Monitoring ecotourism impacts on Toroa | Southern Royal Albatross breeding success on Campbell Island

Johannes H. Fischer & Theo Thompson
Marine Bycatch and Threats team, Department of Conservation
June 2025

Introduction

Toroa | Southern Royal Albatross (*Diomedea epomophora*), as the bird with the largest wingspan on the planet, are an icon of Motu Ihupuku | Campbell Island and the wider Aotearoa New Zealand Subantarctic region. However, the species has declined by >30% since the 1990s (Mischler et al. 2024), which is presumably largely caused by fisheries bycatch across their circumpolar range. Ecotourists, wishing to connect with this charismatic species, particularly at the Cole-Lyall boardwalk on Motu Ihupuku, may have additional impacts on this vulnerable, slow-breeding species, for example, by affecting breeding success during their visits. To better understand the impacts of ecotourism on Toroa, a multi-year nest survival study has been initiated, in which nests are monitored remotely using time-lapse cameras at areas within and outside the area visited by ecotourists.

Methods

Time-lapse camera set-up

Seventy-three time-lapse cameras (Swift Enduro Outdoor Cameras Australia with 32 GB SD cards) were placed in the Col study area across the 2023, 2024, and 2025 breeding seasons to monitor Toroa breeding success in 1) areas within the Col-Lyal boardwalk area visited by ecotourists (i.e., <50 m from the boardwalk; mean distance = 34 m) and 2) areas away from the boardwalk (i.e., the independent area; mean distance = 468 m ; Table 1). Seasons refer to the year in which chicks fledge, as Toroa lay eggs around late November/early December, from which chicks hatch in early February to early March, which ultimately fledge in early October (ACAP 2012). All cameras were placed during the incubation period. Cameras placed within the area visited by ecotourists covered ~86% of nest attempts, cameras placed in the independent area ~10% of breeding attempts (the Col study area held 148 nests in 2024 and 168 nests in 2025; Mischler et al. 2024, 2025). Cameras deployed in the 2023 season were placed in the independent area only, because the ecotourist-impact study was only initiated in the 2024 season, but data are included to increase the sample size. All cameras were deployed to avoid facing directly into the sunlight and programmed to take one photograph every two hours between 0600 and 2100 hours. Cameras deployed in 2023 failed quickly (see results) due to challenging conditions and suboptimal attachment methods. Consequently, cameras deployed in 2024 and 2025 were 1) sealed with Gorilla Roof and Gutter silicone sealant around the edges and in any holes (for plugs or microphone) and 2) attached with wire to a wooden stake with a side stake added for stability (Mischler et al. 2024, 2025). Cameras deployed within the area visited by ecotourists were attempted to be placed in a fashion that minimised detection by ecotourists to avoid the presence of cameras themselves creating confounding effects.

Finally, in 2025, as many nests as possible in the wider Col study area (n = 119) were visually assessed for 2024 breeding success (e.g., several play nests indicate successful fledging, skeletal remains indicate failure), allowing for an overall Col study area nest survival figure to be contrasted with the smaller camera-based sample sizes.

Table 1. Overview of time-lapse cameras deployed to monitor Toroa breeding success.

Breeding season	n Col-Lyall Boardwalk	n independent area
2023	-	12
2024	14	13
2025	16	18
Total	30	43

Data analysis

As the cameras deployed in the 2025 season are still collecting data at the time of writing, only data for the 39 Toroa nests monitored in 2023 and 2024 were considered for analysis, but a further five (four within the boardwalk area, one within the independent area) were excluded to not skew the results (e.g., due to unclear nest outcomes after early camera failure or confounding effects). For the preliminary analysis of nest survival of these nests, data were expressed in the form of nest outcome only (i.e., binary). For cameras that failed prior to fledging, outcome was inferred from the remains of the nest in the following season wherever possible. These data were then fit to a Bayesian generalised linear mixed-effects model with Bernoulli error term and a logit-link function:

$$logit(p_{ns}) = \alpha + \beta \cdot loc_i + \varepsilon_{i,year}$$

in which p_{ns} refers to the probability of nest success, α to the intercept, β to the fixed effect of the presence of ecotourists on nest success, loc_i to whether or not a nest is located within or away from the area visited by ecotourists, and $\varepsilon_{i,year}$ to a random annual effect. This structure allowed for the separation of annual variation from impacts of ecotourists on breeding success, if any. This model was fit in the Bayesian modelling software OpenBUGS (Spiegelhalter et al. 2014), in which Markov Chain Monte Carlo (MCMC) algorithms ensure the propagation of uncertainty. Only vague priors were used (N[0,0.01] for α and β, U[0,3] for σ_ε) and estimates are based on three MCMC chains with 25,000 iterations following a burn-in of 25,000, which was sufficient to reach model convergence. All estimates are reported in means and 95% credible intervals (CIs).

Results

Cameras deployed in 2023 lasted on average 35 days before failure or attachment apparatuses bending over, while cameras deployed in 2024 with improved attachment apparatuses and sealants lasted on average 290 days, with most cameras successfully covering the entire Toroa breeding season (Fig. 1). Toroa nest survival away from the area visited by ecotourists was estimated at 0.665 (0.415-0.870) in 2023 and 0.669 (0.420-0.870) in 2024, while nest survival in the area visited by tourists was estimated at 0.700 (0.401-0.925) in 2024 (Fig. 2). The fixed effect of the presence of ecotourists on Toroa nest survival (β) was estimated at 0.204 (-1.578-2.093), indicating non-significance. Both nest survival estimates align with the raw 2025 nest survival estimate of 0.655.

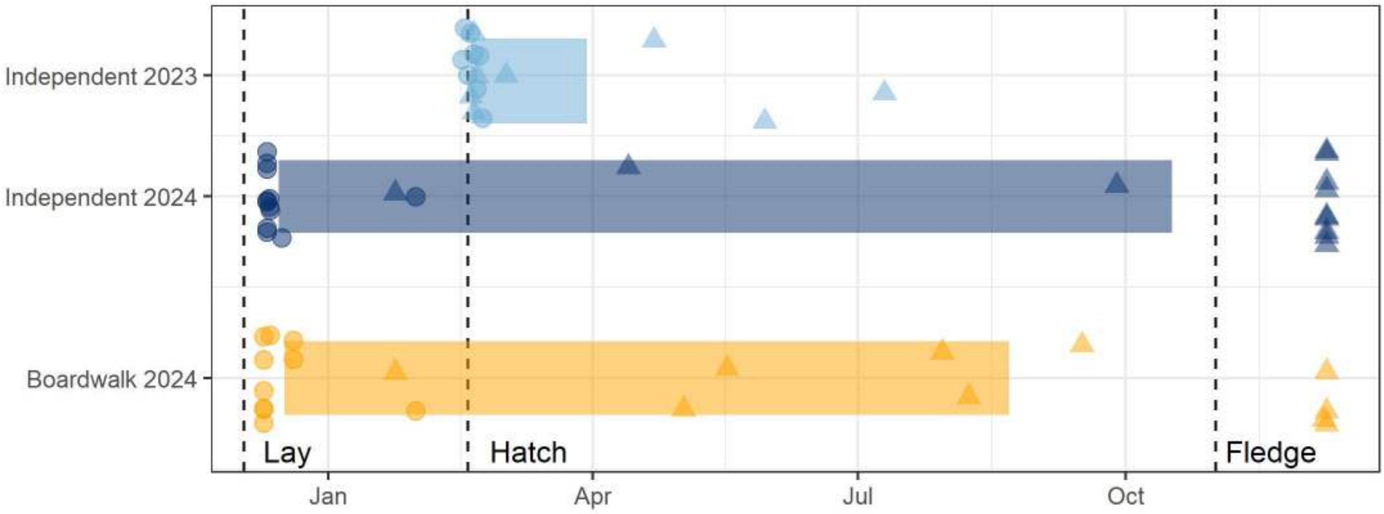

Fig. 1. Performance of remote cameras deployed to monitor Toroa nest success. Circles indicate deployment of individual cameras, triangles indicate failure/retrieval of individual cameras, and rectangles indicate average duration of camera coverage. Dashed lines indicate means of key phenological events during the Toroa breeding season (ACAP 2012).

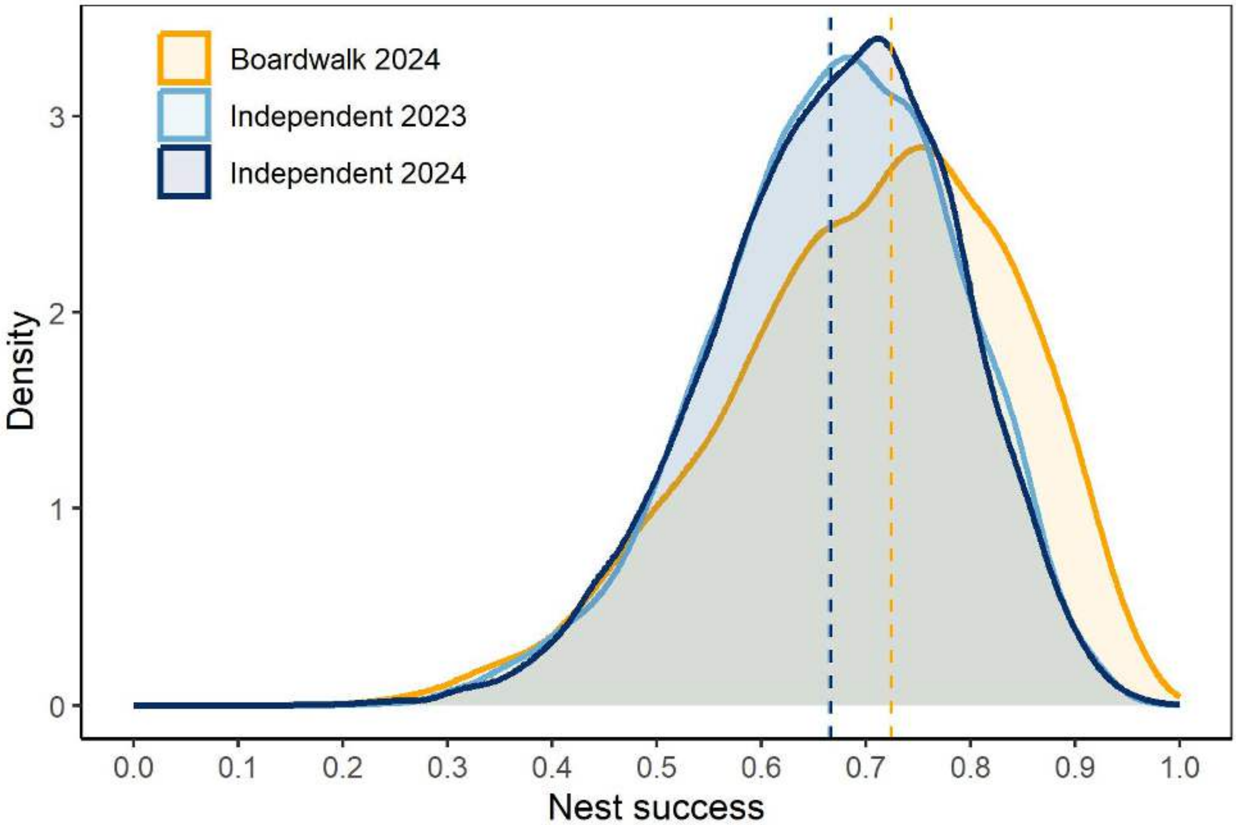

Fig. 2. Estimated Toroa nest success in 2023 and 2024 for areas visited by ecotourists (boardwalk) and away from ecotourists (independent). Dashed lines indicate means.

Discussion

Here, the first insights into Toroa nest survival in relation to ecotourism are presented. While the mean of the nest survival estimate in the area visited by ecotourists is slightly higher than the mean of the estimate in the independent area in both 2023 and 2024, it should be noted that all estimates are surrounded by large amounts of uncertainty. Consequently, while there are initial indications, that ecotourism does not appear to have an impact on Toroa nest survival, further data are required to gain more confidence.

The nest survival estimates presented here are not too dissimilar from past estimates, indicating that breeding success is unlikely to be the main driver of the recent population declines. Specifically, breeding success was estimated at ~0.684 in 2005-2008, at ~0.777 in 1995-1998, at ~0.757 in 1986-1995, and at ~0.647 in 1943-1973 (ACAP 2012, Moore et al. 1997, Waugh et al. 1997) and thus our estimates mirror the most recent estimates in the early 2000s. In comparison with other albatross species, these estimates indicate high levels of nest survival and thus suggest that recent declines are driven by (adult) survival, a symptom typical of bycatch impacts (e.g., Rexer-Huber et al. 2024, Richard et al. 2024). More advanced analyses, such as integrated population models (e.g., Richard et al. 2024), could further disentangle these demographic processes and any temporal trends therein.

To improve on the information presented here, several steps can be taken in future seasons:

1) More seasons of monitoring data are required to reduce uncertainty in the presented estimates. The cameras currently deployed and recording data for the 2025 season will provide further data to improve on the estimates reported on here, but ideally, further data are recorded over the 2026 season as well as an increased dataset will result in a better understanding of interannual variations and reductions of uncertainty.
2) Once a larger dataset has been obtained, an improved nest survival model should be fit to the data. Specifically, the camera data allows for the creation of daily capture histories of each nest to which a daily nest survival model could be fit, providing more robust estimates, including estimates into survival per phenological phase (e.g., see Fischer et al. 2021).
3) To augment the data collected, the outcome of all nests within the Col study area should be recorded by field staff by recording nest remains as was done for the 2024 season. A separate binary GLMM could be fit to these data allowing for a comparison with the daily nest survival model estimates, ultimately facilitating an assessment of potential confounding factors caused by the presence of cameras.
4) To ensure that adequate data are collected by cameras, all cameras should be sealed and fit using the improved wooden stake set-up used from 2024 onwards (e.g., see Mischler et al. 2024).
5) An analysis of changes in nest density over time in areas visited by ecotourists in comparison with changes in nest density away from areas visited by ecotourists should be conducted to provide another data stream to evaluate potential impacts from ecotourism.

Acknowledgements

This study was partially funded by a levy on ecotourism operators visiting the New Zealand Subantarctic, partially funded by Conservation Services Programme of the Department of Conservation which relies on levies on commercial fisheries (project POP2022-09 and POP2023-04) and partially funded by the Department of Conservation. Data collection was only possible thanks to the hard work of field crews, including Claudia Mischler, Chrissy Wickes, Peter Moore, Brodie Philp, Moira Pryde, Kaitlin Hamilton, and Holly Thompson, and the *Evohe,* its skippers, and crew for safe passage to and from this remote location.

References

ACAP. 2012. Southern Royal Albatross. Species Assessment.

Fischer JH et al. 2021. Preparing for translocations of a Critically Endangered petrel through targeted monitoring of nest survival and breeding biology. Oryx 55(4): 564-572.

Mischler C et al. 2024. POP2023-04 Campbell Island Seabird Research Project. Conservation Services Programme, Department of Conservation, Wellington.

Mischler C et al. 2025. POP2023-04 Campbell Island Seabird Research Project. Conservation Services Programme, Department of Conservation, Wellington.

Moore PJ et al. 1997. Southern Royal Albatross *Diomedea epomophora epomophora* census on Campbell Island, 4 January – 6 February 1996, and review of population figures. Science and Research Series no. 101. Department of Conservation, Wellington, New Zealand.

Rexer-Huber K et al. 2024. Antipodean wandering albatrosses and white-chinned petrels 2024. Conservation Services Programme, Department of Conservation, Wellington.

Richard Y et al. 2024. Antipodean Albatross multi-threat risk assessment. New Zealand Aquatic Environment and Biodiversity Report No. 332. 62 p. Fisheries New Zealand, Ministry of Primary Industries, Wellington.

Spiegelhalter, D et al. 2014. OpenBUGS user manual, version 323. BUGS, Cambridge, UK.

Waugh SM et al. 1997. Laying dates, breeding success and annual breeding of Southern Royal albatross *Diomedea epomophora epomophora* at Campbell Island during 1964-69.