Albatross Research

Estimating the number of white-chinned petrels breeding on Antipodes Island

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ABSTRACT

During the summers of 2020-21 and 2021-22 the area of land occupied by white-chinned petrels on Antipodes Island was assessed along with burrow density and burrow occupancy which were combined to produce an estimate of the total population size on the island. This was compared with similar estimates made in 2008-2011, and the potential impact on white-chinned population size of landslides in 2014 and eradication of mice from Antipodes I in 2016 assessed. The most recent estimate of the size of the white-chinned petrel population on Antipodes Island is larger than that made in 2008-2011, but the confidence intervals about both estimates are so large that it is not reasonable to conclude there has been any population change. The 2008-2011 and 2021-22 estimates in combination suggest the population comprises ~46,000 breeding pairs.

The landslides in 2014 destroyed 5.6% of the white-chinned petrel burrows and as birds were incubating at the time of the landslides, up to 2.6% of the breeding population was killed. Subsequently the land on which the landslides occurred has been unsuitable for white-chinned petrel burrows and the birds that used these places have either died, moved, or stopped breeding.

Although mice are known to prey on white-chinned petrels, any improvement in nesting success because of the mouse eradication has not had sufficient time to be reflected in the size of the breeding population.

The use of distance sampling for assessing burrow density, as well as the explicit assessment of the effectiveness of burrow occupancy measurement techniques are useful improvements in white-chinned petrel population size assessment techniques. With greater field effort and increased sample sizes these tools could provide more precise estimates of population size, though even with these improvements, estimates of population size are not precise enough to reliably detect population trends. Detection of population change is likely to be more easily achieved with an intensive mark-recapture study of birds in a representative study population.

INTRODUCTION

White-chinned petrels (*Procellaria aequinoctialis*) nest on 8 island groups between latitudes 46°S and 55°S (ACAP 2009) in the Southern Ocean. Their total population was estimated at more than 1 million breeding pairs in 2009 (ACAP 2009), but they are frequently killed during fisheries interactions and at those few sites with trend information, populations appear to be declining (ACAP 2009). They are regarded as “Vulnerable” by IUCN (Birdlife International 2018).

Until recently the size of the New Zealand populations of white-chinned petrels was unknown, but estimates of the Auckland Island (184,000 pairs) and Campbell Island (22,000 pairs) populations were made in 2015 (Rexer-Huber 2017) and Antipodes Island (43,000 pairs) between 2008–2011 (Thompson 2019).

Since the 2008–2011 population assessments of white-chinned petrels on Antipodes Island there have been two significant changes to the environment: extensive peat avalanches across the island and the eradication of mice (*Mus musculus*). On 6 January 2014 protracted rainfall caused peat avalanches over about 13% of the land area of Antipodes Island during which many white-chinned petrels were killed in their burrows and a large area of land was made unsuitable for nesting. In the winter of 2016 mice were eradicated from the island using aerially broadcast poison (Horn *et al*. 2019). There is no direct evidence that mice have had an impact on white-chinned petrels on Antipodes Island, but elsewhere mice have been identified as significant predators of burrowing petrels (Dilley *et al*. 2015, Dilley *et al.* 2018, Wanless *et al*. 2007).

In this paper we present population estimates of white-chinned petrels on Antipodes Island made during the summers of 2020-21 and 2021-2022, assess whether there has been any detectable change since 2011 and explore the possible impacts of the 2014 landslides and the 2016 mouse eradication.

METHODS

To reliably estimate the size of the breeding population of white-chinned petrels requires:

1. An estimate of the area of land on the island in which petrels burrow, ideally stratified by vegetation and/or topography.
2. Estimates of the density of burrows in each of the identified strata.
3. An estimate of the proportion of the burrows that are occupied by breeding birds.

Area of land occupied by white-chinned petrels

To estimate the area of land occupied by white-chinned petrels, the island was divided into habitat classes based on vegetation and topography types recognisable on a high-quality satellite image of Antipodes Island. Previous researchers had reported that the nesting distribution of grey petrels on

Antipodes Island rarely co-occurred with that of white-chinned petrels (Bell *et al*. 2013) so such land was excluded. Sommer *et al.* (2008, 2009, 2010, 2011) classified the remaining land into vegetation and topography types which were associated with differing white-chinned petrel burrow densities, and these were used as a starting point for assessment of the land occupied by white-chinned petrel.

The areas of habitat classes identifiable on the satellite imagery were corrected for slope using a digital elevation model derived from the contours sourced from the LINZ Data Service and licensed for reuse under the CC BY 4.0 licence.

Burrow density

Burrow density was estimated along 20m distance sampling (Buckland *et al.* 1993) transects between 12/1/2021 and 1/2/2021. Transects were placed along predetermined "routes" chosen to sample representative white-chinned petrel habitat. Transect starting points along the routes were randomised by starting transects at predetermined intervals along the routes (either 20, 30 or 40m).

Transect lengths were measured accurately with a fibreglass tape (Figure1), and burrow distance from the transects was measured using 1m long poles and distances or parts of distances less than 1m were estimated to the nearest 10cm.

White-chinned petrel burrows were distinguished from those of white-headed petrels using the following features:

1. Larger size (Figure 1).
2. Square to oblong burrow entrance shape rather than round (Figure 1).
3. Presence of grey (white-chinned petrel) rather than white (white-headed petrel) feathers.
4. Wet, muddy burrow entrance (Figure 1).

The burrow occupancy analysis was undertaken before assessment of burrow density, providing familiarity with the differences between the burrows occupied by the two species before attempting to designate identity without seeing the bird occupying the burrow.

The dominant vegetation on each transect was classified into the same 10 classes used in previous estimates of white-chinned petrel abundance (Sommer et al. 2008).

Burrow occupancy

Burrow occupancy can be assessed reliably by digging into burrows, or more quickly and less destructively using a burrow scope, or by listening for responses to played-back calls. Burrow occupancy is estimated by dividing the number of burrows in which birds are found by the number of burrows inspected, but some burrows are too long, too crooked or too full of water and/or mud to be usefully assessed with a burrow scope and some birds don't respond to recorded calls.

Figure 1. Differing environments on Antipodes I with white chinned petrel burrows: (**top**) wet burrows amidst megaherbs near the summit of Mt Galloway (**middle**) large square/oblong burrows into mounds on the North Plains, with conspicuous dry "aprons" more typical of the WCP burrows on Antipodes I (**bottom left)** 20 m long distance sampling transect on the North Plains where WCP burrows occur wherever there is a small mound (**bottom right**) raised peat wetland on Central Plateau dominated by *Carex ternaria* and megaherbs. WCP burrows are absent in these flat "clears".

There is no obvious simple correction for burrow occupancy assessed using playback, but burrow occupancy assessed using a burrow scope has sometimes been "corrected" by dividing the number of occupied burrows by the number of "scopeable" burrows – that is all burrows inspected minus those which the burrow scope operator judged could not be assessed (most often because the burrow was too long) (Cuthbert *et al.* 2013). This practice assumes that burrows judged "un-scopeable" have the same occupancy as "scopeable" ones. When no bird is found in a burrow the burrow scope operator carefully explores the burrow to assess its "scopeability". If the operator can see the end of a burrow then they classify it "scopeable", but if they cannot see the end it classed as "un-scopeable". In contrast when a bird is found in a burrow the burrow-scope operator usually withdraws so as not to unnecessarily disturb incubating birds. The burrow is occupied and by default classified as "scopeable". In fact the burrow scope operator makes no assessment of whether they could confidently determine whether occupied burrows are empty and it is likely that many occupied burrows would have been judged "unscopeable" had they been empty. The removal of "un-scopeable" burrows from the sample of burrows results in overestimation of burrow occupancy.

To overcome this problem, a combination of both playback and burrowscope was used on 309 burrows in between 21 December 2020 and 6 January 2021 and this data was then used to produce unbiased estimates of the efficiency of these tools and burrow occupancy. The exercise was repeated on a smaller sample of burrows (44) between 10 and 21 January 2022 to assess interannual variation in burrow occupancy, and to assess the impact of assessing burrow occupancy later in the breeding season.

These three parameters were estimated by assuming that the likelihood of detecting a bird was related to burrow occupancy, and detection efficiency by:

When a bird is detected

$$likelihood = \beta_{bo} \times \left({\beta_{pb}}^{pb} \times \left(1 - \beta_{pb}\right)^{1-pb}\right)^{pbused} \times \left({\beta_{bs}}^{bs} \times (1 - \beta_{bs})^{1-bs}\right)^{bsused}$$

When no bird is detected

$$likelihood = 1 - \beta_{bo} + \beta_{bo} \times \left(1 - \beta_{pb}\right)^{pbused} \times (1 - \beta_{bs})^{bsused}$$

Where

β_{bo} = burrow occupancy

β_{pb} = the efficiency of playback for detecting occupied burrows

β_{bs} = the efficiency of burrow scoping for detecting occupied burrows

pb = 1 when a bird is detected with playback and

= 0 when no bird is detected

bs = 1 when a bird is detected with burrow scope and
= 0 when no bird is detected
pbused = 1 when playback is used to detect birds and
= 0 when playback is not used
bsused = 1 when a burrow scope is used to detect birds and
= 0 when a burrow scope is not used

The parameters β_{bo}, β_{pb} and β_{bs} were estimated by maximising the product of these likelihoods using the function "optim" in program R (R Core Team, 2021) and estimated standard errors from the hessian (Venables & Ripley 2002).

A more complicated model was also constructed with separate estimates of β_{bo}, β_{pb}, and β_{bs} for each of the two summers and a comparison of the two models made using AIC (Burnham & Anderson 2002).

Burrow occupancy was sampled along walking transects through country judged to be representative of the whole island (Figures 1 & 2) and in areas in which occupancy had previously been assessed in 2007–2011. Burrows were located using the same rules used for the distance sampling estimates of burrow density, i.e., only burrows which could be seen from the transects were included. The burrow occupancy checks between 21 December 2020 and 6 January 2021 were timed to occur soon after laying (Sommer *et al*. 2010) and between 10 and 21 January 2022 just before hatching.

To estimate the proportion of birds occupying burrows that were breeding (as opposed to loafing) all the birds seen by burrow scope were classified as loafers if they were moving around and not sitting on an egg or if there was more than one bird in the burrow. Birds seen sitting on eggs, or obvious nests were classed as breeders.

Population size and change

Population size was estimated from the formula

$$pop = burrow\ occupancy \times \sum_{i=1}^{n} burrow\ density_i \times area_i$$

Where the *i*s are one of *n* vegetation types.

The standard error of population size was estimated by bootstrapping.

Population change between estimates made in 2008, 2009, 2010, 2011 (Thompson 2019) and 2020 (this study) was assessed by undertaking a meta-analysis of the 4 population estimates using the metafor package (Viechtbauer 2010) in program R (R Core Team, 2021). This approach enabled estimation of population growth from the 4 population estimates and also incorporated the standard errors of the 4 estimates.

Impact of slips

To assess the likely impact of the 2014 landslides on the Antipodes Island white-chinned petrel population, the estimate of the current population was compared with what it might have been had the landslips not occurred. To do this it was assumed that all the slipped land would have supported white-chinned petrels at the same density they occurred elsewhere in that habitat type.

RESULTS

Area of land occupied by white-chinned petrels

The planar area of Antipodes Island is 2,025 ha, but when corrected for slope is about 2,212 ha. To determine what proportion of Antipodes Island's 2,212 ha was occupied by white chinned petrels (Figure 2) the following areas were excluded:

1. 716 ha of mostly steep deeply vegetated slopes which was found by Bell *et al.* (2013) not to be used by white-chinned petrels, instead being occupied by grey petrels (*Procellaria cinerea*).
2. 99 ha seaward of the "grey petrel" country which comprised cliffs, coastal rocks, and penguin colonies.
3. 125 ha of dense *Polystichum vestitum* fern which was found by both Thompson (2019) and this study, to not support white-chinned petrels. These stands were identified from satellite imagery and appear to mostly occur on old slips and in watercourses.
4. 112 ha of old slips on which there was little or no *Polystichum vestitum* and there was insufficient vegetation and soil for petrels to burrow. Most of these slips occurred before 1995 – the earliest satellite photographs available on Google Earth.
5. 295 ha affected by 2014 slips, identified on the ground (Walker & Elliott 2014) and on a high-resolution satellite image (DigitalGlobe 2009, derived slip layer DOC 2014).
6. 59 ha of raised peat bogs which are dominated by *Carex ternaria* (Figures 1 & 2). These *Carex ternaria* bogs have a smoother surface than most of the rest of the island's vegetation, making it

identifiable in a satellite photo. No white-chinned petrel burrows were found in this vegetation type in the 2021& 2022 burrow-density transects.

All remaining areas were considered white-chinned petrel habitat, including 8.04 ha of high altitude herbfields. These herbfields are dominated by the megaherbs *Pleurophyllum criniferum* and *Anisotome antipoda* and have a smooth appearance which makes them readily identifiable on satellite imagery. Although 10 vegetation classes were distinguished on the ground, only the *Carex ternaria* dominated communities and the high altitude herbfields could be reliably identified in satellite imagery.

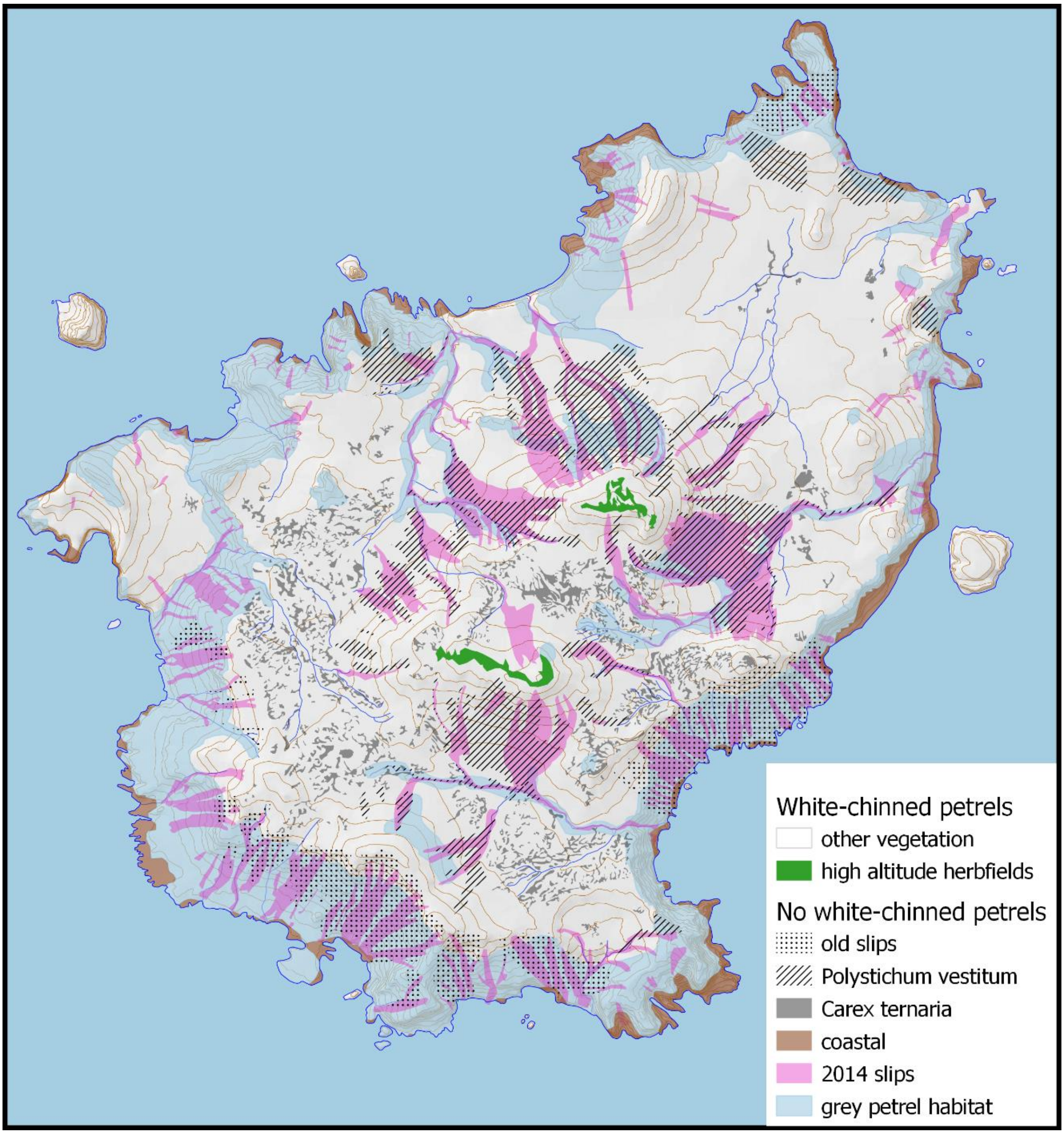

Figure 2. Areas with and without white-chinned petrel burrows on Antipodes Island.

Figure 2 shows the two identifiable vegetation strata in which white-chinned petrels burrows were found and the areas in which it was assumed no white-chinned petrels nested. When corrected for slope, white chinned petrel habitat covered 1073.5 ha including 8.04ha of high altitude herbfield. Most of the white-chinned petrel nesting country was relatively flat so the slope correction increased the estimated land area by only 1.2% for high altitude herbfield, and by 2.9% for the rest of the white-chinned petrel habitat.

Burrow density

Burrow density was assessed along 155 transects in mid-late Jan 2021 (Figure 3). The best and most plausible detection function identified by distance sampling was half normal with cosine (2) adjustments (Miller *et al*. 2019). Burrows were detected up to 14 m from the transects and the distance sampling model estimated that 26% of the burrows within this distance were detected.

Although burrow density transects were classified into 10 vegetation classes, because only 2 (high altitude herbfield and *Carex ternaria* dominated vegetation) were identifiable from satellite imagery, the classification was reduced to 3 classes (high altitude herbfield, *Carex ternaria* and all the rest). No white-chinned petrel burrows were found in *Carex ternaria* dominated vegetation, and burrow density in high altitude herbfield was higher than in the remaining white-chinned petrel habitat (Table 1).

Table 1. White-chinned petrel burrow density in two vegetation types on Antipodes Island.

Vegetation class	Density	SE density	Area (ha.)	Estimated number of burrows	SE burrows
High altitude herbfield	231	59	8.0	1,860	478
Other	86	11	1,065.5	91,762	11,324
Total	**87**	**11**	**1,073.5**	**93,623**	**11,382**

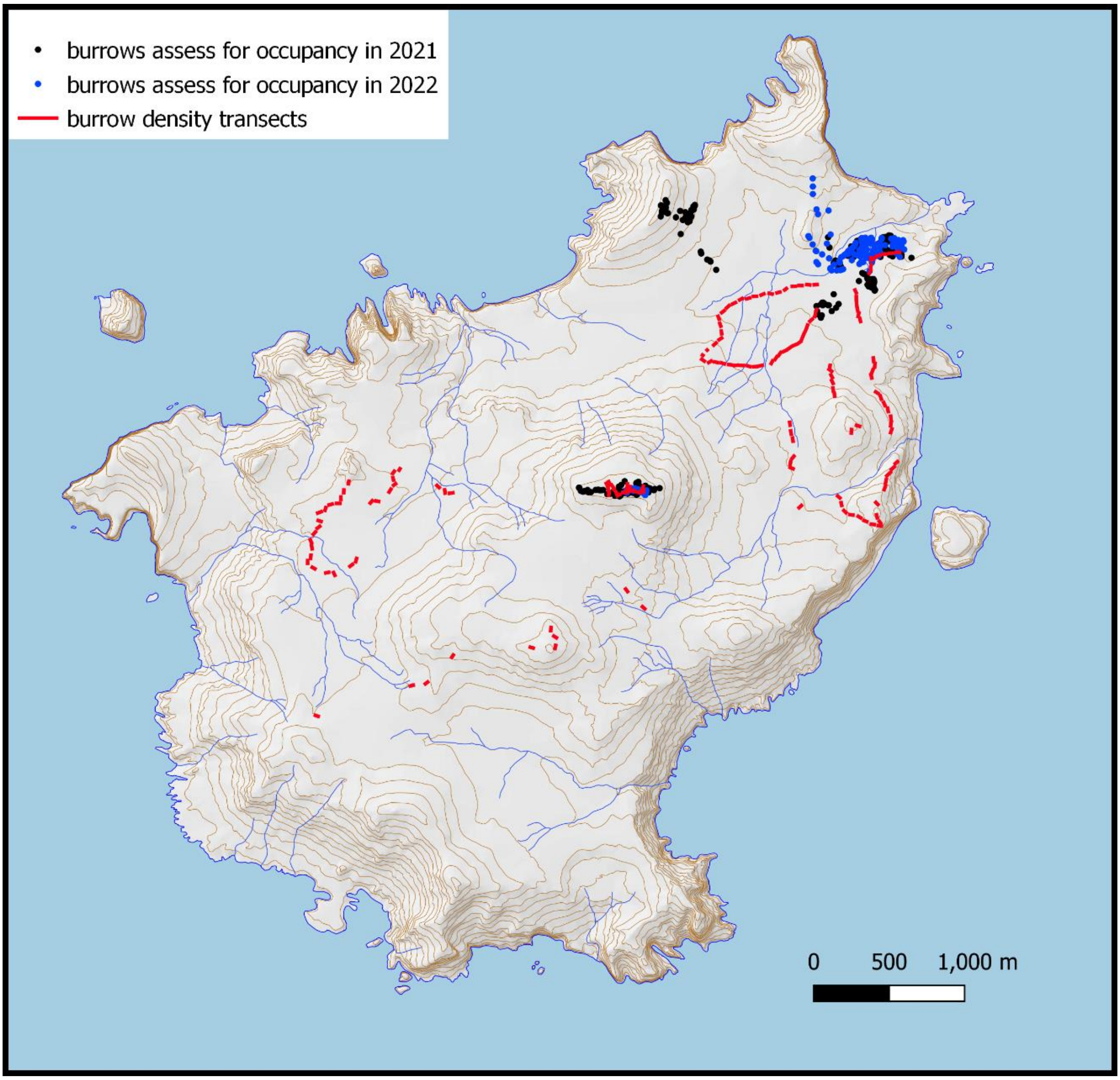

Figure 3. Burrows assessed for occupancy of white-chinned petrels in 2021 (black dots) and 2022 (blue dots), and distance sampling transects (red lines) used in 2021 to assess burrow density.

Burrow occupancy

509 burrows were examined for white-chinned petrel occupancy, 309 in 2020-21 and 200 in 2022. Figure 3 shows the locations of burrows at which occupancy was assessed and Table 2 details the burrow occupancy tools used and outcomes. Estimated burrow occupancy as well as the efficiency of playback and burrow scope at detecting occupied burrows are shown in Table 3.

Table 2. White-chinned petrel burrows examined to determine occupancy.

			Not used	Burrow scope No birds detected	Birds detected
2020-21	Playback	Not used	0	36	11
		No birds detected	75	39	15
		Birds detected	84	10	39
2022	Playback	Not used	0	0	0
		No birds detected	122	7	5
		Birds detected	55	4	7

Table 3. Estimated occupancy and the efficiency of playback and burrow scope at identifying occupied nests

Parameter	2020-2021 Estimate	SE	2022 Estimate	SE
Burrow occupancy	0.636	0.032	0.769	0.131
Playback detection efficiency	0.766	0.035	0.432	0.078
Burrow scope detection efficiency	0.718	0.046	0.642	0.103

Although burrow occupancy measured in 2022 (0.769) is higher than in 2020-21 (0.636), no real change can be concluded due to the large standard error of the 2022 estimate. Similarly, the apparent decrease in burrow scope detection efficiency between 2020-21 and 2022 does not support a conclusion of any real change because of the large standard errors. In contrast the dramatic decline in playback detection efficiency between 2020-21 and 2022 is likely to be real.

Amongst the 65 burrows in which birds were detected with a burrow scope in 2020-21, 60 (92.7%) of them were judged to be nesting rather than loafing (binomial standard error = 3.3%). Breeding bird occupancy in 2020-21 is thereby estimated to have been 58.7% (standard error = 3.7% using parametric bootstrap) after application of this nesting vs loafing proportion.

Population size

Combining the area of land occupied by white-chinned petrels, with burrow density and breeding occupancy leads to an estimate of 54,945 breeding pairs in 2020-21 (Table 4), and when this is compared with earlier counts, suggests there has been 27% increase between 2011 and 2021 (Table 4). However, the standard errors of the population size estimates are large and the population growth rate estimate (2% per annum) derived from the meta-analysis is not significantly different from no population growth (p=0.9830).

Table 4. Burrow density, burrow occupancy and estimated breeding population size of white-chinned petrels on Antipodes Island for 4 summers.

	Burrow Density ha^{-1} (mean ± sd)	Burrow Occupancy % (mean ± sd)	Population Estimate	se	95% CI
2008–09	177.7 ± 144.4	19.2 ± 18.0	45135	6723	31957 - 58313
2009–10	115.3 ± 63.0	27.8 ± 15.8	44924	9979	25366 - 64482
2010–11	133.3 ± 76.0	24.7 ± 20.5	39670	7895	24195 - 55145
2020-21	87.0	58.7	54945	8458	38379 - 71535

Impact of slips

Although the area of slipped land (295 ha) comprised 13% of the island's surface area, it affected only 5.4% of the land white-chinned petrels burrow in and destroyed only 5.2% of the burrows. This was because much of the land that slipped in 2014 was used by grey petrels rather than white-chinned petrels, or was recovering from previous slips and still did not support white-chinned petrels, or it was covered in dense *Polystichum vestitum* which was not favoured habitat of white-chinned petrels. The slips would have destroyed all the eggs and small chicks in burrows at the time and thus reduced productivity that year by 5.2%. Many incubating adults were also killed, comprising up to 2.6% of the breeding population.

The impact of slips is likely to have been much greater for grey petrels as 21 % of their habitat was lost, though they were not in their burrows when the slips occurred.

DISCUSSION

Population size and trends

This study suggests that there was an average of 46,000 pairs of white-chinned petrels nesting on Antipodes Island between 2008 and 2020. Although the population estimates increased by 27% between 2008 and 2020, the confidence intervals about this estimated change are very large so it is not certain there was any real population change.

Perhaps the most significant conclusion is that population estimates such as those made in this study and by Thompson (2019) have little power to detect population trend.

Furthermore, there were five differences in methodology which make comparison of the 2008-2011 population estimates and the 2020 population estimate difficult.

1. Burrow density in 2008-2011 was estimated by rigorously searching a series of 100m^2 plots, whereas in 2020-21 distance sampling protocols were used which only counts burrows which can be seen from transect lines. Occupied and recently active burrows are easy to find as they have an area of recently excavated soil or mud at the burrow entrance or an area of clear ground caused by birds coming and going, whereas at older unused burrows any clear ground, as well as the burrow entrance, is usually overgrown. Distance sampling searches detect only obvious recently active burrows, whereas 100m^2 plot searches will also detect old unused burrows. 100m^2 plots will produce higher estimates of burrow density than distance sampling transects.

2. Burrow occupancy was assessed in 2008-2011 by inspecting burrows found in 100m^2 plots whereas in 2020/21 and 2022 only burrows found along distance sampling transects were examined. Since the burrows found in 100m^2 plots include many old unoccupied burrows, burrow occupancy measured in 100m^2 plots will be lower than assessed on distance sampling transects.

3. Burrow occupancy was “corrected” by combining playback and burrow-scope methods to estimate the proportion of occupied burrows where birds present were missed, whereas in 2008-11 burrow occupancy was “corrected” by excluding burrows which the burrowscope operator judged “un-scopeable”. Excluding “un-scopeable” burrows leads to over-estimation of burrow occupancy.

4. Thompson (2019) estimated population size by multiplying the burrow density and occupancy by the area occupied by white-chinned petrels, and in doing so assumed that white-chinned petrels do not nest in dense *Polystichum* nor where grey petrels nest. While both Thompson and this study used the same area of grey petrel nesting habitat (from Bell *et al.* 2013) for this calculation, the area of dense *Polystichum* fern used in the two studies differed. Thompson *et al*. (2019) calculated the extent of fern from Land Information New Zealand’s 1:50,000 digital map of Antipodes which is turn derived from a map published in 1972, while this study digitised the extent of fern from 2015 satellite imagery.

5. Areas dominated by *Carex ternaria* were excluded from the 2021 estimate of the area of land occupied by white-chinned petrels as no burrows were found in this vegetation type in 2021, whereas they were included in estimates by Thompson (2019).

The methods of assessing population size used in 2021 were easier and quicker than those used in 2008-2011. Distance sampling transects in particular are much quicker to do than measuring burrow density in 100 m^2 plots. Likewise, assessing burrow density using playback is much quicker than using a burrowscope or digging into a burrow. Much less time was spent collecting population density data in the two recent summers than was spent in the earlier study, yet the standard error of the 2021/22 population estimate is similar to those of Thompson (2019). Future attempts to assess population size could make even more precise estimates of population size if distance sampling and playback continued to be used but with greater effort.

The efficiency of detecting occupied burrows using playback appeared to decline between 2020-21 and 2022 and this is likely to be because trip timing meant burrow occupancy surveys had to occur later in the season in 2022 than in 2020-21. Berrow (2000) found variability in detection rates using playback at white-chinned petrel burrows on Bird Island and decline in detectability as incubation progressed. Such variability of playback detection efficiency might suggest that playback is not a reliable way of detecting occupied burrows. However, this variability might be overcome by always undertaking burrow occupancy assessment at the same time of year, or by sampling a subset of burrows using both playback and burrowscope and thus being able to estimate playback detection efficiency. This study suggests that despite its temporal variability, playback may still be the best way of assessing occupancy. Playback is much quicker than using a burrowscope ensuring a larger sample of burrows can be inspected.

The landslides of 2014 reduced the area of useable land for white-chinned petrels by 5.4%, and approximately 2.6% of the breeding birds were killed. This level of mortality is likely to be within the variation in annual mortality that occurs in white-chinned petrels.

It is unlikely there has yet been any increase in white-chinned petrels attributable to the eradication of mice from Antipodes Island. There is good evidence that mice prey on burrow nesting petrels and albatrosses on Gough Island (Dilley *et al.* 2015), and on burrow nesting petrels, including white-chinned petrels, on Marion Island (Dilley *et al.* 2018). It is possible there has been an improvement in white-chinned petrel nesting success (not measured in either the earlier study or this one) following the eradication of mice from Antipodes Island but white-chinned petrels typically first breed when 6 years old (Bell 2013) so there has not yet been enough time for the breeding population to grow sufficiently for this to be detectable.

This study showed that about 46,000 pairs of white-chinned petrels nested on Antipodes Island in recent years but revealed no evidence of recent population change and showed that the tools used to detect trend lacked power. Future attempts to estimate population size on Antipodes Island would do well to use distance sampling transects, and playback, but larger sampling effort would be needed to increase the precision of resulting estimates. Future attempts to estimate population trends in white-chinned petrels on Antipodes Island would be better based on a mark-recapture study in an intensively monitored study area rather than whole island estimates of burrow density and occupancy.

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