Seabird Monitoring Programme

Seabird Population Trends and Causes of Change: 1986–2024

The annual report of the Seabird Monitoring Programme

BTO Birds Science People

JNCC

in association with

RSPB

THE SMP REPORT 2024

Welcome to *Seabird Population Trends and Causes of Change: 1986–2024, the annual report of the Seabird Monitoring Programme*. This report presents the latest seabird population trends in breeding abundance and productivity using data from the Seabird Monitoring Programme (SMP). We are grateful to everyone involved in the SMP, from the surveyors that monitor each breeding season, to those in the offices that coordinate the programme and to the organisations providing knowledge, experience and advice to steer the programme forward. Thank you.

Sarah Harris, SMP Organiser, BTO

SMP GOVERNANCE AND PARTNERSHIP

The SMP is funded jointly by BTO and JNCC, in association with RSPB, with fieldwork conducted by both non-professional and professional surveyors. The programme is also supported by a wide network of organisations that form an Advisory Group and by the SMP Steering Group comprised of Helen Baker (JNCC), Dawn Balmer (BTO), Mark Bolton (RSPB), Niall Burton (BTO), Tim Dunn (JNCC) and Tom Evans (RSPB). Steering Group meetings are also attended by the Statutory Nature Conservation Bodies: Department of Agriculture, Environment and Rural Affairs, Northern Ireland (DAERA), Natural Resources Wales (NRW), Natural England (NE) and NatureScot.

Advisory Group members:

BirdWatch Ireland
British Trust for Ornithology
Department of Agriculture, Environment and Rural Affairs
Department of Housing, Local Government and Heritage
Fair Isle Bird Observatory Trust
Highland Ringing Group
Isle of Man Government
Joint Nature Conservation Committee
Manx BirdLife
Manx National Heritage
Marine Directorate, Scottish Government
National Trust
National Trust for Scotland
Natural England
Natural Resources Wales
NatureScot
Royal Society for the Protection of Birds
Scottish Wildlife Trust
Shetland Oil Terminal Environmental Advisory Group
States of Guernsey
The Seabird Group
UK Centre for Ecology and Hydrology
University of Gloucestershire
Wildlife Trust of South and West Wales

BTO SMP TEAM

The team from BTO included Sarah Harris, the SMP Organiser to September 2025. Erin Taylor joined BTO in November as SMP Organiser and is now the first point of contact. Erin is responsible for running the programme, liaising with professional and volunteer participants, maintaining the database, promoting the programme, and producing the annual report, newsletter and other outputs. Nina O'Hanlon, Senior Research Ecologist in the Wetland and Marine Research Team, is responsible for data analysis and annual trend production. Hala Haddad and Andrew Upton support the Seabird Network in Northern Ireland. Aaron McKay supports the SMP and works closely with Erin. Dawn Balmer, Head of Surveys, provides project management to the SMP, alongside other monitoring schemes. Niall Burton, Head of Wetland and Marine Research, and Liz Humphreys, Head of Marine Research, both in the Wetland and Marine Research Team, are responsible for strategic development of the programme and marine research at BTO. James Pearce-Higgins is the BTO Director of Science and therefore responsible for all survey and research work at BTO. In addition to those above, representatives from a total of 24 organisations form the SMP Advisory Group (listed above).

Birds Science People BTO
British Trust for Ornithology
www.bto.org

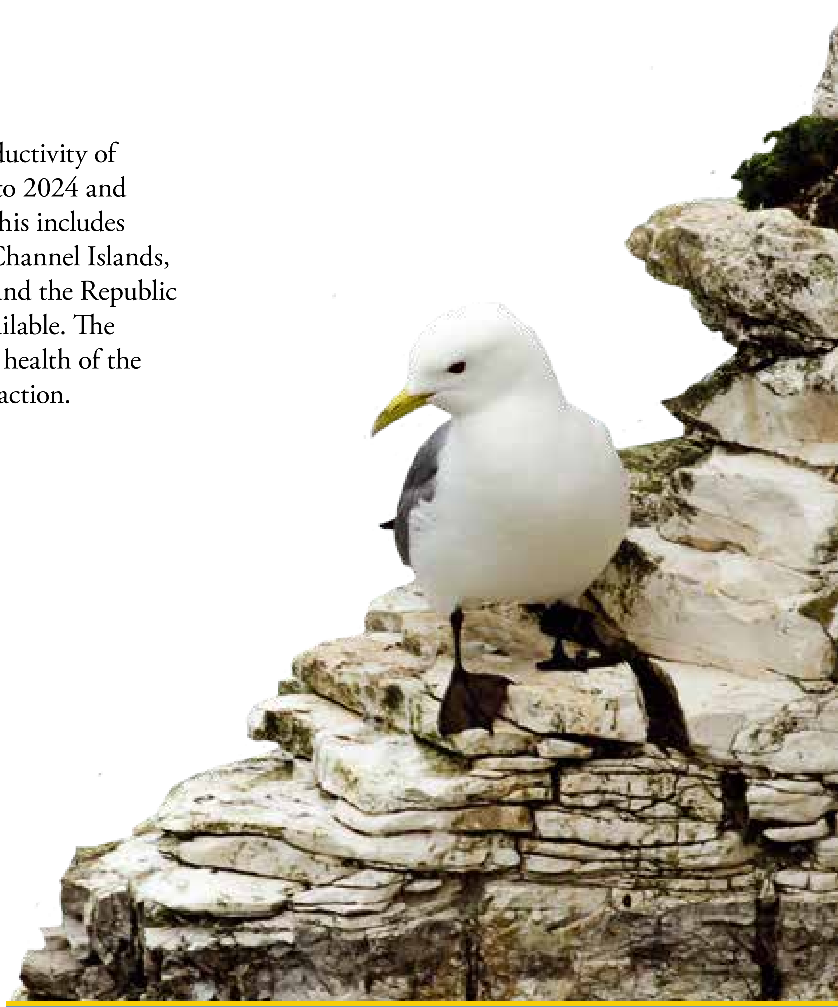
JNCC
Joint Nature Conservation Committee
www.jncc.gov.uk

in association with
RSPB
Royal Society for the Protection of Birds
www.rspb.org.uk

KITTIWAKE COVER IMAGE, BY SAM LANGLOIS / BTO

CONTENTS

This report documents changes in the abundance and productivity of breeding seabird species in Britain and Ireland from 1986 to 2024 and provides a detailed account of the 2024 breeding season. This includes both inland and coastal populations and trends from the Channel Islands, England, Isle of Man, Northern Ireland, Scotland, Wales and the Republic of Ireland which are presented where sufficient data are available. The results from this report are used more broadly to assess the health of the wider environment, to inform policy and for conservation action.



KITTIWAKE, BY RICHARD JACKSON / BTO

CITATION

Harris, S.J., Baker, H., Balmer, D.E., Bolton, M., Burton, N.H.K., Clarke, J.A.E., Dunn, T.E., Evans, T.J., Hereward, H.F.R., Humphreys, E.M., Langlois Lopez, S., Money, S., Taylor, E.A. & O'Hanlon, N.J. 2026. *Seabird Population Trends and Causes of Change: 1986–2024, the annual report of the Seabird Monitoring Programme.* BTO Research Report **795**. British Trust for Ornithology, Thetford.

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ONLINE RESOURCES

SMP email: smp@bto.org
SMP website: www.bto.org/smp
SMP Report: www.bto.org/smp-publications
SMP*news*: www.bto.org/smp-news
Trends Explorer: http://data.bto.org/trends_explorer
Bluesky account: @smp-seabirds.bsky.social

SMP news and coverage

Data from annual surveying of seabirds in Britain and Ireland, submitted to the Seabird Monitoring Programme (SMP), are summarised here. These data are the result of the collective efforts of both professional surveyors and skilled volunteer observers. Coverage since 1986 has enabled the long-term monitoring of seabird populations across these isles.

OVERVIEW OF 2024

The 2024 survey season is the most recent covered by this report. The weather during spring 2024 was characterised as being warm and wet, with the warmest spring on record experienced in Britain. Both Britain and Ireland also experienced above-average rainfall. The summer was the coldest since 2015 in Britain due to persistent northerly winds, interspersed with brief heatwaves and periods of rainfall. In contrast, Ireland continued to experience above-average temperatures, recording its second warmest summer on record. A notably wet and stormy winter preceding the 2024 breeding season saw areas hit by extreme weather events, namely Storms Babet, Ciarán, Henk and Isha, with strong winds and heavy rain (Met Office 2024, Met Éireann 2024, BBC Weather 2024, The Sun Ireland 2024).

As is often the case, the seabird breeding season in 2024 varied considerably between sites and species. The season appeared to offer some respite from the severe impacts of High Pathogenicity Avian Influenza (HPAI), although notable declines in abundance were observed in certain species, particularly those potentially affected by winter storms, such as Shag and auks. Signs of reduced productivity were reported at Scottish north and east coast colonies, with chick deaths later in the chick-rearing period likely due to food shortages, poor quality prey and difficult foraging conditions. Species such as Kittiwake, Great Skua and Guillemot were among those affected.

Challenging weather also coincided with the hatching period of some species, including Arctic Tern and Kittiwake. A delayed start to the breeding season or asynchronous egg laying was noted for some species, including Guillemot and Arctic Tern, alongside increased observations of loafing, non-breeding individuals at colonies. In western regions of Britain, despite some indications of reduced auk numbers at certain sites, Puffins returned early, Fulmar productivity remained stable, and the abundance of Kittiwake, Puffin, Razorbill and Guillemot increased – with productivity also up on recent years for Razorbill and Guillemot. However, large gull species appeared to fare less well in these areas (Key Site Reports 2024 for Canna, Isle of May, Fair Isle and Skomer Island).

▲ **Figure 1:** Coverage map for 2024, showing all sites where Colony Count (blue) and Breeding Success (yellow) surveys were conducted.

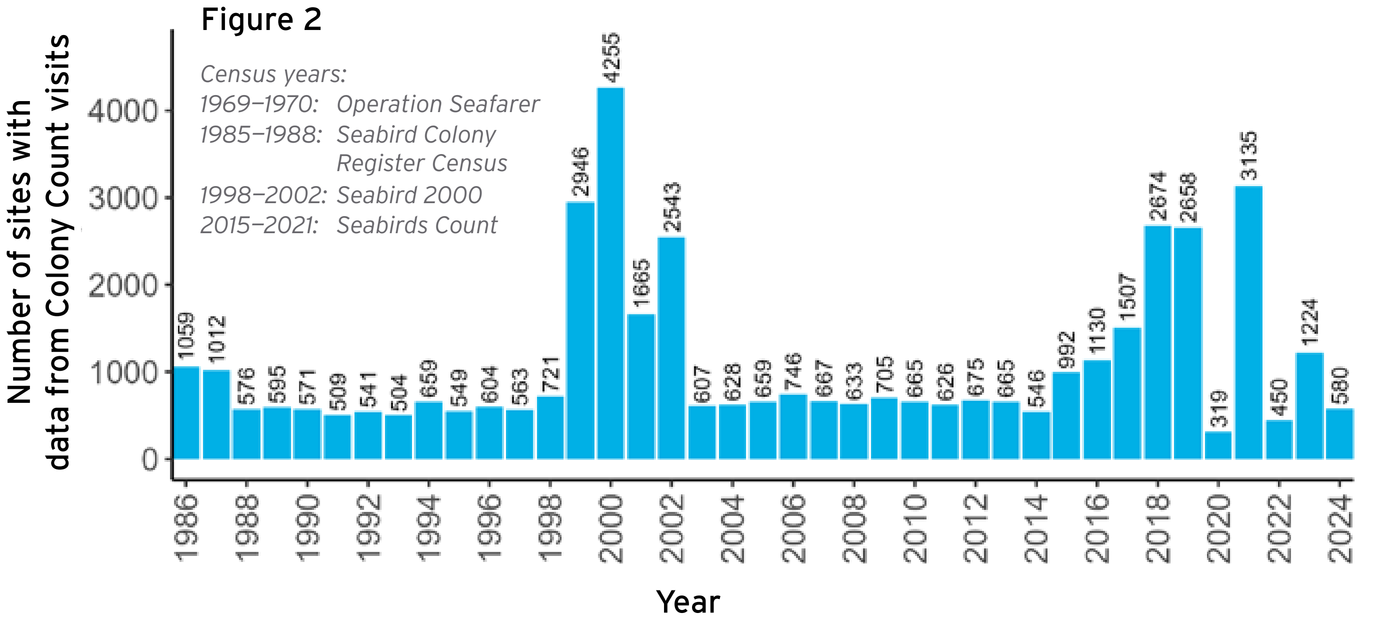
LONG-TERM COVERAGE

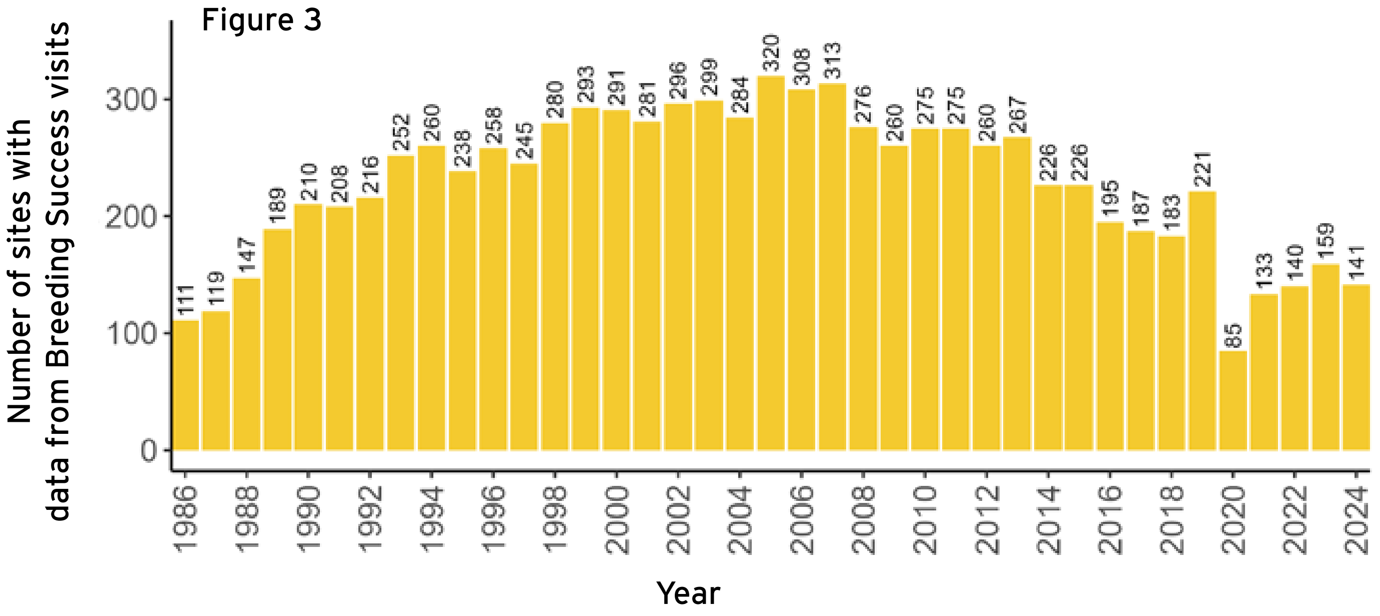
The SMP monitors breeding seabirds throughout the UK, Channel Islands and Isle of Man, and this is supported by the SMP Partnership. Collaboration with BirdWatch Ireland and the National Parks and Wildlife Service enables this report to cover 'Britain and Ireland' (specifically: all of Britain, Ireland, the Isle of Man and the Channel Islands) (Figure 1). Data also feed into the programme from offshore structures, such as oil platforms.

Site coverage for both Colony Counts, used to calculate breeding abundance, and Breeding Success, used to calculate annual productivity values, are illustrated in Table 1. Colony Count coverage shows some large peaks due to additional surveying during census years (listed in Figure 2), and in response to recent HPAI outbreaks. Excluding these years, the long-term coverage is stable for Colony Counts; however, an increase in Breeding Success coverage is needed. A combination of limited funding and the intensive, repeat visits required to measure productivity at sites is thought to be influencing this trend. In future, it is hoped that the situation will improve through a combination of training, increased Plot (sample) monitoring of larger colonies, and the use of new technologies.

▶ **Table 1:** Coverage for 2022–2024. This is the total number of sites where Colony Count or Breeding Success surveys for any seabird species were conducted in each of the years and areas stated.

	Colony Count coverage			Breeding Success coverage		
	2022	2023	**2024**	2022	2023	**2024**
Channel Islands	4	4	**2**	3	2	**2**
England	150	274	**198**	73	79	**66**
Isle of Man	4	4	**3**	1	1	**1**
Northern Ireland	108	75	**84**	4	20	**19**
Scotland	122	791	**222**	45	42	**36**
Wales	42	52	**47**	9	11	**8**
Republic of Ireland	9	12	**24**	5	4	**9**
British and Irish offshore structures	11	12	**0**	0	0	**0**
Britain and Ireland total	**450**	**1,224**	**580**	**140**	**159**	**141**





▲ **Figures 2 and 3:** Site coverage for Britain and Ireland, by year, for Colony Count (blue) and Breeding Success (yellow) monitoring since 1986. Fluctuations in Colony Count coverage have been influenced by census years, Covid-19 and additional HPAI monitoring in 2023.

THE ONLY WAY IS UP

The SMP now has a published sampling strategy to help guide our engagement and monitoring priorities for the programme (O'Hanlon *et al.* 2024). Additionally, work is underway to review the trend analysis and update the *Seabird Monitoring Handbook* (Walsh *et al.* 1995). Improvements to the SMP Online data entry system are being rolled out to the live system as new features become available. These include simplified data entry pages, improved verification to help catch typographical errors or unusual entries, and more detailed mapping functions. The latter has greatly improved the ability to store detailed maps of sites – ensuring repeatability and consistency going forwards, especially if the surveyors for a site change. More improvements are still to come, all with the aim of making data entry and site information easier to navigate and store securely. Spring newsletters and social media (now Bluesky) are also engaging more people with the scheme. Some of these individuals are completely new to seabird surveying; others have been surveying sites for many years and are now entering data retrospectively, as well as for current seasons.

There remains a significant amount of work to improve data storage 'behind the scenes' – ensuring that data are in the correct place and at the correct level (whole colony or a plot within a colony), and that site maps truly represent sites. All this work, alongside historic data entry, may alter coverage figures, but all data will feed into the annual SMP analysis. Additional information and counts are always appreciated – even those dating back to the start of the programme in 1986. The more data there are in the system, the more robust the annual trends will be. The ability to report on specific countries or areas will also be enhanced.

THANK YOU

In June 2024, 263 participants were allocated sites on the SMP Online portal. In June 2025, this figure reached 309. Of course, we know that the true number of surveyors monitoring seabirds and contributing to the SMP database is far greater, with many individuals working in teams and represented in the participant figure by one individual entering data each season. Whether a volunteer or professional surveyor, a lone fieldworker or someone working within a wider team – we are very grateful to you all for monitoring Britain and Ireland's internationally important breeding seabirds.

FIND OUT MORE...

Take Part in the SMP: www.bto.org/smp
Sign up to *SMPnews*: www.bto.org/smp-news
SMP Online: app.bto.org/seabirds/public/index.jsp
Guidance for SMP Online and surveying can be found under the 'Help' tab at the top of the application page.

Seabird population and demographic monitoring in the UK: a review and recommendations for future sampling

Nina J. O'Hanlon, Sarah J. Harris, Chris B. Thaxter, Philipp H. Boersch-Supan, Robert A. Robinson, Dawn E. Balmer and Niall H.K. Burton

Seabird Monitoring Programme

Map Legend

ISSUE 2 | SPRING 2024

SMPnews

Monitoring internationally important seabird populations across the UK

Welcome...

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Tracking bird flu

PARTNERS

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FIELDWORK, BY KATHERINE BOOTH JONES / BTO

BEMPTON SEABIRD COUNTS, BY IZZY FRY

High Pathogenicity Avian Influenza (HPAI): the new normal?

HPAI has had a devastating impact on our seabird populations. Two special issues of *Bird Study* brought together knowledge of the disease and how to respond to it.

By **Phil Atkinson**, Principal Ecologist, BTO

Since the summer of 2021, the impacts of HPAI on the UK's seabirds have been clear to see. Mass mortalities of Great Skuas, Gannets, gulls, terns and auks have been evident across most of the UK's colonies and what was previously a seasonal disease that mostly affected wildfowl in autumn and winter, has turned into a major conservation crisis.

The current HPAI strain that is causing such high mortality emerged in 1996 on a goose farm in China and it suddenly adapted to cross seasons and species to devastating effect. Worryingly, it has also spilled over into both terrestrial and marine mammals.

LIVING WITH BIRD FLU

Despite HPAI still being present both across the UK as well as globally, the emergency response is now over and we have moved into a situation where we are learning to live with lower (but still widespread) levels of disease in wild birds, with occasional flare ups. For example, at the time of writing (July 2025), large numbers of tern and (especially) gull chicks are suddenly being reported dead. Compared with the summers of 2022 and 2023, one bit of good news is that adult seabirds do not seem to have been as impacted, possibly because they are developing levels of immunity which chicks may lack. However, we are still seeing adult mortality this year in the Republic of Ireland and it is possible that adult birds there have less immunity.

BIRD STUDY JOURNAL: HPAI ISSUES

We have marked this move to a 'new normal' with the publication of two HPAI-themed special issues of the journal *Bird Study*, bringing together learnings about the disease and how to respond to it.

These special issues cover wide-ranging aspects of HPAI, including:

- which species may be more or less vulnerable to HPAI;
- what management actions can be taken;
- the impacts on different species groups; and
- how to practically tackle an outbreak when hundreds or thousands of carcasses need to be dealt with and disposed of.

REFINING IMPACT ASSESSMENTS

Using various methods including carcass counts, ringing recoveries, subsequent surveys at seabird breeding colonies, and seawatching data, the evidence base on HPAI's impacts on wild birds has been built up layer by layer. At the start of the outbreak, reports of dead birds highlighted that something major was taking place and then additional studies refined the estimates of its impact.

Over a two year period from October 2021, over 100,000 carcasses were reported, primarily seabirds. The total number of reported carcasses exceeded 1,000 for 11 species, 10 of which were seabirds. In terms of numbers Black-headed Gull and Gannet carcasses exceeded 20,000 showing the scale of the mortality, especially as this must have been a drop in the ocean, when considering not all carcasses will make it to shore. Auks, gulls and terns were also badly affected. A paper in the special issue by Tremlett *et al.* (2024a) from RSPB reported on the surveys of seabird colonies pre- and post-HPAI and these showed the level of impacts. Great Skuas had declined by 77% across the UK, from 6,570 apparently occupied territories to 1,524. Considering that the UK holds c. 60% of the global population, this represents a huge decline. Common Terns were also severely affected, with a 40% decline across the UK over the same period.

WIDE RANGING ISSUES HIGHLIGHTED

The special issues cover a wide range of studies, including one by Coffey & Verspoor on lessons learned from managing an outbreak in real time (Coffey & Verspoor 2025). Removing infected carcasses is suggested to be one possible way to limit the spread of the disease to other birds in the same colony, and the authors highlighted that the lack of an action plan and decision-making delays resulted in carcasses remaining uncollected for 28 days. They stressed the importance of having national action plans in place for future outbreaks to ensure a more timely response.

JOINED UP ACTION

Strong links have been formed between Government and Non-governmental organsiations to share information, formulate a joined up response to future situations, and communicate what people should do when they find dead birds.

WHERE NOW?

Now that the mass mortality of adults is largely in the past, we are looking closely at recovery. Some species, such as Barnacle Goose that underwent a 30% decline have managed to recover by having two good breeding seasons, but the potential for many seabirds to rapidly increase is limited by their breeding biology and also the other pressures they face in the marine environment.

The effects of HPAI on seabird chicks, which are presumed to be more vulnerable than adults, are not yet fully understood. The next few years will undoubtedly see more mortality but we hope to see some recovery in the Britain and Ireland's breeding seabird populations.


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The Science of Pure and Applied Ornithology
The impacts of Highly Pathogenic Avian Influenza on birds: part 1
BTO Birds Science People
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Taylor & Francis Group

◀ ***Bird Study*, Volume 71, Issue 4 (2024):** The impacts of Highly Pathogenic Avian Influenza on birds: part 1

Part 1 discusses topics including HPAI jumping species and seasons (Atkinson & Baillie 2024); declines in UK breeding populations of seabird species of conservation concern (Tremlett *et al.* 2024a); and the impacts of HPAI on seabird populations in the North Sea – detected by sea-watchers' migration counts (Macgregor, *et al.* 2024).

From further afield, mortality among scarce breeding gulls and terns in Poland (Przymencki *et al.* 2024), and closer to home, a case study of the 2023 HPAI outbreak in tern colonies on the east coast of the Republic of Ireland (Burke *et al.* 2024).

Some welcome, yet curious news, with the near-absence of HPAI in Little Terns across 13 European countries (Norman *et al.* 2024) and less welcome epidemics in Baltic Cormorant colonies in 2021 and 2022 (Bregnballe *et al.* 2024).

This issue goes on to report on lower mortalities of seabird and seaduck during the second breeding season of the HPAI H5Nx viruses in eastern Canada (Cormier *et al.* 2024) and a case study from Bird Island in South Georgia of an H5N1 outbreak being the first for the subantarctic region (Bennison *et al.* 2024).

The final two items in the issue document the impacts of HPAI on the wintering Barnacle Goose population of the Solway Firth, UK (Griffin & Peach 2024), and, lastly, the spatial and temporal variation in mortality from avian influenza in Greenland Barnacle Geese in their wintering grounds (Percival *et al.* 2024).

BIRD STUDY COVER IMAGES: GREAT SKUA, BY EDMUND FELLOWES / BTO & BLACK-HEAED GULL, BY DAWN BALMER / BTO


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Bird Study
The Science of Pure and Applied Ornithology
The impacts of Highly Pathogenic Avian Influenza on birds: part 2
BTO Birds Science People
Taylor & Francis
Taylor & Francis Group

◀ ***Bird Study*, Volume 72, Issue 1 (2025):** The impacts of Highly Pathogenic Avian Influenza on birds: part 2

Part 2 of these Special Issues discussed responding to HPAI and where we go from here (Atkinson & Baillie 2025), looked at the vulnerability of wild bird populations to HPAI (Pearce-Higgins *et al.* 2025), and evaluated the use of carcass analysis and testing data to assess HPAI-related mortality in wild birds in the UK and Crown Dependencies (Atkinson *et al.* 2025). The issue also provided insights into the use of bird ringing recovery data to examine the impact of HPAI on wild bird populations (Johnston *et al.* 2025a).

Raptors were a focus with lower breeding success for White-tailed and Golden Eagles noted in Scotland in 2022 correlating with presence of HPAI (Wilson *et al.* 2025); a study in Buzzard population decline during the HPAI outbreak (Swann & Beckmann 2025); and the potential effects of HPAI on occupancy rates, breeding success, age and turnover of breeding Peregrines (Smith *et al.* 2025).

An article looking at the impact of conservation interventions (Pearce-Higgins & Pollock 2025), a case study on the impact of HPAI on a seabird colony in Flintshire, Wales (Coffey & Verspoor 2025), the impact of HPAI on an Italian network of Wildlife Rescue Centres (Pacini *et al.* 2025), and the impact of HPAI on seabird populations in West Africa (Jatta *et al.* 2025) were all covered in the second part of the HPAI species issues of *Bird Study*.

FIND OUT MORE ...
Bird Study journal:
www.bto.org/bird-study

An integrated approach

The SMP plays a critical role in tracking abundance and productivity changes in seabird populations across Britain and Ireland. However, to achieve a full picture of the causes of population change, an integrated monitoring approach is required where SMP data are combined with other data sources, particularly on seabird ecology away from their breeding sites.

By **Sarah Harris**, SMP Organiser, BTO

Seabirds are crucial indicators of ecosystem health, in both terrestrial and marine environments, and are among the most threatened groups of birds. Approximately 30% of seabird species are classified as globally threatened (BirdLife International 2022). Understanding changes in seabird populations, the stressors they face throughout their life cycles, and the threats impacting them is essential for guiding future research and conservation efforts.

Seabird populations are increasingly vulnerable to threats such as climate change, fisheries, disease, invasive predators, and habitat disturbance (Dias *et al.* 2019). Effective monitoring is vital for tracking population changes, habitat use, and threats, thereby informing conservation strategies and policy measures. Recent publications have highlighted advancements in monitoring methodologies through new technologies (Frederiksen *et al.* 2025), while also revealing gaps in ecological data and knowledge. Addressing these gaps is key to optimising conservation efforts – from guiding funding and research priorities to improving monitoring of key biological metrics and identifying under-represented regions.

CURRENT MONITORING

Traditional seabird monitoring has primarily relied on breeding season, field-based surveys, including assessments of breeding productivity and abundance counts through the SMP and periodic censuses (Burnell *et al.* 2023, Harris *et al.* 2024). Although these surveys provide invaluable long-term data, their scope is limited by logistical challenges, resource constraints, the difficulties involved in accurately surveying cryptic species – such as burrow nesters or those coming ashore only at night – and the inaccessibility of many breeding sites. As a result, notable gaps exist in the annual statistics generated by the SMP for some species. Furthermore, understanding seabird movements, threats and population changes, both away from colonies and outside the breeding season – particularly during migration – remains a major barrier to achieving comprehensive knowledge (O'Hanlon *et al.* 2024a, Macgregor *et al.* 2024, Johnston *et al.* 2025b). Although this is not specifically covered by SMP, it is important to note how crucial these times are during the seabird life-cycle.

To overcome these limitations, it is essential to identify gaps and direct funding and monitoring efforts accordingly. Integrating innovative technologies alongside traditional methods and establishing collaborative frameworks will also be crucial to enhancing both spatial and temporal coverage.

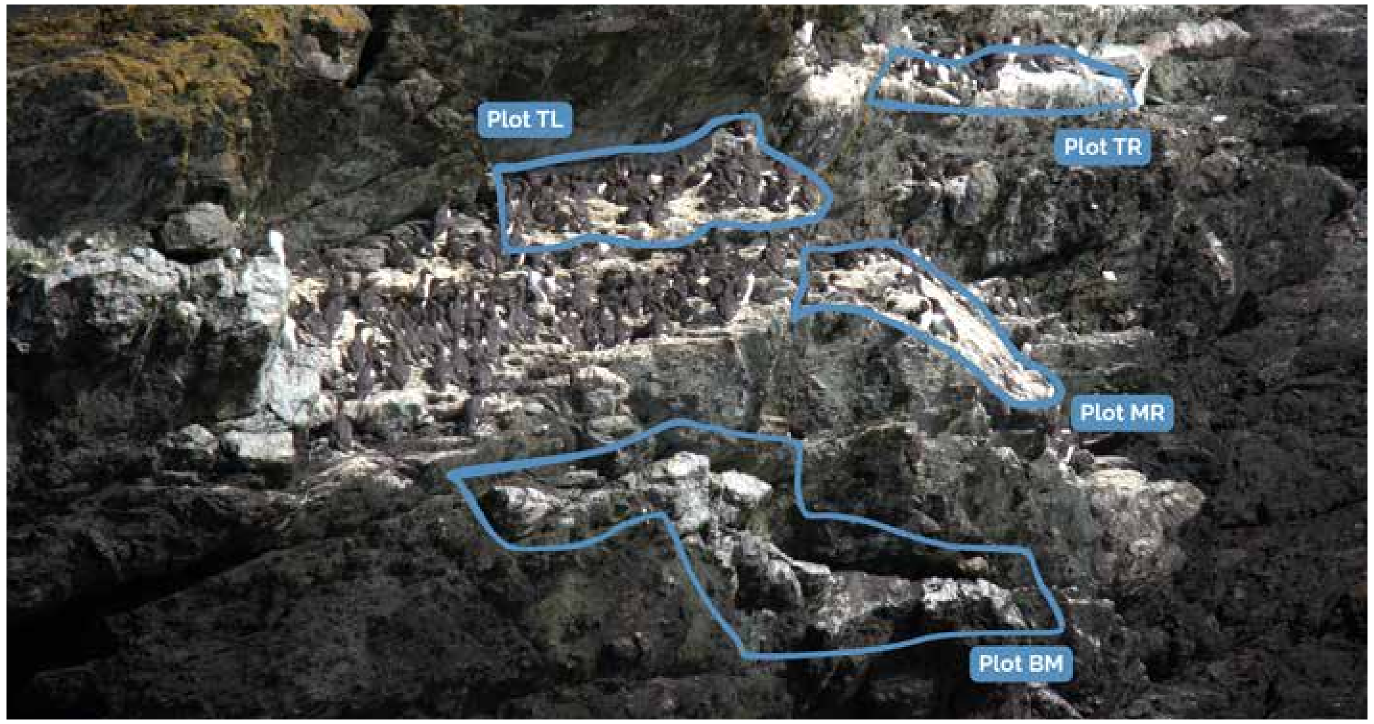
SAMPLING STRATEGY

In 2024, a report titled *Seabird Population and Demographic Monitoring in the UK: A Review and Recommendations for Future Sampling* was published (O'Hanlon *et al.* 2024b), which reviewed the sampling strategy of SMP. It assessed the methods used to produce current trends in abundance and productivity, alongside additional metrics such as diet, survival, and phenology, which were collected at four SMP Key Sites (see page 18). The report highlighted that some trends are imprecise, absent, or geographically limited for several UK breeding seabird species.

The report identified three key areas that, without robust demographic data, would be difficult to understand, underlining the importance of collecting high quality SMP data:

- Identifying key stressors acting on populations and their mechanisms
- Detecting population impacts of known stressors
- Identifying and implementing necessary conservation and management actions

The review recommended steps to improve both the quantity and quality of data collected through the SMP. These include increasing regular monitoring at more sites – particularly in underrepresented countries, regions, and inland areas – and covering a wider range of species and colony sizes. Using multiple sample plots where full Colony Counts are unfeasible was also suggested. Coordination with the BTO Nest Record Scheme should be encouraged so that overall abundance and productivity counts from this scheme are also submitted to the SMP. Additionally, the report recommended that the current method used for trend analysis of abundance and productivity should be revised, alongside the inclusion of informative confidence intervals, to ensure that the SMP trends are robust and properly reflect uncertainty.



▲ Sample plots within a seabird colony – one of the SMP Sampling Strategy's recommendations is to expand monitoring coverage by using multiple sample plots instead of conducting full Colony Counts

GUILLEMOT COLONY, BY KATE FOX

A further recommendation was to encourage and prioritise abundance monitoring at sites where other demographic data – such as survival rate, diet, and phenology – are collected, thereby improving integrated monitoring. This includes creating additional locations with Key Site-style monitoring to expand data submission of survival, phenology and diet metrics. Such data collection opens up opportunities for comprehensive, integrated reporting.

Effective volunteer engagement and training will also be essential in increasing coverage and maintaining data quality. An SMP Engagement Plan is being produced based on the sampling strategy's recommendations, aiming to empower more volunteers to sample more sites across Britain and Ireland through initiatives such as mentoring.

ADVANCING MONITORING METHODS

Visiting seabird colonies during the breeding season to monitor breeding success and productivity is one of the most well-established components of annual seabird monitoring. However, alongside improvements suggested in the sampling strategy review to improve coverage and trend analysis methodologies, a new wave of technologies also has the potential to transform seabird monitoring by revolutionising monitoring methods, as described on Pages 14–17. These include autonomous devices deployed in colonies, using Uncrewed Aerial Vehicles (UAVs), thermal imagers, and the use of satellite imagery – allowing access to remote or hard-to-view areas, and therefore increasing coverage. Such technologies are often less labour-intensive, can reduce disturbance, and enhance monitoring efficiency (Frederiksen *et al.* 2025).

While these technological tools offer significant advantages, their effective deployment depends on establishing standardised protocols, data management systems, and training for volunteers and researchers (see pages 14–17).

INTEGRATED MONITORING

Integrating a more comprehensive, annual monitoring programme with additional ecological metrics beyond abundance and productivity will significantly enhance our understanding of seabird life cycles and the threats these species face.

As highlighted in the Sampling Strategy review, phenological data can reveal changes over time, such as the impacts of climate change on seabirds – including shifts in prey availability, and the timing of breeding and migration (Descamps *et al.* 2019). Diet studies can also provide clues about stressors faced during breeding,

TRAINING, BY DAVID TIPLING / BTO

▲ Engagement and training in both traditional monitoring methods and new technology-based methods will be key to expanding coverage to more sites, especially when aiming to increase Breeding Success data collection and data from inland sites.

potentially explaining low productivity in certain years (Lewis *et al.* 2001). Key studies on the survival rates of seabirds under schemes such as the Retrapping Adults for Survival programme can also provide valuable information about the life stages at which individuals are being lost from the population (Frederiksen *et al.* 2008).

During and outside the breeding season, tracking devices enable insights into individual birds across vast oceanic expanses, revealing migratory routes, key foraging areas, non-breeding distributions, and habitat use (Dunn *et al.* 2019, Merkel *et al.* 2021). Such data are vital for identifying critical habitats during both breeding and non-breeding seasons and understanding threats encountered throughout the annual cycle, especially when linked to demographic data such as abundance and productivity (O'Hanlon *et al.* 2024a).

Other major UK monitoring schemes – such as the Heronries Census (which also covers Cormorants), the BTO/JNCC/RSPB Breeding Bird Survey (colonial nesters), the BTO/JNCC/RSPB Waterways Breeding Bird Survey (colonial nesters), and the BTO/JNCC Nest Record Scheme (individual nests for any nesting species in the UK) – collect valuable data that could be leveraged more effectively for seabird monitoring. A more structured approach to seawatching along our coastlines, throughout the year, and at-sea surveys from vessels, such as the JNCC Volunteer Seabirds at Sea programme, could provide insights outside the breeding season and away from colonies. This would help track population distributions post-fledging and pre-breeding, filling in critical gaps in our knowledge (Macgregor *et al.* 2024).

CONNECTING VOLUNTEERS

Enhancing data sharing among national schemes, standardising survey methods, and fostering collaboration between biological surveys will strengthen collective monitoring efforts. This approach can enable multi-scale research and analysis – from fine-scale to global – supporting evidence-based policy-making. Prioritising monitoring in under-studied regions and for lesser-known species will help address ecological knowledge gaps and guide targeted conservation actions.

Volunteers form the backbone of most long-term monitoring schemes in Britain and Ireland, contributing essential data and practical expertise. Providing training for surveyors in survey methods and in emerging technologies, ensuring adherence to standardised protocols, and broadening engagement are vital for maintaining data quality, increasing coverage, promoting knowledge sharing, and motivating volunteers. It is crucial that policy-makers continue supporting both new and existing initiatives through resource allocation, joint working, and integrating monitoring data into marine spatial planning and protected area designation. Protecting key habitats identified through integrated seabird monitoring is essential for ensuring the resilience of seabird populations.

CONCLUSION

Advancing and integrating seabird monitoring is crucial for protecting these vital indicators of ecosystem health amid increasing environmental threats, and the SMP has a vital role to play in this. While traditional methods have laid a strong foundation, adopting new technologies, expanding geographic and seasonal coverage, and fostering collaborations will deepen our understanding of seabird populations across their life cycle. An integrated approach – combining demographic, behavioural, and movement data – will better identify stressors, conserve critical habitats, and inform targeted management actions.

Realising these goals will require coordinated efforts among researchers, policy-makers, and volunteers. Continued investment in training, standardisation, and data-sharing is essential to maintaining high-quality, comprehensive datasets. Through collaboration and innovation, we can enhance seabird conservation and safeguard ecosystem health for future generations.

FIND OUT MORE ...

Sampling Strategy: www.bto.org/our-work/science/publications/reports/bto-research-reports/00754-seabird-population-and-demographic-monitoring-uk-review

DIFFICULT SURVEY SITE/SPECIES, BY KATHERINE BOOTH JONES / BTO

New technologies and the SMP

How can we ensure SMP keeps up with, and harnesses, the rapidly evolving world of technology to enhance seabird monitoring?

By **Sarah Harris**, SMP Organiser, BTO

Following evidence of seabird declines early in the 20th century, attention shifted towards their conservation and research. Biologists began conducting more studies on this relatively accessible group of birds. Coordinated counts of individual species across different sites, followed by comprehensive seabird censuses covering all species and locations in Britain and Ireland, started to increase in frequency. Technological advances have followed but remain to be implemented on a wider scale.

SEABIRD MONITORING COORDINATION

From the mid-20th century, it became clear that Britain and Ireland hosted internationally significant populations of breeding seabirds, necessitating a coordinated effort to monitor these species annually (Tasker 2000, Burnell *et al.* 2023, Stanbury *et al.* 2024). In 1986, the SMP was established as the first large-scale, structured, and standardised seabird monitoring initiative of its kind (Tasker 2000). Since then, the SMP has inspired similar programmes worldwide, from the Falkland Islands Seabird Monitoring Programme (falklandsconservation.com/seabird-monitoring) to Norway's Seabird Population and Productivity (SEAPOP) scheme (www.seapop.no/en/). However, the methodologies adopted by the programme that kick-started it all, the SMP, are now nearly 40 years old (Walsh *et al.* 1995).

Currently, these methods mainly rely on visual observations – either from boats, land-based vantage points or within colonies, such as using walk-through surveys or by physically checking burrows. The only electronic tool described in the *Seabird Monitoring Handbook for Britain and Ireland* (Walsh *et al.* 1995) is tape-playback, used to elicit call responses from birds nesting in burrows and crevices, such as Manx Shearwaters and petrels.

This highlights that, although technology has advanced significantly, it still needs to be integrated into the fundamental foundations of seabird monitoring.

LIMITATIONS OF CURRENT MONITORING

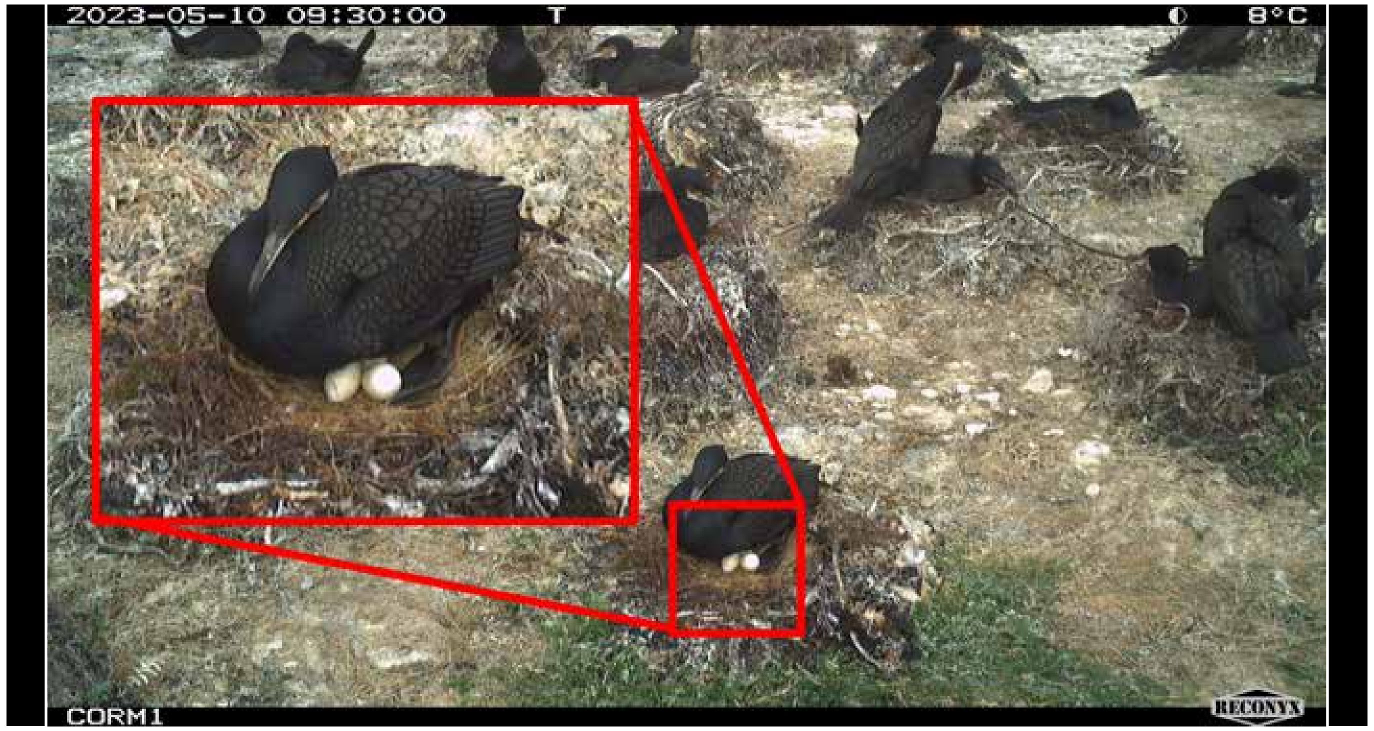
To date, regular SMP coverage has been limited to certain locations. Many sites are remote, expensive to access multiple times in a season, or simply difficult to observe. Consequently, for some species, regions, or countries, annual monitoring is restricted (Harris *et al.* 2024). Training opportunities are also dependent on funding and capacity – both often constrained at remote or hard-to-reach sites (physically or financially). Relying on professional fieldworkers or skilled and dedicated volunteers with limited availability or financial backing can lead to high turnover, resulting in inconsistencies (Schmidt *et al.* 2023).

Monitoring burrow-nesting species is challenging; they nest out of sight, and for storm petrels, they return to their burrows under cover of darkness. Intensive surveys involving checking burrows and crevices for species presence and nest activity are labour-intensive and difficult to sustain over time (Burnell *et al.* 2023). They also almost exclusively breed on islands, presenting additional challenges.

Finally, there has been a decline in the collection of breeding success data – used to calculate productivity – because collecting such data requires multiple visits throughout the season, making it more demanding than the relatively less time-consuming Colony Counts (Harris *et al.* 2024). Reduced funding and engagement are thought to be key factors behind this decline.

TECHNOLOGICAL ADVANCEMENTS

The use of new technologies is already underway, and they have the potential to be used in a wide range of seabird studies, such as monitoring abundance, productivity, survival, phenology, levels of disease and predation. Being able to assess metrics beyond abundance and productivity allows for integrated monitoring, a powerful tool in identifying drivers of change, such as disease.



▲ Time-lapse photography can allow productivity and phenology recording at Cormorant colonies with reductions in disturbance and financial cost compared to monitoring by human observers. This image is from a project led by BTO to examine which seabird metrics can be gathered using camera imagery at Strangford Lough, Northern Ireland.

BIRD ISLAND CORMORANT COLONY THROUGH A TRAIL CAMERA, BY DANIEL JOHNSTON / BTO

SMP-relevant field-based tools (that can also record other metrics) include UAVs or drones for abundance counts of ground-nesting seabirds (Rush *et al.* 2018, Edney *et al.* 2023); fixed cameras for productivity, predation, phenology and survival monitoring (Merkel *et al.* 2016, Johnston *et al.* 2020, Pascalis *et al.* 2018, Hentati-Sundberg *et al.* 2023, Edney *et al.* 2024); thermal imaging for abundance, disease, distribution, and predation monitoring (Black 2018, Hentati-Sundberg *et al.* 2025); and endoscope cameras for checking burrows to record abundance, productivity and phenology (Rexer-Huber *et al.* 2013, Lavers *et al.* 2019).

Space-based technologies, such as satellite imagery (Hughes *et al.* 2011, Fretwell *et al.* 2012, 2017, Fretwell & Trathan 2021), can be used to gather abundance estimates and record distribution. Finally, automated audio recorders can be placed within colonies to study relative abundance, productivity, phenology and predator presence (Buxton *et al.* 2013, Borker *et al.* 2014, Oppel *et al.* 2014).

Bird ringing, including Darvic (plastic uniquely-coded rings) and colour ringing (a unique combination of coloured rings) programmes can also be used alongside new technologies, such as cameras, to provide information on survivial, longevity, site fidelity, dispersal and genetic relationships (Anderson & Green 2009), and some ringing is used in assessments for productivity monitoring.

Tracking devices – from geolocators and radio telemetry to GPS – are now a well-established component of seabird monitoring (Bernard *et al.* 2021). Although tracking does not directly quantify abundance or productivity information for the SMP, it does provide critical insights into seabird behaviour and factors influencing population trends – additional metrics that are already required for integrated monitoring. Tracking can also serve as ground-truthing for other monitoring methods – for instance, identifying routes taken by birds entering and leaving colonies, which can inform passive acoustic monitoring for density and distribution estimates (Arneill *et al.* 2020).

WHY EVOLUTION IN MONITORING IS IMPORTANT

Technological advancements have the potential to improve monitoring of difficult-to-access species and sites, enhance efficiency, reduce disturbance, and increase data precision and comparability (Rush *et al.* 2018, Edney & Wood 2021, Brusa et *al.* 2024). Once deployed, equipment can often remain on-site for extended periods, reducing costs associated with repeated human visits.

Enhanced monitoring capabilities, implemented correctly, can lead to more robust results through increased coverage and data quality. This can then improve our understanding of the overall decline in seabird populations – both in the UK (a 37% decline from 1986 to 2024, Defra 2025) and globally (a 69.7% decline from 1950 to 2010, Paleczny *et al.* 2015). The ultimate goal is to direct funding and research efforts where they are most needed, and inform policy development in order to identify and counteract drivers of decline.

The main causes of population change among seabirds include predation by non-native species, disturbance, fishing activities, renewable energy infrastructure, disease, and climate change – the latter affecting food availability, and increasing the occurrence of extreme weather events (Dias *et al.* 2019, Burnell *et al.* 2023, Harris *et al.* 2024, Tremlett *et al.* 2024a). Increased monitoring effort is therefore critical to understanding the impacts of these pressures on seabird populations.

More comprehensive data would also allow for better regional assessments and trend analyses. Currently, monitoring within Special Protected Areas (SPAs) often lacks the baseline data needed for robust impact assessments – for example, for offshore wind farm developments (Cook *et al.* 2019). Additional data, facilitated by new technologies, can help bridge these gaps.

RISKS ASSOCIATED WITH NEW TECHNOLOGIES

However, there are risks to consider. Data storage, management, processing, and computational development must be carefully planned, both from a technical and financial perspective. Data input systems will also need to adapt to accommodate new data types – such as images, audio, and raw data files. Therefore, data processing systems will also need to be considered in order to prevent a data analysis bottleneck, for example, the use of computer models to analyse UAV imagery and aerial photographs (Descamps *et al.* 2011, Williams & DeLeon 2020, Hayes *et al.* 2021, Tuia *et al.* 2022). Deploying new equipment requires skills that many surveyors may not currently possess, therefore, training will be necessary. Furthermore, reduced human interaction with seabirds – if overlapping with traditionally surveyed sites – could lead to waning interest among surveyors.

Other risks include potential alienation of existing SMP participants, disruption of long-term datasets, and inconsistencies in methods caused by changes in technology. Initial investment in expensive equipment and data storage can be substantial (Hentati-Sundberg *et al.* 2025), and uneven funding might lead to regional disparities in coverage. To ensure comparability across sites, standardised protocols for data collection, storage, and processing should be developed.

IN SUMMARY

Careful planning and budgeting – both medium- and long-term – are essential to mitigate risks. Resources should prioritise filling current spatial and species-specific gaps, with infrastructure for data processing and storage established from the outset.

Before integrating new technologies into long-term monitoring, rigorous testing and standardisation should be carried out alongside traditional methods (DeLeon *et al.* 2023, Dunn *et al.* 2021, Hughes *et al.* 2011, Rexer-Huber *et al.* 2013, Rodway *et al.* 2024). This will involve operating traditional visual methods and new technological methods concurrently, at least at some sites, to ensure data comparability across long-term datasets is maintained (Freeman *et al.* 2007). Additionally, new technological methods should be adopted consistently across national and international programmes, which will require collaboration and standardised protocols.

Maintaining open communication and ongoing training for current fieldworkers (volunteers and professionals) is crucial to ensure data quality, continuity, and motivation of those supporting seabird monitoring.

NEXT STEPS

An assessment of which new technologies are suitable for certain species and sites will need to be carried out, along with conversations with species and technology experts to ascertain what will be included in an updated *Seabird Monitoring Handbook for Britain and Ireland* (Walsh *et al.* 1995). The outcome of any handbook updates will then need to be reflected in the SMP Online methods option list. Any additional information needed for the new methods will need to be added to the system too, for example, image quality for drone images used to count a ground nesting seabird colony, or the variation in the counts made from the images, etc.

As already mentioned, new technologies such as UAVs are currently being used within the SMP. However, the existing data entry system lacks an option to record 'UAV' or 'drone' as a method; instead, 'drone' is entered in the comments section against data so that methods can be updated retrospectively as the system develops. To date, we have secured funding through a BTO appeal for a scoping exercise to identify what is required in order to update the handbook and this is due to commence in 2025. However, in order to produce the actual updated handbook following on from the scoping project, there is a need to source funding, as well as to develop the SMP Online data entry to reflect the necessary changes and additions.

FIND OUT MORE ...

Frederiksen *et al.* 2025. Opportunities and challenges for new technologies in seabird population monitoring. *ICES Journal of Marine Science* **82**: fsaf115. Page 17 contains a summary of Table 1 from this paper, acknowledging the strengths and weaknesses of proposed technological advancements in seabird monitoring.

SUMMARY

UNCREWED AERIAL VEHICLES (UAV)

SMP-specific relevance: Colony Counts and Breeding Success.

Benefits: Can improve coverage in difficult to view or access areas.

Challenges: CAAP regulations; post-fieldwork processing and data storage; weather conditions – wind, rain, harsh light – impacting flight ability and image quality; charging batteries in the field; in-flight failures; initial high cost and ongoing data storage cost; availability of trained pilots; lack of method guidance/species tolerances.

FIXED CAMERAS

SMP-specific relevance: Breeding Success.

Benefits: Time-lapse/motion sensors minimise number of visits (financial benefit), and improve coverage in difficult-to-access areas, reducing disturbance by observers.

Challenges: Power supply in remote locations; lack of best practice guidelines covering set-up, sampling frequency, avoiding biases in site selection; data storage is limited; lack of traditional volunteer involvement and their connection to seabirds.

SATELLITE IMAGERY

SMP-specific relevance: Colony Counts.

Benefits: Can improve coverage and mapping of ground-nesting species in the open in remote or difficult-to-reach areas, from space on a large scale.

Challenges: Disconnects traditional volunteers from fieldwork; cost of high resolution imagery; requires clear skies; not currently suitable for Britain and Ireland due to resolution issues; post-processing and storage costs; lack of guidance.

BIOACOUSTIC MONITORING

SMP-specific relevance: (relative measures of) Colony Counts / Breeding Success.

Benefits: Can use sound to identify colony density change and identify calls from fledged birds where traditional monitoring is difficult, e.g. burrow-nesters.

Challenges: Power supply in remote locations; lack of best practice guidelines; environmental conditions impacting sound detection; post-processing time, development of classifiers and data storage; lacks traditional volunteer involvement.

THERMAL IMAGERY

SMP-specific relevance: Colony Counts.

Benefits: Handheld, fixed or UAV-mounted useful for cryptic (e.g. nocturnal) species or those in dense vegetation/rough terrain; night and bad weather monitoring; reduced background 'noise' for automated computerised detection processing.

Challenges: Generally coarse resolution impacting processing and potentially increased disturbance; post-processing and data storage; lack of guidelines.

BIRD RINGING (USED ALONGSIDE NEW TECHNOLOGIES)

SMP-specific relevance: Breeding Success.

Benefits: Can be deployed to estimate productivity within a colony otherwise hard to observe, e.g. gulls and terns, where walk-through monitoring takes place. Post-breeding benefits to examine survival, longevity, site fidelity, distribution and genetic relationships; benefits integrated monitoring.

Challenges: requires skilled ringers; repeat visits; cost of rings; weather dependent.

TRACKING

SMP-specific relevance: indirect monitoring.

Benefits: Provides opportunities for integrated monitoring alongside SMP data and for ground-truthing alternative monitoring methods, e.g. bioacoustics.

Challenges: Tagging licence required; post-fieldwork processing and data storage; weather conditions; high levels of device malfunctions, small sample sizes, cost and tag effects; lost/irretrievable tags.

Note that this summary covers new technologies and bird ringing *in relation to SMP*. Diet, disease, distribution, phenology, predation and survival, can also be monitored by some of these. Adapted from Frederiksen *et al.* 2025.

Key Site monitoring

A brief overview from SMP Key Sites Canna, Fair Isle, Isle of May and Skomer Island for the 2024 seabird breeding season.

By **Bob Swann** (Canna), **Francis Daunt** (Isle of May), **Leighton Newman** (Skomer Island) and **Alex Penn** (Fair Isle)

Four geographically dispersed seabird sites around the UK collect additional seabird data to complement core SMP monitoring. These data provide further insights into how and why seabird populations change. Information on abundance, productivity, phenology, survival and diet for the species each site monitors will be viewable in annual Key Site reports. Below, is a summary of the 2024 breeding season at these sites.

CANNA

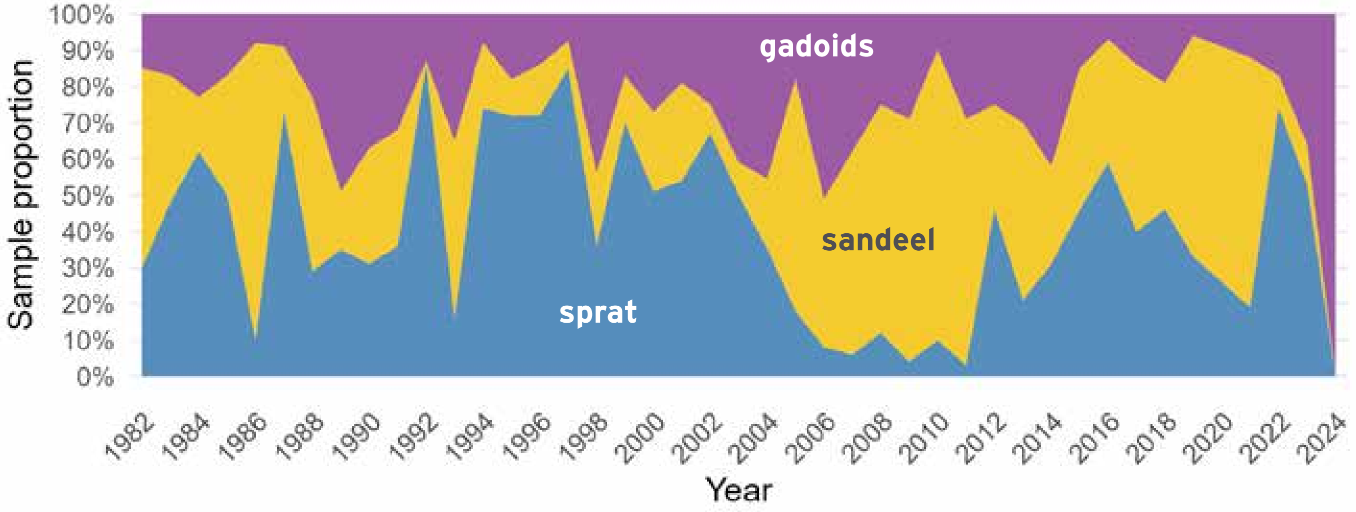
When compared to recent monitoring data, the 2024 breeding season on Canna was arguably the most challenging since record-keeping began in 1969. Several key seabird species, including Herring Gull, Lesser Black-backed Gull, Guillemot, Fulmar, Shag and Great Skua, experienced their lowest counts on record or near-record lows. Great Skua, in particular, declined to its lowest population level since colonising the island in the early 2000s. Although Kittiwake numbers declined from recent peaks, their overall population remains relatively high.

Productivity across many species was significantly below long-term averages; Fulmar suffered total breeding failure, and Shag nearly failed altogether. Similarly, Kittiwake productivity was well below historic norms (Figure 6), while only Herring Gull, Great Black-backed Gull, and Great Skua maintained productivity levels close to long-term averages. Reduced prey abundance was a key factor behind these declines, as evidenced by Guillemots only bringing in low-quality gadoids in 2024, resulting in underweight chicks (Figure 4).

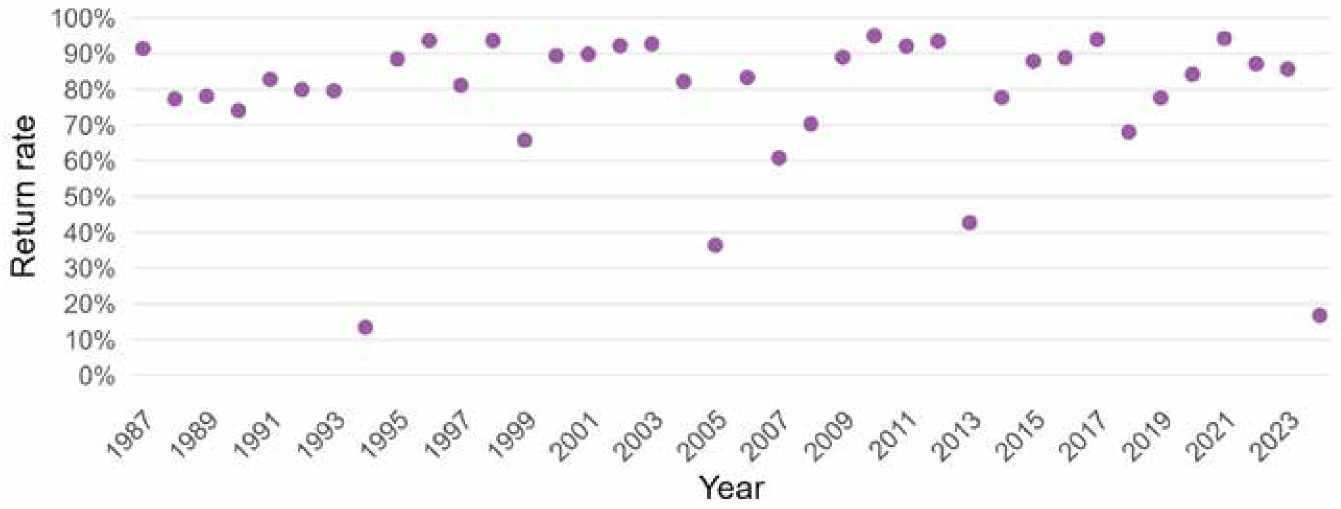
Fortunately, there were no recorded outbreaks of High Pathogenicity Avian Influenza (HPAI) on Canna in 2024, offering some reassurance amid the widespread breeding challenges.

ISLE OF MAY

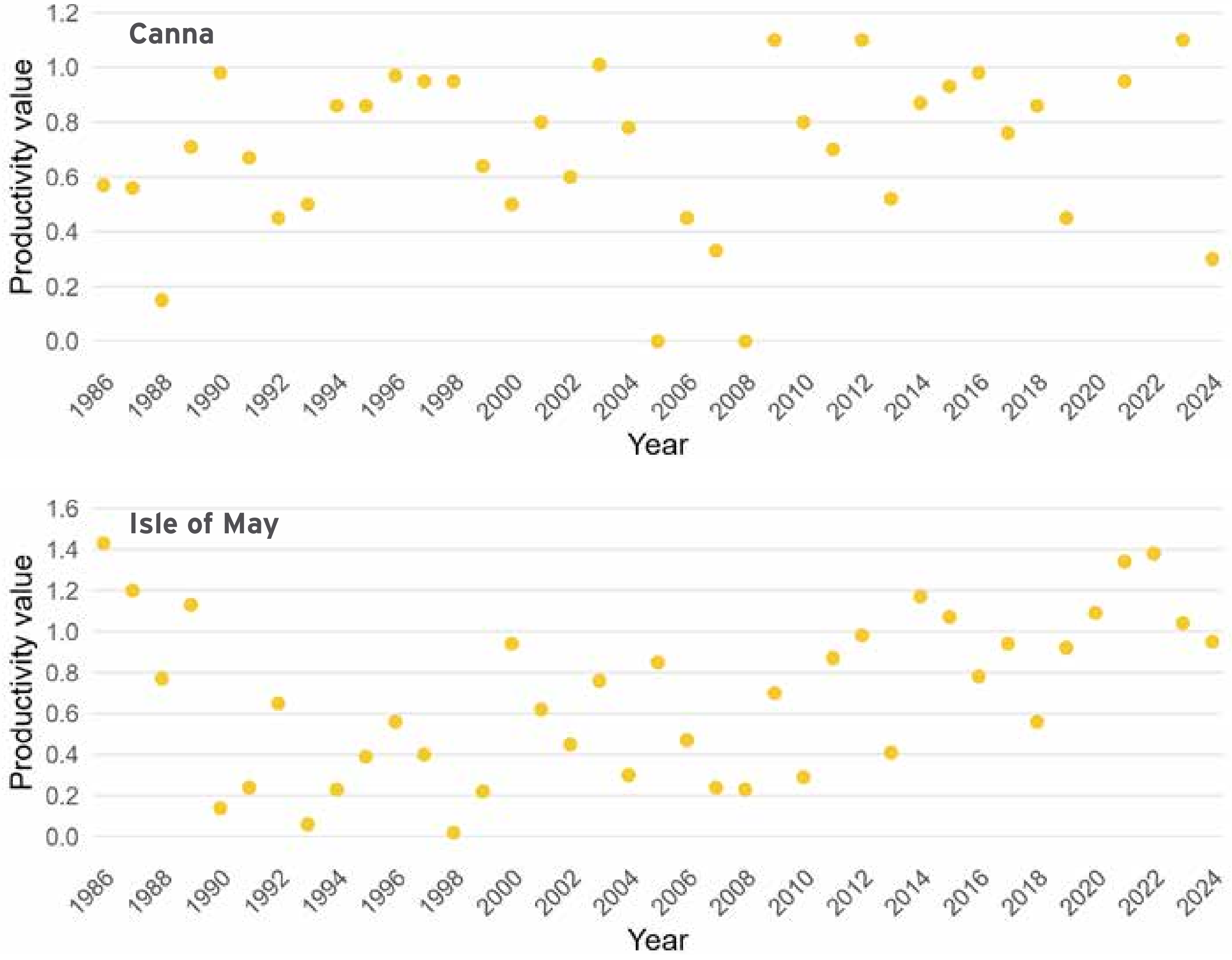
Breeding on the Isle of May in 2024 was late for most species but particularly Shag and Guillemot. Reduced numbers and colony attendance showed that Shag, Guillemot and Razorbill had all been severely affected during the previous non-breeding season. The return rate of individually marked birds was the lowest on record for Guillemot and Razorbill, while it has only been lower in one previous year for Shag (Figure 5). Puffin also returned at lower than average levels, with only Kittiwake experiencing an above average return rate. Productivity is often poor following a winter of low survival (Harris *et al.* 2005), but it was average for Razorbill and Puffin in 2024. Although Guillemot productivity was below average, it was above the level observed following the poor winters in the mid-2000s. Productivity was above average for Shag and Kittiwake (Figure 7); however, for the latter there was high chick mortality recorded at the fledging stage. Sandeels were the main food of Shag, Razorbill and Puffin. Chick diet of Guillemot and Kittiwake was dominated by clupeids, the latter constituting the lowest proportion of sandeel in the diet on record.



▲ **Figure 4:** The proportion of different fish brought back for Guillemot chicks on Canna from 1982 to 2024. Diet is one of several additional metrics collected at Key Sites, allowing integrated monitoring.



▲ **Figure 5:** The return rate for Shag on the Isle of May from 1987 to 2024. Low return rates suggest low survival rates in 2023–2024, a pattern mirrored by Razorbill and Guillemot.



▲ **Figure 6 and 7:** Kittiwake productivity (chicks/pair) on Canna (above) versus the Isle of May (below).

FAIR ISLE

In 2024, the seabirds on Fair Isle faced the worst season for several years, with a number of factors thought to be driving reduced population totals and productivity declines. Many species were slow to return to breeding areas, with a difficult late winter period for Guillemot and Razorbill seemingly affecting the proportion of adults returning in sufficient condition to breed.

Arctic Tern and Kittiwake were both late to settle, with knock-on effects for Arctic Skua, which relies heavily on these two species for food. Just nine pairs of Arctic Skua occupied territories, with only three laying eggs – the lowest total in recorded history. Fulmar, Shag and Black Guillemot numbers remained stable, whilst Great Skua and Gannet bucked the trend, returning in better numbers than 2023, an encouraging sign after previous heavy losses to HPAI in these species (Figure 8).

Productivity however was poor across the board, with several species (Arctic Skua, Guillemot and Shag) returning figures of zero in their monitored plots. A prolonged period of cool and damp weather in early June was lethal for many hatching and newly-hatched young, and this coupled with an apparent lack of food available for some species contributed to this year's poor results.

SKOMER ISLAND

The seabird populations on Skomer in 2024 had varied outcomes, with some species increasing while others showed signs of decline. With whole-colony abundance monitoring, positive news included Razorbill counts totaling 10,639 individuals (+6% on 2023). Guillemot counts yielded 29,358 birds (+1% on 2023) and Kittiwake counts resulted in 1,572 apparently occupied nests (AON), a 22% increase on 2023 and 6% above the five-year mean. Great Black-backed Gulls rose to 122 AON (+1% on 2023, Figure 9) and Shags rose to 11 AON (+120% on 2023), the highest count since 1978.

In less positive news, Fulmar counts recorded 381 apparently occupied sites (AOS), a 19% drop from 2023. Lesser Black-backed Gulls, fell to 5,198 AON/AOT (–12% on 2023) and Herring Gulls declined to 246 AON (–12% on 2023).

The whole-island Puffin count recorded 41,605 individuals (–2% vs. 2023; +19% above the five-year mean). There were no Cormorants breeding on Skomer for a fourth consecutive year.

Study plot abundance counts included Guillemot plots at the Wick that averaged 334 birds, down 6% on 2023; whilst Razorbill plots averaged 1,864 birds, a 3% decline on 2023.

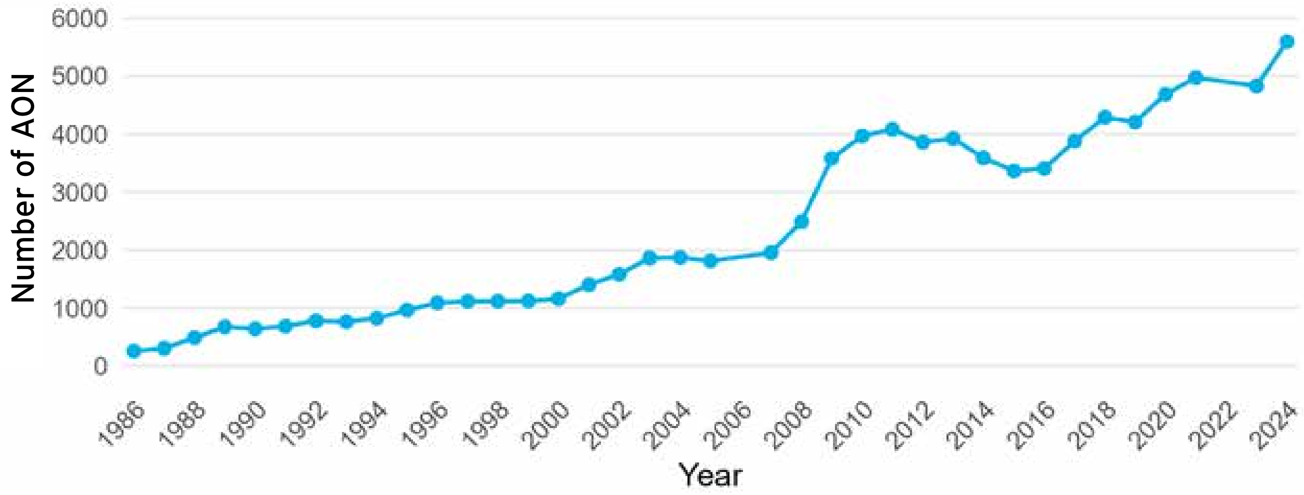
In the 18 fixed 1,000 m² Manx Shearwater plots, MP3 playback resulted in 1,366 responses – a 27% increase on 2023 despite burrow numbers dropping from 4,837 to 4,764.

It was broadly good news for the productivity of seabirds on Skomer in 2024. Fulmar productivity was 0.39 chicks per AOS, up from the five-year mean of 0.37, and Razorbills fledged 0.59 chicks per active site, which was slightly above the long-term average.

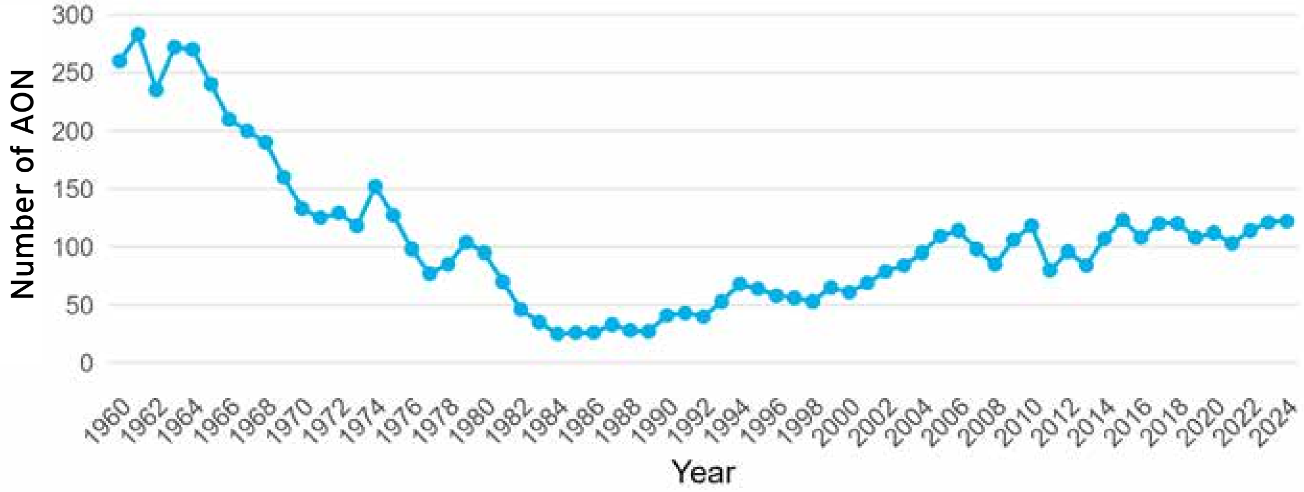
Guillemots fledged 0.75 chicks per egg laid, the highest since a change in methods in 2022 and Kittiwake productivity in study plots reached 0.64 fledglings per nest, +0.03 on the five-year mean. Herring Gulls also did very well, fledging 1.16 chicks per nest, over twice the long-term average of 0.50.

On the other hand, Great Black-backed Gulls produced 1.44 chicks per AON, which was below the five-year mean of 1.61.

GREAT BLACK-BACKED GULL, BY EDMUND FELLOWES / BTO



▲ **Figure 8:** Whole Colony Counts of Gannet Apparently Occupied Nests (AON) on Fair Isle.



▲ **Figure 9:** Whole Colony Counts of Great Black-backed Gull AONs on Skomer Island.

CLIFF COUNTS, BY THE WILDLIFE TRUST FOR SOUTH AND WEST WALES

Background and methods

The SMP is an ongoing annual monitoring programme, established in 1986, covering 25 seabird species that regularly breed in Britain and Ireland.

BACKGROUND

The SMP was established by JNCC (then known as the Nature Conservancy Council) in 1986, working in partnership with 19 other organisations. The aim was to set up an annual monitoring programme for the 25 seabird species which breed regularly in the UK, to allow their conservation status to be assessed. JNCC coordinated the collection, collation, and analysis of data on seabird breeding numbers and success, which were gathered from around the UK, the Channel Islands, the Isle of Man and the Republic of Ireland, by hundreds of skilled non-professional and professional participants. In 2022, JNCC formed a new partnership with BTO and RSPB for funding and management of the SMP. Drawing on its considerable expertise in running bird monitoring projects, BTO now leads on the coordination of the programme, data collation, analysis and outputs.

The SMP (www.bto.org/smp) aims to ensure that sample data on breeding abundance and productivity of a range of seabirds are collected both regionally and nationally, at both the coast and inland, to inform conservation policy and management affecting breeding seabirds.

The SMP Organiser, based at BTO, is responsible for the overall running of the programme, and is the main point of contact for participants. Survey locations are selected by participants based on breeding sites defined within the SMP database. Previously unrecorded sites can also be added to the database. At the end of each breeding season, data entered into the online data entry system, SMP Online (https://app.bto.org/seabirds), are validated, ready for data analysis by a BTO Research Ecologist.

Annual monitoring of breeding abundance and productivity at sample sites forms the core of the SMP and enables annual reporting from the programme. The results published form part of the suite of Official Statistics generated by JNCC. SMP data have helped identify possible drivers of seabird population change and, alongside national censuses, have been crucial for informing conservation policy, research and actions for this group of species.

Previously, SMP statistics were published annually in a report – *Seabird Numbers and Breeding Success in Britain and Ireland* – but in more recent years were presented on the JNCC website. Reporting from 2021 onwards is published on the BTO's SMP webpages.

SURVEY METHODS

Abundance is recorded using whole or plot Colony Counts; simply by counting the number of breeding individual adults, nests, sites, burrows or territories depending on the species. Productivity is the number of chicks to reach fledging age from a nest site or pair of breeding adults. The methods used to monitor abundance and productivity vary by species, and can be found in the *Seabird Monitoring Handbook for Britain and Ireland* (Walsh *et al.* 1995). This also includes the optimum date and time periods when monitoring should be carried out for each species.

For some species, there are multiple methods that can be used depending on the location and accessibility of colonies. *The Seabird Monitoring Handbook for Britain and Ireland* also includes details on how to select and monitor abundance and productivity at sites using plots, rather than whole-colony monitoring.

In addition to this widespread data collection, a Triennial Sites monitoring programme is carried out, whereby a range of seabird species are surveyed at three Scottish sites (St Kilda, Orkney and Bullers of Buchan) every three years by JNCC and the National Trust for Scotland. Data are also collected annually at four Key Sites distributed around the UK: Fair Isle, Canna and the Isle of May in Scotland and Skomer Island in Wales (see pages 18–21). Alongside extensive abundance and productivity studies at these sites, information about phenology (timing of the breeding season), diet and adult survival is also collected. Key Site monitoring is part-funded by JNCC, overseen by BTO, and the sites were chosen to be representative of the major part of the range of most seabird species, and to complement the monitoring carried out by the SMP.

The SMP is complemented by periodic national censuses that provide more comprehensive assessments of the size and overall status of breeding seabird populations across the whole of Britain, Ireland, the Isle of Man and the Channel Islands. These censuses began in 1969, take place at approximately 15–20 year intervals, and have been coordinated by JNCC. The latest and fourth census, *Seabirds Count*, was completed between 2015 and 2021 (Burnell *et al.* 2023).

ANALYTICAL METHODS

COLONY COUNTS (BREEDING ABUNDANCE)

Abundance trends are calculated for the majority of seabird species nesting in Britain and Ireland, monitored using Colony Counts at whole or plot-scale. However, for some species, the annual sample is too small or unrepresentative, or the species is too infrequently monitored to allow for accurate trends to be calculated, and this is discussed in the relevant species accounts.

For those species for which the production of annual trends is considered feasible, all sites within the SMP database with at least three colony counts submitted since its inception in 1986 are included in the annual trend analysis. This therefore excludes a large number of sites that have only been counted once or twice (for example, only during the *Seabird 2000* or *Seabirds Count* censuses).

To ensure results are reliable, breeding abundance trends are only produced for species and regions with sufficient data. To judge this, the number of colonies where data on abundance have been recorded during the trend period is examined. Specifically, trends must be based on data from at least 15% of colonies present within the SMP database (with at least three counts across the monitoring period) to be published i.e. if a trend uses at least 15% of underlying actual data rather than imputed. However, there can be exceptions to this rule. Puffins, for example, are challenging to survey, and coverage is biased towards smaller (potentially unrepresentative) sites. This means that, although the threshold of 15% of sites being covered is met, there remain very wide confidence intervals around the trend and it is therefore not published.

For sites with missing data for a given year, values are currently estimated using an imputation method (Thomas 1993) implemented in 'R', a software used for data science, statistics, and visualisation projects (R Core Team 2024). This approach calculates a value for the missing count using a weighted mean of all the non-missing counts for that site. The weights are based on the relative strengths of year effects across all sites. For a given year the total abundance across colonies is estimated by summing across the available observed data and imputed counts. Indices of abundance are produced by scaling the total abundance for each year relative to the base year (1986).

This imputation approach can introduce uncertainty, which is quantified by bootstrapping (Marchant *et al.* 2004), resampling with replacement across the included colonies. This generates confidence intervals for the estimated total abundance in each year that reflect uncertainty in the imputation of missing counts. Further details on the method behind the trend analysis for the indices of abundance, and estimation of productivity values are provided in *Methods of analysis for production of indices of abundance and estimation of productivity* (JNCC 2014). The analysis therefore produces estimated indices of abundance for each species with 95% confidence intervals, calculated through bootstrapping with replacement across sites (1,000 iterations), which reflects the confidence of the index values based on uncertainty around the imputed missing counts.

For some gull species, results are only presented for a particular subset of habitats. Due to insufficient data from inland colonies for Black-headed Gull and Common Gull, SMP reports have only provided trends for their coastal-nesting populations (sites within 5 km of the Mean High-Water Mark). For Lesser Black-backed Gull and Herring Gull, SMP trends are only presented for natural-nesting birds, given the inherent difficulties in accurately surveying urban nesters of these species.

Colony Count coverage was sparse in 2020 due to the COVID-19 pandemic. Calculation of the abundance trends, therefore, omitted 2020 data, but it was still possible to estimate a trend value for 2020 by interpolating the smoothed trend line between 2019 and 2021.

BREEDING SUCCESS (PRODUCTIVITY)

Productivity is estimated using data submitted from Breeding Success monitoring from within site plots which vary in size and number across sites.

Annual estimates of productivity are calculated using Generalised Linear Mixed Models (GLMMs) in the data analysis software Genstat (VSN International Ltd). For species that lay a single egg, the GLMM is run with a binomial error distribution and logit link function, with the sample size included as a binomial denominator. For species that lay more than one egg, the GLMM is run with a Poisson error distribution and log link function with the sample size included as an offset. Site is included as a random intercept to account for repeated measures of productivity for colonies over multiple years (JNCC 2014).

For each species, up to five models are tested:

1. A full interactive model of year and region/regional sea (subdivisions of the UK, formerly adopted as reporting regions in the SMP) effects;
2. Additive effects of year and region/regional sea;
3. Year only;
4. Region/regional sea only; and
5. Constant productivity (null model).

Model fit is tested using F-ratio statistics and a backward elimination approach to arrive at the minimum adequate model. The parameter estimates are extracted from the minimum adequate model and back transformed to produce estimates of productivity. No confidence intervals are currently implemented for this approach (see JNCC 2014 for further details). Therefore, no measure of uncertainty in the productivity estimates is provided.

Due to the COVID-19 pandemic, Breeding Success coverage was very limited in 2020. To prevent this from affecting the trends, all 2020 data were omitted from the analyses presented in this report (see Harris *et al.* 2021, Harris *et al.* 2022).

Interpreting the results

Pages 32–138 provide breeding abundance and productivity statistics, followed by accounts for the species monitored by the programme.

INFOGRAPHICS

Each species account contains an infographic that illustrates key facts and figures. The icons are as follows:

The **approximate percentage of the species' global population** breeding in Britain and Ireland. If this figure refers to a subspecies, the scientific name is included below the percentage figure in *italics* (Burnell et *al.* 2023).

The status of the species according to the UK **Birds of Conservation Concern 5 addendum** (Stanbury *et al.* 2024) and the *Birds of Conservation Concern in Ireland 4* (Gilbert *et al.* 2021) in *italics* (Red, Amber or Green – from highest to least concern).

International Union for Conservation of Nature (IUCN 2025) Global Red List status. Categories are: Extinct, Extinct in the Wild, Critically Endangered, Endangered, Vulnerable, Near Threatened, Least Concern, Data Deficient and Not Evaluated.

The UK SMP Long-term (LT, 1986–2024) breeding **abundance change** (Increase = increase of >10%, Decline = decrease of >10%, Stable = +/- change of up to 10% and n/a = insufficient SMP data to produce a trend), and the UK **productivity estimate** for 2024, unless specified as otherwise.

The **number of sites in Britain and Ireland** where Colony Count or Breeding Success monitoring was undertaken in 2024. Not all sites are used in the SMP trend analysis.

The **typical lifespan** of the species after reaching breeding age, and the **average age at which they start to breed** (BTO 2025, Burger *et al.* 2020, Horswill & Robinson 2015).

COVERAGE MAPS

Each species account includes a map of the coverage for that species in 2024, regardless of whether data from the site could be used in the abundance or productivity trends. It is important to stress that all data are vital for research projects and potentially future trend calculations. Some seabird species, such as the petrels and Manx Shearwater are considered cryptic and difficult to survey, nesting in hard-to-reach locations, therefore coverage is relatively low most years. Developments in surveying methods and sampling could open up new possibilities for annual monitoring and thus trend calculations – so please keep submitting data for all seabird species.

THRESHOLD FOR ABUNDANCE TRENDS

To ensure results are reliable, breeding abundance trends are only presented where they meet the thresholds described in the 'Background and methods' section of this report (pages 22–33).

TRENDS AND TABLES EXPLAINED

EXAMPLE 1: SMP BREEDING ABUNDANCE CHANGE AND PRODUCTIVITY TABLES

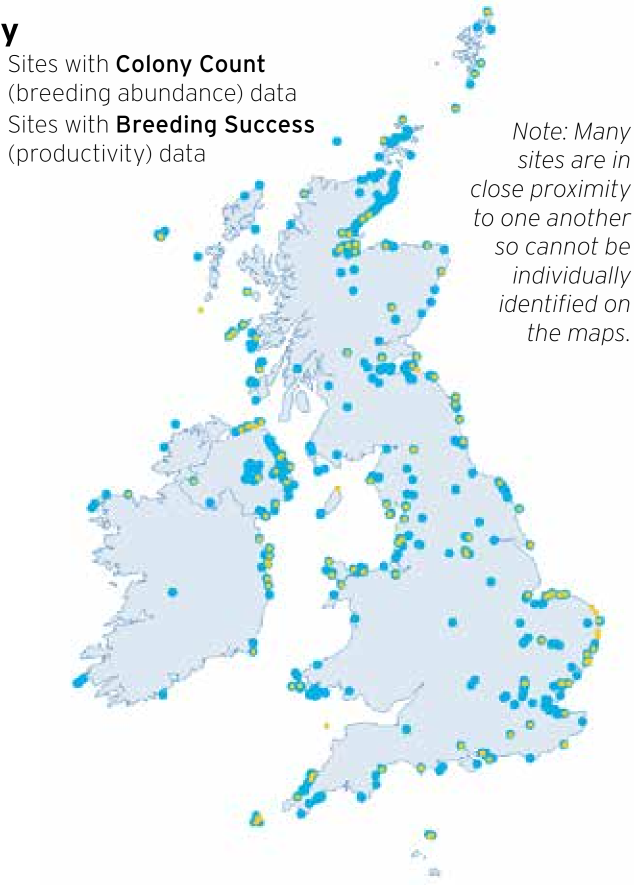
For each species, region-specific population estimates are provided from the *Seabirds Count* (2015–2021) census in the '*Seabirds Count*' column to provide context to the respective SMP-derived change values. Unit values are abbreviated as: AON (Apparently Occupied Nest), AOS (Apparently Occupied Site), AOT (Apparently Occupied Territory), AOB (Apparently Occupied Burrow) and IND (Individual).

All other values in the table are produced using SMP data. In the 'Breeding Abundance %' section of the table, the 'Sites 2024' column refers to the total number of sites in that year that were *used to produce the most recent abundance or productivity trends* for that species and region and this will, therefore, vary from the *total coverage* figures and coverage maps. The UK total includes sites from England, Scotland, Wales and Northern Ireland. For most species, the UK total will be greater than the sum of sites included for the constituent countries within the table, as trends cannot yet be produced for all four countries (specifically Northern Ireland).

Key

● Sites with **Colony Count** (breeding abundance) data

● Sites with **Breeding Success** (productivity) data



Note: Many sites are in close proximity to one another so cannot be individually identified on the maps.

Changes in SMP breeding abundance are presented as the percentage change over two periods: the long-term (LT) and 24-years. Unless stated otherwise, the LT change covers the lifetime of the SMP (1986–2024) and the 24-yr change covers the period 2000 to 2024, with 2000 being the mid-point of the Seabird 2000 (1998–2002) census (Mitchell *et al.* 2004).

Percentage changes are calculated by comparison of the annual abundance estimate for 2024 with those for 1986 (LT change) or 2020 (24-yr change). Changes are considered significant if the 95% confidence intervals of the 2024 estimate do not overlap with the 1986 baseline index of 100 (LT change) or the point estimate for 2000 (24-yr change). Significant changes are marked with an asterisk (*).

The final two columns in the table present the productivity estimates for 2024 and the number of sites from which these were produced. Where it has only been possible to produce figures for one (breeding abundance or productivity) set of results, the tables have been reduced accordingly. The productivity values for 2022 and 2023 can be found on pages 28–31. These values may differ slightly from those reported in the previous report (Harris *et al.* 2024) due to additional data being included in the productivity analysis for 2024.

Variations to the table will occur when abundance changes can only be provided for a particular subset of habitats, or where breeding abundance and/or productivity values are not available for a particular species. Where abundance changes can only be provided for specific nesting habitat types for a given species, this is highlighted in red text in the tables, e.g. COASTAL NESTERS (within 5 km of Mean High Water Mark) or NATURAL NESTERS (on moors, cliffs, marshes, beaches and other areas of semi-natural habitat). See Background and Methods (pages 22–23) for more information.

The SMP sampling and analysis strategy is underway to improve the precision and representativeness of future trends. As a result, in the discussions within each species account, fine spatial-scale analysis of the breeding abundance and productivity trends has not been carried out – with the exception of Gannet.

Note: the term 'region' is used to describe geographic areas including multiple Crown Dependencies (e.g. Channel Islands and Isle of Man) or country groups (e.g. UK, Ireland or Britain and Ireland).

EXAMPLE 2: SEABIRDS COUNT CENSUS RESULTS TABLES

This table shows the abundance change as measured between the *Seabird 2000* (1998–2002) and *Seabirds Count* (2015–2021) censuses, allowing for comparison with the SMP change. For Gannet, a combined result from the most recent Gannet census and the *Seabirds Count* census has been provided as per *Seabirds Count* reporting.

Example 1: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AOS)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	319,508	173	-35*	-32*	0.29	35
England	4,903	34	-40	-36	0.41	12
Scotland	309,545	102	-36*	-32*	0.29	15
Wales	2,494	23	-4	-26	0.49	1

* significant changes

Example 2: Seabirds Count Census Results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	539,977	352,995	-35

The censuses aim to cover the entire population of each species within the whole of Britain, Ireland, Isle of Man and the Channel Islands. The SMP abundance changes trends provide interim values. This is discussed in more detail in the species accounts.

INTERPRETING GRAPHS

All SMP graphs are displayed in the same way throughout the report. The time period starts at 1986 (the SMP baseline year) and ends in 2024 and is illustrated on the x-axis. Please note that both the index of abundance and productivity axes can vary in scale.

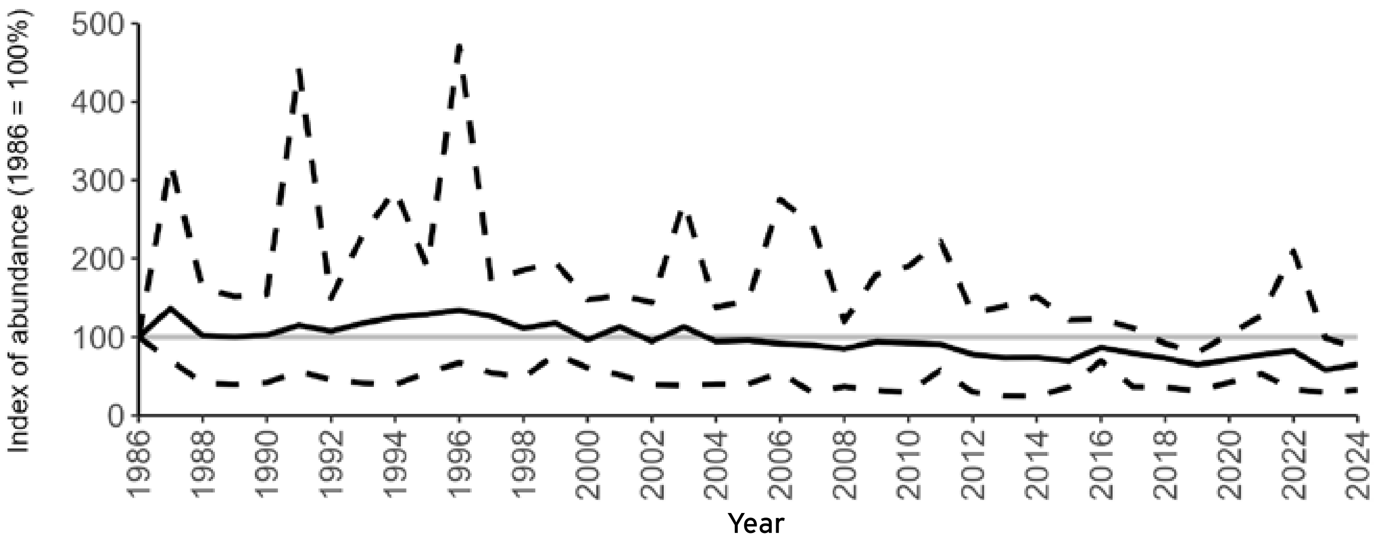
EXAMPLE 3: SMP *BREEDING ABUNDANCE* GRAPHS

The region-specific abundance index graphs show:

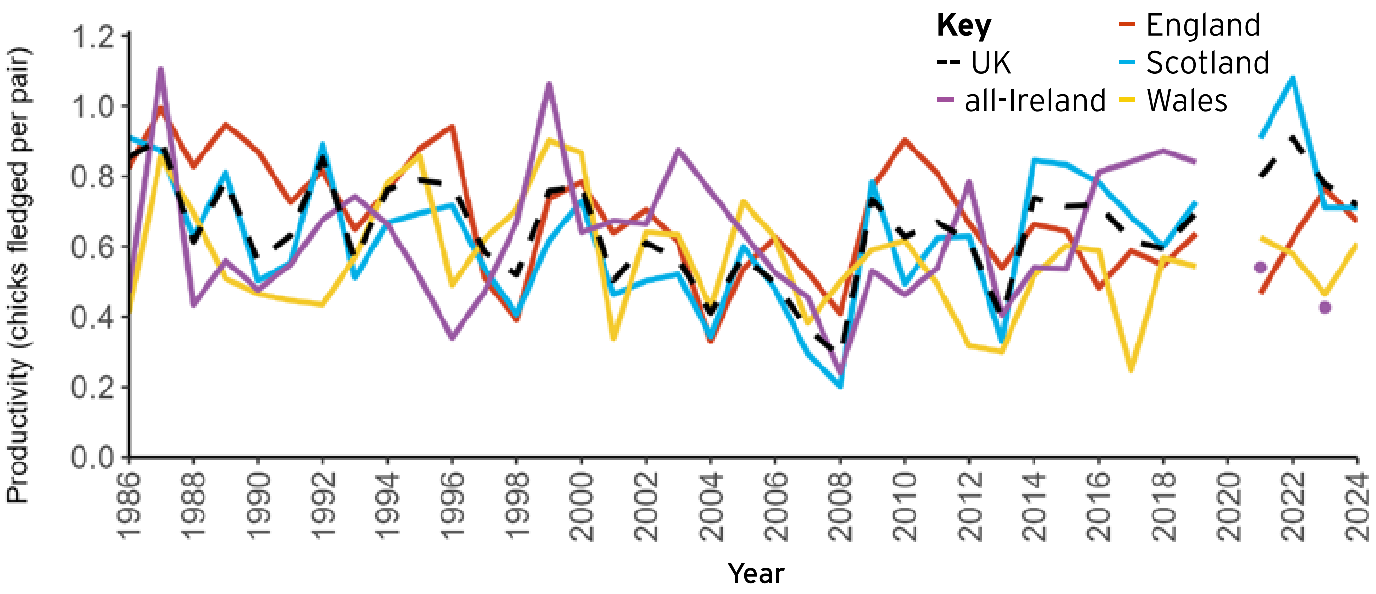
- the abundance trend: solid black line linking values for individual years to illustrate the overall trend over time.
- confidence intervals (95%): black dashed line.

To make it easier to compare trends between species, breeding abundance is expressed as an 'index', set to 100 in the first year (1986), and shown across the monitoring period by a grey solid line.

Example 3: UK SMP Breeding Abundance (1986–2024)



Example 4: SMP Productivity (1986–2024)



EXAMPLE 4: SMP *PRODUCTIVITY* GRAPHS

Multi-country productivity index graphs show the productivity estimates as a dotted black line for the UK, with a solid line for each country and 'all-Ireland'.

This figure is used to illustrate the trend in productivity values over the SMP time period, and also to show where the trends differ between regions, either in their direction or timing.

Due to restricted coverage during the COVID-19 pandemic in 2020 and limited coverage for some species in some years, occasional gaps in productivity results may feature for a given year.

CAVEATS

Results for species with sparse datasets carry wider confidence intervals and should be interpreted cautiously.

Confidence intervals shown in the breeding abundance index graphs only reflect uncertainty in the imputation of missing data. They do not reflect uncertainty in abundance counts and, therefore, the overall uncertainty in the population estimates. Caution should therefore be applied when interpreting the significance of reported population change figures. Due to high amounts of sparsity for several species, especially for region-level data, there is a high risk of false negatives when assessing significance of population changes.

The indices used to assess population change are unsmoothed and therefore percentage changes could be influenced by extreme annual estimates.

GANNETS, BY RICHARD JACKSON / BTO

FIND OUT MORE ...

Trends Explorer: The Bird Trends Explorer is an interactive tool for exploring how bird abundance and other key measures of bird population health are changing through time in the UK and constituent countries. The Bird Trends Explorer accompanies BTO BirdFacts, which provides key information about biology, distribution, phenology, trends and conservation of UK bird species. Visit: https://data.bto.org/trends_explorer

The UK's seabirds: an overview

Table 2: An overview of the breeding abundance changes and productivity values for the UK. For information on interpreting the data presented in this table, see pages 24–27. Information and context regarding the results presented here are available in full within the species accounts (see pages 32–138).

Table 2: UK SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity					
	Abundance	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2022	Sites	2023	Sites	2024	Sites
Fulmar (AOS)	319,508	173	-35*	-32*	0.42	24	0.33	35	0.29	35
Manx Shearwater	-	-	-	-	0.55	1	0.60	3	0.60	1
Gannet	-	-	-	-	0.31	6	0.60	6	0.53	7
Cormorant (AON)	8,829	49	-8	-16	-	-	-	-	-	-
Shag (AON)	20,209	116	-71*	-65*	1.50	13	1.31	20	0.93	17
Arctic Skua (AOT)	727	11	-93*	-89	0.69	3	0.58	2	0.03	4
Great Skua	-	-	-	-	0.09	5	0.44	4	0.17	6
Black-headed Gull (AON)	51,649 (COASTAL NESTERS)	75	-18 (COASTAL NESTERS)	-37	0.36	27	0.30 (ALL NESTERS)	29	0.72	30
Common Gull (ALL NESTERS)	-	-	-	-	0.83	7	0.62	11	0.43	11
Lesser Black-backed Gull (AON) (NATURAL NESTERS)	55,304	76	-72*	-81*	0.40	11	0.48	13	0.40	11
Herring Gull (AON) (NATURAL NESTERS)	61,007	169	-51*	-47*	0.40	16	0.52	15	0.69	19
Great Black-backed Gull (AON)	8,021	98	-16	-20	0.28	12	1.04	11	1.24	9
Kittiwake (AON)	215,913	99	-55*	-38*	0.91	25	0.78	28	0.72	30
Sandwich Tern (AON)	12,980	15	-26*	-24*	0.30	10	0.17	6	0.58	11
Roseate Tern (AON)	120	3	-61	125	0.30	3	-	-	0.85	2
Common Tern (AON)	12,219	103	-63*	-60*	0.40	37	0.43	29	0.27	37
Arctic Tern (AON)	30,451	52	-25	-29	0.14	24	0.11	16	0.11	19
Little Tern (AON)	1,403	37	-19	-2	0.72	43	0.61	44	0.25	20
Guillemot (IND)	1,265,888	105	40	18	0.59	8	0.46	11	0.51	8
Razorbill (IND)	225,015	123	110*	52	0.53	8	0.55	12	0.44	9
Black Guillemot (IND)	35,193	46	3	61	-	-	-	-	-	-
Puffin	-	-	-	-	0.61	2	0.49	6	0.56	8

** Significant abundance changes (where confidence intervals, which reflect imputation uncertainty, do not overlap with the 1986 baseline for the long-term change, or 2000 index value for the 24-year change).*

England's seabirds: an overview

Table 3: An overview of the breeding abundance changes and productivity values for England. For information on interpreting the data presented in this table, see pages 24–27. Information and context regarding the results presented here are available in full within the species accounts (see pages 32–138).

Table 3: England SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity					
	Abundance	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2022	Sites	2023	Sites	2024	Sites
Fulmar (AOS)	4,903	34	-40	-36	0.60	8	0.47	10	0.41	12
Cormorant (AON)	3,333	22	26	-8	-	-	-	-	-	-
Black-headed Gull (AON) (COASTAL NESTERS)	40,398	52	-5	-34	-	-	-	-	-	-
Herring Gull (AON) (NATURAL NESTERS)	11,736	59	-77	-73	-	-	-	-	-	-
Kittiwake (AON)	72,897	22	-43	-26	0.62	7	0.76	8	0.67	9
Sandwich Tern (AON)	9,503	9	-27	-22	0.31	8	0.20	4	0.77	5
Common Tern (AON)	5,478	60	-47*	-51*	0.44	22	0.90	21	0.38	22
Arctic Tern (AON)	6,118	9	-50	-38	0.56	6	0.19	4	0.41	6
Little Tern (AON)	1,004	20	-15	4	0.82	33	0.76	35	0.26	16
Guillemot	-	-	-	-	0.51	2	0.59	2	0.60	2

** Significant abundance changes (where confidence intervals, which reflect imputation uncertainty, do not overlap with the 1986 baseline for the long-term change, or 2000 index value for the 24-year change).*

GUILLEMOTS, BY TOM CADWALLENDER / BTO

Scotland's seabirds: an overview

Table 4: An overview of breeding abundance changes and productivity values for Scotland. For information on interpreting the data presented, see pages 24–27. Information and context regarding the results presented here are available in full within the species accounts (see pages 32–138).

Table 4: Scotland SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity					
	Abundance	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2022	Sites	2023	Sites	2024	Sites
Fulmar (AOS)	309,545	102	-36*	-32*	0.41	12	0.35	12	0.29	15
Gannet		-	-	-	0.22	5	0.59	4	0.64	5
Shag (AON)	16,788	74	-70*	-62*	1.46	9	1.26	13	0.83	10
Arctic Skua (AOT)	727	11	-93*	-89	0.69	3	0.58	2	0.03	4
Great Skua	-	-	-	-	0.09	5	0.44	4	0.17	6
Black-headed Gull (ALL NESTERS)	-	-	-	-	0.43	4	0.49	5	0.49	4
Common Gull (AON)	12,427 (COASTAL NESTERS)	29 (to 2023)	-19 (to 2023) (COASTAL NESTERS)	-38 (to 2023)	0.73	2	0.57 (ALL NESTERS)	2	-	-
Lesser Black-backed Gull (AON) (NATURAL NESTERS)	11,001	15	-42	-46	-	-	-	-	-	-
Herring Gull (AON) (NATURAL NESTERS)	37,349	75	-54*	-43*	-	-	-	-	-	-
Great Black-backed Gull (AON)	5,404	62	-59*	-57*	0.19	3	1.00	3	1.23	5
Kittiwake (AON)	121,082	63	-53*	-41*	1.08	15	0.71	14	0.71	15
Sandwich Tern		-	-	-	0.44	1	0.11	1	0.26	1
Common Tern (AON)	4,071	27	-73*	-67*	0.67	9	0.45	4	0.26	7
Arctic Tern (AON)	19,555	34	-35	-35	0.30	15	0.18	9	0.10	9
Little Tern (AON)	227	16	-75*	-59	1.17	8	0.50	7	0.41	4
Guillemot (IND)	810,645	70	-29	-38	0.59	5	0.46	6	0.55	5
Razorbill (IND)	138,828	78	63	4	0.56	3	0.53	6	0.51	3
Black Guillemot (IND)	33,986	11	-44	-14	-	-	-	-	-	-
Puffin	-	-	-	-	0.63	0[†]	0.48	4	0.58	4

** Significant abundance changes (where confidence intervals, which reflect imputation uncertainty, do not overlap with the 1986 baseline for the long-term change, or 2000 index value for the 24-year change).*

[†] estimated from the productivity analysis. Models can calculate missing years for sites with over three years of previous data

Wales's seabirds: an overview

Table 5: An overview of the breeding abundance changes and productivity values for Wales. For information on interpreting the data presented in this table, see pages 24–27. Information and context regarding the results presented here are available in full within the species accounts (see pages 32–138).

Table 5: Wales SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity					
	Abundance	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2022	Sites	2023	Sites	2024	Sites
Fulmar (AOS)	2,494	23	-4	-26	0.45	3	0.34	4	0.49	1
Cormorant (AON)	1,477	8	-29	-7	-	-	-	-	-	-
Shag (AON)	651	16	-7	-1	1.57	2	1.63	4	1.59	2
Lesser Black-backed Gull (AON) (NATURAL NESTERS)	13,084	17	-76	-79	-	-	-	-	-	-
Herring Gull (AON) (NATURAL NESTERS)	9,815	29	-18	-31	0.58	3	0.55	4	0.81	4
Great Black-backed Gull (AON)	648	12	170*	76	-	-	-	-	-	-
Kittiwake (AON)	4,782	10	-56*	-53*	0.58	3	0.47	2	0.61	2
Razorbill (IND)	23,640	30	256*	141*	-	-	-	-	-	-

** Significant abundance changes (where confidence intervals, which reflect imputation uncertainty, do not overlap with the 1986 baseline for the long-term change, or 2000 index value for the 24-year change).*

Remaining countries or regions covered by the SMP: an overview

In previous reports covering the annual results of the SMP, abundance and productivity values have been presented for particularly important colonies for some species in regions or countries, especially the Channel Islands, Isle of Man, Northern Ireland, the Republic of Ireland and all-Ireland, where insufficient data were available to produce robust trends. Although data may be insufficient to produce these region- or country-specific trends at present, all UK-scale trends include data from England, Northern Ireland, Scotland and Wales.

In the species accounts, some of the productivity graphs show historic data for certain areas where data from more recent years are currently unavailable. Work behind the scenes by both organisational staff and volunteers aims to increase future coverage, collate any missing historic data and improve data flow into the SMP database via SMP Online. As such, future reports are highly likely to feature results from these countries and areas.

Additionally, SMP data analysis methods are currently undergoing development and, together, these advances will increase the ability for greater reporting across Britain and Ireland.

We are particularly grateful to those collecting or collating information from seabird colonies in the Channel Islands, Isle of Man, Northern Ireland, and the Republic of Ireland for their continued support, efforts to align databases and, of course, to everyone for the data submitted to date.

Species accounts

A summary of all seabird species monitored by the SMP follows. This includes survey coverage maps, status, species information, breeding abundance and productivity estimates (where possible), causes of change and conservation initiatives. Please refer to the Interpreting the results pages (24–27) for this section.

By **Sarah Harris**, SMP Organiser, BTO, **Nina O'Hanlon**, Senior Research Ecologist, BTO, **Samuel Langlois**, Research Ecologist, BTO, **Hannah Hereward**, Research Ecologist, BTO, and **Sarah Money**, Marine Ornithologist, JNCC.

PUFFIN, BY EDMUND FELLOWES / BTO

Fulmar

Fulmarus glacialis

c.11%
ssp. glacialis

Abundance: Decline
Productivity: 0.29

Amber-listed
Amber-listed (I)

Colony Count sites: 199
Breeding Success sites: 37

Least Concern

Lifespan: 44 years
Breeding age: 9 years

Britain and Ireland host 5% of the world's breeding Fulmar but around 11% of the subspecies *glacialis* (Burnell *et al.* 2023). They have two colour morphs; one pale and most often encountered around the UK, and the other dark. The latter, referred to as a 'blue Fulmar', is grey all over, and more prevalent in colonies in the high Arctic (van Franeker & Wattel 1982).

DISTRIBUTION

Within Britain and Ireland, Fulmars were originally restricted to the remote archipelago of St Kilda (Scotland), but the breeding population spread rapidly in the 20th century and they are now found breeding around much of the British and Irish coastline (Burnell *et al.* 2023, Balmer *et al.* 2013). An increase in fishery discards at the time has been suggested as one reason for their expansion (Fisher 1952, Bicknell *et al.* 2013, Cordes *et al.* 2015).

Globally, Fulmar are found across the North Atlantic and North Pacific, ranging from the UK to Japan in their southern range, and extending north to the high Arctic (BirdLife International 2024).

The population in Britain and Ireland has no pronounced migration and birds are present offshore during the winter (Quinn *et al.* 2016).

DIET

Fulmar are predominantly surface feeders (Garthe & Furness 2001) and feed on sandeels and zooplankton, but also scavenge on fishery discards (Fisher 1952, Phillips *et al.* 1999, Bicknell *et al.* 2013, Darby *et al.* 2021). The foraging trips of Fulmars can be some of the longest in terms of duration and distance of any UK seabird, often lasting several days. A remarkable trip lasting 14.9 days and covering 6,200 km by a male Fulmar was recorded while its partner was incubating on Eynhallow, Orkney (Edwards *et al.* 2013).

BREEDING

Typically, Fulmars nest on cliffs but will also nest on gentle slopes, under boulders, in the entrance to Puffin burrows, at the base of dry-stone walls or in sand dunes. Nest site opportunities increase on mammalian predator free islands (Anderson 1982, Mitchell *et al.* 2004).

BREEDING ABUNDANCE

The decline of 35% at the UK level recorded by the SMP between 2000 and 2024 is similar to the decline of 37% (UK) recorded by the *Seabirds Count* census since *Seabird 2000.* For England and Scotland, the SMP showed declines of 36% and 32%, respectively, since 2000, whilst the *Seabirds Count* census reported declines of 22% (England) and 37% (Scotland) since *Seabird 2000.* For Wales, the SMP data showed a decline of 26% since 2000, whilst the *Seabirds Count* census showed a similar decline of 27% since the *Seabird 2000* census (Burnell *et al.* 2023).

Regional variation has occurred in Fulmar SMP abundance since 1986 (Figures 10–13). Across most regions, Fulmar trends generally increased between 1986 and the mid-1990s. However, the Scotland trend has since declined markedly. Given that Scotland holds the majority of the UK Fulmar population, the SMP Fulmar trend for the UK as a whole closely matches that for Scotland. In the UK and Scotland, after the previous lowest SMP index values in 2019 since 1986 (of -37% and -42%, respectively), the index for 2022 increased slightly to 13% (UK) and 15% (Scotland) below the baseline. However, the index values have since declined to 35% (UK) and 36% (Scotland) below the baseline in 2024 (Table 6). After the mid-1990s, the abundance trend for Wales fluctuated between periods of stability and noticeable declines, specifically after 2005 and 2017, with an index value of 4% below the baseline in 2024. The trend for England has been relatively stable in recent years, however the index value declined to 40% below the baseline in 2024 (Table 6).

The *Seabirds Count* census also showed declines in Fulmar populations for Northern Ireland, the Isle of Man and the Channel Islands, and a stable population for the Republic of Ireland (Burnell *et al.* 2023). Unfortunately, current data submitted to the SMP for these regions are too sparse to produce SMP abundance trends.

PRODUCTIVITY

Considerable variation has occurred in Fulmar productivity trends across the regions monitored (Figure 14). The productivity trends for the UK and Scotland are relatively stable and follow each other closely, as much of the data have been collected in Scotland across the SMP monitoring period (since 1986). The trends for England and Wales have fluctuated more widely between years. In Wales this is likely to be a result of fewer colonies being monitored on an annual basis. For both the UK and Scotland the mean productivity estimate declined to 0.29 chicks fledged per pair in 2024 (Table 6). The productivity estimates for England and Wales in 2024 were

▶

FULMAR, BY EDMUND FELLOWES / BTO

Table 6: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AOS)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	319,508	173	-35*	-32*	0.29	35
England	4,903	34	-40	-36	0.41	12
Scotland	309,545	102	-36*	-32*	0.29	15
Wales	2,494	23	-4	-26	0.49	1

* significant changes

Table 7: Seabirds Count census results

	Abundance (AOS) *Seabird 2000* (1998–2002)	Abundance (AOS) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	539,977	352,995	-35

Figure 10: UK SMP Breeding Abundance (1986–2024)

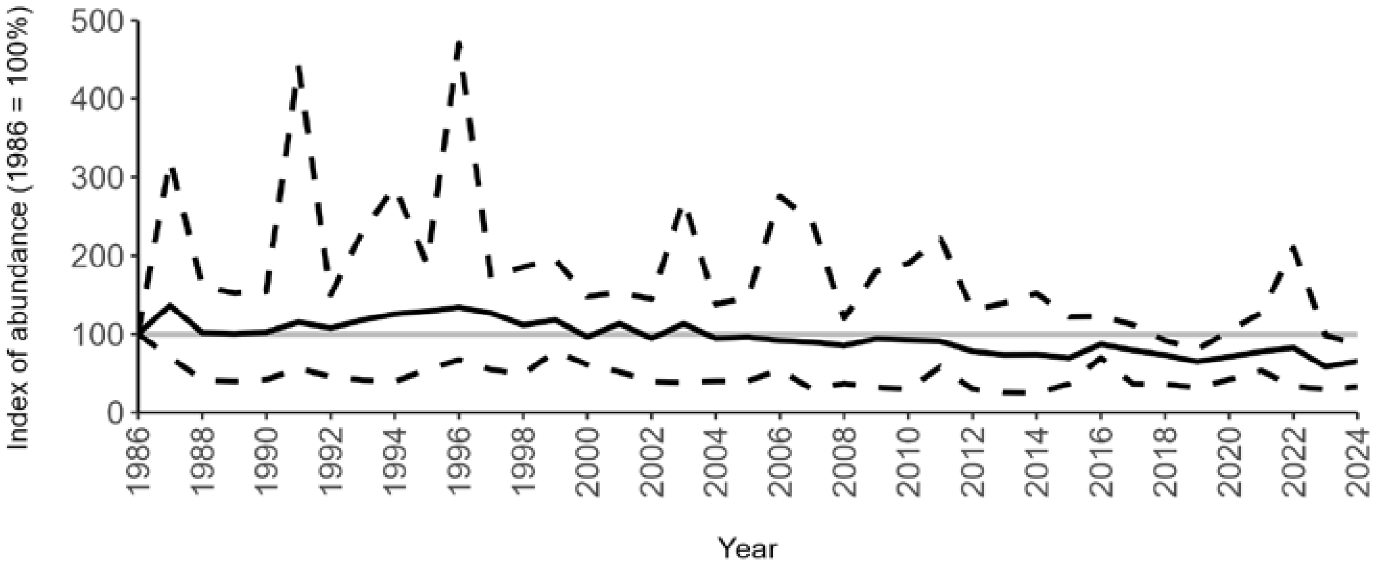


Figure 11: England SMP Breeding Abundance (1986–2024)

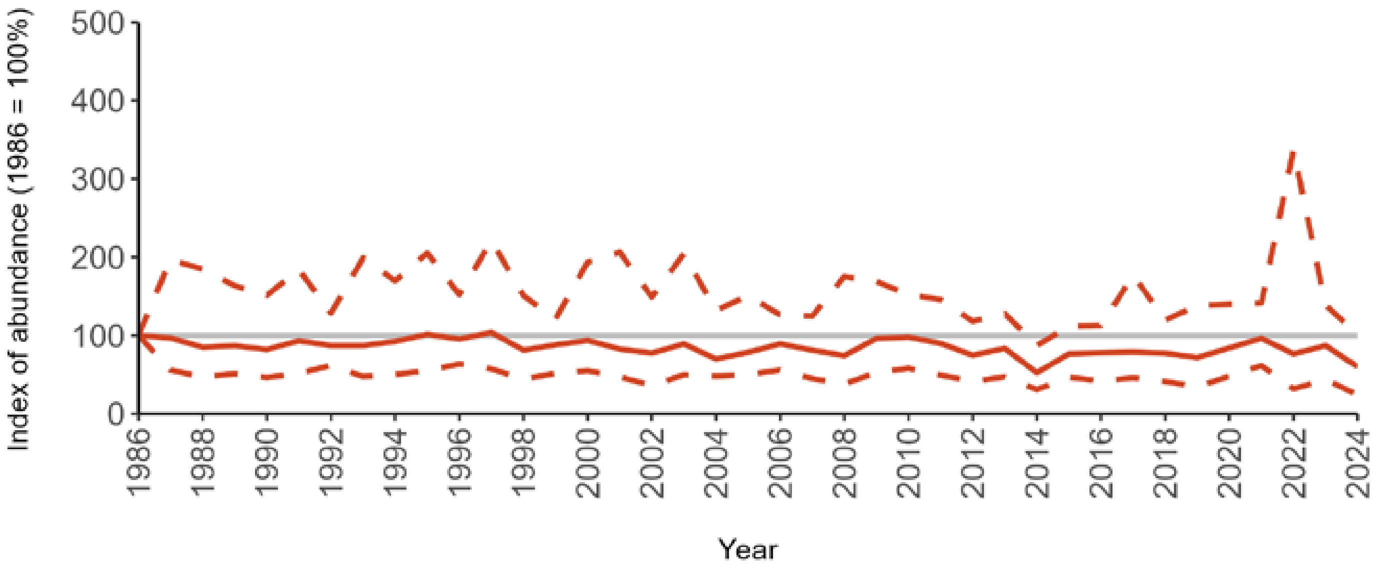


Figure 12: Scotland SMP Breeding Abundance (1986–2024)

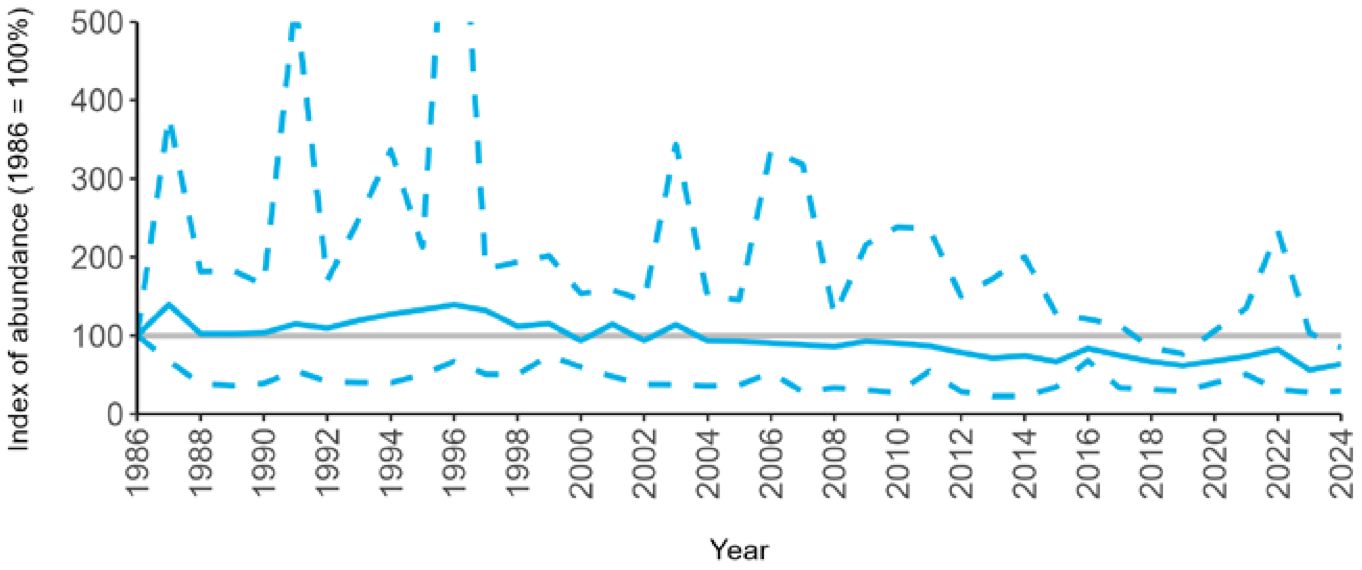
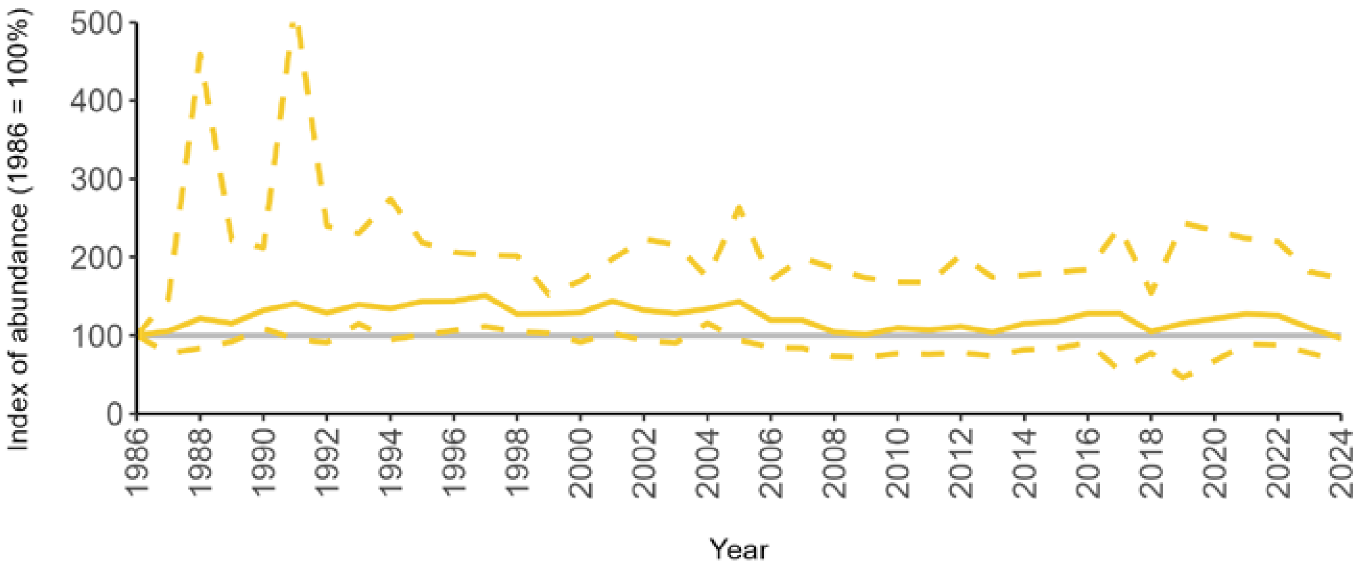
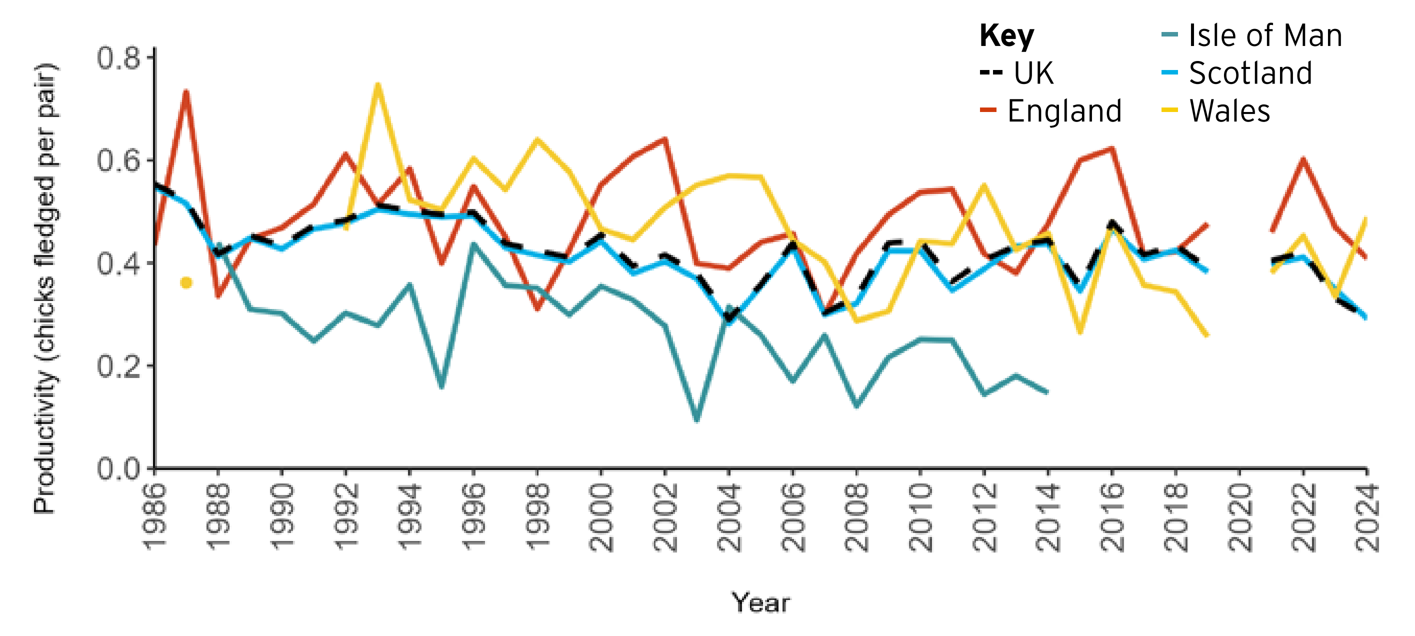


Figure 13: Wales SMP Breeding Abundance (1986–2024)



FULMAR, BY PHILIP CROFT / BTO

Figure 14: SMP Productivity (1986–2024)



FULMAR, BY SARAH KELMAN / BTO

0.41 and 0.49 chicks fledged per pair, respectively (Table 6).

Monitoring of productivity has been low on the Isle of Man throughout the SMP period and no data have been submitted since 2014. Too few data are submitted to the SMP on productivity of Fulmars from other regions to calculate any meaningful productivity values. However, sites with data in Northern Ireland are included within the UK level trend.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

There are a range of pressures Fulmar face, and as a long-lived species, reaching breeding maturity at nine years and laying just one egg per season, populations are particularly vulnerable to pressures acting on adult survival rates and productivity (Burnell *et al.* 2023).

Fulmar have expanded and increased across their range over the last two centuries, potentially due in part to increases in fishery discards and food availability (Burg *et al.* 2003). With the recent banning of fisheries discards it is possible that populations might return to historic levels (Bicknell *et al.* 2013).

Accidental deaths due to bycatch by long-line fisheries in the Norwegian Sea and the North Atlantic (Northridge *et al.* 2020) as well as in gillnets (Žydelis *et al.* 2013) are thought to be having a significant impact on populations. Data analysis on possible future trends identified that if these accidental deaths were stopped, the population could increase between 2% and 17% over a 25-year period (Miles *et al.* 2020).

An increased frequency in extreme weather events has been shown to negatively influence Fulmar breeding success in high arctic Canada, with egg or chick loss following storms (Mallory *et al.* 2009), and climate change-induced changes in sea surface temperatures have altered the distribution of Fulmar's natural prey, such as sandeels, reducing prey availability during the breeding season (MacDonald *et al.* 2015). On the east coast of Scotland, changes in sea surface temperatures have been shown to have a negative impact on Fulmar productivity (Burthe *et al.* 2014).

FULMAR, BY PHILIP CROFT / BTO

Plastic pollution is also a potential issue for Fulmars, with recent studies showing that 80–95% of Fulmars sampled in the UK and Svalbard had ingested at least one piece of plastic. These studies encompass historic records from 1980 to recent records from 2020 (van Franeker *et al.* 2021, Collard *et al.* 2022). However, further research is needed to determine whether this ingestion causes increased mortality (Kühn *et al.* 2020, Neumann *et al.* 2021).

Predation pressure by non-native mammalian species, such as American Mink (*Neovison vison*) (Craik 1997), and from the native White-tailed Eagle (*Haliaeetus albicilla*), numbers of which have increased since reintroductions started in the 1970s (Evans *et al.* 2009), also have the potential to have population level impacts on Fulmars, but further study is needed.

CONSERVATION

The invention and use of bird deterring technology for long-line fisheries has shown to be effective in reducing the accidental catching of Fulmar and other species. These systems include a bird-scaring streamer line, which provides a visual and physical deterrent across the fish bait lines and hooks before they sink (Løkkeborg & Robertson 2002).

As well as general measures to reduce the rate of climate change, Fulmars are likely to benefit from specific policies that aim to reduce pressure caused by other factors e.g. reduction or cessation of commercial fishing for important fisheries. ■

Manx Shearwater

Puffinus puffinus

c.96%

Abundance: n/a
Productivity: 0.60

Amber-listed
Amber-listed (I)

Colony Count sites: 5
Breeding Success sites: 1

Least concern

Lifespan: 15 years
Breeding age: 5 years

Britain and Ireland host approximately 96% of the world's breeding population of Manx Shearwater (Burnell *et al*. 2023). Two island colonies, one on Skomer Island (Wales) and the other on Rum (Scotland), together host around 70% of Britain and Ireland's breeding population (Burnell *et al*. 2023).

DISTRIBUTION

There are c. 50 Manx Shearwater colonies in Britain and Ireland (Burnell *et al.* 2023). They are all on offshore islands, mainly around the Irish Sea and Atlantic coasts. Manx Shearwaters are wide-ranging foragers and so can be seen offshore around the coast throughout the breeding season, only coming ashore to breed (BTO 2025).

Globally, Manx Shearwaters also breed in Iceland, France, Spain, Portugal (the Azores and Madeira), Canada, the eastern United States of America, and the Faroes (del Hoyo *et al.* 1992).

Individuals from the north-east Atlantic, including Britain and Ireland, undertake a clockwise migration of the Atlantic Ocean, migrating south along the west coast of Africa, crossing to Brazil at the narrowest section and overwintering on the Patagonian Shelf off Argentina. They return to breed via the eastern Caribbean, circling near the eastern seaboard of North America before returning to the North Atlantic (Guildford *et al.* 2009). Birds from the west coast of the Atlantic migrate there and back along the eastern seaboard of North and South America (Fayet *et al.* 2020).

DIET

Manx Shearwaters feed by plunging into the sea to depths of around 10 m, pursuing squid and small fish (Brooke 1990, Shoji *et al.* 2016).

BREEDING

Manx Shearwaters nest in burrows and under boulders, often on steep grassy slopes, and lay a single egg. Their breeding colonies are mainly restricted to invasive species-free islands, where there is reduced predation risk. They only visit colonies at night to avoid predation by aerial predators such as gulls and Great Skuas (Mougeot & Bretagnolle 2000).

BREEDING ABUNDANCE

SMP annual abundance trends could not be produced for Manx Shearwater as too few colonies are monitored regularly to allow production of reliable trends. This is due to the difficulties inherent in accessing remote colonies and the costly and labour-intensive nature of the surveys. Consistent (possibly using sample plots) annual monitoring of abundance at colonies across their range may allow trends to be produced in the future. Most Manx Shearwater colonies monitored during the *Seabirds Count* census showed an increase in population estimates across Britain and Ireland compared to the previous census, *Seabird 2000* (Table 9, Burnell *et al.* 2023). However, many of the counts between the two censuses are not directly comparable due to methodological and analytical differences, and therefore considerable caution is required in interpreting these changes.

PRODUCTIVITY

Due to the difficulties involved in monitoring burrow-nesting Manx Shearwaters, productivity is only regularly monitored at a few colonies, but the numbers are sufficient to produce a UK productivity trend. There has been limited fluctuation across the SMP monitoring period (Figure 15). In 2024, 0.60 chicks were fledged per pair from a single location, Skomer (Table 8), the same as the long-term average.

Figure 15: SMP Productivity (1986–2024)

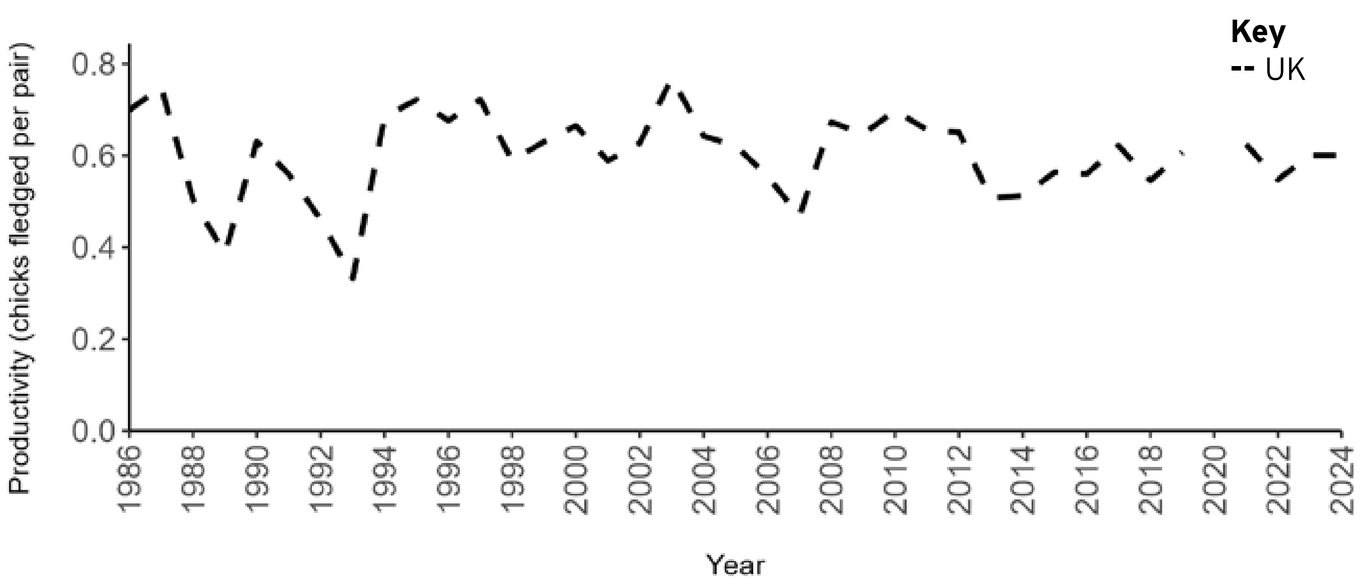


Table 8: SMP Productivity

	Productivity 2024	Sites
UK	0.60	1

PHENOLOGY, DIET AND SURVIVAL RATES

Survival estimates of Manx Shearwater are estimated for the Skomer Island Key Site. No systematic data on phenology or diet have been submitted to the SMP.

CAUSES OF CHANGE

Factors potentially influencing Manx Shearwater numbers include hunting by humans, e.g. legal harvesting in the Faroe Islands (Carboneras *et al.* 2014, Thorup *et al.* 2014), fisheries bycatch in longlines and gillnets (Žydelis *et al.* 2013) and a change to the discards policy reducing fishery discards as a potential food source, although the latter is not thought to be frequently utilised by Manx Shearwaters (Bicknell *et al.* 2013).

The impact of wind farm-related collision and displacement is currently unknown for Manx Shearwater (Bradbury *et al.* 2014, Deakin *et al.* 2022, Dierschke *et al.* 2016).

Predation by species such as rats, mice and Domestic Cats (*Felis catus*) all add to potential pressures (Burnell *et al.* 2023, Mitchell *et al.* 2004). However, the Isle of Rum hosts a large number of breeding Manx Shearwater where they coexist with Brown Rat (*Rattus norvegicus*). This is thought to be due to the breeding colony being above the altitude at which the rats occur on the island (Lambert *et al.* 2015).

The shearwaters' flight efficiency and pelagic nature allows them to explore large areas of sea in search of food, potentially limiting the impact of prey distribution change due to climate change. However, research on Skomer Island suggests that later breeding seasons and lower chick weight at fledging are linked to higher sea surface temperatures (Riou *et al.* 2011). Burrow flooding due to extreme rainfall events in summer could become more frequent with climate change, although the impact that will have on productivity is unknown (Burnell *et al.* 2023). Furthermore, adult survival of Manx Shearwaters may be negatively affected by severe weather conditions during the breeding season, thus highlighting the potentially increasing costs of reproduction if extreme weather events become more frequent (Wood *et al.* 2021). ▶

Table 9: Seabirds Count census results

	Abundance (AOS) *Seabird 2000* (1998–2002)	Abundance (AOS) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	336,538	921,618	174

Artificial light at night can attract fledged chicks, causing them to land on flat ground where it is hard for them to take off again, making them more vulnerable to predation e.g. in Scotland (Syposz *et al.* 2018) and the Canary Islands (Rodriguez & Rodriguez 2009). Adult Manx Shearwaters can also be affected by light pollution, e.g. lights from buildings near nest sites in foggy conditions can cause collisions (Guildford *et al.* 2018). At sea, pollution pressures such as oil spills can be detrimental (Votier *et al.* 2005). A recent study identified 68% of adults and 75% of fledglings had at least one piece of plastic in their stomach contents (Alley *et al.* 2022). The potential impact this has on mortality is unknown in this species.

Also unknown is the population-level impact of Puffinosis, a disease that initially causes blistering of Manx Shearwater feet but which can progress to death (Esmonde *et al.* 2022). The cause of this infection is still being investigated, but a recent study suggests it is likely due to damp nesting burrows causing opportunistic bacterial infections (Esmonde *et al.* 2022).

CONSERVATION

Various islands around the UK coast have had dedicated eradication programmes for invasive predator species, typically targeting rats. The removal of predators from these islands has often led to the successful return and breeding of Manx Shearwater. Lundy Island and some of the Isles of Scilly archipelago (both England), and the Calf of Man (Isle of Man), have all seen an increase in their Manx Shearwater breeding populations since rats were eradicated (JNCC 2021).

Additional eradication programmes may benefit Manx Shearwaters further. It is also critical that effective biosecurity measures and continued monitoring occurs on islands that have undergone successful eradication programmes, or that are currently free from invasive predators to prevent them reaching these places. ■

MANX SHEARWATER SURVEYING, BARDSEY ISLAND BY DAWN BALMER / BTO

Storm Petrel

Hydrobates pelagicus

c.27–35%

Abundance: n/a
Productivity: n/a

Amber-listed
Amber-listed (I)

Colony Count sites: 7
Breeding Success sites: 0

Least concern

Lifespan: 11 years
Breeding age: 4 years

Britain and Ireland host between 27% and 35% of the world's breeding Storm Petrel population (Burnell *et al.* 2023). Storm Petrels are the smallest Atlantic flying seabird, weighing on average 25 g, and the maximum age known from bird ringing records is 38 years and 17 days, set in 2017 (BTO 2025).

DISTRIBUTION

In Britain and Ireland, Storm Petrels breed mainly on offshore islands on the Atlantic fringe in the north and west. They are wide-ranging foragers and can be seen at sea around the British and Irish coastlines throughout the breeding season (BTO 2025).

Globally, the subspecies *pelagicus* breeds widely across small islands in the North Atlantic Ocean (BirdLife International 2024).

Storm Petrels typically migrate out to sea for the winter within the Atlantic Ocean (Militão *et al.* 2022). Bird ringing recoveries have shown some individuals have travelled round the Cape of Good Hope (South Africa), and into the Indian Ocean as far as Mozambique (BTO 2025).

DIET

Storm Petrels forage by pattering with their feet on the surface of the water and pecking up prey items. These include small fish, squid and crustaceans, and they have also been known to feed on jellyfish and fishery discards (del Hoyo *et al.* 1992).

BREEDING

Storm Petrels spend the majority of their life at sea, only coming to land for a few months of the year to breed (BTO 2025). They are nocturnally active at colonies and nest in crevices or burrows, laying a single egg (Bolton *et al.* 2010, BTO 2025). Their breeding colonies are generally restricted to islands free from mammalian predators, as well as some isolated headlands where there is a reduced predation risk (de León *et al.* 2006).

BREEDING ABUNDANCE

Too few Storm Petrel colonies are monitored in Britain and Ireland to enable the production of valid annual breeding abundance trends due to the challenges in monitoring a burrow-nesting species which breeds in remote locations and is only active around their colonies at night. New technologies have been trialled at some colonies to improve colony population estimates. If these become a practical survey method in the future, more regular SMP monitoring of colony abundance and the production of abundance trends may be possible.

The results from the recent *Seabirds Count* census indicate that across Britain and Ireland the Storm Petrel population is thought to have moderately increased in abundance since *Seabird 2000* (Table 10). However, caution is required as the confidence intervals of the population estimates from the *Seabird 2000* and *Seabirds Count* censuses overlap (Burnell *et al.* 2023).

PRODUCTIVITY

Insufficient Storm Petrel colonies are monitored to produce valid SMP productivity trends due to the difficulties involved in monitoring breeding success in this burrow and crevice nesting species. Therefore, very limited productivity data have been submitted to the SMP since 1986. Regular annual monitoring of productivity at colonies across their range would be required to allow productivity trends to be produced in the future.

STORM PETREL, BY JONATHAN DODDS

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Storm Petrels face a range of pressures, one of the major threats being predation. Rat and Domestic Cat predation, for example on the Isles of Scilly (England), has caused population declines and also, as on Orkney and Shetland (Scotland), influenced where Storm Petrel breed (de León *et al.* 2006, Heaney *et al.* 2002).

Seabirds such as Great Skuas (Deakin *et al.* 2018) and gulls (Hey *et al.* 2020) will opportunistically predate Storm Petrels, and this is one of the reasons they are nocturnally active at their colonies. Once leaving the nest, artificial light at night can also attract fledged chicks, causing them to land on the ground where they become more vulnerable to predation (Rodriguez & Rodriguez 2009).

Nest site destruction, caused by trampling by humans or livestock, erosion or disturbance, may also impact productivity in some locations (Cadiou *et al.* 2010, Mitchell *et al.* 2004).

The effects of pollution, climate change, wind farms and the policy changes to end fishery discards on Storm Petrel populations are not well understood, but all have the potential to place additional pressure on this species (Burnell *et al.* 2023).

CONSERVATION

Eradication programmes have been successful in providing rat-free habitat for Storm Petrels to breed again on the Isles of Scilly, Lundy (England), Ramsey Island (Wales), and the Shiants (Scotland) (Heaney *et al.* 2002, Tucker & Heath 1994). Biosecurity measures are also key to ensuring islands remain invasive predator-free (Burnell *et al.* 2023).

The successful uptake of provided nest boxes has benefited Storm Petrel in some locations by providing additional breeding habitat e.g. on Skokholm (Wales), the Shiants and Isles of Scilly (Burnell *et al.* 2023).

The use of GPS tags to monitor Storm Petrel behaviour and movements at sea during the breeding season has identified key foraging areas north of Scotland (Bolton 2021). Statutory protection of these areas in future could potentially be beneficial for this species. ■

Table 10: *Seabirds Count* census results

	Abundance (AOS) *Seabird 2000* (1998–2002)	Abundance (AOS) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	125,722	147,578	17

Leach's Petrel

Hydrobates leucorhous

<1%

Abundance: n/a
Productivity: n/a

Red-listed
Red-listed (I)

Colony Count sites: 0
Breeding Success sites: 0

Vulnerable

Lifespan: 13 years
Breeding age: 5 years

Zero coverage in 2024

Britain and Ireland host fewer than 1% of the world's breeding Leach's Petrel (Burnell *et al.* 2023). In the BoCC5 seabird update of 2024, Leach's Petrel moved from Amber to Red-listed due to declines in their breeding populations, restricted breeding range, global status and the international importance of the UK population (Stanbury *et al.* 2024). Their very restricted range in Britain and Ireland is thought to be limited by the distance from their main foraging grounds at, or beyond, the continental shelf (BTO 2025).

DISTRIBUTION

Only coming to land to breed, Leach's Petrel spend the rest of the time in remote parts of the ocean, converging on upwellings or over the continental shelf edge (Hedd *et al.* 2018, Pollet *et al.* 2014).

They are known to breed on just 11 offshore islands and archipelagos along the Atlantic fringe of Britain and Ireland, all but three of which are in Scotland (Burnell *et al.* 2023).

Globally, Leach's Petrel are found across the Atlantic and Pacific Oceans (BirdLife International 2024). Outside of the breeding season, they can be seen offshore or even blown inshore during autumn and early winter gales (BTO 2025).

Leach's Petrel in the North Atlantic migrate south for the winter as far as Brazil and South Africa (del Hoyo *et al.* 1992, Pollet *et al.* 2019).

DIET

Leach's Petrel are surface feeders, with their diet including small fish, squid and planktonic crustaceans as well as discards from fishery boats (Watanuki 1985, Hedd & Montevecchi 2006). They are also sometimes seen following marine mammals, feeding on leftovers or faeces (IUCN 2025).

BREEDING

Leach's Petrel avoid predation by only visiting colonies nocturnally. Nest sites are found on predator-free offshore islands, and they nest in burrows, under vegetation and in rocky crevices where they lay a single egg (Burnell *et al.* 2023, del Hoyo *et al.* 1992).

During the breeding season, they can travel up to 1,000 km from their colonies, flying over deep parts of the ocean in search of food (Hedd *et al.* 2018, Pollet *et al.* 2014).

Table 11: *Seabirds Count* census results

	Abundance (AOS) *Seabird 2000* (1998–2002)	Abundance (AOS) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	48,357	10,765	-78

LEACH'S PETREL, BY JOE PENDER

BREEDING ABUNDANCE

Very few Leach's Petrel colonies have been monitored over the SMP recording period due to the difficulties in surveying this nocturnal, burrow-nesting species, which only breeds in a small number of remote locations across Britain and Ireland. Consequently, no valid annual abundance trends can be produced. New technologies have been trialled at some petrel colonies to improve colony population estimates. If these become a practical survey method in the future, more regular SMP monitoring of colony abundance and the production of abundance trends may be possible.

Although some methodological differences between censuses make direct comparisons difficult, the *Seabirds Count* census indicates that the Leach's Petrel population has undergone a severe decline of 78% across Britain and Ireland since *Seabird 2000* (Table 11 & Burnell *et al.* 2023). The majority of Leach's Petrel breeding in Britain and Ireland nest on St. Kilda (Scotland) where comparable analytical methods indicate a decline of 68% between 2000 and 2019 (Deakin *et al.* 2021).

PRODUCTIVITY

SMP data on Leach's Petrel breeding success is very limited given the difficulties in monitoring productivity in this burrow and crevice nesting species, therefore, no valid annual productivity trends can be produced. Regular annual monitoring of productivity at colonies would be required to allow productivity trends to be produced in the future.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Native wildlife has the potential to prey on Leach's Petrels. On St Kilda, research suggests that the population has been predated at an unsustainably high level by Great Skuas, and that this, along with other factors such as changes in food supply, has contributed to the decline recorded (Newson *et al.* 2008). Similarly, predation by native species in Canada has been highlighted as an important driver of population dynamics of Leach's Petrels. In Gull Island, Newfoundland, for example, 118,000–143,000 petrels were predated by American Herring Gulls *Larus smithsonianus* in 2012 (Bond *et al.* 2023).

There is some evidence to suggest that grazing can be detrimental to Leach's Petrel (Drury 1973, d'Entremont *et al.* 2022) through modification of the habitat and the risk of trampling. This is supported by the fact that Dun on St Kilda (Scotland), which is ungrazed, has relatively high breeding densities nesting within the dense tussocky grass which has developed there (Burnell *et al.* 2023), and this may also offer additional protection by being unfavourable nesting habitat for the predatory Great Skuas (Miles 2010).

At-sea pollution can be a problem for Leach's Petrel and it is thought that Leach's Petrels are attracted to the flares and lights of oil rigs, which could lead to fatal collisions (Collins *et al.* 2022, Hedd *et al.* 2018). Oil spills are also a concern. There are at least five colonies from the western Atlantic which overlap with offshore oil and gas operations, and three of these colonies have declined in recent years (Hedd *et al.* 2018).

Shifts in prey distributions due to climate change are also considered potential threats to Leach's Petrel (Burnell *et al.* 2023, Pollet *et al.* 2023).

CONSERVATION

Researchers suggest that local, targeted strategies are likely to be the best approach for the conservation of Leach's Petrel (Pollet *et al.* 2023). Invasive species such as mice, rats, Domestic Cats, and Red Fox (*Vulpes vulpes*) have the potential to create predation risk pressures at breeding colonies (Dias *et al.* 2019, Mitchell *et al.* 2004), therefore preventing potential mammalian predator species from reaching the main Leach's Petrel breeding sites through effective biosecurity programmes will be a key conservation measure for this species (Burnell *et al.* 2023). ■

Gannet

Morus bassanus

c.70%

Abundance: n/a
Productivity: 0.53

Amber-listed
Amber-listed (I)

Colony Count sites: 9
Breeding Success sites: 8

Least concern

Lifespan: 17 years
Breeding age: 5 years

The breeding population of Gannet in Britain and Ireland represents 70% of the global total (Burnell *et al.* 2023). The Gannet's iconic plunge-dive, by which they can reach depths of up to 20 m (Garthe *et al.* 2000), makes them a striking feature of British and Irish coastlines.

DISTRIBUTION

Gannets are found across the North Atlantic, and currently are known to breed at 28 colonies around the British and Irish coast (Burnell *et al.* 2023).

They are a wide-ranging species. Globally, Gannets breed from the north-west of Russia, up to the island of Bjørnøya, west to Canada and south to Brittany (BirdLife International 2024). A few pairs also nest in the Mediterranean (Giagnoni *et al.* 2015).

Post-breeding, Gannets move south, with east Atlantic birds typically going to the Bay of Biscay or off the coast of West Africa, and some individuals crossing south of the Equator (Burnell *et al.* 2023, Kubetzki *et al.* 2009).

DIET

Gannets feed on a wide range of pelagic fish and squid (Garthe *et al.* 2000), and these are predominantly caught by plunge-diving. They also scavenge fishery discards. Breeding birds have colony-specific foraging ranges based on density-dependent competition, i.e. birds in the largest colonies forage further from the nest (Hamer *et al.* 2001, Wakefield *et al.* 2013).

BREEDING

Gannets breed on offshore islands and stacks, as well as some mainland cliffs, nesting on small mounds built just outside the pecking range of surrounding pairs. Nests are made from seaweed, terrestrial vegetation and marine debris, in which a single egg is laid (BTO 2025).

BREEDING ABUNDANCE

Up until 2019 an SMP abundance trend was estimated for Gannet at the UK level based on interpolated and extrapolated values from complete censuses since the 1980s (JNCC 2021). This approach was taken, rather than using SMP data, as annual sampling of Gannet is typically only carried out at the smaller, more accessible colonies, which are not representative of the overall trend. Index values had been extrapolated each year as the Gannet population of Britain and Ireland had been experiencing a long-term increase, with the *Seabirds Count* census showing that the population increased by 38% since the previous Gannet census in 2003–05 (Table 12 & Burnell *et al.* 2023). However, given the extent to which Gannet were negatively impacted by the 2022 HPAI outbreak, no extrapolation of the SMP abundance trend was carried out between 2021 and 2024. Too few representative data are submitted to the SMP on abundance of Gannet populations from individual regions to allow for the production of valid abundance trends.

Table 12: *Seabirds Count* and *Gannet census* results

	Abundance (AOS/AON) *Gannet Census* (2003–2005)	Abundance (AOS/AON) *Gannet Census* (2013–2014) & *Seabirds Count* (2015–2021)	Population Change
All Britain, Ireland, Isle of Man and Channel Islands	262,065	360,748	**38**

However, we are able to show the abundance of Gannets from key colonies in England, Wales and the Channel Islands.

Only one Gannet colony exists in England, at Bempton Cliffs in Humberside. The colony increased steadily since its formation in the 1960s until 2005, after which there was a steep increase up to 2017 when 13,392 AOS/AON were recorded (Figure 16). Bempton Cliffs was hit by HPAI in 2022, which is likely responsible for the lower count in that year. The colony has continued to increase in the last two years with 15,794 AOS/AON recorded in 2024.

There is also only one established Gannet colony in Wales, on Grassholm in Pembrokeshire, although single pairs have occasionally set-up territories elsewhere. Since 1986, nest counts have been made from aerial photographs, which have varied in the quality of their resolution and coverage. In 2009, using high resolution digital images, 39,282 AOS/AON were recorded (Figure 17). In 2022, Grassholm was severely hit by HPAI leading to a large drop to 16,482 AOS/AON in 2023. In 2024, a total of 19,199 AOS/AON were recorded.

Two Gannet colonies occur in the Channel Islands, on Ortac and Les Etacs, which became established between 1940 and 1945. Since 1987, Les Etacs increased steadily up until 2016, whilst numbers on Ortac have

▶

Figure 16: Total Bempton Apparently Occupied Nests (1986–2024)

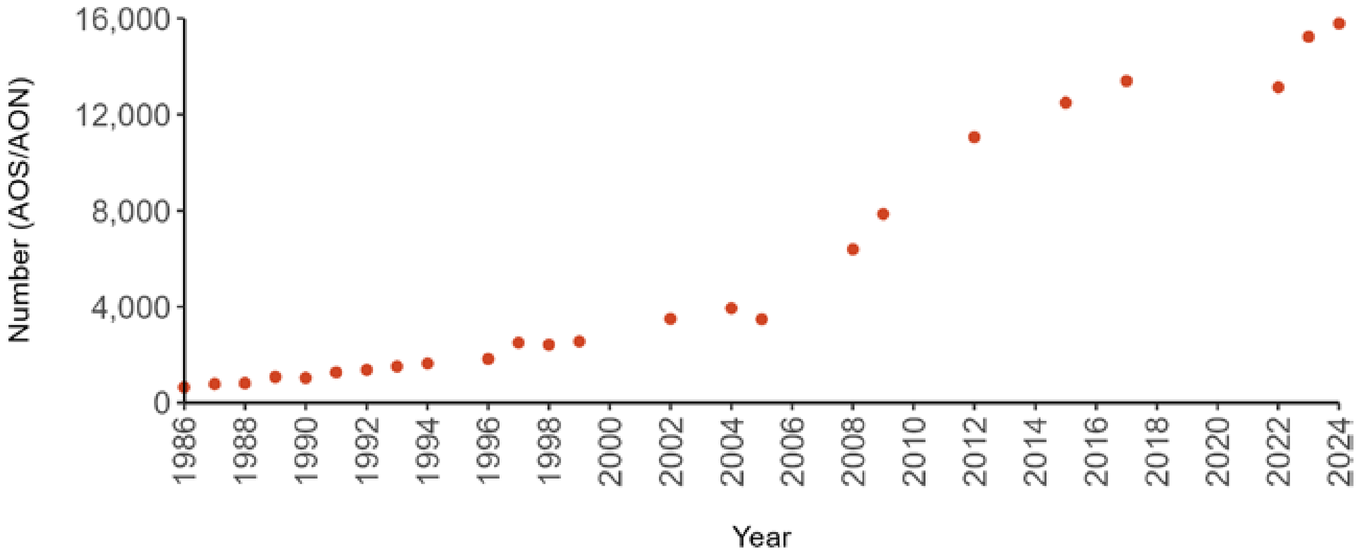


Figure 17: Total Grassholm Apparently Occupied Nests (1986–2024)

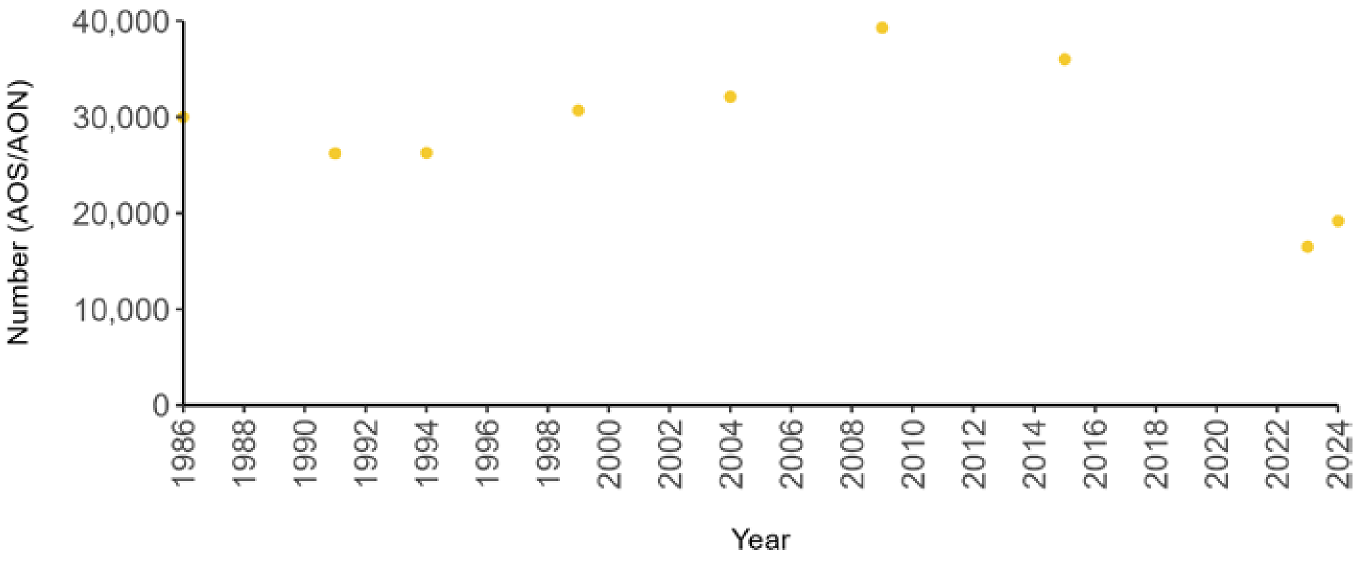
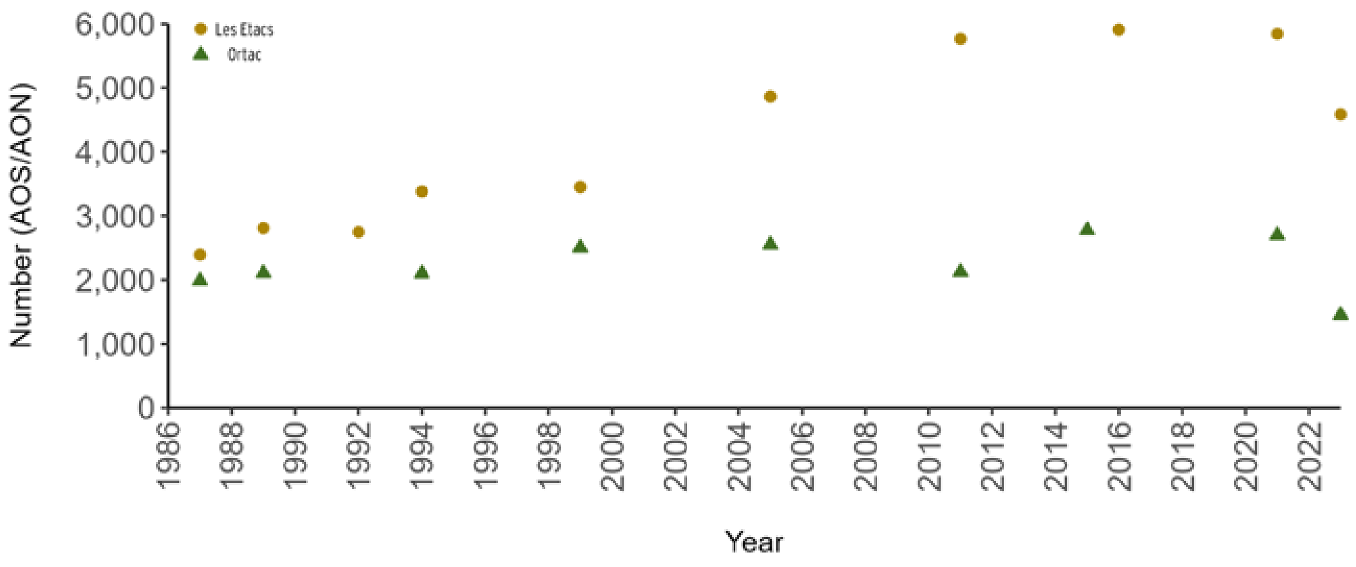


Figure 18: Total Ortac & Les Etacs Apparently Occupied Nests (1987–2023)



been relatively stable over the same period (Figure 18). In 2023, declines were recorded at both colonies, to 4,586 AOS/AON on Les Etacs and 1,451 AOS/AON on Ortac, likely due to the 2022 HPAI outbreak. No data were submitted from the Channel Islands in 2024.

PRODUCTIVITY

The Gannet productivity trends for the UK and Scotland are closely matched, as a large proportion of monitored sites are in Scotland (Figure 19). The trends have been relatively stable in recent years, but 2022 saw the lowest values recorded since SMP monitoring began, with 0.31 and 0.22 chicks fledged per breeding pair in the UK and Scotland, respectively (Tables 2 & 4). This poor productivity is attributable to the 2022 HPAI outbreak, and was probably due to nest abandonment by adults who either died or deserted the breeding colony (Lane *et al.* 2023). Productivity estimates in 2024 were 0.53 and 0.64 chicks fledged per pair in the UK and Scotland, respectively (Table 13). Too few productivity data are submitted to the SMP from the other regions with breeding Gannet to allow for the production of valid annual productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Influencing factors for historical population increases include the cessation of human exploitation during the 19th century and the frequency of oil pollution events decreasing, although mass events still pose a threat (Burnell *et al.* 2023).

Eggshell thinning caused by dichloro-diphenyl-trichloroethane (DDT) resulted in lower productivity in the 1950s and 1960s; this now poses little cause for concern (Chapdelaine *et al.* 1987, Power *et al.* 2021).

Following decades of population growth, the HPAI outbreak caused a severe population decline of 25% at sites surveyed across the UK in 2023 (which covered 75% of the UK breeding population) compared to pre-HPAI baseline counts undertaken between 2014 and 2021 (Lane *et al.* 2023, Tremlett *et al.* 2024b).

Due to their large foraging range and wide range of prey species, Gannets have been less affected than many other seabirds by climate change (Johnston *et al.* 2021). However, increasing sea surface temperatures due to climate change have resulted in a northwards shift in fish prey distribution, especially for Atlantic Mackerel (*Scomber scombrus*) (Montevecchi 1997).

Gannets breeding at the southern limit of their range are particularly at risk in years with higher sea surface temperatures, as they may have to travel further to forage, and a reduction in available prey has been shown to result in fewer chicks fledging (d'Entremont *et al.* 2022).

Plastics have been increasingly incorporated into Gannet nesting materials, causing some birds to become entangled in their nests (O'Hanlon *et al.* 2019). The effects of plastic ingestion on Gannets are largely unknown (Burnell *et al.* 2023).

Across their range, Gannets are common bycatch in longline and fixed gear fisheries, as they often associate with fishing boats during foraging trips (Araújo *et al.* 2022, Barcelona *et al.* 2010, Smith & Morgan 2005), and they have been reported as the most frequently killed bird in bycatch from Portuguese coastal Atlantic waters (Oliveira *et al.* 2015).

A tracking study showed that Gannets may avoid offshore wind farms, especially during the breeding season (Peschko *et al.* 2021) and displacement effects may reduce available foraging areas. However, they have also been identified as highly vulnerable to collisions with offshore wind farms due to their flight height (Bradbury *et al.* 2014, Furness *et al.* 2013).

During autumn migration, adult Gannets are at higher risk than juveniles, as young Gannets tend to hug the coastline and thus interact with wind farms less frequently (Pollock *et al.* 2021).

CONSERVATION

Fishery-related policy actions have the potential to benefit Gannet populations. For example, gillnet fishery closures in Canada in 1992 led to an increase in breeding populations of Gannet due to the removal of tens of thousands of gillnets known to inflict high levels of seabird mortality through fisheries bycatch (Regular *et al.* 2013).

Bird deterrent technologies have the potential to reduce bycatch, one example being 'scarybird' (a visual deterrent deployed above fishing nets to keep away species that are vulnerable to being trapped) which was trialled off the coasts of the Berlengas Islands (Portugal) and discouraged Gannets and large gulls from fishing vessels during operations (Almeida *et al.* 2023).

Monitoring plastics and other debris in Gannet nests could provide a useful indication of the effectiveness of any future fisheries-related policy actions put in to place to reduce fisheries-related plastic pollution (O'Hanlon *et al.* 2019).

Continued monitoring and understanding of population-level impacts of HPAI, particularly since 2022, will also be important moving forward (Burnell *et al.* 2023).

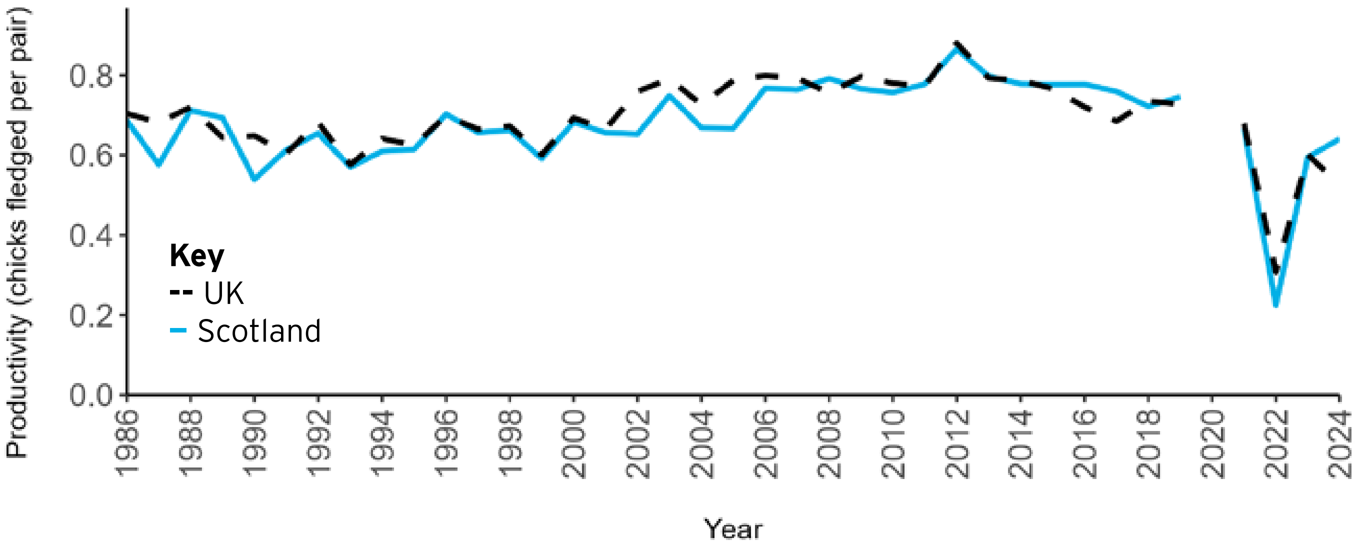
The current development of new methods to automate the detection and counting of Gannets from images taken from Uncrewed Aerial Vehicles (UAVs) is likely to be beneficial for monitoring Gannet numbers in the future. Similar techniques proved useful in monitoring Gannet numbers on the Bass Rock (Scotland) following the 2022 HPAI outbreak (Tyndall *et al.* 2024). ■

Table 13: SMP Productivity

	Productivity	
	2024	Sites
UK	0.53	7
Scotland	0.64	5

GANNET, BY EDMUND FELLOWES / BTO

Figure 19: SMP Productivity (1986–2024)



Cormorant

Phalacrocorax carbo

c.3–4%
ssp. carbo/sinensis

Abundance: Stable
Productivity: n/a

Green-listed
Amber-listed (I)

Colony Count sites: 58
Breeding Success sites: 7

Least concern

Lifespan: 11 years
Breeding age: 3 years

Britain and Ireland host a minimum of 2% of the global breeding population of Cormorant (Burnell *et al*. 2023). Two subspecies of Cormorant, *carbo* and *sinensis*, breed in Britain and Ireland. Throughout Ireland and in coastal Britain, the predominant subspecies is *carbo*, whereas both subspecies are present in breeding colonies in inland areas of Britain. Combined, these subspecies total approximately 3–4% of the global *carbo* and *sinensis* subspecies (BirdLife International 2024).

DISTRIBUTION

Since 1981, Cormorants in Britain and Ireland, thought of as primarily coastal birds, have increasingly used inland lowland lakes and rivers throughout the year and also established successful breeding colonies inland. It is thought that these inland colonies were initially established by the immigration of a subspecies from Continental Europe, *P.c. sinensis*, although some inland sites are currently also populated by the subspecies *P.c. carbo* (Newson *et al.* 2013).

Today, breeding colonies are widely distributed around the coasts of Britain and Ireland, and many inland colonies occur in England and the Republic of Ireland (Burnell *et al.* 2023). Cormorant breeding colony locations may remain constant for long periods, but can also suddenly shift in location. (JNCC 2021, BTO 2025).

With the exception of South America and Antarctica, Cormorants are found across the globe (del Hoyo *et al.* 1992). Their movements vary from being preferentially sedentary, dispersing locally, or having longer migrations (del Hoyo *et al.* 1992).

DIET

Cormorants are generalists, feeding on a range of fish which they catch by pursuit diving (Grémillet *et al.* 2003). This includes species that are popular with anglers, such as Atlantic Salmon (*Salmo salar*) and Sea Trout (*Salmo trutta trutta*). Legal control of Cormorants is permitted for a limited number of individuals under specific licences issued by the Statutory Nature Conservation Bodies.

BREEDING

Coastal nesting birds, predominantly of the subspecies *carbo*, can be found on stacks, rocky islets, cliffs or rocky promontories. Inland, nesting Cormorants of either subspecies (*carbo* or *sinensis*), favour lakes and rivers, nesting in trees, bushes, reedbeds or on bare ground, and artificial structures may also be used. Depending on the location, nest type ranges from a simple depression to a large platform, often using sticks, reeds and seaweed where 3–4 eggs are laid (Bregnballe *et al.* 2014, del Hoyo *et al.* 1992, Newson & Austin 2021).

BREEDING ABUNDANCE

The Cormorant abundance changes for the UK and Ireland, reported by the *Seabirds Count* census, indicated stable populations since 2000 (Table 15 & Burnell *et al.* 2023). The 24-year SMP change for England also indicates a stable population, whilst the change for the UK indicates a decline of 16% (Table 14). For Wales, the *Seabirds Count* census results report a decline of 17% since the *Seabird 2000* census (Burnell *et al.* 2023), whilst the 24-change SMP trend indicates a stable population (Table 14).

The UK SMP abundance index for Cormorant (which includes inland and coastal breeders) has fluctuated over the whole SMP reporting period, with several periods of increase followed by decline (Figure 20). The lowest value was recorded in 2013, when the index was approximately 10% below the baseline, although subsequently it increased markedly and in 2022 was almost 30% above the baseline. In 2024, however, the UK index declined to 8% below the baseline.

CORMORANTS, BY JOHN HARDING / BTO

Cormorant abundance trends for England and Wales have also fluctuated across the SMP reporting period (Figures 21 & 22). In 2024, the England index fell to 26% above the baseline. However, there is high uncertainty (reflected by the wide confidence intervals) around the England trend between 2021 and 2024, due to fewer sites monitored compared to previous years (Figure 21). The abundance index for Wales declined to 29% below the baseline in 2024 (Figure 22).

Too few data are currently submitted to the SMP from other regions to allow for the calculation of valid abundance trends.

PRODUCTIVITY

An insufficient number of Cormorant colonies are monitored across all regions to allow for the production of valid annual productivity trends.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Licensed lethal control and shooting to scare off piscivorous birds, such as Cormorants, are legal actions that may be used to support salmonid conservation. Analysis of the Welsh Cormorant population viability highlighted that the majority of predicted outcomes of different levels of lethal control were population declines, which could impact on their conservation status (Macgregor *et al.* 2022). In England, 2,614 individuals on average were killed under licence each year between 2015/16 and 2018/19, which is suggested to be a major threat to populations there (Newson & Austin 2021). However, population level impacts of control are difficult to quantify.

Disturbance at inland colonies and erosion of cliffs at coastal locations can influence availability of suitable nesting habitat, as can climate-related flooding of nest sites through storms or extreme rainfall (Burnell *et al.* 2023).

Cormorants are also identified as being susceptible to becoming bycatch in gillnets, longlines and trammel net fisheries during operations. It is thought this could have a population level impact (Bregnballe & Frederiksen 2006).

CONSERVATION

Cormorant populations could potentially benefit through the ▶

CORMORANTS, BY PHILIP CROFT / BTO

reduction of conflicts with anglers. A recent paper from Finland highlights that effective stakeholder engagement at a local scale will be important for the continued conservation of Cormorants in light of the conflicts faced with anglers (Nordberg & Salmi 2019). ■

Table 14: SMP Breeding Abundance Change

	Seabirds Count	Breeding Abundance Change %		
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)
UK	8,829	49	-8	-16
England	3,333	22	26	-8
Wales	1,477	8	-29	-7

No significant changes

Table 15: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	14,000	13,330	-5

Figure 20: UK SMP Breeding Abundance (1986–2024)

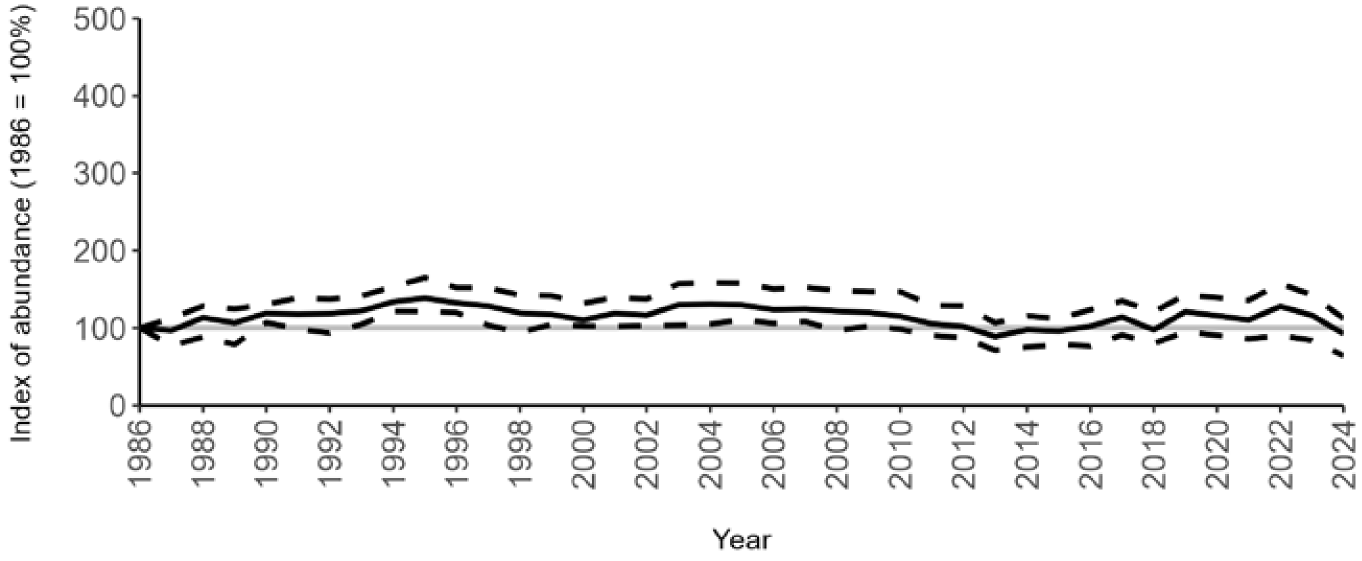


Figure 21: England SMP Breeding Abundance (1986–2024)

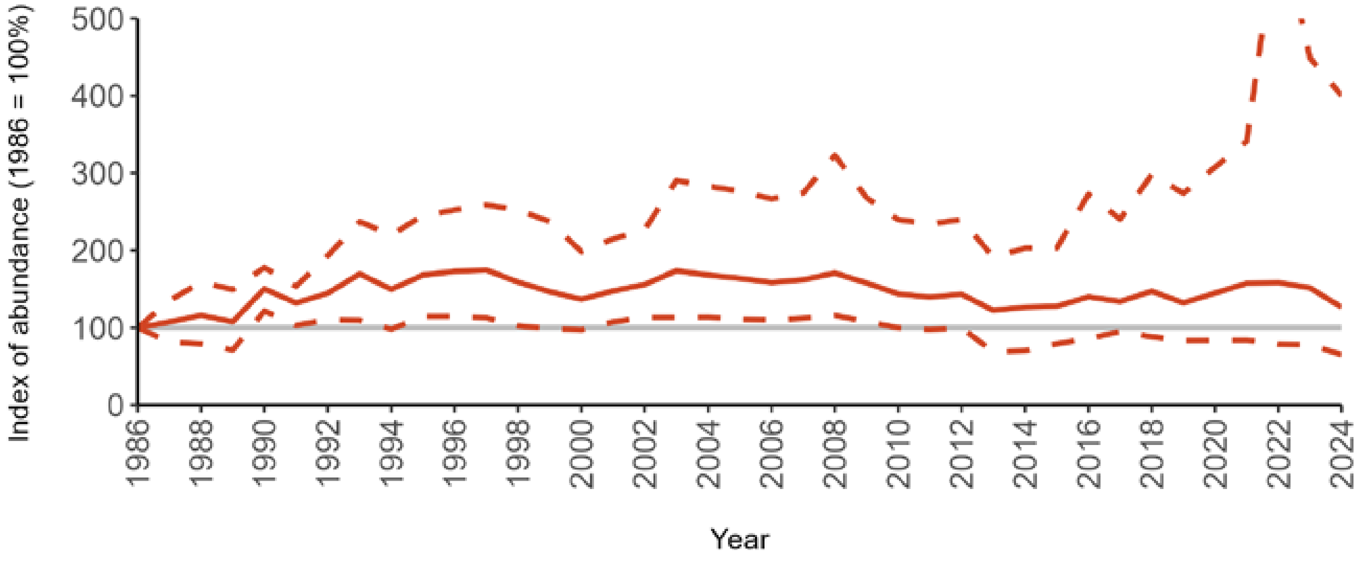
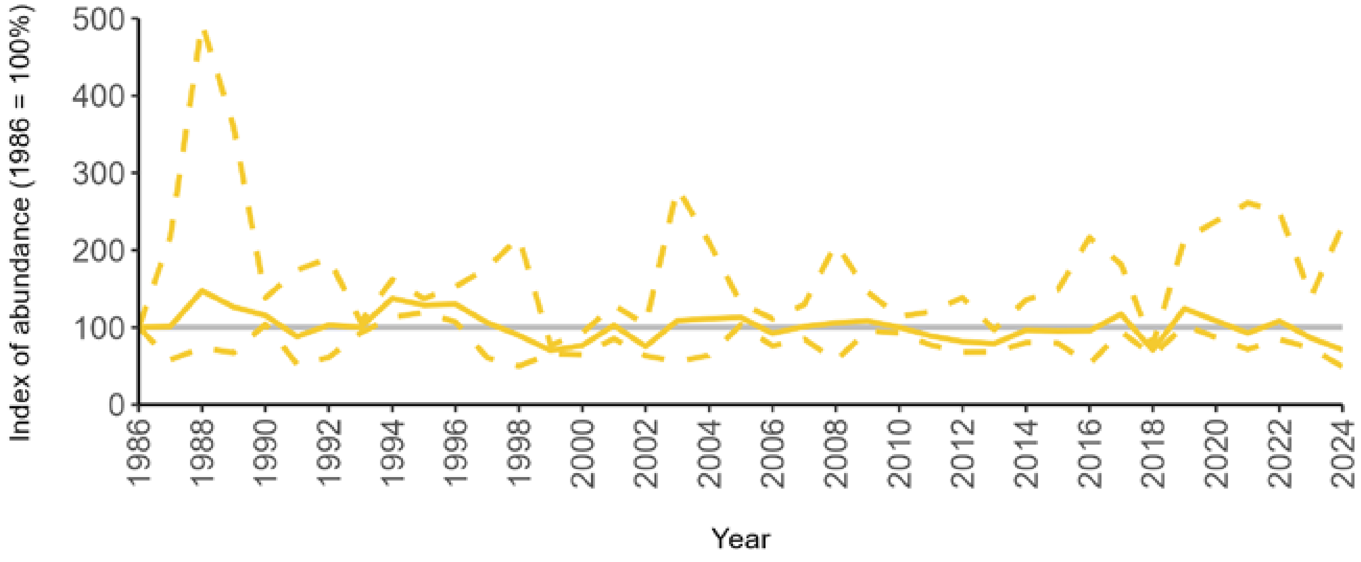


Figure 22: Wales SMP Breeding Abundance (1986–2024)



Shag

Gulosus aristotelis

c.38%
ssp. aristotelis

Abundance: Decline
Productivity: 0.93

Amber-listed
Amber-listed (I)

Colony Count sites: 134
Breeding Success sites: 20

Least Concern

Lifespan: 12 years
Breeding age: 4 years

Britain and Ireland host between 22 and 23% of the world's breeding Shags, and approximately 38% of the subspecies *aristotelis* (Burnell et *al.* 2023). Long-term monitoring of Shag at their breeding colonies has revealed they can be long-lived, with the oldest Shag record being 29 years, 10 months and 25 days (BTO 2025).

DISTRIBUTION

In Britain and Ireland, Shag breeding colonies are mainly found on northern and western coastlines, where suitable cliffs are present (Burnell *et al.* 2023). They are endemic to the north-east Atlantic, Mediterranean, Black Sea and north-west African coasts (BirdLife International 2024). The north and west of Europe is home to the nominate subspecies, *aristotelis* (Gill *et al.* 2023).

Shag are not long-distance migrants, instead dispersing within their range during the non-breeding season (del Hoyo *et al.* 1992). This is true for British and Irish breeding birds, which remain around the coastline, occurring in particularly high densities in northern and western Scotland and western Ireland (BTO 2025).

DIET

Their diet is linked to local prey availability and consists of a range of small fish species, such as sandeel and gadids. Shag forage for prey in open water, with benthic dives over sandy and rocky substrate (Harris & Wanless 1991, BTO 2025).

BREEDING

Shags nest in colonies, ranging in size from a few pairs to several thousand. Nesting sites are generally found on rocky coastlines and islands, in boulder fields, on ledges or in caves, and they lay between one and four eggs (Burnell *et al.* 2023, Wanless & Harris 1997).

BREEDING ABUNDANCE

The Shag abundance changes for the UK, Scotland and Wales, reported by the *Seabirds Count* census, indicated declining populations of between -22 and -29% since 2000 (Burnell *et al.* 2023). In contrast, the 24-year SMP changes for the UK and Scotland show more severe index declines of -65 and -62%, respectively (Table 16). The stable 24-year SMP value of -1% for Wales (Table 16) differed from the 29% decline reported between *Seabird 2000* and *Seabirds Count* (Burnell *et al.* 2023). The marked decline in the Scottish index in 2024 may have been due to the wrecks observed there during the autumn and winter of 2023, with the second lowest adult return rates on record reported on the Isle of May in 2024 (Newell *et al.* 2025).

In recent years, the SMP long-term UK trend had been increasing slightly, following a long period of overall decline. However, in 2024 the index value dropped to 71% below the 1986 baseline (Figure 23). The SMP abundance trend for Scotland (Figure 24) closely matches the UK trend, as many of the colonies monitored are located in Scotland, with a similar decline in the index value to 70% below the 1986 baseline in 2024. The UK and Scotland SMP long-term trends contrast with that for Wales, which has largely fluctuated around the baseline since the late 1990s (Figure 25). The Wales index value in 2024 was 7% below the 1986 baseline (Table 16).

Too few data are submitted to the SMP from other regions to allow for the production of valid abundance trends.

Table 16: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	20,209	116	-71*	-65*	**0.93**	17
Scotland	16,788	74	-70*	-62*	**0.83**	10
Wales	651	16	-7	-1	**1.59**	2

** significant changes*

Table 17: Seabirds Count census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	32,324	25,961	-20

PRODUCTIVITY

The number of sites monitored for productivity in Scotland make up the majority of the UK sample, resulting in both regions following a similar, relatively stable, trend since 1986 (Figure 26). In 2024, the productivity estimates for the UK and Scotland declined to 0.93 and 0.83 chicks fledged per pair, respectively (Table 16). There is greater fluctuation in the Welsh productivity trend across the SMP monitoring period (Figure 26), although this has largely remained higher than the UK and Scotland trends. The productivity estimate was particularly low for Wales in 2021, with just 1.13 chicks fledging per pair. In 2024 the productivity estimate remained similar to the 2023 (1.63) estimate at 1.59 chicks fledged per breeding pair (Table 16).

Data submitted to the SMP on the productivity of Shags from other regions are sparse, so no meaningful productivity values can be given.

PHENOLOGY, DIET AND SURVIVAL RATES

No systematic data on phenology have been collected as part of the SMP. However, diet information has been collected for Shags at the Key Sites of Canna and the Isle of May (Scotland) and adult return rates are estimated for Shags on the Isle of May and are published in the Key Site reports.

CAUSES OF CHANGE

Climate change is indirectly impacting seabird populations through temperature-mediated changes in prey populations (Johnston *et al.* 2021). As a result, Shag diets have changed, adapting to the reduction in available key prey (Howells *et al.* 2018). The situation is complicated by regional variations in prey abundance and local adaptation to changes, making overall impact hard to assess.

A further impact of climate change on this species is the increased frequency of extreme weather events. When there is a sustained period of strong onshore winter winds, this can result in a wreck, where seabirds are unable to feed and are washed ashore, dead or dying (Newell *et al.* 2015). Several pronounced wrecks over the last few decades have involved significant mortality of Shags (Harris & Wanless 1996), the most recent being in the 2023/2024 winter in the North Sea where, for example, the breeding population of the Isle of May was reduced by several hundred pairs. The impacts on Shag populations is cause for concern given that the frequency of such events is predicted to increase (Rahmstorf & Coumou 2011).

While the population-level impacts of plastics on Shag remains a topic of research, plastics have been found to occur in their pellets, e.g. 63% of pellets collected from a Shag colony in north-west Spain contained plastic (Álvarez *et al.* 2018).

An additional potential pressure on the Shag population is accidental bycatch in fisheries (Northridge *et al.* 2020).

It is possible that Shags benefit from offshore wind farms; individuals from colonies in the Irish, North and Baltic Seas use turbine bases as places to rest between foraging bouts (Dierschke *et al.* 2016).

CONSERVATION

As for many other seabird species, measures implemented to reduce climate change, incidental seabird bycatch in the fishing industry and pollution of the marine environment are likely to benefit the UK Shag population.

Only 30% of the British and Irish population of Shag breeds within SPAs (Burnell *et al.* 2023). Designation of additional protected areas could also have benefits for this declining species. Additional studies to identify important non-breeding areas could also prove valuable (Burnell *et al.* 2023). ▶

Figure 23: UK SMP Breeding Abundance (1986–2024)

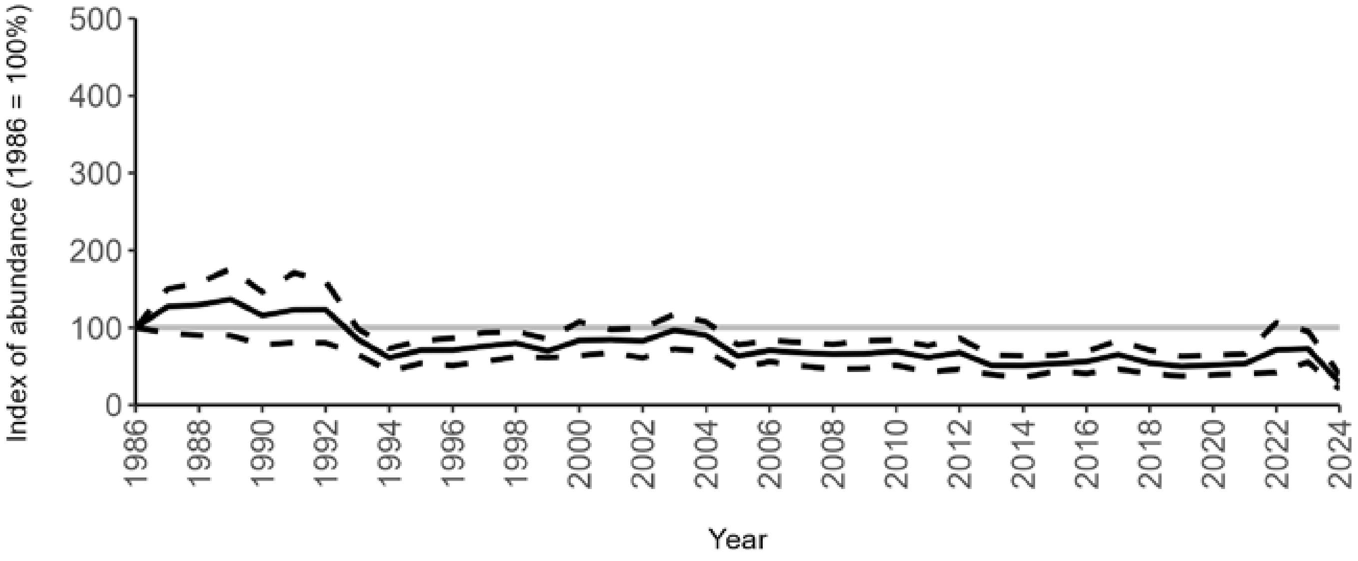


Figure 24: Scotland SMP Breeding Abundance (1986–2024)

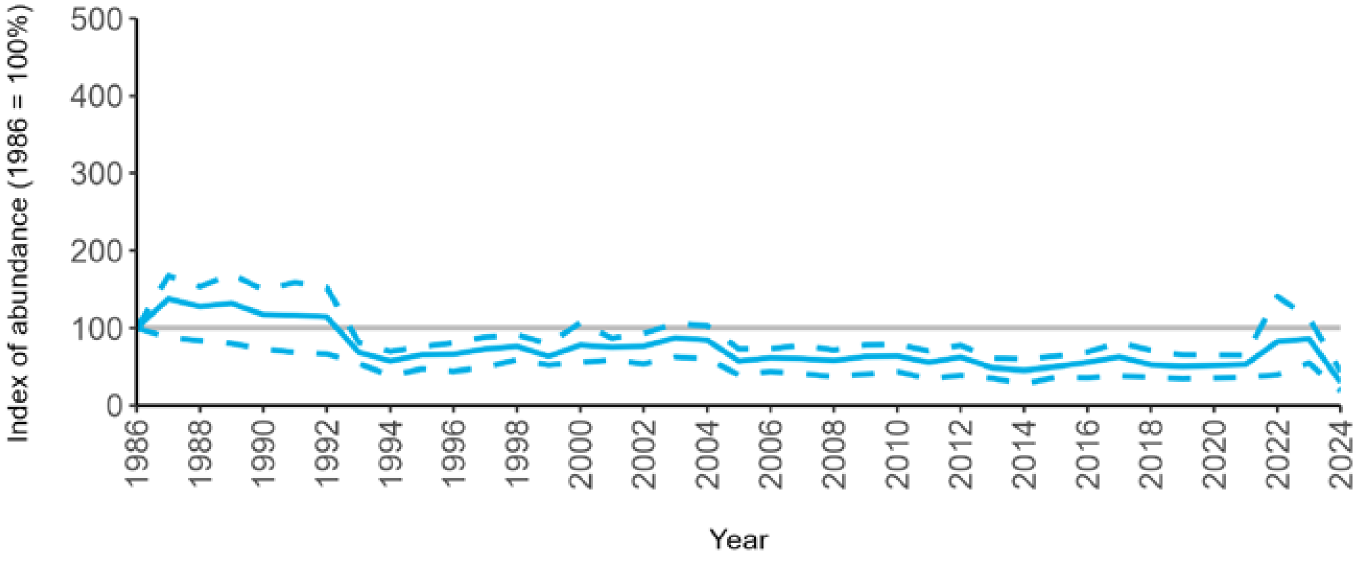


Figure 25: Wales SMP Breeding Abundance (1986–2024)

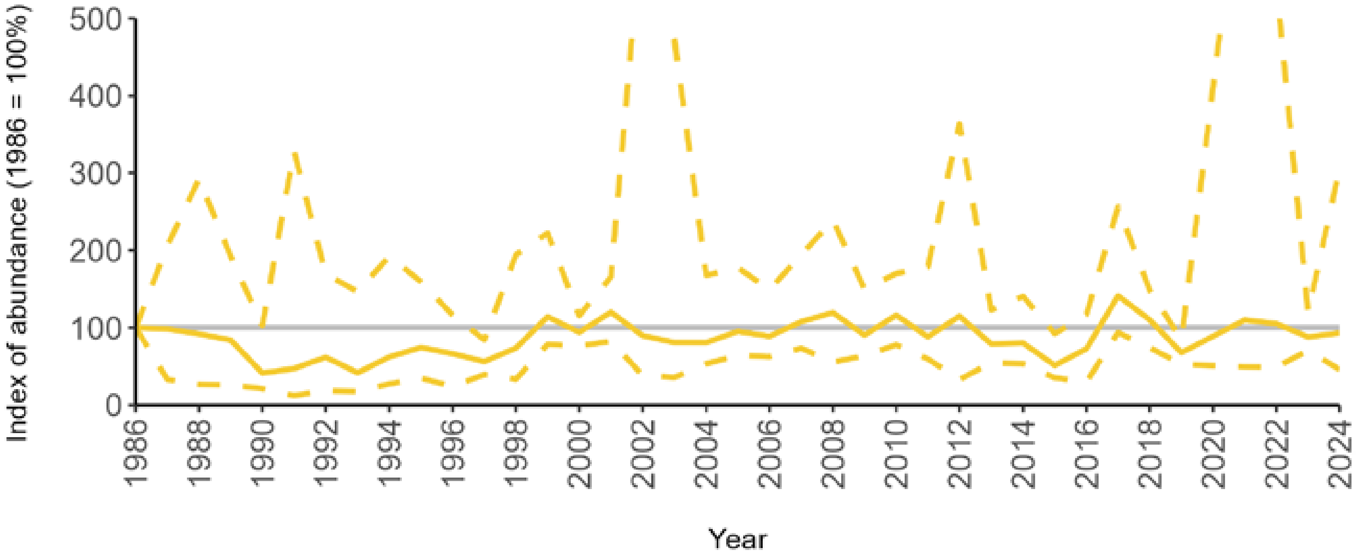
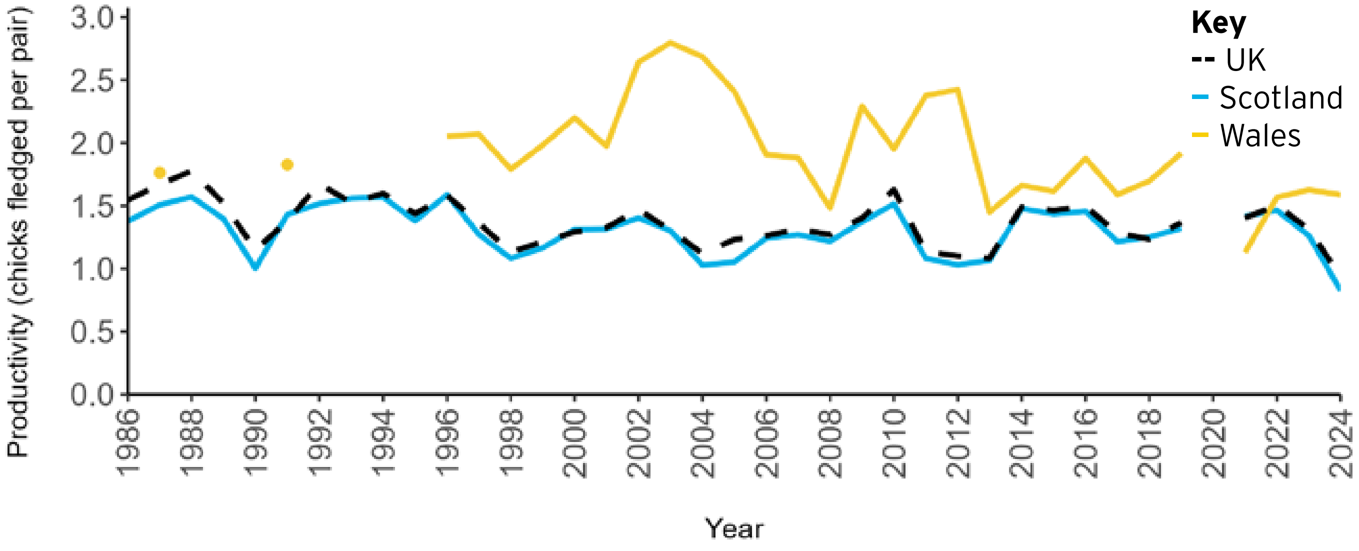


Figure 26: SMP Productivity (1986–2024)



SHAG, BY EDMUND FELLOWES / BTO

Arctic Skua

Stercorarius parasiticus

<0.5%

Abundance: Decline
Productivity: 0.03

Red-listed
n/a (I)

Colony Count sites: 14
Breeding Success sites: 4

Least Concern

Lifespan: 12 years
Breeding age: 4 years

Britain and Ireland host 1–2% of the European population of Arctic Skua and 0.3–0.4% of the global population (Burnell *et al.* 2023). There are two main colour morphs, pale and dark phase. Pale phase birds dominate the northern latitudes of their global range, whereas dark phase birds are more common at the south of their range.

DISTRIBUTION

In Britain and Ireland, breeding Arctic Skuas are restricted to the north and west of Scotland, and are particularly associated with the Northern Isles (Burnell *et al.* 2023). Their global range encompasses the northernmost coasts of Eurasia and North America (del Hoyo *et al.* 1996), and the Scottish population is on the southern edge of its range, together with those along Baltic coasts and the Aleutian Islands.

These long-distance migrants spend the boreal winter in pelagic waters around the Mediterranean, the tropics, and productive seas around South America, Africa, Australasia and in the north-western Indian Ocean (Harrison *et al.* 2022, O'Hanlon *et al.* 2024b, van Bemmelen *et al.* 2024). Scottish-nesting birds winter off western and southern Africa, and South America (van Bemmelen *et al.* 2024).

DIET

In Scotland, Arctic Skua mainly forage by stealing fish from other seabirds (kleptoparisitism), but they have a varied diet with eggs, berries, insects, rodents and small birds also being consumed (Furness 1987). Due to their relatively small size, Arctic Skuas are not commonly found scavenging behind fishing boats alongside larger competitors, or in multi-species feeding flocks (Furness 1987).

BREEDING

In the Northeast Atlantic, Arctic Skuas typically breed in loose colonies on moorlands or coastal grasslands, often close to colonies of the seabirds from which they steal food. In Arctic regions, where most of the global breeding population is found, breeding densities are generally much lower, particularly when breeding inland as individuals forage within their breeding territories on a wide range of prey (Bart & Johnston 2012). Between one and three (most commonly two) eggs are laid in a shallow scrape in the vegetation. (Burnell *et al.* 2023).

BREEDING ABUNDANCE

As the UK distribution of Arctic Skua is entirely restricted to Scotland, the SMP abundance trends for the UK and Scotland are identical, therefore only the graph for Scotland is shown (Figure 27). The SMP 24-year change for Scotland (and therefore the UK) shows a decline of 89% since 2000, far greater than the 66% declines reported in *Seabirds Count* (Tables 18 & 19; Burnell *et al.* 2023). The Scottish population of Arctic Skua has declined steadily since the early 1990s (Figure 27). In 2024, the SMP Arctic Skua abundance index was 93% below the 1986 baseline, the lowest since monitoring began.

PRODUCTIVITY

The Arctic Skua productivity trend for Scotland has shown considerable fluctuation across the recording period, with several years of very poor productivity, particularly between 2004 and 2013 (Figure 28). Following an increase in productivity estimates in recent years, 2024 was a particularly poor year with just 0.03 chicks fledged per breeding pair (Table 18).

ARCTIC SKUA CHICK, BY SARAH HARRIS / BTO

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

In the North Atlantic, Arctic Skuas are primarily reliant on fish species such as sandeels, which they kleptoparasitise from other seabirds. Climate change has reduced the availability of sandeels for a range of seabird species, with knock-on effects that are thought to be contributing to reduced breeding success in Arctic Skuas (Dawson *et al.* 2011, Perkins *et al.* 2018, Phillips *et al.* 1996). Recent studies have highlighted long foraging trip distances during the nesting period, causing additional pressure during the breeding season (Burnell *et al.* 2023, van Bemmelen *et al.* 2021).

There is little information about the trends and potential drivers of population change for Arctic Skuas in Arctic regions, but these populations are thought to be more stable than kleptoparasitic populations of the North Atlantic. However, climate change-related pressures such as heat stress, changes in prey communities, and extreme weather events are known, or potential, drivers of population change across the Arctic (Snell *et al.* 2024, Smith *et al.* 2023, Chmura *et al.* 2018, Oswald & Arnold 2012).

Although installation of onshore wind turbines is designed to reduce the extent of climate change, their placement on the moorland habitats where Arctic Skuas commonly breed may cause detrimental impacts through disturbance and the risk of collision with turbine blades (Burnell *et al.* 2023). Furthermore, collisions with power lines have been recorded at colonies in Norway, the Faroe Islands, and Shetland.

Predation is an additional issue for Arctic Skuas. Arctic Skua chicks are known to be predated by Great Skuas, the population of which had been growing in recent years until the recent HPAI outbreak (Perkins *et al.* 2018, Tremlett *et al.* 2024a). There can also be competition between the two species for breeding territory (Dawson et al. 2011).

In Scotland, species such as Red Fox, Stoat (*Mustela erminea*), European Hedgehog (*Erinaceus europaeus*) and American Mink overlap in range with breeding Arctic Skuas, posing a potential risk by directly predating eggs and chicks, or indirectly by adding pressure to the neighbouring breeding seabirds the Arctic Skuas rely on for prey (Burnell *et al.* 2023).

CONSERVATION

Preventing invasive predators from reaching islands with breeding Arctic Skuas through implementation of biosecurity measures, and removal of invasive species where they do overlap in range could relieve pressure.

The benefits of eradication programmes was demonstrated by the improvement in Arctic Skua breeding success when the non-native American Mink was removed from an archipelago in Finland (Nordström *et al.* 2003).

Given their favoured moorland nesting habitat, Environmental Impact Assessments for onshore wind farms need to take Arctic Skuas into account, as they pose a risk both through collision and disturbance.

Research into year-round and breeding season foraging has taken place in recent years to better understand Arctic Skua's annual movements and foraging ranges. By tracking individuals across their annual cycle we can identify important areas worthy of protection for Arctic Skuas and the seabird species on which they rely (O'Hanlon *et al.* 2024b). Further monitoring of demographic rates (productivity and survival) throughout the global breeding range is key to understanding drivers of population change for populations with different foraging behaviours and prey bases. ►

ARCTIC SKUA & SANDWICH TERN, BY MOSS TAYLOR / BTO

Table 18: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AOT)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	727	11	-93*	-89	0.03	4
Scotland	727	11	-93*	-89	0.03	4

** significant changes*

Table 19: *Seabirds Count* census results

	Abundance (AOT) *Seabird 2000* (1998–2002)	Abundance (AOT) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	2,141	727	-66

Figure 27: Scotland SMP Breeding Abundance (1986–2024)

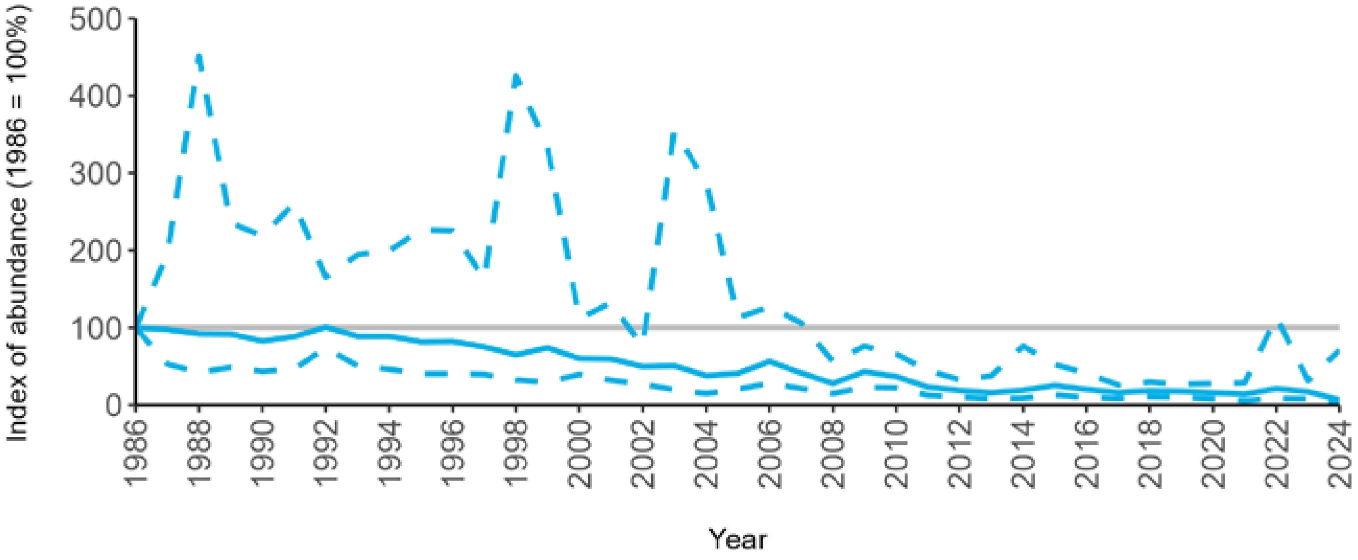
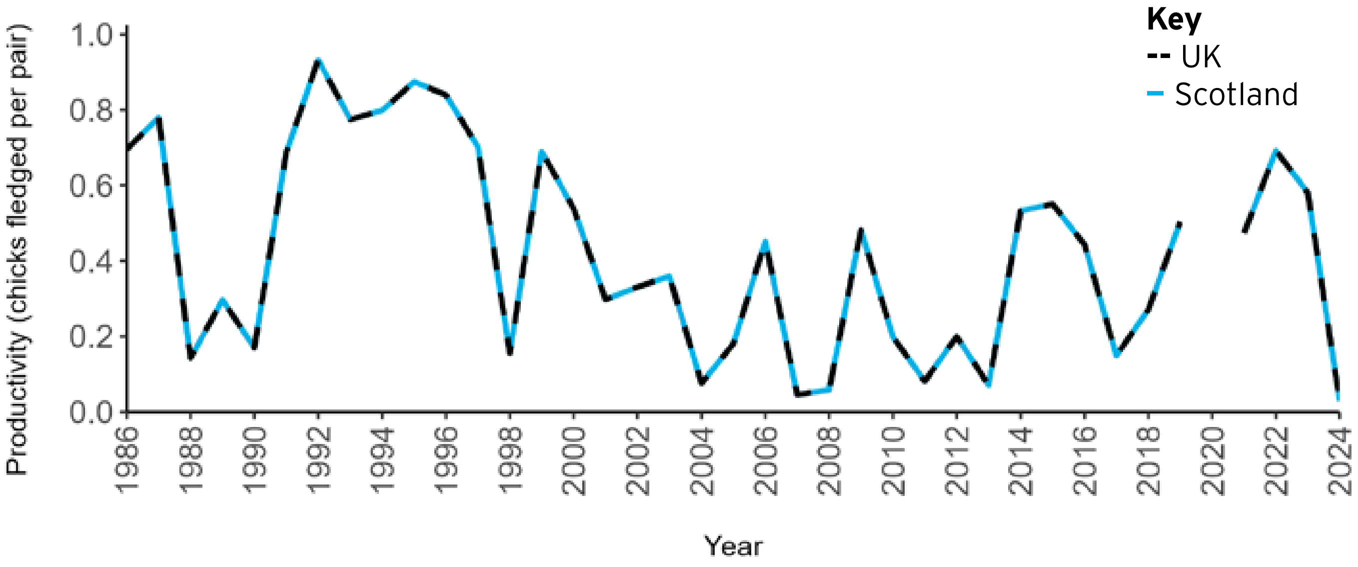


Figure 28: SMP Productivity (1986–2024)



Great Skua

Stercorarius skua

c.64–67%

Abundance: n/a
Productivity: 0.17

Red-listed
Amber-listed (I)

Colony Count sites: 28
Breeding Success sites: 6

Least Concern

Lifespan: 15 years
Breeding age: 7 years

Britain and Ireland host 64–67% of the global breeding population of Great Skua (Burnell *et al.* 2023). They are a highly territorial species and will swoop down on intruders to breeding colonies. Great Skua are colloquially known as 'Bonxies', a Nordic name originating from Shetland and thought to refer to their 'dumpy' posture.

DISTRIBUTION

Great Skua breeding areas in Britain and Ireland are primarily in the north and west of Scotland, with smaller numbers found on the north and west coasts of Ireland (Burnell *et al.* 2023).

Globally, Great Skua breeding areas have been restricted to the north-east Atlantic, although their range is currently expanding both northwards into the Barents Sea and south into Ireland (BirdLife International 2024, Burnell *et al.* 2023), and up to four non-breeding individuals have also spent the summer months on the Calf of Man (Isle of Man) since 2016 (A. Sapsford *pers. comm.*). Strongholds include Scotland, Iceland, Svalbard and the Faroe Islands (Keller *et al.* 2020).

Birds from breeding colonies in Britain and Ireland can be seen off the coast year-round, albeit in low numbers in winter, but are mainly migratory, travelling as far south as West Africa (BirdLife International 2024, Magnusdóttir *et al.* 2011, Wernham *et al.* 2002).

DIET

In small colonies, Great Skua tend to target other seabirds as prey, whereas in larger colonies the majority of individuals consume mainly a range of fish species, including fishery discards (Votier *et al.* 2004). However, they are opportunistic feeders and will adapt their diet to local conditions, utilising whatever prey is readily available e.g. Goose Barnacles (*Lepas anatifera*), eggs or European Rabbits (*Oryctolagus cuniculus*) (Phillips *et al.* 1997, Votier *et al.* 2004).

BREEDING

Nesting on coastal moorland, colonies can range in size from a loose grouping of a few birds to thousands. The nest consists of a grass-lined scrape in which two eggs are laid (Furness 1987).

BREEDING ABUNDANCE

No SMP abundance trends are produced for Great Skua, as too few large colonies are surveyed regularly or in the same year to produce accurate trends. *Seabirds Count* reported a 14% population increase across Britain and Ireland since *Seabird 2000* (Table 20 & Burnell *et al.* 2023). However, this was before the substantial mortality seen during the 2021 and 2022 breeding seasons due to HPAI. An RSPB-led project to assess the population impact of the 2021/22 HPAI outbreak indicated a decline in Great Skua breeding numbers of 76% at the sites surveyed across the UK in 2023 (which covered 81% of the UK breeding population) compared to pre-HPAI baseline counts (Tremlett *et al.* 2024a).

PRODUCTIVITY

The productivity of Great Skua breeding in Scotland (and therefore the UK, as all monitored sites have been in Scotland) has varied considerably over the SMP period (Figure 29), but with an overall decline in productivity since 2006. After low levels of productivity in 2021 and 2022 of 0.10 and 0.09 chicks fledged per pair, respectively (Table 2), attributed to HPAI, the number of chicks fledged per pair increased to 0.44 in 2023. However, in 2024, the estimate declined to 0.17 chicks fledged per pair (Table 21).

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

A significant pressure in recent years is likely to have been a reduction in food availability, although the adaptability of Great Skua to changes in conditions may have protected them to some extent. Competition for food at the local (colony) level was the main driver of population decline recorded in Orkney (Meek *et al.* 2011). A law to reduce fishery discards, fully enforced since 2019, has the potential to negatively affect those populations of Great Skua that rely most heavily on fishery discards for food (Bicknell *et al.* 2013, Votier *et al.* 2008).

Despite competition for food and the decline in available discards from fisheries, Great Skua are highly opportunistic feeders and so are likely to be able to switch their diet depending on prey availability. Studies on Great Skuas breeding in northern Scotland showed that their diet changed from being dominated by sandeels in the 1970s to predominantly discarded whitefish from the 1980s onwards, and the proportion of avian prey in the diet increased significantly between the 1980s and 2010s (Church *et al.* 2018). The avian prey component also changed over time, with Kittiwakes being replaced by mainly auks and Fulmars (Church *et al.* 2018).

Great Skua pellets have been found to contain plastics in the Faroe Islands. This is likely to be a result of secondary uptake from their prey species, Fulmar (Hammer *et al.* 2016, van Franeker *et al.* 2021), which are known to frequently ingest plastic. The effects of this on Great Skuas is unknown.

HPAI caused a substantial loss in the Great Skua population across Scotland in the summer of 2022 (Banyard *et al.* 2022, Camphuysen *et al.* 2022), which became a focus for further surveying in 2023 (Tremlett *et al.* 2024b).

The Great Skua is a cold-adapted species and can suffer from heat stress (Furness 1987), which is likely to increase under climate change, further reducing suitable breeding habitat.

CONSERVATION

The impact of HPAI on seabird species in recent years has been pronounced, particularly for Great Skua. A major recommendation from Tremlett *et al.* (2024a) is continued intensive monitoring of Great Skuas, to determine both the immediate impacts and assess the long-term implications of this disease. ■

Table 21: SMP Productivity

	Productivity	
	2024	Sites
UK	0.17	6
Scotland	0.17	6

Figure 29: SMP Productivity (1986–2024)

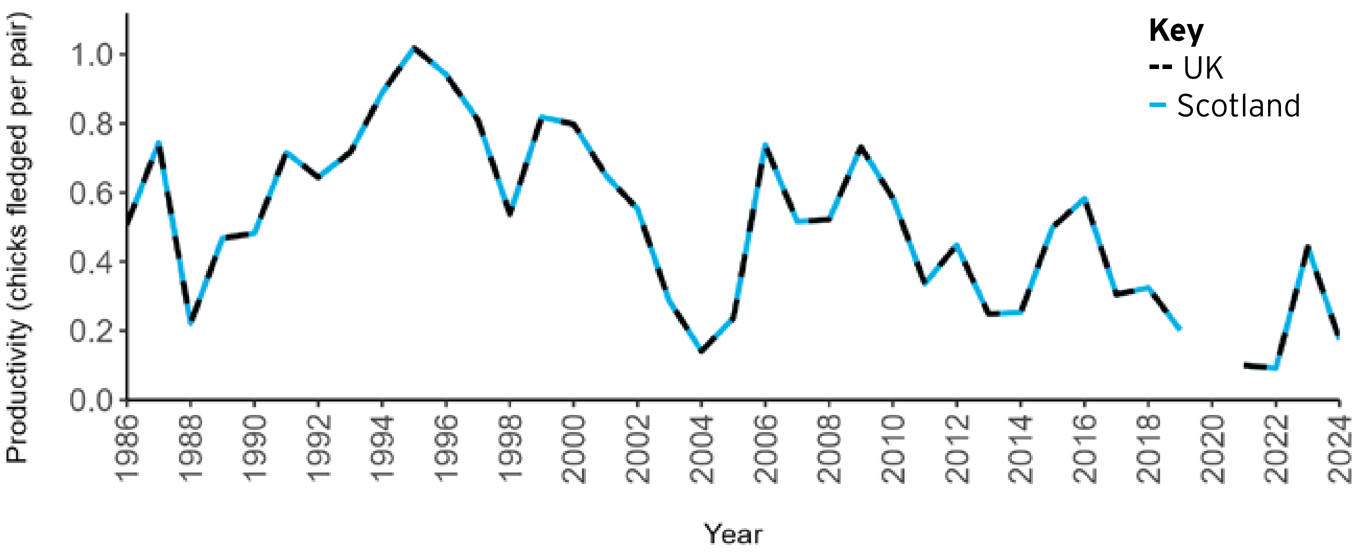


Table 20: *Seabirds Count* census results

	Abundance (AOT) *Seabird 2000* (1998–2002)	Abundance (AOT) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	9,608	10,971	14

Mediterranean Gull

Ichthyaetus melanocephalus

c.1–2%

Abundance: n/a
Productivity: n/a

Amber listed
Amber-listed (I)

Colony Count sites: 38
Breeding Success sites: 10

Least concern

Lifespan: 15 years
Breeding age: 2–3 years

Mediterranean Gulls first bred in Britain in 1968 in Hampshire (England) (Taverner 1970), and in 1995 in Antrim (Northern Ireland) (JNCC 2021), but have rapidly expanded in range and population since then, and 1–2% of the the global population now breed in Britain and Ireland (Burnell *et al*. 2023).

DISTRIBUTION

Most Mediterranean Gull breeding colonies in Britain and Ireland are found in the south and east of England. However, since the *Seabird 2000* (1998–2002) census, there has been a north-westward range expansion, with new colonies in Wales and Ireland and a significant increase in the number of inland colonies in England (Burnell *et al.* 2023).

Globally, the main breeding population is found around the Black Sea and surrounding European countries. From the 1950s onwards, they expanded their range to both the east and west, and Mediterranean Gulls currently breed at scattered locations throughout much of Europe (BirdLife International 2024).

Mediterranean Gulls are predominantly migratory, typically wintering in the Mediterranean Sea, the Black Sea, north-west Europe and Africa, favouring coastal habitats with sheltered waters (del Hoyo *et al.* 1996). Birds that nest in Britain and Ireland can be seen all around the coastline outside of the breeding season, with some additional records inland, including on refuse tips (BTO 2025).

DIET

During the breeding season Mediterranean Gulls typically eat terrestrial and aquatic insects, gastropods, some fish and rodents (del Hoyo *et al.* 1996). In winter, they switch to more marine-related species, such as fish, molluscs and occasional fishery discards, but will also consume insects, earthworms, berries and seeds. They will also forage on refuse tips (Milchev *et al.* 2004, Urban *et al.* 1986).

BREEDING

Most Mediterranean Gull colonies in Britain and Ireland are fairly small, and they often breed alongside Black-headed Gulls (Burnell *et al.* 2023). They nest in a range of habitats, including coastal lagoons, estuaries, saltmarsh, and inland on wetland areas with sparse vegetation (del Hoyo *et al.* 1996). Nests are a shallow scrape lined with grass and feathers into which up to three eggs are laid (Snow & Perrins 1998).

BREEDING ABUNDANCE

Valid annual SMP abundance trends could not be published for Mediterranean Gull due to the scarcity of regular colony monitoring data, resulting in considerable uncertainty around the estimated trends. Many sites are counted well and data are provided by county bird recorders to the Rare Breeding Birds Panel (Eaton *et al.* 2023) so an improved flow of data from local counters to the SMP, together with better monitoring would allow a trend to be produced for this species in the future.

The *Seabirds Count* census demonstrated the rapid recent population growth of this species across Britain and Ireland, with an increase of 1,612% since *Seabird 2000* (Table 22), and a rise in colony numbers from 38 to 61 over the same period (Burnell *et al.* 2023).

FEEDING MEDITERRANEAN GULL, BY THOMAS WILLOUGHBY / BTO

Table 22: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	135	2,311	**1,612**

PRODUCTIVITY

At present too few Mediterranean Gull colonies are monitored regularly enough to produce valid productivity trends. More consistent annual monitoring of productivity at colonies where data have been submitted previously to the SMP would allow a trend to be produced in the future.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Mediterranean Gull populations in Britain and Ireland and across Europe have expanded significantly in recent years, which is likely to be due to a combination of factors, such as climate change, provision and management of suitable habitat, and protection of nesting colonies (Fasola & Canova 1996, Meininger & Flamant 1998, Ausden & Fuller 2009).

However, much of Britain and Ireland's population is sited in just a few colonies, with a single colony in Hampshire holding 68% of the British and Irish population (Burnell *et al.* 2023), meaning local impacts have the potential to cause population-level effects (Eaton *et al.* 2021).

Potential local negative factors include flooding of nests, which can cause desertion of colonies following tidal surges and extreme weather, and this risk is likely to increase due to climate change in the future (JNCC 2021).

Colony disturbance from humans can cause nest desertion and consequent impacts on breeding attempts (James 1984, Burger *et al.* 2020). Egg loss through predation and licensed harvesting of Black-headed Gull eggs can also cause local issues (Burger *et al.* 2020, Burnell *et al.* 2023).

Additional pressures, including outside of the breeding season, facing Mediterranean Gulls include oil pollution, disease and negative impacts resulting from commercial fishing practices and illegal hunting (del Hoyo *et al.* 1996).

CONSERVATION

Conservation schemes aimed at reducing human disturbance, protecting against egg collection, and providing and maintaining suitable nesting habitat through vegetation management, erosion control and nesting substrate provision, have proved successful conservation measures for Mediterranean Gulls (Fasola & Canova 1996, Schwartz *et al.* 2023). Artificial rafts have also occasionally been used by this species (Burgess & Hirons 1992). ■

Black-headed Gull

Chroicocephalus ridibundus

c.2–4%

Abundance: Decline
Productivity: 0.72

Amber-listed
Amber-listed (I)

Colony Count sites: 137
Breeding Success sites: 30

Least Concern

Lifespan: 11 years
Breeding age: 2 years

Approximately 2–4% of the world's Black-headed Gull population breeds in Britain and Ireland (Burnell *et al.* 2023). They are the most widely distributed seabird breeding here, with 46% of the population nesting inland and the remainder on the coast (Burnell *et al.* 2023).

DISTRIBUTION

Breeding Black-headed Gulls are widely distributed across Britain and Ireland, with the exception of large areas of the Scottish and Welsh uplands, and the majority of the breeding population are resident throughout the year (Burnell *et al.* 2023).

Globally, they breed widely across the middle latitudes of the Palearctic, and there is also a small presence on the east coast of Canada (BirdLife International 2024).

Colonies in the milder areas of their range, such as Britain and Ireland, are resident, with the winter population in Britain boosted by birds from northern and eastern Europe (Wernham *et al.* 2002). The remaining populations from colder regions winter in the south of the northern hemisphere (del Hoyo *et al.* 1996).

DIET

Black-headed Gulls are opportunistic feeders, adapting their diet to the local environment. Their main prey items are aquatic and terrestrial insects, earthworms and marine invertebrates, but they will also eat fish, rodents, agricultural grain, berries, fishery discards and human food (del Hoyo *et al.* 1996, Mitchell *et al.* 2004, Scott *et al.* 2015). In the non-breeding season, Black-headed Gulls tend to rely more on artificial food sources, including refuse tips (del Hoyo *et al.* 1996).

BREEDING

Black-headed Gulls usually nest in large colonies near water, e.g. bogs, marshes, gravel pits or on the coast. They will use artificial rafts, occasionally low trees and bushes and will utilise rooftops in some places (Hagemeijer & Blair 1997, Mitchell *et al.* 2004). The nests are made of vegetation, twigs and sticks into which 2–3 eggs are laid (BTO 2025, Snow & Perrins 1998).

BREEDING ABUNDANCE

SMP abundance trends for Black-headed Gull are only produced for coastal nesters, as insufficient inland colonies are monitored annually to produce reliable trends. The SMP 24-year change for the UK shows a decline of 37% since 2000 (Table 23) similar to the decline of 33% for coastal nesters reported by *Seabirds Count* over a similar period (Burnell *et al.* 2023). The England SMP abundance change shows a decline of 34% since 2000 (Table 23), similar to the decline of 38% for coastal nesters shown by *Seabirds Count* (Burnell *et al.* 2023).

Figures 30 and 31 show the long-term trends for the UK and England coastal population over the SMP monitoring period. For both, the abundance index declined from the late 1980s until the mid-2000s, followed by an increase over the next decade. However, the trends for both the UK and England appear to have been in overall decline again since 2017. The 2024 index value was 18% below the 1986 baseline for the UK and 5% below for England (Table 23), likely in part to be due to the recent HPAI outbreak which affected the species in 2022 and 2023. While the English trend closely mirrored the UK trend for much of the SMP monitoring period, a gap between the two has become more evident in recent years, with the index values for England being higher than those for the UK.

Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful abundance trends.

BLACK-HEADED GULL WITH CHICK, BY EDMUND FELLOWES / BTO

PRODUCTIVITY

Over the period of the SMP, Black-headed Gull productivity values, which cover both coastal and inland nesters, have fluctuated markedly for both the UK and Scotland, with no clear trend (Figure 32). This is likely to be in response to local changes in predation, food supply and periods of inclement weather during breeding seasons (JNCC 2021). Whilst productivity estimates were the same in 2023 and 2024 for Scotland (0.49 chicks fledged per pair), the estimate for the UK increased from 0.30 chicks fledged per pair in 2023 to 0.72 in 2024 (Tables 2 & 4).

Too few data are submitted to the SMP on productivity of Black-headed Gull in other regions for the calculation of reliable productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

The *Seabirds Count* census revealed regional differences in the pattern of Black-headed Gull population decline, with losses in England and Wales being concentrated in coastal areas, whereas the declines were more prevalent in inland and often upland sites in Scotland (Burnell *et al.* 2023). There is no clear evidence of the causative factors for the decline, and it is likely that a combination of pressures are acting together. Much of the population decline between the *Seabird 2000* and *Seabirds Count* censuses is accounted for by heavy losses at just four large colonies (Burnell *et al.* 2023).

Predation by a variety of both avian and mammalian predators, including American Mink (Craik 1995, 1997, Coulson 2019), rats, European Otter (*Lutra lutra*), Red Fox and larger gulls, that take eggs and/or chicks, poses a threat to Black-headed Gulls.

Afforestation can remove suitable habitat and introduce opportunities for predatory mammal and bird species, while moorland management and agricultural practices could also be adding pressure at inland locations through habitat changes (Newton 2020, Roos *et al.* 2018).

In England, licences for harvesting Black-headed Gull eggs for human consumption are still being issued, with around 50 licences issued in 2019 with a combined maximum take of more than 60,000 eggs (Burnell *et al.* 2023), but it is unknown if the full numbers permissible are actually taken.

Additional potential pressures are changes in food availability, extreme weather events, chemical pollution and oil spills, disturbance and disease outbreaks (Burnell *et al.* 2023, Gorski *et al.* 1977, Indykiewicz 2015, van de Pol *et al.* 2010). The 2022 HPAI outbreak caused an apparent population decline of 11% in the surveyed sites between the *Seabirds Count* census and 2023. However, the actual impact on the UK Black-headed Gull population is likely to be greater, as birds were also impacted in 2023 and a minimum of 19,969 Black-headed Gulls carcasses (>9% of the UK breeding population) were counted during the outbreak (Atkinson *et al.* 2025, Tremlett *et al.* 2024b).

CONSERVATION

Reduction of the potential impact of predators through both predator fences and direct control, e.g. the removal of American Mink from nesting islands, can lead to increased breeding success and benefit local populations (Hunt & Heffernan 2007, Short 2020).

In England, the creation of new habitat in the form of gravel pits, and appropriate nature reserve management has also proved beneficial for Black-headed Gulls (Burnell *et al.* 2023).

Continued monitoring of populations following the impact of HPAI will be crucial in understanding how outbreaks impact birds nesting in Britain and Ireland, and increased research into the regional differences in population change may also be key to the conservation of this species. ►

Table 23: SMP Breeding Abundance Change and Productivity

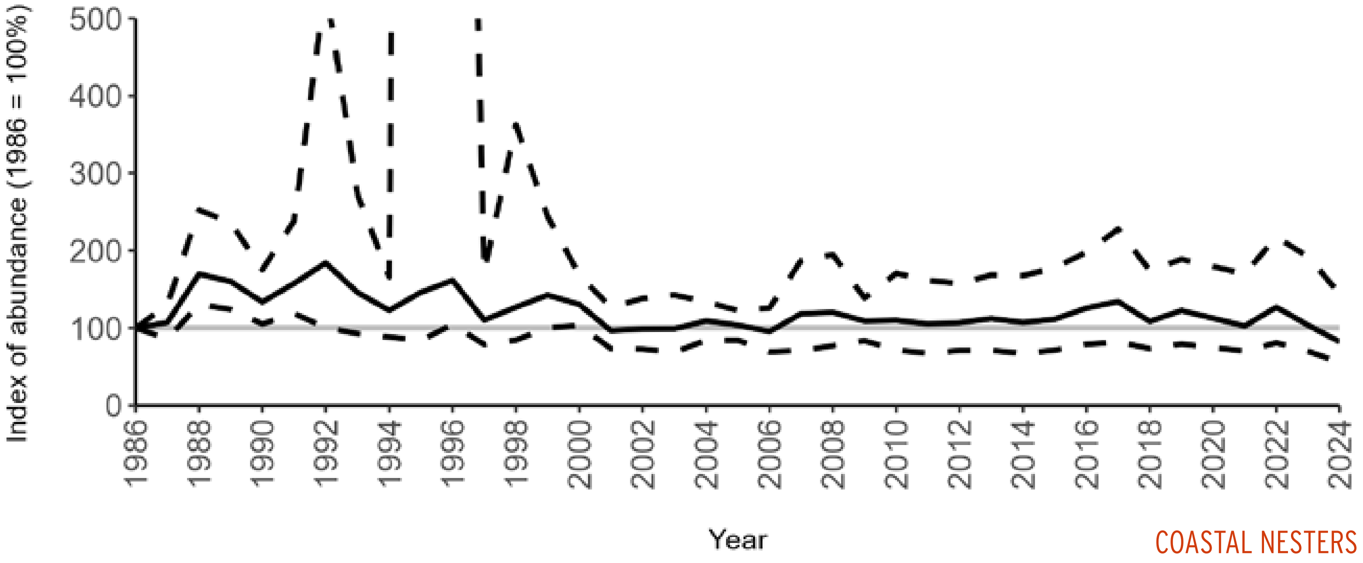
*COASTAL NESTERS **ALL NESTERS	*Seabirds Count**	Breeding Abundance Change %*			Productivity**	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	51,649	75	-18	-37	0.72	30
England	40,398	52	-5	-34	-	-
Scotland	-	-	-	-	0.49	4

No significant changes

Table 24: *Seabirds Count* census results

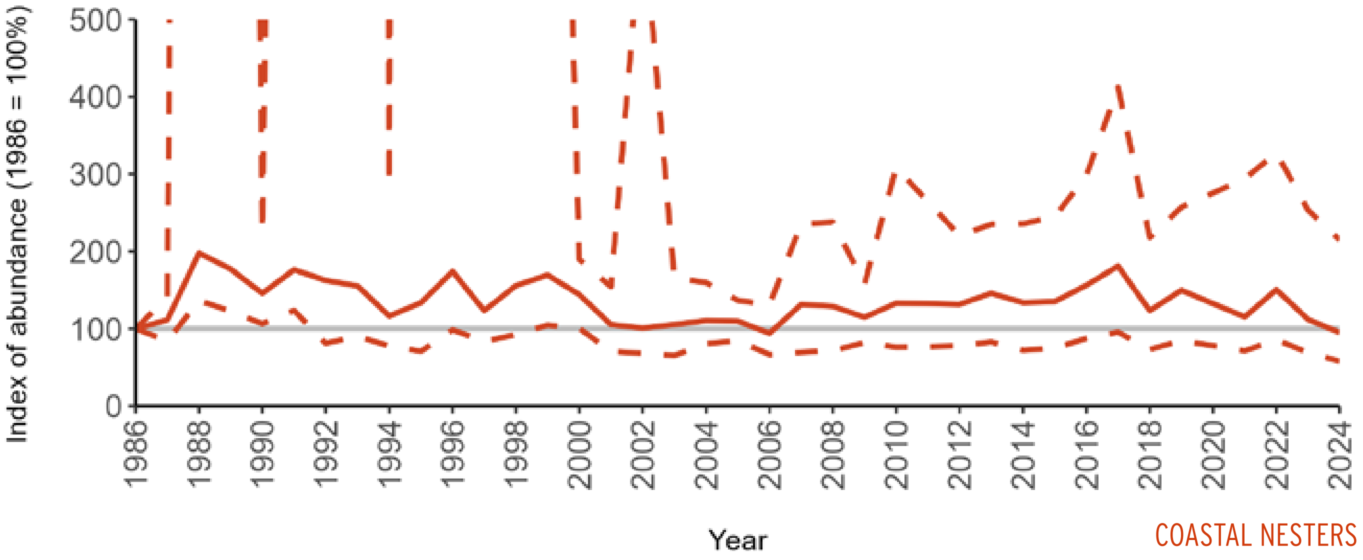
COASTAL NESTERS	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	79,060	56,535	-28

Figure 30: UK SMP Breeding Abundance (1986–2024)



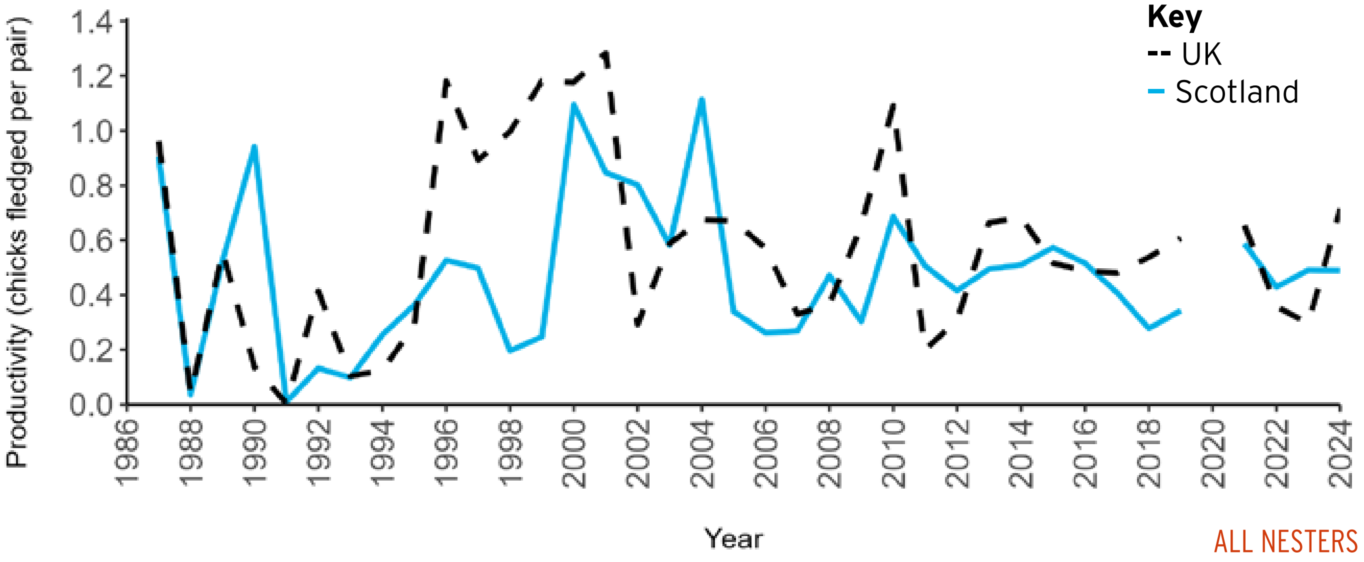
COASTAL NESTERS

Figure 31: England SMP Breeding Abundance (1986–2024)



COASTAL NESTERS

Figure 32: SMP Productivity (1986–2024)



ALL NESTERS

BLACK-HEADED GULL, BY MOSS TAYLOR / BTO

Common Gull

Larus canus

c.2%

Abundance: Decline (Scotland 2023)
Productivity: 0.43

Red-listed
Amber-listed (I)

Colony Count sites: 49
Breeding Success sites: 12

Least Concern

Lifespan: 10 years
Breeding age: 3 years

Britain and Ireland host approximately 2% of the global breeding population of Common Gull (Burnell *et al.* 2023). Key identifying features are their greenish-yellow bill and legs, combined with a grey (*canus* meaning 'whitish-grey') back and upperwings (BTO 2025).

DISTRIBUTION

The majority of Common Gull breeding colonies are in the north and west of Britain and Ireland, with just a few breeding sites recorded further south (Burnell *et al.* 2023).

Globally, breeding Common Gulls are found across the northern Palearctic (BirdLife International 2024). Some areas are home to permanent residents, but other populations migrate south in the winter along their respective coasts, to areas including Portugal and the Mediterranean (del Hoyo *et al.* 1996).

The population in Britain and Ireland is boosted in winter by the arrival of additional migrants from mainland Europe, and they become very widely distributed across lowland and coastal areas (Wernham *et al.* 2002).

DIET

The diet of the Common Gull varies according to their breeding location and with the season (Mudge & Ferns 1982, Kubetzki & Garthe 2003). Inland, they will eat earthworms, beetles and other insects, and in spring have been known to eat grain (del Hoyo *et al.* 1996), whereas coastal birds will take planktonic crustaceans, molluscs, small fish and fishery discards (Burnell *et al.* 2023).

BREEDING

Coastal Common Gulls breed in a range of associated habitats, from beaches to grassy cliff-ledges, whilst inland breeding sites include moorlands, lake shores and river banks (Burnell *et al.* 2023, Skórka *et al.* 2006). They typically nest in small colonies, their nests are shallow cups lined with vegetation and seaweed, and up to three eggs are laid per brood (BTO 2025, del Hoyo *et al.* 1996).

BREEDING ABUNDANCE

Due to insufficient data from inland colonies of Common Gull, it has only been possible to provide trends for their coastal-nesting populations. A valid abundance trend could only be produced for Scotland, which holds the majority of the UK population, as too few colonies have been monitored elsewhere to produce a reliable trend for the UK as a whole. Too few data were submitted for Scotland in 2024 and therefore we only report up to 2023.

For Scotland, the decline in the coastal-nesting Common Gull SMP abundance index of 38% between 2000 and 2023 (Table 25) closely matches the decline of 39% reported by the *Seabirds Count* census for coastal nesters since the *Seabird 2000* census (Burnell *et al.* 2023).

There has been a decline in the Common Gull population trend in Scotland since the mid-2000s, with the lowest index value since monitoring began recorded in 2021, when it was 64% below the 1986 baseline (Figure 33). In 2023 there was an increase in the index to 19% below the baseline (Table 25). Too few data are currently submitted to the SMP in other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

The Common Gull productivity trends for the UK and Scotland were closely matched up to 2016, with most monitored sites being in Scotland (Figure 34). However, fewer sites have been monitored in Scotland in recent years, and the trends have subsequently diverged, with the Scottish values generally being lower than those for the UK as a whole.

The UK and Scotland trends have fluctuated across the SMP monitoring period, with a general decline in values between 1998 and 2019. However, it should be noted that most of the data between 1996 and 2003 came from a study on the effects of American Mink on gulls nesting on the west coast of Scotland (Craik 1995), and may not be representative of the situation in Scotland as a whole (JNCC 2021).

The situation appears to have improved in recent years, with a peak in productivity observed in 2022, with 0.83 and 0.73 chicks fledged per pair for the UK and Scotland, respectively (Tables 2 & 4). Values were lower in 2023, with 0.62 chicks fledged per pair in UK and 0.57 in Scotland (Tables 2 & 4). The productivity estimate for the UK declined further in 2024 to 0.43 chicks fledged per pair (Table 25). Too few data were submitted in 2024 to produce an estimate for Scotland.

Too few data are submitted to the SMP on productivity of Common Gulls in other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Predation is likely to be a significant threat to Common Gull colonies. Non-native and invasive American Mink are known to take Common Gull adults, chicks and eggs (Craik 1997, 2015, Nordström *et al.* 2003), and additional predators include European Otter, other mammals, birds of prey, crows and large gulls (Burnell *et al.* 2023). In extreme cases this can cause colony abandonment. However, it is unknown how much of an impact this has on the overall population trend.

Licences are issued each year by NatureScot allowing the destruction of a number of Common Gull eggs and nests (historically, more than 20% of Scottish nests), and this had the potential to impact on the Scottish breeding population if fully implemented (Burnell *et al.* 2023). In 2024, NatureScot updated the guidance to reduce the number of licences issued for gulls, including Common Gulls.

Detrimental changes in their nesting environments, from afforestation due to changes in moorland management or the construction of developments such as wind farms, can all cause local declines in populations (Burnell *et al.* 2023). Changes in food availability through alterations in agricultural management, climate change and the impact of fishery discard bans, are additional potential pressures whose current impact is unknown (Burnell *et al.* 2023, Mitchell *et al.* 2004).

CONSERVATION

Control of non-native predators, for example removal of American Mink from islands, can lead to higher gull breeding success rates (Hunt & Heffernan 2007), consequently effective predator control is likely to be beneficial for Common Gull populations.

There has been limited research on Common Gulls, and further studies into the impact of pressures such as land management change, alterations in food availability, and licensed nest and egg destruction may be crucial for future conservation of the species, especially in light of the recent declines detected in this species (Burnell *et al.* 2023). ▶

Table 25: SMP Breeding Abundance Change and Productivity

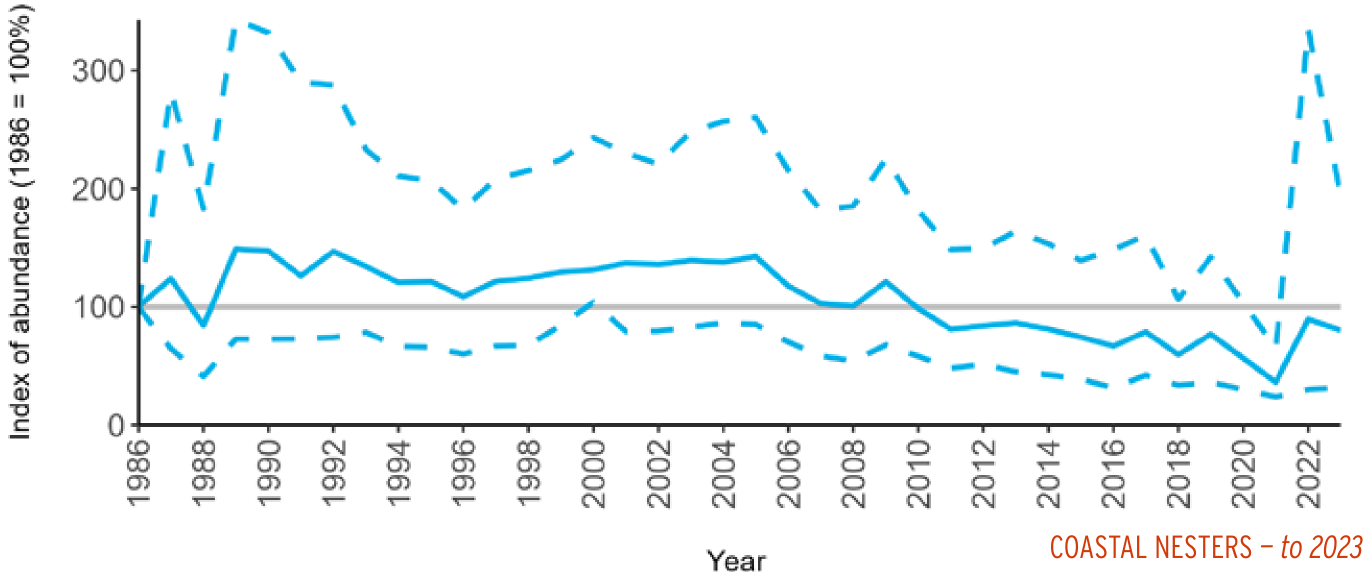
*COASTAL NESTERS **ALL NESTERS	*Seabirds Count**	Breeding Abundance Change %*			Productivity**	
	Abundance (AON)	Sites 2023	LT change (1986–2023)	23-yr change (2000–2023)	2024	Sites
UK	-	-	-	-	0.43	11
Scotland	12,427	29 (2023)	-19 (2023)	-38 (2023)	-	-

No significant changes

Table 26: *Seabirds Count* census results

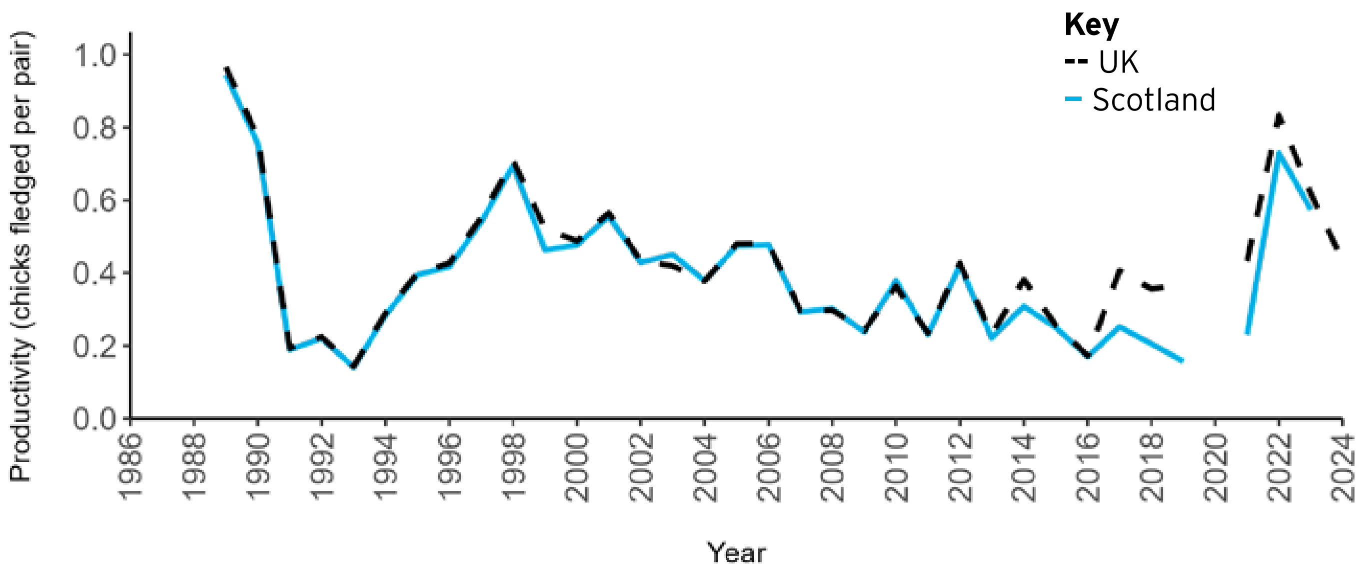
COASTAL NESTERS	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	21,410	14,434	-33

Figure 33: Scotland SMP Breeding Abundance (1986–2023)



COASTAL NESTERS – *to 2023*

Figure 34: SMP Productivity (1986–2024)



ALL NESTERS

COMMON GULL, BY NEIL CALBRADE / BTO

COMMON GULLS, BY PAUL HILLION / BTO

Lesser Black-backed Gull

Larus fuscus

c.87%
ssp. graellsii

Abundance: Decline
Productivity: 0.40

Amber-listed
Amber-listed (I)

Colony Count sites: 92
Breeding Success sites: 11

Least concern

Lifespan: 15 years
Breeding age: 4 years

Britain and Ireland host a minimum of 36% (36–65%) of the global breeding population of Lesser Black-backed Gull and approximately 87% of the subspecies *graellsii* (Burnell *et al.* 2023). Globally, their population has increased and their tendency to migrate has decreased, making them a more common sight year-round in Britain and Ireland (JNCC 2021).

DISTRIBUTION

Breeding Lesser Black-backed Gulls are widespread throughout Britain and Ireland, and have shown a pronounced increase in urban areas in recent years (Burnell *et al.* 2023).

Globally, Lesser Black-backed Gulls are found in north and west Europe and across much of the northern Palearctic (BirdLife International 2024). During the 20th century, their global populations increased and their tendency to migrate has decreased, so that they can now typically be seen throughout the year in much of their breeding range. Part of the breeding population within Britain and Ireland is still migratory, heading towards the coasts of southern Spain, Portugal and northern and western Africa, pausing at many stopovers en route (del Hoyo *et al.* 1996, Klaassen *et al.* 2012, Olsen & Larsson 2003, Wernham *et al.* 2002).

DIET

Lesser Black-backed Gulls are omnivorous, opportunistic feeders, consuming a wide range of natural prey depending on availability within their foraging area, which can include fish, invertebrates, bird eggs and nestlings, carrion, berries and grains. They will also take advantage of refuse tips and fishery discards where available (del Hoyo *et al.* 1996, Langley *et al.* 2022, Olsen & Larsson 2003).

BREEDING

Lesser Black-backed Gulls breed across a variety of coastal and inland habitats, such as cliffs, lakes, moorlands, islands and saltmarshes, and will also nest on artificial structures, such as the roofs of buildings (Raven & Coulson 1997). Locations which are inaccessible to ground predators or where predators are scarce are particularly attractive (JNCC 2021, Rock 2005).

They lay an average of three eggs in a nest which can be a simple lined scrape or constructed from a range of vegetation types (BirdLife International 2015). Lesser Black-backed Gulls often nest in mixed-species colonies, frequently with Herring Gulls, although the two species have different foraging and nesting strategies, with Lesser Black-backed Gulls tending to forage over larger distances and preferring more vegetated areas to nest (Kim & Monaghan 2006, Calladine 1997, JNCC 2021).

BREEDING ABUNDANCE

SMP abundance trends for Lesser Black-backed Gulls are currently only produced for natural nesters (defined as breeding on moors, cliffs, marshes, beaches and other areas of natural or semi-natural habitat) due to the difficulties and uncertainties inherent in monitoring urban nesters (SMP defines this as breeding on human-built structures). Therefore, these trends may not reflect the overall trend of the UK population. The SMP abundance for natural-nesting Lesser Black-backed Gulls in all regions where index values could be produced have decreased markedly since 2000 (Table 27). *Seabirds Count* census results for natural-nesting birds also showed declines in these regions, but the magnitude of the decline was typically less (Burnell *et al.* 2023).

For the UK and Wales, similar declines of 81% and 79%, respectively, were recorded by the SMP for natural-nesters between 2000 and 2024. For Scotland, the 24-year SMP change showed a decline of 46% (Table 27). There is variation between regions in the long-term abundance trends for natural-nesting Lesser Black-backed Gulls over the whole SMP monitoring period, especially during the early years. However, there has been a fairly consistent decline in the index trend for the UK, Scotland and Wales since the late 1990s (Figures 35–37). In 2024, the UK index value was 72% below the 1986 baseline, whilst the values for Scotland and Wales were at 42% and 76% below the baseline, respectively (Table 27). It should be noted for Scotland and Wales that the confidence limits are wide over much of the recording period, therefore these indices should be used with caution.

Too few data are submitted to the SMP on abundance in all other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

The productivity trends for natural-nesting Lesser Black-backed Gulls have fluctuated widely over the SMP recording period for both the UK and Scotland, particularly in recent years (Figure 38). Overall, the UK trend has shown a general increase since recording began, and the productivity estimate was 0.40 chicks fledged per pair in 2024 (Table 27). Productivity estimates for Scotland were typically higher than those in the UK, and showed a slight increasing trend up to 2010. However, no productivity data have been submitted for Scotland since 2014.

Too few data are submitted to the SMP on productivity of Lesser Black-backed Gulls in other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No systematic data on phenology or diet have been collected as part of the SMP. However, adult survival rates of Lesser Black-backed Gull are estimated annually on the Key Site of Skomer Island (Wales) and are published in the Key Site reports for the island.

CAUSES OF CHANGE

Natural-nesting Lesser Black-backed Gulls are facing a wide range of potential pressures, including diseases such as HPAI and botulism (Macdonald & Standring 1978, Tremlett *et al.* 2024a), predation from mammals such as Red Foxes, American Mink and European Badgers (*Meles meles*) (Davis *et al.* 2018), loss of nesting habitat through vegetation changes or rising sea levels (Lock *et al.* 2022, Ross-Smith *et al.* 2015) and emigration to urban areas (Rock 2005).

The legal changes which have reduced the amount of discards from fisheries are likely to be restricting Lesser Black-backed Gull food supplies (Bicknell *et al.* 2013, Furness, *et al.* 1992, Oro 1996, Ross-Smith *et al.* 2014). If this reduces parental body condition, this has been shown to negatively influence chick fledging rates (Nager *et al.* 2000).

Where offshore wind farms are sited within the foraging range of Lesser Black-backed Gulls, there is the potential for collision risk, especially during the breeding season and at some migratory bottlenecks, although studies have shown a degree of avoidance of individual turbines within wind farms (Thaxter *et al.* 2018, 2019).

Chemical pollution can also be an issue, and it has been demonstrated that organic chemical uptake, including increased levels of polychlorinated biphenyls (PCBs) and dichlorodiphenyldichloroethylene (DDE), can increase chick mortality (Burger *et al.* 2018, Bustnes 2006, Hario *et al.* 2000). ▶

Table 27: SMP Breeding Abundance Change and Productivity

NATURAL NESTERS	*Seabirds Count*	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	55,304	76	-72*	-81*	0.40	11
Scotland	11,001	15	-42	-46	-	-
Wales	13,084	17	-76	-79	-	-

No significant changes

Table 28: *Seabirds Count* census results

NATURAL NESTERS	Abundance (AON) *Seabirds 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	112,379	64,267	-43

LESSER BLACK-BACKED GULLS WITH CHICKS, BY EDMUND FELLOWES / BTO

Figure 35: UK SMP Breeding Abundance (1986–2024)

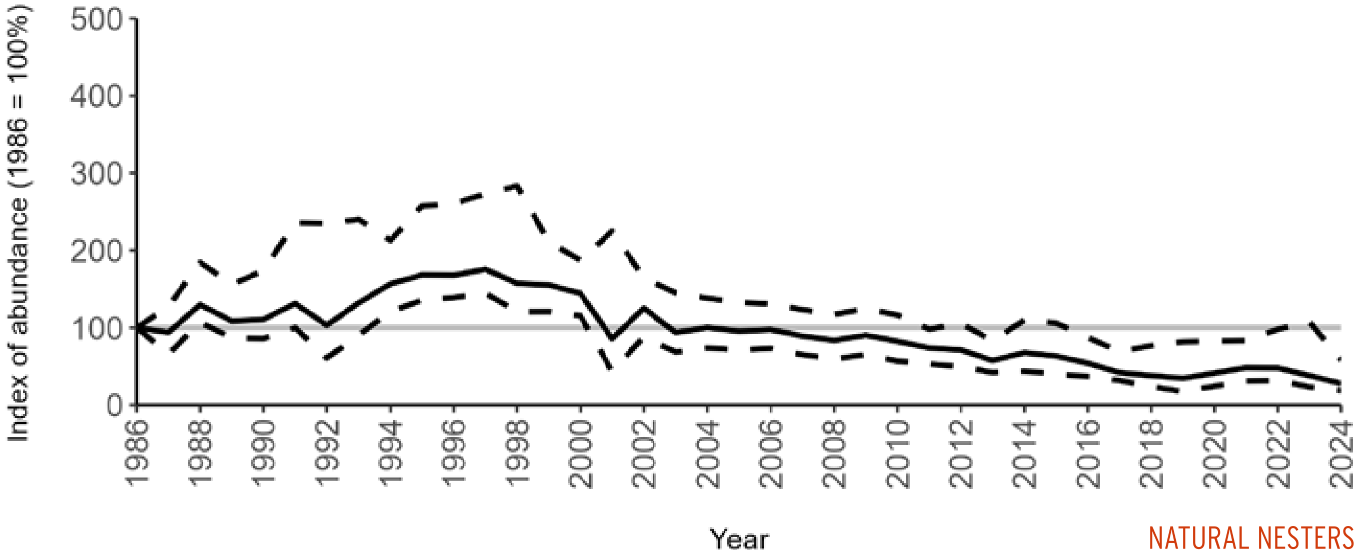


Figure 36: Scotland SMP Breeding Abundance (1986–2024)

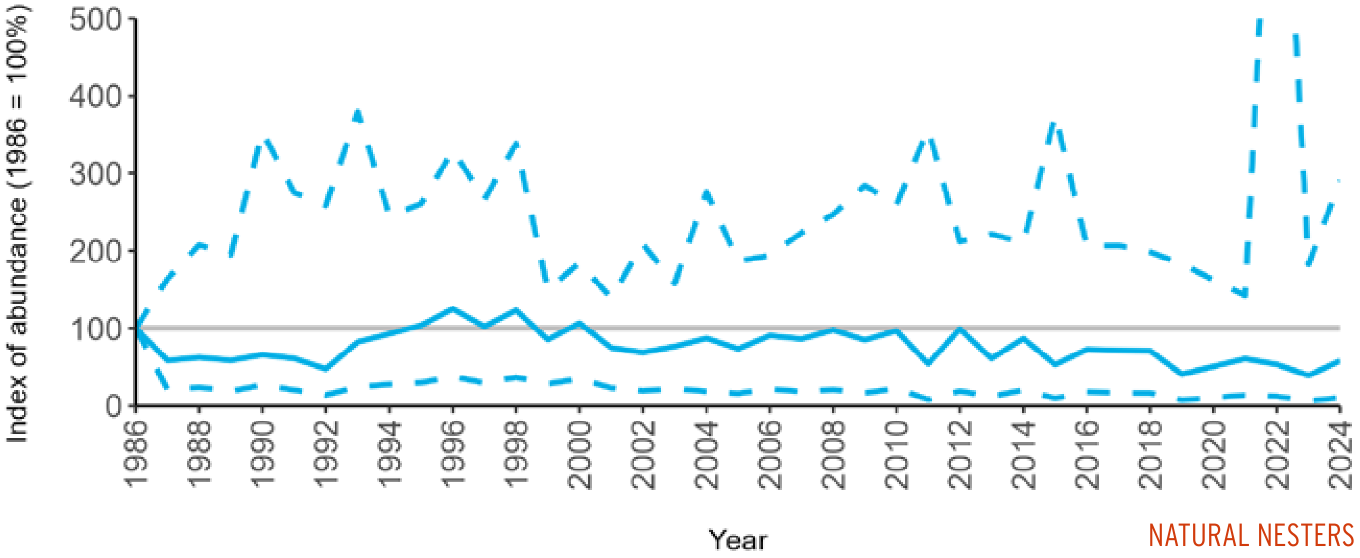


Figure 37: Wales SMP Breeding Abundance (1986–2024)

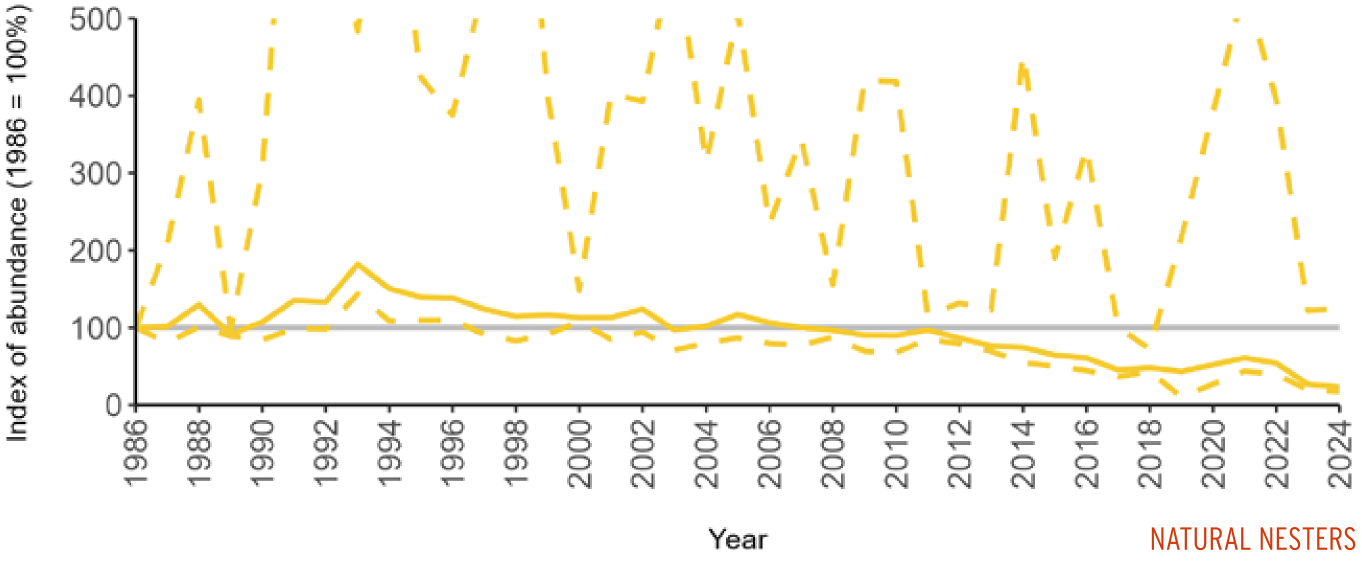
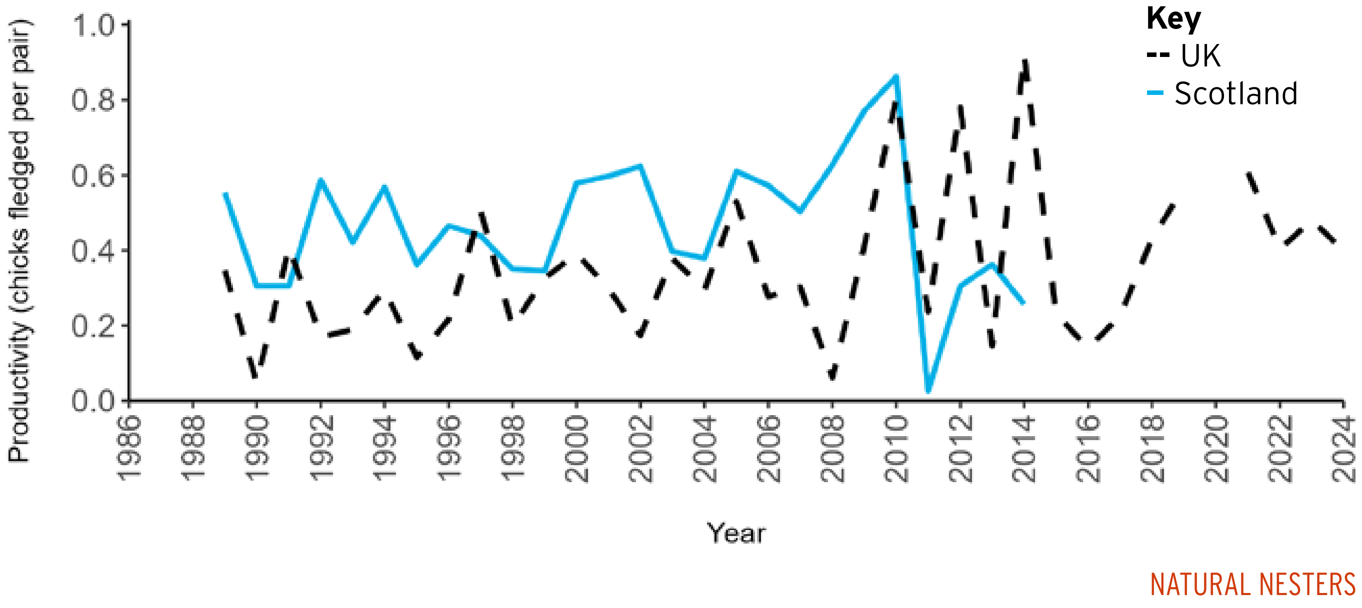


Figure 38: SMP Productivity (1986–2024)



CONSERVATION

Reducing predation levels of Lesser Black-backed Gulls through predator control and the use of predator fences is likely to be beneficial to local populations of this species, alongside appropriate habitat management (Dalrymple 2023). Indeed, the removal of American Mink from islands has been shown to lead to higher gull breeding success rates (Hunt & Heffernan 2007).

In recent years, the removal of Lesser Black-backed Gull from the General Licence scheme throughout the UK and the cessation of mass culling will hopefully have been beneficial for the species. In the future, it should be possible to assess the sustainability of control measures carried out under individual licences to better protect local and regional populations. (Burnell *et al.* 2023).

LESSER BLACK-BACKED GULLS WITH CHICKS, BY EDMUND FELLOWES / BTO

Herring Gull

Larus argentatus

c.70%
ssp. argenteus

Abundance: Decline
Productivity: 0.69

Red-listed
Amber-listed (I)

Colony Count sites: 202
Breeding Success sites: 19

Least Concern

Lifespan: 12 years
Breeding age: 4 years

Britain and Ireland host approximately 44% of the world's breeding Herring Gulls and around 70% of the subspecies *argenteus* (Burnell *et al.* 2023). There are two subspecies of Herring Gull, *argenteus* in the west of their global range and *argentatus* in the east (Keller *et al.* 2020).

DISTRIBUTION

Breeding Herring Gulls are widespread across Britain and Ireland. They typically nest in coastal areas, but are increasing in both inland and urban sites (JNCC 2021, Rock 2005).

In a global context, Herring Gulls breed around northern and western Europe (BirdLife International 2024) They are migratory in the north of their range, but more southern populations (including those in Britain and Ireland) are nomadic or non-migratory (Flint *et al.* 1984).

DIET

Herring Gulls are opportunistic feeders, with natural food sources including fish, marine invertebrates, crustaceans, birds and eggs, but they will also feed on fishery discards and at refuse tips (Bicknell *et al.* 2013, del Hoyo *et al.* 1996, Furness *et al.* 1992, Hüppop & Wurm 2000).

Although Herring Gulls are opportunistic in terms of diet, individuals that breed in natural-nesting sites on the coast predominantly forage in natural habitats, including mussel beds and the intertidal zone. However, some urban nesting individuals will still forage at sea and those on the coast or natural sites may still visit urban areas to forage (Booth Jones *et al.* 2022, Clewley *et al.* 2021, O'Hanlon & Nager 2018, Rock *et al.* 2016).

Individuals targeting high energy food, e.g. human waste food items or fishery discards, tend to have higher productivity than those targeting more natural resources, although this may vary depending on the local availability of food types (van Donk *et al.* 2017).

BREEDING

Herring Gulls nest in a wide variety of both natural and artificial habitats, including cliffs, moorland, farmland, freshwater margins and rooftops (Monaghan & Coulson 1977, Madden & Newton 2004, Raven & Coulson 1997, Sellers & Shackleton 2011).
A nest is built using vegetation, and one to three eggs are laid per nesting attempt (BTO 2025).

BREEDING ABUNDANCE

Herring Gull abundance trends are currently only produced for natural nesters (defined as breeding on moors, cliffs, marshes, beaches and other areas of natural or semi-natural habitat) due to the difficulties and uncertainties inherent in monitoring urban nesters (defined within the SMP as breeding on human-built structures). Therefore, these trends may not reflect the overall trend of the entire UK population.

The declines in SMP abundance trends for natural-nesting Herring Gulls since 2000 (Table 29) agree with those reported by the *Seabirds Count* census (Burnell *et al.* 2023). For the UK and Scotland, similar declines of 47% and 43%, respectively, were recorded by the SMP between 2000 and 2024, whilst the *Seabirds Count* census showed a decrease of 44% for both regions since the *Seabird 2000* census. For England the declines were greater, with the SMP 24-year change showing a decrease of 73% since 2000, whilst the *Seabirds Count* census reported a decline of 60% over a similar period. The SMP 24-year change for Wales shows a decline of 31% since 2000, compared to the decline of 23% reported by *Seabirds Count* since *Seabird 2000* (Burnell *et al.* 2023).


HERRING GULL, BY NEIL CALBRADE / BTO

The long-term SMP abundance trends for natural-nesting Herring Gulls have varied between regions since recording started in 1986, with fluctuations before 2000 for both the UK and England (Figures 39 & 40), but more stability in Scotland (Figure 41). Since 2000, all three regions have experienced overall declines. In 2024, the index values for the UK and Scotland were at 51% and 54% below the 1986 baseline respectively, whilst the England index was at 77% below the baseline (Table 29). The abundance index in Wales has generally been higher than the other regions since the early 2000s, and the index value was at 18% below the baseline in 2024 (Table 29).

Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

Over the SMP recording period, the Scotland productivity trend closely matches the UK trend until 2015, whilst the productivity values for Wales differ markedly from the other regions in some years (Figure 43). Wales has also seen greater fluctuations between years, but with a general decline in values up to the late 2000s. The productivity trend for Wales peaked in 2021 with 1.41 chicks fledged per pair before declining again in more recent years (Table 5). In 2024, the productivity estimates were relatively similar for the UK and Wales, with 0.69 and 0.81 chicks fledged per pair, respectively (Table 29). Insufficient productivity data have been submitted for Scotland since 2015 to calculate any meaningful productivity estimates. This is also true for the other regions where too few data are submitted to the SMP for this calculation. ▶

Table 29: SMP Breeding Abundance Change and Productivity

NATURAL NESTERS	*Seabirds Count*	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	61,077	169	-51*	-47*	0.69	19
England	11,736	59	-77	-73	-	-
Scotland	37,349	75	-54*	-43*	-	-
Wales	9,815	29	-18	-31	0.81	4

* significant changes

Table 30: *Seabirds Count* census results

NATURAL NESTERS	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	126,185	74,926	-41

HERRING GULL, BY GARY CLEWLEY / BTO

Figure 39: UK SMP Breeding Abundance (1986–2024)

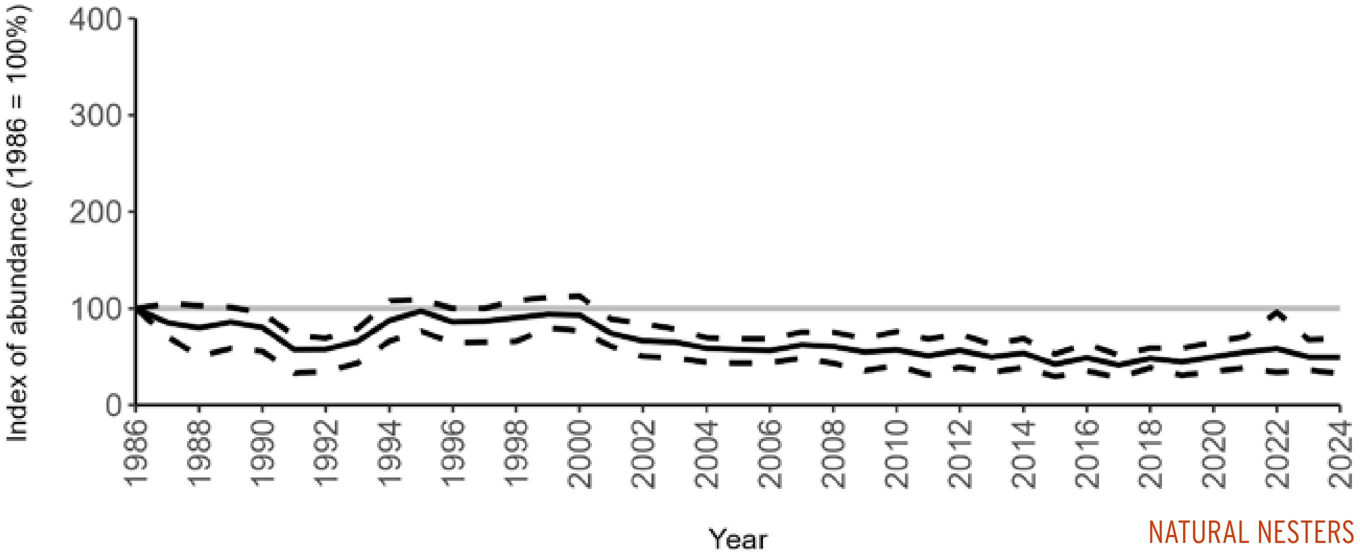


Figure 40: England SMP Breeding Abundance (1986–2024)

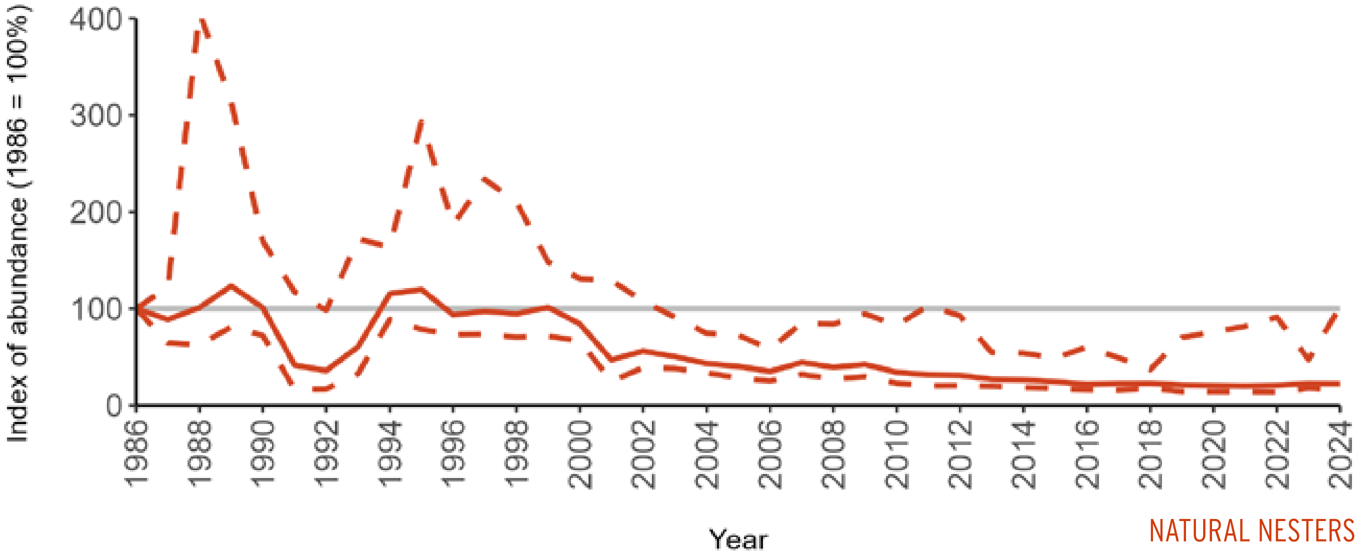


Figure 41: Scotland SMP Breeding Abundance (1986–2024)

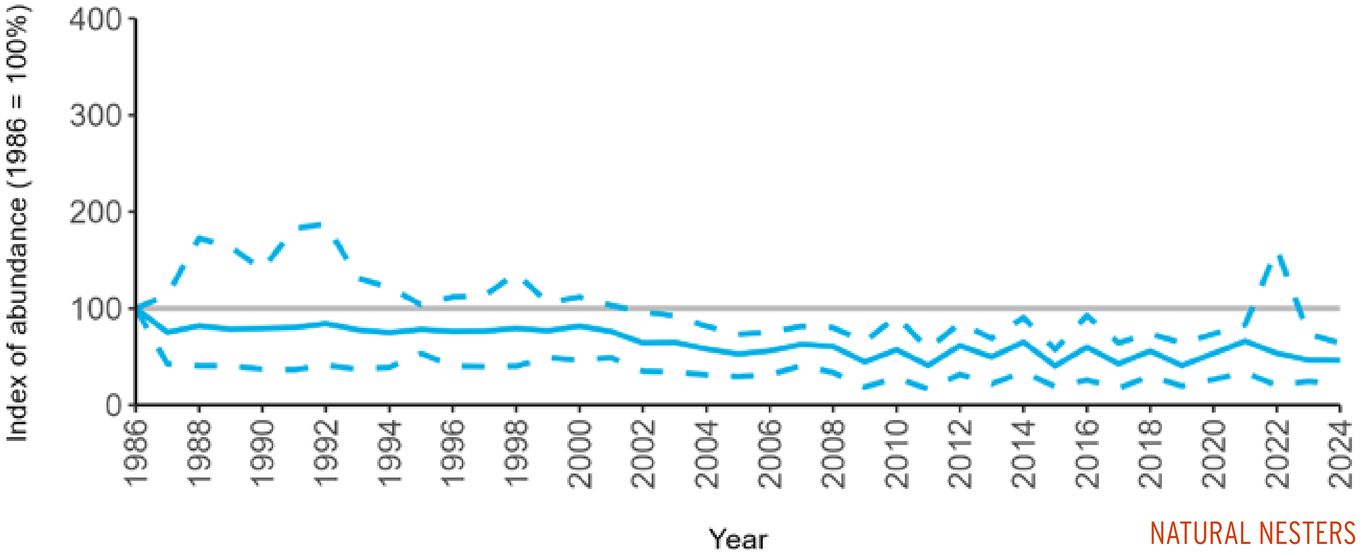


Figure 42: Wales SMP Breeding Abundance (1986–2024)

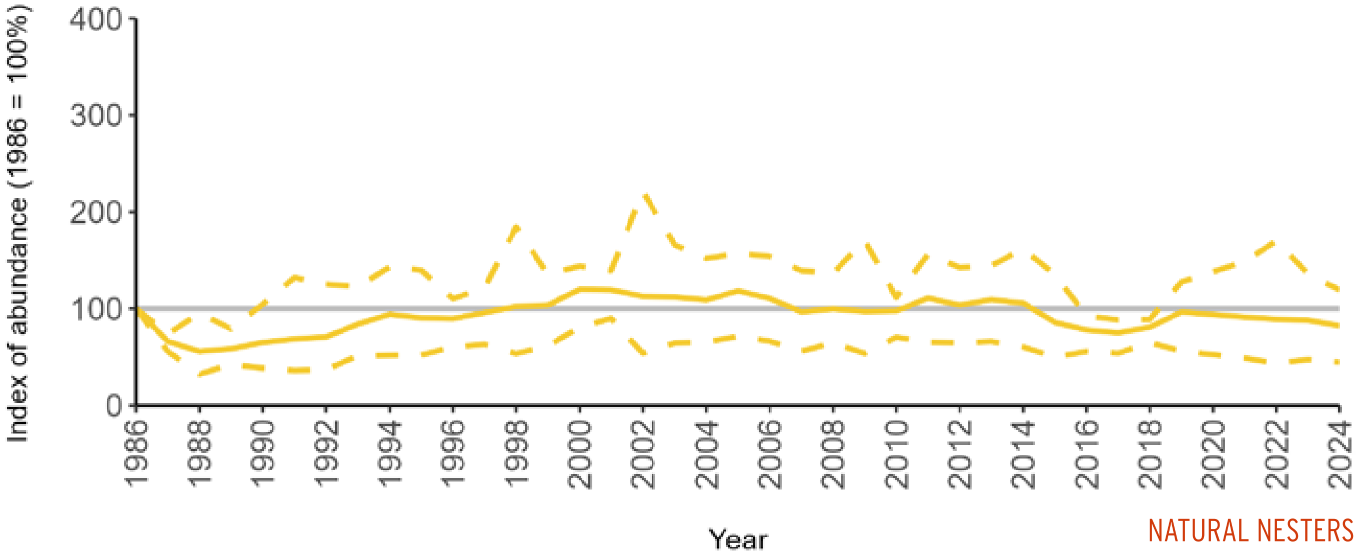
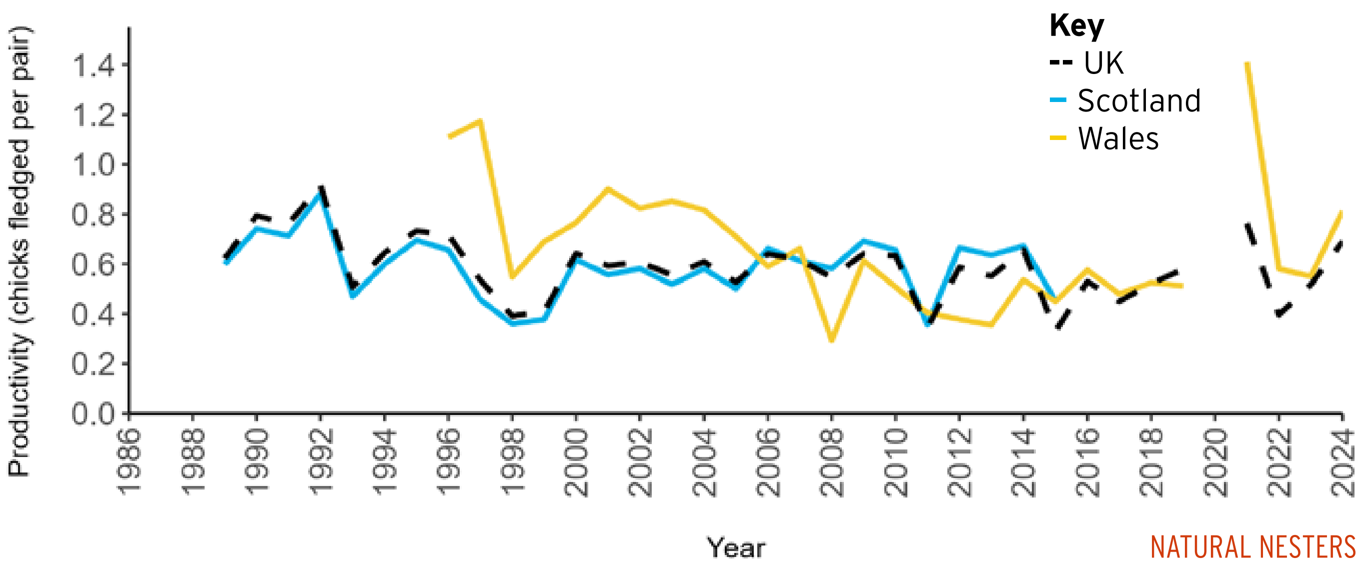


Figure 43: SMP Productivity (1986–2024)



PHENOLOGY, DIET AND SURVIVAL RATES

No systematic data on phenology have been collected as part of the SMP. However, at the Key Site of Canna (Scotland), information on diet is collected and adult survival rates of Herring Gull are estimated on the Key Site of Skomer Island (Wales) and are published in the Key Site reports.

CAUSES OF CHANGE

Pressures affecting Herring Gulls include diseases such as botulism (Coulson 2015, Madden & Newton 2004) and HPAI (Melville & Shortridge 2006, Tremlett *et al.* 2024b). At present, however, HPAI appears to have had less effect on Herring Gulls than on many other seabird species (Tremlett *et al.* 2024b).

Changes in food availability at both national and local levels may also have impacts on Herring Gull populations. Waste management on refuse tips has improved in recent years, reducing the availability of food for gulls by both covering waste soon after arrival at the tip, and diverting food waste for composting (Coulson 2015, Madden & Newton 2004). This is of particular concern for overwintering survival for some populations (Olsson & Hentati-Sundberg 2017, Shlepr *et al.* 2021).

Additionally, the ban on fishery discards is likely to be restricting food supplies for Herring Gulls, which is known to affect their breeding success (Bicknell *et al.* 2013, Foster *et al.* 2017, Furness *et al.* 1992). Reductions in local fishing activity can also result in changes in diet for Herring Gulls, as was shown on Canna (Scotland), and can have a negative impact on their populations (Foster *et al.* 2017).

Herring Gulls are also susceptible to being caught as bycatch by fisheries, including by longlines, trawl nets and gillnets (Anderson *et al.* 2011, Žydelis *et al.* 2013).

Predation from non-native species such as American Mink has affected some colonies (Craik 2015). Predation by other gull species and potentially by reintroduced White-tailed Eagles may also be increasing the pressures Herring Gulls face (Coulson 2019, Evans *et al.* 2009).

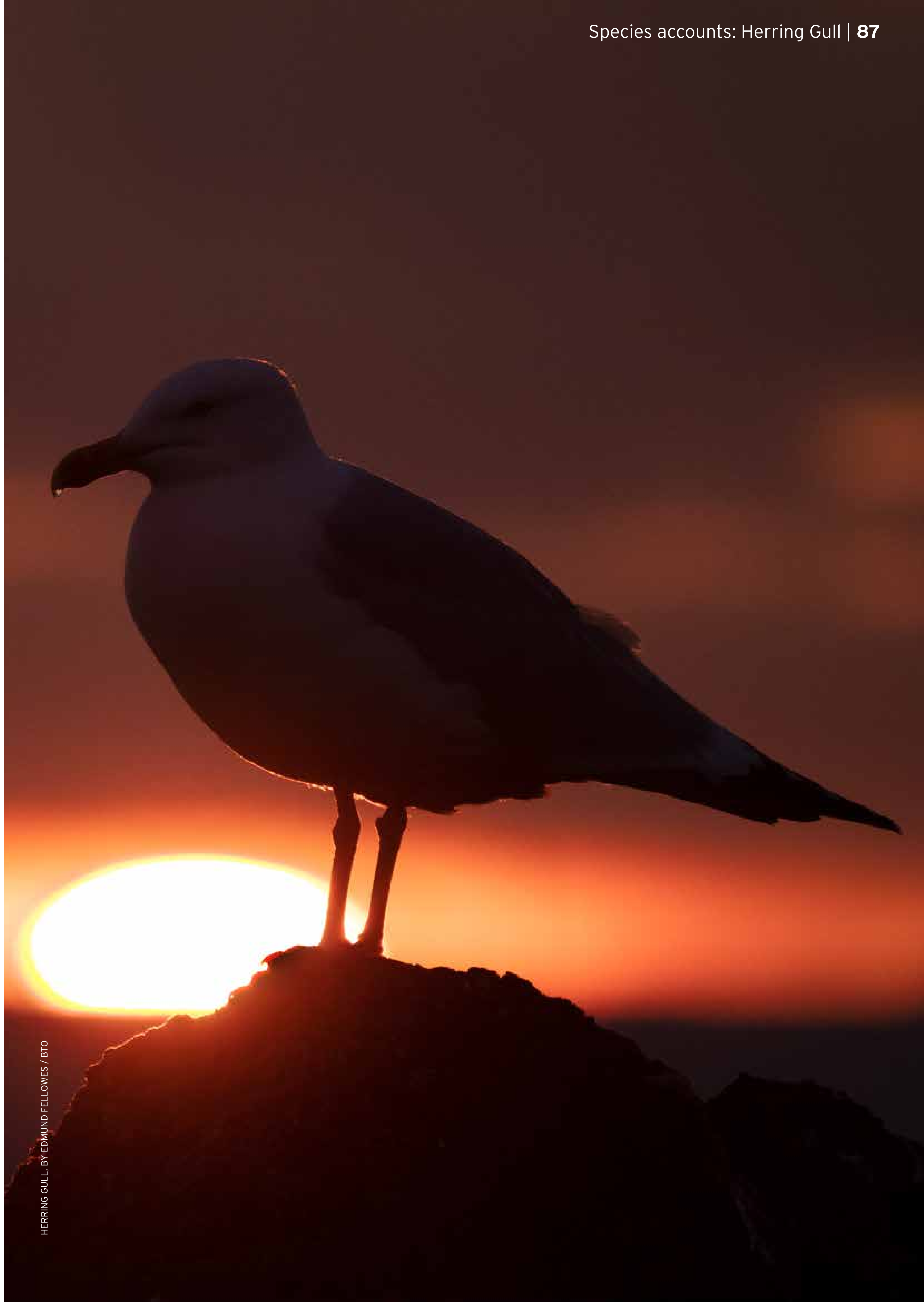
Wind farms can pose an additional threat, with Herring Gulls shown to be attracted to offshore wind farms, potentially for roosting or foraging opportunities (Vanermen *et al.* 2015), and they are considered to have a very high risk of collision mortality (Bradbury *et al.* 2014, Newton & Little 2009).

Historically, deliberate culling is thought to have contributed to declines in Herring Gull numbers (Coulson 2015) and, as they expand more into urban areas, conflict with humans is likely to increase (Rock 2005).

CONSERVATION

Conservation measures which could potentially benefit Herring Gulls include predator control, such as the removal of American Mink, which has led to higher breeding success in some colonies (Coulson 2019).

Maintaining legal limits on Herring Gull control through the licensing process will also be beneficial to this species (2009/147/EC). The expansion of the population into urban environments will require sensitive local management to resolve human-gull conflicts, and education programmes and the use of non-lethal deterrents will be important. The future development of improved monitoring techniques for challenging urban environments will also aid in determining whether proposed control methods are sustainable for Herring Gull populations (Burnell *et al.* 2023, Rock 2005). ■


HERRING GULL, BY EDMUND FELLOWES / BTO

Great Black-backed Gull

Larus marinus

c.7%

Abundance: Decline
Productivity: 1.24

Red-listed
Green-listed (I)

Colony Count sites: 125
Breeding Success sites: 9

Least Concern

Lifespan: 12 years
Breeding age: 5 years


Coverage in 2024

Britain and Ireland host approximately 7% of the global and 9–14% of the European breeding population of Great Black-backed Gull (Burnell *et al.* 2023). In the recent BoCC5a assessment, they were moved from Amber to the Red List (highest concern) (Stanbury *et al.* 2024).

DISTRIBUTION

Following a dramatic expansion in range in the 20th century, Great Black-backed Gulls currently breed across much of northern and western Britain and Ireland, predominantly around the coastline (JNCC 2021, Langlois Lopez *et al.* 2022). Globally, they breed across North Atlantic coasts, and on the Great Lakes of North America, with strongholds in Norway, Canada and Iceland (BirdLife International 2024).

Great Black-backed Gulls that breed in the north of their range typically migrate further south for the winter, and Norwegian and Swedish-ringed Great Black-backed Gulls have been found in the UK during the winter months (Robinson *et al.* 2024). British and Irish Great Black-backed Gulls can be found overwintering around much of the coastline and also at inland sites. Winter numbers are increased by migrants from further north in their range (Wernham *et al.* 2002).

DIET

Great Black-backed Gulls are generalist foragers and eat both natural prey (fish, birds and their eggs, small mammals and marine invertebrates) and human-discarded waste, including fishery discards and scavenged food from landfills (Buckley 1990, Taylor *et al.* 2012).

BREEDING

Great Black-backed Gulls primarily nest on rocky coastlines and coastal grasslands, with small numbers nesting on inland islands and rooftops (del Hoyo *et al.* 1996, JNCC 2021). They nest as single pairs or in small colonies, often within mixed-species colonies (del Hoyo *et al.* 1996). Three eggs are usually laid in a scrape lined with vegetation (Burnell *et al.* 2023).

BREEDING ABUNDANCE

The direction of change in SMP breeding abundance for Great Black-backed Gulls since 2000 (Table 31) agrees with those reported by the *Seabirds Count* census (Burnell *et al.* 2023), with declines in the UK and Scotland, but an increase in Wales. However, the extent of change differs for the UK and Wales. At the UK level, a decline of 20% was recorded by the SMP between 2000 and 2024, whilst the *Seabirds Count* census showed a decrease of 52% since the *Seabird 2000*. For Scotland the SMP change showed a decline of 57% since 2000, whilst the *Seabirds Count* census reported a similar decline of 63% since *Seabird 2000*. By contrast, the SMP change for Wales showed an increase of 76% since 2000, whilst the *Seabirds Count* census reported an increase of 49% over a similar period.

Over much of the SMP reporting period, the UK and Scotland long-term abundance trends for Great Black-backed Gulls have been relatively similar (Figures 44 & 45), as many of the colonies monitored are located in Scotland. Both trends increased during the 1990s but declined thereafter, particularly for Scotland, and both have remained below the 1986 baseline since the early 2000s. In 2024, the index values increased slightly to 16% below the baseline for the UK and 59% below for Scotland (Table 31). By contrast, the trend for Wales increased over the recording period until 2015 (Figure 46). Although it has since declined, the index value for Wales was 170% above the baseline in 2024 (Table 31). However, the wide confidence limits mean this index should be used with caution. Too few data are submitted to the SMP in other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

The UK and Scotland productivity trends for Great Black-backed Gulls are closely matched over the SMP monitoring period (Figure 47), as a large proportion of the monitored nests are located in Scotland. There was an overall decline in productivity for both Scotland and the UK until 2005, after which values increased to a peak in 2021. Following low values in 2022 (Table 4 & 2), which is likely to have been an effect of HPAI, higher productivity estimates were reported in both 2023 and 2024, with 1.24 chicks fledged per pair in the UK and 1.23 chicks fledged per pair in Scotland in 2024 (Table 31). Too few data are submitted to the SMP on productivity of Great Black-backed Gulls in other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No systematic data on phenology or survival have been collected as part of the SMP. However, information on Great Black-backed Gull diet is collected at the Key Site of Skomer Island (Wales), by monitoring prey remains around a sample of nests once chicks have fledged, and is published in the Key Site reports for the island.

CAUSES OF CHANGE

The decline in the Great Black-backed Gull population across Britain and Ireland is also reflected globally. The global population is estimated to have declined by 43%–48% between 1985 and 2021, with the causes largely unclear in all regions (Langlois Lopez *et al.* 2022). Increases observed in the south of the global Great Black-backed Gull range could be a result of immigration from the northern population, although these increases are small compared to the overall declines (Langlois Lopez *et al.* 2022).

In the 19th century, Great Black-backed Gulls were hunted for the millinery trade and since then fell victim to dichloro-diphenyl-trichloroethane (DDT) in the environment (Burnell *et al.* 2023).

Although predation by American Mink on Great Black-backed Gulls appears to be less severe than for other seabirds (Nordström *et al.* 2003), this non-native predator has been shown to cause declines in some years (Craik 2015). White-tailed Eagles, numbers of which are currently increasing in Scotland, may also reduce gull breeding productivity through increased disturbance (Billerman 2020, Hipfner *et al.* 2012).

Although Great Black-backed Gulls have a generalist and adaptable diet, any reductions in food availability, such as the recent ban on fishery discards, may potentially impact on their populations (Reeves & Furness 2002, Reid 2004, Wilhelm *et al.* 2016). Bycatch from fishing activities is an additional potential threat, but current levels are not thought to be significant for this species (Billerman 2020, Christensen-Dalsgaard *et al.* 2022).

Great Black-backed Gulls are considered to have a very high risk of collision mortality with turbines in offshore wind farms (Bradbury *et al.* 2014, Furness *et al.* 2013). This is exacerbated by gulls being attracted to offshore wind farms, potentially for roosting or foraging opportunities (Vanermen *et al.* 2015).

CONSERVATION

Further research is needed to identify the primary drivers of the decline seen in Great Black-backed Gull populations, so that appropriate actions can be implemented. At present, local measures, such as control of non-native predators, where they are demonstrated to have had negative impacts, may be effective (Hunt & Heffernan 2007). ►

Table 31: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	8,021	98	-16	-20	1.24	9
Scotland	5,404	62	-59*	-57*	1.23	5
Wales	648	12	170*	76	-	-

* significant changes

Table 32: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	19,739	11,265	-43

GREAT BLACK-BACKED GULL, BY EDMUND FELLOWES / BTO

Figure 44: UK SMP Breeding Abundance (1986–2024)

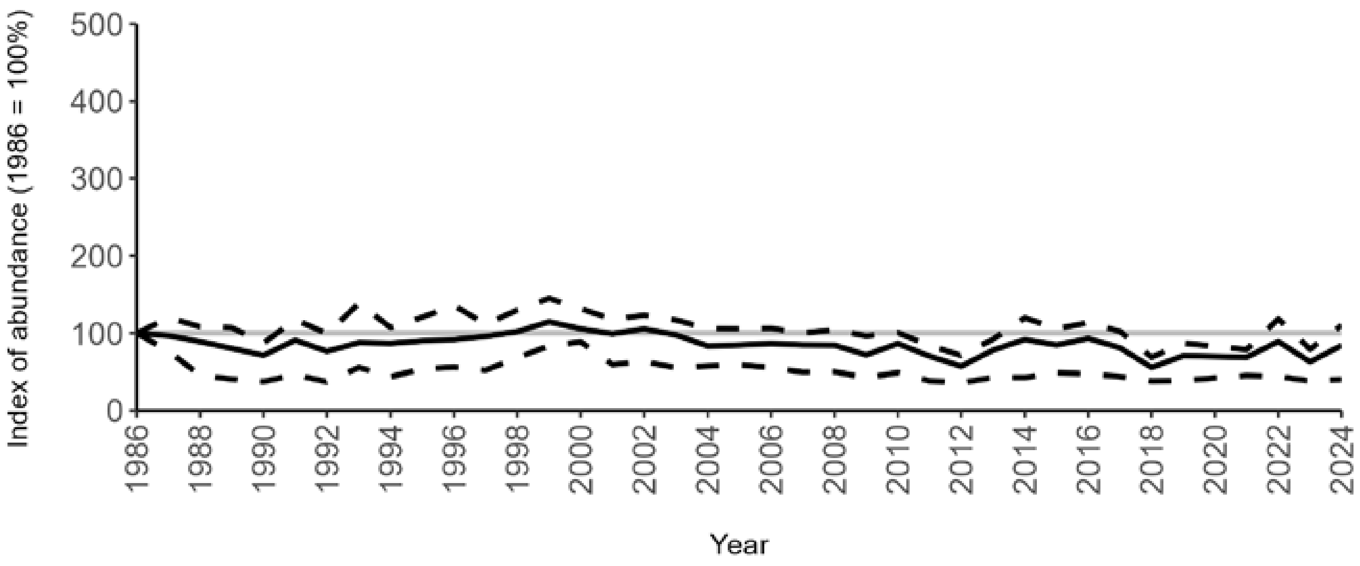


Figure 45: Scotland SMP Breeding Abundance (1986–2024)

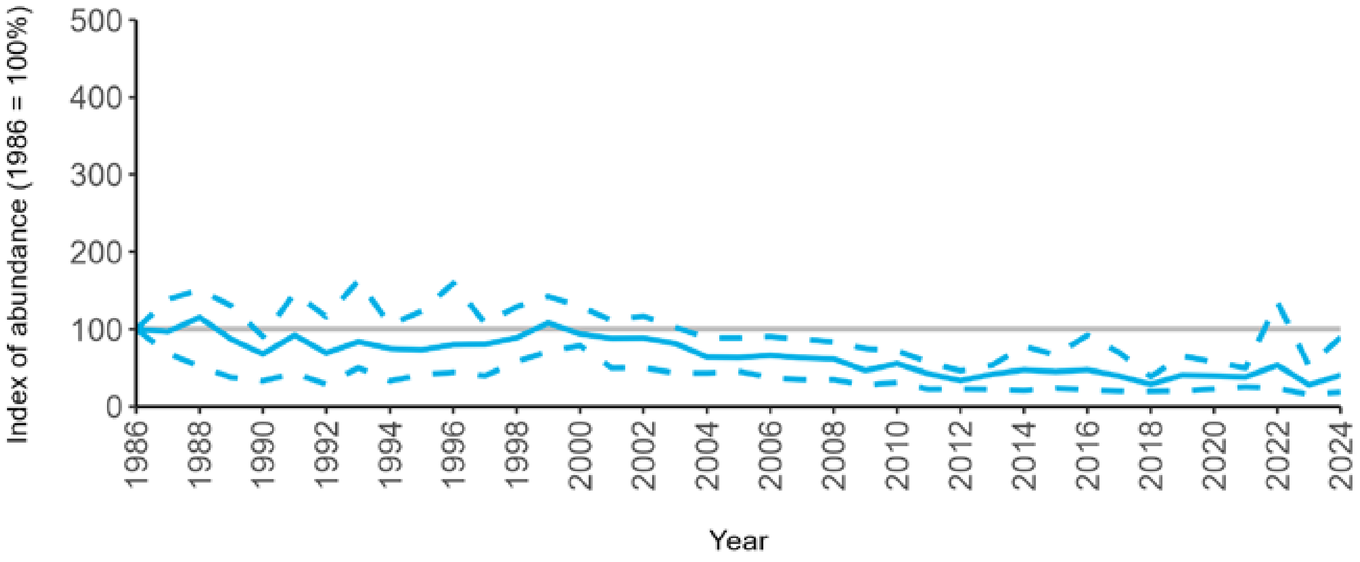


Figure 46: Wales SMP Breeding Abundance (1986–2024)

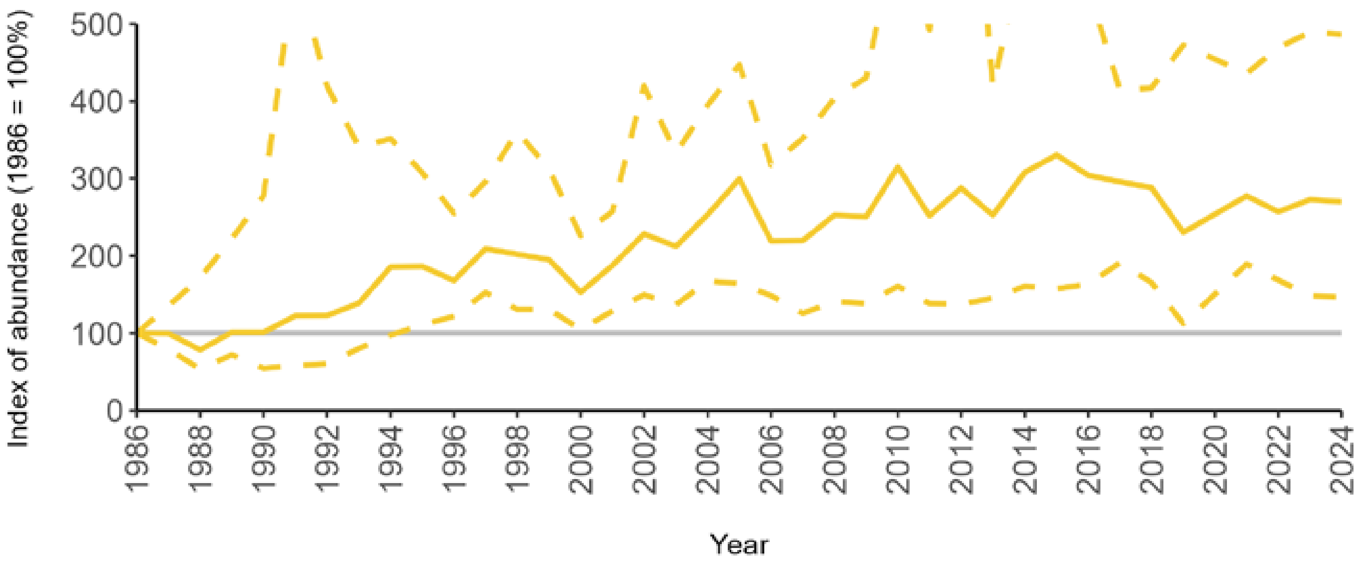
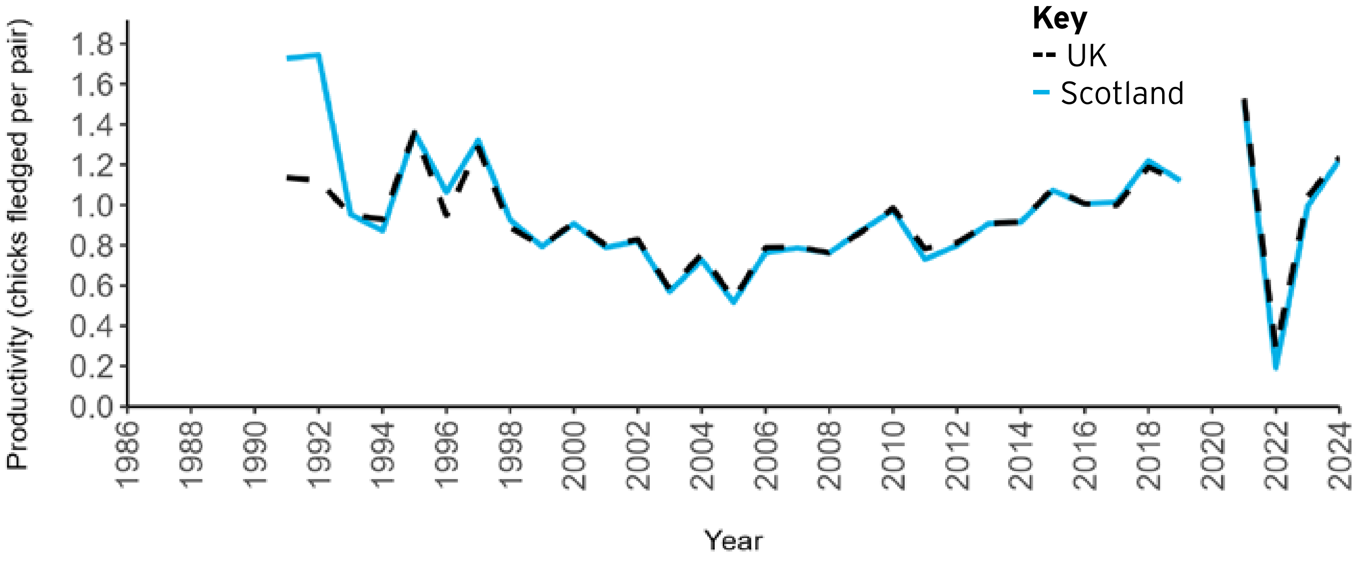


Figure 47: SMP Productivity (1986–2024)



Kittiwake

Rissa tridactyla

c.12%
ssp. tridactyla

Abundance: Decline
Productivity: 0.72

Red-listed
Red-listed (I)

Colony Count sites: 121
Breeding Success sites: 36

Vulnerable

Lifespan: 12 years
Breeding age: 4 years

Britain and Ireland host approximately 6% of the global breeding population of Kittiwake, and around 12% of the subspecies *tridactyla* (Burnell *et al.* 2023). Kittiwake are thought to be the most numerous gull in the world (JNCC 2021).

DISTRIBUTION

Kittiwake colonies are widely distributed around the British and Irish coastline, wherever suitable nesting habitat is available (Burnell *et al.* 2023). Globally, they breed on the coastlines of the North Atlantic, Arctic and North Pacific Oceans (Coulson 2011).

In winter Kittiwakes are generally oceanic, with some individuals travelling over 3,000 km from their nesting colonies to feeding hotspots (Bogdanova *et al.* 2011, Burger *et al.* 2016). Despite this, during the winter, they can be seen around British and Irish coasts (del Hoyo *et al.* 1996) as some birds appear to stay close to their nesting colony during the non-breeding season (Frederiksen *et al.* 2012).

DIET

Kittiwakes are surface feeders and plunge-dive or dip feed to catch their prey. They mainly eat small pelagic shoaling fish such as sandeels, clupeids and gadids, or invertebrates (Bull *et al.* 2004, Chivers *et al.* 2012, Swann *et al.* 2008, Wanless *et al.* 2018), but will also take offal from fishery discards (Coulson 2011). Their foraging range during the breeding season typically covers tens of kilometres (Daunt *et al.* 2002, Trevail *et al.* 2019).

BREEDING

Kittiwakes nest on narrow ledges on vertical rocky sea-cliffs or, less commonly, on artificial structures, e.g. bridges, buildings and offshore oil installations. The nest is made of mud, lined with grass and seaweed, and holds up to three eggs per brood (Coulson 2011, del Hoyo *et al.* 1996, JNCC 2021). Colony size can vary from a few pairs to tens of thousands, and the nests are typically evenly spaced, 30–60 cm apart (Snow & Perrins 1998).

BREEDING ABUNDANCE

The declines in SMP abundance for Kittiwake since 2000 (Table 33) are broadly similar to those reported by the *Seabirds Count* census in most regions (Burnell *et al.* 2023). The decrease of 38% for the UK recorded by the SMP since 2000 is similar to the decline of 43% shown by the *Seabirds Count* census. The SMP 24-year change in England showed a decline of 26%, in contrast to a minor decrease of 4% recorded between the censuses. For Scotland and Wales, declines of 41% and 53%, respectively, were recorded by the SMP since 2000, whilst the *Seabirds Count* census showed decreases of 57% and 34%, respectively, over a similar period (Burnell *et al.* 2023).

The long-term SMP abundance trends for Kittiwake all show declines since the mid-1990s, although the magnitude varies across regions (Figures 48–51). The trends for the UK and Scotland (which are closely matched, as most monitored colonies are in Scotland) declined more severely than those for England and Wales. However, all have consistently remained below the 1986 baseline since the late 1990s, despite some annual fluctuations. In 2024, the index values for all regions were similar, ranging between 43% and 56% below the baseline (Table 33).

Too few Kittiwake abundance data are submitted from other regions to produce valid SMP abundance trends.

KITTIWAKES, BY JONNIE FISK

PRODUCTIVITY

There has been considerable variation in Kittiwake productivity between regions and years across the SMP recording period (Figure 52). All regional trends experienced an overall decline until 2008, following which productivity values increased slightly across all regions. In 2024, productivity estimates for the UK, Scotland, England and Wales were all similar, at between 0.61 and 0.72 chicks fledged per pair (Table 33). Productivity data were submitted for only a few sites in Northern Ireland and the Republic of Ireland in 2024, however, in 2023 the productivity estimate for all-Ireland was 0.43 chicks fledged per pair (Table 33). Too few data are submitted to the SMP on productivity of Kittiwake from other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No systematic data on phenology have been collected as part of the SMP. However, at the Key Sites, information on diet is collected on Canna and the Isle of May (both in Scotland). Data on adult survival are also collected on Canna and Skomer Island (Wales), and on adult annual return rates on the Isle of May.

CAUSES OF CHANGE

Reduced persecution during the early 20th century is thought to have allowed population increases. However, declines have occurred thereafter (Coulson 2011). Food availability is likely to play a key role in Kittiwake population dynamics, and surface-feeding seabirds, such as Kittiwakes, are thought to be more vulnerable to

▶

Table 33: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	215,913	99	-55*	-38*	0.72	30
England	72,897	22	-43	-26	0.67	9
Scotland	121,082	63	-53*	-41*	0.71	15
Wales	4,782	10	-56*	-53*	0.61	2
All Ireland	-	-	-	-	0.43 (2023)	5 (2023)

* significant changes

Table 34: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	418,780	241,321	-42

KITTIWAKES, BY EDMUND FELLOWES / BTO

Figure 48: UK SMP Breeding Abundance (1986–2024)

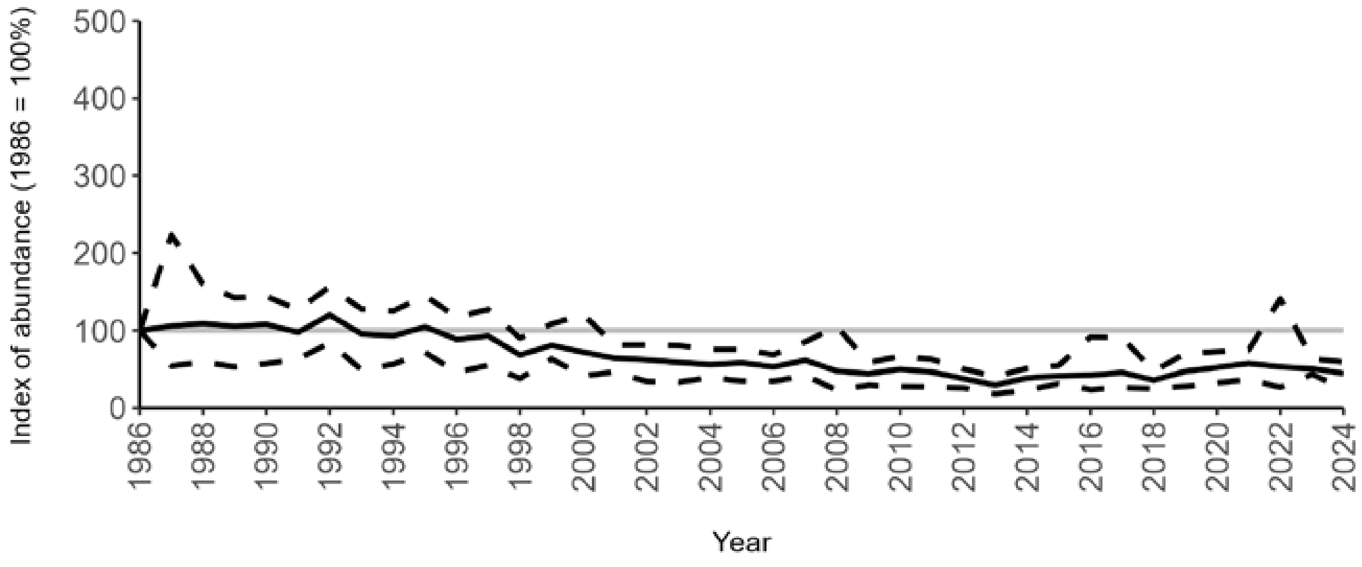


Figure 49: England SMP Breeding Abundance (1986–2024)

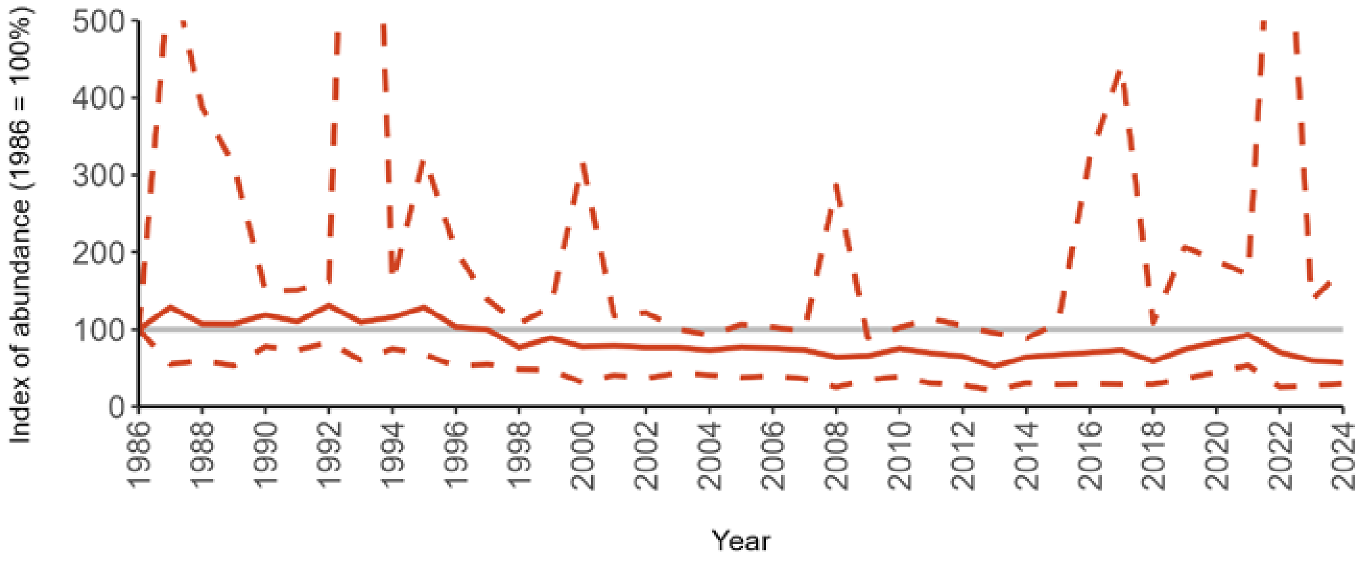


Figure 50: Scotland SMP Breeding Abundance (1986–2024)

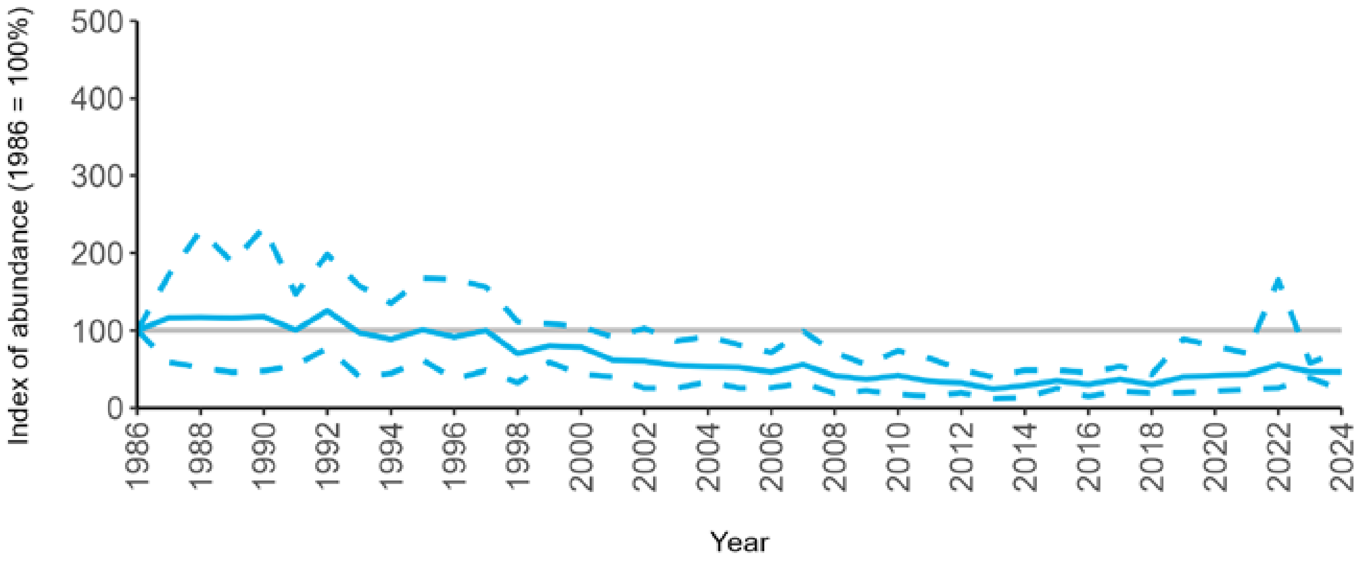


Figure 51: Wales SMP Breeding Abundance (1986–2024)

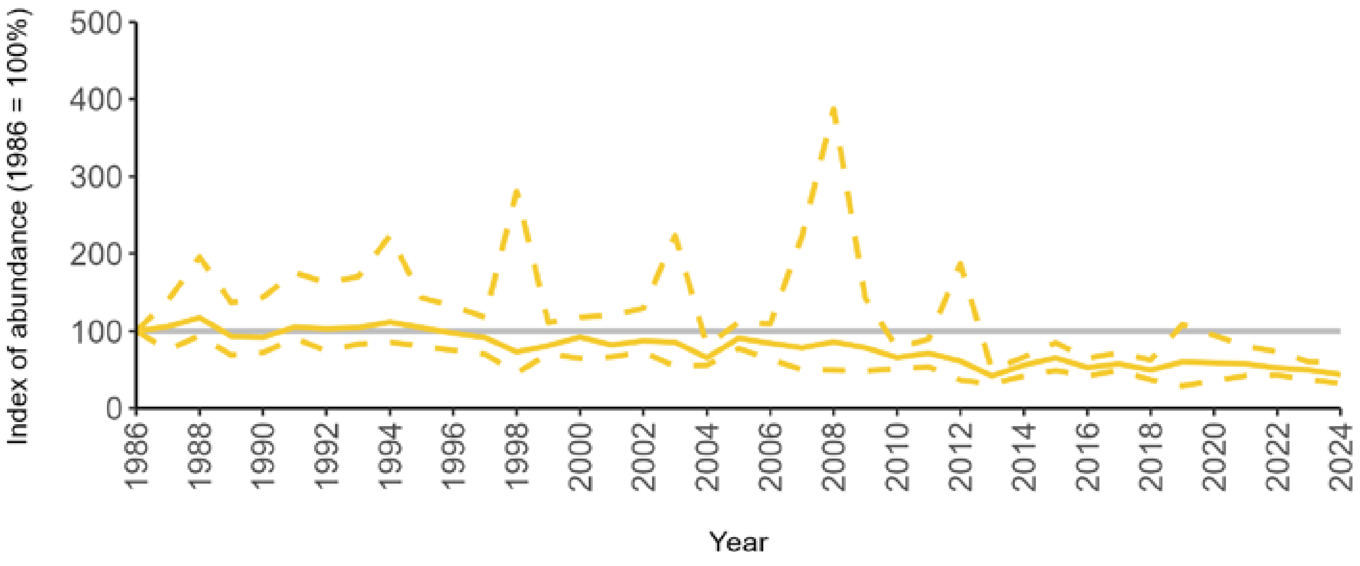
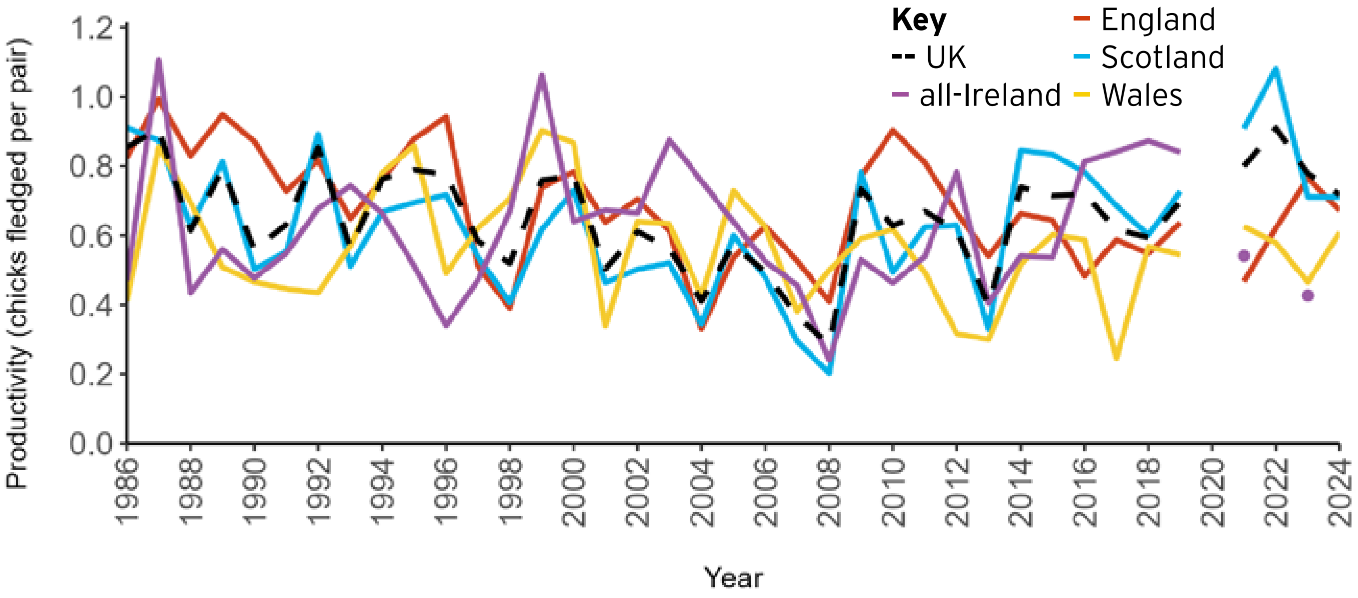


Figure 52: SMP Productivity (1986–2024)



changes in prey availability than deeper diving species (Furness & Tasker 2000, Wanless *et al.* 2007).

Increases in sea surface temperatures due to climate change are causing changes in the distribution and timing of development of the Kittiwake's preferred prey, the Lesser Sandeel (*Ammodytes tobianus*) (Wanless *et al.* 2004), both directly and through indirect effects on the copepod prey of sandeels. The impacts of reduced sandeel abundance have been demonstrated in Kittiwake colonies that have traditionally relied on them. When chicks were fed on alternative prey items, due to sandeel shortages, both adult survival and chick development were reduced (Christensen-Dalsgaard *et al.* 2018, Paredes *et al.* 2014, Régnier *et al.* 2019, Sandvik *et al.* 2005, Wanless *et al.* 2018).

Reductions in food availability may also be caused by the presence of fisheries close to Kittiwake breeding colonies, and studies have shown that increased fishing effort in the Irish, Celtic and North Sea areas may be decreasing the amount of suitable prey, and reducing Kittiwake breeding success as a consequence (Frederiksen *et al.* 2007). Positive effects on breeding success have also been shown following bans on local sandeel fisheries in some areas (Searle *et al.* 2023).

However, the overall picture is complex and likely to differ between regions (Frederiksen *et al.* 2007), and both abundance trends and productivity values can vary considerably between different areas within Britain and Ireland (Burnell *et al.* 2023). Studies have been carried out on the Isle of May (Scotland) which show negative correlations between adult survival and breeding success and increasing sea surface temperatures (Frederiksen *et al.* 2004), but this link is not clear for some colonies in other areas (Cook *et al.* 2014, Carroll *et al.* 2017, Eerkes-Medrano *et al.* 2017).

Climate change can also impact Kittiwake breeding success through an increase in extreme weather events, which can result in nests being washed away during heavy summer rainfall, or hinder foraging ability in winter storms (Clairbaux *et al.* 2021, Newell *et al.* 2015, Stubbings *et al.* 2017).

Further pressures include potential foraging displacement and collision risk following the development of offshore wind farms (Drewitt & Langston 2006, Masden *et al.* 2010).

Cases of HPAI have been recorded at a number of Kittiwake colonies in recent years (Falchieri *et al.* 2022, Tremlett *et al.* 2024b), which has the potential to cause impacts on local population levels in areas which have been severely affected. However, it is also possible that HPAI could have a net positive effect on Kittiwake populations in areas such as Orkney and Shetland, where numbers of Great Skua, which can heavily predate both Kittiwake adults and chicks, have dramatically declined due to HPAI (Burnell *et al.* 2023, Heubeck *et al.* 1999, Heubeck 2002, Votier *et al.* 2004).

CONSERVATION

Management of fisheries, such as the closure of industrial sandeel fishing in English and Scottish waters in 2024, may increase prey availability and is likely to be beneficial to Kittiwake populations e.g. the breeding success rate of Kittiwakes on the Isle of May (Scotland) was shown to increase following the closure of a local sandeel fishery (Daunt *et al.* 2008, Frederiksen *et al.* 2004).

Identification of key Kittiwake foraging areas through tracking technologies may also aid in ensuring these regions are appropriately conserved through statutory protection (Christensen-Dalsgaard *et al.* 2018). ■


KITTIWAKES, BY EDMUND FELLOWES / BTO

Sandwich Tern

Thalasseus sandvicensis

c.9%

Abundance: Decline
Productivity: 0.58

Amber-listed
Amber-listed (I)

Colony Count sites: 20
Breeding Success sites: 12

Least concern

Lifespan: 12 years
Breeding age: 3 years

Britain and Ireland host approximately 9% of the world's breeding Sandwich Terns (Burnell *et al.* 2023). The oldest known Sandwich Tern reached 30 years, 9 months and 14 days (BTO 2025).

DISTRIBUTION

Sandwich Terns are found in a relatively small number of large breeding colonies around the coastlines of Britain and Ireland, with a stronghold in East Anglia (BTO 2025, Burnell *et al.* 2023).

Although overall colony distributions in Britain and Ireland have remained largely the same over time, the species is known to frequently switch breeding sites (Wernham *et al.* 2002). Globally, they breed in western Europe and around the Mediterranean, Black and Caspian Seas (del Hoyo *et al.* 1996).

Most are migratory, and British and Irish birds are known to overwinter around the West African coast, some travelling over 4,000 km to their wintering grounds (BTO 2025, Wernham *et al.* 2002).

DIET

Sandwich Terns feed mainly by plunge-diving up to 2 m in depth, preying on small fish, marine worms and shrimp, but will also take nestling shorebirds (del Hoyo *et al.* 1996). During the breeding season, the dominant prey items are Atlantic Herring (*Clupea harengus*), European Sprat (*Sprattus sprattus*) and sandeel (Green 2017).

BREEDING

Sandwich Tern colonies in Britain and Ireland vary in size and location from year to year. However, they favour coastal habitats, including low-lying islets and spits, remote dunes, or around bays or brackish lagoons, and they typically nest in high densities (JNCC 2021).

Nests are unlined scrapes in dry ground on shingle or low herbage, and they avoid dense or tall vegetation. The usual clutch size is 1–2 eggs, occasionally three (BTO 2025).

BREEDING ABUNDANCE

Sandwich Tern abundance changes since 2000 for the UK and England, reported by the *Seabirds Count* census, indicated stable populations (Burnell *et al.* 2023). In contrast, the 24-year SMP changes for the UK and England show declines of 24% and 22%, respectively (Table 35), likely a consequence of the 2022 HPAI outbreak, which occurred after the census period.

Most Sandwich Tern colonies monitored as part of the SMP are in England, therefore the UK and England long-term abundance trends are closely matched (Figures 53 & 54). Following decline in the trends between 1986 and 2008, there was a gradual increase from 2010 to peak index values in 2021. The 2009 spike in the index was due to an influx of birds, apparently from continental Europe, nesting at Minsmere (Suffolk) (JNCC 2021). However, the indices have since declined to values of 26% and 27% below the 1986 baseline in 2024 for the UK and England, respectively (Table 35). This is likely to have been a result of the 2022 HPAI outbreak, which severely impacted Sandwich Tern populations across north-western Europe (Knief *et al.* 2024).

Too few data are currently submitted to the SMP from Scotland, Northern Ireland and the Republic of Ireland to allow for the calculation of meaningful abundance trends for these countries.

PRODUCTIVITY

SMP productivity trends for the UK and England are relatively similar (Figure 55), as most Sandwich Tern productivity data are collected in England. The productivity trend for Scotland, up to 2019, has values generally considerably lower than for the other regions, particularly prior to the mid-2000s. Within all the regions for which trends can be produced, there are considerable fluctuations in values between years. Gaps in the trends for all-Ireland and Scotland, including since 2019, reflect years where insufficient data were submitted to the SMP to produce robust trends.

Following a peak in productivity values across all regions in 2000, there was a general decline in trends up to 2014. This was followed by an increase for the UK and England until 2017, whilst the trend for Scotland was more stable, albeit with large annual fluctuations (Figure 55). In 2023, productivity estimates were similarly low for the UK and England, with 0.17 and 0.20 chicks fledged per breeding pair, respectively (Tables 2 & 3) with HPAI likely to have been a factor in the decline in productivity since 2021. However, productivity estimates increased in 2024 to 0.58 and 0.77 chicks fledged per breeding pair, for the UK and England respectively (Table 35).

For Scotland, since 2020, data have been submitted only for Sands of Forvie, the country's largest Sandwich Tern colony, where productivity was 0.26 chicks per breeding pair in 2024. Too few data are submitted to the SMP on productivity of Sandwich Terns in other regions to allow for calculation of productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

The Sandwich Tern population has been relatively stable over the course of the SMP monitoring period, but local fluctuations can be experienced due to colony movements in response to human disturbance, reductions in food availability, predation pressure or habitat changes (Burnell *et al.* 2023, Garthe & Flore 2007, Gochfield *et al.* 2018, Wernham *et al.* 2002). Additional human impacts on Sandwich Terns include the trapping of birds in their wintering grounds (Stienen *et al.* 1998), and hunting of adults which has caused severe declines in some areas of their global range (Gochfield *et al.* 2018). ▶

SANDWICH TERN, BY PHILIP CROFT / BTO

Table 35: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	12,980	15	-26*	-24*	0.58	11
England	9,503	9	-27	-22	0.77	5
Scotland	-	-	-	-	0.26	1

* significant changes

Table 36: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	14,257	15,484	9

Red Foxes, European Badgers and European Otters are known to take Sandwich Tern eggs and chicks (Ratcliffe *et al.* 2000, Short 2014). Although large gull species will predate eggs and young, and take food from returning adult birds, the benefits of nesting near larger species, which keep other predators at bay, is thought to outweigh the costs (Stienen 2006).

Climate change has the potential to impact Sandwich Tern populations in Britain and Ireland in the future through the loss of suitable habitat and changes in prey availability. The predicted increases in extreme weather events, alongside erosion, sea level rise and extreme high tides are all potential threats (Johnston *et al.* 2021). The strong winds associated with increased storminess may also reduce their foraging efficiency, impacting on their ability to maintain body condition and feed chicks (Burnell *et al.* 2023, Taylor 1983).

In an attempt to combat climate change, the number of offshore wind farms is increasing, and this has the potential to cause mortalities in Sandwich Terns through collisions with turbines (Furness *et al.* 2012).

HPAI caused severe declines during 2021–22, with high mortality rates being recorded at colonies across northern Europe (Knief *et al.* 2024), a 35% decline in the population at surveyed sites in the UK between 2015–21 and 2023 (Tremlett *et al.* 2024b) and mass mortality at colonies in the Netherlands (Rijks *et al.* 2022).

CONSERVATION

Introduction of legislation to reduce the level of egg collecting and hunting of adults in the early 1900s successfully led to an increasing population trend between the 1920s and mid-1980s (JNCC 2021), prior to SMP monitoring.

The most effective conservation measures for Sandwich Terns are likely to be the continuation of current local management measures, such as predator fencing, site patrols to restrict recreational disturbance (Short 2020), and habitat management to maintain or create suitable nesting habitats (Burgess & Hirons 1992, Fasola & Canova 1996). Decoys can also be used to attract individuals to new, suitable, nesting habitats (del Hoyo *et al.* 1996).

Figure 53: UK SMP Breeding Abundance (1986–2024)

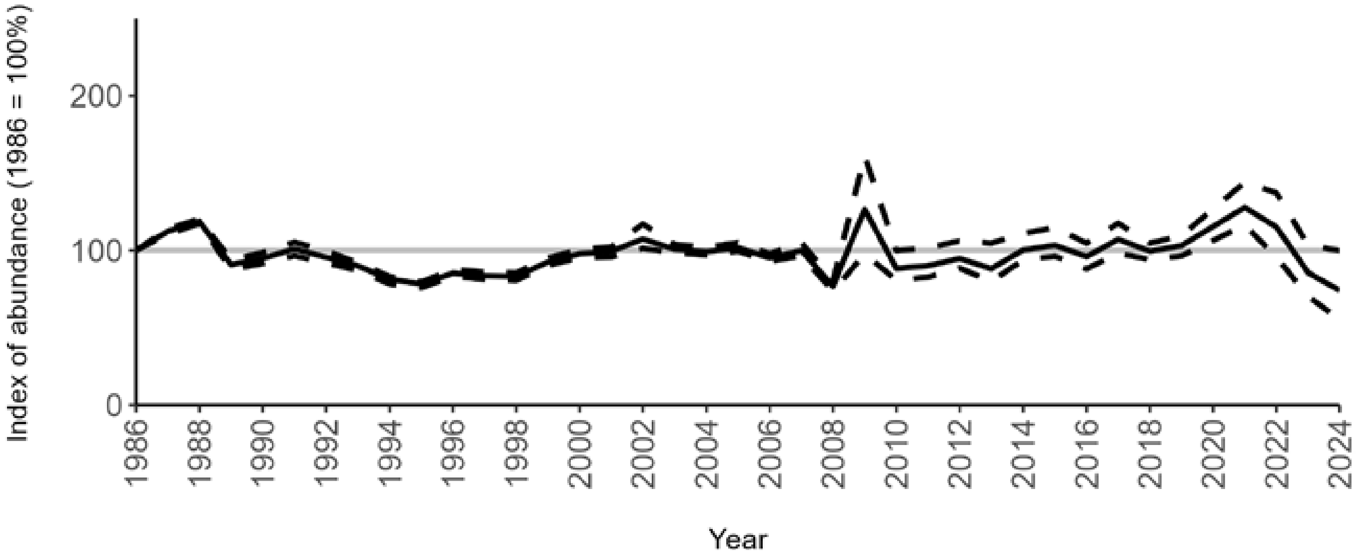


Figure 54: England SMP Breeding Abundance (1986–2024)

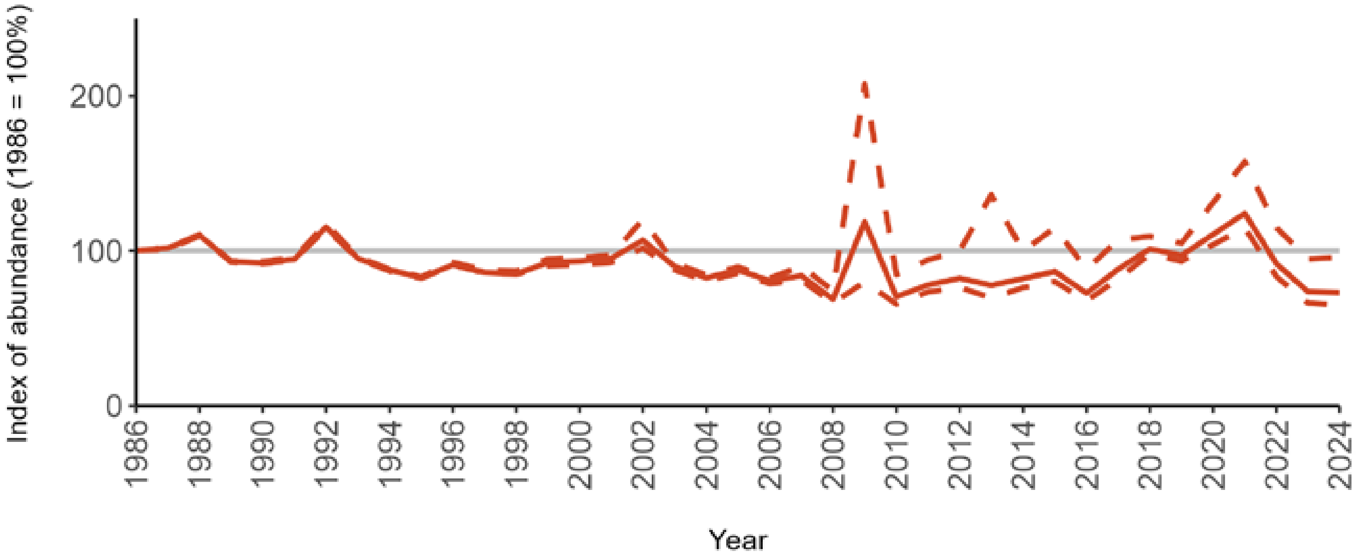
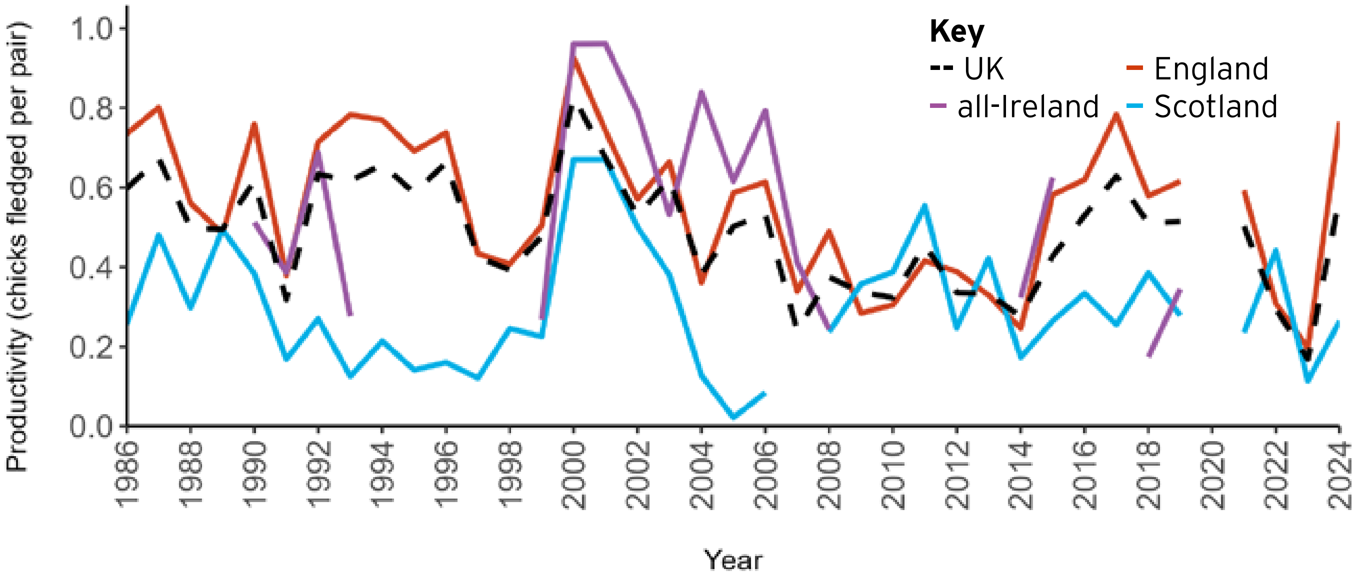


Figure 55: SMP Productivity (1986–2024)



SANDWICH TERN, BY MOSS TAYLOR / BTO

Roseate Tern

Sterna dougallii

c.2%

Abundance: Decline
Productivity: 0.85

Red-listed
Amber-listed (I)

Colony Count sites: 5
Breeding Success sites: 4

Least Concern

Lifespan: 8 years
Breeding age: 2 years

Britain and Ireland host less than 2% of the world's breeding Roseate Terns (Burnell *et al.* 2023). Of those, 94% breed in the Republic of Ireland (Burnell *et al.* 2023).

DISTRIBUTION

In Britain and Ireland, breeding Roseate Terns are restricted to just a few colonies. They are very widely distributed across the globe, and are found in both the northern and southern hemispheres (BirdLife International 2024). Our Roseate Terns migrate south in the non-breeding season (Snow & Perrins 1998).

DIET

Roseate Terns feed by plunge-diving on a variety of species of small fish depending on the location, with sandeels being of particular importance in the North Atlantic (Newton & Crowe 2000), although they may also take insects and marine invertebrates (del Hoyo *et al.* 1996).

BREEDING

Roseate Terns have specialised foraging and nesting behaviours, which restrict their UK colonies to a few suitable areas (JNCC 2021). They nest on coastal lowland sites – beaches, shingle, saltmarsh and islands – especially near to suitable fishing grounds (del Hoyo *et al.* 1996, Snow & Perrins 1998). They create shallow scrapes as nests and tend to favour densely vegetated sites in temperate regions (del Hoyo *et al.* 1996). Nest boxes are utilised at the main colonies in Britain and Ireland, and breeding success in these has been shown to be greater than in open nest sites (Burke *et al.* 2022).

BREEDING ABUNDANCE

As Roseate Terns are restricted to a small number of well-monitored colonies, SMP data cover the entire UK population rather than using a sample of colonies to produce estimated trends. The increase in the UK Roseate Tern population of 125% between 2000 and 2024 calculated from SMP data (Table 37) is similar to the increase of 114% reported by the *Seabirds Count* census since *Seabird 2000* (Burnell *et al.* 2023).

Over the course of the SMP monitoring period, the UK Roseate Tern population underwent a striking decline between 1986 and 1991 (Figure 56), thought to be due to low immature survival rates as a result of trapping and poor food availability on the species' wintering grounds off West Africa (Ntiamoa-Baidu *et al.* 1992). After a subsequent period of stability, the population has slowly started to recover due to a range of conservation measures, including provision of nest boxes for shelter and protection from avian predators, and appropriate habitat management on Coquet Island (England), where the UK population is currently largely found (JNCC 2021, Burnell *et al.* 2023). By 2022, the UK population had risen to 158 AON. The majority of these (154 AON) were breeding on Coquet Island, and an outbreak of HPAI at this colony after the count was carried out caused high levels of mortality (Tremlett *et al.* 2024a). Consequently, in 2023, the number of Roseate Tern breeding on Coquet Island declined to 118 AON, and the UK population to 121 AON. In 2024, the UK population increased slightly to 128 AON, 61% less than the 1986 population figure (Table 37).

Within Britain and Ireland, the Republic of Ireland holds the majority of breeding Roseate Terns at two colonies, Rockabill and Lady's Island Lake. Rockabill is the largest Roseate Tern colony in Europe and has steadily increased over the period, with 1,776 AON in 2024 (Figure 57), aided by habitat management and the provision of nest boxes, despite the colony being impacted by HPAI in 2023.

PRODUCTIVITY

The UK productivity trend for Roseate Tern has fluctuated across the SMP monitoring period (Figure 58), but with a gradual overall increase until recent years, partly due to effective conservation management. However, 2022 saw a decline in productivity to a value of 0.30 chicks fledged per pair (Table 37) due to HPAI on Coquet Island, which caused high mortality of chicks and adults. HPAI was also present in the colony in 2023, therefore no productivity data were submitted to the SMP. In 2024, the UK productivity estimate increased to 0.85 chicks fledged per pair (Table 2).

Too few data are submitted to the SMP on productivity of Roseate Terns in other regions to allow for calculation of productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

In recent decades the Roseate Tern population in Britain and Ireland has been growing. This has been driven mainly by increases in the Irish population through the use of effective local conservation measures. However, recovery in other regions is slow or absent.

Potential impacts on Roseate Terns include human disturbance and egg collecting, but this has been largely prevented by full-time wardening at the main colonies in Britain and Ireland. Predation from species such as Brown Rat, European Otter, Pine Marten (*Martes martes*) and European Badger, large gulls and even Turnstones (*Arenaria interpres*) remains a threat (Acampora *et al.* 2018, Avery *et al.* 1995), although this has been reduced by the use of predator fences, biosecurity measures and predator control where necessary (Burnell *et al.* 2023).

The removal of eggs or hunting of adults has caused severe declines in some areas of their range (Gochfield *et al.* 2018), including deliberate trapping in the non-breeding grounds of Ghana (Ntiamoa-Baidu *et al.* 1992, Ratcliffe & Merne 2002).

Climate change has the potential to impact colonies in the future through extreme weather events, sea level rise, erosion, and impacts on prey populations (Burnell *et al.* 2023).

HPAI caused severe population mortalities in 2022 and 2023 (Falchieri *et al.* 2022), with a 21% decrease in the Roseate Tern population at surveyed sites in the UK between 2015–21 and 2023 (Tremlett *et al.* 2024b).

CONSERVATION

Following the introduction of legislation to reduce the threat from the millinery trade in 19th century, the population increased through the early 20th century, before SMP monitoring began (JNCC 2021).

Successful Roseate Tern conservation measures include education at overwintering sites to reduce trapping of adults (Ratcliffe & Merne 2002), deployment of nest boxes to reduce predation and/or active predator management (Acampora *et al.* 2018), appropriate habitat and vegetation management, and wardening to reduce disturbance (Casey *et al.* 1995, Newton & Crowe 2000, Seward *et al.* 2019).

Examples of successful ongoing conservation management can be found in Ireland on Lady's Island Lake (Daly *et al.* 2020) and on Rockabill (Gill *et al.* 2019), as well as internationally, including restoration of a small islet in the Azores, Portugal (Praia Islet). Work on Praia included European Rabbit eradication, native plant reintroduction and nest box deployment (Bried *et al.* 2009). ▶

Table 37: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	120	3	-61	125	0.85	2

no significant changes

Table 38: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	790	1,989	152

Figure 56: Total UK Apparently Occupied Nests (1986–2024)

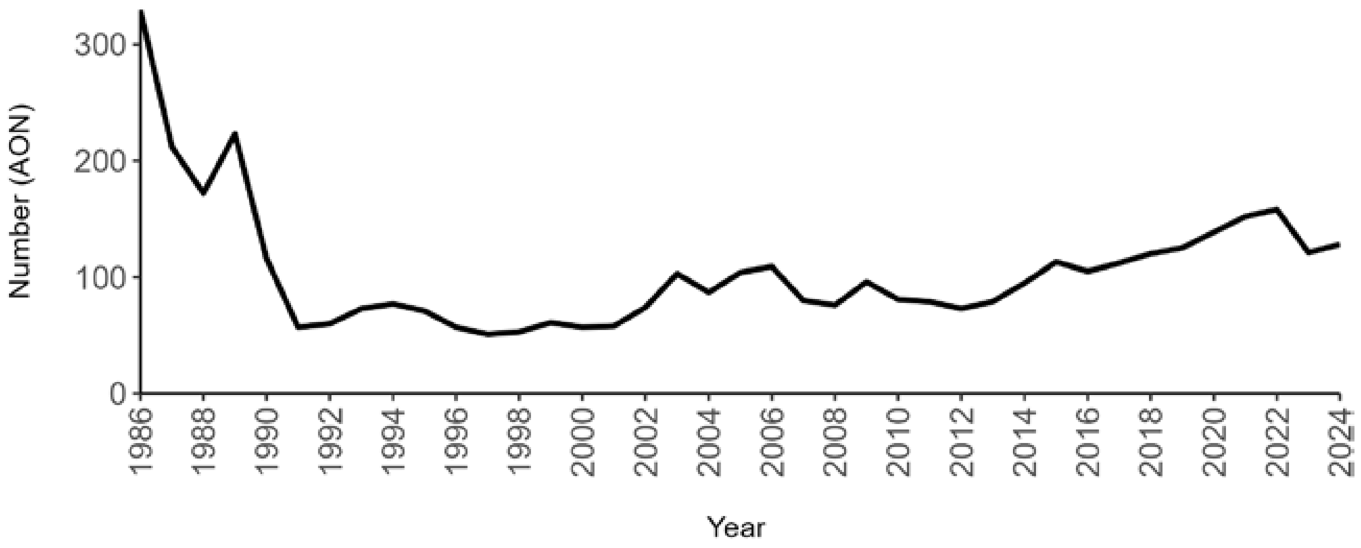


Figure 57: Total Rockabill Apparently Occupied Nests (1986–2024)

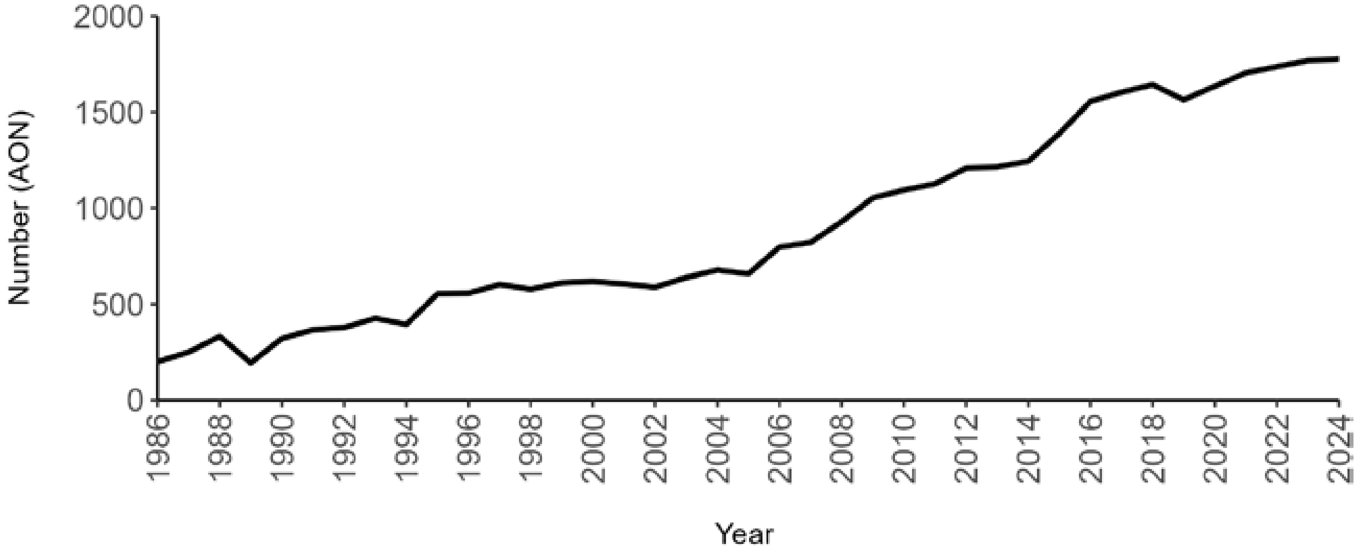
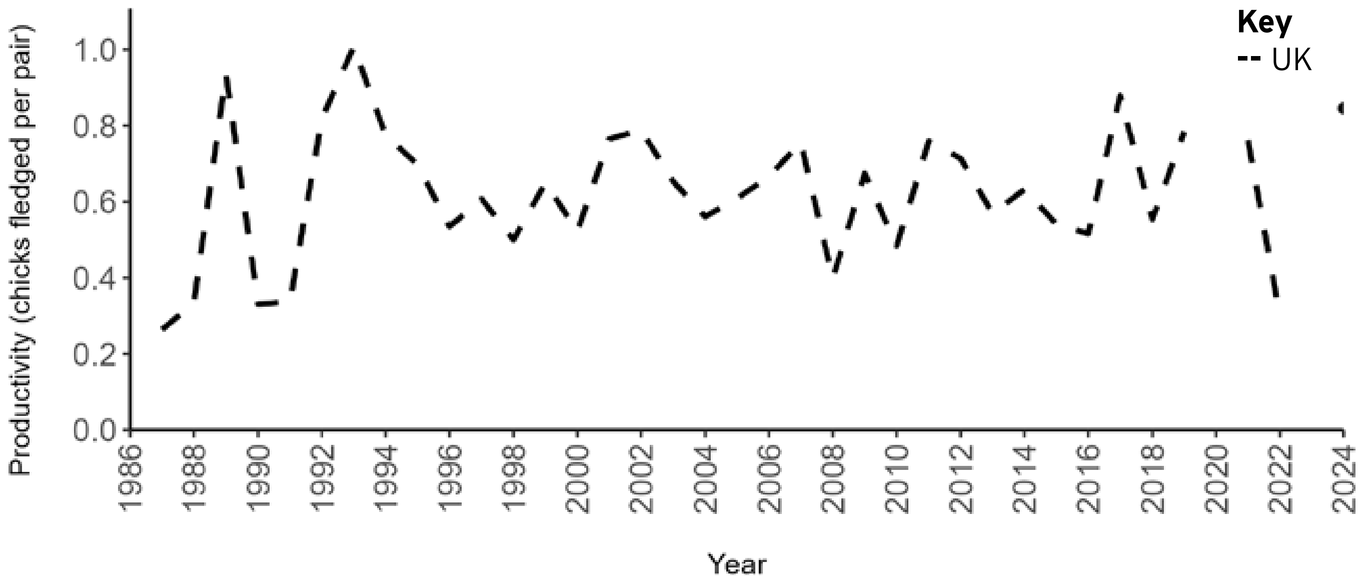


Figure 58: SMP Productivity (1986–2024)



ROSEATE TERN, BY SAM LANGLOIS / BTO

Common Tern

Sterna hirundo

c.1–2%

Abundance: Decline
Productivity: 0.27

Amber-listed
Amber-listed (I)

Colony Count sites: 126
Breeding Success sites: 39

Least Concern

Lifespan: 12 years
Breeding age: 3 years

Coverage in 2024

Britain and Ireland host approximately 1–2% of the world's breeding Common Terns (Burnell *et al.* 2023). Although not the most numerous tern in Britain and Ireland, they are probably the most familiar due to their widespread distribution, both inland and at the coast.

DISTRIBUTION

Common Terns breed around much of the British and Irish coast, although rarely in the south-west of Britain, and are also found at many inland sites (Burnell *et al.* 2023).

Their world distribution stretches around the globe, breeding in most of Europe, Asia and North America (BirdLife International 2024).

British and Irish birds migrate south after breeding to wintering grounds between the coast of Spain and the west African coast (Wernham *et al.* 2002).

DIET

Common Terns are opportunistic foragers, with a broader diet than many other tern species. They forage by plunge-diving over both marine and freshwater bodies, mainly preying on small fish and occasional insects and planktonic crustaceans (del Hoyo *et al.* 1996, Lemmetyinen 1973). They generally forage within 10 km of their breeding colony (Perrow *et al.* 2011, Wilson *et al.* 2014).

BREEDING

Common Terns nest on flat rocks, shingle, sandy beaches, dunes, spits and small islands around the coastline, and inland at gravel pits, lakes, river valleys and marshes (del Hoyo *et al.* 1996, Richards 1990, Snow & Perrins 1998). They are an adaptable species, and artificial structures are also used, such as rafts, docks, barges and rooftops (Burnell *et al.* 2023).

They nest in shallow depressions on open ground with some vegetation cover, and lay 2–3 eggs (del Hoyo *et al.* 1996).

BREEDING ABUNDANCE

The declines in Common Tern SMP abundance since 2000 (Table 39) were greater than those reported by the *Seabirds Count* census (Burnell *et al.* 2023). The 24-year SMP changes showed declines of 60% for the UK, 67% for Scotland and 51% for England. *Seabirds Count* reported smaller declines for the UK (-9%), Scotland (-24%) and England (-3%) since *Seabird 2000*, with the disparity potentially a consequence of the recent HPAI outbreak, which occurred after the census period.

The long-term UK and England trends are closely matched (Figures 59 & 60), as most colonies monitored are within England, although the UK values are typically lower. The UK and England trend indices gradually increased between 2012 and 2022. However, index values declined in 2024 to 63% and 47% below the 1986 baseline for the UK and England, respectively (Table 39). This is likely to have been a result of HPAI, which particularly impacted several Common Tern colonies in England. In contrast to the UK and England, there has been a general decline in the Scottish abundance trend since 1990 (Figure 61), with the index value in 2024 declining further to 73% below the 1986 baseline (Table 39), the lowest value since SMP monitoring began.

Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful abundance trends. ▶

COMMON TERN, BY MOSS TAYLOR / BTO

Table 39: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–24)	24-yr change (2000–24)	2024	Sites
UK	12,219	103	-63*	-60*	0.27	37
England	5,478	60	-47*	-51*	0.38	22
Scotland	4,071	27	-73*	-67*	0.26	7

* significant changes

Table 40: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–02)	Abundance (AON) *Seabirds Count* (2015–21)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	16,028	17,089	7

Figure 59: UK SMP Breeding Abundance (1986–2024)

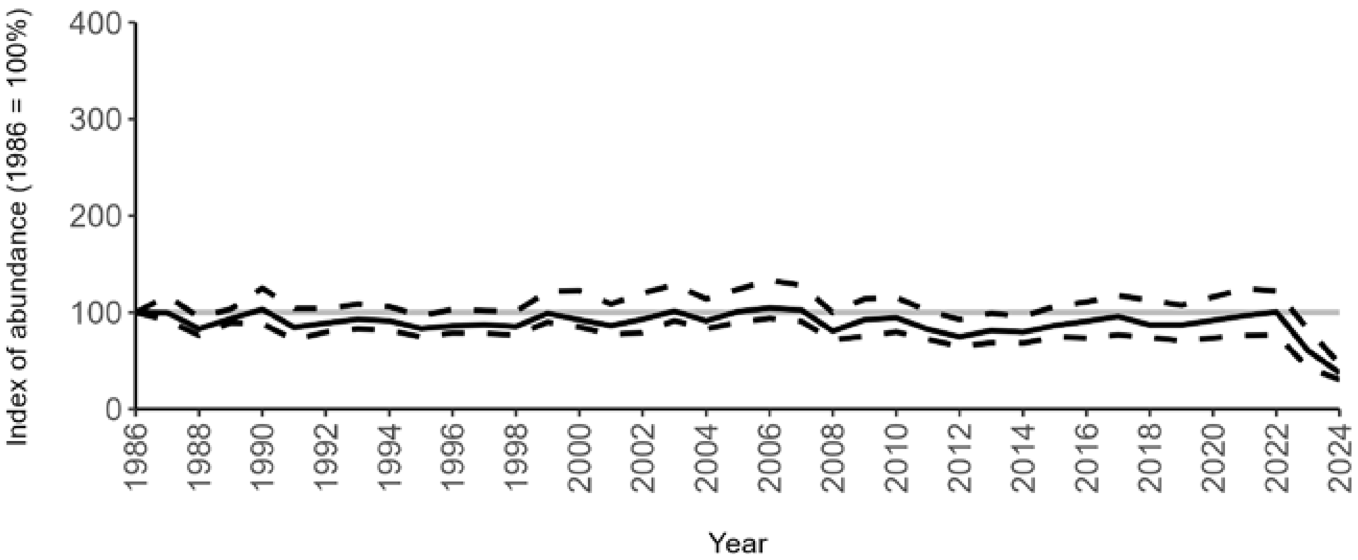
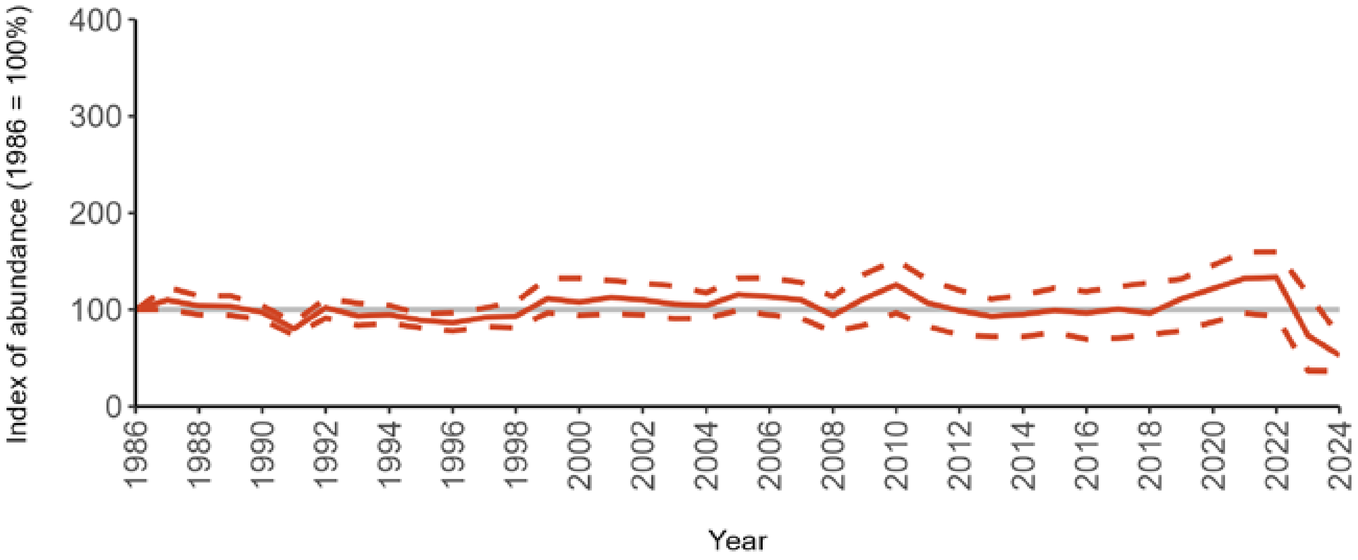


Figure 60: England SMP Breeding Abundance (1986–2024)



COMMON TERN, BY SAM LANGLOIS

Figure 61: Scotland SMP Breeding Abundance (1986–2024)

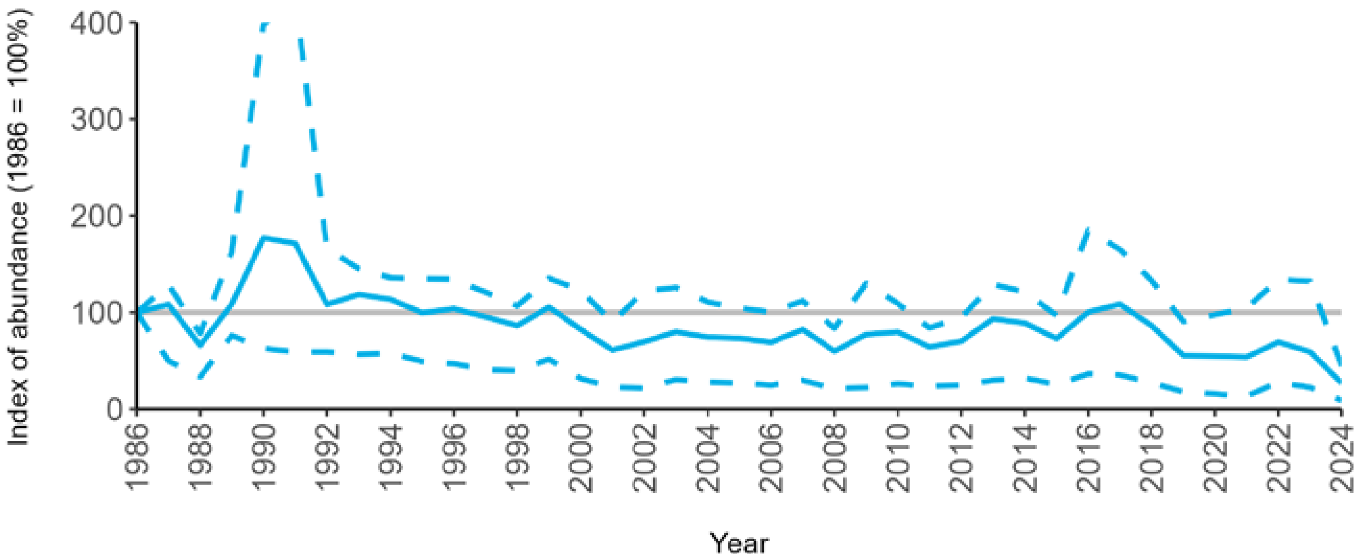
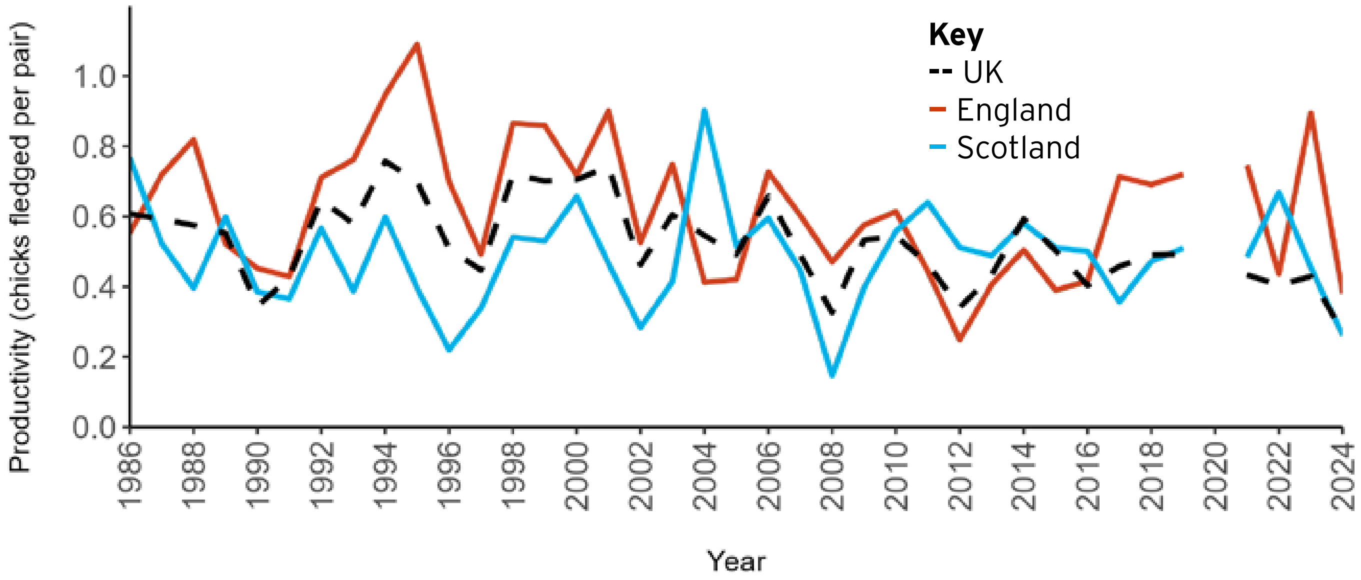


Figure 62: SMP Productivity (1986–2024)



PRODUCTIVITY

Across the SMP recording period, the productivity trends for Common Terns have experienced fluctuations in values between years within all regions (Figure 62). The trend patterns for the UK and England are relatively similar, as a large proportion of monitored sites are in England. However, productivity estimates have been higher in England than in the UK and Scotland over much of the SMP reporting period. In 2024, productivity estimates declined, with similarly low values for the UK and Scotland, with 0.27 and 0.26 chicks fledged per breeding pair, respectively, whilst the estimate for England was 0.38 chicks fledged per pair (Table 39).

Too few data are submitted to the SMP on productivity of Common Terns from other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

The *Seabirds Count* census showed that the British and Irish Common Tern population trend is largely stable (Burnell *et al.* 2023). A large proportion of Britain and Ireland's breeding population occupies just 14 colonies (Burnell *et al.* 2023), and these sites often require intensive management, such as predator control, biosecurity measures, vegetation management, provision of artificial nesting structures or habitats and prevention of human disturbance.

At unwardened sites, human disturbance from recreational activities, such as watercrafts (Burger 1998) and the use of Uncrewed Aerial Vehicles (UAVs), can negatively impact colonies or prevent birds from using otherwise suitable habitat for breeding. An extreme form of human disturbance is the removal of eggs or hunting of adults which led to the Britain and Ireland tern populations almost becoming extirpated at the end of the 1800s (del Hoyo *et al.* 2018).

Predation by American Mink (Craik 1995, 1997), Red Fox and Brown Rat can have significant impacts on local populations, and has caused colony abandonments in the past (Burnell *et al.* 2023). Large gull species and Peregrine Falcons (*Falco peregrinus*), in addition to directly predating Common Terns, may also cause displacement of colonies to less favourable sites through competition or disturbance (Booth & Morrison 2010, Burnell *et al.* 2023, Cuthbert *et al.* 2003).

Climate change is likely to impact Common Tern populations in a variety of ways. Increasing sea surface temperatures will affect prey availability, an increased frequency of storms may reduce foraging ability, and sea level rise, extreme weather events and a longer vegetation growing season may all cause a loss of suitable nesting habitat (Burnell *et al.* 2023, Uttley *et al.* 1989).

HPAI has also had an effect on Common Tern populations over the last few years. Monitoring of 40% of the UK population following the recent outbreak showed a 42% decrease in numbers between 2015–21 and 2023 (Tremlett *et al.* 2024b).

As a migratory species, threats along migration routes and on the wintering grounds may also impact Common Tern breeding populations from Britain and Ireland. Historically, hunting on wintering grounds posed a threat, although the extent to which this persists is unclear. Commercial fishery exploitation of the waters of West Africa could also affect food availability in the non-breeding season (EJF 2020, Grémillet *et al.* 2015).

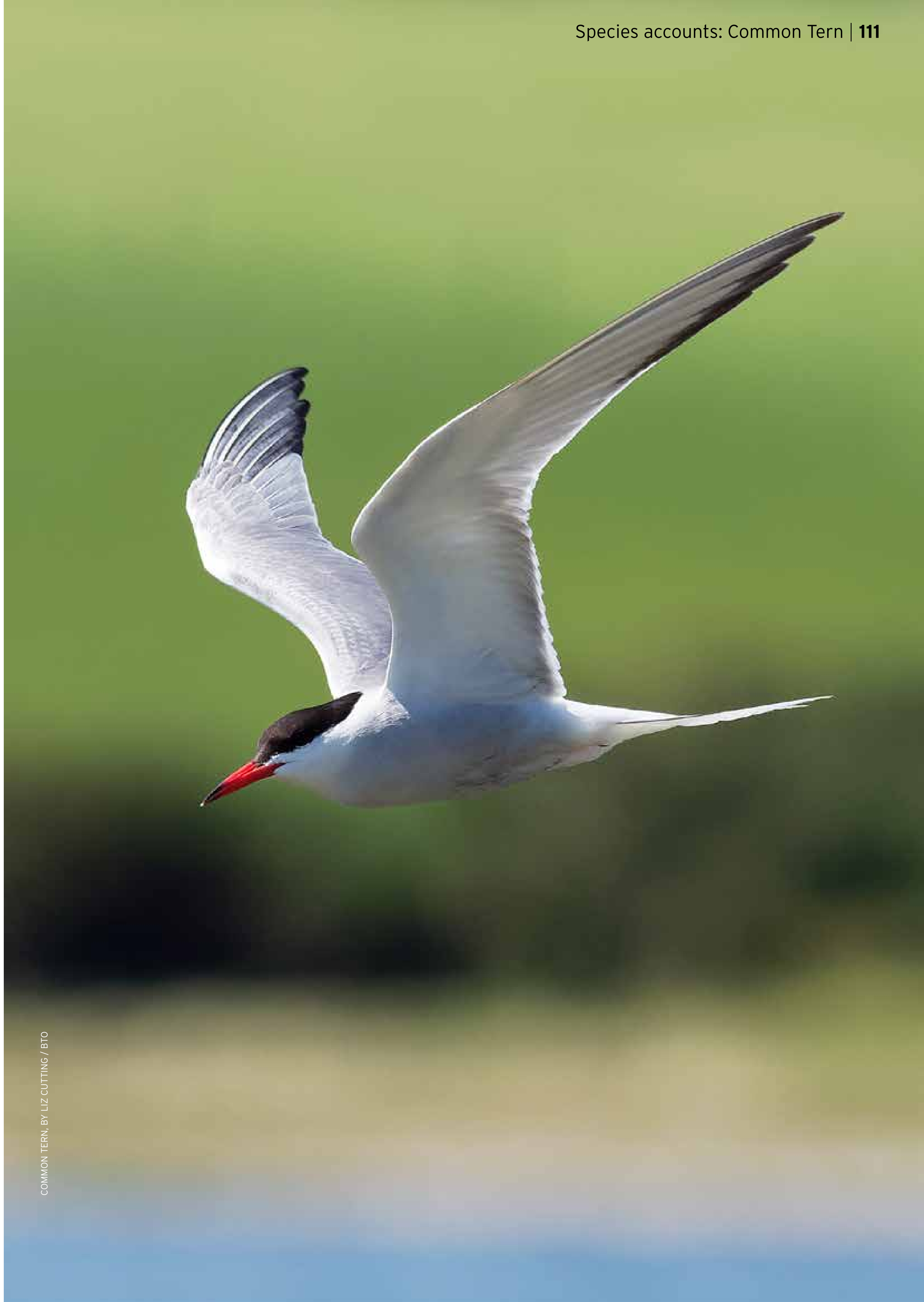
CONSERVATION

The population of Common Terns in Britain and Ireland increased through the 20th century following the introduction of protective legislation to prevent egg collecting and hunting of adult birds (JNCC 2021).

Common Terns are highly adaptable and respond well to a range of conservation management techniques. These include the provision of nest boxes (Bried *et al.* 2009, Burgess & Morris 1992), artificial rafts (Dunlop *et al.* 1991) and islands, habitat management to provide suitable vegetation cover (Cook-Haley & Millenbah 2002), and reduction of disturbance through restrictions on human access (Fasola & Canova 1996).

Predator control also plays an important role in tern conservation. Trapping of American Mink has been shown to significantly improve Common Tern productivity (Ratcliffe *et al.* 2008a), and control of Brown Rat (Amaral *et al.* 2010) and gulls (Blokpoel *et al.* 1997, Donehower *et al.* 2007, Guillemette & Brousseau 2001) have also been shown to be beneficial. Erection of predator fences and effective biosecurity measures are additional measures that are locally effective in protecting Common Tern colonies.

As also described for Roseate Tern, examples of successful conservation management for Common Terns in Ireland include ongoing conservation management at Lady's Island Lake and Rockabill (Gill *et al.* 2019). Internationally, they include an effective conservation scheme in Po Delta, Italy, to protect gull and tern breeding colonies (Fasola & Canova 1996) and restoration of a small islet in the Azores, Portugal (Praia Islet), which included European Rabbit eradication, native plant reintroduction and nest box deployment (Bried *et al.* 2009). ■



COMMON TERN, BY LIZ CUTTING / BTO

Arctic Tern

Sterna paradisaea

c.3%

Abundance: Decline
Productivity: 0.11

Red-listed
Amber-listed (I)

Colony Count sites: 69
Breeding Success sites: 22

Least Concern

Lifespan: 13 years
Breeding age: 4 years

Arctic Terns are the most common breeding tern in Britain and Ireland, and approximately 3% of the global breeding population is hosted here in summer (Burnell *et al.* 2023). They have the longest known annual return migration on earth, of up to 50,000 km (Alerstam *et al.* 2019).

DISTRIBUTION

In Britain and Ireland, breeding Arctic Terns are found predominantly around the northern and western coastlines, with a few colonies further south (Burnell *et al.* 2023).

In a global context, they breed widely across the Arctic and subarctic regions of Europe, Asia and North America (BirdLife International 2024).

Arctic Terns migrate south for the non-breeding season, taking advantage of the long summer daylight hours in the southern hemisphere (del Hoyo *et al.* 1996, Melville & Shortridge 2006, Wernham *et al.* 2002). At this time they can be found throughout the Southern Ocean, from the southern waters off South America and Africa to the edge of the Antarctic ice sheet (BTO 2025).

DIET

Arctic Terns are surface feeders, plunge-diving to depths of 50 cm (Cramp 1985). They preferentially feed on sandeels or other small energy-rich fish species, which are particularly important for chicks (Ewins 1985, Furness 1982, Furness & Tasker 2000, Monaghan *et al.* 1989, Suddaby & Ratcliffe 1997). Additional prey include planktonic species of crustaceans, molluscs, insects and occasionally berries on initial arrival at their breeding grounds. During the non-breeding season, Arctic Terns are known to forage over the open ocean, often near Antarctic Minke Whales (*Balaenoptera bonaerensis*) (Higgins & Davies 1996).

BREEDING

Around Britain and Ireland, Arctic Tern colonies are predominantly coastal, nesting on shingle, beaches, spits or islands with short or sparse vegetation. The nest is a simple scrape in the substrate, in which they lay 1–2 eggs (Mitchell *et al.* 2004).

Elsewhere in the world, Arctic Terns also breed on islets along rivers, inland near water, on grassland, tundra and forest-tundra (del Hoyo *et al.* 1996, IUCN 2025). In these habitats, they nest as solitary pairs or in small colonies (del Hoyo *et al.* 1996).

BREEDING ABUNDANCE

The SMP 24-year change for the UK shows a decline of 29% (Table 41) which was similar to the decrease of 37% recorded by the *Seabirds Count* census since *Seabirds 2000* (Burnell *et al.* 2023). For Scotland, the 24-year SMP change showed a decline of 35% since 2000 (Table 41), compared to the decline of 54% reported by *Seabirds Count*. However, for England, the SMP change indicated a decline of 38% since 2000 (Table 41), whilst the *Seabirds Count* census showed a large increase of 69% over a similar period. This large discrepancy suggests that the colonies sampled in England for the SMP may be unrepresentative of those within the country as a whole.

The long-term SMP abundance trends for Arctic Tern differ considerably between regions (Figures 63–65). Between 1990 and 2012, the UK trend generally fluctuated around the 1986 baseline. After a peak in the index in 2014, the UK index has since declined overall to 25% below the baseline in 2024 (Table 41). Following a peak in 1988, the abundance trend ▶

ARCTIC TERN, BY EDMUND FELLOWES / BTO

Table 41: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2023	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	30,451	52	-25	-29	0.11	19
England	6,118	9	-50	-38	0.41	6
Scotland	19,555	34	-35	-35	0.10	9

No significant changes

Table 42: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	51,293	33,215	-35

Figure 63: UK SMP Breeding Abundance (1986–2024)

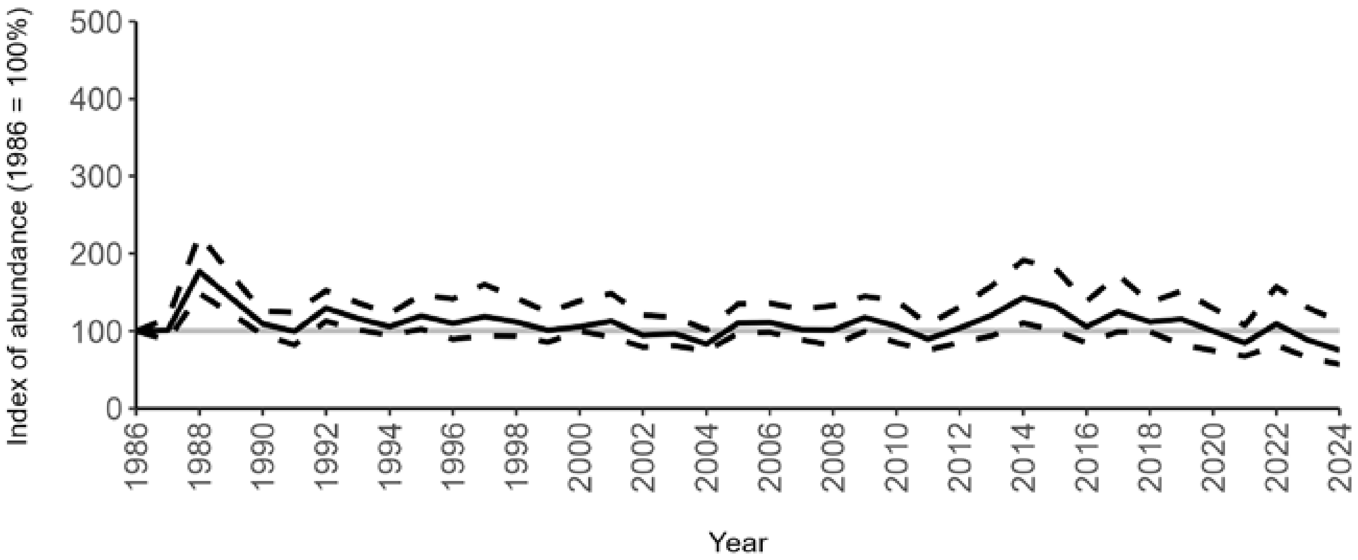


Figure 64: England SMP Breeding Abundance (1986–2024)

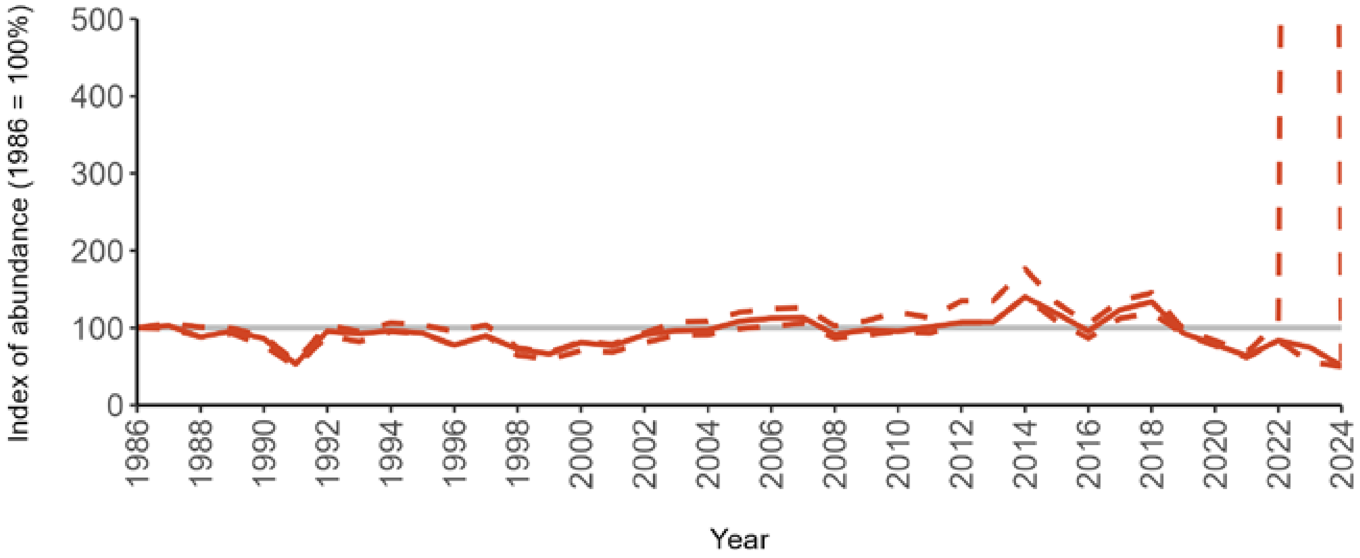


Figure 65: Scotland SMP Breeding Abundance (1986–2024)

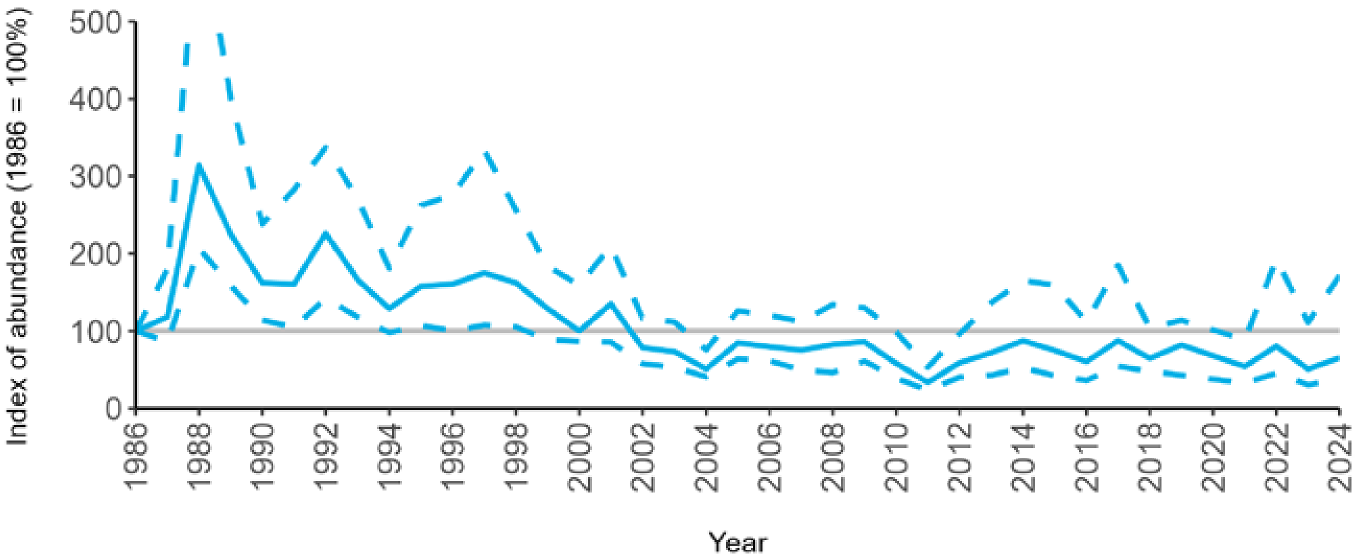
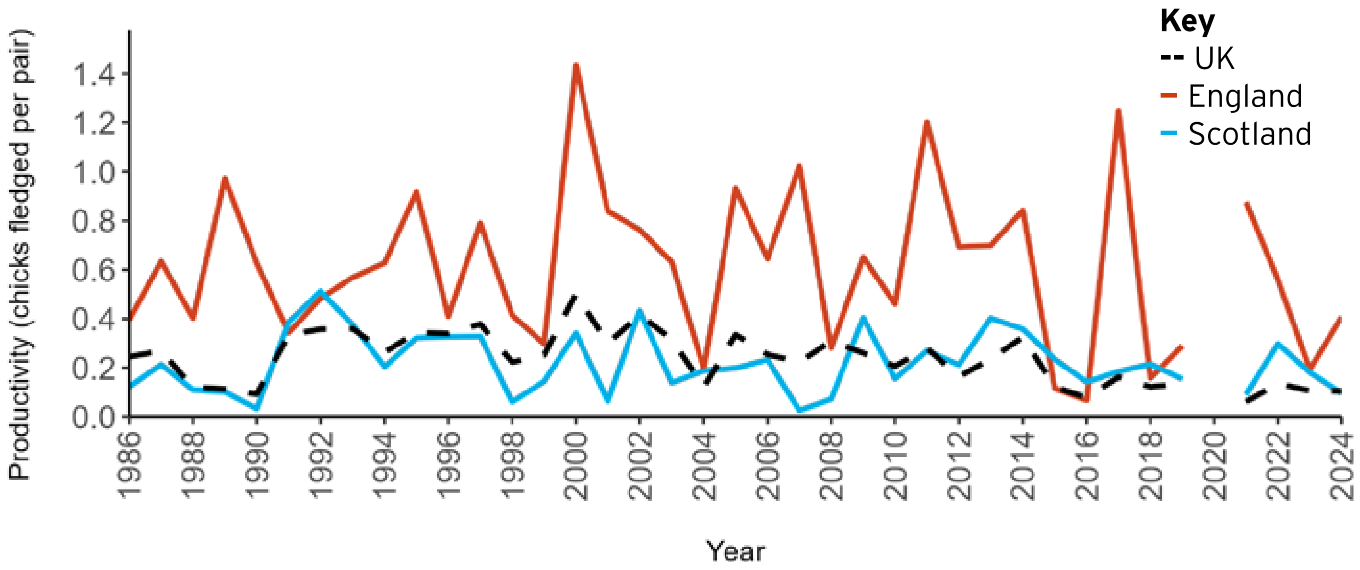


Figure 66: SMP Productivity (1986–2024)



ARCTIC TERN, BY EDMUND FELLOWES / BTO

for Scotland declined overall over the first half of the SMP recording period, and has remained relatively stable, but below the baseline, since the early 2000s, with an index value of 35% below the 1986 baseline in 2024 (Table 41). In comparison, there was a gradual overall increase in the England trend in the early 2000s up to 2014, with a general decline since. The index value for England in 2024 declined to 50% below the 1986 baseline (Table 41). It should be noted that the upper confidence interval for the England index value in 2023 is very large, indicating a high degree of uncertainty, as data was missing from key colonies, and this value should therefore be treated with caution.

Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

The productivity trends for Arctic Tern have been relatively similar for the UK and Scotland over the SMP monitoring period (Figure 66). By contrast, the productivity values for England have generally been higher and show considerable fluctuation in values between years. In 2024, the productivity estimates were similarly low for the UK and Scotland with 0.11 and 0.10 chicks fledged per breeding pair, respectively (Table 41). The trend for England increased slightly in 2024, to 0.41 chicks fledged per pair (Table 41).

Too few data are submitted to the SMP on productivity of Arctic Terns from other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Arctic Terns may be more dependent on sandeels as prey items than the other tern species breeding in Britain and Ireland, and show less flexibility in diet – when their preferred food source declines, they do not readily switch to alternative prey (del Hoyo *et al.* 1996). In the main Arctic Tern breeding areas in Britain, there are also few alternative energy-rich fish available, so any change in sandeel abundance, such as those currently caused by climate change, can have serious effects. Indeed, declines in sandeel availability have been shown to lead to Arctic Tern breeding failures (Monaghan 1992, Schreiber & Kissling 2005, Vigfusdottir *et al.* 2013). After an increase of the sandeel stock around Shetland during the 1970s and early 1980s, a collapse between 1984 and 1990 resulted in declines in the local Arctic Tern population (Bailey 1991, JNCC 2021).

Climate change is also likely to impact Arctic Tern populations through increased storminess, which may hinder foraging ability, and extreme weather events, which can cause nests to flood (Wright & Wilde 2015, Rendell-Read 2016, Short & Watts 2016). The British and Irish population of Arctic Tern is at the southern edge of its range, and predictions of range constrictions for the UK seabird population in response to climate change suggest that in future its population may become restricted to only the most northerly coasts and islands of Scotland (Daunt & Mitchell 2013).

Additional pressures on Arctic Terns include human disturbance through recreational activities, and predation. Loss of Arctic Tern eggs, chicks and adults through predation by American Mink, are thought to have contributed to local population declines, e.g. in western Scotland (Craik 1997). Additional predators include rats, gulls, corvids and European Hedgehogs (Burnell *et al.* 2023, Booth Jones 2020).

CONSERVATION

Legislation to protect Arctic Terns from hunting and egg collecting in the early part of the 20th century led to historic increases in the UK population (JNCC 2021).

Successful Arctic Tern colony site management techniques have included the eradication or control of American Mink (Craik 1997, 2015, Nordström *et al.* 2004, Mavor *et al.* 2006), water level control, prevention of public access, and control and/or exclusion using electric fencing of predatory mammals (Daly *et al.* 2016).

In common with many seabird species, active management of breeding sites alone will be insufficient to deal with all the pressures faced by Arctic Terns. Pressures resulting from the effects of climate change will be particularly challenging to address, given the wide-ranging nature of the likely impacts. These have the potential to affect both their breeding success in summer and their foraging ability in winter months, through a reduction in the pack ice habitat in Antarctica on which they depend for feeding opportunities (Burnell *et al.* 2023). Monitoring and research on Arctic Tern populations will prove vital to assessing the impact of these changes and informing conservation actions.

HPAI caused Arctic Tern mortalities during 2021 and 2022 but these were relatively minor, with a 2% decline in sites surveyed between 2015–21 and 2023 (Buckingham *et al.* 2022, EFSA *et al.* 2022, Falchieri *et al.* 2022, Tremlett *et al.* 2024b). However, continued tracking of the impacts of HPAI across all UK seabird species is advisable. ■

ARCTIC TERN, BY EDMUND FELLOWES / BTO

Little Tern

Sternula albifrons

c.2–3%
ssp. albifrons

Abundance: Decline
Productivity: 0.25

Amber-listed
Amber-listed (I)

Colony Count sites: 47
Breeding Success sites: 23

Least Concern

Lifespan: 12 years
Breeding age: 3 years

Britain and Ireland host around 1–2% of the world's breeding Little Terns, but approximately 2–3% of the subspecies *albifrons* (Burnell *et al.* 2023). Every summer, wardens are employed to help manage and protect key Little Tern breeding areas from predators and human disturbance (BTO 2025).

DISTRIBUTION

Little Terns breed in scattered locations across much of Britain and Ireland, with the largest colonies located in East Anglia, North Wales and south-east Ireland (Burnell *et al.* 2023).

Globally, they are found across much of Europe and Africa, western, central and east coast of Asia, and in parts of Australasia (BirdLife International 2024). They are migratory, with northern birds moving further south in the winter (Tavecchia *et al.* 2006).

DIET

Little Terns feed by plunge-diving or dipping. Their prey varies according to locality, but is primarily small fish, crustaceans and invertebrates (del Hoyo *et al.* 1996, Paiva *et al.* 2008).

BREEDING

In the UK, Little Terns exclusively breed along the coast, on beaches, spits or inshore islets. They have a small foraging range in the breeding season. Active breeders usually feed less than 6 km from the colony, which limits colony location to being within a short distance from suitable feeding grounds, whereas failed breeders have been shown to occupy home ranges of 52 km^2 in Norfolk, England (Perrow *et al.* 2006). Their nests are shallow, well-camouflaged scrapes on the ground, where a clutch of 2–3 eggs are laid (Burnell *et al.* 2023, JNCC 2021).

In other parts of their global breeding range they also nest inland, around marshland habitat, where their nests consist of broken up shells and vegetation (del Hoyo *et al.* 1996).

BREEDING ABUNDANCE

The Little Tern abundance changes for the UK, Scotland and England, reported by the *Seabirds Count* census, indicated declining populations since *Seabird 2000* of between 25 and 32% (Burnell *et al.* 2023). In contrast, the 24-year SMP abundance change for Scotland showed a greater decline of 59%, whilst the index values for the UK and England in 2024, compared to 2000, were more stable at -2% and 4%, respectively (Table 43). It should be noted that the proportion of adult Little Terns choosing to nest each year can fluctuate, and as a consequence it is thought that annual counts, such as those conducted through the SMP, may provide more accurate trends than widely spaced censuses (JNCC 2021).

There has been a continuous overall decline in the long-term abundance trends for Little Tern over most of the SMP monitoring period for the UK, England and particularly Scotland, and values have largely remained below the 1986 baseline (Figures 67–69). The trends have started to improve for the UK and England (where a large proportion of the Little Tern colonies are monitored) since 2019. In 2024, the index values for the UK and England were 19% and 15% below the 1986 baseline, respectively, whilst the index value for Scotland was lower at 75% below the baseline (Table 43).

Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

The UK and England productivity trends for Little Tern are closely matched, as a large proportion of UK monitored sites are in England, whilst the Scottish values are often quite different. However, all have fluctuated widely and there are no apparent trends over the years (Figure 70). Productivity for all regions has been below the figure of 0.70 chicks fledged/pair thought to be needed to maintain population stability (Cook & Robinson 2010) for much of the SMP monitoring period. In 2024, the productivity estimates for the UK and England were similarly low at 0.25 and 0.26 chicks fledged per pair, respectively, whilst in Scotland 0.41 chicks fledged per pair (Table 43).

Too few data are submitted to the SMP on productivity of Little Tern from other regions to allow for calculation of productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

One of the biggest pressures on Little Terns is human disturbance, as their preferred nesting beaches are often busy with recreational users (JNCC 2021, Ratcliffe *et al.* 2008b). However, many Little Tern colonies are wardened to reduce disturbance levels.

Loss of nesting habitat due to increases in extreme weather events alongside erosion, sea level rise and extreme high tides is also becoming an increasing pressure on nesting birds (Rendell-Read 2018, Macleod-Nolan 2020), and further habitat loss through encroachment of vegetation is a problem at some colonies, e.g. Beacon Ponds, East Yorkshire (England) (Hunton 2024).

Little Terns are also subject to predation from a wide range of both mammalian and avian predators, such as Red Foxes, European Badgers, European Hedgehogs, Stoats, Kestrels (*Falco tinnunculus*) and corvids (Rendell-Read 2018, Macleod-Nolan 2020, JNCC 2021).

CONSERVATION

Successful management techniques for Little Tern colonies include wardening, signage and barriers to reduce disturbance, predator fences, diversionary feeding of predators, and habitat management to reduce erosion and vegetation encroachment (BTO 2025, Burnell *et al.* 2023, Fasola & Canova 1996, JNCC 2021, Mederios *et al.* 2007, Ratcliffe *et al.* 2000, Smart & Amar 2018, Wilson *et al.* 2020). Habitat creation, through managed realignment of coastal defences and the creation of artificial islands, is also likely to have beneficial impacts on the Little Tern population of Britain and Ireland. ▶

LITTLE TERN, BY PHILIP CROFT / BTO

Table 43: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (AON)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	1,403	37	-19	-2	0.25	20
England	1,004	20	-15	4	0.26	16
Scotland	227	16	-75*	-59	0.41	4

** significant changes*

Table 44: *Seabirds Count* census results

	Abundance (AON) *Seabird 2000* (1998–2002)	Abundance (AON) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	2,059	1,750	-15

Figure 67: UK SMP Breeding Abundance (1986–2024)

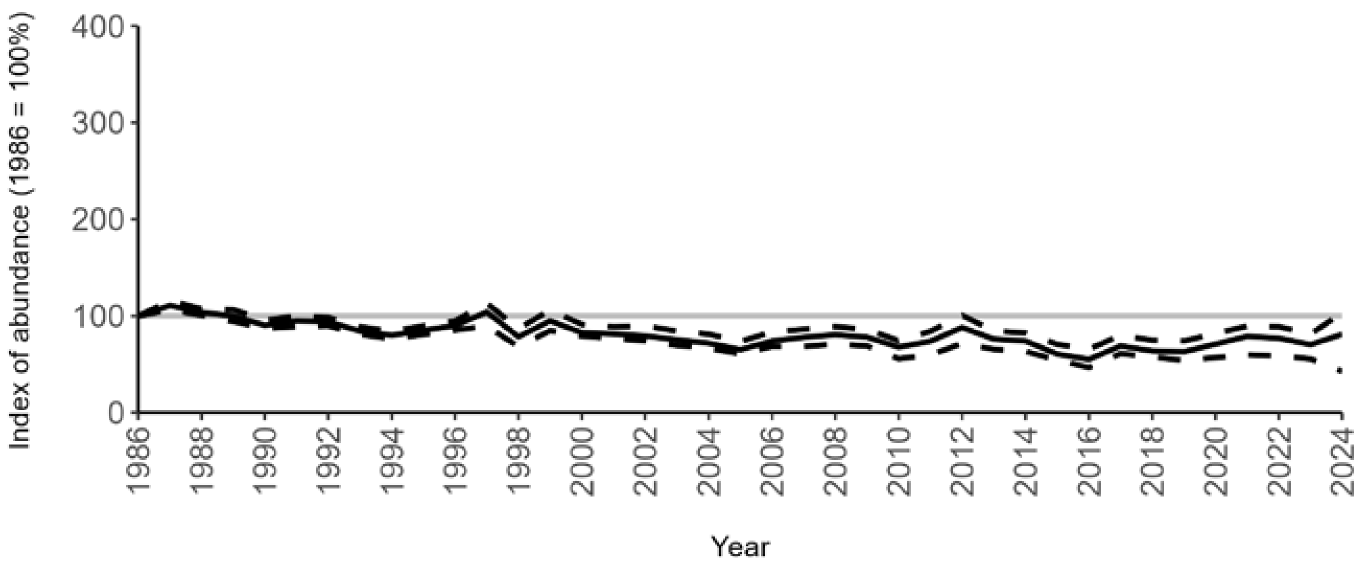


Figure 68: England SMP Breeding Abundance (1986–2024)

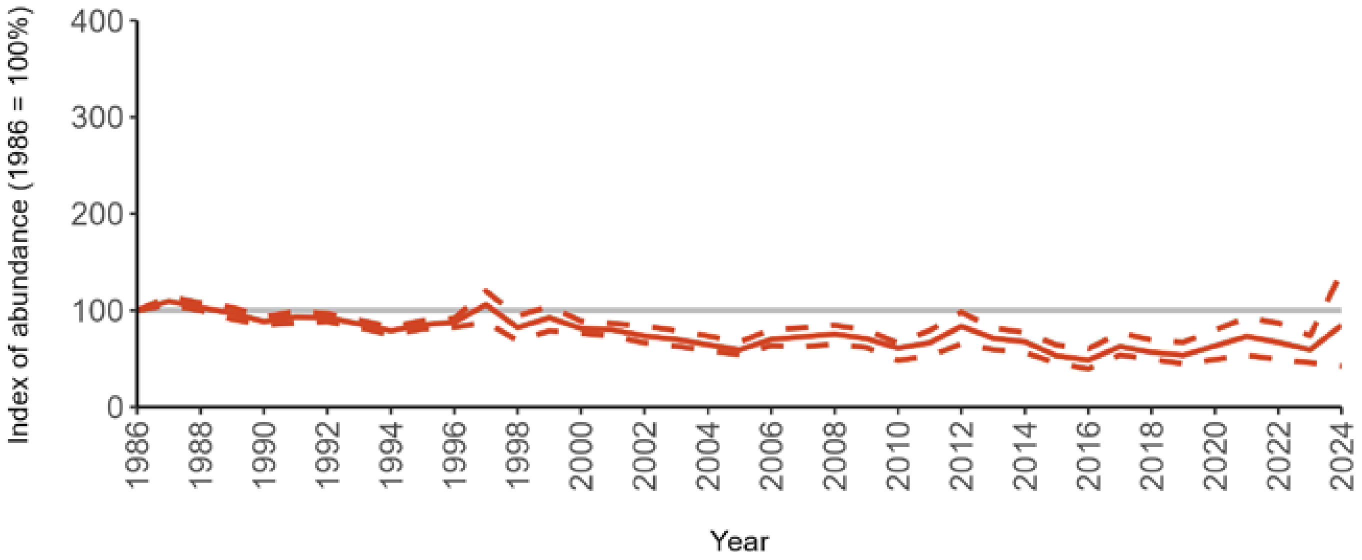


Figure 69: Scotland SMP Breeding Abundance (1986–2024)

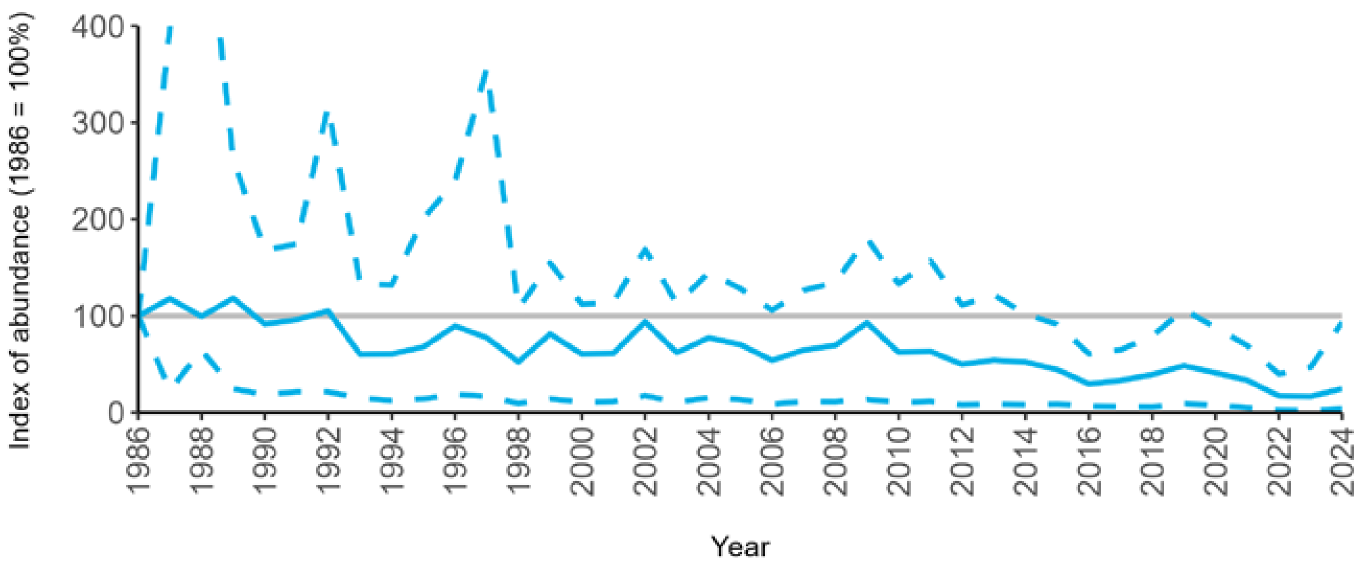
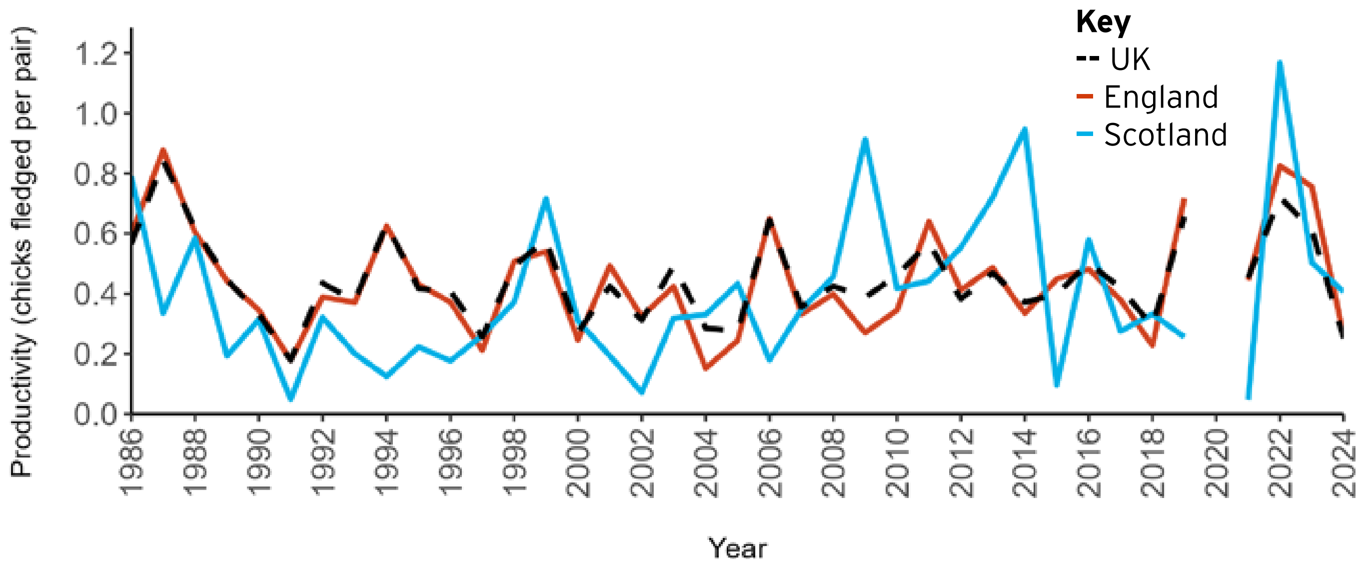


Figure 70: SMP Productivity (1986–2024)



LITTLE TERNS, BY TOM WRIGHT

Guillemot

Uria aalge

c.8%

Abundance: Increase
Productivity: 0.51

Amber-listed
Amber-listed (I)

Colony Count sites: 128
Breeding Success sites: 11

Least Concern

Lifespan: 23 years
Breeding age: 5 years

Britain and Ireland host approximately 8% of the global breeding population of Guillemots (Burnell *et al.* 2023). Two of the five subspecies breed in Britain and Ireland: *Uria aalge aalge*, which is darker and larger and is found across much of Europe as far south as northern England, and *Uria aalge albionis*, which has a browner mantle, is smaller, and is found in the rest of England, Wales and Ireland, as well as Helgoland, parts of France and Iberia. There is also a 'bridled' morph, with a white eye ring and spectacle, which increases in frequency in northern latitudes of the Atlantic Ocean (JNCC 2021).

DISTRIBUTION

Guillemot breeding colonies can be found around the British and Irish coastlines wherever there are suitable cliffs (Burnell *et al.* 2023). Globally they have a circumpolar distribution across the northern Atlantic and Pacific Oceans, and low-arctic areas of the Arctic Ocean (BirdLife International 2024). In winter, British and Irish birds can be seen across all coastal waters, with larger numbers in the north and west (BTO 2025).

DIET

Guillemots predominantly feed on small fish such as sandeels, clupeids and gadids, although crustaceans and molluscs may also be taken (Anderson *et al.* 2014, Lorentsen & Anker-Nilssen 1999, Sonntag & Hüppop 2005). In winter, they venture further offshore to deeper waters, diving up to 180 m in search of prey (Piatt & Nettleship 1985).

BREEDING

Guillemots nest preferentially on ledges on steep sea cliffs, but can also be found in boulder scree. They lay a single egg directly onto the bare rock. They are generally found in large colonies, and breeding success is highest when breeding in higher densities or at sites well protected from predators (JNCC 2021). Breeding densities can reach approximately 20 pairs/m^2 (JNCC 2021).

Juvenile Guillemots leave the nest before their wings have fully grown. When ready to leave the breeding ledge, they jump off the cliff to the beach or waves below and swim out to sea, guided by their male parent (Hjernquist *et al.* 2012).

BREEDING ABUNDANCE

The Guillemot abundance change for the UK, reported by the *Seabirds Count* census, indicated a stable population, with a small decrease of 11% since *Seabird 2000*, whilst, for Scotland, the *Seabirds Count* census showed a decline of 31% (Burnell *et al.* 2023). In contrast, the 24-year SMP change for the UK showed an increase of 18%, whilst the 24-year SMP change for Scotland showed a decline of 38% over a similar period to that of the censuses (Table 45).

The long-term SMP abundance trends differ for the UK and Scotland over the SMP recording period (Figures 71 & 72). The trends were similar until 2001, following which the UK trend gradually increased, whilst the Scotland trend declined. In more recent years the Scotland trend had showed a gradual increase up until 2022, after which the index values dropped. In 2019 and 2022, a large number of small colonies and few large colonies (>10,000 individuals) were counted and included in the abundance index analysis. This caused uncertainty in these estimates, as the large number of smaller colonies included in the sample are likely to have had a disproportionate influence on the index (JNCC 2021). Consequently, these values should be treated with caution. In 2023, a greater number of colonies – including large colonies – were counted than in 2019 and 2022, due to enhanced monitoring for HPAI impacts, and so these values are likely to be more reliable. The 2024 long-term index values were 40% above the 1986 baseline for the UK and 29% below the baseline for Scotland (Table 45).

The *Seabirds Count* census also showed increases in the Guillemot populations for England, Wales, Northern Ireland, the Republic of Ireland and the Isle of Man, with a decline in the Channel Islands (Burnell *et al.* 2023). Unfortunately, current data submitted to the SMP for these regions are too sparse to produce valid SMP abundance trends for comparison.

PRODUCTIVITY

The productivity trends for the UK and Scotland are closely matched (Figure 73), as many of the monitored colonies are in Scotland. In both regions, productivity was relatively stable between 1986 and 2002, before a steep decline until 2007, following which values maintained a higher level. Productivity values in England have shown a less consistent pattern, and there are some gaps in years where no data were submitted. In 2024, productivity estimates for the UK and Scotland remained similar with 0.51 and 0.55 chicks fledged per breeding pair, respectively, whilst the estimate for England was 0.60 chicks fledged per pair (Table 45). Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful productivity trends.

PHENOLOGY, DIET AND SURVIVAL RATES

Data on Guillemot breeding phenology are collected at the Key Site of Skomer Island (Wales), and also at Sumburgh Head in Shetland (Scotland) by the Shetland Oil Terminal Environmental Advisory Group (SOTEAG). Diet information is collected at the Key Sites of Canna, Fair Isle and the Isle of May (all in Scotland), and survival information is collected on Canna and the Isle of May.

CAUSES OF CHANGE

Guillemots face a wide range of current and potential threats in Britain and Ireland. They are particularly susceptible to severe winter storms, which reduce foraging opportunities and can in extreme cases lead to starvation. Climate change is increasing the frequency of these extreme weather events (Field *et al.* 2012), which can lead to many dead birds being washed up on beaches in an event known as a seabird wreck. In the winter of 2013/14, large wrecks occurred along British coasts, and post-mortems suggested starvation and some oil contamination as likely causes (Jessop 2014, Sellers 2014).

Increases in the sea surface temperature are causing changes in the abundance and distribution of Guillemot prey, such as sandeels (Erikstad *et al.* 2013, Heath *et al.* 2009, Régnier *et al.* 2017, Riordan & Birkhead 2018, Wanless *et al.* 2005), potentially resulting in a switch to less energy-rich alternative prey items (Heubeck 2009). This is something Guillemots appear to be able to do more readily than some other seabird species, for example, switching to the nutritionally-poor Snake Pipefish (*Entelurus aequoreus*) in the mid-2000s (Anderson *et al.* 2014). However, they are not able to fully compensate for this change in calorific value, resulting in lower breeding success in these years.

Furthermore, Guillemots are directly susceptible to heat stress, and lower ▶

Table 45: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (IND)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	1,265,888	105	40	18	0.51	8
England	-	-	-	-	0.60	2
Scotland	810,645	70	-29	-38	0.55	5

No significant changes

Table 46: *Seabirds Count* census results

	Abundance (IND) *Seabird 2000* (1998–2002)	Abundance (IND) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	1,571,189	1,449,589	-8

GUILLEMOT COLONY, BY TOM CADWALLENDER / BTO

breeding success has been reported at Baltic colonies where adults are forced to leave their eggs and chicks to cool themselves down (Olin *et al.* 2023).

Guillemot adults, eggs and chicks are vulnerable to predation during the breeding season by a range of species, including Hooded Crows (*Corvus cornix*), Carrion Crows (*Corvus corone*) and Herring Gulls (Booth Jones 2020). This threat increases in smaller auk colonies, which offer less protection for individual birds (Gilchrist 1999).

The commercial fishing industry can also negatively impact Guillemot populations through both bycatch, particularly in gillnets (Northridge *et al.* 2023), and overfishing of important prey species, which is likely to increase foraging pressure and decrease breeding productivity (Nettleship *et al.* 2018a). Offshore wind farms are an additional pressure on Guillemots, through potential displacement of birds from foraging grounds (Peschko *et al.* 2020). In common with other auks, Guillemots are especially vulnerable to oil spills, as a large proportion of their time is spent on the sea surface (Williams *et al.* 1995).

CONSERVATION

As for many species of seabirds, mitigating against climate change is likely to be the most effective conservation action that can be taken to improve Guillemot population numbers. Additional measures that are likely to prove beneficial would be continued research on the impacts of offshore renewable developments, which could then be used to better inform Environmental Impact Assessments, trials of mitigation methods designed to reduce bycatch in fishing activities, and implementation of measures to reduce the frequency or impact of oil pollution incidents.

Although HPAI appears not to have had a major impact on populations so far, continued monitoring will be beneficial to determine whether this remains the case (Tremlett *et al.* 2024b).

Changes to adult survival rates have the potential to alter productivity, which for Guillemot is known to increase with age. Increased monitoring studies of adult survival is likely to be important in understanding both abundance change and productivity (Crespin *et al.* 2006). ■

Figure 71: UK SMP Breeding Abundance (1986–2024)

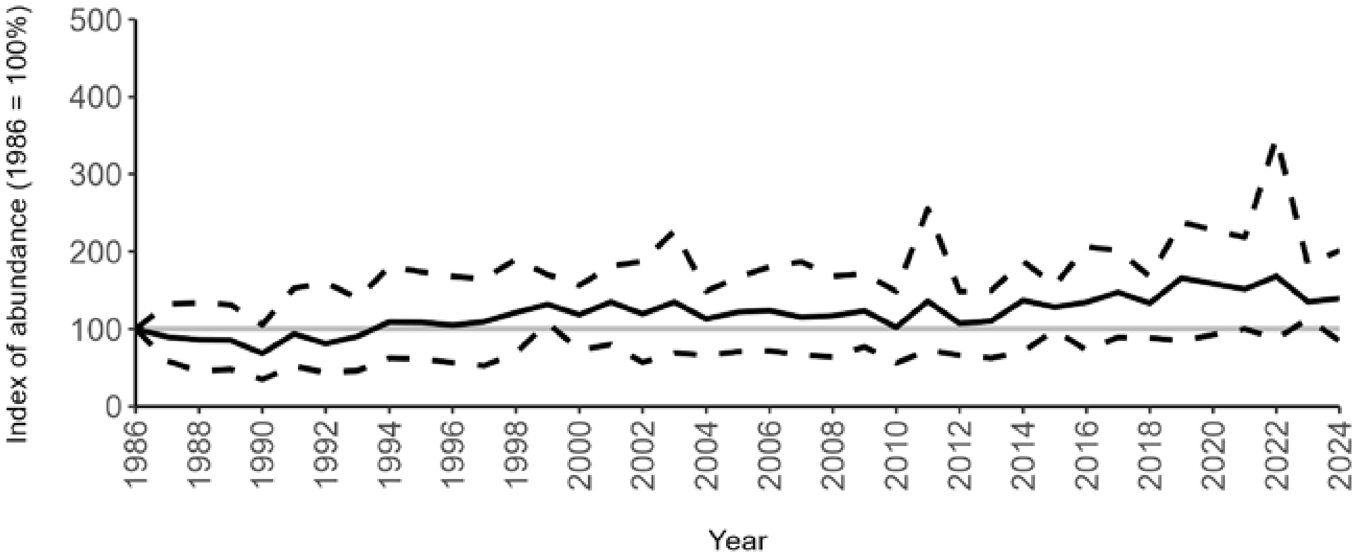


Figure 72: Scotland SMP Breeding Abundance (1986–2024)

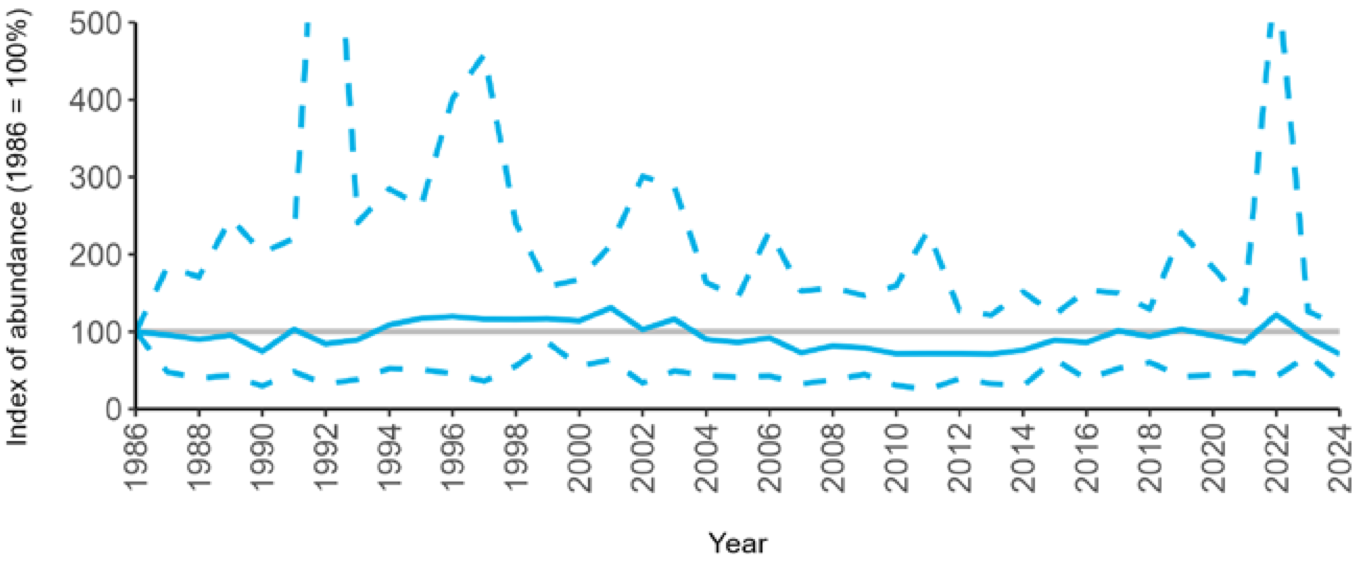
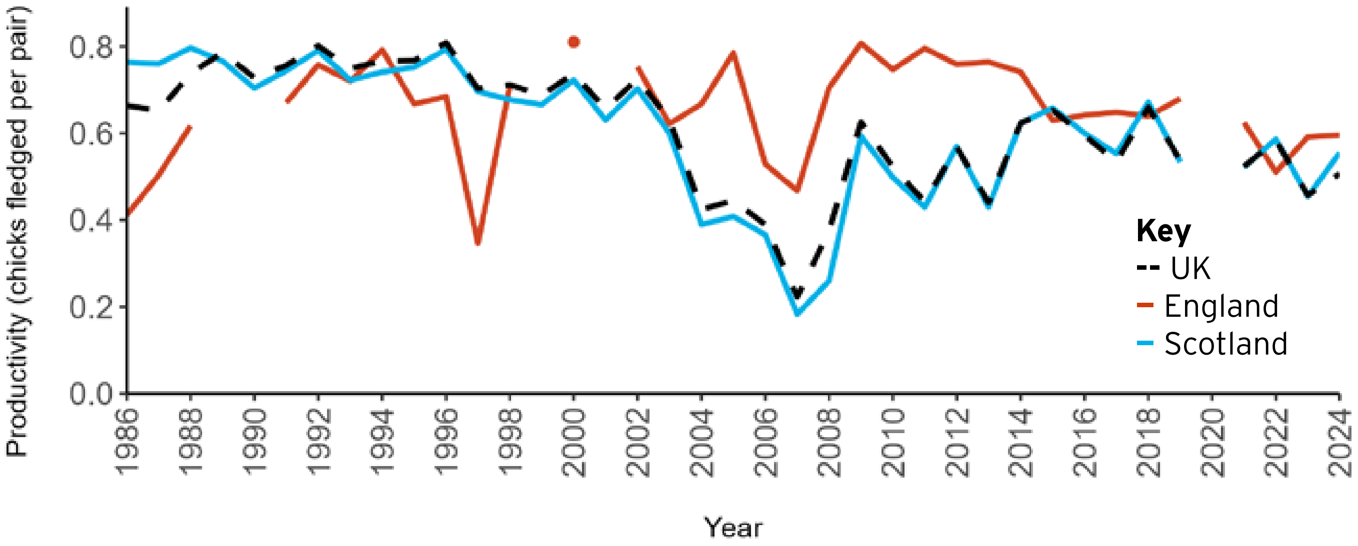


Figure 73: SMP Productivity (1986–2024)



Razorbill

Alca torda

c.29%
ssp. islandica

Abundance: Increase
Productivity: 0.44

Amber-listed
Red-listed (I)

Colony Count sites: 146
Breeding Success sites: 12

Least Concern

Lifespan: 13 years
Breeding age: 4 years

Britain and Ireland host 16–31% of the world's breeding Razorbills and approximately 29% of the subspecies *islandica* (Burnell *et al.* 2023). The oldest known Razorbill, from ringing records, was recorded in 2004 at 41 years, 11 months and 23 days (BTO 2025).

DISTRIBUTION

Razorbills are widespread around the coastlines of Britain and Ireland in the breeding season, with the exception of the south-east of England (Burnell *et al.* 2023). Globally, they breed on north Atlantic coastlines from eastern North America to north-west Russia (BirdLife International 2024).

Razorbills overwinter coastally along both sides of the Atlantic (Lavers *et al.* 2020). British and Irish birds continue to be seen around our coastlines during winter, although many move south to the Atlantic coasts of Europe and North Africa, and some birds move into the western Mediterranean (Wernham *et al.* 2002). Immature birds generally travel greater distances from their colonies than adults in winter, often moving further south or occasionally west to Greenland and the Azores (JNCC 2021).

DIET

Razorbills feed on a variety of prey including krill, clupeids and sandeels (Nettleship 1996, Barrett 2015), catching them by pursuit-diving. Spines within their mouths allow them to hold multiple fish simultaneously, increasing the efficiency of foraging trips and chick provisioning (Burnell *et al.* 2023). Foraging ranges during the breeding season vary between colonies (Isaksson *et al.* 2019).

BREEDING

Razorbills nest on small ledges or in rocky crevices on cliffs, in boulder beaches and in scree (JNCC 2021). They lay a single egg; once the resulting chick fledges, the male parent will feed it for up to two months out at sea (Gaston & Jones 1998).

BREEDING ABUNDANCE

At the UK-level the SMP abundance change for Razorbill since 2000 differed from that reported by the *Seabirds Count* census, with the 24-year SMP change showing an increase of 52% since 2000 (Table 47), whilst the *Seabirds Count* census reported a rise of 18% since *Seabird 2000* (Burnell *et al.* 2023). For Scotland, the 24-year SMP change indicates a small increase of 4% since 2000, similar to the small decline of 2% reported between the censuses.

The 24-year SMP change for Wales showed a large increase of 141%, whilst the *Seabirds Count* census showed an increase of 82% since *Seabird 2000.*

The long-term SMP abundance trends for Razorbill in the UK, Scotland and Wales have all remained largely above the 1986 baseline since the early 1990s (Figures 74–76). Following a period of sustained increase until the early to mid-2000s, the UK and Scotland trends underwent a period of decline, which was most pronounced in Scotland. This was followed by a further period of increasing trends overall for all regions. The highest index values since 1986 were recorded for all three regions in 2022. However, there is a high degree of uncertainty in these values as fewer sites were monitored in 2022, resulting in wide confidence intervals. In all three regions, the abundance values decreased in 2023, when a greater number of colonies, and particularly large colonies, were counted across the UK due to enhanced monitoring for HPAI impacts. In 2024, the UK index value was 110% above the 1986 ▶

RAZORBILLS, BY MIKE TOMS / BTO

Table 47: SMP Breeding Abundance Change and Productivity

	Seabirds Count	Breeding Abundance Change %			Productivity	
	Abundance (IND)	Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)	2024	Sites
UK	225,015	123	110*	52	0.44	9
Scotland	138,828	78	63	4	0.51	3
Wales	23,640	30	256*	141*	-	-

* significant changes

Table 48: *Seabirds Count* census results

	Abundance (IND) *Seabird 2000* (1998–2002)	Abundance (IND) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	219,693	258,629	18

RAZORBILL, BY JOHN HARDING / BTO

Figure 74: UK SMP Breeding Abundance (1986–2024)

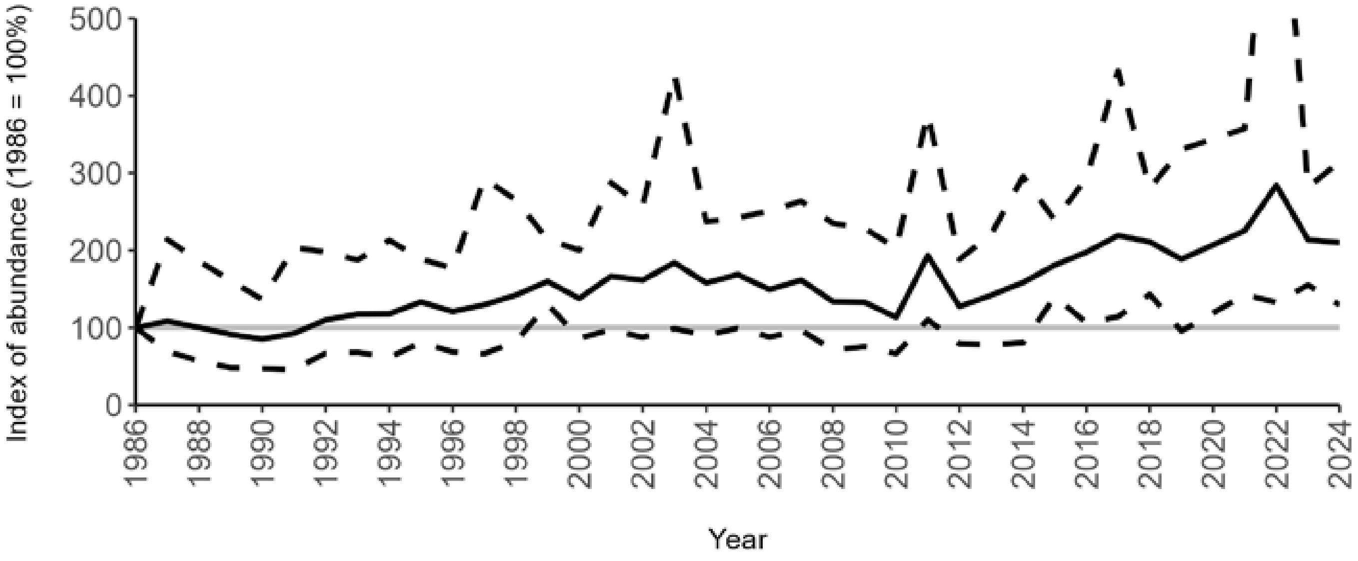


Figure 75: Scotland SMP Breeding Abundance (1986–2024)

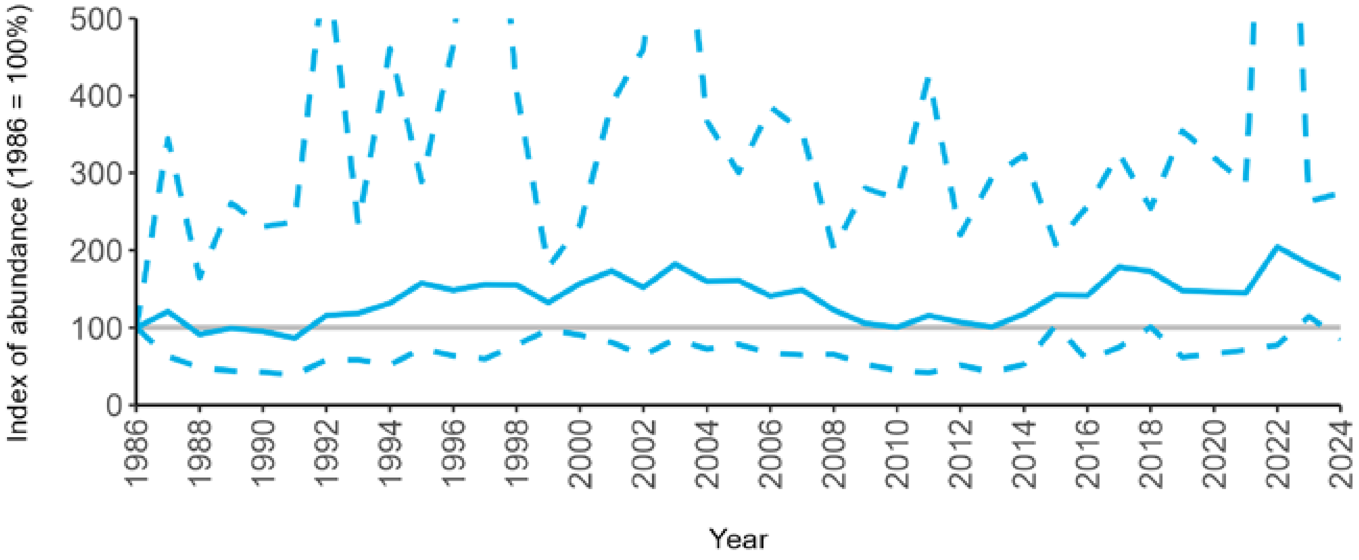


Figure 76: Wales SMP Breeding Abundance (1986–2024)

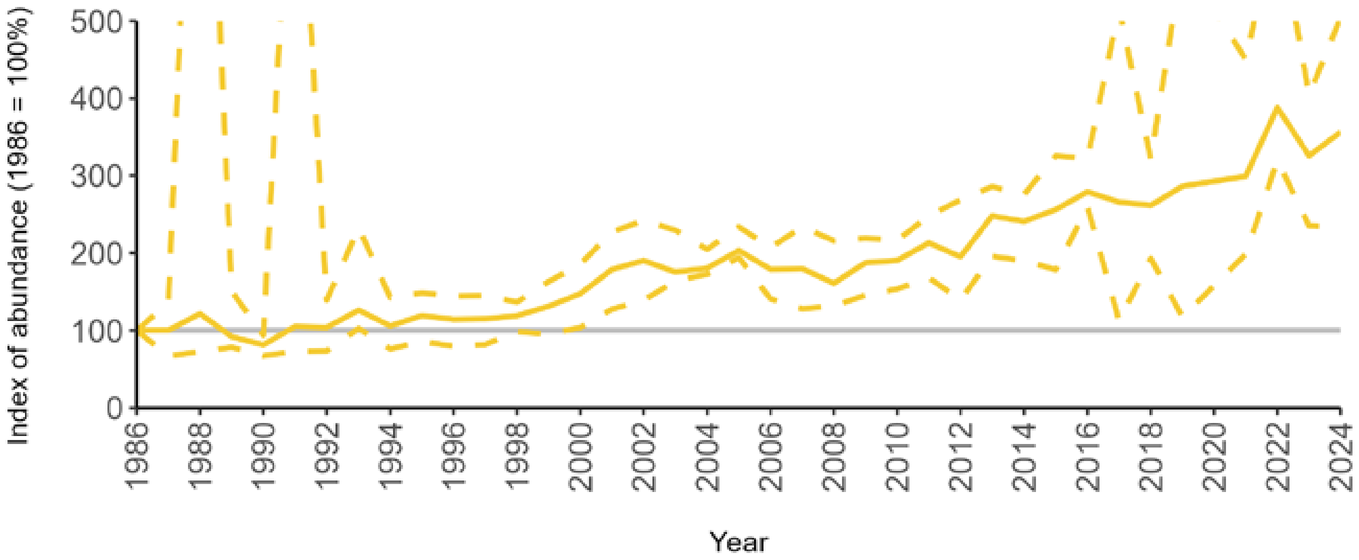
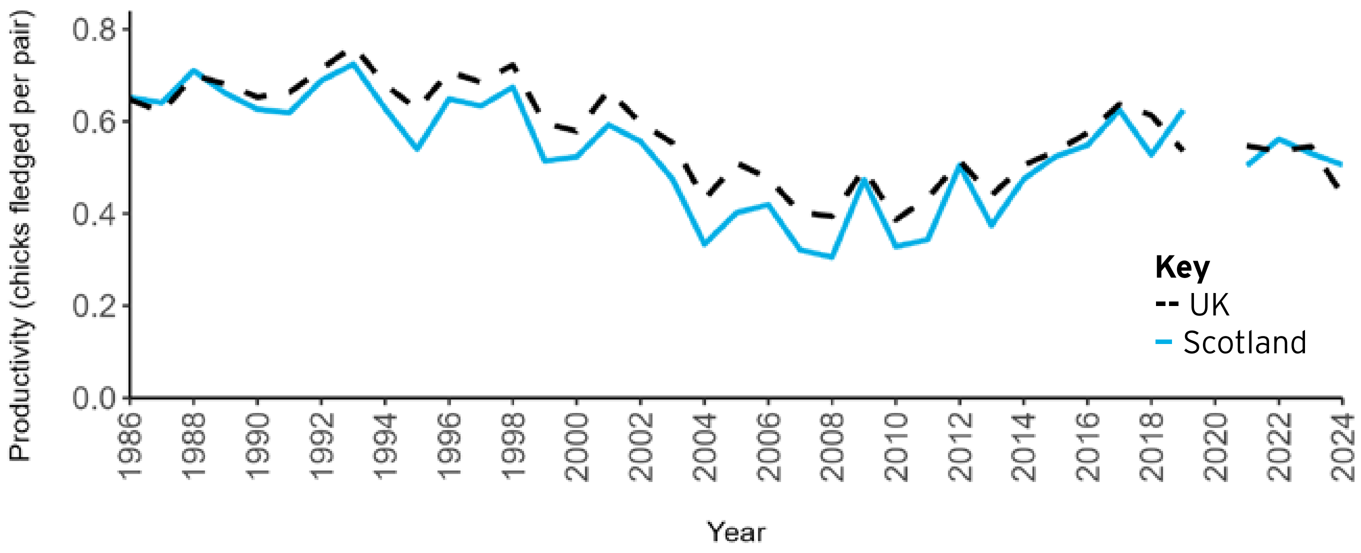


Figure 77: SMP Productivity (1986–2024)



▶

baseline, whilst the index values for Scotland and Wales were 63% and 256% above the baseline, respectively (Table 47). It should be noted that the confidence intervals are wide for several years for Scotland and Wales across the recording period, therefore these indices should be used with caution.

Too few data are currently submitted to the SMP from other regions to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

The productivity trends for Razorbill in the UK and Scotland across the SMP monitoring period have been similar (Figure 77), as most monitored colonies are in Scotland, although UK values have tended to be slightly higher. After an initial period of stability, there was an overall decline in the productivity trends for both regions from the late 1990s to a low point around 2008, after which they started to recover. Productivity estimates have declined since 2019, with 0.44 and 0.51 chicks fledged per breeding pair in 2024, for the UK and Scotland respectively (Table 47).

Too few data are submitted to the SMP on productivity of Razorbills from other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

A range of additional data on Razorbills are collected at the SMP Key Sites. Data on breeding phenology are recorded on Skomer Island (Wales), whilst diet data are collected at the Isle of May (Scotland). Information on adult survival is gathered on Canna (Scotland) and Skomer Island, and on adult annual return rates on the Isle of May.

CAUSES OF CHANGE

Razorbill numbers in Britain and Ireland have continued their steady increase since national seabird censuses began (Burnell *et al.* 2023). However, a few areas showed declines in the *Seabirds Count* census, particularly in the north, and Razorbills, in common with many other seabirds, face a number of pressures.

Extreme climatic events such as intense winter storms can take their toll on Razorbills, through a reduction in their ability to forage and a consequent loss of body condition. This can lead to widespread mortality and their appearance in seabird wrecks (Underwood & Stowe 1984). In 2007, many dead Scottish Razorbills were washed up in the North Sea, Skagerrak and Kattegat (Heubeck *et al.* 2011). A large wreck also occurred in the winter of 2013/14, along the English and Irish Atlantic coasts and down as far as Spain, with post-mortems suggesting starvation as one of the main causes of death (Jessop 2014, Sellers 2014).

Alterations in the distribution and availability of prey species through climate change-induced sea surface temperature changes (Heath *et al.* 2009, Régnier *et al.* 2017, Sandvik *et al.* 2005, Wanless *et al.* 2005) have the potential to impact Razorbill populations in the future, as favoured prey potentially become scarcer. However, studies on the diet of Razorbills have shown a degree of plasticity in food consumption, suggesting that they can switch to alternative prey if the preferred target is less abundant (Gaston & Woo 2008, Barrett 2015).

Razorbills are a pursuit diver, and as a result are often caught in fishing nets as bycatch during commercial fishing (Costa *et al.* 2018, Žydelis *et al.* 2013). Commercial fishing also targets some of the same prey species as Razorbill, so may be reducing the available food supply (Brochet *et al.* 2017, Nettleship *et al.* 2018b).

Although it is thought that Razorbills have a low risk of collision with wind turbines, offshore developments can cause a moderate risk of displacement, impacting on the availability of foraging areas (Bradbury *et al.* 2014). Razorbills are also considered to be vulnerable to the adverse effects of tidal turbines, due to their pursuit diving foraging behaviour (Furness *et al.* 2012).

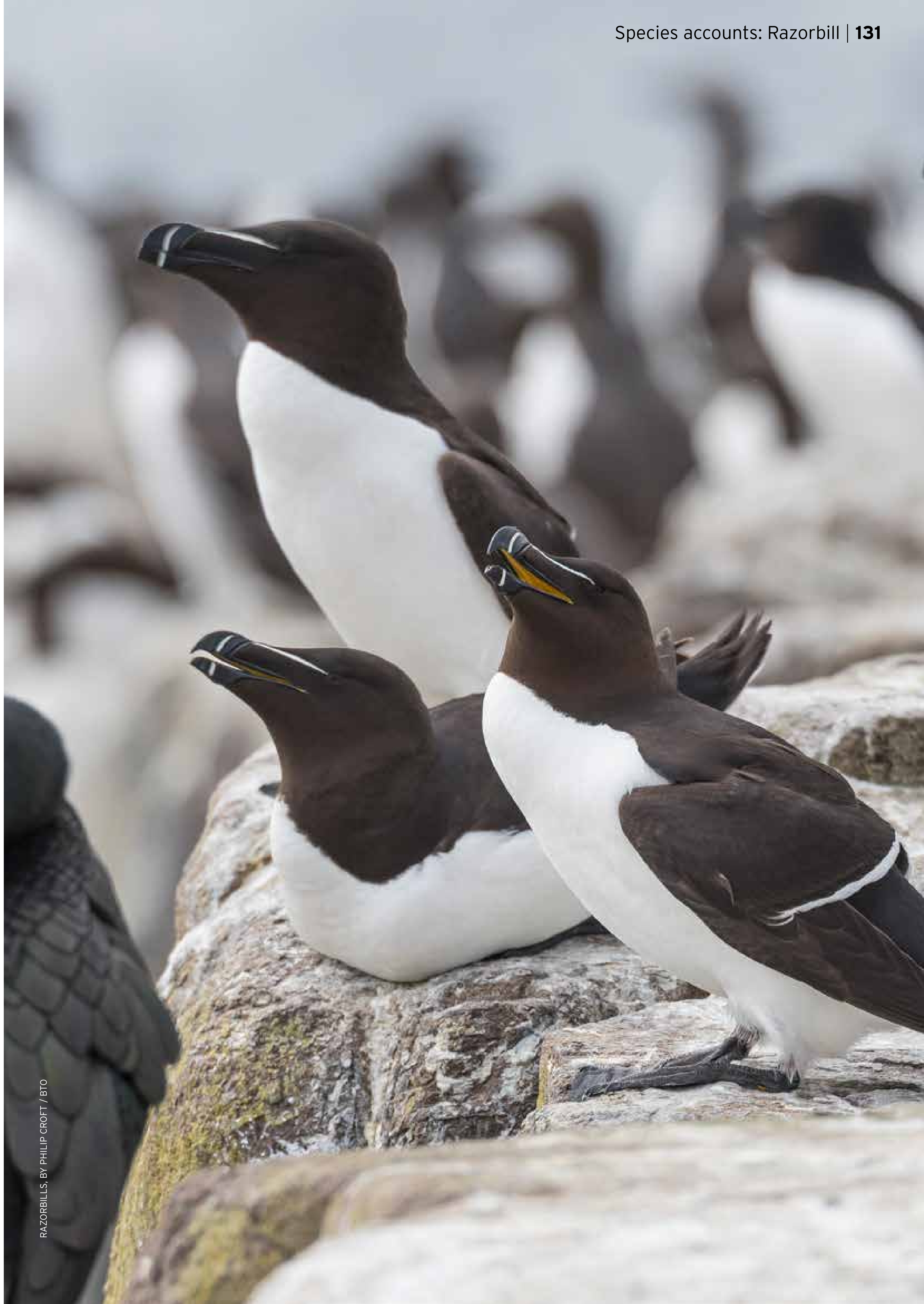
Razorbills are vulnerable to predation from a variety of animals, including rats, Great Skuas and American Mink (Bonesi & Palazon 2007, Swann *et al.* 2016). Brown Rats are known to have caused colony declines in a number of locations, including Canna (Scotland) (Swann 2002), and American Mink have impacted populations in south-west Finland (Nordström *et al.* 2003).

As is the case for all auks, Razorbills are particularly susceptible to the ill effects of oil spills, with oil contamination reducing their ability to fly and forage (Biliavskiy & Golod 2012), often leading to death.

CONSERVATION

Measures likely to have widespread benefits for the Razorbill population include limiting the degree of climate change, assessment and consideration of the full impacts of offshore renewable developments, design and implementation of innovative fishery bycatch mitigation methods, and a reduction in the frequency of oil pollution incidents.

At a local level, rat eradication projects on seabird islands such as Lundy (England) and Canna (Scotland) have been followed by increases in Razorbill numbers (Swann *et al.* 2021). ■



RAZORBILLS, BY PHILIP CROFT / BTO

Black Guillemot

Cepphus grylle

c.3–10%

Abundance: Stable
Productivity: n/a

Green-listed
Amber-listed (I)

Colony Count sites: 62
Breeding Success sites: 1

Least Concern

Lifespan: 11 years
Breeding age: 4 years


Coverage in 2024

Britain and Ireland host approximately 3–10% of the known global breeding population of Black Guillemot (Burnell *et al.* 2023). Also known as ‘Tystie’, they use their wings to propel themselves when hunting for prey under the water (Cairns 1987).

DISTRIBUTION

Black Guillemots nest around the coasts of Ireland, Anglesey (Wales), the Isle of Man, and northern and western Scotland, with a small population in the north-east of Scotland (Burnell *et al.* 2023). Globally, Black Guillemots have a circumpolar distribution in the northern hemisphere, with the British and Irish birds being towards the southern edge of their range (BirdLife International 2024). In Britain and Ireland, they are a resident species that only move short distances offshore during winter (BTO 2025).

DIET

Black Guillemots usually forage within 5 km of their colonies and in sea depths of 10–130 m (Cairns 1987, Dehnhard *et al.* 2023, Durinck *et al.* 1994), although birds in Canada’s Northwest Territories can travel up to 55 km from their colonies to feed (BirdLife International 2000).

They are predominantly benthic foragers and birds from some colonies favour feeding in areas with kelp *Laminaria* spp. (Dehnhard *et al.* 2023). Around British and Irish coasts, their summer diet includes butterfish, sandeels and blennies (Harris & Riddiford 1989, Ewins 1990). Although often foraging in similar areas in winter, their diet may alter to include a higher proportion of invertebrate prey, suggesting these are a more important winter food source (Baak *et al.* 2021, Ewins 1990).

BREEDING

Black Guillemots usually nest in coastal rock crevices or under boulders, although they will also use cavities in artificial structures and purpose-built nest boxes and cairns (BTO 2025, Burnell *et al.* 2023, Leonard & Wolsey 2014). They lay a clutch of two eggs.

BREEDING ABUNDANCE

The Black Guillemot abundance changes for the UK and Scotland, reported by the *Seabirds Count* census, indicated declines of 11% since *Seabird 2000* (Burnell *et al.* 2023). Over a slightly longer time period, the 24-year SMP change for the UK differs, showing an increase of 61% since 2000 (Table 49). This discrepancy is likely due to the smaller proportion of sites covered annually by the SMP. The 24-year SMP change for Scotland is more similar to that of the censuses with a decline of 14% since 2000 (Table 49).

The long-term SMP abundance trends for the UK and Scotland are generally closely matched (Figures 78 & 79), as in most years (although not in recent years) the majority of monitored sites are in Scotland. Since the start of the SMP recording period, the trends had remained below the baseline for both regions, following a decline between 1986 and 1988. However, index values from some of the early years of the SMP recording period should be treated with caution as only a small number of sites submitted data, leading to large confidence intervals, which is also the case in more recent years for Scotland. The trend lines for both regions were relatively stable between the mid-1990s and the end of the 2000s, before increasing slightly in the early 2010s. In 2024, the abundance ▶

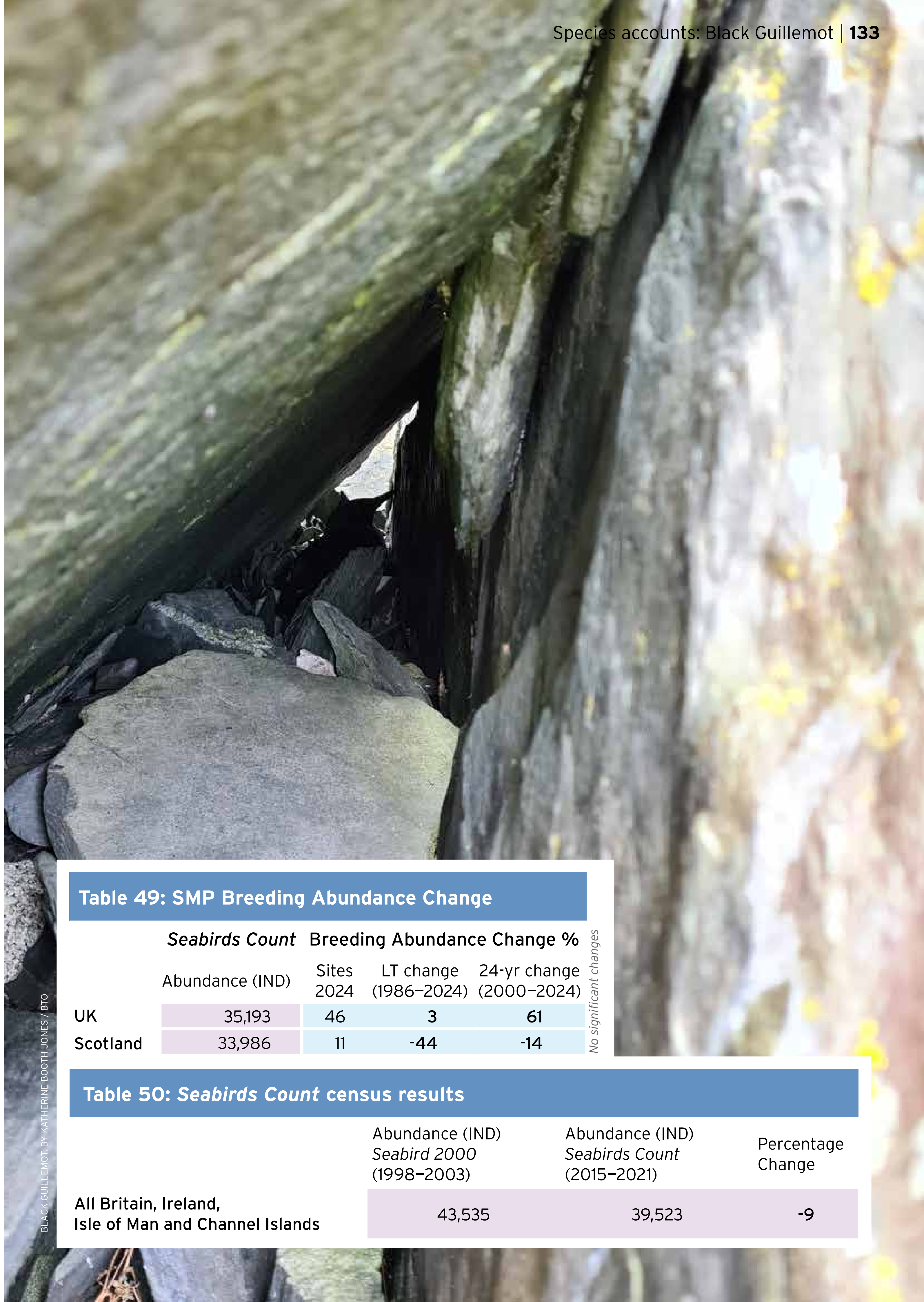

BLACK GUILLEMOT BY KATHERINE BOOTH JONES / BTO

Table 49: SMP Breeding Abundance Change

	Seabirds Count Abundance (IND)	Breeding Abundance Change % Sites 2024	LT change (1986–2024)	24-yr change (2000–2024)
UK	35,193	46	3	61
Scotland	33,986	11	-44	-14

No significant changes

Table 50: *Seabirds Count* census results

	Abundance (IND) *Seabird 2000* (1998–2003)	Abundance (IND) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	43,535	39,523	-9

index value for the UK increased above the 1986 baseline for the first time by 3%, whilst for Scotland the value declined to 44% below the 1986 baseline (Table 49).

Insufficient data are submitted to the SMP on Black Guillemot abundance from other regions and countries to allow for the calculation of meaningful abundance trends.

PRODUCTIVITY

An insufficient number of Black Guillemot colonies are monitored frequently enough to allow for calculation of productivity values for any region.

PHENOLOGY, DIET AND SURVIVAL RATES

No data submitted to the SMP.

CAUSES OF CHANGE

Black Guillemot adults, eggs and chicks are vulnerable to predation during the breeding season due to their ground-nesting, often accessible nesting locations. Predation by various species has been recorded, including mammals such as Brown Rat, European Otter and Domestic Cat, and birds such as Herring Gull and Hooded Crow (*Corvus cornix*) (Greenwood 2014, Johnston *et al.* 2020, JNCC 2021).

Climate change is affecting the abundance, distribution and life cycle timing of a range of seabird prey species through changes in sea surface temperatures (Greenwood 2007, Régnier *et al.* 2017, Wanless *et al.* 2004). In Britain and Ireland, butterfish are the dominant prey species of Black Guillemot (Ewins 1990, Leonard & Wolsey 2014, Shoji *et al.* 2015), and any negative climate change-related impacts on the availability of this species at critical times during the breeding season have the potential to affect Black Guillemot breeding populations.

Climate change may also affect Black Guillemot populations through an increased frequency of extreme weather events and rising sea levels, which may result in challenging foraging conditions and consequent loss of body condition (Hario 2001). Black Guillemots are at the southern edge of their global breeding range in Britain and Ireland and this may also make them more susceptible to climate change (Burnell *et al.* 2023).

Fisheries bycatch poses an additional threat to Black Guillemots, as their diving foraging behaviour renders them vulnerable to being caught in gillnets (Žydelis *et al.* 2013).

Black Guillemots are thought to have a low risk of collision with wind turbines, although offshore wind farms may cause displacement from foraging habitat (Bradbury *et al.* 2014). However, they are thought to be highly vulnerable to the negative impacts of marine tidal renewable energy turbines (Furness *et al.* 2012).

CONSERVATION

Local measures for increasing breeding habitat availability for Black Guillemots have been shown to be effective. These include the provision of nest boxes (Leonard & Wolsey 2014) and the construction of artificial nesting cairns, such as on Grass Holm in Orkney (Burnell *et al.* 2023). There has been an increase in the use of artificial structures for nesting sites in Northern Ireland, benefiting the birds by providing greater protection to the incubating adults, eggs and chicks (Mitchell *et al.* 2004). Redesign of nest boxes to reduce the size of the entrance hole has also been shown to reduce predation by gulls and other predators (Greenwood 2014).

The eradication of ground predators such as rats, Ferret (*Mustela furo*) and stoats from islands on which Black Guillemots breed are also likely to have beneficial effects.

As is the case for all auks, the design of measures which would reduce the threats posed by tidal turbines and commercial fisheries would also benefit Black Guillemots. ■

BLACK GUILLEMOTS, BY KATHERINE BOOTH JONES / BTO

Figure 78: UK SMP Breeding Abundance (1986–2024)

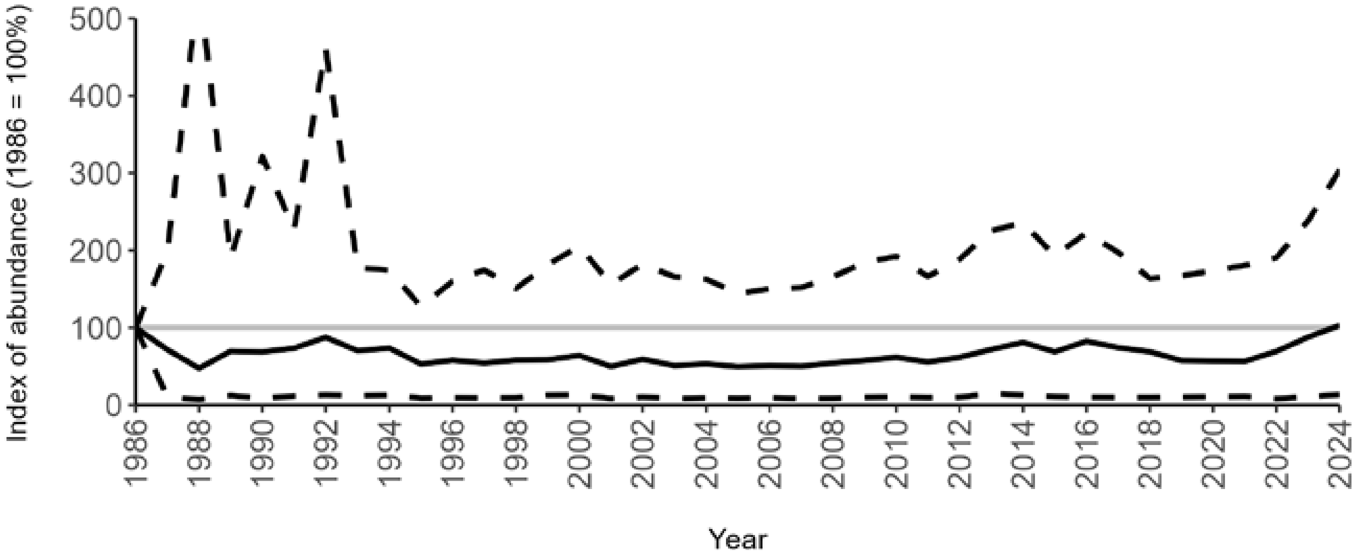
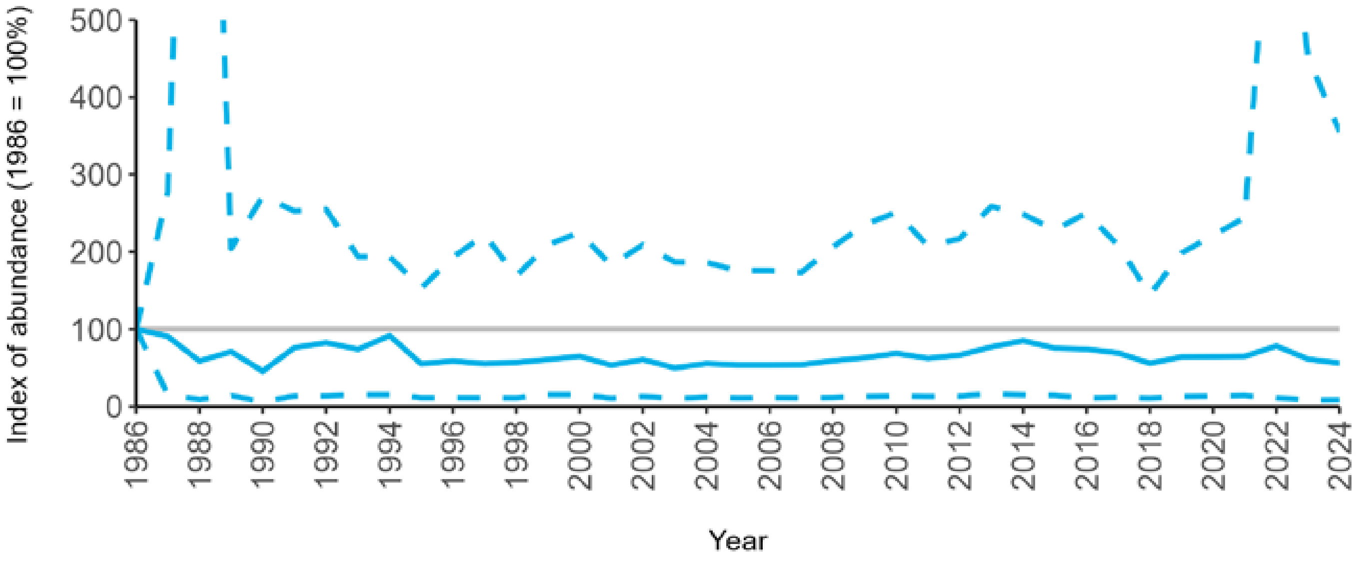


Figure 79: Scotland SMP Breeding Abundance (1986–2024)



Puffin

Fratercula arctica

c.8%

Abundance: n/a
Productivity: 0.56

Red-listed
Red-listed (I)

Colony Count sites: 48
Breeding Success sites: 8

Vulnerable

Lifespan: 18 years
Breeding age: 5 years

Britain and Ireland host approximately 8% of the global and 9% of the European breeding populations of Puffin (Burnell *et al.* 2023). The body size of Puffins is largest in the northern latitudes of their range, and smallest in their southern range. Body size is smallest in the UK, France and southern Norway, and largest in the High Arctic, e.g. Canada, Greenland, Svalbard and Russia (Burnell *et al.* 2023, Kersten *et al.* 2021, Leigh *et al.* 2022).

DISTRIBUTION

Puffin colonies are found around the coastline of much of Britain and Ireland, with the exception of south-east England, and Scotland holds the majority of the population (Burnell *et al.* 2023).

Globally, Puffins breed around the coasts of the North Atlantic and Arctic Oceans, with the highest numbers found in Iceland, Norway, Canada and the Faroe Islands (BirdLife International 2024).

In the non-breeding season, they range extensively across their respective surrounding oceans. British and Irish breeders can be found throughout the North Atlantic Ocean and North Sea, with a few even venturing into the Mediterranean Sea (BTO 2025, JNCC 2021, Fayet *et al.* 2017).

DIET

Puffins forage by pursuit dives and are agile underwater, reaching depths of 60 m (Burger & Simpson 1986). Their predominant prey is Lesser Sandeel, but they will also take European Sprat, Atlantic Herring and juvenile gadoid fish (Harris & Wanless 2011).

BREEDING

Puffins are burrow nesters and, whilst they will excavate their own burrows, they will also use those dug by other species such as Manx Shearwater and European Rabbit (BTO 2025). Extensive burrowing has been known to lead to such extreme soil erosion that the colony can collapse and be forced to move elsewhere, as happened on the island of Grassholm (Wales), which declined from estimates of 250,000 birds in 1890 to just two breeding pairs in 1973 and, thereafter, no breeding was confirmed (Boag & Alexander 1995, Morgan 2012, Pritchard *et al.* 2021).

Puffins are highly colonial and typically nest in soil burrows on the slopes and cliffs of isolated headlands and islands, or less commonly amongst boulder screes or in sheer cliff cracks. They lay a single egg, which is incubated for 36–45 days (Harris & Wanless 2011).

Like other burrow-nesting seabirds, they are easy prey for mammalian predators, so colonies are often in locations where these are absent (JNCC 2021).

BREEDING ABUNDANCE

Too few Puffin colonies are monitored in Britain and Ireland to enable the production of valid annual breeding abundance trends for any region, due to the logistical and financial challenges involved in monitoring this burrow-nesting species. Methodological changes and inconsistencies between national censuses can make it difficult to accurately assess population changes for this species.

The *Seabirds Count* census indicated that the Puffin population in Britain and Ireland had declined by 24% since *Seabird 2000* (Burnell *et al.* 2023) when only sites/records surveyed using the same method, units and sufficiently similar timings between *Seabirds Count* and *Seabird 2000* were included in the analysis (Table 51).

PUFFIN, BY SARAH KELMAN / BTO

PRODUCTIVITY

Puffin productivity trends for the UK and Scotland are closely matched (Figure 80) as a large proportion of monitored sites are in Scotland. Productivity values for the UK and Scotland have fluctuated between years over the SMP recording period. The trends declined from the mid-1990s to a low in 2007, following which they increased overall until 2021. Compared to 2023, there was a small increase in the 2024 productivity estimates for the UK and Scotland, with 0.56 and 0.58 chicks fledged per pair, respectively (Table 52). Too few data are submitted to the SMP on productivity of Puffins from other regions to calculate any meaningful productivity values.

PHENOLOGY, DIET AND SURVIVAL RATES

No systematic data on phenology have been collected as part of the SMP. However, diet information is collected on the Key Sites of Fair Isle and the Isle of May (both in Scotland), whilst information on adult survival is gathered on the Isle of May and Skomer Island (Wales).

CAUSES OF CHANGE

Rising sea surface temperatures are impacting the sandeel populations on which Puffins rely for food (Régnier *et al.* 2017, Wanless *et al.* 2004, 2018). Consequently, adult Puffins need to travel further to forage, which is more energetically costly (Fayet *et al.* 2021). A reduction in food availability during breeding seasons, causing low productivity, is thought to have contributed to population declines at a number of colonies (Owen *et al.* 2018), whilst increases have been noted where favourable foraging conditions occur near colonies (Fayet *et al.* 2021). ▶

Table 51: *Seabirds Count* census results

COMPARABLE FIGURES – NOT COMPLETE COUNTS See main text under 'Breeding Abundance'	Abundance (AOB) *Seabird 2000* (1998–2002)	Abundance (AOB) *Seabirds Count* (2015–2021)	Percentage Change
All Britain, Ireland, Isle of Man and Channel Islands	332,805	254,162	-24

Figure 80: SMP Productivity (1986–2024)

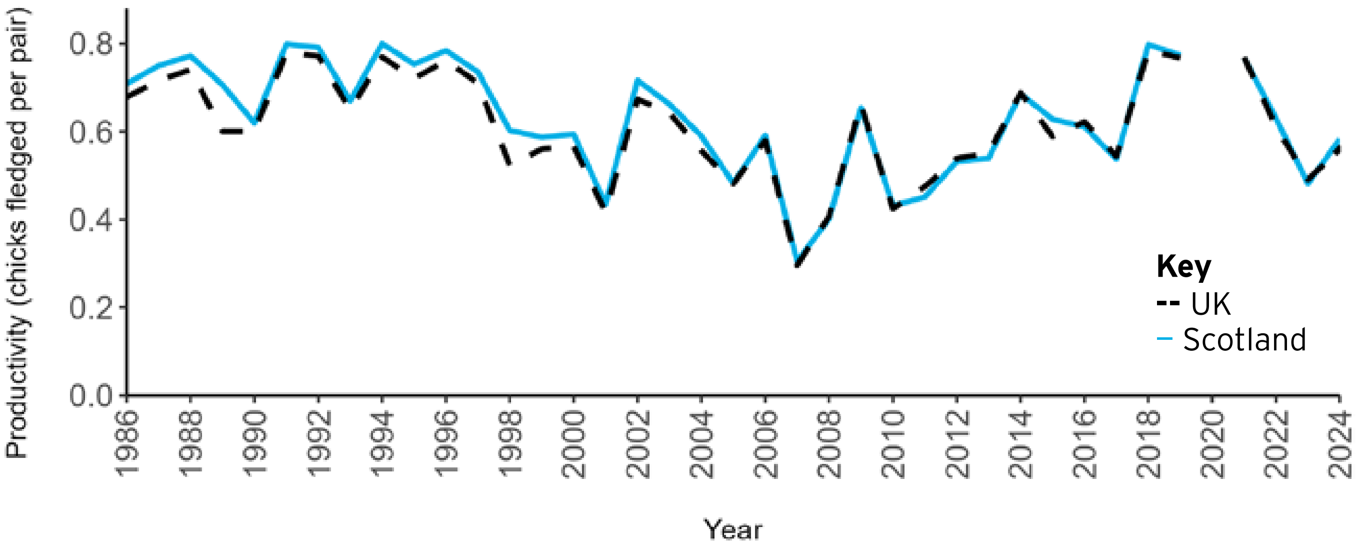


Table 52: SMP Productivity

	Productivity	
	2024	Sites
UK	0.56	8
Scotland	0.58	4

Puffins are vulnerable to mammalian predators, such as Domestic Cats and rats (Brown *et al.* 2011). Avian predators include Great Skua (Miles *et al.* 2015) and large gull species (Langlois Lopez *et al.* 2023, Finney *et al.* 2001). They can also be the target of kleptoparasitic behaviour (Burnell *et al.* 2023, Finney *et al.* 2001), which can reduce the provisioning of prey to chicks.

Increased winter storm frequency due to climate change may also be impacting Puffin populations. These can make foraging more challenging and can lead to a deterioration of body condition, and in extreme cases starvation, resulting in seabird wrecks (Jessop 2014, JNCC 2021).

Offshore wind farms, installed to help tackle climate change, have been identified as a potential threat to the species. These are often located on sandbanks, which are nursery grounds for breeding sandeels (Kenyon & Cooper 2005) and, as such, are favorable Puffin foraging areas. The presence of wind farms can make these areas unsuitable for Puffins due to displacement or barrier effects (Searle *et al.* 2014).

CONSERVATION

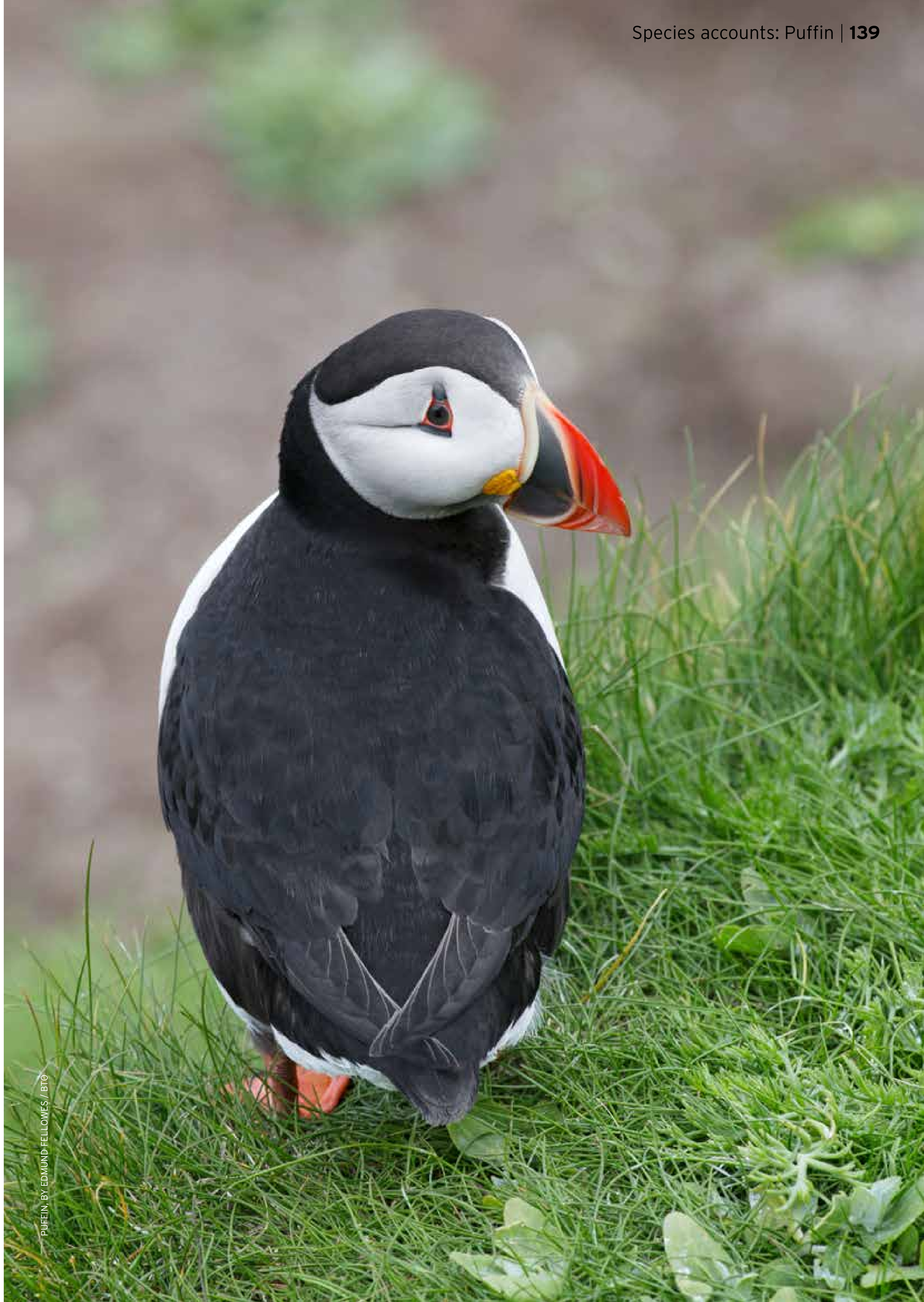
Eradication of predatory mammals from islands with breeding Puffins can prove beneficial. For example, the successful removal of Black Rats (*Rattus rattus*) and Brown Rats from the island of Lundy (England) in 2004 was followed by a rapid increase in Puffin numbers (Brown *et al.* 2011, Lock 2006). Biosecurity measures are also vital, to prevent the colonisation or reinvasion of Puffin breeding islands by invasive predators.

An expansion in the breeding range of Puffins could also be achieved by attracting prospecting adults to predator-free islands before the breeding season. This was successfully achieved at Copeland (Northern Ireland) using tape lures and decoys, resulting in confirmed nesting in 2015 (Booth Jones & Wolsey 2017).

Additional conservation measures likely to benefit Puffins include the careful placement of offshore wind farms in areas away from Puffin colonies and important foraging areas, and reductions in commercial fishing of key prey species, such as sandeels (Burnell *et al.* 2023).

The impact of HPAI on Puffin populations is currently unknown. In the breeding season of 2022, HPAI was confirmed in six out of 25 Puffin carcasses sent for testing, including in a bird from St Kilda, the largest Puffin colony in Britain and Ireland (APHA 2024, Burnell *et al.* 2023). Continued monitoring will be crucial in understanding any impacts from the virus.

Accurate monitoring of the abundance and productivity of Puffins, both through the SMP and national censuses, is currently logistically, methodologically and financially challenging. The development of new monitoring technologies, such as the use of Uncrewed Aerial Vehicles (UAVs), time-lapse photography and burrow sensors, will be critical to better evaluating and understanding Puffin population trends in the future (Burnell *et al.* 2023). ■


PUFFIN, BY EDMUND FELLOWES / BTO

How to get involved

The SMP has collated breeding seabird data for decades, collected and submitted by professional surveyors and enthusiastic volunteers across Britain and Ireland, including the Channel Islands and Isle of Man.

Periodic seabird censuses, such as *Seabirds Count*, provide an invaluable report on the condition of breeding seabird populations. However, censuses can only be delivered every 10 to 15 years due to the time consuming and costly process of completing coordinated counts. As observed with the recent collapse of Gannet and Great Skua breeding colonies due to HPAI, things can change very quickly. Annual seabird surveys like SMP can be more sensitive to capturing stochastic changes in breeding populations which may not be picked up by periodic censuses. Annual monitoring is, therefore, essential for informing effective and timely conservation, and by increasing the coverage of seabird sites across Britain and Ireland we will have more accurate data and a better understanding of how populations are faring.

Additionally, seabird productivity varies considerably from year to year and is highly dependent on the environmental conditions during the chick provisioning period, and is sensitive to extreme weather conditions. Breeding Success surveys are something that, when conducted annually, can deliver a valuable productivity time-series that may offer an early warning system to e.g. the collapse of fisheries and other oceanographic processes that are difficult to monitor.

With exciting advancements in seabird monitoring methodologies and the advancements underway for the SMP – from data entry to data analysis methods and survey sampling – now is the perfect time to get involved in monitoring seabird colonies!

SURVEYS

There are two different surveys you can take part in to contribute to the SMP: Colony Counts and Breeding Success. Colony Counts, depending on the site, generally require one visit to count the nest sites, territories or individual breeding adults within the whole colony. However, when surveying plots, additional visits are required. These visits can last from 20 minutes to several hours depending on the number of birds and the size of the plot. Breeding Success surveys, where the number of chicks that have fledged from active nests are recorded, also require multiple visits to the colony.

Depending on the species you wish to monitor at your site, the survey season starts in late March (for early breeders like Black Guillemot) and ends in October (for late fledging Gannets and Manx Shearwaters).

SIGN UP

If you have never taken part in a BTO survey before, you will need to create a BTO account in order to sign up to take part in the SMP. Visit www.bto.org to create your BTO account and then sign up to the SMP 'project' specifically.

FIND A SITE TO MONITOR

Next, visit SMP Online (app.bto.org/seabirds) and search for SMP sites in your area. There are seabird monitoring sites across Britain and Ireland at both coastal and inland locations, and the map shows you where these sites are.

Sites marked as already having active counters may not have all seabird species covered, so if there is an 'allocated' site that you are interested in monitoring, please get in touch with the SMP Organiser to find out more at smp@bto.org. Finally, sites hosting nesting seabirds not yet on the map can also be added to the programme.

REVIEW THE METHODS

To take part, you will need to be able to identify the seabirds present at your site. You will also need to be able to follow the prescribed methodology in the Seabird Monitoring Handbook. The handbook specifies times, dates and methods for each species which will need to be used to ensure that data are standardised and comparable with previous counts.

COMPLETE SURVEY AND ENTER DATA

After you have completed your survey you will need to input your data through the SMP Online data entry portal – with guidance available online – by the end of October each year.

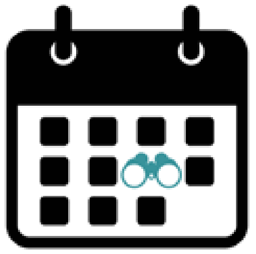
HAVE FUN!

And most importantly, get out into the field and have fun. Monitoring seabirds is a fantastic way to observe this fascinating group of animals first hand. Head to the windswept coastline and see Gannets diving at breakneck speeds to capture prey, or travel to inland colonies where gulls can be seen flying atmospherically over moorland.

SMP ANNUAL TIMELINE

1. February–March
Sign up for SMP and register for your site(s).

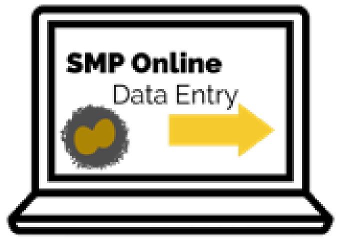
2. Late March
Beginning of monitoring season for early breeders.



3. May–July
Key months for monitoring most seabird species.

4. July–September
Productivity recording ends as the last chicks fledge.

5. October
Deadline for data submission through the SMP online portal.



6. Await results
Data feeds into trend analysis, research and publications.

GRAPHICS, BY SARAH HARRIS/CANVA.COM

▶ A mix of participants, from experienced professional surveyors monitoring seabirds as part of their working life, to volunteers who discovered seabird monitoring recently, and everything in between, describe what drives them to take part in the SMP.

"I've been lucky enough to monitor the seabirds in Scilly since 2000. Monitoring their fortunes and adding this data to the SMP, we are able to put our findings into a national context to understand wider trends."
Dr Vickie Heaney

"Getting involved with the Seabird Monitoring Programme gives you a chance to get lost in the seabird's world!"
Kate Fox

"It gives us an opportunity to learn practical skills in seabird monitoring whilst also knowing we are contributing to an incredibly valuable dataset that underpins our understanding of seabird demography in the UK."
Sam Langlois

"…it's a great excuse to get out and see the incredible seabird colonies that we're privileged to share our islands with!"
Sophie Bennett

"By entering counts on the SMP database, I know I'm providing valuable data towards the national inventory of our seabirds."
Murray Orchard

SLIDE BACKGROUND, BY JOYIMAGE AND OPENCLIPART-VECTORS/CANVA

FIND OUT MORE...

Take Part in the SMP: www.bto.org/smp
Sign up to SMPnews: www.bto.org/smp-news
Follow on Bluesky: @smp-seabirds.bsky.social

References (A–B)

In text ref.	Reference
Acampora *et al.* 2018	**Acampora, H., Ní Dhonnabháin, L., Miley, D. & Newton, S.** 2018. Rockabill Tern Report 2018. BirdWatch Ireland Seabird Conservation Report.
Alerstam *et al.* 2019	**Alerstam, T., Bäckman, J., Grönroos, J., Olofsson, P. & Strandberg, R.** 2019. Hypotheses and tracking results about the longest migration: The case of the arctic tern. *Ecology and Evolution* **9** : 9511–9531.
Alley *et al.* 2022	**Alley, C.L., Arkless, S., Ames, E., Abrahams, M., Gentle, L.K. & Wood, M.J.** 2022. Plastic ingestion in adult and fledgling Manx Shearwaters *Puffinus puffinus* on Skomer Island, Wales. *Seabird Journal* **34** : 33–44.
Almeida *et al.* 2023	**Almeida, A., Alonso, H., Oliveira, N., Silva, E. & Andrade, J.** 2023. Using a visual deterrent to reduce seabird interactions with gillnets. *Biological Conservation* **285** : 110236–110236.
Álvarez *et al.* 2018	**Álvarez, G., Barros, Á. & Velando, A.** 2018. The use of European Shag pellets as indicators of microplastic fibers in the marine environment. *Marine Pollution Bulletin* **137** : 444–448.
Amaral *et al.* 2010	**Amaral, J.J., Almeida, S., Sequeira, M. & Neves, V.C.** 2010. Black rat *Rattus rattus* eradication by trapping allows recovery of breeding roseate tern *Sterna dougallii* and common tern *S.hirundo* populations on Feno Islet, the Azores, Portugal. *Conservation Evidence* **7** : 16–20.
Anderson 1982	**Anderson, A.** 1982. The establishment and growth of a new Fulmar colony on sand dunes. *Bird Study* **23** : 189–194.
Anderson & Green 2009	**Anderson, A. & Green, R.E.** 2009. The value of ringing for bird conservation. *Ringing & Migration* **24** : 205-212.
Anderson *et al.* 2014	**Anderson, H.B., Evans, P.G.H., Potts, J.M., Harris, M.P. & Wanless, S.** 2014. The diet of Common Guillemot *Uria aalge* chicks provides evidence of changing prey communities in the North Sea. *Ibis* **156** : 23–34.
Anderson *et al.* 2011	**Anderson, O.R.J., Small, C.J., Croxall, J.P., Dunn, E.K., Sullivan, B.J., Yates, O. & Black, A.** 2011. Global seabird bycatch in longline fisheries. *Endangered Species Research* **14** : 91–106.
APHA 2024	**APHA (Animal and Plant Health Agency).** 2024. Bird flu *(avian influenza): cases in wild birds*. Available at: https://www.gov.uk/government/publications/avian-influenza-in-wild-birds [Accessed 25/10/2024].
Araújo *et al.* 2022	**Araújo, H., Correia-Rodrigues, P., Debru, P., Ferreira, M., Vingada, J. & Eira, C.** 2022. Balearic shearwater and northern gannet bycatch risk assessment in Portuguese continental waters. *Biological Conservation* **267** : 109463.
Arneill *et al.* 2020	**Arneill, G.E., Critchley, E.J., Wischnewski, S., Jessopp, M.J. & Quinn, J.L.** 2020. Acoustic activity across a seabird colony reflects patterns of within-colony flight rather than nest density. *Ibis* **162** : 416-428.
Atkinson & Bailie 2024	**Atkinson, P.W. & Baillie, S.R.** 2024. Jumping species and seasons – the spread and impact of highly pathogenic avian influenza on seabirds and waterbirds. *Bird Study* **71** 289-292.
Atkinson & Bailie 2025	**Atkinson, P.W. & Baillie, S.R.** 2025. Responding to high pathogenicity avian influenza (HPAI) and the conservation crisis in wild birds – where next? *Bird Study* **72**: 1-4.
Atkinson *et al.* 2025	**Atkinson, P.W., Balmer, D.E., Banyard, A.C., Duggan, J., Falchieri, M., Frost, T.M., Humphreys, E.M., Jones, R., Langlois Lopez, S., Miles, W.T. & Murphy, M.** 2025. Evaluating the use of carcass and testing data to assess the high pathogenicity avian influenza (HPAI) related mortality in wild birds in the United Kingdom and Crown Dependencies between 2021-2023. *Bird Study* 20–38.
Ausden & Fuller 2009	**Ausden, M. & Fuller, R.J.** 2009. Birds and habitat change in Britain part 2: Past and future conservation responses. *British Birds* **102** : 52–71.
Avery *et al.* 1995	**Avery, M.I., Coulthard, N.D., Del Nevo, A.J., LeRoux, A., Medeiros, F., Merne, O., Monteiro, L., Moralee, A., Ntiamoa-Baidu, Y., O'Briain, M. & Wallace, E.** 1995. A recovery plan for Roseate Terns in the East Atlantic: an international programme. *Bird Conservation International* **5** : 441–453.
Baak *et al.* 2021	**Baak, J.E., Leonard, M.L., Gjerdrum, C., Dodds, M.D. and Ronconi, R.A.** 2021. Non-breeding movements and foraging ecology of the Black Guillemot *Cepphus grylle* in Atlantic Canada. *Marine Ornithology* **49** : 57–70.
Bailey 1991	**Bailey, R.S. 1991.** Recent changes in the population of the sandeel (*Ammodytes marinus* Raitt) at Shetland in relation to estimates of seabird predation. *ICES Marine Science Symposium* **193** : 209–216.
Balmer *et al.* 2013	**Balmer, D.E., Gillings, S., Caffrey, B., Swann, B., Downie, I. & Fuller, R.** 2013. *Bird Atlas 2007–11: The Breeding and Wintering Birds of Britain and Ireland.* British Trust for Ornithology, Thetford.
Banyard *et al.* 2022	**Banyard, A.C., Lean, F.Z., Robinson, C., Howie, F., Tyler, G., Nisbet, C., Seekings, J., Meyer, S., Whittard, E., Ashpitel, H.F. & Bas, M.** 2022. Detection of Highly Pathogenic Avian Influenza Virus H5N1 Clade 2.3.4.4b in Great Skuas: A Species of Conservation Concern in Great Britain. *Viruses* **14** : 212.
Barcelona *et al.* 2010	**Barcelona, S.G., Ortiz de Urbina, J.M., de la Serna, J.M., Alot, E. & Macías, D**. 2010. Seabird bycatch in Spanish Mediterranean large pelagic longline fisheries, 2000–2008. *Aquatic Living Resources* **23** : 363–371.
Barrett 2015	**Barrett, R.T.** 2015. The diet, growth and survival of Razorbill *Alca torda* chicks in the southern Barents Sea. *Ornis Norvegica* **38** : 25–31.
Bart & Johnston 2012	**Bart, J. & Johnston, V. (eds.)** 2012. *Arctic shorebirds in North America: a decade of monitoring.* Berkeley, CA: University of California Press.
BBC Weather 2024	**BBC Weather.** 2024. *UK weather summaries and reports.* Available at: www.bbc.co.uk/weather [Accessed: 03/06/2025].
Bennison *et al.* 2024	**Bennison, A., Adlard, S., Banyard, A.C., Blockley, F., Blyth, M., Browne, E., Day, G., Dunn, M.J., Falchieri, M., Fitzcharles, E., Forcada, J., Forster Davidson, J., Fox, A., Hall, R., Holmes, E., Hughes, K., James, J., Lynton-Jenkins, J., Marshall, S., McKenzie, D., Morley, S.A., Reid, S.M., Stubbs, I., Ratcliffe, N. & Phillips, R.A.** 2024. A case study of highly pathogenic avian influenza (HPAI) H5N1 at Bird Island, South Georgia: the first documented outbreak in the subantarctic region. *Bird Study* **71**: 380–391.
Bernard *et al.* 2021	**Bernard, A., Rodrigues, A.S.L., Cazalis, V. & Grémillet. D.** 2021. Toward a global strategy for seabird tracking. *Conservation Letters* **14** : e12804.
Bicknell *et al.* 2013	**Bicknell, A.W.J., Oro, D., Camphuysen, J.C. & Votier, S.C.** 2013. Potential consequences of discard reform for seabird communities. *Journal of Applied Ecology* **50** : 649–658.
Biliavskiy & Golod 2012	**Biliavskiy, G. & Golod, A.** 2012. Oil pollution of the Black Sea. *Proceedings of the National Aviation University* **51** : 92–95.
Billerman *et al.* 2020	**S.M. Billerman., B.K. Keeney., P.G. Rodewald, & T.S. Schulenberg** (eds). 2022. *Birds of the World.* Cornell Laboratory of Ornithology, Ithaca, NY, USA. Available at: https://birdsoftheworld.org/bow/home [Accessed: 25/10/2024]
BirdLife International 2015	**BirdLife International.** 2015. *European Red List of Birds 2015.* Office for Official Publications of the European Communities, Luxembourg. Available at: https://www.iucnredlist.org/resources/birdlife2015#:~:text=2015,16%20Endangered%20and%2055%20Vulnerable [Accessed: 25/10/2024].
BirdLife International 2000	**BirdLife International.** 2000. The Development of Boundary Selection Criteria for the Extension of Breeding Seabird Special Protection Areas into the Marine Environment. OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic. Meeting of the Biodiversity Committee (BDC) Vlissingen (Flushing), 20–24 November, 2000, Agenda Item 8.
BirdLife International 2022	**BirdLife International.** 2022. *State of the World's Birds 2022: Insights and solutions for the biodiversity crisis.* Cambridge, UK: BirdLife International Available at: https://www.birdlife.org/wp-content/uploads/2022/09/SOWB2022_EN_compressed.pdf [Accessed: 20/06/2025].
BirdLife International 2024	**BirdLife International.** 2024. *IUCN Red List for birds.* Available at: https://datazone.birdlife.org/species/search [Accessed: 25/10/2024].
Black 2018	**Black, C.E.** 2018. Spying on seabirds: a review of time-lapse photography capabilities and limitations. *Seabird* **31** : 1–14
Blokpoel *et al.* 1997	**Blokpoel, H., Tessier, G.D. & Andress R.A.** (Bud). 1997. Successful Restoration of the Ice Island Common Tern Colony Requires On-going Control of Ring-billed Gulls. *Colonial Waterbirds* **20** : 98–101.
Boag & Alexander 1995	**Boag, D and Alexander, M.** 1995. *The Puffin*. Blandford Press. London.
Bogdanova *et al.* 2011	**Bogdanova, M.I., Francis, F., Newell, M., Phillips, R.A., Harris, M.P. & Wanless S.** 2011. Seasonal interactions in the black-legged kittiwake, *Rissa tridactyla*: links between breeding performance and winter distribution. *Proceedings of the Royal Society B* **278** : 2412–2418.
Bolton 2021	**Bolton, M.** 2021. GPS tracking reveals highly consistent use of restricted foraging areas by European Storm-petrels *Hydrobates pelagicus* breeding at the largest UK colony: implications for conservation management. *Bird Conservation International* **31** : 35–52.
Bolton *et al*. 2010	**Bolton, M., Brown, J.G., Moncrieff, H., Ratcliffe, N. & Okill, J.D.** 2010. Playback re-survey and demographic modelling indicate a substantial increase in breeding European Storm-petrels *Hydrobates pelagicus* at the largest UK colony, Mousa, Shetland. Seabird Journal **23** : 14–24.
Bond *et al.* 2023	**Bond, A.L., Wilhelm, S.I., Pirie-Hay, D.W., Robertson, G.J., Pollet, I.L. & Arany, J.L.** 2023. Quantifying gull predation in a declining Leach's Storm-petrel (*Hydrobates leucorhous*) colony. *Avian Conservation and Ecology* **18**.
Bonesi & Palazon 2007	**Bonesi, L. & Palazon, S. 2007.** The American Mink in Europe: status, impacts, and control. *Biological Conservation* **134** : 470–483.
Brooke 1990	**Brooke, M.** 1990. *The Manx Shearwater.* T. and A.D. Poyser, London.
Booth Jones 2020	**Booth Jones, K.** 2020. *The Northern Ireland Seabird Report 2019.* British Trust for Ornithology, Thetford.
Booth Jones & Wolsey 2017	**Booth Jones, K & Wolsey, S.** 2017. *Northern Ireland Seabird Report 2016.* British Trust for Ornithology. Thetford.
Booth Jones *et al.* 2022	**Booth Jones, K., Thaxter, C., Clewley, G., Wolsey, S., Calbrade, N., Atkinson, P., Calladine, J. & Burton, N.** 2022. *Belfast's urban gulls: an assessment of breeding populations, breeding season movements and winter population*. BTO Research Report 734. British Trust for Ornithology, Thetford.
Booth & Morrison 2010	**Booth V. & Morrison P.** 2010. Effectiveness of disturbance methods and egg removal to deter large gulls *Larus* spp. from competing with nesting terns *Sterna* spp. on Coquet Island RSPB reserve, Northumberland, England. *Conservation Evidence* **7** : 39–43.
Bond *et al.* 2023	**Bond, A.L., Wilhelm, S.I., Pirie-Hay, D.W., Robertson, G.J., Pollet, I.L. & Arany, J.L.** 2023. Quantifying gull predation in a declining Leach's Storm-petrel (*Hydrobates leucorhous*) colony. *Avian Conservation and Ecology* **18**.
Borker *et al*. 2014	**Borker, A.L., McKown, M.W., Ackerman, J.T. Eagles-Smith, C.A., Tershy, B.R. & Croll, D.A.** 2014. Vocal activity as a low cost and scalable index of seabird colony sites. ***Conservation Biology*** **23** : 1100-1108
Bradbury *et al.* 2014	**Bradbury, G., Trinder, M., Furness, B., Banks, A.N., Caldow, R.W.G. & Hume, D.** 2014. Mapping Seabird Sensitivity to Offshore wind farms. *PLoS ONE* **9** : e106366.
Bregnballe *et al.* 2014	**Bregnballe, T., Lynch, J., Parz-Gollner, R., Marion, L., Volponi, S., Paquet, J.-Y., David N. Carss & van Eerden, M.R.** (eds.) 2014. *Breeding numbers of Great Cormorants Phalacrocorax carbo in the Western Palearctic, 2012–2013.* IUCN-Wetlands International Cormorant Research Group Report, Danish Centre for Environment and Energy No. **99** : 224 Available at: http://dce2.au.dk/pub/SR99.pdf [Accessed 25/10/2024].
Bregnballe & Frederiksen 2006	**Bregnballe, T. & Frederiksen, M.** 2006, Net-entrapment of Great Cormorants *Phalacrocorax carbo sinensis* in relation to individual age and population size. *Wildlife Biology* **12** : 143–150.
Bregnballe *et al.* 2024	**Bregnballe, T., Herrmann, C., Globig, A., Günther, A., Staubach, C., Heise, J. N., Harder, T., Beer, M., Knief, U., Heinicke, T., Leivits, M., Lundström, K., Nurmoja, I., Liang, Y., Larsen, L.E., Hjulsager, C.K., Pohlmonn, A. & Fox, A.D.** 2024. Outbreaks of highly pathogenic avian influenza (HPAI) epidemics in Baltic Great Cormorant *Phalacrocorax carbo* colonies in 2021 and 2022. *Bird Study* **71**: 353–366.
Bried *et al.* 2009	**Bried, J., Magalhães, M.C., Bolton, M., Neves, V.C., Bell, E., Pereira, J.C., Aguiar, L., Monteiro, L.R. & Santos, R.S.** 2009. Seabird habitat restoration on Praia Islet, Azores Archipelago. *Ecological Restoration* **27** : 27–36.

References (B–D)

In text ref.	Reference
Brochet *et al.* 2017	**Brochet, A.-L., Van den Bossche, W., Jones, V. R., Arnardottir, H., Damoc, D., Demko, M., Driessens, G., Flensted, K., Gerber, M., Ghasabyan, M., Gradinarov, D., Hansen, J., Horvath, M., Karlonas, M., Krogulec, J., Kuzmenko, T., Lachman, L., Lehtiniemi, T., Lorgé, P., Lotberg, U., Lusby, J., Ottens, G., Paquet, J.-Y., Rukhaia, A., Schmidt, M., Shimmings, P., Stipnieks, A., Sultanov, E., Vermouzek, Z., Vintchevski, A., Volke, V., Willi, G. & Butchart, S.H.M.** 2019. Illegal killing and taking of birds in Europe outside the Mediterranean: assessing the scope and scale of a complex issue. *Bird Conservation International*, **29** : 10–40.
Brook 1990	**Brooke, M.** 1990. *The Manx Shearwater*. T & AD Poyser, UK.
Brown *et al.* 2011	**Brown, A., Price, D., Slader, P., Booker, H., Lock, L. & Deveney, D. 2011.** Seabirds on Lundy: their current status, recent history and prospects for the restoration of a once important bird area. *British Birds* **104** : 139–158.
Brusa *et al.* 2024	**Brusa, J.L., Farr, M.T., Evenson, J., Silverman, E., Murphie, B., Cyra, T.A., Tschaekofske, H.J., Spragens, K.A. & Converse, S.J.** 2024. Correcting for measurement errors in a long-term aerial survey with auxiliary photographic data. *Ecosphere* **15** : e4961.
BTO 2025	**BTO 2025**. *BirdFacts Species: profiles of birds occurring in the United Kingdom*. BTO, Thetford. Available at: www.bto.org/birdfacts [Accessed: 09/07/2025].
Buckley 1990	**Buckley, N.J.** 1990. Diet and feeding ecology of great black-backed gulls (*Laws marinus*) at a southern Irish breeding colony. *Journal of Zoology* **222** : 363–373.
Buckingham *et al.* 2022	**Buckingham, L., Heward, C., Carter, A., Dixon, T., Micklewright, S., Stace, H. & Ward, R.** 2022. *Treshnish Isles Auk Ringing Report 2022*. Cambridgeshire.
Bull *et al.* 2004	**Bull, J., Wanless, S., Elston, D., Daunt, F., Lewis, S. & Harris, M.** 2004. Local-scale variability in the diet of Black-legged Kittiwakes *Rissa tridactyla*. *Ardea* **92** : 43–52.
Burg *et al.* 2003	**Burg, T. M., Lomax. J., Almond, R., Brooke, M.D.L. & Amos, W.** 2003. Unravelling dispersal patterns in an expanding population of a highly mobile seabird, the northern fulmar (*Fulmarus glacialis*). *Proceedings of the Royal Society of London Series B: Biological Sciences* **270** : 979–984.
Burger *et al.* 2020	**Burger, J., Gochfeld, M. & Garcia. E.F.J.** 2020. *Mediterranean Gull (Ichthyaetus melanocephalus), version 1.0.* In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). 2020. Cornell Lab of Ornithology, Ithaca, NY, USA. Available at: https://doi.org/10.2173/bow.medgul1.01 [Accessed: 25/10/2024].
Burger *et al.* 2018	**Burger, J., Gochfeld, M., Kirwan, G.M., Christie, D.A. & de Juana, E.** 2018. *Lesser Black-backed Gull (Larus fuscus)*. In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). 2018. *Handbook of the Birds of the World Alive*, Lynx Edicions, Barcelona.
Burger *et al.* 2016	**Burger, J., Gochfeld, M., Kirwan, G.M., Christie, D.A. & Garcia, E.F.J.** 2016. Black-legged Kittiwake (*Rissa tridactyla*). In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). 2016. *Handbook of the Birds of the World Alive*, Lynx Edicions, Barcelona.
Burgess & Hirons 1992	**Burgess, N.D., & Hirons, J.M.** 1992. Creation and management of articficial nesting sites for wetland birds. *Journal of Environmental Management* **34** : 285–295.
Burgess & Morris 1992	**Burgess, G.P. & Morris, R.D.** 1992. Shelters decrease gull predation on chicks at a Common Tern colony. *Journal of Field Ornithology* **63** : 186–189.
Burger & Simpson 1986	**Burger, A.E. & Simpson, M.** 1986. Diving depths of Atlantic puffins and common murres. *Auk* **103** : 828–830.
Burke *et al.* 2024	**Burke, B., Adcock, T., Boland, H., Büche, B., Fitzgerald, M., Johnson, G.C., Monaghan, J., Murray, T, Stubbings, E. & Newton, S.** 2024. A case study of the 2023 highly pathogenic avian influenza (HPAI) outbreak in tern (Sternidae) colonies on the east coast of the Republic of Ireland. *Bird Study* **71**: 336–346
Burke *et al.* 2022	**Burke, B., O'Connell, D. P., Kinchin-Smith, D., Sealy, S., & Newton, S. F.** 2022. Nest boxes augment seabird breeding performance in a high-density colony: Insight from 15 years of monitoring data. *Ecological Solutions and Evidence* **3** : e12171
Burnell *et al.* 2023	**Burnell, D., Perkins, A.J., Newton, S.F., Bolton, M., Tierney, T.D., Dunn, T.E.** (eds) 2023. *Seabirds Count: a census of breeding seabirds in Britain and Ireland (2015–2021)*. Lynx Nature Book, Barcelona.
Burthe *et al.* 2014	**Burthe, S., Wanless, S., Newell, M., Butler, A. & Daunt, F.** 2014. Assessing the vulnerability of the marine bird community in the western North Sea to climate change and other anthropogenic impacts. *Marine Ecology Progress Series* **507** : 277–295.
Bustnes 2006	**Bustnes, J.O.** 2006. Environmental pollutants in endangered vs. increasing subspecies of the lesser black-backed gull on the Norwegian Coast. *Environmental Pollution* **144** : 893–901.
Buxton *et al.* 2013	**Buxton, R.T., Major, H.L., Jones, I.L. & Williams, J.C.** 2013. Examining Patterns in Nocturnal Seabird Activity and Recovery Across the Western Aleutian Islands, Alaska, using Automated Acoustic Recording. *The Auk* **130** : 331–341.
Cadiou *et al.* 2010	**Cadiou, B., Bioret, F., & Chenesseau, D.** 2010. Response of breeding European Storm Petrels to habitat change. *Journal of Ornithology* **151** : 317–327.
Cairns 1987	**Cairns, D.K.** 1987. The Ecology and Energetics of Chick Provisioning by Black Guillemots. *The Condor* **89** : 627–635.
Calladine 1997	**Calladine, J.** 1997. A comparison of Herring Gull *Larus argentatus* and Lesser Black-backed Gull *Larus fuscus* nest sites: their characteristics and relationships with breeding success. *Bird Study* **44** : 318–326.
Camphuysen *et al.* 2022	**Camphuysen, C.J., Gear, S.C. & Furness, R.W.** 2022. Avian influenza leads to mass mortality of adult Great Skuas in Foula in summer 2022. *Scottish Birds* **42** : 312–323.
Carboneras *et al.* 2014	**Carboneras, C., Jutglar, F. & Kirwan, G.M.** 2014. Manx Shearwater (Puffinus puffinus). In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). 2014. *Handbook of the Birds of the World Alive*, Lynx Edicions, Barcelona.

In text ref.	Reference
Carroll *et al.* 2017	**Carroll, M.J., Bolton, M., Owen, E., Anderson, G.Q.A., Mackley, E.K., Dunn, E.K. & Furness, R.W.** 2017 Kittiwake breeding success in the southern North Sea correlates with prior sandeel fishing mortality. *Aquatic Conservation: Marine and Freshwater Ecosystems* **27**: 1,164–1,175.
Casey *et al.* 1995	**Casey, S., Moore, N., Ryan, L., Merne, O.J., Coveney, J.A & del Nevo, A.** 1995. The Roseate Tern conservation project on Rockabill, Co. Dublin: a six year review 1989–1994. *Irish Birds* **5** : 251–264.
Chapdelaine *et al.* 1987	**Chapdelaine. G., Laporte, P. & Nettleship, D. N.** 1987. Population, productivity and DDT contamination trends of Northern Gannets (*Sula bassanus*) at Bonaventure Island, Quebec, 1967–1984. *Canadian Journal of Zoology* **65** : 2922–2926.
Chivers *et al.* 2012	**Chivers, L. S., Lundy, M.G., Colhoun, K., Newton, S. F., & Reid, N.** 2012. Diet of Black-legged Kittiwakes (*Rissa tridactyla*) feeding chicks at two Irish colonies highlights the importance of clupeids. *Bird Study* **59** : 363–367.
Chmura *et al.* 2018	**Chmura, H.E., Krause, J.S., Pérez, J.H., Asmus, A., Sweet, S.K., Hunt, K.E., Meddle, S.L., McElreath, R., Boelman, N.T., Gough, L. & Wingfield, J.C.** 2018. Late-season snowfall is associated with decreased offspring survival in two migratory arctic-breeding songbird species. *Journal of avian biology* **49** : 01712.
Christensen-Dalsgaard *et al.* 2018	**Christensen-Dalsgaard, S., May, R.F., Barrett, R.T., Langset, M., Sandercock, B.K. & Lorentsen, S.H.** 2018. Prevailing weather conditions and diet composition affect chick growth and survival in the black-legged kittiwake. *Marine Ecology Progress Series* **604** : 237–249.
Christensen-Dalsgaard *et al.* 2022	**Christensen-Dalsgaard, S., Ytrehus, B., Langset, M., Wiig, J.R. & Bærum, K.M.** 2022. Seabird beachcast events associated with bycatch in the Norwegian purse seine fishery. *Marine Environmental Research* **177** : 105625.
Church *et al.* 2018	**Church, C.E., Furness, R.W., Tyler, G., Gilbert, L. & Votier, S.C.** 2018. Change in the North Sea ecosystem from the 1970s to the 2010s: great skua diets reflect changing forage fish, seabirds, and fisheries. *ICES Journal of Marine Science* **76** : 925–937
Clairbaux *et al.* 2021	**Clairbaux, M., Mathewson, P., Porter, W., Fort, J., Strøm, H., Moe, B., Fauchald, P., Descamps, S., Helgason, H., Bråthen, V., Merkel, B., Anker-Nilssen, T., Bringsvor, I., Chastel, O., Christensen-Dalsgaard, S., Danielsen, J., Daunt, F., Dehnhard, N., Erikstad, K. & Grémillet, D.** 2021. North Atlantic winter cyclones starve seabirds. *Current Biology* **31** : 3964–3971.
Clewley *et al.* 2021	**Clewley, G.D., Barber, L.J., Conway, G.J., Clark, N.A., Donato, B.J., Thaxter, C.B. & Burton, N.H.K.** 2021. Foraging habitat selection by breeding Herring Gulls (*Larus argentatus*) from a declining coastal colony in the United Kingdom. *Estuarine, Coastal and Shelf Science* **261** : 107564.
Coffey & Verspoor 2025	**Coffey, P. & Verspoor, R.S.** 2025. Case study of the impact of an outbreak of high pathogenicity avian influenza (HPAI) on a seabird colony in Flintshire, Wales, United Kingdom. *Bird Study* **72**: 89–97.
Collard *et al.* 2022	**Collard, F., Bangjord, G., Herzke, D. & Gabrielsen, G.W.** 2022. Plastic burdens in northern fulmars from Svalbard: Looking back 25 years. *Marine Pollution Bulletin* **185** : 114333.
Collins *et al.* 2022	**Collins, S.M., Hedd, A., Fifield, D.A., Wilson, D.R. & Montevecchi, W.A.** 2022. Foraging paths of breeding Leach's Storm-Petrels in relation to offshore oil platforms, breeding stage, and year. *Frontiers in Marine Science* **9** : 816659.
Cook *et al.* 2014	**Cook, A.S.C.P., Dadam, D., Mitchell, I., Ross-Smith, V.H. & Robinson, R.A**. 2014 Indicators of seabird reproductive performance demonstrate the impact of commercial fisheries on seabird populations in the North Sea. *Ecological Indicators* **38** : 1–11
Cook *et al.* 2019	**Cook, A.S.C.P., Humphreys, E.M., Robinson, R.A. & Burton, N.H.K.** 2019. *Review of the potential of seabird colony monitoring to inform monitoring programmes for consented offshore wind farm projects.* British Trust for Ornithology, Thetford, Norfolk
Cook & Robinson 2010	**Cook, A.S.C.P & Robinson, R.A.** 2010. *How representative is the current monitoring of breeding seabirds in the UK?* BTO Research Report No. 573, British Trust for Ornithology, Thetford.
Cook-Haley & Millenbah 2002	**Cook-Haley, B.S. & Millenbah, B.F.** 2002. Impacts of vegetative manipulations on common tern nest success at Lime Island, Michagan. *Journal of Field Ornithology* **73** : 174–179.
Cordes *et al.* 2015	**Cordes, L.S., Hedworth, H.E., Cabot, D., Cassidy, M. & Thompson, P.M.** 2015. Parallel declines in survival of adult Northern Fulmars *Fulmarus glacialis* at colonies in Scotland and Ireland. *Ibis* **157** : 631–636.
Cormier *et al.* 2024	**Cormier, T.L., Barychka, T., Beaumont, M., Burt, T.V., English, M.D., Giacinti, J.A., Giroux, J.-F., Guillemette, M., Hargan, K.E., Jones, M., Lair, S., Lang, A.S., Lepage, C., Montevecchi, W.A., Rahman, I., Rail, J.-F., Robertson, G.J., Ronconi, R.A., Seyer, Y., Taylor, L.U., Ward, C.R.E., Wight, J., Wilhelm, S.I. & Avery-Gomm, S.** 2024. Seabird and sea duck mortalities were lower during the second breeding season in eastern Canada following the introduction of highly pathogenic avian influenza A H5Nx viruses. *Bird Study* **71**: 367–379
Costa *et al.* 2018	**Costa, R.A., Pereira, A.T., Costa, E., Henriques, A.C., Miodonski, J., Ferreira, M., Vingada, J.V. & Eira, C.** 2019. Razorbill *Alca torda* mortality in the Portuguese west coast. *European Journal of Wildlife Research* **65** : 1–7.
Coulson 2019	**Coulson, J.C.** 2019. *Gulls*. Harper Collins Publisher, London.
Coulson 2015	**Coulson, J.C.** 2015. Re-Evaluation of the Role of Landfills and Culling in the Historic Changes in the Herring Gull (*Larus argentatus*) Population in Great Britain. *Waterbirds* **38** : 339–354.
Coulson 2011	**Coulson, J.C.** 2011. *The Kittiwake.* T. and A.D. Poyser, London.
Craik 1995	**Craik, J.C.A.** 1995. Effects of North American Mink Mustela vison on the breeding success of terns and smaller gulls in west Scotland. *Seabird Journal* **17** : 3–11.
Craik 1997	**Craik, J.C.A.** 1997. Long-term effects of North American Mink *Mustela vison* on seabirds in western Scotland. *Bird Study* **44** : 303–309.
Craik 2015	**Craik, J.C.A.** 2015. *Results of the mink-seabird project in 2014.* Unpublished. Scottish Association for Marine Science Report, Oban.
Cramp 1985	**Cramp, S.** (ed.) 1985. *Handbook of the birds of Europe, the Middle East and North Africa: the Bird of the Western Palearctic. Volumne 3: Waders to Gulls.* Oxford University Press, Oxford.
Crespin *et al.* 2006	**Crespin, L., Harris, M., Lebreton, J-D. & Wanless, S.** 2006. Increased adult mortality and reduced breeding success with age in a population of Common Guillemot *Uria aalge* using marked birds of unknown age. *Journal of Avian Biology* **37** : 273–282.

References (D–F)

In text ref.	Reference
Cuthbert *et al.* 2003	**Cuthbert, F., Wires, L & Timmerman, K.** 2003. *Status assessment and conservation recommendations for the Common Tern sterna hirundo in the Great Lakes region*. U.S. Department of the Interior, Fish and Wildlife Service, Ft. Snelling, Minnesota, USA.
d'Entremont *et al.* 2022	**d'Entremont, K.J., Guzzwell, L.M., Wilhelm, S.I., Friesen, V.L., Davoren, G.K., Walsh, C.J. & Montevecchi, W.A.** 2022. Northern Gannets (*Morus bassanus*) breeding at their southern limit struggle with prey shortages as a result of warming waters. *ICES Journal of Marine Science* **79** : 50–60.
Dalrymple 2023	**Dalrymple, S.A.** 2023. Predator exclusion fencing improves productivity at a mixed colony of Herring Gulls *Larus argentatus*, Lesser Black-backed Gulls *L. fuscus* and Great Black-backed Gulls *L. marinus*. *Seabird Journal* **35** : 18–29.
Daly *et al.* 2020	**Daly, D., Daly, L. & Murray, M.** 2020. *Lady's Island Lake Tern Report 2019.* Department of Culture, Heritage and the Gaeltacht, Dublin.
Daly *et al.* 2016	**Daly, D., Murphy, B. & Murray, T.** 2016. *Lady's Island Lake Tern Report 2016*. Department of Arts, Heritage, Regional, Rural and Gaeltacht Affairs. Available at: https://www.npws.ie/sites/default/files/publications/pdf/Lady%27s%20Island%20Tern%20Report%202016_0.pdf [Accessed 25/10/2024].
Darby *et al.* 2021	**Darby, J.H., De Grissac, S., Arneill, G.E., Pirotta, E., Waggitt, J.J., Börger, L., Shepard, E., Cabot, D., Owen, E., Bolton, M. & Edwards, E.W.J.** 2021. Foraging distribution of breeding northern fulmars is predicted by commercial fisheries. *Marine Ecology Progress Series* **679** : 181–194
Daunt & Mitchell 2013	**Daunt, F. & Mitchell, I.** 2013. *Impacts of climate change on seabirds*. Marine Climate Change Impacts Partnership: Science Review 125–133.
Daunt *et al.* 2002	**Daunt, F., Benvenuti, S., Harris, M., Dall'Antonia, L., Elston, D.A. & Wanless, S.** 2002. Foraging strategies of the Black-legged Kittiwake *Rissa tridactyla* at a North Sea colony: evidence for a maximum foraging range. *Marine Ecology Progress Series* **245** : 239–247.
Daunt *et al.* 2008	**Daunt, F., Wanless, S., Greenstreet, S.P.R., Jensen, H., Hamer, K.C. & Harris, M.P.** 2008. The impact of the sandeel fishery closure in the north-western North Sea on seabird food consumption, distribution and productivity. *Canadian Journal of Fish Aquatic Science* **65** : 362–381.
Davis *et al.* 2018	**Davis, S., Brown, A., Lock, L., Sharps, E., Bolton, M. & Wilson, L.** 2018. *Productivity of Herring Gulls Larus argentatus and Lesser Black-backed Gulls L. fuscus in relation to fox predation risk at colonies across northern England and Wales in 2012.* RSPB Centre for Conservation Science, Bedfordshire.
Dawson *et al.* 2011	**Dawson, N.M., Macleod, C.D., Smith, M. & Ratcliffe, N.** 2011. Interactions with Great Skuas *Stercorarius skua* as a factor in the long-term decline of an Arctic Skua *Stercorarius parasiticus* population. *Ibis* **153** : 143–153.
de León *et al.* 2006	**de León, A., Mínguez, E., Harvey, P., Meek, E., Crane, J.E. & Furness, R.W.** 2006. Factors affecting breeding distribution of Storm-petrels *Hydrobates pelagicus* in Orkney and Shetland. *Bird Study* **53** : 64–72.
de León *et al.* 2023	DeLeon, E.E., Hook, M.W., Small, M.F. & Tegeler, A.K. 2023. Comparing and combining use of autonomous recording units and traditional counts to monitor Northern Bobwhite. *Journal of Field Ornithology* **94**.
Deakin *et al.* 2018	**Deakin, Z., Gilbert, L., Prior, G. & Bolton, M.** 2019. Assessment of Great Skua *Stercorarius skua* pellet composition to inform estimates of storm petrel consumption from bioenergetics models. *Seabird Journal* **31** : 36–48.
Deakin *et al.* 2021	**Deakin, Z., Hansen, E., Luxmoore, R., Thomas, R., Wood, M., Padget, O,. Medeiros, R., Aitchison, R., Ausden, M., Barnard, R., Booth, V., Hansen, B., Hansen, E., Hey, J., Hilmarsson, J., Hoyer, P., Kirby, W., Luxmoore, A., Mcdevitt, A. & Bolton, M.** 2022. Decline of Leach's Storm Petrels *Hydrobates leucorhous* at the largest colonies in the northeast Atlantic. *Seabird Journal* **33** : 74–106.
Deakin *et al.* 2022	**Deakin, Z., Cook, A., Daunt, F., McCluskie, A., Morley, N., Witcutt, E., Wright, L. & Bolton, M.** 2022. *A review to inform the assessment of the risk of collision and displacement in petrels and shearwaters from offshore wind developments in Scotland.* Scottish Government Report.
Defra 2025	**Defra (Department for Environment Food and Rural Affairs)**. 2025. Wild bird populations in the UK, 1970 to 2024. Department for Environment Food and Rural Affairs. Available at: https://www.gov.uk/government/statistics/wild-bird-populations-in-the-uk/wild-bird-populations-in-the-uk-and-england-1970-to-2024#breeding-seabird-populations-in-the-uk-and-england [Accessed: 28/05/2025]
Dehnhard *et al.* 2023	**Dehnhard, N., Anker-Nilssen, T., Johnston, D., Masden, E.A., Lorentsen, S.H. & Christensen-Dalsgaard, S.** 2023. Foraging behaviour of black guillemots at three Norwegian sites during the breeding season. *Marine Biology* **170** : 1–18.
del Hoyo *et al.* 1992	**del Hoyo, J., Elliot, A. & Sargatal, J.** 1992. *Handbook of the Birds of the World, Vol. 1: Ostrich to Ducks.* Lynx Edicions, Barcelona, Spain.
del Hoyo *et al.* 1996	**del Hoyo, J., Elliott, A. & Sargatal, J.** 1996. *Handbook of the Birds of the World, vol. 3: Hoatzin to Auks*. Lynx Edicions, Barcelona, Spain.
del Hoyo *et al.* 2018	**del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E.** 2018. *Handbook of the Birds of the World Alive*. Lynx Editions. Barcelona. Available at: https://www.hbw.com/node/54016. [Accessed: 25/10/2024].
d'Entremont *et al.* 2020	**d'Entremont, K.J.N., Zitske, L.M., Gladwell, A.J., Elliott, N.K., Mauck, R.A. & Ronconi, R.A.** 2020. Breeding population decline and associations with nest site use of Leach's Storm-Petrels on Kent Island, New Brunswick from 2001 to 2018. *Avian Conservation and Ecology* **15** : 11.
Descamps *et al.* 2011	**Descamps, S., Béchet, Descombes, X., Arnaud, A. & Zerubia, J.** 2011. An automatic counter for aerial images of aggregations of large birds. *Bird Study* **58** : 302–308.
Descamps *et al.* 2019	**Descamps, S., Ramírez, F., Benjaminsen, S., Anker-Nilssen, T., Barrett, R.T., Burr, Z., Christensen-Dalsgaard, S., Erikstad, K.E., Irons, D.B., Lorentsen, S.H. & Mallory, M.L.** 2019. Diverging phenological responses of Arctic seabirds to an earlier spring. *Global Change Biology* **25** : 4081–4091.

In text ref.	Reference
Dias *et al.* 2019	**Dias, M.P., Martin, R., Pearmain, E.J., Burfield, I.J., Small, C., Phillips, R.A., Yates, O., Lascelles, B., Borboroglu, P.G. & Croxall, J.P.** 2019. Threats to seabirds: a global assessment. *Biological Conservation* **237** : 525–537.
Dierschke *et al.* 2016	**Dierschke, V., Furness, R.W. & Garthe, S.** 2016. Seabirds and offshore wind farms in European waters: Avoidance and attraction. *Biological Conservation* **202** : 59–68.
Donehower *et al.* 2007	**Donehower, C.E., Bird, D.M., Hall, C.S. & Kress, S.W.** 2007. Effects of gull predation and predator control on tern nesting success at Eastern Egg Rock, Maine. *Waterbirds* **30** : 29–39.
Drewitt & Langston 2006	**Drewitt, A.L. & Langston, R.H.W.** 2006. Assessing the impacts of wind farms on birds. *Ibis* **148** : 29–42.
Drury 1973	**Drury, W.H.** 1973. Population changes in New England seabirds. *Bird Banding* **44** : 267–313.
Dunlop *et al.* 1991	**Dunlop, C.L., Blokpoel, H. & Jarvie, S.** 1991. Nesting rafts as a management tool for a declining Common Tern (*Sterna hirundo*) colony. *Colonial Waterbirds* **14** : 116–120.
Dunn *et al.* 2021	**Dunn, M.J., Adlard, S., Taylor, A.P., Wood, A.G., Trathan, P.N. & Ratcliffe, N.** 2021. Un-crewed aerial vehicle population survey of three sympatrically breeding seabird species at Signy Island, South Orkney Islands. *Polar Biology* **44** : 717–727.
Dunn *et al.* 2019	**Dunn, D.C., Harrison, A.L., Curtice, C., DeLand, S., Donnelly, B., Fujioka, E.I., Heywood, E., Kot, C.Y., Poulin, S., Whitten, M. & Åkesson, S.** 2019. The importance of migratory connectivity for global ocean policy. *Proceedings of the Royal Society B* **286** : 20191472.
Durinck *et al.* 1994	**Durinck, J., Skov, H., Jensen, F.P. & Pihl, S.** 1994. Important Marine Areas for Wintering Birds in the Baltic Sea. *Colonial Waterbirds* **19** : 157.
Eaton *et al.* 2023	**Eaton, M. & the Rare Breeding Birds Panel.** 2023. Rare breeding birds in the UK in 2021. *British Birds* **116** : 609–684.
Eaton *et al.* 2021	**Eaton, M. & the Rare Breeding Birds Panel.** 2021. Rare breeding birds in the UK in 2019. *British Birds* **114** : 646–704.
Edney *et al.* 2024	**Edney, A.J., Danielsen, J., Descamps, S., Jónsson, J.E., Owen, E., Merkel, F., Stefánsson, R.A., Wood, M.J., Jessopp, M.J. & Hart, T.** 2025. Using citizen science image analysis to measure seabird phenology. *Ibis* **167** : 56–72.
Edney *et al.* 2023	**Edney, A., Hart, T., Jessopp, M., Banks, A., Clarke, L., Cugnière, L., Elliot, K., Martinez, I.J., Kilcoyne, A., Murphy, M., Nager, R., Ratcliffe, N., Thompson, D., Ward, R. & Wood, M.** 2023. Best practices for using drones in seabird monitoring and research. *Marine Ornithology* **51** : 265–280.
Edney & Wood 2021	**Edney, A.J. & Wood, M.J.** 2021. Applications of digital imaging and analysis in seabird monitoring and research. *Ibis* **163** : 317–337.
Edwards *et al.* 2013	**Edwards, E.W., Quinn, L.R., Wakefield, E.D., Miller, P.I. & Thompson, P.M.** 2013. Tracking a northern fulmar from a Scottish nesting site to the Charlie-Gibbs Fracture Zone: Evidence of linkage between coastal breeding seabirds and Mid-Atlantic Ridge feeding sites. *Deep Sea Research Part II: Topical Studies in Oceanography* **98** : 438–444.
Eerkes-Medrano *et al.* 2017	**Eerkes-Medrano, D., Fryer, R.J., Cook, K.B. & Wright, P.J.** 2017. Are simple environmental indicators of food web dynamics reliable: Exploring the kittiwake-temperature relationship. *Ecological Indicators* **75**: 36–47
EFSA *et al.* 2022	**EFSA (European Food Safety Authority), ECDC (European Centre for Disease Prevention and Control), EURL (European Reference Laboratory for Avian Influenza)., Adlhoch, C., Fusaro, A., Gonzales, J.L., Kuiken, T., Marangon, S., Niqueux, É., Staubach, C., Terregino, C., Guajardo, I.M., Chuzhakina K. & Baldinelli F.** 2022. Avian influenza overview June–September 2022. *EFSA Journal* **20** : e07597.
EJF 2020	**EJF (Environmental Justice Foundation).** 2020. *The people's fishery on the brink of collapse. Small pelagics in landings of Ghana's industrial trawl fleet.* Environmental Justice Foundation, London. Available at: https://ejfoundation.org/resources/downloads/EJF-report-small-pelagics-2020-final.pdf [Accessed: 25/10/2024].
Erikstad *et al.* 2013	**Erikstad, K.E., Reiertsen, T.K., Barrett, R.T., Vikebø, F. & Sandvik, H.** 2013. Seabird fish interactions: the fall and rise of a common guillemot *Uria aalge* population. *Marine Ecology Progress Series* **475** : 267–276.
Esmonde *et al.* 2022	**Esmonde, N.P., Hanna, R.E., Patel, J.G., Smyth, V.J., Caplat, P., Smyth, W., Jaggers, P., Padget, O., Guilford, T., Perrins, C. & Reid, N.** 2022. Case Report of *Puffinosis* in a Manx Shearwater (*Puffinus puffinus*) Suggesting Environmental Aetiology. *Animals* **12** : 3457.
2009/147/EC	**European Union, Directive 2009/147/EC**. 2009. *Directive of the European Parliament and the Council on the conservation of wild birds*. Official Journal of the European Communities L20: 7–25.
Evans *et al.* 2009	**Evans, R., Wilson, J.D., Amar, A., Douse, A., MacLennan, A., Ratcliffe, N. & Whitfield, D.P.** 2009. Growth and demography of a re-introduced population of White-tailed Eagles *Haliaeetus albicilla*. *Ibis* **151** : 244–254.
Ewins 1985	**Ewins, P.J.** 1985. Growth, diet and mortality of Arctic tern *Sterna paradisaea* chicks in Shetland. *Seabird Journal* **8** : 59–68.
Ewins 1990	**Ewins, P.J.** 1990. The diet of Black Guillemots *Cepphus grylle* in Shetland. *Holarctic Ecology* **13** : 90–97.
Falchieri *et al.* 2022	**Falchieri, M., Reid, S.M., Ross, C.S., James, J., Byrne, A.M., Zamfir, M., Brown, I.H., Banyard, A.C., Tyler, G., Philip, E. & Miles, W.** 2022. Shift in HPAI infection dynamics causes significant losses in seabird populations across Great Britain. *Veterinary Record* **191** : 294–296.
Fasola & Canova 1996	**Fasola, M., Canova, L.** 1996. Conservation of gull and tern colony sites in north-eastern Italy, an internationally important bird area. Colonial Waterbirds, **19** : 59-67.
Fayet *et al.* 2021	**Fayet, A.L., Clucas, G.V., Anker-Nilssen, T., Syposz, M. & Hansen, E.S.** 2021. Local prey shortages drive foraging costs and breeding success in a declining seabird, the Atlantic puffin. *Journal of Animal Ecology* **90** : 1152–1164.
Fayet *et al.* 2020	**Fayet, A.L., Shannon, P., Lyons, D.E. & Kress, S.W.** 2020. Manx shearwaters *Puffinus puffinus* breeding in the western Atlantic follow a different migration route from their eastern Atlantic conspecifics. *Marine Ornithology* **48** : 179–183
Fayet *et al.* 2017	**Fayet, A.L., Freeman, R., Anker-Nilssen, T., Diamond, A., Erikstad, K.E., Fifield, D., Fitzsimmons, M.G., Hansen, E.S., Harris, M.P., Jessopp, M., Kouwenberg, A.L., Kress, S., Mowat, S., Perrins, C.M., Petersen, A., Petersen, I.K., Reiertsen, T.K., Robertson, G.J., Shannon, P., Sigurðsson, I.A., Shoji, A., Wanless, S. & Guilford, T.** 2017. Ocean-wide drivers of migration strategies and their influence on population breeding performance in a declining seabird. *Currrent Biology* **27** : 3871–3878.
Field *et al.* 2012	**Field, C.B., Barros, V., Stocker, T.F. & Dahe, Q. (eds.)** 2012. *Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation: Special Report of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge.
Fisher 1952	**Fisher, J.** 1952. *The Northern Fulmar.* Collins, London.

References (F–H)

In text ref.	Reference
Finney *et al.* 2001	**Finney, S., Wanless, S., Harris, M. & Monaghan, P.** 2001. The impact of gulls on puffin reproductive performance: An experimental test of two management strategies. *Biological Conservation* **98** : 159–165.
Flint *et al.* 1984	**Flint, V.E., Boehme, R.L., Kostin, Y.V., Kuznetsov, A.A.** 1984. *A field guide to birds of the USSR.* Princeton University Press, Princeton, New Jersey.
Foster *et al.* 2017	**Foster, S., Swann, R.L. & Furness, R.W.** 2017. Can changes in fishery landings explain long-term population trends in gulls? *Bird Study* **64** : 90-97.
Frederiksen *et al.* 2008	**Frederiksen, M., Daunt, F., Harris, M.P. & Wanless, S.** 2008. The demographic impact of extreme events: stochastic weather drives survival and population dynamics in a long-lived seabird. *Journal of Animal Ecology* **77** : 1020-1029.
Frederiksen *et al.* 2012	**Frederiksen, M., Moe, B., Daunt, F., Phillips, R.A., Barrett, R.T., Bogdanova, M.I., Boulinier, T., Chardine, J.W., Chastel, O., Chivers, L.S., Christensen-Dalsgaard, S., Clément-Chastel, C., Colhoun, K., Freeman, R., Gaston, A.J., González-Solís, J., Goutte, A., Grémillet, D., Guilford, T., Jensen, G.H., Krasnov, Y., Lorentsen, S.-H., Mallory, M.L., Newell, M., Olsen, B., Shaw, D., Steen, H., Strøm, H., Systad, G.H., Thórarinsson, T.L. & Anker-Nilssen, T.** 2012. Multi-colony tracking reveals the winter distribution of a pelagic seabird on an ocean basin scale. *Diversity and Distributions* **18** : 530–542.
Frederiksen *et al.* 2007	**Frederiksen, M., Edwards, M., Mavor, R.A. & Wanless, S.** 2007. Regional and annual variation in black-legged kittiwake breeding productivity is related to sea surface temperature. *Marine Ecology Progress Series* **350** : 137–143.
Frederiksen *et al.* 2004	**Frederiksen, M., Wanless, S., Harris, M. P., Rothery, P. & Wilson, L.J.** 2004. The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. *Journal of Applied Ecology* **41** : 1129–1139.
Frederiksen *et al.* 2025	**Frederiksen, M., Layton-Matthews, K., Bennett, S., Funder Castenschiold, J.H., Cruz-Flores, M., Edney, A.J., Fauchald, P., Franklin, K.A., Guímaro, H.R., Hereward, H.F.R., Johnston, D.T., Merkel, B., Molværsmyr, S., Sauser, C., Snell, K.R.S. & Humphreys, E.M** 2025. Opportunities and challenges for new technologies in seabird population monitoring. ICES Journal of Marine Science 82: fsaf115
Fretwell *et al.* 2012	**Fretwell, P.T., LaRue, M.A., Morin, P., Kooyman, G.L., Wienecke, B., Porter, C., Fox, A., Fleming, A., Ratcliffe, N., Morin, P., Trathan, P.** 2012. An Emperor Penguin Population Estimate: The First Global, Synoptic Survey of a Species from Space. *PLoS ONE* **7** : e33751.
Fretwell *et al.* 2017	**Fretwell, P.T., Scofield, P. & Phillips, R.A.** 2017. Using super-high resolution satellite imagery to census threatened albatrosses. *Ibis* **159** : 481–490.
Fretwell & Trathan 2021	**Fretwell, P.T. & Trathan, P.N.** 2021. Discovery of new colonies by Sentinel2 reveals good and bad news for emperor penguins. *Remote Sensing in Ecology and Conservation* **7** : 139–153.
Freeman *et al.* 2007	**Freeman, S.N., Noble, D.G., Newson, S.E. & Baillie, S.R.** 2007. Modelling population changes using data from different surveys: the Common Birds Census and the Breeding Bird Survey. *Bird Study* 54 : 61–72.
Furness 1982	**Furness, R.W.** 1982. *Population, breeding biology and diets of seabirds on Foula in 1980.* Seabird Report 1977–1981. The Seabird Group.
Furness 1987	**Furness, R.W.** 1987. *The Skuas.* T. and A.D. Poyser, Calton.
Furness *et al.* 1992	**Furness, R.W., Ensor, K. & Hudson, A.V.** 1992. The use of fishery waste by gull populations around the British Isles. *Ardea* **80** : 105–113.
Furness & Tasker 2000	**Furness, R.W. & Tasker, M.L.** 2000. Seabird-fishery interactions: quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. *Marine Ecology Progress Series* **202 :** 253–264.
Furness *et al.* 2013	**Furness, R.W., Wade, H.M. & Masden, E.A.** 2013. Assessing vulnerability of marine bird populations to offshore wind farms. *Journal of environmental management* **119** : 56–66.
Furness *et al.* 2012	**Furness, R.W., Wade, H.M., Robbins, A.M.C. & Masden, E.A.** 2012. Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. *ICES Journal of Marine Science* **69** : 1466–1479.
Gaston & Jones 1998	**Gaston, A.J. & Jones, I.L.** 1998. *The Auks: Alcidae.* Oxford University Press, Oxford.
Garthe *et al.* 2000	**Garthe, S., Benvenuti, S. & Montevecchi, W.A.** 2000. Pursuit plunging by northern gannets (*Sula bassana*) feeding on capelin (*Mallotus villosus*). *Proceedings of the Royal Society of London Series B: Biological Sciences*, **267** : 1717–1722.
Garthe & Flore 2007	**Garthe, S. & Flore, B.-O.** 2007. Population trend over 100 years and conservation needs of breeding Sandwich Terns (*Sterna sandvicensis*) on the German North Sea coast. *Journal of Ornithology* **148** : 215–227.
Garthe & Furness 2001	**Garthe, S. & Furness, R.W.** 2001. Frequent shallow diving by a Northern Fulmar feeding at Shetland. *Waterbirds* **24** : 287–289.
Gaston & Woo 2008	**Gaston, A.J. & Woo, K.** 2008. Razorbills (*Alca torda*) follow subarctic Prey into the Canadian Arctic: Colonization Results from Climate Change? *The Auk* **125** : 939–942.
Giagnoni *et al.* 2015	**Giagnoni, R., Conti, C.A., Canepa, P. & Nardelli, R.** 2015. First breeding records of Northern Gannet *Morus bassanus* in Italy. *Avocetta* **39** : 93–95.
Gilbert *et al.* 2021	**Gilbert, G., Stanbury, A & Lewis, L.** 2021. Birds of Conservation Concern in Ireland 2020–2026. *Irish Birds* **9** : 523–544.
Gilchrist 1999	**Gilchrist, H.G.** 1999. Declining Thick-billed Murre *Uria lomvia* colonies experience higher gull predation rates: an inter-colony comparison. *Biological Conservation* **87** : 21–29.
Gill *et al.* 2019	**Gill, L., McManus, A., Tiernan, E. & Newton, S.** 2019. *Rockabill Tern Report 2019.* BirdWatch Ireland Seabird Conservation Report, Kilcoole.
Gill *et al.* 2023	**Gill, F., Donsker, D. & Rasmussen, P.** (eds) 2023. *IOC World Bird List v13.1.* Available at: https://www.worldbirdnames.org/new/ioc-lists/master-list-2/ [Accessed: 25/10/2024]
Gochfield *et al.* 2018	**Gochfeld, M., Burger, J. & Garcia, E.F.J.** 2018. Sandwich Tern (*Thalasseus sandvicensis*). In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). 2018. *Handbook of the Birds of the World Alive*, pp. Lynx Edicions. Barcelona. https://www.hbw.com/node/54016.
Gorski *et al.* 1977	**Gorski, W., Jakuczun, B., Nitecki, C. & Petryna, A.** 1977. Investigation of oil pollution on the Polish Baltic coast in 1974–1975. *Przeglad Zoologiczny* **21** : 20–23.
Green 2017	**Green, E.** 2017. *Tern diet in the UK and Ireland: a review of key prey species and potential impacts of climate change.* RSPB, Royal Society for the Protection of Birds, Sandy.
Greenwood 2007	**Greenwood, J.** 2007. Earlier laying by black guillemots *Cepphus grylle* in response to increasing sea-surface temperature. *Bird Study* **54** 378–379.
Greenwood 2014	**Greenwood, J.** 2014. *A review of black guillemots breeding at Bangor, Co. Down, 1985–2013.* In: Leonard, K. & Wolsey, S. (eds). 2014. *Northern Ireland Seabird Report 2013.* British Trust for Ornithology, Thetford.
Grémillet *et al.* 2003	**Grémillet, D., Liu, H., Le Maho, Y. & Carss, D.N.** 2003. Great cormorants and freshwater fish stocks: a pragmatic approach to an ecological issue. *Cormoran* **13** : 131–136.
Grémillet *et al.* 2015	**Grémillet, D., Péron, C., Provost, P. & Lescroel, A.** 2015. Adult and juvenile European seabirds at risk from marine plundering off West Africa. *Biological Conservation* **182** : 143–147.
Griffin & Peach 2024	**Griffin, L.R. & Peach, W.J.** 2024. Impacts of highly pathogenic avian influenza (HPAI) on a Barnacle Goose *Branta leucopsis* population wintering on the Solway Firth, UK. *Bird Study* **71**: 392–403.
Guildford *et al.* 2009	**Guilford, T., Meade, J., Willis, J., Phillips, R.A., Boyle, D., Roberts, S., Collett, M., Freeman, R. & Perrins, C.M.** 2009. Migration and stopover in a small pelagic seabird, the Manx shearwater *Puffinus puffinus*: insights from machine learning. *Proceedings of the Royal Society B: Biological Sciences* **276** : 1215–1223.
Guildford *et al.* 2018	**Guilford, T., Padget, O., Bond, S. & Syposz, M.** 2018. Light pollution causes object collisions during local nocturnal manoeuvring flight by adult Manx Shearwaters *Puffinus puffinus*. *Seabird Journal* **31** : 48–55.
Guillemette & Brousseau 2001	**Guillemette, M. & Brousseau, P.** 2001. Does culling of predatory gulls enhance the productivity of breeding common terns? *Journal of Applied Ecology* **38** 1–8.
Hagemeijer & Blair 1997	**Hagemeijer, E.J.M. & M.J. Blair** (eds). 1997. *The EBCC Atlas of European Breeding Birds: their distribution and abundance.* T & A.D. Poyser, London.
Hamer *et al.* 2001	**Hamer, K.C., Phillips, R.A., Hill, J.K., Wanless, S. & Wood, A.G.** 2001. Contrasting foraging strategies of Gannets *Morus bassanus* at two North Atlantic colonies. *Marine Ecology Progress Series* **224** : 283–290
Hammer *et al.* 2016	**Hammer, S., Nager, R.G., Johnson, P.C.D., Furness, R.W. & Provencher, J.F.** 2016. Plastic debris in great skua (*Stercorarius skua*) pellets corresponds to seabird prey species. *Marine Pollution Bulletin* **103** : 206–210.
Hario 2001	**Hario, M.** 2001. Chick growth and nest departure in Baltic Black Guillemots *Cepphus grylle*. *Ornis Fennica* **78 :** 97–108. Available at: https://lintulehti.birdlife.fi:8443/pdf/artikkelit/342/tiedosto/of_78_97-108_artikkelit_342.pdf. [Accessed: 24.10.2024]
Hario *et al.* 2000	**Hario, M., Himberg, K., Hollmén, T. & Rudbäck, E.** 2000. Polychlorinated biphenyls in diseased lesser black-backed gull (*Larus fuscus fuscus*) chicks from the Gulf of Finland. *Environmental Pollution* **107** : 53–60.
Harris & Riddiford 1989	**Harris, M.P. & Riddiford, N.J.** 1989. The food of some young seabirds on Fair Isle in 1986–88. *Scottish Birds* **15** : 119–125.
Harris & Wanless 1991	**Harris, M.P. & Wanless, S.** 1991. The importance of the lesser sandeel *Ammodytes marinus* in the diet of the shag *Phalacrocorax aristotelis. Ornis Scandinavica* **22** : 375–382.
Harris & Wanless 1996	**Harris, M.P. & Wanless, S.** 1996. Differential responses of Guillemot *Uria aalge* and Shag *Phalacrocorax aristotelis* to a late winter wreck. *Bird Study* **43** : 220–230.
Harris & Wanless 2011	**Harris, M.P. & Wanless, S.** 2011. *The Puffin.* T. and A.D. Poyser, London.
Harris *et al.* 2005	**Harris, M.P., Anker-Nilssen, T., McCleery, R.H., Erikstad, K.E., Shaw, D.N. & Grosbois, V.** 2005. Effect of wintering area and climate on the survival of adult Atlantic Puffins *Fratercula arctica* in the eastern Atlantic. *Marine Ecology Progress Series* **297**: 283–296.
Harris *et al.* 2024	**Harris, S.J., Baker, H., Balmer, D.E., Bolton, M., Burton, N.H.K., Caulfield, E., Clarke, J.A.E., Dunn, T.E., Evans, T.J., Hereward, H.R.F., Humphreys, E.M., Money, S. and O'Hanlon, N.J.** 2024. *Seabird Population Trends and Causes of Change: 1986-2023, the annual report of the Seabird Monitoring Programme.* BTO Research Report 771. British Trust for Ornithology, Thetford.
Harris *et al.* 2022	**Harris, S.J., Massimino, D., Balmer, D.E., Eaton, M.A., Noble, D.G., Pearce-Higgins, J.W., Woodcock, P. and Gillings, S.** 2021. *The Breeding Bird Survey 2020.* BTO Research Report 736. British Trust for Ornithology, Thetford.
Harris *et al.* 2021	**Harris, S.J., Massimino, D., Balmer, D.E., Kelly, L., Noble, D.G., Pearce-Higgins, J.W., Woodcock, P., Wotton, S. & Gillings, S.** 2022. *The Breeding Bird Survey 2021.* BTO Research Report 745. British Trust for Ornithology, Thetford.
Harrison *et al.* 2022	**Harrison, A.L., Woodard, P.F., Mallory, M.L. & Rausch, J.** 2022. Sympatrically breeding congeneric seabirds (*Stercorarius spp.*) from Arctic Canada migrate to four oceans. *Ecology and Evolution* **12** : e8451.
Hayes *et al.* 2021	**Hayes, M.C., Gray, P.C., Harris, G., Sedgwick, W.C., Crawford, V.D., Chazal, N., Crofts, S., Johnston, D.W.** 2021. Drones and deep learning produce accurate and efficient monitoring of large-scale seabird colonies. *Ornithological Applications* **123**
Heaney *et al.* 2002	**Heaney, V., Ratcliffe, N., Brown, A., Robinson, P.J. & Lock, L.** 2002. The status and distribution of European Storm-petrels *Hydrobates pelagicus* and Manx Shearwaters *Puffinus puffinus* on the Isles of Scilly. *Atlantic Seabirds* **4** : 1–16.
Heath *et al.* 2009	**Heath, M., Edwards, M., Furness, R., Pinnegar, J. & Wanless, S.** 2009. A view from above: changing seas, seabirds and food sources. In: Baxter, J.M., Buckley, P.J. & Frost, M.T. (eds). 2009. Ecosystem Linkages Report Card.
Hedd *et al.* 2018	**Hedd, A., Pollet, I.L., Mauck, R.A., Burke, C.M., Mallory, M.L., McFarlane Tranquilla, L.A., Montevecchi, W.A., Robertson, G.J., Ronconi, R.A., Shutler, D., Wilhelm, S.I. & Buegess, N.M.** 2018. Foraging areas, offshore habitat use, and colony overlap by incubating Leach's storm-petrels *Oceanodroma leucorhoa* in the North-west Atlantic. *PLoS ONE* **13** : e0194389.
Hedd & Montevecchi 2006	**Hedd, A. & Montevecchi, W.A.** 2006. Diet and trophic position of Leach's storm-petrel *Oceanodroma leucorhoa* during breeding and moult, inferred from stable isotope analysis of feathers. *Marine Ecology Progress Series* **322** : 291–301.
Hentati-Sundberg *et al.* 2025	**Hentati-Sundberg, J., Berglund, P-A., Olin, A., Hejdström, A., Österblom, H., Carlsen, A., Queiros, Q. & Olsson, O.** 2025. Technological evolution generates new answers and new ways forward: a progress report from the first decade at the Karlsö Auk Lab. *Marine Ornithology* **53** : 21–33.

References (H–M)

In text ref.	Reference
Hentati-Sundberg *et al.* 2023	**Hentati-Sundberg, J., Olin, A.B., Reddy, S., Berglund, P.-A., Svensson, E., Reddy, M., Kasarareni, S., Carlsen, A.A., Hanes, M., Kad, S. & Olsson, O.** 2023. Seabird surveillance: combining CCTV and artificial intelligence for monitoring and research. *Remote Sensing in Ecology and Conservation* **9** : 568–581.
Heubeck 2002	**Heubeck, M.** 2002. The decline of Shetland's Kittiwake population. *British Birds* **95**: 118–122.
Heubeck 2009	**Heubeck, M.** 2009. Common guillemot *Uria aalge* chick diet and breeding performance at Sumburgh Head, Shetland in 2007–09, compared to 1990–91. *Seabird Journal* **22** : 9–18.
Heubeck *et al.* 2011	**Heubeck, M., Aarvak, T., Isaksen, K., Johnsen, A., Petersen, I.K. & Anker-Nilssen, T.** 2011. Mass mortality of adult Razorbills *Alca torda* in the Skagerrak and North Sea area, autumn 2007. *Seabird Journal* **24** : 11–32.
Heubeck *et al.* 1999	Heubeck, M., Mellor, R.M., Harvey, P.V., Mainwood, A.R. & Riddington, R. 1999. Estimating the population size and rate of decline of Kittiwakes *Rissa tridactyla* breeding in Shetland, 1981–97. *Bird Study* **46**: 48–61
Hey *et al.* 2020	**Hey, J., Hansen, E.S. & Bolton, M.** 2020. Gull Predation on Leach's *Hydrobates leucorhous* and European Storm-petrels *H. pelagicus* on Elliðaey Island, Iceland. *Seabird Journal* **32** : 59–71.
Higgins & Davies 1996	**Higgins, P.J. & Davies, S.J.J.F.** 1996. *Handbook of Australian, New Zealand and Antarctic birds vol 3: snipe to pigeons*. Oxford University Press, Oxford.
Hipfner *et al.* 2012	**Hipfner, J.M., Blight, L.K., Lowe, R.W., Wilhelm, S.I., Robertson, G.J., Barrett, R.T., Anker-Nilssen, T. & Good, T.P.** 2012. Unintended consequences: how the recovery of sea eagle *Haliaeetus spp*. populations in the northern hemisphere is affecting seabirds. *Marine Ornithology* **40** : 39–2.
Hjernquist *et al.* 2012	**Hjernquist, M.B., Hjernquist, M. & Hjernquist, B.** 2012. Behaviour and survival of Common Guillemot *Uria aalge* chicks at departure from a nest site in the Baltic Sea. *Ornis Svecica* **22 :** 3–8.
Horswill *et al.* 2022	**Horswill, C., Miller, J.A. & Wood, M.J.** 2022. Impact assessments of wind farms on seabird populations that overlook existing drivers of demographic change should be treated with caution. *Conservation Science and Practice* **4** : e12644.
Horswill & Robinson 2015	**Horswill, C & Robinson, R.** 2015. Review of seabird demographic rates and density dependence. Available at: https://data.jncc.gov.uk/data/897c2037-56d0-42c8-b828-02c0c9c12d13/JNCC-Report-552-REVISED-WEB.pdf [Accessed 31/07/2024]
Howells *et al.* 2018	**Howells, R.J., Burthe, S.J., Green, J.A., Harris, M.P., Newell, M.A., Butler, A., Wanless, S. & Daunt, F.** 2018. Pronounced long-term trends in year-round diet composition of the European shag *Phalacrocorax aristotelis. Marine Biology* **165** : 1–15.
Hughes *et al.* 2011	**Hughes, B., Martin, G. & Reynolds, S.** 2011. The use of Google Earth satellite imagery to detect the nests of Masked Boobies *Sula dactylatra*. *Wildlife Biology* **17** : 210–216.
Hunt & Heffernan 2007	**Hunt, J. & Heffernan, M.L.** 2007. *Mink trapping at Lough Mask 2007 to protect breeding gulls.* Unpublished report to the Heritage Council and the National Parks and Wildlife Service.
Hunton 2024	**Hunton, R.** 2024. *Beacon Ponds.* SMPnews **2** : 8–13. BTO. Available at: www.bto.org/smp-news [Accessed: 01/07/2024]
Hüppop & Wurm 2000	Hüppop, O. & Wurm, S. 2000. Effects of winter fishery activities on resting numbers, food and body condition of large gulls *Larus argentatus* and *L. marinus* in the south-eastern North Sea. *Marine Ecology Progress Series* **194**: 241–247
Indykiewicz 2015	**Indykiewicz, P.** 2015. Egg losses caused by cold snap in the Black-headed Gull, *Chroicocephalus ridibundus. Polish Journal of Ecology* **63** : 460–466.
Isaksson *et al.* 2019	**Isaksson, N., Evans, T. J., Olsson, O., & Åkesson, S.** 2019. Foraging behaviour of Razorbills *Alca torda* during chick-rearing at the largest colony in the Baltic Sea. *Bird Study* **66** : 11–21.
IUCN 2025	**IUCN. 2025.** *The IUCN Red List of Threatened Species. Version 2025–1*. Available at: www.iucnredlist.org. [Accessed: 09/07/2025].
James 1984	**James, P. C.** 1984. *The status and conservation of seabirds in the Mediterranean Sea.* In: Croxall, J.P., Evans, P.G.H. & Schreiber, R.W. (eds). 1984. Status and conservation of the world's seabirds. International Council for Bird Preservation, Cambridge, UK.
Jatta *et al.* 2025	**Jatta, D., Wille, M., Ballmann, M.Z., Youm, B.N., Camara, F., Dodman, T. & Fox, O.J.L.** 2025. Outbreak of high pathogenicity avian influenza (H5N1) in seabird populations of West Africa. *Bird Study* **72**: 110–120.
Jessop 2014	**Jessop, H.** *Seabird tragedy in the north-east Atlantic winter 2013/14.* Unpublished RSPB Report. Royal Society for the Protection of Birds, Sandy.
JNCC 2021	**JNCC.** 2021. *Seabird Population Trends and Causes of Change: 1986–2023 Report*. JNCC. Available at: www.jncc.gov.uk//out-work/ [Accessed 01/08/2023]
JNCC 2014	**JNCC.** 2014. *Methods of analysis for production of indices of abundance and estimation of productivity.* Available at: https://hub.jncc.gov.uk/assets/701c338f-ed54-43da-a61c-254cb79698b8 [Accessed 31/07/2024]
Johnston *et al.* 2020	**Johnston, D.T., Furness, R.W., Robbins, A., Tyler, G.A. & Masden, E.A.** 2020. Camera traps reveal predators of breeding Black Guillemots *Cepphus grylle. Seabird Journal* **32 :** 72–83.
Johnston *et al.* 2025a	**Johnston, D.T., Atkinson, P.W., Leech, E.I., Burton, N.H.K., Humphreys, E.M., Robinson, R.A., Blackburn, J.R., Blackburn, A.C., Brides, K., Boland, H., Burke, B., Daunt, F., Davies, J.C., Edwards, P.J., Furness, R.W., Holman, D., Redfern, C.P.F., Swann, R.L., Roper, P., Stansfield, S.D., Walsh, A.J. & Pearce-Higgins, J.W.** 2025. Using ring (band) recovery data to examine the impact of high pathogenicity avian influenza (HPAI) on wild bird populations. *Bird Study* **72**: 39–50.
Johnston *et al.* 2025b	**Johnston, D.T., Masden, E.A., Booth Jones, K.A. & Humphreys, E.M.** 2025b. Combining temperature-depth recorder and GPS data improves identification of foraging habitat in diving seabirds. *Marine Biology* **172**.
Johnston *et al.* 2025c	**Johnston, D.T., Humphreys, E.M., Davies, J.G., Evans, T., Howells, R.J. & Pearce-Higgins, J.W.** 2025. Current understanding of how climate change affects seabirds varies between regions and species in the North-East Atlantic. *Marine Ecology Progress Series* **755** : 163–177.
Johnston *et al.* 2021	**Johnston, D.T., Humphreys, E.M., Davies, J.G. & Pearce-Higgins, J.W.** 2021. *Review of climate change mechanisms affecting seabirds within the INTERREG VA area.* MarPAMM project commissioned report.
Keller *et al.* 2020	**Keller, V., Herrando, S., Voříšek, P., Franch, M., Kipson, M., Milanesi, P., Martí, D., Anton, M., Klvanova, A., Kalyakin, M.V., Bauer, H.G. & Foppen, R.P.B.** 2020. *European Breeding Bird Atlas 2: Distribution, Abundance and Change.* Lynx Edicions, Barcelona, Spain.
Kenyon & Cooper 2005	**Kenyon, N. & Cooper, B.** 2005. *Sand banks, sand transport and offshore wind farms. DTI SEA 6 Technical Report.* Department of Trade and Industry, London.
Kersten *et al.* 2021	**Kersten, O., Star, B., Leigh, D.M., Anker-Nilssen, T., Strøm, H., Danielsen, J., Descamps, S., Erikstad, K.E., Fitzsimmons, M.G., Fort, J., Hansen, E.S., Harris, M.P., Irestedt, M., Kleven, O., Mallory, M.L., Jakobsen, K.S. & Boessenkool, S.** 2020. Complex population structure of the Atlantic puffin revealed by whole genome analyses. *Communications Biology* **4** : 922.
Kim & Monaghan 2006	**Kim, S.Y. & Monaghan, P.** 2006. Interspecific differences in foraging preferences, breeding performance and demography in herring (*Larus argentatus*) and lesser black-backed gulls (*Larus fuscus*) at a mixed colony. *Journal of Zoology* **270** : 664–671.
Klaassen *et al.* 2012	**Klaassen, R.H., Ens, B.J., Shamoun-Baranes, J., Exo, K.M. & Bairlein, F.** 2012. Migration strategy of a flight generalist, the Lesser Black-backed Gull *Larus fuscus*. *Behavioral Ecology* **23** : 58–68.
Knief *et al.* 2024	**Knief, U., Bregnballe, T., Alfarwi, I., Ballmann, M., Brenninkmeijer, A., Bzoma, S., Chabrolle, A., Dimmlich, J., Engel, E., Fijn, R., Fischer, K., Hälterlein, B., Haupt, M., Herrmann, C., Veld, R., Kirchhoff, E., Kristersson, M., Kühn, S., Larsson, K. & Courtens, W.** 2024. Highly pathogenic avian influenza causes mass mortality in Sandwich Tern *Thalasseus sandvicensis* breeding colonies across north-western Europe. *Bird Conservation International* **34** : e6.
Kubetzki & Garthe 2003	**Kubetzki, U. & Garthe, S.** 2003. Distribution, diet and habitat selection by four sympatrically breeding gull species in the south-eastern North Sea. *Marine Biology* **143** : 199–207.
Kubetzki *et al.* 2009	**Kubetzki, U., Garthe, S., Fifield, D., Mendel, B. & Furness, R.W.** 2009. Individual migratory schedules and wintering areas of northern gannets. *Marine Ecology Progress Series* **391** : 257–265.
Kühn *et al.* 2020	**Kühn, S., Booth, A.M., Sørensen, L., Van Oyen, A. & van Franeker, J.A.** 2020. Transfer of additive chemicals from marine plastic debris to the stomach oil of northern fulmars. *Frontiers in Environmental Science* **8** : 138.
Lambert *et al.* 2015	**Lambert, M., Carlisle, S. & Cain I.** 2015. *The role of brown rat (Rattus norvegicus) predation in determining breeding success of Manx shearwaters (Puffinus puffinus) on Rum.* Scottish Natural Heritage Commissioned Report, No. 697.
Lane *et al.* 2023	**Lane, J.V., Jana, Avery-Gomm, S., Elmar Ballstaedt, Banyard, A.C., Tatsiana Barychka, Brown, I.H., Brugger, B., Burt, T.V., Careen, N., Johan, Christensen-Dalsgaard, S., Clifford, S., Collins, S.M., Emma, Danielsen, J., Daunt, F., Kyle, Doiron, P. & Duffy, S.** 2023. High pathogenicity avian influenza (H5N1) in Northern Gannets: Global spread, clinical signs, and demographic consequences. *Ibis* **166** : 633–650.
Langley *et al.* 2022	**Langley, L.P., Bearhop, S., Burton, N.H., Banks, A.N., Frayling, T., Thaxter, C.B., Clewley, G.D., Scragg, E. & Votier, S.C.** 2023. Urban and coastal breeding lesser black-backed gulls (*Larus fuscus*) segregate by foraging habitat. *Ibis* **165** : 214–230.
Langlois Lopez *et al.* 2022	**Langlois Lopez, S.L., Bond, A.L., O'Hanlon, N.J., Wilson, J., Vitz, A., Mostello, C., Hamilton, F., Rail, J.F., Welch, L., Boettcher, R., Wilhelm, S., Anker-Nilssen, T., Daunt, F. & Masden, E.** 2022. Global population and conservation status of the Great Black-backed Gull *Larus marinus*. *Bird Conservation International* **33** : 1–11.
Langlois Lopez *et al.* 2023	**Lopez, S.L., Daunt, F., Wilson, J., O'Hanlon, N.J., Searle, K.R., Bennett, S., Newell, M.A., Harris, M.P. & Masden, E.** 2023. Quantifying the impacts of predation by Great Black-backed Gulls Larus marinus on an Atlantic Puffin *Fratercula arctica* population: Implications for conservation management and impact assessments. *Marine Environmental Research* **188** : 105994.
Lavers *et al.* 2019	**Lavers, J., Hutton, I. & Bond, A.** 2019. Changes in technology and imperfect detection of nest contents impedes reliable estimates of population trends in burrowing seabirds. *Global Ecology and Conservation* **17** : e00579.
Lavers *et al.* 2020	**Lavers, J., J. M. Hipfner, & G. Chapdelaine** 2020. Razorbill (*Alca torda*), version 1.0. In: S. M. Billerman (eds). 2020. Birds of the World). Cornell Lab of Ornithology, Ithaca, NY, USA. Available at: https://doi.org/10.2173/bow.razorb.01 [Accessed: 25/10/2024].
Leigh *et al.* 2022	**Leigh, D., Kersten, O., Star, B., Anker-Nilssen, T., Burnham, K., Johnson, J., Provencher, J. & Boessenkool, S.** 2022. Sympatry of genetically distinct Atlantic Puffins *Fratercula arctica* in the High Arctic. *Ibis* **165** : 1022–1030.
Lemmetyinen 1973	**Lemmetyinen, R.** 1973. Feeding ecology of *Sterna paradisaea Pontopp.* and *S. hirundo L.* in the archipelago of south-western Finland. *Annales Zoologici Fennici* **10** : 507–525.
Leonard & Wolsey 2014	**Leonard, K. & Wolsey, S.** 2014. *Northern Ireland Seabird Report 2013*. British Trust for Ornithology, Thetford.
Lewis *et al.* 2001	**Lewis, S., Wanless, S., Wright, P.J., Harris, M.P., Bull, J. & Elston, D.A.** 2001. Diet and breeding performance of black-legged kittiwakes *Rissa tridactyla* at a North Sea colony. *Marine Ecology Progress Series* 221 : 277–284.
Lock *et al.* 2021	**Lock, L., Donato, B., Jones, R. & Macleod-Nolan, C.** 2021. *England's breeding seabirds: a review of the status of their breeding sites and suggested measures for their recovery.* RSPB and Natural England report. Available at: https://www.projectlote.life/uploads/1/3/5/6/135667366/seabird_colony_assessment_lote.pdf [Accessed 25/10/2024].
Lock 2006	**Lock J.** 2006. Eradication of brown rats *Rattus norvegicus* and black rats *Rattus rattu*s to restore breeding seabird populations on Lundy Island, Devon, England. *Conservation Evidence* **3** : 111–113.
Løkkeborg & Robertson 2002	**Løkkeborg, S. & Robertson, G.** 2002. Seabird and longline interactions: effects of a bird-scaring streamer line and line shooter on the incidental capture of Northern Fulmars *Fulmarus glacialis. Biological Conservation* **106** : 359–364.
Lorentsen & Anker-Nilssen 1999	**Lorentsen, S.-H. & Anker-Nilssen, T.** 1999. Diet of Common Murres Wintering in the Northern Skagerrak during 1988–1990: Variation with Sex, Age and Season. *Waterbirds: The International Journal of Waterbird Biology* **22** : 80.
Macdonald & Standring 1978	**MacDonald, S. & Standring, K.T.** 1978. An outbreak of botulism in gulls on Firth of Forth, Scotland. *Biological Conservation* **14** : 149–155.

References (M–P)

In text ref.	Reference
Macdonald *et al.* 2015	**MacDonald, A., Heath, M. R., Edwards, M., Furness, R. W., Pinnegar, J. K., Wanless, S., Speirs, D.C. & Greenstreet, S.P.R.** 2015. Climate driven trophic cascades affecting seabirds around the British Isles. *Oceanography and Marine Biology: An Annual Review* **53** : 55–80.
Macgregor *et al.* 2022	**Macgregor, C.J., Boersch-Supan, P.H., Burton, N.H.K., Carss, D.N., Newson, S.E., Pearce-Higgins, J.W., Robinson, R.A. & Taylor, R.C.** 2022. *Informing decisions on lethal control of Great Cormorant and Goosander in Wales: scenarios from Population Viability Analysis*. Natural Resource Wales Evidence Report Series, Report No. 615.
Macgregor *et al.* 2024	**Macgregor, C.J., Gillings, S., Balmer, D.E., Boersch-Supan, P.H., Harris, S.J., Hereward, H.F.R., Humphreys, E.M., Pearce-Higgins, J.W., Taylor, R.C., Troost, G. & Atkinson, P.W.** 2024. Impacts of highly pathogenic avian influenza on seabird populations in the North Sea are detectable in sea-watchers' migration counts. *Bird Study* **71** : 311–325.
Macleod-Nolan 2020	**Macleod-Nolan, C.** 2020. *Little Tern Newsletter 2019.* Unpublished RSPB Report, Royal Society for the Protection of Birds, Sandy.
Madden & Newton 2004	**Madden, B. & Newton, S.F.** 2004. Herring Gull *Larus argentatus.* In: Mitchell, P.I., Newton, S.F., Ratcliffe, N. & Dunn, T.E. (eds). 2004. *Seabird Populations of Britain and Ireland*: 242–262. Poyser, London.
Magnusdóttir *et al.* 2011	**Magnusdóttir, E., Leat, E.H.K., Bourgeon, S., Strøm, H., Petersen, A., Phillips, R.A., Hanssen, S.A., Bustness, J.O., Herstiensson, P. & Furness, R. W**. 2011. Wintering areas of Great Skuas *Stercorarius skua* breeding in Scotland, Iceland and Norway. *Bird Study* **59** : 1–9.
Mallory *et al.* 2009	**Mallory, M.L., Gaston, A.J., Forbes, M.R. & Gilchrist, H.G.** 2009. Influence of weather on reproductive success of northern fulmars in the Canadian High Arctic. *Polar Biology* **32** : 529–538.
Marchant *et al.* 2004	**Marchant, J.H., Freeman, S.N., Crick, H.Q.P. & Beaven, L.P.** 2004. The BTO Heronries Census of England and Wales 1928–2000: new indices and a comparison of analytical methods. *Ibis* **146** : 323–334.
Masden *et al.* 2010	**Masden, E., Haydon, D., Fox, A. & Furness, R.** 2010. Barriers to movement: Modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. *Marine pollution bulletin* **60** : 1085–1091.
Mavor *et al.* 2006	**Mavor, R.A., Parsons, M., Heubeck, M. & Schmitt. S. 2006.** *Seabird numbers and breeding success in Britain and Ireland, 2005.* Joint Nature Conservation Committee, UK Nature Conservation, Peterborough.
Mederios *et al.* 2007	**Medeiros, R., Ramos, J.A., Paiva, V.H., Almeida, A., Pedro, P. & Antunes, S.** 2007. Signage reduces the impact of human disturbance on little tern nesting success in Portugal. *Biological Conservation* **135** : 99–106.
Meek *et al.* 2011	**Meek, E.R., Bolton, M., Fox, D. & Remp, J.** 2011. Breeding skuas in Orkney: a 2010 census indicates density-dependent population change driven by both food supply and predation. *Seabird Journal* **24** : 1–10.
Meininger & Flamant 1998	**Meininger P.L. & Flamant R.** 1998. Breeding populations of Mediterranean Gull *Larus melanocephalus* in The Netherlands and Belgium. *Sula* **12** : 129–138.
Melville & Shortridge 2006	**Melville, D.S. & Shortridge, K.F.** 2006. Migratory waterbirds and avian influenza in the East Asian-Australasian Flyway with particular reference to the 2003–2004 H5N1 outbreak. In: Boere, G., Galbraith, C. & Stroud, D. (eds). 2006. *Waterbirds around the world*. The Stationery Office, Edinburgh, U.K.
Merkel *et al.* 2021	**Merkel, B., Descamps, S., Yoccoz, N.G., Grémillet, D., Fauchald, P., Danielsen, J., Daunt, F., Erikstad, K.E., Ezhov, A.V., Harris, M.P. & Gavrilo, M.** 2021. Strong migratory connectivity across meta-populations of sympatric North Atlantic seabirds. *Marine Ecology Progress Series* 676 : 173–188.
Merkel *et al.* 2016	**Merkel, F.R., Johansen, K.L. & Kristensen, A.J.** 2016. Use of time-lapse photography and digital image analysis to estimate breeding success of a cliff-nesting seabird. *Journal of Field Ornithology* **87** : 84–95.
Met Éireann 2024	**Met Éireann.** 2024. *Climate statements and weather summaries.* Available at: https://www.met.ie [Accessed: 03/06/2025].
Met Office 2024	**Met Office.** 2024. *UK weather and climate updates.* Available at: https://www.metoffice.gov.uk/about-us/news-and-media/media-centre/weather-and-climate-news [Accessed: 03/06/25].
Milchev *et al.* 2004	**Milchev, B., Kodjabashev, N., Sivkov, Y. & Chobanov, D.** 2004. Post-breeding season diet of the Mediterranean gull *Larus melanocephalus* at the Bulgarian Black Sea coast. *Atlantic Seabirds* **6** : 65–78.
Miles *et al.* 2015	**Miles, W.T., Mavor, R., Riddiford, N.J., Harvey, P.V., Riddington, R., Shaw, D.N., Parnaby, D. & Reid, J.M**. 2015. Decline in an Atlantic Puffin Population: Evaluation of Magnitude and Mechanisms. *PLoS ONE* **10** : e0131527.
Miles 2010	**Miles, W.T.S.** 2010. *Ecology, behaviour and predator-prey interactions of Great Skuas and Leach's Storm-petrels at St Kilda*. PhD Thesis, University of Glasgow, Glasgow.
Miles *et al.* 2020	**Miles, J., Parsons, M. & O'Brien, S.** 2020. *Preliminary assessment of seabird population response to potential bycatch mitigation in the UK registered fishing fleet.* Report prepared for the Department for Environment Food and Rural Affairs.
Militão *et al.* 2022	**Militão, T., Sanz-Aguilar, A., Rotger, A. & Ramos, R.** 2022. Non-breeding distribution and at-sea activity patterns of the smallest European seabird, the European Storm Petrel (*Hydrobates pelagicus*). *Ibis* **164** : 1160–1179.
Mitchell *et al.* 2004	**Mitchell, P.I., Newton, S.F., Ratcliffe, N. & Dunn, T.E.** (eds). 2004. *Seabird Populations of Britain and Ireland*. Poyser, London.
Monaghan 1992	**Monaghan, P.** 1992. Seabirds and sandeels: the conflict between exploitation and conservation in the northern North Sea. *Biodiversity and Conservation* **1** : 98–111.
Monaghan & Coulson 1977	**Monaghan, P. & Coulson, J.C.** 1977. Status of Large Gulls Nesting on Buildings. *Bird Study* **24** : 89–104.
Monaghan *et al.* 1989	**Monaghan, P., Uttley, J.D., Burns, M.D., Thaine, C. & Blackwood, J.** 1989. The relationship between food supply, reproductive effort and breeding success in Arctic Terns *Sterna paradisaea*. *The Journal of Animal Ecology* **58** : 261–274.
Morgan 2012	**Morgan, G.** 2012. The bird populations of Ramsey and Grassholm. *British Birds* **105** : 716–732.
Montevecchi 1997	**Montevecchi, W.** 1997. Centurial and decadal oceanographic influences on changes in northern gannet populations and diets in the north-west Atlantic: implications for climate change. *ICES Journal of Marine Science* **54** : 608–614.

In text ref.	Reference
Mougeot & Bretagnolle 2000	**Mougeot, F. & Bretagnolle, V.** 2000. Predation risk and moonlight avoidance in nocturnal seabirds. *Journal of Avian Biology* **31** : 376–386.
Mudge & Ferns 1982	**Mudge, G.P. & Ferns. P.N.** 1982. The feeding ecology of five species of gulls (Ayes: Larini) in the inner Bristol Channel. *Journal of Zoology* **197** : 497–510.
Nager *et al.* 2000	**Nager, R.G., Monaghan, P., Houston, D.C. & Genovart, M.** 2000. Parent condition, brood sex ratio and differential young survival: an experiment study in gull (*Larus fuscus*). *Behaviour Ecology and Sociobiology* **48** : 452–457.
Nettleship 1996	**Nettleship, D.N.** 1996. *Razorbill (Alca torda).* In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). 1996. *Handbook of the Birds of the World Alive*, Lynx Edicions, Barcelona.
Nettleship *et al.* 2018a	**Nettleship, D.N., Kirwan, G.M., Christie, D.A., Boesman, P. & Garcia, E.F.J**. 2018a. *Common Murre (Uria aalge)*. In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds). *Handbook of the Birds of the World Alive*, Lynx Edicions, Barcelona.
Nettleship *et al.* 2018b	**Nettleship, D.N. Sharpe, C.J. Boesman, P. & Garcia, E.F.J.** *2018b. Razorbill (Alca torda)*. Barcelona. Available at: https://www.hbw.com/node/54062. [Accessed: 25/10/2024].
Neumann *et al.* 2021	**Neumann, S., Harju, M., Herzke, D., Anker-Nilssen, T., Christensen-Dalsgaard, S., Langset, M. & Gabrielsen, G.W.** 2021. Ingested plastics in northern fulmars (*Fulmarus glacialis*): A pathway for polybrominated diphenyl ether (PBDE) exposure? *Science of the Total Environment* **778** : 146313.
Newell *et al.* 2015	**Newell, M., Wanless, S., Harris, M.P. & Daunt, F.** 2015. Effects of an extreme weather event on seabird breeding success at a North Sea colony. *Marine Ecology Progress Series* **532** : 257–268.
Newell *et al.* 2025	**Newell, M.A., Harris, M.P., Wanless, S., Burthe, S.J., Bogdanova, M., Gunn, C.M., Hewitt, J., Taylor, E. & Daunt, F.** 2025. *The Isle of May long-term study (IMLOTS) seabird annual breeding success 1982–2024.* NERC EDS Environmental Information Data Centre.
Newson & Austin 2021	**Newson, S & Austin, G.** 2021. *Definition of favourable conservation status for Great Cormorant.* BTO Report for Natural England, Report No. RP2952. Natural England.
Newson *et al.* 2013	**Newson, S.E., Marchant, J.H., Sellers, R.M., Ekins, G.R., Hearn, R.D. & Burton, N.H.K.** 2013. Colonisation and range expansion of inland-breeding Cormorants in England. *British Birds* **106** : 737–743.
Newson *et al.* 2008	**Newson, S.E., Mitchell, P.I., Parsons, M., O'Brien, S.H., Austin, G.E., Benn, S., Black, J., Blackburn, J., Brodie, B., Humphreys, E. & Leech, D.** 2008. Population decline of Leach's Storm-petrel *Oceanodroma leucorhoa* within the largest colony in Britain and Ireland. *Seabird Journal* **21** : 77–84.
Newton 2020	**Newton, I.** 2020. *Uplands and Birds.* Collins. London.
Newton & Little 2009	**Newton, I. & Little, B.** 2009. Assessment of wind-farm and other bird casualties from carcasses found on a Northumbrian beach over an 11-year period. *Bird Study* **56** : 158–167.
Newton & Crowe 2000	**Newton, S.F. & Crowe, O.** 2000. *Roseate Terns - The Natural Connection. A conservation/research project linking Ireland and Wales.* IWC-BirdWatch Ireland, Monkstown, County Dublin.
Nordberg & Salmi 2019	**Nordberg, K. & Salmi, P.** 2019. Addressing the gap between participatory ideals and the reality of environmental management: the case of the coromorant population in Finland. *Environmental Policy and Governance* **29** : 251–261.
Nordström *et al.* 2003	**Nordström, M., Högmander, J., Laine, J., Nummelin, J., Laanetu, N. & Korpimäki, E.** 2003. Effects of feral mink removal on seabirds, waders and passerines on small islands in the Baltic Sea. *Biological Conservation* **109** : 359–368.
Nordström *et al.* 2004	**Nordström, M., Laine, J., Ahola, M. & Korpimäki, E.** 2004. Reduced nest defence intensity and improved breeding success in terns as responses to removal of non-native American mink. *Behavioral Ecology and Sociobiology* **55** : 454–460.
Norman *et al.* 2024	Norman, D., MacLeod-Nolan, C. & Berthelsen, U.M. 2024. Near-absence of high pathogenicity avian influenza (HPAI) in Little Terns *Sternula albifrons* across 13 European countries. *Bird Study* **71**: 347–352.
Northridge *et al.* 2020	**Northridge, S.P, Kingston, A.R. & Coram, A.J.** 2020. *Preliminary estimates of seabird bycatch by UK vessels in UK and adjacent waters.* Final Report to JNCC – Revised October 2020. Scottish Ocean Institute, University of St Andrews.
Northridge *et al.* 2023	**Northridge, S.P., Kingston, A.R. & Coram, A.J.** 2023. *Regional seabird bycatch hotspot analysis.* JNCC Report 726, JNCC, Peterborough.
Ntiamoa-Baidu *et al.* 1992	**Ntiamoa-Baidu, Y., Nyame, S.K. & Nuah, A.A.** 1992. *Preliminary report on tern trapping in coastal Ghana.* In: Rolland, G. (eds). Proceedings of the Roseate Tern workshop. SEPNB. Brest.
O'Hanlon *et al.* 2019	**O'Hanlon, N.J., Bond, A.L., Lavers, J.L., Masden, E.A. & James, N.A.** 2019. Monitoring nest incorporation of anthropogenic debris by Northern Gannets across their range. *Environmental Pollution* **255** : 113152.
O'Hanlon *et al.* 2024a	**O'Hanlon, N.J., Harris, S.J., Thaxter, C.B., Boersch-Supan, P.H., Robinson, R.A., Balmer, D.E. & Burton, N.H.K.** 2024. *Seabird population and demographic monitoring in the UK: a review and recommendations for future sampling*. BTO Research Report **754**: British Trust for Ornithology, Thetford, UK
O'Hanlon *et al.* 2024b	**O'Hanlon, N.J., van Bemmelen, R.S.A., Snell, K.R.S., Conway, G.J, Thaxter, C.B., Aiton, H., Aiton, D., Balmer, D.E., Are Hanssen, S., Calladine, J.R., Hammer, S., Harris, S.J., Moe, B., Schekkerman, H., Tulp, I. & Humphreys, E.M.** 2024. Cross population comparison of complex migration strategies in a declining oceanic seabird. *Marine Ecology Progress Series* **730** : 113–129.
O'Hanlon & Nager 2018	O'Hanlon, N.J. & Nager, R.G. 2018. Identifying habitat-driven spatial variation in colony size of Herring Gulls *Larus argentatus. Bird Study* **65**: 306–316
Olin *et al.* 2023	**Olin, A.B., Dück, L., Berglund, P.A., Karlsson, E., Bohm, M., Olsson, O. & Hentati-Sundberg, J.** 2023. Breeding failures and reduced nest attendance in response to heat stress in a high-latitude seabird. *Marine Ecology Progress Series.*
Oliveira *et al.* 2015	**Oliveira, N., Henriques, A., Miodonski, J., Pereira, J., Marujo, D., Almeida, A., Barros, N., Andrade, J., Marçalo, A., Santos, J., Oliveira, I.B., Ferreira, M., Araújo, H., Monteiro, S., Vingada, J. & Ramírez, I.** 2015. Seabird Bycatch in Portuguese Mainland Coastal Fisheries: An Assessment through on-Board Observations and Fishermen Interviews. *Global Ecology and Conservation* **3** : 51–61.
Olsen & Larsson 2003	**Olsen, K.M. & Larsson, H.** 2003. *Gulls of Europe, Asia and North America.* Christopher Helm, London.
Olsson & Hentati-Sundberg 2017	**Olsson, O. & Hentati-Sundberg.** 2017. Population trends and status of four seabird species (*Uria aalge, Alca torda, Larus fuscus, Larus argentatus*) at Stora Karlsö in the Baltic Sea. *Ornis Svecica* **27** : 64–93.

References (P–S)

In text ref.	Reference
Oppel *et al.* 2014	**Oppel, S., Parejo, S., Oliveira, N., Pipa, T., Silva, C. Geraldes, P., Goh, M., Immler, E. & Mckown, M.** 2014. Estimating population size of a nocturnal burrow-nesting seabird using acoustic monitoring and habitat mapping. *Nature Conservation* 7.
Oro 1996	**Oro, D.** 1996. Effects of trawler discard availability on egg laying and breeding success in the lesser black-backed gull *Larus fuscus* in the western Mediterranean. *Marine Ecology Progress Series* **132** : 43–46.
Oswald & Arnold 2012	**Oswald, S.A. & Arnold, J.M.** 2012. Direct impacts of climatic warming on heat stress in endothermic species: seabirds as bioindicators of changing thermoregulatory constraints. *Integrated Zoology* **7** : 121–136.
Owen *et al.* 2018	**Owen, E., Prince, O., Cachia-Zammit, C., Cartwright, R., Coledale, T., Elliott, S., Haddon, S., Longmoor, G., Swale, J., West, F. & Hughes, R.** 2018. Counts of Puffins in Shetland Suggest an Apparent Decline in Numbers. *Scottish Birds* **38** : 223–231.
Pacini *et al.* 2025	**Pacini, M. I., Ceccherelli, R., Terregino, C., Bortolami, A. & Silva, L.** 2025. The highly pathogenic avian influenza epidemic 2021–2022 from the point of view of an Italian network of Wildlife Rescue Centres. *Bird Study* **72**: 98–109.
Paiva *et al.* 2008	**Paiva, V.H., Ramos, J.A., Martins, J., Almeida, A. & Carvalho, A.** 2008. Foraging habitat selection by Little Terns *Sternula albifrons* in an estuarine lagoon system of southern Portugal. *Ibis* **150** : 18–31.
Paleczny *et al.* 2015	**Paleczny, M & Hammill, E, Karpouzi, V. & Pauly, D.** 2015. Population Trend of the World's Monitored Seabirds, 1950–2010. *PLoS ONE* **10** : e0129342.
Paredes *et al.* 2014	**Paredes, R., Orben, R.A., Suryan, R.M., Irons, D.B., Roby, D.D, Harding, A.M., Young, R.C., Benoit-Bird, K., Ladd, C., Renner, H. & Heppell, S.** 2014. Foraging responses of Black-legged Kittiwakes to prolonged food-shortages around colonies on the Bering Sea Shelf. *PLoS ONE* **9** : e92520.
Pascalis *et al.* 2018	**De Pascalis, F., Collins P.M. & Green, J.A.** 2018. Utility of time-lapse photography in studies of seabird ecology. *PLoS ONE* **13** : e0208995.
Pearce-Higgins *et al.* 2025	**Pearce-Higgins, J.W., Clewley, G., Bolton, M., Banyard, A.C., Falchieri, M., Lindley, P. & Atkinson, P.W.** 2025. Assessing the vulnerability of wild bird populations to high pathogenicity avian influenza. *Bird Study* **72**: 5–19.
Pearce-Higgins & Pollock 2025	**Pearce-Higgins, J.W. & Pollock, C.J.** 2025. Using expert elicitation to assess the likely effectiveness of conservation interventions during an unprecedented outbreak of high pathogenicity avian influenza (HPAI) in wild birds. *Bird Study* **72**: 74–88.
Percival *et al.* 2024	**Percival, S., Bowler, J., Cabot, D., Duffield, S., Enright, M., How, J., Mitchell, C., Percival, T. & Sigfusson, A.** 2024. Spatial and temporal variation in mortality from avian influenza in Greenland Barnacle Geese *Branta leucopsis* in their wintering grounds. *Bird Study* **71**: 404–411.
Perkins *et al.* 2018	**Perkins, A., Ratcliffe, N., Suddaby, D., Ribbands, B., Smith, C., Ellis, P., Meek, E. & Bolton, M.** 2018. Combined bottom-up and top-down pressures drive catastrophic population declines of Arctic skuas in Scotland. *Journal of Animal Ecology* **87** : 1573–1586.
Perrow *et al.* 2011	**Perrow, M., Skeate, E. & Gilroy, J.** 2011. Visual tracking from a rigid-hulled inflatable boat to determine foraging movements of breeding terns. *Journal of Field Ornithology* **82** : 68–79.
Perrow *et al.* 2006	Perrow, M.R., Skeate, E.R., Lines, P., Brown, D. & Tomlinson, M.L. 2006. Radio telemetry as a tool for impact assessment of wind farms: the case of Little Terns *Sterna albifrons* at Scroby Sands, Norfolk, UK. *Ibis* **148**: 57–75.
Peschko *et al.* 2021	**Peschko, V., Mendel, B., Mercker, M., Dierschke, J. & Garthe, S.** 2021. Northern gannets (Morus bassanus) are strongly affected by operating offshore wind farms during the breeding season. *Journal of Environmental Management* **279** : 111509.
Peschko *et al.* 2020	**Peschko, V., Mercker, M. & Garthe, S.** 2020. Telemetry reveals strong effects of offshore wind farms on behaviour and habitat use of common guillemots (*Uria aalg*e) during the breeding season. *Marine Biology* **167** : 118
Phillips *et al.* 1996	**Phillips, R.A., Caldow, R.W.G. & Furness, R.W.** 1996. The influence of food availability on the breeding effort and reproductive success of Arctic Skuas *Stercorarius parasiticus. Ibis* **138** : 410–419.
Phillips *et al.* 1997	**Phillips, A., Catry, P., Thompson, D., Hamer, K. & Furness, R.W.** 1997. Inter-colony variation in diet and reproductive performance of great skuas Catharacta skua. *Marine Ecology Progress Series* **152** : 285–293.
Phillips *et al.* 1999	**Phillips, R.A., Petersen, M.K., Lilliendahl, K., Solmundsson, J., Hamer, K.C., Camphuysen, C.J. & Zonfrillo, B.** 1999. Diet of the northern fulmar *Fulmarus glacialis*: reliance on commercial fisheries? *Marine Biology* **135** : 159–170.
Piatt & Nettleship 1985	**Piatt, J.F. & Nettleship, D,N.** 1985. Diving Depths of Four Alcids. *The Auk* **102** : 293–297.
Pollet *et al.* 2023	**Pollet, I.L., Lenske, A.K., Ausems, A.N.M.A., Barbraud, C., Bedolla-Guzmán. Y., Bicknell, A.W.J., Bolton, M., Bond, A.L., Delord, K., Diamond, A.W., Fifield, D.A., Gjerdrum, C., Halpin, L.R., Hansen, E.S., Hedd, R., Major, H.L., Mauck, R.A., McClelland, G.T.W., McFarlane Tranquilla, L., Montevecchi, W.A., Parker, M., Pratte, I., Rail, J-.F., Robertson, G.J., Rock, J.C., Ronconi, R.A., Shutler, D., Stenhouse, I.J., Takahashi, A., Watanuki, Y., Welch, L.J., Wilhelm, S.I., Wong, S.M.P. & Mallory, M.L.** 2023. Experts' opinions on threats to Leach's Storm-Petrels (*Hydrobates leucorhous*) across their global range. *Avian Conservation and Ecology* **18** : 1
Pollet *et al.* 2014	**Pollet, I.L., Ronconi, R.A., Jonsen, I.D., Leonard, M.L., Taylor, P.D. & Shutler, D.** 2014. Foraging movements of Leach's storm-petrels *Oceanodroma leucorhoa* during incubation. *Journal of Avian Biology* **45** : 305–314.
Pollet *et al.* 2019	**Pollet, I.L., Ronconi, R.A., Leonard, M.L. & Shutler, D.** 2019. Migration routes and stopover areas of Leach's Storm Petrels *Oceanodroma leucorhoa. Marine Ornithology* **47** : 55–65.
Pollock *et al.* 2021	**Pollock, C.J., Lane, J.V., Buckingham, L., Garthe, S., Jeavons, R., Furness, R.W. & Hamer, K.C.** 2021. Risks to different populations and age classes of gannets from impacts of offshore wind farms in the southern North Sea. *Marine Environmental Research* **171** : 105457.
Power *et al.* 2021	**Power, A., White, P., McHugh, B., Berrow, S., Schlingermann, M., Tannian, M., Newton, S., McGovern, E., Murphy, S., Crowley, D., O'Hea, L., Boyle, B. & O'Connor, I.** 2021. Persistent pollutants in Northern Gannet *Morus bassanus* eggs in Ireland: Levels and colony differences. *Environmental Pollution* **268** : 115723.
Pritchard *et al.* 2021	**Pritchard, R., Hughes, J., Spence, I.M., Haycock, B, & Brenchley, A.** (eds) 2021. *The Birds of Wales Adar Cymru.* Liverpool University Press.
Przymencki *et al.* 2024	**Przymencki, M., Beuch, S., Indykiewicz, P., Litwiniak, K., Bukaciński, D., Bukacińska, M., Zieliński, P., Betleja, J., Marchowski, D., Ledwoń, M., Bzoma, S., Buczyński, A., Wardecki, Ł., Bednarz, Ł., Flis, A., Łożyńska, H., Kusal, B. & Chodkiewicz, T.** 2024. Mortality among scarce breeding gulls and terns during a highly pathogenic avian influenza (HPAI) H5N1 virus outbreak in Poland during 2023. *Bird Study* **71**: 326–335.
Quinn *et al.* 2016	**Quinn, L.R., Meharg, A.A., van Franeker, J,A., Graham, I, M. & Thompson, P.M.** 2016. Validating the use of intrinsic markers in body feathers to identify inter-individual differences in non-breeding areas of Northern Fulmars. *Marine Biology* **163** : 64.
R Core Team 2024	**R: A language and environment for statistical computing.** *R Foundation for Statistical Computing*. Vienna, Austria. Available at: www.R-project.org [Accessed: 25/10/2024].
Rahmstorf & Coumou 2011	**Rahmstorf, S. & Coumou, D.** 2011. Increase of extreme events in a warming world. *Proceedings of the National Academy of Sciences* **108** : 17905–17909.
Ratcliffe & Merne 2002	**Ratcliffe, N. & Merne, O.** 2002. *Roseate tern Sterna dougallii*. In: Wernham, C.V., Toms, M., Marchant, J., Clark, J., Siriwardena, G. & Baillie, S. (eds). 2002. *The Migration Atlas: Movements of the birds of Britain and Ireland.* T and A.D. Poyser, London.
Ratcliffe *et al.* 2008a	**Ratcliffe, N., Craik, C., Helyar, A., Roy, S. & Martin, S.** 2008. Modelling the benefits of American Mink Mustela vison management options for terns in west Scotland. *Ibis* **150** : 114–121.
Ratcliffe *et al.* 2008b	**Ratcliffe, N. Schmitt, S. Mayo, A. Tratalos, J. & Drewitt, A.** 2008. Colony habitat selection by little terns *Stemula albifrons* in East Anglia: Implications for coastal management. *Seabird Journal* **21** : 55–63.
Ratcliffe *et al.* 2000	**Ratcliffe, N., Pickerell, G. & Brindley, E.** 2000. Population trends of Little and Sandwich Terns *Sterna albifrons* and *S. sandvicensis* in Britain and Ireland from 1969 to 1998. *Atlantic Seabird* **21** : 211–226.
Raven & Coulson 1997	**Raven, S.J. & Coulson, J.C**. 1997. The distribution and abundance of Larus gulls nesting on buildings in Britain and Ireland. *Bird Study* **44** : 13–34.
Reeves & Furness 2002	**Reeves, S.A. & Furness, R.W.** 2002. *Net loss-seabirds gain? Implications of fisheries management for seabirds scavenging discards in the northern North Sea.* Unpublished RSPB Report, Royal Society for the Protection of Birds, Sandy.
Régnier *et al.* 2019	**Régnier, T., Gibb FM. & Wright PJ.** 2019. Understanding temperature effects on recruitment in the context of trophic mismatch. *Scientific Reports* **9** : 15179.
Régnier *et al.* 2017	**Régnier, T., Gibb, F.M. & Wright, P.J.** 2017. Importance of trophic mismatch in a winter-hatching species: evidence from lesser sandeel. *Marine Ecology Progress Series* **567** : 185–197.
Regular *et al.* 2013	**Regular, P., Montevecchi, W., Hedd, A., Robertson, G. & Wilhelm, S.** 2013. Canadian fishery closures provide a large-scale test of the impact of gillnet bycatch on seabird populations. *Biology letters* **9** : 20130088.
Reid 2004	**Reid, J.B.** 2004. *Great Black-backed Gull Larus marinus.* In: Mitchell, P.I., Newton, S.F., Ratcliffe, N. & Dunn, T.E. (eds). 2004. *Seabird Populations of Britain and Ireland.* Poyser, London.
Rendell-Read 2016	**Rendell-Read, S.** 2016. *Little Tern Newsletter 2015.* Unpublished RSPB Report, Royal Society for the Protection of Birds, Sandy.
Rendell-Read 2018	**Rendell-Read, S.** 2018. *Little Tern Newsletter 2017.* Unpublished RSPB Report, Royal Society for the Protection of Birds, Sandy.
Rexer-Huber *et al.* 2013	**Rexer-Huber, K., Parker., G.C., Ryan., P.G. & Cuthbert, R.J.** 2014. Burrow occupancy and population size in the Atlantic Petrel *Pterodroma incerta*: a comparison of methods. *Marine Ornithology* **42** : 137–141
Richards 1990	**Richards, A.** 1990. *Seabirds of the northern hemisphere.* Dragon's World Ltd, Limpsfield, U.K.
Rijks *et al.* 2022	**Rijks, J.M., Leopold, M.F., Kühn, S., in't Veld, R., Schenk, F., Brenninkmeijer, A., Lilipaly, S.J., Ballmann, M.Z., Kelder, L., de Jong, J.W. & Courtens, W.** 2022. Mass mortality caused by highly pathogenic influenza A (H5N1) virus in Sandwich terns, the Netherlands, 2022. *Emerging Infectious Diseases* **28** : 2538–2542.
Riordan & Birkhead 2018	**Riordan, J. & Birkhead, T.** 2018. Changes in the diet composition of Common Guillemot *Uria aalge* chicks on Skomer Island. *Ibis* **160** : 470–474.
Riou *et al.* 2011	**Riou, S., Gray, C.M., Brooke, M.de.L., Quillfeldt, P., Masello, J.F., Perrins, C. & Hamer, K.C.** 2011. Recent impacts of anthropogenic climate change on a higher marine predator in western Britain. *Marine Ecology Progress Series* **422** : 105–112.
Robinson *et al.* 2024	**Robinson, R.A., Leech, E.I. & Clark, J.A.** 2024. *The Online Demography Report: bird ringing and nest recording in Britain & Ireland in 2023*. BTO, Thetford. Available at: www.bto.org/ringing-report [Accessed: 09/07/2025].
Rock 2005	**Rock, P.** 2005. Urban gulls: problems and solutions. *British Birds* **98** : 338–355.
Rock *et al.* 2016	**Rock, P., Camphuysen, C. J., Shamoun-Baranes, J., Ross-Smith, V. H. & Vaughan, I. P**. 2016. Results from the first GPS tracking of roof-nesting Herring Gulls *Larus argentatus* in the UK. *Ringing and Migration* **31** : 47–62.
Rodriguez & Rodriguez 2009	**Rodriguez, A. & Rodriguez, B.** 2009. Attraction of petrels to artificial lights in the Canary Islands: effects of the moon phase and age class. *Ibis* **151** : 299–310.
Rodway *et al.* 2024	**Rodway, M.S., Bertram, D.F. & Lalach, L.A.R.** 2024. Mixed metrics and the need to adjust remote-sensing data in the evaluation of Key Biodiversity Areas for colonial-nesting seabirds: An example with Glaucous-winged Gulls *Larus glaucescens. Marine Ornithology* **52** : 341–348.
Roos *et al.* 2018	**Roos, S., Smart, J., Gibbons, D. & Wilson, J.** 2018. A review of predation as a limiting factor for bird populations in mesopredator-rich landscapes: a case study of the UK. *Biological Reviews* **93** : 1915–1937.
Ross-Smith *et al.* 2015	**Ross-Smith, V.H., Johnston, A. & Ferns, P.N.** 2015. Hatching success in Lesser Black-backed Gulls *Larus fuscus* – an island case study of the effects of egg and nest site quality. *Seabird Journal* **28** : 1–16.
Ross-Smith *et al.* 2014	**Ross-Smith, V.H., Robinson, R.A., Banks, A.N., Frayling, T.D., Gibson, C.C. & Clark, J.A.** 2014. The Lesser Black-backed Gull *Larus fuscus* in England: how to resolve a conservation conundrum. *Seabird Journal* **27** : 41–61.

References (S–Z)

In text ref.	Reference
Rush *et al*. 2018	**Rush, G.P., Clarke, L.E., Stone, M. & Wood, M.J.** 2018. Can drones count gulls? Minimal disturbance and semiautomated image processing with an unmanned aerial vehicle for colony-nesting seabirds. *Ecology and Evolution* **8** : 12322–12334.
Sandvik *et al*. 2005	**Sandvik, H., Erikstad, K.E., Barrett, R.T. & Yoccoz, N.G.** 2005. The effect of climate on adult survival in five species of North Atlantic seabirds. *Journal of Animal Ecology* **74** : 817–831.
Schmidt *et al*. 2023	**Schmidt, B., Cruickshank, S., Bühler, C. & Bergamini, A.** 2023. Observers are a key source of detection heterogeneity and biased occupancy estimates in species monitoring. *Biological Conservation* **283** : 110102.
Schreiber & Kissling 2005	**Schreiber, J. & Kissling, W.D.** 2005. Factors affecting the breeding success of Arctic Tern *Sterna paradisaea* in a colony at Kaldbaksbotnur, Faroe Islands. *Atlantic Seabirds* **7** : 97–105.
Schwartz *et al*. 2023	**Schwartz, T., Besnard, A., Pin, C., Scher, O., Blanchon, T., Béchet, A. & Sadoul, N.** 2023. Efficacy of created and restored nesting sites for the conservation of colonial Laridae in the south of France. *Conservation Biology* **37** : e14005.
Scott *et al*. 2015	**Scott, P., Duncan, P. & Green, J.A.** 2015. Food preference of the Black-headed Gull *Chroicocephalus ridibundus* differs along a rural-urban gradient. *Bird study* **62** : 56–63.
Searle *et al*. 2014	**Searle, K., Mobbs, D., Butler, A., Bogdanova, M., Freeman, S., Wanless, S. & Daunt, F.** 2014. *Population consequences of displacement from proposed offshore wind energy developments for seabirds breeding at Scottish SPAs*. Final report to Marine Scotland Science, May. Available at: https://tethys.pnnl.gov/sites/default/files/publications/searleetal.pdf [Accessed: 25/10/2024].
Searle *et al*. 2023	**Searle, K., Regan, C., Perrow, M., Butler, A., Rindorf, A., Harris, M., Newell, M., Wanless, S. & Daunt, F.** 2023. Effects of a fishery closure and prey abundance on seabird diet and breeding success: Implications for strategic fisheries management and seabird conservation. *Biological Conservation* **281** : 109990.
Sellers & Shackleton 2011	**Sellers, R.M. & Shackleton, D.** 2011. Numbers, distribution and population trends of large gulls breeding in Cumbria, north-west England. *Seabird Journal* **24** : 90–102.
Sellers 2014	**Sellers, R.S.** 2014. Mass mortality of razorbills and other seabirds on the coast of Cumbria in February 2014. *Lakeland Naturalist* **2** : 63–71.
Seward *et al*. 2019	**Seward, A., Ratcliffe, N., Newton, S., Caldow, R., Piec, D., Morrison, P., Cadwallender, T., Davies, W. & Bolton, M.** 2019. Metapopulation dynamics of roseate terns: Sources, sinks and implications for conservation management decisions. *Journal of Animal Ecology* **88** : 138–153.
Shlepr *et al*. 2021	**Shlepr, K.R., Ronconi, R.A. Hayden, B., Allard, K.A. & Diamond A.W.** 2021. Estimating the relative use of anthropogenic resources by Herring Gull (Larus argentatus) in the Bay of Fundy, Canada. *Avian Conservation and Ecology* **16**
Shoji *et al*. 2016	**Shoji, A., Dean, B., Kirk, H., Freeman, R., Perrins, C. M. & Guilford, T.** 2016. The diving behaviour of the Manx Shearwater *Puffinus puffinus*. *Ibis* **158** : 598–606.
Shoji *et al*. 2015	**Shoji, A., Elliott, K.H., Greenwood, J.G., McClean, L., Leonard, K., Perrins, C.M., Fayet, A. & Guilford, T.** 2015. Diving behaviour of benthic feeding Black Guillemots. *Bird Study* **62** : 217–222.
Short 2014	**Short, D.** 2014. *Breeding of four species of tern and Black-headed Gull at Forvie National Nature Reserve, 2013*. Unpublished Scottish Natural Heritage Report, Scottish Natural Heritage, Edinburgh.
Short 2020	**Short, D.** 2020. *Breeding of four species of tern and Black-headed Gull at Forvie National Nature Reserve, 2019*. Unpublished Scottish Natural Heritage Report, Scottish Natural Heritage, Edinburgh.
Short & Watts 2016	**Short, D. & Watts E.** 2016. *Breeding of four species of tern and Black-headed Gull at Forvie National Nature Reserve, 2015*. Unpublished Scottish Natural Heritage Report, Scottish Natural Heritage, Edinburgh.
Skórka *et al*. 2006	**Skórka, P., Martyka, R., Wójcik, J.D., Babiarz, T. & Skórka, J.** 2006. Habitat and nest site selection in the Common Gull *Larus canus* in southern Poland: significance of man-made habitats for conservation of an endangered species. *Acta Ornithologica* **41** : 137–144.
Smart & Amar 2018	**Smart, J, & Amar, A.** 2018. Diversionary feeding as a means of reducing raptor predation at seabird breeding colonies. *Journal for Nature Conservation* **46** : 48–55.
Smith *et al*. 2025	**Smith, G.D., McGrady, M.J., Beckmann, B.C. & Oli, M.K.** 2025. Potential effects of HPAI on occupancy rates, breeding success, age and turnover of breeding Peregrine Falcons *Falco peregrinus* in southern Scotland. *Bird Study* **72**: 69-73.
Smith & Morgan 2005	**Smith, J. & Morgan, K.** 2005. *An assessment of Seabird Bycatch in longline and net fisheries in British Columbia*. Technical Report Series Report No. 401. Canadian Wildlife Service.
Smith *et al*. 2023	**Smith, P.A., Smith, A.C., Andres, B., Francis, C.M., Harrington, B., Friis, C., Morrison, R.G., Paquet, J., Winn, B. & Brown, S.** 2023. Accelerating declines of North America's shorebirds signal the need for urgent conservation action. *Ornithological Applications* **125** : 003.
Snell *et al*. 2024	**Snell, K.R., Aldará, J., Hammer, S. & Thorup, K.** 2024. Thermal stress during incubation in an arctic breeding seabird. *Journal of Thermal Biology* **125** : 103967.
Snow & Perrins 1998	**Snow, D.W. & Perrins, C.M.** 1998. *The Birds of the Western Palearctic, Volume 1: Non-Passerines*. Oxford University Press, Oxford.
Sonntag & Hüppop 2005	**Sonntag, N. & Hüppop, O.** 2005. Snacks from the depth: summer and winter diet of Common Guillemots *Urria aalgae* around the Island of Heligoland. *Atlantic Seabirds* **7** : 1–14.
Stanbury *et al*. 2024	**Stanbury, A.J., Burns, F., Aebischer, N.J., Baker, H., Balmer, D.E., Brown, A., Dunn, T.E., Lindley, P., Murphy, M., Noble, G., Owens, R. & Quinn, L.** 2024. The status of the UK's breeding seabirds: an addendum to the fifth Birds of Conservation Concern in the United Kingdom, Channel Islands and Isle of Man and second IUCN Red List assessment of extinction risk for Great Britain. *British Birds* **117** : 471–487.
Stienen *et al*. 1998	**Stienen, E.W.M., Jonard, A. & Brenninkmeijer, A.** 1998. Tern trapping along the Senegalese coast. *Sula* **12** : 19–26.
Stienen 2006	**Stienen, E.** 2006. *Living with gulls : trading off food and predation in the Sandwich Tern Sterna sandvicensis*. Alterra

In text ref.	Reference
Stubbings *et al*. 2017	**Stubbings, E.M., Büche, B.I., Riordan, J.A., Moss, J. & Wood, M.J.** 2017. *Seabird monitoring on Skomer Island in 2017*. Unpublished JNCC Report, Joint Nature Conservation Committee, Peterborough.
Suddaby & Ratcliffe 1997	**Suddaby, D. & Ratcliffe, N.** 1997. The effects of fluctuating food availability on breeding Arctic Terns (*Sterna paradisaea*). *The Auk* **114** : 524–530.
Swann 2002	**Swann, R.L.** 2002. *Canna Seabird Studies 2001*. JNCC, Peterborough.
Swann & Beckmann 2025	**Swann, R.L. & Beckmann, B.C.** 2025. Significant decline of a Common Buzzard *Buteo buteo* population during an outbreak of high pathogenicity avian influenza (HPAI). *Bird Study* **72**: 61-68.
Swann *et al*. 2008	**Swann, R.L., Harris, M.P. & Aiton, D.G.** 2008 The diet of European shag *Phalacrocorax aristotelis*, black-legged kittiwake *Rissa tridactyla* and common guillemot *Uria aalge* on Canna during the chick-rearing period 1981–2007. *Seabird Journal* **21** : 44–54.
Swann *et al*. 2016	**Swann, R.L., Aiton, D.G., Call, A., Foster, S., Graham, A., Graham, K. & Young, A.** 2016. *Canna seabird studies 2015*. JNCC, Peterborough.
Swann *et al*. 2021	**Swann, R.L., Aiton, D.G., Call, A., Foster, S., Graham, A., Graham, K. & Young, A.** 2021. *Canna seabird studies 2021*. Unpublished JNCC Report. Joint Nature Conservation Committee, Peterborough.
Syposz *et al*. 2018	**Syposz, M., Gonçalves, F., Carty, M., Hoppitt, W. & Manco, F.** 2018. Factors influencing Manx Shearwater grounding on the west coast of Scotland. *Ibis* **160** : 846–854.
Tasker 2000	**Tasker, M.L.** 2000. The UK and Ireland seabird monitoring programme – a history and introduction. *Atlantic Seabirds* **2** : 97–102.
Tavecchia *et al*. 2006	**Tavecchia, G., Baccetti, N. & Serra, L.** 2006. *Modelling survival and movement probability of Little Tern Sterna albifrons at a post-breeding moulting site: the effect of the colony of origin*. In: Boere, G., Galbraith, C. & Stroud, D. (eds). *Waterbirds around the world*. The Stationary Office, Edinburgh, UK.
Taverner 1970	**Taverner, H.** 1970. Mediterranean Gulls nesting in Hampshire. *British Birds* **63** : 67–79.
Taylor 1983	**Taylor, I.R.** 1983. Effect of wind on the foraging behaviour of Common and Sandwich Terns. *Ornis Scandinavica* **14** : 90–96.
Taylor *et al*. 2012	**Taylor, C.J., Boyle, D., Perrins, C.M. & Kipling, R.** 2012. *Seabird monitoring on Skomer Island in 2012*. Unpublished JNCC Report, Joint Nature Conservation Committee, Peterborough.
Thaxter *et al*. 2018	**Thaxter, C.B., Ross-Smith, V.H., Bouten, W., Masden, E.A., Clark, N.A., Conway, G.J., Barber, L., Clewley, G.D. & Burton, N.H.** 2018. Dodging the blades: new insights into three-dimensional space use of offshore wind farms by lesser black-backed gulls *Larus fuscus*. *Marine Ecology Progress Series* **587** : 247–253.
Thaxter *et al*. 2019	**Thaxter, C.B., Ross-Smith, V.H., Bouten, W., Clark, N.A., Conway, G.J., Masden, E.A., Clewley, G.D., Barber, L.J. & Burton, N.H.** 2019. Avian vulnerability to wind farm collision through the year: Insights from lesser black-backed gulls (*Larus fuscus*) tracked from multiple breeding colonies. *Journal of Applied Ecology* **56** : 2410–2422.
The Sun Ireland 2024	**The Sun Ireland.** 2024. *News and weather coverage*. Available at: https://www.thesun.ie [Accessed: 03/06/2025].
Thomas 1993	**Thomas, G.E.** 1993. Estimating annual total heron population counts. *Appl. Statistics* **42** : 473–486.
Thorup *et al*. 2014	**Thorup, S.H., Jensen, J-K., Petersen, K.T. & Kasper, D.B.** 2014. *Færøsk Trækfugleatlas. The Faroese Bird Migration Atlas*. Faroe University Press, Tórshavn.
Tremlett *et al*. 2024a	**Tremlett, C.J., Cleasby, I.R., Bolton, M. & Wilson, L.J.** 2024. Declines in UK breeding populations of seabird species of conservation concern following the outbreak of high pathogenicity avian influenza (HPAI) in 2021–2022. *Bird Study* **71** : 293–310.
Tremlett *et al*. 2024b	**Tremlett, C.J., Morley, N. & Wilson, L.J.** 2024. U*K seabird colony counts in 2023 following the 2021–22 outbreak of Highly Pathogenic Avian Influenza*. RSPB Research Report No. 76. RSPB Centre for Conservation Science, RSPB, The Lodge, Sandy, Bedfordshire.
Trevail *et al*. 2019	**Trevail, A.M., Green, J.A., Sharples, J., Polton, J.A., Miller, P.I., Daunt, F., Owen, E., Bolton, M., Colhoun, K., Newton, S., Robertson, G. & Patrick, S.C.** 2019. Environmental heterogeneity decreases reproductive success via effects on foraging behaviour. *Proceedings of the Royal Society B* **286** : 20190795
Tucker & Heath 1994	**Tucker, G.M. & Heath, M.F.** 1994. *Birds in Europe: their conservation status*. BirdLife International, Cambridge, U.K.
Tuia *et al*. 2022	**Tuia, D., Kellenberger, B., Beery, S., Costelloe, B.R., Zuffi, S., Risse, B., Mathis, A., Mathis, M.W., van Langevelde F., Burghardt, T., Kays, R., Klinck, H., Wikelski, M., Couzin, I.D., van Horn, G., Crofoot, M.C., Stewart, C.V., Berger-Wolf, T.** 2022. Perspectives in machine learning for wildlife conservation. *Nature Communications* **13** : 792.
Tyndall *et al*. 2024	Tyndall. A.A., Nichol. C.J., Wade. T., Pirrie. S., Harris. M.P., Wanless. S. & Burton. E. 2024. Quantifying the impact of Avian Influenza on the Northern Gannet colony of Bass Rock using ultra-high-resolution drone imagery and deep learning. *Drones* **8**: 40.
Underwood & Stowe 1984	**Underwood, L.A. & Stowe, T.J.** 1984. Massive wreck of seabirds in eastern Britain, 1983. *Bird Study* **31** : 79–88.
Urban *et al*. 1986	**Urban, E.K., Fry, C.H. & Keith, S.** 1986. *The Birds of Africa, Volume II*. Academic Press, London.
	Uttley, J., Monaghan, P. & White, S. 1989. Differential effects of reduced sandeel availability on two sympatrically breeding species of tern. *Ornis Scandinavica* **20**: 273–277.
van Bemmelen *et al*. 2021	**van Bemmelen, R.S., Schekkerman, H., Hin, V., Pot, M.T., Janssen, K., Ganter, B., Rösner, H.U. & Tulp, I.** 2021. Heavy decline of the largest European Arctic Skua *Stercorarius parasiticus* colony: interacting effects of food shortage and predation. *Bird Study* **68** : 88–99.
van Bemmelen *et al*. 2024	**van Bemmelen, R.S., Moe, B., Schekkerman, Are Hansen, S., Snell, K.R.S., Humphreys, E.M., Mäntylä, E. Hallgrimsson, G.T., Gilg, O., Ehrich, D., Calladine, J., Hammer, S., Harris, S.J., Lang, J., Vignisson, S.R., Kolbeinsson, Y., Nuotio, K., Sillanpää, M., Sittler, B., Sokolov, A., Klaassen, R,H.G., Phillips, R.A. & Tulp, I.A.** 2024. Synchronous timing of return to breeding sites in a long-distance migratory seabird with ocean-scale variation in migration schedules. *Movement Ecology* **12** : 22
van de Pol *et al*. 2010	**van de Pol, M., Ens, B.J., Heg, D., Brouwer, L., Krol, J., Maier, M., Exo, K.M., Oosterbeek, K., Lok, T., Eising, C.M. & Koffijberg, K.** 2010. Do changes in the frequency, magnitude and timing of extreme climatic events threaten the population viability of coastal birds? *Journal of Applied Ecology* **47** : 720–730.

In text ref.	Reference
van Donk *et al.* 2017	**van Donk, S., Camphuysen, C.J., Shamoun-Baranes, J., & van der Meer, J.** 2017. The most common diet results in low reproduction in a generalist seabird. *Ecology and Evolution* **7** : 4620–4629.
Vanermen *et al.* 2015	**Vanermen, N., Onkelinx, T., Courtens, W., van de Walle, M., Verstraete, H. & Stienen, E.W.** 2015. Seabird avoidance and attraction at an offshore wind farm in the Belgian part of the North Sea. *Hydrobiologia* **756** : 51–61.
van Franeker & Wattel 1982	**van Franeker, J.A. & Wattel, J.** 1982. Geographical variation of the fulmar *Fulmarus glacialis* in the North Atlantic. *Ardea* **70** : 31–44.
van Franeker *et al.* 2021	**van Franeker, J.A., Kühn, S., Anker-Nilssen, T., Edwards, E.W., Gallien, F., Guse, N., Kakkonen, J.E., Mallory, M.L., Miles, W., Olsen, K.O. & Pedersen, J.** 2021. New tools to evaluate plastic ingestion by northern fulmars applied to North Sea monitoring data 2002–2018. *Marine Pollution Bulletin* **166** : 112246
Vigfusdottir *et al.* 2013	**Vigfusdottir, F., Gunnarsson, T.G. & Gill, J.A.** 2013. Annual and between-colony variation in productivity of Arctic Terns in West Iceland. *Bird Study* **60** : 289–297.
Votier *et al.* 2008	**Votier, S.C., Bearhop, S., Fyfe, R., & Furness, R.W.** 2008. Temporal and spatial variation in the diet of a marine top predator - links with commercial fisheries. *Marine Ecology Progress Series* **367** : 223–232.
Votier *et al.* 2005	**Votier, S.C., Hatchwell, B.J., Beckerman, A., McCleery, R.H., Hunter, F.M., Pellatt, J., Trinder M. & Birkhead, T.R.** 2005. Oil pollution and climate have wide-scale impacts on seabird demographics. *Ecology Letters* **8** : 1157–1164.
Votier *et al.* 2004	**Votier, S.C., Bearhop, S., Ratcliffe, N., Phillips, R.A. & Furness, R.W.** 2004. Predation by great skuas at a large Shetland seabird colony. *Journal of Applied Ecology* **41** : 1117–1128.
Wakefield *et al.* 2013	**Wakefield, E.D., Bodey T.W., Bearhop, S., Blackburn, J., Colhoun, K., Davies, R., Dwyer, R.G., Green, J.A., Grémillet, D., Jackson, A.L., Jessopp, M.J., Kane, A., Langston, R.H.W., Lescroël, A., Murray, S., Le Nuz, M., Patrick, S.C., Péron, C., Soanes, L.M., Wanless, S., Votier. S.C. & Hamer. K.C.** 2013. Space Partitioning Without Territoriality in Gannets. *Science Journal* **341** : 68–70.
Walsh *et al.* 1995	**Walsh, P.M., Halley, D.J., Harris, M.P., del Nevo, A., Sim, I.M.W. & Tasker, M.L.** 1995. *Seabird monitoring handbook for Britain and Ireland.* Peterborough, JNCC/RSPB/ITE/Seabird Group.
Wanless *et al.* 2018	**Wanless, S., Harris, M.P., Newell, M.A., Speakman, J.R. & Daunt, F.** 2018. Community-wide decline in the occurrence of lesser sandeels *Ammodytes marinus* in seabird chick diets at a North Sea colony. *Marine Ecology Progress Series* **600** : 193–206.
Wanless *et al.* 2007	**Wanless, S., Frederiksen, M., Daunt, F., Scott, B.E. & Harris, M.P.** 2007. Black-legged kittiwakes as indicators of environmental change in the North Sea: Evidence from long-term studies. *Progress in Oceanography* **71** : 30–38.
Wanless *et al.* 2005	**Wanless, S., Harris, M.P., Redman, P. & Speakman, J.R.** 2005. Low energy values of fish as a probable cause of a major seabird breeding failure in the North Sea. *Marine Ecology Progress Series* **294** : 1–8.
Wanless *et al.* 2004	**Wanless, S., Wright, P.J., Harris, M.P. & Elston, D.A.** 2004. Evidence for decrease in size of lesser sandeels Ammodytes marinus in a North Sea aggregation over a 30-yr period. *Marine Ecology Progress Series* **279** : 237–246.
Wanless & Harris 1997	**Wanless. S. & Harris, M.** 1997. *Phalacrocorax aristotelis Shag.* BWP update **1** : 3–13.
Watanuki 1985	**Watanuki, Y.** 1985. Food of breeding Leach's storm-petrels (*Oceanodroma leucorhoa*). *The Auk* **102** : 884–886.
Wernham *et al.* 2002	**Wernham, C., Toms, M., Marchant, J.H., Clark, J., Siriwardena. & G., Baillie, S.R.** 2002. *The Migration Atlas: Movements of the Birds of Britain and Ireland.* British Trust for Ornithology, Thetford.
Wilhelm *et al.* 2016	**Wilhelm, S.I., Rail, J.F., Regular, P.M., Gjerdrum, C. & Robertson, G.J.** 2016. Large-scale changes in abundance of breeding Herring Gulls (*Larus argentatus*) and Great Black-backed Gulls (*Larus marinus*) relative to reduced fishing activities in south-eastern Canada. *Waterbirds* **39** : 136–142.
Wilson *et al.* 2020	**Wilson, L.J., Rendell-Read, S., Lock, L., Drewitt, A.L. & Bolton, M.** 2020. Effectiveness of a five-year project of intensive, regional-scale, coordinated management for little terns *Sternula albifrons* across the major UK colonies. *Journal for Nature Conservation* **53** : 125779.
Wilson *et al.* 2014	**Wilson, S. Bazin, R. Calver, W. Doyle, T.J., Earsom, S.D., Oswald, S.A. & Arnold, J.M.** 2014. Abundance and trends of colonial waterbirds on the large lakes of Southern Manitoba. *Waterbirds* **3**: 233–244.
Wilson *et al.* 2025	**Wilson, M.W., Beckmann, B.C. & Wernham, C.V.** 2025. Reduced breeding success consistent with effects of high pathogenicity avian influenza (HPAI) on raptors in Scotland in 2022. *Bird Study* **72**: 51-60.
Williams & DeLeon 2020	**Williams, H.M. & DeLeon, R.L.** 2020. Deep learning analysis of nest camera video recordings reveals temperature-sensitive incubation behaviour in the purple martin *Progne subis. Behavioral Ecology and Sociobiology* **74** : 1–12.
Williams *et al.* 1995	**Williams, J.M., Tasker, M.L., Carter, I.C. & Webb, A.** 1995. A method of assessing seabird vulnerability to surface pollutants. *Ibis* **137**: s147–152.
Wood *et al.* 2021	**Wood, M.J., Canonne, C., Besnard, A., Lachish, S., Fairhurst, S.M., Liedvogel, M., Boyle, D., Patrick, S.C., Josey, S., Kirk, H. & Dean, B.** 2021. Demographic profiles and environmental drivers of variation relate to individual breeding state in a long-lived trans-oceanic migratory seabird, the Manx shearwater. *PLoS One* **16** : e0260812.
Wright & Wilde 2015	**Wright, D. & Wilde, D.** 2015. *Cemlyn wardens report 2015.* Unpublished NWWT Report, North Wales Wildlife Trust.
Žydelis *et al.* 2013	**Žydelis, R., Small, C., & French, G.** 2013. The incidental catch of seabirds in gillnet fisheries: A global review. *Biological Conservation* **162** : 76–88.

KITTIWAKE, BY SAM LANGLOIS / BTO

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On behalf of the SMP Organiser and the SMP Steering Group, we would like to thank all surveyors for making the SMP the success it is today, and for your continued support as we take the programme forward. Unfortunately, space does not permit all surveyors to be acknowledged individually but we are very grateful for everyone's contributions.

We would also like to thank all the landowners who kindly allow surveyors access to carry out monitoring on their land, and the organisations who support the programme through staff time to carry out the surveys.

Furthermore, there are 24 organisations on the SMP Advisory Group (see page 2 for the full list), including those managing the monitoring work of the four SMP Key Sites of Canna, Fair Isle, the Isle of May and Skomer Island, the programme partners (BTO and JNCC) and the associate partner (RSPB). We are grateful to those representing the organisations on this group for helping to shape the programme and guide its future direction. We would also like to acknowledge those working behind the scenes at these organisations to enable various parts of the programme to function and to be improved and promoted. This includes Information Systems teams, administrators, Communications teams, and data analysts and we thank you all!

In time, we aim to form a network of voluntary Regional Coordinators to join the wider BTO Regional Network to assist the SMP Organiser in local engagement and programme coordination. This is currently in the planning stage and the SMP Online system will need to be adapted to support the administrative aspect of this role. However, we already communicate with some of the most recent census coordinators who helped at a local level with the Seabirds Count project and with the existing BTO network. So thanks are also due to those who continue to, or have started to support the SMP through promotion and the sharing of local knowledge which is invaluable to the programme.

SEABIRD MONITORING, BY KATHERINE BOOTH JONES / BTO

▲ The Seabird Monitoring Programme is only possible thanks to the efforts of both professional fieldworkers and skilled non-professionals dedicated to the monitoring of the UK's seabirds. We are very grateful to all participants who make it possible to report from across the Channel Islands, England, Isle of Man, Northern Ireland, Scotland, the Republic of Ireland and Wales.

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