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An updated census of southern elephant seals *Mirounga leonina* at South Georgia using satellite imagery

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Historically, South Georgia supported the largest breeding population of southern elephant seals *Mirounga leonina* globally. Yet since 1995, this population has remained largely unassessed and was presumed to be stable. Here, we provide an updated assessment of this sub-Antarctic population and demonstrate the application of sub-meter resolution satellite imagery as a valuable approach for censusing an extensive, remote, and challenging breeding population. Across two breeding seasons, 2024 and 2025, we tasked satellite imagery with resolutions between 0.3 m and 0.5 m and integrated image-based counts with extrapolative modeling under a generalised additive framework to estimate abundance at unobserved beaches. Using this approach, we estimate 107,630 (95% CI: 93,945 – 123,342) breeding females at South Georgia, compared to the historical benchmark of 113,444 (se = 4,902) in 1995. This corresponds to a total population of 376,705 individuals (95% CI: 328,808 – 431,697). However, this should not be interpreted as evidence of population stability. The present census followed the spread of highly pathogenic avian influenza through the sub-Antarctic and therefore reflects a recently perturbed population, while methodological differences between surveys and limited validation of the satellite correction framework also affect direct comparability with the 1995 census. In light of the recent IUCN re-assessment of southern elephant seals as Vulnerable, our results highlight both the importance of renewed population assessment at South Georgia and the value for tracking population trajectories across remote Southern Ocean breeding sites.

KEYWORDS
census, marine mammals, pinnipeds, population size, remote sensing, satellite imagery, South Georgia, southern elephant seal

Introduction

Reliable population assessments are fundamental to conservation, but repeated censuses remain challenging for large, remote wildlife populations distributed across inaccessible coastlines. Southern elephant seals *Mirounga leonina* at South Georgia provide a clear example of this challenge. Despite forming island-wide, conspicuous breeding aggregations, population assessments must be conducted in a narrow seasonal window across large geographic areas. South Georgia historically supported the largest breeding population globally (Boyd et al., 1996). However, its scale, remoteness and logistical constraints have limited repeat assessments.

Very-high-resolution satellite imagery now offers a practical means of overcoming traditional limitations of surveying remote wildlife populations by enabling near-synchronous surveys of regions at spatial scales that are difficult to achieve using field or vessel-based methods (Lynch et al., 2012; Fretwell et al., 2017; Weimerskirch et al., 2018; Bowler et al., 2020). Southern elephant seals are particularly suited to image-based surveys because they form dense, conspicuous breeding aggregations on open beaches within a narrow seasonal window, facilitating detection in overhead imagery. Such approaches have previously been applied in other sub-Antarctic regions (McMahon et al., 2014; Fudala and Bialik, 2020; 2022; Laborie et al., 2023).

Southern elephant seals have a circumpolar distribution and were historically exploited at major breeding centers (Bonner, 1984). However, because harvesting was more labor intensive and their products less valuable than those of other species, namely Antarctic fur seals (*Arctocephalus gazella*), exploitation generally caused less severe population declines. Unregulated sealing had largely ceased by the early twentieth century (Headland, 1992), with controlled harvesting continuing until 1961 at Îles Kerguelen and 1964 at South Georgia (Bonner, 1984).

Four distinct breeding populations have been identified for southern elephant seals: (i) at Peninsula Valdés in Argentina; (ii) at South Georgia in the South Atlantic; (iii) the Macquarie population in the South Pacific sector; and (iv) the Heard and Kerguelen populations, including the Crozet and Prince Edward archipelagos, in the south Indian Ocean (Slade et al., 1998; Rus Hoelzel et al., 2001). Across these populations, southern elephant seals breed annually, coming ashore (Hindell and Perrin, 2009) at the end of the austral spring, forming dense colonies on large, open, and typically sandy beaches. The largest males haul-out first, close to the end of September, followed by pregnant females, which give birth to a single pup ~3–5 days after arriving (MacCann, 1980; Le Boeuf and Laws, 1994). Pups are weaned over a period of ~22–23 days, with breeding females coming into oestrus several days prior to weaning (Laws, 1956; Carrick et al., 1962; Arnbom et al., 1993; Fedak et al., 1994; Le Boeuf and Laws, 1994; Fedak et al., 1996; Arnbom et al., 1997). During the breeding season, the species display extreme polygamy with males competing to form harems with breeding access to breeding females (van Aarde, 1980; Campagna et al., 1993; Boyd et al., 1996).

The arrival and departure of breeding female elephant seals follow a Gaussian (normal) distribution reaching ~50% of the maximum numbers two-to-three weeks either side of the peak (MacCann, 1980; Hindell and Burton, 1988; Le Boeuf and Laws, 1994; Boyd et al., 1996). At South Georgia, the peak of breeding falls at the end of October (Rothery and MacCann, 1987; Boyd et al., 1996; Dickens et al., 2021; Bamford et al., 2025). This predictable but temporally constrained breeding phenology provides a defined census window, meaning that satellite imagery must be acquired close to peak haul-out, or counts must be corrected for the expected proportion of breeding females ashore at the time of acquisition. Separately, image-derived counts require careful interpretation to account for variation in attendance around the breeding peak. Given the extreme polygyny exhibited by southern elephant seals, male counts are not meaningfully representative of the wider population, and counts of pups are imprecise due to their small size and juvenile survival (McMahon et al., 1999; Pistorius et al., 1999; McMahon et al., 2000; Pistorius and Bester, 2002; McMahon et al., 2003); consequently, breeding females are the standard demographic unit used in population assessments (MacCann and Rothery, 1988; Boyd et al., 1996; Condit et al., 2007, 2022; Laborie et al., 2023).

Populations show contrasting regional trends over recent decades. The Peninsula Valdés population has increased steadily (1-3.4% annually) for five decades (Ferrari et al., 2013). South Georgia was categorized as stable in 1995 when it was last assessed and represented ~54% of the global breeding population (Boyd et al., 1996). Macquarie Island has been declining overall, with similar trends reflected at Heard Island both linked to sea ice concentration (Slip and Burton, 1999; van den Hoff et al., 2014; Hindell et al., 2025). In the Indian Ocean severe declines occurred at Marion Island (McMahon et al., 2009; Oosthuizen et al., 2015), Îles Crozet (Barrat and Mougin, 1978; Guinet et al., 1992, 1999) and at Îles Kerguelen (Guinet et al., 1992), although the populations at Îles Crozet and Îles Kerguelen may have recently entered growth phases (Laborie et al., 2023). Similarly, smaller populations have seen diverging population trends, with the population at Peninsula Potter, South Shetland Islands first decreasing dramatically by -4.6% annually between 1995-2008, then reversing this decrease and growing at 5% annually from 2008-2018 (Negrete et al., 2022).

This regional variability was abruptly overshadowed by the arrival of highly pathogenic avian influenza (HPAI), first detected in the sub-Antarctic at South Georgia in 2023 (Banyard et al., 2024; Bennison et al., 2024), before spreading eastwards and affecting colonies across much of the species' range over subsequent seasons, irrespective of their prior trends. Significant mortality has been recorded across multiple species and locations, spanning from the South Atlantic to islands in the southern Indian Ocean (Clessin et al., 2025a; 2025b), although the broader demographic consequences remain poorly understood. The clearest evidence of these impacts comes from southern elephant seal populations at Peninsula Valdés (Campagna et al., 2024) and South Georgia (Bamford et al., 2025). Population modeling at Peninsula Valdés has shown that with the aforementioned mortality parameters for pups (>90%) and adult breeding females (~50%), populations will take untill the end of the 21st century to recover (Campagna et al., 2025). Following the spread of HPAI throughout the sub-Antarctic, the latest IUCN species

assessment re-classified southern elephant seals as Vulnerable, down from least concern in 2015 (Hofmeyr, 2026).

South Georgia is therefore both a conservation priority and a methodological test case for repeatable population assessment. The last island-wide census was conducted in 1995 by yacht-based circumnavigation of the archipelago (Boyd et al., 1996), during which breeding female seals were counted, either from aboard the yacht or during beach-walks, along each occupied stretch of coastline. The 1995 census provided an unprecedented spatially explicit baseline, but the logistical and financial demands of repeating such surveys, together with the inherent limitations of oblique-angle counts from a distant or ground-based vantage point, have precluded subsequent island-wide censuses over the past three decades. As a result, the status of a population that once represented almost half of the global breeding population remains uncertain at a time when renewed assessment is increasingly urgent.

Here, we used sub-meter resolution satellite imagery to conduct the first island-wide census of South Georgia's southern elephant seal population since 1995, following the arrival of HPAI. Specifically, we aim to: (i) provide an estimate of the breeding female and total population; (ii) account for temporal variation in breeding female attendance around peak haul-out; (iii) evaluate satellite-image undercounting using paired Unpiloted Aerial Vehicle (UAV) imagery; (iv) estimate abundance at unsurveyed beaches using a spatial modeling framework; and (v) contextualise these updated estimates with the 1995 census baseline. In doing so, we provide an updated population estimate and evaluate the utility of a scalable image-based census framework for monitoring sub-Antarctic southern elephant seal breeding populations.

Methods

Study sites

This study was conducted across the sub-Antarctic archipelago of South Georgia in the southwest Atlantic, approximately 1,500km east of the Falkland Islands (Figure 1). Southern elephant seals breed across the archipelago on wide, open beaches often backed by tussac, grass, rocks or scree (Boyd et al., 1996); these beaches were the targets for the satellite image acquisitions in 2024 and 2025. *In situ* arrival-curve data comprised observations from St Andrews Bay in 2024 and a long-term time series of beach counts from King Edward Point, collected primarily on foot and more recently by UAV counts (Figure 1). Here, individual beaches were considered the primary census unit, with satellite image acquisition and subsequent analysis structured around beach-level counts.

UAV data collection

As in Bamford et al. (2025), overhead UAV imagery were collected at St Andrews Bay using a hand-launched, fixed-wing AgEagle eBee X UAV under fair weather conditions with wind speeds<10m/s. This UAV has a maximum flight time of ~90 minutes and was permitted to fly up to 182m above ground level achieving the target image resolution of ~1-2cm. The camera payload was a 24 megapixel (6000 × 4000 pixel) Aeria X RGB camera designed specifically for photogrammetry, along with a dual-band global navigation satellite system (GNSS) receiver used to determine the position of the image centers accurately to 1.5cm. Data from King Edward Point formed part of the stations long-term monitoring efforts and were primarily collected on foot, supplemented by UAV counts taken from 2019 onwards using a DJI Mavic-2 Pro between 40-70m above sea level (Dickens et al., 2021). UAV datasets are available from the Polar Data Centre (Fenney et al., 2026, 2026a–m).

Satellite imagery collection

Satellite imagery was tasked from Vantor Inc. (formerly Maxar Technologies) during the 2024 and 2025 October/November breeding season. We aimed to collect island-wide imagery over a single breeding season. However, cloud cover, particularly on the southern and western end of the archipelago limited success in 2024 (Figure 1a). Therefore, tasking was repeated in 2025, providing greater coverage (Figure 1b). Both panchromatic and multispectral images with sub-meter resolution were purchased from the WorldView-02 (~0.5m), WorldView-03 (~0.3m) and Legion-01/02 (~0.3m) platforms, detailed in the Supplementary Material (Supplementary Table S1). Satellite imagery were first orthorectified to the REMA Digital Elevation Model (Howat et al., 2022) and then pansharpened using the Brovey Transformation within the *wfsai* package (Gascoyne and Bowler, 2026). Images were then clipped to a 200m buffer of the coastline. After Laborie et al. (2023), three replicate counts of breeding female seals were performed by a single experienced observer over a period of several months to reduce the influence of familiarity bias. These counts were then averaged for each beach producing a total of 46,132 ± 387.23 detected breeding female seals.

Arrival curve

To correct satellite counts taken either side of the peak breeding date, we fitted Gaussian arrival curves to daily breeding female counts taken from overhead UAV imagery of St Andrews Bay captured between 17th October and 1st November 2024. Counts were scaled to the proportion of the observed annual maximum, and a within-season time index was defined as days since the first observation ($t = date - t_0$). A Gaussian arrival curve was fitted

$$f(t) = a \quad exp\left\{-\frac{1}{2}\left(\frac{t-\mu}{\sigma}\right)^2\right\}$$

where a is the fitted peak height (amplitude) on the proportion scale, μ is the peak timing (days post t_0) and σ is the spread or standard deviation of days. The modelled peak date is $t_0 + \mu$ for the St Andrews data.

Elephant seal arrival phenology at South Georgia may vary spatially around the archipelago because difference breeding beaches differ in exposure, width, shelter and topography. Here we use 'outer' beaches to refer to beaches that are relatively open to the ocean, and 'inner' beaches to refer to those located within fjords and more distant to the open ocean. Historically, outer beaches have

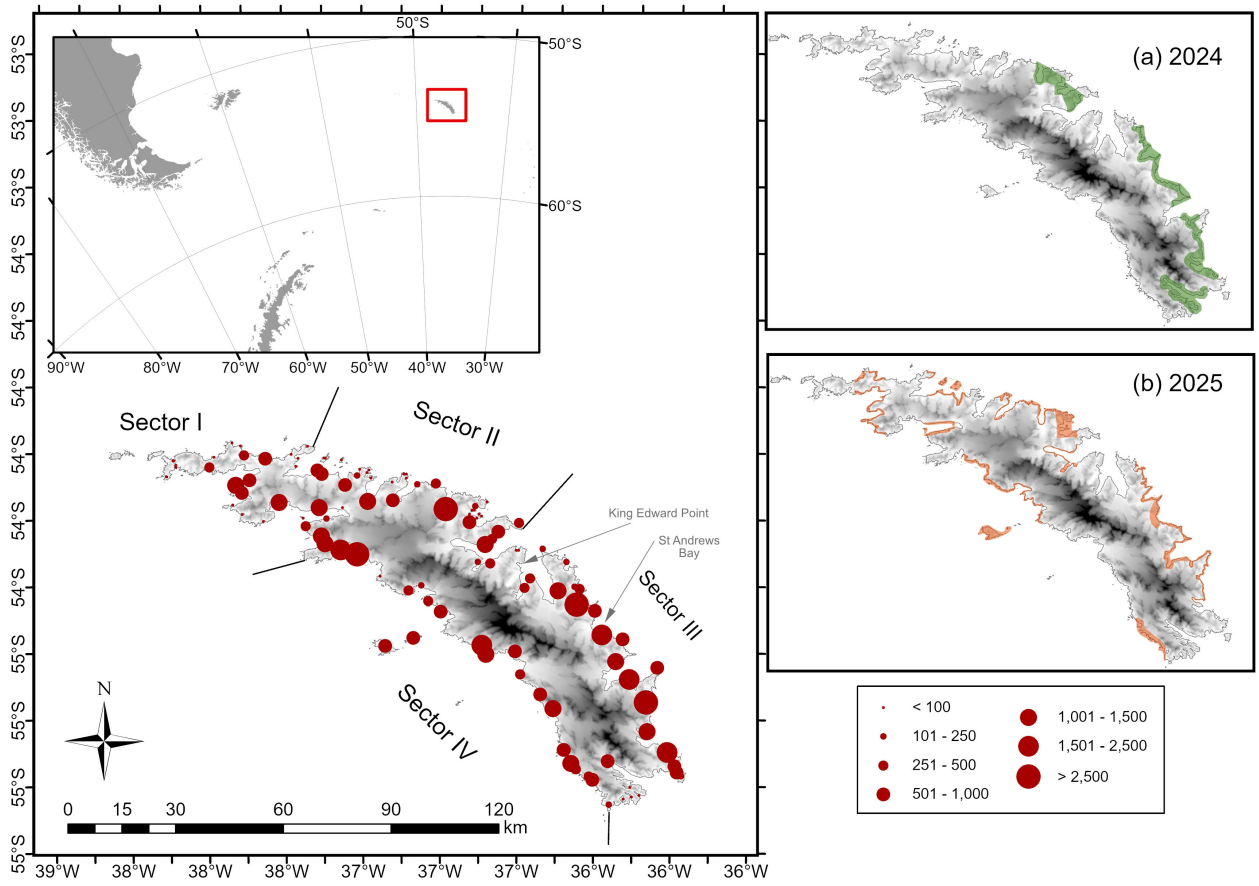

FIGURE 1
South Georgia, located in the southwest South Atlantic ~1,500 km due east of the Falkland Islands. In the lower left, red points represent individual counts from the southern elephant seal census in 1995 (Boyd et al., 1996), displayed with graduated symbology from<100 seals to >2,500 seals. The two subplots indicate the spatial extent of the acquired satellite imagery in **(a)** 2024 (green) and **(b)** 2025 (orange). Former sealing divisions (sectors) indicated.

been dominated as the largest breeding aggregations at South Georgia (Boyd et al., 1996). Interestingly, previous studies indicate that southern elephant seal breeding-site use and harem structure can vary with habitat characteristics, including beach size, shelter, substrate, slope and space availability (Baldi et al., 1996; Boyd et al., 1996; Mulaudzi et al., 2008). Such habitat preference could plausibly lead to beach- or region-specific arrival curves, although these have not been quantified at South Georgia. Further complicating these patterns is the known interannual variation in peak haul out dates at South Georgia, with published peaks between 24th and 27th October (Rothery and MacCann, 1987; MacCann and Rothery, 1988; Boyd et al., 1996; Dickens et al., 2021; Bamford et al., 2025). Therefore, here we derive a weighted arrival curve based on data collected in 2024 at St Andrews Bay alongside the time series data from King Edward Point (KEP) available intermittently between 1995 and 2024; here we opted for an ecologically informed weighting 70:30 in favor of outer (and observed) beaches, given the predominance of these colonies at South Georgia. The sensitivity of the resulting correction was tested using differing weighting scenarios of 60:40 and 80:20, recording compared peak haul out dates, full width at half maximum FWHM (*i.e.*, duration of the season above 50% of peak abundance), and the predicted proportion of breeding females ashore during the satellite acquisition window between 4th October and 6th November. The derived combined arrival curve is used to correct breeding female counts to peak; dates were aligned by day-of-year (DOY) on a non-leap reference year, and the proportion of peak on date d is

$$p(d) = \frac{f(t_d)}{a}$$

where f is the fitted combined Gaussian arrival curve and t_d = DOY(d) – min DOY on the reference-year axis. Uncertainty for the combined curve was obtained by bootstrap propagation (resampling years for KEP; years or residuals for St Andrews, as appropriate, B = 2,000), yielding 95% intervals for μ and σ. For the combined arrival curve, the modelled peak date was taken as the calendar date corresponding to min (DOY) + μ. Thus, the peak-corrected estimate is

$$\widehat{\text{Peak}} = \frac{\text{Count}(d)}{p(d)} = \text{Count}(d)\frac{a}{f(t_d)}$$

Efforts were made to avoid tasking imagery too far outside the peak of breeding and later relying on extrapolating corrections for dates outside the modelled range, where $p(d)$ may be very small and multipliers unstable.

VHR underestimation experiment

Satellite imagery offers the ability to collect information over large spatial scales with a high temporal recapture rate. However, this comes at the cost of resolution, especially when compared to *in situ* UAV imagery. We compared paired counts from very-high-resolution satellite (VHR) WorldView-02 imagery with an approximate ground resolution of 0.5m to UAV imagery over 25 harems split between Hound and St Andrews Bay (11 and 14 harems, respectively). Imagery was acquired from both platforms on 21st October 2024 within ~6 hours at each site (Hound Bay: UAV 12:28, VHR 11:51:50; St Andrews Bay: UAV 17:14, VHR 11:50:57).

Count repeatability was assessed separately for each imagery type by comparing three repeated counts of the same harem. Repeatability was summarised using within-harem coefficient of variation (CV), median and mean percentage change and intraclass correlation coefficients (ICC). $ICC_{(1,1)}$ describes the reliability of a single count, whereas $ICC_{(1,3)}$ describes the reliability of the average of three counts. Here, the first number refers to the one-way random effect ICC model, and the second number represents the number of counts included. Together, these metrics provide complementary measures of precision and repeatability of counts from the two types of imagery.

Potential session drift (*e.g.*, learning or interpretation changes across the three counting sessions/replicates) was tested with a linear mixed model including session (replicate) as a fixed effect and a random intercept for harem nested within beach; P-values were obtained via type-III ANOVA with Satterthwaite degrees of freedom. Given precise repeatability between UAV counts but noisier repeatability for VHR counts, the primary correction factor analysis are based on the mean of the three sessions (replicate) for each beach, harem and modality (platform).

To compare modalities, we formed a paired log-ratio for each harem using session – averaged counts with a small continuity correction ($\varepsilon = 0.5$) to handle zeros:

$$L_{bh} = \log(UAV_{bh} + \epsilon) - \log(VHR_{bh} + \epsilon)$$

where UAV_{bh} and UAV_{bh} are the session-averaged counts for harem h and beach b, respectively. The overall multiplicative correction factor (CF) was set to equal $\exp(\alpha)$, where α is the mean of the log ratios. As data were only available for two beaches and the between-beach variance only weakly identifiable, we estimated α using a fixed effect estimator that combines the two-beach mean log-ratios with equal weighting. Uncertainty in the CF was estimated via a cluster bootstrap, resampling for both beach and within-beach harems (B = 4,000) and took the percentile limits as 95% confidence intervals.

To apply the CF to the breeding female seal counts, we propagated uncertainty using Monte Carlo simulations. For each beach, the seal subtotals were modelled as log-normal calibrated so that the 2.5th and 97.5th percentiles matched the reported 95% CI. CF values were drawn from the bootstrap draws. In each simulation, we sampled one CF (shared over both beaches) and one subtotal for each beach, multiplied them to retrieve the adjusted counts, and repeated this over many runs (matching the number of CF draws). For each beach, we reported the median and 2.5th–97.5th range of the simulated adjusted count. The overall total and its adjusted 95% interval were obtained by summing the adjusted count for each beach within each simulation and then the median and 2.5–97.5% range across simulations.

Model based extrapolation to unsurveyed beaches

As satellite imagery was not available for all beaches on South Georgia within the monitored breeding seasons, we modelled log transformed average counts from observed beaches (Figure 2, green; n = 325) as a function of environmental covariates within a generalized additive model (GAM) framework. Extrapolations were then made to unsurveyed beaches (Figure 2, red; n = 261), whose location was informed in part by the 1995 census data (Boyd et al., 1996) alongside a review and manual digitisation of beaches visible in satellite mosaics of South Georgia (Map data ©2015 Google and ESRI layers via the QuickMapServices plugin v0.21.3). Geodesic beach area was log-transformed, with year, beach type and sealing sector fitted as factors, after Boyd et al. (1996), where:

- Type A – Open beaches. >200m in length with an open aspect landwards where all elephant seals were visible for up to several hundred meters inland.
- Type B – Long beaches. >200m long,<50m wide and backed by tussac grass knolls.
- Type C – Short beaches. backed by tussac,<200m long, <50m wide.
- Type D – Long, narrow beaches. Beaches >200m long and<50m wide, backed by rock and/or scree.
- Type E – Short, narrow beaches. Beaches 200m long and<50m wide, backed by rock and/or scree.

As some beaches (n = 63, 25.2%, see Figures 1a, b) were sampled in both 2024 and 2025, beach identity was included in all candidate models as a random-effect smooth. Additional candidate smooths included Euclidean distance to the fjord entrance (EDFjord, m); Euclidean distance to bathymetric contours (ED50m, ED100m, ED250m, ED500m and ED1000m) and beach aspect in degrees (°). Prior to modeling, pairwise collinearity between continuous covariates was assessed using a threshold of 0.7. Where bathymetric covariates were strongly correlated, variables were selected to retain ecological nearshore *versus* offshore contrast across spatial scales, whilst minimising redundancy.

We modelled log-transformed mean female seal count in a GAM framework, combining both fixed and penalised smooth functions for categorical and continuous environmental covariates, respectively. All models were fitted by restricted maximum likelihood using shrinkage smoothers and low basis dimensions (k = 4) as to favor a parsimonious model. We screened all pairwise tensor-product interactions among all continuous predictors by adding one interaction individually with inclusion assessments being made based on Akaike Information Criterion (AIC), effective degrees of freedom (edf) and P-values. Candidate final models were then compared using AIC. Model fit was assessed using standard

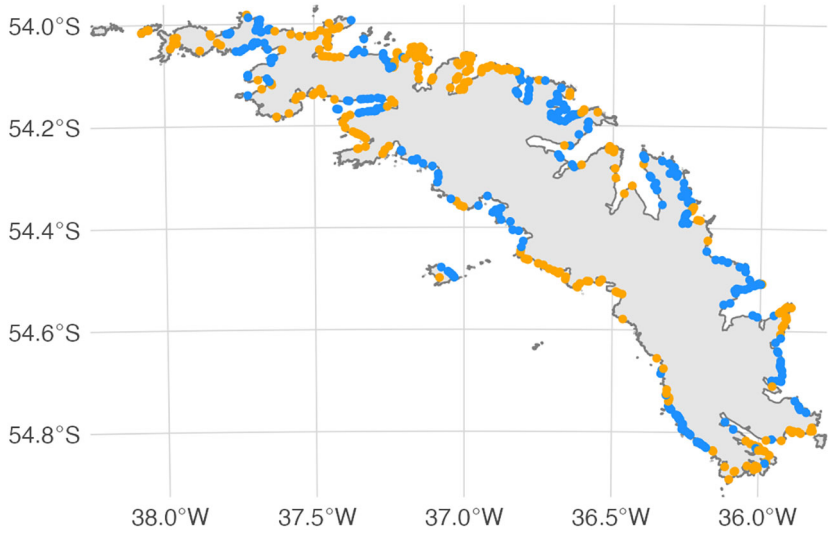

FIGURE 2
Locations of digitised beaches around South Georgia. Satellite imagery was acquired over the 2024 and 2025 seasons for beaches depicted in blue, but not for those in orange.

checks (Zuur et al., 2009). Predictions were back-transformed to the original count scale with uncertainty being estimated by summing beach-level variances. To assess transferability to unsurveyed beaches, we performed leave-one-out cross-validation (after Laborie et al., 2023). In each iteration, all observations from one beach were excluded, the model was refitted to the remaining data, and the omitted beach was predicted, providing a measure of predictive accuracy, bias and uncertainty.

Total population estimation

Despite southern elephant seals annual return to land to breed, at any given time an unknown fraction of the population, whether that be non-breeding females, males or juveniles, remains at sea. This makes estimating total population particularly difficult. To account for this absent component of the population, census counts should be adjusted using a correction factor derived from southern elephant seal life-table analysis. Following MacCann (1985), we applied a fixed demographic scaling factor of 3.5 to estimate the total population aged ≥1 year, explicitly excluding pups born during the respective census years (MacCann and Rothery, 1988).

Results

Arrival curve

The arrival curves were fitted to breeding female count data, with the curve from St Andrews Bay yielding parameter estimates of α = 1.004, μ = 40.35 days, σ = 12.13 days, corresponding to a FWHM of 28.56 days (Figure 3, teal). Across the 14 seasons at KEP, fits were strong (median $R^2 \approx 0.988$), with a mean FWHM of 26.9 ±1.7 days (Figure 3, orange). At KEP, peak dates ranged from the 23rd October in 2023 to the 1st November in 1995, with an average peak date on the 28th October (α = 0.996, μ = 46.10 days, σ = 11.7 days; FWHM 27.5 days; Figure 3, orange).

The 2024 St Andrews arrival curve peaked ~5.75 days earlier than the averaged KEP curve, with a peak on the 23rd October and was slightly broader with a FWHM of 28.56. The two curves were combined (Figure 3, blue), producing an adjusted arrival curve that peaked on the 25th October with α = 0.985, μ = 41.77 days (95% CI: 41.42 - 42.11), σ = 12.2 days (95% CI: 11.94 - 12.64), and a FWHM of 28.74 days (95% CI: 28.21 - 29.77). Alternative weighting scenarios revealed no more than one day variation in fitted peak, along with<0.04 days in FWHM. Across the satellite acquisition window, the maximum deviation from the predicted proportion of breeding females ashore was 2.47 percentage points, corresponding to a maximum difference of 7.45% occurring at the earliest end of the acquisition window (Supplementary Table S2).

VHR underestimation experiment

We analyzed breeding female counts from 25 harems across Hound and St Andrews Bay. Counts derived from UAV imagery were highly repeatable between the three replicate counts, with extremely low within-harem variability (median CV 0.8%, mean CV 1.5%) and near-perfect agreement between each count ($ICC_{(1,3)}$ = 0.99984, $ICC_{(1,1)}$ = 0.99952). In contrast, counts from satellite imagery were substantially more variable, although still highly repeatable overall (median CV 19.5%, mean CV 19.4%; $ICC_{(1,3)}$ = 0.954, $ICC_{(1,1)}$ = 0.873).

Mixed effect models showed no evidence of session drift in UAV counts (P = 0.33), with the session term (*i.e.*, the repeat count) essentially explaining none of the variance (marginal R^2< 0.01) and nearly all the variance attributed to persistent differences among harems (conditional R^2 = 1.00). Estimated differences between the replicates were small for the UAV counts (+0.72 and +0.24, Table 1) indicating that there was little to no variation between counts. However, counts from satellite imagery showed a strong session effect (P = 2.41×10^{-7}, Table 1), with counts increasing considerably

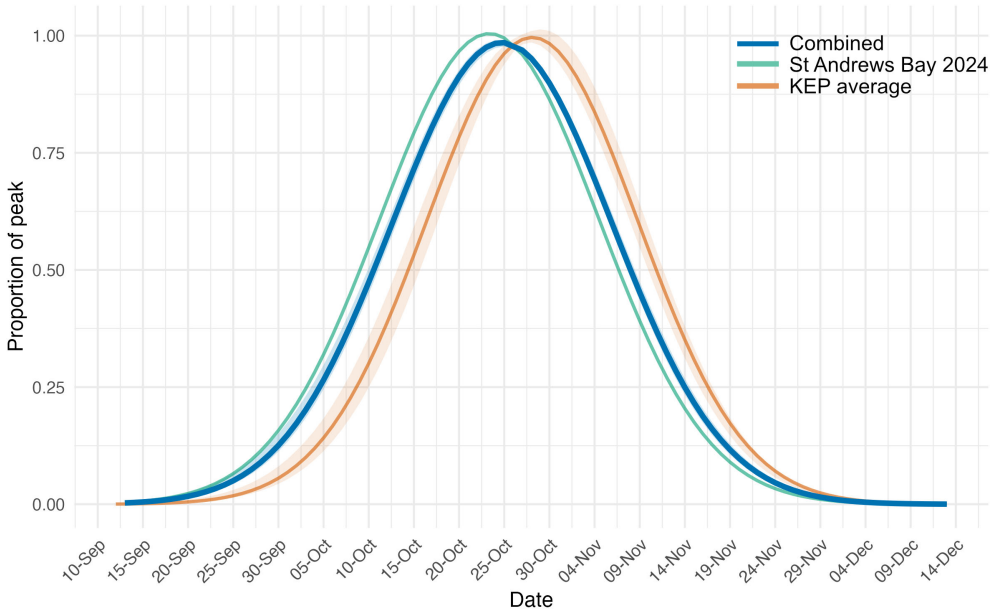

FIGURE 3
Modelled arrival curves for breeding female southern elephant seals at South Georgia, based on data from St Andrews Bay in 2024 (teal); averaged curve from King Edward Point from 1995,1996 & 1998; 2000; 2015-2024 (orange) and their combination on which corrections are based (blue). Shaded ribbons indicate 95% confidence intervals for multi-year data. Model fit available: Bamford et al. (2026).

in later sessions (+18.2 and + 26.6, Table 1). Residual variation was also much larger for satellite than UAV counts, indicating that satellite counts were noisier and influenced by repeated viewing, justifying the averaging of satellite counts sessions for further analyses.

Using session-averaged (replicate-averaged) breeding female counts the correction factor was CF = 1.58 95% CI 1.40–1.83), implying VHR counts underestimate those of a UAV by 36.7% (29 to 45%) on average. Mean UAV counts fell within the approximate 95% CI of mean VHR counts for only eight of 25 harems, consistent with a systematic undercount of breeding female seals in satellite imagery.

Habitat type

A total of 586 beaches were visually assessed to be suitable for southern elephant seals to haul out on at South Georgia. These were classified based on the beach-type criteria from Boyd et al. (1996). Of these beaches, Type A beaches accounted for 5.46%, Type B 29.35%, Type C 37.54%, Type D 10.75% and Type E 16.89% (Figure 4). From the first westerly landfall, South Georgia runs in an approximately south easterly orientation with beaches located with both seaward open aspects and within fjords (Figure 5a). Based on female seals counted on the beaches, those beaches with an aspect facing ENE appeared to be preferred (Figure 5b), hosting 22% of the observed total seals.

Model based extrapolation of the population

GAMs were fitted to a log-transformed mean female seal count, including year, beach type, and sector as parametric terms, and a random-effect smooth for beach identity. Environmental covariates were modelled using thin plate regression splines with shrinkage, with knots set to 4 to minimise overfitting. Only a single interaction term was supported, based on its influence on overall AIC and edf. As such, a tensor product interaction between distance to the 500m bathymetric contour and the aspect of the beach was included (Table 2). In contrast, including distance to the 50m bathymetric contour as an additional marginal term showed little influence (edf ≈ 0, P = 0.795, Table 3) and negligible AIC gain (Δ 0.0009). Thus, this term was dropped, and more parsimonious model was opted for (Table 3). Year showed little to no influence on the model, suggesting counts were consistent between both breeding seasons. Model diagnostics suggested good overall fit, residuals were broadly centered around zero, with QQ-plots indicating some departure from normality in the tails and slight evidence of greater residual spread at lower fitted values.

TABLE 1 Session (replicate) effect from linear mixed models fit. P-values used Satterthwaite degrees of freedom.

Modality	P-value	R^2 (Marginal)	R^2 (Conditional)	SD (Harem)	SD (Residual)	Rep 2 (95% CI)	Rep 3 (95% CI)
UAV	0.33	0.00001	1.00	79.5	1.72	+0.72 [-0.23,1.67]	+0.24 [-0.71,1.19]
VHR	2.41×10^{-7}	0.040	0.93	52.4	14.7	+18.2 [10.0,26.3]	+26.6 [18.4,34.7]

Marginal and conditional R^2 reflect variance explained by fixed effects and fixed + random effects, respectively. Harem SD is the random intercept SD (within beach harem) and SD (residual) is as stated. Contrasts are the changes in counts from replicate 1 with 95% CIs.

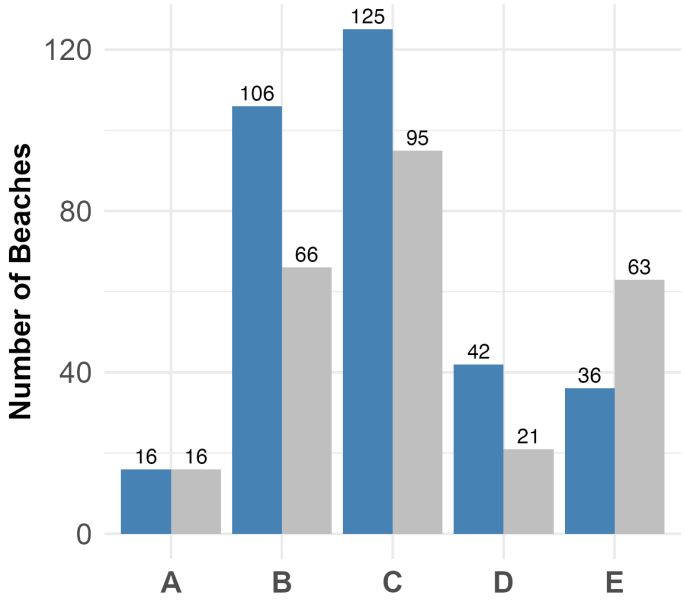

FIGURE 4
Classification of South Georgia beach types based on the criteria set out by Boyd et al. (1996) where Type A beaches are >200 m long with an open landward aspect, allowing seals to be visible for several hundred metres inland; Type B beaches were >200 m long and<50 m wide, backed by tussac grass knolls; Type C beaches were<200 m long and<50 m wide, also backed by tussac; Type D beaches were >200 m long and<50 m wide, backed by rock and/or scree; and Type E beaches were<200 m long and<50 m wide with similar rock and/or scree backing. Blue bars indicate observed beaches where VHR satellite imagery were acquired; grey bars indicate unobserved beaches, not covered by tasked imagery.

The final model explained ~87.8% of the variation in log mean female seal count (adjusted R^2 = 0.801). Beach area had a strong positive effect on average female seal count (edf = 2.039, F = 392.206, P< 0.01). There was substantial between-beach variation, which was captured by the beach id smooth (edf = 109.831, Ref.df = 242, F = 0.903, P< 0.001) (Figure 6). Distance to the fjord mouth showed only weak evidence of an independent effect (P = 0.054), but the smooth was not shrunk to zero and its removal resulted in a slightly less-well supported model. Distance to the 500m bathymetric contour also showed limited support as a standalone smooth (P = 0.061) alongside beach aspect (P = 0.453). However, both were retained to preserve hierarchy within the only clearly supported interaction term. The interaction between distance to the 500m bathymetric contour and beach aspect was significant (F = 5.362, P = 0.001), indicating the distance to the 500m contour varied with beach aspect influence (Figure 7). In terms of the factor covariates, only sector showed a noticeable influence on mean female seal count, with sectors three (estimate =

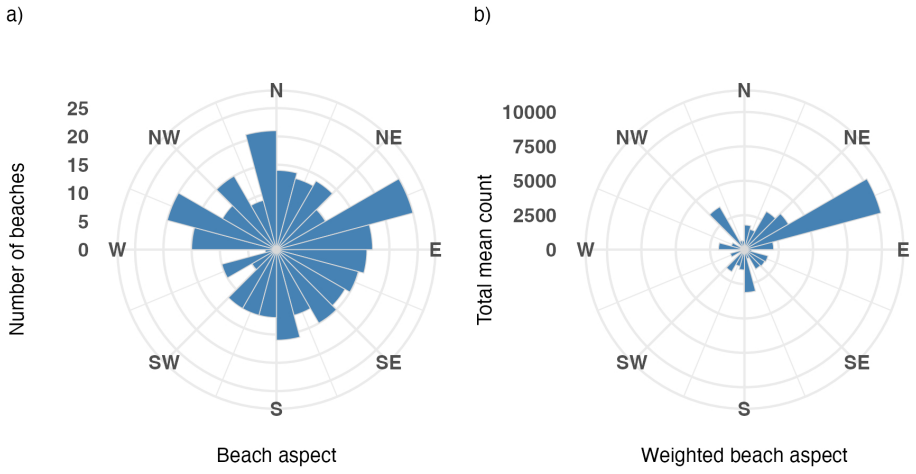

FIGURE 5
(a) Predominant aspect of beaches deemed to be suitable haul outs for southern elephant seals on South Georgia, and **(b)** beaches weighted by mean count of female seals on each 15-degree aspect bin.

TABLE 2 Comparison of candidate interaction terms evaluated for inclusion, with assessments based on Akaike Information Criterion (AIC), effective degrees of freedom (edf) and P-value.

Interaction	AIC	AIC improvement	EDF	P-value
ED500m:beach_aspect	848.983	10.177	3.344	0.001
EDfjord:beach_aspect	858.521	0.639	0.604	0.124
EDfjord: ED50m	858.947	0.213	1.341	0.044
ED50m:ED500m	859.159	0.001	0.000	0.940
area_log:EDfjord	859.159	0.000	0.000	0.981
area_log:ED500m	859.159	0.000	0.000	0.477
area_log:beach_aspect	859.160	0.000	0.001	0.326
ED50m:beach_aspect	859.249	-0.089	0.000	0.349
EDfjord: ED500m	860.374	-1.215	0.829	0.030
area_log:ED50m	862.882	-3.722	1.023	0.168

Euclidean Distance abbreviated to 'ED' and associated with the depth of the bathymetric contour or fjord mouth (i.e., ED50m represents the distance to the 50m bathymetric contour, and EDfjord the distance to the mouth of the fjord that the beach was within).

0.466, SE = 0.188, t = 2.487, P = 0.014) and four (estimate = 0.712, SE = 0.230, t = 3.099, P = 0.002) showing higher predicted log mean female seal count compared to sector one (Table 3).

Leave-one-out cross-validation for beaches indicated low prediction bias across all sites. However, at the level of individual beaches, precision was modest (RMSE = 1.03 and MAE = 0.81 on the log scale; RMSE = 238.03 and MAE = 84.78 on the count scale). When considered across all four sectors, agreement improved but performance remained uneven spatially. Sector two was substantially overpredicted with a proportional error of +64.3%, whereas sectors one and four were reproduced closely (-4.77% and +0.76%, respectively) and sector three was moderately underpredicted (-16.7%). This regional mismatch in sector two is important to consider. However, within this sector, approximately 89.3% of the overprediction was driven by just five beaches, and the median beach-level error within sector two was otherwise small (+4.9 female seals). For reference, sector two represented 24.3% of observed beaches, and 18.2% of observed mean female seal counts in the modelled dataset.

When examined over all sectors, the five worst-predicted beaches contributed 32.9% of the total absolute error, and the top ten contributed 43.3% of the absolute error, together suggesting that error was dominated by a minority of difficult to model beaches. Across all beaches, the mean and median signed errors were low

TABLE 3 Estimated parametric effects and smooths for terms retained in the final parsimonious model.

log_mean ~ year + type.x + sector + s(id, bs = "re") + s(area_log, k = 4, bs = "ts") + s(EDfjord, k = 4, bs = "ts") + s(ED500m, k = 4, bs = "ts") + s(beach_aspect, k = 4, bs = "ts") + ti(ED500m, beach_aspect, k = c(4, 4), bs = c("ts", "ts"))				
Term	Estimate	Std. Error	t-value	P-value
(Intercept)	3.359	0.359	9.363	0.000
Year	-0.012	0.115	-0.107	0.915
beach type B	-0.167	0.321	-0.521	0.603
beach type C	0.182	0.354	0.515	0.607
beach type D	-0.552	0.351	-1.571	0.118
beach type E	-0.204	0.383	-0.531	0.596
Sector 2	0.278	0.201	1.378	0.170
Sector 3	0.466	0.188	2.487	0.014
Sector 4	0.712	0.230	3.099	0.002
Term	**EDF**	**Ref.df**	**F**	**P-value**
s(id)	109.831	242.000	0.903	0.000
s(area_log)	2.039	3.000	392.206	0.000
s(EDfjord)	0.737	3.000	2.628	0.054
s(ED500m)	0.694	3.000	1.641	0.061
s(beach_aspect)	0.001	3.000	0.000	0.453
ti(ED500m,beach_aspect)	3.344	9.000	5.362	0.001

Smooths of covariates of Euclidean Distance abbreviated to 'ED' and associated to the depth of the bathymetric contour or fjord mouth (i.e., ED500m represents the distance to the 500m bathymetric contour, and EDfjord the distance to the mouth of the fjord that the beach was within).

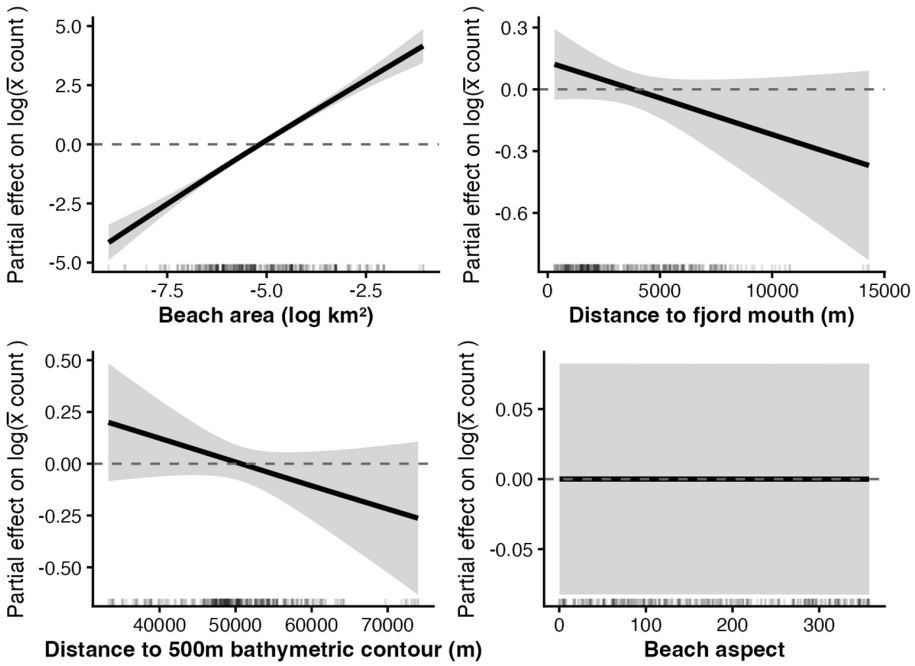

FIGURE 6
Estimated plots, centered on zeros, of the partial effects of the final model's smooth terms on the log of mean female seal count. Grey shading indicates 95% confidence intervals.

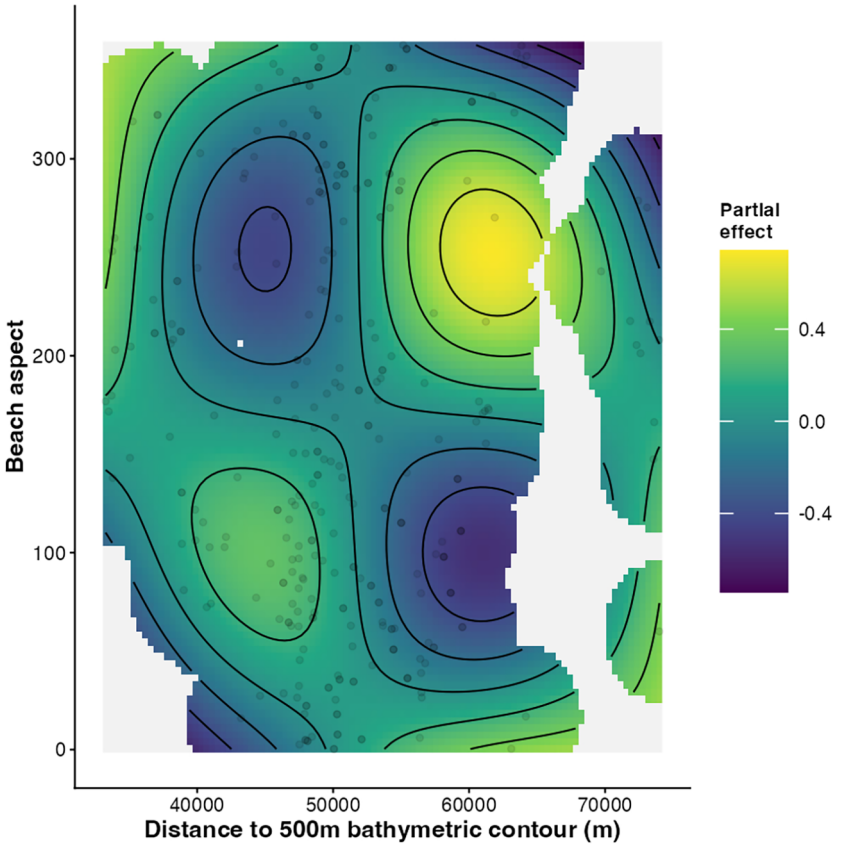

FIGURE 7
Estimated interaction between distance to the 500m bathymetric contour and beach aspect in the final model. Colours and contour lines show the fitted tensor-product smooth for the interaction on the log scale, with warmer colors indicating more positive and cooler colors more negative contributions to predicted log mean female seal count. Points show data locations in covariate space, and gray shading indicates areas with little to no supporting data.

(8.71 and 2.27 female seals per beach, respectively), indicating little overall bias in cross-validated predictions, whereas mean and median absolute error were larger (74.6 and 21.0 female seals per beach, respectively), indicating larger prediction error at a beach-level. This contrast shows that under- and over-predictions are largely balanced across the full dataset. This suggests that the modelled spatial variation in female seal counts with low overall bias, and that local predictive performance is regionally uneven. Therefore, extrapolated predications should be interpreted cautiously, particularly at the level of individual beaches.

Total population estimation

Based on the presented models of 2024 and 2025 data, the total number of breeding females at South Georgia was 68,074 (95% CI: 66,329 – 69,818). When the measured satellite under-count is corrected with uncertainty propagated from both the correction factor and the modelled output, the total rises to 107,630 breeding females (95% CI: 93,945 – 123,342), yielding a total overall population of 376,705 (95% CI: 328,808 – 431,697) after applying MacCann's (1985) demographic scaling factor of 3.5.

Supplementary information further explores these results in the context of recent observed declines associated with the observed population declines linked to HPAI (Bamford et al., 2025) and the trajectory of the South Georgia population since the previous census in 1995 (Boyd et al., 1996).

Discussion

This study provides the first island-wide census of South Georgia's breeding population of southern elephant seals since 1995 and further demonstrates the use of VHR satellite imagery for censusing large populations over remote and challenging environments. By combining tasked sub-meter resolution satellite imagery, UAV count comparison, repeated image counts, breeding-peak corrections and extrapolative modeling we provide an updated estimate of breeding female abundance and overall population numbers alongside an assessment of the opportunities and limitations of image-based monitoring at this scale. The corrected estimate of 107,630 breeding females, and an overall population of 376,705, indicates that South Georgia remains one of the major strongholds for this species in the Southern Ocean.

The timing of this reassessment is important. In 2023, a strain of the HPAI virus that transmits among mammals spread quickly through the sub-Antarctic (Banyard et al., 2024), reaching South Georgia in September 2023 (Bennison et al., 2024), where it has been linked to an average decline of 47% in the number of breeding female southern elephant seals at the three largest breeding sites (Bamford et al., 2025). The estimate presented here should therefore be understood as representing an already perturbed and suppressed population. This context is important when comparing to the previous 1995 population level, because similarity between the two does not necessarily imply long-term stability.

In 1995, South Georgia supported an estimated population of 113,444 (se = 4,902; approximate 95% CI: 103,836 – 123,052, calculated from the reported SE assuming normality) breeding females (Boyd et al., 1996). After correcting for satellite undercounting, our estimate of 107,630 breeding females (95% CI: 93,945 – 123,342) is broadly comparable to the 1995 historical benchmark, corresponding to 94.9% of the 1995 total and with the uncertainty around this estimate spanning both a moderate decline to a slight increase. However, this similarity should not be interpreted as evidence of demographic stability over the intervening three decades, particularly given methodological differences between the two census efforts and recent HPAI-related declines. A formal beach-level comparison was not undertaken because 1995 counts were often recorded between headlands or using poorly reproducible beach markers (*i.e.*, river or glacier front boundaries), leaving too few to confidently match sites for a representative comparison.

Since the 1995 census, comparable island-wide population monitoring of southern elephant seals at South Georgia has been broadly absent. Although localised monitoring and research have continued near research stations, with regular monitoring resuming from King Edward Point Research Station since 2015, these data are not spatially representative of the archipelago and are therefore insufficient to reconstruct island-wide population change reliably between 1995 and the present study. As a result, any interpretation of island-wide population trajectories over this interim period between the two censuses remains, out of necessity, cautious. One plausible interpretation is that the South Georgia population increased after 1995 and was subsequently reduced by HPAI, although the absence of comparable and representative island-wide monitoring prevents this trajectory from being reconstructed directly. This highlights the importance of combining long-term monitoring at key sites with periodic island-wide assessments to capture spatially heterogeneous population change and assess the overall population response robustly, similar to the decadal albatross census conducted around South Georgia (Mackley et al., 2025).

Simulations presented in the Supplementary Material suggest that, under the assumptions applied here, the implied pre-HPAI population from the uncorrected 2024/5 estimate had a mean of approximately 142,000 female seals (95% simulation intervals: 84,737-270,819), indicating that, in order to support the observed HPAI declines reported in Bamford et al. (2025), the population may have increased relative to 1995 before the arrival of HPAI at South Georgia. However, this estimate remains highly uncertain. Interestingly, any apparent population increase across South Georgia appears unlikely to have been driven by continued growth at the larger Barff Peninsula beaches, identified during the 1995 census to support the three largest colonies: St Andrews Bay, Hound Bay and Gold Harbour, together representing ~16% of the population on the archipelago in 1995. Bamford et al. (2025) report that *in situ* counts at these beaches remained stable since the mid-1980s up until the arrival of HPAI. Any population expansion after 1995 may therefore have occurred primarily at other sites around the island.

Despite this recent reduction in numbers, South Georgia remains a globally important breeding population for southern elephant seals. In 1995, it was estimated that South Georgia supported 54% of the global breeding population (Boyd et al., 1996), and the revised estimate presented here maintains its status as one of the major strongholds for the species, comparable to other sub-Antarctic islands. Laborie et al. (2023) provided recent estimates from Îles Kerguelen reporting a pre-HPAI population of 100,300 (±20,840) breeding females and a total population of 347,995 (se = 4,950) seals, which places Îles Kerguelen and South Georgia broadly level in terms of overall population size, and their global significance as breeding sites in the Southern Ocean. However, at the point of submission, HPAI among southern elephant seals has been confirmed on Îles Kerguelen but its impact on the population is, as of yet, unknown. Furthermore, we are also anticipating a significant mortality event at the Macquarie Island population in the upcoming 2026 breeding season.

Beyond the revised abundance estimate, this study demonstrates the practical value of satellite imagery for repeatable monitoring of southern elephant seals. The 1995 census provided an exceptionally detailed baseline, but its yacht-based approach was costly and difficult to repeat at regular intervals under increasing budgetary constraints. Furthermore, subsequent comparisons of ground- and UAV-based counts have shown that ground observations can undercount in dense aggregations, particularly cows and pups (Dickens et al., 2021), highlighting an additional source of uncertainty in ground-based surveys. In contrast, satellite tasking facilitates broad coverage of the archipelago, while avoiding many of the logistical and methodological constraints associated with vessel-based surveys. The combination of direct image counts alongside extrapolative modeling enabled unobserved beaches to be incorporated explicitly. This represents a substantial advance in the repeatability and spatial scale of population assessment for this species, even though further calibration is needed to refine precision and reduce platform-specific bias.

Towards repeatable satellite-based monitoring

As a platform, satellite imagery increases the spatial scale and temporal frequency of census efforts. However, for southern elephant seals, their breeding phenology complicates interpretation because of the asynchrony of breeding female arrivals. As not all breeding female seals are ashore at one time, counts made during the peak breeding window therefore likely do not represent the entire breeding population (Authier et al., 2011). Previous studies suggest that counts made at peak-haul-out represent between 82% to 96% of breeding females (MacCann, 1985; Rothery and MacCann, 1987; Boyd et al., 1996; Laborie et al., 2023), which leaves a proportion uncounted. Here, image-based counts from different days were corrected to a modelled peak with a derived arrival curve (Figure 3).

To construct a picture of island-wide populations, we needed to extrapolate into unsurveyed beaches using a modeling framework that predicted female seal numbers based on environmental and spatial covariates from those beaches that were surveyed. Whilst this model was developed primarily for prediction purposes, the modelled covariates also provide useful context for interpreting broad-scale variations exhibited by the breeding female population. Beach area was the strongest predictor, being positively correlated with greater counts of breeding females. This is intuitive, as larger beaches provide greater physical space for the formation of breeding aggregations. Island-sector also indicated that there is a regional structure in breeding female distribution around South Georgia, with sectors three and four (*i.e.*, the northeast, southeast and southern coastlines of the island) contributing most strongly to predicted abundance. This may reflect broad differences in coastline geology, geomorphology and local habitat between sectors (Dalziel et al., 2021), alongside effects of established colony persistence related to strong natal philopatry and breeding site-fidelity (McMahon and Bradshaw, 2004; Fabiani et al., 2006; Hofmeyr et al., 2012). Distance to the mouth of the fjord and distance to the 500m bathymetric contour were weakly negative, suggesting that outer beaches were slightly more likely to support greater numbers of breeding female seals, although these effects were not as strong as beach areas or sector. The interaction between aspect and distance to the 500m contour further suggests that the relationship between beach orientation and breeding female abundance depends on the offshore setting of the beach, potentially reflecting a preference in exposure, coastal configuration or accessibility from deeper waters.

However, such fine-scale interpretations should be treated cautiously as the model was designed for broad-scale prediction rather than to test specific habitat preferences, and several covariates may act as proxies for unmeasured environmental or historical factors. A further limitation was incomplete image coverage. Although the original plan was to task imagery for the whole island in a single breeding season, limited cloud-free conditions, orbital timing and budget availability necessitated extending image acquisition across two seasons. Despite this extension, a proportion of beaches remained without direct image-based counts, requiring a modeling approach. As with all extrapolative endeavours there are weaknesses commonly associated with how well a model can predict into new locations. Our cross-validation highlighted this issue, with several difficult beaches accounting for a large proportion of the observed prediction error. Future efforts could reduce this uncertainty by increasing the volume of input data (*i.e.*, the number of red dots in Figure 2), and with a larger budget, a larger footprint of the island could be tasked and at a higher priority resulting in more input data. Alternatively, efforts could be made to extend the years over which the census covers; as in Laborie et al. (2023) where nine years of imagery were analyzed. However, the later would introduce enhanced interannual modeling complexity, which is especially pertinent following the arrival of HPAI in the region, and the hypothesised interannual fluctuation in its impact.

A further source of uncertainty is the demographic scaling factor applied here to extrapolate breeding estimates to wider population abundance. MacCann's (1985) factor of 3.5, which has subsequently been applied by (Hindell and Burton, 1988; MacCann

and Rothery, 1988; Lewis et al., 1998; Gil-Delgado et al., 2013), is based on life-history data collected several decades ago, while contemporary survival rates and age structure at South Georgia remain poorly known. Recent studies have also applied slightly different values, including 3.47 in Laborie et al. (2023), while recalculation from MacCann's published survivorship tables gives approximately 3.57, reinforcing that the 3.5 reported in MacCann (1985) should be treated as an approximate demographic scaling factor rather than a precise parameter. As contemporary return and detection probabilities for non-breeders are not available, breeding-female abundance should therefore be regarded as the primary directly estimated demographic quantity, with the wider population estimate considered approximate. This is particularly relevant where population age structure or demographic trajectories differ from that of the population from which the factor was derived (Lewis et al., 1998), and, by extension, where contemporary efforts are temporally distant from the original life-history analysis or have recently been impacted by an atypical demographic perturbation. Our census also pools imagery across two breeding seasons and therefore does not explicitly account for recruitment and mortality between years. Given the lack of contemporary demographic data, together with the concurrent HPAI perturbation linked to declines in adult breeding females at South Georgia (Bamford et al., 2025) and significant pup mortality in adjacent South American populations (Campagna et al., 2024; Uhart et al., 2024), further demographic adjustment would imply a level of certainty not supported by the available data. We therefore retained the published scalar of 3.5 throughout.

Further improvements may also come from incorporating dynamic environmental covariates, such as oceanographic or atmospheric variables, that could influence haul-out dynamics and help explain current unmodelled variation. We did not include such variables here because, given limited interannual resampling, highly variable dynamic covariates could have produced atypical correlations, wider confidence intervals and reduced model parsimony.

Previous studies have differed in the degree of comparability between satellite-derived counts and those obtained from other platforms. McMahon et al. (2014) concluded that 50cm GeoEye-1 satellite and ground counts were broadly comparable (1,790 ±306 vs 1991, respectively), based on strong harem-level relationships and the inclusion of the ground count within the satellite confidence interval, although the absolute satellite counts were still 10.1% lower than the ground count. By contrast, Laborie et al. (2023) reported satellite counts that were 2.6% higher than ground counts in their validation area. When comparisons have been made between aerial platforms with a shared top-down perspective, differences have also been observed. Fudala and Bialik (2022) found that 31cm WorldView-3 satellite counts were 1.5-7% lower than corresponding drone counts for breeding females. One possible explanation for the apparent variation among studies is that agreement between satellite and ground counts may reflect not only satellite detectability, but also limitations inherent to ground-based counting itself. Counts made on foot are subject to oblique viewing angles. From such a perspective, identifying individuals in the center of dense harems is challenging, potentially reducing the reference ground count itself. In this sense, comparisons between satellite and UAV imagery may offer a more direct assessment of platform-specific detection differences, because both methods view colonies from above.

The UAV comparison provided an important calibration step in the satellite census framework. Here we found that UAV counts were highly repeatable, whereas satellite counts were more varied and on average 36.7% lower, suggesting that the principal limitation of 50 cm imagery was incomplete visual detection of breeding females, particularly within dense groups or against complex substrate, as was also concluded by McMahon et al. (2014). The top-down viewing angle of UAV imagery may also provide a useful means of quantifying bias in ground-counts. Ground counts are intrinsically limited by oblique viewing angles, which can obscure individuals within the center of dense harems, particularly in a species that tend to lay in close proximity or on top of one another. Paired ground and UAV surveys could therefore be used to model this undercounting and derive adjustments for ground-based censuses.

A further factor that impacts satellite imagery is atmospheric clarity. In some instances, high-level cloud adds a level of haze to the images that is not readily included in satellite companies' assessments of cloud cover and therefore is not clear when images are purchased. The knock-on effect of this is that these hazy images are acquired and the ability to distinguish features on the ground, particularly when lighting levels are lower, is greatly diminished. Where possible, the opportunity to review imagery before purchase would help reduce this source of error. Furthermore, as satellite providers launch ever increasing numbers of higher resolution commercial satellites, the opportunity to collect finer resolution imagery, and being able to select only the clearest (*i.e.*, cloud/atmospheric haze free) images increases, which will help facilitate further studies that will reduce this uncertainty.

We found evidence that repeated counting of satellite imagery resulted in incrementally higher counts with successive iterations. Given the scale of the counts undertaken (*i.e.*, several hundred beaches and >40,000 female seals) this likely stemmed from increased familiarity with the data format, effectively the observer refined their search image, rather than individual familiarity with specific seals on specific beaches. Nonetheless, this underpins the necessity for repeat counts spread over an extended time-period and the need to streamline workflows for future replication. As counts were conducted by a single observer, the presented replicates quantify intra-observer but not inter-observer variability. Although using one experienced observer ensured consistency through the census, future applications should assess agreement among multiple observers to qualify this source of uncertainty.

A further caveat is that the correction factor used herein was itself derived from a limited number of matched satellite-drone comparisons. Only two temporally overlapping acquisitions were available, and both involved 50cm WorldView-2 imagery, which is likely to yield less precise counts than finer resolution WorldView-3 or Legion imagery. As satellites consistently undercount breeding females compared to UAV counts, a single correction factor was applied. However, another consideration is that the magnitude of

this undercount may also vary spatially between beaches, with larger or more densely packed harems being associated with an increase the likelihood of adjacent individuals being unresolved, while differences in substrate contrast, beach topography, shadowing and viewing conditions may further influence detectability. Consequently, a single island-wide correction may not fully capture spatial variation in detection bias. Furthermore, the use of this correction was likely appropriate for 11 of 21 images (52%) acquired at 50cm resolution, its application to the remaining 10 (48%) ~30cm WorldView-3 and Legion images may have resulted in an overcorrection.

Ideally, separate correction factors would have been estimated for different image resolutions. In practice, obtaining overlapping UAV and satellite imagery within a narrow time-window is extremely difficult, especially in cloud-prone sub-Antarctic environments, where clear-skies, safe UAV flight conditions and field access rarely coincide. The correction applied here should therefore be viewed as a proof-of-concept calibration based on the available paired UAV/satellite data, rather than a universally applicable adjustment. The reported confidence intervals therefore capture uncertainty propagated within this framework, but do not encompass all potential sources of methodological or spatially structured variation. To further refine this, additional paired acquisitions across sensors, image resolutions and colony density and beach characteristics will be required to decrease uncertainties in the future.

Conclusions

This study further demonstrates that sub-meter satellite imagery combined with an extrapolative modeling approach provides a repeatable framework for island-wide monitoring of southern elephant seals at remote sub-Antarctic breeding sites. Satellite imagery greatly increases the spatial scale and potential revisit rate of census efforts and can be implemented at a substantially lower logistical and financial cost when compared to traditional vessel-based surveys. Moving forwards, additional efforts focused on calibration and validation are needed to fine-tune these counts, alongside development into automating workflows, which will be needed to reduce the manual processing burden and platform-specific uncertainty.

Applying this framework at South Georgia provides the first island-wide reassessment of the breeding population since 1995. The corrected estimate of 107,630 breeding females, equivalent to an estimated total population of 376,705 seals, confirms that South Georgia remains a global stronghold for southern elephant seals. However, because this reassessment followed the arrival of HPAI, the estimate should be interpreted as a post-perturbation baseline rather than evidence of long-term stability. Recent regional declines associated with HPAI (Campagna et al., 2024; Bamford et al., 2025), and the recent reclassification of the species as Vulnerable (Hofmeyr, 2026) underscore the need for repeated island-wide censuses alongside long-term monitoring over multiple sites. This will enable the detection of spatially heterogeneous population changes within this globally important population.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Material. Further inquiries can be directed to the corresponding author.

Ethics statement

The animal study was approved by British Antarctic Survey AWERB:1109. The study was conducted in accordance with the local legislation and institutional requirements.

Author contributions

CB: Conceptualization, Formal Analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing. EB: Conceptualization, Data curation, Investigation, Methodology, Software, Writing – review & editing. MG: Data curation, Methodology, Software, Writing – review & editing. JC: Data curation, Methodology, Writing – review & editing. NF: Data curation, Formal Analysis, Writing – review & editing. CF-C: Writing – review & editing. MF: Conceptualization, Writing – review & editing. LH: Conceptualization, Writing – review & editing. PH: Writing – review & editing. JC: Data curation, Writing – review & editing. KO: Data curation, Writing – review & editing. PF: Conceptualization, Writing – review & editing.

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Conflict of interest

The author(s) declared that this work was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Generative AI statement

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