Antipodes Island white-chinned petrel population size and survival study setup

Kalinka Rexer-Huber, Graham Parker, Graeme Elliott, and Kath Walker

Department of Conservation, Conservation Services Programme POP2022-10 Antipodes Island seabird research

Antipodes white-chinned petrel population size and survival study setup

Final report

June 2023

Kalinka Rexer-Huber[1*], Graham C. Parker[1], Graeme Elliott[2], Kath Walker[2]

[1] Parker Conservation, 126 Maryhill Terrace, Dunedin, New Zealand

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

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

Please cite as:

Rexer-Huber K., Parker G.C., Elliott G., Walker K. 2023. Antipodes white-chinned petrel population size and survival study setup. Final report to the Department of Conservation. Parker Conservation, Dunedin. 18 p.

Summary

This study provides an updated estimate of the white-chinned petrel (*Procellaria aequinoctialis*) breeding population size on Antipodes Island. We also detail the setup of a mark-recapture study suitable for estimating key vital rates and detecting population change, adult survival in particular. Lastly, we document blood and feather collection for a wider study on mercury contamination, and deployment and recovery of time-depth recorders for data on dive depth of white-chinned petrels.

Population size estimate. Burrow density is estimated from a representative sample of burrowed areas then corrected for burrow occupancy and extrapolated to the available area of nesting habitat to estimate the breeding population of white-chinned petrels. For an estimate as accurate and precise as possible we built on previous efforts in 2009–11 and 2021–22 (Thompson 2019; Elliott & Walker 2022). To estimate burrow density we used the distance sampling dataset from 2021–22 and expanded the sampling coverage across the whole island, adding 93 transects to a new total of 248 island-wide sampling locations. Distance sampling enabled burrow density estimates that explicitly account for burrow detectability. Occupancy was assessed by inspecting 293 burrows just after laying, calculating rates and corrections using the approach developed for the 2009–11 study (burrow numbers corrected for entrances that are not in fact burrows, and for other species using white-chinned petrel burrows). The area used by white-chinned petrels, with two habitat types distinguishable, was drawn from comprehensive habitat mapping 2021–22. Antipodes Island had an estimated 26,400 (95% CI: 22,200–31,600) white-chinned petrel pairs breeding in Dec 2022 during early incubation. Burrow detectability was different in the two habitat types and occupancy rates differed, so for accuracy the estimate used burrow density, area and occupancy specific to each habitat type. These refinements to 2009–11 and 2021–22 methods result in a population size estimate here that is smaller but more accurate and precise.

Demographic study setup. Population change is more readily detected via intensive study of birds in a representative study population, so we established a mark-recapture study to estimate vital rates, survival in particular. Marked burrows in two study areas contain 169 banded white-chinned petrels. For accurate, precise survival estimates this marked population needs building further, along with recaptures at existing marked burrows for a minimum of three years.

Recommendations. An efficient and effective long-term monitoring strategy could combine annual intensive monitoring effort in a representative study population, as set up here, supplemented by occasional whole-island population size estimates (5–10-year intervals). Ongoing mark-recapture will enable robust trend estimation over time, with whole-island estimates providing occasional more-general overview of breeding numbers.

Contents

Introduction

White-chinned petrels (*Procellaria aequinoctialis*) 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 (Barbraud *et al.* 2008; Perón *et al.* 2010; Anderson *et al.* 2011; Rollinson *et al.* 2017). Domestically, they are listed as Not Threatened (Robertson *et al.* 2021).

Risk assessments that rank seabirds in terms of their risk from fisheries bycatch (e.g., in New Zealand, Richard *et al.* 2020) can be affected profoundly by uncertainty in population size and uncertainty in demographic rate estimates, particularly adult survival (Walker *et al.* 2015). Accurate, robust population data are therefore crucial. Vital rates like survival and productivity can also be important as benchmarks to help understand (bycatch) risk and measure the success of management interventions. Lastly, repeat estimates of population size and survival are needed over time to detect population change.

This project aims to estimate key vital rates and population size for white-chinned petrels on Antipodes Island. At Antipodes, population size estimates in 2009–11 and 2021–22 in combination indicated a breeding population of some 46,000 pairs (Thompson 2019; Elliott & Walker 2022). (Note 2009 here refers to the 2008–2009 breeding season). However, variances were large enough around both estimates to obscure change, if there was any (Elliott & Walker 2022). Large variances around population size estimates are difficult to avoid when dealing with burrowing petrels, but accuracy and precision can be improved via sampling design, and by other aspects of study design (timing, habitat availability, burrow detection, observer bias) (Parker & Rexer-Huber 2015; Dilley *et al.* 2019). Here we aimed to build on the 2009–11 and 2021–22 population size estimates, refining various aspects further to produce a more accurate, precise estimate of the breeding population of white-chinned petrels on Antipodes.

Changes in a population have a better chance of being detected via more intensive study of birds in a representative study population than from whole-island population estimates, as illustrated by the lack of detectable change across previous Antipodes white-chinned petrel population estimates (Elliott & Walker 2022). Indeed, nest count data—like those underpinning population size figures here—are poor at detecting important population changes, with even quite large reductions in population size potentially taking decades to detect in count data (Bakker *et al.* 2018). A study population was closely monitored via mark-recapture 2007–11 to assess survival rates of white-chinned petrels on Antipodes, and showed annual survival of 0.79–0.91 (Thompson 2019). These estimates are now more than a decade old, so given continued high mortality in fisheries it is timely to re-evaluate survival rates at Antipodes to assess whether they have changed in the intervening years.

We had two main objectives:

1. to estimate the size of the white-chinned petrel breeding population on Antipodes Island via a robust island-wide survey; and
2. to establish a mark-recapture study for long-term investigations into demographic parameters, specifically survival.

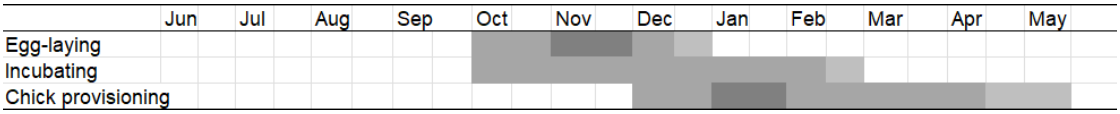
Secondary objectives were to collect dive-depth data from incubating adults, to inform depths and dive rates at which white-chinned petrels could potentially interact with fishing gear; and to collect blood and feathers for a Pacific-wide mercury pollution study.

Methods

Study timing

This work took place on a visit to Antipodes Island between 17 Dec 2022 and 21 Feb 2023. The visit was timed just after the start of the white-chinned petrel breeding season, targeting the end of the main laying period (~14 Dec, NIWA unpubl. data 2007–11; Table 1) when most birds that will breed that year are present and few nesting failures have yet occurred. Burrow occupancy is the most time-sensitive aspect, so burrow inspections took place as soon as possible after arrival, thus were 1–2 weeks after egg lay was complete. Banding for mark-recapture took place during the remainder of incubation and chick-brood (Table 1). Burrow density surveys can take place later without influencing the data in any substantive way, so these took place at the end of the trip (29 Jan–18 Feb).

Table 1. Breeding cycle for white-chinned petrels, following ACAP (2009)



Population size

Study design

To estimate population size in burrowing petrels, a representative sample of burrow density is corrected for burrow occupancy and extrapolated to the available area of nesting habitat (e.g., Lawton *et al.* 2006; Lavers 2015). At Antipodes, we anticipate that accuracy and precision will be maximised by using the careful burrow contents corrections and occupancy calculations from Thompson (2019), the distance sampling and comprehensive habitat mapping from Elliott & Walker (2022), and by building on these with larger sampling effort (coverage expanded to white-chinned petrel habitat across the whole island).

Sampling design took advantage of the thorough spatial analysis of Antipodes habitats by Elliott & Walker (2022). In brief, they classified habitats using a high-quality satellite image of Antipodes, then excluded all habitat types in which white-chinned petrels do *not* occur, and lastly, corrected habitat areas for slope using a digital elevation model. For this study, sampling could therefore be restricted to just the 1073.5 ha that white-chinned petrels occupy (Fig. 2), improving the accuracy of resulting estimates (i.e., not extrapolated to habitat white-chinned petrels do not use or to unsampled habitats) and minimising variance. After excluding areas white-chinned petrels do not occur in, white-chinned petrel habitat contained just one more vegetation type that could be distinguished in satellite images: high-altitude herbfield (8.04 ha). Outside high-altitude herbfield, white-chinned petrel habitats comprise 1065.54 ha (Elliott & Walker 2022) and are referred to from here as primary WCP habitat, largely consisting of low tussock and fern communities. For analyses were therefore consider white-chinned petrel habitat to consist of two vegetation strata: high-altitude herbfield and primary habitat.

Burrow occupancy

Burrow occupancy, or the proportion of burrows that contained a breeding pair (bird on egg), was quantified from four spatially distinct areas on Antipodes that have been previously used to assess occupancy (Sommer *et al.* 2010; Sommer *et al.* 2011; Elliott & Walker 2022) (starred in Fig. 1). All white-chinned petrel burrows found were checked using an infrared burrowscope (Sextant Technologies, Wellington, New Zealand), ensuring that the burrow was inspected throughout. Burrows less than arm's length were not checked by burrowscope, nor were burrows clearly belonging to other species (white-headed petrel *Pterodroma lessonii* or soft-plumaged petrel *Pterodroma mollis*). White-chinned petrel burrows are generally larger than other burrowing petrel species on the island, with more-flattened rather than round entrances, and have tunnels of shallow grade rather than steeply-descending tunnels. In wetter substrates white-chinned petrels burrows typically have a muddy moat at the entrance. Lastly, the presence of white feathers at the entrance is characteristic of white-headed petrels.

The key parameters were whether the burrow was occupied by a white-chinned petrel, and if so, whether the bird was incubating or loafing (bird present without an egg). Some burrow entrances turned out not to lead to a burrow (e.g., collapsed, just a tunnelled cavity under tussock), or led to a nest chamber already inspected, so these were recorded as entrance-not-burrow (ENB). Lastly, white-chinned petrel burrows are sometimes used by other seabirds on Antipodes, so we also recorded when a white-chinned petrel - type burrow contained other species (white-headed petrel or soft-plumaged petrel). White-chinned petrel burrows on Antipodes are large and simple, typically with a single entrance per chamber. Burrows can be inspected in full with confidence that an occupant will be detected, so occupant detection rates were not quantified. To avoid introducing a detection bias, we recorded the few cases where a burrow could not be fully inspected ('unscopable') and excluded these from occupancy estimates.

Burrow inspection took place 23 Dec to 30 Dec 2022. If white-chinned petrel breeding timing remains the same as in the past, burrow occupancy was sampled about a week after egg laying is expected to be complete (~14 December; NIWA unpubl. data 2007–11) (Table 1). Ideally occupancy is quantified when the majority of eggs have been laid, when least failures have yet occurred (Parker & Rexer-Huber 2015). (Note that burrow contents data collected in January and February, while doing study site setup and density transects, were recorded but excluded from calculations of occupancy here as rates become progressively less indicative of the original numbers attempting to breed that season).

The best approach to accurate burrow occupancy calculation for Antipodes white-chinned petrels was workshopped and refined for analysis of the 2007–11 NIWA dataset (Thompson 2019), so we followed that approach. To calculate burrow occupancy, burrows that could not be inspected in their entirety (unscopable) were first discarded. A burrow correction factor (b) to account for entrances that did not lead to a burrow (ENB) was calculated as:

$$b = \frac{burr_{act}}{burr_{total}}$$

where $burr_{total}$ is the total number of fully-inspected burrows and $burr_{act}$ is $burr_{total}$ minus ENB. A small-petrel correction (sm) to account for other species using white-chinned petrel-type burrows (sm petrel) was calculated as:

$$sm = \frac{burr_{wcp}}{burr_{act}}$$

where $burr_{wcp}$ is $burr_{act}$ minus sm petrel. The burrow occupancy rate (c) was then calculated as:

$$c = \frac{burr_{occ}}{burr_{wcp}}$$

where $burr_{occ}$ is the number of burrows occupied by breeding white-chinned petrels (bird on egg).

Because occupancy appeared higher in high-herbfield habitat than elsewhere in white-chinned petrel habitat, we separated the occupancy estimates for population size estimates.

Burrow density

Burrow densities were estimated with distance sampling, to account for detectability decreasing with distance from the line (Buckland *et al.* 2001), since distance sampling improved precision of density estimates relative to previous estimates from count-area sampling (Elliott & Walker 2022).

Distance sampling in the 2021 season was randomised along routes traversing representative habitats, while in 2023 we used a-priori randomisation to increase whole-island coverage. Sampling locations in 2023 were randomised in QGIS 3.4 within white-chinned petrel habitat layers from Elliott & Walker (2022) (Fig. 2). To reach a goal of at least 70 new sampling sites, we generated a surplus of random start points (100 points, minimum separation of 100 m).

At each start point, a transect line was marked using a tape measure (20 m long in 2021, 40 m in 2023). Start points for which part of the transect would fall in unsafe terrain (e.g., off a cliff) were moved uphill until the entire transect could be accessed safely. Each transect was searched thoroughly, moving uphill and looking for burrows under vegetation on the line and in both directions. Burrows less than arm's length were not included, nor were burrows clearly belonging to other species (burrows with white feathers at the entrance, smaller rounder entrances and steeply descending tunnels). Any white-chinned petrel burrow seen from the line was marked by GPS and its perpendicular distance to the line measured to ±10 cm using a pre-marked walking pole. Burrows seen only when the observer was away from the line (e.g., while measuring distance) were not counted, to ensure burrow detections were independent events. Distance sampling at Antipodes took place during 12 Jan–1 Feb 2021 and 29 Jan–18 Feb 2023.

Burrow density was estimated using the package *Distance* in R (Miller *et al.* 2019; R Core Team 2023), combining distance sampling data from 2021 and 2023. The 2021 dataset is as reported in Elliott & Walker (2022), except we excluded transects in a habitat type (clears) in which no white-chinned petrels were found during burrow surveys that year.

The probability of burrow detection as a function of distance was estimated with models using different parametric key functions to define curve shape (uniform, half-normal and hazard-rate), adjusting model fit to data where needed with adjustment terms (cosine, simple polynomial and hermite polynomial) (Burnham & Anderson 2002). Each model's fit to the perpendicular distances was assessed with quantile-quantile plots and Cramer-von Mises goodness-of-fit statistics, and included only if the detection probability coefficient of variation was <20% (Buckland *et al.* 2001). Of these, the model with the lowest AIC value was selected (Burnham & Anderson 2002). Standard errors (SE) and 95% log-normal confidence intervals (CI) are reported throughout.

Number of breeding pairs

To estimate the number of breeding pairs, we first estimated the number of white-chinned petrel burrows in a stratum, calculated as:

$$\widehat{N}_{wcp} = \widehat{D} \times A \times b \times sm$$

where $\widehat{N}_{wcp}$ is the estimated number of white-chinned petrel burrows, $\widehat{D}$ is the estimated density of burrows, A is the slope-corrected surface area of that stratum, b is the burrow correction accounting for entrances that did not lead to burrows (ENB), and sm is the burrow correction accounting for burrows used by small petrels.

The final island-wide estimate of breeding pairs was then calculated using

$$\widehat{N}_{pairs} = \sum_{i=1}^{n} \widehat{N}\mathrm{wcp}_i \times c_i$$

where $\widehat{N}_{pairs}$ is the estimated number of breeding pairs of white-chinned petrels, c is the burrow occupancy rate, and i is one of the vegetation types. Uncertainty in the breeding pairs estimate is calculated using the modelled 95% CI of $\widehat{D}$.

Mark-recapture study

Study areas

One study area, Polar Front, was sited to include a quadrate where birds were banded in the 2007–11 research, for comparability and since birds banded during that study might be recaptured. The area is an almost-flat plateau at 70–90 m asl at the northeast end of North Plains (Fig. 1). (The other two 2007–11 quadrates have become unsuitable, one now having negligible white-chinned petrel numbers and the other being partly demolished by a slip in 2014.) The other study area selected is toward the centre of the island on the lower slopes of Mt Galloway, at 100–130 m asl (Galloway Toe) (Fig. 1). This site also appears representative of white-chinned petrel habitat, is in an area less likely to slip, and is distant enough from Polar Front to be spatially independent but still close enough to the base hut for regular burrow checks to be practical.

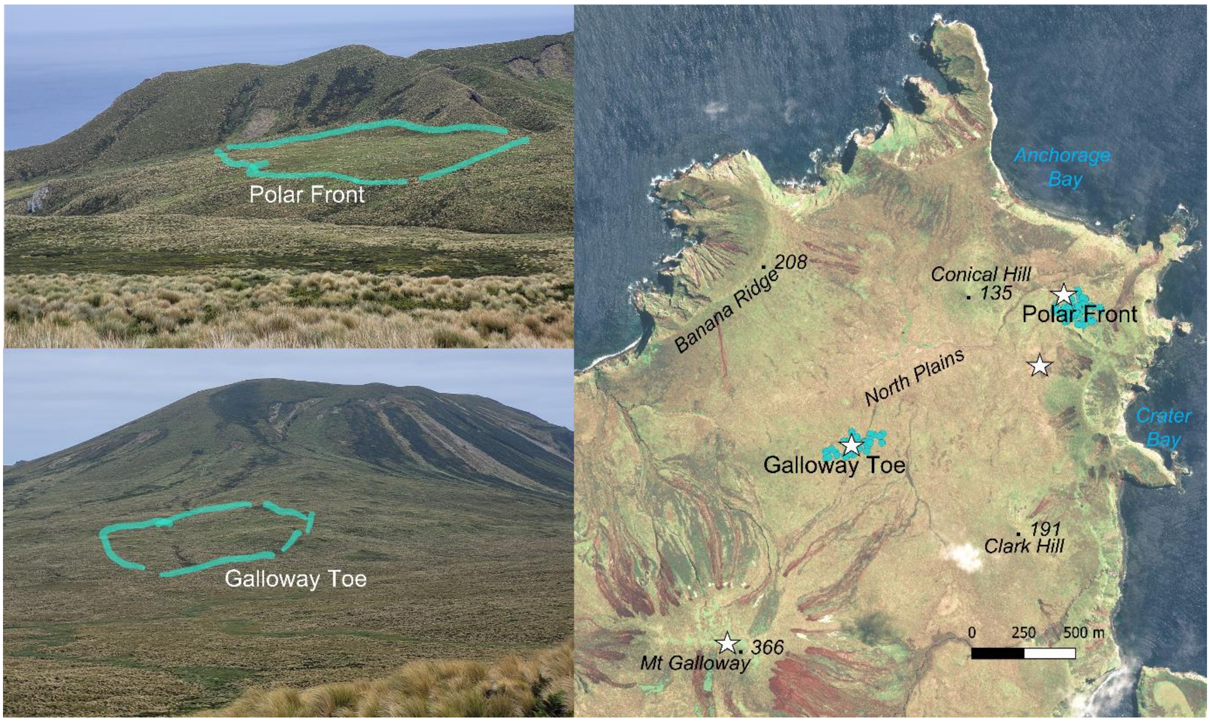


Figure 1. White-chinned petrel study areas (in blue). Top left: Polar Front area viewed from Conical Hill. Bottom left: Galloway Toe area from Banana Ridge. Burrow occupancy areas are starred.

Marked burrows

To establish a study population of banded white-chinned petrels in accessible marked burrows, we first inspected burrows by burrowscope to find occupied burrows and assess 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.

In most cases the nest chamber is well beyond arm's reach, so an access plug is needed. A fibreglass pole probe was used together with the burrowscope guiding its placement to identify the best location for an access plug; that is, within arm's reach but not on top of the nest mound. A pruning saw was used to first cut a tightly-fitting plug into the peat, taking care to remove it as one solid piece, then to cut access into the tunnel/chamber. Study burrows had numbered UV-stable plastic tags wired near the entrance and were marked by GPS. Landmark photographs were also taken to aid burrow relocation if needed.

In marked burrows incubating birds were extracted by hand and banded. Two contour feathers were taken for molecular sexing, then the bird was released back via the burrow entrance. The first bird of a pair was twinked on the forehead to avoid re-extraction. Burrows were revisited every 4–5 days, with just a quick burrowscope check for changeover (newly returned mate lacking twink) to minimise disturbance, until the mate returned and could also be banded.

Dive depth recorders

Time-depth recorders (TDR; G5 Long-Life 10bar 24MB; Cefas Technology, Lowesoft, UK) were fitted on 15 white-chinned petrels using a custom tarsus mount. The best chance of recovering the loggers is during late incubation and the chick-guard period, when adult changeover rates are highest, so we deployed them as soon as possible after arrival. Birds with a tracker were marked with white paint (twink) on the head, and burrows checked by burrowscope every second day until the bird had left and returned. Where possible, the mate also had a TDR fitted. Eleven TDRs were recovered, recording the diving depths reached on flights lasting up to 25 days. The four that did not come back (including one that was redeployed on a new bird) should continue recording, at lower resolution, until next summer.

We also deployed tracking devices (geolocators or GLS; Intigeo C330; Migrate Technology, Cambridge, UK) on 16 birds, fitting GLS to the metal band with UV-stable cable ties. GLS will be recovered next summer, with at-sea movements for the full annual cycle as well as during the interval that complements the depth logger records (for the 14 that had also carried a TDR).

Results

Population size

Burrow occupancy

Burrow contents were inspected in 293 burrows at the ideal time in the breeding season. Only nine burrows were discarded as unscopable (3%). Similarly, only 3% of white-chinned petrel burrows on Disappointment Island could not be checked in entirety (Rexer-Huber *et al.* 2017), and 5–7% on Antipodes Island were unscopable in previous studies (NIWA unpubl. data 2007–11).

Some entrances do not lead to burrows, and other petrel species are sometimes found nesting in white-chinned petrel-type burrows. Accounting for entrances that did not lead to burrows (ENB, entrances to burrows that were collapsed, eroded or already-inspected; n=29) gives a correction factor *b* of 0.8979, where *b* is the proportion of the total inspected that were WCP-type burrows, not ENB. Amongst the 255 WCP-sized burrows, 35 had white-headed or soft-plumaged petrels present, so the correction factor (*sm*) required to subtract burrows in use by small petrels is 0.8627. After applying this white-chinned petrel vs other-species proportion, white-chinned petrel occupancy *c* is therefore estimated to be 0.3818 (Table 2).

Broken down into occupancy sampled in each white-chinned petrel habitat stratum, breeding bird occupancy in 2023 was 0.3161 in primary white-chinned petrel habitat, and 0.6304 in high herbfield habitat (Table 2).

Table 2. White-chinned petrel WCP burrow status and burrow occupancy rates in December 2022 on Antipodes Island

	scoped	known (scoped-unscopable)	burrows (known-ENB)	***b*** (burrows/ known)	WCP burrows (burrows-small petrels)	***sm*** (WCP burrows/ burrows)	***c*** (occ by breeder/WCP burrows)
high herbfield	61	61	52	*0.8525*	46	*0.8846*	*0.6304*
primary WCP	232	223	203	*0.9103*	174	*0.8571*	*0.3161*
Total	293	284	284		220		*0.3818*

Scoped: the total number of burrows inspected, known: scoped minus the number of burrows that could not be inspected in full, burrows: known minus the number of entrances that did not lead to a burrow (ENB), *b*: correction factor, dividing burrows by known, WCP burrows: burrows minus the number containing small petrels; *sm*: correction factor, dividing WCP burrows by burrows; *c*: burrow occupancy, the proportion of WCP burrows occupied by a WCP on egg

Burrow density

A total of 248 transects were visited to assess burrow density; 155 twenty-meter transects in the 2021 season and 93 forty-meter transects in 2023 (Fig. 2). A total of 493 burrows were detected from the line. Burrow detection probabilities varied among vegetation strata, from 0.28 ± 0.02 in high herbfields to 0.43 ± 0.02 in primary white-chinned petrel habitat.

The best detection function identified by distance sampling was half normal with cosine (2) adjustments, which showed excellent fit of the model to the data (Cramér-von Mises test for goodness of fit, p=0.85; Fig. 3, Table 3). Burrows were detected up to 14 m from the transects and the distance sampling model estimated that 25% of the burrows within this distance were detected (Table 3).

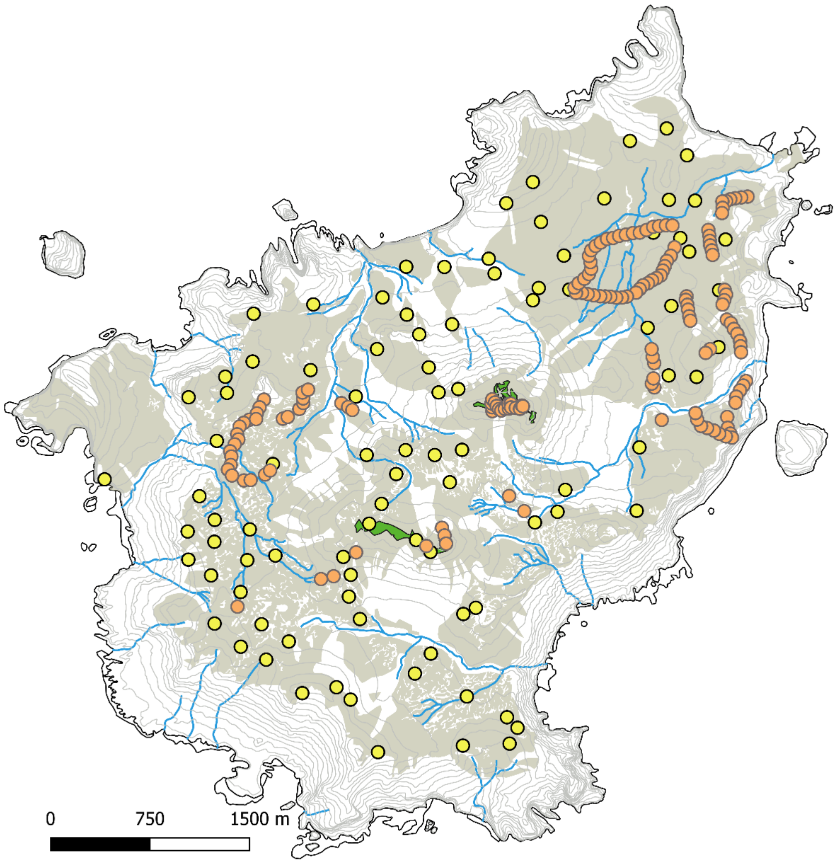


Figure 2. Distance sampling transects to assess white-chinned petrel burrow density on Antipodes Island in 2021 (orange) and 2023 (yellow) breeding seasons. Habitat used by white-chinned petrels is shaded and the part classed as high-herbfield is shown in green (for details see Elliott & Walker 2022).

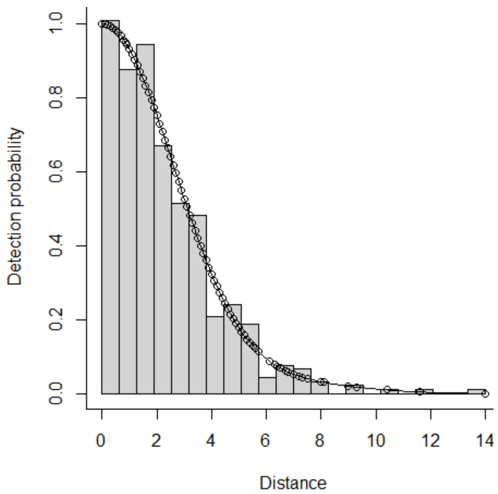


Figure 3. Detection function overlaid on white-chinned petrel burrow detections (n=493) on Antipodes

Table 3. Competing models for estimating white-chinned petrel WCP burrow density on Antipodes Island.

Key + Adj functions	n (Adj)	AIC	GOF	$\hat{p}$	CV ($\hat{p}$)	Strat	$\hat{D}$	95% CI ($\hat{D}$)
Half-normal + cos	2	1788	0.847	0.246	0.03	High herbfield	229.9	143.9–367.4
						Primary WCP	97.2	82.2–115.0
Uniform + cos	5	1791	0.854	0.245	0.04	High herbfield	230.6	143.8–369.6
						Primary WCP	97.5	81.6–116.5

Adj: adjustment terms to adjust detection function fit, AIC: Akaike's Information Criterion, $\hat{D}$: estimated density burrows/ha, 95% CI: lower and upper confidence intervals, Strat: post-stratification used, GOF: detection function goodness-of-fit p value, $\hat{p}$: burrow detection probability (probability that a burrow situated between the line transect and the truncation distance is detected), CV: coefficient of variation

Population size

Antipodes Island had an estimated 82,200 white-chinned petrel burrows (95% CI 69,200–97,800), using density estimated by stratum and stratum surface areas, adjusted by burrow corrections for each stratum to account for entrances that did not lead to burrow, and for burrows with small-petrel residents (Table 4).

The number of breeding white-chinned petrel pairs is based on burrow numbers and rates of burrow occupancy (0.3161 in primary white-chinned petrel habitat and 0.6304 in the small amount of high-herbfield habitat). An estimated 26,400 (95% CI: 22,200–31,600) white-chinned petrel pairs were breeding on Antipodes in Dec 2022 during early incubation (Table 4).

Table 4. White-chinned petrel burrows and estimated numbers of breeding pairs in Antipodes Island white-chinned petrel WCP habitat in December 2022

Vegetation class	Density $\hat{D}$	Area	Correction factors		$\hat{N}$ WCP burrows	Burrow occupancy	$\hat{N}$ pairs	95% CI ($\hat{N}$ pairs)	Active burrow density $\hat{D}_{active}$
	Burrows/ha	ha	*b*	*sm*		*c*			Burrows/ha
High-herbfield WCP	229.9	8.04	0.8525	0.8846	1,394	0.6304	879	550–1,404	109
Primary WCP	97.2	1,065.54	0.9103	0.8571	80,830	0.3161	25,550	21,603–30,217	24
Sum					**82,224**		**26,429**	**22,153–31,621**	**25**

Modelled burrow density is based just on surface inspection during distance sampling, so does not account for what was found when burrows were inspected. Therefore we also provide active burrow density (Table 4), which takes modelled densities and accounts for burrow status and contents inspection, so is a better reflection of the density of white-chinned petrels breeding on Antipodes. Active burrow density was as high as 109 burr/ha in the small amount of high-herbfield habitat, but in most white-chinned petrel habitat was 24 active burr/ha (Table 4).

Mark-recapture study

Across the two study areas, 90 study burrows were marked containing 169 banded white-chinned petrels (40 marked burrows at Galloway Toe and 50 at Polar Front) (Fig. 4). Only 16% of burrows contained breeding birds *and* were suitable for repeat access over time; in other words, 564 burrows were scoped at least once to reach 90 marked burrows.

Burrows were revisited until the mate was also banded, or the chick was left alone in the burrow. This approach meant 77 pairs have both birds banded, with a mate outstanding in only 13 burrows.

Two nesting attempts in the study areas failed during the five-week period of study burrow establishment and checks 23 Dec–31 Jan. This failure rate could in future be compared to that in control burrows (no banding, checked once at the start and again at the end, failure rate compared to that from banded pairs in more-visited burrows) as a control for disturbance. However, control burrows are not a priority until marked burrows contain enough banded birds for robust mark-recapture analyses, so have yet to be marked.

The Polar Front area being sited over a historical NIWA white-chinned petrel study quadrate meant some birds extracted from our marked burrows were already banded. Twenty-two birds were already banded; checking against original banding data (NIWA unpubl. data), these birds were banded between 2007 and 2011. All bands were in good condition, with no repairs required. One bird was still carrying a colour band of the type used to deploy GLS for the 2008–09 tracking study; this was removed.

Feathers were clipped from banded study birds for molecular sexing, including from the already-banded birds. Twenty birds also had blood and additional feathers taken for mercury analysis.

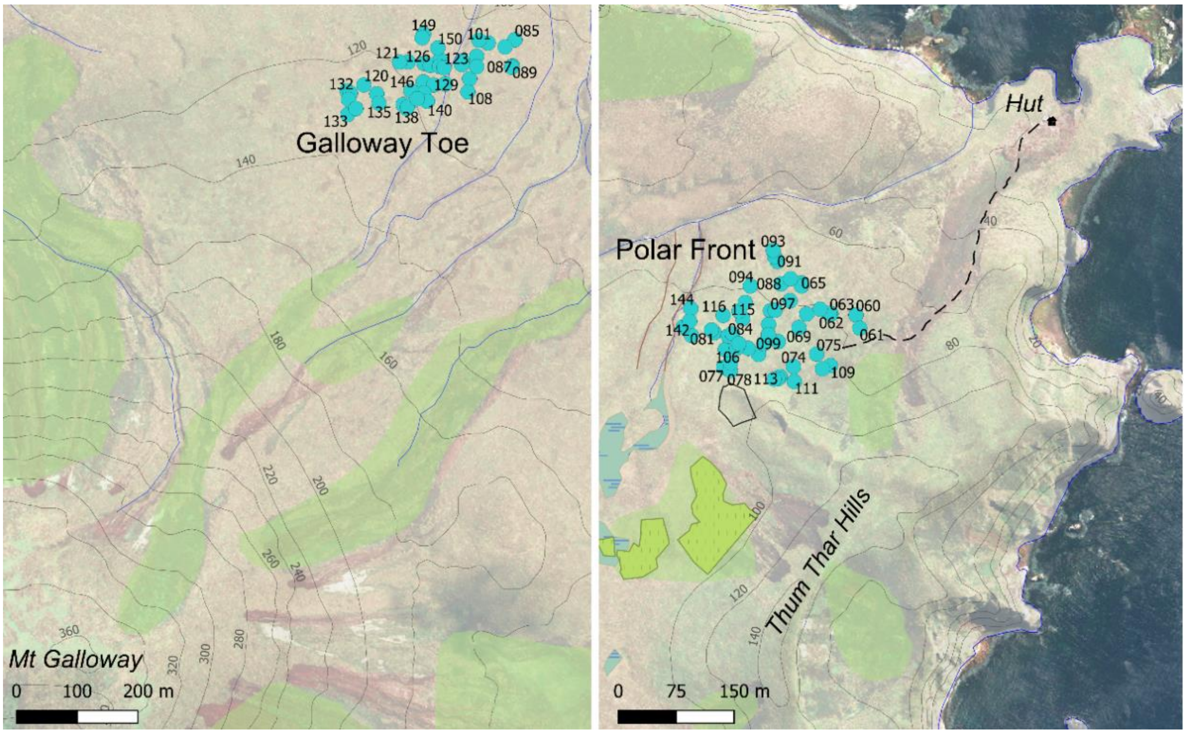


Figure 4. Marked burrows in two study colonies for white-chinned petrel demographic study.

Discussion

Population size

Antipodes Island supports an estimated 26,400 (95% CI: 22,200–31,600) breeding pairs of white-chinned petrel. This figure is smaller but more precise than previous estimates, which ranged from 39,700 (24,200–55,100) to 54,900 (38,400–71,500) pairs (Thompson 2019; Elliott & Walker 2022), and the estimated 26,400 pairs is as accurate as possible. To be clear, this should not be taken as evidence of a decline as it is merely a more accurate estimate.

A smaller, more accurate estimate here was attained by building on the strong points of previous studies. We took the careful burrow contents corrections and occupancy calculations from Thompson (2019) and the comprehensive habitat mapping and distance sampling approach from Elliott & Walker (2022), and built on these with expanded sampling coverage across the whole island (Table 4). Better distance sampling coverage improved the precision of burrow densities modelled for each habitat type. Sampling effort over three seasons included particular effort in high-herbfield habitat where burrow detectability is lower, to ensure the accuracy of modelled burrow densities. Habitat area refinements improved accuracy by allowing densities to be applied only to areas actually used by white-chinned petrels (Elliott & Walker 2022) (Table 4). Lastly, we put particular effort into burrow occupancy, since small differences in occupancy calculations can have large effects on a downstream breeding population estimate. Burrows were inspected for occupancy close to the ideal timing, minimising breeding attempts missed because of early failures, and occupancy calculations used burrow numbers corrected for entrances that are not in fact burrows, and for other species using white-chinned petrel burrows (Table 4). We have almost entirely eliminated the bias issue of "unscopable" burrows raised by Elliott and Walker (2022) by increasing the time spent and the experience levels of the burrowscope operator when assessing occupancy, and finally, habitat-specific occupancy was applied. We believe habitat-specific occupancy is important because if a single whole-island occupancy rate is used, the higher occupancy in high herbfield (0.63) than elsewhere (0.32) would skew upwards a whole-island occupancy rate (to as much as 0.47 here). This would give a substantially larger but less accurate whole-island estimate than when using habitat-specific occupancy as we did here, where the high herbfield occupancy rate is applied only to that (relatively small) habitat type. We note that the number of burrow inspections in high herbfield was substantially lower than in primary

Table 4. Summary of differences between surveys for population estimates in 2009–11, 2021–22 and current

		2009-11	2021-22	2022-23
Habitat area	Comprehensive area estimation so nesting area excludes areas WCP known not to use	roughly	yes	yes
	Comprehensive area estimation: surface area slope corrected	no	yes	yes
Density sampling	Density accounting for burrow detectability (distance sampling)	no	yes	yes
	Spatial coverage of island for density sampling	good	good	best
	Temporal coverage of island for density sampling	single-yr	2-yr	3-yr
	Randomisation for density sampling	mix	systematic	a-priori full
Occupancy sampling	Timing cf end of main lay period (~14 Dec) for occupancy sampling	4-8 wk after	1-3 wk and 4-5 wk after	1-2 weeks after
	Temporal coverage of occupancy estimates	yearly (3 yrs)	yearly (2 yrs)	single year
	Occupancy: visual confirmation bird on egg (burrowscope)	yes	partly (also acoustic)	yes
Occupancy calculations	Burrow numbers corrected for entrances that are not in fact burrows, and for other species using white-chinned petrel burrows	yes	no	yes
	Unscopable burrows bias addressed (contents correction using playback vs scope) or not (unscopable burrows discarded)	no	yes	no
Habitat-specific occupancy	Population size estimate uses habitat-specific occupancy	no	no	yes

Data from Thompson 2019; Elliott & Walker 2022; NIWA unpubl. data 2007–11; this study white-chinned petrel habitat (61 and 232, respectively), but we don't think increasing the numbers inspected in high herbfield would much change the resulting whole island estimate, given that high herbfield is <1% of the area white-chinned petrels use.

The white-chinned petrel breeding population on Antipodes is much smaller than the estimated 153,000 (119,700–195,700) breeding pairs at Disappointment Island in the Auckland Islands (Rexer-Huber *et al.* 2017). Indeed, the Antipodes population of 26,400 (22,200–31,600) pairs is closer to the much less precise estimate of 28,300 (10,400–44,800) breeding pairs for Adams Isl, in the Auckland islands; and to the ~22,000 pairs thought to breed on Campbell Island (Rexer-Huber 2017; Rexer-Huber *et al.* 2020).

The density of active burrows—i.e., only burrows with breeding activity—on Antipodes, at 25 active burrows/ha, was similar or only moderately lower than found in other island-wide estimates for white-chinned petrels: up to 26 active burrows/ha on Îles Kerguelen, and 63 active burrows/ha on South Georgia (Barbraud *et al.* 2009; Martin *et al.* 2009). In contrast, densities of populations in the Auckland Islands are an order of magnitude greater (381 active burrows/ha Adams Island and 394 active burrows/ha Disappointment Island) (Rexer-Huber *et al.* 2020). Low densities of active burrows are driven by low occupancy rates, which are unusually low on Antipodes among white-chinned petrel breeding islands. Burrow occupancy of 38% across all sites, here, is in fact higher than the 19–28% recorded on Antipodes 2009–11 (Thompson 2019), but in both cases Antipodes rates remain very low compared to occupancy of white-chinned petrels of 59–73% in the Auckland Islands (Rexer-Huber *et al.* 2017; Rexer-Huber *et al.* 2020) and 60–70% on Kerguelen and Crozet (Barbraud *et al.* 2008; Barbraud *et al.* 2009). Two factors likely contribute to low-to-very-low occupancy rates on Antipodes. Firstly, Antipodes has little of the more-typical wet steep herbfield habitat (habitat that on Antipodes has higher, more-typical occupancy) than other breeding islands; indeed, high herbfield makes up <1% of white-chinned petrel habitat on Antipodes. Secondly, substrate may play a part in why a high rate of habitable-looking burrows stand empty. The dry and firm substrate found in most white-chinned petrel habitat on Antipodes means a burrow may stay intact for years after being abandoned by a pair. In contrast, white-chinned petrel burrows elsewhere are typically wetter, thus collapsing sooner without maintenance by a resident pair. In support of this, the areas on Antipodes where burrows are more-typically wet and collapsible are high herbfields, where many unused burrows are obviously collapsed and burrow occupancy rates are closer to the rates seen elsewhere.

Mark recapture study

Important population changes are more likely to be detected via intensive study of birds in a representative study population than by comparison of whole-island population size estimates, with even quite large reductions in seabird population size potentially taking decades to detect in repeat counts (Bakker *et al.* 2018; Elliott & Walker 2022). Therefore, we established a mark-recapture study focused on estimating vital rates, survival in particular. Two study areas comprising 169 banded birds in 90 marked burrows are a good first step toward a marked population large enough for accurate survival estimates precise enough to detect changes.

Important next steps are to build the number of marked burrows to reach at least 400 banded birds, alongside recaptures at existing marked burrows to start building a capture history dataset. New burrows should preferably be marked within and around the perimeter of current study areas, for efficiency of monitoring, rather than establishing an additional study area involving more travel time. Secondarily, control burrows to assess investigator disturbance should be marked (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.

We anticipate that marked burrows will need to be visited for at least three years before band resighting data start giving useful estimates of survival, given the nature of mark-recapture studies for long-lived seabirds like white-chinned petrels. To illustrate, the 366 birds banded and resighted 2007–11 yielded annual survival estimates of adequate precision after four years of banding and resightings, ranging from 0.79 (95% CI 0.69–0.87) to 0.91 (0.76–0.97) (Thompson 2019).

Recommendations

For an efficient and effective long-term monitoring strategy, we recommend annual intensive monitoring effort in a representative study population. Ongoing mark-recapture of banded birds in a closely-monitored study area will have the resolution to detect population changes that can be difficult to detect even in repeat whole-island population size estimates.

Annual study population monitoring should still be supplemented by periodic whole-island population size estimates, population size being such a widely used metric in conservation management. It must be emphasised that whole-island population estimates are not a precise enough tool for monitoring trends over time, and are instead best seen as point estimates, or snapshots. However, whole-island estimates every five to ten years are attainable alongside annual monitoring, given the relatively sensible effort involved for whole-island coverage of distance sampling on Antipodes. Occupancy estimation is time-consuming, given the very low occupancy rates, but robust occupancy estimates (repeating methods as closely as possible for comparability) are vital given the outsized influence this parameter can have on estimates.

Efficient long-term monitoring requires a study large enough to provide accurate, precise estimates of vital rates, but small enough for thorough, intensive monitoring effort to be feasible, and requires that the study be continued for long enough to yield robust survival estimates. Continued effort is needed in the two white-chinned petrel mark-recapture areas established here to build the population of marked birds to be large enough for survival estimates to be useful, with ongoing annual effort to collect the crucial resightings data allowing survival to be estimated over time.

Acknowledgements

Collection of data in 2023 was supported by funding from the Department of Conservation's Conservation Services Programme (project POP2022-10). This is partially funded through a levy on the quota holders of relevant commercial fish stocks, so we thank the fishing industry for their contribution to the funding. We were supported by Johannes Fischer (DOC Wellington) who managed the contract, and by Ros Cole and the DOC Murihiku team who provided logistic support. We are grateful to Paul Sagar for access to the data from the 2007–11 NIWA study, and to David Thompson for workshopping best-suited burrow occupancy corrections for Antipodes. Thanks also to Steve Kafka (skipper) and the crew of Evohe for getting us safely to and from Antipodes Island.

References

ACAP. 2009. ACAP Species assessment: White-chinned petrel *Procellaria aequinoctialis*. Agreement on the Conservation of Albatrosses and Petrels.

Anderson, O.R.J.; Small, C.J.; Croxall, J.P.; Dunn, E.K.; Sullivan, B.J.; Yates, O.; Black, A. 2011. Global seabird bycatch in longline fisheries. *Endangered Species Research 14*: 91–106.

Bakker, V.J.; Finkelstein, M.E.; Doak, D.F.; VanderWerf, E.A.; Young, L.C.; Arata, J.A.; Sievert, P.R.; Vanderlip, C. 2018. The albatross of assessing and managing risk for long-lived pelagic seabirds. *Biological Conservation 217*: 83–95.

Barbraud, C.; Delord, K.; Marteau, C.; Weimerskirch, H. 2009. Estimates of population size of white-chinned petrels and grey petrels at Kerguelen Islands and sensitivity to fisheries. *Animal Conservation 12*: 258–265.

Barbraud, C.; Marteau, C.; Ridoux, V.; Delord, K.; Weimerskirch, H. 2008. Demographic response of a population of white-chinned petrels *Procellaria aequinoctialis* to climate and longline fishery bycatch. *Journal of Applied Ecology 45*: 1460–1467.

BirdLife International. 2023. Species factsheet: *Procellaria aequinoctialis*. Downloaded from http://www.birdlife.org on 19/04/2023.

Buckland, S.T.; Anderson, D.R.; Burnham, K.P.; Laake, J.L.; Borchers, D.L.; Thomas, L. 2001. Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford, Oxford University Press, 432 pp.

Burnham, K.P.; Anderson, D.R. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. Second. New York, Springer.

Elliott, G.; Walker, K. 2022. *Estimating the number of white-chinned petrels breeding on Antipodes Island. Report to Department of Conservation*. Nelson, Albatross Research.

Lavers, J.L. 2015. Population status and threats to Flesh-footed Shearwaters (*Puffinus carneipes*) in South and Western Australia. *ICES Journal of Marine Science 72*: 316–327.

Lawton, K.; Robertson, G.; Kirkwood, R.; Valencia, J.; Schlatter, R.; Smith, D. 2006. An estimate of population sizes of burrowing seabirds at the Diego Ramirez archipelago, Chile, using distance sampling and burrow-scoping. *Polar Biology 29*: 229–238.

Martin, A.R.; Poncet, S.; Barbraud, C.; Foster, E.; Fretwell, P.; Rothery, P. 2009. The white-chinned petrel (*Procellaria aequinoctialis*) on South Georgia: population size, distribution and global significance. *Polar Biology 32*: 655–661.

Miller, D.L.; Rexstad, E.; Thomas, L.; Marshall, L.; Laake, J.L. 2019. Distance Sampling in R. *Journal of Statistical Software 89*: 1–28, 10.18637/jss.v089.i01.

Parker, G.C.; Rexer-Huber, K. 2015. *Literature review of methods for estimating population size of burrowing petrels based on extrapolations from surveys. Report prepared by Parker Conservation*. Wellington, Department of Conservation.

Perón, C.; Authier, M.; Barbraud, C.; Delord, K.; Besson, D.; Weimerskirch, H. 2010. Interdecadal changes in at-sea distribution and abundance of subantarctic seabirds along a latitudinal gradient in the Southern Indian Ocean. *Global Change Biology 16*: 1895–1909.

R Core Team. 2023. *R: A language and environment for statistical computing*. Vienna, R Foundation for Statistical Computing Available at: https://www.R-project.org/.

Rexer-Huber, K. 2017. White-chinned petrel distribution, abundance and connectivity have circumpolar conservation implications. PhD, Dunedin, University of Otago. Available at: http://hdl.handle.net/10523/7778.

Rexer-Huber, K.; Parker, G.C.; Sagar, P.M.; Thompson, D.R. 2017. White-chinned petrel population estimate, Disappointment Island (Auckland Islands). *Polar Biology 40*: 1053–1061.

Rexer-Huber, K.; Thompson, D.R.; Parker, G.C. 2020. White-chinned petrel burrow density, occupancy and population size at the Auckland Islands. *Notornis 67*: 387–401.

Richard, Y.; Abraham, E.R.; Berkenbusch, K. 2020. *Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006-07 to 2016-17. New Zealand Aquatic Environment and Biodiversity Report 237. https://www.mpi.govt.nz/dmsdocument/39407/direct*. Wellington, Ministry for Primary Industries.

Rollinson, D.P.; Wanless, R.M.; Ryan, P.G. 2017. Patterns and trends in seabird bycatch in the pelagic longline fishery off South Africa. *African Journal of Marine Science 39*: 9–25.

Sommer, E.; Boyle, D.; Baer, J.; Fraser, M.J.; Palmer, D.; Sagar, P.M. 2010. *Antipodes Island white-chinned petrel and grey petrel field work report, 2009-10. Unpublished final report to Ministry of Fisheries*. Wellington, National Institute of Water & Atmospheric Research.

Sommer, E.; Boyle, D.; Fraser, M.; Sagar, P.M. 2011. *Antipodes Island white-chinned petrel field work report, 2011. Unpublished Final Research Report to Ministry of Fisheries*. Wellington, National Institute of Water & Atmospheric Research.

Thompson, D.R. 2019. *Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability: synthesis of population and demographic work. New Zealand Aquatic Environment and Biodiversity Report No. 22*. Wellington, Fisheries New Zealand.

Walker, N.; Smith, N.; Sharp, B.; Cryer, M. 2015. *A qualitative review of New Zealand's 2013 level two risk assessment for seabirds. New Zealand Fisheries Science Review 2015/1*. Wellington, Ministry for Primary Industries.