THE PERUVIAN DIVING PETREL *PELECANOIDES GARNOTII* BOUNCES BACK: A 300-FOLD POPULATION INCREASE ON CHOROS ISLAND, NORTHERN CHILE

MARIÁN A. MÉNDEZ[1], NICOLÁS LUNA[2], & GUILLERMO LUNA-JORQUERA[1,3]*

[1]*Department of Marine Biology, Center for Ecology and Sustainable Management of Oceanic Islands ESMOI, Facultad de Ciencias del Mar, Universidad Católica del Norte, Larrondo 1281, Coquimbo, Chile*
[2]*Research and Technology Centre (FTZ), Kiel University, Hafentörn 1, 25761, Büsum, Germany*
[3]*Centro de Estudios Avanzados en Zonas Áridas (CEAZA), Coquimbo, Chile *(gluna@ucn.cl)*

Received 20 November 2025, accepted 09 July 2026

ABSTRACT

Méndez, M. A., Luna, N., & Luna-Jorquera, G. (2026). The Peruvian Diving Petrel *Pelecanoides garnotii* bounces back: A 300-fold population increase on Choros Island, Northern Chile. *Marine Ornithology*, *54*(2), 467–473. http://doi.org/00.0000/0000-0000.00.0.0000

The Peruvian Diving Petrel *Pelecanoides garnotii* experienced a dramatic population decrease from an estimated 100,000 breeding pairs in the early 20th century to just 300 pairs by 1980. Since the 1990s, conservation efforts have promoted a remarkable recovery. We updated population estimates on Choros Island, the species' primary breeding site, through a census conducted in November 2024. We estimated 89,876 breeding pairs (95% CI: 80,952–98,800), representing a 7.5-fold increase since 2013 and a 300-fold recovery over 44 years. The expansion of breeding habitat and consolidation of nesting patches reflect successful long-term conservation measures, particularly habitat protection and eradication of European Rabbit *Oryctolagus cuniculus*. This recovery demonstrates the critical value of sustained, multi-decade conservation strategies for threatened seabird populations. However, proposed mining and port development adjacent to the island would undermine this recovery and reverse decades of progress.

Key words: seabirds, abundance, habitat restoration, habitat protection, conservation, *Pelecanoides garnotii*, Humboldt Current

INTRODUCTION

Over recent decades, global seabird populations have decreased substantially, resulting from multiple anthropogenic stressors, including exotic species introductions at breeding sites, fisheries bycatch, pollution, and habitat degradation (Dias et al., 2019). Given these challenges, habitat restoration—particularly the recovery of breeding sites—has emerged as a critical conservation priority, with systematic population monitoring proving essential for tracking recovery trajectories (Dunn et al., 2024). The Peruvian Diving Petrel *Pelecanoides garnotii*, a species endemic to the Humboldt Current System, exemplifies the value of implementing such interventions. Historically, various anthropogenic disturbances have directly impacted the breeding sites of this species throughout its distribution in Chile. These include nest damage caused by unrestricted human transit within breeding islands, guano extraction, harvesting of adults and eggs, light pollution (Cristofari et al., 2019), and introduction of invasive mammals such as European Rabbits *Oryctolagus cuniculus* (introduced to Choros and Chañaral islands) and foxes *Lycalopex culpaeus* or *L. griseus* (introduced to Chañaral Island; Araya & Duffy (1987), as *Canis culpaeus* or *C. griseus*, respectively). These pressures had severe consequences. The largest colony on Chañaral Island, once estimated at over 100,000 breeding pairs, was completely extirpated (Araya & Duffy, 1987). According to Araya & Duffy (1987), European Rabbits were introduced to nearby Chañaral Island in the 20th century and were likely brought to Choros Island around the same period, causing severe habitat degradation on both islands. However, no data exist for the population prior to rabbit introduction; the earliest available estimate, from 1980, reported only around 300 breeding pairs (Fernández et al., 2019), and it remains unknown whether the population was stable or already decreasing at that time. As a consequence of the decrease in numbers of breeding colonies in both Peru and Chile, the Peruvian Diving Petrel was classified as Threatened in 1988 (BirdLife International, 2020). More recently, the species has been downlisted to Near Threatened, reflecting population recovery, although its breeding colonies remain highly localized and vulnerable (BirdLife International, 2020).

Currently, four Peruvian Diving Petrel breeding colonies are known in Chile: Grande de Atacama Island, Pan de Azúcar Island, Pájaros II Island, and Choros Island (see Figure 1 in Fernández et al., 2019). Choros Island is recognized as the largest breeding colony of Peruvian Diving Petrel in Chile, with 11,934 breeding pairs estimated in 2013. This estimate represented 95% of the Chilean population recorded that year, while the remaining 5% were distributed across the other three islands (Fernández et al., 2019). Both Chañaral and Choros islands are of high conservation value: Chañaral because of its historical significance as the former largest colony, and Choros for its current role as the species' primary breeding site (Fernández et al., 2019).

Conservation measures have progressively enhanced protection on both islands and their surrounding waters. The Pingüino de Humboldt National Reserve was established in 1990 to protect the Humboldt Penguin *Spheniscus humboldti*, encompassing both Choros and Chañaral islands and indirectly benefiting Peruvian Diving Petrel breeding sites by restricting public access (Ministerio de Minería, 1990). Marine protection was further strengthened in 2005 with the designation of two separate marine reserves: the Islas Choros y Damas

Marine Reserve and the Isla Chañaral Marine Reserve, each providing a one-nautical-mile buffer around their respective islands (Subsecretaría de Pesca [SUBPESCA], 2005a, 2005b). Targeted restoration actions accomplished the successful eradication of European Rabbits from both islands in 2013 (Island Conservation, 2014), followed by installation of artificial nests and an acoustic attraction system on Chañaral Island in 2019 to promote breeding recolonization (Wolf & Vilches, 2024). Most recently, the "Archipiélago de Humboldt" Multiple-Use Conservation Area was established in 2023, expanding marine and coastal protection to over 574,000 ha (5,740 km^2) and encompassing all eight islands of the archipelago (Ministerio del Medio Ambiente, 2023). Despite these varied restoration approaches, the colony on Chañaral Island has not yet re-established to pre-collapse levels. However, the detection of breeding pairs on the island represents the first reproductive activity recorded on the island since the 1980s (Wolf & Vilches, 2024). Therefore, Choros Island remains the most important colony for the species' recovery and serves as the primary reference for monitoring conservation outcomes. This study aims to provide an updated estimate of the breeding population of the Peruvian Diving Petrel on Choros Island.

STUDY AREA AND METHODS

Choros Island is located in the Coquimbo Region, Northern Chile, at 29°13′S and 71°31′W, and has an approximate area of 2.92 km^2 (Fig. 1). The island is characterized by a Mediterranean, semi-arid climate and a maximum elevation of 67 meters above sea level. Most of the colony is easily accessible, except for several difficult-to-access areas on the eastern slope characterized by loose rocks. In this area, a total of seven patches were identified, ranging in size from 196 to 1,290 m^2 and covering a total area of 3,970 m^2, equivalent to ca. 3.7% of the entire colony. Each patch contained fewer than 20 nests located in crevices and cavities between rocks, with a total of more than 140 breeding pairs.

To estimate the abundance of breeding pairs of Peruvian Diving Petrel, we strictly followed the standardized census protocol described by Fernández et al. (2019), which we have applied since 2010. This protocol integrates three key parameters: nesting patch size, occupancy rate, and nest density. Fieldwork was conducted between 08 and 12 November 2024. Patches were mapped with a handheld GPS (Garmin GPSMAP 62), excluding difficult-to-access areas. The perimeter of each patch was walked to delineate its boundaries, and its area was calculated using the GPS's built-in area measurement function. Patches were classified as small (≤ 59 m^2 or < 30 nests) or large (> 59 m^2). In small patches, nests were counted directly by two independent observers, and counts were averaged when they differed by ≤ 5%; otherwise they were repeated. In larger patches, circular plots of 0.55 m^2 were used for subsampling. After measuring the area of each nesting patch, we established circular sampling plots at both the center and the periphery of the patch. Count points were systematically arranged within each patch by alternating center and periphery locations every 20 steps until the entire perimeter was covered. The number of sampling plots per patch ranged from six to 164, depending on patch size. At each count point, the circular plot attached to a rope was randomly deployed and then retrieved using the rope. This procedure allowed us to avoid entering the patches and thus reduced the risk of nest collapse. We used a 3-m rope for the periphery and a 10-m rope for the center. A nest was recorded as present when its entrance lay entirely within the boundaries of the sampling point. This methodology minimized errors associated with double counting and allowed calculation of 95% confidence intervals for the population estimate. The total sampling area for each patch was calculated by multiplying the area of the circular plot by the number of times it was applied. The number of nests in the sampled area was calculated by summing the counts registered every time the plot was applied. The number of active nests for each patch was obtained correcting for both the occupancy rate (the ratio between active/inactive nests) and the percentage of nests with double entrances. Nest density for each patch was obtained by dividing the total number of corrected nests by the total sampled area. Mean nest density and 95% confidence intervals were calculated from all sampled patches.

To quantify the temporal trend of the Peruvian Diving Petrel population on Choros Island, we compiled all available census estimates of breeding pairs between 1980 and 2024 (n = 11 census years). Because surveys were not conducted at regular intervals, we modelled population growth using a log-linear regression approach, which is robust to irregular sampling and has been used to estimate finite rates of increase in seabird populations recovering after invasive mammal eradications (Bird et al., 2024; Brooke et al., 2018). Assuming exponential population growth, abundance at time t can be expressed as $N_t = N_0 \cdot \lambda^t$, where λ is the finite (annual) rate of increase. Applying natural logarithms, λ was estimated from the slope (b) of a linear regression of ln(pairs) on years, as $\lambda = e^b$. The corresponding annual proportional growth rate was computed as $r = (\lambda - 1) \cdot 100$ and expressed as percent per year. The 95% confidence intervals for λ were estimated by exponentiating the 95% confidence limits of b derived from the regression.

To evaluate whether the eradication of European Rabbits in 2013 was associated with a change in the Peruvian Diving Petrel population trajectory, we fitted three separate log-linear models: one to the full time series, one to the pre-eradication period (1980–2013), and one to the post-eradication period (2013–2024); the 2013 census was included in both partial models as the common boundary point. The temporal trend in total nesting-patch area was tested with a robust linear regression of total patch area against year; a robust regression was used because of the high heterogeneity detected in the total patch-area data. All analyses were carried out in R (R Core Team, 2024) using the base function *lm()* for the log-linear population models and the *rlm()* function of the "MASS package" (Venables & Ripley, 2002) for the robust regression; figures were produced with the package "ggplot2" (Wickham, 2016).

The study was conducted under authorization from the Chilean National Forest Corporation (CONAF; Permit No. 032/2022) to conduct fieldwork within the reserve. Capture and handling of wild birds were authorized by the Chilean Agricultural and Livestock Service (SAG; Exempt Resolution No. 6906/2024). All procedures were approved by the Scientific Ethics Committee of the Universidad Católica del Norte, Coquimbo Campus (CEC UCN Resolutions No. 16/2022 and No. 19/2024).

RESULTS AND DISCUSSION

A total of 55 patches—six small and 49 large—were georeferenced on Choros Island, primarily on the western side of the island (Fig. 1). Patch size ranged from 14 to 29,926 m^2, with a total colony area of 108,369 m^2. Mean patch size increased from 1,743 m^2 (95% CI: 827–2,659) in 2010 to 1,970 m^2 (95% CI: 727–3,214) in 2024; however, this difference was not statistically significant when mean

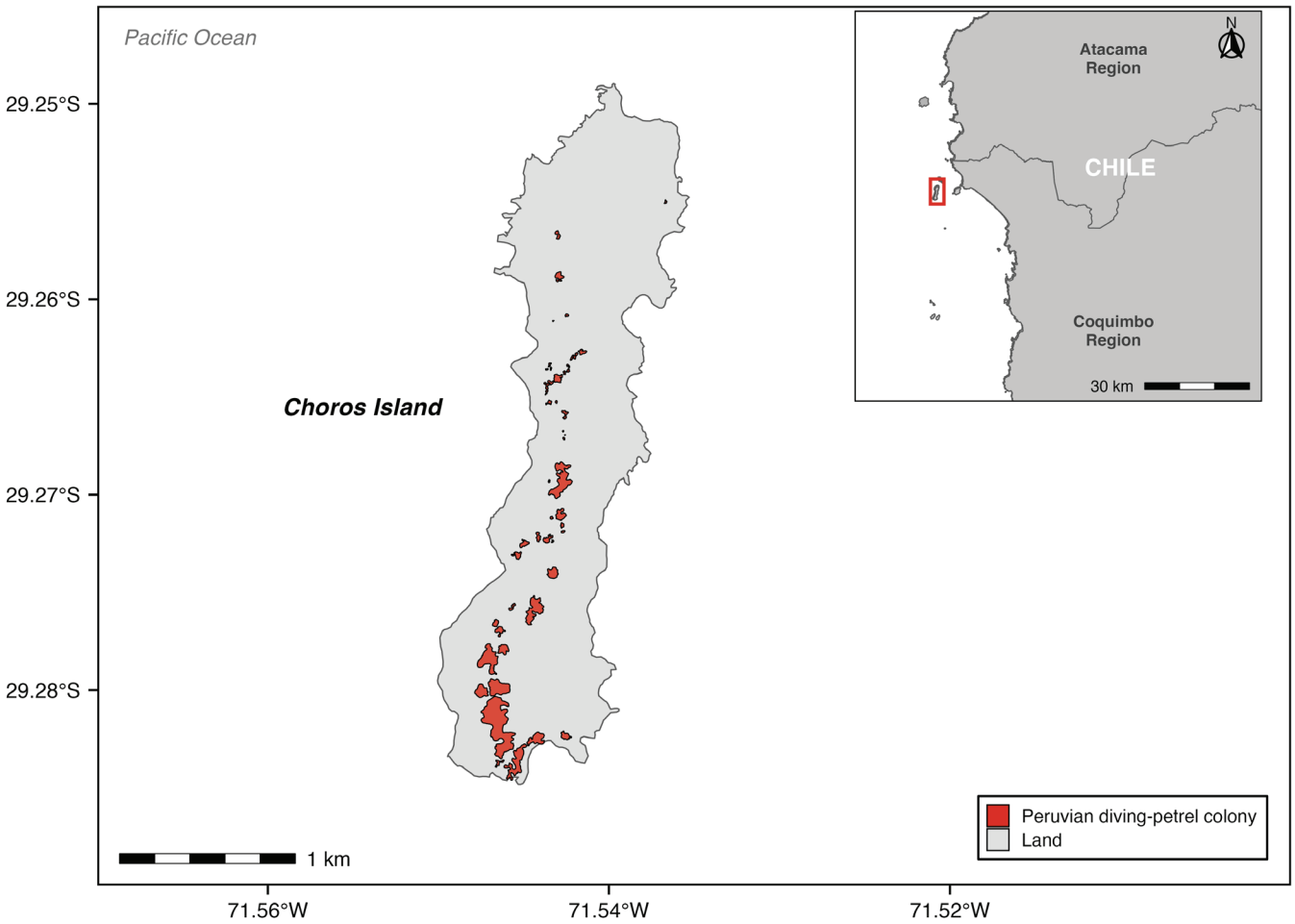


Fig. 1. Nesting patches of the Peruvian Diving Petrel *Pelecanoides garnotii* colony on Choros Island, Chile, shown in red. Records were obtained during the spring 2024 breeding season.

patch sizes were compared on the log scale (ANOVA, $F = 0.018$, $p = .892$). Nevertheless, total nesting-patch area increased over time (Fig. 2). After relatively low values in 2010–2012, the total area occupied by nesting birds more than tripled between 2013 and 2014 and continued to rise through 2024, when it reached its maximum. This temporal increase in total colony area was statistically significant based on a robust linear regression of total patch area against year ($p = .006$).

Regarding nest occupancy, 42.4% of nests were classified as inactive, of which 41.3% were empty, 0.4% contained a dead adult, and 0.7% contained dead chicks. In contrast, 57.6% of nests were active, with the highest proportion corresponding to nests with chicks alone (17.2%), followed by nests with an adult and an egg (16.3%), nests with only an egg (15.3%), nests with an adult and a chick (5.4%), and finally nests with an adult only (3.4%). The higher proportion of unhatched eggs (31.6%) compared to chicks (22.6%) suggests that the census was conducted during an intermediate to advanced breeding stage, when most pairs had already completed egg-laying and a substantial proportion of eggs had hatched. Nest occupancy and the proportion of active and inactive nests are known to vary seasonally in species of the genus *Pelecanoides* as the breeding cycle progresses, reflecting the transition from incubation to chick rearing. In this context, the 2024 census results were compared with those obtained in 2013, when an occupancy rate of approximately 60% was estimated during the peak of the breeding season in November, corresponding to an inactive nest

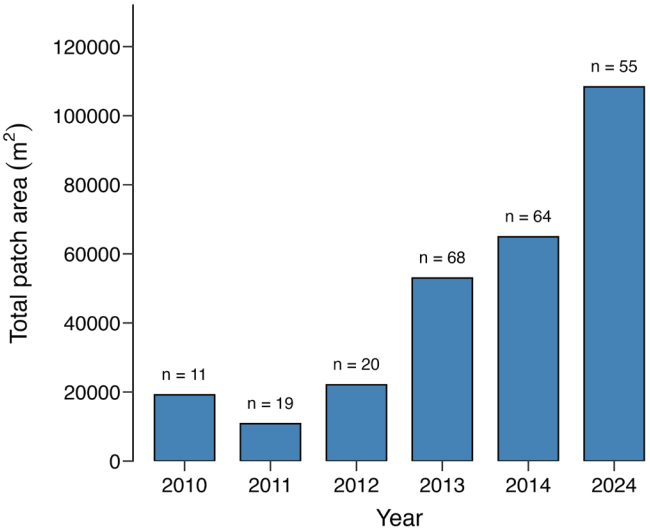


Fig. 2. Temporal variation in the total area of nesting patches (m^2) of the Peruvian Diving Petrel *Pelecanoides garnotii* colony on Choros Island, Chile. The number of patches (n) surveyed per year is indicated.

proportion of approximately 40%. This value is comparable to the 42.4% of inactive nests estimated in 2024. Consequently, the similar occupancy rates suggest that both censuses were conducted at a similar temporal stage of the spring breeding cycle, minimizing potential biases associated with survey timing and supporting direct comparison of occupancy parameters.

The average nest density was 0.91 nests·m^{-2}, with an occupancy rate of 57%. Because approximately 4% of nests had two or three entrances (Fernández et al., 2019), we applied a correction factor to account for multiple entrances. After correction, the average density decreased to 0.81 nests·m^{-2} and the occupancy rate to 51%. A one sample *t*-test indicated that the mean nest density in 2024 did not differ significantly from the mean of 0.89 nests·m^{-2} (range 0.47–1.50 nests·m^{-2}) reported by Fernández et al. (2019) for the 2010–2014 period. During that period, nest density also showed no significant temporal variation according to a general linear model (p = .672; based on data from Table S1 in Fernández et al., 2019).

The current breeding population size of Peruvian Diving Petrel on Choros Island was calculated to be 89,876 breeding pairs (95% CI: 80,952–98,800). This estimate is consistent with the 98,000 pairs reported in 2023, although that figure was obtained using a different methodological approach (Wolf & Vilches, 2024). After remaining relatively low through the early 2000s, the population began to increase steadily from 2008 onward, reaching 11,934 pairs in 2013 (Fig. 3). No census data are available for 2015–2022 because of initial funding limitations and subsequent COVID-19-related restrictions. Over the full time series (1980–2024), the population grew at λ = 1.144 (r = 14.4% per year; 95% CI: 1.104–1.186). Growth was moderate before rabbit eradication (1980–2013: λ = 1.108, r = 10.8% per year; 95% CI: 1.074–1.143) and markedly higher afterwards (2013–2024: λ = 1.232, r = 23.1% per year; 95% CI: 1.123–1.350).

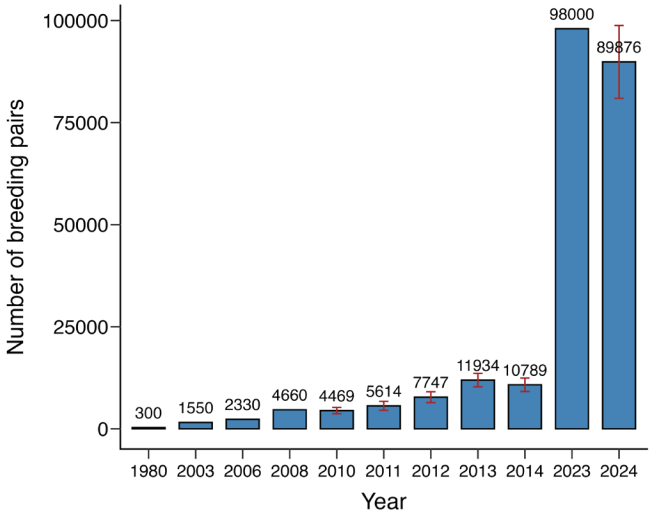


Fig 3. Estimates of the breeding population of Peruvian Diving Petrel *Pelecanoides garnotii* on Choros Island, Chile. Historical records from 1980 to 2008 were compiled from published sources and are presented without 95% confidence intervals because the original studies did not report measures of uncertainty. Estimates from 2010–2014 and 2024 were obtained using the same methodology (Fernández et al., 2019) and include vertical error bars representing 95% confidence intervals. The 2023 value, reported by Island Conservation (Wolf & Vilches, 2024), was obtained using a different methodology and does not include confidence intervals.

The magnitude of the increase documented in this study—corresponding to a ~300-fold rise in breeding pairs over the 1980–2024 period and a ~7.5-fold increase relative to the 2013 census—is consistent with recovery trajectories described for seabirds following the eradication of introduced mammals (Brooke et al., 2018; Jones et al., 2016).

In burrow-nesting Procellariiformes, recovery has been linked to three concurrent mechanisms: the release of nesting burrows previously occupied or destroyed by invaders, the regeneration of vegetation that can stabilize the burrowing substrate, and immigration from neighboring or formerly extirpated colonies (Bird et al., 2024; Brooke et al., 2018). These mechanisms may also operate on Choros Island, although their relative contributions are not known with certainty. The release of burrows from invasive pressure is likely the principal driver of this recovery, whereas vegetation recovery may play a comparatively minor role because Peruvian Diving Petrels generally do not nest beneath dense plant cover. Immigration is also plausible; however, genetic evidence indicates that Choros Island has historically acted as a source of colonization for the Chilean and Peruvian range rather than as a sink (Cristofari et al., 2019). Therefore, the island is more likely to export individuals to neighboring colonies than to depend on immigration for its own recovery. On Choros, European Rabbits had occupied a substantial portion of the *P. garnotii* breeding area for decades, trampling and enlarging petrel burrows for their own use and degrading the nesting habitat. Their eradication in 2013 halted rabbit-induced burrow destruction and enlargement and was followed by vegetation recovery and a substantial increase in the total area occupied by the colony in subsequent years. The slow pre-eradication growth and the steep post-2013 increase are therefore temporally consistent with the removal of the main limiting factor and extend the positive trend previously reported by Fernández et al. (2019).

Comparable post-eradication recoveries have been documented in other burrow-nesting seabirds. The Bermuda Petrel *Pterodroma cahow* increased from 18 breeding pairs in 1961 to 85 pairs by 2008, corresponding to a mean growth rate of ~3.3% per year (λ = 1.033) sustained under predator control, the provision of artificial burrows, and the translocation of nestlings to a secure new colony on Nonsuch Island (Carlile et al. 2012; Madeiros et al., 2012). On Macquarie Island, the eradication of cats in 2001, followed by that of rabbits, rats *Rattus* spp., and mice *Peromyscus* spp. between 2011 and 2014, triggered clear population responses in the four resident burrow-nesting petrels. Antarctic Prions *Pachyptila desolata* and White-headed Petrels *Pterodroma lessonii* increased by ~1% per year, whereas Blue Petrels *Halobaena caerulea* and Grey Petrels *Procellaria cinerea*, which had been extirpated from the main island in the early 1900s, recolonized and increased by ~14% and ~10% per year, respectively, following cat eradication. Immigration was identified as the main driver of this rapid response (Bird et al., 2024). Similar post-eradication increases have been reported for three seabird species on Surprise Island, New Caledonia, two of which responded primarily through immigration (Philippe-Lesaffre et al., 2023); for Scripps's Murrelet *Synthliboramphus scrippsi* at Anacapa Island, California (Whitworth et al., 2013); and for several burrow-nesting petrels on Saint-Paul Island, southern Indian Ocean, where the number of breeding seabird species more than doubled (from six to 13) two decades after the eradication of rats and rabbits (Barbraud et al., 2021).

Over the full time series (1980–2024), the Peruvian Diving Petrel population on Choros grew at r = 14.4% per year (λ = 1.144),

a long-term rate that exceeds that reported for most recovering Procellariiformes (Brooke et al. 2018) and is comparable in magnitude to the rates documented for the recolonizing species at Macquarie Island (Bird et al. 2024), although the underlying mechanisms likely differ, as detailed below. The overall pattern at Choros, however, masks a marked shift associated with rabbit eradication: growth was moderate before eradication (1980 to 2013: r = 10.8% per year) and more than doubled afterwards (2013 to 2024: r = 23.1% per year). The post-eradication estimate is based on four census years (2013, 2014, 2023, and 2024) and carries a comparatively wide confidence interval; it should therefore be interpreted with some caution. Nevertheless, the results consistently indicate an acceleration of population growth following eradication. Both the robust long-term trend and the steeper post-eradication trajectory point in the same direction. Importantly, the mechanism underlying this rapid increase most likely differs from that invoked for the recolonizing species at Macquarie Island, where immigration was identified as the primary driver (Bird et al., 2024). At Choros, by contrast, net immigration is unlikely to be the main driver because the colony was never extirpated and, as noted above, has historically acted as a source rather than a sink (Cristofari et al., 2019). The increase reported here is therefore best interpreted as the outcome of improved local reproductive success and survival, facilitated by reduced burrow competition and habitat recovery following rabbit eradication, together with the recruitment of previously non-breeding individuals produced within the colony. The rapid increase in breeding pairs may also reflect the recruitment of a floating population of mature but non-breeding Peruvian Diving Petrels that took advantage of newly available nesting habitat following rabbit eradication. Similar dynamics involving cavity-nesting seabird species have been documented elsewhere, where floating adults rapidly occupy newly created or restored burrow habitat and contribute disproportionately to colony growth (Ainley et al., 2024).

Choros currently holds ~95% of the Chilean breeding population of Peruvian Diving Petrels (Fernández et al., 2019) and, given its role as a regional source, may increasingly export individuals to neighboring colonies such as the recently restored Chañaral Island, located only ~23 km away, where social attraction recently yielded the first natural hatching in decades. Additional processes may also contribute and warrant further investigation, including variability in breeding phenology and the possibility of more than one breeding attempt per year under favorable conditions. Although these mechanisms remain hypothetical, they underscore the dynamic nature of the population and the need for long-term demographic and behavioral studies.

The marked increase in breeding pairs at Choros Island highlights the effectiveness of multi-decadal conservation strategies that combine strict protection with active habitat restoration. Long-term protection and access restrictions implemented since the establishment of the Humboldt Penguin National Reserve in the 1990s have played a fundamental role in reducing chronic human disturbance at breeding sites. These measures were subsequently reinforced by the establishment of the Choros-Damas Marine Reserve and four Benthic Resource Management Areas decreed in 1998 in adjacent zones, which, although focused on benthic fisheries management, contributed to improved regulation and organization of fishing activities (Romero et al., 2025). The population increase observed in 2013 likely reflects the cumulative effects of these protection measures, while the exponential growth recorded in 2023 and 2024 is consistent with habitat restoration achieved through the eradication of European Rabbits completed in 2013. Despite these conservation gains, the recovery of the Peruvian Diving Petrel remains highly vulnerable to anthropogenic threats that intersect directly with the species' ecological and behavioral traits. At present, the most immediate threat is artificial light pollution (Silva et al., 2020). The breeding islands lie only ca. 7 km from small towns and coves along the continental coast. Because Peruvian Diving Petrels are nocturnally active around the colony, individuals are attracted to these artificial light sources, leading to collisions and grounding events that cause mortality in both fledglings and adults (Rodríguez et al., 2017). This mortality intensifies during January, when tourist activity along the coast increases (Silva et al., 2020).

Two additional and more severe threats would emerge if the Dominga Mining-Port Project (https://dominga.cl), proposed adjacent to Choros Island, is approved. First, the facilities would substantially increase maritime traffic within the spatially concentrated foraging areas used by Peruvian Diving Petrel and other seabird species, elevating the risk of vessel strikes and hydrocarbon spills, which are known to cause long-lasting damage to coastal and benthic communities (Castège et al., 2007; Urbina et al., 2021). Second, brine discharge from the project's desalination plant could degrade marine environmental quality within and adjacent to the Humboldt Penguin National Reserve (Riera et al., 2022), potentially disrupting prey availability in the highly productive upwelling system on which the Peruvian Diving Petrel, and other seabirds and marine mammals, depend. Critically, the Dominga project would also amplify the light-pollution threat that already affects the species. Additional light sources associated with the development—including vessels, port infrastructure, and the mining and desalination facilities—could compound the existing coastal lighting, increasing the cumulative attraction of petrels and the resulting collision-related mortality. Together, these threats underscore that the long-term persistence of this recovering population will depend not only on maintaining predator-free and structurally intact breeding habitats, but also on safeguarding key marine foraging areas and limiting light pollution around colonies.

CONCLUSION

Our findings provide robust evidence of the recovery of the Peruvian Diving Petrel breeding population on Choros Island, as reflected by the consolidation of nesting patches, a substantial expansion of colony area, and a marked rise in breeding pairs over the past decade. This recovery trajectory underscores the cumulative and long-term benefits of sustained conservation measures, particularly habitat restoration through invasive species eradication and the protection of breeding sites. It also provides critical information for designing new management plans and updating existing ones. Despite this encouraging trajectory, key demographic processes remain poorly understood. Future studies addressing demography, breeding success, and survival are therefore essential to clarify the mechanisms driving this recovery, project future population trends, and identify the conditions under which similar outcomes could be achieved at other coastal islands across the species' distributional range. This study not only documents the recovery of a single species but also highlights a broader conservation opportunity: the island-based approaches applied here may guide management actions for the recovery of other seabird species facing critical population declines.

ACKNOWLEDGEMENTS

We are grateful to Alvaro López for his valuable assistance in the

field. This research was supported by the DAAD/BECAS Chile doctoral scholarship, 2022 (ANID Folio: 62210024) awarded to NL. We thank reviewers who provided comments to help us to improve this paper.

CREDIT AUTHORSHIP CONTRIBUTION STATEMENT

MAM: Investigation, conceptualization, methodology, data curation, software, formal analysis, visualization, writing—original draft. NL: Investigation, project administration, funding acquisition, resources, methodology, supervision, validation, writing—review & editing. GL-J: Investigation, conceptualization, methodology, formal analysis, supervision, validation, writing—review & editing.

REFERENCES

Ainley, D. G., Divoky, G. J., Baird, P., & Spencer, G. C. (2024). 'Floating populations' of seabirds: The bane of demographic modelers and managers. *Marine Ornithology*, *52*(2), 375–382. http://doi.org/10.5038/2074-1235.52.2.1589

Araya, B., & Duffy, D. C. (1987). Animal introductions to Isla Chañaral, Chile: Their history and effects on seabirds. *Cormorant*, 15, 3–6.

Barbraud, C., Delord, K., Le Bouard, F., Harivel, R., Demay, J., Chaigne, A., & Micol, T. (2021). Seabird population changes following mammal eradication at oceanic Saint-Paul Island, Indian Ocean. *Journal for Nature Conservation*, *63,* Article 126049. https://doi.org/10.1016/j.jnc.2021.126049

Bird, J. P., Fuller, R. A., & Shaw, J. D. (2024). Patterns of recovery in extant and extirpated seabirds after the world's largest multipredator eradication. *Conservation Biology*, *38*(4), Article e14239. https://doi.org/10.1111/cobi.14239

BirdLife International. (2020). *Pelecanoides garnotii. The IUCN Red List of Threatened Species 2020: e.T22698280A179971538*. Retrieved October 15, 2025, from https://www.iucnredlist.org/species/22698280/179971538

Brooke, M. de L., Bonnaud, E., Dilley, B. J., Flint, E. N., Holmes, N. D., Jones, H. P., Provost, P., Rocamora, G., Ryan, P. G., Surman, C., & Buxton, R. T. (2018). Seabird population changes following mammal eradications on islands. *Animal Conservation*, *21*(1), 3–12. https://doi.org/10.1111/acv.12344

Carlile, N., Priddel, D., & Madeiros, J. (2012). Establishment of a new, secure colony of Endangered Bermuda Petrel *Pterodroma cahow* by translocation of near-fledged nestlings. *Bird Conservation International*, *22*(1), 46–58. https://doi.org/10.1017/S0959270911000372

Castège, I., Milon, E., & Pautrizel, F. (2007). Response of benthic macrofauna to an oil pollution: Lessons from the "Prestige" oil spill on the rocky shore of Guéthary (South of the Bay of Biscay, France). *Deep Sea Research Part II: Topical Studies in Oceanography*, *54*, 1303–1318.

Cristofari, R., Plaza, P., Fernández, C. E., Trucchi, E., Gouin, N., le Bohec, C., Zavalaga, C., Alfaro-Shigueto, J., & Luna-Jorquera, G. (2019). Unexpected population fragmentation in an endangered seabird: The case of the Peruvian diving-petrel. *Scientific Reports*, *9*(1), Article 2021. https://doi.org/10.1038/s41598-019-38682-9

Dias, M. P., Martin, R., Pearmain, E. J., Burfield, I. J., Small, C., Phillips, R. A., Yates, O., Lascelles, B., Borboroglu, P. G., & Croxall, J. P. (2019). Threats to seabirds: A global assessment. *Biological Conservation*, *237*, 525–537. https://doi.org/10.1016/j.biocon.2019.06.033

Dunn, R. E., Benkwitt, C. E., Maury, O., Barrier, N., Carr, P., & Graham, N. A. J. (2024). Island restoration to rebuild seabird populations and amplify coral reef functioning. *Conservation Biology*, *39*(1), Article e14313. https://doi.org/10.1111/cobi.14313

Fernández, C. E., Portflitt-Toro, M., Miranda-Urbina, D., Plaza, P., Luna, N., & Luna-Jorquera, G. (2019). Breeding abundance and population trend of the Peruvian Diving-petrel *Pelecanoides garnotii* in Chile: Recovery of an endangered seabird? *Bird Conservation International*, *30*, 423–433.

Island Conservation. (2014). *Restauración ecológica de la Reserva Nacional Pingüino de Humboldt: Erradicación de conejo europeo de la Isla Choros* [Final report]. Island Conservation.

Jones, H. P., Holmes, N. D., Butchart, S. H. M., Tershy, B. R., Kappes, P. J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D. P., Bonnaud, E., Burbidge, A. A., Campbell, K., Courchamp, F., Cowan, P. E., Cuthbert, R. J., Ebbert, S., Genovesi, P., Howald, G. R., Keitt, B. S., Kress, S. W., … Croll, D. A. (2016). Invasive mammal eradication on islands results in substantial conservation gains. *Proceedings of the National Academy of Sciences*, *113*(15), 4033–4038. https://doi.org/10.1073/pnas.1521179113

Madeiros, J., Carlile, N., & Priddel, D. (2012). Breeding biology and population increase of the Endangered Bermuda Petrel Pterodroma cahow. *Bird Conservation International*, *22*(1), 35–45. https://doi.org/10.1017/S0959270911000396

Philippe-Lesaffre, M., Thibault, M., Caut, S., Bourgeois, K., Berr, T., Ravache, A., Vidal, E., Courchamp, F., & Bonnaud, E. (2023). Recovery of insular seabird populations years after rodent eradication. *Conservation Biology*, *37*(6), Article e14107. https://doi.org/10.1111/cobi.14107

Ministerio de Minería. (1990). *Decreto 4: Crea Reserva Nacional Pingüino de Humboldt en terrenos fiscales de la III y la IV Región y la declara lugar de interés científico*. Biblioteca del Congreso Nacional de Chile. https://bcn.cl/3ohq7

Ministerio del Medio Ambiente. (2023*). Decreto 31: Declaración Área Marina Costera Protegida de Múltiples Usos denominada "Archipiélago de Humboldt"*. Biblioteca del Congreso Nacional de Chile. https://bcn.cl/ieNVPM

R Core Team. (2024). *R: A language and environment for statistical computing* (Version 4.4.1) [Computer software]. R Foundation for Statistical Computing. https://www.R-project.org/

Riera, R., Sanchez-Jerez, P., Rodríguez, M., Monterroso, O., & Ramos, E. (2022). Long-term monitoring of fish assemblages around a seawater desalination plant. *Marine Pollution Bulletin*, *174*, Article 113240. https://doi.org/10.1016/j.marpolbul.2021.113240

Rodríguez, A., Holmes, N. D., Ryan, P. G., Wilson, K. J., Faulquier, L., Murillo, Y., Raine, A. F., Penniman, J. F., Neves, V., Rod ríguez, B., Negro, J. J., Chiaradia, A., Then, P., Anderson, A., Metzger, B., Shirai, M., Deppe, L., Wheeler, J., Hodum, P., Gouveia, C., Carmo, V., Carreira, G. P., Delgado-Alburqueque, L., Guerra-Correa, C., Couzi, F.-X., Travers, M., & Le Corre, M. (2017). Seabird mortality induced by land-based artificial lights. *Conservation Biology*, *31*(5), 986-1001. https://doi.org/10.1111/cobi.12900

Romero, P., Bularz, B., Arenas-Proaño, G., & Moreno, D. (2025). Does the Chilean TURFs achieve the sustainability of its fisheries? Evaluation of its performance considering the administrative, biological, and economic dimensions of this fisheries regime. *Frontiers in Marine Science*, *12*, Article

1502422. https://doi.org/10.3389/fmars.2025.1502422

Silva, R., Medrano, F., Tejeda, I., Terán, D., Peredo, R., Barros, R., Colodro, V., González, P., González, V., Guerra-Correa, C., Hodum, P., Keitt, B., Luna-Jorquera, G., Malinarich, V., Mallea, G., Manríquez, P., Nevins, H., Olmedo, B., Páez-Godoy, J., … Toro-Barros, B. (2020). Evaluación del impacto de la contaminación lumínica sobre las aves marinas en Chile: Diagnóstico y propuestas. *Ornitología Neotropical*, *31*(1), 13–24. https://doi.org/10.58843/ornneo.v31i1.575

Subsecretaría de Pesca. (2005a). *Decreto 150: declara reserva marina espacio marítimo en torno a Isla Chañaral, III Región*. Subsecretaría de Pesca, Ministerio de Economía, Fomento y Reconstrucción. https://bcn.cl/3m2so

Subsecretaría de Pesca. (2005b). *Decreto Supremo N° 151: Declara Reserva Marina espacio marítimo en torno a Islas Choros-Damas, IV Región*. Subsecretaría de Pesca, Ministerio de Economía, Fomento y Reconstrucción. https://www.subpesca.cl/portal/615/w3-article-6308.html

Urbina, M. A., Guerrero, P. C., Jerez, V., Lisón, F., Luna-Jorquera, G., Matus-Olivares, C., Ortiz, J. C., Pavez, G., Pérez-Alvarez, M. J., Riquelme-Bugueño, R., Santos-Carvallo, M., Sepúlveda, M., Victoriano, P. F., & Gomez-Uchida, D. (2021). Extractivist policies hurt Chile's ecosystems. *Science*, *373*(6560), 1208–1209. https://doi.org/10.1126/science.abm0157

Venables, W. N., & Ripley, B. D. (2002). *Modern applied statistics with S* (4th ed.). Springer. https://doi.org/10.1007/978-0-387-21706-2

Whitworth, D. L., Carter, H. R., & Gress, F. (2013). Recovery of a threatened seabird after eradication of an introduced predator: Eight years of progress for Scripps's murrelet at Anacapa Island, California. *Biological Conservation*, *162*, 52–59. https://doi.org/10.1016/j.biocon.2013.03.026

Wickham, H. (2016). *ggplot2: Elegant graphics for data analysis.* Springer. https://doi.org/10.1007/978-3-319-24277-4

Wolf, C. A., & Vilches, M. J. (2024). *10-years post-eradication of invasive European rabbits on Isla Choros, Chile, Oct 2023* [Report]. Island Conservation.