PaCSWG8 Doc 03
Agenda Item 5.1

Agreement on the Conservation of Albatrosses and Petrels

Eighth Meeting of the Population and Conservation Status Working Group

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An update on the New Zealand large-scale monitoring and tracking programme with improved insights into trends and distribution

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SUMMARY

New Zealand hosts a large proportion of the world's Procellariiform seabirds. Given this responsibility, New Zealand has developed and maintains a large-scale monitoring and tracking programme with the aim to better understand population dynamics, distributions, and trends. We analysed 184 population counts across 1945-2024 and 1,151 tracks for 11 selected Procellariiform taxa to: I) estimate long-term population trends (in the form of annual growth rates r_t) for and across these taxa during several time periods using Bayesian GLMMs, II) review current IUCN Red List statuses of these taxa based on their estimated long-term population trends, and III) improve understanding and accessibility of the at-sea distribution of the selected taxa by generating single year-round distribution maps. Our results show that 73% of the focal taxa have exhibited concerning declines over the monitoring period available, with biannual and semi-biannual breeders showing the greatest rates of decline. Evaluating changes in growth rates for individual taxa and this Procellariiform community combined revealed that rates of declines have not improved (e.g., mean $r_{1990-2010}$= -1.07% (-1.56; -0.58), mean $r_{post-2010}$ = -0.84% (-1.29; -0.40)). Given these results, some species may warrant changes in their IUCN Red List assignments (e.g., Antipodean Albatross and Salvin's Albatross may warrant listing as CR, while Southern Royal Albatross may warrant listing as EN). The produced year-round distribution maps highlight that this New Zealand Procellariiform seabird community utilises virtually every ocean on Earth and that all taxa utilise the high seas extensively. In addition, the produced distribution maps improve accessibility for key target audiences, such as fisheries managers. The data underpinning our trend assessments are relatively simple and imperfect count data, and our results should be interpreted alongside other data streams (e.g., demographic data) where available. Yet, our work highlights the concerning state of the New Zealand Procellariiform community and the global responsibility for improving their currently declining trends.

RECOMMENDATIONS

We recommend that PaCSWG:

1. reviews the methodologies applied for assessing long-term population trends and developing simple distribution maps from complex and varied data sources.
2. considers how long-term population trend estimates, based on methods described here, may be useful to inform the reporting of key indicators of ACAP-species conservation status.
3. reviews the adequacy of current IUCN Red List Status for relevant ACAP species in light of the reported long-term population trends assessments.
4. considers how ACAP may wish to use single distribution maps, using methods based on those described here, to communicate more effectively with target audiences, such as fisheries managers.

Información actualizada sobre el programa de seguimiento y localización a gran escala de Nueva Zelanda con información mejorada sobre las tendencias y la distribución

RESUMEN

Nueva Zelanda alberga una gran proporción de las aves marinas procellariiformes del mundo. Dada esta responsabilidad, dicho Estado ha elaborado y mantiene un programa de seguimiento y localización a gran escala con el objetivo de comprender mejor la dinámica, la distribución y las tendencias de la población. Analizamos 184 recuentos poblacionales entre 1945 y 2024 y 1151 datos de seguimiento de 11 taxones procellariiformes seleccionados para I) estimar las tendencias poblacionales a largo plazo (en forma de tasas r_t de crecimiento anual) para todos estos taxones durante varios períodos de tiempo utilizando modelos lineales generalizados mixtos (GLMM) bayesianos, II) revisar el estado actual de la Lista Roja de la UICN de estos taxones en función de sus tendencias poblacionales estimadas a largo plazo y III) mejorar la comprensión y accesibilidad de la distribución en el mar de los taxones seleccionados mediante la generación de mapas de distribución únicos durante todo el año. Nuestros resultados muestran que el 73 % de los taxones focales han exhibido disminuciones preocupantes durante el período de seguimiento disponible, siendo los mejoradores semestrales y semisemestrales los que muestran las mayores tasas de disminución. La evaluación de los cambios en las tasas de crecimiento de los taxones individuales y de esta comunidad procallariiforme combinada reveló que las tasas de disminución no han mejorado (p. ej., media $r_{1990-2010}$= -1,07 % [-1,56; -0,58]; media $r_{post-2010}$ = -0,84 % [-1,29; -0,40]). Dados estos resultados, algunas especies pueden requerir cambios en su clasificación de la Lista Roja de la UICN (por ejemplo, el *Diomedea antipodensis* y el *Thalassarche salvini* pueden requerir la inclusión como En Peligro Crítico [CR], mientras que el *Diomedea epomophora* puede requerir la inclusión como En Peligro [EN]). Los mapas de distribución producidos durante todo el año destacan que esta comunidad de aves marinas procellariiformes de Nueva Zelanda utiliza prácticamente todos los océanos de la Tierra y que todos los taxones utilizan ampliamente la alta mar. Además, los mapas de distribución producidos mejoran la accesibilidad para los principales destinatarios, como los administradores de pesquerías. Los datos en los que se basan nuestras evaluaciones de tendencias son datos de recuento relativamente simples e imperfectos, y nuestros resultados deben interpretarse junto con otros flujos de datos (p. ej., datos demográficos) cuando estén disponibles. Sin embargo, nuestro trabajo pone de relieve el preocupante estado de la comunidad procellariiforme de Nueva Zelanda y la responsabilidad mundial de mejorar sus tendencias actuales a la baja.

RECOMENDACIONES

Recomendamos que el GdTPEC tome las siguientes medidas:

1. Revisar las metodologías aplicadas para evaluar las tendencias demográficas a largo plazo y desarrollar mapas de distribución simples a partir de fuentes de datos complejas y variadas.
2. Considerar cómo las estimaciones de las tendencias de la población a largo plazo, basadas en los métodos descritos aquí, pueden ser útiles para informar sobre los indicadores clave del estado de conservación de las especies amparadas por el ACAP.
3. Examinar la idoneidad de la situación actual de la Lista Roja de la UICN para las especies pertinentes del ACAP a la luz de las evaluaciones de las tendencias poblacionales a largo plazo informadas.
4. Considerar la forma en que el ACAP puede desear utilizar mapas de distribución únicos, utilizando métodos basados en los aquí descritos, para comunicarse de manera más eficaz con los destinatarios, como los administradores de pesquerías.

Mise à jour concernant le programme de surveillance et de suivi à grande échelle de la Nouvelle-Zélande avec une meilleure compréhension des tendances et de la répartition

RÉSUMÉ

La Nouvelle-Zélande abrite une grande proportion mondiale des oiseaux de mer de l’ordre Procellariiformes. Compte tenu de cette responsabilité, la Nouvelle-Zélande a élaboré, et poursuit à ce jour, un programme de surveillance et de suivi à grande échelle dans le but de mieux comprendre la dynamique, la répartition et les tendances des populations. Nous avons analysé 184 comptages de population entre 1945 et 2024, et 1 151 suivis de 11 taxons Procellariiformes sélectionnés, afin de : I) estimer les tendances démographiques à long terme (sous forme de taux de croissance annuels r_t) pour et entre ces taxons sur plusieurs périodes à l’aide de GLMM bayésiens ; II) réexaminer le statut actuel de ces taxons sur la Liste rouge de l’UICN en fonction de leurs tendances démographiques estimées à

long terme ; et III) améliorer la compréhension et l'accessibilité de la répartition en mer des taxons sélectionnés en générant des cartes de répartition uniques sur l'ensemble de l'année. Nos résultats montrent que 73 % des taxons focaux ont présenté un déclin inquiétant au cours de la période de suivi disponible, les reproducteurs semestriels et trimestriels affichant les taux de déclin les plus élevés. L'évaluation des changements dans les taux de croissance des taxons individuels, combinée à l'évolution cette communauté de Procellariiformes, a révélé que les taux de déclin ne se sont pas améliorés (par ex. moyenne $r_{1990-2010}$= -1,07 % (-1,56 ; -0,58), moyenne $r_{post-2010}$ = -0,84 % (-1,29 ; -0,40)). Compte tenu de ces résultats, certaines espèces peuvent justifier une modification de leur statut sur la Liste rouge de l'UICN (par exemple, les albatros *Diomedea antipodensis* et *Thalassarche salvini* pourraient être inscrits en tant que CR, tandis que l'albatros *Diomedea epomophora* peut justifier une inscription en tant qu'EN). Les cartes de répartition produites toute l'année soulignent que la communauté de Procellariiformes de Nouvelle-Zélande utilise pratiquement tous les océans de la Terre, tandis que les taxons dans leur ensemble utilisent largement la haute mer. De plus, les cartes de répartition produites améliorent l'accessibilité pour les principaux publics cibles, tels que les gestionnaires de pêcheries. Les données qui sous-tendent nos évaluations des tendances proviennent de comptage et sont relativement simples et imparfaites ; nos résultats doivent donc être interprétés en parallèle avec d'autres flux de données (par ex., données démographiques) lorsqu'ils sont disponibles. Notre travail met pourtant en évidence l'état préoccupant de la communauté de Procellariiformes néo-zélandaise ainsi que la responsabilité mondiale en matière d'amélioration de ces tendances actuellement déclinante.

RECOMMANDATIONS

Nous recommandons que le GTSPC :

1. Examine les méthodologies appliquées pour évaluer les tendances à long terme des populations et élaborer des cartes de répartition simples à partir de sources de données complexes et variées.
2. Examine de quelle manière les estimations des tendances démographiques à long terme, fondées sur les méthodes décrites ici, peuvent être utiles pour éclairer la production d'indicateurs clés de l'état de conservation des espèces inscrites à l'ACAP.
3. Examine l'adéquation du statut actuel de la Liste rouge de l'UICN pour les espèces pertinentes inscrites à l'ACAP, à la lumière des évaluations des tendances démographiques à long terme rapportées.
4. Examine comment l'ACAP pourrait faire usage des cartes de répartition uniques, en utilisant des méthodes basées sur celles décrites ici, pour communiquer plus efficacement avec les publics cibles, tels que les gestionnaires de pêcheries.

1. INTRODUCTION

New Zealand hosts a large proportion of the world's seabirds, and particularly a large proportion of the world's large Procellariiform seabirds (Croxall et al. 2012, Beal et al. 2021). Consequently, New Zealand holds a considerable global responsibility on the conservation of these species. Therefore, the New Zealand Department of Conservation (DOC) coordinates a large-scale monitoring and tracking programme with the aims to better understand population dynamics, distributions, and trends.

Here, we leverage both the New Zealand large-scale monitoring programme and the tracking programme to:

I. Estimate long-term population trends for and across selected taxa during several time periods,
II. Review current IUCN Red List statuses of selected taxa based on their estimated long-term population trends,
III. Improve understanding and accessibility of the at-sea distribution of selected taxa by generating single year-round distribution maps for key target audiences, such as fisheries managers.

2. METHODS

2.1. Selection of taxa

We selected Albatross (Diomedeidae) and Petrel (Procellariidae) taxa for inclusion in our analyses based on:

I. The presence of significant breeding populations in New Zealand (taxa endemic to New Zealand were considered high priority, while for taxa that also breed outside of New Zealand, only the New Zealand breeding population was considered),
II. Their known threat status (e.g., ACAP listing IUCN Red List Status and New Zealand Threat Classification System Status; ACAP 2024, IUCN 2024, Robertson et al. 2021),
III. Their vulnerability to longline bycatch (e.g., as illustrated by Edwards et al. 2023a,b),
IV. The availability and quality of year-round tracking data (based on the BirdLife International Seabird Tracking Database; www.seabirdtracking.org and unpublished datasets) and population count data at representative colonies (based on the ACAP database and existing reports published by the New Zealand Department of Conservation; DOC).

Following this approach, we selected 11 taxa, ten of which are ACAP-Annex-1-listed taxa, and one is an ACAP candidate taxon (Flesh-footed shearwater *Ardenna carneipes*), for our analyses. We treated Antipodean (*Diomedea antipodensis antipodensis*), Gibson's (*D. a. gibsoni*), and Southern Buller's albatross (*Thalassarche bulleri bulleri*) (Table 1) as separate taxa, in alignment with the New Zealand Threat Classification System (Robertson et al. 2021). Further details on selected taxa and the associated tracking and count data are presented in Table 1

2.2. Population count data

Following the selection of our focal taxa, we sourced population count data (in number of breeding pairs) from monitoring programmes at 11 colonies for these 11 taxa to assess population trends. We used count data, rather than other potentially more accurate and precise data sources, such as capture-recapture/resight data (e.g., Oppel et al. 2022, Richard et al. 2024a), as such data of adequate density are much sparser than count data and we aimed to assess both population and community-level trends and changes therein. Population count data were sourced from both the ACAP database as well as published DOC reports (and unpublished DOC reports for 2024). Count data were generated both through full island counts (e.g., Frost 2022) as well as through counts from representative study/index sites (e.g., Rexer-Huber et al. 2023, Walker et al. 2023) depending on the study system. For systems that consist of several representative study sites/index sites (e.g., Antipodean, Gibson's, Southern Royal Albatross *D. epomorphora* and Southern Buller's Albatross), only years in which all representative study sites were counted were included. Many population counts rely on extrapolations and correction factors to account for various confounding factors (e.g., Frost 2019) and consequently, we assumed that all reported counts were of the best possible quality and included the reported data at face value. As the reporting of uncertainty around population counts in the ACAP database and the original reports is inconsistent or absent (e.g., Bell et al. 2023, Rexer-Huber et al. 2023, Walker et al. 2023, Sagar et al. 2024), we could not include a justifiable, uniform estimation process for the uncertainty surrounding each count in our modelling process (see below), and thus we only sourced reported means for counts that included uncertainty reporting. For taxa breeding over more than one calendar year (i.e., virtually all), we report year as the year in which chicks fledge, as per ACAP convention. In one case (White-capped Albatross *T. steadi*), two conflicting sources of count data exist (based on different correction factors applied to aerial counts; Walker et al. 2020, Baker et al. 2023) and we chose here to select the more precautionary analysis (i.e., Walker et al. 2020). After compilation, our dataset consisted of 184 population counts for the 11 selected taxa (Table 1, Fig. 1).

2.3. Estimating long-term population trends and reviewing IUCN statuses

To estimate the population trends of each taxon over time, we followed the approach of Paxton et al. (2016) and Fischer et al. (2020) and fitted a Bayesian generalised linear mixed-effects models (GLMM) with a Poisson error to the population count data available. Specifically, we fitted the following model to the population count data:

1) $Log(N_t) = \alpha + r_t + \varepsilon_t$,

in which N_t refers to the population count (in breeding pairs) in year t, α is $log(N_0)$ (the population count at year 0), r is the annual population growth rate (i.e., $log(\lambda)$, in which λ is the finite rate of increase), t is the number of years between 0 and t, and ε_t is a random annual variation (Fischer et al. 2020). We thus assumed that population trajectories would follow exponential growth (positive r_t) or decline (negative r_t) trajectories. We fitted this equation to the data per taxa, while specifying different time periods, to evaluate growth rates during specific periods (Fig. 1):

I. The total monitoring period available (r_{total}),
II. The monitoring period prior to 1990 (if available; $r_{pre-1990}$),
III. The 1990-2010 monitoring period (if available; $r_{1990-2010}$),
IV. The 2010-present monitoring period (if available; $r_{post-2010}$), and
V. The period following the highest population count available (referred to as N_{max}) ($r_{N_{max}}$).

We only calculated growth rates if more than two counts were available in any given period. However, we allowed for some flexibility in the definitions of time periods II, III, and IV to maximize information gained from the available data. Specifically, if only two data points were available for a given period, but a datapoint existed one year preceding or following the period under assessment, we extended the time period (i.e., we allowed some flexibility for Grey-headed Albatross *T chrysostoma*, Campbell Albatross *T. impavida*, and Flesh-footed Shearwaters for the post-2010 time period; Fig. 1). In addition to our GLMMs, for visualisation purposes, we also generated plots of standardised population sizes over time in which we set N_{max} to 1 and scaled all other counts accordingly. Finally, to gain insights into community-level population trends, we calculated the community-level means for r_t for the same time periods as for the taxon-specific estimates, apart from V, as this time period depended on a taxon-specific N_{max}.

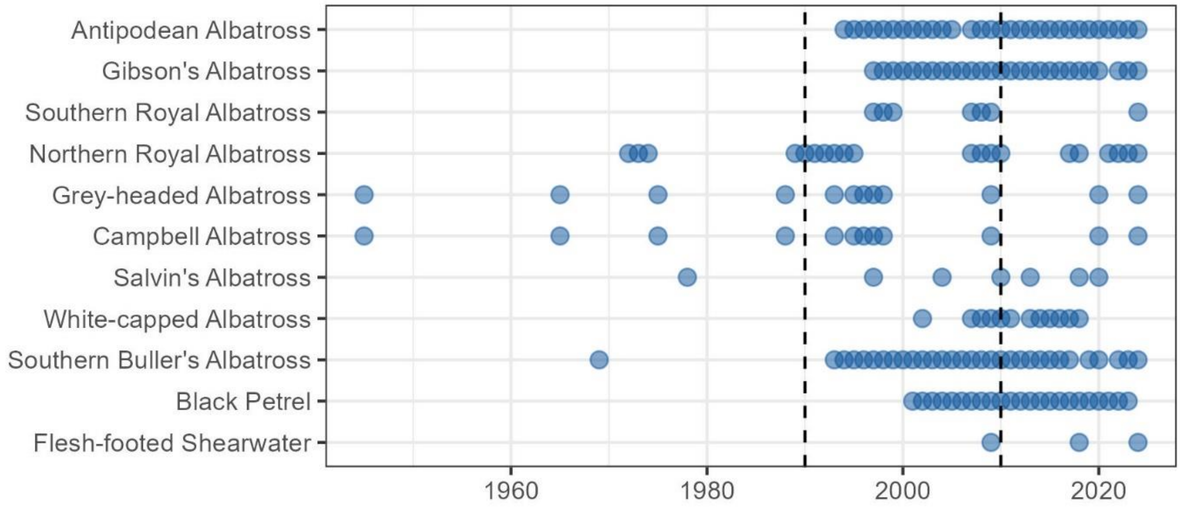

Fig. 1. Count periods for the eleven taxa included in our analyses. Each symbol relates to a population count. Dotted lines indicate the different time periods for population growth analyses.

We reviewed current IUCN Red List statuses for our focal taxa using our population growth rate estimates and IUCN Red List Criterion A (reduction in population size; IUCN 2012). As generation length is the primary time determinant for Red List assessments under Criterion A, we sourced generation lengths for all selected taxa from Bird et al. (2020). We then assessed the estimated population declines as following Shaw et al (2024) using:

2. $trend_{IUCN,s} = -(1-(1+r_{s,3G})^{3G}$

in which $trend_{IUCN,s}$ refers to the trend assessment for Criterion A for species s, $r_{s,3G}$ refers to the estimated population growth rate over the course of three generations (*3G*). For almost all taxa, the monitoring period did not exceed three generations, so we treated r_{3G} as r_{total}. However, for Grey-headed and Campbell Albatrosses, the monitoring period exceeded three generations (68 and 65 years respectively) and thus we separately estimated r_{3G} for these two taxa. As the IUCN Red List taxonomy does not treat Gibson's and Antipodean Albatross as two separate taxa, we estimated r_{3G} for the two taxa jointly. We then evaluated how the $trend_{IUCN,s}$ estimate fitted the different IUCN Red List Categories as per Criterion A. If the trend equalled a decline of ≥80% over three generations, the species status arising from our review would be considered Critically Endangered (CR), based on A234. Similarly, if the trend equalled a decline of ≥50% or ≥30%, the species status would be considered Endangered (EN) or Vulnerable (VU), respectively. As a precautionary approach, we considered the median of the estimate as the cut-off for the category evaluation. While for most of our taxa, the monitored population represents either the sole population or the vast majority of the population (i.e., >90%) of the taxon under consideration, this was not the case for Grey-headed Albatross (which breeds circumpolar), Southern Buller's Albatross (which has a second large population on Solander), and Flesh-footed Shearwater (which has multiple colonies across New Zealand and Australia). Consequently, for these taxa, our evaluation of current IUCN criteria assignments should only be considered at a population level.

We fitted our GLMMs in the Bayesian modelling programme OpenBUGS 3.2.3 (Spiegelhalter et al. 2014), which employs Markov Chain Monte Carlo (MCMC) algorithms to generate posterior distributions for parameters, while simultaneously allowing for uncertainty to be adequately propagated in those distributions. We used uninformative priors for α and r (N[0,0.01]) (Fischer et al. 2020). To obtain posterior distributions for our parameters of interest, we pooled two MCMC chains of 50,000 after a burn-in of 25,000 iterations. We evaluated trace plots and the Gelman-Rubin statistic (R<1.05) to confirm model convergence. For easy interpretation, we transformed all r_t estimates into percentages and report estimates as medians with 95% credible intervals, unless otherwise stated.

2.4. Tracking data and processing thereof

To update the available information and improve the accessibility of the at-sea distributions of our 11 focal taxa, we collated 1,151 tracks from 16 different colonies of the focal taxa from either datasets published on the BirdLife International Seabird Tracking Database or unpublished datasets currently held by DOC. Specifically, we sourced 48 datasets from the Seabird Tracking Database (reference numbers 427, 429, 430, 469, 470, 471, 472, 474, 476, 478, 479, 532, 533, 556, 618, 619, 620, 621, 624, 631, 632, 636, 640, 648, 658, 659, 666, 669, 949, 951, 999, 1082, 1257, 1258, 1259, 1324, 1325, 2057, 2058, 2069, 2070, 2072, 2074, 2075, 2077, 2081, 2172, 2173) and compiled additional unpublished data (e.g., because tracking is still ongoing) for Gibson’s Albatross, Southern Royal Albatross, Black Petrel, and Southern Buller’s Albatross. We did not differentiate between breeding vs. non-breeding stages or adult vs. juvenile stages, as we wanted to describe distributions as comprehensively as possible for each taxon (e.g., datasets of 4/11 taxa contained juvenile tracking data).

The compiled location data originated from both GPS/PTT tracking as well as GLS tracking, which are characterised by fundamental differences. Specifically, GPS and PTT tracking devices have high accuracy (~30-170 m for GPS devices, ~50-20,000 m for PTT devices; Hazel 2009, Irvine et al. 2020) at a variable temporal resolution (1-40 locations/day), but rarely provide year-round tracking data as devices either fall off when birds moult their feathers (almost all GPS/PTT devices are attached using feather mounts), their battery runs out, or their memory space is depleted. On the other hand, GLS tracking devices have low accuracy (~145 km; Merkel et al. 2016) at a regular temporal resolution (2 locations/day) but also have low weights, battery and memory requirements, and consequently GLS devices can be deployed on birds’ leg bands, providing year-round tracking data, even across years. Our dataset contained 695 (60%) GPS/PTT tracking tracks and 456 40% GLS tracks (Table 1) and each taxon’s dataset contained at least several tracks covering the full annual cycle (i.e., each dataset contained location data for each month of the year).

The fundamental differences between GPS/PTT and GLS data listed above also translate in key differences in terms of data processing. For unpublished, and thus unprocessed, GPS/PTT data, we followed Rowley et al. (2024) and discarded PTT-derived locations with an Argos quality of A, B and Z and locations with an Argos-generated error ellipse variable of >10km error radius (Bose & Debski 2021). We then subjected both published and unpublished GPS/PTT data to a visual inspection and a speed filter of >25 m/s to remove any erroneous positions, matching previous work (e.g., Carneiro et al. 2020). We then linearly interpolated all GPS/PTT data (i.e. re-discretization every 12 hours) to match GLS sampling frequency and obtain regular locations. For unpublished GLS datasets, we processed and cleaned data using an iterative probabilistic algorithm which includes several speed filters and spatial masks, which we fitted through *probGLS* (Merkel et al. 2016). For published GLS data, we removed locations around the equinoxes (March equinox: −21, +7 days; September equinox: −7, +21 days) prior to analysis where necessary. We subjected all GLS data to the same visual inspection and speed filter of >25 m/s as the GPS/PTT data. Consequently, we obtained cleaned data from various tag types, ready for integration into year-round distribution maps.

2.5. Generating year-round distribution maps

To improve the accessibility of the at-sea distribution of the focal taxa, we generated single year-round distribution maps that accounted for I) different tracking data types, sample sizes, and the error associated with them, II) different colony sizes and therefore different representativeness of tracking data at a taxon level, and III) changes in sample size across the annual cycle (e.g., due to ceased data transmission). Specifically, we estimated utilization distributions (UDs) using kernel analyses in the adehabitatHR package (Calenge 2006) for each data group, which consisted of unique combinations of taxon, colony (e.g., island group), device type and calendar month. We used a fixed smoothing parameter (*h*) of 50 km for GPS/PTT data and 200 km for GLS data (Carneiro et al. 2020). To control for differences in the number of trips per device type, we combined UDs based on GPS/PTT data and UDs based on GLS data by weighting each by the proportion of individuals represented. When data were available from several colonies of a single taxon, we combined UDs for each colony into a single UD based on the percentage of the total population involved based on data sourced from ACAP (2024). We then summed monthly UDs to generate balanced annual distributions per taxon. From these taxon-level year-round UDs, we calculated the 50, 75, 95 and 99% isopleths to categorise different levels of intensity in use. We delineated core areas as those areas enclosed by the lowest percent of these isopleths.

As a final step, we merged all 11 year-round taxon-level UDs into a single year-round UD and calculated the same isopleths to present the combined distribution of this New Zealand Procellariiform community and further improve the accessibility of these data.

3. RESULTS

3.1. Long-term population trends

Our analyses have shown that 8/11 (73%) of the focal taxa have declined at least at some point over the monitoring period available (Fig. 2). Biannually and semi-biennially breeding species (Antipodean Albatross, Gibson’s Albatross, Southern Royal Albatross, Northern Royal Albatross *D sanfordi*, Grey-headed Albatross, and White-capped Albatross), in addition to the annually breeding Salvin’s Albatross (*T. salvini*), showed the greatest rates of decline over the entire monitoring period (Fig. 3A, Table 2). In contrast, Southern Buller’s Albatross, Black Petrel (*P. parkinsoni*), and Flesh-footed Shearwater showed increasing population trends over the monitoring period available.

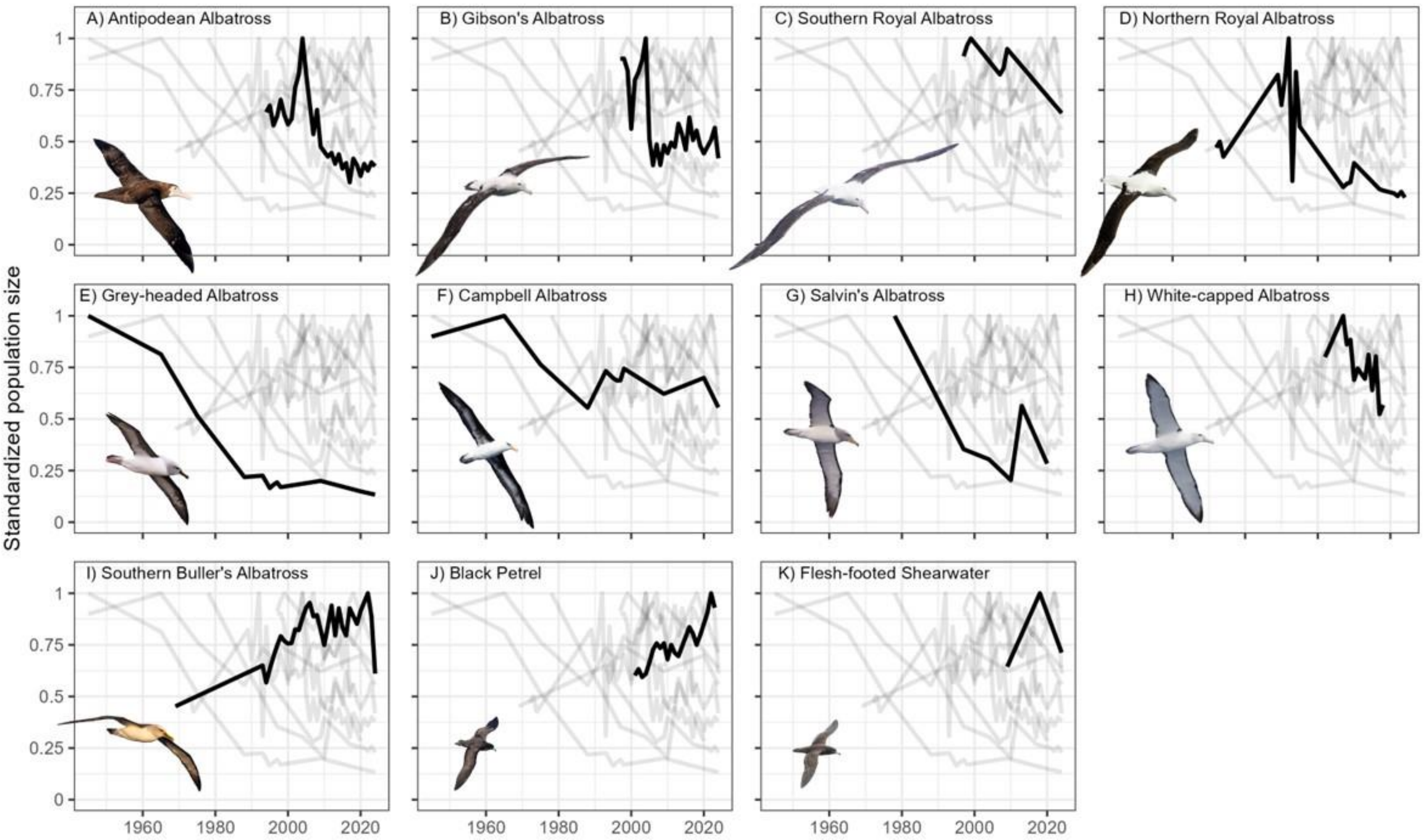

Fig. 2. Standardized population trajectories for the entire monitoring period available for the 11 focal taxa. Black lines represent the trajectory of the focal taxon per panel, translucent lines represent other taxa

When evaluating taxon-specific trends over the predetermined time periods, specific patterns arise, but for most taxa, annual growth rates did not improve (Fig. 3B, Table 2). Growth rates for Northern Royal Albatross deteriorated after 1990s, while for Grey-headed Albatross and Campbell Albatross, growth rates improved temporarily between 1990 and 2010, after which they deteriorated again. Reduction in growth rates post-2010 was also evident in the Antipodean Albatross, White-capped Albatross, and Southern Buller's Albatross. Contrastingly, Gibson's Albatross and Salvin's Albatross population growth rates improved post-2010.

Across taxa, this large Procellariform community has been declining by 1.22% (-1.39; -1.05) per annum for the monitoring period available. When decomposing this trajectory into the three major time-periods under consideration, our results show that this decline has not slowed down. Specifically, the estimated community level annual rate of decline was -0.33% (-0.42; -0.25) pre-1990 (but note that sample size for this earlier period was limited), while the annual rate of decline for 1990-2010 was -1.07% (-1.56; -0.58) and the annual rate of decline post-2010 was -0.84% (-1.29; -0.40). The overlapping CIs for the latter two time periods indicate uncertainty in any improvements of the community-level trend.

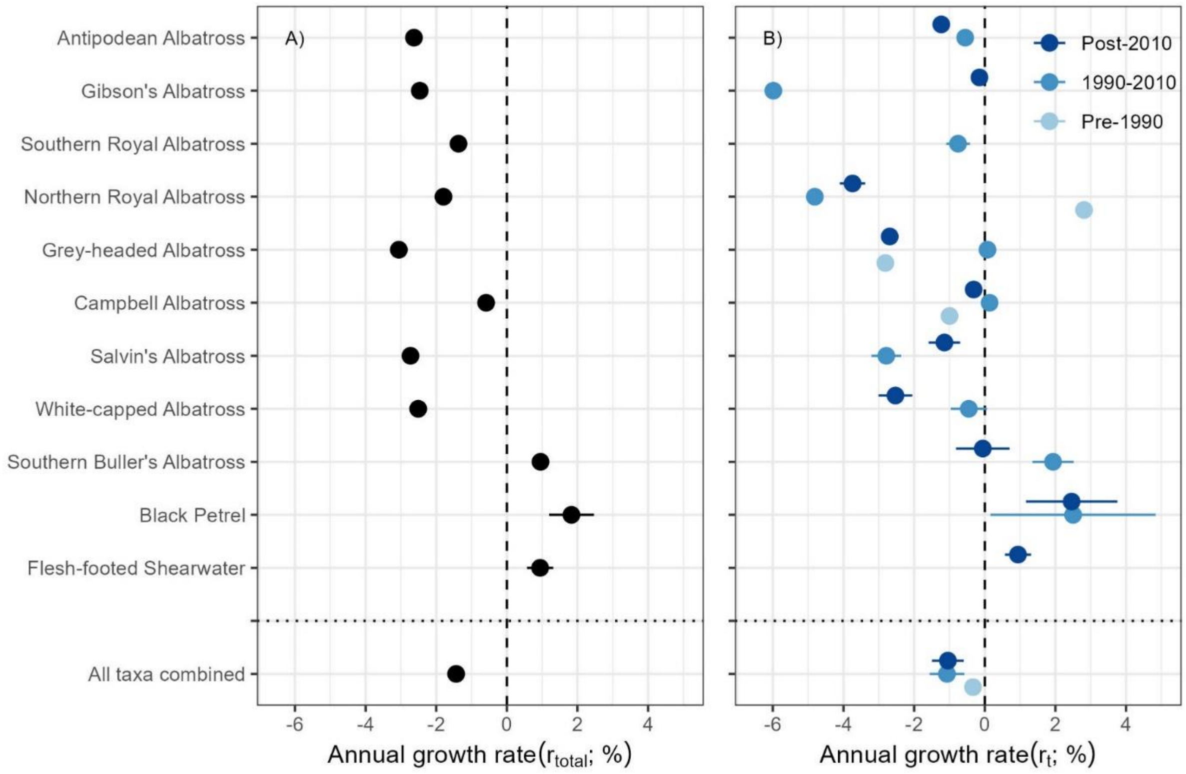

Fig. 3. Annual growth rate estimates (in %) for the entire monitoring period available (A) and for specific time periods (B).

3.2. Review of current IUCN statuses

Using population growth rate estimates over three generations and generation lengths revealed mismatches between current IUCN statuses and the IUCN status qualifications under Category A (Fig. 4). Specifically, the estimated decline over three generations indicated that Antipodean (including Gibson's Albatross) and Salvin's Albatross may warrant listing as CR, and Southern Royal and White-capped Albatross may warrant listing as EN. In addition, the New Zealand population of Grey-headed Albatross may warrant listing as CR. Conversely, the estimated growth over three generations of Flesh-footed Shearwater (the Lady Alice Island population), Southern Buller's Albatross (the Snares population) did not match current IUCN statuses either.

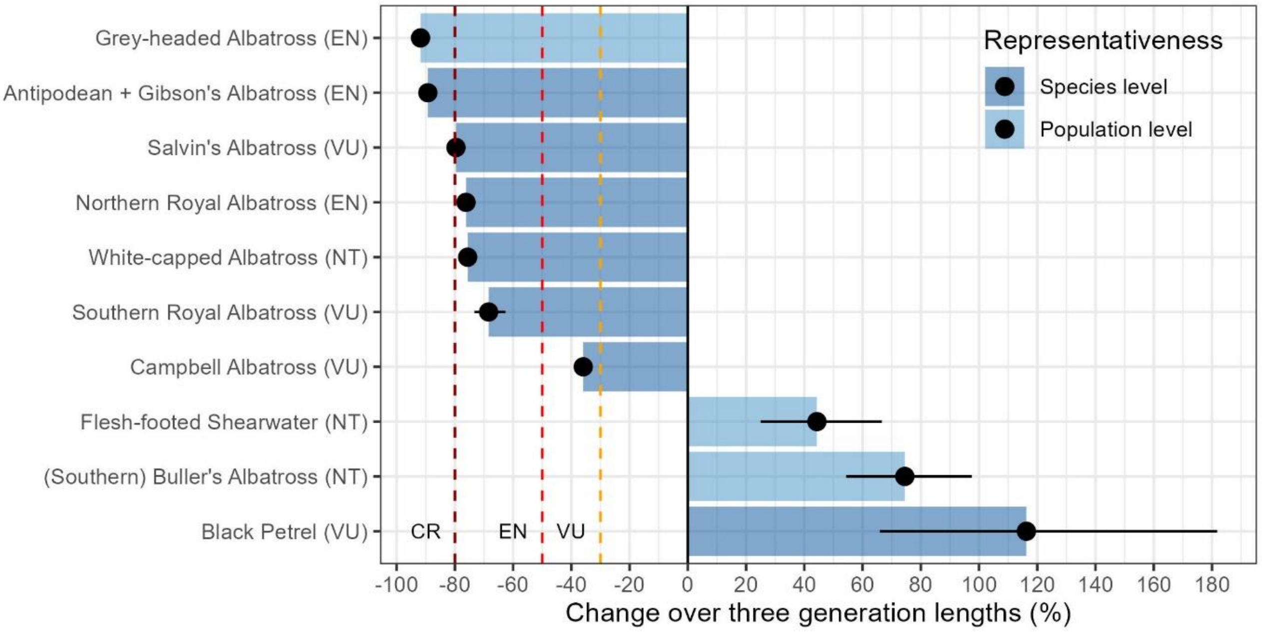

Fig. 4. Estimated population change over three generations in relation to IUCN Category A classifications. Current IUCN status per species is given in brackets.

3.3. Year-round distributions

Year-round distribution maps for each of the 11 focal taxa can be found in Figures 5-15 and distributions per taxon are described here in brief.

Antipodean Albatross utilised waters from the Great Australian Bight to Chile, with core areas located in the Tasman Sea, the New Zealand subantarctic extending north up to the east and northeast of New Zealand, and west of Chile. Gibson's Albatross exhibited a more confined distribution, ranging from Western Australia to east of New Zealand, with core areas encompassing the Tasman Sea, the New Zealand Subantarctic, and the Chatham Rise. Southern Royal Albatross exhibited a circumpolar distribution across the Southern Ocean with core areas located in the New Zealand Subantarctic extending up to the east of the South Island of New Zealand as well as the Patagonian Shelf off Argentina. Northern Royal Albatross exhibited a similar circumpolar distribution, but core areas of use encompassed the Chatham Rise, waters to the east of the North Island of New Zealand, waters west of Chile, as well as the Patagonian Shelf.

Grey-headed Albatross also exhibited circumpolar distributions in the Southern Ocean, with core areas of use located in the New Zealand Subantarctic, extending towards Polar waters south of New Zealand, east of the Chatham Rise, and west of Southern Chile. Campbell Albatross largely utilized waters from the eastern Indian Ocean to the eastern Pacific west of Chile, but some individuals exhibited circumpolar movements as well. Core areas of use were located in the Great Australian Bight, the Northern Tasman Sea, and the New Zealand Subantarctic, extending into the Polar waters south of New Zealand. Salvin's Albatross distribution extended from the Tasman Sea east to waters off South America, with core areas of use including the New Zealand Subantarctic, the Southern Tasman, The Chatham Rise, waters of Chile, and the Northern Humboldt Current System off Peru. White-capped Albatross ranged from waters west of Namibia and South Africa to the New Zealand Subantarctic, with core areas being located in the Alguhas Upwelling System, off eastern Australia and Tasmania, including the southern Tasman Sea, and the waters around New Zealand including the New Zealand Subantarctic. Southern Buller's Albatross ranged from west of Tasmania to the waters west of South America, with core areas of use encompassing the waters surrounding Tasmania, the Southern Tasman Sea, the waters surrounding the New Zealand Subantarctic and South Island as well as the waters off Chile and Southern Peru.

Black Petrels ranged from eastern Australia to waters west of Costa Rica, Colombia, Ecuador and Peru, with core areas being located in the northern Tasman Sea, the waters north and north-east of New Zealand, the Galapagos, and waters off Colombia, Ecuador, and northern Peru. Flesh-footed Shearwaters ranged from northern New Zealand across the Tropics up until waters east off Japan and Russia, with core areas of use encompassing the northern Tasman Sea, the waters surrounding the North Island, the Chatham Rise and Louisville Ridge, various key areas in the tropics and waters east off Hokkaido.

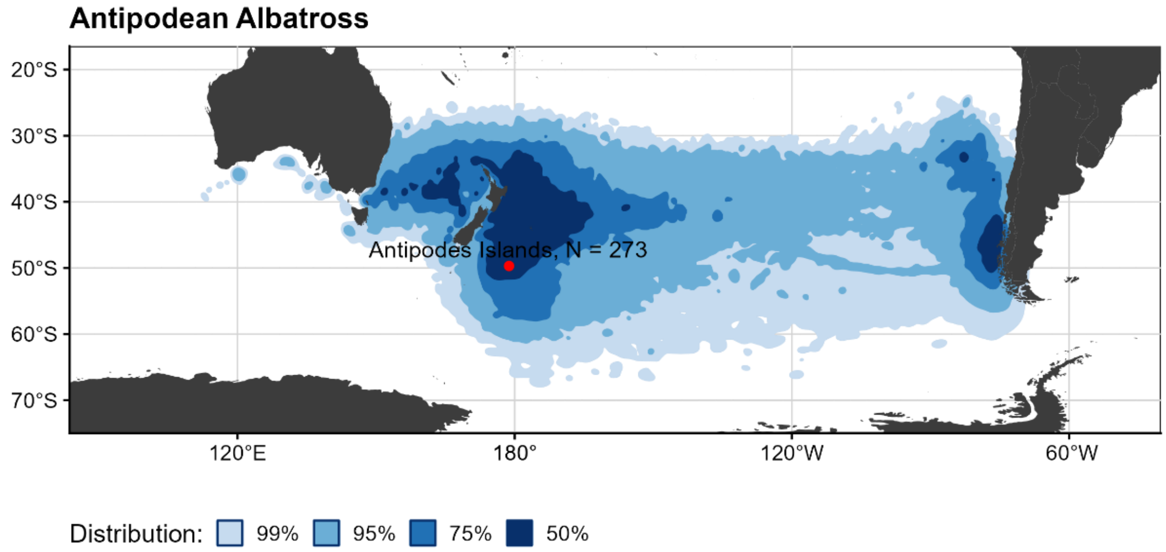

Fig. 5. Year-round distribution of Antipodean Albatross.

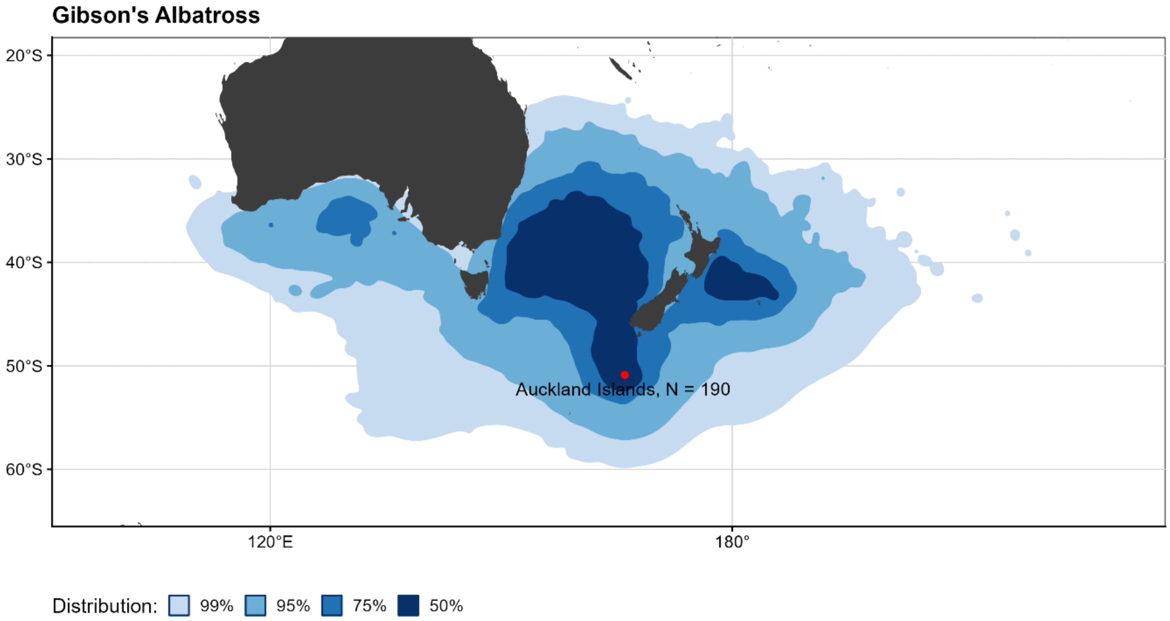

Fig. 6. Year-round distribution of Gibson's Albatross.

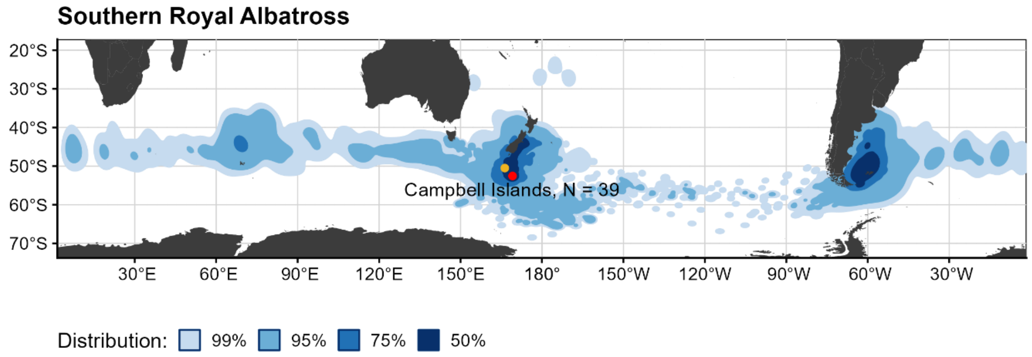

Fig. 7. Year-round distribution of Southern Royal Albatross. Yellow symbol represents the untracked Enderby Island colony (<1% of the total population).

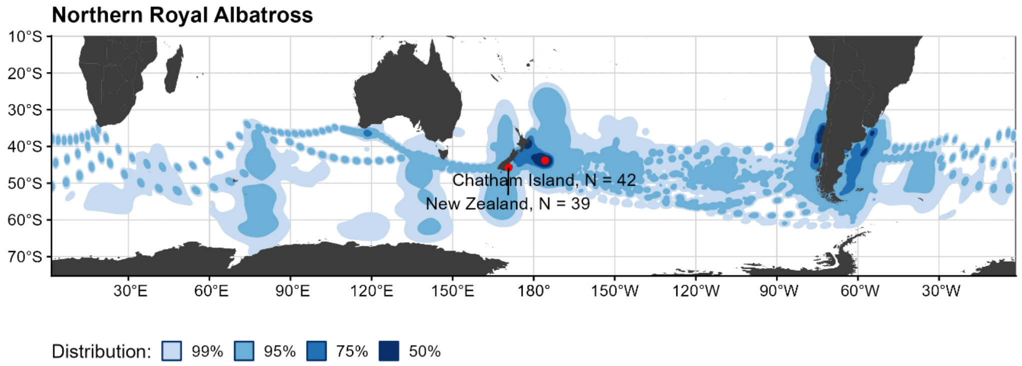

Fig. 8. Year-round distribution of Northern Royal Albatross.

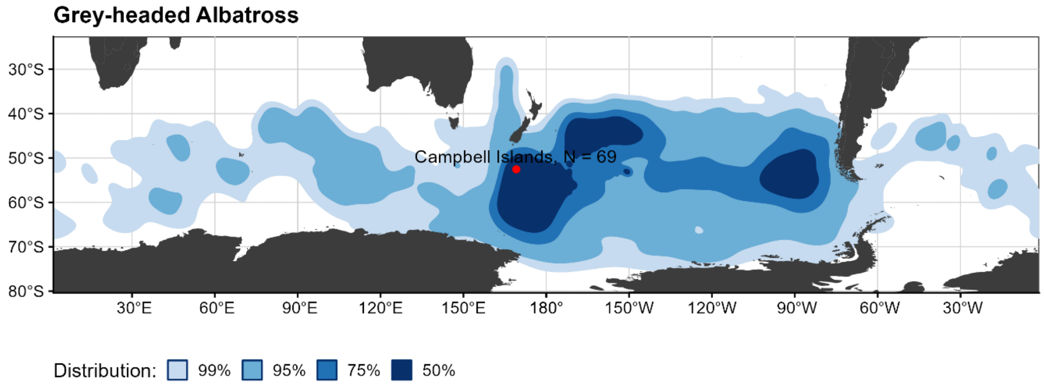

Fig. 9. Year-round distribution of Grey-headed Albatross.

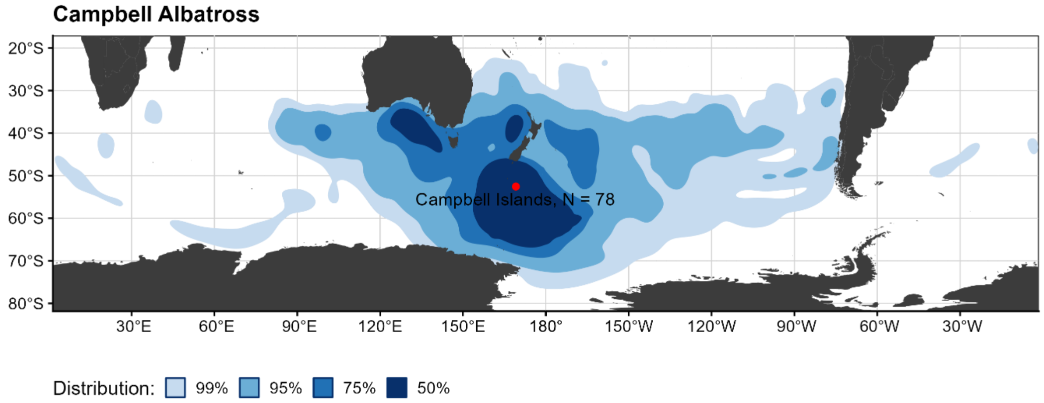

Fig. 10. Year-round distribution of Campbell Albatross.

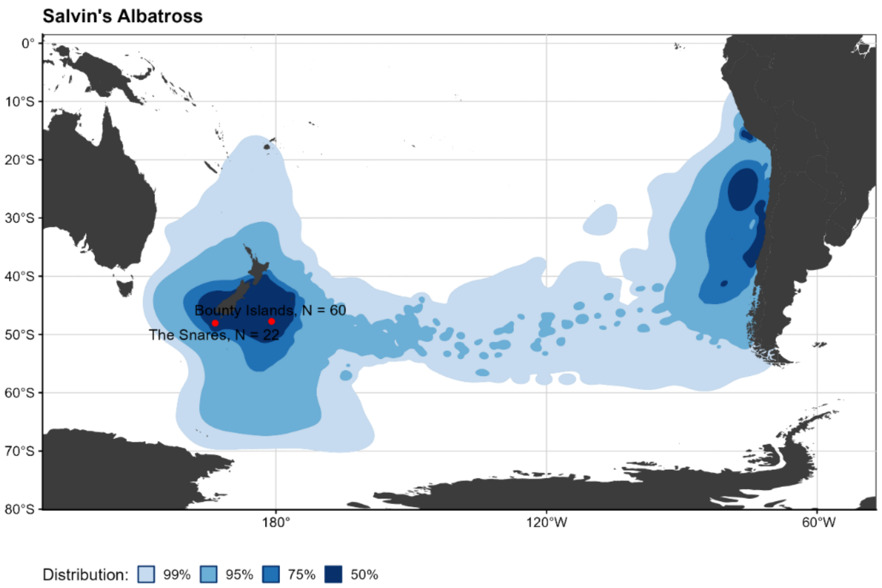

Fig. 11. Year-round distribution of Salvin's Albatross.

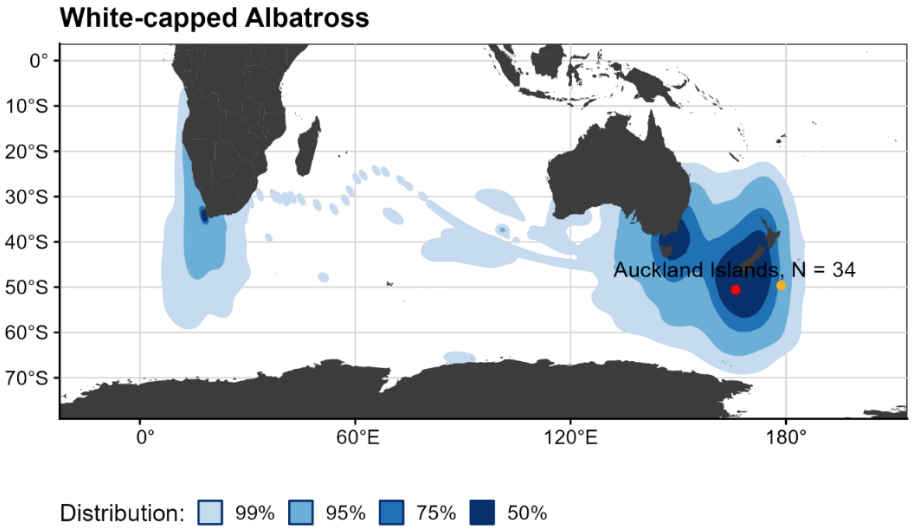

Fig. 12. Year-round distribution of White-capped Albatross. Yellow symbol represents the untracked Bollon's Island colony (<1% of the total population).

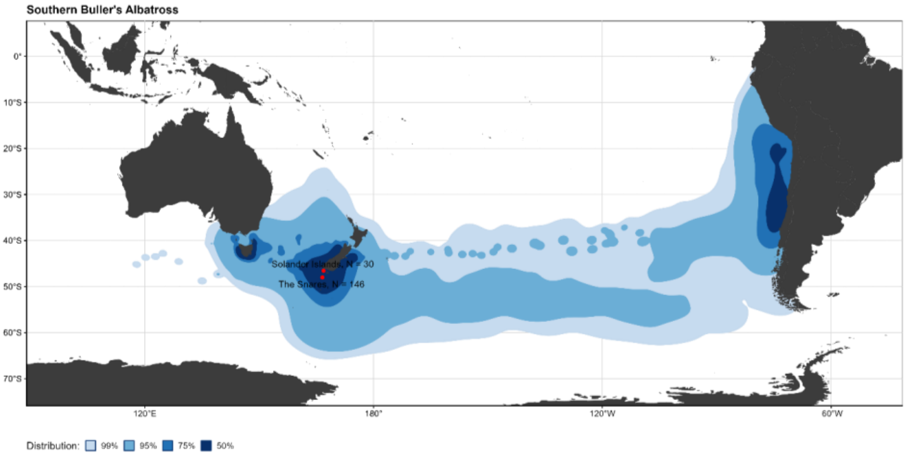

Fig. 13. Year-round distribution of Southern Buller's Albatross.

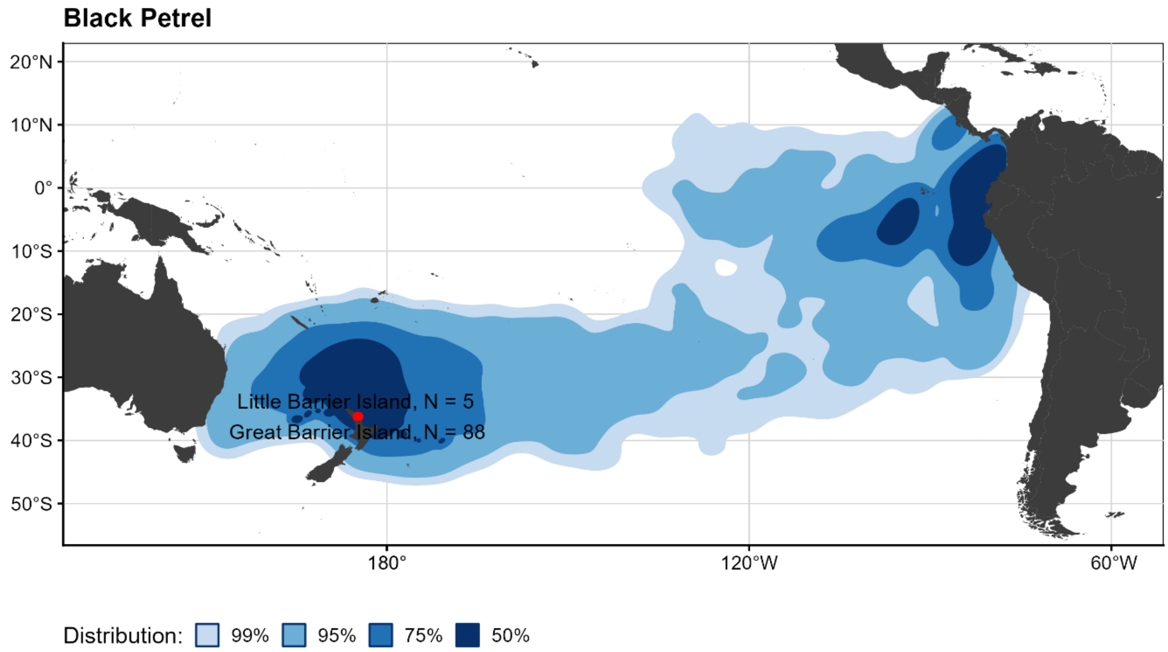

Fig. 14. Year-round distribution of Black Petrel.

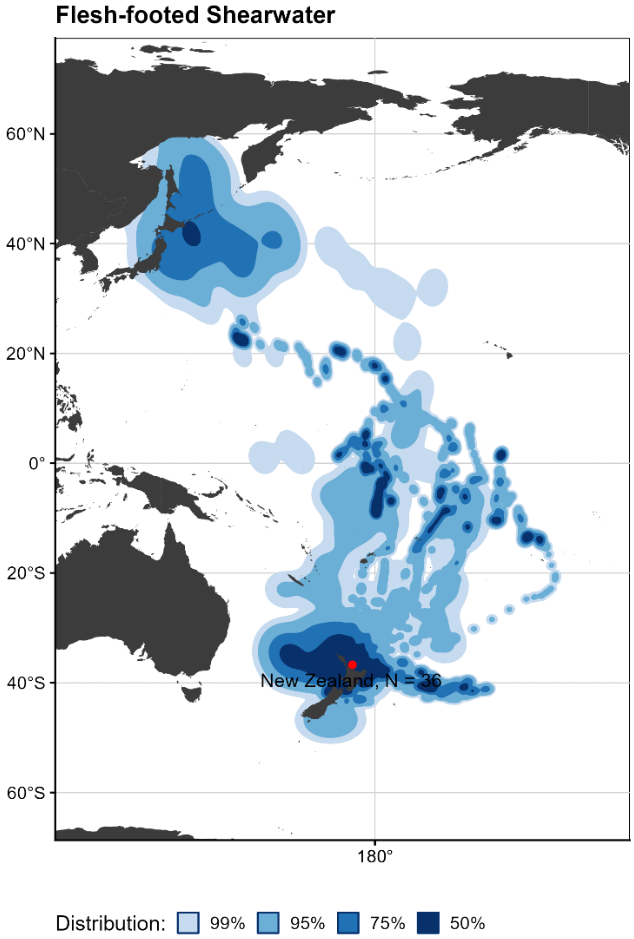

Fig. 15. Year-round distribution of Flesh-footed Shearwater tracked from New Zealand (Lady Alice Island and Ohinau Island, jointly encompassing ~5% of the total population).

Combined, the year-round distribution maps highlighted the wide distribution of this New Zealand Procellariiform community and its connection to the rest of the world (Fig. 16). Most of the oceans on Earth were utilized at least to some extent by some taxa. Core areas of use highlighted the importance of waters around New Zealand, including the Subantarctic and Polar waters, the Tasman Sea, and the Chatham Rise, but also of waters east of Australia and Tasmania, waters west of Chile and along the Humboldt Current System, as well as the Patagonian Shelf, and off Australia.

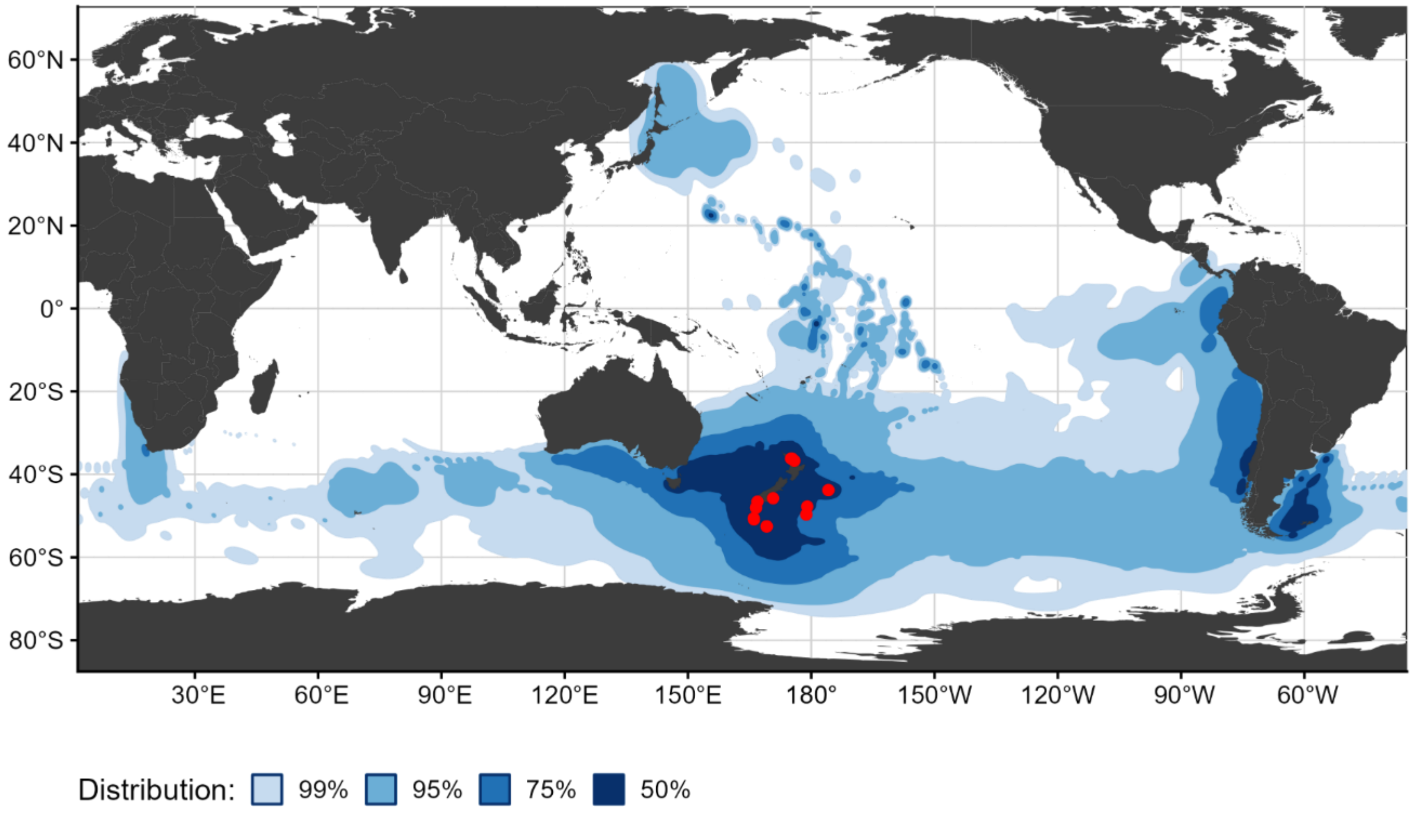

Fig 16. Distribution of all 11 focal taxa combined based on 1,151 tracks. Colonies where birds were tracked from are indicated in red.

4. DISCUSSION

Our study presented updated estimates of long-term trends based on count data, and easily interpretable year-round distributions based on a variety of tracking data, both on a taxon-level for the 11 focal taxa, as well as for the New Zealand Procellariiform community that these taxa jointly form. Our analyses highlight that the majority of the selected taxa have suffered long-term and severe declines and that these declines have not been reduced so far. Consequently, the IUCN status of some taxa may warrant re-evaluation based on the information presented here. Furthermore, the year-round distribution maps highlighted that these taxa connect New Zealand with virtually all of Earth's oceans, and therefore underscore the international responsibility to implement change in order to reverse the trends presented here.

4.1 Long-term population trends

The estimated population declines of these New Zealand Procellariiform taxa over long time periods is of great concern. Our approach utilised log-linear models similar to those in TRIM (Pannekoek & van Strien 2005; which is the ACAP recommended approach to trend analyses) and mirrors previous work by Paxton et al. (2016) and Shaw et al. (2024) on other taxa (Hawaiian songbirds and African raptors, respectively), which in both cases illustrated similarly concerning population collapses. As such, we consider the methods applied appropriate for the data available. However, it should be noted that count data is imperfect, and thus our results should be interpreted with this consideration in mind. While we endeavoured to generate uncertainty as appropriately as possible in our trend estimates, we did not specifically generate uncertainty for the count data, largely because of the inconsistent, or absent reporting of uncertainty surrounding these data. This is particularly the case for long-term studies employing study/index sites (Bell et al. 2023, Rexer-Huber et al. 2023, Walker et al. 2023, Sagar et al. 2024). Consequently, the uncertainty surrounding some of our trend estimates is unrealistically small (e.g., for Southern Royal and Salvin's Albatross). Future extensions of the work presented here could include the generation of additional uncertainty surrounding individual count estimates using the ACAP reliability and accuracy categories and informative priors within a hierarchical modelling framework (e.g., Fischer et al. 2020). The specification of such priors, however, would remain a subjective exercise. Despite these shortcomings, our analyses cover some of the most extensive seabird monitoring time periods on Earth (up to 80 years) and the best available count data. Additionally, for several taxa, these trends are supported by trends in demographic rates. For example, adult survival, breeding probability, and breeding success has been reduced in both Antipodean and Gibson's Albatross for several decades (Rexer-Huber et al. 2023, Walker et al. 2023). Consequently, the estimated population declines appear an adequate reflection of reality.

The estimated population trends are subject to the time period under consideration, a common challenge in trend analyses (Pannekoek & van Strien 2005). We attempted to overcome this challenge by not only analysing trends over one, but six different time periods. This exercise highlighted that there was no "one size fits all" period that best described the trend of a taxon. For instance, there are stark differences between the $r_{N_{max}}$ estimates and the $r_{post-2010}$ for Antipodean and Gibson's Albatross, most likely due to the different severities of the sudden declines that both taxa suffered around 2007 (Rexer-Huber et al. 2023, Walker et al. 2023). Consequently, for Gibson's Albatross, the N_{max} was treated more as an outlier, but not for Antipodean Albatross, resulting in an underestimate of the decline in the former, and a more adequate estimate in the latter. As all taxa are extremely long-lived *k*-selected species, the best way forward may thus be to analyse the full monitoring period available (or at least trends across three generations, see below), while analysing sections of adequate length (e.g., 20 years, roughly corresponding to a generation time for these species) to detect changes in trends over time. This approach highlighted the concerning, ongoing, and apparently unabated declines that these taxa are suffering.

4.2. Review of current IUCN statuses

Our trend analyses indicate that some species may warrant uplisting on the IUCN Red List, but ideally, these results should be interpreted alongside other data (e.g., capture-recapture/resight data where available), yet in the absence of additional data, changes to Red List status may be appropriate.

Antipodean Albatross (including Gibson's Albatross) may warrant listing as CR under Criterion A, as our analyses show a decline of >90% across three generations, which has not ceased for the species as a whole. Richard et al. (2024a) provide a more detailed analysis in analysis in the shape of an integrated population model for the Antipodes Island population, which projected a future decline of 6% per annum and extinction within three generations and highlighted that the most likely cause of decline, (bycatch in international fisheries) has not been abated, and thus CR appears adequate.

Salvin's Albatross and Southern Royal Albatross may warrant listing as CR and EN, respectively, as our analyses show a decline of 80% and 68% across three generations respectively. Unlike for the Antipodean Albatross, no additional demographic data at an adequate density exists to further evaluate our results. However, short-term visits to Campbell Island in 2020 and 2023 had already indicated a surprising and concerning population decline in the Southern Royal Albatross population (Mischler 2020, Mischler & Wickes 2023, Mischler et al. 2024) and thus, in the absence of other supporting data, EN may be an adequate assessment. Additional data for Salvin's Albatross is even sparser and as such, listing as CR may be the appropriate precautionary approach given our trend estimates.

White-capped Albatross, which may warrant listing as EN based on an estimated decline of 76% over three generations, conversely has multiple, contradicting, sources of information exist. Baker et al. (2023) used TRIM to assess the population trends of this species and estimated a non-significant decline of ~1%. However, this assessment utilised different correction factors fitted to count data compared to Walker et al. (2020), which we used here, and which may present the more precautionary approach. In addition, some demographic data for White-capped Albatross exists, indicating variable, but regularly concerningly low adult survival (Parker et al. 2024). In the light of these various other sources of information, VU may be an adequate IUCN category for this species.

The New Zealand population of Grey-headed Albatross suffered the most detrimental decline of all taxa analysed (~92% decline across three generations) and at least one of the Atlantic populations (Pardo et al. 2017) has shown similarly concerning declines, including declines in vital rates. However, until further insights from the Indian Ocean populations is available, CR may be hard to justify.

The Snares population of Southern Buller's Albatross and Black Petrel, in contract to the previous species, show increasing trend estimates over three generations. These estimates, however, conflict with other data streams. In the case of Southern Buller's Albatross, adult survival of the Snares population has been declining for several years and is regularly estimated <0.90 (Sagar et al. 2024). It has been shown that in Albatross populations, declines in demographic rates may only become evident in count data following a delay, as the proportion of pre-breeders and non-breeders is gradually depleted (e.g., see Oppel et al. 2022). Consequently, the very recent, and severe decline in this population (see Fig 2I) may be a manifestation of reduced adult survival in previous years and as such, any IUCN category changes, certainly towards LC, for this species would be premature. Similarly, while the Black Petrel population has been consistently increasing at the study site, recruitment (and thus juvenile survival) has been exceedingly low (Bell et al. 2023) and consequently, the increase observed may be a consequence of immigration into the study population. Therefore, any IUCN category changes may be premature for this species as well.

In summary, for some taxa, the trend estimates presented here are supported by additional data and thus should instil faith in the estimates and the adequacy of any warranted IUCN status changes, while for others are in conflict with additional data and any changes to statuses would thus be premature. Ideally, all taxa are analysed using integrated population models, fusing various data streams, such as count and capture-recapture/resight data (e.g., Oppel et al. 2022, Fischer et al. 2023, Richard et al. 2024a), but given the remoteness of many of the colonies of these taxa, collecting adequate data for such models for all taxa is unlikely. In scenarios where only count data are available, and trends based on these indicate that IUCN status changes are warranted, we believe that the precautionary approach would be to apply those changes indeed.

4.3. Year-round distributions

The distribution maps we generated provide simple and accessible overviews of the annual ranges of these taxa. Tracking data is inherently complex and is subject to spatiotemporal and technical biases and specifications (e.g., different tag types with different error radii and tracking durations; Carneiro et al. 2020) and thus providing easy and accessible insights for non-technical, but key audiences (e.g., industry, fisheries managers, NGOs etc.) has been challenging. The approach we used here provides an answer to this challenge, as the maps provided in Fig, 5-15 and Fig. 16 provide easily interpretable single-pane figures of the annual distribution of each taxon and the community, respectively. In addition, our approach is relatively straightforward and thus readily repeatable. We note that this approach, as many other approaches too, is subject to sample size issues, which is highlighted by the distribution maps of the two Royal Albatross species. We also acknowledge that there are other approaches to generating year-round insights into the distributions of seabirds, including more complex approaches (e.g., Richard et al. 2024b), but such approaches require extensive datasets and technical knowledge, and are thus not necessarily suitable for many taxa or multi-taxa analyses as ours here. Alternatively, simply plotting tracks (e.g., Rowley et al. 2024) could also present an alternative, but this approach does not account for the various shortcomings, typical of tracking data (e.g., differing tag types, different colony sizes, and spatiotemporally varying sample sizes). Consequently, we believe that our approach here provides a middle ground of various analyses and visualisation suited for the complexity of tracking data, while retaining accessibility and interpretability, crucial for key audiences.

We here present two disparate data analyses, trend and spatial analyses, together, as they are inherently intertwined. For the focal species, the single-most pressing threat, and the likely driver of the presented population declines, is bycatch in commercial fisheries, particularly in the high seas. Various more sophisticated and detailed analyses have further highlighted the ongoing pressure that bycatch risk is placing on these taxa (e.g., Edwards et al. 2023a,b, Richard et al. 2024, Rowley et al. 2024). While it may be tempting to speculate on impacts of other threats such as impacts of climate change, invasive species, or plastic pollution, no evidence exists for the taxa analysed here that indicates that the observed declines are due to threats other than bycatch in commercial fisheries (e.g., Clark et al. 2023, ACAP 2024, Richard et al. 2024). Consequently, a logical continuation on the analyses presented here is to merge the two disparate analyses into one single spatial representation, illustrating where the most severely declining taxa are occurring and subsequently communicate this to decision-makers in order to address the most likely driver of the declines, commercial fisheries bycatch.

5. CONCLUSION

The trend estimates and distribution maps we presented here highlighted the dire state of these New Zealand Procellariiform taxa, their dependency on the high seas, and thus the international responsibility for reversing the current and ongoing collapse. Our results show that 73% of the focal taxa exhibited highly concerning rates of decline, with biannual and semi-biannual breeders (e.g., the taxa that are most *k*-selected and thus most vulnerable to decreases in adult survival) showing the greatest rates of decline. Rates of declines have not improved over the decades. Some taxa may thus warrant changes in their IUCN Red List assignments. This Procellariiform seabird community utilises virtually every ocean on Earth and that all taxa depend on the high seas extensively. All evidence indicates that the underlying drivers of the presented declines are ongoing, unsustainable levels of bycatch in commercial fisheries in the high seas (Richard et al. 2024, Rowley et al. 2024). Decades of bycatch mitigation efforts have not resulted improvements in population trends (for many taxa, not even stabilisation). Therefore, further concerted and improved efforts to ensure the implementation of best practice seabird bycatch mitigation methods globally, particularly in the high seas, as well as ensuring adequate monitoring of and compliance with these mitigation methods are crucial to preserving what is left of both this, and the global, large Procellariiform seabird community.

6. ACKNOWLEDGMENTS

The data and analyses presented here would not exist without the commitments and hard work of dozens of people over decades, too numerous to exhaustively name here. Key data contributors and analysts include B Baker, B Bell, D Bell, M Bell, S Bose, D Burgin, A Carneiro, M Charteris, P Crowe, G Elliott, P Frost, C Mischler, P Moore, G Parker, B Philp, S Ray, K Rexer-Huber, C Robertson, P Sagar, G Taylor, A Tennyson, D Thompson, T Thompson, A Waipoua, S Waugh, K Walker, J Watts, E Whitehead & C Wickes.

We also thank Fisheries New Zealand for (co-)funding various tracking devices, including some for Antipodean, Gibson’s, Southern Royal Albatross and Black Petrels. Finally, many of the data presented here we obtained through funding provided by the Conservation Services Programme and the relevant fisheries that were levied through this programme, and thus we thank the industry for their contributions over the years.

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Table 1. Sample size and characteristics of analysed population count and tracking data per taxon.

Taxon		**NZ endemic**	**IUCN status**	***N* tracks (*N* GPS/PPT; *N* GLS)**	**Tracked colonies**	**Tracking period**	***N* counts**	**Count period (*N* years)**	**Count frequency (year^{-1})**	**Counted colonies**	**Count method**
Antipodean Albatross	*Diomedea antipodensis antipodensis*	✓	EN[2]	273 (223; 50)	Antipodes Is.	1996-2023	30	1994-2024 (31)	0.97	Antipodes Is.	Study/index site counts
Gibson’s Albatross	*Diomedea antipodensis gibsoni*	✓	EN[2]	190 (116; 74)	Adams Is.	1994-2024	27	1997-2024 (28)	0.96	Adams Is.	Study/index site counts
Southern Royal Albatross	*Diomedea epomorphora*	✓	VU	39 (36; 3)	Campbell Is.	2024	7	1997-2024 (26)	0.27	Campbell Is.	Study/index site counts
Northern Royal Albatross	*Diomedea sanfordi*	✓	EN	81 (76; 5)	Motuhara & Pukekura	1993-2021	20	1972-2024 (30)	0.66	Motuhara	Whole island counts
Grey-headed Albatross	*Thalassarche chrysostoma*		EN	69 (17; 52)	Campbell Is.	1997-2013	12	1945-2024 (80)	0.15	Campbell Is.	Whole island counts
Campbell Albatross	*Thalassarche impavida*	✓	VU	78 (10; 68)	Campbell Is.	1997-2011	12	1945-2024 (80)	0.15	Campbell Is.	Whole island counts
Salvin’s Albatross	*Thalassarche salvini*	✓[1]	VU	82 (29; 53)	Proclamation Is., Western Chain	2008-20	7	1978-2020 (43)	0.16	Proclamation Is.	Whole island counts
White-capped Albatross	*Thalassarche steadi*	✓[1]	NT	34 (13; 21)	Disappoint. Is.	2005-10	12	2002-18 (17)	0.71	Disappoint. Is.	Study/index site counts
Southern Buller’s Albatross	*Thalassarche bulleri bulleri*	✓	NT[2]	176 (118; 58)	Solander Is., Snares Is.	1994-2024	31	1969-2024 (56)	0.55	Snares Is.	Study/index site counts
Black Petrel	*Procellaria parkinsoni*	✓	VU	93 (21; 72)	Aotea, Hauturu	2007-20	23	2001-23 (23)	1.00	Aotea	Study/index site counts
Flesh-footed Shearwater	*Ardenna carneiceps*		NT	36 (30; 6)	Lady Alice Is., Ohinau Is.	2006-23	3	2009-24 (16)	0.19	Lady Alice Is.	Whole island counts

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[1] A small number of breeding pairs breed outside of New Zealand, but given the vast majority of the population breeds in New Zealand, these taxa are considered endemic.

[2] IUCN threat status is assigned at species level and reported here at taxa level.

Table 2. Population trend estimates over varying time periods (median and 95% credible intervals) as well as the highest recorded population counts (N_{max} in breeding pairs). Note that the N_{max} can represent the highest count for the study/index site only.

Taxon	r_{total} (%)	$r_{pre-1990}$ (%)	$r_{1990-2010}$ (%)	$r_{post-2010}$ (%)	$r_{N_{max}}$ (%)	Year N_{max}	N_{max}
Antipodean Albatross	-2.63 (-2.69; -2.57)	-	-0.55 (-0.69; -0.42)	-1.23 (-1.44; -1.02)	-4.55 (-4.67; -4.42)	2004	8,153
Gibson's Albatross	-2.47 (-2.53; -2.40)	-	-5.99 (-6.16; -5.81)	-0.15 (-0.33; 0.00)	-1.60 (-1.70; -1.49)	2004	8,728
Southern Royal Albatross	-1.37 (-1.57; -1.17)	-	-0.76 (-1.09; -0.42)	-	-1.68 (-1.92; -1.43)	1999	2,323
Northern Royal Albatross	-1.80 (-1.84; -1.75)	2.81 (2.64; 2.98)	-4.81 (-4.96; -4.67)	-3.74 (-4.10; -3.38)	-3.81 (-3.90; -3.72)	1992	6,800
Grey-headed Albatross	-3.06 (-3.08; -3.04)	-2.81 (-2.85; -2.78)	0.08 (-0.10; 0.25)	-2.69 (-2.90; -2.48)	-3.06 (-3.08; -3.04)	1945	43,000
Campbell Albatross	-0.59 (-0.60; -0.57)	-1.00 (-1.03; -0.96)	0.14 (0.05; 0.22)	-0.31 (-0.44; -0.19)	-0.69 (-0.71; 0.66)	1965	34,800
Salvin's Albatross	-2.73 (-2.81; -2.65)	-	-2.79 (-3.21; -2.37)	-1.14 (-1.59; -0.69)	-2.73 (-2.81; -2.65)	1978	8,655
White-capped Albatross	-2.51 (-2.71; -2.30)	-	-0.45 (-0.96; 0.06)	-2.53 (-3.02; -2.05)	-3.97 (-4.26; -3.68)	2007	4,478
Southern Buller's Albatross	0.95 (0.57; 1.31)	-	1.93 (1.35; 2.52)	0.00 (-0.82; 0.70)	-23.63 (-32.54; -14.77)	2022	303
Black Petrel	1.83 (1.20; 2.46)	-	2.50 (0.16; 4.84)	2.46 (1.17; 3.75)	-	2023	128
Flesh-footed Shearwater	0.94 (0.57; 1.31)	-	-	0.94 (0.57; 1.31)	-	2018	3,217
Community-level mean	-1.22 (-1.39; -1.05)	-0.33 (-0.42; -0.25)	-1.07 (-1.56; -0.58)	-0.84 (-1.29; -0.40)	-	-	-