After pilots in 2013 and 2014, a systematic collection of environmental data of all the waterbird count sites along the East Atlantic Flyway was conducted in 2017. Questionnaires with pre-defined lists of environmental factors, pressures and conservation measures were scored by local observers and site managers for 88 important sites (i.e. the sites supporting highest concentrations of waterbirds) along the flyway from NW Europe to southern Africa. The results were reported in van Kleunen *et al.* (2018) and Dodman *et al.* (2018). This monitoring approach was promising; the collected data gave some interesting insights in the environmental status of the waterbird sites. However, there were some gaps in the data collection, and the design of the questionnaires showed some shortcomings. In 2020, monitoring assessments were carried out again, at a larger selection of sites, and with improvements made to the questionnaire. In this annex, the basic results of the questionnaires conducted in 2020 are reported . In chapter 3 of this report, the conservation status of important sites along the flyway is discussed in relation to pressures and conservation measures. In annex 3, results of a pilot study exploring the use of remote sensing to monitor specific environmental factors are presented.

A2.2. Study sites

IWC National Coordinators of the countries involved in flyway monitoring were asked to coordinate the filling out of environmental monitoring forms for each site. This was preferably done by local observers and site managers. The results were then validated by the National Coordinators and further checked and validated by us. We received environmental data from 115 sites spread over 31 countries and five regions (see figure 1). The data collection was focused on coastal sites used by waterbirds in January. Many of these sites are also important for breeding birds and many of the pressures and conservation measures will also apply to them. However, breeding sites in the Arctic are not included in these analyses, as they are not used during the winter counts. A description of conditions in the Russian Arctic can be found in chapter 4 of this report.

A2.3. Content of the questionnaire

The questionnaire has three sections, starting with the characterisation of some natural factors including habitat at the site. It is followed by questions about the presence of human activities and pressures at the site, and on the habitats and waterbird populations . It ends with questions about conservation measures needed and implemented at the sites to counteract these pressures. Compared with 2017, some changes were made to the habitat, pressure and conservation measures categories and their scoring, and these are summarised below. For a more elaborate overview of the methodology and categories used, refer to van Kleunen *et al.* (2018).

Data on habitat, the natural characteristics of sites and natural processes that could be relevant for waterbird abundance were assigned, simply by scoring 'present' (yes) or 'absent' (no). This is a more simplified approach compared with the scoring used until 2017 when assessors were asked to assign 'many', 'some' or 'no' against each.

Data on human presence and pressures were first col-

Region	n sites	agriculture lands (%)	beach and open sea (%)	estuarine / intertidal areas(%)	freshwater lakes / marshes (%)	man-made fishponds (%)	man-made saltponds (%)	man- made water reservoir (%)	mudflats (%)	riverine areas (%)	saltmarshes (%)	seagrass beds (%)	seminatural grass-land / ricefields (%)	shallow water (%)	small uninhabitated islands (%)
NW Europe	37	46	89	76	41	3	3	16	70	8	76	81	57	100	54
Iberia–N Africa	8	38	50	88	38	38	38	13	100	38	100	75	75	100	63
W Africa	37	57	78	84	19	5	5	8	78	32	57	11	35	97	57
Gulf of Guinea	22	55	68	100	41	32	5	14	86	64	36	14	45	95	77
Southern Africa	11	27	91	91	18	0	36	0	91	27	100	18	0	100	55
Total	**115**	**49**	**79**	**85**	**31**	**11**	**10**	**11**	**80**	**30**	**66**	**39**	**43**	**98**	**60**

Table A2.1. Overview of habitats (elements) recorded at the selected sites per region of the flyway (excluding the (sub) arctic breeding areas). Figures are percentages of all sites where the feature was present.

lected through questions related to the types of human uses present at the site. These were scored for each site on a scale between 0 (absent) and 10 (everywhere). This was similar to the approach in 2017. For scoring the pressures (which human activities are considered to affect the waterbird populations of the site) the methodology of the IBA monitoring protocol was used, which includes scores for timing (when does it take place), scope (where does it take place) and severity (the strength of the pressure) (BirdLife International 2006).

Figure A2.1. Overview of coastal sites from which questionnaires were returned in 2020. Background colours of the coastline denote the regions recognised.

In 2020, coordinators were asked about the need for conservation measures in general (yes or no) and if conservation measures were implemented (yes or no). Thereafter, coordinators commented on a list of conservation measures which were implemented, and at what scale (whole of the site, most, some or little of the site). The effectiveness of the measure was scored in terms of 'good', 'some effect' or 'no effect'. In 2017, coordinators were asked instead to score these measures on a scale from 010.

A2.4. Results

The overall sample of sites has increased compared to 2017, when we received questionnaires for 88 sites, of which data from 73 sites were complete and included. In 2020, we received complete monitoring forms from 115 sites. In 2017 most of our information came from Africa and included also some inland sites and/or small sites. Sites from Europe with complete information were underrepresented in 2017.

A2.4.1. Natural factors

All sites selected for analysis were located at/near the coast. Estuarine areas, mudflats, saltmarshes and beaches are dominant habitat types in the waterbird sites across the flyway. 40% of the sites have been cultivated to some extent: agricultural lands, (semi-natural) grasslands or rice fields. Other artificial (man-made) habitat elements such as fish ponds, salt ponds and water reservoirs are found in a minority (about 10%) of the sites, most in the Iberia – N Africa region (Fig. A2.2 and table A2.1).

A2.4.2. Human uses and pressures

Human use of wetlands can be incompatible with the

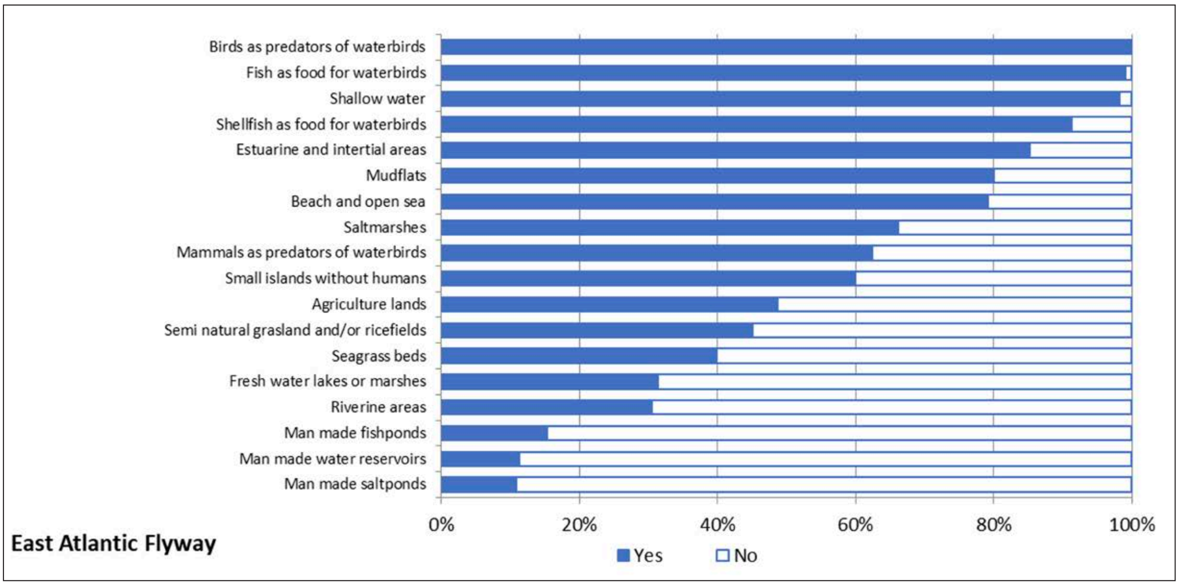


Figure A2.2. Occurrence of habitats and other natural factors at the 115 sites along the East Atlantic Flyway, 2020.

function of the area as a staging site or a breeding location for waterbirds, by affecting the area or quality of the habitat, by causing disturbance or by causing direct mortality of waterbirds. In this section an overview is presented of the existing pressures along the East Atlantic Flyway with particular reference to the assessed sites.

Expansion/intensification of agriculture

Farming is a strong pressure in more than 40% of the sites (Fig. A2.3). However, there is some contrast between regions. It is rather dominant in countries between Iberia and the Gulf of Guinea. On the other hand, in NW Europe and Southern Africa only 20-30% of the sites are affected heavily. Habitat loss due to conversion of wetlands to farmland has had a direct impact on many marshland birds and shorebirds. On the other hand, some species have benefitted from the increase of highly productive croplands and grasslands, which are an important driver of the

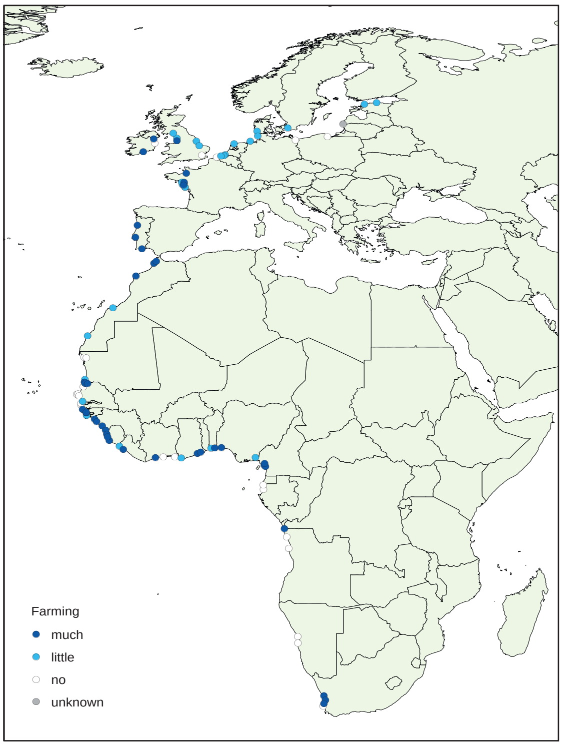


Figure A2.3. Farming as a pressure at the assessed sites.

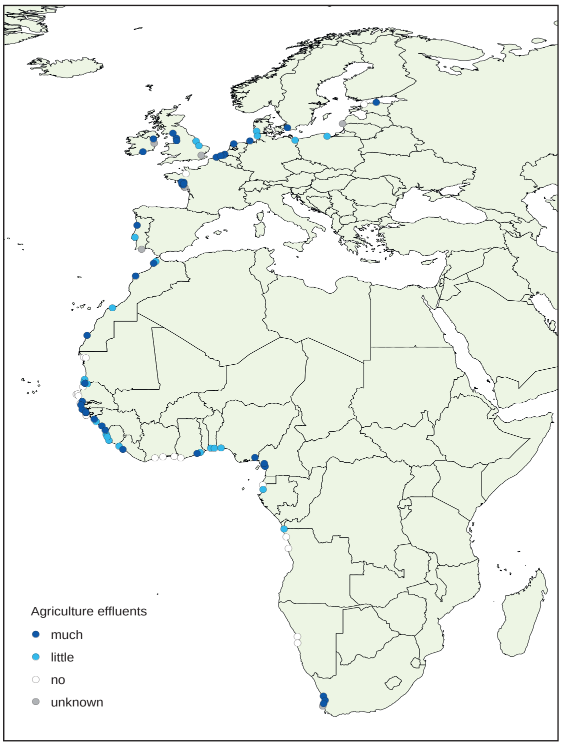


Figure A2.4. Agricultural effluents as a pressure at the assessed sites.

increase of goose populations in NW Europe. Indirectly, pollution by herbicides and pesticides and eutrophication caused by fertilisers (Fig. A2.4) typically increase when

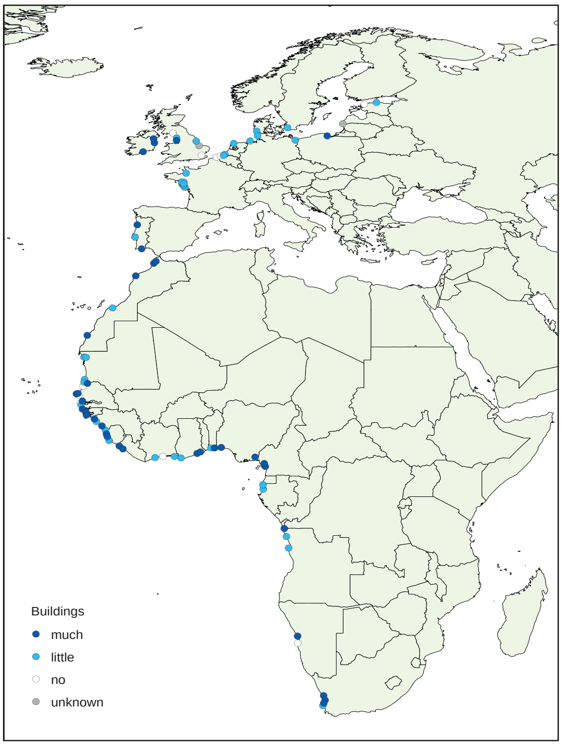


Figure A2.5. Presence and/or expansion of buildings as a pressure at the assessed sites.

farming expands and intensifies. These cause water quality problems, and in the case of fertilizers may lead to the encroachment of aquatic and littoral vegetation. A recent report from The Gambia indicated that open water habitats for waterbirds are declining due to the expansion of reed beds (Sawo *et al.* 2020).

Expansion of built-up areas

More than 40% of the assessed sites are affected strongly by the presence and/ or expansion of cities or settlements (Fig. A2.5). Although a comparison with the 2017 monitoring assessments shows some decline, this is still a dominant pressure across the flyway with the exception of NW Europe. Directly it results in habitat destruction. Indirectly expansion of buildings at or near wetlands often leads to an increase of other pressures linked to human presence, such as disturbance and pollution.

Development/expansion of energy production/ mining

Drilling for fossil fuels or minerals is reported as a strong pressure for only 15% of the assessed sites (Fig. A2.6), particularly in the Gulf of Guinea, for instance in Nigeria, where oil fields at or near the coast are exploited. Sand mining leading to habitat destruction is increasingly reported from some African countries (van Roomen *et al.* 2020). Overall, there has been little change since 2017. Exploitation of renewable energy sources has increased strongly in the last decade, as it is seen as an effective means of combatting global warming. However, these

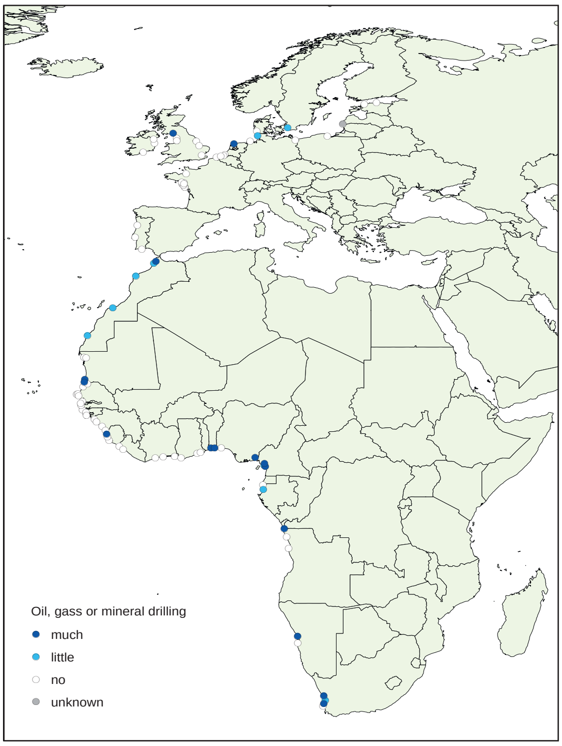


Figure A2.6. Presence and/or expansion of oil, gas or mineral drilling as a pressure at the assessed sites.

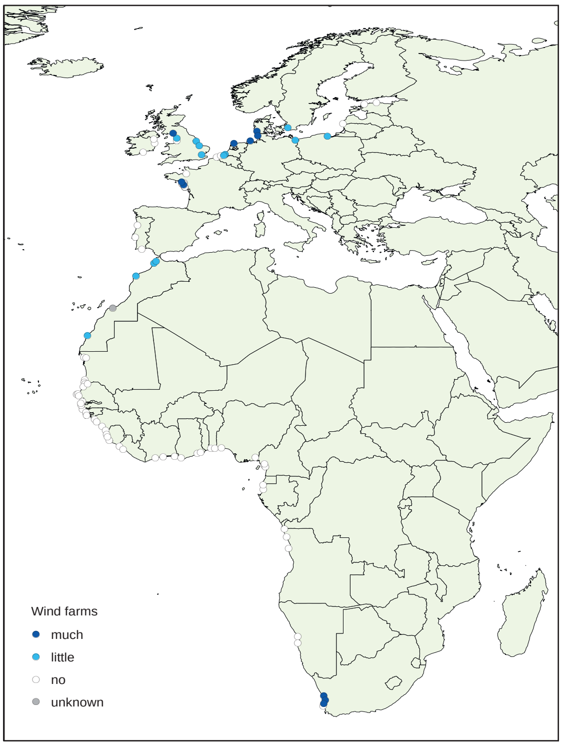


Figure A2.7. Wind farms as a pressure at the assessed sites.

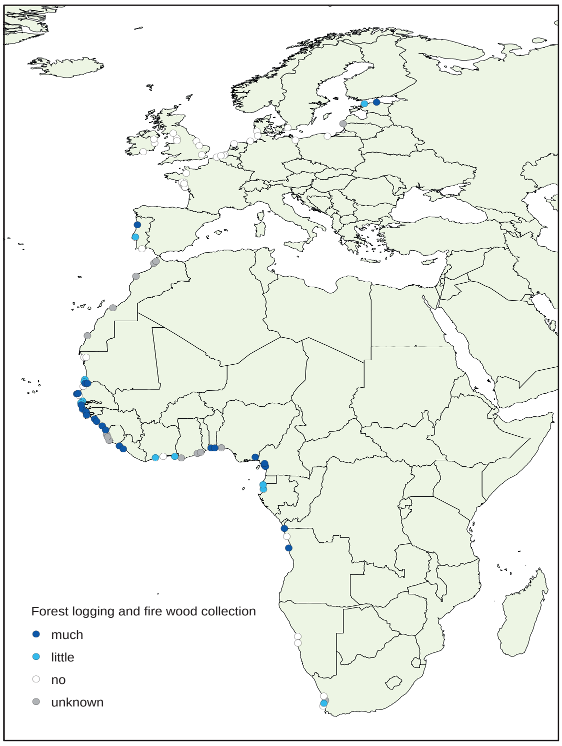


Figure A2.8. Presence of forest logging and firewood gathering at the assessed sites.

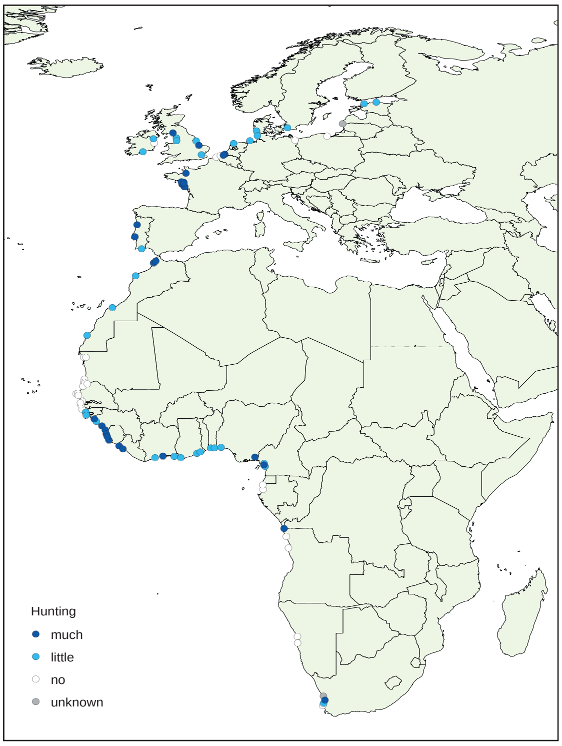


Figure A2.9. Presence of hunting as a pressure at the assessed sites.

operations can have negative side effects on biodiversity when placed in sensitive areas, directly by affecting the habitat or direct mortality among birds, or indirectly by causing disturbance. Coastal areas are favourable locations for wind farms, as they are usually exposed to more and stronger winds. Nowadays wind turbines dominate coastal landscapes in many European countries. However, the environmental monitoring assessments indicate a relatively low impact on waterbirds: about 10% of the assessed sites were affected strongly, mostly in NW Europe (Fig. A2.7). Furthermore, we do not see an increased impact of wind farms on waterbirds compared to the 2017 monitoring results. This suggests that wind farms occur at very few sites of importance to waterbirds, perhaps because of effective nature legislation. In many protected areas in Europe, it is mandatory to conduct a comprehensive environmental impact assessment prior to the construction of any wind farms. Locally, construction of hydropower dams upriver can affect the hydrology of wetlands further downstream.

Utilisation of natural resources

Forest logging and firewood collection is a strong pressure in almost 40% of the assessed sites (Fig. A2.8). Most are located in W Africa and the Gulf of Guinea, where mangrove habitats are more prevalent, and the impacts of logging and deforestation are therefore stronger. Apart from commercial logging in coastal forests, the inhabitants of wetlands or nearby areas are often dependent on wood for cooking or construction, and may gather their firewood in coastal scrublands or in mangroves. Cutting these natural forests reduces the availability of roosting and foraging areas for birds, makes the shoreline more vulnerable to erosion, and makes mudflats more accessible to people, leading to increased disturbance and/or shellfish gathering (van Roomen *et al.* 2020). However, it is noteworthy that, despite ongoing pressures of deforestation, there has been very little change in the extent of mangrove forests between 1996 and 2016 (Annex 3).

Hunting, whether legally or illegally, is considered to be a serious pressure in about 35% of the sites. It can be a direct factor reducing the numbers of individuals. Indirectly it affects bird numbers by causing disturbance, and is especially harmful when birds are disturbed at staging or roosting sites. Hunting is distributed rather evenly across the regions of the flyway, with the exception of Southern Africa where it is a minor pressure or does not play a role at all in most assessed sites (Fig. A2.9). Compared to 2017, hunting was reported as a pressure more frequently.

Fishing is one the strongest pressures for waterbirds across the East Atlantic Flyway (Fig. A2.10). It is by far the most dominant factor affecting waterbirds in W Africa and in the Gulf of Guinea. However, compared to 2017 it seems to have declined slightly. Offshore commercial fishing affects fish stocks and therefore fish-eating waterbirds. Small-scale artisanal fishing by local people in wetlands can cause some disturbance to foraging or roosting waterbirds.

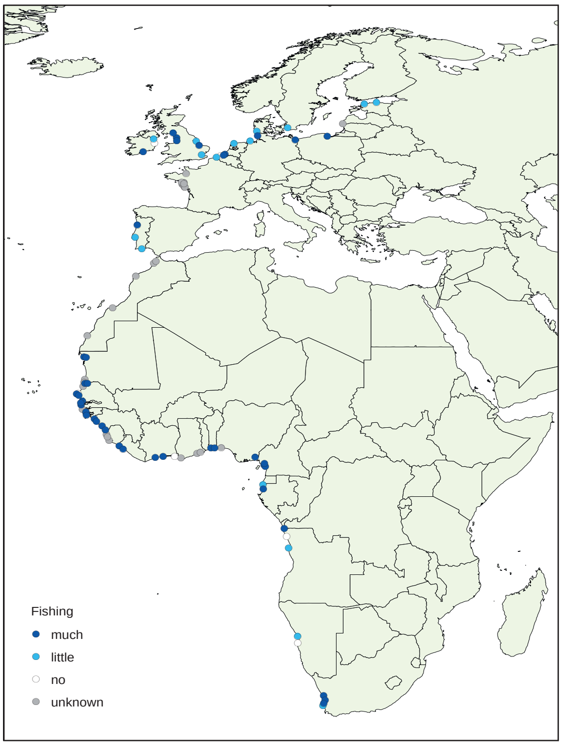


Figure A2.10. Presence of fishing as a pressure at the assessed sites.

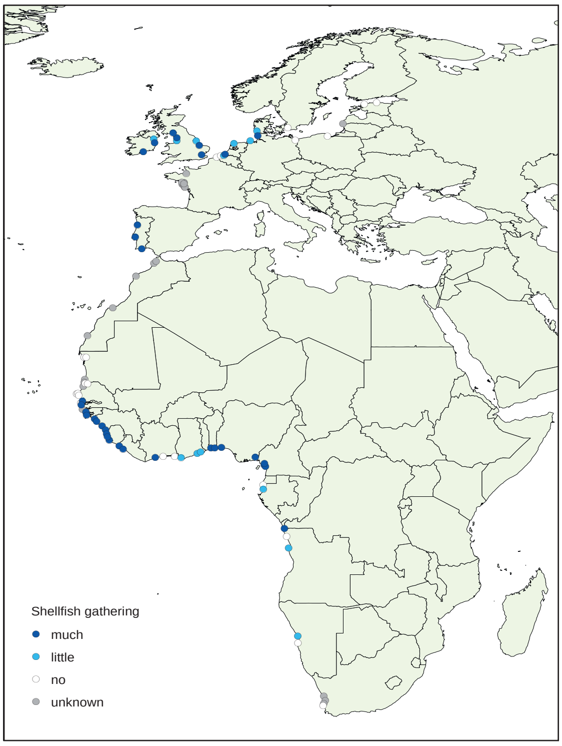


Figure A2.11. Presence of shellfish gathering as a pressure at the assessed sites.

Shellfish gathering is a highly ranked pressure as well (Fig. A2.11). It is dominant in Iberia and W Africa, though is considered to be of little relevance to waterbirds in Southern Africa. It shows little change compared to 2017. The commercial harvesting of shellfish in intertidal areas has caused notable food shortage for some wader and duck species (e.g. Ens *et al.* 2004). Shellfish operations in African countries are smaller in scale than in Europe, but shellfish constitute an important food source for local people. Their presence can cause some disturbance to wader species at mudflats.

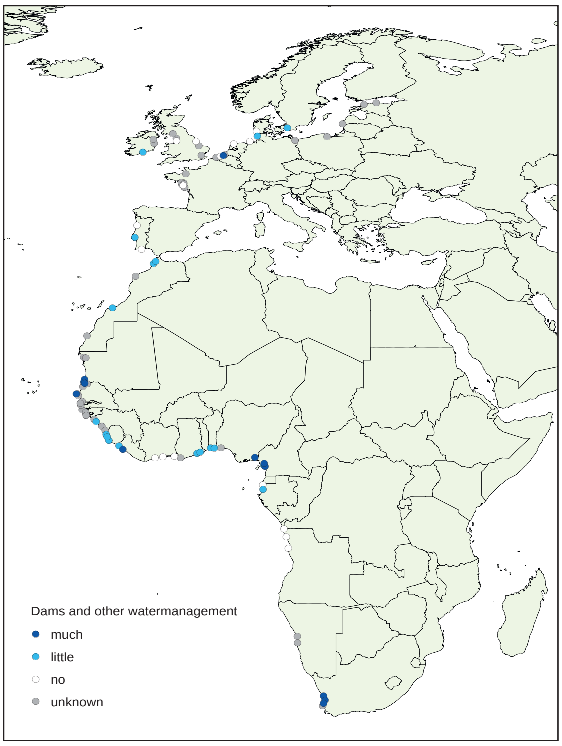


Figure A2.12. Presence of dams and other water management as a pressure at the assessed sites.

Natural system modifications

The foremost reported pressures in this category are the construction of dams and the management of artificial hydrological regimes (Fig. A2.12). As a consequence of these, especially wader species dependent on intertidal habitats have faced a significant loss of habitat. The most notable example is the construction of dams and sluices in the Delta area in the SW Netherlands, which turned several intertidal estuaries into stagnant lakes. On the other hand, it created a new sort of ecosystem with permanent surface waters, which has led to an increase of some other waterbird species (Arts *et al.* 2019).

Human intrusion and disturbance

Several types of activities cause disturbance of waterbirds, including ship traffic, air traffic (Fig. A2.13) and locally military exercises. However, the main source of disturbance reported is recreation/tourism, which featured as a highly dominant pressure across assessed sites in the flyway (Fig. A2.14). It is particularly significant in NW Europe, Iberia - N Africa and Southern Africa. In densely populated coun-

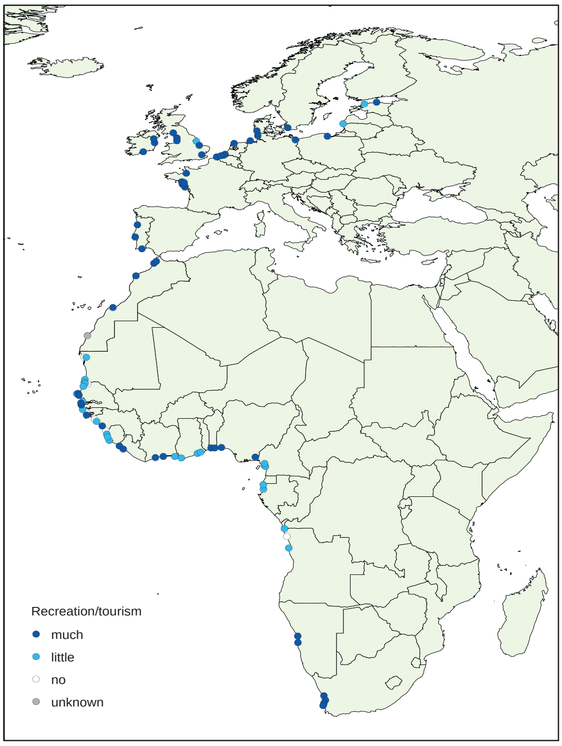


Figure A2.13. Presence of recreation, tourism as a pressure at the assessed sites.

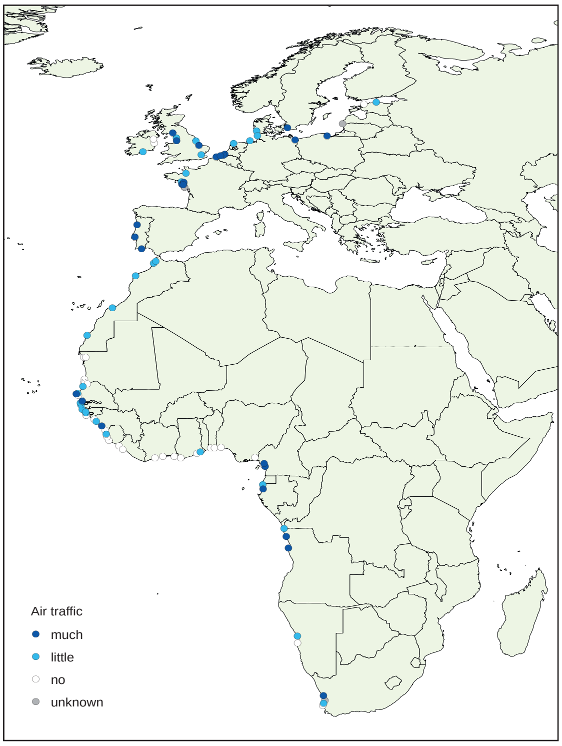


Figure A2.14. Presence of air traffic as a pressure at the assessed sites.

tries, such as Belgium and The Netherlands, coastal sites are used intensively for all sorts of recreational purposes, resulting in unfavourable breeding and staging conditions outside nature reserves where recreation is prohibited or restricted (Devos 2020, Hornman 2020). Recreation is more and more prevalent in African sites also. For instance,

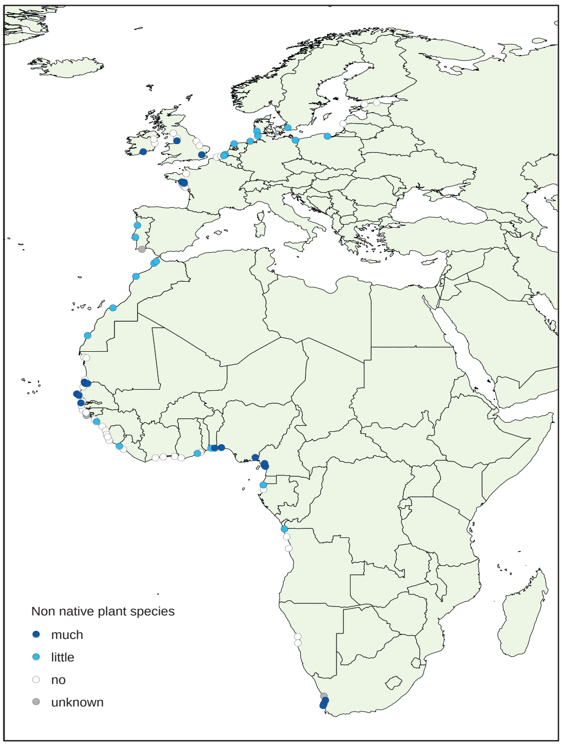


Figure A2.15. Presence of non-native plant species as a pressure at the assessed sites.

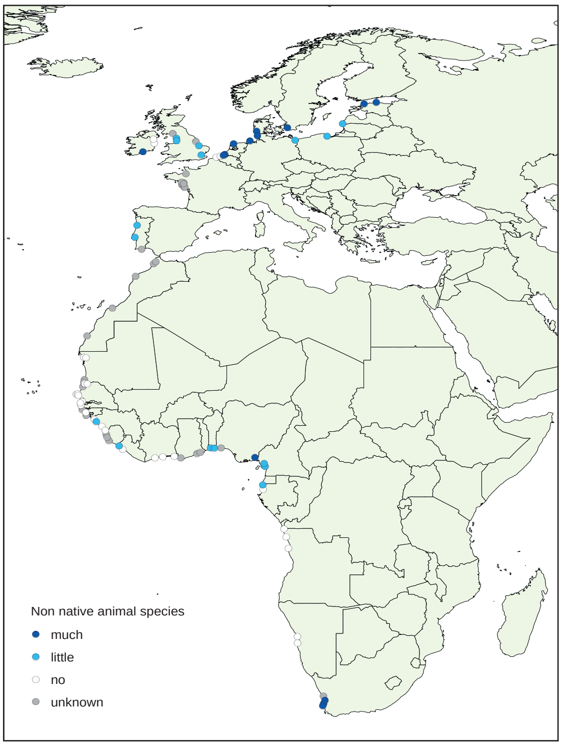


Figure A2.16. Presence of non-native animal species as a pressure at the assessed sites.

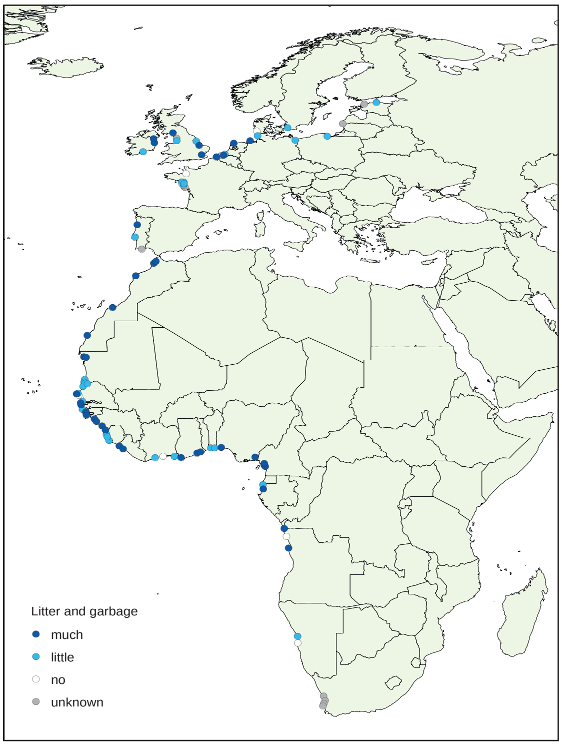


Figure A2.17. (Presence of litter and garbage as a pressure at the assessed sites.

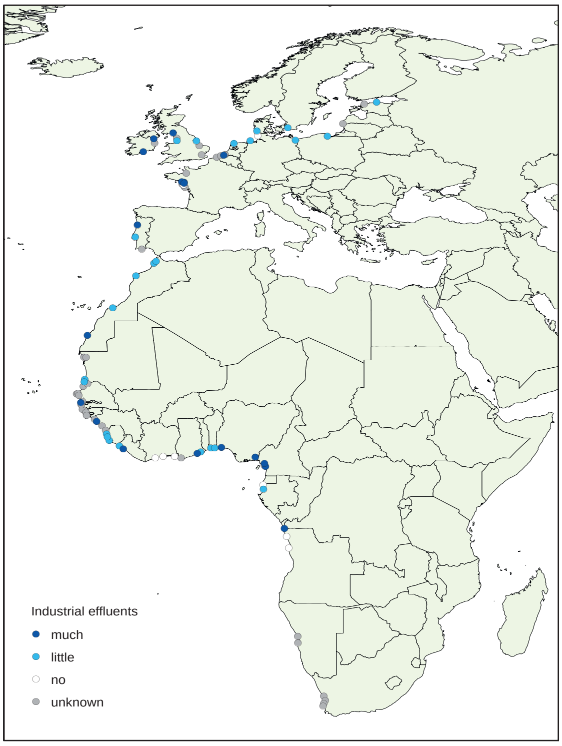


Figure A2.18. Presence of industrial effluents as a pressure at the assessed sites.

near Lagos in Nigeria, tourist facilities are being developed (Onoja 2020).

Invasive and other problem species

The spread of invasive plant and animal species is enhanced by globalisation and climate change. Globally this is considered a major threat to birds, especially from invasive animal species that can affect birds through predation, competition or by spreading diseases. This is particularly relevant to birds restricted to islands. Invasive plants can alter habitats significantly (BirdLife International 2018). Along the flyway, in more than 20% of the assessed sites invasive plants are considered a pressure with high impact (Fig. A2.15). Highest percentages are found in the Gulf of Guinea and Southern Africa. At some sites in Africa aquatic plants such as *Typha* colonise and spread in wetlands, for instance in Diawling in Mauritania (Daf *et al.* 2020). Such plants can grow at a high density and may completely take over large parts of a wetland, significantly reducing the foraging area for waterbirds. Invasive animals are considered a pressure in more than 20% of the assessed sites too. This is particularly relevant in NW Europe and Southern Africa (Fig. A2.16). Locally in Africa stray dogs can pose a problem to waterbirds (van Roomen *et al.* 2020). In NW Europe some alien benthic species are spreading. For instance, Japanese Oysters *Crassostrea gigas* have colonised intertidal areas of the Wadden Sea, where they may affect the availability of shellfish to foraging waders such as Oystercatchers *Haematopus ostralegus* (Waser 2018).

Pollution

Litter and garbage constitute the fourth most prevalent and impacting pressure reported in assessed flyway sites (Fig. A2.17). It is a dominant pressure in Iberia - N Africa, W Africa and the Gulf of Guinea. Water pollution with chemicals is also a substantial pressure (Fig. A2.18). In NW Europe and Southern Africa, agricultural effluents are the most prevalent source (both toxic chemicals and eutrophicating substances), while in the other regions urban waste water is the main pollutant. Industrial effluents were reported less frequently as a serious pressure than in 2017. Overall, the impact of pollution has increased since 2017.

Arnold Meijer / Blue Robin

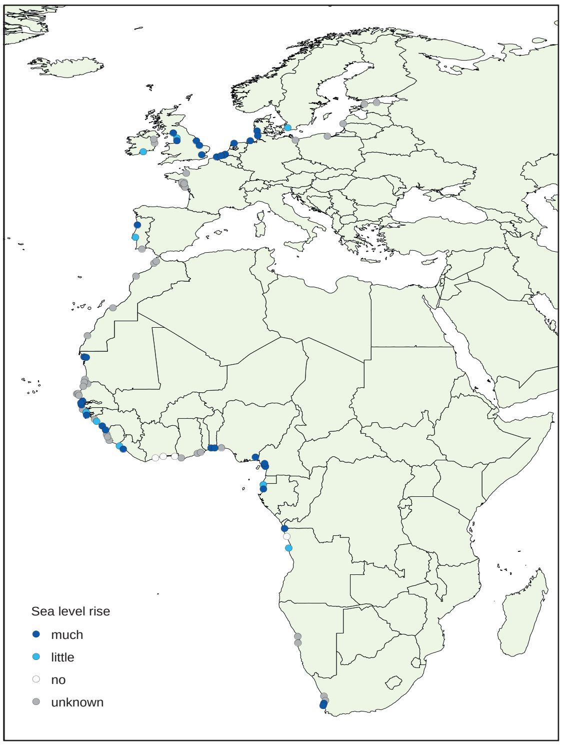


Figure A2.19. Presence of sea level rise as a pressure at the assessed sites.

Climate change

Pressures related to global warming were reported frequently along the flyway, notably the impact of sea level rise (Fig. A2.19). It is often regarded as a future and potentially long-term threat for many sites. The current impacts that are already visible in particular in African countries include flooding of mudflats and shifts of sand banks (van Roomen *et al.* 2020). Some sites face the consequences of extreme weather events. This was, for instance, the case in Southern Africa that suffered from severe drought, affecting freshwater and inland wetlands in particular. The frequency of spring storms seems to be increasing in W Europe, causing flooding to some shorebird colonies (van de Pol *et al.* 2010).

Wintering waterbird numbers in Scandinavian and Baltic countries are increasing because milder winter weather has rendered their wintering habitats more favourable (see Chapter 2).

A2.3. Conservation measures

Conservation of coastal wetlands is vital for the continued survival of waterbirds, especially for migratory birds that depend on a network of key sites. As is shown in the previous sections, waterbirds face several pressures at most of their sites along the flyway. At more than 90% of the assessed sites measures are needed to counteract pressures (Fig. A2.20). The results also show that conservation action has made progress: at least some measures have been taken at more than 80% of assessed sites (Fig. A2.21). About 80% of the assessed sites are formally protected by national or international laws/agreements (Fig. A2.22). This protection is considered highly effective at most European

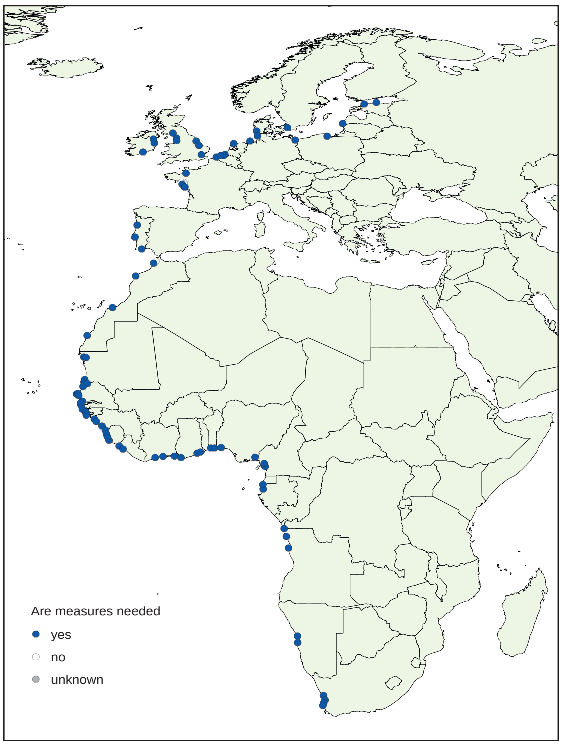


Figure A2.20. Overview of sites where conservation action is needed.

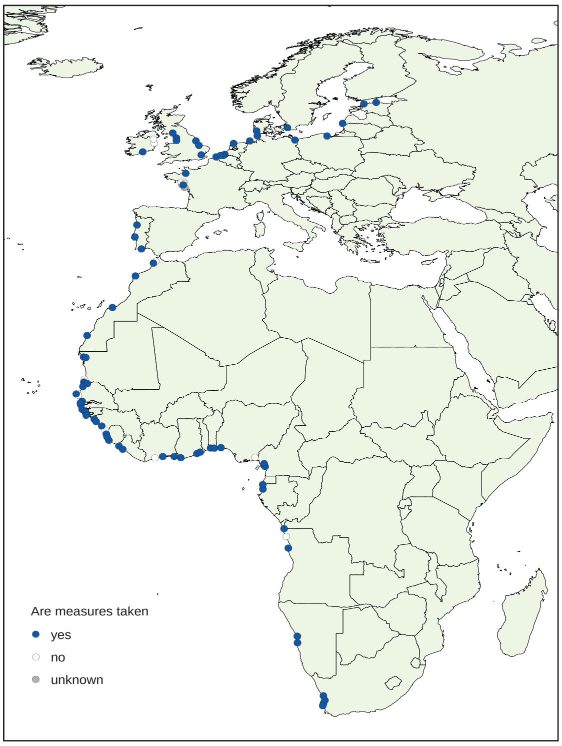


Figure A2.21. Overview of sites where conservation measures have been taken.

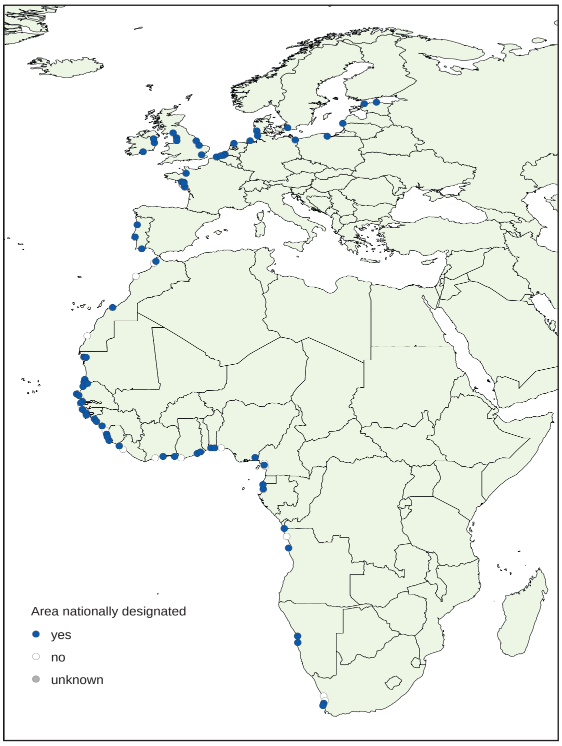


Figure A2.22. Overview of sites which have formal national protection.

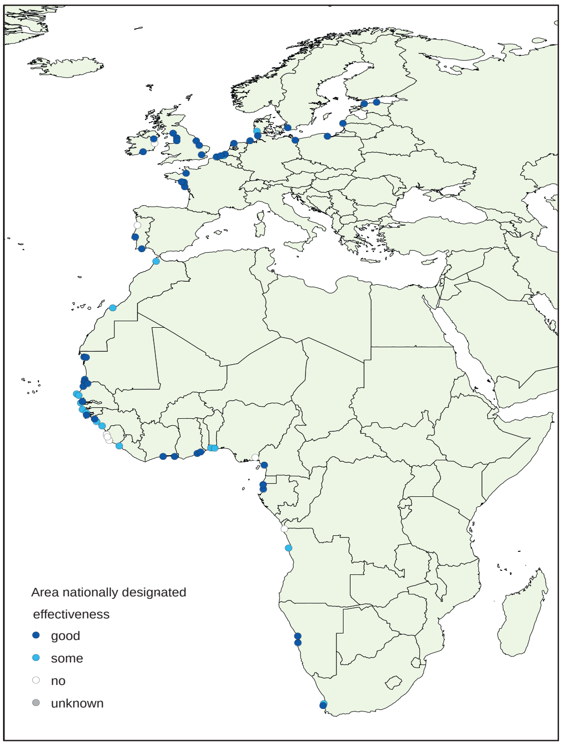


Figure A2.23. Overview of the effectiveness of the national protection per site.

sites. However, in Africa, there are some sites where protection has proven to be less effective or in some cases has no impact at all (Fig. A2.23).

Expansion/intensification of agriculture

Measures to regulate agricultural land-use have been implemented in about 50% of the assessed sites

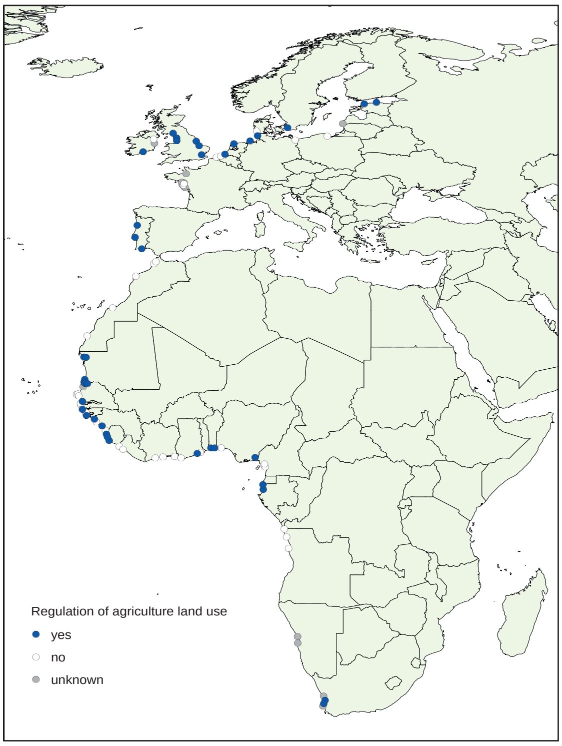


Figure A2.24. Overview of sites where measures have been taken to regulate agricultural land-use.

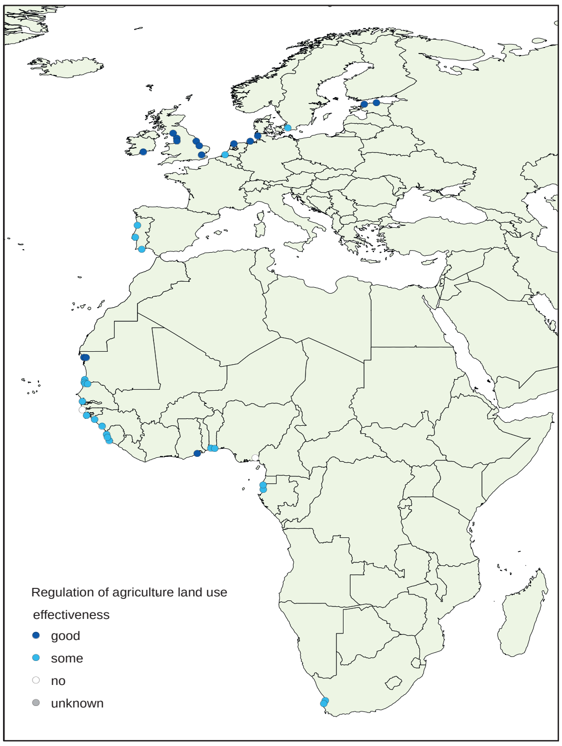


Figure A2.25. Overview of the effectiveness of the regulation of agricultural land-use per site.

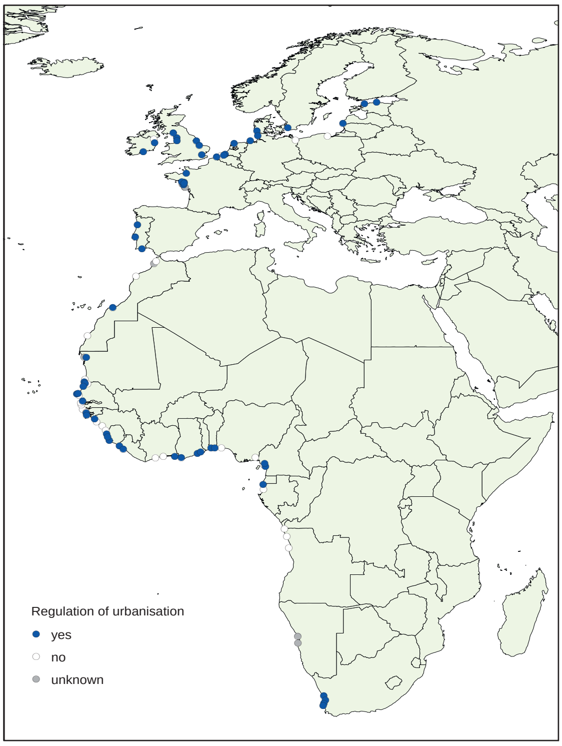


Figure A2.26. Overview of sites where measures have been taken to regulate urbanisation.

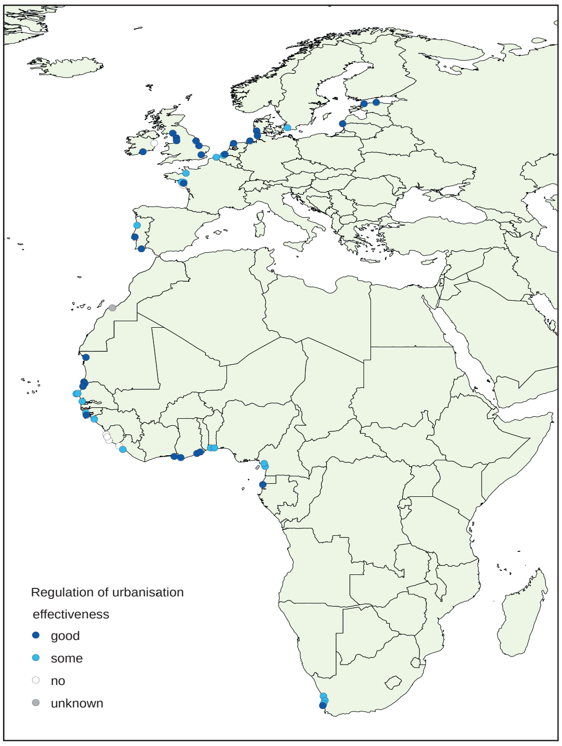


Figure A2.27. Overview of the effectiveness of regulation of urbanisation per site.

(Fig. A2.24). The effectiveness seems highest in NW Europe, where 97% of the assessed sites are designated as protected areas (Fig. A2.25). There it is generally prohibited to convert land of importance to nature to farmland. Existing farmland is increasingly managed to increase biodiversity. At some places farmland is even given back to nature, for instance to restore saltmarshes or intertidal areas.

Expansion of built-up areas

Measures to regulate urbanisation have been taken in about 50% of the assessed sites (Fig. A2.26). In most sites these measures have shown at least some positive impact (Fig. A2.27). The main exception is Sierra Leone.

Development/expansion of energy production/mining

In almost 50% of the assessed sites measures have been taken to regulate the impact of fossil energy production. For instance, in and near the Wadden Sea the impact of gas drilling on the hydrology is assessed (Kleefstra *et al.* 2021). Measures to regulate wind farms have been taken widely along the flyway (Fig. A2.28). Plans to build new wind farms near protected areas have to be evaluated in an environmental impact assessment. These measures show at least some effect at most sites across the flyway (Fig. A2.29).

Han Bouwmeester / Agami

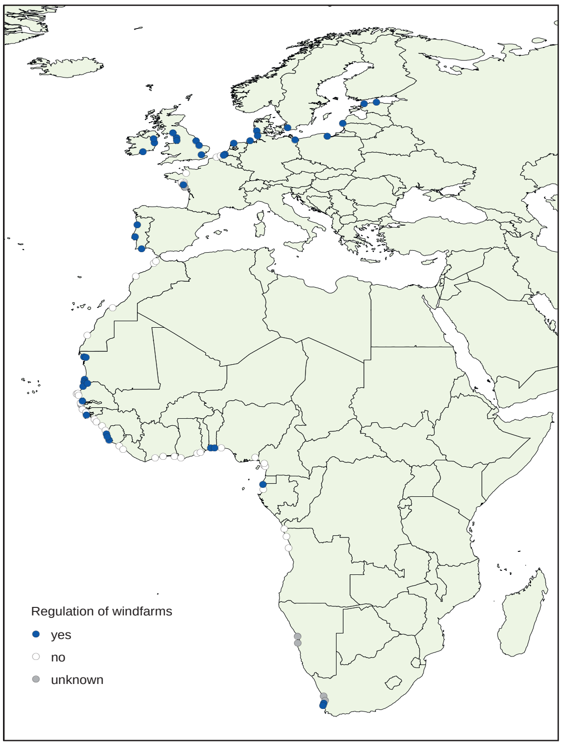


Figure A2.28. Overview of sites where measures have been taken to regulate wind farms.

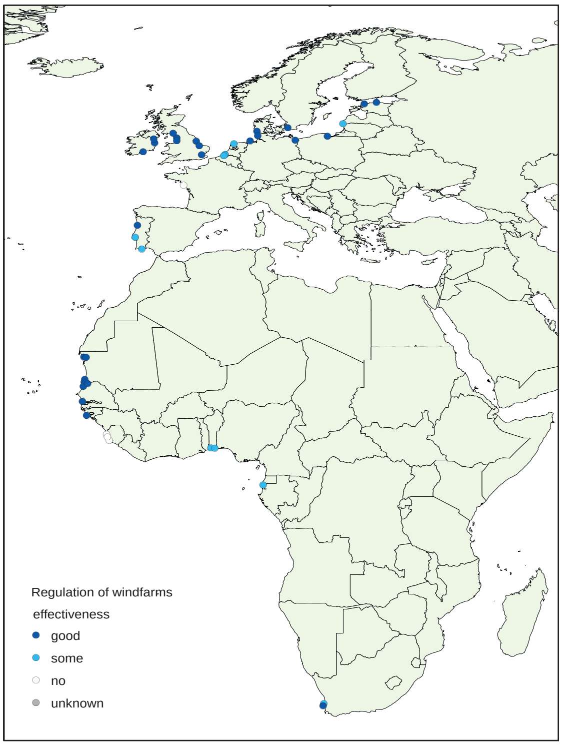


Figure A2.29. Overview of the effectiveness of the regulation of wind farms per site.

Natural system modifications

Measures to improve hydrological regimes have been undertaken in almost 40% of the assessed sites, particularly in Europe (Fig. A2.30). Most measures have proven to be effective to at least some extent (Fig. A2.31). In NW European countries, such as Belgium and the Netherlands,

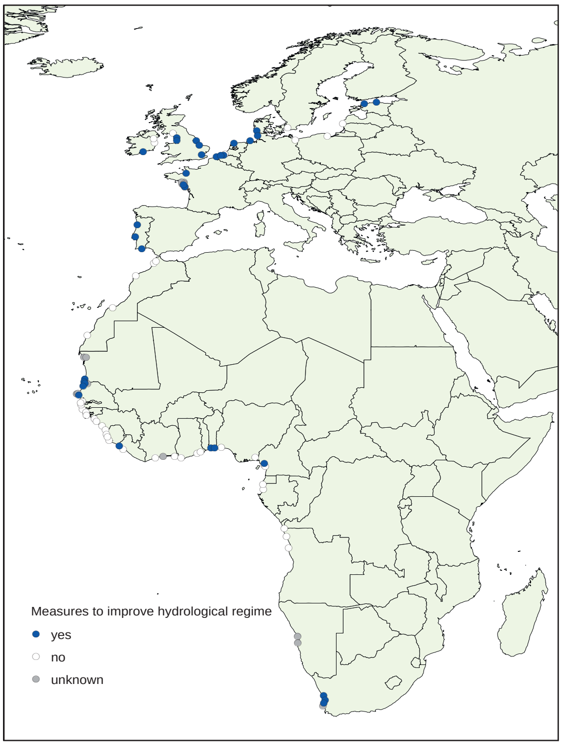


Figure A2.30. Overview of sites where measures have been taken to improve the hydrological regime.

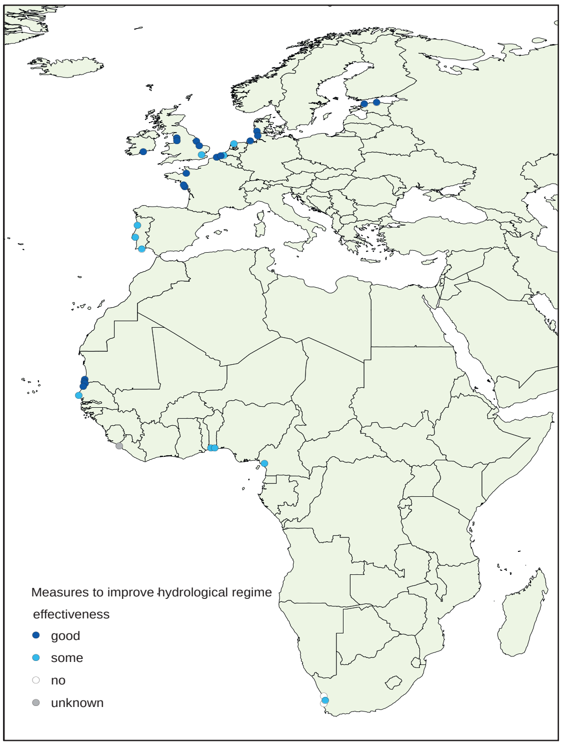


Figure A2.31. Overview of the effectiveness of measures to improve the hydrological regime.

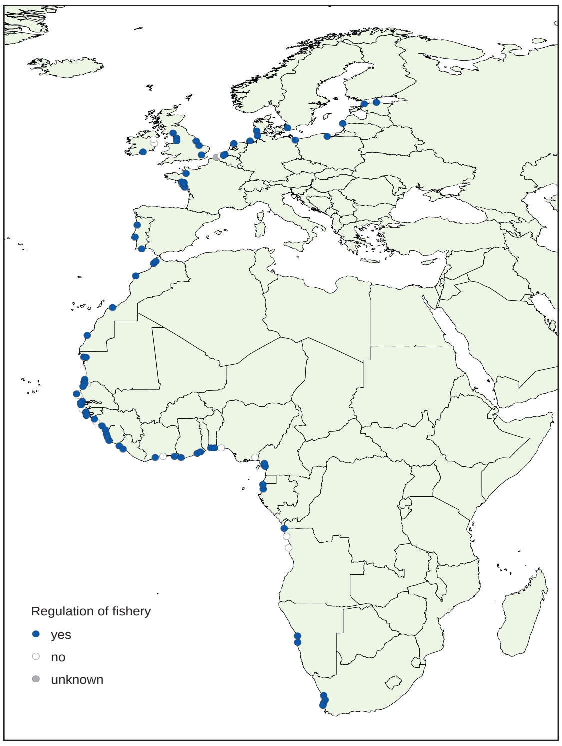


Figure A2.32. Overview of sites where measures have been taken to regulate fisheries.

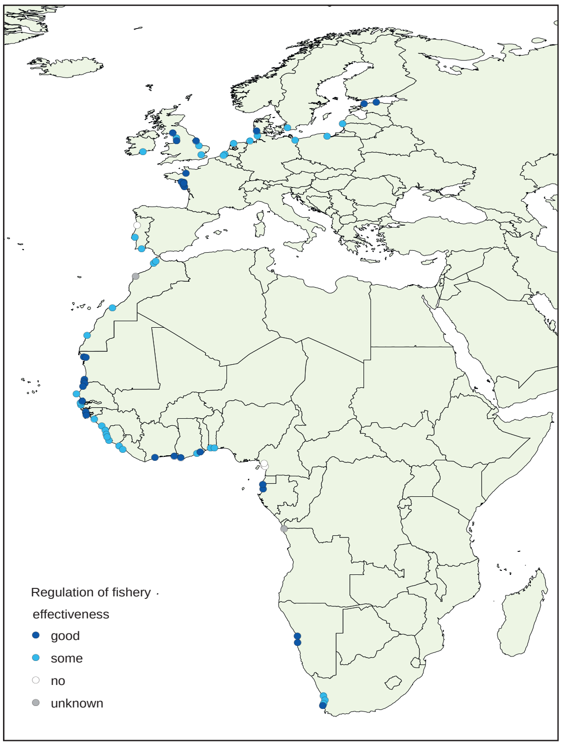


Figure A2.33. Overview of the effectiveness of measures to regulate fisheries per site.

some projects have been carried out to create or restore salt marsh habitats.

Utilisation of natural resources

Formally, fishing is regulated in about 80% of the assessed sites (Fig. A2.32). This is to be expected, given that 80% of the sites are under national or international protection. This regulation was considered highly effective in around half of the sites, and have at least some effect in most of the sites (Fig. A2.33). Comparable statistics apply to hunting. For instance, in Denmark, 90 shooting-free reserves have been designated (Clausen *et al.* 2020) and this seems effective to some extent. Measures to regulate forest cutting have been taken in around 50% of the assessed sites (Fig. A2.34). This is particularly relevant in W Africa and in the Gulf of Guinea, where large stretches of mangroves are found along the coast. At most of the sites the measures have had some effect (Fig. A2.35), though it takes time for forests to recover or re-establish after improved protection and/or replanting.

Tim Dodman

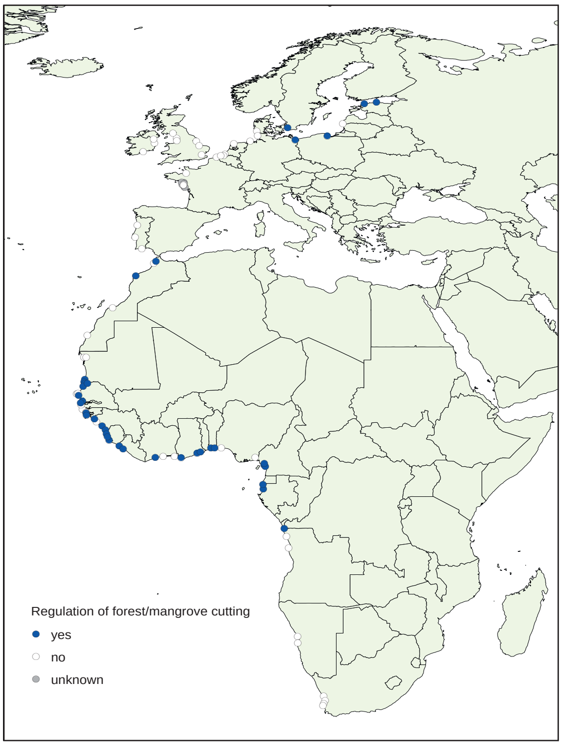


Figure A2.34. Overview of sites where measures have been taken to regulate forest/mangrove cutting.

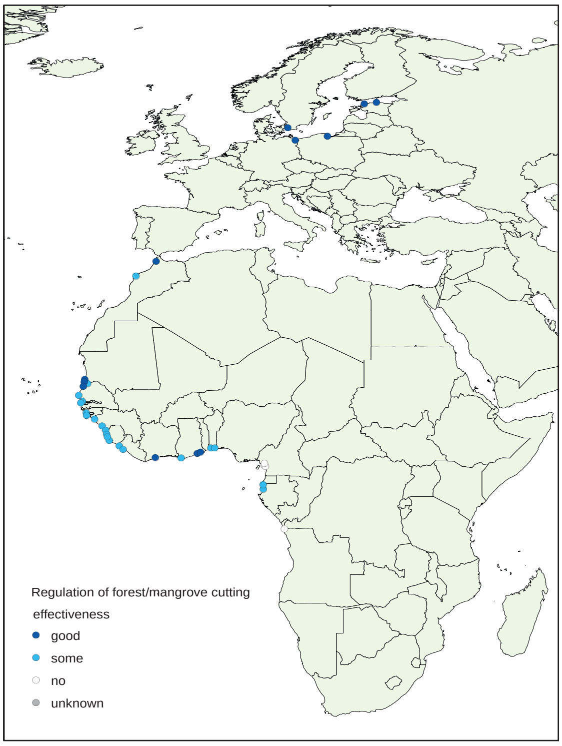


Figure A2.35. Overview of the effectiveness of measures to regulate forest/mangrove cutting per site.

Human intrusions and disturbance

Recreational and tourism activities are regulated at around 60% of the assessed sites (Fig. A2.36). Measures seem to be effective, especially in Southern Africa and W Africa (Fig. A2.37). Recreation and tourism is a key component of most assessed sites in NW Europe, with most of them being managed for nature and recreation. Some countries such as Denmark and the Netherlands restrict or zone the more disturbing recreational activities, such as the use of motorboats and wind- and kitesurfing. Such activities are uncommon at most sites in Africa.

Invasive and other problem species

Measures have been taken to control invasive plant and/or

Tim Dodman

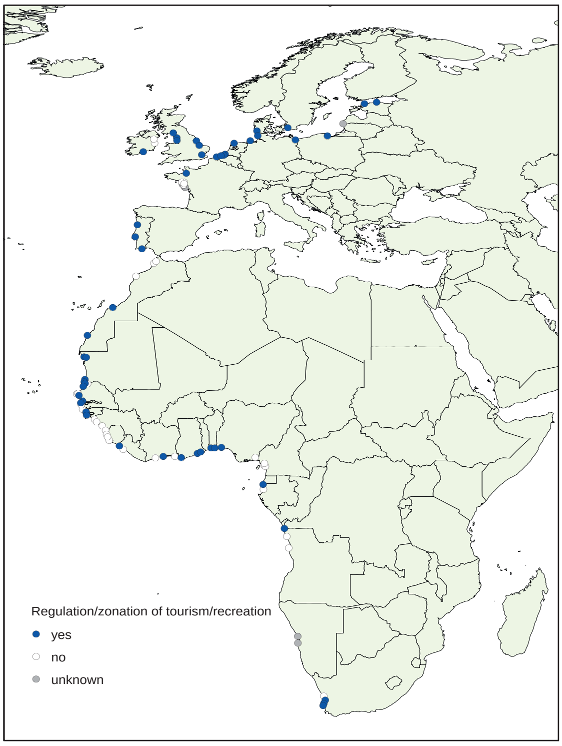


Figure A2.36. Overview of sites where measures have been taken to regulate tourism or recreation.

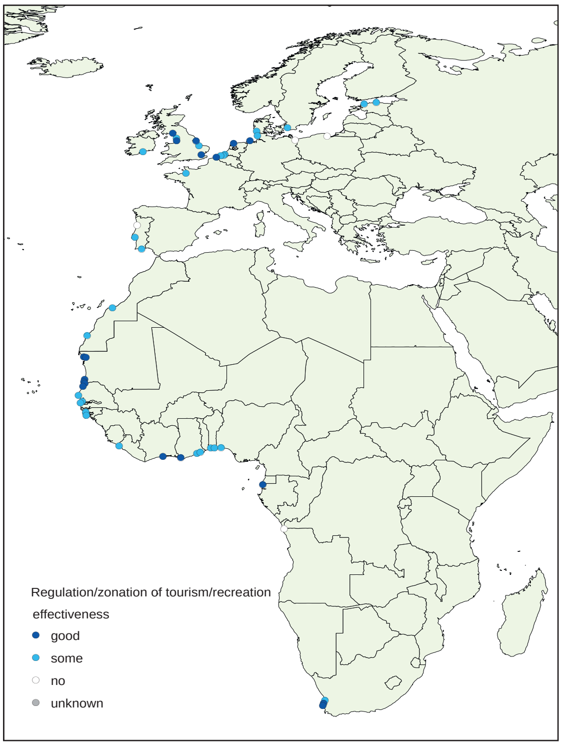


Figure A2.37. Overview of the effectiveness of measures taken to regulate tourism of recreation per site.

animal species in one-third of assessed sites (Fig. A2.38). At European sites, these measures seem to be having some effect, but are generally not highly effective (Fig. A2.39). Fewer or no measures have been taken at sites across Africa, where non-native species often were not listed as major pressures. Extensive measures have been

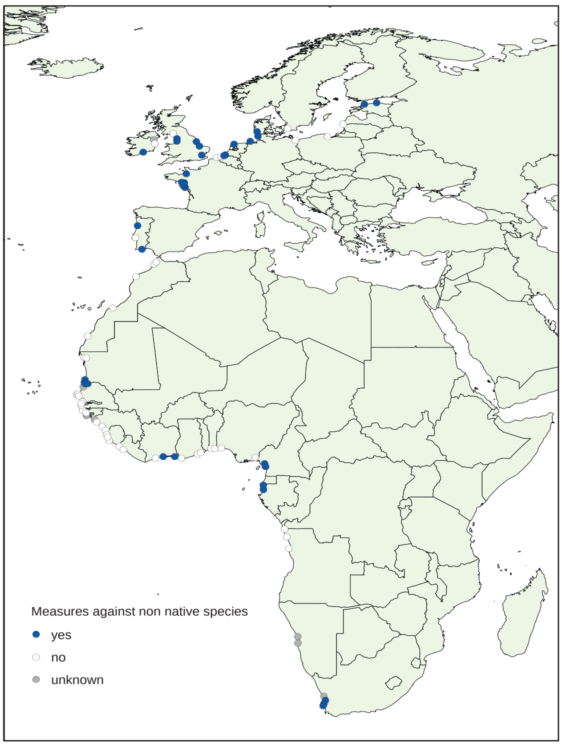


Figure A2.38. Overview of sites where measures have been taken against non-native species.

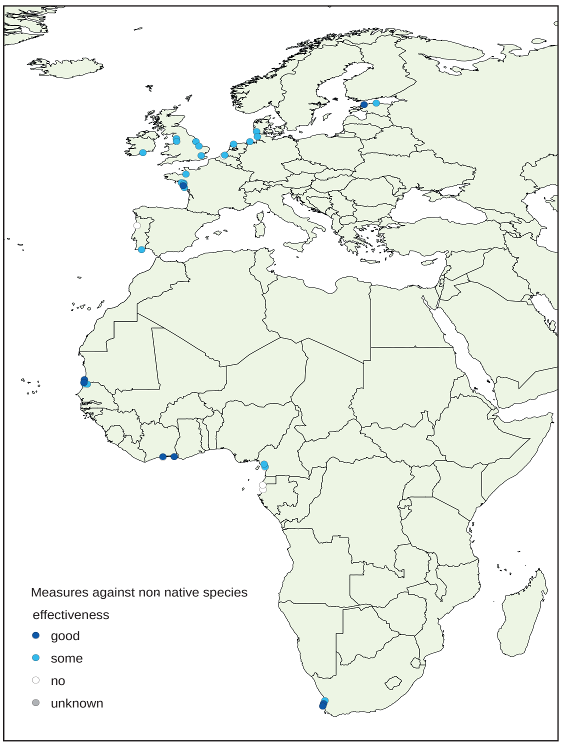


Figure A2.39. Overview of the effectiveness of measures taken against non-native species.

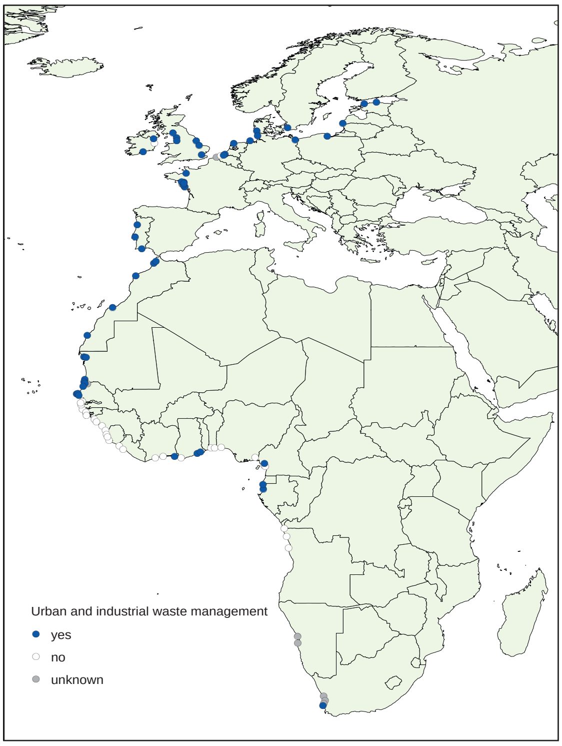


Figure A2.40. Overview of sites where measures have been taken to regulate urban and industrial waste.

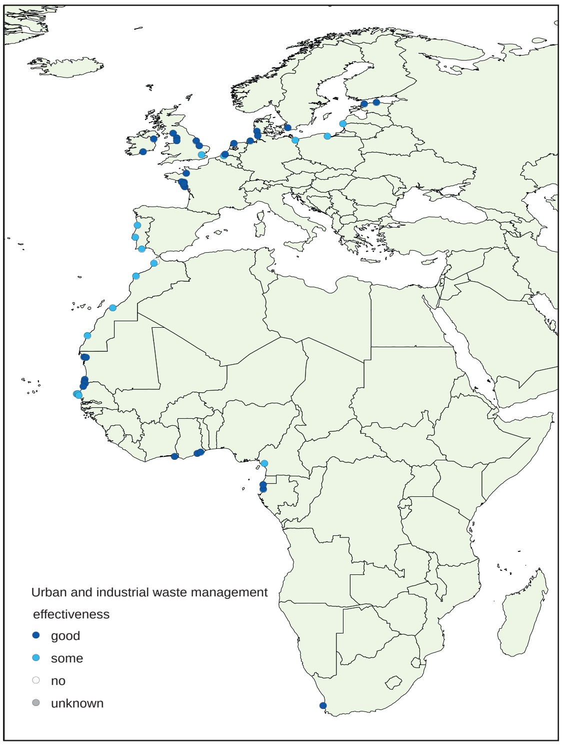


Figure A2.41. Overview of the effectiveness of measures taken to regulate urban and industrial waste.

taken however over the years to control invasive plant species in the Lower Senegal Valley in W Africa, which continue to some extent.

Pollution

Measures to regulate urban and industrial waste have been taken at more than 50% of the assessed sites (Fig. A2.40). These were widely recorded as being effective, especially in Southern Africa and the Gulf of Guinea, where implemented (Fig. A2.41).

People

An important part of conservation work is the involvement of the local people in site policy and management, e.g. through site support groups (SSGs). Local communities were engaged in conservation work at almost 70% of the assessed sites (Fig. A2.42). Raising awareness about nature and conservation measures was conducted in around 60% of the assessed sites. Conservation research was carried out in around 50% of the sites, all along the flyway (Fig. A2.43).

Miguel Xavier

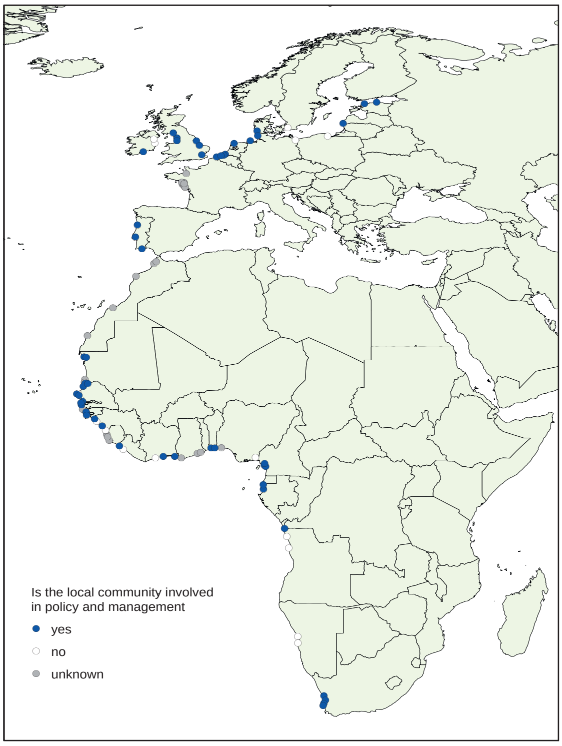


Figure A2.42. Overview of sites where the local community is involved in policy and management.

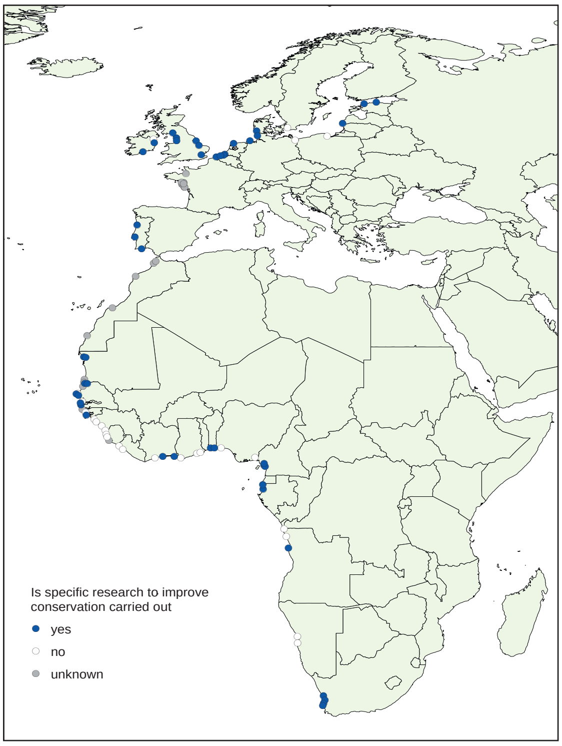


Figure A2.43. Overview of sites where conservation research is carried out.

A2.4. Discussion and recommendations

This was the third time a significant effort was made to collect environmental data from across the flyway in a coordinated manner, using a consistent format. It is promising that the number of forms received increased from 73 sites in 2017 to 115 in 2020. The different regions of the flyway are represented rather well in the sample. The most notable omissions were the lack of data from Spain, and very little data from Sweden. Some attention should be given to obtaining data from these countries in the future.

This monitoring approach is strongly dependent on expert opinion, and it is acknowledged that different experts can interpret pressures differently and have different views on their impacts. Therefore, we recommend that improvements to the outputs could be made with the help of a panel of experts who would review the scores and threats independently, and follow up directly with the network of participants. Data from remote sensing and other global datasets could also be usefully support and inform monitoring assessments.

The format of the questionnaires has improved. However, in the habitat section, one important habitat type, namely mangroves / forest, was not included. The data processing of the monitoring forms when returned from the coordinators is labour-intensive, and we recommend an online form with an underlying central database for future assessments.

Erik van Winden

For literature references mentioned in this Annex, see the main reference list in Chapter 15

Wadden Sea, Netherlands (Marc Guyt / Agami)

Annex 3. Remote sensing and other global datasets included in the analyses

Olivia Crowe, Emma Teuten & Adrian Hughes

A3.1 Introduction

Remote sensing allows to determine changes in land cover (and other environmental indicators) from a range of high-resolution datasets now available. Some of these datasets are hosted on dedicated platforms, which reduces the need for downloads of very large datasets and also often enables analyses within the platforms using these datasets.

A selection of key remote-sensing layers, and other spatial datasets with relevance to (changes in) the environmental state, pressures and human responses in wetlands along the EAF (EAF) were downloaded, or accessed through Google Earth Engine (GEE), a cloud computing platform that enables users to run large scale, complex geospatial analysis on Google's servers. Datasets were managed using the Mollwiede equal area projection for all area calculations in the analysis, with the exception of some conducted in Google Earth Engine which used the Lambert Azimuthal equal area projection.

Data from these layers were extracted at two spatial scales, local (at the site level) and regional. The seaward and terrestrial (inland) boundaries for the regions were defined by the coastline, including the area 5 km out to sea and 40 km inland, respectively, hereafter referred to as 'regional subunits' for the purpose of describing the statistical results at the broader regional scale. The site analyses were based on those IWC sites for which boundaries were available, drawing upon others where available and appropriate (Ramsar Sites and Important Bird and Biodiversity Areas (IBAs)). In total 72 sites were included. The Arctic region was removed from these analyses due to poor data coverage of some of the key datasets.

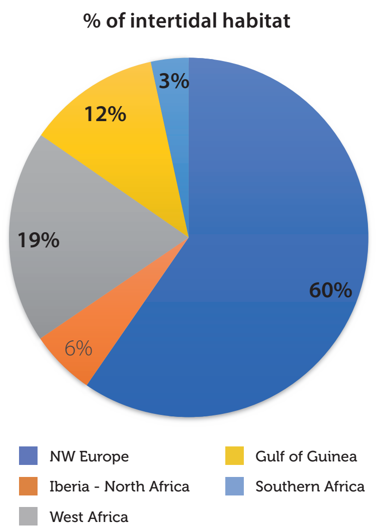
The remote sensing datasets included and further details about the usage, limitations and analyses relating to these datasets are described in the following paragraphs.

A3.2 Global Intertidal Change

About the layer: Global Intertidal Change (https://www.intertidal.app/): The spatial extent of the non-vegetated areas of Earth's coastline that undergo regular tidal inundation (Murray et al. 2019).

The layer was developed using 56 predictor layers of which many were Landsat composite metrics designed to identify individual pixels that undergo frequent wetting and drying over a 30-year period (1984-2016) divided into 10 three-year interval layers. Version 1.1 of the data is available on Google Earth Engine and also as a direct download. A new version (1.2) was made available for use in this project by Nicholas Murray. It includes a more recent interval layer (2017-2019) and covers a shorter time period (1999-2019) but with greater accuracy due to more images being used in the analysis.

Limitations: The layers achieved >82% accuracy when compared to independent, globally distributed valida-



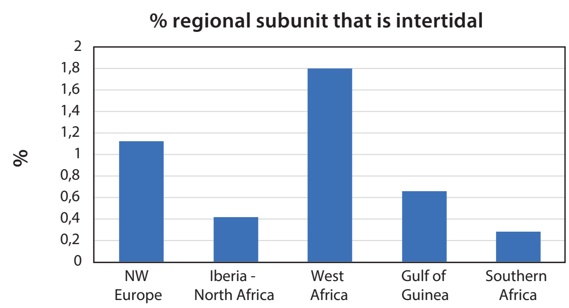


Figure A3.1. Extent of intertidal area across the regions, as % of the overall area in the EAF (left), and by regional subunit (limited by a 5km offshore zone).

tion data. Some errors occur as intertidal areas share spectral characteristics with other land cover such as coastal development and aquaculture. Also, the number of images used in the analysis varies over space and time, leading to further losses of accuracy. While the layer is accurate at the global and regional scale, its usefulness at site level is more limited. Ideally, local intertidal layers should be created using local training data, but this effort is beyond the scope of this report.

Extreme caution needs to be taken when comparing different time intervals. Owing to the variable availability of Landsat images over the study period, each time step in the intertidal change data has a varying coverage and precision. An appropriate statistical model is needed to accurately compare time periods. This was beyond the scope of this report.

Analyses and results: We used the Reduce Regions method in Google Earth Engine to calculate the extent of intertidal areas in the EAF regions for the 2017-2019 time period. Overall, the largest amount (by area) of intertidal area is found in NW Europe followed by W Africa (Fig. A3.1), although the percentage of occurrence compared to other habitat types is highest in W Africa. Limitations with respect to the quality of the data from earlier years prevented a longer-term assessment of change.

Potential value to future EAF environmental monitoring assessments: This layer will inform the extent, and change in extent, of intertidal habitats over time (state).

A3.3 Global Surface Water

About the layer

Global Surface Water (https://global-surface-water.appspot.com/): Monthly data on the location and temporal distribution of surface water from 1984 to 2020 using Landsat imagery (Pekel *et al.* 2016). This dataset was used mostly to inform the extent of possible wetlands on the inland side of the coastal boundary. The European Commission's Joint Research Centre developed this water dataset in the framework of the Copernicus Programme. It contains maps of the location and temporal distribution of surface water from 1984 to 2020 using Landsat imagery and provides statistics on the extent and change of those water surfaces. Each pixel was individually classified into water / non-water using an expert system, and the results were collated into monthly data points.

Limitations

Variations in accuracy and availability of water detection are dependent on the data archive of Landsat imagery, which varies over space and time. Generally, the further you go back in time the fewer detections are documented due to less suitable imagery being available. Also, due to the low angle of the sun in the northern hemisphere in winter, there are no data for these periods in northern areas, e.g. NW Europe.

Analyses and results

To identify wetland areas, we used the monthly datasets to identify areas where water was detected in any month between December and February in three-year periods: 1997-2000, 2007-2010 and 2017-2020. Due to the lack of winter detections at northern latitudes (see Limitations above) we needed to use a different approach for NW Europe. We extracted the monthly dataset and selected areas where there had been water detected in at least 50% of the months in the three-year period, i.e. detection in 18 or more months out of 36. We did this for 2018-2020 to give an indication of current water extent but found that results for the periods 1998-2000, 2008-2010 were not reliable due to lack of data (no detections possible). For all analyses we used the Reduce Regions method in Google Earth Engine to calculate surface water extent in the regional subunits and the sites.

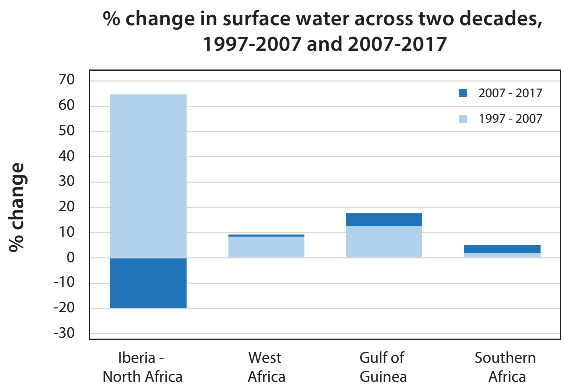


Figure A3.2. Percentage change in surface water occurring in IWC sites along the EAF between 1997-2007 and 2007-2017.

Surface water at sites collectively increased in all regions between 1997 and 2017 (note that NW Europe was not included because of limited data quality from earlier periods). It was highest in Iberia - N Africa and the Gulf of Guinea. In most regions the highest increase was between 1997 and 2007 (Fig. A3.2), with a much smaller increase between 2007 and 2017. It is interesting to note that surface water in sites in Iberia - N Africa actually decreased between 2007-2017.

Potential value to future EAF environmental monitoring assessments

This layer will inform the extent, and change in extent, of surface water within sites over time (state).

A3.4 Global Mangrove Watch

About the layer

Global Mangrove Watch (https://data.unep-wcmc.org/

datasets/45): Mangrove extent and change, using a baseline for mangrove coverage in 2010 and comparing this against seven time periods between 1996 and 2016 (derived from Bunting *et al.* 2018). Global Mangrove Watch (GMW) is a collaboration between Aberystwyth University (U.K.), solo Earth Observation (soloEO; Japan), Wetlands International, the World Conservation Monitoring Centre (UNEP-WCMC) and the Japan Aerospace Exploration Agency (JAXA). The GMW provides geospatial information about mangrove extent and change. It developed a baseline for global mangrove coverage in 2010 using ALOS PALSAR and Landsat (optical) data. Changes from this baseline for seven time periods between 1996 and 2016 were derived using JERS-1, ALOS and ALOS-2.

Limitations

The Global Mangrove Watch dataset has an overall accuracy of 94% (Bunting *et al.* 2018). This provides reliable results at a regional scale, but caution should be taken when analysing data at the site scale, especially where the total mangrove area is small. A single pixel in the GMW data has an area of 900 m^2.

Analyses and results

The Tabulate Area in ArcGIS Pro v2.6 was used to calculate the area (m^2) of mangrove in each site and regional subunit, using the mangrove coverage in 1996, 2010 and 2016. These values were used to calculate the % change between 1996 and 2010, 2010 and 2016, and overall (1996 – 2016). A processing cell size of 30 m was used, in the Mollweide projection.

The analyses inferred that very little change has taken place between 1996 and 2016 in the extent of mangroves, both at the regional and the site (1.3% decline) levels. It is possible that the extent of decline has been higher, but that this has been masked by replanting, or natural regeneration, which probably explains the increase in mangrove extent in W Africa between 1996 and 2010. Furthermore, if protected, mangroves can re-establish themselves quite well. So, site protection alone could also result in an increase in mangrove cover.

Potential value to future EAF environmental monitoring assessments

This layer will inform the extent, and change in extent, of mangroves within sites over time (state).

A3.5 Urbanisation

About the layer

Urbanisation (https://www.arcgis.com/home/item.html?id=1453082255024699af55c960bc3dc1fe): Spatial extent and change in urbanisation over time. This global urbanisation analysis covers the period between 1992-2015 (ESA 2017). The urbanisation analysis employed the European Space Agency Climate Change Initiative (ESA-CCI) land cover, which covers the period 1992 – 2015. Subsequently, this has been updated annually by the Copernicus Climate Change Service (C3S). Every effort has been made to ensure consistency across the entire time series.

Limitations

While the data are reasonably accurate at the regional subunit scale (user accuracy for the urban class is 86-88%), they are less reliable at the site scale. For example, urbanisation appears to have been overestimated at Songhor Lagoon (GH00007) and Keta Lagoon complex (GH00003).

Analyses and results

The data were accessed directly in ArcGIS Pro 2.6, through ESRI's Living Atlas. The data were provided as a multidimensional raster, and a definition query was applied to access data from a single year. The Tabulate area tool was used to calculate the area in the Urban Areas category (code 190). The area of urban pixels in each regional subunit was calculated for the years 1992, 2000, 2010 and

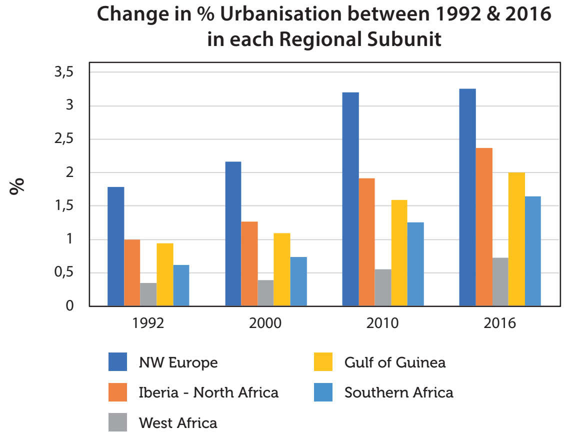


Figure A3.3a. Change in the percentage of regional subunits covered by urbanisation between 1992 and 2016.

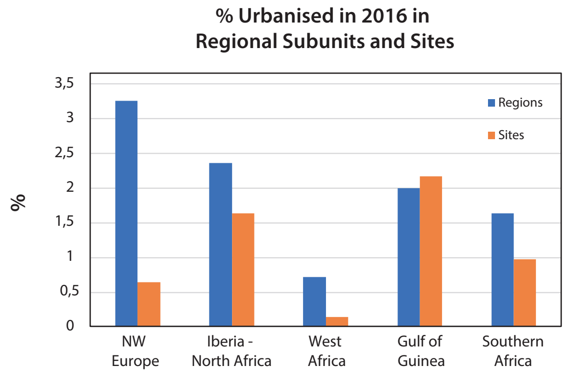


Figure A3.3b. Percentage of regional subunits and sites that were urbanised in 2016.

2019. Change in area between each of these years was determined. The analysis was repeated at the site level using the same methodology.

There has been an increase in urbanisation across all regions between 1992 and 2016 (Fig. A3.3a), and in most regions the highest rate of increase was between 2000 and 2010; especially in NW Europe. For most regions the extent of urbanisation is highest in the regional subunits compared with sites (Fig. A3.3b), except in the Gulf of Guinea where the proportion of sites that has been urbanised is slightly higher than the proportion across the regional subunits.

Potential value to future EAF environmental monitoring assessments

This layer will inform the extent, and change in extent, of urbanisation within sites over time, allowing measures of habitat loss, as well as inferences about human-related pressures. For example, increased urbanisation may lead to an increase in recreation, increased pollution etc. Such analyses at a broader, regional level are also appropriate, e.g. conclusions may be drawn that waterbird sites may be under increasing levels of human pressure where an increase in urbanisation at the regional-level is shown.

A3.6 Global Fishing Watch

About the layer

Global Fishing Watch (GFW, https://globalfishingwatch.org/data-download/datasets/public-fishing-effort): Dataset that informs the temporal and spatial extent of vessels at sea and fishing. It is based on the detection of a ship's Automatic Identification System (AIS) which generates information about fishing effort and vessel presence (Kroodsma *et al.* 2018). The International Maritime Organization requires large ships to broadcast their position with Automatic Identification System (AIS) in order to avoid collisions. The GFW Effort dataset contains AIS-based information about fishing effort and vessel presence. This dataset is based on fishing detections of >114,000 unique AIS devices on fishing vessels. Fishing vessels are identified via a neural network classifier, vessel registry databases and manual review by GFW and regional experts.

Limitations

AIS was made mandatory for vessels over 15 m length in the EU in May 2014. The observed apparent increase in vessel and fishing hours over time probably reflects the implementation of this legislation in the regions that include Europe (NW Europe and Iberia - N Africa).

While the GFW contains the most comprehensive information on global fishing effort, there are limitations of using AIS in its measurement. Not all fishing vessels have / use AIS. It is not required for smaller vessels. Its use can be hindered by coverage and transmission gaps. The system can be deliberately switched off to conceal illegal fishing activities.

Satellite imagery (AIS data harvested from GFW) can detect vessels above a certain size, meaning that coverage is much higher for larger vessels (> 24 meters) and for the high seas. Some areas have also a poor AIS reception. AIS works best in general for the high seas. At the point of conducting this analysis, GEE only contained data for 2012 – 2016. Due to the logistical challenge of processing a dataset of this size, GEE was used where possible, and data from 2020 were subsequently extracted.

The datasets prior to 2017 are not complete, thereby limiting trend assessments, but they have been improving ever since. The limitations above, together with the known underestimation resulting from artisanal, illegal and/ or unregulated fishing activities in some of the regions limit our ability to use these datasets with much confidence. Into the future, for EEZs, it may be better to look at Vessel Monitoring Systems (VMS) data. These data are currently not easy to obtain. GFW also have other datasets including for monitoring ports, anchorages and transhipment which can provide further insight into the movements of individual vessels. Alternatively, it may be worth looking at a combination of different datasets, including logbooks from RFMOs, VMS and AIS (from GFW).

Potential value to future EAF environmental monitoring assessments

Stricter regulations and improvements in technology across the EAF are needed to deliver accurate results about the extent of vessel operations at sea and fishing.

A3.7 Gridded Population of the World

About the layer

Global populations (Gridded Population of the World, GPW (UN WPP-adjusted 2015), https://sedac.ciesin.columbia.edu/data/collection/gpw-v4): Modelled distribution of human populations (counts and densities) on a continuous global raster surface (CIESIN 2016). This dataset informs extent and change in population occurrence over time, thereby inferring change in human-related pressures. Since the release of the first version of this global population layerin 1995, the essential inputs to GPW have been population census tables and corresponding geographic boundaries. The purpose of GPW is to provide a spatially disaggregated population layer that is compatible with data sets from social, economic and Earth science disciplines, and remote sensing. It provides globally consistent and spatially explicit data for use in research, policy-making and communications. We used the UN WPP-adjusted dataset which is based on counts from national censuses and population registers, but are

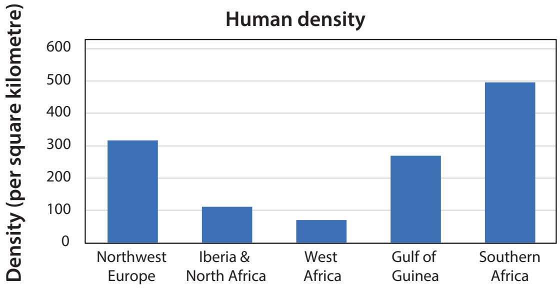


Figure A3.4a. Human density inside IWC sites in each region in 2020.

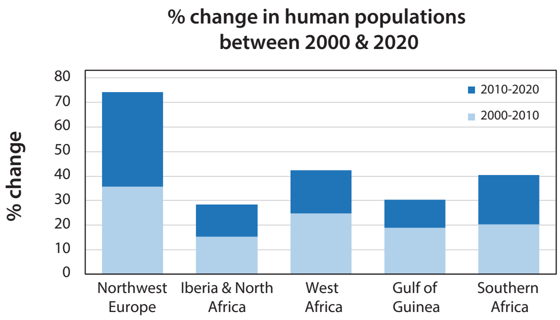


Figure A3.4b. Change in human populations inside IWC sites in each region between 2000 and 2020.

adjusted to match official UN population estimates. As a result, these rasters will have greater consistency across countries for regional or global analyses.

Limitations

Care is needed if reviewing results at small scales, such as at site level, particularly for some of the smaller IWC sites.

Analyses and results

We used Zonal Statistics tools in ArcGIS to aggregate population counts across sites in years 2000, 2010, and 2020. We also calculated percentage of change metrics for 2000-2020 and 2010-2020.

At the regional subunit level, the density of human populations in IWC sites is highest in Southern Africa (Fig. A3.4a) and lowest in W Africa. Densities were shown to increase in all regions between 2000 and 2020, and the rate of increase has been highest in NW Europe (Fig. A3.4b).

Potential value to future EAF environmental monitoring assessments

The effects of human-related pressures as inferred by changes in human population levels are not necessarily curtailed to what is happening within the site boundary. Future analyses should also include a buffer zone beyond the site boundary, perhaps treated separately from the site-level assessment. Future analyses should also include a broader regional-level component.

Like urbanisation, changes in human population levels can be used as a proxy for human-related pressure. However, its application is somewhat limited in terms of interpreting which pressure and the magnitude of its effects. For example, an increase in human density within a site could be associated with an increase recreational disturbance, urban waste, litter and garbage, pollution, among others. It would not be immediately possible to identify from this dataset alone which pressure, or whether the change in human population levels has actually caused a change in the effects of any of the pressures.

A3.8 World Database on Protected Areas

About the layer

World Database on Protected Areas (https://www.protectedplanet.net/en): The World Database on Protected Areas (WDPA) is the most comprehensive global database of marine and terrestrial protected areas, and was used to determine the extent of coverage of protected area cover of wetlands at regional and local scales. It is a joint project between the UN Environment Programme and the International Union for Conservation of Nature (IUCN), and is managed by UN Environment Programme World Conservation Monitoring Centre (UNEP-WCMC), in collaboration with governments, non-governmental organisations, academia and industry.

Limitations

The quality of the WDPA dataset is dependent on the quality of the data provided by the many data providers. Data quality can vary greatly between countries and over time. Some providers may provide poorly digitized boundaries or may provide data many years after a site is designated. Some older sites may not have information in the year of designation. Some sites only exist as point locations and are therefore not included in the polygon dataset used in our analysis. Due to some data providers taking years to provide data on designated sites it is generally best practice not to use older versions of the WDPA for time comparisons but to use the latest version and use the 'status_yr' field to filter out sites designated in previous years. However, this method does not take into account boundary changes to existing sites in the intervening period.

Analyses and results

We used the August 2021 version of the WDPA and removed proposed sites and those designated as UNESCO MAB Biosphere Reserves. We created a 'dissolved' layer to remove all overlaps across the protected area types to produce one layer of protected area cover. We estimated the proportion of intertidal area across each regional sub-

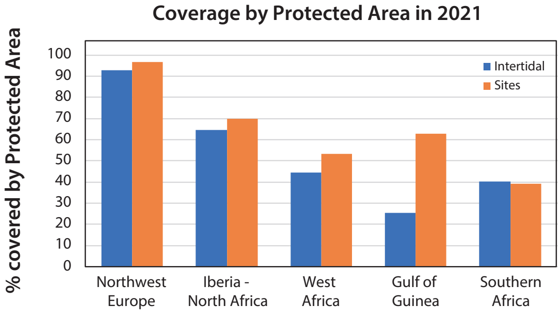


Figure A3.5a. Coverage by protected area of the intertidal areas (at regional subunit scale) and of sites overall in 2021.

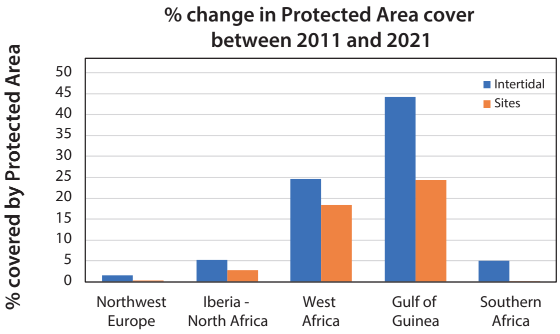


Figure A3.5b. Percentage of change in protected area cover between 2011 and 2021 of the intertidal areas (at regional subunit scale) and of sites overall.

unit covered by protected areas; for the sites we estimated the proportion covered by protected areas. We repeated this process using a layer with sites designated in 2011 or before to show change in protection over the last 10 years.

For the regions we used the Tabulate Area tool in ArcGIS to calculate the amount of intertidal area (Global Intertidal 2014-2016 see above) protected in each region for all protected areas and protected areas designated in 2011 or before. The percentage of change was then calculated from 2011-2021. For sites we used the Tabulate Intersection tool in ArcGIS to calculate the amount of protected area in sites for all protected areas and protected areas designated in 2011 or before. A percentage change was then calculated from 2011-2021.

In total, 11,185 km^2 of intertidal area was protected in 2021, representing 72% of intertidal areas overall (based on the intertidal area generated for the 2017-19 period). Coverage is especially high in NW Europe, much of this due to the designation of a large extent of intertidal habitats under the European Union's Birds and Habitats Directives.

In most regions, the extent of protected area coverage of intertidal areas (regional subunit level) and of sites was similar, the biggest exception being the Gulf of Guinea where sites were well covered but intertidal areas much less so (Fig. A3.5a). The proportion of intertidal area (regional subunit al level) and sites covered by protected areas was highest in NW Europe and lowest between W Africa and Southern Africa (Fig. A3.5a).

Overall, there was a slight increase in protected area cover between 2011 and 2021 in intertidal areas across each regional subunit (5%) and across the sites (8%), but there was quite a lot of variation between the regions (Fig. A3.5b), with coverage increasing substantially in both the Gulf of Guinea and W Africa, and much less so in NW Europe, presumably because much of the important sites in this region have already been designated several decades ago. The increase in protected area coverage between 2011 and 2021 was also relatively small in Iberia – N Africa and in Southern Africa.

Potential value to future EAF environmental monitoring assessments

This layer will continue to inform the protected area coverage of the intertidal habitats and other sites in the region (response to pressures). Perhaps future analyses could be improved if the outer boundary of the regional subunit was delimited by a 5 km border and extended to include the extent of intertidal habitats where they extend beyond this limit.

While it is useful to be able to measure the extent of protected area coverage, and change over time, it should be noted that the extent to which sites covered by protected areas are actually protected from potentially damaging activities varies widely, both within and between countries and regions across the EAF.

For literature references mentioned in this Annex, see the main reference list in Chapter 15