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Global population and conservation status of the Great Black-backed Gull *Larus marinus*

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Summary

The Great Black-backed Gull *Larus marinus* is a generalist species that inhabits temperate and arctic coasts of the north Atlantic Ocean. In recent years, there has been growing concern about population declines at local and regional scales; however, there has been no attempt to robustly assess Great Black-backed Gull population trends across its global range. We obtained the most recent population counts across the species' range and analysed population trends at a global, continental, and national scale over the most recent three-generation period (1985–2021) following IUCN Red List criteria. We found that, globally, the species has declined by 43%–48% over this period (1.2–1.3% per annum, respectively), from an estimated 291,000 breeding pairs to 152,000–165,000 breeding pairs under two different scenarios. North American populations declined more steeply than European ones (68% and 28%, respectively). We recommend that Great Black-backed Gull should be uplisted from 'Least Concern' to 'Vulnerable' on the IUCN Red List of Threatened Species under criterion A2 (an estimated reduction in population size >30% over three generations).

Introduction

Gulls in the genus *Larus* are found in a wide variety of habitats, from urban environments and agricultural land to remote, uninhabited islands. They are omnivorous and feed on anthropogenic waste, naturally hunted fish, fish and offal scavenged from the fishing industry, intertidal organisms, and other seabirds and small mammals (Harris 1980, Buckley 1990, Steenweg *et al.* 2011, Westerberg *et al.* 2019). This flexibility has made them highly resourceful and adaptable, and a common feature of many ecosystems. As observed by Anderson *et al.* (2016), this adaptability has perhaps led ecologists and conservationists to a feeling of complacency regarding the conservation status and marked ecological knowledge gaps of *Larus* gulls. However, accurate and contemporary information about the ecology and status of populations is key to shape conservation policy.

Limited attention to the population trends of *Larus* species has been compounded by their status as "nuisance species" resulting in persistent negative perceptions towards them. They may negatively impact agriculture and fisheries or create human-wildlife conflict by stealing food, defending their offspring, or defecating on vehicles and street furniture. Gulls have been "managed" and culled in great numbers in the early 20th century. For example, over 800,000 Herring *Larus argentatus* and Great Black-backed Gull *L. marinus* eggs were destroyed in New England, United States, between 1934 and 1950 due to concerns over the potential impacts on crops and fishing weirs (Anderson *et al.* 2016). Similarly, non-lethal and lethal gull control is a common tool used at local scales in the event of wildlife conflicts, typically when gull populations negatively impact other bird populations (Anker-Nilssen and Tatarinka 2000, Bosch *et al.* 2000, Guillemette and Brousseau 2001, Finney *et al.* 2003). Oro and Martínez-Abraín (2007) examined the use of lethal control in gull management, the most prevalent approach, finding that it is often ineffective in the long-term and that its implementation is generally driven by the negative perception of gulls rather than scientific evidence. Large knowledge gaps regarding the foraging ecology of some *Larus* species can result in faulty assumptions about gulls' impact on anthropogenic resources, further challenging the justification for these management measures.

Additionally, it has been argued that recent declines in *Larus* populations should be of no concern after increases in abundance observed in the 20th century were linked to anthropogenic activities, and populations may at present be returning to "baseline" levels of abundance. However, although the distribution of gull species was relatively well known prior to the 20th century, quantitative records of population size are scarce during that time (Holloway 2010). Therefore, it is extremely challenging to identify a "natural" baseline abundance of *Larus* populations, particularly for species that have long been associated with human settlements. Although there is evidence that anthropogenic resources have led to a significant increase of some populations, others do not rely on these resources and have experienced declines due to environmental change and anthropogenic impacts in a similar fashion to other seabird species (Regehr and Rodway 1999, Stenhouse and Montevecchi 1999, Davoren and Montevecchi 2003). Accurate and objective classifications of the status and population trends of the species are therefore needed to inform management and conservation of their populations.

Accordingly, concern has grown regarding the declines seen in several, generalist *Larus* species across their range (Anderson *et al.* 2016). One such species is the Great Black-backed Gull, *Larus marinus,* which inhabits temperate and Arctic coastal areas on either side of the Atlantic Ocean, their core breeding area being the United States, Canada, Iceland, Scandinavia, and the British Isles. They are primarily diurnal foragers and exploit a wide variety of coastal and terrestrial habitats and food resources (Coulson 2019). The number of breeding Great Black-backed Gulls has fluctuated greatly throughout the 20th and 21st centuries. During most of the former, populations increased in both numbers and geographical range across the North Atlantic basin (Nisbet *et al.* 2013). A remarkable example of this increase was reported by Drury (1974) in New England, USA, where the breeding population of Great Black-backed Gulls increased from 30 to 12,400 breeding pairs between 1930 and 1972. This trend observed in Great Black-backed Gulls was mirrored in other *Larus* species and was associated with a decrease in persecution and an increase in the availability of anthropogenic food resources, particularly landfill waste and fishery discards (Wilhelm *et al.* 2016). More recently, however, steep declines have been recorded in the last 20–40 years. For example, between 2008 and 2013, the number of Great Black-backed Gulls in Maine, United States, declined by 30% from approximately 10,000 to 6,000 breeding pairs (Mittelhauser *et al.* 2016). In eastern Canada, the estimated number of breeding pairs declined from approximately 41,000 to 12,000 between 1990 and 2014 (Wilhelm *et al.* 2016). In Europe, similar declines have been observed in some countries such as Iceland, where the population declined from 22,500 to 7,000 pairs between 1980 and 2016 (BirdLife International 2015). Although these local and regional estimates suggest that the species is in decline, the species' extensive breeding range and difficulty in censusing all populations has meant that accurate estimates of global breeding population trends have not been undertaken. Accurate and contemporary information about the status of Great Black-backed Gull populations is key to inform conservation and research priorities, as well as environmental impact assessments required in the regulation of some marine activities (e.g., renewable energy generation). Accordingly, in this study we sourced the most recent breeding population estimates from throughout the species' range and analysed the change in abundance at a global, continental, and national scale over the most recent three-generation period, a time frame used to evaluate species against the IUCN Red List criteria.

Methods

Population counts

Population count data were sourced from 17 countries, spanning the entirety of the Great Black-backed Gull's breeding range (Figure 1 and Table 1). For a small number of countries, states, or regions, where total population counts were not available, a sample from selected breeding colonies was used (see "Data analysis" for how missing populations were incorporated in the analysis). All population counts represented the number of breeding pairs only, excluding data from the non-breeding season and non-mature birds, in line with IUCN criteria.

The geographical resolution of population counts varied across the species' range. Country-level counts were available for Europe, whereas for the United States and Canada counts were generally available at a higher resolution (state or region; Table 1). Population estimates were collected using a variety of methods of different accuracy. Methods of higher accuracy included individual counts carried out at ground level, as well as boat and aerial surveys of breeding individuals. Methods of lower accuracy were typically linked to some European countries, where the data had already been classed by BirdLife International as "based on extrapolation from a subset of colonies" or "based on expert opinion" (see references from Table 1 for sources).

Selection of counts and uncertainty

Generation length, defined as "the average age of parents of the current cohort", is used in conservation for assessing extinction risk. The IUCN Red List criteria contrast changes in population size over the most recent three-generation period of a species to quantitatively assign each species to an extinction risk category (IUCN 2012). Generation length in Great Black-backed Gulls has been estimated as 12 years (Bird *et al.* 2020, IUCN 2022), so we estimated changes in Great Black-backed Gull abundance from 1985 to 2021 (36 years).

Because population counts were available at different geographical scales (country, state, regions), the term "population" is used hereafter to represent any geographical scale for which there was an individual set of counts (i.e. each input to the analysis presented in Table 1). We chose to use two counts per population, as this was the common denominator for all populations. As we were interested in presenting the overall change in abundance over the three-generation period rather than changes in abundance at a higher temporal resolution, the two counts closest to the beginning and end of the study period (1985 and 2021, respectively) were chosen for populations where more than two counts were available. This ensured consistency and the highest degree of accuracy in the analysis (see Data Analysis, below).

Each population count was allocated an estimate of uncertainty based on the method used to produce the count. This was done according to a five-point scale (Croxall and Kirkwood 1979, Lynch *et al.* 2013) ranging from low uncertainty (N1) to high uncertainty (N5). We redefined the N5 category from Lynch et al. (2013) to

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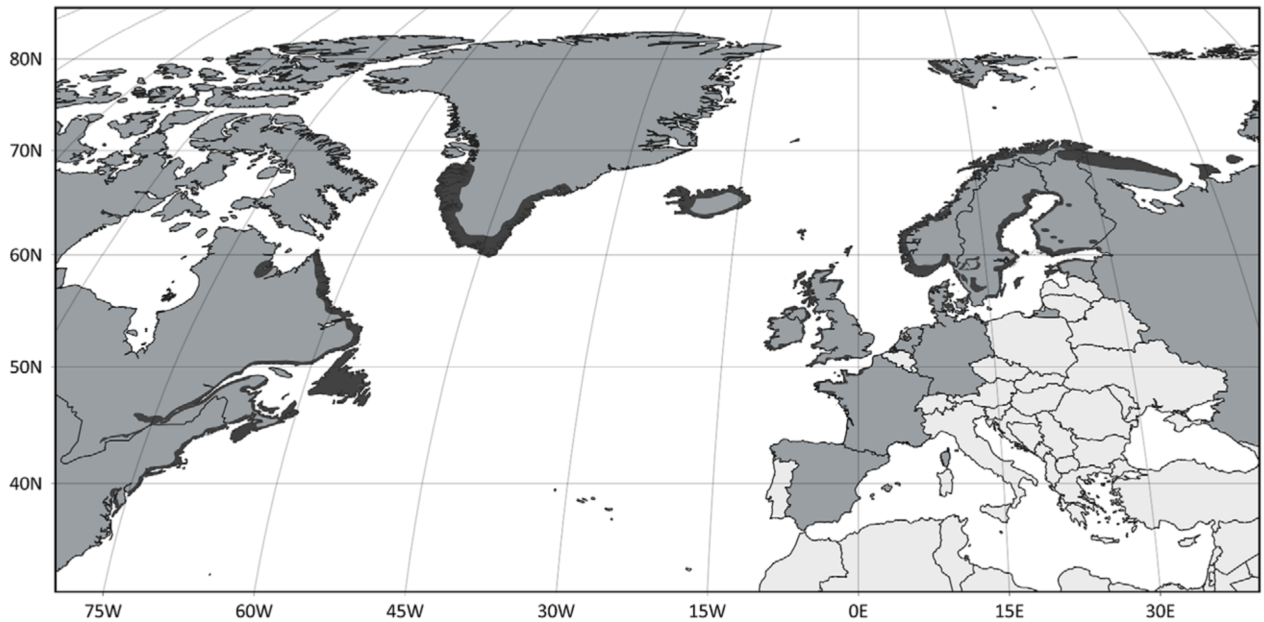

Figure 1. Global breeding range of Great Black-backed Gulls *Larus marinus* (areas coloured black) and countries where population counts were obtained (dark grey). Breeding range data were obtained from BirdLife International and Handbook of the Birds of the World (Del Hoyo *et al.* 2019).

Table 1. Parameters used in the analysis. These include: two population counts per region (Count 1 and Count 2), the years these counts were obtained (Year 1 and Year 2), the error associated with each count (Error 1 and Error 2; see Table 2 for more information about error categories), and the locations each population count represents.

Country/State/Province	Region	Year 1	Year 2	Count 1	Count 2	Error 1	Error 2
Europe							
Russia	Russia	2000[1]	2013[2]	3,500	4,000	N4	N4
Estonia	Estonia	1998[1]	2015[2]	4,000	1,250	N4	N4
Finland	Finland	1986[11]	2016[2]	2,800	1,511	N4	N4
Sweden	Sweden	2000[1]	2016[2]	12,500	8,400	N4	N4
Germany	Germany	1997[1]	2015[2]	23	100	N2	N2
Denmark	Denmark	2000[1]	2017[2]	2,500	2,500	N4	N2
Norway	Mainland	1990[9]	2013[7]	40,000	41,688	N4	N2
Netherlands	Netherlands	1999[1]	2015[2]	13	70	N2	N2
France	France	1999[1]	2011[2]	3,850	6,502	N2	N2
United Kingdom	Great Britain	1970[14]	2015[2]	22,000	13,109	N2	N4
Spain	Spain	2000[1]	2016[2]	4	18	N2	N5
Republic of Ireland	Ireland	2000[1]	2018[2]	2,200	3,081	N2	N2
Faroe Islands	Faroe Islands	1995[1]	2015[2]	1,200	1,200	N5	N5
Iceland	Iceland	1980[2]	2016[2]	22,500	7,000	N5	N4
Norway	Svalbard	2000[1]	2017[2]	100	200	N5	N4
Greenland	Greenland	1995[1]	2016[2]	10,500	5,000	N5	N5
Canada							
Newfoundland and Labrador	Labrador	1978[6]	2014[23]	2,080	2,561	N3	N3
	Newfoundland	2002[24]	2017[23]	6,123	6,679	N3	N3
	Witless Bay	1979[4]	2012[4]	198	48	N1	N1
Quebec	Gaspé Peninsula	1989[5]	2018[6]	1,337	1,247	N2	N2
	Îles-de-la-Madeleine	1990[6]	2017[6]	1,211	626	N2	N2
	St Lawrence Estuary	1990[19]	2016[19]	2,623	782	N2	N2

(*Continued*)

Table 1. (*Continued*)

Country/State/Province	Region	Year 1	Year 2	Count 1	Count 2	Error 1	Error 2
	North Shore of Gulf of St. Lawrence	1988[21]	2015[6]	1,100	768	N2	N2
Nova Scotia	Cape Breton and mainland	1987[25]	2013[26]	50,767	7,711	N3	N3
	Sable Island	1970[16]	2013[16]	527	398	N2	N2
Prince Edward Island	Prince Edward Island	1986[24]	2019[23]	2,326	1,200	N3	N3
New Brunswick	Gulf of St Lawrence	1986[27]	2015[10]	1,134	1,541	N3	N3
	Bay of Fundy	1979[25]	2013[26]	743	1,258	N3	N3
Ontario	Great Lakes	1990[6]	2008[6]	10	100	N1	N1
United States							
Maine	Maine	1977[13]	2013[13]	9,846	6,934	N2	N2
Massachusetts	Massachusetts	1995[12]	2018[12]	14,746	3,658	N3	N3
New Jersey	New Jersey	1985[20]	2013[20]	226	2,113	N3	N3
New York	Long Island	2001[8]	2019[8]	3,918	1,728	N2	N2
Virginia	Virginia	1993[18]	2018[17]	514	1,123	N1	N1
North Carolina	North Carolina	1988[15]	2020[15]	3	122	N1	N1
Unsurveyed populations (Scenario A)	**Unsurveyed populations (Scenario A)**	1985[22]	2021[22]	29,100	29,100	NA	NA
Unsurveyed populations (Scenario B)	**Unsurveyed populations (Scenario B)**	1985[22]	2021[22]	29,100	15,100	NA	NA

[1]BirdLife International (2000)
[2]BirdLife International (2015)
[3]BirdLife International/European Bird Census Coucil (2020)
[4]Bond *et al.* (2016)
[5]Cotter and Rail (2007)
[6]Lock (1979)
[7]Fauchald *et al.* (2015)
[8]New York State Department of Environmental Conservation; unpubl. data
[9]Lorentsen (1994)
[10]Guzzwell *et al.* (2015)
[11]Hario and Rintala (2016)
[12]Massachusetts Division of Fisheries and Wildlife; unpubl. data
[13]Mittelhauser *et al.* (2016)
[14]Nager and O'Hanlon (2016)
[15]North Carolina Wildlife Resources Commission; unpubl. data
[16]Ronconi *et al.* (2016)
[17]Watts *et al.* (2019)
[18]Watts *et al.* (1998)
[19]Rail (2018)
[20]Washburn *et al.* (2016)
[21]Chapdelaine and Brousseau (1991)
[22]These population estimates were calculated as 10% of the global population in 1985. See "Data analysis" section.
[23]CWS unpubl. data (Wilhelm)
[24]Cotter *et al.* (2012)
[25]Lock (2003)
[26]Regular *et al.* (2015)
[27]Boyne *et al.* (2006).

account for a potential margin of error of up to 100% in some instances as we were confident no population estimate would be erroneous by more than twice its size (Table 2). For data from the USA, Canada, and some European countries, where raw population counts were available from count coordinators or published literature, error categories were either assigned by the count coordinators themselves or selected based on the methodology detailed in the literature. Counts from the remaining 12 European countries were obtained from BirdLife International European Red List species factsheets (2021). These data had already been allocated an accuracy/quality classification by BirdLife International according to their own criteria. We therefore translated these classifications into our study's error categories using a conservative approach whereby the highest possible degree of accuracy issued by BirdLife International ("Good - Based on reliable and complete or representative quantitative data") was allocated to the third error category in this study ("N3- Estimate of nests with an accuracy of 10–25%") (see Table 2 for conversion criteria). Such a conservative approach was taken to limit the potential to overestimate the accuracy of population trend estimates calculated from BirdLife International data.

Data analysis

All analyses were carried out in R version 4.0.4 (R Core Team 2020). A stochastic simulation was conducted for each population, which allowed us to account for the error associated with each population count to present confidence intervals around the estimated changes in population size. Random draws from the simulation provided two "estimated" counts per population, which were then used to calculate the finite rate of change of the population.

Simulation parameters were adapted from Lynch *et al.* (2013) to fit our data in the following way: A fractional error (representing

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Table 2. Error categories, BirdLife International criteria conversion, and the descriptions used to classify population counts in terms of uncertainty.

This study's criteria	Birdlife International criteria	Description
N1	NA	Nests are individually counted with an accuracy of 0-5%
N2	NA	Nests are individually counted with an accuracy of 5-10%
N3	Good ("Based on reliable and complete or representative quantitative data")	Estimate of nests with an accuracy of 10-25%
N4	Medium ("Based on reliable but incomplete or partially representative quantitative data")	Rough estimate of nests with an accuracy of 25-50%
N5	Poor ("Based on qualitative information, but no (or potentially unreliable/ unrepresentative) quantitative data")	Rough estimate of nests with an accuracy of 50-100%

the uncertainty of the count) was randomly sampled from a uniform distribution defined by the error category associated with each population count.

$$\begin{aligned} &(\mathrm{N1})E \sim Unif(0.00, 0.05)\\ &(\mathrm{N2})E \sim Unif(0.05, 0.10)\\ &(\mathrm{N3})E \sim Unif(0.10, 0.25)\\ &(\mathrm{N4})E \sim Unif(0.25, 0.50)\\ &(\mathrm{N5})E \sim Unif(0.50, 1.00) \end{aligned} \quad \text{(Equation 1)}$$

1. An estimated population count $(\widehat{n})$ was randomly sampled from a truncated normal distribution limited at 0 in the lower tail to avoid sampling negative population counts. This truncated normal distribution was defined by the original population count (n) and the randomly sampled error (E) from the previous step.

$$\widehat{n} \sim N\left(n, \left[\frac{E}{2}n\right]^2\right) \quad \text{(Equation 2)}$$

2. Equations 1 and 2 were bootstrapped 50,000 times for the two counts from each population. Using the estimated population counts in each year, we calculated the finite rate of population change (λ):

$$\lambda = \left(\frac{\widehat{n}_2}{\widehat{n}_1}\right)^{\frac{1}{Year2 - Year1}} \quad \text{(Equation 3)}$$

where *Year1* was the year of the first population estimate (n_1), and *Year2* was the year of the most recent estimate (n_2).

3. Lastly, the estimated population sizes in 1985 and 2021 were calculated as follows:
   a. The time gaps between n_1 and 1985, and n_2 and 2021 were calculated:

$$Gap_1 = Year1 - 1985 \quad \text{(Equation 4)}$$

$$Gap_2 = 2021 - Year2 \quad \text{(Equation 5)}$$

   b. Once the time gaps had been obtained, the fractional change ($Change_1$ and $Change_2$) in population size between $\widehat{n}_1$ and 1985 and $\widehat{n}_2$ and 2021 was then calculated as follows:

$$Change_1 = \lambda^{Gap_1} \quad \text{(Equation 6)}$$

$$Change_2 = \lambda^{Gap_2} \quad \text{(Equation 7)}$$

   c. Lastly, the 1985 start (*StartPopulation1985*) and the 2021 end populations (*EndPopulation2021*) were calculated, as well as the fractional change in population size over the three-generation period (*3GenPeriod*):

$$StartPopulation1985 = \frac{\widehat{n}_1}{Change1} \quad \text{(Equation 8)}$$

$$EndPopulation2021 = \widehat{n}_2 \times Change2 \quad \text{(Equation 9)}$$

$$3GenPeriod = \frac{(End\ Population\ 2021 - Start\ Population\ 1985)}{StartPopulation1985} \quad \text{(Equation 10)}$$

4. Equations 3 to 10 were repeated for each of the 50,000 bootstrapped population estimates to obtain distributions of estimates for each population for 1985 and 2021.

Due to the large uncertainty associated with the counts of some European countries and the need to use a normal distribution with a truncated lower limit of 0, the resulting distributions for some regions were marginally non-normal. To stop extreme, positive outliers from heavily influencing the mean statistics, counts outside the 2.5% and 97.5% quantiles were excluded in figures involving European data, which resulted in normal distributions.

There were populations for which no trend data were available (unsurveyed populations). These were: 27% of the Quebec population (Canada), approximately 40% of the New York population (USA), and the entirety of Delaware (USA). These figures accounted for <10% of our 1985 global population estimate. Two scenarios were used to include these figures in the analysis. The first approach (Scenario A) was conservative and assumed these populations had remained stable over the three-generation period. The second approach (Scenario B), likely more realistic, assumed unsurveyed populations declined at the same rate as the global average of the surveyed populations (Table 1). Mean statistics for global and continental estimates were calculated from the sum of every iteration from all populations that fell within that geographical area. Unsurveyed populations were only included in global estimates and were excluded from continental, national, and regional estimates. Finally, we compared overall population changes over three generations to IUCN Red List criterion A2, defined as "Population reduction observed, estimated, inferred, or suspected in the past where the causes of reduction may not have ceased OR may not be understood OR may not be reversible" (IUCN 2012). We did not apply criterion A1 because the causes of decline were not known, did not appear to have ceased, and were not reversible (see "Discussion" section). We also did not apply criterion A3 because we did not project populations into the future. Lastly, we did not apply criterion A4 because we used two count estimates per population and therefore a "moving window" scenario would not yield any additional decline estimates.

Table 3. Summary of changes in the abundance of Great Black-backed Gull pairs globally, in North America (Canada and USA), and in Europe (rest of countries from Table 1). Change is represented as the estimated total number of breeding pairs in 1985 and 2021, and the corresponding percentage change in population size and finite rate of change (λ) over that time. For global estimates, "Scenario A" assumes unsurveyed populations have remained stable between 1985-2021, whereas "Scenario B" assumes unsurveyed populations have declined at the same rate as the global average.

	Breeding pairs 1985	Breeding pairs 2021	Three-gen change (%)	Finite rate of change (λ)
Population	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)
Global (Scenario A)	291,093	165,469	−42.80	0.9844
	(261,037; 323,788)	(146,275; 186,768)	(−31.43; −52.93)	(0.9792; 0.9895)
Global (Scenario B)	290,921	151,595	−47.56	0.9820
	(260,884; 324,182)	(132,158; 172,642)	(−36.69; −57.31)	(0.9766; 0.9874)
North America	123,590	38,564	−68.68	0.9681
	(109,857; 137,487)	(36,188; 41,028)	(−63.94; −72.78)	(0.9644; 0.9720)
Europe	137,915	97,693	−28.06	0.9905
	(110,462; 169,920)	(78,641; 118,729)	(−49.74; −0.42)	(0.9811; 0.9999)

Because we had population trend data for virtually the entire range (90% of the global population), we assumed that population trends appropriately reflected the global population.

Results

a. Global population trend

Under "Scenario A" (unsurveyed populations assumed to have remained stable between 1985 and 2021), the number of Great Black-backed Gull pairs assessed in this study declined from approximately 291,000 (95% CI: 261,000; 324,000) to 165,000 (95% CI: 146,000; 187,000), a three-generation change of -42.8% (95% CI: -31.4%; –52.9%), and an annual decline of 1.2% (95% CI: 0.8%; 1.5%) (Table 3). Under "Scenario B" (unsurveyed populations assumed to have declined at the same rate as the global average), the number of Great Black-backed Gull pairs declined from approximately 291,000 (95% CI: 261,000; 324,000) to 152,000 (95% CI: 132,000; 173,000), a three-generation decline of 47.5% (95% CI: -36.7%; -57.3%), and an annual decline of 1.3% (95% CI: 1.0%; 1.6%). According to IUCN Red List criterion A2 (applicable when the population decline has not ceased), the results from both scenarios correspond to a listing of Vulnerable (a decline of ≥30% over 10 years or three generations, whichever is longer), and Scenario B approaches the threshold of an Endangered listing (decline of ≥50%). Indeed, the upper 95% CIs from both scenarios overlapped with an Endangered listing and the lower 95% CIs did not drop below a Vulnerable listing.

b. North American and European population trends

The three-generation decline was greater in North America than Europe (two-sample t-test: df =50,070, t = –772.37, $P < 0.001$), with a mean three-generation change of -68.7% (95% CI: –63.9%; –72.8%) compared to -28.1% (95% CI; –49.7%; –0.4%), respectively (Table 3).

c. Country and sub-country population trends

While there were populations that showed growth between 1985 and 2021, these tended to be smaller and/or were present within a larger state or country where most populations declined, such as the Bay of Fundy (75% increase) in Canada or New Jersey (1686% increase) in the United States. Generally, increases were recorded in populations at the edges of the species' range, or in areas that have been recently colonised such as Spain, Germany, Netherlands, North Carolina (United States) or Virginia (USA) (Figure 2). Overall, however, declines were recorded across most of the species' strongholds: Atlantic Canada, United States, United Kingdom, and Iceland. The exception was Norway, which apparently remained relatively stable, but large uncertainty surrounded the estimated change in abundance (Table 4; Figure 2).

Discussion

Globally, Great Black-backed Gull populations were estimated to have declined between 42.8% and 47.5% from 1985 to 2021, depending on the assumption regarding unsurveyed populations. This overall decline is consistent with trends observed in local and regional analyses from recent years (Mittelhauser *et al.* 2016, Nager and O'Hanlon 2016, Washburn *et al.* 2016, Wilhelm *et al.* 2016), though North American populations declined more steeply than European ones (-68.7% and -28.1% per annum, respectively).

Most Great Black-backed Gull populations increased for most of the 20th century until peaking sometime in its later decades. In areas where large-scale surveys were undertaken with some regularity, this peak was reached between the 1970s and 1990s (Cotter *et al.* 2012, Anderson *et al.* 2016, Mittelhauser *et al.* 2016). Following this peak, our results showed declines across most populations that historically held a large percentage of the global population: Atlantic Canada and the United States, Iceland, Sweden, United Kingdom, and Ireland. Norway, which supports almost half of the breeding European Great Black-backed Gulls, remained relatively stable over that period, though confidence intervals around estimates of λ dip into negative growth (95% CI: 0.9817; 1.0222), and most of the colonies that are monitored annually are in decline (Anker-Nilssen *et al.* 2021). Increasing trends were typically observed in countries and states with smaller populations, generally at the southern or northern edges of the species' range such as Germany, the Netherlands, Spain, Svalbard, North Carolina (United States), and Virginia (United States). France showed a

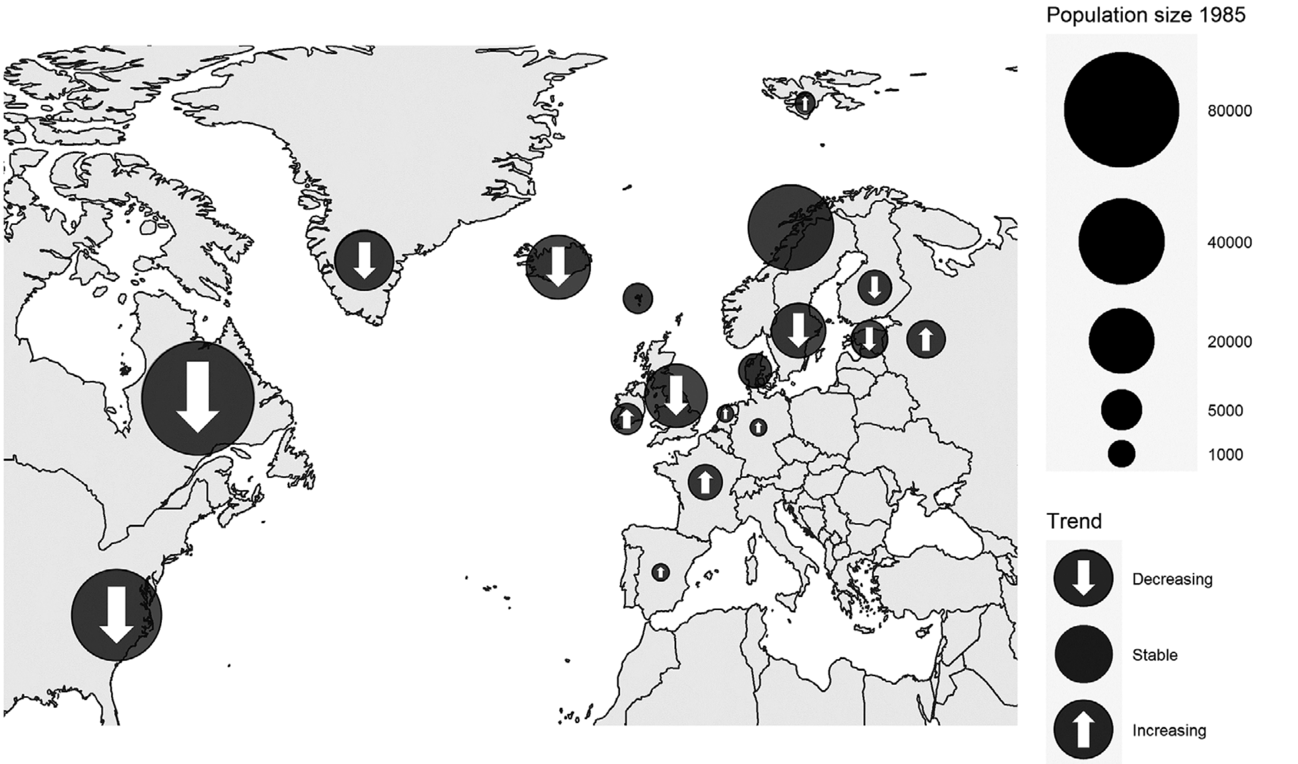

Figure 2. Bubble map showing the size and trends of Great Black-backed Gull populations per country (except Svalbard, which is shown separate to Norway) between 1985 and 2021. Circles reflect the size of populations and symbols reflect the population trend. Stable populations are depicted as those where the three-generation change has been less than 5% between 1985 and 2021.

remarkable increase from approximately 2,000 to 10,000 pairs. Increases seen at these locations could perhaps be due to birds emigrating from larger populations; Great Black-backed Gulls from Canadian colonies have replenished US populations in the past (Anderson *et al.* 2016).

Due to the vast distribution and inaccessibility of some Great Black-backed Gull populations, absolute counts of breeding pairs are rarely undertaken. Therefore, the number of available counts differed between countries, as did methodology and count accuracy. Using two population counts and standardising the uncertainty associated with each count following the five-point scale detailed in Lynch *et al.* (2013; Table 2) was the approach we used to factor in the limited count data and varying levels of accuracy. The main caveat of this approach was the potential to over- or underestimate 1985 and 2021 population estimates. Depending on how close the two population estimates per region were in time to the initial (1985) and final (2021) years of the most recent three-generation period, estimates could have been over- or underestimated. We minimized this bias by choosing the two counts closest to 1985 and 2021. Additionally, our estimated global population size for 1985 (291,000) was within the range estimated by BirdLife International in previous species assessments (283,000–403,000; BirdLife International 2015, Wetlands International 2015). Our approach also meant we were not able to provide finer-scale changes in abundance over time periods shorter than three generations. For example, Mittelhauser *et al.* (2016), reported an annual rate of change of -6.3% between 2008 and 2013 in Maine (USA, a figure much higher than the longer-term three-generation decline of -1% reported in this study. However, our focus was on linear change in abundance over a three-generation period in line with the IUCN Red List criteria which aimed to provide an overall change in abundance over a standardised time period.

The causes of decline remain largely unexplored. The foraging plasticity and overall adaptability of Great Black-backed Gulls is presumed to be a buffering mechanism against variable availability of resources (Coulson 2019). Identifying the individual causes of population change for the Great Black-backed Gull would be challenging due to existing knowledge gaps regarding their foraging ecology and population dynamics (lack of demographic rate data) (Bicknell *et al.* 2013). However, significant contributors to the above-mentioned declines of populations have been suggested. Reductions in the availability of fish discards may be a primary reason that some *Larus* populations have declined. In the North Sea, Great Black-backed Gulls have been identified as one of the species at risk of being impacted by an ongoing, gradual reduction in discards since 1990 and a newly implemented discard ban in European Union waters (Bicknell *et al.* 2013, Sherley *et al.* 2020). However, the impacts of these changes on North Sea Great Black-backed Gull populations remain largely unknown. In eastern Canada, a reduction in landed groundfish (which provided offal and discards) due to a moratorium in 1992 was correlated with a decline in the number of Great Black-backed Gull pairs in the following years (Cotter *et al.* 2012). Conversely, some populations forage mostly on naturally occurring habitats and food such as seabirds, mammals, intertidal organisms, and forage fish (Harris 1980, Borrmann *et al.* 2019, Westerberg *et al.* 2019) and their declines are likely unrelated to changes in anthropogenic fish and waste management. Between the 1970s and 2000, delayed spawning of capelin *Mallotus villosus* was correlated with lower breeding success in several seabird species in Canada, including Great

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Table 4. Changes in Great Black-backed Gull abundance over the three-generation period 1985-2021 per population. Change is represented by the estimated number of breeding pairs at the beginning and end of the period, and the corresponding percentage change in the number of breeding pairs and finite rate of change (λ).

Population	Breeding pairs 1985 Mean (95% CI)	Breeding pairs 2021 Mean (95% CI)	Three-gen change (%) Mean (95% CI)	Finite rate of change (λ) Mean (95%CI)
Europe				
Russia	3,209	4,473	79.77	1.0106
	(1,256; 6,267)	(2,321; 7,227)	(−59.86; 432.17)	(0.975; 1.0475)
Estonia	10,117	842	−90.34	0.9340
	(4,853; 17,169)	(495; 1,238)	(−96.83; −77.04)	(0.9086; 0.9599)
Finland	2,866	1,372	−50.40	0.9796
	(1,937; 3,806)	(874; 1,901)	(−72.83; −16.25)	(0.9645; 0.9951)
Sweden	19,076	7,517	−52.75	0.9756
	(8,315; 34,573)	(4,497; 10,887)	(−85.80; 18.16)	(0.9472; 1.0046)
Germany	9	163	1,797.47	1.0851
	(8;10)	(149; 178)	(1,478.30; 2,161.81)	(1.0797; 1.0905)
Denmark	2,627	2,521	12.87	1.0000
	(1,192; 4,651)	(1,564; 3,562)	(−62.98; 165.10)	(0.9728; 1.0275)
Norway	39,972	42,848	14.04	1.0018
	(25,080; 56,003)	(25,172; 62,575)	(−48.65; 120.07)	(0.9817; 1.0222)
Netherlands	3	132	4,340.21	1.1110
	(3;3)	(120; 144)	(3,501.02; 5,293.34)	(1.1047; 1.1171)
France	2,097	10,072	383.50	1.0446
	(1,786; 2,437)	(8,833; 11,389)	(267.30; 528.26)	(1.0368; 1.0524)
United Kingdom	18,510	12,239	−33.87	0.9886
	(17,640; 19,381)	(11,362; 13,117)	(−38.48; −29.07)	(0.9866; 0.9905)
Spain	1	29	2,866.25	1.0986
	(1;1)	(26; 31)	(2,308.88; 3,504.74)	(1.0924; 1.1047)
Republic of Ireland	1,664	3,260	96.99	1.0189
	(1,459; 1,884)	(3,016; 3,504)	(63.8; 134.53)	(1.0138; 1.0240)
Faroe Islands	1,315	1,272	52.92	1.0006
	(291; 2,883)	(362; 2,456)	(−84.14; 565.89)	(0.9501; 1.0541)
Iceland	18,892	6,097	−64.59	0.9681
	(8,292; 29,274)	(2,047; 10,675)	(−89.07; −10.48)	(0.9404; 0.9969)
Svalbard	66	245	671.65	1.0421
	(9; 189)	(76; 451)	(−52.47; 3,876.30)	(0.9796; 1.1077)
Greenland	16,454	4,369	−59.05	0.9655
	(3,734; 36,009)	(1,328; 8,095)	(−95.37; 67.14)	(0.9182; 1.0144)
Canada				
Labrador	2,162	2,671	23.94	1.0058
	(1,890; 2,431)	(2,196; 3,163)	(−0.60; 52.91)	(0.9998; 1.0119)
Newfoundland	5,627	6,852	27.50	1.0059
	(3,793; 7,853)	(5,565; 8,214)	(−25.85; 106.20)	(0.9917; 1.0203)
Witless Bay	142	18	−87.38	0.9441
	(139; 145)	(17;18)	(−87.38; −86.89)	(0.9431; 0.9451)

(*Continued*)

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Table 4. (*Continued*)

Population	Breeding pairs 1985	Breeding pairs 2021	Three-gen change (%)	Finite rate of change (λ)
	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)	Mean (95%CI)
Gaspé Peninsula	1,353	1,238	−8.36	0.9975
	(1,252; 1,455)	(1,151; 1,325)	(−18.42; 2.62)	(0.9944; 1.0007)
Îles-de-la-Madeleine	1,369	568	−58.43	0.9759
	(1,266; 1,472)	(527; 610)	(−63.14; −53.25)	(0.9727; 0.9791)
St Lawrence Estuary	3,311	620	−81.24	0.9545
	(3,061; 3,564)	(573; 667)	(−83.45; −78.8)	(0.9513; 0.9578)
North Shore Gulf of St. Lawrence	1,145	709	−37.94	0.9868
	(1,065; 1,226)	(655; 765)	(−45.03; −30.24)	(0.9835; 0.9900)
Cape Breton and mainland Nova Scotia	58,674	4,328	−92.56	0.9301
	(49,411; 68,111)	(3,479; 5,229)	(−90.10; −94.52)	(0.9225; 0.9378)
Sable Island	478	378	−20.89	0.9935
	(455; 500)	(349; 406)	(−26.62; −14.85)	(0.9914; 0.9955)
Prince Edward Island	2,608	1,147	−55.59	0.9775
	(2,151; 3,071)	(964; 1,333)	(−66.30; −42.50)	(0.9702; 0.9847)
Gulf of St Lawrence	1,035	1,620	59.34	1.0125
	(850; 1,229)	(1,319; 1,932)	(14.94; 113.13)	(1.0039; 1.0212)
Bay of Fundy	820	1,432	75.30	1.0155
	(716; 923)	(1,170; 1,704)	(39.37; 118.12)	(1.0093; 1.0219)
Great Lakes	5	528	9,906.54	1.1365
	(5; 5)	(503; 552)	(9,221.98; 10,630.27)	(1.1342; 1.1387)
United States				
Maine	9,105	6,415	−29.50	0.9903
	(8,639; 9,571)	(5,913; 6,922)	(−35.64; −22.95)	(0.9878; 0.9928)
Massachusetts	27,129	3,055	−88.54	0.9412
	(21,288; 33,408)	(2,540; 3,579)	(−91.90; −84.27)	(0.9326; 0.9499)
New Jersey	226	4,009	1,686.4	1.0831
	(193; 260)	(3,240; 4,812)	(1,246.40; 2,225.33)	(1.0749; 1.0913)
New York (Long Island)	8,114	1,578	−80.54	0.9555
	(7,705; 8,530)	(1,535; 1,621)	(−81.84; −79.15)	(0.9537; 0.9574)
Virginia	400	1,234	208.25	1.0318
	(387; 414)	(1,200; 1,267)	(193.14; 223.89)	(1.0303; 1.0332)
North Carolina	2	137	6,364.32	1.1228
	(2; 2)	(134; 140)	(6,115.85; 6,619.99)	(1.1216; 1.1240)
Unsurveyed (Scenario A)	29,100	29,100	0.00	1.0000
	(29,100; 29,100)	(29,100; 29,100)	(0.00; 0.00)	(1.0000; 1.0000)
Unsurveyed (Scenario B)	29,100	15,100	−47.56	0.9820
	(29,100; 29,100)	(15,100; 15,100)	(−47.56; −47.56)	(0.9820; 0.9820)

Black-backed Gulls (Stenhouse and Montevecchi 1999, Davoren and Montevecchi 2003, Regehr and Rodway 1999). In Finland, where a three-generation decline of 50% occurred, the eutrophication of water bodies and culling could be contributory factors (Hario and Rintala 2016). Lastly, predation can directly or indirectly impact populations. In North America, Bald Eagles *Haliaeetus leucocephalus*, whose populations are currently recovering, depredate Herring and Great Black-backed Gulls and exclude them from areas near eagle nest sites (White *et al.* 2006, Hipfner *et al.* 2012). Overall, the drivers of change remain largely unknown

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across most of the Great Black-backed Gull populations reported in this study and the causes of decline appear not have ceased; see annually monitored Norwegian (Anker-Nilssen *et al.* 2021) and United Kingdom populations (Seabird Monitoring Program 2022). It is likely that changes in abundance are driven by several factors acting simultaneously upon populations due to the widely acting nature of human threats.

We estimated that Great Black-backed Gulls declined globally in the range of 43% to 48% between 1985 and 2021. We suggest that uplisting Great Black-backed Gulls to 'Vulnerable' on the IUCN Red List is appropriate and further assessments should be carried out regularly since it is approaching an 'Endangered' listing. Causes of decline remain largely unknown and unexplored for most populations. Further research should focus on reducing knowledge gaps to support a better understanding of factors influencing population trends and status across its range. A key knowledge gap is the lack of estimates of demographic parameters. These data, in particular survival and productivity, are key to understanding the drivers and mechanisms of population change and can be used to model population trajectories and inform targeted conservation action. Additionally, improved and consistent monitoring of Great Black-backed Gull populations (including standardisation of survey methodologies, more frequent population counts, and coordination between countries) would allow for more accurate assessments of population size, status, and trends.

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