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Endurance or extinction: long-term declines in albatrosses at South Georgia highlight threats from South Atlantic fisheries and climate change

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ABSTRACT: Marine ecosystems face multiple human threats, and many species are declining. The wandering albatross *Diomedea exulans*, black-browed albatross *Thalassarche melanophris* and grey-headed albatross *T. chrysostoma* are categorised globally as Vulnerable, Least Concern and Endangered, respectively, by the IUCN. The populations at South Georgia are listed by the Agreement on the Conservation of Albatrosses and Petrels as High Priority Populations for conservation, and to determine their current status and trends, we surveyed all breeding sites of wandering albatrosses, and ~30% and ~73% of black-browed and grey-headed albatrosses, respectively. Comparisons with previous surveys indicated considerable variation in trends among sites and slower rates of decline from 2014/2015 to 2023/2024 than from 2003/2004 to 2014/2015: wandering albatross −0.1 vs. −1.7% yr^{-1}; black-browed albatross −1.1 vs. −1.8% yr^{-1}; grey-headed albatross: −4.1 vs. −5.0% yr^{-1}. Updated population estimates for South Georgia were 1278, 55 119 and 18 475 breeding pairs of wandering, black-browed and grey-headed albatrosses, comprising 13.3, 7.6 and 28.7% of revised global totals and reflecting major declines of 39, 46 and 66%, respectively, in just 32 to 40 yr. The main threats are bycatch in fisheries outside South Georgia waters and climate change, including the southerly shift of Antarctic krill *Euphausia superba* for the *Thalassarche* species. There are no current terrestrial threats other than highly pathogenic avian influenza (HPAI) for wandering albatrosses. Addressing bycatch is therefore a clear management priority, which needs to overcome the main barriers of weak governance, reluctance to mandate best-practice bycatch mitigation and poor monitoring and enforcement of compliance.

KEY WORDS: Seabird conservation · Bycatch · Fisheries management · Long-term monitoring · Climate change

1. INTRODUCTION

Human activities pose multiple threats to marine vertebrates, including seabirds, marine mammals, turtles and sharks (Avila et al. 2018, Clay et al. 2019, Pimiento et al. 2020, Dulvy et al. 2021). Conserving these species is challenging because many are highly mobile, with distributions that extend across multiple jurisdictions, and so cooperation is required among diverse stakeholders that often have conflicting economic, social and political interests (Lewison et al. 2018, Beal et al. 2021). Nevertheless, the current international focus on the plight of the oceans indicates growing political will and a realisation that functioning ecosystems provide direct and indirect services to humanity that have widespread socio-

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economic benefits as well as the capacity to improve quality of life (Sumaila et al. 2015, Glaum et al. 2020).

Pelagic seabirds are amongst the most threatened of all marine vertebrates, related to their delayed maturity and low reproductive output, such that small increases in adult mortality due to human activities lead to population decreases from which it can take decades to recover (Rolland et al. 2008, Tuck et al. 2011). The main threats are invasive alien species, incidental mortality (bycatch) in fisheries and climate change or severe weather (Dias et al. 2019, Phillips et al. 2023). Invasive vertebrates have been eliminated from 100s of islands worldwide, but challenges remain in particular in the tropics and on inhabited islands (Spatz et al. 2022). There are effective technical and operational measures to address bycatch, but implementation is patchy and dependent on good fisheries governance (Good et al. 2020, Baker et al. 2024). Other threats, such as climate change, are much harder to mitigate because of the concerted changes needed by far more people and authorities at a global scale. Moreover, new threats are emerging, including the H5N1 strain of highly pathogenic avian influenza (HPAI), which originated from commercial poultry farms in 1996 and has had catastrophic impacts on some seabird species in recent years (Leguia et al. 2023, Pohlmann et al. 2023, Bennison et al. 2024, Lane et al. 2024).

South Georgia holds globally important populations of many seabirds, including the wandering albatross *Diomedea exulans*, black-browed albatross *Thalassarche melanophris* and grey-headed albatross *T. chrysostoma* (Poncet et al. 2017), which are listed as Vulnerable, Least Concern and Endangered, respectively, by the International Union for the Conservation of Nature (IUCN) (BirdLife International 2025). These species have circumpolar breeding distributions, and in each case, there were at least 3 island groups which held >10% of global numbers in the mid 2010s (Phillips et al. 2016). Trends vary, but in general are stable or increasing; the exceptions are at South Georgia, where the steep long-term declines of these globally important populations prompted their inclusion as 3 of 9 High Priority Populations for conservation by the Agreement on the Conservation of Albatrosses and Petrels (ACAP) (Phillips et al. 2016).

Previous surveys at South Georgia revealed considerable variation in trends across sites for all 3 species (Poncet et al. 2006, 2017). The main drivers in the wandering albatross population appear to be spatial segregation in at-sea distributions, greater overlap with fisheries and likely higher bycatch rates of adults at the most populous and westernmost breeding site, Bird Island (Warwick-Evans et al. in press). Differences among sites in breeding success may also be a factor (Rackete et al. 2021). Our aims were to (1) update the trends and status of the 3 ACAP High Priority Populations at South Georgia by comparing results of surveys in austral summer 2023/2024 with previous years, (2) revise the global totals for these species, (3) contrast the likely drivers with those affecting populations at other island groups and (4) discuss the implications and priorities for conservation management.

2. MATERIALS AND METHODS

2.1. Wandering albatross *Diomedea exulans* survey

Between 16 January and 12 February 2024, all breeding sites occupied by wandering albatrosses identified in 3 previous surveys at South Georgia (53° to 55° S, 34° to 42° W) (Clark 1984, Poncet et al. 2006, 2017), south Atlantic Ocean, were resurveyed during the early incubation period (Fig. 1). The largest sites were surveyed on foot, and nest GPS location, nearby vegetation cover, seal abundance and other data were recorded *in situ* using a bespoke 'Albatross Survey' app. As HPAI was confirmed prior to the survey, contact with birds was minimised and all nests with an adult in incubating position were assumed to contain an egg. Counts of some distant sites where boat landing or access on foot was problematic were made from vantage points on land or a vessel using binoculars, or using a DJI Mavic 2 Pro unmanned aerial vehicle (UAV) flown at 60 to 100 m above ground level.

Long-term monitoring of albatrosses has taken place at Bird Island for several decades, and annually since the early-mid 1970s (Pardo et al. 2017). This includes an intensive study of wandering albatrosses in an area on Wanderer Ridge (n = 100 nests, corresponding to 14.3% of the Bird Island total in 2023/2024). Here, nests are visited daily throughout the laying period (December to early January) to record eggs, and weekly thereafter to record breeding failures. All other areas on Bird Island are visited several times in December and January to mark nests. All of Bird Island was checked for active nests on 31 January 2024, and the total corrected for the proportion of nest failures by that date in the intensive study plot. Numbers of wandering albatrosses found dead prior to or during the ground survey — almost all of which would have resulted from HPAI infection — were proportionately higher on Bird Island than at other sites in the island group (55 vs. 8 adults; A. Bennison &

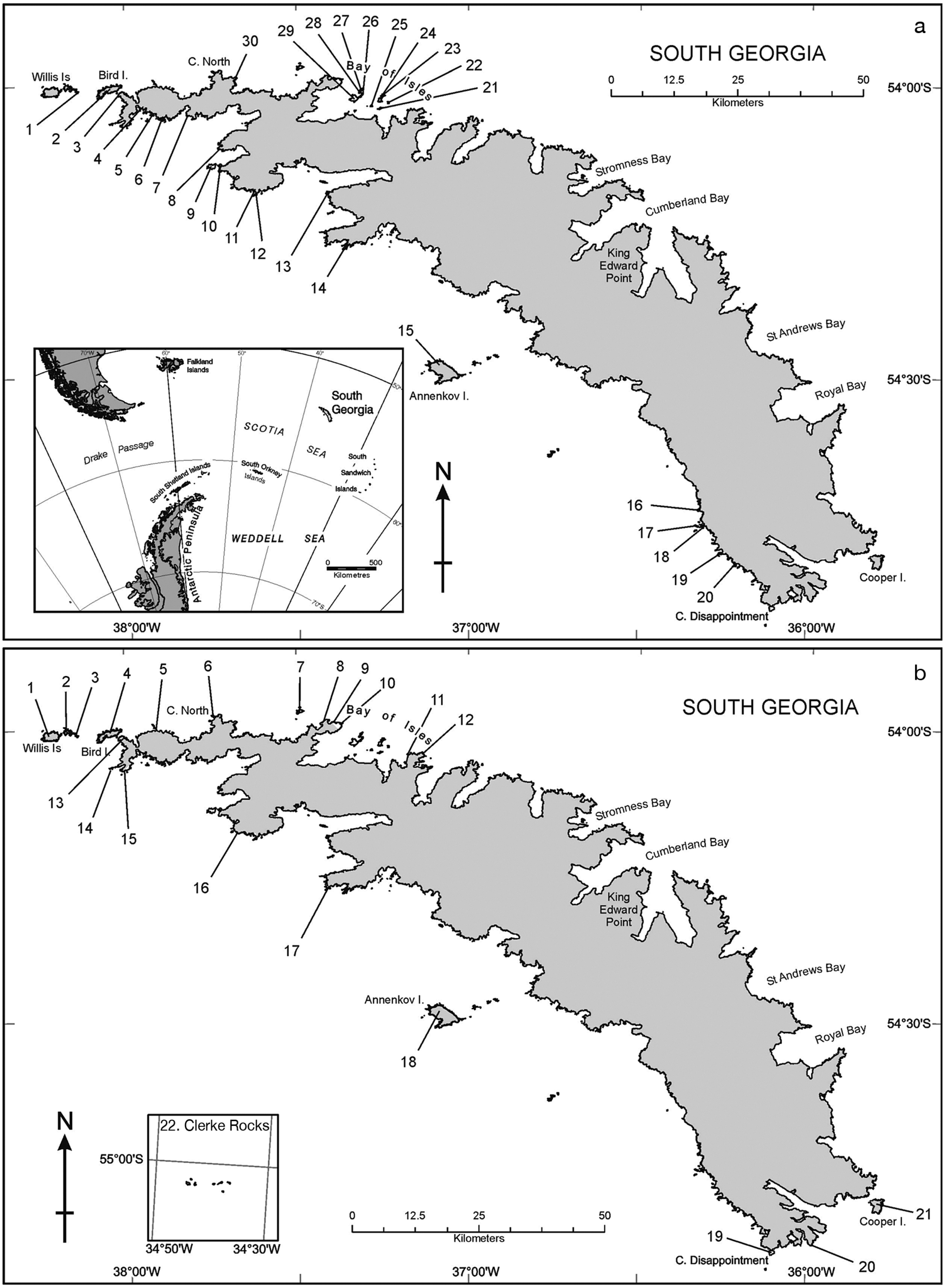

Fig. 1. Breeding sites of (a) wandering albatrosses, and (b) black-browed and grey-headed albatrosses surveyed at South Georgia in 2023/2024. Numbers refer to sites listed in Tables 1–3. Maps redrawn from Poncet et al. (2017)

R. A. Phillips unpubl. data); as such, the counts at all other sites were corrected using the mean failure rate from the 5 previous breeding seasons (2018/2019 to 2022/2023) at Bird Island, which was considered to be more applicable.

Corrected counts in 2023/2024 were compared with those from previous surveys presented in Poncet at al. (2006, 2017), but a different correction for previous failures was applied to chick counts from 1983/1984 in Clark (1984). Poncet et al. (2006) applied a correction factor of 1.571 based on the failure rate on Wanderer Ridge from egg laying to 1 October 1984. However, as this rate was considerably higher than on the rest of Bird Island in that season, in the current study we applied a more representative correction factor based on the failure rates on Wanderer Ridge by 1 October over the 5 breeding seasons from 1982/1983 to 1985/1986. There was no survey at Annenkov Island (Fig. 1) in 2014/2015 and so the count there was estimated from the mean trend between the previous and following surveys at all sites other than Bird Island, where the decrease was higher than elsewhere.

2.2. Black-browed albatross *Thalassarche melanophris* and grey-headed albatross *T. chrysostoma* survey

At South Georgia, both species breed at 7 sites, and either black-browed albatrosses or grey-headed albatrosses breed at 20 and 9 sites, respectively. In summer 2003/2004, all breeding sites were surveyed (Poncet et al. 2006), but in 2014/2015, only a subset of 13 sites were surveyed that had held 30.3 and 72.5% of the total populations of black-browed and grey-headed albatrosses, respectively, in the earlier census (Poncet et al. 2017). This same subset of sites was surveyed in 2023/2024.

At Bird Island, nests with an incubating bird were counted on the ground (3 November), or from vantage points with binoculars or telescope (grey-headed albatrosses on 10 to 18 November and black-browed albatrosses on 17 to 21 November), and the means of 3 counts (which were always within 5%) for each colony used in further analyses. These counts were corrected to the original number of breeding pairs using the reciprocals (1.10 and 1.047) of the proportions of nests that had failed by the same dates in 2 intensive-study colonies which were visited daily throughout the laying period; Colony J (116 pairs of black-browed albatrosses) and Colony E (45 pairs of grey-headed albatrosses).

Colonies not on Bird Island, and a few colonies on Bird Island that were not completely visible from vantage points on land, were photographed from 21 to 30 November 2023 between 08:00 and 20:00 h UTC from a ship or rigid inflatable boat (RIB) at a mean distance of 350 m, using a Nikon D610 camera with a Nikon 28–300 mm f3.5–5.6 lens, or, for 4 sub-colonies only, a Sony A7II camera with Sony 24–105 mm f4 and Sigma 100–400 mm f5–6.3 lenses. Multiple close-up photographs (focal length 300 mm) were taken in RAW (for lossless adjustment) and jpeg formats at a minimum shutter speed of 1/2000 s and underexposed to retain highlight detail needed to separate the species. The jpeg images were stitched using Microsoft Image Composite Editor software (v 2.0.3; softonic 2024), and counted manually using DotDotGoose (v 1.7.0; Ersts 2023). The unstitched photographs were used to check for errors and to confirm species in some cases. Albatrosses were recorded to species or labelled as unidentified when there was insufficient detail of the head or bill. The count of unidentified birds was then assigned to species assuming the same ratio as those identified in that colony. Birds in all images were counted twice, using vertical then horizontal sweeps to ensure complete coverage. All vertical sweeps were completed before the horizontal sweeps to maximise the experience and consistency of counters.

Photo-counts were corrected for diurnal variation in attendance of non-breeding birds (of any age) and failed breeders as in Poncet at al. (2006, 2017). This was based on data from Bird Island as there were no dead adult black-browed or grey-headed albatrosses (from any cause, including HPAI) seen anywhere at South Georgia before the end of the survey period. For the diurnal correction, study Colony F1, which held 100 pairs of black-browed albatrosses, was photographed daily using a Canon 90D camera with a Canon 55–200 mm f4.5–5.6 lens at 2 h intervals between 08:00 and 20:00 h UTC, between 20 November and 4 December. Birds in clear photographs (i.e. not obscured by mist or rain) were counted, providing 10 to 15 photo-counts for each 2 h interval from which mean attendance was calculated relative to peak attendance at solar noon. The means ranged from 0.9361 to 0.9866, and the reciprocal was used to standardise all photo-counts to 12:00 h UTC. To correct for the presence of non-breeding and failed birds, and account for previous nest failures, albatrosses in 10 colonies at Bird Island were counted in photographs taken at the start, middle and end of the survey period (21 and 27 November and 4 December). These were corrected for diurnal variation in attendance as above, and the relationship with the original number of breeding pairs was determined using analysis of covariance, with date and species included as independent variables. As the effects of date and species were not significant, a single correction was used for photo-counts by applying the following equation:

$$\text{Total breeding pairs} = 0.902\ (\pm 0.009\ \text{SE}) \times (\text{photocount standardised to 12:00 h UTC}) \quad (1)$$

Consistency in counting of birds in photographs between 4 observers in 2023 was tested by comparing totals for 12 colonies using the Kruskal-Wallis rank sum test in R version 4.3.2 (R Core Team 2023). These colonies were selected at random and represented the range of colony sizes and conditions. To check for consistency with the previous 2 surveys, counts of albatrosses in 35 images from 16 colonies in 2003, and 14 images from 11 colonies in 2014 (when there was higher snow cover) were compared with the original photo-counts using non-parametric Wilcoxon rank sum tests in R version 4.3.2 (R Core Team 2023). The overall error for the photo-counts in 2023 was estimated by comparing the corrected photo-counts from the ship or RIB to the corrected ground counts on Bird Island for 13 colonies (2674 pairs) of black-browed albatrosses and 3 colonies (226 pairs) of grey-headed albatrosses.

2.3. Spatial analysis

To assess whether there was spatial clustering in population trends, i.e. rates were more or less similar at neighbouring breeding sites, Moran's I statistic, which is an index of global spatial autocorrelation, was calculated for the trend data in an equal area projection in ArcMap 10.6. A significant positive Moran's I indicates spatial clustering, a significant negative value indicates spatial dispersion and a near zero value indicates that no spatial structure is detected. In addition, possible linear relationships within each site between the trends in 2003/2004 to 2014/2015 vs. 2014/2015 to 2023/2024 were tested using Pearson correlations.

3. RESULTS

3.1. Wandering albatross *Diomedea exulans*

Counts of wandering albatrosses in the all-islands census of South Georgia in 2023/2024 summed to 1278 breeding pairs (Table 1). There was considerable variability among breeding sites in the magnitude and direction of the recent population trend; at the major sites (≥20 pairs in 2014/2015), rates were −1.1 (Bird Island), 2.3 (Cape Alexandra), 1.9 (Saddle Island), −0.3 (Prion Island) and 0.2% yr^{-1} (Albatross Island). Annenkov Island was not surveyed in 2014/2015, but the count in 2023/2024 was very similar to 2003/2004. There was no indication that trends were more similar at neighbouring sites that held ≥10 pairs in 2014/2015 (Moran's I = 0.350, p = 0.124), and nor were the rates consistent in the intervals between the 2 surveys within these sites (Pearson correlation $r_8 = -0.165$, p = 0.651).

Including the interpolated count at Annenkov Island, the overall annual trend at South Georgia between 2014/2015 and 2023/2024 was −0.1% yr^{-1}, which is considerably better than the trends between previous censuses (−1.5% yr^{-1} from 1983/1984 to 2003/2004, and −1.7% yr^{-1} from 2003/2004 to 2014/2015) (Table 1). Nevertheless, the total population of wandering albatrosses at South Georgia including Annenkov Island has decreased in 40 yr by 39.3% (−1.2% yr^{-1}); from 2105 pairs in 1983/1984 to 1278 pairs in 2023/2024 (Table 1).

3.2. Black-browed albatross *Thalassarche melanophris*

Trends in numbers of black-browed albatrosses at the 12 breeding sites at South Georgia surveyed in 2023/2024 and 2014/2015 varied from −4.7 to 3.3% yr^{-1} (Table 2). There was no obvious spatial pattern in the trends (Moran's I = −0.103, p = 0.641), and nor were the rates consistent within sites in the intervals between these 2 surveys (Pearson correlation r_{10} = 0.160, p = 0.620). Numbers at this subset of sites summed to an estimated 16 725 breeding pairs, indicating a decrease of 1.1% yr^{-1} since 2014/2015, which is slower than the decrease of 1.8% yr^{-1} from 2003/2004 to 2014/2015. Applying the trend in this subset of colonies to those not counted since 2003/2004, and assuming they represented the same proportion (30.3%) of the total, the population of black-browed albatrosses at South Georgia has decreased from 101 488 pairs in 1984/1985–1991 to 55 119 pairs in 2023/2024, which is a decrease of 45.7% (−1.6 to −1.3% yr^{-1}) in 33 yr.

3.3. Grey-headed albatross *Thalassarche chrysostoma*

Trends in numbers of grey-headed albatrosses at the 6 breeding sites at South Georgia surveyed in 2023/2024 and 2014/2015 varied from −2.4 to −8.4% yr^{-1}

Table 1. Population sizes (breeding pairs; n) and trends of wandering albatrosses at South Georgia between surveys in 1983/1984, 2003/2004, 2014/2015 and 2023/2024. The 1983/1984 data are from Clark (1984), adjusted as in Poncet at al. (2006) to include 31 failed nests observed at Cape Alexandra and using a revised correction factor for previous failures (see Section 2.1). The 2003/2004 and 2014/2015 counts are from Poncet et al. (2006, 2017). The locations of breeding sites are shown in Fig. 1. Survey methods as follows: G: ground count; Vpt: vantage point on land; S: from survey vessel; UAV: unmanned aerial vehicle

No.	Breeding site	1983/1984 (n)	2003/2004 (n)	Change 1983/1984 to 2003/2004 (% yr^{-1})	2014/2015 (n)	Change 2003/2004 to 2014/2015 (% yr^{-1})	2023/2024 (n)	Change 2014/2015 to 2023/2024 (% yr^{-1})	Survey method in 2023/2024
1	Proud Island		6		3	−6.1	1	−11.5	S
2	Bird Island	1370	948	−1.8	772	−1.8	699	−1.1	G
3	Cape Alexandra	49	40	−1.0	35	−1.2	43	2.3	G/S
4	Coal Harbour	16	16	0.0	18	1.1	37	8.3	G
5	Frida Hole	8	6	−1.4	3	−6.1	3	0.0	G
6	Chaplin Head	3	0	−100.0			0		S
7	Weddell Point	41	10	−6.8	10	0.0	19	7.4	G/UAV
8	Kade Point	35	23	−2.1	10	−7.3	17	6.1	G
9	Saddle Island	46	40	−0.7	32	−2.0	38	1.9	G
10	Laws Point (Demidov Isthmus)		2		1	−6.1	1	0.0	G
11	Granat Point (Bomford Peninsula)	20	15	−1.4	7	−6.7	6	−1.7	G
12	Tidespring Island (Samuel Island)	7	1	−9.3	5	15.8	2	−9.7	UAV
13	Cape Rosa	11	4	−4.9	4	0.0	2	−7.4	G
14	Nuñez Peninsula	0	3	–	1	−9.5	1	0.0	G
15	Annenkov Island	228	193	−0.8	[166][a]		190		G
16	Diaz Cove North		0		0				
17	Diomedea Island (Kupriyanov Island outer)	7	5	−1.7	9	5.5	4	−8.6	G
18	Poncet Island (Kupriyanov Island inner)	1	0	−100.0	0	0.0	0	0.0	UAV
19	Ranvik	1	0	−100.0			1		S
20	Trollhul	8	3	−4.8	2	−3.6	2	0.0	G
21	Inner Lee	8	9	0.6	15	4.8	9	−5.5	G
22	Outer Lee	24	9	−4.8	3	−9.5	12	16.7	G
23	Skua Island	0	0	0.0	0	0.0	2		G
24	Prion Island	52	43	−0.9	37	−1.4	36	−0.3	G
25	Petrel Island	0	1		0	−100.0	0	0.0	UAV
26	Invisible Island	1	1	0.0	1	0.0	0	−100.0	UAV
27	Mollyhawk Island	5	3	−2.5	1	−9.5	7	24.1	G
28	Crescent Island	8	15	3.2	11	−2.8	4	−10.6	G
29	Albatross Island	148	155	0.2	139	−1.0	141	0.2	G
30	Nameless Point	8	2	−6.7	0	−100.0	0	0.0	G
31	Mirnyy Island (Trollhul North)						0		Vpt
32	Lazarev Island (Kupriyanov Islet)				0		0	0.0	UAV
33	Nilse Hullet						0		S
34	Aucellina Point						1		Vpt/UAV
	Total excl. Annenkov Island	**1877**	**1360**	**−1.6**	**1119**	**−1.8**	**1088.0**	**−0.3**	
	Total South Georgia	**2105**	**1553**	**−1.5**	**1285**	**−1.7**	**1278**	**−0.1**	

[a]No ground survey took place, so the count is estimated from the mean trend from the previous and following surveys at all sites other than Bird Island (see Section 2.1)

(Table 3). There was no obvious spatial patterns in the trends (Moran's $I = -0.260$, $p = 0.587$), and nor were the rates consistent within sites in the intervals between these 2 surveys (Pearson correlation $r_4 = -0.751$, $p = 0.085$). Numbers at this subset of sites summed to an estimated 13 397 breeding pairs, indicating a decrease of 4.1% yr^{-1} since 2014/2015, which is slower than the decrease of 5.0% yr^{-1} from 2003/2004 to 2014/2015. Applying the trend in this subset of colonies to those not counted since 2003/2004, and assuming they represented the same proportion (72.5%) of the total, the population of grey-headed albatrosses at South Georgia has decreased from 54 752 pairs in 1984/1985–1992 to an estimated 18 475 pairs in 2023/2024, which is a decrease of 66% (−3.9 to −4.7% yr^{-1}) in 32 yr.

Table 2. Population sizes and (breeding pairs; n) trends of black-browed albatrosses at different sites at South Georgia between surveys in 1984/1985–1991, 2003/2004, 2014/2015 and 2023/2024. The 1984/1985–1991, 2003/2004 and 2014/2015 counts are from Poncet et al. (2006, 2017). The locations of breeding sites are shown in Fig. 1

No.	Site	1984/ 1985–1991 (n)	2003/2004 (n)	Change 1985–1992 to 2003/2004 (% yr^{-1})	2014/2015 (n)	Change 2003/2004 to 2014/2015 (% yr^{-1})	2023/2024 (n)	Change 2014/2015 to 2023/2024 (% yr^{-1})
1	Main Island	[a]	14 559					
2	Trinity Island	[a]	13 960					
3	Hall Island	[a]	0					
1–3	**Willis Islands**	**33 570[a]**	**28 519**	**−0.9**				
4a	Bird Island (subset)[b]	5606	3192	−3.9	2714	−1.5	2219	−2.2
4b	Bird Island (total)	14695	8264	−4.3			5767	
5	Sorn & Bernt	84	74	−0.7	60	−1.9	54	−1.1
6	Cape North	2050	1546	−1.5	1642	0.5	1284	−2.7
7	Welcome Islands	250	188	−1.6	152	−1.9	184	2.1
8	Sheathbill Bay	[c]	481		345	−3.0	352	0.2
9	Sitka Bay	[c]	816		588	−2.9	544	−0.9
10	Cape Buller	[c]	177		93	−5.7	60	−4.7
8–10		**1630[c]**	**1474**	**−0.6**				
11	Cape Wilson	[d]	205		200	−0.2	197	−0.2
12	Cape Crewe	[d]	42		31	−2.7	30	−0.3
11–12			**322[d]**	**247**	**−1.5**			
13	Paryadin Peninsula North	[e]	1428		1079	−2.5	883	−2.2
14	Jomfruene	[e]	0		0	0.0	0	0.0
15	Paryadin Peninsula South	[e]	3789		2802	−2.7	3764	3.3
13–15			**6466[e]**	**5217**	**−1.1**			
16	Klutschak Point	850	784	−0.4				
17	Cape Nunez	962	981	0.1				
18	Annenkov Island	17 500	9398	−3.4				
19	Green Island	7924	3404	−4.6				
20	Rumbolds Point	2000	2340	0.9				
21	Cooper Island	11 785	10 606	−0.6	8772	−1.7	7154	−2.2
22	Clerke Rocks	1400	1254	−0.8				
	Total South Georgia subset[f]	**28 193**	**22 544**	**−1.2**	**18 478**	**−1.8**	**16 725**	**−1.1**
	Total South Georgia	**101 488**	**74 296**	**−1.7**	**[60 896][g]**		**[55 119][g]**	

[a]Included all Willis Islands; [b]Subset of 8 colonies (38 to 39% of the Bird Island totals in 1984/1985–1991, 2003/2004 and 2023/2024); [c]Count for Cape Buller included Sheathbill Bay and Sitka Bay; [d]Count for Cape Crewe included Cape Wilson; [e]Count for Northwest South Georgia included Paryadin Peninsula North and South, and Jomfruene; [f]Only includes the 13 sites counted in all surveys; [g]Extrapolated from the ratio of the subset counted in 2023/2024 to the total South Georgia population in 2003/2004

3.4. Count consistency

Photo-counts were consistent among the 4 counters in 2023 (12 colonies, 1.7% mean difference, Kruskal-Wallis: $H = 0.1897$, $p = 0.979$), and with previous counts in 2003 (16 colonies, 2.9% mean difference, Wilcoxon rank sum test: $W = 2422$, $p = 0.9055$) and 2014 (11 colonies, 6.2% mean difference, Wilcoxon rank sum test: $W = 265$, $p = 0.6319$). The estimated error of estimates based on boat-based photo-counts compared with those based on ground and vantage-point counts on Bird Island was 2.04% overall (1.6% for black-browed albatrosses and 6.4% for grey-headed albatrosses).

4. DISCUSSION

The results from these recent surveys indicate that the rates of decline of all 3 albatross species were slower from 2014/2015 to 2023/2024 than from 2003/2004 to 2014/2015: wandering albatross *Diomedea exulans* −0.1 vs −1.7% yr^{-1}; black-browed albatross *Thalassarche melanophris* −1.1 vs. −1.8% yr^{-1}; grey-headed albatross *T. chrysostoma* −4.1 vs. −5.0% yr^{-1}. Although these are positive signs, the overall decreases in the last 32 to 40 yr at South Georgia are substantial — 39, 46 and 66% for wandering, black-browed and grey-headed albatrosses — such that these

Table 3. Population sizes (breeding pairs; n) and trends of grey-headed albatrosses at different sites at South Georgia between surveys in 1984/1985–1992, 2003/2004, 2014/2015 and 2023/2024. The 1984/1985–1992, 2003/2004 and 2014/2015 counts are from Poncet et al. (2006, 2017). The locations of breeding sites are shown in Fig. 1

No.	Site	1984/ 1985–1992 (n)	2003/2004 (n)	Change 1985–1992 to 2003/2004 (% yr^{-1})	2014/2015 (n)	Change 2003/2004 to 2014/2015 (% yr^{-1})	2023/2024 (n)	Change 2014/2015 to 2023/2024 (% yr^{-1})
1	Main Island	[a]	5177					
2	Trinity Island	[a]	3309					
3	Hall Island	[a]	2686					
1–3	**Willis Islands**	**15 000**[a]	**11 172**	**−1.5**				
4a	Bird Island (subset)[b]	4782	3189	−2.4	2248	−3.1	1237	−6.4
4b	Bird Island (total)	6857	5120	−2.4			1888	
5	Sorn & Bernt	1480	1625	0.5	616	−8.4	447	−3.5
6	Cape North	415	488	0.9	324	−3.7	212	−4.6
13	Paryadin Peninsula North	[c]	6721		3740	−5.2	2996	−2.4
14	Jomfruene	[c]	490		389	−2.1	176	−8.4
15	Paryadin Peninsula South	[c]	22 058		12 251	−5.2	8329	−4.2
13–15	**Northwest South Georgia**	**31 000**[c]	**29 269**	**−0.3**				
	Total South Georgia subset[d]	**[37 677]**	**34 571**	**−0.4**	**19 568**	**−5.0**	**13 397**	**−4.1**
	Total South Georgia	54 752	47 674	−0.7	[26 985][e]		[18 475][e]	

[a]Included all Willis Islands; [b]Subset of 11 colonies (62 to 70% of the Bird Island totals in the previous and following surveys); [c]Count for Northwest South Georgia included Paryadin Peninsula North and South, and Jomfruene; [d]Only includes the 9 sites counted in all surveys; [e]Extrapolated from the ratio of the subset to the total South Georgia population in 2003/2004

populations now represent only 13.3, 7.6 and 28.7% of revised global totals of 9616, 724 717 and 64 408 breeding pairs, respectively (Table 4). These percentages and global totals are approximations given the uncertainties associated with the counts at each island group, some of which are several decades old. In terms of the number of birds at South Georgia, the declines are such that there were 1654, 92 738 and 72 554 fewer breeding wandering, black-browed and grey-headed albatrosses, respectively, in 2023/2024 than at the time of the first censuses in the mid-1980s (Tables 1–3).

Although trends based on surveys in single breeding seasons several years apart may be unreliable, particularly for species that breed biennially, all evidence suggests that the data collected in 2023/2024 were representative. Methodologies were the same in the last 3 surveys, involving corrections for previous nest failures and for diurnal variation in attendance of non-breeders and failed birds (Poncet et al. 2006, 2017), the counts of all species at Bird Island were in line with the long-term trends based on annual monitoring, despite mortalities caused by HPAI infection in wandering albatrosses (Bennison et al. 2024), and the same observers carried out the last 3 wandering albatross surveys. Although there were more assumptions in the corrections applied to the photo-counts for black-browed and grey-headed albatrosses, variability among counters was low, the error in the photo-counts vs. direct ground and vantage-point counts on Bird Island was 2.0% overall, and the subset of colonies surveyed in 2023/2024 represented substantial portions of the South Georgia populations (30.3 and 72.5% of the black-browed and grey-headed albatrosses, respectively, during the full census in 2003/2004).

4.1. Wandering albatross trends and threats

The census in 2023/2024 indicates a more positive outlook for wandering albatrosses at South Georgia than in previous decades. The decrease has slowed to −0.1% yr^{-1} from 2014/2015 to 2023/2024, compared to −1.7% yr^{-1} from 2003/2004 to 2014/2015, and −1.5% yr^{-1} from 1983/1984 to 2003/2004 (Table 1). At the major sites (≥20 pairs in 2014/2015), the trend from 2014/2015 to 2023/2024 was considerably poorer at Bird Island (−1.1% yr^{-1}), which is by far the largest breeding site (699 pairs in 2023/2024), than at Cape Alexandra, Saddle, Prion and Albatross islands (−0.3 to 2.3% yr^{-1}). This is consistent with the considerably steeper decline at Bird Island than in the Bay of Isles (Albatross and Prion islands) in 1998/1999 to 2017/2018 from annual counts (−3.0 vs. −1.4% yr^{-1}, Rackete et al. 2021). Annenkov Island was not surveyed in 2014/2015, but the count in 2023/2024 was very simi-

Table 4. Regional and global population sizes and recent trends of wandering, black-browed and grey-headed albatrosses. Trend is indicated by arrows; ~: broadly stable

Island group	Estimated annual breeding pairs	Survey seasons	Trend (last 1–2 decades)	Global proportion (%)	Source
Wandering albatross					
Atlantic Ocean					
South Georgia	1278	2023/2024	↓	13.3	This study
Indian Ocean					
Prince Edward Island	1800	2008/2009	~	18.7	(Ryan et al. 2009)
Marion Island	2979	2023/2024	↑	31.0	(ACAP 2025)
Îles Crozet	2189	2016/2017 to 2022/2023	↑	22.8	(Weimerskirch et al. 2018, ACAP 2025)
Îles Kerguelen	1361	1984/1985 to 2022/2023	↑	14.2	(Weimerskirch et al. 1989, 2018, ACAP 2025)
Pacific Ocean					
Macquarie Island	9	2022/2023		0.1	(ACAP 2025)
Total	**9616**				
Black-browed albatross					
Atlantic Ocean					
Falklands	530 791	2010/2011	↑	73.2	(Wolfaardt 2012)
South Georgia	55 119[a]	2023/2024	↓	7.6	This study
Indian Ocean					
Îles Crozet	710	1981/1982, 2016/2017	↓	0.1	(Weimerskirch et al. 2018, ACAP 2025)
Îles Kerguelen	2945	2013/2014 to 2017/2018	↑	0.4	(Weimerskirch et al. 2018, ACAP 2025)
Heard Island	600	2000/2001	↑	0.1	(Woehler et al. 2002)
Pacific Ocean					
Macquarie Island[c]	152	1993/1994, 2023/2024	~	<0.1	(Brothers & Ledingham 2008, ACAP 2025)
Campbell Islands	31	1994/1995		<0.1	(ACAP 2025)
Antipodes Islands (Bollons)	115	1995/1996		<0.1	(Tennyson et al. 1998)
Islas Ildefonso	54 284	2014/2015	↑	7.5	(Robertson et al. 2017)
Islotes Evangelistas	4819	2014/2015	↑	0.7	(Robertson et al. 2017)
Diego de Almagro	14 814[b]	2001/2002		2.0	(Lawton et al. 2003)
Islas Diego Ramirez	59 746[b]	2002/2003, 2014/2015	↑	8.2	(Robertson et al. 2007, 2017)
Islote Albatros	46	2019/2020	↓	<0.1	(Droguett et al. 2023)
Islote Leonard	545	2014/2015	↓	0.1	(Robertson et al. 2017)
Total	**724 717**				
Grey-headed albatross					
Atlantic Ocean					
South Georgia	18 475[a]	2023/2024	↓	28.7	This study
Indian Ocean					
Prince Edward Island	1506	2008/2009	↓	2.3	(Ryan et al. 2009)
Marion Island	8462	2023/2024	↑	13.1	(ACAP 2025)
Îles Crozet	5319	1981/1982, 2016/2017	↑	8.3	(Weimerskirch et al. 2018, ACAP 2025)
Îles Kerguelen	6680	1986/1987 to 2014/2015	↓	10.4	(ACAP 2010, 2025, Weimerskirch et al. 2018)
Pacific Ocean					
Macquarie	89	2023/2024	~	0.1	(ACAP 2025)
Campbell Islands	6429	2019/2020	↓	10.0	(Frost 2020)
Islas Diego Ramirez	17 440[b]	2002/2003 to 2014/2015	↑	27.1	(Robertson et al. 2007, 2017)
Islas Ildefonso	8	2001/2002		<0.1	(Robertson et al. 2008)
Total	**64 408**				

[a]Extrapolated from the ratio of the subset counted in 2023/2024 to the South Georgia total in 2003/2004; [b]Corrected counts assuming same ratio of pairs:birds (0.95) as in Robertson et al. (2017); [c]Including Bishop Islet

lar to 2003/2004, so the trend there is in line with breeding sites other than Bird Island.

The long-term decrease in wandering albatrosses at South Georgia compares with the gradual recovery since the mid-1980s that followed earlier declines at Indian Ocean islands, including Crozet (except le de l'Est), Kerguelen, Marion and Prince Edward islands, although counts at this last site in 2001/2002 and

2008/2009 suggest that recovery there has stalled (Table 4). The main driver of the earlier declines at these sites and Macquarie Island is considered to be the advent of demersal and pelagic longline fisheries in the Southern Ocean in the 1960s (Weimerskirch et al. 1997, Nel et al. 2003, Terauds et al. 2006, Ryan et al. 2009). The contrasting trend at le de l'Est may relate to differential overlap with fisheries (Weimerskirch et al. 2018). There were no known terrestrial threats to wandering albatrosses at South Georgia until the recent HPAI outbreak (Bennison et al. 2024) and all breeding sites were designated as Specially Protected Areas in 2022. Breeding success at South Georgia is very high (mean 73% since 1980/1981) and was higher in the last 2 decades than previously, possibly related to stronger winds, greater availability of discards from demersal fisheries, and reduced competition as a consequence of the population decline (Pardo et al. 2017). As such, there is no reason to expect that prey availability or other conditions at sea have deteriorated. The major ongoing threat to the South Georgia population, even more than to the wandering albatrosses in the Indian Ocean, is therefore bycatch in fisheries. This occurs outside the South Georgia and South Sandwich Islands Marine Protected Area, where seabird bycatch has been reduced to negligible levels (Collins et al. 2021).

During breeding at South Georgia, wandering albatrosses remain predominantly in the southwest Atlantic Ocean, whereas nonbreeding adults and immatures have a circumpolar distribution, overlapping with fisheries in all southern oceans (Clay et al. 2019). Based on reported fishing effort at large spatial scales (5 × 5° cell) available until the early 2010s, the overlap was greatest with pelagic longline vessels flagged to Japan, Taiwan, Brazil, Uruguay, Spain and Portugal, and demersal longline vessels flagged to Argentina, Chile, South Africa, Falklands and New Zealand or operating in the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR) area at different times of year, with hotspots in the southwest Atlantic and southwest Indian oceans (Jiménez et al. 2016, Clay et al. 2019). Recent analyses of fine-scale overlap of various life-history classes with fisheries confirmed these results for the southwest Atlantic Ocean but also identified high overlap with demersal longliners flagged to South Korea and Chile, trawlers flagged to Argentina and Uruguay, and undeclared vessels without associated vessel automatic identification system (AIS) positions that may have been from the Brazilian small-scale hook-and-line fisheries, or illegal, unreported and unregulated (IUU) vessels (Carneiro et al. 2022).

Seabird bycatch rates are very variable, but the fleets of particular concern for wandering albatrosses are pelagic longliners managed by the International Commission for the Conservation of Atlantic Tuna (ICCAT), Brazilian small-scale fisheries and IUU vessels (Bugoni et al. 2008a,b, Tuck et al. 2011, Jiménez et al. 2016). Annual survival of adults at Bird Island from 1980 to 2012 showed a negative correlation with pelagic longline fishing effort in the nonbreeding period, and juvenile survival was low and correlated negatively with both pelagic and demersal longline effort (Pardo et al. 2017). Pelagic longline fishing effort peaked in the 1990s and has since declined, and in the Indian Ocean has shifted north (Clay et al. 2019). Shifts in distribution, reduced effort in some fisheries, and greater use of seabird-bycatch mitigation (Favero et al. 2013, Jiménez et al. 2020, Collins et al. 2021) may therefore explain the slowed rate of decline of the South Georgia population in the last decade. Adult females, and juveniles and immatures of both sexes likely remain at greater risk of bycatch than adult males, as their distributions extend further north into subtropical waters and hence show greater overlap with pelagic tuna fisheries (Jiménez et al. 2016, Clay et al. 2019). Tracking indicates that the steeper population decrease at Bird Island than in the Bay of Isles (and likely elsewhere) reflects the greater overlap of breeding adults and nonbreeders from that site with fisheries, particularly at the Subtropical Convergence and Patagonian Shelf break (Warwick-Evans et al. in press, British Antarctic Survey unpubl. data).

HPAI is now a threat, and was the likely cause of death of 58 adult wandering albatrosses on Bird Island, of which at least 9 were breeding birds (leading to failure of those nests), during the incubation period in 2023/2024 (Bennison et al. 2024), and of 15 adult wandering albatrosses in 2024/2025 (authors' unpubl. data). Whether it will be a major source of mortality in the future is unknown. The other potential threats are likely minor. Adult wandering albatrosses feed marine plastics, other debris and hooks discarded inside non-target fish to their chicks, the long-term consequences of which are unknown (Phillips et al. 2010, Phillips & Waluda 2020). Mercury contamination of wandering albatrosses at South Georgia is high and appears to have increased in recent decades (Tavares et al. 2013, Mills et al. 2024). However, studies in the Indian Ocean provide mixed support for an effect of mercury concentrations on demography of wandering albatrosses (Goutte et al. 2014, Bustamante et al. 2016).

4.2. Black-browed albatross trends and threats

Black-browed albatrosses continue to decline at South Georgia but at a slower rate than previously (-1.1% yr^{-1} from 2014/2015 to 2023/2024 vs. -1.8% yr^{-1} from 2003/2004 to 2014/2015). This contrasts with increases in black-browed albatrosses over the last 1 to 2 decades at the Falklands, Kerguelen, Ildefonso, Evangelistas and Diego Ramirez islands (Table 4). The recent increases in the Chilean populations are attributed to reduced mortality in demersal fisheries for Patagonian toothfish *Dissostichus eleginoides* following a switch to a new fishing system that prevents depredation of the catch by whales (Robertson et al. 2014, 2017). Although the trend of the Kerguelen population is now positive, the decline in the mid-late 1990s and early 2000s was attributed to unfavourable environmental changes and bycatch in pelagic longline fisheries off Australia during the nonbreeding season, and IUU demersal longline fisheries around Kerguelen (Rolland et al. 2008, Michael et al. 2017). The small populations at Crozet, Albatros and Leonard islands have declined, and that at Macquarie Island is stable, but together these represent <0.5% of global numbers (Table 4).

The black-browed albatrosses at South Georgia therefore represent the only sizeable population (>1% of global) that is decreasing (Table 4). The local demersal fishery for Patagonian toothfish is closed for most of their breeding season, and implements best-practice mitigation when it operates, hence seabird bycatch rates around South Georgia are now negligible (Collins et al. 2021). Overlap with fisheries is much higher for adults during the nonbreeding season, and year-round for juveniles and immatures, which use waters in the Benguela Upwelling region and northern Patagonian Shelf. The main fleets in these regions are pelagic longliners flagged primarily to Taiwan, and demersal longliners and trawlers flagged to Namibia and South Africa (Clay et al. 2019). Seabird bycatch rates have been greatly reduced in longline and trawl fisheries in Namibia and South Africa in the last 1 to 2 decades (Maree et al. 2014, Rollinson et al. 2017, Da Rocha et al. 2021), which likely explains the slower decline at South Georgia in recent years. However, the birds remain susceptible to bycatch elsewhere in their range, including in the tuna fisheries in the ICCAT region that have still not adopted best-practice bycatch mitigation or effective monitoring of compliance (Jiménez et al. 2020).

Analyses of data collected until the early 2010s found significant effects of climate and prey availability on demography of black-browed albatrosses at South Georgia (Pardo et al. 2017). This included a positive influence of sea-ice extent during the preceding nonbreeding period on breeding success and a negative influence of sea-surface temperature on return probability. Recent changes in wind regime appeared to be detrimental, with poorer breeding success associated with more poleward winds during the breeding period. The Southern Annular Mode had a negative influence on juvenile survival, and direct and indirect indices of krill availability had positive influences on juvenile and adult survival. Overall, although adult survival was the main influence on population trends, poor breeding success also made a major contribution to the long-term population decline (Pardo et al. 2017). The main mechanism by which environmental changes will continue to have negative repercussions on breeding success of black-browed albatrosses at South Georgia is reduced densities and a southerly shift of Antarctic krill *Euphausia superba* in the southwest Atlantic Ocean, possibly related to lower sea-ice extent (Kawaguchi et al. 2024).

4.3. Grey-headed albatross trends and threats

Results from the recent survey indicate that the precipitous decrease in grey-headed albatrosses at South Georgia has continued, albeit at -4.1% yr^{-1} from 2014/2015 to 2023/2024, compared with -5.0% yr^{-1} from 2003/2004 to 2014/2015. There are contrasting trends in this species across the Southern Ocean, with decreases at Prince Edward, Kerguelen and Campbell islands, and increases at Marion, Crozet, and possibly Diego Ramirez islands (Table 4). Ryan et al. (2009) suggested that the decrease at Prince Edward Island was potentially related to greater heat stress as this is the most northerly breeding colony. In addition, high bycatch rates in the local toothfish fishery in the mid-1990s (Nel et al. 2002) coincided with a population decrease at Marion Island (Stevens et al. 2024). There is no obvious explanation for the opposing trends at Crozet and Kerguelen islands, particularly as this species is now rarely killed in fisheries in the Indian Ocean (Weimerskirch et al. 2018), or the possible recent increase at Diego Ramirez (Robertson et al. 2017). The massive decline at Campbell Island since the 1940s, which began long before the expansion of pelagic longline fishing in the region, was considered to have been driven by oceanographic changes (Waugh et al. 1999). This remains the most likely explanation given numbers continued to decrease until the recent survey in 2019/2020 (Frost 2020), and the overlap with fisheries is less than that of many other albatross populations in New Zealand (Goetz et al. 2022).

The steep ongoing decrease in grey-headed albatrosses at South Georgia is of particular concern because the islands formerly held >50% of the global population. Population modelling showed a negative correlation between adult survival at Bird Island and fishing effort (Pardo et al. 2017). As with black-browed albatrosses, overlap of breeding grey-headed albatrosses with local fisheries has been relatively low since the introduction of the closed season at South Georgia, and in any case the latter was never a common bycatch species in the local area (Collins et al. 2021). The greatest overlaps at large spatial scales are with pelagic longliners flagged to Japan, Taiwan and South Korea, and demersal longliners or trawlers flagged to Argentina, Chile, Falklands or in the CCAMLR area (Clay et al. 2019). Juvenile grey-headed albatrosses dispersing from South Georgia, and potentially immatures, use a staging area in the southeast Atlantic Ocean that is a recorded bycatch hotspot for the Japanese pelagic longline fishery, and this species dominates that bycatch in the late summer (Inoue et al. 2012, Frankish et al. 2021). Mean juvenile survival of cohorts fledged up until 2000 was just 0.764, which, along with relatively low adult survival and breeding success, contributed to the ongoing population decline (Pardo et al. 2017). Grey-headed albatrosses of all age classes remain at risk from ICCAT fisheries given the sub-optimal bycatch-mitigation regulations and weak enforcement (Jiménez et al. 2020).

Mercury concentrations in grey-headed albatrosses at South Georgia have increased threefold in the last 25 yr, and male parents that failed had significantly higher feather mercury concentrations than successful birds (Mills et al. 2020a). Rates of ingestion of marine debris are 6× higher in grey-headed albatrosses than black-browed albatrosses but still likely to be below a level at which there might be population-level impacts (Phillips & Waluda 2020). There were effects of changing climate and prey availability on grey-headed albatrosses during the breeding but not the nonbreeding season, with a positive relationship between adult survival and krill density in a long-term survey area northwest of South Georgia, and a negative relationship between return probability and more poleward winds (Pardo et al. 2017). There were also relationships between adult survival, juvenile survival and breeding success, and climatic indices (the Southern Oscillation Index and the Southern Annular Mode), although not necessarily in the same direction. Overall, adult survival was the dominant driver of population trends, but the low and highly variable rates for different reproductive parameters, especially breeding success, also made major contributions to the long-term population decline (Pardo et al. 2017). Reduced densities and the retreat south of Antarctic krill will continue to be problematic for grey-headed albatrosses, as for black-browed albatrosses, given its importance in the diet of both species (Mills et al. 2020b).

5. CONCLUSIONS

Although the rates of decline of all 3 albatross species at South Georgia have decreased since the previous survey, the populations have decreased overall by 39 to 66% in just 32 to 40 yr. Apart from HPAI for wandering albatrosses, the main threats are climate change — which is a global challenge — and bycatch, which is the clear management priority given there are effective technical and operational solutions. The designation of the birds at South Georgia as ACAP High Priority Populations should be providing the impetus for action within national fisheries bodies and regional fisheries management organisations to improve bycatch-mitigation regulations and implementation, greatly increase levels of observer coverage to monitor their effectiveness and, if necessary, to impose penalties for non-compliance (Phillips et al. 2016, Good et al. 2020, Baker et al. 2024).

In terms of extinction risk, continuation of the observed rates of decline at South Georgia over 3 generations (~70 yr, Bird et al. 2020) would justify regional IUCN Red-listing of wandering and black-browed albatrosses as Endangered, and grey-headed albatrosses as Critically Endangered. Given the generally much poorer status of these populations than the same species in other ocean basins, attention should be focused on ICCAT particularly as a recent study clearly showed the benefits of the adoption of ACAP best-practice bycatch mitigation by all pelagic longliners in the south Atlantic Ocean (Bell et al. 2025), on demersal longline and trawl fisheries in the southwest Atlantic, including Brazilian small-scale hook-and-line fisheries (Bugoni et al. 2008a,b), and on assessing the scale and impact of IUU fishing (Agnew et al. 2009). If stronger action to reduce seabird bycatch is not taken by ICCAT, other regional fishery management organisations and some national fisheries bodies, then the prospects for wandering, black-browed and grey-headed albatrosses at South Georgia will remain bleak. Any attempts to predict the future for these populations would also need to consider the implications of ongoing climate change and other human threats (Barbraud et al. 2012, Michael et al. 2017).

Data availability. Stitched images of breeding sites of black-browed and grey-headed albatrosses, and counts derived from the images are available at https://doi.org/10.5285/3f71209a-33d5-4d48-8899-d8a6d8d295e6.

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