Agreement on the Conservation
of Albatrosses and Petrels

**Ninth Meeting of the Population and Conservation
Status Working Group**

Swakopmund, Namibia, 25 May 2026

A standardised framework for processing abundance data of albatrosses and petrels and modelling their multi-decadal trends

JH Fischer, JA Chambon, T Thompson, PP Serafini, JA Arata, GB Baker, C Barbraud, E Bell, MD Bell, DJ Bell, J Black, A Bennison, DP Boyle, S Broni, K Carlyon, APB Carneiro, M Charteris, S Childerhouse, TA Clay, K Clemens-Seely, J Cooper, M Connan, K Delord, GP Elliott, C Facer, M Frazer, P Frost, K Hamilton, S Hamilton, KP Huyvaert, K Jensz, G Jiménez-Uzcátegui, S Lamb, S Latimer, AB Makhado, MJ Masotla, T Mattern, H McGovern, JC McInnes, S Micol, W Misiak, C Mischler, P Moore, GC Parker, S Poncet, K Pütz, N Pacoureau, K Ramm, S Ray, A Rebergen, K Rexer-Huber, CJR Roberston, G Robertson, OC Rowley, P Ryan, R Sagar, PM Sagar, M Schofield, P Seddon, RB Sherley, K Simister, JC Stahl, K Stevens, PA Street, C Suazo, GA Taylor, S Thalmann, DR Thompson, M Tierney, J Tinnemans, H Thompson, KJ Walker, J Watts, E Whitehead, C Wickes, J Whyte, AG Wood, RA Phillips, MC Double, I Debski & S Wotherspoon

Supplementary Material 5 *ACAPT Implementation*

Supplementary Material 6 *Exploration of ACAPT performance under varying q approaches*

Supplementary Material 8 *Exploration of Performance of density-based IUCN category evaluations under diminishing data quality*

SUMMARY

The ACAP database is the central repository for abundance data for ACAP species, yet these data are disparate and subject to a range of errors and idiosyncrasies. We developed a standardised framework for processing and modelling abundance data of ACAP species, consisting of 1) updating definitions of metrics, methods, and errors, and restructuring data, 2) standardising error corrections and incorporation of uncertainty, 3) converting annual breeding pairs into mature individuals, and 4) modelling of trends through a bespoke R package (*ACAPT*). Our framework then enabled population size and trend estimates comparable across space, time, and species, and facilitated IUCN Red List and ACAP High Priority Population assessments. We processed and modelled abundance data (471 counts) for 10 New Zealand endemics (3 *Diomedea*, 5 *Thalassarche*, and 2 *Procellaria* species), which varied in data quality, time-series length (38-91 years), and life-history characteristics.

Our results revealed complex, contrasting population trends within New Zealand's marine mega-avifauna. All *Diomedea* species exhibited ongoing declines (-35% since 1980, -21% since 2006). Trends of *Thalassarche* species varied; two species declined, one of which drastically (Salvin's Albatross; *T. salvini*; -31% since 1980, -6% since 2006), but one species remained stable, and one has increased. Both *Procellaria* petrel species showed long-term increases, and one species warrants downlisting. These contrasting trends cumulated in a net change of 1,400 (-187,300; 94,400) mature individuals since 1980, equating to a relative change of 0% (-17%; 17%) suggesting considerable uncertainty. Since 2006, declines became more certain with a net loss of -19,200 (-53,200; 35,800) mature individuals, equating to a relative change of -3% (-10%; 4%). Our framework and the resultant estimates represent a major advance in understanding and communicating the divergent fates of these seabirds. In doing so, we provide a crucial step towards mitigating losses and ultimately reversing the identified declines.

RECOMMENDATIONS

We recommend that PaCSWG:

1. *Endorse* the updated definitions of metrics, count methods, and errors for ACAP abundance data in Table 1-3.
2. *Endorse* the proposed abundance data processing, modelling framework, and decision rules, including of standardised approaches to 1) generate baseline uncertainty and account for sampling error, 2) account for detection error where possible, 3) account for occupancy error to ensure breeding pairs as the primary abundance metric, 4) account for phenology-related error using breeding success data, 5) convert breeding pairs to mature individuals using breeding probability estimates, and 6) modelling trends through the custom-built R package *ACAPT*.
3. *Endorse* the proposed IUCN Red List Criterion A2/A4 assessment approach under different timelines and *recommend* the most suitable statistical threshold.
4. *Recommend* that AC assigns resource in the AC Work Programme to improve the ACAP database data structure, mirroring the examples in the Supplementary Tables, through clearer separation of metrics, methods, and error treatment, and count adjustment history and incorporation of additional auxiliary data.
5. *Recommend* that AC commits resource in the AC Work Programme to apply the agreed data processing and modelling framework to the remaining ACAP species as well as completing the complementary spatial analyses required for the IUCN Red List assessments.
6. *Request* ACAP Secretariat to advise BirdLife International on IUCN Red List Assessments of ACAP species by 1 October 2026.
7. *Update* the monitoring guidelines to include the insights obtained through the data processing framework (e.g., ≥10% of breeding sites should be monitored).
8. Recommend that PaCSWG9 encourage ongoing research that will aid future abundance estimation processes, including breeding probability monitoring, retrospective assessments of errors, and (density-based) extrapolations.

9. *Request* AC approves the updated Terms of Reference of the Intersessional Group on ACAP Trends (Supplementary Material 0).

Un marco estandarizado para el procesamiento de datos sobre abundancia de albatros y petreles y la modelización de sus tendencias a lo largo de varias décadas

RESUMEN

La base de datos del ACAP es el repositorio central de datos sobre abundancia para las especies del ACAP. Sin embargo, estos datos son dispares y están sujetos a una serie de errores e idiosincrasias. Desarrollamos un marco estandarizado para el procesamiento y la modelización de datos sobre abundancia de especies del ACAP que consiste en 1) actualizar las definiciones de métricas, métodos y errores, y reestructurar los datos, 2) estandarizar las correcciones de errores y la incorporación de la incertidumbre, 3) convertir las parejas reproductoras anuales en ejemplares maduros, y 4) modelizar las tendencias a través de un paquete R personalizado (ACAPT). Luego, nuestro marco permitió realizar estimaciones de tamaño y tendencia de las poblaciones comparables en el espacio, el tiempo y las especies, y facilitó las evaluaciones de la Lista Roja de la UICN y las poblaciones de mayor prioridad del ACAP. Procesamos y modelizamos datos sobre abundancia (471 conteos) para 10 especies endémicas de Nueva Zelanda (3 de *Diomedea*, 5 de *Thalassarche* y 2 de *Procellaria*), que variaron en calidad de datos, longitud de la serie temporal (de 38 a 91 años) y características de la historia de vida. Nuestros resultados revelaron tendencias poblacionales complejas y contrastantes dentro de la megaavifauna marina de Nueva Zelanda. Todas las especies de *Diomedea* presentaron una disminución continua (-35 % desde 1980, -21 % desde 2006). Las tendencias de las especies de *Thalassarche* variaron; dos especies disminuyeron, una de ellas drásticamente (*Thalassarche salvini*; -31 % desde 1980, -6 % desde 2006), una se mantuvo estable y otra aumentó. Las dos especies de petreles *Procellaria* registraron aumentos a largo plazo, y una de ellas justifica una reclasificación a una categoría de menor riesgo en la lista. Estas tendencias contrastantes se acumularon en un cambio neto de 1400 (-187 300; 94 400) ejemplares maduros desde 1980, equivalente a un cambio relativo del 0 % (-17 %; 17 %), lo que sugiere una incertidumbre considerable. Desde 2006, las disminuciones se volvieron más ciertas con una pérdida neta de -19 200 (-53 200; 35 800) ejemplares maduros, lo que equivale a un cambio relativo de -3 % (-10 %; 4 %). Nuestro marco y las estimaciones resultantes representan un avance importante en la comprensión y comunicación de los destinos divergentes de estas aves marinas. Al hacerlo, proporcionamos un paso crucial hacia la mitigación de las pérdidas y, en última instancia, la reversión de las disminuciones identificadas.

RECOMENDACIONES

Recomendamos que el GdTPEC tome las siguientes medidas:

1. Aprobar las definiciones actualizadas de métricas, métodos de conteo y errores para los datos sobre abundancia del ACAP en las Tablas 1 a 3.
2. 2. Aprobar el marco propuesto para el procesamiento y modelado de datos sobre abundancia, que consiste en enfoques estandarizados para 1) generar incertidumbre de base y tener en cuenta el error de muestreo, 2) tener en cuenta el error de detección cuando sea posible, 3) tener en cuenta el error de ocupación para que las parejas reproductoras sean la principal métrica de abundancia, 4) tener en cuenta el error relacionado con la fenología utilizando datos de éxito reproductivo, 5) convertir las parejas reproductoras en ejemplares maduros utilizando estimaciones de probabilidad de reproducción y 6) modelar las tendencias a través del paquete R personalizado ACAPT.
3. Revisar las evaluaciones propuestas del Criterio A2/A4 de la Lista Roja de la UICN según diferentes cronogramas y distintos enfoques estadísticos para los estudios de caso y recomendar el enfoque más adecuado y las propuestas asociadas de reclasificación a categorías de mayor o menor riesgo.
4. Revisar los criterios para identificar las poblaciones de mayor prioridad del ACAP y acordar un criterio de λ y período de tiempo apropiado, teniendo en cuenta la incertidumbre.
5. Recomendar que el CA asigne recursos en el Programa de Trabajo del Comité Asesor para mejorar la estructura de datos de la base de datos del ACAP, imitando los ejemplos en las Tablas Suplementarias, con una separación más clara de las métricas, los métodos y el tratamiento de errores, y la incorporación de datos auxiliares adicionales.
6. Recomendar que el CA asigne recursos en el Programa de Trabajo del Comité Asesor para extender estos análisis a las demás especies del ACAP.
7. Solicitar que el CA actualice los Términos de Referencia del Grupo Intersesional sobre las Tendencias del ACAP para continuar trabajando, entre otros asuntos, en 1) la estandarización de las asignaciones de poblaciones de mayor prioridad del ACAP, 2) análisis espaciales de tendencias mediante la integración de datos de rastreo y 3) la elaboración de materiales de comunicación científica del ACAP que ilustren los análisis de tendencias para su inclusión en futuros productos del ACAP.
8. Solicitar al ACAP que asesore a BirdLife International sobre las evaluaciones de la Lista Roja de la UICN de las especies del ACAP para el 1 de octubre de 2026.

Un cadre standardisé pour le traitement des données d’abondance des albatros et des pétrels et la modélisation de leurs tendances pluri-décennales

RÉSUMÉ

La base de données de l’ACAP est le dépôt central des données d’abondance des espèces de l’ACAP ; toutefois, ces données sont hétérogènes et sujettes à divers types d’erreurs et de particularités. Nous avons développé un cadre standardisé pour le traitement et la modélisation des données d’abondance des espèces de l’ACAP, comprenant : (1) la mise à jour des définitions des indicateurs, des méthodes et des erreurs, ainsi que la restructuration des données ; (2) la standardisation des corrections d’erreurs et l’intégration de l’incertitude ; (3) la conversion des couples reproducteurs annuels en individus matures ; et (4) la modélisation des tendances à l’aide d’un package R dédié (ACAPT). Ce cadre a permis d’obtenir des estimations de la taille des populations et de leurs tendances comparables dans l’espace, dans le temps et entre espèces, et a facilité les évaluations de la Liste rouge de l’UICN et des populations hautement prioritaires de l’ACAP. Nous avons traité et modélisé des données d’abondance (471 comptages) pour 10 espèces endémiques de Nouvelle-Zélande (3 *Diomedea*, 5 *Thalassarche* et 2 *Procellaria*), présentant des variations en termes de qualité des données, de longueur des séries temporelles (38–91 ans) et de caractéristiques du cycle de vie. Nos résultats ont révélé des tendances démographiques complexes et contrastées au sein de la mégafaune aviaire marine de Nouvelle-Zélande. Toutes les espèces de *Diomedea* ont présenté des déclins continus (-35 % depuis 1980, -21 % depuis 2006). Les tendances des espèces de *Thalassarche* variaient : deux espèces ont décliné, dont une fortement (albatros de Salvin ; *T. salvini* ; -31 % depuis 1980, -6 % depuis 2006), tandis qu’une espèce est restée stable et une autre a augmenté. Les deux espèces de pétrels *Procellaria* ont montré des augmentations à long terme, et l’une d’entre elles justifie un déclassement. Ces tendances contrastées ont abouti à un changement net de 1 400 (-187 300 ; 94 400) individus matures depuis 1980, soit une variation relative de 0 % (-17 % ; 17 %), ce qui indique une incertitude considérable. Depuis 2006, les déclins sont devenus plus certains, avec une perte nette de -19 200 (-53 200 ; 35 800) individus matures, soit une variation relative de -3 % (-10 % ; 4 %). Notre cadre et les estimations qui en résultent représentent une avancée majeure dans la compréhension et la communication des trajectoires divergentes de ces oiseaux marins. Ce faisant, nous apportons une contribution essentielle aux efforts visant à atténuer les pertes et, à terme, à inverser les déclins identifiés.

RECOMMANDATIONS

Nous recommandons que le PaCSWG :

1. Approuve les définitions mises à jour des indicateurs, des méthodes de comptage et des erreurs pour les données d’abondance de l’ACAP figurant aux tableaux 1 à 3.
2. Approuve le cadre proposé pour le traitement et la modélisation des données d’abondance, consistant en des approches standardisées visant à : (1) générer

l'incertitude de référence et prendre en compte l'erreur d'échantillonnage ; (2) prendre en compte l'erreur de détection lorsque cela est possible ; (3) prendre en compte l'erreur d'occupation afin de garantir que les couples reproducteurs constituent la principale mesure d'abondance ; (4) prendre en compte l'erreur liée à la phénologie à l'aide de données de succès reproducteur ; (5) convertir les couples reproducteurs en individus matures à l'aide d'estimations de probabilité de reproduction ; et (6) modéliser les tendances à l'aide du package R dédié ACAPT.

3. Examine les évaluations proposées des critères A2/A4 de la Liste rouge de l'UICN selon différents horizons temporels et différentes approches statistiques pour les études de cas, et recommande l'approche la plus appropriée ainsi que les propositions correspondantes de reclassement (rehaussement/déclassement).
4. Examine les critères permettant d'identifier les populations prioritaires élevées de l'ACAP et convenir d'un λ approprié ainsi que d'un critère de période, tout en tenant compte de l'incertitude.
5. Recommande que le Comité consultatif alloue des ressources dans son programme de travail afin d'améliorer la structure des données de la base ACAP, en s'inspirant des exemples figurant dans les tableaux supplémentaires, notamment par une séparation plus claire des indicateurs, des méthodes et du traitement des erreurs, ainsi que par l'intégration de données auxiliaires supplémentaires.
6. Recommande que le Comité consultatif consacre des ressources dans son programme de travail afin d'étendre ces analyses aux autres espèces de l'ACAP.
7. Demande au Comité consultatif de mettre à jour le mandat du groupe intersessionnel sur les tendances de l'ACAP afin de poursuivre les travaux portant notamment sur : (1) la standardisation de l'identification des populations prioritaires élevées de l'ACAP ; (2) les analyses spatiales des tendances fondées sur l'intégration de données de suivi ; et (3) le développement de supports de communication scientifique de l'ACAP illustrant les analyses de tendances, en vue de leur intégration dans les futurs produits de l'ACAP.
8. Demande à l'ACAP de conseiller BirdLife International sur les évaluations de la Liste rouge de l'UICN concernant les espèces de l'ACAP d'ici au 1er octobre 2026.

A standardised framework for processing abundance data of albatrosses and petrels and modelling their multi-decadal trends

Johannes H Fischer[1,2] *, Johannes A Chambon[3], Theo Thompson[1,3], Patricia P Serafini[2,4], Javier A Arata[5], G Barry Baker[2,6,7,8], Christophe Barbraud[,9], Elizabeth Bell[10], Mike D Bell[11], Dave J Bell[11], Jennifer Black[12], Ashley Bennison[12], Dave P Boyle[14], Sharyn Broni[1], Kris Carlyon[15], Ana P B Carneiro[2,16], Matt Charteris[17], Simon Childerhouse[18], Tommy A Clay[,19, 20], Katie Clemens-Seely[1], John Cooper[21], Maelle Connan[22], Karine Delord[2,9], Graeme P Elliott[1], Colin Facer[1], Mark Frazer[23], Peter Frost[24], Kaitlyn Hamilton[1], Sheryl Hamilton[2,15], Kate P Huyvaert[2,25,26], Katrina Jensz[8], Gustavo Jiménez-Uzcátegui[2,27], Simon Lamb[10], Scott Latimer[1], Elizabeth K Mackley[13], Azwianewi B Makhado[2,21,28], Makhudu J Masotla[28], Sarah J Manthorpe[13], Thomas Mattern[3,29,30], Hollie McGovern[1], Julie C McInnes[2,6,31], Susan Micol[16], Wiesława Misiak[32], Claudia Mischler[1], Peter Moore[1], Graham C Parker[33], Sally Poncet[12,34,35] Klementz Pütz[35], Nathan Pacoureau[36], Kris Ramm[1], Samantha Ray[10], Aalbert Rebergen[1], Kalinka Rexer-Huber[33], Christopher JR Roberston[1], Graham Robertson[31], Olivia C Rowley[1], Peter Ryan[21], Rachael Sagar[1], Paul M Sagar[2,37], Matt Schofield[38], Phillip Seddon[3], Richard B. Sherley[39,40], Kate Simister[1], Jean-Claude Stahl[41], Kim Stevens[21], Phillip A Street[25], Cristian Suazo[,16,42], Graeme A Taylor[1], Sam Thalmann[15], David R Thompson[37], Megan Tierney[2,43,44], Joris Tinnemans[1], Holly Thompson[1], Kath J Walker[1], Jim Watts[1], Edin Whitehead[45], Chrissy Wickes[1], Jordana Whyte[1], Andrew G Wood[12], Richard A Phillips[2,13], Mike C Double[31], Igor Debski[1,2] & Simon Wotherspoon[31]

[1] Department of Conservation, Wellington, New Zealand.
[2] Population and Conservation Status Working Group, Agreement on the Conservation of Albatrosses and Petrels, Hobart, Australia.
[3] Department of Zoology, Otago University, Dunedin, New Zealand.
[4] Laboratório de Biomarcadores de Contaminação Aquática e Imunoquímica, Universidade Federal de Santa Catarina, Brazil.
[5]Centro de Investigación en Dinámica de Ecosistemas Marinos de Altas Latitudes (IDEAL), Valdivia, Chile.
[6] Institute of Marine and Antarctic Studies, University of Tasmania.
[7] Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin 0909, Northern Territory, Australia.
[8] Latitude 42 Environmental Consultants, Hobart, Tasmania, Australia.
[9] Centre d'Etudes Biologiques de Chizé – CNRS UMR7372, France.
[10] Wildlife Management International Ltd., Blenheim, New Zealand.
[11] Toroa Consulting Ltd., Blenheim, New Zealand.
[12] Government of South Georgia & the South Sandwich Islands, Stanley, Falkland Islands.
[13] British Antarctic Survey, Natural Environment Research Council.
[14] Chatham Island Taiko Trust. Chatham Islands, New Zealand.
[15] Department of Natural Resources and Environment, Tasmania, Australia.
[16] BirdLife International, Cambridge, UK.
[17] Waybacks Ltd., Westport, New Zealand.
[18] Blue Planet Marine, Canberra, Australia.
[19] Environmental Defence Fund, Monterrey, USA.
[20] Institute of Marine Sciences, University of California, Santa Cruz, Santa Cruz, USA.
[21] FitzPatrick Institute of African Ornithology, University of Cape Town, Rondebosch, South Africa.
[22] Department of Zoology, Institute for Coastal and Marine Research, Nelson Mandela University, Gqeberha, South Africa.
[23] Canterbury Museum, Christchurch, New Zealand.
[24] Science Support Service, Whanganui, New Zealand.
[25] Department of Fish, Wildlife, and Conservation Biology, Colorado State University, Fort Collins, Colorado, USA.
[26] Department of Veterinary Biology Microbiology and Pathology, Washington State University, Pullman, Washington, USA.
[27]Charles Darwin Research Station, Charles Darwin Foundation, Puerto Ayora, Galápagos, Ecuador.
[28] Department of Forestry, Fisheries, and the Environment, South Africa.
[29] Tawaki Trust, Dunedin, New Zealand.
[30] Global Penguin Society, Chubut, Argentina.
[31] Australian Antarctic Division, Department of Climate Change, Energy, The Environment and Water.

[32] Secretariat of the Agreement on the Conservation of Albatrosses and Petrels, Hobart, Australia.
[33] Parker Conservation, Karitane, New Zealand.
[34] South Georgia Surveys.
[35] Antarctic Research Trust, Forch, Switzerland.
[36] Department of Biological Sciences, Simon Fraser University, Burnaby, Canada.
[37] New Zealand Institute for Earth Science, Wellington, New Zealand.
[38] Department of Mathematics and Statistics, Otago University, Dunedin, New Zealand.
[39] Centre for Ecology and Conservation/Environment and Sustainability Institute, University of Exeter, Penryn, UK.
[40] Sharjah Marine Science Research Centre, University of Khorfakkan, Khor Fakkan, Sharjah, United Arab Emirates.
[41] Te Papa Tongarewa, Wellington, New Zealand.
[42] Department of Animal Ecology and Systematics, Justus Liebig University Giessen, Giessen, Germany.
[43] South Atlantic Environmental Research Institute, Stanley, Falkland Islands.
[44] Joint Nature Conservation Committee, UK.
[45] Auckland University, Auckland, New Zealand.

* Corresponding Author: johannesfischer@live.nl

Acknowledgements

While the authorship is extensive, it is not exhaustive. This study is the culmination of the dedication, hard work, and perseverance of generations of seabird scientists. We are immensely grateful to everyone who has contributed to each individual count of albatrosses or petrels that ultimately led to the timeseries collated here, and we are humbled to stand on their collective shoulders. We are also grateful to ACAP for their foresight to establish and maintain the ACAP abundance database. Finally, we thank Caio Kenup for his assistance in processing expert-elicited data and all experts that participated in elicitations. For Aotearoa New Zealand specifically, we are most grateful to mana whenua mana moana (People of the Land and Sea) and we thank Kāi Tahu, Te Poari a Pukekura, Ngāti Waiwai, Ngāti Rehua Ngātiwai ki Aotea, Ngāti Mutunga, and the traditional owners of Te Tara Koi Koia, Motuhara, and Rangitutahi for allowing us to work on their taonga (treasures, applicable to species, islands, land, and sea) across decades. Access to many of these islands would not have been possible without the expertise and skills of generations of skippers and pilots and we thank the skippers and crew of the *Evohe*, *Tiama*, *Baltazar*, and *Awesome*, and the pilots of Southern Lake Helicopters in particular. Many of the islands on which work was undertaken are kept free of invasive predators (sometimes after successful eradications) and thus we thank the local Department of Conservation staff in Murihiku and the Hauraki for their intergenerational dedication to biosecurity. Further thanks goes to Armada de Chile (III Zona Naval) for support fieldwork on Diego Ramírez, all the crews from their units Lautaro, Cabrales, Oscar Viel, POV Fuentealba, and the lighthouse keepers at Gonzalo Island.

Funding Statement

Just as it is impossible to exhaustively incorporate all data contributors as authors, it is not possible to exhaustively compile all sources of funding, cumulating to dozens if not hundreds of millions of USD, that enabled the generation of some of the longest animal abundance timeseries on the planet. Therefore, funding sources stated here should be treated as the most prominent ones only. Most New Zealand counts were funded through the Department of Conservation in general, and its Conservation Services Programme (CSP) in particular. CSP is partially funded through a levy on the quota holders of relevant commercial fish stocks in New Zealand, and thus we thank the New Zealand fishing industry for their substantial contribution. Additional New Zealand counts were supported by the Ministry of Primary Industries, Albatross Research, the DeepWater Group, Wildlife Management International, the Vontobel Foundation, Antarctic Research Trust, and the Global Penguin Society. Counts on South Georgia (Islas Georgias del Sur)[1] were funded by South Georgia Surveys, the Government of South Georgia and the South Sandwich Islands, and the Antarctic Research Trust, counts on the Prince Edward Islands were funded through the National Research Foundation, counts on Macquarie Island were funded by the Department of Climate Change, Energy the Environment and

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[1] A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty of the Falkland Islands (Islas Malvinas), South Georgia and the South Sandwich Islands (Islas Georgias del Sur e Islas Sandwich del Sur) and the surrounding maritime areas.

Water with logistic support provided by the Tasmanian National Parks and Wildlife Service and the Australian Antarctic Division.

Ethics Statement

All fieldwork in New Zealand was approved and supported by mana whenua mana moana (People of the Land and Sea), specifically, Kāi Tahu, Te Poari a Pukekura, Ngāti Waiwai, Ngāti Rehua Ngātiwai ki Aotea, Ngāti Mutunga, and the owners of Te Tara Koi Koia, Motuhara, and Rangitutahi, as well as the Department of Conservation through the relevant Wildlife Authority and Entry Permits. Fieldwork on Diego Ramirez was approved by Res. Nº 959 and Nº 6093 from Agriculture and Livestock Service. Fieldwork on South Georgia (Islas Georgias del Sur)[1] was permitted by the Government of South Georgia and the South Sandwich Islands. Fieldwork at the Prince Edward Islands was conducted under permit from the South African National Antarctic Programme. Fieldwork on Macquarie Island was performed under NRE Tasmania Standard Operating Procedures in association with reserve access authority issued by Tasmanian Parks and Wildlife Service.

Conflict of Interests

The authors have no conflicts of interest to declare.

ACAP IG Trends workshop 24 May outcome:

Each update from the ACAP IG Trends workshop on 24 May is documented within context in text boxes.

ABSTRACT

Estimating marine megafauna abundance and trends is challenging due to the difficulty of ensuring data coherency across space, time, and species. Seabird abundance data are disparate and subject to errors and idiosyncrasies. We developed a standardised framework for processing and modelling abundance data of species listed by the Agreement on the Conservation of Albatrosses and Petrels (ACAP), consisting of the restructuring of data, incorporating uncertainty, standardising error corrections, converting into mature individuals, and modelling of trends through a bespoke R package (*ACAPT*). Our framework enabled estimates comparable across space, time, and species, and facilitated conservation assessments. We processed and modelled abundance data (893 counts) for thirteen ACAP species (4 *Diomedea*, 6 *Thalassarche*, 1 *Phoebastria*, and 2 *Procellaria*) from 36 breeding sites with varying data quality, time-series lengths (40-91 years), and life-history characteristics. Results revealed complex, contrasting trends among different species. New Zealand *Diomedea* albatrosses exhibited ongoing declines (combined -25% since 1970), while Wandering Albatrosses (*D. exulans*) declined globally and then recovered following growth at Indian Ocean sites. *Thalassarche* albatross trends varied: three species declined considerably (Grey-headed *T. chysostoma* -66%, Campbell *T. impavida* -51%, and Salvin's Albatross *T. salvini* -31%), two species appeared stable, although White-capped Albatross *T. steadi* may be declining, and one increased. Grey-headed Albatross warrant uplisting on the IUCN Red List. Both *Procellaria* petrels showed long-term increases. Westland Petrel may warrant downlisting on the IUCN Red List. These trends cumulated into a net change of -553,800 (95% credible interval: -1,342,600; -186,800) mature individuals since 1970, a relative change of -38% (-52%; -24%), predominantly driven by Grey-headed Albatross declines. Since 2006, declines became more certain with a net change of -153,900 (-206,000; -78,600) mature individuals i.e., -14% (-20%; -9%). Despite these declines, only one site warranted listing as an ACAP High Priority Population due to the high decline threshold (-46% within 20 years) and thus policy revisions may be warranted. Our framework and the resultant estimates represent a major advance in understanding and communicating the divergent fates of these seabirds. In doing so, we provide a crucial step towards mitigating losses and ultimately reversing the identified declines.

Key words

Population counts; Seabirds; Conservation; Agreement for the Conservation of Albatrosses and Petrels; state-space model; Bayesian inference.

CONTENTS

1. INTRODUCTION

Understanding population size and trends is a fundamental objective of conservation biology, particularly in the face of the ongoing, human-induced biodiversity loss (Ceballos et. al 2015). Defaunation of global marine ecosystems is accelerating (e.g., Paleczny *et al.* 2015, Pacoureau *et al.* 2021, Edgar *et al.* 2024). However, obtaining confident estimates of abundance and trends for marine megafauna is particularly difficult (e.g., Dulvy *et al.* 2024). Decades of industrial-scale sampling coupled with highly advanced statistical models (e.g., stock assessments) have enabled the estimation of abundance and trends for marine megafauna of commercial interest, such as tunas and billfishes (Juan-Jorda *et al.* 2022) and, historically, whales (Baker & Clapham 2004). Despite these efforts, regularly obtaining accurate estimates remains challenging (e.g., Edgar *et al.* 2024). Yet, some marine megafauna species return to land to breed (e.g., sea turtles, pinnipeds, and seabirds), creating opportunities for collecting abundance data at breeding/aggregation sites (Paleczny *et al.* 2015, LaReu *et al.* 2021, Hays *et al.* 2024). However, marine megafauna species are usually also characterised by highly complex life histories (i.e., delayed maturity, high longevity, low productivity, with breeding frequencies below once per annum; e.g., Weimerskirch 2018, Roast *et al.* 2023), creating considerable challenges to inferring true trends from abundances observed at terrestrial breeding/aggregation sites (e.g., Oppel *et al.* 2022). Despite these challenges, abundance estimates are crucial for marine megafauna conservation, as, for example, for determining conservation status (e.g., Sherley *et al.* 2020ab, IUCN 2012) or understanding risk posed by threats such as fisheries bycatch (Edwards *et al.* 2023, Anon. 2025, JB Bell *et al.* 2025) are highly dependent on these data and their accuracy. Consequently, obtaining estimates of abundance and trends of marine megafauna, including for seabirds, remains a global conservation priority.

Like all marine megafauna, seabirds are charismatic, culturally and ecologically important, and highly threatened (Dias *et al.* 2019, Jones *et al.* 2025), yet estimates of seabird abundance and trends that are comparable across space, time, and species remain elusive. Paleczny *et al.* (2015) highlighted a ~70% decline in seabird abundance during 1950-2010 yet did not ensure direct comparability among abundance data. Fischer *et al.* (2024) also highlighted ongoing, long-term declines in New Zealand albatross and petrel populations, but equally did not ensure direct comparability. Not ensuring comparability among counts is problematic as this omission risks propagation of multiple errors, ignores uncertainty, and therefore prevents confident insights into trends and conservation status.

Among seabirds, albatrosses and large petrels are of particular conservation concern, predominantly due to the global impacts of bycatch and invasive species (Phillips *et al.* 2016), leading to the ratification of the Agreement on the Conservation of Albatrosses and Petrels (ACAP) in 2004, which strives to conserve albatrosses and petrels by coordinating international activities to mitigate threats to their populations (Cooper *et al.* 2006). Obtaining abundance data for these species are particularly challenging as they breed on some of the most remote, hostile, and inaccessible islands on the planet (Phillips *et al.* 2016, Rodrigues *et al.* 2019) and thus individual counts are often extremely costly. For example, New Zealand invested >$400,000 USD in a single year to obtain abundance data for nine species (CSP 2025). To overcome these challenges, various methods are used to estimate albatrosses and petrel abundance, including from the ground, vessels, airplanes, helicopters, drones, and satellites (e.g., Poncet *et al.* 2006, Robertson *et al.* 2008, Fretwell *et al.* 2017, Frost *et al.* 2025, Elliott *et al.* 2025). These counts, however, are rarely conducted at the same time and consequently count metrics (e.g., nests, breeding adults, eggs, or chicks) also vary, including among counts of the same species at the same breeding site (Parker & Rexer-Huber 2020, Wolfaardt & Phillips 2020). Due to these different count methods, metrics, and timings, abundance data of albatrosses and petrels are subject to several different sources of error and inconsistent accounting thereof. Consequently, count data of ACAP species are disparate and subject to considerable idiosyncrasies. A standardised framework is thus required to process these challenging count data to 1) obtain comparable abundance estimates, 2) enable unified modelling of their trends, and ultimately, 3) gain comparable insights into their conservation status.

Here, we developed a standardised framework to process and estimate albatross and petrel abundance and model their multi-decadal trends using Bayesian statistics. We illustrate the utility and flexibility of our framework by providing the first comparable estimates of abundance for 13 albatross and petrel species listed on ACAP Annex 1 (ACAP 2025a), 11 of which are listed as threatened (i.e., Critically Endangered, Endangered, or Vulnerable) on the IUCN Red List (BirdLife International 2025). Our standardisation framework is applicable to a wide range of other seabird species, and our trend model is applicable to a wide range of animal species. Consequently, our work paves a way forward towards obtaining better population estimates of marine megafauna and their global trends.

2. METHODS AND MATERIALS

Our framework to process and model abundance data (i.e., counts) of albatrosses and petrels consisted of five distinct steps: 1) restructuring existing data and redefining metrics, methods, and errors, 2) standardising incorporation of uncertainty and error corrections to express all data in a uniform metric: annual breeding pairs at the start of the breeding period, 3) converting annual breeding pairs into mature individuals, 4) modelling of trends, and 5) assessing their conservation status. Our framework was implemented through a Bayesian modelling approach, enabling the integration of both modelled data as well as additional data through informative priors where required. Below, we detail the methodology for each step (section 2.1-2.6), after which we present 13 case studies (section 2.7, Supplementary Material 1 & 2) using data from ten albatrosses and petrel species endemic to New Zealand (Antipodean Albatross *Diomedea antipodensis*, Northern Royal Albatross *D. sanfordi*, Southern Royal Albatross *D. epomophora*, Campbell Albatross *Thalassarche impavida*, Buller's Albatross *T. bulleri*, White-capped Albatross *T. steadi*, Chatham Albatross *T. eremita*, Salvin's Albatross *T. salvini*, Westland Petrel *Procellaria westlandica*, and Black Petrel *P. parkinsoni*), two circumpolar species (Wandering Albatross *D. exulans* and Grey-headed Albatross *T. chrysostoma*), and one tropical species (Waved Albatross *Phoebastria irrorrata*).

2.1. Definitions of metrics, methods, and errors, and developing a suitable data structure

ACAP is the central repository for abundance data for all 31 albatross and petrel species listed on its Annex 1 and collates these data in a centralised database (the ACAP database from hereon; ACAP 2025a). For each count, the location (breeding site and part-site), year, method, reliability, accuracy, and reference are recorded. Breeding site is defined as an individual island or a specific colony on the mainland (but the definition takes conservation management into account too) of which part-sites are subsections (e.g., specific study or index areas within a larger colony; Wolfaardt & Phillips 2020). Year refers to a biologically relevant year, which varies between species depending on their phenology, and as per ACAP convention, the calendar year in which chicks fledge is recorded. Method is expressed as A-H, each letter representing a different method (and sometimes metric and error(s) accounted for), reliability is recorded as 1-5 each representing a different category of error estimation or absence thereof, and accuracy scores consist of "high" (≤10% of stated figure), "medium"

(≤50%), or “low” (≤100%) (ACAP 2009). In 2025, the database held 3,884 counts across two centuries (1871-2025) for all 31 ACAP species, and 815 (21%) counts across eight decades (1951-2025) for the ten species endemic to New Zealand (ACAP 2025a,b).

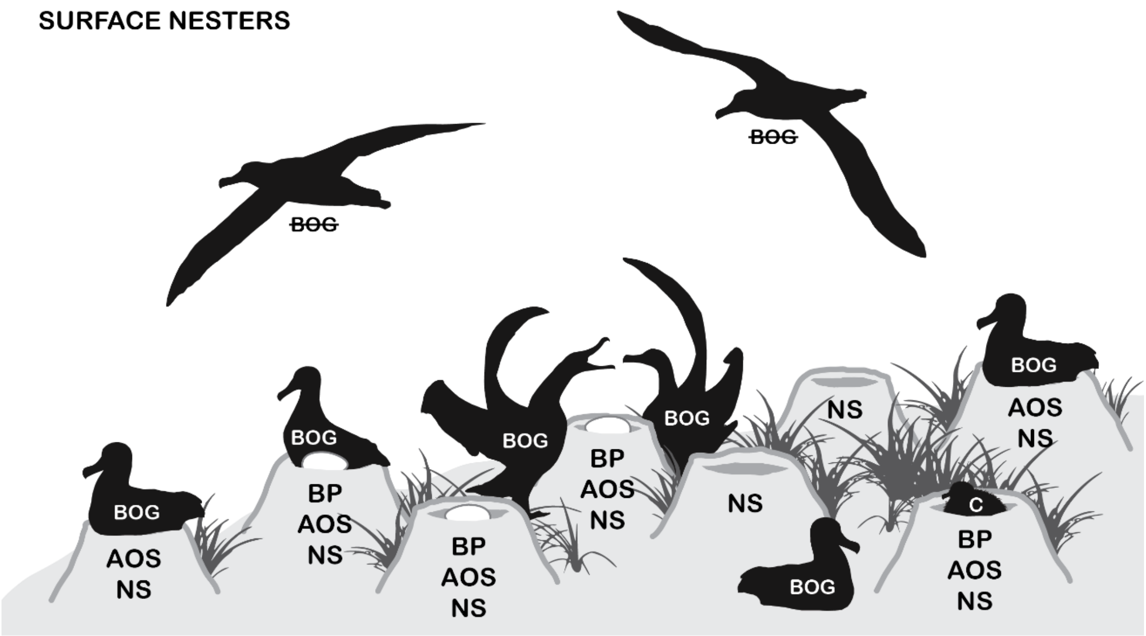
We reviewed the data structure within the ACAP database and its utility using an anonymous online questionnaire (https://forms.gle/k7uBuBpRDz3k5qAUA), an expert elicitation, and several online discussions during 2024-2025. In total, 39 experts on albatross and petrel counts from around the world (including all ACAP Population and Conservation Status Working Group members) were invited, and 17 (44%) provided insights. Feedback highlighted key concerns with the current data structure of the ACAP database: 1) recording of the count metric is unclear, and method, metric, and error(s) are often conflated, 2) methods, metrics, and errors are poorly defined and open to multiple interpretations, 3) the reliability score is of limited utility and is also conflated with method and metric, and 4) accuracy scores were interpreted differently by experts when requested to express their beliefs numerically, with responses differing depending on whether surface or burrow/crevice nesters were considered.

To address the identified concerns, we revised the structure of the ACAP abundance data and updated definitions of key terms. Specifically, we 1) explicitly separated the target metric, the count method, and the associated errors (whether accounted for or not), 2) agreed on updated definitions for metrics (and the hierarchy therein), methods, and errors (Table 1-3, Fig. 1-2), 3) defined a standardised framework for how metric, method, and errors interacted for surface and burrow/crevice nesters by mapping all logical combinations within a decision tree, and 4) updated the numerical representation of the ACAP accuracy scores based on the mean elicited values (for counts of surface nesters, “high” ≤10% of the stated figure, “medium” ≤33%, and “low” ≤67%; for burrow/crevice nesters, “high” ≤16%, “medium ≤46%, “low” ≤88%). We only considered counts conducted since 1970 (inclusive) as prior counts were considered of poor accuracy.

ACAP IG Trends workshop 24 May outcome:
Only counts conducted since 1970 (inclusive) should be included as older counts were perceived of lower accuracy. In addition, the IG Trends is to engage with data custodians to identify any poor-quality counts since 1970, document the underlying reasons for why counts should be considered of poor quality, and jointly decide on their inclusion.

We did not redefine the spatial or temporal resolutions of counts in the ACAP database. Instead, we agreed that 1) breeding site represents the finest spatial resolution for expressing abundance data, and 2) year is retained, as per ACAP convention, as the finest temporal resolution. While a breeding site is usually defined as an island or specific colony in the ACAP database, here we integrated monitoring considerations as well. For example, if two breeding sites are usually monitored at the same time (e.g., because they are closely located, neighbouring islands), we consider them jointly as a single breeding site to increase efficiency (i.e., Little Solander and Solander data were processed as Solander Islands data, instead of separately). In addition, we considered large archipelagos consisting of several breeding sites jointly to further increase efficiency.

ACAP IG Trends workshop 24 May outcome:
For lumped archipelago breeding sites that may exhibit divergent trends based on clear underlying drivers (e.g., differences in invasive species composition), modelling of trends for part-sites of archipelagos separately should be investigated, provided ≥3 counts for each site exist.



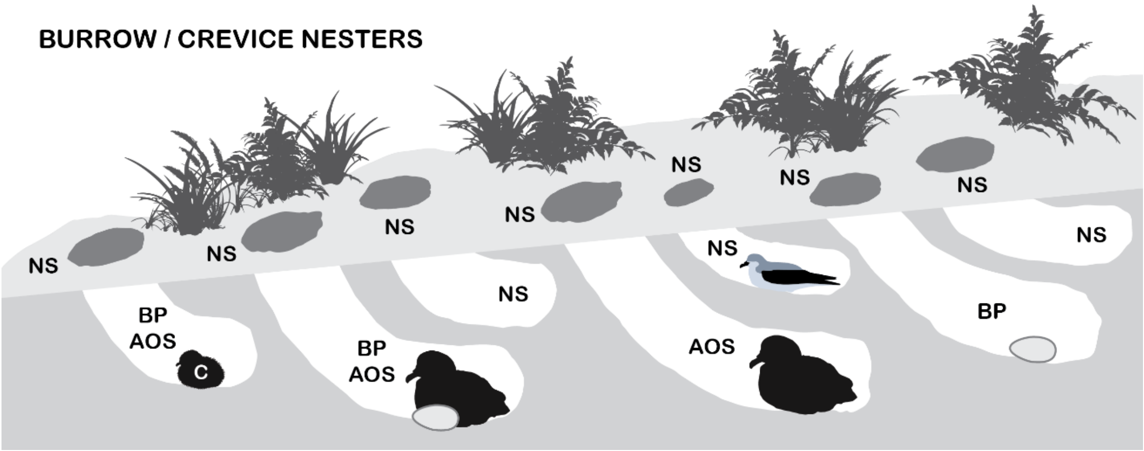


Fig. 1. Schematic illustrating the different metrics for both surface and burrow/crevice nesters and how they relate to each other, creating a hierarchy (Table 1). BP: annual breeding pairs, AOS: apparently occupied sites, BOG: birds on ground, NS: nest sites, C: chicks. Note the non-target *Pterodroma* petrel (grey) among the target *Procellaria* petrels (black) to highlight the challenge of mixed-species colonies. Credit: J. Chambon.

Following these adjustments to the structure of the ACAP database abundance data, we recorded annual abundance data at the breeding site level for the ten ACAP species endemic to New Zealand. For each record, we documented the metric in which the count originally was conducted and ultimately expressed in, method(s) applied, and errors accounted or unaccounted for, as well as the updated ACAP accuracy score, additional auxiliary data (e.g., date, breeding phenology, and breeding success (section 2.2.3), and breeding probability (section 2.3)), and references (Supplementary Material 1 & 2). This reformatting exercise required revisiting the original publications or reports. In doing so, we regularly identified additional abundance data not currently captured in the ACAP database, which we then also included in our data. Specifically, we identified 237 additional counts for these ten species, leading to a total data set of 893 annual breeding site counts (note the ACAP database records counts of part-sites separately, while we combined these, so our counts represent combinations of year and breeding site). In revisiting original sources, metric descriptors did regularly not match our definitions, and thus we engaged with data holders to ensure adequate data interpretation. Ultimately, our restructuring resulted in improved and more transparent datasets suitable for standardised processing and modelling of long-term trends and conservation status assessments (section 2.2-2.5).

Table 1. Updated, standardised definitions of metrics for counts of albatrosses and petrels. See Fig. 1 for a graphical representation.

Metric (acronym)	**Description**	**Example**
Annual Breeding pairs (BP)	The number of pairs successfully laying an egg in the year[A] the count is conducted. For this metric, the presence of an egg or chick will usually need to be confirmed systematically. This is the ACAP standard metric in which species abundance is expressed. This metric is sometimes referred to as occupied nest sites or expressed in number of eggs but to ensure consistency, is referred to as BP hereafter.	Wolfaardt & Phillips 2020, Elliott *et al.* 2025, Rexer-Huber *et al.* 2025
Apparently occupied	The number of nest sites where birds appear to be breeding, e.g., because they are sitting on a nest or occupying a burrow, but the contents thereof have not been confirmed, in the year[A] the count	Baker *et al.* 2023, Stevens *et al.* 2024

nest sites (AOS)	is conducted. Nest site is defined as the location where the adult bird sits on the egg or chick.	
Birds on ground (BOG)	The number of birds on the ground present in the year[A] the count is conducted, regardless of 1) whether they are, or appear to be, breeding, 2) whether they are, or appear to be, associated with a nest site, and 3) regardless of their age (i.e., this metric can include immatures, prebreeders, non-breeders, and breeders).	Hayes *et al.* 2021
Nest sites (NS)	The number of nest sites (e.g., burrows) present in the year[A] the count is conducted regardless of their contents. This metric may not be suitable for mobile incubators (Awkerman *et al.* 2005) and care must be applied for burrow-nesting species that can have multiple burrow entrances.	Parker & Rexer-Huber 2020, Bell *et al.* 2017
Chicks (C)	The number of chicks present in the year[A] the count is conducted. Chicks are defined as young birds on the nest site between hatching and fledging age.	Frost 2017, 2024,

[A] Year refers to a biologically relevant year, which varies between species depending on the species' phenology. As per ACAP convention, the year reflects when chicks fledge.

Table 2. Updated, standardised definitions of count methods for albatrosses and petrels.

Method (acronym)	Definition	Example
Ground count (GC)	Count of the target metric from the ground, including the use of aids such as tally counters, GPSs, burrowscopes, or RFID readers.	Waugh *et al.* 2020, Mischler *et al.* 2025,
Vantage point count (VPC)	Count of the target metric from a ground-based vantage point, including the use of aids such as binoculars or photography and orthomosaics.	Cuthbert & Sommer 2004, Stevens *et al.* 2024, Mischler *et al.* 2025
Aerial count – using fixed-wing (AC^{fw})	Count of the target metric on the ground from a fixed-wing plane including the use of aids such a photography and orthomosaics.	Fisher 1966, Robertson *et al.* 2008, Frost 2017
Aerial count – using helicopter (AC^{heli})	Count of the target metric from a helicopter including the use of aids such a photography and orthomosaics.	Baker *et al.* 2023, Frost *et al.* 2025
Aerial count – using UAV (AC^{UAV})	Count of the target metric from an unmanned aerial vehicle (UAV; e.g., drone or aerial kite), usually using photography and orthomosaics.	Rexer-Huber *et al.* 2025, Virtue *et al.* 2025
Aerial count – using satellite imagery (AC^{sat})	Count of the target metric under consideration from satellite imagery.	Fretwell *et al.* 2017, Attard *et al.* 2025

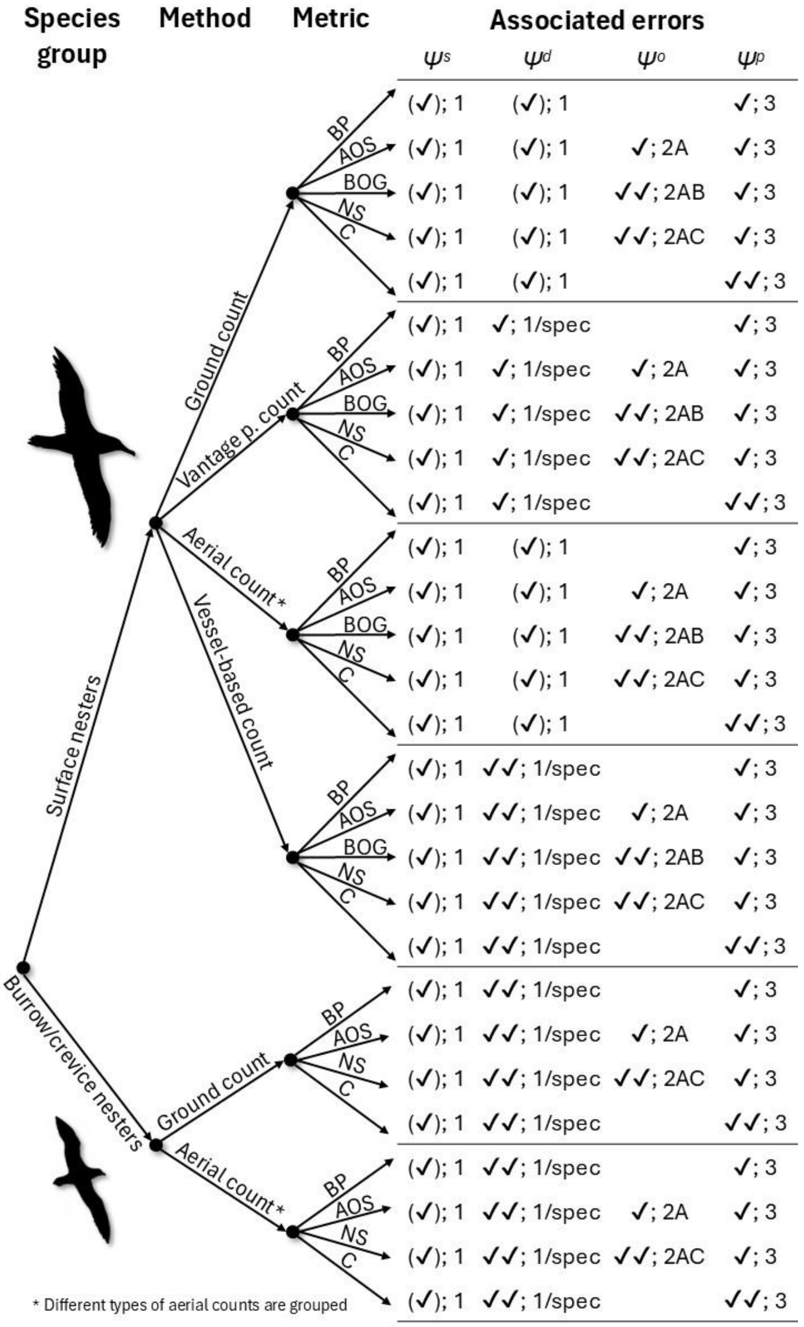
Vessel-based count (VBC)	Count of the target metric from a vessel at sea, including the use of aids such as binoculars or photography.	Poncet *et al.* 2006, Rexer-Huber *et al.* 2020

Table 3. Updated, standardised definitions of common [A] sources of error in counts of albatrosses and petrels. Overarching sources of error are indicated by Ψ, subcategories thereof are indicated by ψ. ↓ indicates an underestimate, ↑ indicates an overestimate.

Source of error (symbol)	**Definition**	**Impact on estimate**	**Example**
Sampling error (Ψ^{s})	The error associated with counting only a sample of the population and associated decisions (e.g., randomisation, representativeness, and coverage).	↓↑	Elliott *et al.* 2016
Detection error ($\Psi^{d}; \psi^{obs}$)	The error associated with the probability of not detecting the target metric, despite it being present, including obstruction error (ψ^{obs}) when the metric under consideration is being obstructed from view. When using some methods (e.g., AC^{sat}) false positives (objects incorrectly counted as birds) can also occur as well.	↓↑	Parker & Rexer-Huber 2020
Occupancy error (Ψ^{o}; ψ^{ABA}, ψ^{BOG} & ψ^{occ})	The error associated with the probability of counting the target metric as a different metric. This error category consists of different sub-categories (Fig. 1 & 2): - Counting AOS (e.g., birds appearing to breed, while not actually breeding, sometimes referred to as pretend-breeders or triers) as BP (ψ^{ABA}). Relevant to both surface and burrow/crevice nesters. Example: BP ~ Bin(AOS, ψ^{ABA}). - Counting loafers (birds not associated with a nest site) as BP or AOS (ψ^{BOG}). Relevant to surface nesters only. Example: AOS ~ Bin(BOG, ψ^{BOG}); BP ~ Bin(BOG, $\psi^{BOG} \cdot \psi^{ABA}$). - Counting NS (e.g., burrows) as BP or AOS while not actually being occupied (ψ^{occ}). Relevant to both surface and burrow/crevice nesters. Example: AOS ~ Bin(NS, ψ^{occ}); BP ~ Bin(NS, $\psi^{occ} \cdot \psi^{ABA}$). For burrow/crevice nesters, ψ^{occ} (and ψ^{ABA}) are often quantified when assessing the proportion of counted nest sites/burrows that contain the target species and target metric (often referred to as burrow occupancy). In certain cases, this error can include other species (e.g., because burrow contents were not checked; Fig. 1).	↑	Poncet *et al.* 2006, Robertson *et al.* 2008, Walker *et al.* 2020, Baker *et al.* 2023, Rexer-Huber *et al.* 2025
Phenology-related error ($\Psi^{p}; \psi^{lay}$ & ψ^{fail})	The error associated with the probability of not counting the target metric, because the nest failed prior to the count (ψ^{fail}) or because the egg had not been laid at the time of the survey (ψ^{lay}). Both sub-categories of phenology-related error can occur within the same count, if the count period spans a long enough time.	↓	Frederick *et al.* 2006, Poncet *et al.* 2006, Elliott *et al.* 2025

[A] Other sources of error raised included observer error (i.e., the error associated with different observers conducting counts; Parker & Rexer-Huber 2020) and interpretation error (i.e., the error associated with the potential incorrect retrospective interpretation of the count data). While these errors do occur, they appear less common, and when the four common sources of error are accounted, the associated uncertainty should capture the less common sources of error as well. However, care must be taken with area-based extrapolations, as these approaches can introduce different sources of error (e.g., 2D vs. 3D and homogeneous vs. heterogeneous extrapolation approaches).

ACAP IG Trends workshop 24 May outcome:
Minor adjustments to the definitions in Table 1-3 were agreed upon.



Species group	Method	Metric	Associated errors			
			ψ^s	ψ^d	ψ^o	ψ^p
Surface nesters	Ground count	BP	(✓); 1	(✓); 1		✓; 3
		AOS	(✓); 1	(✓); 1	✓; 2A	✓; 3
		BOG	(✓); 1	(✓); 1	✓✓; 2AB	✓; 3
		NS	(✓); 1	(✓); 1	✓✓; 2AC	✓; 3
		C	(✓); 1	(✓); 1		✓✓; 3
	Vantage p. count	BP	(✓); 1	✓; 1/spec		✓; 3
		AOS	(✓); 1	✓; 1/spec	✓; 2A	✓; 3
		BOG	(✓); 1	✓; 1/spec	✓✓; 2AB	✓; 3
		NS	(✓); 1	✓; 1/spec	✓✓; 2AC	✓; 3
		C	(✓); 1	✓; 1/spec		✓✓; 3
	Aerial count*	BP	(✓); 1	(✓); 1		✓; 3
		AOS	(✓); 1	(✓); 1	✓; 2A	✓; 3
		BOG	(✓); 1	(✓); 1	✓✓; 2AB	✓; 3
		NS	(✓); 1	(✓); 1	✓✓; 2AC	✓; 3
		C	(✓); 1	(✓); 1		✓✓; 3
	Vessel-based count	BP	(✓); 1	✓✓; 1/spec		✓; 3
		AOS	(✓); 1	✓✓; 1/spec	✓; 2A	✓; 3
		BOG	(✓); 1	✓✓; 1/spec	✓✓; 2AB	✓; 3
		NS	(✓); 1	✓✓; 1/spec	✓✓; 2AC	✓; 3
		C	(✓); 1	✓✓; 1/spec		✓✓; 3
Burrow/crevice nesters	Ground count	BP	(✓); 1	✓✓; 1/spec		✓; 3
		AOS	(✓); 1	✓✓; 1/spec	✓; 2A	✓; 3
		NS	(✓); 1	✓✓; 1/spec	✓✓; 2AC	✓; 3
		C	(✓); 1	✓✓; 1/spec		✓✓; 3
	Aerial count*	BP	(✓); 1	✓✓; 1/spec		✓; 3
		AOS	(✓); 1	✓✓; 1/spec	✓; 2A	✓; 3
		NS	(✓); 1	✓✓; 1/spec	✓✓; 2AC	✓; 3
		C	(✓); 1	✓✓; 1/spec		✓✓; 3

* Different types of aerial counts are grouped

Fig. 2. Decision tree illustrating the interaction of metric (Table 1), method (Table 2), and errors associated with counts of albatrosses and petrels (Table 3). BP: annual breeding pairs, AOS: apparently occupied sites, BOG: birds on ground, NS: nest sites, C: chicks (Fig. 1). Symbols indicate the importance of accounting for errors: (✓) = important, depending on

context; ✓ important; ✓✓ crucially important. Numbers following tick marks indicate the equations in section 2.2 that should be applied. spec indicates that specific, custom, context-dependent approaches may be necessary, see case studies for examples thereof. Silhouettes adapted from Fischer *et al.* (2023).

2.2. Standardised error corrections and incorporation of uncertainty

We applied a series of equations and models within a Bayesian hierarchical framework to standardise the disparate ACAP count data and prepare them for the standardised modelling of trends. Depending on how the target species was counted, through which metric the count was expressed, and following the decision tree in Fig. 2, we 1) incorporated baseline uncertainty and accounted for sampling error Ψ^{s}, 2) accounted for detection error Ψ^{d} where possible, 3) accounted for occupancy error Ψ^{o} when required, and 4) accounted for phenology-related error Ψ^{p}. This approach ensured that all adjusted counts were ultimately expressed in the standardised number of breeding pairs at the start of the breeding period while accounting for uncertainty. We completed step 1, 3, and 4 using a standardised approach consisting of several hierarchical models that were fit jointly to the data as required (see below). However, we could not develop a standardised methodology to account for Ψ^{d} (step 2), and as such, specific, custom, context-dependent approaches were included in our hierarchical framework where possible (see Supplementary Material 1 & 2). In our framework, we accounted for Ψ^{o} (step 3) and Ψ^{p} (step 4) separately but in parallel. This contrasts with some prevailing assumptions held on Ψ^{o} (e.g., AOS = BP, and ψ^{ABA} changes predictably over the breeding period; Poncet *et al.* 2006, Robertson *et al.* 2008, Baker *et al.* 2023) and we provide additional analyses in Supplementary Material 3 to justify our parallel approach. It should be noted that our standardised framework did not seek to overwrite any existing, already applied, efforts to account for errors (e.g., see Moore 2004, Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). Rather, our processing and standardisation steps are intended to enable the accounting for errors and incorporation of uncertainty where this has not yet been done, ensuring that abundance data are comparable across space, time and species, ultimately enabling estimates of trends. A glossary of all acronyms and symbols used in our framework can be found Table 4.

Table 4. Glossary of acronyms and symbols.

Acronym/symbol	Brief description (Section; Table; Figure)
Count metrics	
BP	Annual breeding pairs (2.1; Table 1; Figure 1, 3)
AOS	Apparently occupied sites (2.1; Table 1; Figure 1, 3)
BOG	Birds on ground (2.1; Table 1; Figure 1, 3)
NS	Nest sites (2.1; Table 1; Figure 1, 3)
C	Chicks (2.1; Table 1; Figure 1, 3)
Count methods	
GC	Ground count (2.1; Table 2)
VPC	Vantage point count (2.1; Table 2)
$\mathrm{AC}^{\mathrm{fw}}$/ $\mathrm{AC}^{\mathrm{heli}}$ / $\mathrm{AC}^{\mathrm{UAV}}$ /$\mathrm{AC}^{\mathrm{sat}}$	Aerial count from a fixed-wing plane / a helicopter / an unmanned aerial vehicle / a satellite (2.1; Table 2)
VBC	Vessel-based count (2.1; Table 2)
Errors	
Ψ^{s}	Sampling error (2.1; Table 3)
Ψ^{d}	Detection error (2.1; Table 3)
ψ^{obs}	Obstruction error, subcategory of Ψ^{d} (2.1; Table 3)
Ψ^{o}	Occupancy error (2.1; Table 3; Figure 3)
ψ^{ABA}	Counting AOS as BP, subcategory of Ψ^{o} (2.1; Table 3; Fig. 1 & 4)
ψ^{BOG}	Counting BOG as AOS or BP, subcategory of Ψ^{o} (2.1; Table 3; Fig. 1 & 4)
ψ^{occ}	Counting NS as AOS or BP, subcategory of Ψ^{o} (2.1; Table 3; Fig. 1 & 4)
Ψ^{p}	Phenology-related error (2.1; Table 3)
ψ^{fail}	Error caused by nests failing prior to count, subcategory of Ψ^{p} (2.1; Table 3; Fig. 5)
ψ^{lay}	Error caused by eggs being laid after count, subcategory of Ψ^{p} (2.1; Table 3)
Data processing	
$\rho_{t,i}$	Reported abundance of sample regardless of metric (2.2.1)
$\xi_{t,i}$	Proportion of the breeding site counted (2.2.1)
$P_{t,i}$	Extrapolated abundance of total breeding site regardless of metric (2.2.1)
$n_{t,i}$	Extrapolated abundance of total breeding site with standardised uncertainty regardless of metric (2.2.1; Table 4; Fig. 3)
$S(a_{t,i})$	Scaling factor associated with the ACAP accuracy score (2.2.1; Table 4; Fig. 3)
$\mathrm{bp}_{t,i}$	Number of nests for which egg/chick presence has been confirmed in (sub-)count (2.2.3)
$\mathrm{aos}_{t,i}$	Total number of AOS regardless of nest contents in (sub-)count (2.2.3)
$\mathrm{bog}_{t,i}$	Number of birds on the ground in (sub-)count (2.2.3)
$\mathrm{m}_{t,i}$	Number of apparent partners next to AOS (2.2.3)
$\mathrm{ns}_{t,i}$	Number of nest sites checked (2.2.3)
$\mathrm{n}^{\mathrm{bs}}_{t,i(,k)}$	Number of nests monitored for breeding success that were successful (2.2.4)
$\mathrm{o}^{\mathrm{bs}}_{t,i(,k)}$	Nests monitored for breeding success (2.2.4)
$f_{t,i(,k)}$	Annual breeding success (fecundity) (2.2.4; Figure 4)
$\theta_{t,i(,k)}$	Count timing in relation to the species' phenology (2.2.4; Fig. 5)
$\Theta_{i(,k)}$	Duration of total breeding period (or phenological stage *k*) (2.2.4; Fig. 5)
$\upsilon_{t,i(,k)}$	Start time of count in relation to the species' phenology (2.2.4)
$\omega_{t,i(,k)}$	End time of count in relation to the species' phenology (2.2.4)
$y_{i,t}$	Standardised abundance estimate, as derived from the data processing steps, expressed in BP at the start of the incubation period (2.2.5)
$\mathrm{M}_{i,t}$	Total number of mature individuals (2.3)
$\mathrm{u}_{i,t}$	Mature individuals not breeding (2.3)
$p^{b}_{i(,t)}$	Breeding probability (2.3)
$s_{i(,t)}$	Sex ratio (2.3)
Trend model	
$N_{t(,i)}$	Abundance of breeding site *i* in year *t* as derived through trend model Eq. 5 (2.4)
$\lambda_{t,i}$	Annual growth rate (2.4)
q	Mean log growth rate (2.4)
$x_{t,i}$	Logarithm of $N_{t,i}$ (2.4)

Acronym/symbol	Brief description (Section; Table; Figure)
$r_{t,i}$	Logarithm of $\lambda_{t,i}$ (2.4)
$\zeta_{i,t}$	Standardised component variance, as derived from the data processing (2.4)
Conservation assessments	
CR	Critically Endangered (2.5.1)
EN	Endangered (2.5.1)
VU	Vulnerable (2.5.1)
NT	Near Threatened (2.5.1)
LC	Least Concern (2.5.1)
GL	Generation length (2.5.1)
R_t	Fractional overall population change over predefined time period of 3 GL
T	Predetermined reference time interval based on GL
δ	Number of years within T during which abundance estimates are available
W	Time window (e.g., 2005-…)
Indexing	
t	Year: 1, 2, 3, … (the calendar year during which fledging occurs)
i	Breeding site: 1, 2, 3, …
k	Phenological stage: incubation, guard, post-guard

2.2.1. Incorporating baseline uncertainty and accounting for sampling error

Many counts of albatrosses and petrels (and seabirds in general) are recorded without explicit, consistent or comparable expressions of uncertainty. This is particularly common in abundance data, as absolute censuses of all breeding sites are extremely rare (Parker & Rexer-Huber 2020, Wolfaardt & Phillips 2020). However, communicating uncertainty associated with counts of these species is challenging due to the idiosyncrasies associated with the data. Most counts of ACAP species consist of a single figure (e.g., Sagar & Stahl 2005, Mackley *et al.* 2025), though some estimates are paired with efforts to express uncertainty mathematically (e.g., Waugh *et al.* 2020, Frost *et al.* 2025). Even if uncertainty is expressed, some approaches represent only a subset of the uncertainty in the count methodology (e.g., a single assumption in the estimation process), rather than the full uncertainty surrounding the actual count (e.g., Moore *et al.* 2012, Elliott *et al.* 2025). Moreover, even if uncertainty is quantified, the resulting estimates may not be comparable across years (e.g., Elliott *et al.* 2016 vs. Elliott *et al.* 2025). Finally, many counts of albatrosses and petrels are conducted at smaller study or index sites, which are assumed to be representative of the larger breeding site population (e.g., Moore *et al.* 2012, Mischler *et al.* 2025), yet there is no standardised mechanism to incorporate information on what proportion of the overall population is counted (e.g., 1% vs. 10%), and how this proportion affects uncertainty surrounding the counts, i.e., sampling error Ψ^{s}.

Here, we formulate a standardised approach to incorporating baseline uncertainty surrounding counts of albatrosses and petrels in a consistent and comparable manner to overcome the challenges of the disparate data subject to varying idiosyncrasies while simultaneously accounting for Ψ^s. Specifically, we applied Eq. 1 to all counts, which leveraged 1) the accuracy scores within the ACAP database (ACAP 2025b) representing an existing semi-standardised mechanism that qualitatively reflects prior beliefs in uncertainty, and 2) knowledge on the proportion of a breeding site that is counted. This approach assumes that the proportion of the overall breeding site counted does not influence the accuracy score of the count. This assumption is appropriate, as high accuracy counts of small proportions are common in the ACAP database.

1.
$$P_{t,i} = (\frac{\rho_{t,i}}{\xi_{t,i}})$$
$$n_{t,i} \sim \mathrm{N}(P_{t,i}, \tau^{n}_{t,i})$$
$$\tau^{n}_{t,i} = \frac{1}{S(a_{t,i}) \cdot P^{2}_{t,i} + \frac{(1 - \xi_{t,i})}{\xi^{2}_{t,i}} \cdot P_{t,i}}$$

in which $P_{t,i}$ is the extrapolated abundance in year *t* at site *i* (irrespective of the metric the count is expressed in, see section 2.2.3), $\rho_{t,i}$ is the reported abundance, $\xi_{t,i}$ is the known proportion of the breeding site that has been counted, $n_{t,i}$ is the extrapolated abundance, truncated below $P_{t,i}$, with standardised uncertainty expressed through precision $\tau^{n}_{t,i}$, and $S(a_{t,i})$ is a scaling factor that was identified iteratively to define asymptotes matching prior beliefs associated with the ACAP accuracy scores (section 2.1, Table 5, Fig. 3). For example, for a count of 5,000 BP of a surface nester categorised as "High Accuracy", our expert elicitation identified that the true value of abundance should fall within 10% of the reported value, and thus, that uncertainty bounds should equate to 4,500-5,500 based on the held prior beliefs, provided a large enough proportion of the population was counted. We calculated the values of $S(a)$ reported in Table 5 iteratively in order to meet the identified prior beliefs assuming 100% of a population was counted. For a graphical representation, see Fig. 3.

Our approach leverages an asymptotic function that generates uncertainty around the count based on the pre-assigned accuracy category and proportion of the breeding site counted. This approach thus enables the generation of uncertainty in a standardised fashion, while adhering to the principle that counting a larger proportion of a breeding site reduces uncertainty (e.g., Elliott *et al.* 2016) - up to a certain extent - while retaining the prior information on count accuracy contained within the ACAP database. In most cases, we treated $\xi_{t,i}$ as fixed. However, in certain cases the fraction of the overall population was uncertain (e.g., Imber 1987, Bell *et al.* 2022), and we therefore constructed informed priors to incorporate the additional uncertainty surrounding $\xi_{t,i}$. Additionally, if part-sites of breeding sites exhibited divergent trends, $\xi_{t,i}$ was interpolated linearly where possible to capture changing proportions over time. We applied Eq. 1 to all counts to incorporate standardised baseline uncertainty, while simultaneously accounting for Ψ^s. As uncertainty was expressed as a function of ξ, we only included counts in which $\xi_{t,i} > 0.01$ (i.e., counts that covered more than 1% of the breeding site's population). This approach was also perceived to address other, less prevalent sources of error, e.g., among-observer error (Parker & Rexer-Huber 2020).

Table 5. Scaling factors used in Eq. 1 to translate qualitative uncertainty into numerical uncertainty using the accuracy score categories within the ACAP database.

	Surface nesters		**Burrow/crevice nesters**	
Accuracy score	% within stated value (expert elicited)	*S(a)*	% within stated value (expert elicited)	*S(a)*
High	10%	0.0026	16%	0.0067
Medium	33%	0.0283	46%	0.0551
Low	67%	0.1169	88%	0.2016

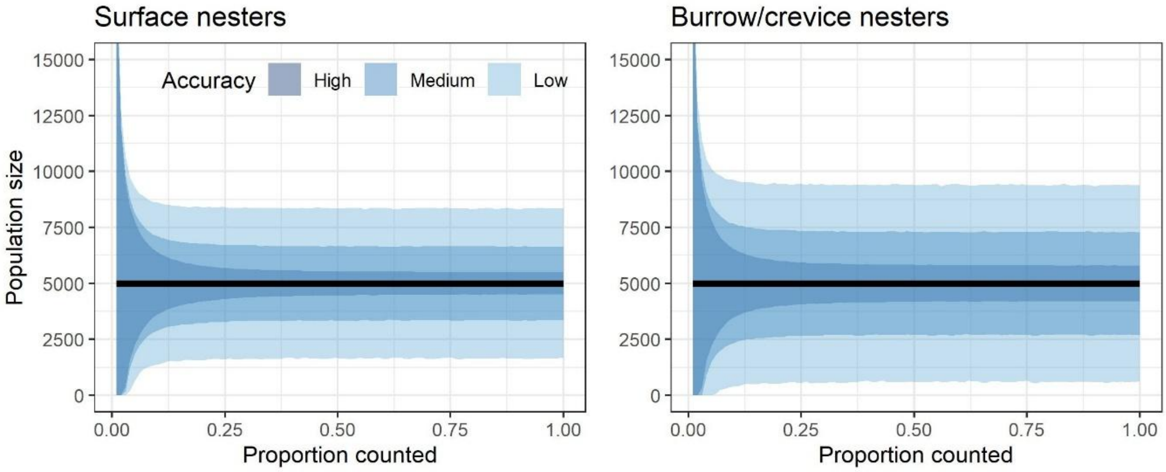


Fig. 3. Graphical representation of Eq. 1 applied to hypothetical counts of a surface and a burrow/crevice nester breeding site with a population size of 5,000 BP (horizontal black line). Shaded areas represent the uncertainty (represented by 95% CIs) as produced by Eq. 1 under varying ACAP accuracy scores ($S(a)$) and proportions of the population counted (ξ

2.2.2. Accounting for detection error

Imperfect detection of target metrics is an inherent, ubiquitous challenge in ecological monitoring, and is common for ACAP species, and thus, detection error Ψ^{d} is a common source of error associated with counts of albatrosses and petrels (Parker & Rexer-Huber 2020, Wolfaardt & Phillips 2020). The target metric can be obscured from view during counts for various reasons, which are species-, site- (or even within-site-), and context-dependent; thus, Ψ^{d} can be subject to considerable variation among counts. However, in general, Ψ^{d} is a more important consideration for the more cryptic burrow/crevice nesters (e.g., burrowing petrels; Parker & Rexer-Huber 2020) than surface nesters (e.g., albatrosses; Wolfaardt & Phillips 2020) (Fig. 2), yet cliff-nesting albatrosses can also be subject to considerable detection challenges (Ryan *et al.* 2009). Common approaches to address Ψ^{d} for individual counts include distance sampling and repeat surveys in a mark-resight framework, comparing multiple methods (e.g. ground counts of an aerial-count block), or by perpendicular validation transects (Buckland *et al.* 2001, Walker & Elliott 2005, Frederick *et al.* 2006, Parker & Rexer-Huber 2020).

However, to date, no approaches exist to account for Ψ^{d} retrospectively that effectively allows for improved comparability across counts, space, time, and species, and we were equally unable to identify an explicit standardised step in our framework to do this. It may be possible to infer Ψ^{d} for counts retrospectively when Ψ^{d} has been estimated for the same breeding site and species in previous/subsequent counts. Yet even if comparable information on Ψ^{d} exists, care needs to be taken to ensure that the area of extrapolation remains comparable (Parker & Rexer-Huber 2020). However, if no prior data exists, quantifying Ψ^{d} may prove highly challenging due to the site-, species-, observer- and context-specific nature of this source of error (Parker & Rexer-Huber 2020, Wolfaardt & Phillips 2020). Expert elicitations may provide some general solutions to estimate Ψ^{d} retrospectively (Fischer *et al.* 2020) including for a specific form of Ψ^{d}, obstruction error ψ^{obs}, but such approaches are time consuming and remain case- and context-specific.

We here follow the decision tree in Fig. 2 and we assume that the uncertainty incorporated through Eq. 1 is sufficient to account for Ψ^{d} for situations and species that inherently have a high detection probability (e.g., surface-breeding *Diomedea* albatrosses for which ground-counting Ψ^{d} has been shown to be >0.95; e.g., Walker & Elliott 2005, Rexer-Huber *et al.* 2024). In other cases (i.e., counts that are not ground or aerial counts of surface breeders), we either used averages of previously reported Ψ^{d} (e.g., Buller's Albatross VPCs) or count-specific modified Delphi expert elicitations (e.g., White-capped Albatross VBCs; Hemming *et al.* 2018, Fischer *et al.* 2020; Supplementary Material 1). Through this approach, we still attempted to account for Ψ^d where it was pressing to do so, while not overly inflating uncertainty for counts in contexts when detection is known to be high.

2.2.3. Accounting for occupancy error

Another common source of error associated with counts of albatrosses and petrels is occupancy error Ψ^o. The ACAP convention is to express abundance in annual breeding pairs (BP, i.e., the number of pairs successfully laying an egg in the year the count is conducted; Table 2; Wolfaardt & Phillips 2020). However, abundance of ACAP species can be expressed in range of other metrics (Fig. 1). For example, surface-nesters, like albatrosses, can be present at the breeding site, and occupy nest sites, even when not actively breeding, and such individuals typically consist of a virtually indistinguishable mix of immatures, pre-breeders, non-breeders, and failed breeders (e.g., Stahl & Sagar 2006, Wolfaardt & Phillips 2020). This phenomenon is subject to local, diurnal, intra- and interannual variation (e.g., Poncet *et al.* 2006, Robertson *et al.* 2008, Walker *et al.* 2020, Baker *et al.* 2023). Consequently, counts of surface breeders, particularly from aerial, vantage point, or vessel-based counts, are often expressed in BOG or AOS, as it is not possible to confirm nest contents and therefore obtain a direct count in BP (Wolfaardt & Phillips 2020). Similar challenges occur for burrow/crevice nesters, like burrowing petrels (Parker & Rexer-Huber 2020). While these species cannot be counted as BOG, counts of nest sites with (AOS or BP) or without (NS) confirmation of occupancy by the target species are common (Fig. 1). Even when occupancy is confirmed, verification of the presence of an egg or chick is still required to express counts in BP as, burrow/crevice nesters can also occupy nest sites without actively breeding (Parker & Rexer-Huber 2020, Waugh *et al.* 2020). Thus, for both surface and burrow/crevice nesters, adjustments to any counts expressed in metrics other than BP are required to obtain uniform, directly comparable, abundance data.

To overcome the vexing challenge of Ψ^o, we leveraged the logical hierarchy of abundance metrics together with information contained in sub-counts associated with abundance counts to enable conversions of AOS, BOG, and NS into BP. The different abundance metrics are inter-related and follow a hierarchy (e.g., not all AOS equal BP, but all BP equal AOS; Fig. 4). This hierarchy logically guides conversions of other metrics into BP, provided information exists to enable these conversions. This information is typically collected in sub-counts associated with overall abundance counts. Examples of such sub-counts are ground transects of surface nesters concurrent to wider aerial counts to obtain a ratio of BP:AOS (e.g., Elliot *et al.* 2025, Frost *et al.* 2025, Rexer-Huber *et al.* 2025), close-up photographs during aerial counts to obtain a ratio of BOG:AOS (Baker *et al.* 2023), or burrow scoping transects/plots of burrow/crevice nesters during wider ground counts to obtain ratios of BP:NS or AOS:NS (e.g., Rexer-Huber *et al.* 2023).

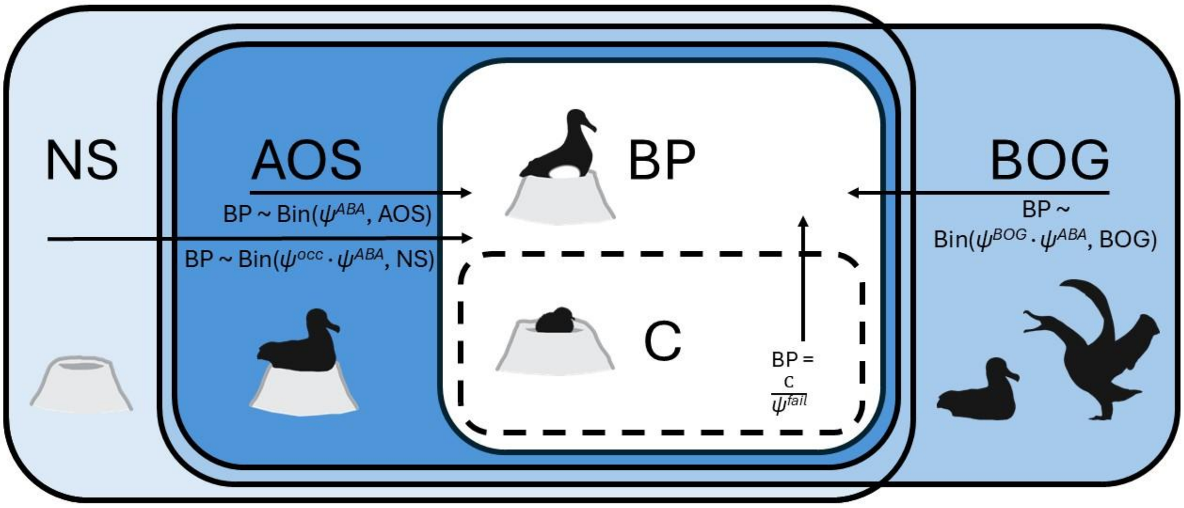


Fig. 4. Hierarchy of abundance metrics for ACAP species and how these can be converted to the standard, BP. BP: annual breeding pairs, AOS: apparently occupied sites, BOG: birds on ground, NS: nest sites, C: chicks. Equations illustrate Eq. 2ABC (and Eq. 3 for converting C into BP). Note, BOG is relevant to surface nesters only (Fig. 1). Silhouette credit: J. Chambon.

Here, we developed a hierarchical framework of binomial generalised linear mixed-effect models (GLMMs; with logit-link functions) to leverage the information contained in such sub-counts to standardize accounting for Ψ^o. Specifically, we specified 1) a GLMM to adjust AOS to BP through estimates of ψ^{ABA} (Eq. 2A), which is relevant to both surface and burrow/crevice nesters, 2) a GLMM to adjust BOG to AOS through estimates of ψ^{BOG} (Eq. 2B), which is relevant to surface nesters only, and 3) a GLMM to adjust NS to AOS through estimates of ψ^{occ} (Eq. 2C), which is relevant to both surface and burrow/crevice nesters (Table 3, Fig. 1 & 4). We used these GLMMs in combination as required (e.g., to adjust NS to BP we combined Eq. 2A and 2C). Ψ^o is subject to breeding site, local, diurnal, intra- and interannual variation in an apparently not (fully) predictable manner (e.g., Supplementary Material 3; Poncet *et al.* 2006, Robertson *et al.* 2008, Stahl & Sagar 2006, Walker *et al.* 2020, Baker *et al.* 2023, Elliott *et al.* 2025, Rexer-Huber *et al.* 2025, Thompson 2025), and thus we simply indexed Eq. 2A-C per year *t* and breeding site *i* and modelled each conversion using random effects as following:

2A.
$$\mathrm{bp}_{t,i} \sim \mathrm{Bin}\left(\mathrm{aos}_{t,i},\ \psi_{t,i}^{ABA}\right)$$
$$\mathrm{logit}\left(\psi_{t,i}^{ABA}\right) = \alpha_i^{ABA} + \varepsilon_{t,i}^{ABA}$$
$$\varepsilon_{t,i}^{ABA} \sim \mathrm{N}\left(0, \sigma_{t,i}^{ABA^{-2}}\right)$$

in which $\mathrm{bp}_{t,i}$ is the number of nests in which the presence of an egg or chick has been confirmed in (sub-) count *c* (e.g., during ground transects; Frost *et al.* 2025), $\mathrm{aos}_{t,i}$ is the number of apparently occupied sites checked for the presence of an egg or chick in the same (sub-) count, $\psi_{t,i}^{ABA}$ is the count-specific adjustment for AOS to BP, α_i^{ABA} is the equation intercept, and $\varepsilon_{t,i}^{ABA}$ is a count-specific random effect with precision $\sigma_{t,i}^{ABA^{-2}}$ to be estimated from the data.

2B.
$$\mathrm{aos}_{t,i} \sim \mathrm{Bin}\left(\mathrm{bog}_{t,i} - \mathrm{m}_{t,i},\ \psi_{t,i}^{BOG}\right)$$
$$\mathrm{logit}\left(\psi_{t,i}^{BOG}\right) = \alpha_i^{BOG} + \varepsilon_{t,i}^{BOG}$$
$$\varepsilon_{t,i}^{BOG} \sim \mathrm{N}\left(0, \sigma_{t,i}^{BOG^{-2}}\right)$$

in which $\mathrm{aos}_{t,i}$ is the number of apparently occupied sites in (sub-) count *c* (e.g., in close-up images; Baker *et al.* 2023), $\mathrm{bog}_{t,i}$ is the number of all birds on the ground in the same (sub-) count, $\mathrm{m}_{t,i}$ is the number of apparent partners to apparently occupied sites in the same sub-count, $\psi_{t,i}^{BOG}$ is the count-specific adjustment for BOG to AOS, α_i^{BOG} is the equation intercept, and $\varepsilon_{t,i}^{BOG}$ is a count-specific random effect with precision $\sigma_{t,i}^{BOG^{-2}}$ to be estimated from the data. Eq. 2B can be combined with Eq. 2A to adjust BOG to BP directly (e.g., Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). Note, birds of unknown status should be excluded from Eq. 2B (e.g., Baker *et al.* 2023, Frost *et al.* 2025).

2C.
$$\mathrm{aos}_{t,i} \sim \mathrm{Bin}\left(\mathrm{ns}_{t,i},\ \psi_{t,i}^{occ}\right)$$
$$\mathrm{logit}\left(\psi_{t,i}^{occ}\right) = \alpha_i^{occ} + \varepsilon_{t,i}^{occ}$$
$$\varepsilon_{t,i}^{occ} \sim \mathrm{N}\left(0, \sigma_{t,i}^{occ^{-2}}\right)$$

in which $\mathrm{aos}_{t,i}$ is the number of apparently occupied sites in (sub-) count *c* (e.g., burrow scoping transects), $\mathrm{ns}_{t,i}$ is the number of nest sites in the same (sub-)count, $\psi_{t,i}^{occ}$ is the count specific adjustment for NS to AOS, α_{i}^{occ} is the equation intercept, and $\varepsilon_{t,i}^{occ}$ is a count-specific random effect with precision $\sigma_{t,i}^{occ^{-2}}$. Eq. 2C can be fused with Eq. 2A to adjust NS directly to BP (e.g., Rexer-Huber *et al.* 2023). We used vague priors for all intercepts ($\alpha \sim$ N(0,0.001), where 0.001 refers to precision) and standard deviations ($\sigma_{t,i} \sim$ U(0,3)). For breeding sites where the sub-count data covered ≤25% of the abundance timeseries, we removed $\varepsilon_{t,i}$, as exploratory analyses based on such limited data and high incidence of imputing became uninformative (e.g., $\psi_{t,i}^{ABA}$ ranged 0-1 when $\varepsilon_{t,i}^{ABA}$ was included).

Our approach to address Ψ^{o} overcomes shortfalls of using simple ratios derived from sub-counts, which are often based on small sample sizes and do not acknowledge uncertainty explicitly (e.g., Frost *et al.* 2025), and which have been used to correct abundance estimates in other years (e.g., see Walker *et al.* 2020 vs. Baker *et al.* 2023). Instead, our approach incorporates uncertainty surrounding Ψ^{o} (and therefore ψ_{c}^{ABA}, ψ_{c}^{BOG}, and ψ_{c}^{occ}), allows for inference of Ψ^{o} for abundance counts for which no data on Ψ^{o} was collected while increasing uncertainty, and vice-versa, enables for incorporation of information from years during which sub-counts did take place but no abundance counts were conducted (rarer but possible).

2.2.4. Accounting for phenology-related error

Albatrosses and petrels generally breed in remote locations that are difficult to reach, and this creates challenges to consistently conduct counts that coincide with the target species' ideal count time (i.e., early incubation; Wolfaardt & Phillips 2020). As the timing of counts often varies across years, this introduces another common source of error: phenology-related error Ψ^{p}. Most importantly, as nests fail over time from incubation onwards, the number of BP, AOS, NS, or C that can be counted is reduced relatively predictably: ψ^{fail} (e.g., Frederick *et al.* 2006, Rexer-Huber *et al.* 2021, Thompson 2025). We accounted for ψ^{fail} in a standardised manner by leveraging the principles of daily nest survival models (e.g., Fischer *et al.* 2021, Rexer-Huber *et al.* 2021, Thompson 2025), together with any data that was available on nests monitored to derive annual breeding success of the target species (i.e., the probability that a laid egg turned into a fledged chick), and insights into the timing of each specific count in relation to the target species' phenology. Note, nests monitored for breeding success usually consist of a smaller sample of overall abundance counts, so we necessarily assumed that the observed breeding success data was representative of the overall breeding site. Specifically, we applied the following binomial GLMM framework to account for ψ^{fail} for each breeding site *i* for each year *t* a count was conducted:

3A.
$$
\begin{aligned}
&\mathrm{n}_{t,i}^{\mathrm{bs}} \sim \mathrm{Bin}\left(\mathrm{o}_{t,i}^{\mathrm{bs}}, f_{t,i}\right)\\
&\mathrm{logit}\left(f_{t,i}\right) = \alpha_i^f + \varepsilon_{t,i}^f\\
&\varepsilon_{t,i}^f \sim \mathrm{N}\left(0, {\sigma_{t,i}^f}^{-2}\right)\\
&\Psi_{t,i}^{fail} = {f_{t,i}}^{\frac{\theta_{t,i}}{\Theta_i}}\\
&\theta_{t,i} \sim \mathrm{U}\left(|\upsilon_{t,i} - 1|, \omega_{t,i} + 1\right)
\end{aligned}
$$

in which $\mathrm{n}_{t,i}^{\mathrm{bs}}$ is number of nests monitored for breeding success that were successful (i.e., produced a fledgling), $\mathrm{o}_{t,i}^{\mathrm{bs}}$ is the total number of nests monitored for breeding success, $f_{t,i}$ is the breeding success (fecundity) estimate, α_i^f is the equation intercept, $\varepsilon_{t,i}^f$ is an annual random effect on breeding success with precision ${\sigma_{t,i}^f}^{-2}$ estimated from the data, Θ_i is the total breeding period (in weeks, rounded to the nearest integer, values for which were sourced from the literature), and $\theta_{t,i}$ is the timing the count took place in relation to the mean lay date of the breeding site of the target species (in weeks, rounded to the nearest non-negative integer), which was uniformly distributed between start time $\upsilon_{t,i}$ and end time $\omega_{t,i}$ of the count. We incorporated additional variation by subtracting 1 week of $\upsilon_{t,i}$ and adding one week to $\omega_{t,i}$, to acknowledge that species' breeding phenology can fluctuate from year to year (e.g., Lewis *et al.* 2012). We chose to express timings in weeks instead of days as to not portray an inflated level of precision. We used vague priors (α_i^f~ N[0,0.001] and $\sigma_{t,i}^f$ ~ U[0,3]) unless otherwise stated (see section 2.7). In short, through Eq. 3 we first estimated breeding success *f*, which we then used within a hazard function to estimate the likely nest failure that had occurred at the time the count took place (Fig. 5).

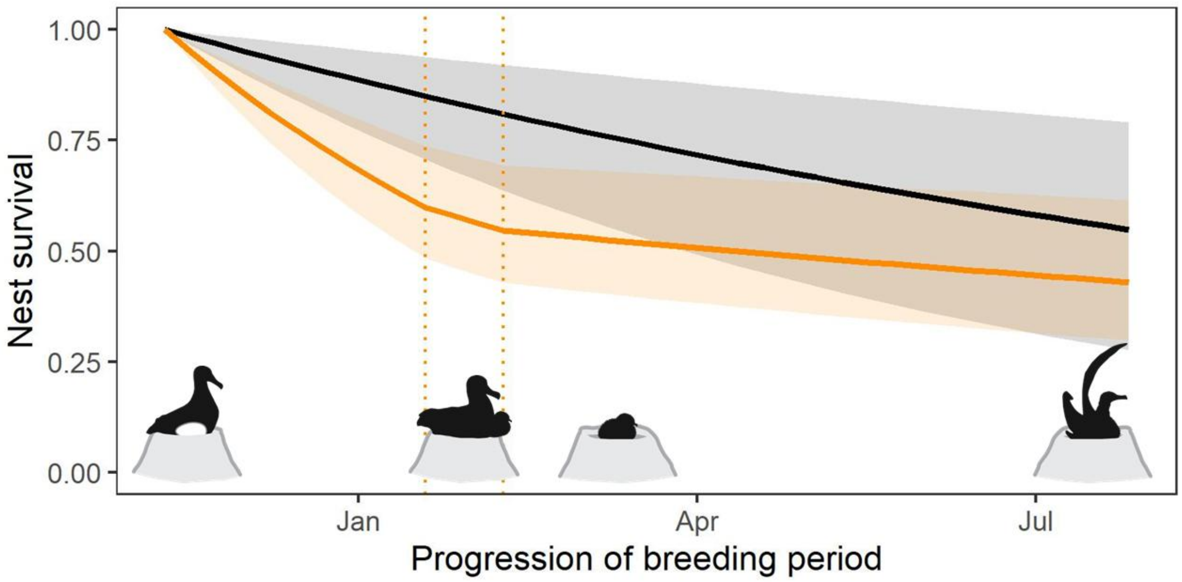


Fig. 5. Graphical representation of Eq. 3A (black) and Eq. 3B (orange). Dotted lines indicate changes between phenological stages with differing f_k for Eq. 3B. Silhouette Credit: J. Chambon.

Equation 3A relies on the assumption of an exponential hazard function, meaning that the rate of breeding failure is considered a consistent function of time (Fig. 5). In many cases, this assumption may have been violated (e.g., due to increased failures during adverse weather events or during specific phenological stages; Cleeland *et al.* 2020, Rexer-Huber *et al.* 2021). Occasionally, sufficient monitoring data exists to obtain survival estimates for individual phenological stages (i.e., incubation, guard, or post-guard respectively; Rexer-Huber *et al.* 2021, Thompson 2025). When such data exists, Eq. 3A can be adapted to estimate $\psi^{fail}_{t,i}$ as a product of the different stage-specific survival estimates $f_{k,j}$, enabling the incorporation of differing rates of breeding failure for each phenological stage (Fig. 5; Thompson 2025) as:

3B.

$$\mathrm{n}^{\mathrm{bs}}_{t,i,k} \sim \mathrm{Bin}\left(\mathrm{o}^{\mathrm{bs}}_{t,i,k}, f_{t,i,k}\right)$$

$$\mathrm{logit}\left(f_{t,i,k}\right) = \alpha^{f}_{i,k} + \varepsilon^{f}_{t,i,k}$$

$$\varepsilon^{f}_{t,i,k} \sim \mathrm{N}\left(0, {\sigma^{f}_{t,i,k}}^{-2}\right)$$

$$\psi^{fail}_{t,i,k} = f_{t,i,k}^{\frac{\theta_{t,i,k}}{\Theta_{i,k}}}$$

$$\theta_{t,i,k} \sim \mathrm{U}\left(|\upsilon_{t,i,k} - 1|, \omega_{t,i,k} + 1\right)$$

$$\psi^{fail}_{t,i} = \prod_{k} \psi^{fail}_{t,i,k}$$

Our approach to address Ψ^{p} thus leverages existing data on species' breeding success and knowledge on phenology. Additionally, the approach in both Eq. 3A and Eq. 3B allows for inference of breeding success for years during which no specific breeding success monitoring took place, but for which abundance counts are available, and vice-versa, allows for incorporation of information for years during which breeding success monitoring took place, but no abundance counts were conducted. We applied Eq. 3AB to all counts of albatross and petrel species, where possible, depending on whether the required data were available (see section 2.7, Table 7). It should be noted that when both active and failed nests are reported, Eq. 3 should be only fit to active nests (e.g., Sagar *et al.* 2014), as it would otherwise result in double-accounting for failed nests. Eq. 3 can be extended with additional fixed effects on f to incorporate additional nuance, e.g., known environmental impacts (e.g., Northern Royal Albatross; Supplementary Material 1). For breeding sites where breeding success data covered ≤25% of the abundance count timeseries, we removed $\varepsilon_{t,i}^{f}$, as preliminary analyses highlighted that estimates with such limited data and high incidence of imputing became uninformative (i.e., $f_{t,i}$ ranged 0-1 when $\varepsilon_{t,i}^{f}$ was included). In scenarios where abundance was expressed in both BP/AOS + failed nests (e.g., Bell *et al.* 2017, Bell 2023), we excluded counts of failed nests, as it is unclear for how long failed nests remain noticeable, and once Eq. 3 is applied to BP/AOS, all failures will be accounted for. Where no breeding success data existed, we used informative priors for $f_{t,i}$ either based on published figures, or closely related species where necessary. Species- and breeding-site-specific approaches are detailed in Table 4. For our rationale on why we applied Ψ^{o} (Eq. 2) and (Eq. 3) in parallel, see Supplementary Material 3.

Through Eq. 3AB we developed a standardised solution for counts that were conducted at varying times after mean laying, ψ^{fail}. However, a different form of Ψ^{p} can occur: the error that is associated when counts are conducted prior to all eggs having been laid, ψ^{lay} (Wolfaardt & Phillips 2020). In scenarios where counts are conducted around laying, but prior to mean lay dates, conversions using the count date and the mathematical distribution of the lay period of the species and breeding site can be applied to adjust the count (e.g., Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). As counts subject to ψ^{lay} are relatively rare, we adhere to the count-specific approach applied to such instances instead of developing a second standardised model for ψ^{lay}.

2.2.5. Hierarchical modelling to account for various sources of error

In our standardised abundance processing framework, various sources of error can be accounted for simultaneously. Specifically, after accounting for Ψ^s to incorporate baseline uncertainty, and obtaining an abundance estimate of the total breeding site in the original metric at the time of the count through Eq. 1, i.e., $n_{t,i}$, the outputs of Eq. 2 and 3, and, if applicable, appropriate estimates of Ψ^d can be combined to convert $n_{t,i}$ into the desired standardised measure of abundance for ACAP species, i.e., BP at the beginning of the breeding season, $y_{t,i}$, as following:

4. $$y_{t,i} = \frac{\frac{n_{t,i}}{\Psi^d_{t,i}} \cdot \Psi^o_{t,i}}{\Psi^p_{t,i}}$$

In which $\Psi^d_{t,i}$ is either considered 1.000 if it warrants to do so (i.e., if Eq. 1 has already provided sufficient uncertainty) or estimated through a custom approach (section 2.2.2), $\Psi^o_{t,i}$ is expressed as either $\psi^{ABA}_{t,i}$ (e.g., for a AC of AOS of albatrosses), $\psi^{BOG}_{t,i}$ (e.g., for a VPC of BOG of albatrosses), $\psi^{occ}_{t,i}$ (e.g., for a GC of NS of a burrow-nesting petrel), or a combination thereof (section 2.2.3), and $\Psi^p_{t,i}$ is usually expressed as $\psi^{fail}_{t,i}$ (section 2.2.4). While Eq. 4 here is formulated to sequentially account for Ψ^d, then for Ψ^o, and then for Ψ^p, conversions of $n_{t,i}$ to $y_{t,i}$ will produce the same result if Eq. 4 is formulated in a different order. The order as in Eq. 4 however is beneficial in scenarios where breeding success has traditionally been estimated through metrics other than BP (e.g., Robertson 1991, Scofield 2011) as this approach allows for first adjusting to BP, then calculate $f_{i,t}$, and thus $\psi^{fail}_{t,i}$, and then estimating $y_{t,i}$ if so required (e.g., Salvin's Albatross; Supplementary Material 1). Through this equation, the various equations accounting for errors are combined hierarchically within the same modelling framework. Ultimately, our approach enabled us to convert all counts of ACAP species into abundance data truly comparable across time, space, and species.

2.3. Conversions from annual breeding pairs to mature individuals

Abundance data on albatrosses and petrels is recorded in annual breeding pairs (ideally, or derivatives thereof; Table 1) as per ACAP convention, yet other global conservation processes require abundance to be expressed in mature individuals, including BirdLife International and the International Union for the Conservation of Nature (IUCN) Red List (IUCN 2012, BirdLife International 2025). Mature individuals are defined as individuals known, estimated or inferred to be capable of producing offspring, excluding juveniles, senescent individuals, suppressed individuals, or individuals in subpopulations whose densities are too low for fertilization to occur (BirdLife International 2025). Note, this definition does not allow for gradual exclusion of senescent individuals a function of age (e.g., Froy *et al.* 2013), but we did not seek to update this widely used definition. To elevate the utility of the abundance data collated in the ACAP database for wider global conservation processes, including Red List assessments (IUCN 2012) and fisheries risk assessments (e.g., Edwards *et al.* 2023, Anon. 2025), we also transformed the standardised annual breeding pair estimate, $y_{t,i}$, (see section 2.2) into an estimate of mature individuals as part of our overall framework.

Albatrosses and petrels have some of the most complex life histories of all bird species (Carneiro *et al.* 2020, Oppel *et al.* 2022, Richards *et al.* 2024), and as such, transforming annual breeding pairs into mature individuals requires careful consideration. For example, at least 32% of ACAP species are semi-biennial or biennial breeders once mature, and thus, transformations to mature individuals requires incorporation of breeding probability (the probability than an adult female lays an egg in a given year; Edwards *et al.* 2023). Furthermore, some species exhibit skewed sex-ratios due to sex-specific impacts on adult survival (Weimerskirch *et al.* 2005, Awkerman *et al.* 2007, Gianuca *et al.* 2019, Richard *et al.* 2024), which requires further consideration beyond breeding probabilities within these transformations. Building upon the approach by Edwards *et al.* (2023), we established two stochastic nodes to transform of annual breeding pairs (i.e., the metric in which $\mathrm{y}_{t,i}$ is expressed) to mature individuals M:

5A. $$\mathrm{u}_{t,i} \sim \mathrm{NegativeBinomial}\left(p_i^b, 2 \cdot \mathrm{y}_{t,i}\right)$$
$$\mathrm{M}_{t,i} = \mathrm{u}_{t,i} + 2 \cdot \mathrm{y}_{t,i}$$

5B. $$\mathrm{u}_{t,i} \sim \mathrm{NegativeBinomial}\left(p_i^b, \tfrac{1}{s_i} \cdot \mathrm{y}_{t,i}\right)$$
$$\mathrm{M}_{t,i} = \mathrm{u}_{t,i} + \frac{1}{s_i} \cdot \mathrm{y}_{t,i}$$

In which p_i^b refers to the annual breeding probability (which can be indexed on *t* if such data exist, see below), $\mathrm{u}_{t,i}$ is the unknown number of mature individuals that do not participate in breeding in year *t*, and s_i is the sex ratio (which also can be indexed on *t* if such data exist). *s* should be skewed towards the less abundant sex, so that if there is a deviation from a 50:50

ratio, $s < 0.5$. This approach is an improvement on Edwards *et al.* (2023) through the more explicit propagation of uncertainty in the negative binomial formulation, reflecting the challenge of translating BP to M (e.g., Oppel *et al.* 2022). To establish the performance of both equations under varying conditions of data quality and sex ratio, we completed a sensitivity testing exercise (Supplementary Material 4). Our sensitivity testing indicated that, if a population has an even sex ratio, the simpler conversion, Eq. 5A, provides adequate approximations for M, but if a skewed sex ratio is present, Eq. 5A results in an underestimate of M, and Eq. 5B provides a more adequate estimate. However, when using a fixed estimate of p^b instead of time-varying model estimate of p^b estimates, uncertainty increased substantially, to the extent that the improvements provided by incorporation of s were no longer beneficial. Consequently, we converted BP into M using Eq. 5A if no evidence exists for a skewed sex ratio, or using Eq. 5B, if such evidence exists and time-varying p^b estimates were available.

It should be noted that these estimates of M do not incorporate juveniles and pre-breeders (sub-adults) as per the definition of mature individuals (BirdLife International 2025). To incorporate those life-stage classes, further advanced modelling approaches, such as matrix population models (Waugh *et al.* 2015, Pardo *et al*, 2017) or integrated population models (e.g., Ventura *et al.* 2021, Oppel *et al.* 2022, Richard *et al.* 2024), are required, for which the necessary data is generally sparser or unavailable.

2.4. Standardised modelling of trends

Once the albatross and petrel abundance data was adjusted to be comparable across time, space, and species, we fit custom Bayesian state-space models to timeseries of each breeding site of each species to obtain standardised trend estimates. Specifically, we developed a bespoke Bayesian state-space trend model that we fit through the custom R package *ACAPT*, which allowed for the necessary flexibility and considerations when working with extremely long-lived species such as albatrosses and petrels (Bird *et al.* 2020). The model was a modified version of the generalised state-space decision support tool *JARA* (Winker *et al.* 2020, Sherley *et al.* 2020a,b). Fundamentally, both *JARA* and our own state-space model enable the modelling of trends of multiple abundance timeseries simultaneously even if those timeseries span different time periods, contain missing values, and have different scales through a hierarchical process that separates process variation from observation error (Winker *et al.* 2020). Our trend model was structured as follows:

6.
$$N_{(t+1),i} = \lambda_{t,i} \cdot N_{t,i}$$
$$\lambda_{t,i} \sim \mathrm{LogNormal}(q, \sigma_q^2)$$
$$N_t = \sum_i N_{t,i}$$

In which $N_{t,i}$ is the trend model estimated population size for breeding site *i* and year *t* (expressed as either BP or M), $\lambda_{t,i}$ is the annual growth rate, with mean log growth rate q that is constant and common to all breeding sites, constant variance σ_q^2, and total population size N_t. In log space, our trend model can be expressed in the form of a normal random walk:

$$x_{(t+1),i} = x_{t,i} + r_{t,i}$$
$$r_{t,i} \sim \mathrm{N}(q, \sigma_q^2)$$
$$y_{t,i} \sim \mathrm{LogNormal}(x_{t,i}, \sigma_y^2 + \zeta_{t,i}^2)$$

In which $x_{t,i}$ equals $\mathrm{Log}(N_{t,i})$, $r_{t,i}$ equals $\mathrm{Log}(\lambda_{t,i})$, $y_{t,i}$ is the standardised abundance estimate as derived through the steps detailed in sections 2.2-2.3 for breeding site *i* at year *t* with mean $x_{t,i}$ and variance consisting of the count-specific variance $\zeta_{t,i}^2$ as derived through the processes in sections 2.2-2.3, and additional variance common to all surveys σ_y^2. We used vague priors, specified via mean and precision ($q \sim \mathrm{N}(0, 0.01)$, $\sigma_q^{-2} \sim \mathrm{Gamma}(0.01, 0.01)$, and $\sigma_y^{-2} \sim \mathrm{Gamma}(0.01, 0.01)$) and the initial year in each timeseries was specified as $x_{t_0 i} \sim \mathrm{N}(\mu_{t_0 i}, \tau_{t_0 i}^{-1})$. We assumed that q as constant over time. In exploratory analyses, we varied our model specification by defining q either as an additional random walk, a smoothed cubic spline, or a linear model. These variations of our model became problematic with intermittent data (i.e., unrealistically high extents of uncertainty), a feature that is common of ACAP abundance data, and thus the simplest formulation in which q_t was treated as constant was considered the most appropriate. Our trend model up to this point is essentially identical to the trend model implemented through *JARA* (Winker *et al.* 2020, Sherley *et al.* 2020a,b), but our approach as implemented through *ACAPT* deviates slightly when estimating overall population change for the conservation status assessments due to a differing approach to projections (see section 2.5). Our trend model can be freely accessed and used through the custom R package *ACAPT* via https://github.com/SWotherspoon/ACAPT. For more information on *ACAPT*, see Supplementary Material 5 & 6.

We fit our trend model in Eq. 5 to all our standardised breeding site timeseries of the ten case studies both in terms of BP and M to estimate trends in both metrics and through summing the estimates of individual breeding sites, we provide estimates of current population size in BP and M per species regardless of the duration since the last count at a given breeding site. Furthermore, to evaluate overall trends on a genus level, we reran our trend model across the breeding sites of *Diomedea*, *Thalassarche*, and *Procellaria* species in our case studies jointly (M only). Finally, to provide overall insights into the fate of New Zealand's endemic large Procellariiform community, we reran our trend model across all breeding sites jointly (M only). Consequently, our flexible trend model provided the opportunity to obtain unprecedented insights into multi-decadal fates of ACAP species and has the potential to facilitate various conservation status assessments.

2.5 Conservation status assessments

2.5.1 IUCN Red List

The IUCN Red List is the world's most comprehensive conservation status assessment, providing key insights into species' extinction risk and through this, guide conservation prioritisation and policy (IUCN 2025). IUCN Red List categories are assigned through several objective criteria relevant to populations and their extinction risk (IUCN 2012). The categories Critically Endangered (CR; extremely high risk of extinction), Endangered (EN; very high risk), and Vulnerable (VU; high risk) jointly indicate that a species is threatened, in contrast with Near Threatened (NT; not threatened now, but is likely at risk in the near future) and Least Concern (LC; not threatened). A key assessment criterion, Criterion A, is population trend over three generation lengths (GL; the average age of parents of the current cohort; BirdLife International 2025). The other criteria evaluate geographic range in the form extent of occurrence and/or area of occupancy (Criterion B), small population size in decline (Criterion C), very small population (Criterion D), and direct estimates of extinction probability (Criterion E). Each criterion consists of several sub-criteria, which for Criterion A revolve around the causes of population reduction and period over which the reduction is estimated (Table 6, Supplementary Material 7).

All ACAP species have been assessed on the IUCN Red List, but most are due for re-assessment (BirdLife International 2025) and to address this shortcoming, we built upon *JARA* (Winker *et al.* 2020, Sherley *et al.* 2020a,b) and modelled population trends within our

R package *ACAPT* so that our results also provided insights into IUCN Red List re-assessment against Criterion A. For ACAP species, Criterion A generally is the most important criterion as 1) most species are showing ongoing, long-term declines, causes thereof have generally not yet ceased (Fischer *et al.* 2024, ACAP 2025b), 2) ACAP species range widely and thus geographic range is mostly relevant to single island endemics, 3) most populations exceed 10,000 (VU under Criterion C) and thus also 1,000 (VU under Criterion D) (ACAP 2025b), and 4) direct estimates of extinction probability only exist for a select few species (e.g., Genovart *et al.* 2016, Oppel *et al.* 2022, Richard *et al.* 2024). We focussed on assessing species against Criterion A4 (Table 6) as this 3 GL "moving window" is the most flexible of the four sub-criteria and the most relevant for species with extremely long GLs that usually are not fully covered by the available timeseries (e.g., 3 GL for Antipodean Albatross approximate a century; Table 7). However, for some species, very long timeseries that cover 3 GL do exist, enabling assessments against Criterion A2. While we here focussed on Criterion A as the most important Criterion for these species, it is not the sole criterion to assess species against for the IUCN Red List. However, evaluations against all IUCN Red List Criteria should be executed through formal assessments and are beyond the scope of the analyses presented here.

Table 6. Overview of IUCN Red List Criterion A, its four sub-criteria, and what percentage of population reduction results in what Red List category (IUCN 2012). GL = generation length, CR = Critically Endangered, EN = Endangered, VU = Vulnerable, NT = Near Threatened. IUCN (2012) does not provide an explicit threshold for NT, so we sourced thresholds for consideration of NT from JARA (Winker *et al.* 2020). For an overview of the other criteria, see Supplementary Material 5.

Sub-criteria	Brief description	CR	EN	VU	NT
A1	Population reduction estimated over 3 GL in the past, for which causes are clearly reversible and understood and have ceased.	≥90%	≥70%	≥50%	≥25%
A2	Population reduction estimated over 3 GL in the past, for which causes may not have ceased or may not be understood or may not be reversible.	≥80%	≥50%	≥30%	≥20%
A3	Population reduction projected to be met over 3 GL in the future.	≥80%	≥50%	≥30%	≥20%
A4	Estimated and/or projected population reduction over 3 GL, where the time period must include both the past and the future, and for which causes may not have ceased or may not be understood or may not be reversible.	≥80%	≥50%	≥30%	≥20%

We modelled population change in mature individuals M to facilitate the Criterion A assessments as following:

7. $$R_t = (\frac{M_t}{M_{t-\delta}})^{\frac{T}{\delta}}$$
$$t \in W$$

In which R_t is the (fractional) overall population change over a predetermined reference time interval *T* based on GL (e.g., 75 based on a GL of 25), δ is the number of years within *T* for which abundance estimates as derived through the trend model are available (e.g., 20 as per ACAP standard), and *W* is the time window (e.g., 2006-2081). We specified three different *W* to facilitate three different evaluation approaches: 1) the full timeseries available up to 3 GL and inferred forwards to obtain 3 GL (if 3 GL were fully covered, this equated to an assessment against Criterion A2), 2) the last GL and then inferred for the next 2 GL (Criterion A4), and 3) the last 20 years and then inferred forward up to 3 GL (Criterion A4). We included the latter assessment approach as 20 years is the ACAP standard period to provide statements on population change; ACAP 2025b). We sourced GL estimates for each of our case study species directly from BirdLife International (2025) after an update of Bird *et al.* (2020) (BirdLife International unpubl.; Table 7). These calculations were conducted in our custom R package *ACAPT*.

Different statistical approaches may result in different categories being suggested due to the explicit incorporation of our approach. Preliminary data explorations highlighted that the zero-bound on the lower end, and the absence of a boundary on the upper end of R_t could bias density-based evaluations either towards CR or LC in cases of high uncertainty (Supplementary Material 8). We thus report the median of R_t, the mode of R_t, and a density-based approach that calculates the proportion of iterations per IUCN category (c.f. *JARA*; Winker *et al.* 2020). In cases where >50% of the iterations is allocated in threatened categories but LC contains the highest percentual allocation, we follow *JARA* and consider a species NT (Winker *et al.* 2020). Finally, we excluded modes where they were subject to zero-bound biases, as assessed visually in density-plots. We chose to report these three statistical approaches to facilitate discussion on Red List category assignments in the face of uncertainty and, ultimately, aiding formal IUCN Red List assessments (during which species should be assessed against other criteria as well).

ACAP IG Trends workshop 24 May outcome:
Modes and densities are poor statistical approaches for identifying suitable IUCN Red List criteria and instead medians, or slightly more precautionary percentile approaches (e.g., 40% or 45% percentiles) should be considered.

Our approach to calculating population change and evaluating Red List categories against Criterion A slightly diverges from the approach in *JARA* (Winker *et al*. 2020, Sherley *et al*. 2020a,b). In the *JARA* approach, the change in total population size over 3 GL is estimated by projecting forward from the fitted model. However, for long lived species, like all ACAP species, this entails projecting far into the future, and the model projections for a long time horizon can be highly variable, leading to a large uncertainties in population trend. Instead, the approach we took here is to determine the trend over 3 GL by extrapolating the trend observed in recent history to 3 GL, under the assumption that the current trend will persist into the future.

2.5.2 ACAP High Priority Populations

Next to the IUCN Red List conservation status assessments, ACAP has its own conservation prioritisation mechanism in which it identified High Priority populations of ACAP species (ACAP 2025b). A population (i.e., breeding site) can be considered an ACAP High Priority population if it 1) holds ≥10% of the world population of the species, 2) is at risk of fisheries bycatch, and crucially, 3) has been declining at a rate ≥3% per annum for 20 years (ACAP 2025b). While these criteria are objective, ACAP High Priority populations have not uniformly been identified in a standardised fashion, and designation has largely been driven by ACAP Parties making a case for candidate High Priority populations. Our standardised modelling of trends of ACAP species provides an opportunity for standardised designation of ACAP High Priority populations as well, and we showcase this through our case studies. We first identified which breeding sites hold ≥10%, or have held ≥10%, of the global population at some point over the available timeseries (Table 7). We then consulted the literature to identify which species are at risk from fisheries bycatch (i.e., Abraham *et al*. 2019, Peatman *et al*. 2019, Edwards *et al*. 2023, ACAP 2024, Anon. 2025). Most importantly, we adapted W and T in Eq. 7 to reflect the last 20 years and applied the equation at the breeding site level rather than the species level. If a population met the first two criteria and the median of R_{20} fell below 0.544 (3% population decline means $\lambda_{t,i}$ = 0.970, and 0.970^{20} = 0.544), we proposed that this population be considered an ACAP High Priority population. To enable

ACAP to make decisions on High Priority Populations in the face of uncertainty, we provided estimates in the form of both medians and modes as well as density-based estimates. Finally, to provide further context, we also provided estimates in relation to other potential thresholds (e.g., $\lambda_{t,i}$ $0.980^{20} = 0.668$ and $\lambda_{t,i}$ $0.990^{20} = 0.818$).

2.6 Model implementation

We completed the Bayesian modelling underpinning our framework in two steps. The first step, the series of hierarchical models enabling the standardised error corrections, explicit incorporation of uncertainty (section 2.3), and the conversion into M (section 2.4), was completed using JAGS (Plummer 2003) within R (R Core Team 2024). Within this first step, we ran our hierarchical models for 100,000 iterations using two MCMC chains with a burn-in of 50,000 iterations. Evaluation of trace plots and $\hat{R} < 1.05$ indicated successful convergence of all monitored parameters. We then extracted median, CV, and 95% credible intervals (CrI) from the 100,000 posterior samples for each standardised count for $y_{t,i}$ (BP) and $M_{i,t}$. We used the median and the CV for the second modelling step, the modelling of trends and the subsequent conservation prioritisations assessments through the custom R package *ACAPT* which was also implemented through JAGS with R. Within this second step, trend models were run for 10,000 iterations using four MCMC chains with a burn-in of 5,000 iterations, which were sufficient to achieve model convergence, confirmed through posterior predictive checks. We report medians and 95% CrIs from the 20,000 posterior samples for all trend estimates obtained through *ACAPT*.

For reporting purposes, population changes are presented as changes over the full timeseries and since 2006 (i.e., the last 20 years as per ACAP convention; ACAP 2025b). To mirro similar work on other marine megafauna (Pacoureau *et al.* 2021, Juan-Jorda *et al.* 2022, Dulvy *et al.* 2024), we also calculated population change since 1970. Where timeseries did not start in ≤1970, we back-cast timeseries. For evaluations against IUCN Red List Criterion A, changes are presented over the full timeseries up to 3 GL, the last GL, and since 2006. For the ACAP High Priority Population evaluation, change was only assessed since 2006 as per ACAP convention (ACAP 2025b).

2.7 Case studies

We illustrate the utility of our framework to process and model abundance data of albatrosses and petrels using ten New Zealand endemics from 22 breeding sites, two circumpolar species from 12 breeding sites, and one tropical species from one breeding site (Fig. 6). These case studies varied considerably in terms of data availability, quality, and time span (40-91 years, mean = 54 years) as well as life history characteristics, such as breeding probability (e.g., Carneiro *et al.* 2020, Edwards *et al.* 2023), and generation length (e.g., Bird *et al.* 2020). Thus, these case studies enabled a thorough evaluation of how the framework could overcome different challenges associated with idiosyncratic and disparate abundance data from a range of species. Details of each case study's data and the abundance modelling steps taken can be found in Supplementary Material 1 and 2, a summary is provided in Table 4.

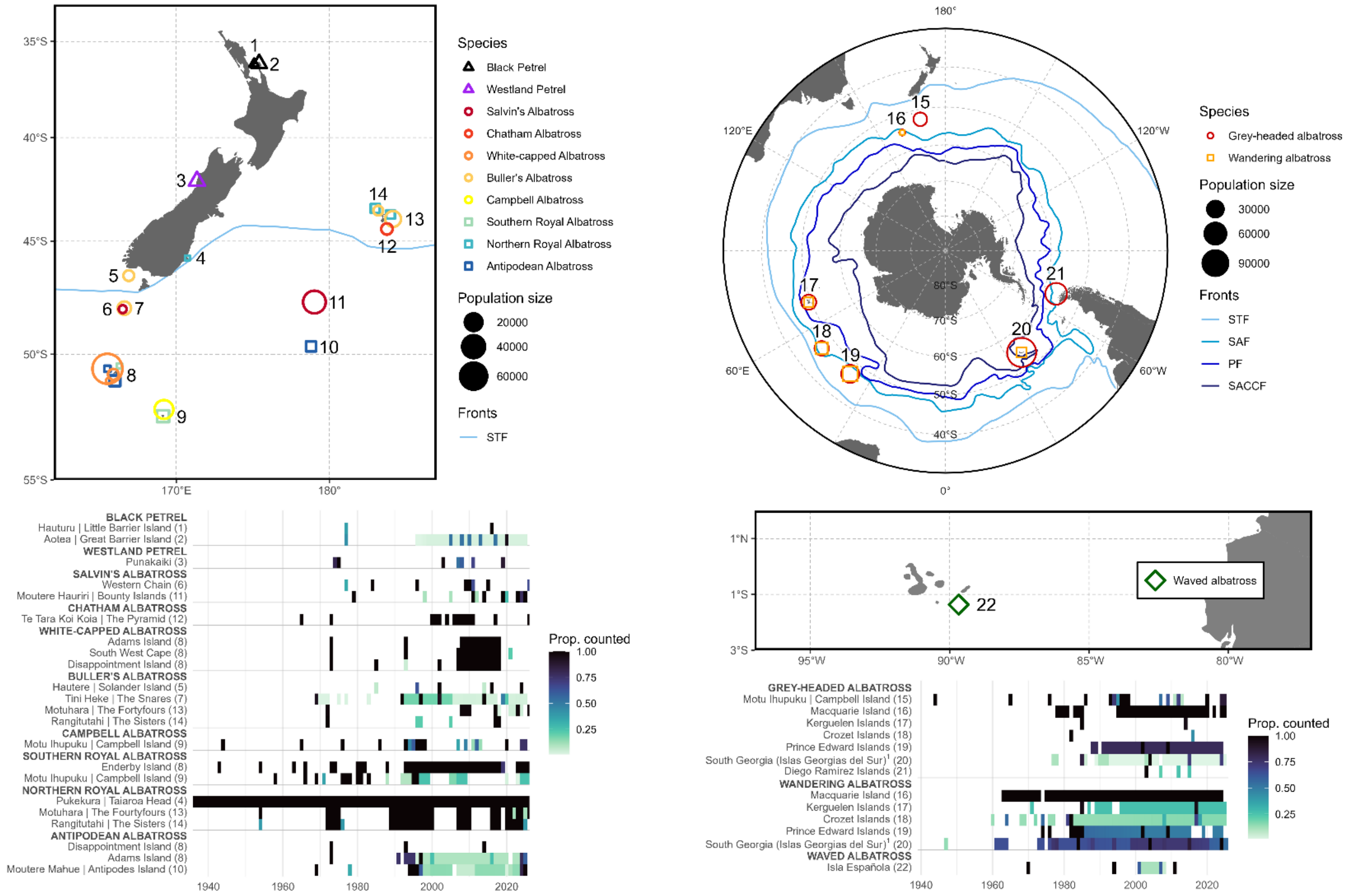


Fig. 6. Breeding sites of *Diomedea* albatrosses (squares), *Thalassarche* albatrosses (circles), *Phoebastria* albatrosses (diamonds), and *Procellaria* petrels (triangles) and proportion (ξ) of each breeding site counted during 1936-2026. Symbol size approximates relative colony size.

Table 6. Overview of data and processing steps for the ten New Zealand endemic albatrosses and petrels that are listed on ACAP Annex 1. Proportions of total population are based on the most recent estimates. For definitions of acronyms see Table 4. **Bold** breeding sites indicate that they contain(ed) ≥10% of the species' population. Numbers in brackets indicate equations applied (see main text and Table 3). ~ O indicates that original approaches in the source references were carried over, ~ t indicates that time-sensitive models were used to apply year-specific estimates to make corrections/transformations, ~ α indicates that mean estimates from models were applied due to limited data, ~ U indicates the application of uniform priors, ~ N indicates the application of normal priors, ~ β indicates the application of beta-distributed priors. GL is based on a BirdLife International (2025) update of Bird *et al.* 2020. For further details, see Supplementary Material 1and 2.

Species	Breeding site	Prop. pop.	Time series	No. counts	Metric	Method	Ψ^d	Ψ^o (Eq. 2)	Ψ^p (Eq. 3AB)	p^b (Eq. 5AB)	s (Eq. 5B)	GL
Antipodean Albatross	**Moutere Mahue | Antipodes I.**	42%	1969-2025	32	BP, BOG	GC, VPC, AC^{UAV}	-	$\psi^{BOG}\cdot\psi^{ABA}$ ~ O	ψ^{fail} ~ O (1969, 2024-25) ψ^{fail} ~ t (1978-2023) (3A)	~ t (5B)	~ U (pre- & post-2005)	31
	Adams I.	54%	1973-2025	34	BP, BOG	GC, VPC, AC^{UAV}	-	$\psi^{BOG}\cdot\psi^{ABA}$ ~ O	ψ^{fail} ~ t (1973-1996) (3A) ψ^{fail} ~ O (1997- 2025)	~ t (5B)	~ U (pre- & post-2005)	
	Disappointment I.	4%	1973-2014	3	BP, AOS	GC, VPC, AC^{heli}	-	$\psi^{BOG}\cdot\psi^{ABA}$ ~ α (Adams; 2AB)	ψ^{fail} ~ α (Adams; 3A)	~ α (Adams; 5A)	-	
Wandering Albatross	**South Georgia Is. (Islas Georgias del Sur)[1]**	13%	1947-2026	59	BP, AOS, BOG, C	GC, VPC, AC^{heli}, AC^{UAV}, VBC	-	$\psi^{BOG}\cdot\psi^{ABA}$ ~ O	ψ^{fail} ~ t (3A)	~ β (5A)	-	27
	Prince Edward Is.	48%	1974-2024	40	BP, AOS, C	GC	-	ψ^{ABA} ~ O	ψ^{fail} ~ α (3A)	~ β (Crozet Is.) (5A)	-	
	Crozet Is.	27%	1960-2025	53	BP, AOS, BOG	GC, AC^{heli}, AC^{sat}	-	ψ^{ABA} ~ α (2A) ψ^{BOG} ~ α (Kerguelen Is.) (2B)	ψ^{fail} ~ t (3A)	~ β (5A)	-	
	Kerguelen Is.	12%	1971-2025	38	BP, AOS	GC, AC^{heli}	-	ψ^{ABA} ~ α (Crozet Is.) (2A)	ψ^{fail} ~ t (3A)	~ β (Crozet Is.) (5A)	-	
	Macquarie I.	<1%	1963-2024	61	BP	GC	-	-	-	~ β (Crozet Is.) (5A)	-	
Northern Royal Albatross	**Motuhara | Fortyfours**	38%	1972-2026	30	BP, AOS, BOG, C	GC, AC^{fw}, AC^{sat}	-	ψ^{ABA} ~ U (2A) ψ^{BOG} ~ α (2B)	ψ^{fail} ~ t + fixed effect for storm impacts (3A)	~ N (with/ without storm impacts (5A)	-	28
	Rangitutahi | Sisters	61%	1972-2025	30	BP, AOS, BOG, C	GC, AC^{fw}, AC^{sat}	-	ψ^{ABA} ~ U (2A) ψ^{BOG} ~ α (2B)	ψ^{fail} ~ t + fixed effect for storm impacts (3A)	~ N (with/ without storm impacts (5A)	-	
	Pukekura | Taiaroa head	<1%	1938-2026	91	BP	GC	-	-	-	~ N (5A)	-	
Southern Royal Albatross	**Motu Ihupuku | Campbell I.**	>99%	1958-2026	29	BP, AOS	GC, VPC	ψ^{d}_{vpc} ~ U	ψ^{ABA} ~ α (2A) ψ^{BOG} ~ α (2B)	ψ^{fail} ~ O (1995-1996, 2005-2009)	~ N (5A)	-	27

Species	Breeding site	Prop. pop.	Time series	No. counts	Metric	Method	Ψ^{d}	Ψ^{o} (Eq. 2)	Ψ^{p} (Eq. 3AB)	p^{b} (Eq. 5AB)	s (Eq. 5B)	GL
	Enderby I.	<1%	1943-2026	41	BP, AOS	GC, AC^{heli}	-	-	$\psi^{fail} \sim t$ (1958-1994, 1997-1999, 2020-2026) (3A) $\psi^{fail} \sim \alpha$ (3A)	~ N (5A)	-	
Campbell Albatross	**Motu Ihupuku | Campbell I.**	100%	1944-2025	18	AOS, BOG, C	GC, VPC, AC^{heli}, AC^{UAV}, VBC	$\psi^{d}_{vpc} \sim U$ (1944-1986, 1994, 2008-2025) $\psi^{d}_{vpc} \sim O$ (1993, 1995-1998)	$\psi^{ABA} \sim t$ (2A)	$\psi^{fail} \sim O$	~ N (varying between 1994-2020 and 2024-2025) (5A)	-	21
Grey-headed Albatross	**Diego Ramirez Is.**	22%	2003-2015	4	AOS, BOG	GC, VPC, VBC, AC^{fw}	-	$\psi^{ABA} \sim \alpha$ (Motu Ihupuku) (2A) $\psi^{BOG} \sim \alpha$ (Motu Ihupuku) (2B)	$\psi^{fail} \sim \alpha$ (Motu Ihupuku, Macquarie I., Prince Edward Is., South Georgia (Islas Georgias del Sur)[1]) (3A)	~ α (Motu Ihupuku, Prince Edward Is., South Georgia (Islas Georgias del Sur)[1]) (5A)	-	25
	South Georgia Is. (Islas Georgias del Sur)[1]	44%	1977-2025	40	BP, AOS, BOG	GC, VPC, VBC	-	$\psi^{BOG} \cdot \psi^{ABA} \sim \alpha$ (1985-1986) $\psi^{BOG} \cdot \psi^{ABA} \sim O$ ^{A}	$\psi^{fail} \sim O$ (1977-1978) ^{A} $\psi^{fail} \sim \alpha$ (1985-1986) $\psi^{fail} \sim O$ (1989-2004) ^{A}	~ α (5A)	-	
	Prince Edwards Is.	13%	1988-2024	36	AOS	VPC	-	$\psi^{ABA} \sim t$ (2A)	$\psi^{fail} \sim t$ (3A)	~ α (5A)	-	
	Crozet Is.	7%	1982-2016	2	AOS, C	GC, VPC, AC^{heli}	-	$\psi^{ABA} \sim \alpha$ (Prince Edward Is.) (2A)	$\psi^{fail} \sim \alpha$ (Motu Ihupuku, Macquarie I., Prince Edward Is., South Georgia (Islas Georgias del Sur)[1]) (3A)	~ α (Prince Edward Is.) (5A)	-	
	Kerguelen Is.	8%	1985-2014	2	AOS	GC, AC^{heli}	-	$\psi^{ABA} \sim \alpha$ (Prince Edward Is.) (2A)	$\psi^{fail} \sim \alpha$ (Motu Ihupuku, Macquarie I., Prince Edward Is., South Georgia (Islas Georgias del Sur)[1]) (3A)	~ α (Prince Edward Is.) (5A)	-	
	Macquarie I.	<1%	1978-2025	36	BP	GC	-	-	$\psi^{fail} \sim t$ (3A)	~ α (Motu Ihupuku, Prince Edward Is., South Georgia (Islas Georgias del Sur)[1]) (5A)	-	
	Motu Ihupuku | Campbell I.	6%	1944-2025	18	AOS, BOG, C	GC, VPC, AC^{heli}, AC^{UAV}, VBC	$\psi^{d}_{vpc} \sim U$ (1944-1986, 1994, 2008-2025) $\psi^{d}_{vpc} \sim O$ (1993, 1995-1998)	$\psi^{ABA} \sim t$ (2A)	$\psi^{fail} \sim O$	~ N (5A)	-	
Buller's Albatross	**Hautere | Solander I.**	13%	1985-2024	6	AOS, BOG, C	GC, VPC, AC^{heli}	$\psi^{d}_{vpc} \sim U$	$\psi^{ABA} \sim t$ (2A)	$\psi^{fail} \sim t$ (3A)	~ N (5A)	-	24
	Tini Heke | Snares Is.	28%	1969-2025	39	BP, AOS,	GC, VPC	$\psi^{d}_{vpc} \sim U$	$\psi^{ABA}_{VPC} \sim \alpha$ (2A)	$\psi^{fail}_{VPC} \sim t$ (3A)	~ N (5A)	-	

Species	**Breeding site**	**Prop. pop.**	**Time series**	**No. counts**	**Metric**	**Method**	$\boldsymbol{\Psi^d}$	$\boldsymbol{\Psi^o}$ (Eq. 2)	$\boldsymbol{\Psi^p}$ (Eq. 3AB)	$\boldsymbol{p^b}$ (Eq. 5AB)	$\boldsymbol{s}$ (Eq. 5B)	**GL**
	Motuhara | Fortyfours	49%	1972-2026	11	BP, BOG, AOS, NS	GC, VPC, AC^{fw}	-	$\psi^{ABA} \sim$ t (2A) $\psi^{BOG} \sim \alpha$ (2B) $\psi^{occ} \sim$ t (2C)	$\psi^{\mathrm{fail}} \sim$ N (3A)	~ N (5A)	-	
	Sisters | Rangitutahi	9%	1972-2018	6	BP, AOS, BOG, NS	GC, AC^{fw}	-	$\psi^{ABA} \sim \alpha$ (2A) $\psi^{BOG} \sim \alpha$ (2B) $\psi^{occ} \sim \alpha$ (2C))	$\psi^{\mathrm{fail}} \sim$ N (3A)	~ N (5A)	-	
White-capped Albatross	**Disappointment I.**	95%	1973-2018	16	BOG	GC, VPC, AC^{heli}, VBC	$\psi^{d}_{VBC} \sim \beta$	$\psi^{ABA} \sim$ t (2A) $\psi^{BOG} \sim$ t (2B)	$\psi^{\mathrm{fail}} \sim \alpha$ (3A)	~ N (5A)	-	22
	Southwest Cape	5%	1973-2021	15	BOG	GC, VPC, AC^{heli}, AC^{UAV}, VBC	$\psi^{d}_{VBC} \sim \beta$	$\psi^{ABA} \sim$ t (2A) $\psi^{BOG} \sim$ t (2B)	$\psi^{\mathrm{fail}} \sim \alpha$ (3A)	~ N (5A)	-	
	Adams I.	<1%	1973-2018	13	BOG	GC, VPC, AC^{heli}, VBC	-	$\psi^{ABA} \sim$ t (2A) $\psi^{BOG} \sim$ t (2B)	$\psi^{\mathrm{fail}} \sim \alpha$ (3A)	~ N (5A)	-	
Chatham Albatross	**Te Tara Koi Koia | Pyramid**	100%	1965-2026	14	AOS, NS	GC, VPC, AC^{fw}, VBC	$\psi^{d}_{VBC} \sim \beta$, $\psi^{d}_{AC} \sim \beta$	$\psi^{ABA} \sim \alpha$ (1965-1974) (2A)	$\psi^{\mathrm{fail}} \sim$ N (3A)	~ N (5A)	-	21
Salvin’s Albatross	**Moutere Hauriri | Bounty Is.**	97%	1979-2025	11		GC, AC^{fw}, AC^{UAV}	-	$\psi^{ABA} \sim$ t (2A) $\psi^{BOG} \sim$ t (2B)	$\psi_{\mathrm{k}}^{\mathrm{fail}} \sim$ N (3B)	~ N (5A)	-	21
	Western Chain	3%	1977-2026	8		GC, AC^{fw},	-	-	$\psi_{\mathrm{k}}^{\mathrm{fail}} \sim$ N (3B)	~ N (5A)	-	
Waved Albatross	**Isla Espanola**	<99%	1970-2011	11	BP, C	GC	~ U (1970-2008) ~ O (2011)	-	$\psi^{\mathrm{fail}} \sim$ O (1970, 1994, 2001, 2007) $\psi^{\mathrm{fail}} \sim$ N (2002-2006, 2008) (3A)	~ N (5A	-	21
Westland Petrel	**Punakaiki**	100%	1974-2019	8	BP, AOS, NS	GC	-	$\psi^{ABA} \sim (\mathrm{N} \cdot \alpha)$ (1974-1975, 2007-2011) (2A) $\psi^{occ} \sim$ O (1974-1975, 2007-2011) (2C) $\psi^{occ} \cdot \psi^{ABA} \sim$ O (2003, 2019) (2AC)	$\psi^{\mathrm{fail}} \sim$ N (1974-1975, 2003, 2007-2019, separately) (3A)	~ U (1974-2011) (5A) ~ α (2019) (5A)	-	18
Black Petrel	**Aotea | Great Barrier I.**	85%	1977-2025	31	BP, NS	GC	-	$\psi^{occ} \cdot \psi^{ABA} \sim$ O	$\psi^{\mathrm{fail}} \sim$ t (3A)	~ N (5A)	-	16
	Te Hauturu-o-Toi | Little Barrier I.	15%	1977-2016	2	BP, NS	GC	-	$\psi^{occ} \cdot \psi^{ABA} \sim$ O	$\psi^{\mathrm{fail}} \sim \alpha$ (3A)	~ N (5A)	-	

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[1] A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty of the Falkland Islands (Islas Malvinas), South Georgia and the South Sandwich Islands (Islas Georgias del Sur e Islas Sandwich del Sur) and the surrounding maritime areas.

[A] The original correction factor used South Georgia (Islas Georgias del Sur)[1] Grey-headed Albatross counts is an inseparable compound factor of ψ^{BOG}, ψ^{ABA}, and ψ^{fail} the approach of which we carried over here without further adjustments (Poncet *et al.* 2006, 2017, Mackley *et al.* 2025).

3. RESULTS

Our standardised framework to process and model abundance data of ACAP species performed well, with the necessary flexibility, and ensured that all abundance data available for our case studies (893 counts) were subjected to standardised error corrections resulting in unified expression of data in BP at the start of the breeding season (tables in Supplementary Material 2). Data were also converted from BP into M, both of which allowed for the standardised modelling of trends, but the latter was used for conservation assessments.

3.1 Population trends

ACAP IG Trends workshop 24 May outcome:
Only counts conducted since 1970 (inclusive) should be included as older counts were perceived of lower accuracy. In addition, the IG Trends is to engage with data custodians to identify any poor-quality counts since 1970, document the underlying reasons for why counts should be considered of poor quality, and jointly decide on their inclusion.
For lumped archipelago breeding sites that may exhibit divergent trends based on clear underlying drivers (e.g., differences in invasive species composition), modelling of trends for part-sites of archipelagos separately should be investigated, provided ≥3 counts for each site exist.

3.1.1. Diomedea *albatrosses*

Antipodean Albatross abundance declined from an estimated 21,594 (95% credible intervals: 10,857; 45,216) breeding pairs (BP) in 1969 to 8,425 (7,180; 10,037) BP in 2025, which equated to a net change of -36,162 (-86,242; -13,356) mature individuals (M), or a relative change of -49.1 (-66.4; -29.3) %, over the entire timeseries, and a net change of -11,168 (-13,398; -9,359) M, or a relative change of -23.0 (-23.4; -22.5) % in the last 20 years (Fig. 7; Table 7). This collapse was predominantly driven by large declines on Adams Island between the 1970s and 1990s, followed by further declines on both Adams Island and Moutere Mahue | Antipodes in the mid-2000s.

Wandering Albatross abundance declined from an estimated 12,365 (7,714; 21,418) BP in 1947 to 10,942 (9,017; 13,496) BP in 2026, which equated to a net change of -9,073 (-30,166; 1,268) mature individuals (M), or a relative change of -19.7 (-40.9; 4.2) %, over the entire timeseries, and a net increase of 6,794 (3,850; 10,985) M, or a relative growth of 34.5 (8.5; 55.8) % in the last 20 years. This changing overall population trajectory reflected the various interacting trajectories from different breeding sites, in which the South Georgia (Islas

Georgias del Sur)[1] population has suffered a continuous decline, while the Indian Ocean populations declined until the 1990s, and subsequently recovered.

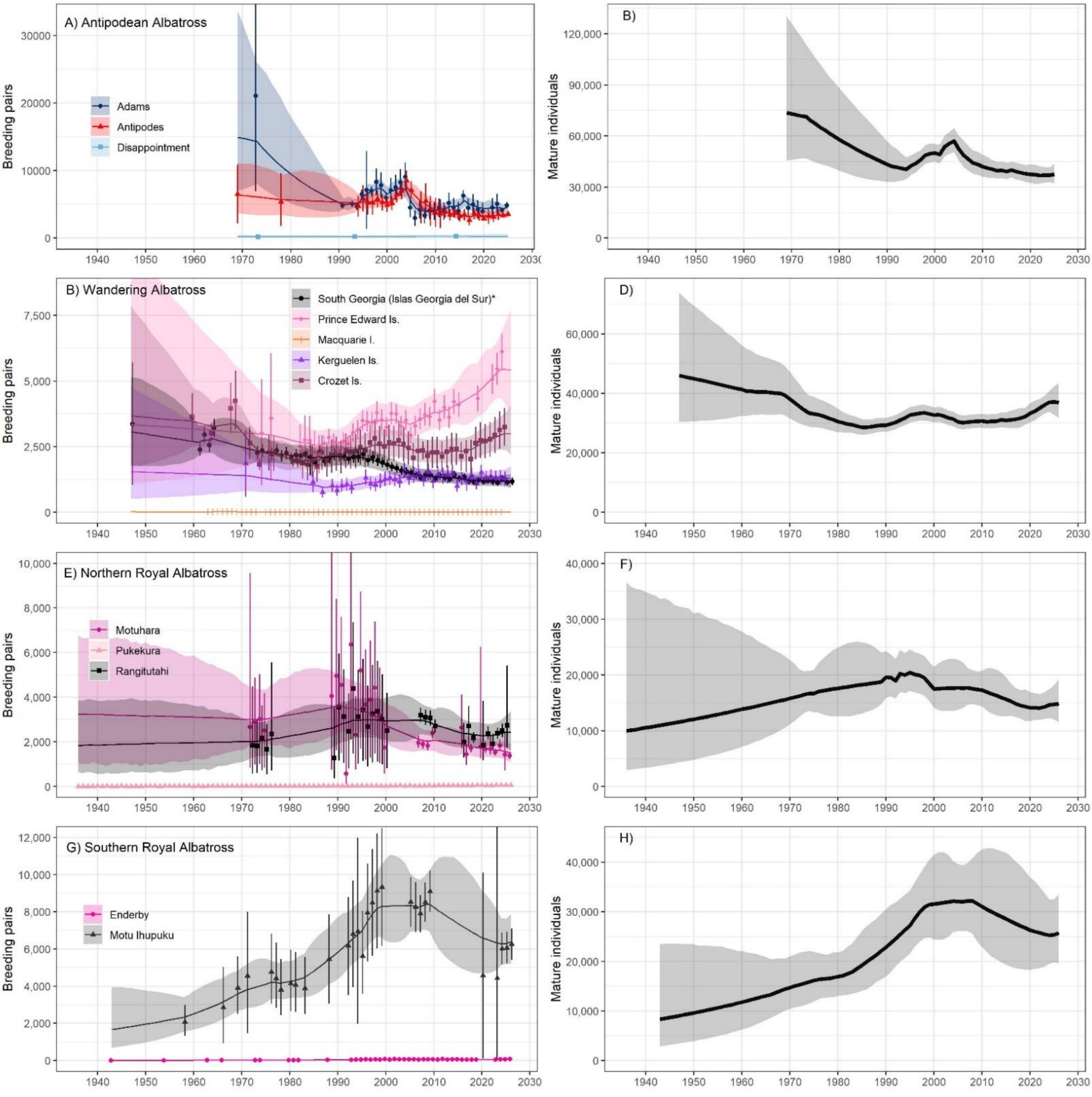

Fig. 7. Standardised abundance and trend estimates of Antipodean (AB), Wandering (CD), Northern Royal (EF), and Southern Royal Albatross (GH) expressed in annual breeding pairs per breeding site (ACEG) and total mature individuals (BDFH). Symbols with error bars (ACEG) represent $y_{t,i}$ estimates produced through the standardised processing and modelling framework (step 2.2) expressed in medians with 95% CrIs. Trend lines and translucent ribbons represent *ACAPT* trend model estimates (also expressed in medians with 95% CrIs). Note y-axes vary between species.

Northern Royal Albatross abundance changed from an estimated 5,067 (1,678; 10,605) BP in 1936 to a peak of 6,502 (4,601; 8,729) BP in 1993, after which a decline became apparent, leaving 3,981 (2,779; 5,351) BP in 2026. The trend equated to a net change of 4,835 (-17,356; 8,557) M. or a relative change of 48.5 (-47.3; 291.0) %, over the entire timeseries, indicating that this species likely increased, but given the wide credible interval, there is a possibility of a considerable decline as well. However, in the last 20 years, Northern Royal Albatrosses suffered a net change of -2,817 (-2,994; -2,626) M, equating to a relative change of -16.0 (-18.6; -13.4) % (Table 7). This decline was pronounced on Motuhara | the Forty-fours in particular, but the Rangitutahi | The Sisters population declined as well.

Southern Royal Albatross abundance changed from an estimated 1,679 (693; 3,955) BP in 1943 to a peak in abundance at 8,396 (6,969; 10,156) BP in 2005, after which the species declined, leaving 6,450 (5,250; 7,948) BP in 2026. The trend equated to a net increase of 19,394 (16,741; 18,894) M, or a relative change of 208.8 (42.5; 578.6) %, over the entire timeseries. However, Southern Royal Albatrosses suffered a net change of -6,336 (-6,525; -5,852) M, or a relative change of -19.8 (-25.0; -14.9) %, in the last 20 years (Table 7). The species-level decline was predominantly driven by declines on Motu Ihupuku | Campbell Island, while the small Enderby population continued to slowly increase.

3.1.2. Thalassarche *albatrosses*

Campbell Albatross abundance decreased from an estimated 29,989 (14,728; 59,905) BP in 1944 to 9,425 (6,633; 13,612) BP in 2025 (Fig. 8). This trend equated to a net change of -40,102 (-92,802; -16,623) M, or a relative change of -60.4 (-70,7; -49.8) %, over the entire timeseries and -16,353 (-30,895; -10,484) M, or -38.3 (-44.6; -38.5) %, in the last 20 years (Table 7). While this decrease was mostly steady over the majority of the timeseries, it may have accelerated in the 2020s.

Grey-headed Albatrosses exhibited the most dramatic decline of all species evaluated here. The species declined from an estimated 202,311 (67,380; 666,898) BP in 1944 to 56,255 (36,765; 88,396) BP in 2025. This trend equated to a net change of -790,587 (-2,217,616; -286,28) M, or a relative change of -74.8 (-86.8; -57.1) %, over the entire timeseries, indicating large uncertainty of the magnitude of the decline. However, certainty increased from the late 1970s onwards, and loss was estimated at -128,122 (-128,620; -120,659) M, or -32.5 (-40,0; -27.7) %, in the last 20 years. The decline was predominantly driven by severe, and ongoing long-term declines at South Georgia (Islas Georgias del Sur)[1] and Motu Ihupuku, while the population at Islas Diego Ramirez appears stable, and the population at the Prince Edward Islands are increasing, but these sites are dwarfed by the larger declining sites.

Buller's Albatross abundance increased from an estimated 19,408 (11,629; 34,792) BP in 1969 to 32,958 (26,056; 41,989) BP in 2026. This trend equated to a net increase of 31,263 (16,114; 35,078) M, or a relative change of 60.3 (20.2; 96.2) %, over the entire timeseries and 13,212 (10,737; 15,484) M, or 18.9 (17.7; 19.2) %, in the last 20 years (Table 7). While generally all four Buller's Albatross breeding sites showed increases across the entire timeseries, for Tini Heke | the Snares, this increase stagnated around the mid-2000s and reverted to a slow decline, contrasting with the other breeding sites.

Table 7. Absolute (net) and relative change in mature individuals (M) since 1970 and 2006 (i.e., the last 20 years), per species, genus, and overall (in median; 95% CrIs).

Species	Absolute change since 1970	Rel. change since 1970 (%)	Absolute change since 2006	Rel. change since 2006 (%)
Antipodean Albatross	-35,659 (-85,522; -13,762)	-48.8 (-65.4; -29.9)	-11,168 (-13,398; -9,359)	-23.0 (-23,4; -22,5)
Wandering Albatross	-1,115 (-3,696; 205)	-2.9 (-7.8; 0.6)	6,794 (3,857; 10,985)	22.5 (13.8; 33.7)
Northern Royal Albatross	-986 (-3,431; 361)	-6.2 (-15.1; 3.2)	-2,817 (-2,994; -2,626)	-16.0 (-18,6: -13.4)
Southern Royal Albatross	11,002 (9,228; 12,491)	74.7 (59.4; 88.8)	-6,336 (-6,525; -5,852)	-19.8 (-25.0; -14.9)
Diomedea total	-26,758 (-80,420; -706)	-18.9 (-35.5; -4.0)	-13,526 (-19.060; -6,852)	-10.5 (-13.3; -7.4)
Buller's Albatross	30,525 (16,328; 34,348)	58.2 (20,5; 92.4)	13,212 (10,737; 15,484)	18.9 (17.7; 19.2)
Campbell Albatross	-27,652 (-15,103; -53,345)	-51.3 (-58.2; -47.4)	-16,353 (-30,895; -10,484)	-38.5 (-44.6; -38.3)
Grey-headed Albatross	-519,505 (-876,616; 311,094)	-66.2 (-72.3; -59.1)	-128,122 (-128,620; -120, 659)	-32.5 (-36.0; -27.7)
White-capped Albatross	6,520 (-89,311; 44,456)	2.8 (-21.1; 34.4)	-23,228 (-39,540; 1,577)	-8.8 (-18.6; 0.05)
Chatham Albatross	2,142 (-2,776; 4,022)	19.0 (-13.7; 64.3)	1,793 (819; 3,594)	15.4 (8.6; 25.9)
Salvin's Albatross	-51,703 (-270,733; 9,711)	-33.2 (-64.9; 15.0)	-7,150 (-7,524; -4,908)	-6.2 (-6.4; -4.9)
Thalassarche total	-559,672 (1,276,453; -233,660)	-43.3 (-56.8; -29.4)	-159,848 (-195,023; -115,396)	-17.9 (-22.6; -13.1)
Waved Albatross	1,924 (-1,880; 1,836)	7.1 (-4.6; 9.8)	3,781 (2,890; 5,740)	16.3 (15.0; 17.1)
Westland Petrel	16,265 (7,135; 27,037)	533.8 (250.4; 842.4)	9,125 (2,690; 18,401)	89.6 (50.8; 94,7)
Black Petrel	14,434 (9.050; 18,660)	293.0 (95.8; 687.5)	7,394 (2,485; 19,498)	61.8 (31.5; 104.6)
Procellaria total	30,699 (16,185; 45,697)	385.0 (151.0; 748.1)	16,519 (5,175; 37,899)	74.6 (39.3; 99.6)
Total	**-553,807** **(-1,342,567; -186,833)**	**-37.7** **(-51.7; -24.0)**	**-153,074** **(-206,017; -78,610)**	**-14.3** **(-19.7; -8.5)**

White-capped Albatross abundance changed from an estimated 78,906 (44,751; 137,867) BP in 1973 to a peak in abundance of 93,410 (70,642; 129,510) BP in 2002, after which a shallow decline may have become apparent, leaving 81,625 (57,533; 116,719) BP in 2021. The trend equated to a net change of 7,333 (-55,915; 35,177) M, or a relative change of 3.2 (-14.3; 25.4) %, over the entire timeseries, highlighting considerable uncertainty around the fate of this species. However, uncertainty decreased more recently, and White-capped Albatrosses experience a net change of -23,228 (-39,540; 1,577) M, or a relative change of -8.8 (-18.6; 0.05) %, in the last 20 years (Table 7). This decrease was predominantly driven by declines on Disappointment Island, but the Southwest Cape population appeared in decline as well.

Contrasting with most other species, Chatham Albatross abundance on the sole breeding site at Te Tara Koi Koia | the Pyramid remained nearly unchanged, with an estimated 4,273 (2,238; 8,066) BP in 1965 and an estimated 5,388 (4,189; 6,871) BP 2026, equating to a net change of 2,777 (-2,935; 4,760) M, or a relative change of 26.1 (-14.4; 86.3) %, over the entire timeseries, and 1,793 (819; 3,593) M, or a relative change of 15.4 (8.6; 25.9) %, in the last 20 years.

Salvin’s Albatross abundance declined from an estimated 72,098 (36,342; 146,793) BP in 1973 to 48,285 (35,107; 66,763) BP in 2026, which equated to a net change of -49,747 (-174,483; -3,819) M, or a relative change of -32.4 (-4.9; -54.3) %, over the entire timeseries, and a net change of -7,150 (-7,524; -4,908) M, or a relative change of -6.2 (-6.4; -4.9) %, in the last 20 years (Table 7). This concerning decrease was predominantly driven by declines on the large Moutere Hauriri | Bounty Islands population, but the small Western Chain population declined since the mid-1990s as well.

3.1.3. Phoebastria *albatrosses*

Waved Albatross abundance changed from an estimated 10,420 (6,398; 19,043) BP in 1970 to 11,973 (7,963; 18,314) BP in 2011 (Fig. 9AB). This growth equated to a net change of 1,924 (-1,880; 1,836) M, or a relative change of 7.1 (-4.6; 9.8) %, over the entire timeseries, and a change of 3,781 (2,890; 5,740) M, or a relative change of 16.3 (15.0; 17.1) %, in the last 20 years (Table 7).

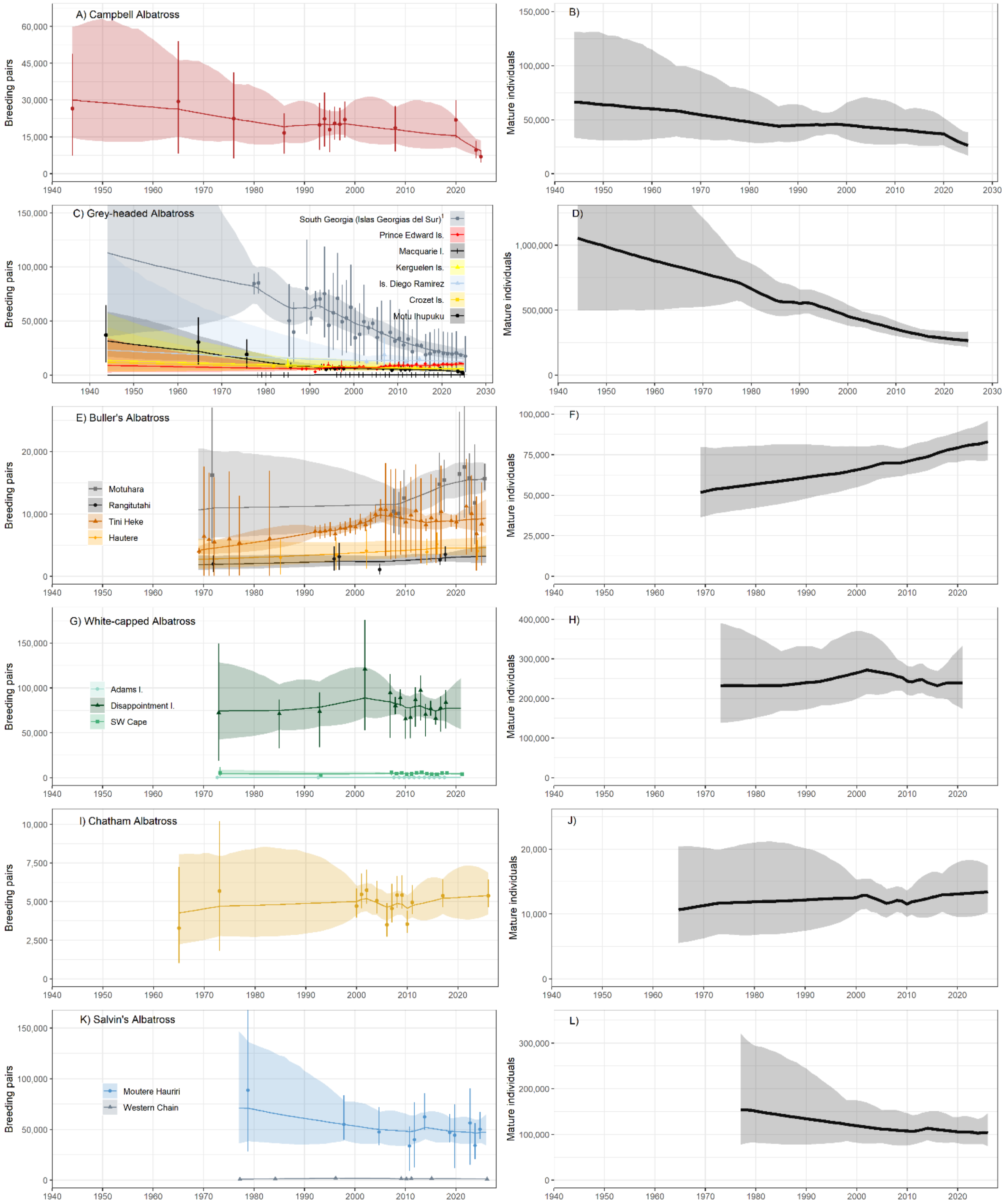

Fig. 8. Standardised abundance and trend estimates of Campbell (AB), Grey-headed (CD), Buller's (EF), White-capped (GH), Chatham (IJ), and Salvin's Albatross (KL), expressed in annual breeding pairs per breeding site (ACEGIK) and total mature individuals (BDFHJL). Symbols with error bars (ACEGIK) represent $y_{t,i}$ estimates produced through the standardised processing and modelling framework (step 2.2) expressed in medians with 95% CrIs. Trend lines and translucent ribbons represent *ACAPT* trend model estimates (also expressed in medians with 95% CrIs). Note y-axes vary between species.

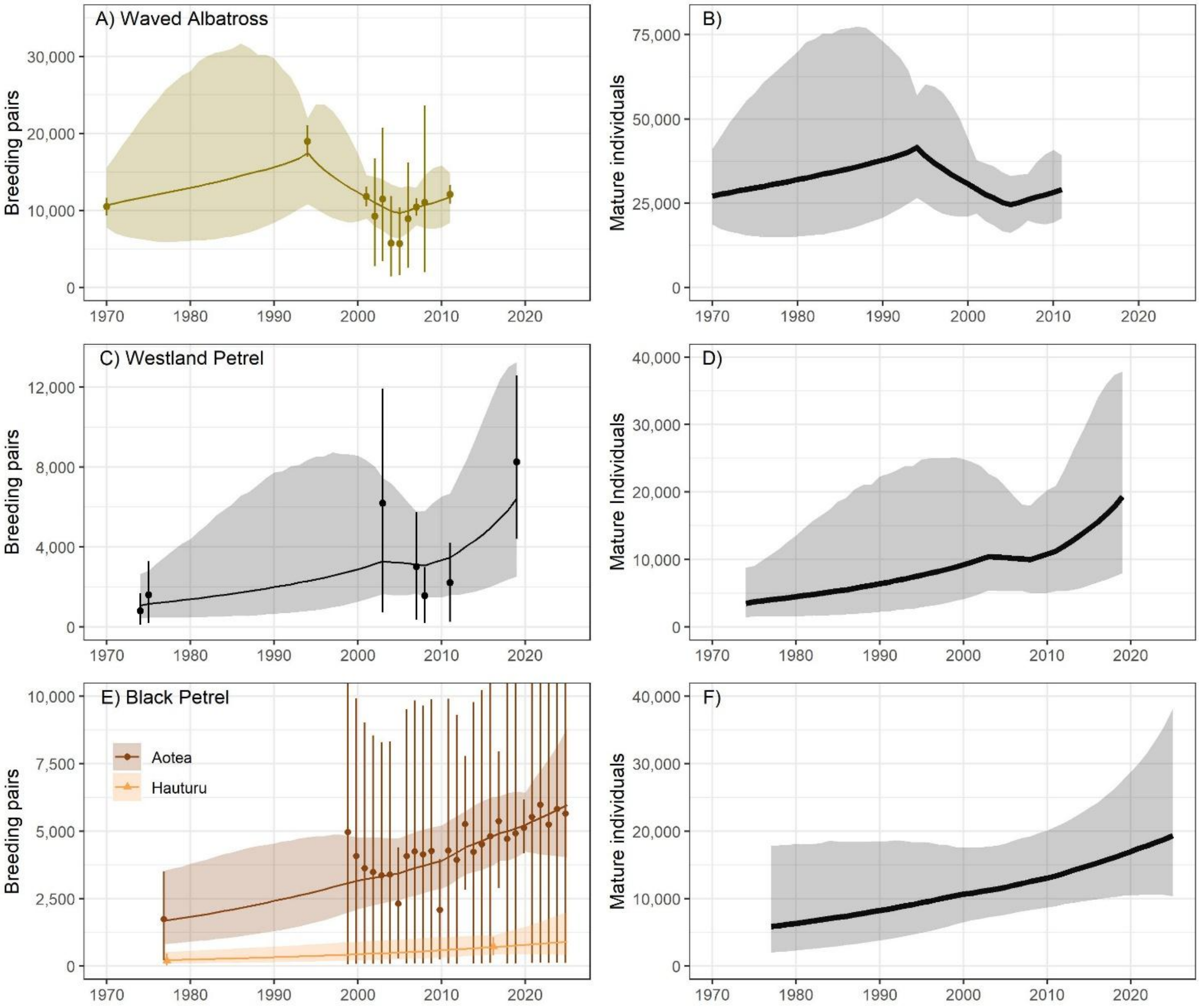

Fig. 9. Standardised abundance and trend estimates of Waved Albatross (AB), Westland (CD) and Black Petrel (EF) expressed in breeding pairs per breeding site (AC) and total mature individuals (BD). Symbols with error bars (AC) represent $y_{t,i}$ estimates produced through the standardised processing and modelling framework (step 2.2) expressed in medians with 95% CrIs. Trend lines and translucent ribbons represent *ACAPT* trend model estimates of mature individuals (also expressed in medians with 95% CrIs). Note y-axes vary slightly between species.

3.1.4. Procellaria *petrels*

Westland Petrel abundance increased from an estimated 1,909 (912; 4055) BP in 1974 to 6,430 (3,981; 13,255) BP in 2019 (Fig. 9). This growth equated to a net increase of 15,802 (6,553; 29,071) M, or a relative change of 450.2 (331.7; 458.7) %, over the entire timeseries, and 9,058 (2,688; 17,094) M, or a relative change of 89.6 (50.8; 61.8) %, in the last 20 years (Table 7). This increase appeared to be predominantly driven by the count in 2019, and considerable uncertainty surrounded the trajectory of this species.

Black Petrel abundance changed from an estimated 1,691 (822; 3,540) BP in 1977 to 6,883 (4,439; 10,834) BP in 2026. This growth equated to a net increase of 13,466 (8,366; 20,307) M, or a relative change of 228.4 (113.9; 418.2) %, over the entire timeseries, and 7,393 (2,485; 19,498) M, or a relative change of 61.8 (31.5; 99.6) %, in the last 20 years (Table 7). This increase was surrounded by considerable uncertainty, similar to its sister species, yet the uncertainty surrounding the Black Petrel population trajectory was driven by the low representativeness of counts, not a low temporal resolution of counts.

3.2 Conservation assessments

ACAP IG Trends workshop 24 May outcome:
Modes and densities are poor statistical approaches for identifying suitable IUCN Red List criteria and instead medians, or slightly more precautionary percentile approaches (e.g., 40% or 45% percentiles) should be considered.

3.2.1. IUCN Conservation status

When evaluating each species against IUCN Red List Criterion A, different species fell into different categories depending on what 1) section of the timeseries in the past was considered and 2) what statistical approach was used to identify the appropriate category. Some species' timeseries were sufficiently long enough to enable evaluation against Criterion A2 (Northern Royal, Southern Royal and, Campbell Albatross, and Black Petrel), but as most evaluations were made against Criterion A4, results refer to this criterion unless explicitly stated otherwise.

Antipodean Albatross (currently EN) was evaluated as EN when using medians or a density-based approach, but as CR when using the mode of population change over the last 20 years of observed data (Table 8; Fig. 10). Wandering Albatross (VU) was evaluated as NT when using the median or mode of population change for the entire timeseries available, but all other approaches resulted in categorisation as LC. The timeseries for Northern Royal Albatross (EN) was sufficiently long to enable assessments against Criterion A2 and against this criterion, the species' status was evaluated as LC by all three statistical approaches. Under Criterion A4, the species' status was evaluated as EN when using the last 20 years of observed data for all three statistical approaches. The timeseries for Southern Royal Albatross (VU) was also sufficiently long to enable evaluations against Criterion A2, under which, the species was categorised as LC. Under Criterion A4, the species was evaluated as LC when using a density-based or median approach, but as EN when using the mode of population change over the last 20 years of observed data.

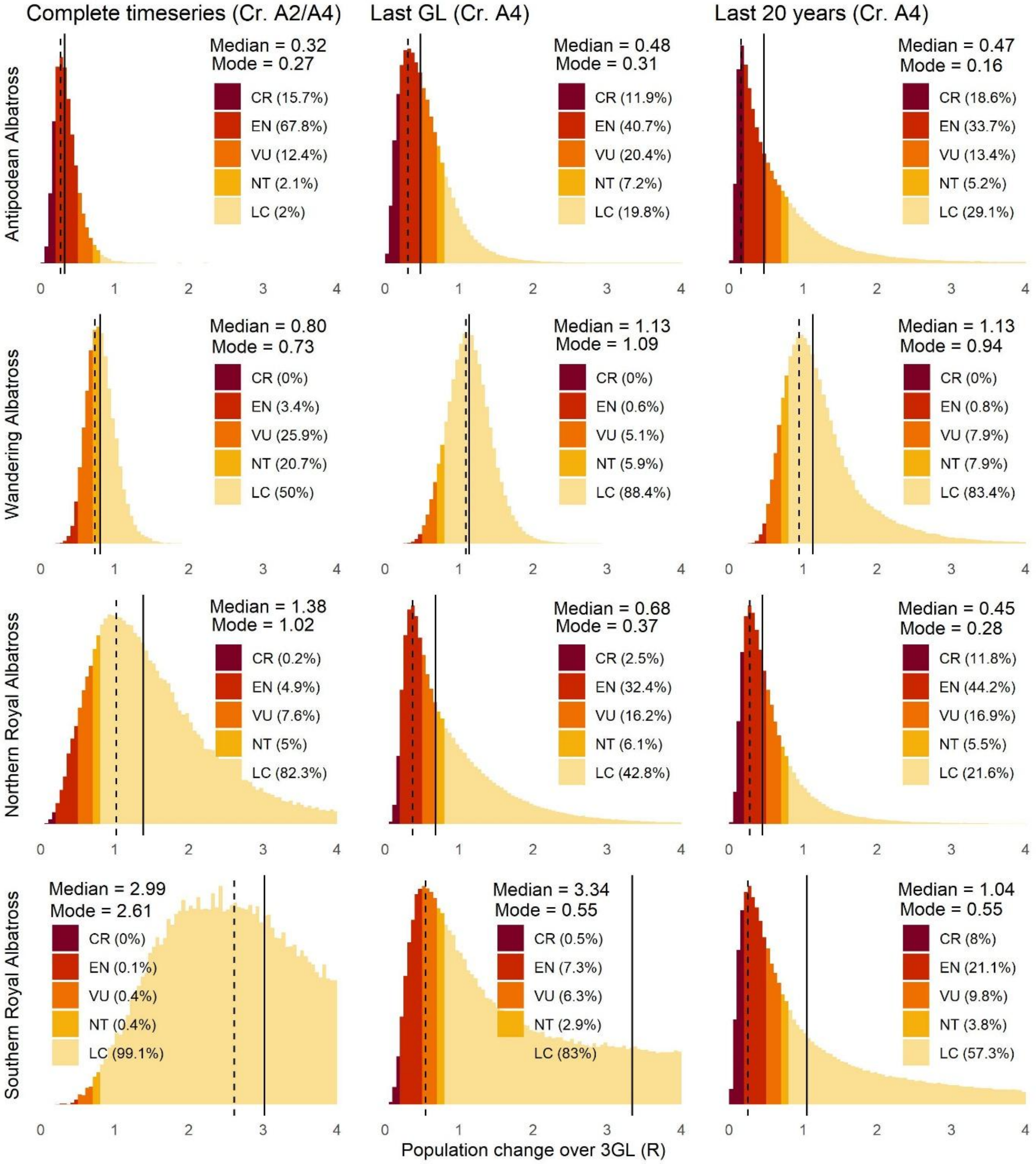

Figure 10. IUCN Red List evaluations using Criterion A for Antipodean (top row), Wandering (second), Northern Royal (third), and Southern Royal Albatross (bottom) based on population change evaluated over the entire timeseries available up until 3 generation lengths (GL) (left column), over the last GL (middle), or over the last 20 years (right). Legends provide the percentage allocation of iterations to IUCN Red List categories; solid vertical lines represent medians; dashed vertical lines represent modes. The total timeseries for Northern and Southern Royal Albatross were long enough (>3 GL) to enable evaluation against Criterion A2, while for the others, evaluation using the total timeseries represents another assessment approach under Criterion A4, as the timeseries did not cover the full 3 GL.

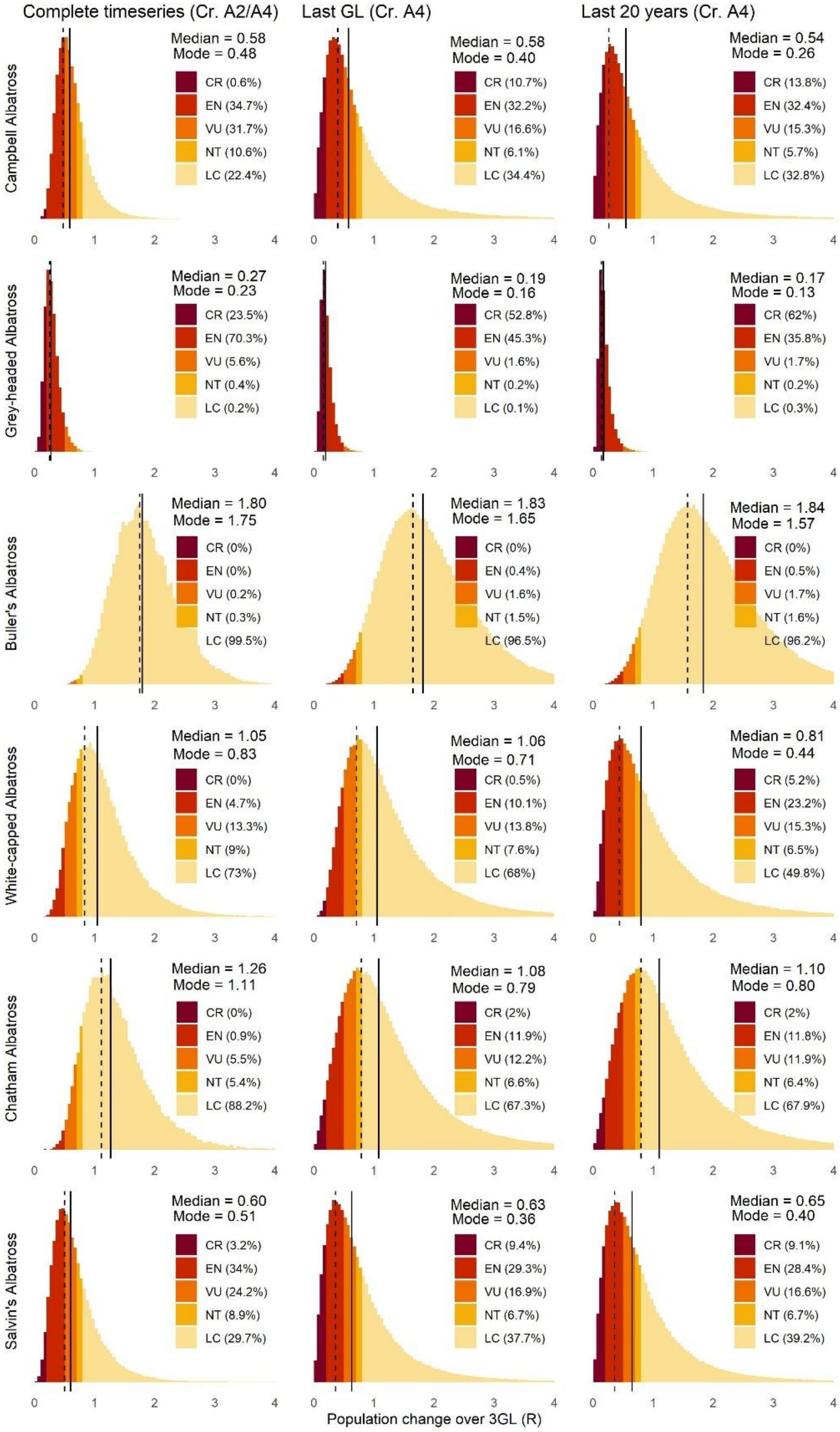

Figure 11. IUCN Red List evaluations using Criterion A for Campbell (top row), Grey-headed (second row), Buller's (third) White-capped (fourth, Chatham (fifth), and Salvin's Albatross (bottom row) based on population change evaluated over the entire timeseries available up until 3 generation lengths (GL) (left column), over the last GL (middle), or over the last 20 years (right). Legends provide the percentage allocation of densities to IUCN Red List categories; solid vertical lines represent medians; dashed vertical lines represent modes. The total timeseries for Campbell and Grey-headed Albatross were long enough (>3 GL) to enable evaluation against Criterion A2, while for the others, evaluation using the total timeseries represents another assessment approach under Criterion A4.

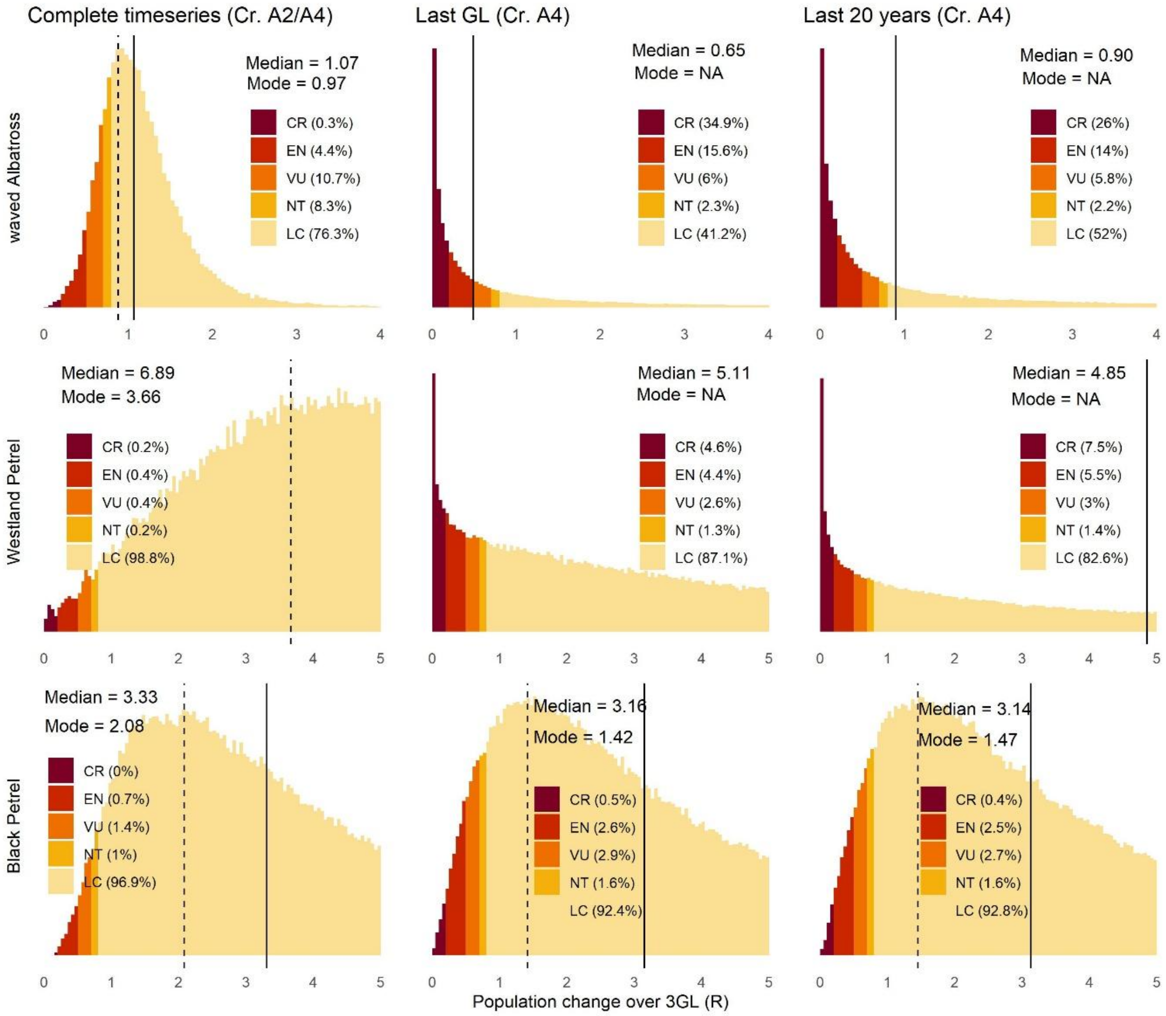

Figure 12. IUCN Red List evaluations using Criterion A for Waved Albatross, (top row), Westland Petrel (middle), and Black Petrel (bottom) based on population change evaluated over the entire timeseries available up until 3 generation lengths (GL) (left column), over the last GL (middle column), or over the last 20 years (right column). Legends provide percentual allocation of densities into IUCN Red List categories; vertical lines represent medians. The total timeseries for Black Petrels was long enough (>3 GL) to enable evaluation against Criterion A2, while for Waved Albatross and Westland Petrel, the evaluation using the total timeseries represents another assessment approach under Criterion A4. Westland Petrel medians < 5, and therefore outside the illustration. Waved Albatross and Westland Petrel modes were subject to uncertainty artefacts caused by the zero-bound for R estimates over the last GL and last 20 years and therefore excluded.

The timeseries for Campbell Albatross (currently VU) was sufficiently long to enable evaluations against Criterion A2 and against this criterion, the species' status was evaluated as EN when using the mode and the density-based approach, and as VU when using the median (Fig. 11). Under Criterion A4, the species' status was similarly evaluated as EN, VU, or LC depending on the statistical threshold. Similarly, the timeseries for Grey-headed Albatross (EN) enabled evaluations against Criterion A2. Against this criterion, the species' status was evaluated as EN. However, under Criterion A4, the species' status was evaluated as CR regardless of statistical approach or period of observations considered. Buller's Albatross (NT) was evaluated as LC regardless of the period of observations or the statistical approach. White-capped Albatross (NT) was evaluated as NT when considering the last 20 years of observations and using the median or a density-based approach. However, when using the mode, the species was evaluated as EN. Chatham Albatross (VU) was evaluated as LC when using medians and density-based approaches regardless of the period of data used, but as NT when considering the mode of R over the last GL. Salvin's Albatross (VU) was evaluated as EN when using the mode for of R over the last GL and the last 20 years or the density-based approach for the entire timeseries, but as VU when using the median.

Waved Albatross (currently CR) was evaluated as LC when using the entire observation period, but as VU or NT when using more recent observation periods. However, our analyses highlighted the high degree of uncertainty associated with this species which resulted in uncertainty artefacts caused by zero-bounds biasing modes.

Westland Petrel (currently EN) was evaluated as LC category regardless of the period under assessment and the statistical approach (Fig. 12). Similar to Waved Albatross, results underscored the uncertainty associated with this species resulting in uncertainty artefacts caused by zero-bounds biasing modes, despite the reasonable confidence in an increasing trend. The timeseries for Black Petrel (VU) was sufficiently long to enable evaluation against Criterion A2 and against this criterion, the species' status was evaluated as LC regardless of the statistical approach. Under Criterion A4, Black Petrel (VU) also fell into the LC category regardless of the period under consideration or the statistical approach employed.

When comparing the three different statistical approaches across species, approaches aligned when certainty was high, but when uncertainty was prevalent, modes were generally more precautionary, followed by medians, with density-based evaluations being generally (but not always) the least precautionary. Out of all 117 R_t comparisons, the three statistical approaches were in agreement in 70% cases; median and modes agreed in 63% of cases, densities and modes agreed in 66% of cases, while medians and densities agreed in 79% of the cases. When comparing the highest category assigned out of the three sections of the timeseries considered only (Table 8), approaches agreed in 58% of cases; median and modes agreed in 53% of cases, densities and modes as well as medians and densities agreed in 65% of cases.

Table 8. Highest IUCN Red List categories per species returned using different statistical approaches under Criterion and Criterion A4 (regardless of section of timeseries considered).

Species	Current Status (crit.)	Highest cat. under A2 (median; mode; density)	Highest cat. under A4 (median; mode; density)
Antipodean Albatross	EN (A4)	-	EN; CR; EN
Wandering Albatross	VU (A4)	-	NT; NT; LC
Northern Royal Albatross	EN (A4, B2)	LC; LC; LC	EN; EN; EN
Southern Royal Albatross	VU (D2)	LC; LC; LC	LC; EN; LC
Campbell Albatross	VU (D2)	VU; VU; EN	VU; EN; EN
Grey-headed Albatross	EN (A4)	EN; EN; EN	CR; CR; CR
Buller’s Albatross	NT (D2)	-	LC; LC; LC
White-capped Albatross	NT (A4)	-	NT; EN; LC
Chatham Albatross	VU (D2)	-	LC; NT; LC
Salvin’s Albatross	VU (D2)	-	VU; EN; EN
Waved Albatross	CR (B2)	-	VU; LC*; NT
Westland Petrel	EN (B2)	-	LC; LC*; LC
Black Petrel	VU (D2)	LC; LC; LC	LC; LC; LC

* Zero-bound bias affecting modes were not included (Fig. 12).

3.2.2. ACAP High Priority Populations

Of the 36 breeding sites of the 13 case studies, 24 met the first two criteria for inclusion as an ACAP High Priority Population. Specifically, these 24 breeding sites held ≥10% of the species’ world population (Table 7) and all these populations were deemed at risk of bycatch in fisheries (e.g., Awkerman *et al.* 2006, Abraham *et al.* 2019, Peatman *et al.* 2019, Edwards *et al.* 2023, ACAP 2024, Anon. 2025). One breeding site had to be excluded from the trend criterion evaluation (Black Petrels on Te Hauturu-o-toi | Little Barrier Island) as only two data points existed for this population. Of the remaining 23 sites, only one showed a median population change since 2006 (R_{20}) below 0.544. Grey-headed Albatross at South Georgia (Islas Georgias del Sur)[1] (a current ACAP High Priority Population; ACAP 2025b) exhibited a median decline of ≥3% per year decline over the last 20 (Fig. 13). Under a median annual decline threshold of ≥2% per year (i.e., $\lambda = 0.980$, $R_{20} = 0.688$), Grey-headed Albatross at Motu Ihupuku would qualify as an ACAP High Priority Population as well. Under a median annual decline threshold of ≥1% per year ($\lambda = 0.990$, $R_{20} = 0.818$), three further breeding sites qualified: Wandering Albatross at South Georgia (Islas Georgias del Sur)[1] and Antipodean Albatross at Moutere Mahue (both current ACAP High Priority Populations; ACAP 2025b) as well as Northern Royal Albatross at Motuhara. If modes would be considered the deciding statistical approach, no other High Priority Populations would be identified under a of ≥3%

per year decline threshold. Under a mode of ≥2% per year decline threshold, a total of five populations would qualify including the Isla Espanola Waved Albatross population (currently an ACAP High Priority Population; ACAP 2025b). Under a mode of ≥1% per year decline a total of 11 populations would qualify. These results were largely consistent with density-based evaluations (Supplementary Material 9).

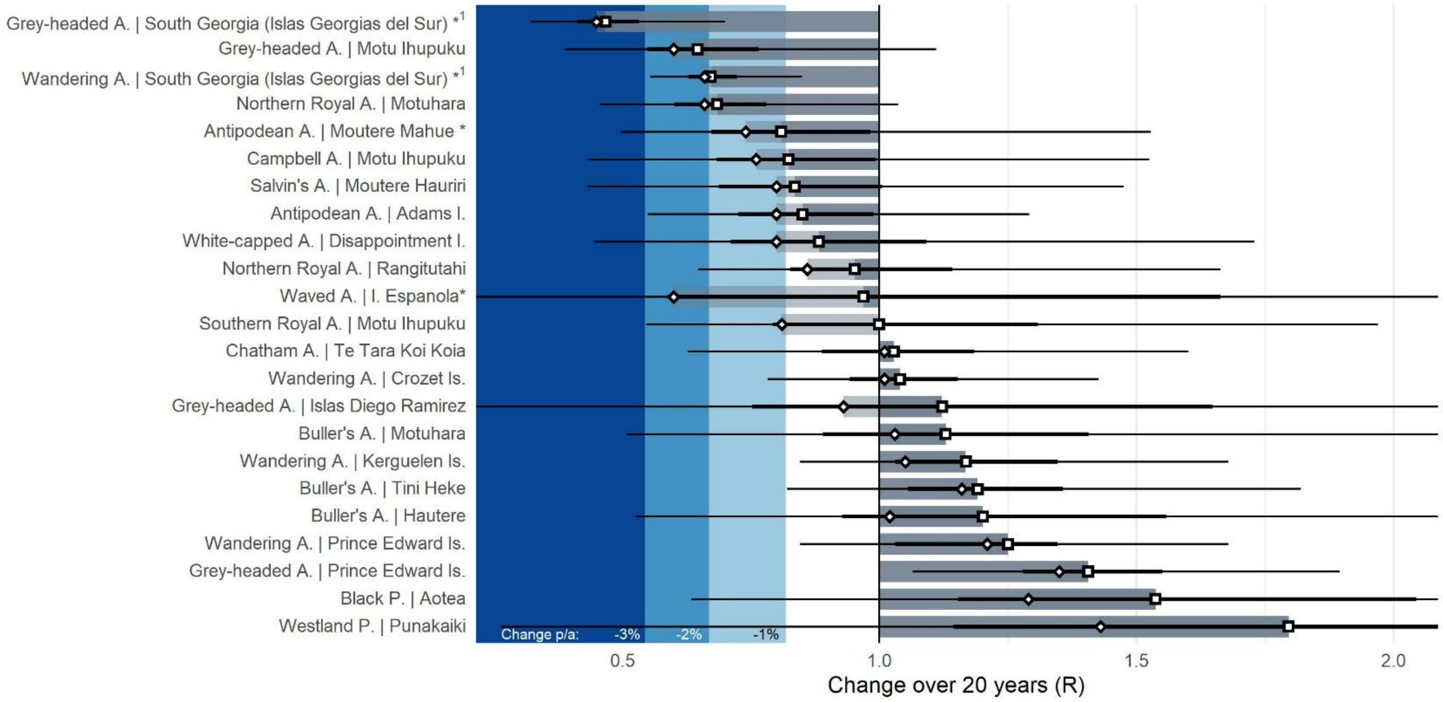

Figure 13. ACAP High Priority Population evaluations for case study breeding sites holding ≥10% of the world population (Table 7) that area at risk of bycatch in fisheries. Estimates illustrate population change (R_{20}) over the last 20 years through by medians (squares) and modes (diamonds) with 50% (thick error bars) and 95% credible intervals (thin error bars). Blue shades illustrate different levels of population decline; dark blue: $\lambda = 0.970$ (3% decline p/a) with $R_{20} = 0.544$, blue: $\lambda = 0.980$ (2% decline p/a) with $R_{20} = 0.688$, and light blue: $\lambda = 0.990$ (1% decline p/a) with $R_{20} = 0.818$. The Hauturu Black Petrel population was excluded entirely due to insufficient data. * Indicates current ACAP High Priority Populations (ACAP 2025b).

4. DISCUSSION

We here present the most comprehensive and coherent approach to evaluate albatross and petrel abundance and population status to date, which provided fully comparable trend insights across space, time, and species, while incorporating uncertainty explicitly, thereby providing a considerable advancement in understanding of the conservation status of these species. By 1) incorporating prior information on count accuracy, 2) adjusting for imperfect detection where required, 3) transforming different proxy metrics into a standardised abundance metric (annual breeding pairs), and 4) inferring the proportion of failed nests, our abundance data processing framework ensured that each individual count was transformed into the number of annual breeding pairs at the beginning of the breeding period. Using further auxiliary information on breeding probability, we then provided abundance data expressed in mature individuals as well, as required for conservation (IUCN 2012) and fisheries risk assessments (e.g., Edwards *et al.* 2023, Anon. 2025). After generating abundance estimates, comparable across species, time, and space, we implemented a flexible trend model through our bespoke R package *ACAPT* that enabled inference of population trends for each breeding site and species, while incorporating uncertainty surrounding individual counts. Our trend models have documented the dynamics of albatrosses and large petrels in unprecedented detail, highlighting the complex fates of these species. Our results in part provide further documentation of the extinction crisis as announced by ACAP in 2025 (ACAP 2025c) but also provides site- and species-specific nuance. For instance, our results highlight the apparent stability of circumpolar species in the Indian Ocean that are declining elsewhere. Our analyses already directly informed conservation assessments and will aid targeted management efforts necessary to reverse the documented declines. Furthermore, the framework we developed provides a critical stepping stone towards obtaining similarly detailed insights into the population trajectories of all 31 ACAP species, and perhaps, ultimately, all seabird species.

4.1 Abundance data processing and insights into future monitoring and reporting

Our abundance processing framework mirrors the modelling approach used in integrated population models by utilising related sources of information, enabling a holistic assessment whereby the results are greater than the sum of all parts. Integrated population models fuse different datasets across potentially different timeseries to enable more coherent insights into the dynamics of populations while incorporating uncertainty (e.g., Schaub & Abadi 2000, Zipkin & Saunders 2018, Armstrong *et al.* 2021, Richard *et al.* 2024). Similarly, leveraging data beyond abundance ensured a higher level of confidence in our estimates while retaining flexibility within our analysis approach. Sub-models within our framework leveraged 1) information on occupancy to enable the transformation of proxy metrics to BP while incorporating uncertainty, 2) data on eggs laid and chicks fledged to estimate breeding success and back-cast to estimate the number of annual breeding pairs at the start of the breeding period, and 3) the individuals' status to estimate breeding probability and ultimately, M, while acknowledging the biennial nature of some of the species. These sub-models proved highly flexible, enabling the integration of nuanced information where required. For example, we accounted for the development of a skewed sex ratio in Antipodean Albatross populations (Elliott *et al.* 2025) when estimating mature individuals, and the temporary impact of a catastrophic storm event on Northern Royal Albatross demographics (Robertson 1991, 1998). Similarly, we capitalised on the unique data available for Salvin's Albatross stage-specific breeding success (Thompson 2025) when adjusting for nest failure. However, data in many cases were limited and thus we had to resort regularly to using informative priors. While the use of informative priors is not unusual (e.g., Edwards *et al.* 2023, Anon. 2025), parameter estimates rooted in data are certainly preferred as this allows for more informed estimation of uncertainty. Considering the comprehensive nature of our analyses, our framework provides opportunities to reflect on current data availability and what monitoring and reporting considerations for these species are relevant for the future.

4.1.1. Future monitoring considerations

ACAP has produced guidelines for counts of both surface nesters (Wolfaardt & Phillips 2020) and burrow/crevice nesters (Parker & Rexer-Huber 2020), both of which are valuable resources, and our analyses of abundance data here build on this advice. For example, the ACAP recommended optimum monitoring approach consists of annual data collection (including counts) at representative study sites/plots/quadrats in combination with full breeding site counts every 5-10 years (Wolfaardt & Phillips 2020). Our modelled trajectories support this recommendation as still the best approach for obtaining high-quality abundance estimates. However, our results suggest that the representative study sites should exceed 10% of the breeding site population due to the nature of decreasing uncertainty under the asymptote specified in Eq. 1 (which mirrors the Poisson error distribution patterns, equally inflecting around 10%; Elliott *et al.* 2016). The impact of the uncertainty generation for counts that only represent small fractions was particularly evident for Buller's Albatross on Tini Heke (study sites represent 1.4-2.7% of the breeding site) and Black Petrel on Aotea | Great Barrier Island (2.3%) (Fig 9A & 10A). Consequently, both population's datasets would benefit from increased coverage of their representative study sites (at least for abundance data). Furthermore, some species are monitored annually, but not for abundance (e.g., Westland Petrels or Waved Albatross; Street 2013, Simister *et al.* 2023) and thus would benefit from the establishment of abundance study sites as well (e.g., plots or quadrats).

When annual abundance monitoring is not possible, it is preferable to conduct counts in several successive years at larger time intervals rather than single counts over smaller time intervals, partially due to the impact breeding probability variation can have on single counts (Wolfaardt & Phillips 2020), and partially due to the decrease in uncertainty following successive counts as propagated by trend models. Finally, Wolfaardt & Phillips (2020) and Parker & Rexer-Huber (2020) recommended to count albatrosses and petrels as closely as possible to the end of laying, and this recommendation is supported here as well. As such, our work here builds on the previous recommendations, provides increased statistical rigor, which will hopefully reduce uncertainty in the future.

Our work also revealed which species and sites should be considered the highest priority for obtaining updated abundance estimates to reduce uncertainty. Within New Zealand, White-capped Albatrosses at Disappointment Island, Westland Petrels at Punakaiki, and Black Petrels at Hauturu are monitoring priorities. White-capped Albatrosses are a high priority to gain further insights into the likely shallow decline that this species is suffering, while the two petrels require further data to confirm the likely population growth. For Wandering Albatrosses at Peninsule Rallier du Baty at Kerguelen and Ile aux Cochons, Ile de l'Est, and Ile des Apotres, and Ile des Pingouins appear relevant priorities to ensure that trajectories observed on the smaller study sites in each archipelago are indeed representative of overall trajectories. For Grey-headed Albatrosses, updated counts from Crozet, Kerguelen, and Islas Diego Ramirez appear pressing monitoring priorities given the low number of data points for each of these sites. Finally, a full island count of Waved Albatrosses, expressed in breeding pairs following the methodology of Anderson *et al*. (2008) would be of great value.

4.1.2. Future reporting considerations

ACAP recommends to clearly record the unit (or metric) of counts (Wolfaardt & Phillips 2020), but despite this recommendation, it remained a challenge to interpret some counts due to the lack of reporting detail. Ideally, each count has an exact description of what was counted. Here, we provide the necessary terminology with updated definitions of count metrics and their hierarchy. Future reporting of ACAP species counts should follow the terminology and ensure exact recording of what has been counted to facilitate future integration into joint modelling processes such as ours. Similarly, metadata reporting associated with counts can be improved as well. For instance, reporting of count dates is surprisingly rare. Furthermore, to enable and promote the use of auxiliary data in joint modelling exercises (and their integration into the ACAP database; see below), reporting of these auxiliary data could be improved. Key auxiliary data consist of 1) the number of NS/BOGs/AOS/BPs counted in transects, close-up photographs, or burrow inspections (Baker *et al.* 2023, E Bell *et al.* 2025, Frost *et al.* 2025), 2) the number of nests monitored and number of chicks that have fledged from these (or eggs hatched, chicks survived until post-guard, and subsequently fledged; Fischer *et al.* 2021, Thompson 2025), and 3) summaries of individuals' status to provide insights into breeding probability (Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). However, auxiliary data are often reported estimates alone, limiting their utility. For example, reports do not always contain data summaries (e.g., number of eggs laid, number of chicks fledged, and f estimate) while multi-year means only or conditional estimates are common but problematic. Conditional estimates in particular cannot directly be integrated into the analysis framework with ease (e.g., p^b for successful or failed breeders, without reporting overall p^b; e.g., Waugh *et al.* 2015, Fu & Sagar 2016). Similarly, compound estimates, e.g., recording BOG and BP only for surface nesters informing $\psi^{BOG} \cdot \psi^{ABA}$ only in combination (Elliott *et al.* 2025, Rexer-Huber *et al.* 2025), should be avoided where possible, as such reporting misses a step in the occupancy metric hierarchy (Fig. 4). Ideally, auxiliary data summaries are recorded to the finest resolution possible (e.g., for Ψ^o: 967 BOG, 865 AOS, 850 BP; c.f. Baker *et al.* 2017) alongside estimates. Some further consideration should be given to reporting of information that enables easy estimation of annual p^b. One option could be to report p^b as a product of demographic analyses (e.g., Pardo *et al.* 2017) more explicitly (e.g., in table form). Alternatively, reporting the estimated number of individuals following demographic analyses as well as recorded breeding pairs would enable the estimation of p^b as well (e.g., Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). Neither option provide easy access to p^b and thus further work is required to elucidate this parameter for most species. Adjustments in reporting of auxiliary data would enable the inclusion of these auxiliary data in the ACAP database allowing for more direct integration into joint abundance modelling exercises and therefore improved insights into species' status.

4.1.3. ACAP database considerations

The ACAP database (ACAP 2025b) has been fundamental to our analyses, and while our study underscores its importance, it also revealed several avenues for improving this crucial resource and ensuring its continued relevance and utility. Our work suggests that the database is not exhaustive, even for some very well studied species. Specifically, 27% of the counts collated here were not in the ACAP database despite being relevant and as such further efforts are required to ensure comprehensive coverage. More pressingly though, the current data structure within the ACAP database is not fully fit for the purpose of transparently archiving abundance data. The most problematic aspect of the current data structure was the conflation of metric, method, and errors corrections. Clear, coherent definitions of metrics, methods, and errors were also lacking. Further challenges included the limited utility of the reliability score (which also is conflated with method and metric) and differing numerical interpretations of the ACAP accuracy score. Our framework has provided solutions to each of these challenges. We coined more transparent definitions of metrics, methods, and errors (Table 1-3), which subsequently allow for clear separation between the three in the data structure (Supplementary Material 2). Additionally, while we discarded the reliability score, we formulated a numerical approach using the ACAP accuracy score to communicate prior beliefs on uncertainties associated individual counts. The approach in Eq. 1 appeared a promising avenue going forward, with applications extending beyond solely ACAP species. However, assignment of the accuracy scores still relies on consistency from those archiving counts. Further improvements could consist of providing clear definitions of what *Low*, *Medium*, and *High* counts (i.e., what combination of metric, method, and timing, and other additional considerations would result in which score).

Our data structure allowed for more transparent recording of metadata as well. The current ACAP database does not have the option to record count dates (and how these relate to the species' phenology) or how representative (i.e., ξ) a count is, and both should be considered crucial additions. Finally, to capitalise on the power that the joint modelling of abundance and demographic information beyond abundance in our framework highlighted, recording relevant auxiliary data in the ACAP database should be considered as well. Key additional data include occupancy counts (e.g., Baker *et al.* 2023, Frost *et al.* 2025) and the number of nests monitored and number of chicks that have fledged (e.g., E Bell *et al.* 2025). However, as mentioned above, further consideration should be given on the integration of p^b. The estimation of M is sensitive to p^b and despite decades of investment into demographic understanding of Wandering Albatrosses (e.g., Pardo *et al.* 2017, Weimerskirch 2018), limited information in available on this particular parameter and thus better site-, and season-specific reporting on p^b is required. Potentially, the number of individuals in a population as estimated through capture-mark-resight modelling (Elliott *et al.* 2025, Rexer-Huber *et al.*

2025) could be integrated into the database. Alternatively, direct annual estimates of p^b from demographic analyses (e.g., Pardo *et al.* 2017) could be captured within the database. Improvement of the ACAP database, including by incorporating auxiliary data, would further underscore the database as an impressive and highly valuable abundance resource, and would encourage ongoing analyses and collaborations across borders such as ours to better understand the fates of these species.

4.1.4. Future abundance data processing considerations

We also identified key areas for future study and improvement of data standardisation and processing framework and thus further enhancing insights into abundance and population trends. Firstly, despite our best efforts, we were unable to formulate a standardised approach that enables accounting for Ψ^d in counts that have not done so originally. We have attempted to overcome this challenge by using informative priors when other counts reported Ψ^d estimates (e.g., Sagar & Stahl 2005), by increasing uncertainty surrounding counts through downscaling the associated ACAP accuracy score in Eq. 1, or by applying custom adjustments for specific counts through formal expert elicitations (Hemming *et al.* 2018). While these custom approaches can be powerful (e.g., Fischer *et al.* 2020, 2022), they are also time consuming and place considerable burden on experts and thus are only suitable when it is most pressing to account for Ψ^d, e.g., for VBCs. Ideally, a similar standardised approach mirroring our process for accounting for Ψ^o (Eq. 2) and Ψ^p (Eq. 3) would be developed for Ψ^d, enabling retrospective adjustments. Secondly, while we developed a standardised modelling process to account for ψ^{fail}, we have not established such a process for ψ^{lay}. Counts conducted prior to (mean) lay dates are uncommon, but they are not absent (e.g., Baker *et al.* 2020, Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). To adjust such counts, we've either relied on the ψ^{lay} corrections applied in the original sources (e.g., Elliott *et al.* 2025, Rexer-Huber *et al.* 2025) or we've increased the uncertainty by adjusting the associated ACAP accuracy score. However, a standardised modelling process for account for ψ^{lay} is yet to be developed. Third, we formulated the proportion of the population ξ being directly representative of the population of the breeding site to enable extrapolation in Eq. 1. However, extrapolation is often based on density calculations and surface area estimations particularly with burrow/crevice nesters, which can be particularly sensitive to various considerations including 2D vs. 3D surface extrapolations (Parker & Rexer-Huber 2020). Our approach's strength is its simplicity, yet future work could investigate how to integrate density-based estimation processes into our framework, which may hold a potential solution to the challenge of retrospectively accounting for Ψ^d (e.g., by integrating distance-sampling; Buckland *et al.* 2001).

4.2 Trend modelling through *ACAPT*

Most of the insights generated by our work were produced by our flexible trend model as implemented through *ACAPT* which shows great promise for wider applicability for future abundance, trend, and conservation assessments. Specifically, after our standardisation modelling steps, we fit our Bayesian state-space model ACAPT to the breeding pair and mature individual datasets for each species providing the trend and conservation status evaluations that form the core of our results. Our state-space model builds on a similar decision-tool, JARA (Winker *et al*. 2020, Sherley *et al*. 2020a,b), but improves on this model, particularly through more explicit treatment of uncertainty within projections that are relevant to the IUCN Red List assessments. In addition, we adapt the IUCN Red List evaluations, which can be implemented through *JARA* as well, to suit other conservation assessments, such as the ACAP High Priority Population evaluations, further underscoring the flexibility of *ACAPT*. Yet, there are further improvements that can be made to ACAPT and our modelling approach. Firstly, we separated the hierarchical modelling to standardise abundance and the trend modelling in *ACAPT*, rather than fully integrate these two. We made this decision to ensure that 1) any of the preparatory standardisation steps did not have possibly overly inflated impacts on the overall trend estimates and 2) that *ACAPT* remained a more general trend modelling package like its predecessor *JARA* (Winker *et al.* 2020). However, there could be benefits to a fully integrated approach as well, including the continuous propagation of uncertainty, including accounting for the asymmetry therein. Once the ACAP database has been improved to enable better data integration, an update to ACAPT could potentially facilitate a more integrated approach. Other future improvements and expansions of ACAPT could also include the integration of other abundance metrics such as the Living Planet Index, as has been done for other marine megafauna (e.g., Pacoureau *et al.* 2021), or the inclusion of options that enable specific evaluation of efficacy of policy changes within the modelling framework (e.g., Pacoureau *et al*. 2023).

4.3 Long-term albatross and petrel trends

The abundance trends presented here provide a stepping stone to better understand and communicate the fate of these charismatic species. Our results are a considerable improvement on previous studies (e.g., Fischer *et al.* 2024), underscore the concerning, long-term, and ongoing declines of several ACAP species but also provide improved site- and species-specific nuance. Previous work had not integrated uncertainty in the underlying abundance data and the trend models, and as such, our overall results appear less severe than those reported in Fischer *et al.* (2024). Nevertheless, our results have several commonalities with previous work, and while the underlying causes of the documented trends were not a focus of our analyses, some insights can be gained from the literature and will hopefully guide future work to provide detailed insights into causes, and, ultimately, inform direct conservation actions.

Our approach also offers the opportunity to improve the nature of seabird abundance assessments and bring understanding of seabird abundance closer in line with abundance insights into other marine megafauna (e.g., Pacoureau *et al.* 2021, Juan-Jorda *et al.* 2022, Dulvy *et al.* 2024). Next steps should focus on expanding our analyses to all ACAP species to provide complete and coherent insights into the population trajectories of all albatrosses and large petrels. Furthermore, considering similar efforts that have been completed for other species-groups (e.g., Che-Castaldo *et al.* 2017, Talis *et al.* 2023, Sherley *et al.* 2019, 2024) obtaining continuous time-series for all seabird species, may be possible, and would indeed result in seabird abundance assessments mirroring those of other marine mega-fauna communities. An additional future opportunity includes the provision of these continuous timeseries to fisheries risk assessments to disentangle the improvements that bycatch mitigation may afford species from the reduced number of individuals available for capture due to the decline of populations (e.g., Edwards *et al.* 2023, Anon. 2025). Perhaps more importantly though, our trends provide more tangible and communicable metrics to the general public, managers, policy makers, funding bodies and others to illustrate the on-water impact of the current extinction crisis (ACAP 2025c).

4.3.1. Diomedea *albatrosses*

A common theme of both our study and previous work is the continuing decline of the *Diomedea* albatross populations in New Zealand. The ongoing and dire status of the Antipodean Albatross has been well documented and studied (e.g., Richard *et al.* 2024, Rowley *et al.* 2024, Elliott *et al.* 2025, Rexer-Huber *et al.* 2025) and insights from other studies generally match our estimates (e.g., -49% since 1970, -23% since 2006). However, the more recent, but similar declines of Northern and recently Southern Royal Albatross (-16% and -20%, respectively) have received less attention. The decline of the Antipodean Albatross is tightly linked to bycatch in pelagic longline fisheries, particularly in the high seas of the Tasman Sea and the southwestern Pacific east of New Zealand (Richard *et al.* 2024, Rowley *et al.* 2024, Anon. 2025). The impact of royal albatross bycatch in pelagic longline fisheries is less well understood, and direct links may be more difficult to establish as these species are caught in trawl fisheries as well (Edwards *et al.* 2023), including off the Patagonian Shelf (Jimenez *et al.* 2014, Tamini *et al.* 2020) and South Africa (Petersen *et al.* 2009, Rollinson *et al.* 2017). In addition, both species are circumpolar (Morten *et al.* 2025) and thus, both encounter various fleets in various jurisdictions around the Southern Ocean that jointly impact upon their populations. Irrespective of the marine threats, the conservation gains achieved for Southern Royal Albatross through the invasive species eradications on Motu Ihupuku and the resultant restoration (Towns & Broome 2010, Brown *et al.* 2023), are now coming undone. Similarly, both Chatham Island strongholds for Northern Royal Albatrosses have now recovered from the impacts of the severe storm in the 1980s (Roberston 1991, 1998, Bell 2026) and while sufficient time has passed for demographic impacts from this storm to be no longer evident, both populations are still declining. More concerted monitoring of all three species, as well as more concerted efforts to mitigate bycatch in the relevant fishing fleets is required to stem the current declines (Jiminez *et al.* 2026). The key fisheries that impact Antipodean Albatross are known (Richard *et al.* 2024, Rowley *et al.* 2024), but they are not for the two royal albatrosses, and this should be seen as a research and conservation priority.

Contrasting to the New Zealand *Diomedea* albatrosses, the Wandering Albatross population trajectory is composed of contrasting population trends from the various breeding sites around the Southern Ocean. Despite the huge investment into understanding the demography of this species from a range of different sites (e.g., Pardo *et al*. 2017, Weimerskirch 2018), our study appears to be the first to fully align the population trajectories of all breeding sites in fully comparable manner. Our trend analyses show that generally, most sites declined between the 1970s (or earlier for some sites) and 1990s, but that subsequently the breeding sites in the Indian Ocean stabilised (the Kerguelen archipelago) or started to increase (Crozet and Prince Edward Islands). In contrast to these Indian Ocean populations, the South Georgia (Islas Georgias del Sur)[1] population of Wandering Albatrosses has continued to decline, albeit at an increasingly slower rate (Mackley *et al.* 2025). These contrasts are remarkable especially when considering that the most rapidly increasing population at the most rapidly increasing site (Prince Edward Islands, which includes Marion) still has invasive mice (Mus musculus) that are known to predate on chicks and even adults (Jones & Ryan 2009, Connan *et al.* 2024). Understanding the underlying drivers of the divergent Wandering Albatross population trends between ocean basins appears a global research priority as it may pave the way for improving marine policy on bycatch mitigation implementation to benefit seabirds globally. To date, some studies have investigated the drivers of differing population trends within archipelagos (e.g., Warwick-Evans *et al.* 2025, 2026) but between-ocean basin studies of this nature are yet to be completed and should be encouraged.

4.3.2. Thalassarche *albatrosses*

Contrasting with the *Diomedea* albatrosses, not all *Thalassarche* albatrosses exhibited declines, and some species showed stable or increasing trends. Of the five *Thalassarche* species evaluated, and in fact, of all species evaluated, Grey-headed Albatrosses exhibited the most significant decline, both in terms of absolute (>500,000 mature individuals) and relative numbers (>70% over the last 3 GL). Similar to the Wandering Albatrosses, our results are the first to enable full comparability among all Grey-headed Albatross populations providing crucial insights. Specifically, our results show that while some population indeed appear stable to increasing as well (e.g., Islas Diego Ramirez and Prince Edwards Islands; Robertson *et al.* 2017, Stevens *et al.* 2024) the declines at South Georgia (Islas Georgias del Sur)[1] and Motu Ihupuku are so large that the overall species has suffered dramatic population losses. Just like for Wandering Albatrosses, further evaluating the underlying drivers of the contrasting population trajectories appears key to stemming further decreases at the declining populations.

Campbell Albatrosses also exhibited a concerning relative decline (-51% since the 1970). This decline is not a recent phenomenon (e.g., Moore & Moffat 1990, Waugh *et al.* 1999, Moore 2004) and in the mid-2000s the decline may have temporarily halted (Sagar 2014, Frost 2020). However, a renewed and apparently steeper decline appeared in the 2020s. This increased decline may be inflated due to temporarily low breeding probability (Mischler *et al.* 2025), yet we accounted for this possibility in our modelling framework through adjustments in p^b (Supplementary Material 1). Traditionally, it has been suggested that the decline of Campbell Albatross has been driven by climate-related drivers, though pelagic longline fisheries may have played a key role as well (Waugh *et al.* 1999). The species is not bycaught in large numbers within New Zealand fisheries (Edwards *et al.* 2023), nor does the species appear to overlap excessively with fisheries (though overlap increases during the non-breeding season; Thompson *et al.* 2021). However, misidentification with the abundant Black-browed Albatross (*T. melanophris*) may be obscuring fisheries-related drivers of the observed decline. Given the potential for exacerbated declines in recent years and continuing uncertainty on the underlying drivers of the observed loss, further study of this species appears a priority.

Salvin's Albatross also exhibited a severe decline (-33% since 1970). Yet, despite our best efforts to align challenging, historic records (e.g., Robertson & van Tets 1982) with modern counts, the extent of loss remains surrounded by uncertainty, and the population on Moutere Hauriri continues to show considerable interannual variation. The latest count (2026) on the smaller Western Chain islets also suggested further concerning declines. The underlying causes of the decline of this species are uncertain. Neither breeding site suffers from the presence of any invasive species, and the species is not commonly caught in pelagic longline fisheries, unlike the *Diomedea* albatrosses, but can be subject to considerable trawl bycatch (Adasme *et al.* 2019, Edwards *et al.* 2023, Anon. 2025). It is possible that the use of net-sonde cables in New Zealand trawl fisheries in the 1980s and 1990s underpinned the decline of this species, which would align with a reduced rate of the decline following the ban of these cables in New Zealand in 1992 (Bartle 1991, Adasme *et al.* 2019). However, most of the observed mortalities associated with net-sonde cables in the New Zealand trawl fleet were assigned to the White-capped Albatrosses (Bartle 1991), so uncertainty remains prominent.

White-capped Albatrosses are the most abundant albatross species in New Zealand, and their trends have been subject to considerable debate in the literature, with some analyses showing non-significant declines, while others indicated a severe population decline (e.g., Walker *et al.* 2021 vs. Baker *et al.* 2023). Our analyses provide a more nuanced understanding of the fate of this species, suggesting that the species may recently have started to decline, i.e., a middle ground between previous analyses. However, the last count of this species at its stronghold on Disappointment Island dates to 2018 (Baker *et al.* 2023). Mortality estimates in New Zealand domestic fisheries alone exceed 2,600 adults annually

(Edwards *et al.* 2023) to which several thousand mortalities caused by high seas fisheries should be added (e.g., Rollinson *et al.* 2017, Peatman *et al.* 2019, Anon. 2025). Therefore, reigniting previous efforts to quantify the species' abundance (next to demographics; Parker *et al.* 2024) appears a global research priority. Contrasting with its two sister species, the small Chatham Albatross population has remained remarkably stable for the entire ~60-year timeseries available, indicating that this species is potentially at carrying capacity at its sole breeding ground, and is not too impacted by bycatch in New Zealand (Edwards *et al.* 2023) or South American waters (Quinones *et al.* 2022).

Finally, of the *Thalassarche* species analysed here, Buller's Albatross has shown the most positive trend, but this trend, at least in recent years, appears largely driven by the Chatham Island breeding sites. In contrast, the Tini Heke population has apparently been in decline since the mid-2000s, resulting a surprisingly similar population trajectory to both Antipodean Albatross populations, albeit less severe. Adult survival continues to be low for this population (Sagar *et al.* 2025) and thus continuing declines appear likely. The contrasting trend of the Tini Heke population may suggest challenges associated with pelagic longline bycatch in the Tasman Sea high seas (where this population forages more prominently than the other populations; Fischer *et al.* 2023), yet recent analyses suggest that trawler impacts within New Zealand waters cannot be ignored (Rutter *et al*. 2026).

4.3.3. Phoebastria *albatrosses*

The only *Phoebastria* species analysed here was Waved Albatross and our results build on insights provided by previous work (e.g., Anderson *et al*. 2002, 2008). This species is listed as CR and is consequently of high conservation concern. However, the trend analyses presented here provide some further nuance. The species does not appear to exhibit the catastrophic decline projected by Anderson *et al* (2008) (-4% per annum) and instead showed a shallow decline, potentially towards stabilisation. It should be noted however that the population trajectory of this species continues to be shrouded in a large amount of uncertainty despite our best efforts to include additional surveys. As such, further work, ideally mirroring the methods employed by previous surveys while taking our monitoring and reporting recommendations (above) into account are strongly encouraged as the species may be at considerable risk from bycatch (Awkerman *et al*. 2006).

4.3.4. Procellaria *petrels*

A surprising result of our analyses were the consistent, long-term increases of the two endemic *Procellaria* species. Both species are considered at a reasonably high risk of suffering population level impacts due to bycatch (Edwards *et al.* 2023), and thus these trends contradict current perceptions. While trends for both species evaluated here remain surrounded by considerable uncertainty, there is supporting evidence for increasing trends in both species. Adult survival is reasonably high, particularly Westland Petrels (Waugh *et al.* 2015). Additionally, for both species, terrestrial threats appeared to have decreased over the last 50 years. Westland Petrel chicks were harvested for subsistence at least until the mid-1950's (Wilson 2016, Waugh & Wilson 2017), a practice which since has ceased. Black Petrels may have benefited from improved predator control on Aotea and the complete eradication of invasive predators from Hauturu (Veitch 2001, Wade & Veitch 2019). Furthermore, considerable efforts have been invested into reducing seabird bycatch in demersal longline fisheries, at least in the north of New Zealand, to relieve pressure on Black Petrels (Goad & Williamson 2015, Goad & Debski 2017). Consequently, it is not unlikely that the presented trends are a true reflection of reality, but ideally, our abundance trends are corroborated with supporting demographic studies (e.g., an update of Zhang *et al.* 2020) and further, high-quality abundance estimates in the near future, to reduce uncertainty.

4.4 Conservation status assessments

ACAP IG Trends workshop 24 May outcome:
Modes and densities are poor statistical approaches for identifying suitable IUCN Red List criteria and instead medians, or slightly more precautionary percentile approaches (e.g., 40% or 45% percentiles) should be considered.

4.4.1. IUCN Red List

Our work resulted in the most detailed IUCN Red List assessments of these 13 ACAP species to date. Our evaluations against Criterion A4 (and A2 where relevant) presented here largely align with the existing categories the species already have been assigned to (BirdLife International 2025), especially when considering that evaluation against the other criteria is still required. Evaluation against these remaining criteria will likely result in the continued listing of species as threatened even if our evaluations here allocate them in NT or LC (e.g., Chatham Albatrosses, Waved Albatrosses, and Black Petrels, which remain highly localised breeders and therefore likely qualify as at least VU against Criterion D). Follow-up work to complete IUCN Red List assessments for ACAP species is strongly encouraged. Our analyses should provide the baseline for such assessments, even if they are beyond the scope of the development of the modelling framework presented here. Furthermore, our conservation status assessments through *ACAPT* also bear relevance and opportunities for local Red List assessments (e.g., Roberston *et al.* 2021).

Despite broad alignment with current categories (particularly when considering that evaluations against other criteria are still pending), two species stand out as requiring a likely category change on the IUCN Red List: Grey-headed Albatross and Westland Petrel. Our analyses suggest that Grey-headed Albatross should be uplisted from EN to CR due to the ongoing, severe declines over the last GL/last 20 years at several key breeding sites, most pressingly South Georgia (Islas Georgias del Sur)[1] and Motu Ihupuku. While these declines are not a novel insight (e.g., Waugh *et al.* 1999, Moore 2004, Poncet *et al.*, 2006, 2017, Mackley *et al.* 2025), more positive growth rates at other breeding sites (e.g., Robertson *et al.* 2017, Cleeland *et al.* 2021, Stevens *et al.* 2024) were considered to offset some of these declines. Our analyses here are the first to ensure full comparability of counts among sites and years, ensuring a holistic evaluation of the species' threat status, revealing that despite smaller breeding sites showing stability or even increases, this circumpolar breeder warrants listing as CR. Westland Petrels on the other hand, may warrant downlisting from EN to VU. Our analyses suggest that this species has been showing long term increases and the argument breeding site fragmentation under Criterion B (Waugh & Wilson 2017) may not be suitable for a wide-ranging species like a seabird. Yet, the species remains a highly restricted breeder (Waugh *et al.* 2020), and thus likely qualifies as VU against other criteria (e.g., D).

Other ACAP species may also warrant category changes following our evaluations against Criterion A2 and A4 (including the CR Waved Albatross), yet those changes are less clear-cut due to the explicit incorporation of uncertainty in our analyses, and thus the statistical thresholds for category assignments in the face of uncertainty require further discussion. Uncertainty is common in abundance data of virtually all species, and ACAP species are no exception. We have incorporated this uncertainty explicitly in our modelling framework and therefore our species evaluations against IUCN Red List criteria can be uncertain. As per the IUCN Red List guidelines (IUCN 2012), we here explicitly specify the range of possible criteria. Still, a single category must be chosen and the basis for the decision, which must be precautionary as well as credible, have to be documented (IUCN 2012). We explored three statistical approaches for assigning IUCN Red list categories: medians, modes, and density-based assignments. Our work has shown that the different statistical approaches are in agreement more often than not, but still differ regularly and thus the choice of stastical approach will have consequences for category assignments. Density-based approaches may not be the most suitable approach, even if it is the approach used in *JARA* (Winker *et al.* 2020, Sherley *et al.* 2020a,b). The combination of a left-hand zero-bound, the lack of a right-hand bound, and uneven categories renders assigning categories using a probabilistic density-based approach suboptimal, particularly when uncertainty abounds (Supplementary Material 8). Instead, a simpler central-tendency approach may be more robust, but whether a more, or less precautionary approach (e.g., mode vs. median, respectively) is preferred, depends on risk attitude. The median may be a more statistically defensible approach, yet it risks being influenced towards lower categories if many iterations fall within LC (e.g., see Southern Royal Albatross). The mode on the other hand may be more precautionary but can be influenced by the opposite effect of uncertainty within a one-sided bounded system: zero-bounds unrealistically inflating allocation of iterations as CR (e.g., Waved Albatross and Westland Petrel). While the interpretation of the uncertainty and the recommended category, and therefore the risk attitude, ultimately is at the discretion of form Red List assessors (IUCN 2012), ACAP does appear to have a role to play and should consider developing policies on how to recommend category assignments in the face of uncertainty. Once such policies are developed, our standardised abundance processing and trend modelling framework may be expanded to enable standardised conservation assessments of seabirds beyond ACAP species, adding to the work completed on Benguela Current species (e.g., Sherley *et al.* 2019, 2024), and ultimately paving the way towards comparable assessments of large communities as completed for other marine megafauna (e.g., Pacoureau *et al.* 2021, Dulvy *et al.* 2024).

4.4.2. ACAP High Priority Populations

Our results resulted in the most detailed and standardised evaluation of potential ACAP High Priority Populations to date. However, our results revealed potential challenges with the current population trend threshold for High Priority Populations assignments. Specifically, a population can only be considered if it exhibits a population growth rate of $\lambda < 0.970$ over the last 20 years. This threshold is arguably extremely high, as a sustained λ at this rate would translate into a population loss of ~46% over 20 years, and if it was extended to 3 GL for these long-lived species, it would equate to a devastating population loss of 77-94%. Despite this high bar, nine populations are perceived to have met this threshold leading to their listing as High Priority Populations (ACAP 2025b). For example, the growth rate of Antipodean

Albatross at Moutere Mahue was estimated at 0.943 (0.928-0.956) based on an integrated population model (Richard *et al.* 2024), the growth rate of Waved Albatross at Espanola Island was estimated at 0.962 (0.961-0.962) based on a matrix model (Anderson *et al.* 2008), and the growth rate of Tristan Albatrosses (*D. dabbanena*) at Gough Island under current conditions was estimated at 0.989 (0.973-1.005) and under worsening conditions at 0.939 (0.924-0.954) using an integrated population model (Oppel *et al.* 2022). Other populations have been assigned as High Priority Populations through abundance-based evaluations of population growth. Yet these abundance-based evaluations usually do not explicitly account for uncertainty (e.g., $\lambda = 0.970$ for Wandering Albatrosses at Bird Island; Poncet *et al.* 2006, 2017). Contrasting with previous evaluations, our analyses revealed that when using abundance-based assessments that explicitly account for uncertainty, only Grey-headed Albatrosses at South Georgia (Islas Georgias del Sur)[1] would qualify for listing as an High Priority Population out of the case studies evaluated here. This breeding site is currently listed as a High Priority Population (ACAP 2025b) supporting previous assessments (e.g., Poncet *et al.* 2006, 2017). However, two other existing High Priority Populations did not exhibit the qualifying population growth rate of <0.970: Antipodean Albatross at Moutere Mahue and Wandering Albatross at South Georgia (Islas Georgias del Sur)[1].

Considering our results and how they contrast with previous evaluations, including those of current ACAP High Priority populations, the current λ criterion threshold may be too high to effectively identify all relevant High Priority Populations, at least when explicitly accounting for uncertainty in abundance assessments. There are several potential explanations for the apparent mismatch. Firstly, due to the sensitivity of demographic assessments to a range of assumptions (in particular, immigration and emigration; Paquet *et al.* 2021), population growth estimates derived from these analyses may be too pessimistic. On the other hand, however, albatross and petrel populations can obscure trends due to their complex population structure, resulting in limited to absent signs of declines in abundance at breeding sites while total abundance is indeed in decline (e.g., Oppel *et al.* 2022, Sagar *et al.* 2025). Considering these challenges, the λ criterion threshold for identifying ACAP High Priority Populations could be adjusted to include two tiers: A) $\lambda < 0.970$ over the last 20 years based on a demographic assessment, or B) median $\lambda < 0.990$ over the last 20 years based on a standardised abundance assessment that explicitly accounts for uncertainty, such as the analyses presented here. For the populations assessed here, this two-tiered approach would mean that Antipodean Albatrosses on Moutere Mahue and Wandering Albatrosses on South

Georgia (Islas Georgias del Sur)[1] continue to be recognised as ACAP High Priority Populations (Walker & Elliott 2017), but that Grey-headed Albatrosses on Motu Ihupuku and Northern Royal Albatrosses on Motuhara are also recognised as such given the steep decline this population faced over the last 20 years.

Further discussions within ACAP are required to improve the ACAP High Priority Population assessment process. Specifically, ACAP will need to identify 1) the statistical cut-off that the threshold(s) could apply to (e.g., median, mode, or density; Supplementary Material 9), 2) the appropriate frequency (e.g., every 5-10 years) over which all ACAP populations are reviewed against the High Priority Population criteria to ensure the list remains current, and 3) what a process for de-listing an ACAP High Priority Population should look like. Finally, the criterion stipulating that a population should represent 10% of the global population also requires reviewing and, potentially, the inclusion of a timeframe qualifier (e.g., similar to the one we applied here, stipulating that a population meets the criterion if it held ≥10% at any point during the available timeseries). Without such a qualifier, there may be a risk that population declines substantially up to <10% of the world population and subsequently no longer qualifies listing as a High Priority Population (e.g., Grey-headed Albatrosses as Motu Ihupuku). Taking all these considerations into account however, our analyses provide a promising pathway to ensure more coherency within the list of ACAP High Priority Populations and therefore improve the communication around priorities of the Agreement and the conservation of these species.

ACAP IG Trends workshop 24 May outcome:
Further work over the intersessional period is required to formulate the most suitable, standardised approach for identifying ACAP High Priority Populations.

6. CONCLUSION

We presented the most comprehensive abundance assessment of albatrosses and petrels to date. Our work built upon decades of fieldwork across the most remote and inhospitable islands on the planet, representing the investment of generations of scientists into the understanding of these seabirds. We coined updated definitions, designated an improved data structure, designed a novel data processing framework, and developed a flexible trend model implemented through our bespoke R package. These combined advancements resulted in trend estimates and assessments of conservation status comparable across space, time, and species, illuminating over half a century of seabird abundance change. Our work illustrated and quantified the concerning, long-term, and ongoing declines affecting NZ *Diomedea*, Grey-headed, Campbell, and Salvin's Albatrosses, but also stable trends or recent increases of other, less impacted, species that appeared to have benefited from decades of conservation investments (e.g., predator eradications and improvements in fisheries bycatch mitigation). These estimates provide a path towards improved monitoring, reporting, research prioritisations, further expansions of the analyses presented here, and investigations into the underlying causes of uncovered trends. Most importantly however, our work presents the opportunity to both communicate the current fate of these species and to develop and enforce strategies that reverse declines to once again ensure that all albatrosses and petrels species thrive.

7. REFERENCES

Abraham E, Richard Y, Walker N, Gibson W, Daisuke O, Tsuji S, Kerwath S, Winker H, Parsa M, Small C & Waugh S. 2019. Assessment of the risk of surface longline fisheries in the Southern Hemisphere to albatrosses and petrels for 2016. 13th Meeting of the Ecologically Related Species Working Group of the Commission for the Conservation of Southern Bluefin Tuna (CCSBT).

ACAP 2024. Report of the Taxonomy Working Group. Fourteenth meeting of the Advisory Committee, Lima, Peru. Agreement on the Conservation of Albatrosses and Petrels.

ACAP. 2009. ACAP species assessment: Southern Royal Albatross *Diomedea epomophora*. https://www.acap.aq/. Agreement on the Conservation of Albatrosses and Petrels. Accessed 1 December 2025.

ACAP. 2025a. Agreement on the Conservation of Albatrosses and Petrels – Data Portal. https://data.acap.aq/. Agreement on the Conservation of Albatrosses and Petrels. Accessed 1 December 2025.

ACAP. 2025b. ACAP Priority Conservation Actions for 2026-2028. Eight Meeting of Parties, Dunedin, New Zealand. Agreement on the Conservation of Albatrosses and Petrels.

ACAP. 2025c. Agreement on the Conservation of Albatrosses and Petrels: Report of the eighth session of the meeting of parties. Dunedin, New Zealand. Agreement on the Conservation of Albatrosses and Petrels.

Adasme LM, Canales CM & Adamse NA. 2019. Incidental seabird mortality and discarded catches from trawling off far southern Chile (39-57°**S).** ICES Journal of Marine Science 76: 848-858.

Anderson DJ, Huyvaert KP, Awkerman JA, Proano CB, Milstead WB, Jimenez-Uzcategui G, Cruz S & Grace JK. 2008. Population status of the critically endangered waved albatross Phoebastria irrorata, 1999-2007. Endangered Species Research 5: 185-192.

Anon. 2025. CCSBT collaborative risk assessment for seabird bycatch with surface longlines in the Southern Hemisphere 2025. CCSBT, Canberra, Australia.

Armstrong DP, Parlato EH, Egli B, Dimond WJ, Berggren A, McCready M, Parker KA & Ewen JG. 2021. Capturing the dynamics of small populations: a retrospective assessment using long-term data for an island reintroduction. Journal of Animal Ecology 90: 2915-2927.

Attard MRG, Phillips RA, Oppel S, Bowler E & Fretwell PT. 2025. Feasibility of using very high-resolution satellite imagery to monitor Tristan albatrosses *Diomedea dabbenena* on Gough Island. Endangered Species Research 56: 187-199.

Awkerman JA, Huyvaert JP, Mangel J, Shigueto JA & Anderson DJ. 2006. Incidental and intentional catch threatens Galapagos waved albatross. Biological Conservation 133: 483-489.

Awkerman JA, Huyvaert KP & Anderson DJ. 2005. Mobile incubation in Waved Albatross (*Phoebastria irrorata*): associated hatching failure and artificial mitigation. Avian Conservation and Ecology 1: 1.

Awkerman JA, Westbrock MA, Huyvaert KP & Anderson DJ. 2007. Female-biased sex ratio arises after parental care in the sexually dimorphic Waved Albatross (*Phoebastria irrorata*). The Auck 124: 1336-1346.

Baker CS & Clapham PJ. 2004. Modelling the past and future of whales and whaling. Trends in Ecology and Evolution 19: 365-371.

Baker GB, Hedley G, Cunningham R & Waugh S. 2020. Estimated population size of the Westland Petrel 2007-2011. New Zealand Aquatic Environment and Biodiversity Report No. 242. Fisheries New Zealand, Wellington, New Zealand.

Baker GB, Jensz K, Bell M, Fretwell PT & Phillips RA. 2017. Seabird Population Research, Chatham Islands 2016/17 aerial photographic survey. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Jensz K, Cunningham R, Robertson G, Sagar P, Thompson TR & Double MC. 2023. Population assessment of White-capped Albatross *Thalassarche steadi* in New Zealand. Emu – Austral Ornithology 123: 60-70.

Bartle JA. 1991. Incidental capture of seabirds in the New Zealand subantarctic squid trawl fishery, 1990. Bird Conservation International 1: 351-359.

Bell E, Lamb S, Ray S. 2022. Population trends and breeding population size of black petrels (*Procellaria parkinsoni*) – 2020/2021 operational report. New Zealand Aquatic Environment and Biodiversity Report No. 280. Fisheries New Zealand, Wellington, New Zealand.

Bell EA, Lamb S & Ray S. 2025. Key demographic parameters and population trends of tākoketai/black petrels (*Procellaria parkinsoni*) on Aotea/Great Barrier Island: 2024/25. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell JB, Fischer JH, Carneiro APB, Griffiths S, Bielli A, Jimenez S, Oppel S, Phillips RA, Wade HM, Yates O & Reeves SA. 2025. Evaluating the effectiveness of seabird mitigation measures for pelagic longlines in the South Atlantic. Biological Conservation 302: 110981.

Bell M. 2023. Motuhara seabird research: field trip report Dec 2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2026. Seabird research – Chatham Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell MD, Bell DJ & Boyle DP. 2017. Chatham Island Mollymawk research on Te Tara Koi Koia: November 2016. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bird JP, Martin R, Akcakaya HR, Gilroy J, Burfield IJ, Garnett ST, Symes A, Taylor J, Sekercioglu CH & Butchart SHM. 2020. Generation lengths of the world's birds and their implications for extinction risk. Conservation Biology 34: 1252-1261.

BirdLife International. 2025. BirdLife International Data Zone. BirdLife DataZone. Accessed 1 December 2025.

Brown DA, Cox FS & Fergus AJ. 2022. The eradication of Campbell Island sheep and subsequent ecological responses. New Zealand Journal of Ecology 46: 3483.

Buckland ST, Anderson DR, Burnham KP, Laake JL, Borchers DL & Thomas L. 2001. Introduction to distance sampling: estimating abundance of biological populations. Oxford University Press.

Carneiro APB *et al.* 2020. A framework for mapping the distribution of seabirds by integrating tracking, demography and phenology. Journal of Applied Ecology 57: 514-525.

Ceballos G, Ehrlich PR, Barnosky AD, Garcia A, Pringle PM & Palmer TM. 2015. Accelerated modern human-induced species losses: Entering the sixth mass extinction. Science Advances 1: 1400253.

Che-Castaldo C, Jenouvrier S, Youngflesh C, Shoemaker KT, Humphries G, McDowall P, Landrum L, Holland MM, Li Y, Ji R & Lynch HJ. 2017. Pan-Antarctic analysis aggregating spatial estimates of Adelie penguin abundance reveals robust dynamics despite stochastic noise. Nature Communications 8: 832.

Cleeland JB, Pardo D, Raymond B, Terauds A, Alderman R, McMahan CR, Phillipps RA, Lea MA & Hindell MA. 2020. Introduced species and extreme weather as key drivers of reproductive output in three sympatric albatrosses. Scientific Reports 10: 8199.

Cleeland JB, Pardo D, Raymonds B, Tuck GN, McMahon CR, Phillips RA, Alderman R, Lea MA & Hindell MA. 2021. Disentangling the influence of three major threats on the demography of an albatross community. Frontiers in Marine Science 8: 578144.

Connan M, Jones CJ, Risi MM, Smyth LK, Oppel S, Perold V, Stevens KL, Daling R & Ryan PG. 2024. First evidence of mouse predation killing adult great albatrosses. Biological Invasions 26: 25-31.

Cooper J, Baker GB, Double MC, Gales R, Papworth W, Tasker ML & Waugh SM. 2006. The agreement on the conservation of albatrosses and petrels: rationale, history, progress and the way forward. Marine Ornithology 34: 1-5.

CSP. 2025. Conservation Services Programme Annual Plan 2025/26. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Cuthbert RJ & Sommer ES. 2004. Population size and trends of four globally threatened seabirds at Gough Island, South Atlantic Ocean. Marine Ornithology 32: 97-103.

Dias MP, Martin R, Pearmain EJ, Burfield IJ, Small C, Phillips RA, Yates O, Lascelles B, Borboroglu PG & Croxall JP. 2019. Threats to seabirds: a global assessment. Biological Conservation 237: 525-537.

Dilley BJ, Schoombie S, Schoomie J & Ryan P. ‘Scalping’ of albatross fledglings by introduced mice spreads rapidly at Marion Island. Antarctic Science 28: 73-80.

Dulvy NK, Pacoureau N, Matsushiba JH, Yan HF, Vanderwright WJ, Rigby CL, Finucci B, Sherman CS, Jabado RW, Carlson JK, Carlson JK, Pollom RA, Charvet P, Pollock CM, Hilton-Taylor C & Simpfendorfer CA. 2024. Ecological erosion and expanding extinction risk of sharks and rays. Science 386: 6726.

Edgar GJ, Bates AE, Krueck NC, Baker SC, Stuart-Smith RD & Brown CJ. 2024. Stock assessment models overstate sustainability of the world’s fisheries. Science 385: 860-865.

Edwards CTT, Peatman T, Goad D & Webber DN. 2023. Update to the risk assessment for New Zealand seabirds. New Zealand Aquatic Environment and Biodiversity Report No. 314. Fisheries New Zealand, Wellington, New Zealand.

Elliott G, Waker K, Rexer-Huber K, Tinnemans J, McDonald A, Rawlence T & Parker G. 2025. Gibson’s wandering albatross: drone-based population estimate, demography and at-sea distribution. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Walker K, Parker G & Rexer-Huber K. 2016. Gibson's wandering albatross census and population study 2015/16. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Fischer JH, Bell M, Frost P, Sagar PM, Thompson DR, Middlemiss KL, Debski I & Taylor GA. 2023. Year-round GLS tracking of Northen Buller's Albatross and comparison with Southern Buller's Albatross. Conservation Services Programme, Wellington, New Zealand.

Fischer JH, Carneiro A, Rowley O & Debski I. 2024. An update on the New Zealand large-scale monitoring and tracking programme with improved insights into trends and distribution. Eight meeting of the Population and Conservation Status Working Group, Agreement for the Conservation of Albatrosses and Petrels, Lima, Peru.

Fischer JH, Taylor GA, Cole R, Debski I, Armstrong DP & Wittmer HU. 2020. Population growth estimates of a threatened seabird indicate necessity for additional management following invasive predator eradications. Animal Conservation 23: 94-103.

Fischer JH, Wittmer HU, Kenup CF, Parker KA, Cole R, Debski I, Taylor GA, Ewen JG, Armstrong DP. 2022. Predicting harvest impact and establishment success when translocating highly mobile and endangered species. Journal of Applied Ecology 59: 2071-2083.

Fischer JH, Wittmer HU, Taylor GA, Debski I & Armstrong DA. 2021. Preparing for translocations of a Critically Endangered petrel through targeted monitoring of nest survival and breeding biology. Oryx 55: 564-572.

Fisher HI. 1966. Aerial census of Layan Albatrosses breeding on Midway Atoll in December, 1962. The Auk 83: 670-673.

Frederick PC, Heath JA, Bennetts R & Hafner H. 2006. Estimating nests not present at the time of breeding surveys: an important consideration in assessing nesting populations. Journal of Field Ornithology 77: 212-219.

Fretwell PT, Scofield P & Phillips RA. 2017. Using super-high resolution satellite imagery to census threatened albatross. Ibis 159: 481-490.

Frost PGH, Baker GB, Fischer JH & Sagar PM. 2025. Population survey of Southern Buller's Albatross *Thalassarche bulleri bulleri* on the Solander Islands | Hautere, March 2024. Marine Ornithology 53: 361-371.

Frost PGH. 2017. Aerial census of Northern Royal Albatross (*Diomedea sanfordi*) fledglings on Rangitatahi (The Sisters) and Motuhara (Fortyfours), July 2017. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2020. Status of Campbell Island and Grey-headed Mollymawks on the Northern Coast of Campbell Island, November 2019. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2024. Aerial survey of Northern Royal Albatross | Toroa (*Diomedea sanfordi*) populations on the Chatham Islands, December 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Froy H, Phillips RA, Wood AG, Nussey DH & Lewis S. 2013. Age-related variation in reproductive traits in the wandering albatross: evidence for terminal improvement following senescence. Ecology Letters 16: 642-649.

Fu D & Sagar P. 2016. The 2014 demographic assessment of the Snares islands population of Southern Buller's albatross (*Diomedea bulleri bulleri*). New Zealand Aquatic Environment Biodiversity Report 165. Ministry for Primary Industries, Wellington, New Zealand.

Genovart M, Arcos MJ, Alvarez D, McMinn M, Meier Rm Wynn RB, Guilford T & Oro D. 2016. Demography of the critically endangered Balearic Shearwater: the impact of fisheries and time to extinction. Journal of Applied Ecology 53: 1158-1168.

Gianuca D, Votier SC, Pardo D, Wood AG, Sherley RB, Ireland L Choquet R, Townley S, Forcada J, Tuck GN & Phillips RA. 2019. Sex-specific effects of fisheries and climate on the demography of sexually dimorphic seabirds. Journal of Animal Ecology 88: 1366-1378.

Goad D & Debski I. 2017. Bird-scaring line designs for small longline vessels. Eighth Meeting of the Seabird Bycatch Working Group, Agreement on the Conservation of Albatrosses and Petrels, Wellington, New Zealand.

Goad D & Williamson J. 2015. Improving and documenting seabird bycatch mitigation practices in the North Eastern New Zealand longline fishery. Conservation Services Programme, Wellington, New Zealand.

Hayes M, Gray PC, Harris G, Sedgwick WC, Crawford VD, Chazal N, Crofts S & Johnston DW. 2021. Drones and deep learning produce accurate and efficient monitoring of large-scale seabird colonies. Ornithological Applications 123: duab022.

Hays GC, Schofield G, Papazekou M, Chatzimentor A, Katsanevakis S & Mazaris AD. 2024. A pulse check for trends in sea turtle numbers across the globe. iScience 27: 109071.

Hemming V, Burgman MA, Hanea AM, McBride MF & Wintle BC. 2018. A practical guide to structured expert elicitation using the IDEA protocol. Methods in Ecology & Evolution 9: 169-180.

Imber MJ. 1987. Breeding ecology and conservation of the black petrel (*Procellaria parkinsoni*). Notornis 34: 19–39.

IUCN. 2012. IUCN Red List categories and criteria, version 3.1, second edition. IUCN Species Survival Commission, IUCN, Cambridge, UK.

Jimenez S, Forselledo R, Mas F, Domingo A. 2026. Preliminary analysis of the overlap of southern royal albatross with fisheries in the southwest Atlantic. Joint thirteenth meeting of the Seabird Bycatch Working Group and the ninth meeting of the Population and Conservation Status Working Group, Agreement on the Conservation of Albatrosses and Petrels, Swakopmund, Namibia.

Jimenez S, Phillips RA, Brazeiro A, Defeo O & Domingo A. 2014. Bycatch of great albatrosses in pelagic longline fisheries in the Southwest Atlantic: Contributing factors and implications for management. Biological Conservation 171: 9-20.

Jones HP, Appoo J, Benkwitt CE, Borrelle SB, Dunn RE, Epstein HE, Fowlke LA, Holmes ND, Jeannot LL, Malhi Y, Ranking LL, Steibl S, Wails CN, Will DJ & Graham NAJ. 2025. The circular seabird economy is critical for oceans, islands and people. Nature Reviews Biodiversity 1: 689-702.

Juan-Jorda MJ, Murua H, Arrizablaga H, Merino G, Pacoureau N & Dulvy NK. 2022. Seventy years of tunas, billfishes, and sharks as sentinels of global ocean health. Science 378: 6620.

Lewis S, Nussey DH, Wood AG, Croxall JP & Phillips RA. 2012. Intrinsic determinants of a population trend in timing of breeding in the wandering albatross. Oikos 121: 2061-2071.

Mackley EK, Poncet S, Black A, Black J, Floyd K, Hall RM, Holmes EE, Manthorpe SJ, Passfield K, Bennison A & Phillips RA. 2025. Endurance or extinction: long-term declines in albatrosses at South Georgia highlight threats from South Atlantic fisheries and climate change. Endangered Species Research 57: 437-451.

Mischler C, Moore P, Thompson H, Hamilton K & Pryde M. 2025. Campbell Island Seabird Research. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Moore PJ & Moffat RD. 1990b. Mollymawks on Campbell Island. Science & Research internal report 59. Department of Conservation, Wellington, New Zealand.

Moore PJ, Larsen EJ, Chateris M & Pryde M. 2012. Southern Royal Albatross on Campbell Island/Motu Ihupuku; Solving a band injury problem and population survey 2004-08. DOC Research and Development Series 333, Department of Conservation, Wellington, New Zealand.

Moore PJ. 2004. Abundance and population trends of mollymawks on Campbell Island. Science for Conservation 242. Department of Conservation, Wellington, New Zealand.

Morten JM, *et al.* 2025. Global marine flyways identified for long-distance migrating seabirds from tracing data. Global Ecology and Biography 34: e70004.

Oppel S, Clark BL, Risi MM, Horswill C, Converse SJ, Jones CW, Osborne AM, Stevens K, Perold V, Bond AL, Wanless RM, Cuthbert R, Cooper J & Ryan PG. 2022. Cryptic population decrease due to invasive species predation in a long-lived seabird supports need for eradication. Journal of Applied Ecology 59: 2059-2070.

Pacoureau N, Rigby CL, Kyne PM, Sherly RB, Winker H, Carlson JK, Fordham SV, Barreto R, Fernando D, Francis MP, Jabado RW, Herman KB, Liu KM, Marshall AD, Pollom RA, Romanov EV, Simpfendorfer CA, Yin JS, Kindsvater HK & Dulvy NK. 2021. Half a century of global decline in oceanic sharks and rays. Nature 589: 567-571.

Pacoureau N, Carlson JK, Kindsvater HK, Rigby CL, Winker H, Simpfendorfer CA, Charvet P, Pollom RA, Barreto R, Sherman S, Talwar BS, Skerrit DJ, Sumaila RS, Matsushiba JH, VanderWright WJ, Yan HF & Dulvy NK. 2023. Conservation successes and challenges for wide-ranging sharks and rays. PNAS 120: e2216891120.

Paleczny M, Hammill E, Karpouzi V & Pauly D. 2015. Population trend of the world's monitored seabirds, 1950-2010. PLoS ONE 10: e0129342.

Paquet M, Knape J, Arlt D, Forslund P, Part T, Flagstad O, Jones CG, Nicoll MAC, Norris K, Pemberton JM, Sand H, Svenson L, Tatayah V, Wabakken P, Wikenros C, Akesson M & Low M. 2021. Integrated population models poorly estimate the demographic contribution of immigration. Methods in Ecology and Evolution 12: 1899-1910.

Pardo D, Forcada J, Wood AG, Tuck GN, Ireland L, Pradel R, Croxall JP & Phillips RA. 2017. Additive effects of climate and fisheries drive ongoing declines in multiple albatross species. PNAS 114: E10829-10837.

Parker GC & Rexer-Huber K. 2020. Guidelines for designing population surveys of burrowing petrels. Agreement for the Conservation of Albatrosses and Petrels.

Parker GC, Osborne J, Sagar R, Schultz H & Rexer-Huber K. 2024. White-capped albatross population study, Disappointment Island 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Peatman T, Abraham E, Ochi D, Webber D & Smith N. 2019. Project 68: estimation of seabird mortality across the WCPFC Convention Area. Fifteenth regular session of the Scientific Committee of the Western and Central Pacific Fisheries Commission, Pohnpei, Federated States of Micronesia.

Petersen SL, Honig MB, Ryan PG & LG Underhill. 2010. Seabird bycatch in the pelagic longline fishery off Southern Africa. African Journal of Marine Science 31: 191-204.

Phillips RA, Gales R, Baker GB, Double MC, Favero M, Quintana F, Tasker ML, Weimerskirch H, Uhart M & Wolfaardt A. 2016. The conservation status and priorities for albatrosses and large petrels. Biological Conservation 201: 169-183.

Plummer M. 2003. JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. Proceedings of the 3rd international workshop on distributed statistical computing. 124: 1-10.

Poncet S, Robertson G, Phillips RA, Lawton K, Phalan B, Trathan PN & Croxall JP. 2006. Status and distribution of wandering, black-browed and grey-headed albatrosses breeding at South Georgia. Polar Biology 29: 772-781.

Poncet S, Wolfaardt AC, Black A, Browning S, Lawton K, Lee J, Passfield K, Strange G & Phillips RA. 2017. Recent trends in numbers of wandering (Diomedea exulans), black-browed (Thalassarche melanophris) and grey-headed albatrosses (T. chrysostoma) breeding at South Georgia. Polar Biology 40: 1347-1358.

Quinones J, Romero C, Mangel JC, Alfaro-Shigueto J, Moreno C & Zavalaga C. 2022. At-sea surveys reveal new insights of fine-scale behaviour and foraging behaviour of Chatham Albatrosses (Thalassarche eremita) in central southern Peru. Notornis 69: 72-78.

R Core Team. 2024. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna Austria.

Rexer-Huber K, Parker G, Elliott G & Walker K. 2023. Antipodes Island white-chinned petrel population size and survival study setup. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Parker G, Sager P & Thompson D. 2021. Salvin's Albatross breeding dates and productivity: nest-camera analysis. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Walker KJ, Elliott GP, Baker GB, Debski I, Jensz K, Sagar PM, Thompson DR & Parker GC. 2020. Population trends of light-mantled sooty albatross (*Phoebetria palpebrata*) at Adams Island and trial of ground, boat, and aerial methods for population estimates. Notornis 67: 341-355.

Rexer-Huber K, Whitehead E, Parker G, Patterson E, Walker K, Welch J & Elliott G. 2024. Antipodean wandering albatrosses and white-chinned petrels 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Whitehead E, Walker K & Elliott G. 2025. Antipodean wandering albatrosses and white-chinned petrels 2025. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Richard Y, Berkenbusch K, Crawford E, Tornquist M, Walker K, Elliott G, & Tremblay-Boyer L. 2024. Antipodean Albatross multi-threat risk assessment. New Zealand Aquatic Environment and Biodiversity Report No. 332. Fisheries New Zealand, Wellington, New Zealand.

Roast MJ, Martins S, Fernandez-Peralta L, Beaz JC, Diame A, March D, Ouled-Cheikh, Marco A, Gonzales-Solis & Cardona Luis. 2023. Hidden demographic impacts of fishing and environmental drivers of fecundity in a sea turtle population. Conservation Biology 37: e 14110.

Robertson CJR & van Tets GF. 1982. The status of birds at the Bounty Islands. Notornis 29: 311-336.

Robertson CJR. 1991. Questions on the harvesting of Toroa in the Chatham Islands. Science & Research Series 35. Department of Conservation, Wellington.

Robertson G, Moreno CA, Lawton K, Kirkwood D & Valencia J. 2008. Comparison of census methods for Black-browed Albatrosses breeding at Ildefonso Archipelago, Chile. Polar Biology 31: 153-162.

Robertson G, Wienecke B, Suazo CG, Lawton K, Arata JA & Moreno C. 2017. Continued increase in the number of black-browed albatrosses (*Thalassarche melanophris*) at Diego Ramirez, Chile. Polar Biology 40: 1035-1042.

Rodriguez A, Arcos JM, Bretagnolle V, Dias MD, Holmes ND, Louzao M, Provencher J, Raine AF, Ramirez F, Rodriguez B, Ronconi RA, Taylor RS, Bonnaud E, Borrelle SB, Cortes V, Descamps S, Friesen VL, Genovart M, Hedd A, Hodum P, Humpries GRW, Le Corre M, Lebarbenchon C, Martin R, Melvin EF, Montevecchi WA, Pinet P, Pollet IL, Ramos R, Russel JC, Ryan PG, Sanz-Aguillar A, Spatz DR, Travers M, Votier SC, Wanless RM, Woehler E, Chiaradia A. 2019. Future directions in conservation research on petrels and shearwaters. Frontiers in Marine Science 6: https://doi.org/10.3389/fmars.2019.00094

Rollinson DP, Wanless RM & Ryan PG. 2017.Patterns and trends in seabird bycatch in the pelagic longline fishery off South Africa. African Journal of Marine Science 39: 9-25.

Rowley O, Waipoua TA, Debski I, Elliott G, Parker G, Rexer-Huber K, Walker K & Fischer J. 2024. Fine scale overlap analysis of Antipodean and Gibson's albatross with pelagic longline fishing effort. Joint Twelfth meeting of the Seabird Bycatch Working Group and Eight meeting of the Population and Conservation Status Working Group, Agreement for the Conservation of Albatrosses and Petrels, Lima, Peru.

Rutter JD, Carneiro APB, Debski I, Düssler MD, Fischer JH, Parker G, Rexer-Huber K, Rowley O, Sagar P, Schultz H, Simister K, Guilford T, & Davis KJ. 2026. Navigating uncertainty in fine-scale fisheries overlap to assess bycatch risk of a wide-ringing pelagic seabird. Joint meeting of the Ninth Population and Conservation Status Working Group and Thirteenth Seabird Bycatch Working Group, Agreement on the Conservation of Albatrosses and Petrels, Swakopmund, Namibia.

Ryan PG, Jones MGW, Dyer BM, Upfold L & Crawford RJM. 2009. Recent population estimates and trends in numbers of albatrosses and giant petrels breeding at the sub-Antarctic Prince Edward Islands. African Journal of Marine Science 31: 409-417.

Sagar P, Charteris M & Scofield P. 2014. Salvin's albatross population size and survival at the Snares Western Chain. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Rexer-Huber K, Schultz H, Simister K, Thompson D & Parker G. 2025. Population studies of southern Buller's Albatross at Tini Heke | The Snares Islands. Conservation Services Programme, Wellington, New Zealand.

Sagar P. 2014. Population estimates and trends of Campbell and grey-headed albatrosses at Campbell Island. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar PM & Stahl J-C. 2005. Increases in the numbers of breeding pairs in two populations of Buller's Albatross (*Thalassarche bulleri bulleri*). Emu – Austral Ornithology 105: 49-55.

Schaub M & Abadi F. Integrated population models: a novel analysis framework for deeper insights into population dynamics. Journal of Ornithology 152: 227-237.

Scofield P. 2011. Data collection of demographic, distributional, and trophic information on selected seabird species to allow estimation of effects of fishing on populations viability. Ministry of Fisheries, Wellington, New Zealand.

Sherley RB, Crawford RJM, de Blocq AD, Dyer BM, Geldenhuys D, Hagen C, Kemper J, Makhado AB, Pichegru L, Tom D, Upfold L, Visagie J, Waller LJ & Winker H. 2020a. The conservation status and population decline of the African penguin deconstructed in space and time. Ecology and Evolution 10: 8506-8516.

Sherley RB, Crawford RJM, Dyer BM, Kemper J, Makhado AB, Masotla M, Pichegru L, Pistorius PA, Roux JP, Ryan PG, Tom D, Upfold L & Winker H. 2019. The status and conservation of the Cape Gannet *Morus capensis*. Ostrich 90: 335-346.

Sherley RB, Makhado AB, Crawford RJM, Hagen C, Kemper J, Ludynia K, Masolta MJ, McInnes A, Pichegru L, Tom D, Upfold L & Waller LJ. 2024. The African Penguin *Spheniscus demersus* should be considered Critically Endangered. Ostrich 95: 181-187.

Sherley RB, Winker H, Rigby CL, Kyne PM, Pollom Rm Pacoureau N, Herman K, Carlson JK, Yin JS, Kndsvater HK & Dulvy NK. 2020b. Estimating IUCN Red List population reduction: JARA – A decision-support tool applied to pelagic sharks. Conservation Letters 13: e12688.

Stahl JC & Sagar PM. 2006. Behaviour and patterns of attendance of non-breeding birds at the breeding colony in a Buller's Albatross *Thalassarche bulleri* population at The Snares. Notornis 53: 327-338.

Stevens KL, Altwegg R, Connan M & Ryan P. 2024. Population growth of the grey-headed albatross population on Marion Island inferred using three analysis methods. Antarctic Science 36:1-11.

Street PA. 2013. Abundance, survival, and breeding probabilities of the critically endangered Waved Albatross. MSc Thesis. Department of Fish, Wildlife, and Conservation Biology, Colorado State University, USA

Suazo CG, Anguita C, Letelier CG, Martinez A & Quillfeldt P. 2024. The white-capped albatross: a new breeding record from the Diego Ramirez Islands, Chile. Polar Biology 47: 771-777.

Talis EJ, Che-Castaldo C, Hart T, McRae L & Lynch HJ. 2023. Penguindex: a living planet index for Pygoscelis species penguins identifies key areas of population change. Polar Biology 46: 707-718.

Tamini LL, Chaves LN, Dellacasa RF, Crawford R & Frere E. 2021. Incidental capture of seabirds in Argentinean side-haul trawlers. Bird Conservation International 31: 591-604.

Thompson DR, Goetz KT, Sagar PM, Torres LG, Kroeger CE, Sztokowksi LA, Orben RA, Hoskins AJ & Phillips RA. 2021. The year-round distribution and habitat preferences of Campbell Albatross (*Thalassarche impavida*). Aquatic Conservation: Marine and Freshwater Ecosystems 31: 2967-2978.

Thompson T. 2025. Advancing remote monitoring of albatrosses through emerging technologies to inform conservation. MSc thesis. University of Otago, Dunedin, New Zealand.

Towns DR & Broome KG. 2003. From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands. New Zealand Journal of Zoology 30: 377-398.

Veitch CR. 2001. The eradication of feral cats (*Felis catus*) from Little Barrier Island, New Zealand. New Zealand Journal of Zoology 28: 1-12.

Ventura F, Lukacs PM, Granadeiro JP, Matano R & Catry P. 2021. Demographic responses to environmental change of black-browed albatross, sentinel of the Patagonian Shelf Large Marine Ecosystem. Marine Ecology Progress Series 668: 107-120.

Virtue J, Turner D, Williams G, Zeliadt S & Lucieer A. 2023. Thermal drone observations capture fine-scale population decline of short-tailed shearwaters. Remote Sensing in Ecology and Conservation https://doi.org/10.1002/rse2.70020

Wade R & Veitch CR. 2019. Costs and benefits for biodiversity following rat and cat eradication on the Hauturu-o-toi/Little Barrier Island. Island invasives: scaling up to meet the challenge 62: 558.

Walker K & Elliott G. 2005. Population changes and biology of the Antipodean wandering albatross. Notornis 52: 206-214.

Walker K & Elliott G. 2017. ACAP priority population assessment: Antipodean Albatross at Antipodes Island. Fourth Meeting of the Population and Conservation Status Working Group, Wellington, New Zealand. Agreement on the Conservation of Albatrosses and Petrels.

Walker KJ, Elliott GP, Rexer-Huber K, Parker GC, Sagar PM & McClelland PJ. 2021. Shipwrecks and mollymawks: an account of Disappointment Island Birds. Notornis 67: 213-245.

Warwick-Evans V, Pearmain EJ, Phillips RA. 2026. Overlap of nonbreeding wandering albatrosses with fisheries and implications for colony-specific population trajectories at South Georgia. Conservation Biology e70260.

Warwick-Evans V, Pearmain EJ, Thorne L, Phillips RA. 2025. Spatial segregation and bycatch risk as potential drivers of population rends of wandering albatrosses at South Georgia. Conservation Biology 40: e70126.

Waugh SM & Wilson KJ. 2017. Threats and threat status of the Westland Petrel *Procellaria westlandica*. Marine Ornithology 45: 195-203.

Waugh SM, Barbraud C, Adams L, Freeman AND, Wilson K-J, Wood G, Landers TJ & Baker GB. 2015. Modelling the demography and population dynamics of a subtropical seabird, and the influence of environmental factors. The Condor: Ornithological Applications 117: 147-164.

Waugh SM, Barbraud C, Delord K, Simister KLJ, Baker GB, Hedley GK, Wilson K-J & Rands DRD. 2020. Trends in density, abundance, and response to storm damage for Westland Petrels *Procellaria westlandica*, 2007-2019.

Waugh SM, Weimerskirch H, Moore PJ & Sagar PM. 1999. Population dynamics of Black-browed and Grey-headed Albatrosses *Diomedea melanophrys* and *D. chrysostoma* at Campbell Island, New Zealand, 1942-1996. Ibis 141: 216-225.

Weimerskirch H, Delord K, Barbraud C, Le Bouard F, Ryan PG, Fretwell P & Marteau C. 2018. Status and trends of albatrosses in the French Southern Territories, Western Indian Ocean. Polar Biology 41: 1963-1972.

Weimerskirch H, Lallemand J & Martin J. 2005. Population sex ratio variation in a monogamous long-lived bird, the wandering albatross. Journal of Animal Ecology 74: 285-291.

Weimerskirch H. 2018. Linking demographic processes and foraging ecology in wandering albatross – Conservation implications. Journal of Animal Ecology 87: 945-955.

Wilson KJ. 2016. A review of the biology and ecology and evaluation of threats to the Westland Petrel *Procellaria westlandica*. Unpublished report.

Winker H, Pacoureau N & Sherley RB. 2020. JARA: 'Just Another Red-list Assessment'. BioRxiv: https://doi.org/10.1101/672899.

Wolfaardt A & Phillips R. 2020. Guideline census methodologies for albatrosses and petrels. Agreement on the Conservation of Albatrosses and Petrels.

Zhang J, Bell EA & Roberts JO. 2020. Demographic assessment of black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island). New Zealand Aquatic Environment and Biodiversity Report N. 244. Fisheries New Zealand, Wellington, New Zealand.

Zipkin EF & Saunders SP. 2018. Synthesising multiple data types for biological conservation using integrated population modelling. Biological Conservation 217: 240-250.

SUPPLEMENTARY MATERIAL 0.

Draft updated ACAP IG Trends Terms of References.

To robustly estimate and communicate the trends of all ACAP species, the Intersessional Group will:

1. Review trajectories truncated at 1970, poor quality counts as identified by data holders, and review percentile cut-offs for IUCN Red List category assignments.
2. Publish the methodology developed in PaCSWG9 Doc 3, including placing *ACAPT* on *GitHub*.
3. Complete standardised abundance data processing, trend modelling, and conservation assessments for all ACAP species.
4. Lead development of ACAP input into IUCN Red List re-assessment process led by BirdLife International.
5. Develop updated, standardised criteria for identifying ACAP High Priority Populations.
6. Investigate trends spatially by integrating trend estimates with tracking and fishing effort data.
7. Develop outputs for reporting population counts and trends, including the use of clear and easy to understand figures and graphics.
8. Evaluate changes in trends following RFMCO resolutions and relevant domestic legislation changes.
9. Work with the ACAP secretariat to improve the ACAP population database.
10. Provide recommendations to refine ACAP population status indicators.
11. Explore options to streamline updates to abundance and trend analyses.
12. Report back to PaCSWG10 (Aug 2027; papers due in May 2027) on the process, results, and provide recommendations for the integration of results into different ACAP products.

SUPPLEMENTARY MATERIAL 1.

Case study details per species

Antipodean Albatross (Diomedea antipodensis)
The Antipodean Albatross is arguably the best studied seabird species in New Zealand. We collated data for this species from the three key breeding sites (Moutere Mahue | Antipodes Island for ssp. *antipodes*, Adams and Disappointment Island for ssp. *gibsoni*) across 1969-2025, which jointly cover >99% of the overall population. Counts for this species have largely been conducted annually following standardised protocols since the mid-1990s using GCs at designated study and index sites on Moutere Mahue and Adams Island (representing 2.8-14.9% and 2.6-9.5% of the breeding sites, depending on study site combinations, respectively). These counts have documented concerning declines in both breeding sites from the mid-2000s onwards (e.g., Elliott *et al.* 2025, Rexer-Huber *et al.* 2025), which resulted in the species being listed as EN on the IUCN Red List in 2018 (BirdLife International 2025) and the Moutere Mahue population as an ACAP High Priority Population in 2017 (Walker & Elliott 2017). Full island counts, however, are rare and were only conducted in the 1960s, 1970s and 1990s (Robertson 1975, Walker & Elliott 1999, 2005), and recently, when the world's largest AC^{UAV} enabled the total breeding site estimates on Moutere Mahue and Adams Island (Elliott *et al.* 2024, 2025, Rexer-Huber *et al.* 2024, 2025). In contrast, the Disappointment Island colony has only been counted three times, through a combination of GC, VPC, and AC^{heli} (Robertson 1975, Baker & Jensz 2014a). This species also breeds on Motu Ihupuku | Campbell Island and Rangihaute | Rangiāuria | Pitt Island, but both populations are very small (<1% of the world population combined) and counts are irregular and incomplete (Miskelly *et al.* 2008, Mischler *et al.* 2025), so we did not include these data in our analyses.

Most Antipodean Albatross counts required minimal adjustments, as most of the reported counts were already aligned with our framework in the original references (e.g., Elliott *et al.* 2025, Rexer-Huber *et al.* 2025). As typically $\Psi^d > 0.950$ for this species (Walker & Elliott 2005), we did not explicitly account for detection and assumed that Eq. 1 would suffice to address this error. Moutere Mahue counts were not adjusted for Ψ^p during 1978-2023, while the Adams Island counts were not adjusted for Ψ^p during 1973-1996. Consequently, we used an annually varying model (Eq. 3A) to make the necessary adjustments for ψ^{fail}. The Disappointment Island counts required adjustments for Ψ^o and Ψ^p, for which we used the intercepts of the Adams Island models (Eq. 2AB & 3A, respectively). Both the Moutere Mahue and Adams Island populations have a known history of skewed sex-ratios (Elliott *et al.* 2025, Rexer-Huber *et al.* 2025), and, as data enabling insights on time-varying p^b exist for both, we applied conversions from $y_{t,i}$ to $M_{t,i}$ that specifically accounted for these factors (Eq. 5B). No p^b data for Disappointment Island exists and thus we used the model intercept for Adams Island for the conversions on Disappointment Island (Eq. 5A).

Wandering Albatross (D. exulans)
We collated data for the five major Wandering Albatross breeding sites Macquarie Island, Kerguelen Islands, Crozet Islands, Prince Edward Islands, South Georgia (Islas Georgias del Sur)[1] corresponding jointly to >99% of the world population, over a notably long period: 1947-2026.

On South Georgia (Islas Georgias del Sur)[1], Wandering Albatrosses breed at multiple locations throughout the archipelago, making complete archipelago-wide counts difficult to undertake. Bird Island supports the largest population of the archipelago, and it has been counted almost annually since 1972 (GC of BP), with several earlier counts in the 1960s (Croxall *et al.* 1990, Poncet *et al.* 2006, 2017, Mackley *et al.* 2025). Full archipelago censuses were undertaken in 1984, 2004, 2015, and 2024 (Croxall *et al.* 1990, Poncet *et al.* 2006, 2017, Mackley *et al.* 2025) using mainly GC of BP but occasionally VPC, VBC, AC^{heli}, and AC^{UAV} for a very limited number of locations (representing <1% of the archipelago population, which for simplicity were assumed to be BP). The populations on Albatross and Prion Islands have been counted (GC of BP) almost annually since 1999, whereas counts at other locations were conducted only intermittently. We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. As population trends appeared to differ among breeding sites, the proportional contribution of each location to the total archipelago population was linearly interpolated between each full archipelago census years (1984, 2004, 2015, and 2024). Prior to 1984, location proportions were assumed to remain constant at 1984 values, and after 2024, proportions were assumed to remain constant at 2024 values. For simplicity, proportions used as inputs to Eq. 1 were calculated outside the JAGS model. All counts from Bird Island had already been adjusted for ψ^{fail} with breeding monitoring data from Bird Island. Counts from other locations were adjusted for ψ^{fail} using Eq. 3A with time-varying adjustment based on breeding monitoring data from Prion Island, as population trends and breeding success at Bird Island were considered to differ from those elsewhere in the archipelago (S. Poncet, unpubl. data). Finally, we calculated p^b as the product of return and breeding probabilities (following Carneiro *et al.* 2020) and constructed informative priors for return probability (beta[20.83,8.93]) and breeding probability (beta[60.97,31.41]) derived from estimates in Pardo *et al.* (2017). We used this estimate of p^b to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

On Prince Edward Islands, Wandering Albatrosses breed on both Marion Island and Prince Edward Island. The population on Marion Island has been monitored almost annually since 1982 (GC of BP) with a couple of previous counts in the 1970s (Nel *et al.* 2003). The Prince Edward population has only been monitored in 1974, 1976, 1982, 1983, 1984, 2002, and 2009, all through GC of C or BP (Ryan *et al.* 2003, 2009). We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. The proportion of the island-group population occurring at each island used for extrapolation was derived from counts obtained in years when multiple islands were surveyed concurrently, with proportions

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[1] A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty of the Falkland Islands (Islas Malvinas), South Georgia and the South Sandwich Islands (Islas Georgias del Sur e Islas Sandwich del Sur) and the surrounding maritime areas.

linearly interpolated between survey years to account for differing rates of population change among islands. After 2009, the last year with Prince Edward Island count, the proportion of Marion Island was assumed to remain constant at the 2009 value. For simplicity, proportions used as inputs to Eq. 1 were calculated outside the JAGS model. We adjusted for ψ^{fail} using breeding monitoring data collected at Marion Island between 2006 and 2009 (Jones *et al.* 2017) through Eq. 3A. Finally, no site-specific information on p^b was available for Prince Edward Islands, so we used the prior for Crozet Islands (see below) to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

On Crozet Islands, Wandering Albatrosses breed on the five main islands: Île aux Cochons (the largest population), Île de la Possession, Île des Apôtres, Île aux Pingouins, and Île de l'Est. The population on Île de la Possession has been monitored almost annually since 1981 (GC of BP), with earlier counts also available from the 1960s and 1970s (Weimerskirch *et al.* 2018). Different combinations of the other islands were counted more sporadically over the years (1964, 1972, 1974, 1981, 1982, 2008, 2017, and 2018) with varying methods (GC of BP; AC^{heli} and AC^{sat} of BOG) (Weimerskirch *et al.* 2018). We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. The proportion of the island-group population occurring at each island used for extrapolation was derived from counts in 1981 and 1982, and 2017 and 2018, and linearly interpolated in between, allowing for varying rates of population change on the different islands. Subsequent counts for islands other than Île de la Possession were obtained in 2008 (Îles Apôtres and Île aux Pingouins; GC) and in 2017 (Île aux Cochons in 2017, AC^{heli}) and 2018 (Île de l'Est and Îles Apôtres, AC^{sat}). For Îles Apôtres and Îles aux Pingouins, abundance in 2017 (for the purpose of proportion calculation) was estimated by applying annual rates of change derived from the 1981–1982 to 2008 period and extrapolating linearly to 2017 (as 2017 AC^{sat} counts for Îles Apôtres were associated with substantial uncertainty). For Île aux Cochons, the 2017 count was first adjusted for ψ^{ABA} using occupancy data from Île de la Possession before estimating proportional distribution (C. Barbraud and K. Delord, unpubl. data). For the Île de l'Est counts in 2017, where AC^{sat} counts were associated with substantial uncertainty, abundance in 2017 (for proportional calculation only) was estimated by applying the mean annual rate of change observed across all other sites, extrapolated linearly from the 1982 baseline to 2017. Proportions were then calculated for 1981-1982 and 2017-2018. Proportions for intermediate years were linearly interpolated between 1982 and 2017, with proportions before 1982 assumed to be equal to 1982 values and those after 2017 assumed to remain constant at 2017 values. For simplicity, proportions used as inputs to Eq. 1 were calculated outside the JAGS model. In the JAGS model, we adjusted for ψ^{ABA} through Eq. 2A using occupancy (i.e., n_{BP} out of n_{AOS}) from Île de la Possession (C. Barbraud and K. Delord, unpubl. data). We adjusted for ψ^{BOG} using data from Péninsule Rallier du Baty in Kerguelen Islands, which was counted via AC^{heli} (AOS) and AC^{sat} (BOG) in the same year (2017; Weimerskirch *et al.* 2018), through Eq. 2B. All counts needed adjustment for ψ^{fail} which was modelled time varying through Eq. 3A using breeding monitoring data from Île de la Possession available for all years with counts since 1968 (C. Barbraud and K. Delord, unpubl. data). Finally, we constructed an informative prior for p^b (Beta[24.3,18.6]) centred on 0.566 from information on Crozet Islands in Carneiro *et al.* (2020). As no uncertainty was reported we assumed a

95% credible interval of approximately 0.42–0.72, allowing moderate uncertainty around the long-term average breeding frequency. We used this prior for p^b to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

On the Kerguelen Islands, Wandering Albatrosses breed at multiple locations across the archipelago, including two major locations — Péninsule Courbet, counted almost annually since 1985, with one earlier count in 1971, and Péninsule Rallier du Baty, with only two counts in 1987 and 2017 — as well as several smaller locations comprising the Île Howe group, Presqu'île Joffre and adjacent islands, Baie Larose, Îles Nuageuses, and Îles Leygues (Weimerskirch *et al.* 1989, 2018). Most counts were GC of BP, however, counts of Péninsule Rallier du Baty where either GC (1987) or AC$^{\text{heli}}$ (2017) of AOS. We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. The proportion of the island-group population occurring at each location used for extrapolation was derived from counts in the two years in which Péninsule Rallier du Baty was counted (1987 and 2017), and linearly interpolated in between, allowing for varying rates of population change at the different locations. All locations were counted in 1985 and 1987 (although not all locations were surveyed in both years), so counts from these two years were used to estimate the proportion of each location in the 1980s. The 2017 Péninsule Rallier du Baty count, conducted by AC$^{\text{heli}}$, was first adjusted for ψ^{ABA} using data from Crozet (see below) prior to estimating proportions. In 2017, the smaller locations were not counted, so rates of change for Péninsule Courbet and Péninsule Rallier du Baty were calculated between the 1980s and 2017. Abundance at the smaller locations in 2017 was then estimated by applying the mean rate of change observed at the two major sites over this period, after which site proportions were calculated for 2017. Proportions for intermediate years were linearly interpolated between 1985 and 2017, with proportions before 1985 assumed to be equal to 1985 values and those after 2017 assumed to remain constant at 2017 values. Occasional counts of the smaller locations were used only to estimate their proportional contribution and were excluded from the remainder of the model to simplify its structure, as they contribute a very small fraction of the Kerguelen population (c. 5%, with the remainder concentrated in Péninsule Courbet and Péninsule Rallier du Baty). For simplicity, proportions used as inputs to Eq. 1 were calculated outside the JAGS model. All counts needed adjustment for ψ^{fail} which was modelled time varying through Eq. 3A using breeding monitoring data from Péninsule Courbet available for all years with counts since 2000 (C. Barbraud and K. Delord, unpubl. data). Finally, no site-specific information on p^b was available for Kerguelen Islands, so we used the prior for Crozet Islands (see below) to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

The small population on Macquarie Island has been studied intensively since 1963, with comprehensive nearly annual ground counts of breeding pairs at all colonies since (Terauds *et al.* 2006). We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. Counts were generally conducted throughout the breeding season and therefore did not require adjustment for Ψ^p. Finally, no site-specific information on p^b was available for Macquarie Island, so we used the prior for Crozet Islands (see below) to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

Northern Royal Albatross (D. sanfordi)

We collated count data for the three Northern Royal Albatross breeding sites, the privately-owned Motuhara | Forty-fours and Rangitutahi | Sisters archipelagos within the Chatham Islands and the Pukekura | Taiaroa Head colony on the Otago Peninsula – the world's only "mainland" albatross colony (jointly comprising >99% of the world population), over a remarkably long period: 1936-2026. Pukekura is small (<1% of the global population), but its accessibility has resulted in a truly unique time-series: a virtually continuous GC census of >90 years, documenting every breeding attempt since the establishment of the breeding site in the 1930's (Richdale 1939, Robertson 1993, DOC unpubl. data). This breeding site is subject to concerted management and breeding success is maximised through targeted interventions (e.g., predator control, artificial incubation, and cross fostering; Robertson 2001, Richard *et al.* 2015). Both Chatham Islands breeding sites on the other hand are challenging to access, but hold >99% of the species' population. Both sites have been counted intermittently, with periods of concerted effort during the 1970s, 1990s, 2000s, and 2020s. Counts are predominantly completed as AC^{fw}, with both breeding sites counted simultaneously, but GCs do occur (e.g., Frost 2017, 2024, Bell 2024). Therefore, counts are expressed in BP, AOS, BOG, and C. In 1985, a severe storm stripped most soil and vegetation from the Motuhara (and to a lesser extent, Rangitutahi), caused near-complete breeding failure, and disrupted the alternating cycle of this biennial breeder, making trends difficult to detect, at least for a period (Roberston 1991, 1998). Both sites have been subjected to multiple counts within years to compare counting methods (e.g., GCs vs AC^{fw} vs AC^{sat}), which included the establishment of study quadrats representing 11% of the breeding site, but direct comparability between methods remained challenging (e.g., Scofield 2011, Baker *et al.* 2017, Bell 2021, Frost 2022). Despite irregularity, environmental stochasticity, and differing methods, these data indicated concerning declines over time (e.g., Bell *et al.* 2018), resulting in the species being listed as EN in 2000 (BirdLife International 2025).

Uniquely, the Pukekura time-series represents continuous monitoring of all breeding attempts and is thus expressed as all BP at the start of the breeding period for the entire breeding site. Therefore, no adjustments were required apart from applying Eq. 1. For the Chatham Island breeding sites, we first selected the counts that required the least data processing for years during which >1 count was conducted. Then, as Bell *et al.* (2018) estimated $\Psi^{d} > 0.950$ (on Rangitutahi), as is typical for *Diomedea* albatrosses, we did not explicitly account for imperfect detection and assumed that Eq. 1 would suffice. Less than 25% of counts on both the Motuhara and Rangitutahi provided explicit data on ψ^{BOG} despite BOG being the common metric of counting this species and thus we used an intercept only model to first transform BOG into AOS through Eq. 2B. Only two counts provided explicit data on ψ^{ABA} and thus we used these two data points to generate a uniform prior to then transform AOS into BP as per Eq. 2A. The traditional metric to express counts of Northern Royal Albatrosses on the Chatham Islands is not BP, and f is estimated through whole breeding site counts of chicks (e.g., Robertson 1991, Frost 2017), and thus, obtaining accurate estimates of f was not straightforward. Pukekura estimates could not be used as this site is subject to management targeted to maximize f (e.g., DOC Coast Otago Operations 2026).

Consequently, we first transformed all counts into BP, and then used the BP estimates at the time of the count and whole island counts of chicks to obtain f. We then included a fixed effect for the years 1989-2000 to capture the substantial impacts of the severe storm on the productivity of both sites (Roberston 1991, Scofield 2011, Bell *et al.* 2018). After these steps, we successfully applied Eq. 3A to estimate BP at the start of the breeding season for both breeding sites, i.e., $y_{t,i}$. Finally, as the only information on p^b for this species is based on the Pukekura long-term capture-mark-recapture dataset, we used the information contained in Richard *et al.* (2015) and Edwards *et al.* (2025) to first devise an informative prior for years without storm impacts (N[0.567,125]). We then used the long-term f estimate during 1989-2000 of both Chatham Island sites with the estimates p^b of failed and successful breeders in Richard *et al.* (2015) to devise a second informative prior for p^b for years with storm impacts for both Chatham Island sites during 1989-2000 (N[0.664,125]). No evidence exists for skewed sex ratios in this species and thus we applied Eq. 5A to convert $y_{t,i}$ to $M_{t,i}$ for all three breeding sites.

Southern Royal Albatross (D. epomophora)

We collated count data for two Southern Royal Albatross breeding sites, Motu Ihupuku | Campbell Island and Enderby Island (jointly comprising >99% of the world population) over a notably long period: 1943-2025. These two breeding sites contrast in size, with Motu Ihupuku holding >99% of the overall population and Enderby Island holding <1%. This contrast is reflected in the count data, with Enderby Island GCs typically covering the entire site, and being repeated almost annually since 1993 (Childerhouse *et al.* 2003), while Motu Ihupuku counts (GCs, sometimes combined with VPCs) were more intermittent and irregular, with full island counts being conducted roughly every decade until the mid-2000s (Moore *et al.* 1997, 2012). However, standardised study and index site GCs (representing 2.5-25.7% of the Motu Ihupuku population, depending on combinations) provided additional, more regular data since the late 1980s up to 2026, which recently highlighted a substantial population decline (Mischler *et al.* 2025). This species also breeds on Adams Island, but the population is minute (<1% of the world population) and counts are irregular and incomplete (Elliott *et al.* 2025), so we did not include these data in our analyses.

To standardise the Southern Royal Albatross data, we first selected the counts that required the least amount of adjustment in the few years that several counts were conducted on Enderby and then we fit Eq. 1 to both the Enderby and the Motu Ihupuku timeseries. Various studies have shown that $\Psi^d > 0.950$ for GCs of this species (e.g., Moore *et al.* 1997, 2012, Mischler *et al.* 2025) and thus for GCs, we did not explicitly account for imperfect detection and assumed that Eq. 1 would suffice. However, $\Psi^d < 0.950$ for VPCs on Motu Ihupuku (Moore *et al.* 1997, 2012), and thus we used the few estimates that exist to construct an informative prior for ψ^d_{VPC} (U[0.720-1.000]) to account for imperfect detection in VPCs (1958-1983 and 1995). Similarly, those older VPCs did not account for Ψ^o and these counts consisted of an indistinguishable mix of BP, AOS, and BOG. Data on ψ^{ABA} and ψ^{BOG} for this species is rare (e.g., Baker *et al.* 2014, 2020a) thus, we used an intercept-only formulations of Eq. 2A and 2B to estimate ψ^{ABA} and ψ^{BOG}, respectively. Some counts from Motu Ihupuku were paired with explicit estimates of ψ^{fail} (Moore *et al.* 2012), and these original adjustments were carried over. Where no adjustments had been made, we fit a time-varying model to existing f data from Motu Ihupuku to estimate annually varying ψ^{fail} through Eq. 3A. However, no detailed data on f exist from Enderby (Childerhouse *et al.* 2003) and thus we used the f intercept estimate from the Campbell data to adjust the Enderby Island counts. Information on p^b for this species is limited, and thus we used the little information contained in Waugh *et al.* (1997), Moore *et al.* (2012), and Edwards *et al.* (2025) to construct an informative prior (N[0.515,100]), and as no evidence exists for a skewed sex ratios in this species, we applied Eq. 5A to convert $y_{t,i}$ to $M_{t,i}$ for all three breeding sites.

Campbell Albatross (Thalassarche impavida)

We collated data for the only breeding site for this species from Motu Ihupuku across a remarkable timeseries encompassing 1944-2025. Campbell Albatrosses breed in a series of ~30 colonies on rugged cliffs and peninsulas on the northern coast of Motu Ihupuku, which they share with Grey-headed Albatrosses (*T. chysostoma*) and a small number of Black-browed Albatrosses (*T. melanophris*), both of which breed at the same time (Moore *et al.* 1997, Moore 2004). Although Moore (2004) covered ~81% through GCs in the late 1990s, most counts are conducted through VPCs (standardised photo points) as well as through some AC^{heli}, AC^{UAV}, and VBCs (Moore & Moffat 1990, Moore & Blezard 1999a,b, Moore 2004, Sagar 2014, Frost 2020). Due to the mixed nature of the colonies, several counts are simply of albatrosses, without specifying which species counts pertain to, and as both species have exhibited different trajectories at different sections, great care is required in disentangling counts of *Thalassarche* spp. (Moore 2004). Modern counts fortunately do separate explicitly between both species (Frost 2020, Mischler 2024, 2025). Despite investments into demographic studies (Waugh *et al.* 1999, 2001), no standardised study or index sites exist that would enable standardised GCs. Notwithstanding the challenges associated with obtaining insights into the abundance and trends of Campbell Albatrosses, the standardised VPCs have signalled concerning declines of this species (Moore 2004, Frost 2020, Mischler *et al.* 2025).

To standardise Campbell Albatross counts, we first disentangled decadal estimates reported for the 1990s and 2000s (Moore 2004, Sagar 2014, respectively) which, for the latter, required using explicit data from suitable part-sites from which extrapolation was possible. This approach resulted in an increase of data points from nine to 18. Still, the decadal data points for the 1940s to 1980s had been subjected to substantial re-analyses by Moore (2004) that further disentanglement was not possible. Following this step, we incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. Most counts did not feature in the ACAP database, but those that did were categorised as “High”. However, upon discussion with experts and due to the challenges associated with separating different *Thalassarche* albatrosses within the same VPCs, all counts were re-categorised to “Medium” or lower (Moore, Mischler pers. comms.). Moore (2004) provided estimates for Ψ^d for the various approaches that can be taken to VPCs, which we used to formulate informative priors (U[0.815,0.994]) to account for Ψ^d for the counts where this had not been incorporated explicitly (1944-1986, 1994, 1996-2025). Due to the nature of the VPCs, most counts are initially expressed in BOG, but generally adjusted to reflect AOS (Moore 2004, Frost 2020, Mischler *et al.* 2024, 2025). Some ground transects provided data to estimate ψ^{ABA} (Moore

2004, Sagar 2014, Mischler *et al.* 2024, 2025) and so we applied Eq. 2A (using a time-sensitive formulation) to these data to convert AOS to BP while incorporating further uncertainty as all transect counts were based on small sample sizes. Following efforts by Moore (2004) on accounting for Ψ^p, virtually all counts had been adjusted following the same regression models (e.g, Sagar 2014, Frost 2020, Mischler *et al.* 2024, 2025) and so we adapted this approach for the counts that had not been adjusted to ensure comparability. As all counts contained 30-100 Black-browed Albatrosses which cannot be distinguished from Campbell Albatrosses at the distance the VPCs are conducted (Moore *et al.* 1997, Moore 2004), we formulated an informative prior (U[15, 50]), which we subtracted from the BP estimate. Finally, we formulated an informative prior based on data contained in Waugh *et al.* (1999) for p^b (logit(N[1.7, 25]), corresponding to 0.846 [0.786-0.891]). However, as there was an indication that p^b was potentially lower in 2024 and 2025 (Mischler *et al.* 2024, 2025), we adjusted this prior accordingly (logit(N[1.1, 5]), corresponding to 0.750 [0.551-0.880]). We then used these priors to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

Grey-headed Albatross (T. chrysostoma)
We collated data for the seven major grey-headed albatross breeding sites, Motu Ihupuku | Campbell Island, Macquarie Island, Kerguelen Islands, Crozet Islands, Prince Edward Islands, South Georgia (Islas Georgias del Sur)[1], and Diego Ramírez Islands, corresponding jointly to >99% of the world population, over a notably long period: 1944-2026.

On Diego Ramírez Islands, Grey-headed Albatrosses breed on eight different islands. The whole archipelago was counted using different methods (GC, VPC, VBC, AC^{fw}) in 2003 (Robertson *et al.* 2007). Then subsets of the islands were subsequently counted in 2007, 2012, and 2015, all AC^{fw} (Robertson *et al.* 2014, 2017). GC recorded AOS, while other methods recorded BOG. We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. The proportion of the island-group population breeding on each island used for the extrapolation was derived from the 2003 counts. However, as counts were completed using different methods and at different time of the breeding cycle, the proportion was calculated from counts adjusted for ψ^{BOG}, ψ^{ABA}, and ψ^{fail}. As no occupancy data was available for Diego Ramírez Islands, we used estimates of ψ^{ABA} and ψ^{BOG} from Motu Ihupuku (see below) to adjust counts through Eq.2A and Eq.2B, respectively. As no breeding success estimate was available for Diego Ramírez Islands, we modelled adjustment for ψ^{fail} following the approach used for Kerguelen Islands (see below). No estimate of p^b was available for Diego Ramírez Islands, so we used the mean p^b across Motu Ihupuku, Prince Edward Islands, and South Georgia (Islas Georgias del Sur)[1] to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

On South Georgia (Islas Georgias del Sur)[1], Grey-headed Albatrosses breed in several parts of the north-western end of the archipelago. On one of the major site, Bird Island, two study plots were established in 1976 and counted almost annually since (Prince *et al.* 1994). The only full archipelago census of the species was conducted in 2004 (Poncet *et al.* 2006), and was followed up by counts of about 72.5% of the archipelago in 2015 and 2024 (Poncet *et al.* 2017, Mackley *et al.* 2025). Other counts included surveys of some colonies in 1985 and 1986, categorised as "*Low*" due to limited information on methods and counts rounded to the nearest thousand, as well as island-wide Bird Island censuses conducted in 1977, 1978, 1990, 1991, 1992, and 2003 (Prince *et al.* 1994). Counts of the study plots on Bird Island were GC of BP; counts of island-wide Bird Island were GC (BP) and VPC (<10% of the island population; adjusted to BP); and other counts included a mixture of GC, VPC, and VBC, with VPC and VBC using photographs and recorded as BOG. We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. The proportion of island-group population breeding at each part of South Georgia (Islas Georgias del Sur)[1] used for the extrapolation was derived from the 2004 full archipelago census (Poncet *et al.* 2006). For the study plots on Bird Island, their proportion to the Bird Island population, and eventually to the island-group population (~1.5%), was estimated from their mean proportional representation over four years using data from Prince *et al.* (1994). All counts from Bird Island had been adjusted for ψ^{fail} (Poncet *et al.* 2006, Mackley *et al.* 2025). Counts from 2004, 2015, and 2024 were already adjusted for ψ^{BOG}, ψ^{ABA}, and ψ^{fail} using a composite correction factor derived from paired GC and VPC of several colonies on Bird Island in 2004 and 2024, with the 2015 counts adjusted using the correction factor estimated in 2004 (Poncet *et al.* 2006, 2017, Mackley *et al.* 2025). We adjusted the counts from the 1980s (other than Bird Island) for ψ^{BOG}, ψ^{ABA}, and ψ^{fail} using the mean of the composite correction factor estimated across the two years for which it was available, as a binomial process. Finally, p^b was estimated using data from Ryan *et al.* (2007) describing the number of birds breeding in successive years over the four years following a breeding attempt, conditional on breeding success. Specifically, we modelled counts of outcome-specific breeding interval probabilities (i.e., number of birds breeding in k years after a breeding attempt depending on the breeding outcome) within each breeding outcome group, $y^o = (y_1^o, \dots, y_K^o)$, as arising from a multinomial distribution with total N^o and outcome-specific probabilities $r^o = (r_1^o, \dots, r_K^o)$, where r^o was assigned a Dirichlet prior distribution:

$$y^o \sim multinomial(N^o, r^o)$$
$$r^o \sim Dirichlet(\alpha^o)$$
$$N^o = \sum_{k=1}^{K} y_k^o$$

where $k \in \{1, \dots, K\}$ indexes breeding intervals ($K = 4$), and $o \in \{1,2\}$ indexes breeding outcome (1 = successful, 2 = failed), with $\alpha_k^o = 1$. The probability of a k-year inter-breeding interval (p_k) was modelled as a mixture of the interval distributions associated with successful (r_k^1) and failed (r_k^2) breeding events, weighted by the long term mean breeding success probability (γ):

$$p_k = \gamma \cdot r_k^1 + (1 - \gamma) \cdot r_k^2$$

The expected interval between breeding attempts E was calculated as the probability-weighted mean of interval lengths:

$$E = \sum_{k=1}^{K} p_k \cdot k$$

The breeding frequency p^b was then defined as the inverse of the expected inter-breeding interval:

$$p^b = \frac{1}{E}$$

We then used the estimate of p^b to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

On Prince Edward Islands, Grey-headed Albatrosses breed on both Marion Island and Prince Edward Island. The population on Marion Island was counted almost annually since 1988 (Nel *et al.* 2002, Ryan *et al.* 2009, Stevens *et al.* 2024), while the colonies on Prince Edward Island have only been counted in 2002 and 2009 (Ryan *et al.* 2003, 2009). All counts across both islands were recorded as AOS from VPC, however the timing of the counts varied over time and between islands. We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. The proportion of the island-group population breeding on each island used for the extrapolation was derived from the 2002 and 2009 counts from both islands. However, as counts from Marion Island and Prince Edward Island were completed at different times of the breeding cycle, the proportion was calculated from counts adjusted for ψ^{fail} and ψ^{ABA}, and averaged across the two years. Occupancy data (i.e., n_{BP} out of n_{AOS}) has been collected on a subset of the Marion Island population almost annually since 1988 (from c. 430 AOS on average; K. Stevens, unpubl. data) and were used to derive time-varying estimates of ψ^{ABA} for Marion Island counts adjustment through Eq. 2A. Following Ryan *et al.* (2009), who estimated that 25-30% of AOS on Prince Edward Island were not on eggs, we used an informative prior (U[0.7,0.75]) for ψ^{ABA} for Prince Edward Island. Breeding success data were available for most years as additional counts of chicks in the Marion Island population towards the end of each breeding season. These data were used to derive time-varying estimates of ψ^{fail} for adjustment through Eq.3A, with counts adjusted for ψ^{ABA} used as estimates of the number of eggs present when counts were conducted. Finally, p^b was estimated using from Ryan *et al.* (2007) following the method described for South Georgia (Islas Georgias del Sur)[1] (see above). We then used the estimate of p^b to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

On the Crozet Islands, Grey-headed Albatrosses breed on four islands (Île de la Possession, Île de l'Est, Île des Apôtres, and Île des Pingouins), all of which were ground-counted in 1982, although at different stages of the breeding cycle, thus recording AOS or chicks (Croxall 1984). The only subsequent count for the island group was conducted in 2016 by helicopter and covered only Île des Apôtres and Île des Pingouins (Weimerskirch *et al.* 2018). Counts from 1982 were used to estimate the proportion of the island-group population occurring on each island. However, as these data were not collected in a consistent metric, we used the adjusted counts accounting for ψ^{fail} and ψ^{ABA} to derive island-level proportions, incorporating baseline uncertainty, and extrapolate all counts to the breeding site level using Eq. 1. We modelled adjustments for ψ^{fail} (Eq. 3A) and ψ^{ABA} (Eq. 2A), and transformation from $y_{t,i}$ into $M_{t,i}$ (Eq. 5A) following the approach used for Kerguelen Islands, as no data were available for Crozet Island.

On the Kerguelen Islands, Grey-headed Albatrosses breed primarily on Îles Nuageuses, with smaller colonies on the Loranchet Peninsula. Both sites were counted in 1985 (GC on Îles Nuageuses, VPC on the Loranchet Peninsula). The only subsequent count for the island group was conducted by helicopter in 2014. All counts from the Kerguelen Islands recorded AOS. We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1, with colony proportions derived from the 1985 counts. All counts required adjustment for ψ^{fail}, however, as no breeding data were available for Kerguelen Islands, we used breeding success from Motu Ihupuku, Macquarie Island, Prince Edward Islands, and South Georgia (Islas Georgias del Sur)[1] (Waugh *et al.*1999, Sagar & Amey 2013, K. Stevens, unpubl. data). We modelled site-specific mean breeding success on the logit scale within a hierarchical normal model. The observed mean breeding success for the four sites were assumed to arise from a global mean with breeding-site-level random effects, with additional uncertainty incorporated through observation-specific standard errors. Site effects were treated as random and normally distributed around zero, with a shared standard deviation governing between-site variation. A diffuse Normal(0, 0.001) prior was assigned to the global mean, and a Uniform(0, 1) prior was used for the standard deviation of site effects. Predicted breeding success for Kerguelen Islands was obtained by combining the global mean and the corresponding site-level random effect, with the latter informed indirectly through the shared distribution of breeding site effects, and back-transforming from the logit scale. We then used this estimate to adjust Kerguelen Islands counts for ψ^{fail} through Eq. 3A. As no data were available for ψ^{ABA} or p^b for the Kerguelen Islands, and given their proximity to the Prince Edward Islands, we used the Marion Island intercept estimate for ψ^{ABA} (see below) to adjust from AOS to BP (Eq. 2A) and Marion Island estimate of p^b to transform Kerguelen Islands $y_{t,i}$ into $M_{t,i}$ through Eq. 5A, assuming no sex bias.

The small population on Macquarie Island has been studied intensively since 1978, with comprehensive nearly annual ground counts of breeding pairs at all colonies from 1978 to 1985, and again from 1995 to 2025 (Copson 1988, Terauds *et al.* 2005). We incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. Since 1995, counts were generally conducted throughout the breeding season and therefore did not require adjustment for Ψ^p. The only two discrete counts (2000 and 2003) had already been adjusted for ψ^{fail}. We adjusted counts from the 1970s and 1980s for ψ^{fail} using breeding data from Macquarie Island through Eq. 3A. Finally, no site-specific information on p^b was available for Macquarie Island; we therefore used the mean of site-specific estimates for Motu Ihupuku, Prince Edward Islands, and South Georgia (Islas Georgias del Sur)[1] to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

To standardise Grey-headed Albatross counts from Motu Ihupuku, we mirrored the approach employed for Campbell Albatross, as the two species are traditionally counted at the same time. First, we disentangled decadal estimates reported for the 1990s and 2000s (Moore

2004, Sagar 2014, respectively) but retained the decadal data points for 1940s-1980s (Moore 2004). We then incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. Similar to the Campbell Albatross counts, we re-categorised all counts to "Medium" or lower (Moore, Mischler pers. comms.). Moore (2004) provided estimates for Ψ^d for the various approaches that can be taken to VPCs, which we used to formulate informative priors (U[0.815,0.994]) to account for Ψ^d for the counts where this had not been incorporated explicitly (1944-1986, 1994, 1996-2025). Most counts were expressed in BOG, but generally adjusted to reflect AOS (Moore 2004, Frost 2020, Mischler *et al.* 2024, 2025). Ground transects provided data to estimate ψ^{ABA} (Moore 2004, Sagar 2014, Mischler *et al.* 2024, 2025) using Eq. 2A (using a time-sensitive formulation) and enabled us to convert AOS to BP while incorporating further uncertainty. Following efforts by Moore (2004) on accounting for Ψ^p, virtually all counts had been adjusted following the same regression models (e.g, Sagar 2014, Frost 2020, Mischler *et al.* 2024, 2025). Finally, we formulated an informative prior based on data contained in Waugh *et al.* (1999) for p^b (logit(N[-0.24, 7]), corresponding to 0.440 [0.272-0.622]). We then used this prior to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A as there is no indication for a sex bias.

Buller's Albatross (T. bulleri)

We collated data for the four major Buller's Albatross colonies, Hautere | Solander and Tini Heke | Snares Islands in the New Zealand Subantarctic (ssp. *bulleri*) and the Motuhara and Rangitutahi in the Chatham Island archipelago (ssp. *platei*) (jointly comprising >99% of the world population) across 1969-2025. The Tini Heke population has been studied intensively since 1948 (e.g., Sagar *et al.* 2000, Francis & Sagar 2012). Population counts, consisting of a combination of GCs and VPCs started in 1969, but varied across years depending on what sections were counted (study and index areas on North East Island cover 1.4-18.5% depending on their combination), and full archipelago counts have been intermittent (i.e., 1969, 1992, 1997, 2002, 2014, 2019, and 2020; Sagar *et al.* 1999, Sagar & Stahl 2005, Sagar 2014, Thompson & Sagar 2020). The remote and difficult to access Hautere, Motuhara, and Rangitutahi populations, the latter two of which are privately-owned, have been counted sporadically using a variety of approaches (GCs, VPCs, AC^{heli} and AC^{fw}; e.g., Baker *et al.* 2017, Bell 2021, Frost *et al.* 2025). Of these three, only Motuhara holds established index plots (representing 1.8-3.9%; Bell 2021, 2023, 2026), while the other two breeding sites are largely counted at a full island or archipelago level. Buller's Albatrosses also breed on Rosemary Rock in the Three Kings archipelago (ssp. *platei*; Rayner *et al.* 2020), but the population is minute (<1% of the world population) and counts are irregular, so we did not include these data in our analyses.

For Hautere and Tini Heke counts (i.e., ssp. *bulleri*), we incorporated baseline uncertainty and extrapolated all counts to breeding site level through Eq. 1. Attempts have been made to estimate Ψ^d for various Hautere counts (0.900-0.960) and Tini Heke VPCs (0.900-0.950) (Sagar & Stahl 2005, Thompson *et al.* 2017, Thompson & Sagar 2020) and so we incorporated these estimates explicitly through informed uniform priors. For the Tini Heke, we separated the sections of each count that were conducted using GCs and those that were conducted using VPCs using Thompson & Sagar (2020) and Sagar (unpubl. data) and adjusted the latter only. Most of the AC from Hautere were already adjusted from BOG to AOS in the original references (e.g., Frost *et al.* 2025), but none accounted for ψ^{ABA} and so we sourced data on Hautere transect counts (Baker & Jensz 2014b, Thompson *et al.* 2017, Frost *et al.* 2025) and applied Eq. 2A to these data to account for ψ^{ABA} retrospectively while incorporating further uncertainty as all transect counts were based on small sample sizes (Frost *et al.* 2025). As no direct data on ψ^{ABA} from Tini Heke was directly available (despite the detailed work on this topic; Stahl & Sagar 2006), we used the intercept estimate from Hautere to adjust the Tini Heke VPCs (as these produced data expressed in AOS as well). Subsequently, we sourced the data on f from Hautere (JC Stahl unpubl. data) and Tini Heke (Sagar & Stahl 2005) and applied Eq. 3A to these data to account for ψ^{fail} for all Hautere counts and the Tini Heke VPCs only. We did not adjust the GCs on Tini Heke as signs of failed nests (e.g., eggshells, dead chicks etc.) remain in place for an unusually long time under the forest canopy on this breeding site and thus data had already been adjusted for this source of error (P. Sagar pers. comm.). Finally, we formulated an informative prior based on Edwards *et al.* (2025) for p^b (N[0.800, 250]), which was based on the information contained in Fu & Sagar (2016). We then used this prior to transform $y_{t,i}$ into $M_{t,i}$ through Eq. 5A for both breeding sites, as there is no indication for a sex bias in any population and no further information to differentiate p^b among sites.

For Motuhara and Rangitutahi counts (i.e., ssp. *platei*), we did not adjust for Ψ^d as the habitat on these islands differs considerably from Hautere and Tini Heke and Bell *et al.* (2018) estimated Ψ^d at 0.996-0.999 on Rangitutahi. As such, for these two islands, we assumed that the uncertainty incorporated through Eq. 1 was sufficient. Motuhara and Rangitutahi counts have been expressed in virtually all possible metrics over the years and as such it was necessary to estimate ψ^{ABA}, ψ^{BOG}, and ψ^{occ}, but data for the estimation of these parameters were sparse, partially as potentially definitions of failed and empty nests differed between teams (alternatively the situation has changed on the islands over time; e.g., Fraser *et al.* 2010 vs. Bell 2026). We sourced five estimates with data for ψ^{ABA}, three for ψ^{BOG}, and four for ψ^{occ} (Fraser *et al.* 2010, Baker *et al.* 2017, Bell 2026). Based on how these were distributed among the years in the timeseries, we used Eq. 2A (ψ^{ABA}) and Eq. 2C (ψ^{occ}) while accounting for annual variation for Motuhara but used an intercept-only model for Eq. 2B (ψ^{BOG}) for Motuhara. As almost all data on Ψ^o originated from Motuhara instead of Rangitutahi, we used intercept only models for Eq. 2ABC for the latter breeding site. Information on f for Motuhara and Rangitutahi is exceptionally sparse and only one estimate that covers the entire breeding period exists (fused from two different sources; Bell 2021, Frost *et al.* 2021), which we used to construct a mildly informed prior (N[0.510,50]) for integration into Eq. 3A. We then used the same approach (i.e., the same prior for p^b within Eq. 5A) as for Hautere and Tini Heke, as no information for Motuhara or Rangitutahi on p^b exists, nor is there any indication for a sex imbalance. Subsequently, all counts were expressed in $y_{t,i}$ and $M_{t,i}$ suitable for the our trend modelling steps.

White-capped Albatross (T. steadi)

Population counts and trends of this species, the most numerous albatross species in New Zealand, have been subject to considerable debate, as trend estimates vary considerably depending on how error adjustments are applied (see Walker *et al.* 2021 vs. Baker *et al.* 2023). We collated data for the three major White-capped Albatross colonies, Disappointment Island, Southwest Cape on Maukahuka | Auckland Island, and Logan Point on Adams Island (jointly comprising >99% of the world population), across 1973-2021. All three colonies are difficult to survey, and counts have been conducted intermittently and irregularly (although regular counts did occur during 2007-2018; Baker *et al.* 2023), using a variety of different methods, including GCs, VPCs, VBCs, $\mathrm{AC}^{\mathrm{heli}}$ and $\mathrm{AC}^{\mathrm{UAV}}$. While a study area exists on the largest colony (Castaway Bay representing ~4.7% of the Disappointment Island colony), this area is not fully geographically demarcated, is largely used for demographic studies, and has not been widely used as an index count site (Walker *et al.* 2021, Parker *et al.* 2024). White-capped Albatrosses also breed on Bollon's Island off Antipodes Island (Tennyson *et al.* 1998), the Western Chain Islets (Miskelly *et al.* 2001), and Motuhara (Bell *et al.* 2017), but these populations are very small (<1% of the world population) and counts are very rare, so we did not include these data in our analyses. Furthermore, we note that this species recently bred on the Diego Ramirez archipelago in Chile (Suazo *et al.* 2024), but we continue to treat this species as a New Zealand breeding endemic here.

We fit Eq. 1 to all White-capped Albatross counts under the assumption that the incorporated uncertainty would also cover the associated Ψ^{d}. However, the 1973 VBCs, some of which were combined with GCs and VPCs, were documented with rough uncertainty estimates (Robertson 1975), but required further Ψ^{d} adjustments due to the influence of this error on VBCs (Fig 2; e.g., Rexer-Huber *et al.* 2020). To address this gap, we completed an online modified Delphi expert elicitation (https://forms.gle/kXCoSfzbGkAPzYvZ9; Hemming *et al.* 2018) with 11 experts familiar with the breeding sites. After round two, we aggregated the four-point estimates to derive priors following Fischer *et al.* (2022) for the Disappointment Ψ^{d}_{VBC}: β(9.044, 2.681) and the Southwest Cape Ψ^{d}_{VBC}: β(6.388, 2.585) (corresponding to modes of 0.771 and 0.711, respectively). Due to the small population size and considerable uncertainty surrounding the Logan Point VBC in 1973, no attempts were made to infer Ψ^{d} for this count. We then used all available data on ψ^{ABA} (Walker *et al.* 2021, Parker *et al.* 2022, Elliott *et al.* 2023) and ψ^{BOG} (Walker *et al.* 2021, Parker *et al.* 2022, Baker *et al.* 2023, Elliott *et al.* 2023) to fit to Eq. 2A and Eq. 2B, respectively, each while accounting for interannual effects. As virtually all information on ψ^{ABA} and ψ^{BOG} were obtained on Disappointment Island, we followed the approach from Baker *et al* (2023) and considered these values representative for the other breeding sites as well. Despite this shortcoming, this approach still allowed us to transform counts expressed in BOG or AOS into BP while incorporating uncertainty explicitly. We collated all available data on f (e.g., Thompson & Sagar 2008ab, Francis 2012, Frost 2026) and fit Eq. 3A to these. As only three of the six years for which data on f were available covered years in the abundance timeseries (i.e., <25% of the timeseries), we used an intercept only model to obtain $y_{t,i}$. Again, we did not differentiate between different breeding sites due to data limitations. Finally, we sourced the only available estimate of p^{b} for White-capped Albatross from Francis (2012), and constructed an informative prior (N[0.680, 250]) to convert the generated $y_{t,i}$ into $\mathrm{M}_{\mathrm{t,i}}$ using Eq. 5A to obtain both standardised abundance metrics required for our trend models.

Chatham Albatross (T. eremita)

Counts of this species, the rarest of the New Zealand endemic albatrosses, are infrequent and intermittent. In general, this species is one of the least studied albatross species globally, which is in part caused by the extreme difficulty of accessing its main breeding site, Te Tara Koi Koia | the Pyramid within the Chatham Islands archipelago, which is privately owned (Bell *et al.* 2017, Bell 2026). We collated the intermittent and irregular abundance data for this breeding site across 1965-2026. Data were predominantly collected between 2000 and 2026 during GCs with a small proportion of VPCs (Bell *et al.* 2017), aside from two early counts, a VBC in 1965 (Dawson 1973) and an AC^{fw} in 1973 (Robertson *et al.* 1991). All counts covered the entire island, and no abundance index sites exist (although there are two study areas for demographic studies; Bell *et al.* 2017). During 2014-2018, the site was visited to translocate pre-fledging chicks to main Chatham Island in order to establish a second colony, but no abundance data were collected during these visits, and ultimately, no second colony was established through these translocations, as all chicks that survived to maturity recruited back to the source site (or the Western Chain; Bell 2026, P Sagar unpubl. data). A few Chatham Albatross pairs also breed on the Western Chain islets (Miskelly *et al.* 2001), but their number is so small (<1% of the world population) and counts are rare, so we did not include these data in our analyses.

Chatham Albatross counts were almost entirely expressed in NS rather than BP, despite indications that NS do not equal BP (Robertson *et al.* 2003, Bell *et al.* 2017). For 2000-2026, Ψ^d was considered sufficiently high that we did not explicitly account for detection and assumed that Eq. 1 would suffice. For these counts, the proportion counted through VPCs was known and thus this was subtracted to derive ξ for Eq. 1 and simplify subsequent steps. The 1965 VBC and 1973 AC^{fw} each had existing Ψ^d estimates, but these were potentially inaccurate (Dawson 1973, Robertson *et al.* 1991). We therefore completed an online modified Delphi expert elicitation (https://forms.gle/79gRwTEzwEViXgUD6; Hemming *et al.* 2018) with nine experts familiar with the breeding site. We aggregated the obtained four-point estimates following Fischer *et al.* (2022), including the original Ψ^d estimates (obtaining a total of six estimates), to derive updated beta-distributed priors for the Ψ^d associated with the VBC and AC^{fw} ($\beta[7.119, 4.079]$, $\beta[18.200, 2.870]$, corresponding to modes of 0.636 and 0.864, respectively). While interpretation of empty vs. failed nests appeared to have changed over time, all GCs recorded nest contents (e.g., Fraser *et al.* 2011, Bell *et al.* 2017, Bell 2026), enabling direct interpretation of BP, and thus no adjustments of Ψ^o for these data were required. However, the VBC and AC^{fw} presumably recorded AOS (as no landings took place) and to convert these into BP in the absence of species-specific data, we calculated the mean of the intercept estimates for ψ^{ABA} for both White-capped and Salvin's Albatross as the most closely related species, and then used this estimate in Eq. 2A. Similarly, no breeding success data exists for Chatham Albatross, but it may mirror the closely related Salvin's Albatross, which also breeds on similar substrates (Bell *et al.* 2017). Thus, to account for ψ^{fail}, we generated an informative prior using all available data on f for Salvin's Albatross (N[0.352,250]); Thompson 2025), after which we fit Eq. 3 to all Chatham Albatross counts. Finally, no data p^b exist for Chatham Albatross and thus we used the informative priors we had generated for p^b for the two most closely-related species, White-capped and Salvin's

Albatross (N[0.680, 250]), and N[0.927,100], respectively) and averaged these, after which we fit Eq. 4A to all counts. Following these steps, all Chatham Albatross data were in a suitable, comparable format to enable the coherent modelling of trends.

Salvin's Albatross (T. salvini)
Counts of this species, the second-most numerous albatross species in New Zealand, are infrequent and intermittent, and depending on how early counts are interpreted, trend estimates vary considerably, leading to contrasting conservation status assessments (IUCN Red List = VU, New Zealand Threat Classification System = CR; BirdLife International 2025, Robertson *et al.* 2021). We collated data for the two major Salvin's Albatross breeding sites, Moutere Hauriri | Bounty Islands, and the Western Chain Islets (jointly comprising >99% of the world population) across 1979-2026. Salvin's Albatross breeding sites are characterised as densely populated, but barren, isolated granite rock stacks that are extremely difficult to access and survey, and thus various methods have been used to obtain abundance data, ranging from GCs, to AC^{fw} and AC^{heli}, to more recently, AC^{UAV} (Thompson 2025). The largest Salvin's Albatross population is located on Moutere Hauriri (97% of the global population), and while whole archipelago counts have been conducted (as ACs), there has been a focus on Proclamation Island in some years due to the accessibility of this islet (10.3% of the breeding site) (Parker & Rexer-Huber 2020a, Mattern 2022, 2024ab). The smaller breeding site, the Western Chain islets, consists of two main islets (Toru; 66.7%; and Rima; 33.3%) which have largely been counted through GCs (Sagar *et al.* 2014, Thompson 2025). Salvin's Albatrosses also breed on Te Tara Koi Koia, but in such low numbers (<1% of the world population; Bell *et al.* 2017) that we did not include these data in our analyses. Furthermore, we note that this species also breeds on Penguin Island and Apôtres Island in the Crozet Archipelago, but these populations are extremely small (<1%) (Jouventin 1990; Weimerskirch *et al.* 2018), and thus we continue to treat this species as a New Zealand breeding endemic here.

While almost all counts of Salvin's Albatross could be processed and adjusted relatively straightforwardly through our standardised framework, the 1979 Bounty Islands GC needed additional consideration. Specifically, Robertson & van Tets (1982) estimated a density of 0.5 BP/m^2 and then extrapolated across all suitable areas (15.3 ha). This approach may have produced an overestimate due to sampling in high density areas only, the assumption of uniform density, and an absence of high-resolution maps to enable fine-scale delineation of suitable areas (Amy & Sagar 2013, Thompson 2025). It was not possible to re-evaluate the densities in 1979. Instead, we updated the delineation of suitable areas by placing a 2 m^2 circular buffer around each manually identified Salvin's Albatross BOG in high-resolution UAV footage of the entire archipelago collected in 2023 and 2025 (Mattern 2022, 2024a, 2024b), merging all buffers, recalculating the mean area between both years (9.85 ha), and ultimately repeating the original extrapolation (see Thompson 2025 for details). We then assigned the lowest ACAP accuracy score to this count to further acknowledge the continuing, unavoidable uncertainty.

After this additional work, we followed our framework step by step as outlined above. Salvin's Albatross breeding sites are barren in nature, and Sagar & Amey (2013) estimated Ψ^d

> 0.990 on Moutere Hauriri, so we did not explicitly account for detection and assumed that Eq. 1 would suffice. We then collated all information on Ψ^o obtained through transects or close-up photographs (e.g., Sagar & Amey 2013, Baker & Jensz 2019) and fit Eq. 2A and Eq. 2B to the collated data on ψ^{ABA} and ψ^{BOG}, respectively, allowing us to convert BOGs into BPs, while allowing for annual variation. All counts from the Western Chain were obtained during GCs and expressed in BP and thus this step was not necessary for this breeding site. As Salvin's Albatross is one of the few species for which stage-specific f values have been quantified, we were able to account for Ψ^p through Eq. 3B (Thompson 2025). However, the number of years for which f_k has been quantified are few, and thus, we used the mean f_k across years to account for Ψ^p. Specifically, we used the information in Thompson (2025) on T_k and f to construct informative priors (f_{inc} = N[0.562,125], f_{guard} = N[0.960,30000], $f_{\text{post-guard}}$ = N[0.697, 600]) and use these in Eq. 3B. Information on f has been obtained from Moutere Hauriri only and to date no estimates for the Western Chain are available, and thus we used Moutere Hauriri f to adjust both sites. Finally, the only information available on p^b for Salvin's Albatross is based on three years of research on Toru islet (Sagar *et al.* 2011). We used the data contained in Sagar *et al.* (2011) to fit a binomial GLMM with a logit-link function to estimate p^b and construct an informative prior (N[0.927,100]), which we included in Eq. 5A to estimate M for all counts (Thompson 2025).

Waved Albatross (Phoebastria irrorrata)

We collated data covering 1970-2011 for the main breeding site for Waved Albatrosses: Isla Espanola, within the Galapagos archipelago. The species also breeds on Isla de la Plata of the Ecuadorian mainland and occasionally attempts breeding on Isla Genovesa, but both hold <1% of the breeding population of the species and were therefore not included here (Anderson *et al.* 2002, 2008, Street 2013). There are two key study areas on Isla Espanola, Punta Suarez (22.9% of overall population) and Punta Ceballos (37.6 %) with additional work being conducted in the centre of the island as well (ACAP 2009, Awkerman *et al.* 2014). All data were obtained through GCs. The species is listed as CR and receives some attention, yet full island counts are rare and more recently the focus has been on obtaining CMR datasets (G. Jimenez and K Huyvaert unpubl. data). A further complication is that the species is a mobile incubator and therefore part-site counts may come with a lower level of confidence (Awkerman *et al* 2005).

Two counts of Waved Albatrosses covered the entire island (1970 and 1994), two covered Punta Suarez and Punta Ceballos (2001 and 2007), while one covered Punta Ceballos only (2011) (Anderson *et al.* 2002, 2008, Street 2013). In addition, the CMR datasets from Punta Suarez and Punta Ceballos provided some further abundance data for plots within each study area (5.7 and 5.5% of the overall population respectively). We used Eq. 1 to extrapolate all counts to the whole island but assigned the CMR data as "Low" accuracy as these data were not explicitly collected for abundance estimates and due to the mobile incubation behaviour of the species (Awkerman *et al.* 2005). The 2011 estimate accounted for Ψ^d and thus we used the information contained in (Street 2013) to construct an informative prior (U[0.923,0.996]) to account for Ψ^d for the other counts as well. All counts were expressed in either BP or C and thus no accounting for Ψ^O was required. Anderson *et al.* (2002) and (2008) did account for ψ^{fail} but the CMR-based abundance data had not been accounted for ψ^{fail}. Consequently, we used information in ACAP (2009) on f to construct an informative prior (N[0.229, 200]) and then used Eq. 3A to account for ψ^{fail} for these counts that required adjustments. Finally, we sourced information from Street (2013) and Awkerman *et al*. 2007 to construct an informative prior for p^b (logit(N[1.38,5])) and s (N[0.457,12500]), which we included in Eq. 5B to estimate M for all counts.

Westland Petrel (Procellaria westlandica)

We collated data covering 1974-2019 for the sole breeding site for Westland Petrels, Punakaiki, on the Te Waipounamu | the South Island of New Zealand. The species breeds in a rugged area across ~28 sub-colonies that can be difficult to access, define, and may change over time (Waugh *et al.* 2020). Due to these challenges, population counts for this species are rare and gaps in the timeseries are common and long. Counts can only realistically be conducted as GCs, but despite this commonality, differences between counts remained in terms of timing, correction for Ψ^{o}, and subsequent extrapolation steps (Wood & Otley 2013, Baker *et al.* 2020, Waugh *et al.* 2020). Consequently, considerable uncertainty surrounded most data. Nevertheless, Waugh & Wilson (2017) successfully argued that the species should be uplisted to EN on the IUCN Red List due to the perceived impact of extreme weather events on the fragmentation of breeding habitat.

Westland Petrel GCs during 2002-2005 and 2010, were not documented with sufficient information to disentangle individual count years, and thus we used 2003 as a mid-point for these counts (Wood & Otley 2013). No Westland Petrel GCs explicitly accounted for Ψ^{d} (e.g., through distance sampling) and since we were unable to identify a unified mechanism to incorporate Ψ^{d} retrospectively, we attempted to acknowledge the absence of accounting for this error by downscaling the accuracy of all counts by one category. All Westland Petrel GCs have been accompanied with some efforts to account for Ψ^{o}. However, some GCs explicitly accounted for ψ^{ABA} and ψ^{occ} (2003 and 2019; Wood & Otley 2013, Waugh *et al.* 2020), while the other GCs only accounted for ψ^{occ} (e.g., Baker *et al.* 2020). Consequently, we used the data reported in Baker *et al.* (2020) to model ψ^{ABA} and information in Waugh *et al.* (2020) to construct an informative prior (N[0.480,200]), which we then fused into an overall estimate for Westland Petrel $\psi^{ABA} = 0.658$ (0.570-0.743). We then used this overall estimate within Eq. 2A to adjust all counts that had not explicitly accounted for this source of error. Breeding success in this species may have changed over time (Bartle 1985, Freeman & Wilson 2002, Waugh *et al.* 2006, 2020) and thus we used the available information on multi-year averages in Freeman & Wilson (2002) to generate an informative prior for $f_{1974\text{-}1975}$ (N[0.390, 100]), Waugh *et al.* (2006) to generate a prior for f_{2003} (N[0.620, 700]), and Waugh *et al.* (2020) to generate a prior for $f_{2007\text{-}2019}$ (N[0.685, 80]). We then used these different estimates to account for Ψ^{P} in Eq. 3A, while explicitly incorporating change in f over time. Only limited information was available on Westland Petrel p^b. Specifically, two, quite different deterministic estimates were available (Waugh *et al.* 2015, Waugh & Wilson 2017), which we used to construct a uniform prior (U[0.460,0.768]) and estimate M for the 1974-2011 counts. However, recently, more data has been collected due to the perception that p^b has changed and thus we used recent (2021-2024) intensive burrow monitoring data (K

Simister unpubl. data) together with a binomial GLMM with a logit-link function to obtain an updated estimate for $p^b = 0.706$ (0.693-0.720) and estimate M for the 2019 count. No evidence exists for a skewed sex ratio in this species (e.g., Waugh *et al.* 2015) and thus we used Eq. 5A for all estimates of M. Following these steps, all Westland Petrel data (both in $y_{t,i}$ and $M_{t,i}$) were in a suitable format to allow for the modelling of trends.

Black Petrel (P. parkinsoni)

This species has been subject to a long-standing and ongoing population study at its largest breeding site, Aotea | Great Barrier Island, since 1996 (Bell *et al.* 2025), rendering it one of the best studied species of its genus, rivalled only by White-chinned Petrels (*P. aequinoctialis*) in other ocean basins (e.g., Dasnon *et al.* 2022). We collated data across 1977-2025 for the two Black Petrel breeding sites, Aotea | Great Barrier Island (~87%) and Te Hauturu-o-toi | Little Barrier Island (~13%) which jointly hold 100% of the world population. Note, the first count for both sites was here labelled as 1977 as a midpoint of the 1972-1975, 1977, 1979, and 1982-1983 counts that could not be disentangled (Imber 1987). The two breeding sites contrast in their data availability, with Aotea having been monitored annually for >20 years (e.g., Bell & Sim 2000, Bell *et al.* 2025), while only two data points exist for Hauturu (Imber 1987, Bell *et al.* 2016). Only one comprehensive effort to estimate the true population of the entirety of Aotea exists, spanning three years of GCs (2019-2021; Bell *et al.* 2022), which we here used to scale all other counts (i.e., calculate ξ). Aside from this effort, GCs on Aotea are usually conducted using nine census squares within the traditional Mt Hirakimata study area ($\xi = 2.3\%$). However, during 1996-1998, only three census grids existed, representing <1% of the breeding site population, which we therefore excluded from processing in our framework (see section 2.2.1). Occasionally, stratified transect GCs have been conducted throughout the wider study area ($\xi = 53\%$), and these data have provided contrasting results when directly compared to data from the census grids (e.g., Bell *et al.* 2013, 2018). When both were available, we favoured counts that represented a larger proportion of the breeding site, i.e., transects. We excluded area-based extrapolations using the census-grids only due to the known bias of these grids towards high-density areas (e.g., Bell & Sim 2000, Bell *et al.* 2013, 2018). The two data points that exist for Hauturu were both obtained through GCs, of which only the latter consisted of concerted efforts using transects and dog surveys (Bell *et al.* 2016). Other Black Petrel work has taken place on Hauturu (e.g., Bell *et al.* 2011, 2015), most of which involved using existing study burrows, but as the proportion study burrows represent is unknown, we could not collate additional abundance data for this site. Using two data points to estimate trends is far from optimal, but we include the Hauturu data here for completeness' sake (and posterior predictive checks did not indicate any poor fit of the trend model).

Modern Black Petrel GCs explicitly accounted for Ψ^d, e.g., through distance sampling (Bell *et al.* 2022) and during several other counts, researchers have gone through great lengths to maximize Ψ^d, e.g., through the use of petrel detection dogs (e.g., Bell *et al.* 2015, 2016). As we did not identify a unified mechanism to incorporate Ψ^d retrospectively, we did not explicitly account for this error. However, we did adjust the ACAP accuracy score of some counts to reflect the lack of specific accounting for Ψ^d. Specifically, following comparison with the latest transect counts (Bell *et al.* 2022), the 2005 and 2010 transect counts in the 35 ha study area were re-categorised *Low* accuracy and the 2013 and 2017 transect counts in the same area were re-categorised as *Medium* accuracy. Furthermore, to acknowledge the uncertainty of the 1977 Aotea and Hauturu counts (Imber 1987), we used expert judgement to adjust ξ using uniform priors (ξ_{Aotea} = U[0.333-0.531], $\xi_{Hauturu}$ = U[0.250-0.500]). Black Petrel GCs on both Aotea and Hauturu have always been accompanied with concerted efforts to count BP directly while accounting for Ψ^{o} ($\psi^{occ} \cdot \psi^{ABA}$ specifically; Bell *et al.* 2016, 2022, 2025), and thus the original efforts were carried over for both sites rather than applying retrospective adjustments through Eq. 2AC. However, generally, Black Petrel counts were not adjusted for Ψ^{p}, even though virtually all counts from Aotea and even some Hauturu counts are accompanied with data on f (Bell *et al.* 2016, 2025). Consequently, we fit a time-varying model to f data from Aotea and to estimate annually varying ψ^{fail} through Eq. 3A. Due to the limited data for Hauturu, we only used the f intercept estimate to adjust the one Hauturu count that needed it (2016). Concerted population modelling has been conducted for Black Petrels as well (e.g., Zhang *et al.* 2020), but annual estimates of p^{b}, not conditional on breeding success, are not currently available. Thus, we used information contained in Zhang *et al.* (2020) to construct an informative prior (N[0.719,7000]). As no evidence exists for a skewed sex ratios in this species, we applied Eq. 5A to convert $y_{t,i}$ to $M_{t,i}$ for both breeding sites.

References

ACAP. 2009. ACAP species assessment: Waved Albatross Phoebastria irrorata.

Anderson DJ, Huyvaert KP, Apanius V, Townsend H, Gillkin CL, Hill LD, Juola F, Porter ET, Wood DR, Lougheed C & Vargas H. 2002. Population size and trends of the Waved Albatross Phoebastria irrorata. Marine Ornithology 30: 63-69

Anderson DJ, Huyvaert KP, Awkerman JA, Proano CB, Milstead WB, Jimenez-Uzcategui G, Cruz S & Grace JK. 2008. Population status of the Critically Endangered waved albatross Phoebastria irrorate, 1999 to 2007. Endangered Species Research 5: 185-192.

Awkerman JA, Cruz S, Proano, Huyvaert KP, Jimenez-Uzcategui G, Baquero, Wikelski M & Anderson DJ. 2014. Small range and distinct distribution in a satellite breeding colony of the critically endangered Waved Albatross. Journal of Ornithology 155: 367-378.

Awkerman JA, Huyvaert KP & Anderson DJ. 2005. Mobile incubation in Waved Albatross (*Phoebastria irrorata*): associated hatching failure and artificial mitigation. Avian Conservation and Ecology 1: 1.

Awkerman JA, Westbrock MA, Huyvaert KP & Anderson DJ. 2007. Female-biased sex ratio arises after parental care in the sexually dimorphic Waved Albatross (*Phoebastria irrorata*). The Auck 124: 1336-1346.

Baker GB & Jensz K. 2014a. Gibson's Albatross at Disappointment Island - Analysis of aerial photographs. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB & Jensz K. 2014b. Buller's albatross aerial survey at the Solander Islands 2014. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB & Jensz K. 2019. Aerial survey of Salvin's albatross at the Bounty Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Elliott GP, French RK, Jensz K, Muller CG & Walker KJ. 2020a. Development of aerial monitoring techniques to estimate population size of great albatrosses (Diomedea spp.). Notornis 67: 321-331.

Baker GB, Hedley G, Cunningham R & Waugh S. 2020. Estimated population size of the Westland Petrel 2007-2011. New Zealand Aquatic Environment and Biodiversity Report No. 242. Fisheries New Zealand, Wellington, New Zealand.

Baker GB, Jensz K & Hamer D. 2014. Southern royal albatross at Enderby Island – analysis of aerial photographs 2014. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bartle JA 1985. Report on research on Westland black petrels 1969-83 with an outline of future objectives and a brief summary of management problems. Unpublished report of the National Museum of New Zealand, Wellington, New Zealand.

Bell E & Sim J. 2000. Surveying and monitoring of black petrels on Great Barrier Island, 1997/1998. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell E, Lamb S, Ray S. 2022. Population trends and breeding population size of black petrels (*Procellaria parkinsoni*) – 2020/2021 operational report. New Zealand Aquatic Environment and Biodiversity Report No. 280. Fisheries New Zealand, Wellington, New Zealand.

Bell EA, Lamb S & Ray S. 2025. Key demographic parameters and population trends of tākoketai/black petrels (*Procellaria parkinsoni*) on Aotea/Great Barrier Island: 2024/25. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Mischler C & Sim JL. 2015. Preliminary survey and population monitoring of black petrels (Procellaria parkinsoni) on Hauturu-o-toi/Little Barrier Island, 2014,15. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Mischler CP, MacArthur N & Sim JL. 2016. Black petrel (*Procellaria parkinsoni*) population study on Hauturu-o-Toi/Little Barrier Island, 2015/16. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Sim J, Scofield P, Francis C & Landers T. 2013. At-sea distribution and population parameters of the black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island), 2012/13. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Sim JL, Bull L, Pierre J, Torres L & Shaffer S. 2011. At-sea distribution of the black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island), 2009/10: Part 1 – environmental variables. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M, Bell D, Boyle D & Tuanui-Chisholm. 2018. Rangitatahi Seabird Research: December 2017. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2021. Motuhara seabird research: field trip report January 2021. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2024. Motuhara seabird research: field trip report January 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2026. Seabird research – Chatham Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell MD, Bell DJ & Boyle DP. 2017. Chatham Island Mollymawk research on Te Tara Koi Koia: November 2016. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Carneiro APB, Pearmain EJ, Oppel S, Clay TA, Phillips RA, Bonnet-Lebrun A, Wanless RM, Abraham E, Richard Y, Rice J, Handley J, Davies TE, Dilley BJ, Ryan PG, Small C, Arata J, Arnould JPY, Bell E, Bugoni L, Campioni L, Catry P, Cleeland J, Deppe L, Elliott G, Freeman A, González-Solís J, Granadeiro JP, Grémillet D, Landers TJ, Makhado A, Nel D, Nicholls DG, Rexer-Huber K, Robertson CJR, Sagar PM, Scofield P, Stahl J, Stanworth A, Stevens KL, Trathan PN, Thompson DR, Torres L, Walker K, Waugh SM, Weimerskirch H, Dias MP (2020) A framework for mapping the distribution of seabirds by integrating tracking, demography and phenology. Journal of Applied Ecology 57:514–525.

Childerhouse S, Robertson CJR, Hockley W & Gibbs NJ. 2003. Royal Albatross (*Diomedea epomophora*) on Enderby Island, Auckland Islands. Department of Conservation, Wellington, New Zealand.

Copson G (1988) The status of the black-browed and grey-headed albatrosses on Macquarie Island. PPRST 122:137–141.

Croxall JP (ed) (1984) Status and conservation of the world's seabirds: based on the proceedings of the ICBP Seabird Conservation Symposium, held in Cambridge, England, August 1982. International Council for Bird Preservation, Cambridge.

Croxall JP, Rothery P, Pickering SPC, Prince PA (1990) Reproductive Performance, Recruitment and Survival of Wandering Albatrosses Diomedea exulans at Bird Island, South Georgia. Journal of Animal Ecology 59:775–796.

Dasnon A, Delord K, Chaigne A & Barbraud C. 2022. Fisheries bycatch mitigation measures as an efficient tool for the conservation of seabird populations. Journal of Applied Ecology 59: 1674-1685.

Dawson EW. 1973. Albatross populations at the Chatham islands. Notornis 20: 210–230.

Department of Conservation Coastal Otago District Operations. 2026. Pukekura annual report. 2024/25 Season. Department of Conservation, Dunedin, New Zealand.

Edwards CTT, Peatman T, Fischer J & Gibson W. 2025. Inputs to the 2024 seabird risk assessment for the Southern Bluefin Tuna surface longline fishery. Fisheries New Zealand, Wellington, New Zealand.

Elliot G, Walker K, Rexer-Huber K, Tinnemans J, Long J, Sagar S, Osborne J & Parker G. 2024. Gibson's wandering albatross: demography, satellite tracking and census. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Waker K, Rexer-Huber K, Tinnemans J, McDonald A, Rawlence T & Parker G. 2025. Gibson's wandering albatross: drone-based population estimate, demography and at-sea distribution. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Walker K, Rexer-Huber K & Parker G. 2023. White-capped Albatross population study: February 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Fischer JH, Wittmer HU, Kenup CF, Parker KA, Cole R, Debski I, Taylor GA, Ewen JG, Armstrong DP. 2022. Predicting harvest impact and establishment success when translocating highly mobile and endangered species. Journal of Applied Ecology 59: 2071-2083.

Francis RIC & Sagar PM. Modelling the effect of fishing on southern Buller's albatross using a 60-year dataset. New Zealand Journal of Zoology 39: 3-17.

Francis RICC. 2012. Fisheries risks to the population viability of White-capped Albatross *Thalassarche steadi*. Fisheries New Zealand, Wellington, New Zealand.

Fraser M, Cameron N, Scofield P, Robertson CJRR. 2010A. Population assessment of Northern Buller's Albatross and Northern Giant Petrels at the Fortyfours, Chatham Islands, 1-8 December 2009. Ministry of Fisheries, Wellington, New Zealand.

Fraser M, Henderson G, Robertson CJR & Scofield P. 2011. Population dynamics of the Chatham Albatross at the Pyramid, 19 November – 2 December 2010. Ministry of Fisheries, Wellington, New Zealand.

Freeman AND & Wilson K-J. 2002. Westland petrels and hoki fishery waste: opportunistic use of a readily available resource. Notornis 49: 139-144.

Frost PGH, Baker GB, Fischer JH & Sagar PM. 2025. Population survey of Southern Buller's Albatross *Thalassarche bulleri bulleri* on the Solander Islands | Hautere, March 2024. Marine Ornithology 53: 361-371.

Frost PGH, Bell MD & Taylor GA. 2021. Trial-camera assessment of the fates of Northern Royal Albatross and Northern Buller's Mollymawk chicks: 2021 breeding season, Motuhara. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2017. Aerial census of Northern Royal Albatross (*Diomedea sanfordi*) fledglings on Rangitatahi (The Sisters) and Motuhara (Fortyfours), July 2017. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2020. Status of Campbell Island and Grey-headed Mollymawks on the Northern Coast of Campbell Island, November 2019. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2021. Status of Northern Royal Albatross Diomedea sanfordi nesting on the Chatham Islands, December 2020, Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2022. A census of Northern Royal Albatross nesting on the Chatham Islands, February 2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2024. Aerial survey of Northern Royal Albatross | Toroa (*Diomedea sanfordi*) populations on the Chatham Islands, December 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2026. Trail-camera monitoring of White-capped Albatross (Thalassarche steadi) on Disappointment Island: 2018, 2022 and 2023) breeding seasons. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Fu D & Sagar P. 2016. The 2014 demographic assessment of the Snares islands population of Southern Buller's albatross (*Diomedea bulleri bulleri*). New Zealand Aquatic Environment Biodiversity Report 165. Ministry for Primary Industries, Wellington, New Zealand.

Hemming V, Burgman MA, Hanea AM, McBride MF & Wintle BC. 2018. A practical guide to structured expert elicitation using the IDEA protocol. Methods in Ecology & Evolution 9: 169-180.

Imber MJ. 1987. Breeding ecology and conservation of the black petrel (*Procellaria parkinsoni*). Notornis 34: 19–39.

Jones MGW, Dilley BJ, Hagens QA, Louw H, Mertz EM, Visser P, Ryan PG (2017) Wandering Albatross Diomedea exulans breeding phenology at Marion Island. Polar Biol 40:1139–1148.

Jouventin P. 1990. Shy albatrosses Diomedea cauta salvini breeding on Penguin Island, Crozet Archipelago, Indian Ocean. Ibis 132: 126-127.

Mackley EK, Poncet S, Black A, Black J, Floyd K, Hall RM, Holmes EE, Manthorpe SJ, Passfield K, Bennison A, Phillips RA (2025) Endurance or extinction: long-term declines in albatrosses at South Georgia highlight threats from South Atlantic fisheries and climate change. Endang Species Res 57:437–451.

Mattern T. 2022. Tawaki Project: Bounty-Antipodes expedition 2022.

Mattern T. 2024a. Tawaki Project: Antipodes-Bounty expedition 2023/24.

Mattern T. 2024b. Tawaki Project: Bounty-Antipodes expedition 2024.

Mischler C, Moore P, Thompson H, Hamilton K & Pryde M. 2025. Campbell Island Seabird Research. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Miskelly CM, McNally N, Seymour R, Gregory-Hunt D & Lanauze J. 2008. Antipodean wandering albatross (*Diomedea antipodensis*) colonising the Chatham Islands. Notornis 55: 89-95.

Miskelly CM, Sagar PM, Tennyson AJD & Scofield RP. 2001. Birds of the Snares Islands, New Zealand. Notornis 48: 1-40.

Moore PJ & Blezard R. 1999a. Photographs of Campbell Island mollymawk colonies. A guide to photopoints, historical comparisons, and counting mollymawks Department of Conservation, Technical Series 17. Department of Conservation, Wellington, New Zealand.

Moore PJ & Blezard R. 1999b. Counting Campbell Island mollymawk colonies form photographs – data supplements. Science & Research internal report 169. Department of Conservation, Wellington, New Zealand.

Moore PJ & Moffat RD. 1990a. Research and management projects on Campbell Island 1987-88. Science and Research Internal Report 57. Department of Conservation, Wellington, New Zealand.

Moore PJ & Moffat RD. 1990b. Mollymawks on Campbell Island. Science & Research internal report 59. Department of Conservation, Wellington, New Zealand.

Moore PJ, Larsen EJ, Chateris M & Pryde M. 2012. Southern Royal Albatross on Campbell Island/Motu Ihupuku; Solving a band injury problem and population survey 2004-08. DOC Research and Development Series 333, Department of Conservation, Wellington, New Zealand.

Moore PJ, Scott JJ, Joyce LJ & Peart M. 1997. Southern royal albatross *Diomedea epomophora epomophora* census on Campbell Island, 4 January – 6 February 1996. Science & Research Series 101. Department of Conservation, Wellington, New Zealand.

Moore PJ, Taylor GA & Amey JM. 1997. Interbreeding of Black-browed Albatross *Diomedea m. melanophris* and New Zealand Black-browed Albatross *D. m. impavida* on Campbell Island. Emu 97: 322-324.

Moore PJ. 2004. Abundance and population trends of mollymawks on Campbell Island. Science for Conservation 242. Department of Conservation, Wellington, New Zealand.

Nel DC, Ryan PG, Crawford RJ, Cooper J, Huyser OA (2002) Population trends of albatrosses and petrels at sub-Antarctic Marion Island. Polar Biol 25:81–89.

Pardo D, Forcada J, Wood AG, Tuck GN, Ireland L, Pradel R, Croxall JP, Phillips RA (2017) Additive effects of climate and fisheries drive ongoing declines in multiple albatross species. Proc Natl Acad Sci USA 114.

Parker G & Rexer-Huber K. 2020a. Drone-based Salvin's albatross population assessment: feasibility at the Bounty Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Parker GC, Elliott G, Walker K & Rexer-Huber K. 2022. Gibson's albatross and white-capped albatross in the Auckland Islands 2021-2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Parker GC, Osborne J, Sagar R, Schultz H & Rexer-Huber K. 2024. White-capped albatross population study, Disappointment Island 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Poncet S, Robertson G, Phillips RA, Lawton K, Phalan B, Trathan PN, Croxall JP (2006) Status and distribution of wandering, black-browed and grey-headed albatrosses breeding at South Georgia. Polar Biol 29:772–781.

Poncet S, Wolfaardt AC, Black A, Browning S, Lawton K, Lee J, Passfield K, Strange G, Phillips RA (2017) Recent trends in numbers of wandering (Diomedea exulans), black-browed (Thalassarche melanophris) and grey-headed (T. chrysostoma) albatrosses breeding at South Georgia. Polar Biol 40:1347–1358.

Prince PA, Rothery P, Croxall JP, Wood AG (1994) Population dynamics of Black-browed and Grey-headed Albatrosses Diomedea melanophris and D. chrysostoma at Bird Island, South Georgia. Ibis 136:50–71.

Prince PA, Rothery P, Croxall JP, Wood AG (1994) Population dynamics of Black-browed and Grey-headed Albatrosses Diomedea melanophris and D. chrysostoma at Bird Island, South Georgia. Ibis 136:50–71.

Rayner MJ, Parker KA, Neho T & Hvid T. 2020. Buller's mollymawk (*Thalassarche bulleri platei*) count at Rosemary Rock, Manawatāwhi (Three Kings Islands). Notornis 67: 580-582.

Rexer-Huber K, Walker KJ, Elliott GP, Baker GB, Debski I, Jensz K, Sagar PM, Thompson DR & Parker GC. 2020. Population trends of light-mantled sooty albatross (*Phoebetria palpebrata*) at Adams Island and trial of ground, boat, and aerial methods for population estimates. Notornis 67: 341-355.

Rexer-Huber K, Whitehead E, Parker G, Patterson E, Walker K, Welch J & Elliott G. 2024. Antipodean wandering albatrosses and white-chinned petrels 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Whitehead E, Walker K & Elliott G. 2025. Antipodean wandering albatrosses and white-chinned petrels 2025. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Richard Y, Perriman L, Lalas C & Abraham ER. 2015. Demographic rates of northern royal albatross at Taiaroa Head, New Zealand. PeerJ 3:e906.

Richdale LE 1939. A Royal Albatross nesting on the Otago Peninsula, New Zealand. Emu 38: 467-488.

Robertson CJR & van Tets GF. 1982. The status of birds at the Bounty Islands. Notornis 29: 311-336.

Robertson CJR, Bell D & Scofield P. 2003. Population Assessment of the Chatham Albatross at The Pyramid, December 2001. DOC Science Internal Series 91. Department of Conservation, Wellington.

Robertson CJR. 1975. Report on the distribution, status, and breeding biology of the Royal Albatross, Wandering Albatross and White-capped Mollymawk on the Auckland Islands. In Yaldwyn JC (ed). Preliminary Results of the Auckland Islands Expedition 1972-1973. Department of Lands and Survey, Wellington, New Zealand.

Robertson CJR. 1991. Questions on the harvesting of Toroa in the Chatham Islands. Science & Research Series 35. Department of Conservation, Wellington.

Robertson CJR. 1993. Survival and longevity of the Northern Royal Albatross *Diomedea epomorphora sanfordi* at Taiaroa Head 1937-93. Emu 93: 269-276.

Robertson CJR. 1998. Factors influencing the breeding performance of the Northern Royal Albatross. In: Robertson G & Gales R. (eds), Albatross biology and conservation, pp. 99-104. Surrey Beatty & Sons, Chipping Norton, Australia.

Robertson CJR. 2001. Effects of intervention on the royal albatross population at Taiaroa Head, Otago, 1937-2001. DOC Science Internal Series 23. Department of Conservation, Wellington, New Zealand.

Robertson G, Moreno C, Arata JA, Candy SG, Lawton K, Valencia J, Wienecke B, Kirkwood R, Taylor P, Suazo CG (2014) Black-browed albatross numbers in Chile increase in response to reduced mortality in fisheries. Biological Conservation 169:319–333.

Robertson G, Moreno CA, Lawton K, Arata J, Valencia J, Kirkwood R (2007) An estimate of the population sizes of Black-browed (Thalassarche melanophrys) and Grey-headed (T. chrysostoma) Albatrosses breeding in the Diego Ramírez Archipelago, Chile. Emu - Austral Ornithology 107:239–244.

Robertson G, Wienecke B, Suazo CG, Lawton K, Arata JA, Moreno C (2017) Continued increase in the number of black-browed albatrosses (Thalassarche melanophris) at Diego Ramírez, Chile. Polar Biol 40:1035–1042.

Robertson HA, Baird KA, Elliott GP, Hitchmough RA, McArthur NJ, Makan TD, Miskelly CM, O'Donnell CFJ, Sagar PM, Scofield RP, Taylor GA & Michel P. 2021. Conservation status of birds in Aotearoa New Zealand, 2021. Department of Conservation, Wellington, New Zealand.

Ryan PG, Cooper J, Dyer BM, Underhill LG, Crawford RJM, Bester MN (2003) Counts of Surface-nesting Seabirds Breeding at Prince Edward Island, Summer 2001/02. African Journal of Marine Science 25:441–451.

Ryan PG, Jones MG, Dyer BM, Upfold L, Crawford RJ (2009) Recent population estimates and trends in numbers of albatrosses and giant petrels breeding at the sub-Antarctic Prince Edward Islands. African Journal of Marine Science 31:409–417.

Ryan PG, Phillips RA, Nel DC, Wood AG (2007) Breeding frequency in Grey-headed Albatrosses Thalassarche chrysostoma. Ibis 149:45–52.

Sagar P, Amey J (2013) Salvin's albatross population trend at the Bounty Islands, 1997-2011. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Charteris M & Scofield P. 2014. Salvin's albatross population size and survival at the Snares Western Chain. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P. 2014a. Population estimates and trends of Campbell and grey-headed albatrosses at Campbell Island. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P. 2014b. Population studies of Southern Buller's Albatrosses on the Snares. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar PM & Stahl J-C. 2005. Increases in the numbers of breeding pairs in two populations of Buller's Albatross (*Thalassarche bulleri bulleri*). Emu – Austral Ornithology 105: 49-55.

Sagar PM, Molloy J, Weimerskirch H & Warham J. 2000. Temporal and age-related changes in survival rates of Southern Buller's Albatrosses (*Thalassarche bulleri bulleri*) at the Snares, New Zealand, 1948-1997. The Auk 117: 699-708.

Sagar PM, Stahl J-C, Molloy J, Taylor GA & Tennyson AJD. 1999. Population size and trends within the two populations of Southern Buller's Albatross *Diomedea bulleri bulleri*. Biological Conservation 89: 11-19.

Scofield P. 2011. Data collection of demographic, distributional, and trophic information on selected seabird species to allow estimation of effects of fishing on populations viability. Ministry of Fisheries, Wellington, New Zealand.

Stahl JC & Sagar PM. 2006. Behaviour and patterns of attendance of non-breeding birds at the breeding colony in a Buller's Albatross *Thalassarche bulleri* population at The Snares. Notornis 53: 327-338.

Stevens KL, Altwegg R, Connan M, Ryan PG (2024) Population growth of the grey-headed albatross population on Marion Island inferred using three analysis methods. Antarctic Science 36:449–459.

Street PA. 2013. Abundance, survival, and breeding probabilities of the critically endangered Waved Albatross. MSc Thesis. Department of Fish, Wildlife, and Conservation Biology, Colorado State University, USA.

Tennyson A, Imber M & Taylor R. 1998. Numbers of Black-browed Mollymawk (*Diomedea m. melanophrys*) and White-capped Mollymawk (*D. cauta steadi*) at the Antipodes Islands in 1994-1995 and their population

Terauds A, Gales R, Alderman R (2005) Trends in numbers and survival of Black-browed (Thalassarche melanophrys) and Grey-headed (T. chrysostoma) Albatrosses breeding on Macquarie Island. Emu - Austral Ornithology 105:159–167.

Terauds A, Gales R, Baker GB, Alderman R (2006) Population and survival trends of Wandering Albatrosses (Diomedea exulans) breeding on Macquarie Island. Emu - Austral Ornithology 106:211–218.

Thompson D & Sagar P. 2008a. Annual report 2006/07: A population and distributional study of white-capped albatross (Auckland Islands). Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D & Sagar P. 2008b. Annual report 2007/08: A population and distributional study of white-capped albatross (Auckland Islands). Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D & Sagar P. 2020. Population studies of southern Buller's albatrosses on the Snares Islands | Tini Heke. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D, Sagar P, Baker B & Jensz K. 2017. Southern Buller's Albatross survey at Solander Islands 2016. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson T. 2025. Advancing remote monitoring of albatrosses through emerging technologies to inform conservation. MSc thesis. University of Otago, Dunedin, New Zealand.

Walker K & Elliott G. 1999. Population changes and biology of the wandering albatross *Diomedea exulans gibsoni* at the Auckland Islands. Emu 99: 239-247.

Walker K & Elliott G. 2005. Population changes and biology of the Antipodean wandering albatross. Notornis 52: 206-214.

Walker K & Elliott G. 2017. ACAP priority population assessment: Antipodean Albatross at Antipodes Island. Fourth Meeting of the Population and Conservation Status Working Group, Wellington, New Zealand. Agreement on the Conservation of Albatrosses and Petrels.

Walker KJ, Elliott GP, Rexer-Huber K, Parker GC, Sagar PM & McClelland PJ. 2021. Shipwrecks and mollymawks: an account of Disappointment Island Birds. Notornis 67: 213-245.

Waugh SM & Wilson KJ. 2017. Threats and threat status of the Westland Petrel *Procellaria westlandica*. Marine Ornithology 45: 195-203.

Waugh SM, Barbraud C, Adams L, Freeman AND, Wilson K-J, Wood G, Landers TJ & Baker GB. 2015. Modelling the demography and population dynamics of a subtropical seabird, and the influence of environmental factors. The Condor: Ornithological Applications 117: 147-164.

Waugh SM, Barbraud C, Delord K, Simister KLJ, Baker GB, Hedley GK, Wilson K-J & Rands DRD. 2020. Trends in density, abundance, and response to storm damage for Westland Petrels *Procellaria westlandica*, 2007-2019.

Waugh SM, Doherty PF, Freeman AND, Adams L, Wood GC, Bartle JA & Hedley GK. 2006. Demography of Westland Petrels (*Procellaria westlandica*), 1995–2003. Emu 106: 219-226.

Waugh SM, Sagar PJ & Cossee RO. 2001. New Zealand Black-browed Albatross *Diomedea melanophris impavida* and Grey-headed Albatross *D. chrysostoma* banded as Campbell Island: Recoveries from the Pacific Region. Emu 99: 29-35.

Waugh SM, Sagar PM & Paull D. 1997. Laying dates, breeding success and annual breeding of southern royal albatrosses *Diomedea epomophora* at Campbell Island during 1964–69. Emu 97: 194–199.

Waugh SM, Weimerskirch H, Moore PJ & Sagar PM. Population dynamics of Black-browed and Grey-headed Albatrosses *Diomedea melanophrys* and *D. chrysostoma* at Campbell Island, New Zealand, 1942-1996. Ibis 141: 216-225.

Weimerskirch H, Delord K, Barbraud C, Le Bouard F, Ryan PG, Fretwell P & Marteau C. 2018. Status and trends of albatrosses in the French Southern Territories, Western Indian Ocean. Polar Biology 41: 1963-1972.

Weimerskirch H, Zotier R, Jouventin P (1989) The Avifauna of the Kerguelen Islands. Emu - Austral Ornithology 89:15–29.

Wood GC & Otley HM. 2013. An assessment of the breeding range, colony sizes and population of the Westland Petrel (*Procellaria westlandica*). New Zealand Journal of Zoology 40: 186–195. Marine Ornithology 48: 273-281.

Zhang J, Bell EA & Roberts JO. 2020. Demographic assessment of black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island). New Zealand Aquatic Environment and Biodiversity Report N. 244. Fisheries New Zealand, Wellington, New Zealand.

SUPPLEMENTARY MATERIAL 2. DATA TABLES

Antipodean Albatross

Supplementary Table 1. Antipodes Island counts of Antipodean Albatross (ssp. *antipodensis*), including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π^B	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1969*	VPC & GC	BOG & BP	Antipodes I.	1,867	7 Feb – 1 Mar	2-5	NA	NA	NA	0.688^C	✓; ?	0.333^C	1.000	Unknown (low)	5,000-8,000^C	BP	6,500 (2,170-10,840)	Warham & Bell 1979; Walker & Elliott 2005
1978*	GC	C	Antipodes I.	1,096	Nov-Dec	40-49	NA	NA	-	-	-	NA	0.333^C	Unknown (low)	3,800^C	BP	5,384 (1,776-9,604)	Bell 1978; Walker & Elliott 2005
1994	GC	BP	Antipodes I.	4,635	Feb	1-5	0.748	NA	NA	-	-	NA	1.000	Medium	-	-	4,727 (3,175-6,283)	Walker & Elliott 2005
1995	GC	BP	Antipodes I.	5,757	Feb	1-5	0.748	NA	NA	-	-	NA	1.000	Medium	-	-	5,858 (3,921-7,795)	Walker & Elliott 2005
1996	GC	BP	Antipodes I.	5,148	Feb	1-5	0.791	0.538	NA	-	-	NA	1.000	Medium	-	-	5,234 (3,496-6,967)	Walker & Elliott 2005
1997	GC	BP	Antipodes I.	3,167	Feb	1-5	0.797	0.497	-	-	-	NA	0.612	Medium	-	-	5,258 (3,511-7,004)	Walker & Elliott 2005
1998	GC	BP	SA, MCBA	694	5-19 Feb	2-4	0.748	0.535	-	(0.985^D)	-	NA	0.121	High	5,736	BP	5,845 (4,535-7,146)	Walker & Elliott 2005
1999	GC	BP	SA, MCBA	621	5-19 Feb	2-4	0.626	0.472	-	(0.985^D)	-	NA	0.121	High	5,132	BP	5,271 (4,037-6,514)	Walker & Elliott 2005
2000	GC	BP	SA, B32, MCBA	711	5-19 Feb	2-4	0.774	0.472	-	(0.985^D)	-	NA	0.148	High	4,740	BP	4,890 (3,891-5,876)	Walker & Elliott 2005
2001	GC	BP	SA, B32, MCBA	744	5-19 Feb	2-4	0.784	0.571	-	(0.985^D)	-	NA	0.147	High	4,960	BP	5,147 (4,120-6,177)	Walker & Elliott 2005
2002	GC	BP	SA, B32, MCBA	931	5-19 Feb	2-4	0.678	0.448	-	(0.985^D)	-	NA	0.147	High	6,207	BP	6,487 (5,286-7,679)	Walker & Elliott 2005
2003	GC	BP	SA, B32, MCBA	1,009	5-19 Feb	2-4	0.731	0.518	-	(0.985^D)	-	NA	0.147	High	6,727	BP	7,004 (5,758-8,251)	Walker & Elliott 2005
2004	GC	BP	SA, B32, MCBA	1,220	5-19 Feb	2-4	0.719	0.474	-	(0.985^D)	-	NA	0.146	High	8,133	BP	8,535 (7,089-9,979)	Walker & Elliott 2005
2005	GC	BP	SA, B32, MCBA	1,010	5-19 Feb	2-4	NA	0.483	-	(0.985^D)	-	NA	0.146	High	6,733	BP	7,086 (5,811-8,371)	Walker & Elliott 2005
2007	GC	BP	SA, B32	246	Feb	1-5	0.575	-	-	-	-	NA	0.056	High	4,393	BP	4,530 (2,147-6,909)	Elliott & Walker 2017

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π [B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
2008	GC	BP	SA, B32	300	Feb	1-5	0.655	0.618	-	-	-	NA	0.056	High	5,357	BP	5,490 (2,894-8,105)	Elliott & Walker 2017
2009	GC	BP	SA, B32	218	Feb	1-5	0.550	0.283	-	-	-	NA	0.056	High	3,893	BP	4,016 (1,798-6,253)	Elliott & Walker 2017
2010	GC	BP	SA, B32	207	Feb	1-5	0.660	0.366	-	-	-	NA	0.056	High	3,696	BP	3,788 (1,658-5,959)	Elliott & Walker 2017
2011	GC	BP	SA, B32	196	Feb	1-5	0.517	0.413	-	-	-	NA	0.056	High	3,500	BP	3,618 (1,506-5,738)	Elliott & Walker 2017
2012	GC	BP	SA, B32, MCBA	544	Feb	1-5	0.557	0.397	-	-	-	NA	0.144	High	3,627	BP	3,896 (3,001-4794)	Elliott & Walker 2017
2013	GC	BP	SA, B32, MCBA	478	Feb	1-5	0.567	0.466	-	-	-	NA	0.144	High	3,187	BP	3,425 (2,596-4,247)	Elliott & Walker 2017
2014	GC	BP	SA, B32, MCBA	535	Feb	1-5	0.681	0.453	-	-	-	NA	0.143	High	3,567	BP	3,831 (2,943-4,718)	Elliott & Walker 2017
2015	GC	BP	SA, B32, MCBA	450	Feb	1-5	0.527	0.412	-	-	-	NA	0.143	High	3,000	BP	3,253 (2,447-4,068)	Elliott & Walker 2017
2016	GC	BP	SA, B32, MCBA	483	Feb	1-5	0.667	0.532	-	-	-	NA	0.143	High	3,220	BP	3,457 (2,632-4,297)	Elliott & Walker 2017
2017	GC	BP	SA, B32, MCBA	369	5-10 Feb	2	0.804	0.365	-	-	-	NA	0.142	High	2,460	BP	2,628 (1,915-3,336)	Elliott & Walker 2017
2018	GC	BP	SA, B32, MCBA	509	5-10 Feb	2	0.660	0.564	-	-	-	NA	0.142	High	3,393	BP	3,644 (2,780-4,502)	Elliott & Walker 2018
2019	GC	BP	SA, B32, MCBA	471	5-10 Feb	2	0.590	0.458	-	-	-	NA	0.142	High	3,140	BP	3,384 (2,562-4,205)	Elliott & Walker 2019
2020	GC	BP	SA	75	15-28 Mar	7-9	0.615	0.397	-	-	-	NA	0.028	High	2,679	BP	2,876 (75-6,758)	Elliott & Walker 2020
2021	GC	BP	SA, B32, MCBA	307	5-10 Feb	2	0.625	0.465	-	-	-	NA	0.096	High	3,198	BP	3,259 (2,087-4,421)	Walker & Elliott 2022
2022	GC	BP	SA, B32, MCBA	438	5-10 Feb	2	0.723	0.497	-	-	-	NA	0.141	High	2,920	BP	3,148 (2,359-3,943)	Walker & Elliott 2022
2023	GC	BP	SA, B32, MCBA	487	6-12 Feb	2	0.711	0.466	-	-	-	NA	0.140	High	3,247	BP	3,528 (2,678-4,383)	Parker et al. 2023
2024*	GC & AC^{UAV}	AOS	Antipodes I.	2,766	2 Feb – 2 Mar	2-5	0.681	0.500	-	(0.980[D])	✓[E]	$\psi^{ABA} = 0.698$ (0.489; 0.872)[E]	0.824	High	3,383 (3,182; 3,585)[F]	BP	3,384 (3,044-3,723)	Rexer-Huber et al. 2024
2025*	GC & AC^{UAV}	AOS	Antipodes I.	4,389	25 Jan-10 Feb	0-2	NA	NA	-	-	✓[E]	$\psi^{ABA} = 0.600$ (0.324; 0.723)[E]	0.985	High	3,546 (3,048; 4,044)[F]	BP	3,546 (3,192-3,901)	Rexer-Huber et al. 2025

[A] Mean lay is ~24 Jan and breeding period is considered ~50 weeks. [B] Proportion of breeding site based on retrospective assessment based on full island drone counts in Rexer-Huber et al. 2024, 2025.[C] Retrospectively estimated in Walker & Elliott 2005. [D] E_d estimated but deemed negligible and not included in reported adjusted count. [E] Day-specific $\psi^{lay} \cdot \psi^{fail}$ estimates were included for each daily count. [F] Based on habitat-related extrapolation.

Appendix Table 2. Adams Island counts of Antipodean Albatross (ssp. *gibsoni*), including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π ^{B}	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1973*	GC	BP	Adams I.	7,000	Early Feb	3-4	NA	NA	-	0.195^C	-	NA	1.000	Unknown (low)	20,000^C	BP	21,040 (6,928-35,150)	Robertson 1975, Walker & Elliott 2002
1991	GC & VPC	BP	RR, AA, AB	3,531	Early Feb	3-4	0.648	NA	-	-	-	✓; ?	0.754	High	-	-	4,832 (4,337-5,329)	Walker & Elliott 2002
1993	GC & VPC	BP	RR, AA, AB	3,746	Early Feb	3-4	0.778	NA	-	-	-	✓; ?	0.754	High	-	-	5,066 (4,551-5,583)	Walker & Elliott 2002
1994	GC & VPC	BP	RR, AA, AB	3,433	Early Feb	3-4	0.713	0.606	-	-	-	✓; ?	0.754	High	-	-	4,667 (4,187-5,149)	Walker & Elliott 2002
1995	GC, VPC & AC^{heli}	BP & AOS	RR, AA, AB	4,750	Early Feb	3-4	0.634	0.704	-	-	-	✓; ?	0.754	High	-	-	6,510 (5,850-7,173)	Walker & Elliott 2002
1996	GC	BP	SA	200	Early Feb	3-4	0.644	0.618	-	-	-	NA	0.029	High	-	-	7,119 (1,342-12,880)	Walker & Elliott 2002, Elliott et al. 2016
1997	GC, VPC & AC^{heli}	BP & AOS	Adams I.	6,993	23-31 Jan	2-3	0.676	0.622	NA	(0.994)^D	$\psi^{lay} \cdot \psi^{fail} = 0.944$	✓; ?	1.000	High	8,414	BP	6,992 (6,293-7,689)	Hamilton et al. 2000, Walker et al. 2002, Elliott et al. 2016
1998	GC	BP	RR, AA, FS	791	25 Jan – 20 Feb	2-6	0.637	0.604	-		$\psi^{lay} \cdot \psi^{fail} = 0.991$	NA	0.095	High	8,436	BP	8,324 (6,358-10,300)	Walker et al. 2002, Elliott et al. 2016
1999	GC	BP	RR, AA, FS	743	3 Jan – 6 Feb	0-4	0.612	0.564	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.996$	NA	0.095	High	7,886	BP	7,818 (5,907-9,720)	Walker & Elliott 2005, Elliott et al. 2016
2000	GC	BP	RR, AA, AB, FS	1,556	Jan	0-3	0.521	0.404	-	✓; ?	$\psi^{lay} \cdot \psi^{fail} = 0.982$	NA	0.259	High	4,904^E	BP	6,010 (5,225-6,792)	Walker & Elliott 2005, Elliott et al. 2016
2001	GC	BP	RR, AA, FS	675	Jan	0-3	0.650	0.684	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.956$	NA	0.095	High	7,463	BP	7,106 (5,301-8,909)	Walker & Elliott 2005, Elliott et al. 2016
2002	GC	BP	RR, AA, FS	714	Jan	0-3	0.620	0.513	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.965$	NA	0.095	High	7,822	BP	7,510 (5,649-9,373)	Walker & Elliott 2005, Elliott et al. 2016
2003	GC	BP	RR, AA, FS	784	Jan	0-3	0.530	0.604	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.991$	NA	0.095	High	8,362	BP	8,255 (6,299-10.220)	Walker & Elliott 2005, Elliott et al. 2016
2004	GC	BP	RR, AA, FS	862	Jan	0-3	0.540	0.603	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.975$	NA	0.095	High	9,345	BP	9,072 (6,990-11,150)	Walker & Elliott 2005, Elliott et al. 2016
2005	GC	BP	RR, AA, FS	429	Jan	0-3	0.379	0.476	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.949$	NA	0.095	High	4,778	BP	4,518 (3,118-5,908)	Walker & Elliott 2005, Elliott et al. 2016
2006	GC	BP	RR, AA, FS	279	Jan	0-3	0.255	0.460	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.818$	NA	0.095	High	3,605	BP	2,940 (1,834-4,039)	Elliott & Walker 2013, Elliott et al. 2016
2007	GC	BP	RR, AA, FS	400	Jan	0-3	0.243	0.533	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.930$	NA	0.095	High	4,545	BP	4,206 (2,865-5,554)	Elliott & Walker 2013, Elliott et al. 2016
2008	GC	BP	RR, AA, FS	318	Jan	0-3	0.326	0.570	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.933$	NA	0.095	High	3,605	BP	3,345 (2,156-4,533)	Elliott & Walker 2013, Elliott et al. 2016
2009	GC	BP	RR, AA, FS	386	Jan	0-3	0.242	0.634	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.906$	NA	0.095	High	4,503	BP	4,060 (2,742-5,381))	Elliott & Walker 2013, Elliott et al. 2016

Year	**Method**	**Counted metric**	**Part site**	**Reported raw count**	**Count date**	**Weeks post μ lay**[A]	**f**	**p^b**	**Ψ^s**	**Ψ^d**	**Ψ^p**	**Ψ^o**	**π** [B]	**Acc. category**	**Reported adjusted count**	**Reported adjusted metric**	**Standardised abundance $(y_{i,t})$**	**References**
2010	GC	BP	RR, AA, FS	383	Jan	0-3	0.310	0.656	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.977$	NA	0.095	High	4,144	BP	4,031 (2,726-5,338)	Elliott & Walker 2013, Elliott et al. 2016
2011	GC	BP	RR, AA, FS	425	Jan	0-3	0.432	0.624	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.970$	NA	0.095	High	4,630	BP	4,472 (3,100-5,858)	Elliott & Walker 2013, Elliott et al. 2016
2012	GC	BP	RR, AA, FS	379	Jan	0-3	0.417	0.665	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.907$	NA	0.095	High	4,419	BP	3,988 (2,685-5,289)	Elliott & Walker 2013, Elliott et al. 2016
2013	GC	BP	RR, AA, FS	492	23-31 Jan	2-3	0.497	0.696	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.948$	NA	0.095	High	5,486	BP	5,181 (3,680-6,692)	Elliott & Walker 2013, Elliott et al. 2016
2014	GC	BP	RR, AA, FS	430	23-31 Jan	2-3	0.302	0.553	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.909$	NA	0.095	High	5,000	BP	4,529 (3,126-5,922)	Elliott & Walker 2014, Elliott et al. 2016
2015	GC	BP	RR, AA, FS	381	23-31 Jan	2-3	0.419	0.661	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.938$	NA	0.095	High	4,292	BP	4,012 (2,702-5,315)	Walker & Elliott 2015, Elliott et al. 2016
2016	GC	BP	RR, AA, AB, FS	1,628	23 Jan – 9 Feb	2-5	0.676	0.656	-	✓; ?	$\psi^{lay} \cdot \psi^{fail} = 0.952$	$(\psi^{BOG} \cdot \psi^{ABA} = 0.740)^F$	0.259	High	5,385[E]	BP	6,283 (5,477-7,101)	Elliott et al. 2016, Baker et al. 2017
2017	GC	BP	RR, AA, FS	424	23-31 Jan	2-3	0.677	0.529	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.946$	NA	0.095	High	4,736	BP	4,461 (3,070-5,842)	Walker et al. 2017
2018	GC	BP	RR, AA, FS	475	Jan	0-3	0.576	0.574	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.971$	NA	0.095	High	5,169	BP	4,999 (3,533-6,479)	Rexer-Huber et al. 2019
2019	GC	BP	RR, AA, FS	403	23-31 Jan	2-3	0.544	0.636	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.953$	NA	0.095	High	4,471	BP	4,243 (2,902-5,592)	Rexer-Huber et al. 2019
2020	GC	BP	RR, AA, FS	376	23-31 Jan	2-3	NA	0.478	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.962$	NA	0.095	High	4,133	BP	3,959 (2,659-5,248)	Rexer-Huber et al. 2020
2022	GC	BP	RR, AA, FS	428	23-31 Jan	2-3	0.736	NA	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.953$	NA	0.095	High	4,746	BP	4,505 (3,120-5,888)	Parker et al. 2022
2023	GC & AC^{UAV}	BP	RR, AA, FS	481	23-31 Jan	2-3	0.599	0.749	-	-	$\psi^{lay} \cdot \psi^{fail} = 0.960$	$(\psi^{ABA} = 0.729)^G$	0.095	High	5,296	BP	5,063 (3,583-6,551)	Walker et al. 2023
2024	AC^{UAV}	BOG	Adams I.	5,559	21 Jan – 25 Feb	2-7	0.468	0.533	-	-	✓[G]	$\psi^{BOG} \cdot \psi^{ABA} = 0.540\ (0.350; 0.880)^H$	0.794	High	4,130 (3,904-4,361)	BP	4,130 (3,715-4,539)	Elliott et al. 2024
2025*	AC^{UAV}	BOG	Adams I.	-	13 Jan-2 Feb	0-3	NA	NA	-	-	✓[G]	$\psi^{BOG} \cdot \psi^{ABA} = 0.670\ (0.360\text{-}0.900)^H$	0.590	High	4,865 (4,793-4,929)	BP	4,864 (4,377-5,350)	Elliott et al. 2025

[A] Mean lay is ~10 Jan and breeding period is considered ~50 weeks. [B] Proportion of breeding site based on full island counts completed in 1997 and then updated in 2025 (Elliott et al. 2025). [C] Retrospectively estimated by Walker & Elliott 2002 by repeating the methodology in 1997. [D] Estimated but considered negligible. [E] Estimated but without the counts at AB included in the extrapolation. [F] Estimated during aerial survey trials. [G] Estimated during drone trials. [H] Day-specific $\psi^{lay} \cdot \psi^{fail}$ estimates were included for each daily count.

Appendix Table 3. Disappointment Island counts of Antipodean Albatross (ssp. *gibsoni*), including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1973	GC	BP	Disappointment I.	200	6 Jan & 15 Feb	0-5	NA	NA	-	-	-	-	1.000	Unknown (Low)	-	-	207 (68-346)^{1,2}	Robertson 1975
1993	GC & VPC	BP & AOS	Disappointment I.	-	17-18 Jan	1	NA	NA	-	-	-	0.911	1.000	Medium	226	BP	229 (154; 305)^{1,2}	Walker et al. 2021
2014	AC^{heli}	AOS	Disappointment I.	463	20 Jan	1	NA	NA	-	-	-	($\psi^{BOG} \cdot \psi^{ABA} = 0.560$)[B]	1.000	Unknown (Low)	452	AOS	293 (97; 497)^{1,2,3}	Baker & Jensz 2014A

[A] Mean lay is ~10 Jan and breeding period is considered ~50 weeks. [B] Ψ^o measured on Adams Island on the same day but not included in this count (Walker et al. 2021).

Northern Royal Albatross

Table 4. Motuhara | Fortyfours counts of Northern Royal Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database. **Bold** indicates count selected for inclusion in the data processing framework in scenarios where >1 counts were completed in a year.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^{A}	f	p^{b}	Ψ^{s}	Ψ^{d}	Ψ^{p}	Ψ^{o}	π^{B}	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1954*	GC	BP	Motuhara	2,000	1 Feb		-	-	-	-	-	NA	1.000	Unknown (Low)	-	-		Dawson 1955, Robertson 1991
1972*	AC^{fw}	C	Motuhara	1,752^{C}	Sep	42-46	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1973*	AC^{fw}	BOG	Motuhara	3,285	28 Nov	2	0.557^{D}	-	NA	-	-	$\psi^{BOG} = 0.688^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1974*	AC^{fw}	BOG	Motuhara	3,450^{F}	24 Oct	0	0.446^{D}	-	NA	-	-	$\psi^{BOG} = 0.688^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991
1975*	AC^{fw}	BOG	Motuhara	2,846	28 Nov	2	0.609^{D}	-	NA	-	-	$\psi^{BOG} = 0.688^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1989*	AC^{fw}	C	Motuhara	719	25 Sep	45	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1990*	AC^{fw}	BOG	Motuhara	4,562	2 Dec	3	0.027^{D}	-	NA	-	-	$\psi^{BOG} = 0.650^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1991*	AC^{fw}	BOG	Motuhara	4,520	2 Dec	3	0.091^{D}	-	NA	-	-	$\psi^{BOG} = 0.650^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1992*	AC^{fw}	C	Motuhara	118	28 Aug	41	-	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Rebergen 2000
1993*	AC^{fw}	BOG	Motuhara	5,982	3 Dec	3	0.038^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Rebergen 2000
1994*	AC^{fw}	BOG	Motuhara	2,178	20 Dec	5	-	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Rebergen 2000
1995*	AC^{fw}	BOG	Motuhara	5,542	1 Dec	2	0.133^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Rebergen 2000
1996*	AC^{fw}	BOG	Motuhara	3,828	5 Dec	3	0.181^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Rebergen 2000
1997*	AC^{fw}	BOG	Motuhara	3,893	5 Dec	3	0.098^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
1998*	AC^{fw}	BOG	Motuhara	4,893	30 Nov	2	0.344^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
1999*	AC^{fw}	BOG	Motuhara	3,570	23 Nov	1	0.254^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
2000*	AC^{fw}	BOG	Motuhara	1,957	27 Dec	2	0.449^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
2007	AC^{fw}	BOG	Motuhara	1,879	10 Dec	4	0.542^{D}	-	NA	-	-	-	1.000	High	-	-		Scofield 2011

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π [B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
2008	**GC**	BP	Motuhara	2,125	29 Nov	2	0.479	-	NA	-	-	$\psi^{ABA} \cdot \psi^{BOG}$ =	1.000	High	-	-		Scofield 2011
		BOG		2,212	29 Nov	2	0.460[D]	-	-	-	-	0.961	1.000	High	-	-		
	AC^{fw}											-						
2009	AC^{fw}	BOG	Motuhara	2,055	28 Nov	2	0.504[D]	-	NA	-	-	-	1.000	High	-	-		Scofield 2011
2010	AC^{fw}	BOG	Motuhara	2,692	28 Nov	2	0.402[D]	-	NA	-	-	-	1.000	High	-	-		Scofield 2011
2016	AC^{sat}	BOG	Motuhara	2,632	12-19 Feb	13-14	-	-	NA	-	-	-	1.000	Medium	-	-		Fretwell et al. 2017
2017	AC^{fw}	BOG	Motuhara	1,803	23 Nov	1	0.580[G]	-	NA	-	-	ψ^{BOG} = 0.943[H]	1.000	Medium	1,581	AOS		Baker et al. 2017,
												ψ^{BOG} = 0.864[I]			(1,502-1,661)			Frost 2017
	GC	BP	Motuhara	1,400	8 Dec	3	0.716	-	NA	-	-	ψ^{ABA} =0-916-	1.000	Medium	-	-		M Bell et al. 2017B
	AC^{sat}	BOG	Motuhara	2,533	20 Dec	5	-	-	NA	-	-	0.957	1.000	Medium	-	-		
												-						
2018	AC^{fw}	BOG	Motuhara	1,850	4 Dec	3	0.667[G]	-	NA	-	-	ψ^{BOG} = 0.967[H]	1.000	High	1,789	AOS		Frost 2019, 2021A
															(1,704-1,874)			
2020	AC^{fw}	C	Motuhara	1,059	15 Sep	44	-	-	NA	-	✓[J]	NA	1.000	High	1,203	C		Frost 2021A
2021	**AC^{fw}**	BOG	Motuhara	1,782	18 Dec	5	-	-	NA	-	-	ψ^{BOG} = 0.954[H]	1.000	High	1,696	AOS		Frost 2021B
															(1,616-1,779)			
	GC	BP	NRA1-5	146	28 Jan	11	-	-	-	-	-	-	0.110	High	-	-		Bell 2021
2022	GC	BP	NRA1-5	148	29 Jan	11	-	-	-	-	-	NA	0.110	High	-	-		Bell 2022
	AC^{fw}	BOG	Motuhara	1,705	1 Feb	11	-	-	NA	-	-	ψ^{BOG} = 0.957[H]	1.000	High	1,604	AOS		Frost 2022
															(1,501-1,704)			
2023	**GC**	BP	Motuhara	1,498	13 Dec	4	0.804	-	NA	-	-	NA	1.000	High	-	-		Bell 2023
	AC^{fw}	BOG	Motuhara	2,366	20 Dec	5	0.692[G]	-	NA	-	-	ψ^{BOG} = 0.748[H]	1.000	High	1,744	AOS		Frost 2024A
															(1,461-2,041)			
2024	**AC^{fw}**	BOG	Motuhara	2,178	9 Dec	4	-	-	NA	-	-	ψ^{BOG} = 0.944[H]	1.000	High	1,853	AOS		Frost 2024B
															(1,770-1,939)			
	GC	BP	Motuhara	1,472	20 Jan	10	-	-	NA	-	-	NA	1.000	High	-	-		Bell 2024
2025*	GC	C	NRA1-5	134	24 Feb	15	-	-	NA	-	-	NA	0.110	High	-	-		M Bell unpub.
2026*	GC	BP	Motuhara	1,344	1 Dec	2	-	-	NA	-	-	NA	1.000	High	-	-		Bell 2026

[A] Mean lay date ~ 14 Nov and breeding period is considered ~46 weeks (Robertson & Richdale 1993). [B] Proportion represented by aerial study grids NRA1-5 based on 2023-2024 averages (Bell 2023, Frost 2024A, Bell 2024). [C] Robertson (1991) only reports C for all Chatham Islands breeding sites in 1972, so mean proportion between breeding sites over 1973-1975 was used to allocate chicks to breeding sites. [D] Calculated using BOG rather than BP. [E] Robertson (1991) applies two estimates of ψ^{BOG} based on Pukekura data as an academic exercise, but these estimates were not based on local data, and counts are considered expressed in BOG. [F] Disagreement between Robertson (1991) and Rebergen (2000) on this count and we favoured the original count here. [G] Calculated using AOS rather than BP. [H] Based on close-up photographs. [I] Based on ground transects. [J] Adjustments made based on Pukekura fledging curve (Frost 2021A).

Table 5. Sisters counts of Northern Royal Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database. **Bold** indicates count selected for inclusion in the data processing framework in scenarios where >1 counts were completed in a year.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^{A}	*f*	p^{b}	Ψ^{s}	Ψ^{d}	Ψ^{p}	Ψ^{o}	π^{B}	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1954*	GC	BP	Little Sister	1,000	26 Dec		-	-	-	-	-	NA	0.395	Unknown (Low)	-	-		Dawson 1954, Robertson 1991
1972*	AC^{fw}	C	Little & Big Sister	848^{C}	Sep	42-46	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Robertson 1991
1973*	AC^{fw}	BOG	Little & Big Sister	1,996	28 Nov	2	0.261^{D}	-	NA	-	-	$\psi^{BOG} = 0.688^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1974*	AC^{fw}	BOG	Little & Big Sister	2,500^{F}	24 Oct	0	0.517^{D}	-	NA	-	-	$\psi^{BOG} = 0.688^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1975*	AC^{fw}	BOG	Little & Big Sister	1,868	28 Nov	2	0.417^{D}	-	NA	-	-	$\psi^{BOG} = 0.688^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991
1976*	AC^{fw}	C	Little Sister	446	?	37-46^{G}	-	-	-	-	-	NA	0.395	Unknown (Low)	-	-		Robertson & Gales 1998
1989*	AC^{fw}	C	Little & Big Sister	380	25 Sep	45	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1990*	AC^{fw}	BOG	Little & Big Sister	3,763	2 Dec	3	0.204^{D}	-	NA	-	-	$\psi^{BOG} = 0.650^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1991*	AC^{fw}	BOG	Little & Big Sister	3,380	2 Dec	3	0.270^{D}	-	NA	-	-	$\psi^{BOG} = 0.650^{E}$	1.000	Unknown (Low)	-	-		Robertson 1991, Rebergen 2000
1992*	AC^{fw}	BOG	Little Sister	1,101	30 Nov	1	0.262^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Robertson & Gales 1998, Rebergen 2000
1993*	AC^{fw}	BOG	Little & Big Sister	4,428	3 Dec	3	0.096^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Robertson & Gales 1998, Rebergen 2000
1994*	AC^{fw}	BOG	Little & Big Sister	3,133	20 Dec	5	0.227^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Robertson & Gales 1998, Rebergen 2000
1995*	AC^{fw}	BOG	Little & Big Sister	3,816	1 Dec	2	0.343^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Robertson & Gales 1998, Rebergen 2000
1996*	AC^{fw}	BOG	Little & Big Sister	2,902	5 Dec	3	0.268^{D}	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Rebergen 2000
1997*	AC^{fw}	BOG	Little & Big Sister	3,535	5 Dec	3	0.292^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
1998*	AC^{fw}	BOG	Little & Big Sister	3,764	30 Nov	2	0.351^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
1999*	AC^{fw}	BOG	Little & Big Sister	3,396	23 Nov	1	0.221^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
2000*	AC^{fw}	BOG	Little & Big Sister	2,774	27 Dec	2	0.293^{D}	-	NA	-	-	-	1.000	Unknown (Low)				Rebergen 2000
2007	AC^{fw}	BOG	Little & Big Sister	3,509	10 Dec	4	0.439^{D}	-	NA	-	-	-	1.000	High	-	-		Scofield 2011

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π [B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
2008	AC^{fw}	BOG	Little & Big Sister	3,389	29 Nov	2	0.213[D]	-	NA	-	-	-	1.000	High	-	-		Scofield 2011
2009	AC^{fw}	BOG	Little & Big Sister	3,397	28 Nov	2	0.288[D]	-	NA	-	-	-	1.000	High	-	-		Scofield 2011
2010	AC^{fw}	BOG	Little & Big Sister	3,052	28 Nov	2	0.404[D]	-	NA	-	-	-	1.000	High	-	-		Scofield 2011
2016	AC^{sat}	BOG	Little & Big Sister	1,805	29 Dec	13-14	-	-	NA	-	-	-	1.000	Medium	-	-		Fretwell et al. 2017
2017	**AC^{fw}**	BOG	Little & Big Sister	3,269	23 Nov	1	0.394[H]	-	NA	-	-	ψ^{BOG} = 0.932[I]	1.000	Medium	2,824 (2,718-2,931)	AOS		Baker et al. 2017, Frost 2017
	AC^{sat}	BOG	Little & Big Sister	2,578	20 Dec	5	-	-	NA	-	-	-	1.000	Medium		-		
2018	**GC**	BP	Little & Big Sister	2,092	30 Nov-4 Dec	2-3	0.457	-	NA	1.000	-	ψ^{ABA} = 0.941	1.000	High	-	-		M Bell et al. 2018
	AC^{fw}	BOG	Little & Big Sister	2,165	4 Dec	3	0.448[H]	-	NA	-	-	ψ^{BOG} = 0.982[J] ψ^{BOG} = 0.970[K]	1.000	High	2,130	AOS		Frost 2019, 2021A
2020	AC^{fw}	C	Little & Big Sister	739	15 Sep	44	-	-	NA	-	✓[K]	NA	1.000	High	840	C		Frost 2021A
2021	AC^{fw}	BOG	Little & Big Sister	2,392	18 Dec	5	-	-	NA	-	- -	ψ^{BOG} = 0.953[J] ψ^{BOG} = 0.959[K]	1.000	High	2,298 (2,168-2,435)	AOS		Frost 2021B
2022	AC^{fw}	BOG	Little & Big Sister	1,841	1 Feb	11	-	-	NA	-	-	ψ^{BOG} = 0.944[J] ψ^{BOG} = 0.942[K]	1.000	High	1,668 (1,517-1,843)	AOS		Frost 2022
2023	AC^{fw}	BOG	Little & Big Sister	2,733	20 Dec	5	0.569[H]	-	NA	-	-	ψ^{BOG} = 0.741[J] ψ^{BOG} = 0.844[K]	1.000	High	2,405 (1,922-2,991)	AOS		Frost 2024A
2024	AC^{fw}	BOG	Little & Big Sister	2,795	9 Dec	4	-	-	NA	-	-	ψ^{BOG} = 0.906[J] ψ^{BOG} = 0.953[K]	1.000	High	2,469 (2,333-2,609)	AOS		Frost 2024B
2025*	GC	C	Little Sister	517	26 Aug	41	-	-	-	-	-	NA	0.395	High	-	-		Bell 2026

[A] Mean lay date ~ 14 Nov and breeding period is considered ~46 weeks (Robertson & Richdale 1993). [B] Proportions between both islands based on 2007-2024 average. [C] Robertson 1991 only reports C for all Chatham Islands breeding sites in 1972, so mean proportion between breeding sites over 1973-1975 was used to allocate chicks to breeding sites. [D] Calculated using BOG rather than BP. [E] Robertson (1991) applies two estimates of ψ^{BOG} based on Pukekura data as an academic exercise, but these estimates were not based on local data, and counts are considered expressed in BOG. [F] Disagreement between Robertson (1991) and Rebergen (2000) on this count and we favoured the original count here. [G] Without information on count date, assumed to have occurred between 1 Aug and 30 Sep. [G] Calculated using AOS rather than BP. [H] Estimated using close-up photographs. [I] Estimated on Big Sister using close-up photographs. [J] Estimated on Little Sister using close-up photographs. [K] Adjustments made based on Pukekura fledging curve (Frost 2021A).

Table 6. Pukekura counts of Northern Royal Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1936*	GC	BP	Pukekura	1	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1937*	GC	BP	Pukekura	1	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1938*	GC	BP	Pukekura	2	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1939*	GC	BP	Pukekura	4	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1940*	GC	BP	Pukekura	5	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1941*	GC	BP	Pukekura	3	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1942*	GC	BP	Pukekura	5	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1943*	GC	BP	Pukekura	2	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1944*	GC	BP	Pukekura	3	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1945*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1946*	GC	BP	Pukekura	0	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1947*	GC	BP	Pukekura	3	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1948*	GC	BP	Pukekura	3	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1949*	GC	BP	Pukekura	4	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1950*	GC	BP	Pukekura	5	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1951*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1952*	GC	BP	Pukekura	7	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1953*	GC	BP	Pukekura	4	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1954*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1955*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1956*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1957*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1958*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1959*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1960*	GC	BP	Pukekura	3	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1961*	GC	BP	Pukekura	5	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1962*	GC	BP	Pukekura	4	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1963*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1964*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1965*	GC	BP	Pukekura	7	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1966*	GC	BP	Pukekura	7	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1967*	GC	BP	Pukekura	6	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1968*	GC	BP	Pukekura	8	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1969*	GC	BP	Pukekura	9	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1970*	GC	BP	Pukekura	8	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1971*	GC	BP	Pukekura	8	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1972*	GC	BP	Pukekura	7	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1973*	GC	BP	Pukekura	10	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1974*	GC	BP	Pukekura	7	Continuous ^{B}	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1975*	GC	BP	Pukekura	10	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1976*	GC	BP	Pukekura	6	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1977*	GC	BP	Pukekura	10	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1978*	GC	BP	Pukekura	8	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1979*	GC	BP	Pukekura	7	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1980*	GC	BP	Pukekura	9	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1981*	GC	BP	Pukekura	8	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1982*	GC	BP	Pukekura	11	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1983*	GC	BP	Pukekura	9	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1984*	GC	BP	Pukekura	11	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1985*	GC	BP	Pukekura	12	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1986*	GC	BP	Pukekura	11	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1987*	GC	BP	Pukekura	18	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1988*	GC	BP	Pukekura	15	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1989*	GC	BP	Pukekura	19	Continuous^B	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
1990*	GC	BP	Pukekura	22	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1991*	GC	BP	Pukekura	15	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1992*	GC	BP	Pukekura	17	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1993*	GC	BP	Pukekura	19	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1994*	GC	BP	Pukekura	20	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1995	GC	BP	Pukekura	23	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1996*	GC	BP	Pukekura	21	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1997*	GC	BP	Pukekura	18	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1998*	GC	BP	Pukekura	24	Continuous^B	0	0.458^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
1999*	GC	BP	Pukekura	12	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2000*	GC	BP	Pukekura	11	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2001*	GC	BP	Pukekura	20	Continuous^B	0	-	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2002*	GC	BP	Pukekura	24	Continuous^B	0	0.174^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2003*	GC	BP	Pukekura	34	Continuous^B	0	0.774^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2004	GC	BP	Pukekura	17	Continuous^B	0	0.588^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2005*	GC	BP	Pukekura	30	Continuous^B	0	0.655^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2006*	GC	BP	Pukekura	17	Continuous^B	0	0.500^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2007*	GC	BP	Pukekura	15	Continuous^B	0	0.786^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2008*	GC	BP	Pukekura	21	Continuous^B	0	0.400^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2009	GC	BP	Pukekura	32	Continuous^B	0	0.563^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2010	GC	BP	Pukekura	21	Continuous^B	0	0.762^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2011	GC	BP	Pukekura	30	Continuous^B	0	0.700^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2012	GC	BP	Pukekura	31	Continuous^B	0	0.548^D	0.567^C	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2013	GC	BP	Pukekura	36	Continuous^B	0	0.694^D	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2014*	GC	BP	Pukekura	32	Continuous^B	0	0.742^D	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2015*	GC	BP	Pukekura	32	Continuous^B	0	0.813^D	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2016	GC	BP	Pukekura	35	Continuous^B	0	0.743^D	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026

Year	**Method**	**Counted metric**	**Part site**	**Reported raw count**	**Count date**	**Weeks post μ lay**[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	**Acc. category**	**Reported adjusted count**	**Reported adjusted metric**	**Standardised abundance** $(y_{i,t})$	**References**
2017	GC	BP	Pukekura	38	Continuous[B]	0	0.632[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2018	GC	BP	Pukekura	33	Continuous[B]	0	0.394[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2019	GC	BP	Pukekura	51	Continuous[B]	0	0.529[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2020	GC	BP	Pukekura	45	Continuous[B]	0	0.727[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2021	GC	BP	Pukekura	41	Continuous[B]	0	0.732[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2022	GC	BP	Pukekura	36	Continuous[B]	0	0.694[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2023	GC	BP	Pukekura	49	Continuous[B]	0	0.653[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2024	GC	BP	Pukekura	45	Continuous[B]	0	0.733[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2025*	GC	BP	Pukekura	44	Continuous[B]	0	0.864[D]	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026
2026*	GC	BP	Pukekura	47	Continuous[B]	0	-	-	NA	-	-	NA	1.000	High	-	-		DOC Coastal Otago 2026

[A] Mean lay date before 1968 = 15 Nov, 1978-2000 = 12 Nov, >2000 = 10 Nov and breeding period is considered ~46 weeks (Robertson 1993, DOC unpub.). [B] Breeding site virtually continuously monitored since its establishment and thus no clear count date apply, and instead breeding attempts are assumed to be recorded at or near laying (DOC unpub.). [C] Reported here but not included in any of the models for other breeding sites as Pukekura breeding success is manipulated to achieve maximum *f* through management (Robetson 2001, Richard et al. 2015) [D] Multi-year average estimated by Richard et al. (2015).

Southern Royal Albatross

Supplementary Table 7. Campbell Island counts of Southern Royal Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π^B	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1958	GC & VPC	BP, AOS & BOG	Campbell I.	2,278	7 Jan-19 Feb	4-11	0.743	-	NA	-	-	-	1.000	Medium	-	-		Westerkov 1963, Dilks & Wilson 1979, Dilks & Grindell 1983
1966*^{C}	GC & VPC	BP, AOS & BOG	Campbell I.	3,216 ^{C}	Nov-Jan	0-8	0.639^C	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Guard 1968, Paul & Surrey 1969, Moore et al. 1997a, Waugh et al. 1997
1969	GC & VPC	BP, AOS & BOG	Campbell I.	4,344	8-26 Jan	5-7^D	-	0.510	NA	-	-	-	1.000	Medium	-	-		Taylor et al. 1970, Dilks & Wilson 1979, Waugh et al. 1997
1971*	GC & VPC	BP, AOS & BOG	Campbell I., South of fence	1,709	31 Jan-10 Mar	8-14	0.610	-	-	-	-	-	0.361	Unknown (Low)	-	-		Dilks & Wilson 1979, Moore et al. 1997a
1976	GC & VPC	BP, AOS & BOG	Campbell I.	5,336	12 Dec-14 Feb	2-8^D	-	-	NA	-	-	-	1.000	Medium	-	-		Dilks & Wilson 1979, Moore et al. 1997a
1977	GC & VPC	BP, AOS & BOG	Campbell I.	4,906	1 Jan-13 Feb	4-8^D	-	-	NA	-	-	-	1.000	Medium	-	-		Dilks & Wilson 1979, Moore et al. 1997a
1978	GC & VPC	BP, AOS & BOG	Campbell I.	4,208	23 Dec-11 Feb	3-10	-	-	NA	-	-	-	1.000	Medium	-	-		Dilks & Dunn 1978, Moore et al. 1997a
1980	GC & VPC	BP, AOS & BOG	Campbell I.	4,575	23 Dec-15 Jan	3-11	-	-	NA	-	-	-	1.000	Medium	-	-		Wilson & Elliott 1981, Moore et al. 1997a
1981	GC & VPC	BP, AOS & BOG	Campbell I.	4,493	29 Dec-7 Feb	4-9	-	-	NA	-	-	-	1.000	Medium	-	-		Wilson & Elliott 1981, Moore et al. 1997a
1983	GC & VPC	BP, AOS & BOG	Campbell I.	4,243	14 Jan-11 Feb	6-10	-	-	NA	-	-	-	1.000	Medium	-	-		Dilks & Grindell 1983, Moore & Moffat 1990, Moore et al. 1997a
1988	GC	BP	Col, Moubray	472	3-8 Jan	4-5	0.744	-	-	-	-	NA	0.089	Medium	-	-		Moore & Moffat 1990
1992	GC	BP	Col, Moubray	534	Early Jan	4-6	0.782	-	-	-	-	NA	0.089	Medium	-	-		Moore et al. 1997a
1993	GC	BP	Col, Moubray	587	1-7 Jan	4-5	-	-	-	-	-	NA	0.089	Medium	-	-		Moore et al. 1997a
1994	GC	BP	Col, Moubray	605	Early Jan	4-6	0.815	-	-	-	-	NA	0.089	Low	-	-		Moore et al. 1997a
1995	GC & VPC	BP, AOS & BOG	Campbell I.	6,308	10 Jan-5 Feb	5-9	0.765	0.520^E	NA	ψ^d_{GC} =0.967, ψ^d_{VPC} = 0.867	ψ^{fail} = 0.943-1.000	-	1.000	Medium	6,939-7,302	BP		Moore et al. 1997ab, 2012, ACAP 2009
1996	GC (& VPC) ^{F}	BP	Campbell I.	7,787	4 Jan-6 Feb	5-9	0.769	0.520^E	NA	ψ^d_{GC} = 0.998, ψ^d_{VPC} = 0.720-0.770	ψ^{fail} = 0.960-1.000	-	1.000	Medium	8,075-8,414	BP		Moore unpub., Moore et al. 1997a, 2012, ACAP 2012

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1997	GC	BP	Col, Moubray, Faye, Paris, Honey	2,121	26 Nov-19 Feb	0-11	0.778	0.520[E]	-	0.996	-	NA	0.257	Medium	8,250-8,560	BP		Moore unpub., ACAP 2012, Moore et al. 2012
1998	GC	BP	Col, Moubray, Faye, Paris, Honey	2,247	Jan-Feb	4-12	0.797	0.520[E]	-	0.990	-	NA	0.257	Medium	-	-		Moore unpub., ACAP 2012, Moore et al. 2012
1999	GC	BP	Col, Moubray, Faye, Paris, Honey	2,323	Dec-Jan	0-8	-	-	-	0.992	-	NA	0.257	Medium	-	-		Moore et al. 2012
2005	GC	BP	Col, Moubray, Faye	1,311	1 Jan-20 Feb	4-11	0.705	-	-	0.997	ψ^{fail} = 0.943	NA	0.163	High	8,300-8,700[G]	BP		Moore et al. 2012
2006	GC	BP	Col, Moubray, Faye	1,184	1 Jan-15 Feb	4-11	0.726	-	-	1.000	ψ^{fail} = 0.880	NA	0.163	High	8,300-8,700[G]	BP		Moore et al. 2012
2007	GC	BP	Col, Moubray, Faye, Paris, Honey	1,916	21 Dec-17 Feb	3-11	0.682	-	-	1.000	ψ^{fail} = 0.943	NA	0.257	High	8,300-8,700[G]	BP		Moore et al. 2012
2008	GC	BP	Col, Moubray, Faye, Paris, Honey	1,987	31 Dec-20 Feb	4-11	0.621	-	-	1.000	ψ^{fail} = 0.907	NA	0.257	High	8,300-8,700[G]	BP		Moore et al. 2012
2009	GC	BP	Col, Moubray, Faye, Paris, Honey	2,204	21 Dec-9 Feb	3-10	-	-	-	-	ψ^{fail} = 0.942	NA	0.257	High	8,300-8,700[G]	BP		Moore et al. 2012
2020	GC	BP	Col	104	12-16 Mar	14	-	-	-	-	(ψ^{fail} = 0.759)[G]	NA	0.025	High	137	BP		Mischler 2020
2023	GC	BP	Col	49[I]	16 Feb	11	-	-	-	-	-	NA	0.012[H]	Medium	-	-		Mischler & Wickes 2023
2024	GC	BP	Col, Moubray, Faye, Paris, Honey	1,480	3 Dec-7 Feb	0-9	0.667	-	-	0.987	-	NA	0.257	High	-	-		Mischler et al. 2024, Fischer & Thompson 2025
2025*	GC	BP	Col, Moubray, Faye, Paris, Honey	1,502	5 Dec-4 Feb	0-9	0.655	-	-	1.000	-	NA	0.257	High	-	-		Mischler et al. 2025, Fischer & Thompson 2025
2026*	GC	BP	Col, Moubray, Faye, Paris, Honey	1,564	14 Dec-22-Jan	1-7	-	-	-	-	-	NA	0.257	High	-	-		Mischler et al. in prep.

[A] Mean lay is ~3 December (Waugh et al. 1997). [B] Proportions estimated based on 2004-08 means. [C] Only a multi-year estimate for 1965-1968 available, here labelled 1966 (Moore et al. 1997b). [D] Weeks since laying directly sourced from Dilks & Wilson 1979 instead of calculating based on reported survey dates. [E] Multi-year estimate available only (Moore et al. 2012). [F] Only a very small section of the overall island was counted using VPCs due to recognition of inadequacies in this count method (Moore et al. 1997a) and thus considered negligible here. [G] Multi-year estimate only reported (Moore et al. 2012). [G] Estimate based on previous counts and thus not include here and instead repeated to enable propagation of uncertainty. [H] Only part of part site counted and subsequently survey effort was used to estimate proportion of part site counted (Mischler & Wickes 2023).

Supplementary Table 8. Enderby Island counts of Southern Royal Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database. **Bold** indicates count selected for inclusion in the data processing framework in scenarios where >1 counts were completed in a year.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1943*^{B}	GC	BP	Enderby I.	8	?^{C}	?^{C}	-	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Bell 1963, French et al. 2020
1954	GC	BP?	Enderby I.	3	12-23 Mar	14-16	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Falla 1954, Dell 1954, Childerhouse et al. 2003
1963*	GC	BP?	Enderby I.	10	?^{C}	?^{C}	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Taylor 1971, Childerhouse et al. 2003
1966*	GC	BP	Enderby I.	15	Jan	4-8	-	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Taylor 1971, Childerhouse et al. 2003
1973	GC	BP	Enderby I.	17	?^{C}	?^{C}	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Russ 1974, Childerhouse et al. 2003, French et al. 2020
1974*	GC	BP	Enderby I.	20	5-20 Dec	1-3	-	-	NA	-	-	-	1.000	Unknown (Low)	-	-		Russ 1974, Childerhouse et al. 2003
1980*	GC	BP	Enderby I.	16	31 Jan	8	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Mitchell & Ensor 1986, French et al. 2020
1981*	GC	BP	Enderby I.	15	?^{C}	?^{C}	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Bell 1982, Childerhouse et al. 2003
1982*	GC	BP	Enderby I.	16	4 Feb	9	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		French et al. 2020
1988*	GC	BP	Enderby I.	33	7-15 Feb	9-11	-	-	NA	-	-	NA	1.000	Unknown (Low)	-	-		Taylor 1988, Childerhouse et al. 2003
1993	GC	BP	Enderby I.	32	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
1994*	GC	BP	Enderby I.	45	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
1995*	GC	BP	Enderby I.	52	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
1996*	GC	BP	Enderby I.	55	Late Dec-Feb	3-12	0.740^D	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
1997*	GC	BP	Enderby I.	47	Late Dec-Feb	3-12	0.740^D	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
1998*	GC	BP	Enderby I.	56	Late Dec-Feb	3-12	0.740^D	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
1999*	GC	BP	Enderby I.	42	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
2000*	GC	BP	Enderby I.	67	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	**f**	**p^b**	**Ψ^s**	**Ψ^d**	**Ψ^p**	**Ψ^o**	**π**	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
2001	GC	BP	Enderby I.	41	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
2002*	GC	BP	Enderby I.	69	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		Childerhouse et al. 2003
2003	GC	BP	Enderby I.	54	1 Dec-20 Mar	0-15	-	-	NA	-	-	NA	1.000	High	-	-		Wilkinson unpub.
2004	GC	BP	Enderby I.	55	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2005	GC	BP	Enderby I.	56	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2006	GC	BP	Enderby I.	66	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2007	GC	BP	Enderby I.	49	3 Dec-26 Feb	0-12	-	-	NA	-	-	NA	1.000	High	-	-		Chilvers 2007
2008	GC	BP	Enderby I.	64	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2009	GC	BP	Enderby I.	53	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2010	GC	BP	Enderby I.	57	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2011	GC	BP	Enderby I.	38	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2012	GC	BP	Enderby I.	77	?[C]	?[C]	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2013	AC^{heli} **GC**	BOG BP	Enderby I. Enderby I.	74 52	17 Jan 17-31 Jan	6 6-8	- -	- -	NA NA	- -	- -	ψ^{BOG} = 0.761 -	1.000 1.000	High High	54 -	AOS -		Baker & Jensz 2013
2014	AC^{heli} **GC**	BOG BP	Enderby I. Enderby I.	80 52	20 Jan 8-15 Feb	7 10-12	- -	- -	NA NA	- -	- -	ψ^{BOG} = 0.768 NA	1.000 1.000	High High	63 -	AOS -		Baker et al. 2014A
2015	GC	BP	Enderby I.	60	28 Jan-16 Mar	8-15	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2016	GC	BP	Enderby I.	44	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2017	AC^{heli} **GC**	BOG BP	Enderby I. Enderby I.	69 47	21 Jan 26 Jan	7 7	- -	- -	NA NA	- -	- -	ψ^{BOG} = 0.851 ψ^{BOG} = 0.839, ψ^{ABA} = 0.904	1.000 1.000	High High	57 -	AOS -		Baker et al. 2020
2018	GC	BP	Enderby I.	49	Late Dec-Feb	3-12	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2019	GC	BP	Part of Enderby I.	29	Late Jan–early Feb	7-10	-	-	-	-	-	NA	0.740[E]	Low	39	BP		DOC unpub.
2023	GC	AOS	Enderby I.	49	27 Dec	3	-	-	NA	-	-	ψ^{ABA} = 0.959	1.000	High	47	BP		DOC unpub.
2024	GC	BP	Enderby I.	63	17-30 Dec	2-4	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2025*	GC	BP	Enderby I.	56	19-23 Dec	2-3	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.
2026*	GC	BP	Enderby I.	70	19 Dec-13 Jan	2-6	-	-	NA	-	-	NA	1.000	High	-	-		DOC unpub.

[A] Mean lay is ~3 December (Waugh et al. 1997. [B] Only a multi-year estimate for 1941-1945 available, here labelled 1943 (French et al. 2020). [C] For unknown count dates prior to the formalisation of the methodology (Childerhouse et al. 2003) the full range of reported count dates (0-16 post mean laying) is used, after the formalisation of the methodology, the prescribed survey time period (3-12 weeks post mean laying) is used. [D] Multi-year estimate only reported (Childerhouse et al. 2003). [E] Incomplete coverage of the island corrected using the mean distribution of nesting area in 2023 and 2024 (P Frost unpub.).

Campbell Albatross

TBC

Buller's Albatross

Table 10. Hautere | Solander counts of Southern Buller's Albatross (ssp. *bulleri*), including updated count methods, metrics, and error specifications, as well as subsequent standardised data for ACAP IG Trends. Superscript numbers refer to numbered equations in main text applied to standardise counts. * Indicates counts absent from the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1985	GC	C	Little Solander	143	Late July	26	-	-	-	0.960	ψ^{fail} = 0.696[C]	NA	0.067	Unknown (Low)	205	BP		Cooper et al. 1986, Sagar & Stahl 2005
1996	AC^{heli} & GC & VPC	AOS	Great & Little Solander	?[D]	15-22 Feb	4-5	0.703	-	-	-	-	ψ^{BOG} = 0.965	0.633[E]	Unknown (Low)	2,625	AOS		Sagar et al. 1999, Sagar & Stahl 2005, Stahl unpub.
2002	AC^{heli} & GC & VPC	BOG	Great & Little Solander	?[D]	20 Feb-8 Mar	4-7	0.676	-	NA	ψ^d_{VPC} = 0.900-0950	-	ψ^{BOG} = 0.800 (0.760-0.850)	1.000	Unknown (Low)	4,912 (4,628-5,145	AOS		Sagar & Stahl 2005, Stahl unpub.
2014	AC^{heli}	BOG	Little Solander	182	20 Jan	0	-	-	-	-	ψ^{lay} = 0.420	ψ^{BOG} = 0.970 ψ^{ABA} = 0.805	0.067	Medium	305	AOS		Baker & Jensz 2014B
2016	AC^{heli} & GC & VPC	BOG	Great & Little Solander	6,852[F]	25-29 Feb	5-6	-	-	NA	ψ^d_{VPC} = 0.950	-	ψ^{BOG} = 0.813 ψ^{ABA} = 0.821[G]	1.000	Medium	5,620	AOS		Thompson et al. 2017
2024*	AC^{heli}	BOG	Great & Little Solander	6,761	9 Mar	7	-	-	NA	-	-	ψ^{BOG} = 0.969 ψ^{ABA} = 0.611[G]	1.000	Unknown (Medium)	5,377 (5,234-5,523)	AOS		Frost et al. 2025

[A] Mean lay date used 21 Jan based on "late January in Sagar & Warham (1998). [B] Proportion represented by Little Solander based on average proportion across the full site counts (1996, 2002, 2016, and 2024). [C] Based on a multi-year *f* estimate covering 1996, 1997, 2002, and 2003 (Sagar & Stahl 2005). [D] Adjusted counts of AOS only reported. [E] Some colonies on Big Solander were missed in 1996 (Sagar & Stahl 2005). [F] Raw counts of BOGs only reported for AC^{heli} counts (86%) of the survey, but not for GC or VPC. [G] ψ^{ABA} reported but not incorporated in the reported estimates due to small sample sizes.

Table 11. Tini Heke | Snares counts of Southern Buller's Albatross (ssp. *bulleri*), including updated count methods, metrics, and error specifications, as well as subsequent standardised data for ACAP IG Trends. Superscript numbers refer to numbered equations in main text applied to standardise counts. * Indicates counts absent from the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1969	GC & VPC	AOS & BP	North-east Is., Alert Stack	4,024	11-19 Feb	3-4	-	-	-	0.950[C]	0.929[C]	-	0.931	High	4,448	AOS & BP		Warham & Bennington 1983, Sagar et al. 1999, Sagar & Stahl 2005
1970*	GC	BP	Mbay	90	?[D]	3-14[D]	-	-	-	-	✓; ?	NA	0.014	Unknown (high)	-	-		Sagar & Stahl 2005
1971*	GC	BP	Mbay	83	?[D]	3-14[D]	-	-	-	-	✓; ?	NA	0.014	Unknown (high)	-	-		Sagar & Stahl 2005
1972*	GC	BP	Mbay	77	?[D]	3-14[D]	-	-	-	-	✓; ?	NA	0.014	Unknown (high)	-	-		Sagar & Stahl 2005
1975*	GC	BP	Mbay	84	?[D]	3-14[D]	-	-	-	-	✓; ?	NA	0.014	Unknown (high)	-	-		Sagar & Stahl 2005

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance ($y_{i,t}$)	References
1977*	GC	BP	Mbay, Rback	102	? [D]	3-14 [D]	-	-	-	-	✓; ?	NA	0.019	Unknown (high)	-	-		Sagar & Stahl 2005
1983*	GC	BP	Mbay	85	? [D]	3-14 [D]	-	-	-	-	✓; ?	NA	0.014	Unknown (high)	-	-		Sagar & Stahl 2005
1992	GC & VPC	AOS & BP	North-east Is., Alert Stack, Broughton I.	7,683	5-17 Mar	6-8	0.758	-	NA	0.953-0.977 [E]	-	-	1.000	High	-	-		Sagar et al. 1994, Sagar & Stahl 2005
1993	GC & VPC	AOS & BP	N Prom, Mbay	1,326	2-9 Mar	6-7	0.756	-	-	0.976-0.988 [E]	✓; ?	-	0.180	High	-	-		Sagar & Stahl 2005
1994	GC & VPC	AOS & BP	N Prom, Mbay	1,350	2-9 Mar	6-7	0.674	-	-	0.976-0.988 [E]	✓; ?	-	0.180	High	-	-		Sagar & Stahl 2005
1995	GC & VPC	AOS & BP	N Prom, Mbay	1,390	2-9 Mar	6-7	0.682	-	-	0.976-0.988 [E]	✓; ?	-	0.180	High	-	-		Sagar & Stahl 2005
1996	GC & VPC	AOS & BP	N Prom, Mbay	1,258	2-9 Mar	6-7	0.686	-	-	0.976-0.988 [E]	✓; ?	-	0.180	High	-	-		Sagar & Stahl 2005
1997	GC & VPC	AOS & BP	North-east Is., Alert Stack, Broughton I.	8,242	2-14 Mar	5-7	0.761	-	NA	0.956-0978 [E]	-	-	1.000	High	-	-		Sagar & Stahl 2005
1998	GC & VPC	AOS & BP	N Prom, Mbay, Rback	1,460	2-9 Mar	6-7	0.671	-	-	0.976-0.988 [E]	✓; ?	-	0.185	High	-	-		Sagar et al. 1999, Sagar & Stahl 2005
1999	GC & VPC	AOS & BP	N Prom, Mbay, Rback	1,539	2-9 Mar	6-7	0.644	-	-	0.976-0.988 [E]	✓; ?	-	0.185	High	-	-		Sagar & Stahl 2005
2000	GC & VPC	AOS & BP	N Prom, Mbay, Rback	1,504	2-9 Mar	6-7	0.712	-	-	0.976-0.988 [E]	✓; ?	-	0.185	High	-	-		Sagar & Stahl 2005
2001	GC & VPC	AOS & BP	N Prom, Mbay, Rback	1,666	2-9 Mar	6-7	0.735	-	-	0.976-0.988 [E]	✓; ?	-	0.185	High	-	-		Sagar & Stahl 2005
2002	GC & VPC	AOS & BP	North-east Is. Alert Stack	8,165	25 Feb-2 Mar	5-6	0.772	-	-	0.959-0980 [E]	✓; ?	-	0.931	High	8,713	AOS & BP		Sagar & Stahl 2005
2003	GC & VPC	AOS & BP	N Prom, Mbay, Rback	1,724	2-9 Mar	6-7	0.863	-	-	0.976-0.988 [E]	✓; ?	-	0.185	High	-	-		Sagar & Stahl 2005
2004	GC & VPC	AOS & BP	N Prom, Mbay, Rback	1,903	2-9 Mar	6-7	0.727	-	-	0.976-0.988 [E]	✓; ?	-	0.185	High	-	-		Sagar & Stahl 2005
2005	GC & VPC	AOS & BP	N Prom, Mbay, Rback	2,001	Feb-Mar	2-10	-	-	-	0.976-0.988 [E]	✓; ?	-	0.180	High	-	-		Sagar unpub.
2006	GC	BP	Mbay, L & U Punui	289	6-9 Apr	11	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar & Thompson 2007
2007	GC	BP	Mbay, L & U Punui	269	4-8 Apr	10-11	-	0.580-0.840 [F]	-	-	✓; ?	NA	0.027	High	-	-		Sagar & Thompson 2007, Francis & Sagar 2012
2008	GC	BP	Mbay, L & U Punui	271	Late Mar-late Apr	8-14	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar et al. 2009
2009	GC	BP	Mbay, L & U Punui, Rback	249	6-12 Apr	11-12	-	-	-	-	✓; ?	NA	0.032	High	-	-		Sagar et al. 2009

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	*f*	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π^B	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
2010	GC	BP	Mbay, L & U Punui, Rback	227	8-15 Apr	11-12	-	-	-	-	✓; ?	NA	0.032	High	-	-		Sagar et al. 2010
2011	GC	BP	Mbay, L & U Punui, Rback	255	Late Mar-late Apr	8-14	-	-	-	-	✓; ?	NA	0.032	High	-	-		Sagar et al. 2012
2012	GC	BP	Mbay, L & U Punui	285	21-27 Apr	13-14	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar et al. 2012
2013	GC	BP	Mbay, L & U Punui	241	11-29 Apr	11-14	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar 2014
2014	GC & VPC	AOS & BP	North-east Is., Alert Stack, Broughton Is.	8,704	22 Feb – 4 Mar	5-6	-	0.566-0.833 [F]	NA	0.962-0.981 [E]	-	-	1.000	High	-	-		Sagar 2014, Fu & Sagar 2016
2015	GC	BP	Mbay, L & U Punui	253	23-29 Mar	9-10	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar 2015
2016*	GC	BP	Mbay, L & U Punui	241	Late Mar-late Apr	8-14	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar et al. 2017
2017	GC	BP	Mbay, L & U Punui	281	3-6 Apr	10-11	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar et al. 2017
2019	GC & VPC	AOS & BP	N Prom, W Coast, Rback, E Coast, Broughton I.	3,668	2-10 Mar	6-7	-	-	-	0.976-0.988 [E]	-	-	0.469	High	-	-		Thompson & Sagar 2020, Sagar unpub.
2020	GC & VPC	AOS[X] & BP	S Coast[G], Mbay, L & U Punui,	2,668	11-17 Mar	7-8	-	-	-	0.973-0.986 [E]	-	-	0.228	High	-	-		Thompson & Sagar 2020, Sagar unpub.
2022	GC	BP	Mbay, L & U Punui	303	27 Mar-10 Apr	9-11	-	-	-	-	✓; ?	NA	0.027	High	-	-		Thompson & Sagar 2022
2023	GC	BP	Mbay, L & U Punui, Rback	269	25 Mar-7 Apr	9-11	-	-	-	-	✓; ?	NA	0.032	High	-	-		Thompson & Sagar 2023
2024	GC	BP	Mbay, L & U Punui	185	7-16 Apr	11-12	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar et al. 2024
2025*	GC	BP	Mbay, L & U Punui	227	10-15 Apr	11-12	-	-	-	-	✓; ?	NA	0.027	High	-	-		Sagar et al. 2025

[A] Mean lay date used 21 Jan based on "late January in Sagar & Warham (1998). [B] Proportions based on means calculated using the 1992, 1997, and 2014 full islands counts (Sagar & Stahl 2005, Sagar 2014). [C] Ψ^d and Ψ^p reported but not used to correct counts (Warham & Barrington 1983). [D] Dates unknown so full range during which counts have been conducted used. [E] Ψ^d calculated using the estimate Ψ^d_{VPC} = 0.900-0.950, the assumption that Ψ^d_{GC} = 1.000, and the proportions of each method per count as reported in Thompson & Sagar (2020) with additional insights (P Sagar unpub). [F] Multi-year estimates only reported, and ranges are based on separate estimates of p^b for breeders and non-breeders (Francis & Sagar 2012, Fu & Sagar 2016). [G] Only 72% of the South Coast section was counted (Thompson & Sagar 2020; Sagar unpub.).

Table 12. Motuhara | Fortyfours counts of Northern Buller's Albatross (ssp. *platei*), including updated count methods, metrics, and error specifications, as well as subsequent standardised data for ACAP IG Trends. Superscript numbers refer to numbered equations in main text applied to standardise counts. * Indicates counts absent from the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	*f*	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1972*	GC	BP?	Motuhara	16,000	January	9-13	-	-	-	-	✓; ?	✓; ?	0.985	Unknown (Low)	-	-		Robertson 1991
2008	GC	NS	Motuhara	15,238	13-19 Nov	2-3	-	-	-	-	-	$\psi^{occ} = 0.886$ $\psi^{ABA} = 0.714$	0.985	High[C]	-	-		Scofield et al. 2008, Thompson 2019
2009	GC	NS	Motuhara	14,674	9-11 Nov	1-2	-	-	-	-	$\psi^{fail} = 0.915$	$\psi^{occ} = 0.942$ $\psi^{ABA} = 0.691$	0.985	High[C]	-	-		Fraser et al. 2009A, Thompson 2019
2010	GC	NS	Motuhara	14,185	1-8 Dec	4-5	-	-	-	-	$\psi^{fail} = 0.957$	$\psi^{occ} = 0.917$ $\psi^{ABA} = 0.857$	0.985	High[C]	-	-		Fraser et al. 2010A, Thompson 2019
2017	**AC^{fw}**	BOG	Motuhara + stack 1-4	15,702	23 Nov	3	-	-	NA	-	-	$\psi^{BOG}{}_{AC} = 0.983$, $\psi^{BOG}{}_{GC} = 0.895$, $\psi^{ABA}{}_{GC} = 0.983$	1.000	Medium	13,771 (13,537-14,006)	BP		Baker et al. 2017
	GC	NS	Motuhara + stack 1-4	17,682	5-7 Dec	5	-	-	NA	-	$\psi^{fail} = 0.791$	-[E]	1.000	Medium	13,047	BP		Bell et al. 2017, Baker et al. 2017
2018*	AC^{fw}	BOG	Quadrats 1-5	304	4 Dec	5	-	-	-	-	-[E]	$\psi^{BOG} = 0.974$	0.019	Unknown (High)	296	AOS		Frost 2019
2021*	GC	NS	Quadrats 1-10	647	19-31 Jan	11-13	0.510[F]	-	-	-	-[E]	($\psi^{ABA} = 0.674$[G])	0.038	Unknown (High)	489	BP		Bell 2021, Frost et al. 2021C
2022[H]	GC	NS	Quadrats 1-10	709	29 Jan	13	-	-	-	-	-[E]	-[E]	0.038	High	510	BP		Bell 2022, Frost 2022
2023	GC	NS	Motuhara	15,809	10-12 Dec	6	-	-	-	-	$\psi^{fail} = 0.721$	-[E]	0.985	High	-	-		Bell 2023
2024	GC	NS	Quadrats 1-10	611	18 Feb	16	0.573[I]	-	-	-	$\psi^{fail} = 0.573$	-[E]	0.038	High	322	BP		Bell 2024
2026*	GC & VPC[J]	NS	Motuhara	14,224	28 Nov-1 Dec	4	-	-	-	-	$\psi^{fail} = 0.673$	$\psi^{occ} = 1.000$[K] $\psi^{ABA} = 1.000$[K]	0.985	High	-	-		Bell 2026

[A] Mean lay date used 31 Oct (ACAP 2012). [B] Proportions of quadrats based on means calculated using the 2008, 2009, 2017, 2023, and 2026 full island counts, with additional insights on the offshore stacks from the 2017 count (Fraser et al, 2009A, 2010A, Bell et al. 2017, Bell 2023, 2026). [C] Recorded as *High* accuracy in the ACAP database but here adjusted to Medium due to the lack of *in situ* marking during counts (M Bell pers. comm.) and the fact that the quadrat counts for the same years have also been recorded as *Medium*. [E] Without clear separation between empty and failed nests and explicit statement if statement whether empty nests were occupied or not, assessing Ψ^o is not possible. [F] *f* estimated using a composite of field-based (Bell 2021) and camera-based (Frost et al. 2021) estimates. [G] Camera-based assessment of ψ^{ABA} but not used in the estimates of BP (Frost et al. 2021). [H] AC^{fw} and AC^{UAV} of quadrats completed as well on 1 Feb but not included here as these counts were completed for comparative purposes, rather than count purposes and thus GC are considered here. [I] *f* estimated at hatching (Bell 2024). [J] 1.6% of the overall count conducted using VPCs so 1.6% of the reported BP are AOS, but given this small proportion, the overall count is treated as BP (Bell 2026). [K] estimated in the index plots and multiple instances recorded of two eggs being present in nests, indicating that more breeding pairs than nest sites may be present, but due to the nature of probabilities, here treated as 1.000 (Bell 2026).

Table 13. Rangitutahi | Sisters counts of Northern Buller's Albatross (ssp. *platei*), including updated count methods, metrics, and error specifications, as well as subsequent standardised data for ACAP IG Trends. Superscript numbers refer to numbered equations in main text applied to standardise counts. * Indicates counts absent from the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^{A}	f	p^{b}	Ψ^{s}	Ψ^{d}	Ψ^{p}	Ψ^{o}	π^{B}	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance $(y_{i,t})$	References
1972	GC	BP?	Little & Big Sister	2,000	January	9-13	-	-	NA	-	✓; ?	✓; ?	1.000	Low	-	-		Robertson 1991
1996*	GC	BP	Little Sister	565	Late Nov	3-4		-	-	-	-	NA	0.219	Unknown (Low)	-	-		Bell et al. 2018
1997*	GC	BP	Little Sister	636	Late Nov	3-4		-	-	-	-	NA	0.219	Unknown (Low)	-	-		Bell et al. 2018
2005	GC	NS?	Little Sister	200	27 Jan	13		-	-	-	-	-	0.219	Unknown (Low)	-	-		Scofield unpub.
2017	AC^{fw}	BOG	Little & Big Sister	2,692	4 Dec	5	-	-	NA	-	-	$\psi^{BOG} = 0.978$	1.000	Medium	2,366 (2,269-2,464)	BP^{C}		Baker et al. 2017
2018	GC	NS	Little & Big Sister	3,158	30 Nov-4 Dec	4-5	-	-	NA	0.999-0.996	$\psi^{fail} = 0.637$-0.752	$\psi^{occ} \cdot \psi^{ABA} = 0.965$	1.000	Medium	3,158	BP		Bell et al. 2018

^{A} Mean lay date used 31 Oct (ACAP 2012).B Proportions calculated using all counts during which both islands were counted (1972, 2017, and 2018). ^{C} ψ^{ABA} estimated on Motuhara during GCs (Baker et al.2017).

White-capped Albatross

Supplementary Table 14. Disappointment Island counts of White-capped Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^{A}	f	p^{b}	Ψ^{s}	Ψ^{d}	Ψ^{p}	Ψ^{o}	π^{B}	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1973*	GC, VPC, VBC	BOG	Disappointment I.	60,000	15 Feb	13	-	-	NA	0.771 (0.504-0.952)^{C}	-	-	1.000	Unknown (Low)	-	-		Robertson 1975, Walker et al. 2020
1985	AC^{heli}	BOG	Disappointment I.	71,799	18 Feb	14	-	-	NA	-	-	$\psi^{BOG} \cdot \psi^{ABA}$ = 0.670-720^{D}	1.000	High	50,259	BP		Walker et al. 2021
1993	AC^{heli}	BOG	Castaway Bay	3,740	22 Jan	10	-	-	-	-		$\psi^{BOG} \cdot \psi^{ABA}$ = 0.670-720^{D}	0.047^{B}	High	-	BP		Walker et al. 2021
2002	AC^{heli}	BOG	Disappointment I.	125,476	21 Feb	14	-	-	NA	-		$\psi^{BOG} \cdot \psi^{ABA}$ = 0.670-720^{D}	1.000	Medium	87,833	BP		Walker et al. 2021
2007	AC^{heli}	BOG	Disappointment I.	110,649	16 Dec	4	0.622^{E}	-	NA	-	-	ψ^{BOG} = 0.990^{F}	1.000	High	109,543^{G}	AOS		Thompson & Sagar 2008a, Francis 2012, Walker et al. 2021, Baker et al. 2023
2008	AC^{heli}	BOG	Disappointment I.	86,080	13 Dec	4	-	-	NA	-	-	ψ^{BOG} = 0.995 (ψ^{ABA} = 0.810)^{H}	1.000	High	85,650^{G}	AOS		Walker et al 2020, Baker et al. 2023
2009	AC^{heli}	BOG	Disappointment I.	91,694	14 Dec	4	-	-	NA	-	-	ψ^{BOG} = 0.982 (ψ^{ABA} = 0.935)^{H}	1.000	High	90,044^{G}	AOS		Walker et al 2021, Baker et al. 2023
2010	AC^{heli}	BOG	Disappointment I.	72,275	3 Dec	3	-	-	NA	-	-	ψ^{BOG} = 0.986	1.000	High	71,263^{G}	AOS		Walker et al 2021, Baker et al. 2023
2011	AC^{heli}	BOG	Disappointment I.	72,635	15 Dec	4	-	-	NA	-	-	ψ^{BOG} = 0.987	1.000	High	71,691^{G}	AOS		Walker et al 2021, Baker et al. 2023
2012	AC^{heli}	BOG	Disappointment I.	93,752	11 Jan	8	-	-	NA	-	-	ψ^{BOG} = 0.929	1.000	High	87,096^{G}	AOS		Walker et al 2021, Baker et al. 2023
2013	AC^{heli}	BOG	Disappointment I.	111,312	14 Jan	9	-	-	NA	-	-	ψ^{BOG} = 0.866	1.000	High	96,396^{G}	AOS		Walker et al 2021, Baker et al. 2023
2014	AC^{heli}	BOG	Disappointment I.	89,552	20 Jan	9	-	-	NA	-	-	ψ^{BOG} = 0.777	1.000	High	69,582^{G}	AOS		Walker et al 2021, Baker et al. 2023
2015	AC^{heli}	BOG	Disappointment I.	96,864	14 Jan	9	-	-	NA	-	-	ψ^{BOG} = 0.942^{I} (ψ^{ABA} = 0.849-0.851)^{H}	1.000	High	91,246^{G}	AOS		Walker et al 2021, Baker et al. 2023
2016	AC^{heli}	BOG	Disappointment I.	91,554	13 Jan	8	-	-	NA	-	-	ψ^{BOG} = 0.901^{I} (ψ^{ABA} = 0.775)^{H}	1.000	High	82,490^{G}	AOS		Walker et al 2021, Baker et al. 2023
2017	AC^{heli}	BOG	Disappointment I.	85,510	19 Jan	9	-	-	NA	-	-	ψ^{BOG} = 0.900	1.000	High	76,959^{G}	AOS		Walker et al 2021, Baker et al. 2023
2018*	AC^{heli}	BOG	Disappointment I.	95,688	5 Jan	7	0.263^{E}	-	NA	-	-	ψ^{BOG} = 0.892^{J}	1.000	High	85,354^{G}	AOS		Rexer-Huber et al. 2019, Walker et al. 2021, Baker et al. 2023, Frost 2026

[A] Mean lay is ~15 Nov. [B] Based on multi-year average proportions calculated from Walker et al. 2021. [C] Estimated through expert elicitation following modified Delphi protocol. [D] Ψ^o based on 1993 ground counts (Walker et al. 2021). [E] Additional *f* estimates exist for 2006 (0.526; Thompson & Sagar 2008a), 2022 (0.467; Elliott et al. 2023, Frost 2026), and 2023 (0.727, based on chicks only; Frost 2026). [F] ψ^{BOG} based on multi-year average (Baker et al. 2023). [G] Adjusted counts from Baker et al. 2023, additional adjusted counts available in Walker et al. 2021. [H] ψ^{ABA} estimates available in Francis 2012, Parker et al. 2017, and Walker et al. 2021, but not included in Baker et al. 2023, of which the 2008 estimate originates from SW Cape. [I] Additional ψ^{BOG} estimates available in Parker et al. 2017 and Walker et al. 2021 but not included in Baker et al. 2023. [J] Further Ψ^o estimates available in Parker et al. 2022 and Elliott et al. 2023.

Supplementary Table 15. South-west Cape (Maukahuka | Auckland Island) counts of White-capped Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π^B	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1973*	GC, VPC. VBC	BOG	SW Cape	4,000	Early Feb	12-13	-	-	NA	0.712 (0.397-0.939[B]	-	-	1.000	Unknown (Low)	-	-		Robertson 1975, Walker et al. 2020
1993	AC^{heli}	BOG	SW Cape	3,000	Late Jan – Feb	10-14	-	-	NA	-			1.000	Low	-	-		Robertson & Gales 1998
2007	AC^{heli}	BOG	SW Cape	6,548	16 Dec	4	-	0.680 (0.580; 0.810)	NA	-	-	(ψ^{BOG} = 0.990) [D]	1.000	High	6,483	AOS		Francis 2012, Walker et al. 2021, Baker et al. 2023
2008	AC^{heli}	BOG	SW Cape	4,786	13 Dec	4	0.627	0.680 (0.580; 0.810)	NA	-	-	E_{aba} = 0.810 (ψ^{BOG} = 0.995) [D]	1.000	High	4,762	AOS		Francis 2012, Thompson & Sagar 2008b, Baker et al. 2023
2009	AC^{heli}	BOG	SW Cape	5,264	14 Dec	4	-	0.680 (0.580; 0.810)	NA	-	-	(ψ^{BOG} = 0.982) [D]	1.000	High	5,169	AOS		Francis 2012, Baker et al. 2023
2010	AC^{heli}	BOG	SW Cape	4,161	3 Dec	3	-	0.680 (0.580; 0.810)	NA	-	-	(ψ^{BOG} = 0.986) [D]	1.000	High	4,103	AOS		Francis 2012, Baker et al. 2023
2011	AC^{heli}	BOG	SW Cape	4,370	15 Dec	4	-	-	NA	-	-	(ψ^{BOG} = 0.987) [D]	1.000	High	4,313	AOS		Baker et al. 2023
2012	AC^{heli}	BOG	SW Cape	5,846	11 Jan	8	-	-	NA	-	-	(ψ^{BOG} = 0.929) [D]	1.000	High	5,431	AOS		Baker et al. 2023
2013	AC^{heli}	BOG	SW Cape	6,571	14 Jan	9	-	-	NA	-	-	(ψ^{BOG} = 0.866) [D]	1.000	High	5,690	AOS		Baker et al. 2023
2014	AC^{heli}	BOG	SW Cape	5,542	20 Jan	9	-	-	NA	-	-	(ψ^{BOG} = 0.777) [D]	1.000	High	4,306	AOS		Baker et al. 2023
2015	AC^{heli}	BOG	SW Cape	4,741	14 Jan	9	-	-	NA	-	-	(ψ^{BOG} = 0.942) [D]	1.000	High	4,466	AOS		Baker et al. 2023
2016	AC^{heli}	BOG	SW Cape	4,893	13 Jan	8	-	-	NA	-	-	(ψ^{BOG} = 0.901) [D]	1.000	High	4,409	AOS		Baker et al. 2023
2017	AC^{heli}	BOG	SW Cape	5,436	19 Jan	9	-	-	NA	-	-	(ψ^{BOG} = 0.900) [D]	1.000	High	4,892	AOS		Baker et al. 2023
2018*	AC^{heli}	BOG	SW Cape	5,834	5 Jan	7	-	-	NA	-	-	(ψ^{BOG} = 0.892) [D]	1.000	High	5,204	AOS		Baker et al. 2023
2021*	AC^{UAV}	BOG	East gulch	764	22 Mar	18	-	-	NA	-	-	ψ^{BOG} = 0.955	0.220[B]	High	730	AOS		R. Sagar unpub.

[A] Mean lay is ~15 Nov. [B] Proportion of the east gulch estimated using 2016-2018 AC^{heli} (Baker unpub.). [C] Estimated through expert elicitation following modified Delphi protocol. [D] ψ^{BOG} based on estimates from Disappointment Island.

Supplementary Table 16. Adams Island counts of White-capped Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1973*	? [B]	BOG	Logan Point	200	Early Feb	12-13	-	-	NA	?	-	-	1.000	Unknown (Low)	-	-		Robertson 1975, Walker et al. 2020
1993	AC^{heli}	BOG	Logan Point	100	Late Jan – Feb	10-14	-	-	NA	-	-	-	1.000	Medium	-	-		Robertson & Gales 1998
2008	AC^{heli}	BOG	Logan Point	79	13 Dec	4	-	-	NA	-	-	(E_{loaf} = 0.995) [C]	1.000	High	79	AOS		Baker et al. 2023
2009	AC^{heli}	BOG	Logan Point	131	14 Dec	4	-	-	NA	-	-	(E_{loaf} = 0.982) [C]	1.000	High	129	AOS		Baker et al. 2023
2010	AC^{heli}	BOG	Logan Point	132	3 Dec	3	-	-	NA	-	-	(E_{loaf} = 0.986) [C]	1.000	High	130	AOS		Baker et al. 2023
2011	AC^{heli}	BOG	Logan Point	117	15 Dec	4	-	-	NA	-	-	(E_{loaf} = 0.987) [C]	1.000	High	115	AOS		Baker et al. 2023
2012	AC^{heli}	BOG	Logan Point	178	11 Jan	8	-	-	NA	-	-	(E_{loaf} = 0.929) [C]	1.000	High	165	AOS		Baker et al. 2023
2013	AC^{heli}	BOG	Logan Point	215	14 Jan	9	-	-	NA	-	-	(E_{loaf} = 0.866) [C]	1.000	High	186	AOS		Baker et al. 2023
2014	AC^{heli}	BOG	Logan Point	184	20 Jan	9	-	-	NA	-	-	(E_{loaf} = 0.777) [C]	1.000	High	143	AOS		Baker et al. 2023
2015	AC^{heli}	BOG	Logan Point	193	14 Jan	9	-	-	NA	-	-	(E_{loaf} = 0.942) [C]	1.000	High	182	AOS		Baker et al. 2023
2016	AC^{heli}	BOG	Logan Point	176	13 Jan	8	-	-	NA	-	-	(E_{loaf} = 0.901) [C]	1.000	High	159	AOS		Baker et al. 2023
2017	AC^{heli}	BOG	Logan Point	171	19 Jan	9	-	-	NA	-	-	(E_{loaf} = 0.900) [C]	1.000	High	154	AOS		Baker et al. 2023
2018*	AC^{heli}	BOG	Logan Point	194	5 Jan	7	-	-	NA	-	-	(E_{loaf} = 0.892) [C]	1.000	High	173	AOS		Baker et al. 2023

[A] Mean lay is ~15 Nov. [B] Reported as a potential VBC in Roberston (1975) but VPCs may also have been conducted. Unclear if Logan Point can be seen during a VBC. [C] ψ^{BOG} based on estimates from Disappointment Island.

Chatham Albatross

Supplementary Table 17. Te Tara Koi Koia | Pyramid counts of Chatham Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1965*	VBC	AOS	Te Tara Koi Koia | Pyramid	2,000	17 Oct	6	-	-	NA	0.636 (0.349-0.878)[B]	-	-	1.000	Unknown (low)	2,500	BP		Dawson 1973
1973*	AC^{fw}	AOS	Te Tara Koi Koia | Pyramid	3,965	28 Nov	11	-	-	NA	0.864 (0.693-0.971)[B]	-	-	1.000	Unknown (low)	4,200	BP		Robertson et al. 1991
2000	GC & VPC[C]	NS	Te Tara Koi Koia | Pyramid	5,333	25 Nov-5 Dec	11-13	-	-	NA	-	-	(ψ^{occ} = 0.590) [D]	1.000	High	-	-		Robertson et al. 2003
2001	GC & VPC[C]	NS	Te Tara Koi Koia | Pyramid	5,326	6-10 Dec	13-14	-	-	NA	-	-	(ψ^{occ} = 0.650) [D]	1.000	High	-	-		Robertson et al. 2003
2002	GC & VPC[C]	NS	Te Tara Koi Koia | Pyramid	5,304	30 Nov-9 Dec	12-13	-	-	NA	-	-	(ψ^{occ} = 0.710) [D]	1.000	High	-	-		Robertson et al. 2003
2004*	GC & VPC[C]	NS	Te Tara Koi Koia | Pyramid	5,341	3-12 Dec	12-14	-	-	NA	-	-	(ψ^{occ} = 0.610) [D]	1.000	Unknown (High)	-	-		Robertson et al. 2004
2006	GC & VPC[E]	NS	Te Tara Koi Koia | Pyramid	5,206	28 Jan-13 Feb	20-23	-	-	NA	-	(Ψ^{fail} = 0.931)	(ψ^{occ} = 0.396)	1.000	High	-	-		Fraser et al. unpub.
2007	GC & VPC[E]	NS	Te Tara Koi Koia | Pyramid	5,412	22-28 Jan	19-20	-	-	NA	-	(Ψ^{fail} = 0.987)	(ψ^{occ} = 0.448)	1.000	High	-	-		Fraser et al. unpub.
2008	GC & VPC[E]	NS	Te Tara Koi Koia | Pyramid	5,247	19-29 Nov	10-12	-	-	NA	-	(Ψ^{fail} = 0.960)	(ψ^{occ} = 0.753)	1.000	High	-	-		Scofield et al. 2008
2009	GC & VPC[E]	NS	Te Tara Koi Koia | Pyramid	5,407	22 Nov-7 Dec	11-13	-	-	NA	-	(Ψ^{fail} = 0.967)	(ψ^{occ} = 0.702)	1.000	High	-	-		Fraser et al. 2009B
2010	GC & VPC[E]	NS	Te Tara Koi Koia | Pyramid	5,194	9-12 Dec	13-14	-	-	NA	-	(Ψ^{fail} = 0.970)	(ψ^{occ} = 0.505)	1.000	High	-	-		Fraser et al. 2010B
2011	GC & VPC[E]	NS	Te Tara Koi Koia | Pyramid	5,245	24-30 Nov	11-12	-	-	NA	-	(Ψ^{fail} = 0.973)	(ψ^{occ} = 0.702)	1.000	High	-	-		Fraser et al. 2011
2017	GC & VPC[F]	NS	Te Tara Koi Koia | Pyramid	5,296	9-13 Nov	9-10	-	-	NA	-	(Ψ^{fail} = 0.744)	(ψ^{occ} = 0.966)	1.000	High	-	-		M Bell et al. 2017A
2026*	GC & VPC[G]	NS	Te Tara Koi Koia | Pyramid	5,227	8-13 Nov	9-10	-	-	NA	-	(Ψ^{fail} = 0.666)	-	1.000	High	-	-		Bell 2026

[A] Mean lay is ~7 Sep based on M Bell et al. 2017A. [B] Estimated through a modified Delphi expert elicitation, in which the original value was included as one of the estimated values. [C] 0.04% of the population was counted through VPC (Roberson et al. 2003). [D] Empty nests and failed nests not differentiable. [E] 11.0-11.9% of the population was counted through VPC. [F] 14.0% of the population was counted through VPC. [G] 8.9% of the population was counted through VPC.

Salvin's Albatross

Supplementary Table 18. Moutere Hauriri | Bounty Island counts of Salvin's Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	*f*	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1979	GC[C]	BP	Moutere Hauriri	76,352	7-20 Nov	9-11		-	NA	-	-	NA	1.000	Low	49,250[C]	BP		Robertson & van Tets 1982, Amey & Sagar 2013, Thompson 2025
1998	GC	BP	Proclamation I.	3,065	12-16 Nov	10		-	-	>0.990	-	($\psi^{BOG} \cdot \psi^{ABA}$ = 0.668$^{\underline{D}}$; ψ^{ABA} = 0.627$^{\underline{D}}$)	0.103	High	-	-		Clark 1998, Amey & Sagar 2013, Thompson 2025
2005	GC	BP	Proclamation I.	2,634	16-23 Nov	10-11		-	-	>0.990	-	($\psi^{BOG} \cdot \psi^{ABA}$ = 0.579$^{\underline{D}}$)	0.103	High	-	-		Amey & Sagar 2013, Thompson 2025
2011	AC^{fw}	BOG	Moutere Hauriri	42,832	12 Oct	5		-	NA	-	-	-[E]	1.000	High	31,786[G]	BP		Baker et al. 2014B, Thompson 2025
2012	GC	BP	Proclamation I.	2,211	14 Nov	10		-	-	>0.990	-	($\psi^{BOG} \cdot \psi^{ABA}$ = 0.622$^{\underline{D}}$; ψ^{ABA} = 0.790$^{\underline{D}}$)	0.103	Low	-	-		Amey & Sagar 2013, Thompson 2025
2014	AC^{fw}	BOG	Moutere Hauriri	53,893	23 Oct	7		-	NA	-	-	ψ^{BOG} = 0.826$^{\underline{D}}$ ψ^{ABA} =0.899$^{\underline{D}}$	1.000	High	39,995	BP		Baker et al. 2014B, Thompson 2025
2019	AC^{fw}	BOG	Moutere Hauriri	60,419	25 Oct	7	0.350 (0.230-0.480)[F]	-	NA	-	-	ψ^{BOG} = 0.716[G] ψ^{BOG} = 0.648$^{\underline{D}}$ ψ^{ABA} =0.724$^{\underline{D}}$	1.000	High	26,955 (26,626-27,283)	BP		Sagar et al. 2018, Baker & Jensz 2019, Thompson 2025
2020	AC^{UAV}	BOG	Proclamation, Spider, Tunnel Is.	12,684	28 Oct	8	0.350 (0.230-0.480)[F]	-		-	-	-[H]	0.272	High	5,962 (5,200-6,595)	BP		Parker & Rexer-Huber 2020, Thompson 2025
2023*	AC^{UAV}	BOG	Moutere Hauriri	69,709	10-15 Nov	5-6	0.350 (0.230-0.480)[F]	-	NA	-	-	-	1.000	High	-	-		Mattern 2022, Thompson 2025, Mattern
2024*	AC^{UAV}	C	Depot, Tunnel Ruatara, Proclamation, Spider Is.	11,546	31 Jan	21	0.350 (0.230-0.480)[F]	-	-	-	-	NA	0.777	Medium	-	-		Mattern 2024a, Thompson 2025
2025*	AC^{UAV}	BOG	Moutere Hauriri	50,789	13 Oct	6	0.350 (0.230-0.480)[F]	-	NA	-	-	ψ^{BOG} = 0.746$^{\underline{D}}$ ψ^{ABA} =0.903$^{\underline{D}}$	1.000	High	-	-		Mattern 2024b, Thompson 2025

[A] Mean lay date used 4 Sep (Thompson 2025). [B] Proportions based on averages across all full breeding site counts (Thompson 2025). [C] Robertson & van Tets 1979 extrapolation proved challenging and required additional spatial re-analyses (see main text and Thompson 2025). [D] Estimated during ground transects. [E] Ψ^o estimate from 2014 used rather than concurrent data. [F] Only Multi-year mean estimates could be derived through daily-survival modelling of remote camera data (Thompson 2025). [G] Estimated from close-up photographs. [H] Ψ^o estimate from Sager et al. (2018) used rather than concurrent data.

Supplementary Table 19. Western Chain islets counts of Salvin's Albatross, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database. **Bold** indicates count selected for inclusion in the data processing framework in scenarios where >1 counts were completed in a year.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^A	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π^B	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1977	GC	BP	Rima	122	21 Nov	11	-	-	-	-	-	NA	0.333	High	-	-		Sagar 1977, Thompson 2025
1984	GC	C	Toru, Rima	585	Feb	21-25	-	-	NA	-	-	NA	1.000	High	-	-		Miskelly 1984, Thompson 2025
1996	**GC**	BP	Toru	1,021	5 Oct	4	-	-	NA	-	-	NA	0.667	High	-	-		Clark 1996, Thompson 2025
	GC	BP	Rima	189	8 Nov	9	-	-	NA	-	-	NA	0.333	High				Clark 1996, Thompson 2025
2009	GC	BP	Toru, Rima	1,195	6-7, 16 Oct	5-6	-	0.927 (0.662-0.997)^C	NA	-	-	NA	1.000	High	-	-		Sagar et al. 2011, Thompson 2025
2010	GC	BP	Toru, Rima	1,116	30 Sep, 2 Oct	4	-	0.927 (0.662-0.997)^C	NA	-	-	NA	1.000	High	-	-		Sagar et al. 2011, Thompson 2025
2011	GC	BP	Toru	780	28-29 Sep	3-4	-	0.927 (0.662-0.997)^C	-	-	-	NA	0.667	High	-	-		Sagar et al. 2011, Thompson 2025
2015	AC^{heli}	BOG	Toru, Rima	2,307			-	-	NA	-	-	-	1.000	High	1,486	AOS		Baker et al. 2015
	GC	BP	Toru, Rima	1,125	17 Oct	6	-	-	NA	-	-	$\psi^{BOG} = 0.667^D$ $\psi^{ABA} = 0.877^D$	1.000	High	-	-		Sagar et al. 2014, Thompson 2025
2026*	GC	BP	Toru	415	1 Nov	8	-	-	-	-	-	NA	0.667	High	-	-		Rexer-Huber et al. in prep.

^{A} Mean lay date used 4 Sep (Thompson 2025). ^{B} Proportion based on average proportion across the full site counts. ^{C} p^b re-estimated by Thompson (2025). ^{D} Estimated during ground transects.

Westland Petrel

Supplementary Table 20. Punakaiki counts of Westland Petrel, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post µ lay[A]	f	p^{b}	Ψ^{s}	Ψ^{d}	Ψ^{p}	Ψ^{o}	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1974*	GC	AOS	Punakaiki	818	Apr-Jul	0-10	0.390 (0.200; 0.630)[C]	-	-	-	-	✓; ?	0.810	Unknown (Low)	818	AOS		Best & Owen 1976, Freeman & Wilson 2002, Wood & Otley 2013
1975*	GC	AOS	Punakaiki	2,000 (1,000; 5,000)	?[D]	?[D]	0.390 (0.200; 0.630)[C]	-	-	-	-	✓; ?	1.000	Unknown (Low)	2,000 (1,000; 5,000)	AOS		Bartle 1985, Marchant & Higgins 1990, Freeman & Wilson 2002, Wood & Otley 2013
2003 (2002-05; 2010)[E]	GC	NS	Punakaiki	12,843	Oct-Dec	19-26	0.620 (0.542; 0.698)[F]	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.303$ (0.264; 0.342)	0.977	Medium[G]	2,954-5,137	BP		Waugh et al. 2006, Wood & Otley 2013, Waugh & Wilson 2017
2007*	GC	AOS	Punakaiki	5,476 (4,511; 6,443)	15 May - 15 Jun	0-3	-	0.460[G]	-	-	-	$\psi^{occ} = 0.435$ (0.405; 0.465)[H] ($\psi^{ABA} = 0.837$)[I]	0.536	Medium[G]	2,385 (2,220; 2,549)	AOS		Waugh & Wilson 2017, Baker et al. 2020
2008*	GC	AOS	Punakaiki	4,718 (3,587; 5847)	17-28 Mar; 12-23 May	0	-	0.460[G]	-	-	-	$\psi^{occ} = 0.299$ (0.257; 0.341)	0.596	Medium[G]	1,403 (1,085; 1,718)	AOS		Waugh & Wilson 2017, Baker et al. 2020
2011	GC	AOS	Punakaiki	4,334 (3,327; 5346)	17 Jan– 3 Feb; 23 May- 2 Jun	0-1	-	0.460[G]	-	-	-	$\psi^{occ} = 0.546$ (0.505; 0.587)	0.693	Medium[G]	2,289 (1,975; 2,628)	AOS		Waugh & Wilson 2017, Baker et al. 2020
2019	GC	BP	Punakaiki	31,306	Jul-Aug	8-14	0.685 (0.469; 0.901)[J]	0.768 (0.670; 0.844)[K]	-	-	-	$\psi^{occ} = 0.385$ (0.340; 0.500), $\psi^{ABA} = 0.480$ (0.337; 0.623)[L]	0.886	High[G]	6,223 (5,478; 6,968)	BP		Waugh et al. 2015, 2020

[A] Mean lay is ~23 May. [B] Proportion calculated based on sub-colonies counted and their population size using information from Wood & Otley (2013) and Waugh et al. (2020). [C] 1976-1996 average. [D] Dates of count unknown, and thus date range of the 1974 count was used instead. [E] Count effort distributed across 2002-2005, with some additional effort as 2010, but information provided in Wood & Otley (2013) is insufficient to disentangle individual survey years and as such 2003 is used here as a midpoint for Wood & Otley (2013). [F] 1995-2003 average. [G] Downscaled one accuracy category in the modelling framework to acknowledge the lack of accounting for Ψ^{d}. [H] 2007-2014 average. [I] 2008-2011 average. [J] ψ^{ABA} considered negligible in Baker et al. (2020). [K] 2010-2019 average. [L] 1995-2012 average. [M] 2015-2019 average.

Black Petrel

Supplementary Table 21. Aotea counts of Black Petrel, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay^{A}	*f*	p^{b}	Ψ^{s}	Ψ^{d}	Ψ^{p}	Ψ^{o}	π^{B}	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1977^{C}	GC	BP	Aotea	500-1,000^{C}	-^{C}	-^{C}	-	-	NA	-	-	-	0.333-0.531^{D}	Unknown (Low)	-	-		Imber 1987
1996	GC	NS	3 census grids	43	?	5-12	0.869	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.535$	0.008^{F}	High	23	BP	NA^{F}	Bell & Sim 2000A, Bell et al. 2025
1997	GC	NS	3 census grids	53	?	5-12	0.686	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.660$	0.008^{F}	High	35	BP	NA^{F}	Bell & Sim 2000A, Bell et al. 2025
1998	GC	NS	3 census grids	52	21 Jan-24 Feb	6-11	0.772	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.673$	0.008^{F}	High	35	BP	NA^{F}	Bell & Sim 2000A
1999	GC	NS	6 census grids	95	25 Jan-28 Feb	7-12	0.708	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.684$	0.015	High	65	BP		Bell & Sim 2000B
2000	GC	NS	9 census grids	118	17 Jan-23 Feb	6-11	0.671	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.695$	0.023	High	82	BP		Bell & Sim 2000C
2001	GC	NS	9 census grids	121	15 Jan-16 Feb	6-10	0.816	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.628$	0.023	High	76	BP		Bell & Sim 2002
2002	GC	NS	9 census grids	124	25 Nov-5 Dec	0	0.763	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.645$	0.023	High	80	BP		Bell & Sim 2003A
2003	GC	NS	9 census grids	130	4-18 Dec	0-2	0.600	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.577$	0.023	High	75	BP		Bell & Sim 2003B
2004	GC	NS	9 census grids	132	1-12 Dec	0-1	0.793	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.583$	0.023	High	77	BP		Bell et al. 2005
2005	GC	NS	Mt Hobson study area (35 ha)	192	?	0-12^{G}	0.750	0.719 (0.690-0.742)^{E}	✓	✓	-	$\psi^{occ} \cdot \psi^{ABA} = 0.575$	0.531	Low^{G}	1,126 (814-1,438)	BP		Bell et al. 2007, 2018
2006	GC	NS	9 census grids	147	1-11 Dec	0-1	0.706	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.626$	0.023	High	92	BP		Bell et al. 2009
2007	GC	NS	9 census grids	148	?	0-2^{G}	0.812	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.649$	0.023	High	96	BP		Bell et al. 2025
2008	GC	NS	9 census grids	149	11-18 Dec	1-2	0.710	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.624$	0.531	High	93	BP		Bell et al. 2011
2009	GC	NS	9 census grids	151	?	0-2^{G}	0.698	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.636$	0.023	High	96	BP		Bell et al. 2025
2010	GC	NS	Mt Hobson study area (35 ha)	298	?	0-12^{G}	0.732	0.719 (0.690-0.742)^{E}	✓	✓	-	$\psi^{occ} \cdot \psi^{ABA} = 0.566$	0.531	Low^{G}	1,010 (698-1,322)	BP		Bell et al. 2010, 2018
2011	GC	NS	9 census grids	153	?	0-2^{G}	0.605	0.719 (0.690-0.742)^{E}	-	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.627$	0.023	High	96	BP		Bell et al. 2025

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π[B]	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
2012	GC	NS	9 census grids	154	?	0-2[G]	0.820	0.719 (0.690-0.742) [E]	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.578	0.023	High	89	BP		Bell et al. 2025
2013	GC	NS	Mt Hobson study area (35 ha)	510	?	0-12[G]	0.809	0.719 (0.690-0.742) [E]	✓	✓	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.567	0.531	Medium [I]	2,587 (1,026-4,148)	BP		Bell et al. 2013, 2018
2014	GC	NS	9 census grids	158	6-13 Dec	0-2	0.685	0.719 (0.690-0.742) [E]	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.601	0.023	High	95	BP		Bell et al. 2014
2015	GC	NS	9 census grids	163	6-15 Dec	0-2	0.683	0.719 (0.690-0.742) [E]	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.620	0.023	High	101	BP		Bell et al. 2015
2016	GC	NS	9 census grids	166	2-14 Dec	0-2	0.626	0.719 (0.690-0.742) [E]	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.645	0.023	High	107	BP		Bell et al. 2016B
2017*	GC	NS	Mt Hobson study area (35 ha)	298	Feb	8-12	0.670	-	✓	✓	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.582	0.531	Medium [I]	2,427 (1,713-3,140)	BP		Bell et al. 2018
2018	GC	NS	9 census grids	177	12 Jan-9 Feb	5-9	0.636	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.542	0.023	High	96	BP		Bell et al. 2020
2019	GC	NS	9 census grids [J]	177	7 Jan-13 Feb	4-10	0.696	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.576	0.023	High	102	BP		Bell et al. 2020
2020	GC	NS	High quality and core medium quality habitat [J]	293	Jan-Feb	4-12	0.715	--	✓	✓	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.452-0.462	0.900-1.000 [K]	Medium	4,336 (3,770-4,993) [L]	BP		Bell et al. 2020, 2021, 2022A
2021	GC	NS	9 census grids [J]	192	8 Jan-1 Feb	5-8	0.724	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.604	0.023	High	116	BP		Bell et al. 2022A
2022	GC	NS	9 census grids	195	19 Dec-8 Feb	2-9	0.774	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.656	0.023	High	128	BP		Bell et al. 2022B
2023	GC	NS	9 census grids	196	28 Nov-5 Dec	0	0.563	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.607	0.023	High	119	BP		Bell et al. 2023
2024*	GC	NS	9 census grids	198	4-14 Dec, 7-28 Feb	0-12	0.703	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.611	0.023	High	121	BP		Bell et al. 2024
2025*	GC	NS	9 census grids	200	13-18 Dec, 22 Jan-4 Feb	1-8	0.650	-	-	-	-	$\psi^{occ} \cdot \psi^{ABA}$ = 0.600	0.023	High	120	BP		Bell et al. 2025

[A] Mean lay ~ 7 Dec (Imber 1987, Bell et al. 2025). [B] Proportions for census grids were calculated using the 2019-2021 population estimate as a baseline, assuming that this estimate itself represented ~0.95 of the total population (Bell et al. 2022A), proportions of the 35 ha Mt Hobson study area was calculated using a retrospective estimate of the overall Aotea population based on the proportion of the census grids and the most accurate estimate of the study area (Bell et al. 2018). [C] Based on fieldwork during 1972-1975, 1977, 1979, and 1982-1983 across breeding seasons, and as such, the median year, 1977, was chosen here as the representative year (Imber 1987). [D] estimated using expert judgement (Bell pers. comm.). [E] Only multi-year (1996-2016) estimates available (Zhang et al. 2020). [F] Due to the large uncertainties associated with counts that represent <1% of the breeding site, these counts are not included in the trend modelling (see main text). [G] Initially recorded *High* in the ACAP database but based on comparison with other counts and re-analyses (e.g., Bell et al. 2018), best re-categorised as *Low*. [H] When count date is not available, we assumed that census block counts would take place during Dec and that transect counts took place during Dec-Feb (Zhang et al. 2020). [I] Initially recorded *High* in the ACAP database but based on comparison with other counts and re-analyses (e.g., Bell et al. 2018), best re-categorised as *Medium*. [J] Across 2019-2021, the most robust population estimate was generated (Bell et al. 2022A), which we here assign to the middle year (2020), while we use the census grids and extrapolation to estimate the overall population in the other two years (2019 and 2021), rather than disaggregating the robust overall estimate. [K] Bell et al. (2022A) could not estimate the population in low quality habitat and suggested that <500 BP would remain in the hinterland and as such, we built additional uncertainty into the estimate through π. [L] The truncated, high and core-medium quality habitat estimate is used here (Bell et al. 2022A).

Supplementary Table 22. Hauturu counts of Black Petrel, including count methods, metrics, and error specifications, as well as subsequent standardised data $y_{i,t}$ expressed in BP at the start of the breeding season (for which superscript numbers refer to equations in main text applied). * Indicates counts not (yet) included in the ACAP database.

Year	Method	Counted metric	Part site	Reported raw count	Count date	Weeks post μ lay[A]	f	p^b	Ψ^s	Ψ^d	Ψ^p	Ψ^o	π	Acc. category	Reported adjusted count	Reported adjusted metric	Standardised abundance data ($y_{i,t}$)	References
1977[B] *	GC	BP	Hauturu	50-100	-[B]	-[B]	-	-	NA	-	?	-	0.250-0.500[C]	Unknown (Low)	-	-		Imber 1987
2016	GC	BP	Hauturu	170	28 Apr-19 May	20-23	0.852	-	NA	-	-	$\psi^{occ} \cdot \psi^{ABA} = 0.559$	1.000	Medium	620	BP		Bell et al. 2016A

[A] Mean lay ~ 7 Dec (Imber 1987, Bell et al. 2025). [B] This estimate is based on fieldwork during 1972-1975, 1977, 1979, and 1982-1983 across breeding seasons, and as such, the median year, 1977, was chosen here as the representative year (Imber 1987). [C] estimated using expert judgement (Bell pers. comm.).

References

ACAP. 2009. ACAP Species assessment: Southern Royal Albatross *Diomedea epomophora*. Agreement of Albatrosses and Petrels http://www.acap.aq.

Amey J & Sagar P. 2013. Salvin's albatross population trend at the Bounty Islands 1997-2011; Salvin's albatross population at the Bounty Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB & Jensz K. 2013. Southern royal albatross at Enderby Island – analysis of aerial photographs 2013. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB & Jensz K. 2014A. Gibson's Albatross at Disappointment Island - Analysis of aerial photographs. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB & Jensz K. 2014B. Buller's albatross aerial survey at the Solander Islands 2014. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB & Jensz K. 2019. Aerial survey of Salvin's albatross at the Bounty Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Elliott GP, French RK, Jensz K, Muller CG & Walker KJ. 2020. Development of aerial monitoring techniques to estimate population size of great albatrosses (Diomedea spp.). Notornis 67: 321-331.

Baker GB, Hedley G, Cunningham R & Waugh SM. 2020. Estimated population size of the Westland Petrel, 2007-2011. Fisheries New Zealand, Wellington, New Zealand.

Baker GB, Jensz K & Hamer D. 2014A. Southern royal albatross at Enderby Island – analysis of aerial photographs 2014. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Jensz K & Sagar P. 2014B. 2013 Aerial survey of Salvin's albatross at the Bounty Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Jensz K & Sagar P. 2015. 2014 Aerial survey of Salvin's albatross at The Snares, Western Chain. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Jensz K, Bell M, Fretwell PT & Phillips RA. 2017. Seabird population research, Chatham Islands; 2016/17 aerial photographic survey. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Baker GB, Jenzs K, Cunningham R, Robertson G, Sagar P, Thompson TR & Double MC. 2023. Population assessment of White-capped Albatross *Thalassarche steadi* in New Zealand. Emu – Austral Ornithology 123: 60-70.

Bartle JA 1985. Report on research on Westland black petrels 1969-83 with an outline of future objectives and a brief summary of management problems. Unpublished report of the National Museum of New Zealand, Wellington, New Zealand.

Bell BD. 1963. Classified notes on birds on and about Auckland Islands. Unpublished report.

Bell BD. 1982: Auckland Island expedition 21.2.82–24.3.82. Wildlife Service, Wellington, New Zealand.

Bell E, Lamb S, Ray S. 2022A. Population trends and breeding population size of black petrels (*Procellaria parkinsoni*) – 2020/2021 operational report. New Zealand Aquatic Environment and Biodiversity Report No. 280. Fisheries New Zealand, Wellington, New Zealand.

Bell E, Ray S & Crowe P. 2021. Population trends, at-sea distribution, and breeding population size of black petrels (*Procellaria parkinsoni*) on Great Barrier Island/Aotea: 2019/2020 operational report. New Zealand Aquatic Environment and Biodiversity Report No. 270. Fisheries New Zealand, Wellington, New Zealand.

Bell E, Ray S, Crowe P, Butler D, Bell M & McArthur N. 2020. Population trends, at-sea distribution, and breeding population size of black petrels (*Procellaria parkinsoni*) on Great Barrier Island/Aotea: 2018/2019 operational report. New Zealand Aquatic Environment and Biodiversity Report No. 246. Fisheries New Zealand, Wellington, New Zealand.

Bell EA & Sim JL. 2000A. Survey and monitoring of black petrels on Great Barrier Island, 1997/98. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA & Sim JL. 2000B. Survey and monitoring of black petrels on Great Barrier Island, 1998/99. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA & Sim JL. 2000C. Survey and monitoring of black petrels on Great Barrier Island, 1999/2000. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA & Sim JL. 2002. Survey and monitoring of black petrels on Great Barrier Island, 2000/01. DOC Science Internal Series 48. Department of Conservation, Wellington, New Zealand.

Bell EA & Sim JL. 2003A. Survey and monitoring of black petrels on Great Barrier Island, 2001/02. DOC Science Internal Series 134. Department of Conservation, Wellington, New Zealand.

Bell EA & Sim JL. 2003B. Survey and monitoring of black petrels on Great Barrier Island, 2002/03. DOC Science Internal Series 135. Department of Conservation, Wellington, New Zealand.

Bell EA & Sim JL. 2005. Survey and monitoring of black petrels on Great Barrier Island, 2003/04. DOC Science Internal Series 213. Department of Conservation, Wellington, New Zealand.

Bell EA, Burgin D, Sim J, Dunleavy K, Fleishman A & Scofield RP. 2018. Population trends, breeding distribution and habitat use of black petrels (Procellaria parkinsoni) - 2016/2017 operational report. New Zealand Aquatic Environment and Biodiversity Report No. 198. Fisheries New Zealand, Wellington, New Zealand.

Bell EA, Lamb S & Maclean C. 2023. Key demographic parameters and population trends of tākoketai/black petrels (*Procellaria parkinsoni*) on Aotea/Great Barrier Island: 2022/23. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Lamb S & Ray S. 2024. Key demographic parameters and population trends of tākoketai/black petrels (*Procellaria parkinsoni*) on Aotea/Great Barrier Island: 2023/24. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Lamb S & Ray S. 2025. Key demographic parameters and population trends of tākoketai/black petrels (*Procellaria parkinsoni*) on Aotea/Great Barrier Island: 2024/25. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Mischler C, Sim JL & Scofield P. 2015. Population parameters of the black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island), 2014/15. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Mischler C, Sim JL, Scofield P, Francis C, Edward A & Landers T. 2014. At-sea distribution and population parameters of the black petrels (Procellaria parkinsoni) on Great Barrier Island (Aotea Island), 2013/14. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Mischler CP, MacArthur N & Sim JL. 2016A. Black petrel (*Procellaria parkinsoni*) population study on Hauturu-o-Toi/Little Barrier Island, 2015/16. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Mischler CP, McArthur N, Sim JL, Scofield RP. 2016B. Black petrel (*Procellaria parkinsoni*) population study on Aotea/Great Barrier Island, 2015/16. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Sim J, Scofield P, Francis C & Landers T. 2013. At-sea distribution and population parameters of the black petrels (Procellaria parkinsoni) on Great Barrier Island (Aotea Island), 2012/13. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Sim JL & Scofield P. 2007. Demographic parameters of the black petrel (Procellaria parkinsoni). DOC Science Internal Series 273. Department of Conservation, Wellington, New Zealand.

Bell EA, Sim JL & Scofield P. 2009. Population parameters and distribution of the black petrel (Procellaria parkinsoni), 2005/06. DOC Science Internal Series 307. Department of Conservation, Wellington, New Zealand.

Bell EA, Sim JL & Scofield P. 2011. Population parameters and distribution of the black petrel (Procellaria parkinsoni), 2007/08. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Sim JL, Scofield P & Francis C. 2010. Population parameters o the black petrels (Procellaria parkinsoni) on Great Barrier Island (Aotea Island), 2009/2010. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell EA, Welch M & Lamb S. 2022B. Key demographic parameters and population trends of tākoketai/black petrels (*Procellaria parkinsoni*) on Aotea/Great Barrier Island: 2021/22. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M, Bell D & Boyle D. 2017A. Chatham Island Mollymawk research on Te Tara Koi Koia: November 2016. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M, Bell D, Boyle D & Tuanui-Chisholm H. 2017B. Motuhara seabird research: December 2016. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M, Bell D, Boyle D & Tuanui-Chisholm. 2018. Rangitatahi Seabird Research: December 2017. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2021. Motuhara seabird research: field trip report January 2021. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2022. Motuhara seabird research: field trip report January 2021. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2023. Motuhara seabird research: field trip report Dec 2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2024. Motuhara seabird research: field trip report January 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Bell M. 2026. Seabird research – Chatham Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Best HA & Owen KL 1976. Distribution of breeding sites of the Westland black petrel (*Procellaria westlandica*). Notornis 23: 233 242.

Childerhouse S, Robertson C, Hockly & Gibbs N. 2003. Royal albatross (*Diomedea epomophora*) on Enderby Island, Auckland Islands. DOC Science Internal Series 144.Department of Conservation, Wellington, New Zealand.

Chilvers L. 2007. NZ sea lion research trip, Auckland Islands, November 28th 2006 to February 27th 2007. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Clark G, Andrea B, & Amey J. 1998. The "Totorore" Expedition to the Bounty Islands, New Zealand. October 1997 to January 1998 Report. New Zealand Maritime Museum, Auckland, New Zealand.

Clark G. 1996. The "Totorore" expedition to the Snares Western Chain; September 1995 to December 1995. New Zealand Maritime Museum, Auckland, New Zealand.

Cooper WJ, Miskelly CM, Morrison K & Peacock RJ. 1986. Birds of the Solander Islands. Notornis 33: 77-89.

Dawson EW. 1973. Albatross populations at the Chatham islands. Notornis 20: 210–230.

Dell RK 1954. Field notes. ANARE to Enderby Island (Auckland Islands) – Holmlea. MS Papers 2366 297.

Department of Conservation Coastal Otago District Operations. 2026. Pukekura annual report. 2024/25 Season. Department of Conservation, Dunedin, New Zealand.

Dilks PJ & Grindell JM. 1983. Visit to Campbell Island by P.J. Dilks and J.M. Grindell, 11 January - 23 February 1983. Unpublished file report, Ecology Division.

Dilks PJ. 1979: Observations on the food of feral cats on Campbell Island. New Zealand Journal of Ecology 2: 64–66.

Elliot G, Walker K, Rexer-Huber K, Tinnemans J, Long J, Sagar S, Osborne J & Parker G. 2024. Gibson's wandering albatross: demography, satellite tracking and census. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G & Walker K. 2013. Gibson's wandering albatross research Adams Island 2013. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G & Walker K. 2014. Gibson's wandering albatross at Adams Island – population study. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G & Walker K. 2017. Antipodean wandering albatross census and population study 2017. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G & Walker K. 2018. Antipodean wandering albatross census and population study 2018. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G & Walker K. 2019. Antipodean wandering albatross census and population study 2019. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G & Walker K. 2020. Antipodean wandering albatross satellite tracking and population study: Antipodes Island 2020. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Waker K, Rexer-Huber K, Tinnemans J, McDonald A, Rawlence T & Parker G. 2025. Gibson's wandering albatross: drone-based population estimate, demography and at-sea distribution. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Walker K, Parker G & Rexer-Huber K. 2016. Gibson's wandering albatross census and population study 2015/16. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Walker K, Rexer-Huber K & Parker G. 2023. White-capped Albatross population study: February 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Falla RA. 1954: Field notes. Auckland Islands trip, March 12–23 1954. MS Papers 2366–297.

Fischer JH & Thompson T. 2025. Monitoring ecotourism impacts on Toroa | Southern Royal Albatross breeding success. Department of Conservation, Wellington.

Francis RIC & Sagar PM. 2012. Modelling the effect of fishing on southern Buller's albatross using a 60-year dataset. New Zealand Journal of Zoology 39: 3-17.

Francis RICC. 2012. Fisheries risks to the population viability of White-capped Albatross *Thalassarche steadi*. Fisheries New Zealand, Wellington, New Zealand.

Fraser M, Henderson G, Robertson CJR & Scofield P. 2011. Population dynamics of the Chatham Albatross at the Pyramid, 19 November – 2 December 2010. Ministry of Fisheries, Wellington, New Zealand.

Fraser M, Bell M, Scofield P & Robertson CJR. 2009A. Population assessment of Northern Buller's Albatross and Northern Giant Petrel at the Fortyfours. Ministry of Fisheries, Wellington, New Zealand.

Fraser M, Hunt S, Scofield P & Robertson CJR. 2009B. Population dynamics of the Chatham Albatross at the Pyramid, 9 November – 7 December 2008. Ministry of Fisheries, Wellington, New Zealand.

Fraser M, Cameron N, Scofield P & Robertson CJR. 2010A. Population assessment of Northern Buller's Albatross and Northern Giant Petrels at the Fortyfours, Chatham Islands, 1-8 December 2009. Ministry of Fisheries, Wellington, New Zealand.

Fraser M, Palmer D, Deppe L, Scofield P & Robertson CJR. 2010B. Population dynamics of the Chatham Albatross at the Pyramid, 20 November – 14 December 2009. Ministry of Fisheries, Wellington, New Zealand.

Freeman AND & Wilson K-J. 2002. Westland petrels and hoki fishery waste: opportunistic use of a readily available resource. Notornis 49: 139-144.

French RK, Miskelly CM, Muller CG, Russ RB, Taylor GA & Tennyson AJD. 2020. Birds of Enderby Island, Auckland Islands, New Zealand subantarctic. Notornis 67: 189-212.

Fretwell PT, Scofield P & Phillips RA. 2017. Using super-high resolution satellite imagery to census threatened albatross. Ibis 159: 481-490.

Frost PGH, Baker B, Fischer JH, Sagar PM. 2025. Population survey of southern Buller's albatross *Thalassarche bulleri bulleri* on the Solander Islands | Hautere, March 2024. Marine Ornithology 53: 361-371.

Frost PGH, Bell MD & Taylor GA. 2021C. Trial-camera assessment of the fates of Northern Royal Albatross and Northern Buller's Mollymawk chicks: 2021 breeding season, Motuhara. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2017. Aerial census of Northern Royal Albatross (*Diomedea sanfordi*) fledglings on Rangitatahi (The Sisters) and Motuhara (Forty-fours), July 2017. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2019. Aerial surveys of Northern Royal Albatross (*Diomedea sanfordi*) on the Chatham Islands: 2017-2018 breeding season. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2021A. Status of Northern Royal Albatross *Diomedea sanfordi* nesting on the Chatham Islands, December 2020. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2021B. Numbers of Northern Royal Albatross chicks and Northern Giant Petrel adults on the Chatham Islands, September 2020. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2022. A Census of Northern Royal Albatross nesting on the Chatham Islands, February 2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2024A. Nesting of Northern Royal Albatross | Toroa *Diomedea sanfordi* on the Chatham Islands: 2022-23 breeding season. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2024B. Aerial survey of Northern Royal Albatross | Toroa (*Diomedea sanfordi*) populations on the Chatham Islands, December 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH. 2026. Trail-camera monitoring of White-capped Albatross (*Thalassarche steadi*) on Disappointment Island: 2018, 2022 and 2023) breeding seasons. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Fu D & Sagar P. 2016. The 2014 demographic assessment of the Snares islands population of Southern Buller's albatross (*Diomedea bulleri bulleri*). New Zealand Aquatic Environment Biodiversity Report 165. Ministry for Primary Industries, Wellington, New Zealand.

Guard A. 1968. Wildlife breeding locations: albatross distribution. No. I in a series of two unpublished maps drawn from observations by members of the 1964-65-66-67-68 expeditions; fieldwork and research by CG Surrey and D Paull.

Hamilton S, Kemp J, Elliott G, Walker K, Wiltshire A 2000. Monitoring Gibson's wandering albatross; 1999/2000. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Imber MJ. 1987. Breeding ecology and conservation of the black petrel (*Procellaria parkinsoni*). Notornis 34: 19–39.

Marchant S & Higgins PJ. 1990. Handbook of Australian, New Zealand and Antarctic Birds. Volume 1: Ratites to ducks. Part A, Ratites to petrels. Oxford University Press, Melbourne, Australia.

Mattern T. 2022. Tawaki Project: Bounty-Antipodes expedition 2022.

Mattern T. 2024a. Tawaki Project: Antipodes-Bounty expedition 2023/24.

Mattern T. 2024b. Tawaki Project: Bounty-Antipodes expedition 2024.

Michler C & Wickes C. 2023. POP2022-11. Campbell Island/Motu Ihupuku Seabird Research & Operation Endurance February 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Mischler C, Moore P, Thompson H, Hamilton K, Pryde M. 2025. Campbell Island Seabird Research. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Mischler C, Thompson T, Moore P, Philip B & Wickes C. 2024. Campbell Island Seabird Research. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Mischler C. 2020. Campbell Island/ Motu Ihupuku Seabird Research; Operation Endurance March 2020. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Miskelly CM. 1984. Birds of the Western Chain, Snares Islands 1983-84. Notornis, 31: 209–223.

Mitchell SJ & Ensor PH. 1986. Hooker's sea lion survey, Enderby Island, Auckland Islands, January 1980. In: Penniket A, Garrick A & Breese E. (compilers) Preliminary reports of expeditions to the Auckland Islands Nature Reserve 1973–1984. Reserve Series. Department of Lands and Survey. Wellington, New Zealand.

Moore PJ & Moffat RD. 1990. Research and management projects on Campbell Island 1987-88. Science and Research Internal Report 57. Department of Conservation, Wellington, New Zealand.

Moore PJ, Larsen EJ, Charteris M & Pryde M. 2012. Southern royal albatross on Campbell Island/Motu Ihupuku: solving a band injury problem and population survey, 2004-08. DOC Research and Development Series 333. Department of Conservation, Wellington, New Zealand.

Moore PJ, Scott JJ, Joyce LJ & Peart M. 1997a: Southern royal albatross *Diomedea epomophora epomophora* census on Campbell Island, 4 January–6 February 1996, and a review of population figures. Science and Research Series 101. Department of Conservation, Wellington, New Zealand.

Moore PJ, Waugh SM, West C & Mitchell G. 1997b. Preliminary results of a Southern royal albatross *Diomedea epomophora epomophora* census, Campbell Island, 12 January - 10 February 1995. Science and Research series No. 100. Department of Conservation, Wellington, New Zealand.

Parker G & Rexer-Huber K. 2020. Drone-based Salvin's albatross population assessment: feasibility at the Bounty Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Parker G, Sagar P, Thompson D & Rexer-Huber K. 2017. The establishment of a marked population of white-capped albatross to allow estimation of adult survival & other demographic parameters, Disappointment Island, Auckland Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Parker GC, Elliott G, Walker K & Rexer-Huber K. 2022. Gibson's albatross and white-capped albatross in the Auckland Islands 2021-2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Parker GC, Rexer-Huber K, Walker K & Elliott G. 2023. Antipodean wandering albatross population study 2023. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Paull D & Surrey G. 1969. Southern royal albatross and giant petrel on Campbell Island over the years 1966, 1967, 1968, 1969. Unpublished nest records and reports, Volumes 1-3.

Rexer-Huber K, Elliott G, Thompson D, Walker K & Parker GC. 2019. Seabird populations, demography and tracking: Gibson's albatross, white-capped albatross and white-chinned petrels in the Auckland Islands 2018-19. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Elliott G, Walker K, Thompson D & Parker GC. 2020. Seabird population research: Gibson's albatross and white-capped albatross in the Auckland Islands 2019-20. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Mattern T, Ellenberg U & Sagar P. In prep. Salvin's Albatross Western Chain population study. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Whitehead E, Parker G, Patterson E, Walker K, Welch J & Elliott G. 2024. Antipodean wandering albatrosses and white-chinned petrels 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Whitehead E, Walker K & Elliott G. 2025. Antipodean wandering albatrosses and white-chinned petrels 2025. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rebergen A. 2000. Royal albatross counts Chatham Islands. Period November 1972-August 1975 and from September 1989-April 2000. Unpublished report. Department of Conservation, Wellington, New Zealand.

Richard Y, Perriman L, Lalas C & Abraham ER. 2015. Demographic rates of northern royal albatross at Taiaroa Head, New Zealand. PeerJ 3: e906.

Robertson CJR & Richdale LE. 1993. The breeding phenology of the Royal Albatross (*Diomedea epomophora sanfordi*) 1937-1974. Conservation Advisory Science Notes No. 48. Department of Conservation, Wellington, New Zealand.

Robertson CJR & van Tets GF. 1982. The status of birds at the Bounty Islands. Notornis 29: 311-336.

Robertson CJR, Bell D & Scofield P. 2003. Population Assessment of the Chatham Albatross at The Pyramid, December 2001. DOC Science Internal Series 91. Department of Conservation, Wellington.

Robertson CJR, Bell D & Scofield P. 2004. Population assessment of the Chatham mollymawk at the Pyramid, September and December, 2003. Unpublished report.

Robertson CJR. 1975. Report on the distribution, status, and breeding biology of the Royal Albatross, Wandering Albatross and White-capped Mollymawk on the Auckland Islands. In Yaldwyn JC (ed). Preliminary Results of the Auckland Islands Expedition 1972-1973. Department of Lands and Survey, Wellington, New Zealand.

Robertson CJR. 1991. Questions on the harvesting of Toroa in the Chatham Islands. Science & Research Series 35. Department of Conservation, Wellington.

Robertson CJR. 1993. Timing of egg laying in the Royal Albatross (*Diomedea epomophora*) at Taiaroa Head 1937-1992. Conservation Advisory Science Notes 50. Department of Conservation, Wellington, New Zealand.

Robertson CJR. 2005. Aerial Photography of Northern Royal Albatross on the Forty-Fours and Sisters Islands, Chatham Islands. Department of Conservation, Wellington, New Zealand.

Robertson G & Gales R. 1998. Albatross biology and conservation. Surrey Beatty & Sons, Chipping Norton, Australia.

Russ R. 1974. Report on activities during the December 1973 trip to Auckland Islands. 5 Dec – 20 Dec. New Zealand Wildlife Service, Wellington, New Zealand.

Sagar P & Thompson D. 2007. The impact of fisheries bycatch on Southern Buller's Mollymawks. Report on the 2006 & 2007 field season.

Sagar P, Charteris M, Parker G, Rexer-Huber K & Thompson D. 2018. Salvin's Albatross: Bounty Island population project – ground component. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Rexer-Huber K, Schultz H, Simister K, Thompson D & Parker G. 2025. Population studies of Southern Buller's Albatrosses at Tini Heke | The Snares Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Rexer-Huber K, Thompson D & Parker G. 2024. Population studies of southern Buller's albatrosses at Tini Heke / The Snares Islands and Hautere / Solander Islands. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Thompson D & Scofield P. 2017. Population study of Southern Buller's Albatross on the Snares. Deepwater Group Ltd. Nelson, Aotearoa New Zealand.

Sagar P, Torres L & Thompson D. 2009. Demography and distribution of Buller's Albatrosses Thalassarche bulleri bulleri: final research report of the 2009 field season. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Torres L & Thompson D. 2010. Demography and distribution of Buller's Albatrosses Thalassarche bulleri bulleri: final research report of the 2010 field season. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Torres L & Thompson D. 2012. Demography and distribution of Buller's Albatrosses Thalassarche bulleri bulleri: final research report of the 2012 field season. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P, Torres L & Thompson D. 2013. Demography and tracking of Buller's Albatrosses at The Snares, and tracking of Snares Crested Penguins and Rockhopper Penguins from The Snares and Campbel Island respectively: Final research report of the 2013 field season. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P. 2014. Population studies of Southern Buller's Albatrosses on the Snares. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar P. 2015. Population study of Southern Buller's Albatrosses on the Snares. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar PM & Stahl J-C. 2005. Increases in the numbers of breeding pairs in two populations of Buller's Albatross (*Thalassarche bulleri bulleri*). Emu – Austral Ornithology 105: 49-55.

Sagar PM, Charteris MR & Scofield RP. 2014. Salvin's albatross population size and survival at the Snares Western Chain. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Sagar PM, Charteris MR, Carroll JWA & Scofield PR. 2011. Population size, breeding frequency and survival of Salvin's albatrosses (*Thalassarche salvini*) at the Western Chain, The Snares, New Zealand. Notornis 58: 57-63.

Sagar PM, Molloy J, Tennyson AJD & Butler D. 1994. Numbers of Buller's Mollymawks breeding at the Snares Islands. Notornis 41: 85-92.

Sagar PM, Stahl J-C, Molloy J, Taylor GA & Tennyson AJD. 1999. Population size and trends within the two populations of Southern Buller's Albatross *Diomedea bulleri bulleri*. Biological Conservation 89: 11-19.

Sagar PM. 1977. Birds of the Western Chain, Snares Islands, New Zealand. Notornis 24: 178-183.

Scofield P, Fraser M & Robertson CJR. 2008. Population dynamics of the Chatham Albatross at the Pyramid, 19-29 November 2007. Unpublished report.

Scofield P. 2011. Data collection of demographic, distributional, and trophic information on selected seabird species to allow estimation of effects of fishing on populations viability. Ministry of Fisheries, Wellington, New Zealand.

Taylor GA. 1988. Report to the Department of Conservation on an expedition to the Auckland Islands National Reserve, February 1988. Department of Conservation, Wellington, New Zealand.

Taylor GA. 1988. Report to the Department of Conservation on an expedition to the Auckland Islands National Reserve, February 1988. Department of Conservation, Wellington.

Taylor RH. 1971. Influence of man on vegetation and wildlife of Enderby and Rose Islands, Auckland Islands. New Zealand Journal of Botany 9: 225–268.

Taylor RH. Bell BD & Wilson PR. 1970. Royal albatrosses, feral sheep and cattle on Campbell Island. New Zealand Journal of Science 13: 78–88.

Thompson D & Sagar P. 2008a. Annual report 2006/07: A population and distributional study of white-capped albatross (Auckland Islands). Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D & Sagar P. 2008b. Annual report 2007/08: A population and distributional study of white-capped albatross (Auckland Islands). Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D & Sagar P. 2020. Population studies of southern Buller's albatrosses on the Snares Islands | Tini Heke. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D & Sagar P. 2022. Population studies of southern Buller's Albatross on the Snares. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D & Sagar P. 2023. Population studies of southern Buller's albatrosses on Tini Heke the Snares. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson D, Sagar P, Baker B & Jensz K. 2017. Southern Buller's Albatross survey at Solander Islands 2016. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Thompson DR. 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. 228. Fisheries New Zealand, Wellington, New Zealand.

Thompson T. 2025. Advancing remote monitoring of albatrosses through emerging technologies to inform conservation. MSc thesis. University of Otago, Dunedin, New Zealand.

Walker K & Elliott G. 2002. Population changes and biology of the wandering albatross *Diomedea exulans gibsoni* at the Auckland Islands. DOC Science Internal Series 68. Department of Conservation, Wellington, New Zealand.

Walker K & Elliott G. 2005. Population changes and biology of the Antipodean wandering albatross. Notornis 52: 206-214.

Walker K & Elliott G. 2015. Gibson's wandering albatross population study 2014-15. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Walker K & Elliott G. 2022. Antipodean wandering albatross satellite tracking and population study on Antipodes Island in 2021 and 2022. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Walker K, Elliott G, Parker GC & Rexer-Huber K. 2023. Gibson's wandering albatross: population study and potential for drone-based while-island census. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Walker K, Elliott G, Rexer-Huber K & Parker G. 2017. Gibson's wandering albatross population study and census 2016/17. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Walker KJ, Elliott GP, Rexer-Huber K, Parker GC, Sagar PM & McClelland PJ. 2021. Shipwrecks and mollymawks: an account of Disappointment Island Birds. Notornis 67: 213-245.

Warham J & Bell BD. 1979. The birds of Antipodes Island, New Zealand. Notornis 26: 121-169.

Warham J & Bennington SL. 1983. A census of Buller's Albatross Diomedea bulleri at the Snares Islands. Emu 83: 112-114.

Waugh SM & Wilson KJ. Threats and threat status of the Westland Petrel *Procellaria westlandica*. Marine Ornithology 45: 195-203.

Waugh SM, Barbraud C, Adams L, Freeman AND, Wilson K-J, Wood G, Landers TJ & Baker GB. 2015. Modelling the demography and population dynamics of a subtropical seabird, and the influence of environmental factors. The Condor: Ornithological Applications 117: 147-164.

Waugh SM, Barbraud C, Delord K, Simister KLJ, Baker GB, Hedley GK, Wilson K-J & Rands DRD. 2020. Trends in density, abundance, and response to storm damage for Westland Petrels *Procellaria westlandica*, 2007-2019.

Waugh SM, Doherty PF, Freeman AND, Adams L, Wood GC, Bartle JA & Hedley GK. 2006. 2006. Demography of Westland Petrels (*Procellaria westlandica*), 1995–2003. Emu 106: 219-226.

Waugh SM, Sagar PM & Paull D. 1997. Laying dates, breeding success and annual breeding of southern royal albatrosses *Diomedea epomophora* at Campbell Island during 1964–69. Emu 97: 194–199.

Westerskov K. 1963. Ecological factors affecting the distribution of a nesting royal albatross population. Pp. 795–811 in Proceedings of the 13th International Ornithological Congress.

Wilson PR & Elliott GP. 1981. Report on sheep, cattle and royal albatross counts at Campbell Island during 1979/80 and 1980/81: with other biological notes recorded during the 1979/80 visit. Unpublished file report, Ecology Division.

Wood GC & Otley HM. 2013. An assessment of the breeding range, colony sizes and population of the Westland Petrel (*Procellaria westlandica*). New Zealand Journal of Zoology 40: 186–195.j

Zhang J, Bell EA & Roberts JO. 2020. Demographic assessment of black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island). New Zealand Aquatic Environment and Biodiversity Report N. 244. Fisheries New Zealand, Wellington, New Zealand.

SUPPLEMENTARY MATERIAL 3.

Are there predictable intra-annual patterns in the proportion of apparently occupied sites that can be integrated into the abundance data processing framework?

In our abundance data processing framework, we apply Ψ^o and Ψ^P in parallel (Fig. 2) where relevant (e.g., counts of AOS conducted after mean laying). This contrasts with a widespread assumption that Ψ^o and Ψ^p are inherently linked, particularly for ψ^{ABA}. Various studies on albatrosses have argued that the number of apparently occupied sites may increase over time as a function of nest failure (e.g., Poncet *et al.* 2006, Robertson *et al.* 2008, Baker *et al.* 2023, Frost *et al.* 2025). Some studies have attempted to investigate the influence of various factors on the proportion of non-breeders, apparently breeding adults, and/or BOGs present on colonies throughout the breeding period to better understand intra-annual change in ψ^{ABA}. For example, Stahl & Sagar (2006) showed that the proportion of Buller's Albatross BPs compared to BOGs ($\psi^{BOG} \cdot \psi^{ABA}$) remained stable during incubation and chick-rearing despite the onset of nest failures. Walker *et al.* (2020) illustrated that annual variation, not intra-annual variation, had the greatest influence on the proportion of White-capped Albatross BP compared to AOS (ψ^{ABA}). Elliott *et al.* (2024, 2025) and Rexer-Huber *et al.* (2024, 2025) illustrated that the proportion of BPs compared to BOGs ($\psi^{BOG} \cdot \psi^{ABA}$) also does not vary in a predicable manner for both subspecies of Antipodean Albatross. Thus, a range of studies have indicated the importance of considering species-specific, diurnal, and interannual variation as well as any intra-annual variation in occupancy error, but no study has been able to show conclusively that the proportion of non-breeders, apparently breeding adults, and/or BOGs present on colonies changes predictably throughout the breeding period.

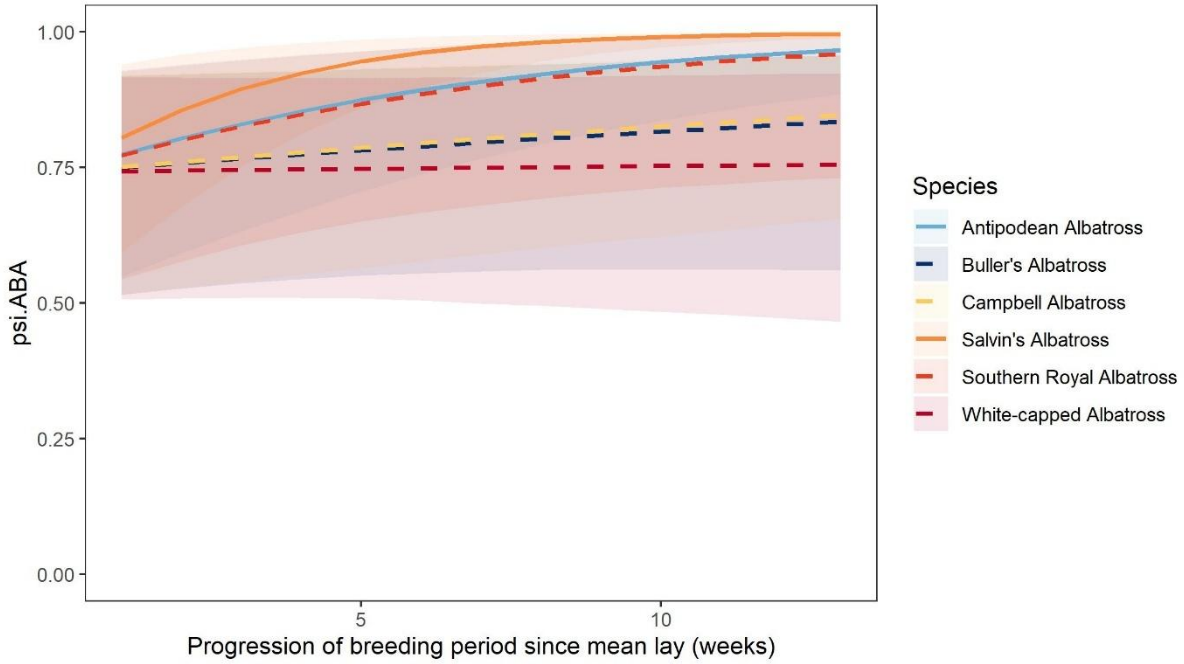
We leveraged all the information we collated on Ψ^o for our case studies (Supplementary Tables 1-22) to explicitly investigate if Ψ^o is subject to predictable intra-annual patterns. Specifically, we used data from 48 albatross sub-counts quantifying BP as a proportion of AOS (ψ^{ABA}) to fit a Bayesian binomial GLMM with a logit-link function:

1.
$$\begin{aligned} &\mathrm{bp}_c \sim \mathrm{Bin}(\mathrm{aos}_c,\ \psi_c^{ABA}) \\ &\mathrm{logit}(\psi^{ABA}) = \alpha_{ABA} + \theta_s^{\beta^{ABA}} \cdot s_{s,c} \cdot t_c + \varepsilon_{y,c}^{ABA} \end{aligned}$$

In which bp_c is the number of BP counted per sub-count c investigating Ψ^o, aos_c is the number of AOS counted per sub-count, α_{ABA} is the model intercept, $\theta_s^{\beta^{ABA}}$ is a design-matrix of species-specific fixed effects on intra-annual variation, $s_{s,c}$ is the species under consideration, t_c is the timing (in relation to species-specific phenology of the count), and $\varepsilon_{y,c}^{ABA}$ is a random annual effect. Thus, we leveraged the information-sharing ability of Bayesian statistics, while investigating if there are general patterns in intra-annual variation in ψ^{ABA}.

The results of this supplementary analysis are illustrated in Supplementary Figure 1., which highlight that the majority of the species included in this analysis did not show any significant intra-annual patterns in ψ^{ABA} and highlighted uncertainty above all. Antipodean

Albatross did show a significant intra-annual pattern in ψ^{ABA}, in which the proportion of BP in relation to AOS increases as the breeding period progresses, presumable as pre-breeders become less common on the colony. Salvin's Albatross showed a similar significant pattern, but this potentially requires more investigation and the Salvin's Albatross sample size was particularly small <5 data points.



Supplementary Figure 1. Model estimates of the proportion breeding pairs in relation to apparently occupied sites (ψ^{ABA}) over the course of the breeding period for six species of albatross. Dashed lines indicate non-significance.

Our additional analysis in combination with the pre-existing body of evidence illustrates the lack of strong evidence for predictable intra-annual patterns in ψ^{ABA}. Consequently, the most parsimonious approach to account for ψ^{ABA} (and by extension Ψ^{o}) is to use random effects within relatively simple binomial GLMMs that adjust individual counts using model estimates of ψ^{ABA} without additional attempts to incorporate intra-annual effects, i.e., as per Eq. 2ABC. Naturally, there are clear, predictable intra-annual patterns in Ψ^{p} (Fig. 4), and thus, we account for this source of error while incorporating intra-annual patterns. Ultimately, we therefore accounted for these two counter-acting sources of error (Ψ^{o} results in overestimation, Ψ^{p} results in underestimation) in parallel, but separately.

References

Baker GB, Jenzs K, Cunningham R, Robertson G, Sagar P, Thompson TR & Double MC. 2023. Population assessment of White-capped Albatross *Thalassarche steadi* in New Zealand. Emu – Austral Ornithology 123: 60-70.

Elliot G, Walker K, Rexer-Huber K, Tinnemans J, Long J, Sagar S, Osborne J & Parker G. 2024. Gibson's wandering albatross: demography, satellite tracking and census. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Elliott G, Walker K, Rexer-Huber K, Tinnemans J, McDonald A, Rawlence T & Parker G. 2025. Gibson's wandering albatross: drone-based population estimate, demography and at-sea distribution. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Frost PGH, Baker GB, Fischer JH & Sagar PM. 2025. Population survey of Southern Buller's Albatross *Thalassarche bulleri bulleri* on the Solander Islands | Hautere, March 2024. Marine Ornithology 53: 361-371.

Poncet S, Robertson G, Phillips RA, Lawton K, Phalan B, Trathan PN & Croxall JP. 2006. Status and distribution of wandering, black-browed and grey-headed albatrosses breeding at South Georgia. Polar Biology 29: 772-781.

Rexer-Huber K, Whitehead E, Parker G, Patterson E, Walker K, Welch J & Elliott G. 2024. Antipodean wandering albatrosses and white-chinned petrels 2024. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Whitehead E, Walker K & Elliott G. 2025. Antipodean wandering albatrosses and white-chinned petrels 2025. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Robertson G, Moreno CA, Lawton K, Kirkwood D & Valencia J. 2008. Comparison of census methods for Black-browed Albatrosses breeding at Ildefonso Archipelago, Chile. Polar Biology 31: 153-162.

Stahl JC & Sagar PM. 2006. Behaviour and patterns of attendance of non-breeding birds at the breeding colony in a Buller's Albatross *Thalassarche bulleri* population at The Snares. Notornis 53: 327-338.

Walker KJ, Elliott GP, Rexer-Huber K, Parker GC, Sagar PM & McClelland PJ. 2021. Shipwrecks and mollymawks: an account of Disappointment Island Birds. Notornis 67: 213-245.

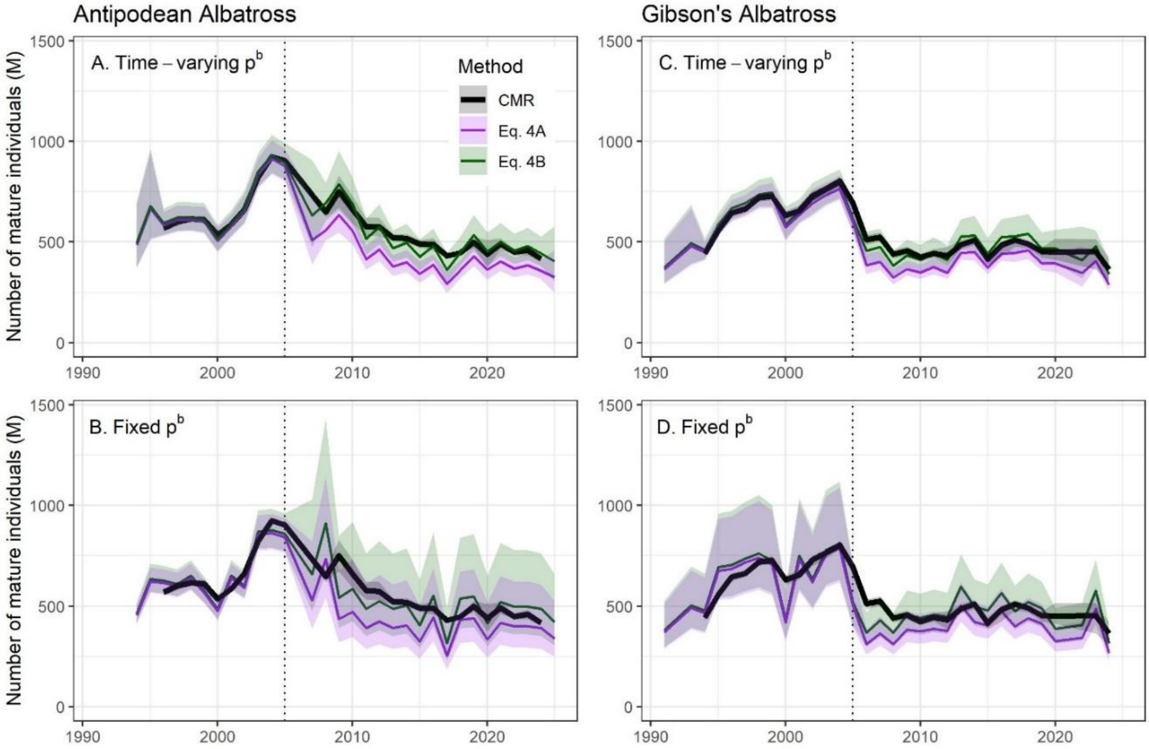
SUPPLEMENTARY MATERIAL 4.

Sensitivity testing conversions of BP to M under varying conditions

To establish which approach for converting BP to M was most suitable under varying conditions of data availability and skewed sex ratio, we evaluated both equations using data on Antipodean Albatross (*D. a. antipodensis*) from the Antipodes and Gibson's Albatross (*D. a. gibsoni*) from Adams Island, two biennially breeding taxa with a skewed sex ratio that has become evident after a population crash in 2005 (e.g., Richards *et al.* 2024) for which conversions to M are not straightforward.

To evaluate both conversions, we relied on the long-term capture-mark-recapture (CMR) data for Antipodean Albatross (including the Gibson's Albatross subspecies) and contrasted M estimates derived thereof using multi-state modelling (Elliott *et al.* 2025, Rexer-Huber *et al.* 2025) with estimates derived from Eq. 4A and 4B. We sourced BP from the ACAP database and extracted CMR-based estimates of $p_{t,i}^{b}$ (i.e., annually varying) and estimates of s_i pre- and post-2005 from Elliott *et al.* (2025) and Rexer-Huber *et al.* (2025) which we used to construct informative priors, specified via mean and precision ($s_{\leq 2005}^{antips}$ ~ N(0.491, 10000); $s_{\geq 2006}^{antips}$ ~ N(0.403, 2500); $s_{\leq 2005}^{gibsons}$ ~ N(0.487, 7500); $s_{\geq 2006}^{gibsons}$ ~ N(0.423, 1500)). Annual variation captured in $p_{t,i}^{b}$ is a product of in-depth multi-event CMR modelling and such information is not available for all populations of albatrosses and petrels. We therefore also repeated both equations with a fixed p_i^b using informative priors for the periods pre- and post-2005: ($p^{b}{}_{\leq 2005}^{antips}$ ~ N(0.501, 1500); $p^{b}{}_{\geq 2006}^{antips}$ ~ N(0.450, 150); $p^{b}{}_{\leq 2005}^{gibsons}$ ~ N(0.583, 150); $p^{b}{}_{\geq 2006}^{gibsons}$ ~ N(0.600, 300)).

The results of our evaluation (Fig. 6) indicated that, if a population has an even sex ratio, the simpler conversion, Eq. 4A, provides adequate approximations for M, but if a skewed sex ratio is present, Eq. 4A results in an underestimate of M, and Eq. 4B provides a more adequate estimate. Both conversions could be readily applied to years for which CMR-based M estimates did not exist (early 1990s), enabling longer timeseries, while incorporating uncertainty. However, when using fixed p^b instead of time-varying p^b estimates, uncertainty increased substantially, to the extent that the improvements provided by incorporation of s were no longer beneficial, resulting in an arguably lower performance of Eq. 4B. Based on this evaluation, we converted BP into M using Eq. 4A if no evidence exists for a skewed sex ratio, or using Eq. 4B, if such evidence exists and time-varying p^b estimates were available.



Supplementary Figure 2. Comparison of CMR-derived estimates of M (black lines) with two BP-to-M conversions using time-varying (AC) and fixed p^b (BD) for Antipodean (AB) and Gibson's Albatross (CD). The dotted vertical line indicates the population crash in both breeding sites after which a different *s* was applied.

References

Elliott G, Walker K, Rexer-Huber K, Tinnemans J, McDonald A, Rawlence T & Parker G. 2025. Gibson's wandering albatross: drone-based population estimate, demography and at-sea distribution. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Rexer-Huber K, Whitehead E, Walker K & Elliott G. 2025. Antipodean wandering albatrosses and white-chinned petrels 2025. Conservation Services Programme, Department of Conservation, Wellington, New Zealand.

Richard Y, Berkenbusch K, Crawford E, Tornquist M, Walker K, Elliott G, & Tremblay-Boyer L. 2024. Antipodean Albatross multi-threat risk assessment. New Zealand Aquatic Environment and Biodiversity Report No. 332. Fisheries New Zealand, Wellington, New Zealand.

SUPPLEMENTARY MATERIAL 5.

ACAPT Implementation via bespoke R package

See separate HTML file.

SUPPLEMENTARY MATERIAL 6.

Exploration of ACAPT performance under varying q modelling approaches

See separate HTML file.

SUPPLEMENTARY MATERIAL 7.

Brief description of IUCN Red List categories B-E (IUCN 2012).

No numeric thresholds for NT are provided in IUCN (2012). Instead, IUCN (2012) states "*A taxon is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future.*"

Sub-criteria	**Brief description**	**CR**	**EN**	**VU**
B1	Extent of occurrence	<100 km^2, with two out of: - Fragmented or at 1 site - Continuing decline - Extreme fluctuations	<5,000 km^2, with two out of: - Fragmented or at <5 sites - Continuing decline - Extreme fluctuations	<20,000 km^2, with two out of: - Fragmented or at <10 sites - Continuing decline - Extreme fluctuations
B2	Area of occupancy	<10 km^2, with two out of: - Fragmented or at 1 site - Continuing decline - Extreme fluctuations	<500 km^2, with two out of: - Fragmented or <5 sites - Continuing decline - Extreme fluctuations	<2,000 km^2, with two out of: - Fragmented or <5 sites - Continuing decline - Extreme fluctuations
C1	Small population size and pop. decline	M <250 and estimated pop. and decline of >25% over 1 GL (or 3 y)	M <2,500 and estimated pop. and decline of >20% over 2 GL (or 5 y)	M <10,000 and estimated pop. and decline of >10% over 3 GL
C2	Small pop. size and vulnerable pop. structure	M <250, a continuing decline, and one out of: - No pop. >50 M or 90% of M in one pop. - Extreme fluctuations	M <2,500, a continuing decline, and one out of: - No pop. >250 M or 95% of M in one pop. - Extreme fluctuations	M <10,000, a continuing decline, and one out of: - No pop. >1,000 M or all in one pop. - Extreme fluctuations
D1	Small pop. size	M <50	M <250	M <1,000
D2	Small area of occupancy	-	-	Restricted area of occupancy (<20 km^2) or <5 sites such that the species could become CR or EX in a very short period.
E	Probability of extinction	p^{ex} <50% within 3 GL (or 10 y)	p^{ex} <20% within 5 GL (or 20 y)	p^{ex} <10% within 100 y

References

IUCN. 2012. IUCN Red List categories and criteria, version 3.1, second edition. IUCN Species Survival Commission, IUCN, Cambridge, UK.

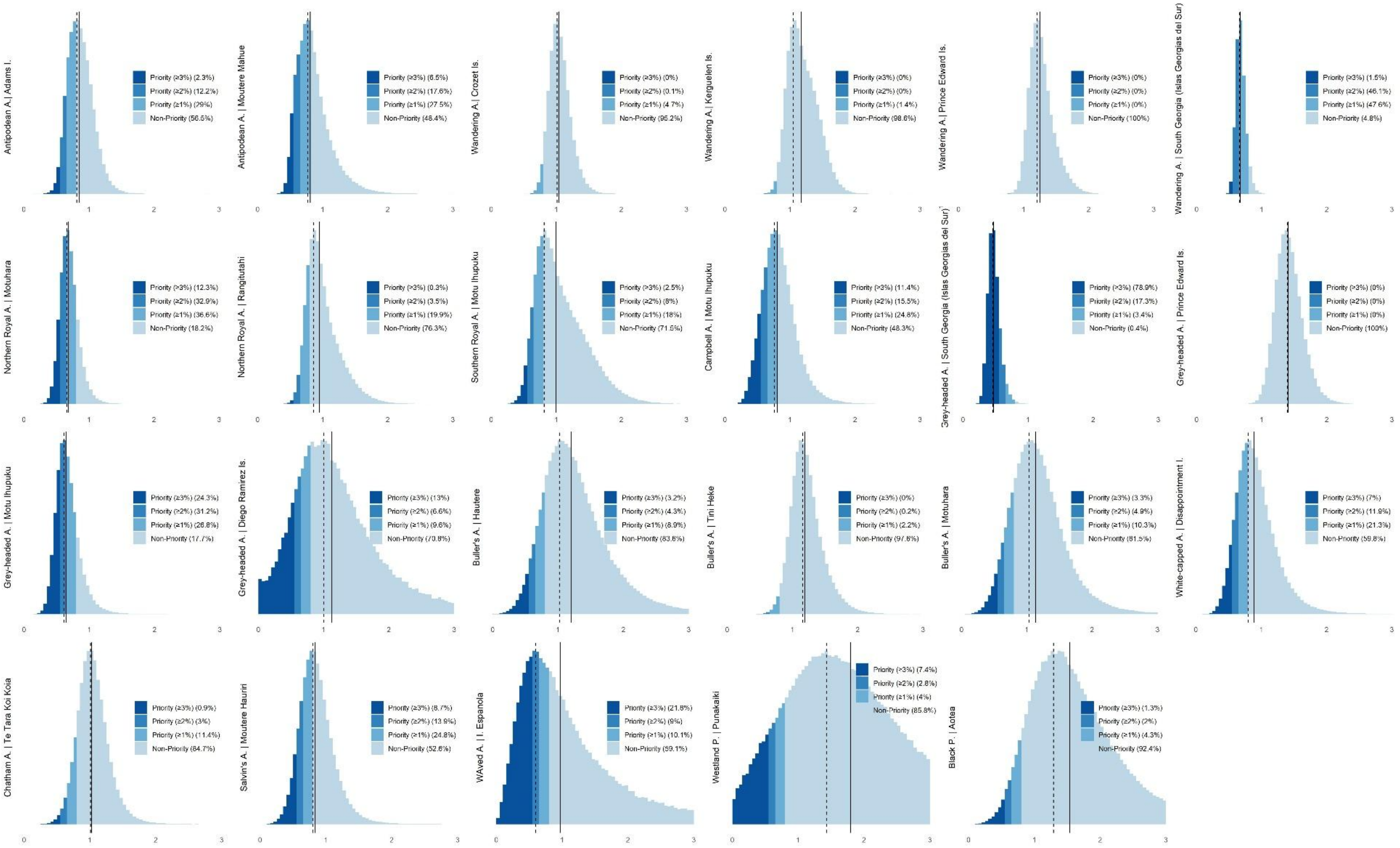
SUPPLEMENTARY MATERIAL 8.

Exploration of performance of density-based IUCN category evaluations under diminishing data quality.

See separate HTML file.

SUPPLEMENTARY MATERIAL 9.

Density-based evaluation of ACAP High Priority Populations.



Supplementary Figure 3. ACAP High Priority Population assessments. Legends provide the percentage allocation of iterations to different levels of population decline over 20 years (≥3% decline p/a, ≥2% decline p/a, ≥1% decline p/a or less); solid vertical lines represent medians; dashed vertical lines represent modes.