Changes in population density and response to storm damage for Westland petrels 2007 – 2019

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Abstract

The density and distribution of petrel burrows was assessed over an 12-year period, forming the baseline population estimate for the endangered Westland petrel at around 6,200 breeding pairs in 2019 and a world population of between 13800 - 17600 individuals for the areas surveyed. Density transects were conducted in 2007 - 2011 to establish the location and density of 17 colonies of petrels nesting in rugged untracked terrain in Paparoa National Park, West Coast, New Zealand. During 2014 – 2018, major storms caused widespread treefall and landslides, destroying breeding habitat throughout the species' range. The surveys were repeated at several accessible colonies in 2012 – 2017, and a comprehensive survey in 2019 covering 95% of the estimated population. Burrow density has increased over the 12-year timeframe, and this is commensurate with a 1.022% annual increase (0.971 – 1.076: 95% Credibility Interval) in the population from Mark Recapture data from the demography study area. Demographic influences of the major, and ongoing habitat loss may be continuing, with depressed occupancy rates in at least 3 years following the 2014 storm events, as birds re-establish nests and partnerships following loss of their habitat in landslips and treefall. The study presents a picture of the complex effects of climate-related disruption to long-lived species biology. With an expected increase in the frequency of severe weather events including ex-tropical cyclones and abnormal fluctuations in sea surface temperatures associated with climatic change, in the West Coast region, and a single nesting area vulnerable to landslip and treefall, it is likely that climate change will have an ongoing and negative influence of Westland petrel breeding habitat. Current indications from the demographic study suggest that the species has some flexibility to adapt, albeit with potentially lower breeding outputs immediately following habitat disturbance.

Introduction

For Westland petrels, (*Procellaria westlandica*), an endangered, single-site endemic species (Birdlife International 2018), there is a need to establish a robust population estimate, using repeatable methodologies (Dilley et al. 2019). These are necessary to enable monitoring of population changes in response to management actions, and to detect threats, such as habitat destruction or depredations by introduced invasive species (Dilley et al. 2017, Waugh & Wilson 2017). A key feature of survey robustness is the estimation of error, and documentation of areas surveyed, and a methodology that enables repeated measures through time (Thompson et al. 1998). This species nests in rugged, untracked terrain characterized by karst features and sheer bluffs, therefore despite its relative accessibility (within 60 km of an urban centre), it remains poorly known. The species has several identified management issues – the level and influence of fishery bycatch and threats in its terrestrial habitat (Waugh & Wilson 2017).

For long-lived species such as petrels, adverse effects of climate change on population growth can be difficult to track. These can include catastrophic influences of ice and weather on breeding seabirds (e.g. Barbraud & Weimerskirch 2001), or can be more subtle, such as through disruption to breeding cycles and displacement of optimal breeding and feeding habitat (e.g. Durant et al. 2005). The aim of the study was to estimate changes in population density and burrow occupancy for the endangered Westland Petrel *Procellaria westlandica*. First, the study assessed the trend in burrow density at 17 colonies followed between 2007 – 2019 but excluded four known colonies (Wood & Otley 2013) which were deemed difficult and unsafe to access due to storm damage to the colony and surrounding area in 2019. These colonies comprise of less than 5% of the estimated total population in 2013. Second, a minimum breeding population estimate for the Westland Petrel was derived from the 2019 surveys. Third, the population trend was assessed based on Capture – Mark – Recapture analyses at the largest colony where demographic studies have been undertaken for over 50 years (Waugh et al. 2015a). These provided an independent assessment of the trends observed for the population based on burrow density and occupancy. Finally, a major storm event, caused by ex-tropical cyclone *Ita* and subsequent storms destroyed breeding habitat in most of the Westland petrel colonies in 2014. This enabled a unique opportunity to study the change in the petrel's colonies and breeding parameters in response to these perturbations (Waugh et al. 2015b).

Methods

The Westland petrel nests in steep, forested terrain in Paparoa National Park, near Punakaiki (43.144ºS, 171.343ºE) (Figure 1). Each colony survey consisted of mapping a series of strip transects, of 20 – 200 m length and 2 m width. Colonies were defined by the survey teams, and initially located based on knowledge from previous surveys, these comprised clusters of nests which were contiguous on the landscape. Typically, these areas featured 'gardened' soil with little undergrowth, as a result of the petrel's activity. Areas around the periphery of colonies were searched, and where burrows ceased to be detected with 20 – 50 m of the last burrows, these edges were mapped. Transects were randomly placed along the slopes of each colony and separated by at least 20 m. Teams of 1 – 4 people conducted the surveys, and using hand-held Garmin GPS, accurate to ± 5 m, marked the start and end of each transect. Area estimates were taken from QGIS estimates of area for each colony perimeter polygon (QGIS 2019). Area of colonies and damaged zones were not corrected for slope. With thorough searching under vegetation, the numbers of burrows found along each transect were noted. The

surveys were conducted primarily in the summer (non-breeding: January – April) period in 2007 – 2011. In 2014 – 2019, the surveys were undertaken whilst birds were incubating (May – July), or early during the chick rearing period (August). As burrows endure between years, the estimates of burrow density taken during the summer were considered representative of the breeding period density for the year following. The number of transects was 212 in 2007 – 2011, 211 from 2014 – 2018, and 367 in 2019.

The occupancy rate was assessed using burrow-scopes at 3 colonies in 2008 – 2011, and at 12 of the 17 colonies in 2019 with sample sizes of more than 25 burrows at all but 3 of the 31 samples taken (Supplementary Material Table 2). The occupancy rate was the number of burrows occupied by one or more adult bird. The data were normally distributed (Shapiro-Wilk Test = 0.98, n.s.). Between year and colony differences in occupancy rate were tested using ANOVA, with 12 colonies and 8 years of data. In 2019, burrows in three establishing areas were surveyed to assess occupancy.

A ballpark estimate of breeding pairs was calculated for each colony from the undamaged surface-area, the average density of burrows by colony, the burrow occupancy at the colony assessed during the incubation period in 2019, with colony-by-colony data shown in Table 3. The ratio of breeding and non-breeding birds was not assessed at each colony. This was corrected globally for the proportion of breeders to non-breeders (0.480 ± 0.073) found in study nests at the study colony estimated during 2015 – 2019 during 4 years in 62 – 86 burrows per year. The estimated number of breeding pairs was calculated as $A \times (1-I) \times D \times O$, where A is Area of Occupancy, I is the percentage of area impacted by storm damage, D is mean burrow density and O is colony specific occupancy rate (Table 4). Standard errors for the number of occupied burrows and number of breeding pairs were calculated using the delta method (Seber 1982).

Following storm damage during 2014 - 2018, visual assessments and aerial photography was used to identify colonies that had been impacted by landslips or treefall. Most of these areas were visited in 2019, and where possible, the perimeter of each landslip was assessed by taking GPS points at some of the margins. Not all these areas were fully mapped, due to the danger and difficulty of working in unstable terrain. Slip and windfall area estimates are therefore considered to be qualitative.

Statistical methods

Burrow density analyses

Burrow density was modelled using generalized additive mixed models (GAMM) in the 'mgcv' library in R version 3.6.1 (Wood et al. 2017). This allowed for the possibility of nonlinear responses to covariates. GAMM were specified with a Gaussian family, an identity link function, and restricted maximum likelihood estimation (REML). The response variable was the burrow density and several models were fitted to test different hypotheses. First, burrow density was modelled as constant across years. Second, burrow density was modelled as a function of year with year specified as a categorical variable. The year 2015 was not included in the model since only two small colonies were monitored and observed densities were clearly outliers compared to other densities obtained in different years, probably due to the perturbations from the 2014 storm events. Third, we tested for the effect of colony

on burrow density, where colony was specified as a random factor. Fourth, we tested for spatial autocorrelation in burrow density by modelling spatial autocorrelation as an isotropic thin plate spline, set up as a two-dimensional smoother based on both x and y coordinates of the starting point of each transect (Cleasby et al. 2015). Finally, we tested for a temporal trend in burrow density by replacing the year effect by a continuous temporal covariate corresponding to the number of years monitored, and which was modelled using a smoother. Models were compared using the Akaike Information Criterion (AIC, Akaike 1973) and the model with the lowest AIC was selected. Model validation was assessed using residuals versus fitted values to verify homogeneity, QQ-plots and histograms of the residuals for normality, and residuals versus each explanatory variable to check independence (Zuur et al. 2009).

Estimating population growth rate using capture-mark-recapture data

We used individual capture-mark-recapture (CMR) data from 2010 to 2019 at study colony to estimate the population growth rate of marked individuals. Study colony is a large, densely burrowed, and well-studied colony of Westland petrels, and subject to the demographic research projects conducted since the 1970s. From 2010 – 2019, 40 – 104 individual birds yearly found in 36 - 60 marked burrows with access lids were banded with stainless steel leg bands and recaptured, and their breeding activity, band number, weight, partner identity were noted at study colony. These were used to estimate the proportion of all breeding to non-breeding Westland petrels. We used Pradel's models (Pradel 1996) with the survival and lambda formulation in program MARK (White & Burnham 1999). We tested several models where the adult survival probability (s) and the recapture probability were either constant or time dependent, but where the population growth rate (λ) was constant. Model selection was performed using AIC. Estimates were obtained from the Markov Chain Monte Carlo algorithm in MARK, with 20000 tuning samples, 20000 burn-in samples, and 50000 stored samples.

We used CMR data from burrows that were surveyed throughout the whole study period to avoid biases in population growth rate due to expansion or contraction of the study area following storm damage. In addition, since we used CMR data from 2010 to estimate λ and since the CMR study started earlier (Waugh et al. 2015a), we were confident that λ was not biased high due to individuals missed in the first years of the study.

Survival, recapture and population growth rate were estimated from a sample size of 234 individuals for the study colony. Over the study period, no band wear or loss was found, so we assumed that band loss was negligible.

Results

Burrow densities and occupancy rates

Densities varied between 0.005 - 0.203 burrows per m^2 with ¾ of colonies having densities between 0.015 – 0.075 burrows per m^2. Examining the raw data shows a complex picture of increases and decreases in comparable areas (Supplementary Appendix 3). The selected model (model 4, Table 1) indicated that burrow density varied as a function of year, colony and was spatially auto correlated. This model explained 52% of the deviance in burrow density. Both the random intercept for colony and the spatial smoother were significant (Table 2). Model validation indicated that there was no evidence for non-normality in residuals and that residuals were not related to fitted values. Year-specific burrow densities from model 4 indicated that densities were higher in 2016 and 2017 than in 2007 (Table 2, Figure 2). GAMM results indicated that there was a nearly quadratic temporal trend in burrow density with an increase in density from 2007 to 2016 and a slight decrease between 2016 and 2019 (Table 3, Figure 3).

There were no significant differences in burrow occupancy between years ($F_7 = 0.94$, n.s.), and colonies ($F_{10}=0.228$, n.s.), or between new and established areas ($F_1 = 0.023$, n.s.). Average occupancy was 0.385 (±0.116, n = 66) (Supplementary Table X).

Population growth rate using CMR data

For study colony, the selected model indicated that adult survival and recapture probabilities were time dependent. The estimated population growth rate was 1.022 (95% higher posterior density credibility intervals HPDCI: 0.971 – 1.076), and the probability that λ was higher than 1 was 0.80.

Number of breeding pairs and total population size

We estimated the number of breeding pairs based on burrow density estimates on a colony by colony basis (Table 4). At three colonies (middle bluff, studio and three bluffs), field teams had low confidence in the estimate produced by multiplying the area of the colony by the densities, as they felt there was around an order of magnitude difference higher or lower in the total number of burrows estimate compared to those they observed in the field. The number of burrows was estimated at ca. 31,306 in 2019, of which around 12,964 were occupied, giving an estimated 6,223 ± 380 breeding pairs in 2019 for the areas surveyed. Using the known age structure of the population to take into account the non-breeding proportion of the population (21%) from the matrix population model in Waugh et al. (2015a) and assuming a 1:1 sex ratio, this gives a total population of 15,711 (95% CI: 13,830 – 17,591) individuals in 2019.

Storm damage and its impacts on breeding Westland petrels

The transect surveys identified the perimeters of each colony and qualitatively assessed the amount of land lost to landslips and treefall in 2019 (Table 5). These assessments include the areas damaged in the extensive 2014 storms, and subsequent events such as Cyclones Fehi

and Gita in 2018 (Newshub 2020). A qualitative assessment of the areas lost to landslips showed that most of the 17 colonies surveyed in 2019 suffered some damage. Three had more than 10% of their area affected, 9 colonies had 1 – 10%, and 5 had no signs of storm damage.

Discussion

The Westland petrel population appears to have grown in the last 12 years, with independent measures of burrow density, area occupied, and CMR data indicating increases in the population. Our transect surveys showed an overall increase in burrow density in areas surveyed since 2007, with an average increase of 2.2% per annum modelled. Within the study period there was fluctuation in the modelled densities, which increased between 2007 and 2016 and slightly declined from 2016 to 2019. The estimated stochastic population growth rate from the CMR data between 2010 and 2019 (1.022) also suggested an increase in population size and was similar to the one obtained in Waugh et al. (2015a). These CMR data, were collected at study colony, where density and occupancy were measured multiple times in 12 years. It has an estimated 4,100 burrows, but our extensive transect sampling of other colonies indicate that these changes can be generalized across most colonies. The exceptions include two colonies which suffered extensive storm damage between 2014 – 2019, and which have decreased in area occupied and the estimated number of burrows (Dougies bluff and Bees nest), these colonies were previously estimated to hold 13% of the population. There were also decreases in density at three small colonies (less than 10,000m^2) – Three bluffs, Power barrow and Rob's colonies, which have small numbers of estimated burrows, and are unlikely to have affected the overall trends in numbers observed (Supplementary Table X).

In addition to population growth inferred from demographic datasets, the transect surveys identified new areas with burrows in 2019, some not found in 2007 – 2011, others with small estimated populations from the preliminary analysis of our dataset (Baker et al. 2011). Although surveys were thorough in 2007 – 2011, and the field teams comprised several of the same workers, it is possible that burrow density in these new areas may have been at the edge of detectability in the pre-2012 data collection period. Many of these areas were in-between neighboring colonies (e.g. between Middle and Study colonies, between Noisy Knob and Solomon's), or were extensions or densification of the known colonies s (e.g. Dougies bluff, Liddy's, Rowe and Studio colonies) into adjacent areas. One new area, Nuggety, was identified for the first time as part of our surveys since 2012 and in all probability is a newly established colony.

These changes may be due to natural population growth observed between 2007 – 2013 with individuals digging burrows in new areas, and as a result of displacement of numerous individuals by landslips and treefall from 2014 onwards. The growth in density at several medium to large colonies (see Supplementary material) shows that the petrels are adapting to these perturbations and establishing new breeding areas. Landslips are likely to have been

a feature of their breeding habitat across time, with reports of important areas of slips seen historically in petrel breeding areas (J.A. Bartle pers comm in Waugh & Wilson 2017).

We conclude a real increase in both the area occupied and density of burrowed areas has occurred since this series of surveys began. In colonies visited in several years between 2010 – 2019 (e.g. Middle, Rowe, Liddy's and Study colonies), we are confident there has been growth in the area occupied by breeding birds, with areas becoming more densely burrowed at the peripheries of established breeding zones. This has been combined with increases in density in the core areas of colonies surveyed in multiple years.

A well-defined estimate of the total number of breeding pairs of Westland petrels is difficult to provide, given the dispersed nature of their breeding, the inaccessible terrain, and the fact that an unknown proportion of nests occur outside recognizable colonies, although at very low density. Our rough estimates of area occupied, density, and occupancy suggest a minimum number of around 6,223 breeding pairs.

It is important to acknowledge that there are limitations to the methodologies applied. We consider that our estimates of changes in density and population growth in core areas of the colonies through CMR studies are robust. The area occupied by breeding petrels and total number of breeding pairs would benefit from further refinement through time. For example, the effort required to estimate burrow density over large areas of cliffy and untracked terrain meant that in some areas, we were not able to sample intensely enough to estimate within colony heterogeneity in density (e.g. for newly established areas). Thus, for statistical robustness, it would be advisable to refine the density measure for these areas. We place low confidence in these estimates for some colonies, which concern a small number (< 200) of burrows, and there was concern by field workers that the number estimated from density estimates was an order of magnitude different than what they observed on the ground.

However, we consider that improving this estimate would require a very heavy input of resources, which may be better deployed in addressing threats to the species such as reducing the possibility that pig predation occurs in the future, reducing habitat degradation and trampling of nests by goats, stabilization of the slip-damaged terrain within and adjacent to colonies, and possible fishery bycatch in poorly observed fleets or times of year, along with reducing mortality on birds grounded by lights near dwellings (Waugh and Wilson 2017).

While an estimate of the total population size may be a necessity for some threat-status assessment and management questions, we consider that the present study presents an order of magnitude-only estimate of the petrel population. In the future, efforts would be optimally deployed monitoring and understand the drivers of population growth parameters, such as the spread of colonies and through measures of density, occupancy and mark-recapture studies, to provide indices of population change, while the total population number may remain elusive.

Storm damage influences nesting of Westland petrels

Storm damage in Cyclone Ita in 2014 caused significant perturbations at several Westland petrel colonies (Waugh et al 2015b). These were tracked in detail at Rowe colony and Study colony. Rowe and Study Colonies, which were visited annually, continued to suffer treefall and slipping with subsequent major storm events, such as Cyclone Gita in 2018.

At two remote colonies, Dougies Bluff and Bees Nest, an estimated 38 – 50% of the colony area was lost to landslip and windfall by 2019, when the first on-the-ground survey was undertaken following the 2014 storm events. This is particularly important for Dougies Bluff colony, as it represents currently the largest colony in terms of surface area in 2019, thus may contain a sizable proportion of the 2019 Westland petrel population. Estimates of the areas damaged varied from < 10% (9 colonies), 10 – 50% (3 colonies) and no damage (5 colonies) of the 17 surveyed in 2019 and comprising over 95% of the breeding population of Westland petrels. At Rowe colony, 11% in the colony area was estimated lost in 2019 and 6% at Study colony. These areas lost contained 42% and 27% of monitored demographic study nests in 2014 and were in higher-density parts of the colonies (Waugh et al 2015b). At these two colonies, and Noisy Knob colony, very high-density areas for breeding were affected by landslips, indicating that the proportion of area affected may be relatively slight, but affect a disproportionately high number of breeding birds within the colony. Despite the need to establish new burrows by the petrels, following these perturbations, we were unable to detect differences in burrow occupancy between new and established areas. This may be because Westland petrels have a low breeding frequency and burrow occupancy habitually (noted since the early 2000s; Waugh et al. 2003), and their characteristics enable adaptation to change at a certain level. They show very high survivorship, and yet their breeding outputs put them among those species that could be considered to exemplify "life-in-the-slow-lane (Waugh et al. 2015a).

Conclusions

The study presents the first long-term estimate of population trend for the endangered Westland Petrel, and establishes a baseline estimate of breeding population at around 6,200 breeding pairs in 2019, corresponding to a world population size between 13800 and 17600 individuals. Independent measures of burrow density, and CMR estimates indicate that the population continues to grow by around 2% per year. Despite significant perturbations in the habitat of the Westland Petrel, the population growth has continued, and our surveys from 2007 – 2019 it is apparent the space occupied by the petrel colonies has extended, demonstrating the resilience of this long-lived species, giving a positive perspective for its future. It appears they can cope with disturbance of their breeding habitat,

by relocating to new colony areas. However, this will likely affect their reproductive output, and climatic change may also affect their food availability, breeding and migration phenology as well as their inter-breeding foraging areas.

References

Barbraud, C. and Weimerskirch, H., 2001. Emperor penguins and climate change. *Nature*, *411*(6834), p.183.

BirdLife International 2018. Procellaria westlandica . The IUCN Red List of Threatened Species 2018: e.T22698155A132629809. https://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T22698155A132629809.en. Downloaded on 20 January 2020.

Dilley, B.J., Hedding, D.W., Henry, D.A., Rexer-Huber, K., Parker, G.C., Schoombie, S., Osborne, A. and Ryan, P.G., 2019. Clustered or dispersed: testing the effect of sampling strategy to census burrow-nesting petrels with varied distributions at sub-Antarctic Marion Island. *Antarctic Science*, *31*(5), pp.231-242.

Dilley, B.J., Schramm, M. and Ryan, P.G., 2017. Modest increases in densities of burrow-nesting petrels following the removal of cats (Felis catus) from Marion Island. *Polar Biology*, *40*(3), pp.625-637.

Durant, J.M., Hjermann, D.O., Anker-Nilsen, T., Beaugrand, G., Mysterud, A., Pettorelli, N., Stenseth, N.C. 2005. Timing and abundance as key mechanisms affecting trophic interactions in variable environments. Ecology Letters 8:952-958.

Newshub 2020. https://www.newshub.co.nz/home/new-zealand/2018/02/live-updates-cyclone-gita-to-bring-new-zealand-to-standstill.html. Accessed 20 Jan. 2020.

QGIS Development Team (2019). QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org. Accessed 15 June 2019.

Thompson, W.L., White, G.C., Gowan, C. 1998. Monitoring vertebrate populations. Academic Press, San Diego, USA

Waugh, S.M., Barbraud, C., Adams, L., Freeman, A.N., Wilson, K.J., Wood, G., Landers, T.J. and Baker, G.B., 2015a. Modeling the demography and population dynamics of a subtropical seabird, and the influence of environmental factors. *The Condor: Ornithological Applications*, *117*(2), pp.147-164.

Waugh, S.M.; Cabrera, H.; Wood, G.C.; Davis, L.S. 2003. Burrow occupancy in Westland petrels (Procr:llaria westlandica). Notornis 50(3): 123-127.

Waugh, S.M., Poupart T, Wilson K-J. 2015b. Storm damage to Westland petrel colonies in 2014 from cyclone ITA. *Notornis* 62: 165 – 168. https://www.notornis.osnz.org.nz/node/4321

Waugh, S.M. & Wilson, K-J. 2017. Threats and threat status of the Westland Petrel Procellaria westlandica. Marine Ornithology 45: 195-203.

Wood, G.C. & Otley, H.M. (2013) An assessment of the breeding range, colony sizes and population of the Westland petrel (*Procellaria westlandica*), New Zealand Journal of Zoology 40: 3, 186-195, DOI: 10.1080/03014223.2012.736394