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Antipodean wandering albatrosses and white-chinned petrels 2025

Kalinka Rexer-Huber, Edin Whitehead, Kath Walker, and Graeme Elliott

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Report to Department of Conservation, Conservation Services Programme

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Kalinka Rexer-Huber[1*], Edin Whitehead[2], Kath Walker[3], Graeme Elliott[3]

[1] Parker Conservation, 13 Kerr St, Karitane, New Zealand

[2] Northern New Zealand Seabird Trust, 174 Ti Point Road, Warkworth, New Zealand

[3] Albatross Research, 549 Rocks Road, Nelson, New Zealand

* Corresponding author: k.rexer-huber@parkerconservation.co.nz

Cover photo: drone photo with nesting albatross (blue dot), sitting albatross (orange dot) and overlay of 15-m grid (white-dashed line) and block boundaries (purple line)

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Summary

The Antipodean wandering albatross *Diomedea antipodensis antipodensis* has been in decline since a population crash in 2005–07. Declining numbers appear to have been largely driven by high female mortality, but low chick production—with fewer birds breeding and reduced breeding success—has compounded the problem. To tease out the causes of falling numbers of Antipodean wandering albatrosses and identify the effectiveness of potential solutions, research includes an annual visit to the breeding grounds on Antipodes Island. Alongside this core annual study, we present results from a whole-island survey to estimate the size of the Antipodean albatross population nesting island-wide. Alongside the albatross research, we are developing a mark-recapture study for white-chinned petrels. This report describes the results of the annual field programme in the 2024/25 breeding season for both Antipodean albatrosses and white-chinned petrels.

Antipodean wandering albatross. The core annual study involves mark-recapture in an intensively monitored study area and census of the annual count areas. This season's field programme allowed updates to the trend in nesting population size, survival, productivity and recruitment. There are some signs that rates of decline are slowing; for example, the population of breeding pairs was declining at -5.2% per annum (2008–2013), which has slowed to a -1.5% decline in the last decade 2014–2023. The number of Antipodean wandering albatrosses breeding has been roughly stable for the past four seasons, and female survival improved 2010–2019 to approach male survival rates. Female survival has reached 91.2% (most recent 4-year average), but this is still lower than for males (92.6%) and remains lower than females' pre-crash average of 95.9%. Breeding success in 2024 at 68% approached the average pre-crash nesting success of 74%, although the mean 2006–2024 rate remains comparatively low at 63%. However, the actual number of chicks produced remains small, even in years with good breeding success, since numbers nesting remains low. Recruitment is starting to draw from the (much smaller) cohorts produced since the crash, so population numbers will soon no longer be supplemented by higher recruitment rates seen over the past decade.

The last whole-island count of nesting Antipodean albatross took place 1994–96. To update the whole-island estimate, we built on last year's effort which combined ground counts and drone aerial photography producing orthomosaics of 77% of the Antipodean albatross breeding habitat. This season drones were used to obtain photographs of the entire Antipodean albatross breeding habitat on Antipodes Island. Orthomosaic images were constructed from the photos and the number of albatrosses counted and corrected for pretend-nesters (apparently-nesting birds with no egg) using data from nest-content checks conducted during drone overflight (has-egg rate). A second correction used the proportion of eggs not yet laid or nests that had failed at the time the photographs were taken (lay-fail rate), using data from regular visits to the study area. Just 1% of the 1546-ha Antipodean albatross breeding range was not overflown in 2025. Numbers in these unphotographed 22 ha were estimated by categorising nesting-habitat quality across the island, then extrapolating nest densities by habitat-quality class to the unphotographed areas. The number nesting island-wide in 2025 estimated from drone counts (3,546±254 breeding pairs) and the 3,383±201 annual breeding pairs in 2024 provide two successive estimates that together account for biennial breeding and resulting year-on-year differences. This is the first time since 1996 that the number of breeding pairs nesting on Antipodes Island have been comprehensively assessed across the island. The proportion nesting in annual count blocks in 2024 (14%) and 2025 (14.7%) are similar to that recorded 1994–96 (14.9%), indicating that the annual count blocks remain representative of whole-island trends in nest numbers.

Trends in nest numbers and demographic parameters from the core annual study indicate that the population has been approximately stable for the last four years. However, there is so far no evidence of any sustained improvement in Antipodean wandering albatross demography, as required for the population to recover, with tentative improvements recorded here merely slowing the decline. Recommendations include ongoing mark-recapture monitoring of demographic and population-size trends, and research into causes of declines. More-targeted ongoing engagement with fishers is also needed to achieve better bycatch mitigation in line with ACAP best practice.

White-chinned petrel. A mark-recapture study to estimate vital rates, survival in particular, was established in the 2022–23 season. Substantial effort to grow the mark-recapture study this year mean there are now 367 banded white-chinned petrels in 203 marked burrows in the two study areas. For accurate, precise survival estimates this marked population needs recaptures at existing marked burrows for a minimum of two more years. The two years of resighting data obtained to date are not yet enough for mark-recapture modelling to produce a useful survival estimate. However, summary statistics highlighted the importance of quality monitoring data: startlingly low year-on-year return rates recorded last year (24% of birds that had been in the colony the previous season returned) were 76% this year, closer to the return rates expected for annual breeders. Burrow reoccupancy was also better than the year prior, with 39% of burrows marked last year reoccupied this season, compared to just 27% the year before, although reoccupancy still appears low compared to the 44–68% recorded at Antipodes Island in a 2007–11 study.

Keywords: Antipodes, mark-recapture, survival, productivity, population trend, drone, aerial photo-counts, Antipodean wandering albatross, white-chinned petrel

Contents

Introduction

Assessments of the risk of commercial fisheries to seabird populations can be affected profoundly by uncertainty in population size and uncertainty in demographic rate estimates, particularly adult survival (e.g. Richard et al., 2020; Walker et al., 2015). To reduce uncertainty or bias in estimates of risk from fishing, robust population status information is needed (key aspects like survival, productivity, recruitment, trends). Long-lived, slow-breeding seabirds that are vulnerable to accidental capture in commercial fisheries are the focus here: Antipodean wandering albatross *Diomedea antipodensis antipodensis* and white-chinned petrels *Procellaria aequinoctialis*.

Antipodean wandering albatross

The Antipodean wandering albatross is a biennially breeding seabird virtually endemic to the Antipodes Islands, New Zealand. A few pairs also breed on Campbell and Chatham Islands, but those make up less than 1% of the population. A sister subspecies, *D. antipodensis gibsoni*, breeds in the Auckland Islands and has morphological, distributional, and breeding timing differences with *D. a. antipodensis*. Genomic work has further showed distinct breeding units via significant genome-wide differentiation (Foote, 2024).

The New Zealand threat classification lists Antipodean wandering albatrosses as Threatened; Nationally Critical (Robertson et al., 2021), with the five qualifiers Conservation Dependent, Climate Impact, Conservation Research Needed, Island Endemic and One Location.

Since the eradication of house mice from the Antipodes Islands in 2016 (Horn et al., 2022) the island group is free of introduced mammals, and HPAI (Highly Pathogenic Avian Influenza) has not yet reached subantarctic New Zealand (Waller et al., 2025), so the current major conservation threats to Antipodean wandering albatrosses are in the marine environment. The species forages mainly in the Pacific Ocean to the east of New Zealand, and to a lesser extent in the Tasman Sea (Bose and Debski, 2020; Richard et al., 2024b; Walker and Elliott, 2006, 2022). In New Zealand domestic fisheries, there are an estimated 38 Antipodean wandering albatross deaths each year, mostly in surface or pelagic longline gear (Edwards et al., 2023). In international waters, Antipodean wandering albatrosses forage in the southern Pacific Ocean (Bose and Debski, 2020; Parker et al., 2023; Richard et al., 2024b; Walker and Elliott, 2006) in areas where there is a large amount of surface longline fishing effort with no or very little observer coverage (Peatman et al., 2019). Antipodean wandering albatrosses are caught in those fisheries, but the lack of observer coverage and fisher-reported captures makes the overall number of annual captures of Antipodean wandering albatrosses impossible to quantify accurately.

The size of the breeding population has been followed closely since the mid-90s via both intensive monitoring for mark-recapture modelling, and via census in annual count areas (e.g. Elliott and Walker, 2018; Walker and Elliott, 2022, 2005). However, the last whole-island count of Antipodean albatrosses was 30 years ago, conducted 1994–96 (Clark et al., 1995; Walker and Elliott, 2005). We revisit the effort to obtain a new whole-island estimate, with the overall aim to ensure that trends detected via high-intensity monitoring remain representative. This is important because the Antipodean wandering albatross population has been declining since a population crash 2006–07, and is expected to continue doing so unless vital rates recover (Edwards et al., 2017; Richard et al., 2024a; Walker and Elliott, 2022).

White-chinned petrel

White-chinned petrels breed on eight subantarctic islands around the Southern Ocean. Although there are an estimated 1.2 million pairs globally (BirdLife International 2023), they are listed as Vulnerable by the IUCN due to documented declines on land and at sea, and very high rates of bycatch in commercial fisheries (Anderson et al., 2011; Barbraud et al., 2008; Péron et al., 2010; Rollinson et al., 2017).

Domestically, they are listed as Not Threatened (Robertson et al., 2021). The white-chinned petrel population of Antipodes is estimated at 26,400 breeding pairs (Rexer-Huber et al., 2023). A large study on Antipodes Island in 2007–11 allowed key vital rates for New Zealand's white-chinned petrels to be estimated, particularly adult survival (Thompson, 2019), but these estimates are now more than a decade old. Given continued high mortality in fisheries, where it is documented, and poorly documented or unknown bycatch rates in areas where Antipodes white-chinned petrels overlap with substantial fishing fleets (e.g. pelagic longliners and artisanal longline and gillnet fisheries off Chile and Peru) (Rexer-Huber et al., 2025), it is timely to re-evaluate key vital rates at Antipodes to gauge whether there have been important changes. A white-chinned petrel mark-recapture study was re-established on Antipodes Island in December 2022 (Rexer-Huber et al., 2023), specifically designed for adult survival modelling and detecting population change.

Aims and objectives

This work aimed to understand the conservation status and estimate key demographic parameters of Antipodean albatross and white-chinned petrels.

The primary objective of this study was to update the key demographic parameters of the Antipodean wandering albatross (survival, productivity, recruitment and population trend), and to estimate the total population size of the Antipodean albatross on Antipodes Island. A secondary objective was collecting samples towards describing the diet of Antipodean wandering albatross and assessing signatures of nutritional stress. The objective of white-chinned petrel work was to update key demographic parameters (adult survival). Additional work included collection of morphometric data from known-sex Antipodean albatrosses for taxonomic assessment.

Methods

Timing and logistics

Albatross and petrel research on Antipodes Island took place during nine weeks over the period December 2024–February 2025. The first 6 weeks (17 December–24 January) focused on the core annual study for Antipodean albatross (including effort on diet sampling and morphometrics), and on final trials to prepare for whole-island count work. During this first phase we also built on the white-chinned petrel study, resighting and banding for mark-recapture. The following 3 weeks (25 January–15 February) focused primarily on whole-island Antipodean albatross nest counts, while maintaining core work in the Antipodean albatross study area.

The SV *Evohe* brought the researchers (Edin Whitehead and Kalinka Rexer-Huber) from Bluff to Antipodes Island 12–16 December 2024, and picked them up from Antipodes on 16 February and returned to Dunedin 19 February 2025.

Antipodean albatross

Mark-recapture study

In summer 1994 and every year thereafter except 2006, a 29-ha study area on Antipodes Island (Fig. 1) has been visited to count nests, check nest contents, record the band numbers of previously banded birds, band nesting albatrosses, and band chicks just before fledging.

All nests found within the study area are marked and monitored, so that a year later the nesting outcome—failed or fledged—can be determined to estimate productivity. All Antipodean wandering albatrosses found nesting within the study area are double-banded with individually numbered metal and large coloured plastic (darvic) bands, one on each leg. Since 1995 most chicks in 68% of cohorts have also been banded. Ideally, all chicks in the study area are banded, but the proportion of chicks that are banded each year depends on the timing of the field trips, which in turn is dependent on the availability of transport. In 21 of the years since 1994 researchers arrived just before, at, or soon after the date at which the first chicks fledge (26 December) when more than 90% of the chicks are still present and can be banded. In nine of the years since 1994 late trips meant up to 45% of the chicks had already fledged without being banded, and no chicks were banded in 2006 (no trip) or 2020 (very late trip). Arrival on Antipodes for the 2025 season was 16 December 2024 so all chicks produced in 2024 could be banded.

Survival rates are estimated from resightings of banded birds using maximum likelihood mark-recapture statistical methods in program MARK via the R package RMark (Laake, 2013; R Core Team, 2025; White and Burnham, 1999). Following Gibson's albatross analyses (e.g. Elliott et al., 2024), adult birds are categorised by sex and by breeding-status state for modelling: non-breeders, successful breeders, failed breeders and sabbatical birds taking a year off after a successful breeding attempt. Birds in each of these states have different probabilities of being seen on the island, and potentially different survival rates, so the models test whether it is important to estimate survival rates separately for breeding and non-breeding birds as well as estimating resighting probabilities separately for each state. In addition, models estimate the probabilities of transition between the strata; for example, the probability of being a non-breeder one year and then a breeder in the following year. Some transition probabilities are fixed rather than estimated: birds may not change sex; after successful breeding birds invariably become "sabbatical" birds; and they may not transition to "sabbatical" from any stratum other than "successful breeder" (or $psi_{(\text{successful breeding to sabbatical})}=1$; $psi_{(\text{failed breeding to sabbatical})}=0$; $psi_{(\text{non-breeding to sabbatical})}=0$; and $psi_{(\text{sabbatical to sabbatical})}=0$). We created models where survival and resighting probability varied by time, sex and breeding-status state, and tested whether:

- survivorship differed for breeders and non-breeders as well as between years and sexes;
- survivorship differed between years and sexes, but not by breeding status;
- survivorship was constant between years, by sex and by breeding status.

Annual survival and resighting probabilities were then estimated using the best-fitting model.

The number of birds in the study area in each breeding-status state (non-breeders, successful breeders, and failed breeders) is estimated by dividing the actual counts of birds in each state by their respective resighting probabilities and uncertainties estimated from the best-supported model in mark-recapture analysis, propagating uncertainties via the delta method. Sabbatical birds are the only state where this calculation cannot use actual in-colony counts, since the birds are not present on the island. Instead, the number of sabbatical birds is estimated as the number of successful birds in the previous year multiplied by their estimated survival. The estimated total population of birds that have bred in the study area is the sum of the estimates of all states.

The survival and resighting probability estimates assume no emigration although there is detectable emigration (Richard et al., 2024a). In our models, birds that emigrate then subsequently return will contribute to low detectability, and birds that permanently emigrate will contribute to low survival and these models will underestimate survival. However, wandering albatrosses have strong nest site fidelity: a pair's separate nesting attempts are rarely more than a few hundred metres apart, and birds nesting at new sites within a few hundred metres of the study area are usually detected during the census of surrounding country. In other words, the under-estimate is small, unquantified but consistent from year to year.

Trends in the mark-recapture-based estimates of population size are calculated as annual rates of population growth or decline (λ or lambda), using $\lambda = (\frac{N_{t+y}}{N_t})^{\frac{1}{y}} - 1$ where *N* represents the number of albatrosses, *t* the first year in the time series, and *y* the number of years in the time series.

Annual count areas

Since 1994, all the nests in two representative annual count areas (Fig. 1) additional to the study area have been counted most years. Counts are carried out between 6 and 12 February, just after the completion of laying, and as close as possible each year to the same time at each place. A strip-search method is used in which researchers walk back and forth across the area to be counted, each within a 25-m wide strip loaded in a mapping GPS, and count all the nests with eggs in their strip. VHF contact is maintained between searchers to ensure no double-ups or gaps. Every bird on a nest is checked for the presence of an egg, and each bird found with an egg is marked by GPS. All non-breeding birds on the ground are also counted, and they and most breeding birds on eggs are checked for bands. Once the whole block has been counted, the accuracy of the census is checked by walking straight transects (validation lines) at right angles to the strips, checking that all nests within 10–15 m of the transect have a GPS point thus have already been counted.

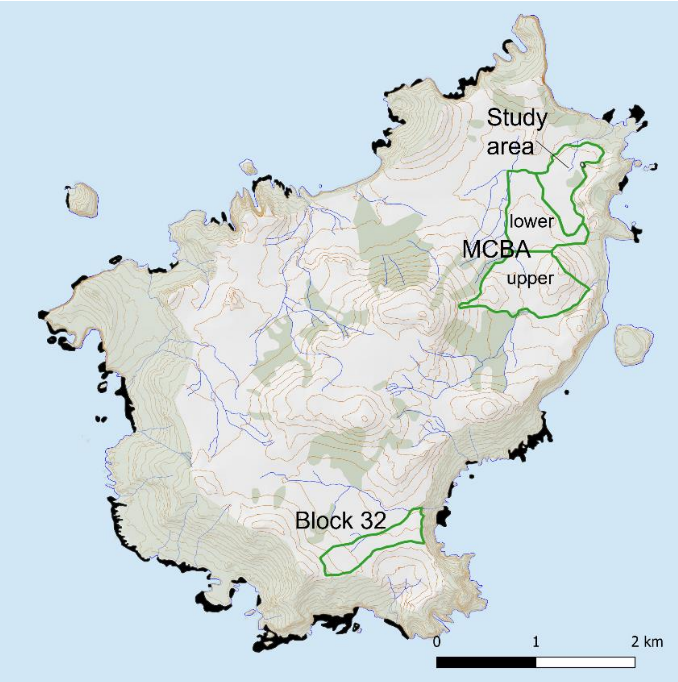

Figure 1. The core annual study for Antipodean wandering albatross takes place in the study area on Antipodes Island (top right) and in the two annual count areas. The area in which albatrosses do not nest is shaded.

By extrapolating the sample of nests counted in annual count areas, we can estimate the total number of pairs of wandering albatross nesting on Antipodes Island. The annual count areas hold 14.9% of all the nests on Antipodes Island, based on the last whole-island count 1994–96 (Clark et al., 1995; Walker and Elliott, 2022, 2002a, 2002b). The 2024 and 2025 whole-island counts of Antipodes Island nests provided the opportunity to examine and update the proportion that are in the annual count areas.

The proportion of the total population in 1994–96 that was nesting in those parts of the island counted in subsequent years is used to estimate the total number of nests on the island each year 1998–2023 using the following equation:

$$\widehat{total}_i = \frac{pairs_i}{proportion_i}$$

Where:

$\widehat{total}_i$ is the estimated total number of pairs nesting in year *i*.

$proportion_i$ is the proportion of pairs counted nesting in 1994–96 that were in those parts of the island that were counted in year i (see column '% of population counted' in Table 2).

$pairs_i$ is the number of pairs counted nesting in year *i*.

For 2024 and 2025, $proportion_i$ is simply the proportion recorded during each year's respective whole-island population estimate.

This estimate assumes that the relative abundance of nests in the counted blocks is constant from year to year, which is supported by the repeated counts of the study area and census blocks on Antipodes Island (Elliott and Walker, 2018), and confirmed by updating counts across the whole island that showed the ratio remains very similar to that recorded 1994–96 (see Results).

Whole-island breeding pair estimate

To estimate the number of nesting albatrosses across the whole island, we took drone photographs over all Antipodean albatross nesting habitat, building on last year's estimate that covered 77% of the island (Rexer-Huber et al., 2024).

The technique and drones used in both 2024 and 2025 were the same, as follows. The 1,546 ha nesting distribution was subdivided into 138 drone blocks in qGIS, each covering ~11 ha or one battery-worth of flying time in moderate wind conditions, for systematic aerial photography by drone (Fig. 2).

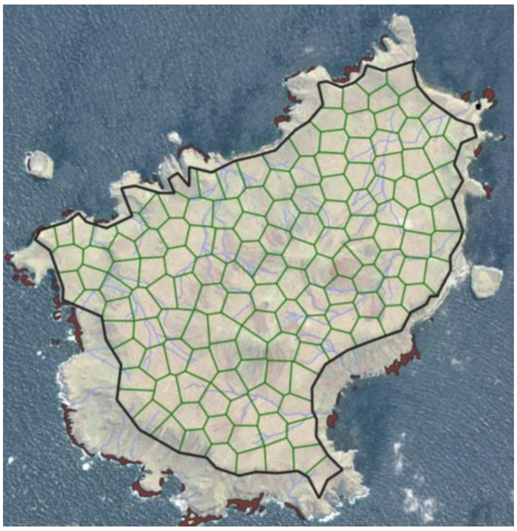

Figure 2. Drone blocks (green polygons) for coverage of Antipodean wandering albatross nesting distribution (bold line)

Two DJI Mavic 2 Pro drones were used. Drone flight was programmed individually for each block in the flight planning software UgCS (https://www.ugcs.com) to obtain overlapping nadir photographs of the entire area at an appropriate ground resolution (1.3 cm/px, via flight no faster than 8.25 m/s, taking a photo every 16.6 m in order to get 65% front and side overlaps). The 65% overlap was sufficient to produce orthomosaic images using photogrammetry, and the 1.3 cm/pixel resolution was sufficient to confidently identify albatrosses on the ground and distinguish whether they were on a nest or not.

Drone flights started earlier than last year, to capitalise on what is often more-stable weather in January. However, starting before the end of lay means some nests will be missed, so we started 25 January (when 60% of eggs are laid, average of last 9 years) to ensure a decent sample of nests for lay-fail correction modelling (see below). To fly a block, the pilot and ground-counter rushed to a take-off point when a weather window opened. Photography was prioritised in areas with no 2024 count data, where possible. In practise, destinations for drone flight were determined by wind and fog with areas somewhat sheltered from the day's wind direction, and areas below the low-cloud layer the only blocks flyable at times. All but three of the blocks could be overflown before the boat arrived to remove the team from the island. Some practicalities of drone flight on Antipodes are discussed in Appendix A.

Best-practise drone use around wildlife requires careful assessment of the risk of adverse effects on animals (Borrelle and Fletcher, 2017; Hughes et al., 2018; Mustafa et al., 2018; Rexer-Huber and Parker, 2020). No concerning animal responses were seen during trials in 2022 and 2023 (authors' unpubl. data) nor during drone flights in 2024 (Rexer-Huber et al., 2024). Nonetheless, we monitored carefully for animal responses to the drone during all flights, particularly during take-off and landing when drones come nearer to animals on the ground.

A major assumption involved in aerial-photo nest counts is that all apparently incubating birds are breeding (nest contents bias). Because a highly variable proportion of birds that look like they are incubating do not in fact have an egg (range 16–29% during 2023 trials and 13–51% in 2024) (Parker et al., 2023; Rexer-Huber et al., 2024), we focussed on collecting nest-contents data concurrent with drone photography. During drone flights, the ground-counter collected concurrent nest-contents data in transects across the area being droned. Nest contents were recorded for all apparently-nesting albatross (ANA; sitting on nest mounds in incubation/brooding posture.). The nest correction factor for a given area is calculated as the proportion of ANA that are sitting on an egg at the time nests are overflown (has-egg rate); that is, calculated as

$$has\ egg\ rate = \frac{n\ on\ egg}{total\ \mathrm{ANA}_{\mathbf{ground}}}$$

where total $\mathrm{ANA}_{\mathrm{ground}}$ is $total\ \mathrm{ANA}_{\mathbf{ground}} = n\ on\ egg + n\ on\ empty$

The has-egg rate variance (lower and upper 95% confidence intervals) is calculated using a t-distribution instead of a normal distribution to reflect small sample sizes.

In addition, a 'lay-fail' correction was made for nests that might have failed before, or for eggs that might be laid after the block was photographed. The lay-fail rate (or the proportion of the total number of nests laid that are active on each day of the season) was derived from the closely monitored study area where the total number of eggs laid by the end of the laying season was known, and the number of active nests was checked every few days from the start of laying until the end of the drone-based photography. Lay-fail correction curves were generated by fitting spline curves to the relationship between the proportion of nests that have eggs and date using R and the glmmTMB package (Brooks et al., 2017; R Core Team, 2025), providing estimates of the lay-fail correction for every day of the drone-based photography as well as their standard errors.

Drone-photo counts

To check coverage and image quality while on the island, image positioning was viewed using the 'import geotagged photos' tool in qGIS to look for gaps and ensure block coverage; further, image file sizes smaller than 10,000 kb (typical of images affected by cloud cover) were checked individually to determine whether image quality was sufficient to detect albatrosses.

Orthomosaics were made off-island (WebODM; https://www.opendronemap.org/webodm/). Between 305 and 618 photos were required to cover each block (mean 471 photos) and resulting orthomosaics were 0.43–3.77 gb in size. All but four blocks had full coverage, with composite images of suitable quality (not clag-affected); one orthomosaic had a hole partially covered by neighbouring blocks, and three blocks were not flown.

Orthomosaics were each counted in qGIS, overlaying the drone block boundaries with a 15-m grid for count accuracy. Image quality at this year's slightly higher 1.3 cm/px resolution was good enough to allow an experienced observer counting every albatross to easily differentiate apparently nesting albatrosses

(ANA, or birds that look like they are incubating) from albatrosses *not* on nests (sitting or standing) and from the few remaining fledglings. Categories were ANA; BOG sit (bird not on nest, sitting); BOG stand (bird not on nest, standing), fledgling. Giant petrels could also be distinguished (see Appendix B). Counting all orthomosaics took 13 days.

To estimate the number of breeding pairs in each block in 2025, the number of albatrosses on nest in each orthomosaic was adjusted to account for pretend-breeders and earlier nest failures and eggs yet to be laid. That is, the drone-photo count of apparently-nesting albatrosses (ANA_{drone}) in each block was corrected with the has-egg rate at the time photos were taken to provide the number of active nests at the time the block was droned. The number of active nests was divided by the lay-fail rate for the day on which each block was droned:

$$active\ nests = \mathrm{ANA}_{\mathrm{drone}} \times has\ egg\ rate$$

$$breeding\ pairs\ 2025 = \frac{active\ nests}{layfail\ rate}$$

The standard error of the number of breeding pairs was estimated using standard errors of the has-egg rate and lay-fail correction and the delta method for propagation of errors. Finally, the total number of pairs breeding across all drone-count areas in 2025 was estimated as the sum of 2025 pairs from all drone blocks.

<u>Nesting-habitat quality extrapolation</u>
Despite best efforts, 1.4% of the 1,546 ha breeding range (22 ha) had not been overflown by the time the team were picked up from the island. To estimate the number of pairs breeding in not-counted areas, we used last year's nesting-habitat quality classifications (low, medium or high quality nesting habitat for each drone block island-wide) (Rexer-Huber et al., 2024), then used habitat-class nest densities to estimate nest numbers in the three uncounted blocks. That is, using the number of breeding pairs in each counted block in 2025 as above (failure-rate and nest-contents corrected) and the GIS-derived area of each block, we calculated average nest density for each habitat-quality class. Average densities were applied to the area of the three blocks lacking photographs.

Antipodean wandering albatross samples and measures

<u>Diet: feathers</u>. For a study aiming to assess changes in diet and stress during the non-breeding period using corticosterone and stable isotope analysis (Brendon Dunphy, University of Auckland), feather samples were taken from 20 fledglings, 15 adult females and 14 adult males.

All samples and data derived from it will be stored and managed according to protocols agreed between the Department of Conservation and Te Rūnanga o Ngāi Tahu.

<u>Taxonomy: standard measures</u>. Contributing toward re-assessment of Gibson's and Antipodean wandering albatross taxonomy, we took standard wandering albatross measures from known-sex Antipodean albatrosses. The same protocol was used as for Gibson's albatrosses to enable direct comparison. 13 females and 11 males were measured. This work is part of a paper being progressed by Kath Walker.

White-chinned petrels

The mark-recapture study of white-chinned petrels that was started in 2022 to follow trends in demographic parameters (especially survival) was developed further, marking more study burrows in the two established study areas (Fig. 3).

New study burrows within the existing study colony areas (Galloway Toe and Polar Front, Fig. 3) were identified by first using a burrowscope to find whether burrows were occupied, and for assessment of their suitability as long-term study burrows. Some burrows were excluded if the nest chamber was too far beneath the surface to reach by access plug, or too shallow to withstand regular checks over time without being damaged by the investigator. As before, burrows were all marked with blue cattle tags wired to the substrate near the burrow entrance, with small access plugs to reach the nest chamber (cut by hand saw) in almost all marked burrows.

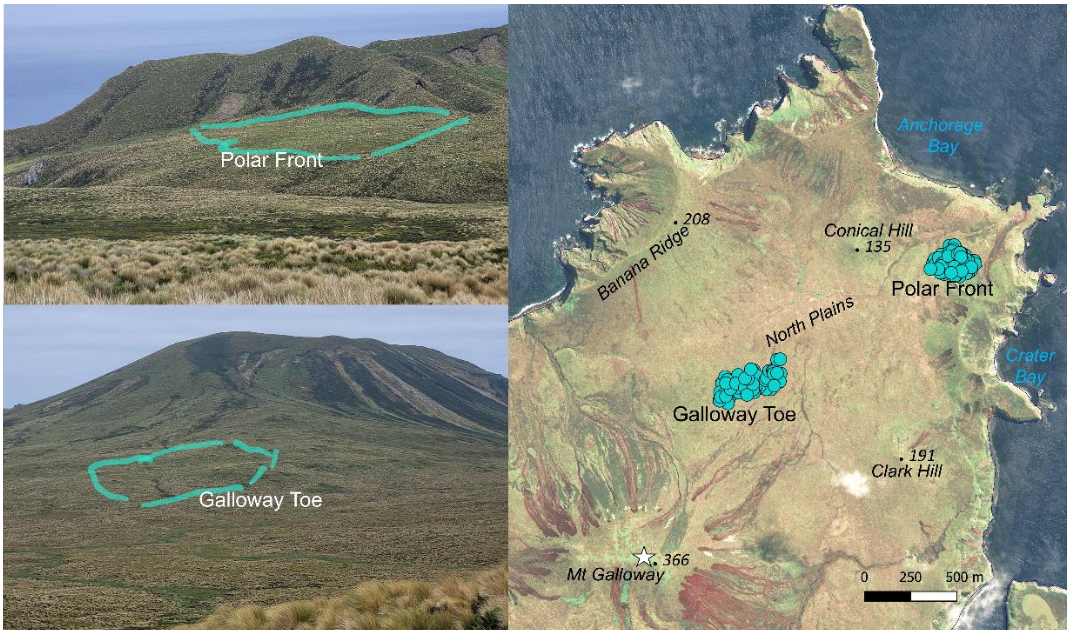

Figure 3. White-chinned petrel study areas on Antipodes Island (in blue). Top left: Polar Front area viewed from Conical Hill. Bottom left: Galloway Toe area from Banana Ridge.

The existing study burrows were checked for returned banded birds. The first bird found in an active burrow had twink applied to the forehead for ease of identifying changeovers. In all cases—existing and new marked burrows—burrows were revisited every 5–10 days (based on the observed changeover routine at this time of year) for a quick inspection by burrowscope to check for a newly-returned mate lacking twink from handling, until the mate was found and also banded, or the chick was left alone in the burrow when brooding ended. Most pairs have both birds banded.

Results

Antipodean albatross

Mark-recapture modelling

A range of mark-recapture models were compared using AICc (Table 1). Antipodean albatrosses in the colony are seen in different states (non-breeders, successful breeders, failed breeders, and sabbatical birds taking a year off after a successful breeding attempt). The best supported multi-state model showed that survival rate differs over time and by sex, with resighting probability differing over time between states and sexes, and the probability of transitioning from one state differs between states, sexes and over time (model 1 in Table 1). Models distinguishing survival for breeding and non-breeding birds had poorer fit to the data (model 2, Table 1). There was less support again for models not distinguishing survival of females and males (model 3, Table 1).

Table 1: Comparison of the top models of Antipodean albatross survival (S). All models have detection probabilities (p) and transitions (Psi) that vary with time, sex and state { p(~-1 + time:sex:stratum)Psi(~-1 + time:sex:stratum) }, where strata are the following breeding-status states: breeders, successful breeders, failed breeders, and sabbatical birds.

Model	npar	AICc	ΔAICc
1. Survival varies with time and sex { S(~-1 + time:sex) }	482	43942.01	0.000
2. Survival varies with time, sex and for breeding and non-breeding birds { S(~-1+time:sex:breed) }	542	43979.74	37.7261
3. Survival varies with time { S(~-1 + time) }	241	44316.33	374.3206

Population size trend from mark-recapture

The size of the breeding population in the study area estimated by mark-recapture (Fig. 4) was increasing up until 2005 at an average rate of about 4.8% per annum for both sexes (1996–2005). The increase was initially slow, then rapid in 2002–2005. After 2007 the population of breeding pairs declined, initially very rapidly with an average 2008–2013 of -5.3% per annum. In recent years the rate of decline has abated to an average of -1.4% per annum decline for both sexes (2014–2023), and the population of breeding females has been roughly stable for the last three years (Fig. 4).

The sex ratio before 2005 was about 1:1 but for the following decade averaged 1.5 times as many males as females (average 2008–2018). The sex ratio appears to have improved a bit since 2019, sitting around 1.4 times as many males as females for five successive years.

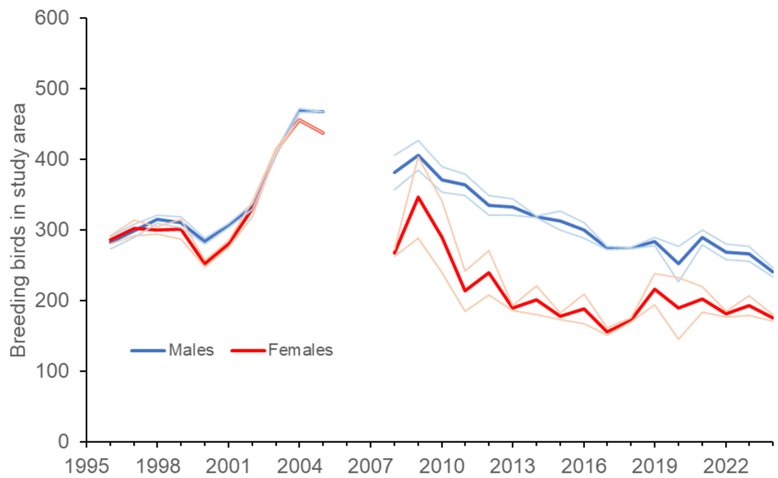

Figure 4. The number of breeding female (red) and male (blue) Antipodean wandering albatross in the study area on Antipodes Island estimated by mark-recapture. Pale lines show lower and upper 95% lower and upper confidence limits. Note: population estimates by mark-recapture are not reliable for the last two year's data, so results for 2023 and 2024 should be taken with caution

Survival

Adult survival varied around a mean of about 95.9% up until 2004 and during this period male and female breeder and non-breeder survival was not significantly different. Since 2004 annual survival of both males and females has declined, with female survival significantly lower and more variable than that of males (Fig. 5). Since 2014 female annual survival has been particularly variable with both the lowest and second highest female survival occurring in that period. When that volatility is smoothed via 4-year rolling averages, average female survival appears to have increased since 2010–2019 to approach male survival rates. However, although average female survival has reached 91.2% (most recent 4-year average), this is still lower than for males (92.6%) and remains lower than the pre-crash average of 95.9% (Fig. 5).

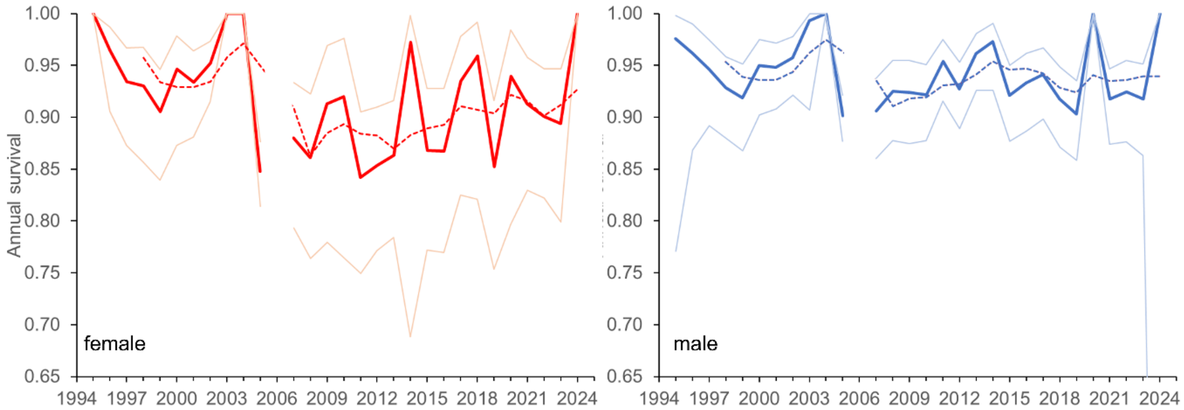

Figure 5. Estimated annual survival of female (red, left) and male (blue, right) Antipodean wandering albatross on Antipodes Island since 1996. Dashed lines represent 4-year rolling means; pale lines show lower and upper 95% lower and upper confidence limits. Estimates from the best-supported survival-by-sex model. Mark-recapture estimates are not reliable for the last two year's data, so results for 2023 and 2024 should be taken with caution

Productivity

Nesting success in 2024 was 68%. Nesting success has been higher in the three most recent cohorts than it has been for several years. However, this spike has not improved the average nesting success since the 2006 crash; the current average remains 63%, notably lower than the average pre-crash nesting success of 74% (blue in Fig. 6). The number of chicks produced in the study area continues to be much lower than that before the crash (red in Fig. 6) mostly because of the much smaller size of the breeding population.

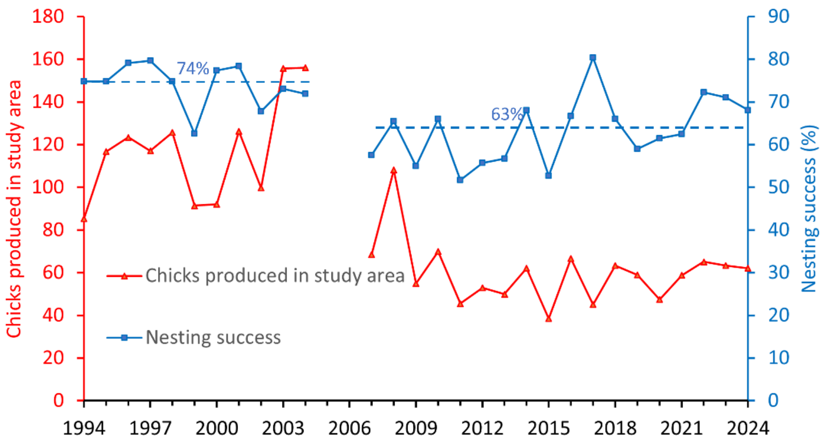

Figure 6. Nesting success and the number of chicks fledged from the study area on Antipodes Island since 1994. The dashed lines indicate average nesting success in two periods, 1994–2004 and 2007–24.

Population trend from annual count areas

Nests were again counted in the study area, Block 32 and MCBA (Fig. 1) in 2025. From these counts the total number of breeding pairs on the island was estimated (Table 2). After an increase between 2000 and 2005, the number of nests dropped sharply by about 36% between 2005 and 2007 (Fig. 7). In the following decade this reduction slowed, and since 2017 the numbers of pairs nesting have remained fairly similar from year to year (Fig. 7). There is no sign of recovery.

Table 2: Antipodean wandering albatross nests with eggs in February in three annual count areas on Antipodes Island in 1994–2025. The estimated number nesting on the island uses either observed ratios (for 1994–96, 2024, and 2025, for which there are direct data on the proportion nesting in those areas relative to island-wide counts) or estimated ratios interpolated over the interval between island-wide counts (for 1997–2023, estimated ratios grey-shaded). For 2024 and 2025, we also show corrected counts (*italicised*), correcting February nest numbers using lay-fail rates from the intensively-monitored study area.

Year	Study area	Block 32	MCBA total	Lower MCBA†	% of Feb population counted	Total counted	Estimated nests on island
1994	114	125	544*		14.9	783	4635
1995	156	185	482*		14.9	823	5757
1996	154	133	418*		14.9	705	5148
1997	150		464*		12.1	614	5074
1998	160		534		12.1	694	5736
1999	142		479		12.1	621	5132
2000	119	130	462		14.8	711	4813
2001	160	141	443		14.7	744	5048
2002	148	178	605		14.7	931	6330
2003	214	187	608		14.7	1009	6876
2004	216	249	755		14.6	1220	8332
2005	211	186	613		14.6	1010	6913
2006							
2007	119	127			5.6	246	4393
2008	165	135			5.6	300	5357
2009	98	120			5.6	218	3893
2010	106	101			5.6	207	3696
2011	88	108			5.6	196	3500
2012	95	104	345	145	14.4	544	3782
2013	88	93	297	127	14.4	478	3330
2014	91	103	341	130	14.3	535	3736
2015	73	86	291	124	14.3	450	3149
2016	100	92	291	97	14.3	483	3388
2017	57	82	230	85	14.2	369	2594
2018	97	97	315	136	14.2	509	3586
2019	99	96	276	107	14.2	471	3326
2020	75				2.8	75	2679
2021	94	89		124	9.6	307	3198
2022	90	74	274	113	14.1	438	3114
2023	90	87	310	112	14.0	487	3471
2024	88	80 (*84*)	296 (*324*)	121	14.0	464 (*496*)	3313 (*3383*)
2025	76	94 *(107)*	246 *(274)*		14.7	416 *(457)*	2834 *(3546)*

* estimated (see Walker and Elliott, 2002b)
† Lower MCBA is a subarea of the overall MCBA count block, shown to enable comparison when only a part-count of MCBA was possible

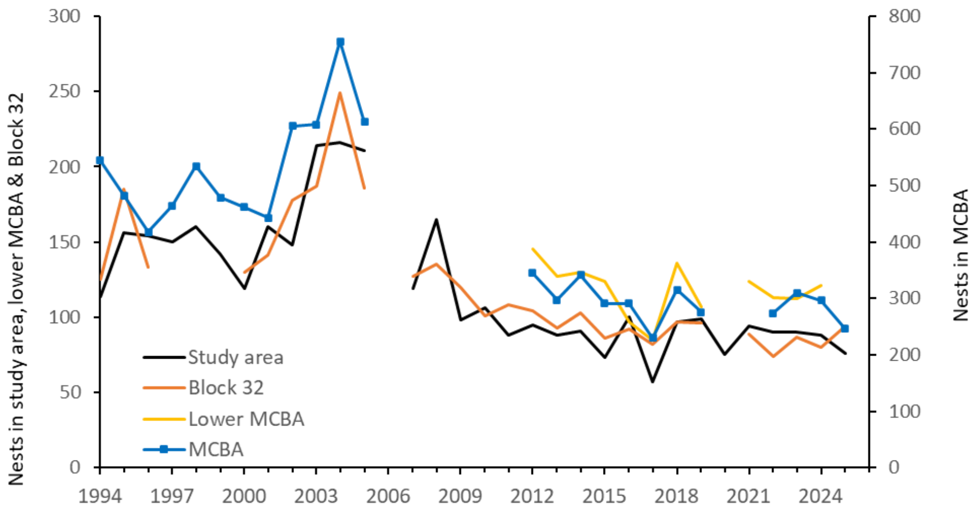

Figure 7. The number of Antipodean wandering albatross nests in three areas on Antipodes Island since 1994. The MCBA subarea 'lower MCBA' is shown to enable comparison in years when only a part-count of MCBA was possible.

The 2024 and 2025 estimates of the number of nesting pairs on Antipodes Island use the ratios observed during each year's whole-island breeding pair estimate (14% of nests island-wide were in annual count areas in 2024, and 14.7% in 2025) (Table 2). Estimates are 220 pairs and 61 pairs greater, respectively, than if they had been estimated based on the ratio from the 1994–97 whole island ground census.

Going forward we recommend using the average of the 2024 and 2025 estimates; that is, a ratio of 14.3% of the island's nests occurring in annual count areas. The Study Area contained 2.7% of all nests 2024–25, MCBA had 8.8% and Block 32 comprised 2.8% of nests island-wide.

Whole-island breeding-pair estimate

Drone photography

During the three weeks available for the whole-island counts (from 25 January to the team's pickup on 16 February), there were six days suitable for drone flight for most of the day and another three days where flying occurred in windows of only a few hours of suitable flying conditions before the low-cloud clag layer descended. In practise this meant a lot of standby, rushing for windows, and sitting watching the clag layer hoping for it to lift. Nonetheless, 135 of the 138 blocks were overflown at least once (Fig. 8), covering 1,524 ha or 99% of the albatross nesting area.

Drones have potential to disturb wildlife, so we monitored carefully for animal disturbance throughout all drone flights. Close observation of the drone during flight (including take-off and landing) revealed little obvious disturbance to wildlife in the area. Over the course of 45 hours of drone flight time, animal responses were limited to:

- head-tilting by an Antipodean albatross on ground to watch the drone landing from where it sat 20 m away (1 occasion)
- drone came within ~5 m of a bird (2 occasions; once skua, once Antipodean albatross) during vertical climb after launching. The drone stopped until the bird flew past.
- Birds followed or flew around the drone (3 occasions; twice skua, once light-mantled sooty albatross), remaining 10–20 m away then leaving.

Nest-contents checks during drone overflight covered ~65 km (Fig. 10), checking 1,307 nests along up to 12 transects per day for 10 days. Nest-contents data showed that on average 60% of apparently-nesting albatrosses had eggs (has-egg rate), with the rest being birds with no egg that appeared to be incubating.

However, the has-egg rate ranged widely from 32% to 72% across the 10 distinct days and areas checked (median sample size 86 nests), and not in an entirely predictable way (e.g. becoming smaller as the breeding season progressed) (Table 3). The variability and unpredictability of the has-egg rate highlights the importance of nest-contents checks being concurrent with drone overflight of a site to keep estimated nesting numbers as accurate as possible.

Failed nests and yet-to-be-laid eggs were accounted for via regular visits to the study area throughout the droning programme. A fitted lay-fail curve provides estimates of the lay-fail correction for every day of drone-based photography (Fig. 9).

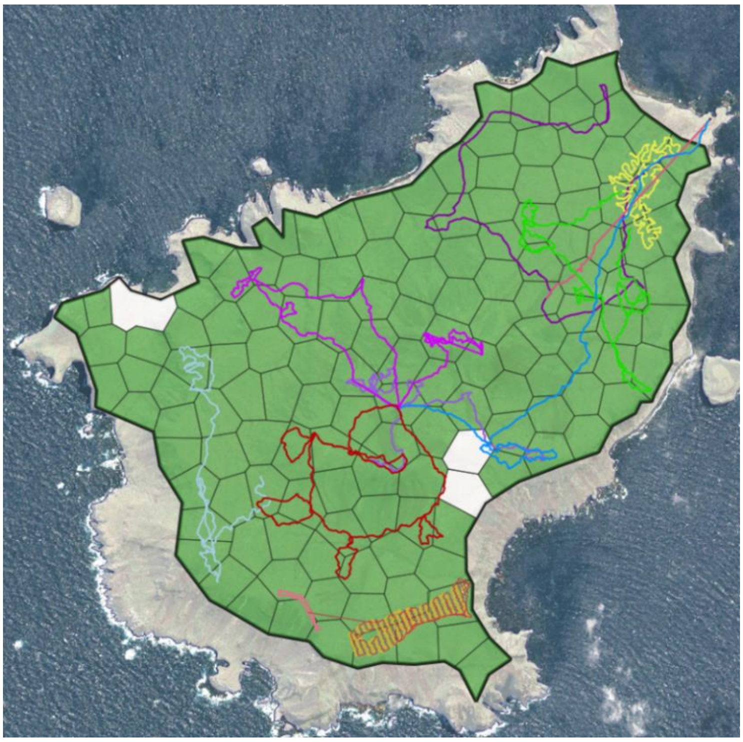

Figure 8. Drone coverage and nest-contents checks for Antipodean albatross across their nesting distribution (bold black line) January–February 2025. Green: blocks with high-quality orthomosaics (complete coverage, no clag or blur issues); White: blocks with no aerial imagery in 2025 (but covered in 2024). Coloured lines: coverage and search effort of nest-contents checks during each day's drone overflight

Table 3: Nest contents of apparently-nesting Antipodean wandering albatross in 2025

Date	Description	Bird on egg	Bird on empty nest	ANA (apparently nesting albatross)	Has-egg rate (bird on egg / ANA)	
					Has-egg rate	SE has-egg rate
25-Jan	SA round	11	23	34	0.3235	
28-Jan	Waterhouse circuit	55	39	94	0.5851	0.0454
30-Jan	Galloway, Central, Ramparts, Windward	53	23	76	0.6974	0.0629
31-Jan	Melville	22	17	39	0.5641	0.0531
1-Feb	MCBA, Clark, Pipit, Galloway	80	42	122	0.6557	0.0386
5-Feb	NSD, Stack, Windward	54	33	87	0.6207	0.0331
6-Feb	Pipit	5	5	10	0.5000	
7-Feb	Melville, Central, Waterhouse	56	28	84	0.6667	0.0329
8-Feb	Reliance, Block32	107	54	161	0.6646	0.1024
10-Feb	SA round	68	26	94	0.7234	

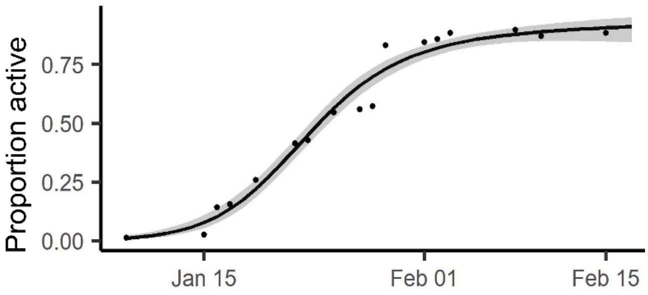

Figure 9. Fitted curve of the relationship between date and the proportion of nests that have eggs in the study area in 2025. Grey shaded areas are 95% confidence intervals

Whole-island breeding pair estimate

Overall, 6,341 Antipodean albatrosses were counted in drone photographs, including all those standing or sitting and obviously not nesting, as well as those albatrosses apparently nesting (Fig. 10). Antipodean albatrosses are spread relatively evenly across the island (Fig. 10), with only seven blocks containing no nests.

Excluding birds not on nests, the 4,389 ANA were corrected with the data from concurrent nest-contents inspections (has-egg rate, Table 3), giving an estimated 2,834 albatrosses on egg in February. Then accounting for nest failures before, and eggs still to be laid after, the drone-photo date (lay-fail rate, Fig. 9), we calculate that there were 3,491 breeding pairs in the 2025 season in the drone-photographed areas (up to 59 breeding pairs per 11 ha block).

Estimating nesting numbers for the three unphotographed blocks (14.4 ha of low-quality habitat and 23.3 ha of high-quality habitat) using the average nest densities in low- and high-quality nesting habitat (0.5±0.1 and 3.1±0.1 nests/ha, respectively, mean±se) added an estimated 55±4 pairs.

Taken together, summed figures from all 135 photographed blocks and the three habitat-estimated blocks gives an estimated 3,546±254 pairs of Antipodean albatrosses breeding island-wide in 2025.

Figure 10. Distribution and density of apparently-nesting Antipodean albatrosses counted on Antipodes Island in February 2025.

White-chinned petrels

The mark-recapture study of white-chinned petrels that was started in the 2022–23 season was improved with another 25 burrows marked, bringing the study to 203 marked burrows containing 367 banded birds. Despite greater effort, yield has been decreasing: we scoped 669 burrows at least once this year, marking 4 new burrows/100 explored, while 552 burrows were scoped at least once in 2024 (hit-rate of 12 marked burrows/100 explored). This is not unexpected, since searches have stayed within the bounds of already-marked burrows (to avoid the study area growing beyond the size achievable in a fieldwork day), meaning that a progressively higher proportion of the burrows available *and* attractive to white-chinned petrels are already marked. Burrow occupancy tells a similar story: burrow occupancy was lower in new (unmarked, non-study) burrows than in previous years (8%, cf 17% and 30% in December 2023 and 2022 respectively), suggesting a diminishing-returns effect where most 'quality' burrows in the searched areas have already been marked.

Existing study burrows were checked for returned white-chinned petrels with bands, with 71 of the 263 birds banded to date recovered, giving an overall resighting rate of 0.27. Last year's concerningly-low year-on-year return rate (24% of birds banded in the 2022–23 season returned in 2023–24) has improved, with 76% of the 94 birds banded last year back in the current breeding season. Burrow reoccupancy rates also appear better than recorded last year (27% of 2022–23 study burrows reoccupied in the 2023–24 season), with 39% of 2024 study burrows reoccupied this year.

Last year burrow-shifting was not detected, despite all burrows in the area around marked burrows being checked, so it is unlikely the low return rates were because birds were missed because they had shifted to other (unchecked) burrows. This year burrow-shifting by 20 pairs was recorded, moving 26 m (median) from the burrow where last recorded. Because of comprehensive burrow searches (inspecting all burrows within the marked-burrow area and in a 50 m buffer around it), it is unlikely that return rates recorded this season were underestimated due to some study birds shifting beyond the area searched. However, it does mean that going forward, researchers need to keep looking for banded birds in nearby unmarked burrows, in case of burrow switching.

The two years of resighting data available so far are not yet enough for mark-recapture modelling to produce an accurate, useful survival estimate. Based on simulation modelling for a similar-scale study on white-capped albatrosses (Rexer-Huber et al., 2018; Roberts et al., 2015), we expect that an estimate produced with three years of resightings data from a marked population this size will provide a more-accurate (albeit preliminary) estimate, with precision improving once four years of resightings can be incorporated.

Discussion

Antipodean wandering albatross

Population size trends/trajectory

After an extended period of decline starting in 2005, the number of breeding-age Antipodean wandering albatrosses has been roughly stable over the last four to five years. This is seen in annual nest counts and confirmed in the estimates derived from mark-recapture, a method that is more sensitive to changes in the population than are simple counts of nests.

Although a stable breeding population is preferable to one that is declining, stable is not enough for Antipodean wandering albatrosses. Simulations predict that without intervention to the current situation, the breeding population will decrease from ~3,300 pairs to 400 breeding pairs after 30 years (Richard, 2021; Walker and Elliott, 2022). Without recovery of key demographic vital rates—survival, productivity and recruitment—to pre-crash levels, population projections show that numbers will continue to decline (Edwards et al., 2017; Richard et al., 2024a). This is seen in wandering albatrosses elsewhere: an albatross population can only grow when survival, breeding and success probabilities are higher than the long-term means (Pardo et al., 2017). Securing the population against stochastic events (landslips like those occurring in 2014, disease like HPAI) and halting the projected decline requires an increase in the size of the Antipodean wandering albatross breeding population, but there is no indication of such an increase in nesting numbers. Even taking into account birds absent because they are on breeding sabbatical, different detection rates for breeding and non-breeding birds, and different survival rates (using data from the mark-recapture study), no growth in the breeding population is detectable. Indeed, it is now two decades since the breeding population was last increasing (in the 2004 season).

Several factors together explain the state of the Antipodean wandering albatross population: low survival rates, differing between males and females; suboptimal recruitment, and productivity. Of these, productivity and recruitment have improved or are stable. Although the average nesting-success rate remains lower (63%) than the average pre-crash rate of 74%, productivity has been relatively high in the three most recent cohorts, varying around 70%. However, more-normal looking productivity rates cannot change that the actual number of chicks produced each year is still much lower than before the 2005 crash, because the number of birds breeding each year is now smaller. Recruitment—the number of birds breeding in the study area for the first time—has remained (on average) steady since 2007, despite the declining number of breeding pairs over that period; this higher rate of recruitment will have been slowing the population decline (Elliott and Walker, 2020). Most birds recruiting in recent years hatched around or just before the 2005–07 population crash, so recruitment going forward will be drawing on the (much smaller) pool of chicks produced since then.

Before 2004 survival rates were similar for male and female Antipodean wandering albatrosses, varying around a mean of 95.9%, but since then female survival has been much lower and more variable than male survival. At 90.2% average female survival over the last decade (since 2012) remains 6% lower than the average in the decade before the population crash, and very low for such a K-selected species (Véran et al., 2007; Weimerskirch and Jouventin, 1987). Further spikes in female mortality would drive further population decline and prevent recovery: wandering albatross populations declined with survival rates between 84% and 92% (stable when survival rates of 96-97% were recorded) (Cuthbert et al., 2004; Pardo et al., 2017; Weimerskirch and Jouventin, 1987). Although the occasional year with relatively high female survival appears to have pushed upward the average 2010–2019, and has been closing the gap between male and female survival rates, the sex difference in survival remains. However, when the most-recent five-year average of female mortality was used in the simulation model developed in 2021 (Richard, 2021; Walker and Elliott, 2022), the improved female survivorship seen to date was not great enough to cause population increase in the Antipodean wandering albatross without other intervention.

Whole-island breeding pair estimate

To contribute to the bigger picture for Antipodean wandering albatrosses, we worked across all parts of the albatross distribution for a whole-island breeding pair estimate. Building on over a decade of planning and several years of targeted method testing and workflow development (Rexer-Huber et al., 2024, 2020; Walker et al., 2023; Walker and Elliott, 2022), we conducted comprehensive, fully repeatable counts underpinned by technologies that allow a more accurate picture of the number of breeding pairs than was possible in the whole-island ground counts of the 1990s (GIS and GPS to ensure areas not missed, drones for completeness of coverage).

The whole island drone-based population size estimate was 3,546 annual breeding pairs in 2025. However, since Antipodean albatrosses are biennial breeders, the two successive estimates of annual breeding pairs in 2024 and 2025 better account for expected year-on-year differences than would a single year's estimate: 3,383±201 pairs breeding in 2024 and 3,546±254 annual breeding pairs in 2025. The 2025 estimate is less precise but we believe more accurate, because in this second year, we were able to obtain direct data on nesting numbers from all parts of the Antipodean albatross nesting distribution, removing the need for extrapolations from habitat-quality-based densities to large uncounted areas, as was the case last year. With whole-island coverage and comprehensive concurrent nest-contents data we have produced an estimate as accurate and precise as possible.

As anticipated, pretend-breeders—birds on nest with no egg—have potential to greatly affect the accuracy of aerial photo-counts. We found that 40% of apparently-nesting Antipodean wandering albatrosses were actually pretend-breeders with no egg, varying substantially by site from 28% to 50%, similar to last year's range of 13–51% (Rexer-Huber et al., 2024). Nest-contents sampling this season was more comprehensive than ever before (median 86 nests per sample) and better linked temporally and spatially to aerial photography. Variability in pretend-nester rates between sites and among years at the same sites remains pronounced, confirming that a single nest-contents correction factor would be inappropriate (e.g. average of sampling across sites, or over time). Instead, nest-contents checks should be concurrent with the photography they are applied to, to reduce the error introduced to aerial photo-counts as much as possible, making estimated nest numbers as accurate as possible.

Updated estimates of island-wide nest numbers also enable the proportions used for ongoing annual monitoring to be updated. Importantly, we find that the ratio used to estimate the number of nests via the annual monitoring approach remains similar to that from the 1994–96 whole island estimate (annual count areas contained 14.9% of island-wide nests in 1994–96; this year, they contained 14.7% of nests). This confirms that the annual count areas have remained representative of island-wide trends over the last 30 years of monitoring, and indicates that the approach used for the annual estimate remains a relevant cost-effective way to provide accurate annual estimates to follow the trend of the island-wide breeding population. Going forward, we recommend applying an updated ratio. Given biennial breeding and two successive years of direct assessment of the ratio to whole-island estimate, it seems appropriate to use the average of the ratios recorded in 2024 and 2024 (14.3%) in future.

White-chinned petrels

A mark-recapture study focused on estimating vital rates, survival in particular, established on Antipodes in 2022 now comprises two study areas with 367 banded white-chinned petrels in 203 marked burrows. This is a marked population large enough for survival estimates that are accurate and precise enough to detect important population changes. In the previous white-chinned petrel study at Antipodes, the 366 white-chinned petrels banded and resighted 2007–11 yielded annual survival estimates of adequate precision after four years of banding and resightings (Thompson, 2019).

This second season of band-resighting checks highlighted the need for quality monitoring data. Last year, only 24% of birds that had been in the colony the previous season were back, a startlingly low rate of return for an annually-breeding species (Rexer-Huber et al., 2024). This year, return rates were 76%, closer to the return rate expected for annual breeders, suggesting that 2023–24 may indeed simply have been a bad year and not the 'new normal'. Burrow reoccupancy was also better than the year prior, with 39% of burrows marked last year reoccupied this season, compared to just 27% the year before, although reoccupancy still appears low compared to the 44–68% recorded in the 2007–11 study (NIWA unpubl. data).

The main thrust of the study going forward should be recaptures at existing marked burrows to build a capture-history dataset of sufficient quality to produce useful survival estimates via mark-recapture modelling. Secondarily, control burrows to assess investigator disturbance should be marked within the study areas (no banding), with a nest failure check at the end of a given trip across all marked burrows to allow comparison of failure rates in study and control burrows. In the coming season, marking control burrows should take precedence over adding to the number of marked burrows with newly-banded birds. Once control burrows are established, building the population of banded birds can take priority again.

We anticipate that marked burrows will need to be visited for at least two more years before band resighting data start giving useful estimates of survival (that is, accurate and with acceptable precision), given the nature of mark-recapture studies for long-lived seabirds like white-chinned petrels. As discussed above, four years of resightings were required for a similarly-sized banded population to yield annual survival estimates of adequate precision (Thompson, 2019).

Recommendations

Antipodean wandering albatross

Overall, some gradual improvements in Antipodean wandering albatross demography appear to have slowed the rate of population decline over the past four years. However, this slowing is fragile, being underpinned by tentative improvements in female survival—which is vulnerable to large fluctuations year to year—and driven by high recruitment rates from before the crash that cannot be sustained since recruitment is starting to draw from the (much smaller) cohorts produced since the crash. While the improvements detected are welcome, on their own they are insufficient to halt the population decline. For Antipodean albatross to have resilience against future significant rapid change—ongoing spikes in fisheries mortality, but also stochastic events such landslips or HPAI-type disease (Elliott and Walker, 2020; Richard et al., 2024a)—requires the breeding population to grow again after a population decline, else shrinking populations become progressively more vulnerable to stochastic events.

With nest numbers in stasis, more than a decade of low chick production, annual mortality remaining high for such a K-selected species, and impending shortage of recruiting birds, the situation for Antipodean wandering albatross is concerning. Monitoring the trends in population size and demographic parameters remains a high priority, as does research into causes of declines. More-targeted ongoing interventions with fishing fleets known to overlap with Antipodean albatrosses is also needed to achieve better bycatch mitigation that is in line with ACAP best practice.

For effective population size/trend monitoring and research into causes, we recommend:

- Ongoing intensive effort to monitor the core marked population to ensure consistency and continuity in the dataset with most power to detect trend and determine the cause of any change detected.
- Ground counts of the three representative blocks to be carried out annually for uninterrupted population trend-monitoring
- Island-wide nesting population estimate be repeated every 15–20 years to ensure the intensively-monitored part of the population remains representative.
- Other potential issues (pollution, diet changes) be explored further.

White-chinned petrels

Ongoing mark-recapture will enable robust trend estimation over time, provided the study is large enough, small enough, and goes for long enough. That is:

- large enough numbers of banded birds to provide accurate, precise estimates of vital rates;
- small enough marked-burrow colony size for thorough, intensive monitoring effort to be feasible; and
- be continued for long enough to yield robust survival estimates.

Specifically, continued effort is needed in the two white-chinned petrel mark-recapture areas now established to build the population of marked birds for reliable survival estimates, and ongoing annual effort for at least two more years for the crucial resighting data underpinning survival rate estimates.

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Appendix A: Drone process practicalities

Timing. This season the weather shifted after 25 January from the dense clag dominating Dec and Jan to be more dominated by south-westerly cycles than previously. Although the timing of nests counts is ideally after all eggs are laid (i.e. from the 2nd week Feb), drone work started earlier to take advantage of the more settled conditions that often occur in Jan, pegged to 25 Jan, being the average date at which 60% of eggs are laid (to ensure data quality for lay-fail rate correction). Therefore, drone flight started before all eggs are laid, but eggs not yet laid at the time of drone flight were accounted for using data from the regular study area nest checks, fitting a curve to the proportion of active nests on each date.

Single-day coverage. On good days we learned that a single pilot with two DJI Mavic 2 Pro drones flown concurrently can fly 25 blocks in a day, including travel time between bocks. To minimise travel time to the southern end of the island (thus maximising the number of blocks flyable when good conditions occurred) we established a base camp at the southern side of Central Plateau, walking there ahead of a good forecast. In practise, arriving on a storm's heels often meant sitting out the end of it until conditions became flyable.

Division of labour. In theory a pilot flying one drone could also do nest-contents checks while their drone is in the air, but affect how well the pilot can monitor drones and animal interactions (affecting flight efficiency and safety). Keeping an eye on the drone also reduces how many nests are able to be checked (affecting quality of nest-contents data). Drone coverage was maximised with a pilot flying—and monitoring—two drones and then rushing to the next launch spot, independently of the nest-checker who could focus on spatial and temporal coverage of transects to best capture variability in the estimates.

Launch position. In steeper areas or areas containing bluffs, drone launch had to be from near the upper elevation. In practise that meant some blocks in steeper country could only be flown one at a time, compared to those over plains and plateaus where 3–5 blocks could usually be flown from a given takeoff point before needing to move.

Wind. In moderate- to high windspeeds, more than one battery was required to complete a programmed flight with enough power to return-to-home RTH partway through a flight and resume the programmed flight after battery replacement. This was relatively common in 2024, so parameter adjustments in 2025 allowed for (among other things) faster flight speeds such that the same areas could be covered using less battery. This battery savings meant that typical wind experienced was better accounted for in 2025; ie. fewer occasions where more than one battery required to cover a block than last year.

As before, the flight software warned of dangerous wind conditions in just light- to moderate winds, and flight could continue with surprisingly strong winds (30–35 kn based on forecast in Windy). However, up to two batteries are needed to complete coverage of such blocks when persisting into the wind. The main issues found when flying in very strong winds were around bluffs (gusts shunting drone into bluffs) and on return-to-home: when RTH trajectory was into a strong head-wind, the calculation of battery required for safe RTH proved insufficient, meaning the drone ran out of battery before reaching the pilot. So if flying in strong winds, be aware that more batteries are required, and plan the takeoff point so the RTH path is not into a headwind.

Focus. At programmed flight speeds a high shutter speed proved necessary. We settled on shutter priority 1/1000. Despite this, a few flights returned blurred images, or parts blurred. The simplest fix was to press the focus button (usually top right on controller) as the drone approached the start of its programmed lines, when returning with a new battery, and occasionally in between.

Appendix B: Giant petrel distribution

Northern giant petrels could be differentiated in drone orthomosaics from albatrosses and skuas. Their distribution across the island is shown below.

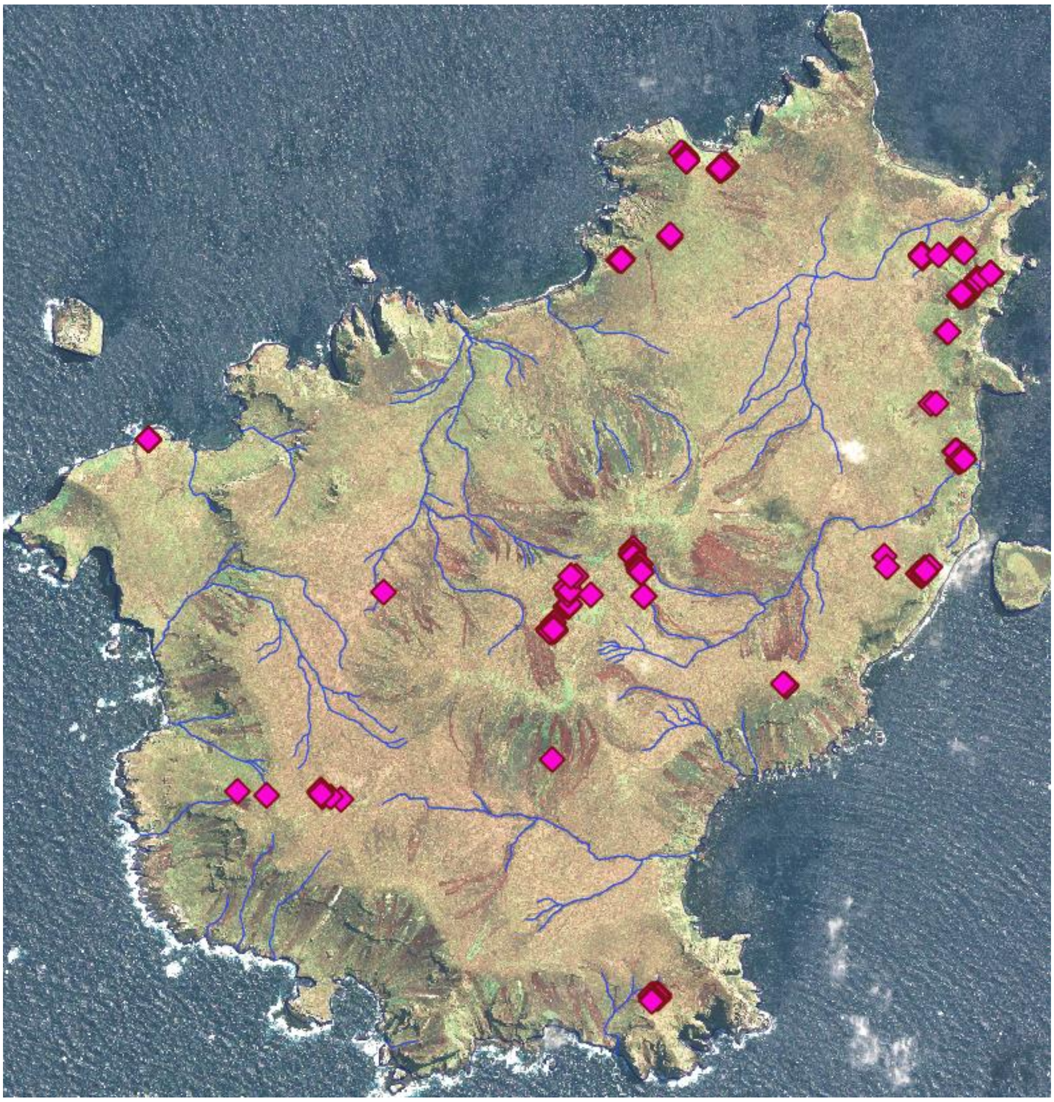

Figure. Distribution and density of northern giant petrels on Antipodes Island in February 2025

The number of giant petrels counted (262 individuals) is only a rough index of the island's population—since chicks could not be separated from adults, the count is NOT of the number of breeding pairs, non-breeding adults or chicks. This is becuase of timing and image resolution: in February, giant petrel chicks are in the process of fledging with many having already fledged and left the island, so counts in February underestimate chick production. Further, at the image resolution achieved (1.3 cm/px, see Methods), it is impossible to distinguish chicks from adult giant petrels, inflating adult numbers to an unknown extent.

Because of count timing and image resolution, these data are primarily useful to show northern giant petrel distribution and current locations across Antipodes Island, providing only a coarse indication of potential population size.