Conservation Services Programme (CSP)

POP2025-06 Updated population assessment for New Zealand fur seal

Report prepared by Dr. Alasdair Hall, Hall Marine Consulting, and Dr. Jody Weir, Department of Conservation.

Photo: Alasdair Hall

Department of Conservation
Te Papa Atawhai

Te Kāwanatanga o Aotearoa
New Zealand Government

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Any questions can be directed to: **csp@doc.govt.nz**

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Executive Summary

- The protected kekeno/New Zealand fur seal (*Arctocephalus forsteri*) is the most commonly bycaught marine mammal species in New Zealand's commercial fisheries (MacKenzie et al. 2022), with 1,351 recorded bycatch deaths between the 2019/20 – 2025/26 fishing years[1] (to date: 15/04/2026) (Ministry for Primary Industries 2026), and 1,908 recorded captures. Additional threats to kekeno include emerging diseases (Taylor et al. 2026, Ashley et al. 2026) and changes to prey availability (Devine et al. 2025).

- This report details activities undertaken to update knowledge of kekeno abundance and distribution. It includes updates to abundance estimates in four regions (Kaikōura, Mātakitaki-a-Kupe/Cape Palliser, The West Coast of the South Island (WCSI) and The Bounty Islands), an updated nationwide population estimate and an evidence-based framework for future kekeno population monitoring.

- A mark-recapture study to estimate kekeno abundance at Mātakitaki-a-Kupe/Cape Palliser in January 2026 was postponed to 2027 due to forecasted extreme weather. A preliminary direct count indicated at least 1,000 pups had been produced in the 2025/26 breeding season. This figure should be treated with caution, and mark-recapture will be conducted at this site to provide a robust assessment.

- Mark-recapture and direct counts at six Kaikōura colonies estimated a total of 9,465 – 10,027 pups had been produced in the 2025/26 breeding season. All Kaikōura colonies experienced increases in pup production relative to 2024/2025. This may be a continued response to significantly reduced breeding in 2023/24, associated with an Unusual Mortality Event (UME).

- Continued pup production declines were identified at Wekakura Point, Cape Foulwind and Taumaka Island (WCSI) from an analysis of a multi-decade population monitoring programme. The 2025 pup abundance estimates were 186 (178 – 194 95% CI) at Wekakura Point, 131 (122 – 140 95% CI) at Cape Foulwind and 566 (555 – 577 95% CI) at Taumaka Island. These figures represent declines of 83%, 71% and 61% respectively from each colony's recorded maximum in the 1990s.

- Drone surveys at the Bounty Islands (January 2024) estimated 4,252 – 4,341 pups had been produced, the first local assessment in three decades.

[1] Fishing Years run from 1 October to 30 September

- An updated nationwide population estimate for New Zealand's kekeno was calculated using a pre-existing modelling framework, producing an estimate of 208,536 – 251,644 kekeno in 2026. Colonies where pup production data have never been collected could not be included in this estimate. The estimate could be improved by increasing the spatial coverage of regular kekeno colony assessments, and through the collection of demographic data.

- To track trends in kekeno abundance and consolidate understandings of what drives these trends, data collection is recommended for the following parameters:
    - Nationwide disease prevalence: to identify and respond to new diseases in New Zealand wildlife.
    - Pup production and body condition: to monitor trends in population abundance and health, and facilitate threat detection.
    - Marine distribution and foraging behaviour: to improve understandings of spatial and temporal overlap with commercial fisheries, and the impacts of climate change on prey availability.
    - Diet: to detect changes in prey availability and assess resource competition with commercial fisheries.
    - Survival and fecundity: to facilitate population viability analyses to understand how the population may respond to threats.

- A recommended framework for future kekeno monitoring involves combining studies at focal sites with coordinated nationwide surveys to track large-scale population trends. Suggested focal sites are:
    - West Coast South Island (Wekakura Point, Cape Foulwind and Taumaka Island colonies): due to an existing long-term dataset and longstanding concerns over bycatch rates.
    - Kaikōura (Ōhau Point, Kaikōura Peninsula and South Coast colonies): due to the recent UME, the detection of canine distemper virus (CDV) (Taylor et al. 2026), and the existence of a recently established monitoring programme.
    - Mātakitaki-a-Kupe/Cape Palliser: due to concerns over high bycatch rates (Pavanato et al. 2023) and the recent detection of CDV (Taylor et al. 2026).
    - Sandymount Colony (Otago Peninsula): due to an existing long-term dataset (Seddon et al., unpublished data; Appendix 4).

1 Introduction

1.1 Context

Kekeno/New Zealand fur seal (*Arctocephalus forsteri*) are the most commonly bycaught marine mammal in New Zealand waters, comprising 86% of observed marine mammal bycatch between 1995/96 – 2018/19 (MacKenzie et al. 2022). Between the 2019/20 – 2025/26 fishing years[2] (to date: 15/04/2026) there have been 1,908 recorded kekeno captures in commercial fisheries, with 1,351 of these resulting in mortality (Ministry for Primary Industries 2026) (Figure 1). A Risk Atlas platform using model outputs from the spatially-explicit fisheries risk assessment (SEFRA) (MacKenzie et al. 2022) found that trawl fisheries have the highest predicted annual adjusted exploitation rate for kekeno, particularly in Fishing Management Areas (FMAs) 2, 3 and 7 (Underwood et al. 2025). Set net fisheries with relatively high annual adjusted kekeno bycatch rates were within FMAs 3, 5, 8 and 9 (Underwood et al. 2025). The southern bluefin tuna surface longline fishery in FMAs 2, 3 and 7 was noted as a key surface longline fishery (Underwood et al. 2025). In the Fisheries Act (1996), emphasis is placed on avoiding, remedying and mitigating the adverse effects of fishing on protected marine mammal species at the population level. The Marine Mammals Protection Act (1978) and Marine Mammals Protection Regulations (1992), administered by The Department of Conservation (DOC), provide for the protection of all marine mammals in New Zealand.

Kekeno are one of two pinniped (seals, sea lions and walrus) species permanently resident on mainland New Zealand, and are also found on the country's nearshore, offshore and sub-Antarctic islands, as well as in Australia (where the nationwide population has been estimated to be 117,400 individuals (Chilvers & Goldsworthy, 2015)). Worldwide, pinnipeds experienced historic overharvesting. This included kekeno (Ling 1990), and while the last commercial hunt was in 1946 (Lalas & Bradshaw 2001), the species' distribution and abundance remain considerably reduced compared to pre-exploitation (Lalas & Bradshaw 2001, Emami-Khoyi et al. 2018, Hall et al. 2025b). There were potentially as many as three million kekeno in New Zealand prior to human arrival (Emami-Khoyi et al. 2018), with the most recent estimate of the contemporary population being ~182 – 239,000 (Hall et al. 2025b). Some otariids (eared-seals) impacted by overharvesting have recovered relatively rapidly – for example, there are now estimated to be over two million Cape fur seals (*Arctocephalus pusillus*) in southern Africa, compared to fewer than 100,000 individuals after sealing (Robbertse et al. 2025). By contrast, Australian fur seals (*Arctocephalus pusillus doriferus*) were recovering slowly (Kirkwood et al. 2010), with 21,400 live pups estimated in 2007, before declines in the following decade (16,903 pups were estimated in 2017 (McIntosh et al. 2022)). A similar reversal of a previously increasing abundance trend has been recorded in Antarctic fur seals

[2] Fishing Years run from 1 October to 30 September

(*Arctocephalus gazella*) on the South Shetland Islands (Krause et al. 2024). Post-sealing, kekeno expanded northwards from refugia colonies (Bradshaw et al. 2000b, Bouma et al. 2008, Dussex et al. 2016), with re-established breeding first recorded on the North Island in the early 1990s (Dix 1993).

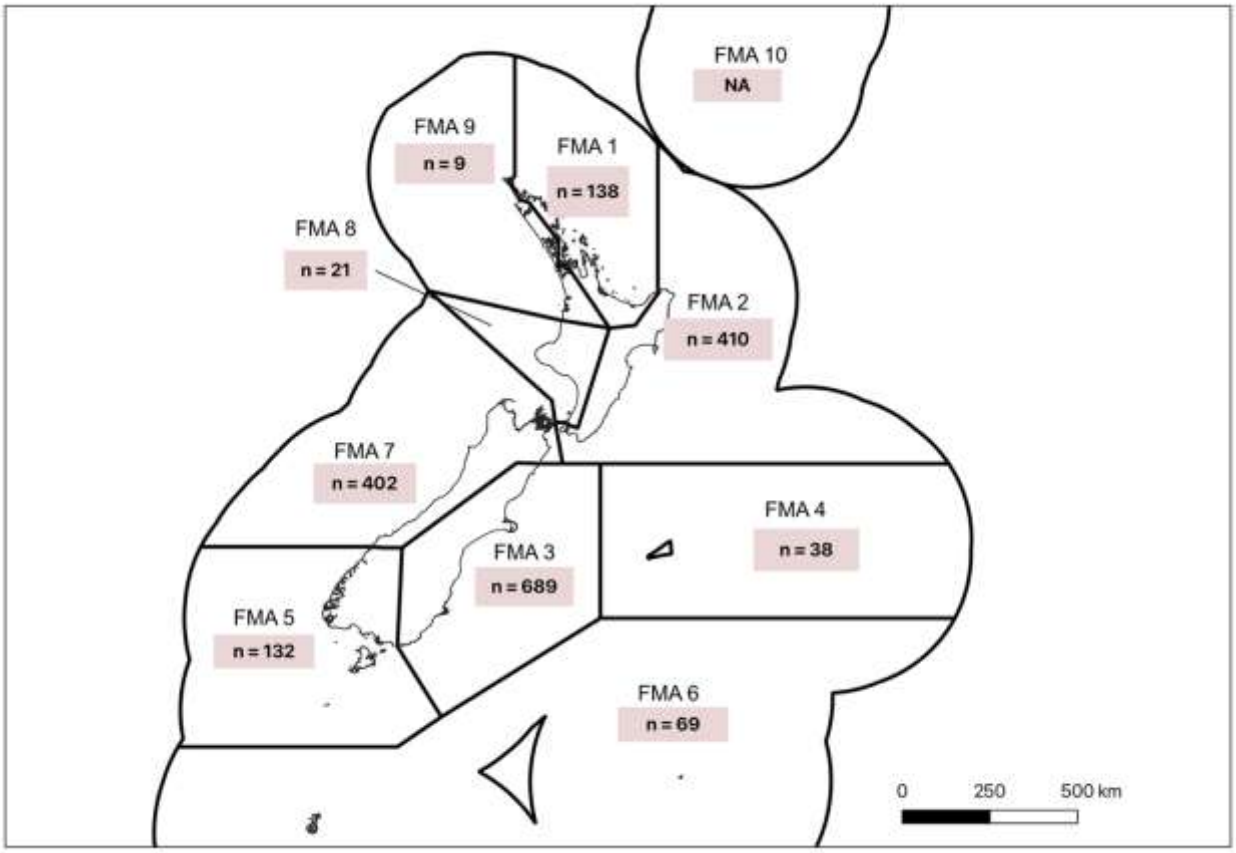

Figure 1 Reported kekeno/New Zealand fur seal bycatch (n) in commercial fisheries in the 2019/20 to 2025/26[2] fishing years by Fisheries Management Areas (FMA). Data accessed on 15/04/2026 from https://www.mpi.govt.nz/fishing-aquaculture/sustainable-fisheries/managing-the-impact-of-fishing-on-protected-species/accidental-captures-of-seabirds-and-protected-marine-species-by-commercial-fishers-from-1-october-to-31-december-2025

The biggest human threat to many otariids is entanglement and drowning due to fishing gear interactions (Chilvers 2018). These interactions not only cause pinniped mortality, but also engender gear damage and catch loss (Cosgrove et al. 2015, Barrios-Guzmán et al. 2024). Substantial effort has been dedicated to mitigating pinniped-fisheries interactions where they regularly occur (Hamilton & Baker 2019, Jounela et al. 2024, Underwood et al. 2025). Common approaches involve gear modifications, such as seal exclusion devices (Hamer & Goldsworthy 2006, Iriarte & Winter 2025), closures (Bamford et al. 2021, Jounela et al. 2024) and acoustic deterrents (Jepson et al. 2007, Lehtonen et al. 2022).

The additional pressure placed on pinniped species/populations through direct (bycatch) and indirect (resource competition) fisheries' impacts may lower their resilience to concurrent threats (Ashley et al. 2026). These include disease – particularly, currently, the threat from high pathogenicity avian influenza (HPAI) (Plaza et al. 2024, Romero et al. 2025, Ashley et al. 2026) and climate change – including its impacts on prey distribution and abundance (Cruz-Vallejo et al. 2024, Bas et al. 2025), and elevated heat stress while ashore (De Villiers & Roux 1992). Examples of disease and climate impacts on kekeno have been observed. A novel strain of canine distemper virus (CDV) was recorded in colonies in both Kaikōura and Mātakitaki-a-Kupe/Cape Palliser in 2025, associated with unusually high mortality (Wilson et al. 2025, Taylor et al. 2026). On the West Coast of the South Island (WCSI), declining kekeno pup abundance and mass have been associated with years of high sea surface temperature (Roberts et al. 2022). Some kekeno prey species on the WCSI have shifted into deeper waters (Devine et al. 2025).

Kekeno have been a lower conservation priority than other marine mammals (Lalas & Bradshaw 2001), and monitoring has been spatially inconsistent across New Zealand (Boren et al. 2006, Baird 2011, Hall et al. 2025b). There have been two long-term monitoring programmes for the species – on the WCSI (Hall et al. 2026b) and in south-eastern South Island (MacDiarmid & Lalas 2014; Seddon et al., unpublished data). An important opportunity exists to develop our understanding of the at-sea movements and diving behaviour of New Zealand's kekeno, as published data are limited in temporal and geographic scale, and outdated (Sinclair & Wilson 1994, Harcourt et al. 1995, 2001, 2002, Mattlin et al. 1998). Where such data have been collected for New Zealand seabirds (Rowley et al. 2024, Waipoua et al. 2026), and for pinnipeds overseas (Bamford et al. 2021, Goldsworthy et al. 2022, Riaz et al. 2023), they have been used to determine the spatial overlap with commercial fisheries, identify high risk periods and make recommendations on priority bycatch mitigation actions.

This report provides a summary of the results of recent kekeno population abundance monitoring efforts from four areas: Mātakitaki-a-Kupe/Cape Palliser, Kaikōura, The WCSI and the Bounty Islands. It also collates kekeno abundance data from around New Zealand to provide an updated nationwide population estimate. Finally, it draws on international best practices to suggest a framework for future kekeno monitoring in New Zealand.

1.2 Objectives

The objectives of this project are:

1. To provide robust estimates of the abundances of kekeno at several important sites.
2. To provide an updated nationwide abundance estimate for kekeno in New Zealand.
3. To provide an evidence-based framework for nationwide kekeno population monitoring.

2 Estimates of kekeno abundance at specific sites in New Zealand

2.1 Kekeno abundance estimate methodologies

Kekeno population size is assessed via pup production (Boren et al. 2006, Watson et al. 2015, Chilvers 2021, Hall et al. 2025a). This is typical in otariids (Berkson & DeMaster 1985, Chilvers et al. 2007, Kirkman et al. 2011, McIntosh et al. 2018), as pups are easily identifiable, and largely confined to areas within and around their natal colonies until weaning, meaning all are theoretically available to count. In contrast, unknown numbers of weaned individuals may be at sea at the time of surveying. A kekeno colony is defined as a breeding aggregation of kekeno, and when such aggregations occur within two kilometres of each other, they are categorized as sub-colonies within a single colony (Shaughnessy et al. 2015; Hall et al. 2024).

Two methods have been used to estimate kekeno pup production: direct counts and mark-recapture (also known as mark-resighting). Direct counts involve head counts of pups, and are quicker, cheaper and less disruptive than mark-recapture. Direct counts of kekeno pups have been conducted via ground, boat, manned aircraft and drone surveys (Taylor et al. 1995, Baker et al. 2009, Hall et al. 2025a, 2026a). Direct counts may produce less precise estimates than mark-recapture, particularly at colonies with large numbers of pups and/or complex terrain (Watson et al. 2009, McIntosh et al. 2018).

Mark-recapture involves capturing and marking a subset of the pups born at a colony before releasing them to mingle with non-marked pups. Subsequently, numbers of marked and unmarked pups are tallied during visual recapture counts (Boren et al. 2006, Chilvers 2021, Hall et al. 2025a). Marking has typically been achieved through fur-trimming (Boren et al. 2006, Hall et al. 2024) or flipper tagging (Bradshaw et al. 2000b, Hall et al. 2026b). In both direct counts and mark-recapture, dead pups should be added to the live totals. The ratio of marked to unmarked pups can be input into a modified Peterson estimate (Chapman 1952).

2.2 Kaikōura kekeno population abundance update

Kaikōura's coastline is closest to FMA 3. Between the 2019/20 – 2025/26 (to date: 15/04/2026) fishing years, there have been 689 reported kekeno captures in FMA 3, including 528 mortalities (Ministry for Primary Industries 2026). Of these, 403 were in trawls, 149 were in surface long lines and 135 were in set nets (Ministry for Primary Industries 2026). Other threats to Kaikōura's kekeno include disease (Taylor et al. 2026), vehicle collisions (Hall et al. 2023) and deliberate illegal killings. First observed in late 2023, Kaikōura's kekeno experienced an unusual mortality event (UME), which associated with both high observed rates of malnutrition and the eventual discovery of CDV in deceased individuals (Taylor et al. 2026).

2.2.1 Kaikōura 2025/26 pup production and condition assessment methods

2.2.1.1 Pup production

Since 2022/23, consistent pup production monitoring has occurred annually at five colonies: Ōhau Point, South Rakautara, the Kaikōura Peninsula, South Coast and Otumatu (Hall et al. 2024) (Figure 2). The Oaro colony was first assessed in 2024/25. The Kaikōura Peninsula and South Coast colonies consist of four and five sub-colonies respectively (Hall et al. 2024).

Direct counts and mark-recapture have been used to assess Kaikōura's kekeno population, with the choice of method influenced by the colony/sub-colony size (both in terms of geographic extent and pup production) and terrain complexity. Pups are marked by fur trimming (Figure 3). The methods used at each site in the 2025/26 breeding season are presented in Appendix 1.1.

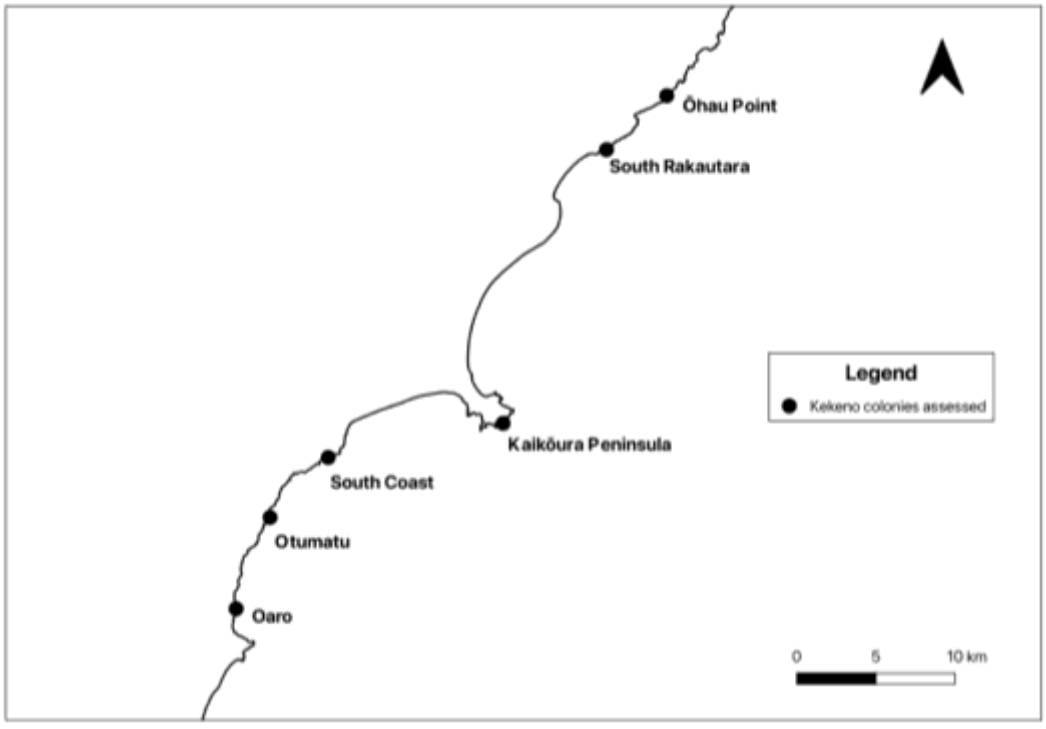

Figure 2 Locations of kekeno colonies assessed in Kaikōura in the 2025/26 breeding season

Figure 3 Example of fur trimming to facilitate mark-recapture with kekeno pups. Photo: Fiona Wardle

2.2.1.2 Pup condition

Measurements of pup dorsal straight length and mass were conducted at Ōhau Point, the Kaikōura Peninsula and South Coast colonies to quantify pup condition. Pups were weighed in a hessian sack suspended from an angling scale (Berkley 50LB Dial Scale), and measured from nose tip to tail tip using a tailor's tape measure, with the pup restrained in an outstretched position. At Ōhau Point, 101 pups were assessed (51 males and 50 females). At the Kaikōura Peninsula, 101 pups were assessed (51 males and 50 females, at Rhino Horn and Whalers Bay) and at the South Coast colony 100 pups were assessed (50 males and 50 females, at SC2 and SC4). In addition to mass and length, a body condition index (BCI) was calculated by dividing a pup's mass by its length (Boren et al. 2006).

2.2.2 Kaikōura 2025/26 pup production and condition assessment results

2.2.2.1 Pup production

Pup production estimates derived from mark-recapture and direct counts are shown in Table 1. A total of 1,656 – 1,740 (mean ± SE) pups were counted by direct count, and 7,809 – 8,287 (mean ± SE) estimated from mark recapture. Combined, these methods provide a pup production estimate of 9,465 – 10,027 pups for Kaikōura in the 2025/26 breeding season.

Table 1 Pup production estimates for Kaikōura kekeno colonies and sub-colonies assessed in the 2025/26 breeding season.

Site	Mean pup direct count ± SE	Mean pup mark-recapture estimate ± SE
Ōhau Point		5,346 ± 102
South Rakautara		591 ± 42
Lynch's Reef (Kaikōura Peninsula)	89 ± 1	
KP 2 (Kaikōura Peninsula)	381 ± 7	
Rhino Horn (Kaikōura Peninsula)	367 ± 17	
Whalers Bay (Kaikōura Peninsula)	471 ± 7	
SC 1 (South Coast)		616 ± 22
SC 2 (South Coast)		971 ± 45
SC 3 (South Coast)	168 ± 5	
SC 4 (South Coast)		524 ± 28
SC 5 (South Coast)	114 ± 4	
Otumatu	62 ± 0	
Oaro	46 ± 1	
Total by method	1,656 – 1,740	7,809 – 8,287
Total	9,465 – 10,027	

2.2.2.2 Pup condition

On average, male pups were longer and heavier than female pups. Average male pup length was 76.9 (± 4.88) cm compared to 74.1 (± 4.85) cm for females. Average male pup weight was 7.93 (± 1.55) kg, compared to 7.17 (± 1.41) kg for females. Average male BCI was 103 (± 15.4) g/cm compared to 96.2 (±14.9) g/cm for females. Welch's t-tests for pan-colony length, mass and BCI, found that male pups were significantly longer ($t = -5.08$, $df = 299.98$, $p < 0.0001$) and heavier ($t = -4.51$, $df = 298.08$, $p < 0.0001$) than female pups and had significantly higher BCI ($t = -3.63$, $df =$

299.91, p-value < 0.001). T-tests were also run to compare length, mass and BCI between sexes at each colony. At Ōhau Point, there was no significant sex-based difference in pup length (t = -1.72, df = 98.95, p = 0.09), but male pups had significantly higher mass (t = -2.62, df = 97.02, p = 0.01) and BCI (t = -2.76, df = 96.08, p < 0.01) than female pups. At the Kaikōura Peninsula, males were significantly longer (t = -3.77, df = 96.03, p < 0.001) and heavier (t = -2.95, df = 98.95, p < 0.01) than female pups, but there was no significant difference in pup BCI by sex (t = -1.87, df = 96.07, p = 0.06). Similarly, South Coast male pups were significantly longer (t = -3.56, df = 93.82, p < 0.001) and heavier (t = -2.48, df = 97.47, p = 0.01) than female pups, but there was no significant difference in pup BCI by sex (t = -1.80, df = 97.83, p = 0.07).

Pups from different colonies were weighed and measured on different dates, meaning the effect of 'colony' on raw morphometric measurements cannot be separated from the effect of 'measurement date'. Therefore, the results in Table 2 should not be used to directly compare between the condition of pups at the different assessed colonies, with modelling required to account for variability in dates.

Table 2 Results of pup condition assessments from three Kaikōura kekeno colonies in the 2025/26 breeding season.

Site	Measurement date(s)	Mean length (cm) ± SD	Mean mass (kg) ± SD	Mean BCI (g/cm) (± SD)
Ōhau Point (all)	17th Feb, 2026	75.30 ± 4.71	7.33 ± 1.51	96.8 ± 15.20
Ōhau Point (M)		76 ± 4.65	7.71 ± 1.59	101 ± 16
Ōhau Point (F)		74.40 ± 4.67	6.95 ± 1.35	92.7 ± 13.20
Kaikōura Peninsula (all)	21st Feb, 2026	74.30 ± 5.37	7.33 ± 1.33	98.3 ± 13.50
Kaikōura Peninsula (M)		76.10 ± 5.51	7.70 ± 1.31	101 ± 12.30
Kaikōura Peninsula (F)		72.40 ± 4.52	6.95 ± 1.26	95.8 ± 14.30
South Coast (all)	22nd and 24th	77 ± 4.73	8 ± 1.64	103 ± 17
South Coast (M)	Feb, 2026	78.60 ± 3.97	8.39 ± 1.66	106 ± 17.10
South Coast (F)		75.50 ± 4.92	7.60 ± 1.54	100 ± 16.40

2.3 Mātakitaki-a-Kupe/Cape Palliser kekeno population abundance update

Mātakitaki-a-Kupe/Cape Palliser, in the Wairarapa, was the first North Island colony where kekeno breeding re-established post-sealing (Dix 1993). The area within and around the colony is of great significance to Māori, containing the remains of stone walls, middens, pā and urupā sites, and

features tied to histories of Kupe and early Polynesian explorers (Greater Wellington Regional Council 2021).

As in Kaikōura, CDV has been detected in kekeno at Mātakitaki-a-Kupe/Cape Palliser, again associated with higher than typical mortality rates (Taylor et al. 2026). Additionally, high bycatch rates in the region have led to recommendations that a long-term monitoring programme is established for the colony (Pavanato et al. 2023). The Mātakitaki-a-Kupe/Cape Palliser kekeno colony is closest to FMA 2, on the Cook Strait (Figure 4), a region with high levels of kekeno bycatch, particularly in hoki (*Macruronus novaezelandiae*) trawls (Pavanato et al. 2023). Between the 2019/20 – 2025/26 (to date: 15/04/2026) fishing years, 410 kekeno captures have been reported in FMA 2, with 299 mortalities.

A mark-recapture and body condition study was planned for Mātakitaki-a-Kupe/Cape Palliser in January of 2026, but a forecast extreme weather event caused its postponement until January 2027. Therefore, the only new data available for this colony come from a preliminary direct count undertaken as part of a site recce on January 7th, 2026. This count was only intended to be an approximation to inform the subsequent planned work. It was coarse in nature, did not cover all available breeding space and did not account for pup mortality. However, it represents the most recent approximate pup production estimate for the colony, and establishes the breeding distribution in the 2025/26 season (Figure 4).

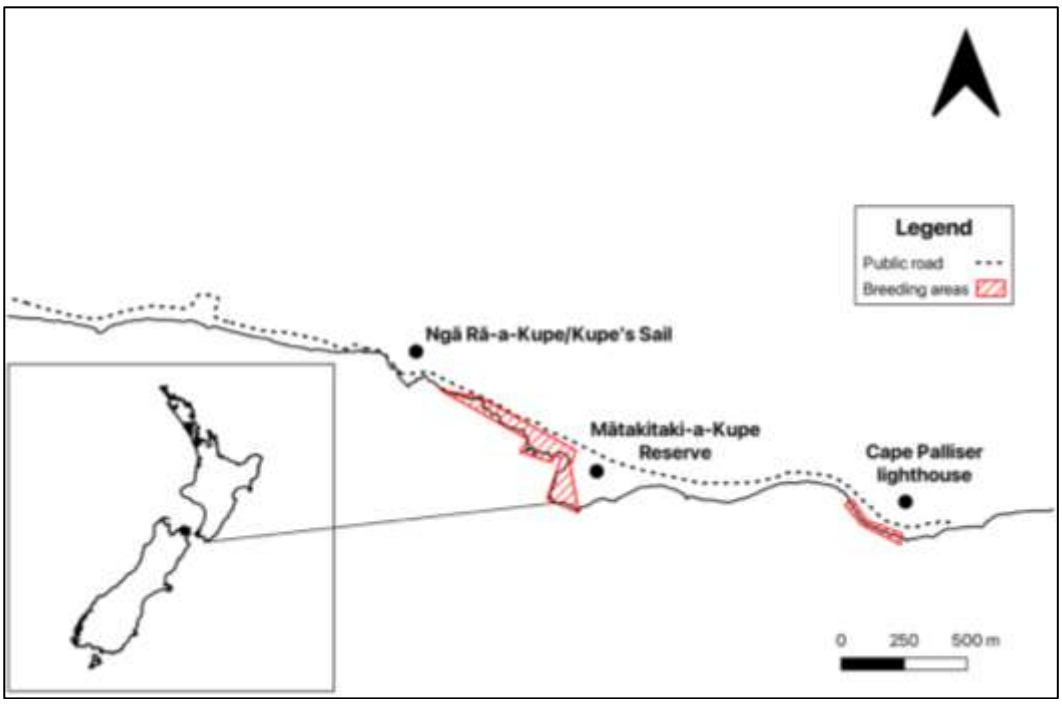

Figure 4 The distribution of kekeno breeding at Mātakitaki-a-Kupe/Cape Palliser in January 2026.

From the direct count conducted on January 7th, 2026, approximately 1,000 pups were estimated to have been produced at Mātakitaki-a-Kupe/Cape Palliser in the 2025/26 breeding season. Almost 80% of pups were counted between Ngā-Rā-o-Kupe/Kupe's Sail and the Mātakitaki-a-Kupe Reserve (Figure 4). The remaining ~20% were counted on the shoreline below the Cape Palliser lighthouse.

2.4 West Coast of the South Island kekeno population abundance update

Three kekeno colonies on the West Coast of the South Island (WCSI) – Wekakura Point, Cape Foulwind and Taumaka Island – are the subjects of a longitudinal monitoring programme. The programme began in 1991 at Cape Foulwind and Taumaka Island, and 1992 at Wekakura Point, due to concerns over high kekeno bycatch rates, particularly in hoki trawls (Roberts & Neale 2016). All three colonies are closest to FMA 7, where 402 kekeno have been recorded captured between the 2019/20 – 2025/26 fishing years (to date: 15/04/2026), including 235 mortalities (Ministry for Primary Industries 2026). Of the 402 captures, 244 were in trawls, 156 were in surface longlines and 2 were in set nets (Ministry for Primary Industries 2026). A previous report (Roberts & Neale 2016) demonstrated that all three colonies were experiencing declining pup production.

The WCSI kekeno monitoring programme has comprised consistently executed mark-recapture estimates and body condition analyses. A full description of the field methods and analyses undertaken can be found in Hall et al. (2026b). Generalized additive models (GAMs) used to assess the impact of year on live pup abundances had strong explanatory power, with the smooth terms for year highly significant for each of the colonies (Figure 5). Compared to each colony's maximum pup production estimate in the 1990s, the results from the most recent year (2025) represent declines of 83% for Wekakura Point, 71% for Cape Foulwind and 61% for Taumaka Island. All three colonies experienced particularly low pup abundances in 2000, 2010–2012 and 2023, with relative peaks in the mid-1990s, around 2005 and 2016–2020. The most recent pup abundance estimates are shown in Table 3, with overall trends shown in Figure 5.

Table 3 Live kekeno pup abundance estimates for the West Coast South Island study colonies in 2025

Colony	2025 live pup estimate (95% confidence interval)
Wekakura Point	186 (178–194)
Cape Foulwind	131 (122–140)
Taumaka Island	566 (555–577)

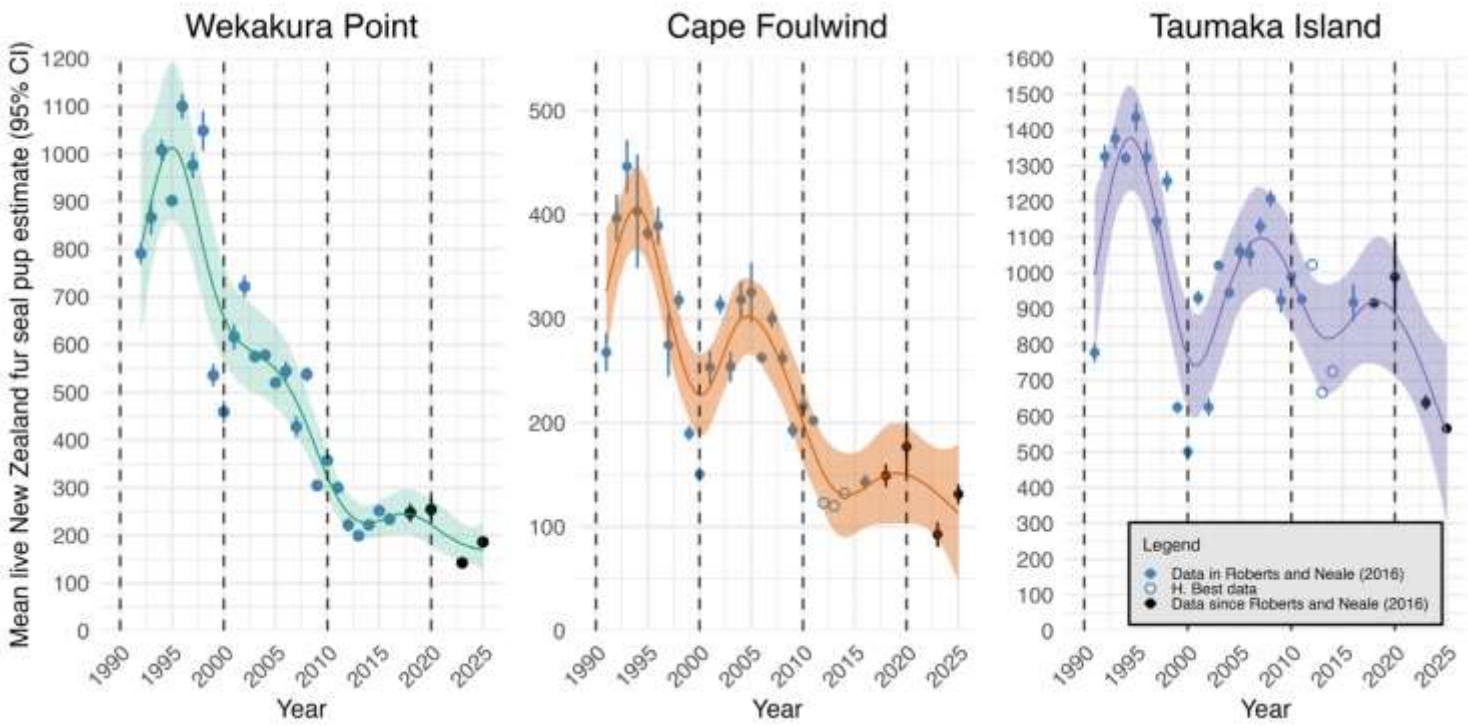

Figure 5 Mean live kekeno pup abundance estimates from the three West Coast study colonies. Smooths from the GAMs are overlaid, with shaded areas representing the 95% confidence interval. Hollow points are used for years when error ranges are unavailable. Figure from Hall et al. (2026b).

Between 1997 – 2025, on average, dead pups as a percentage of live pups were 12.7% (range 4.2–25.2%) at Taumaka Island, 2.3% (range 0–5.4%) at Wekakura Point and 3% (0–9%) at Cape Foulwind.

Pearson's product-moment correlations used to explore the relationship between pup mass and the inter-survey difference in live pup estimates found a weak, non-significant correlation between these estimates at Wekakura Point ($r = 0.19$, $p = 0.32$), and moderate, significant correlations at Cape Foulwind ($r = 0.51$, $p < 0.01$) and Taumaka Island ($r = 0.49$, $p < 0.05$).

2.5 Bounty Islands kekeno population abundance estimate

The Bounty Islands/Moutere Hauriri are closest to FMA 6, where 69 bycaught kekeno have been recorded between the 2019/20 – 2024/25 fishing years (to date: 15/04/2026), with 64 fatalities (Ministry for Primary Industries 2026). Sixty-five recorded captures were in trawls, and four in bottom long lines. The Bounty Islands' kekeno population was last assessed in 1994 using aerial photography (Taylor 1996). Drone images taken during a January 2024 survey of erect-crested penguins (*Eudyptes sclateri*) provided an opportunity to estimate pup production within the 'Main Group' of the Bounties archipelago. This is where ~93% of all Bounty Islands kekeno were observed by Taylor (1996) and ~97% of all Bounty Islands kekeno were observed by Mattern et al.

(unpublished data) in a survey conducted in November 2022 (immediately prior to the start of pupping).

A full description of the methods for the drone surveys and subsequent analyses can be found in Hall et al. (2026a). Total pup production for the Main Group in the 2023/24 breeding season was estimated at 4,252 – 4,341 pups (mean ± SD). Breeding, evidenced by pup presence, was observed on 8/13 of the islands/islets surveyed. Depot Island, had the highest pup count (1,910 ± 10 SD), followed by Penguin Island (809 ± 6 SD) and then Proclamation Island (491 ± 1 SD).

2.6 Discussion - estimates of kekeno abundance at specific sites in New Zealand

2.6.1 Summary of findings

The results described in Sections 2.2 – 2.5 provide important updates to our understandings of kekeno abundance and population trends in four regions. The time series of data available from each region varies considerably, impacting the inferences that can be derived. The WCSI study is New Zealand's most comprehensive kekeno monitoring programme (Hall et al. 2026b). In Kaikōura, recent abundance monitoring has consistently occurred since the 2022/23 breeding season, before which it was sporadic since 2005 (Boren et al. 2006). The updated Bounty Islands and Mātakitaki-a-Kupe/Cape Palliser estimates provide single year snapshots. Long-term data sets are particularly beneficial for advancing ecological knowledge and informing relevant policy (Hughes et al. 2017), and care should be taken not to over-interpret the results from any of the shorter-term/snapshot studies (Kaikōura, Mātakitaki-a-Kupe/Cape Palliser and The Bounty Islands) presented here. The potential dangers of doing so can be illustrated hypothetically using the WCSI study. For example, had WCSI monitoring occurred only between 2015 – 2020, pup production would have appeared stable or increasing, masking the long-term declines. The diverging colony dynamics in the regions studied also highlight the importance of not transposing trends from one region onto another.

The results from the past four seasons of monitoring in Kaikōura (Appendix 2), are shown in Figures 6 – 10. All colonies surveyed in the 2025/2026 breeding season experienced considerable increases in pup production compared to 2024/2025, with increases also observed from 2023/2024 – 2024/25. For example, Ōhau Point produced 5,346 ± 102 (SE) pups in the 2025/26 breeding season, compared to 3,137 ± 87 (SE) in 2024/25, an increase of 70.4%. The combined pup production estimates from each colony/sub-colony in 2025/26 was 9,465 – 10,027 compared to 5,507 – 5,951 in 2024/25. The pup production increases in the two most recent seasons come after a significant pup production decline between 2022/23 and 2023/24, associated with a starvation event impacting multiple age classes, and the discovery of CDV (Taylor et al. 2026). It is possible, therefore, that the recent increases represent population responses to these stressors. Pup production increases were recorded in Galápagos fur seals (*Arctocephalus galapagoensis*) and

Galápagos sea lions (*Zalophus wollebaeki*) in years following population declines associated with reduced prey availability (Páez-Rosas et al. 2021) - a driver known to impact otariid fecundity (Boyd 1991, Geeson et al. 2022). Similar responses have been observed following disease outbreaks, such as after a phocine distemper virus outbreak which caused mass mortality in northern European harbour seals (*Phoca vitulina*). This episode was followed by several years of high pup production (Brasseur et al. 2018), which were not maintained long-term. Future data collection in Kaikōura is required to determine whether the post-UME increases are sustained.

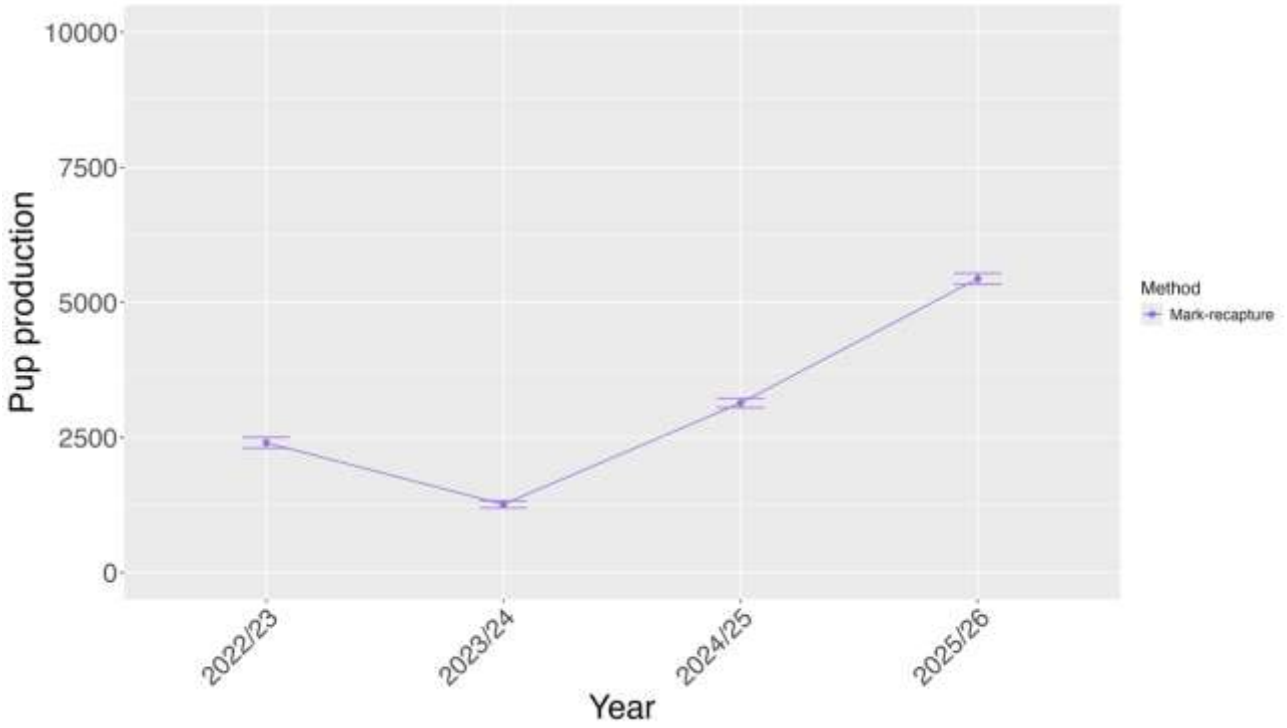

Figure 6 Kekeno pup production trends at Ōhau Point from 2022/23 – 2025/26. Error bars represent standard errors.

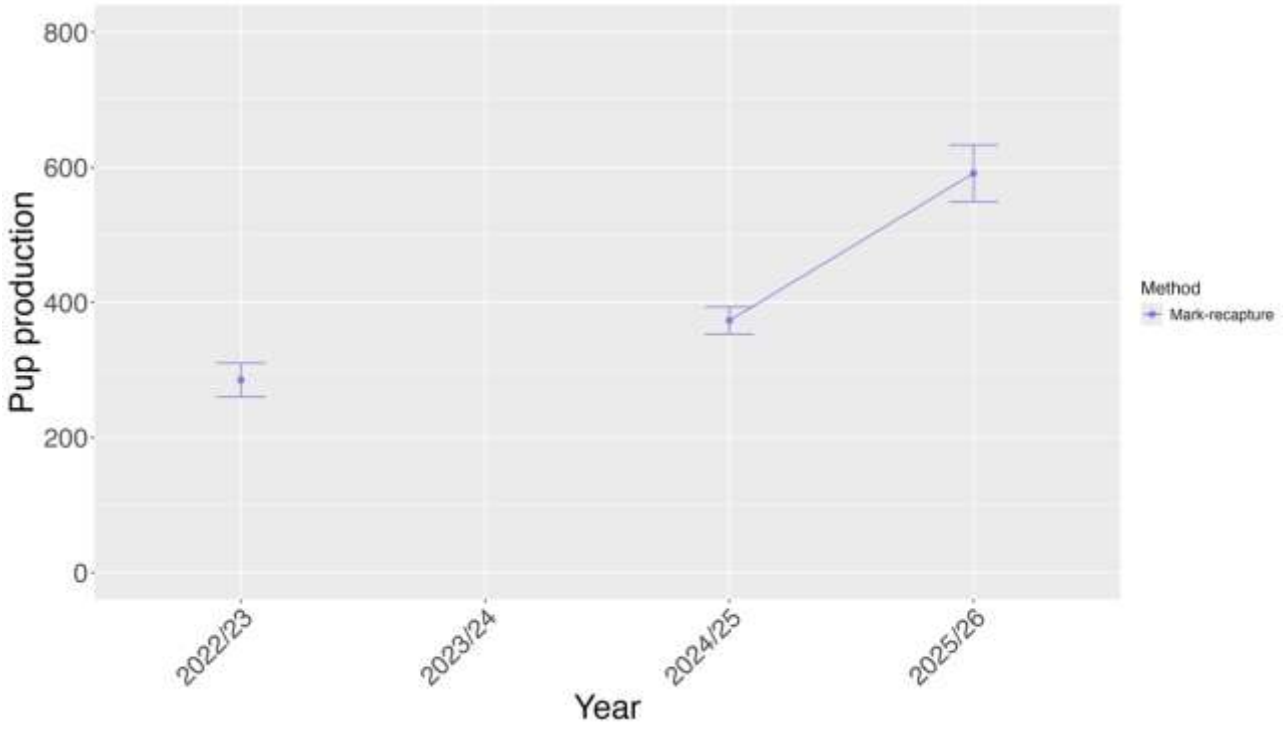

Figure 7 Kekeno pup production trends at South Rakautara from 2022/23 – 2025/26. Error bars represent standard errors.

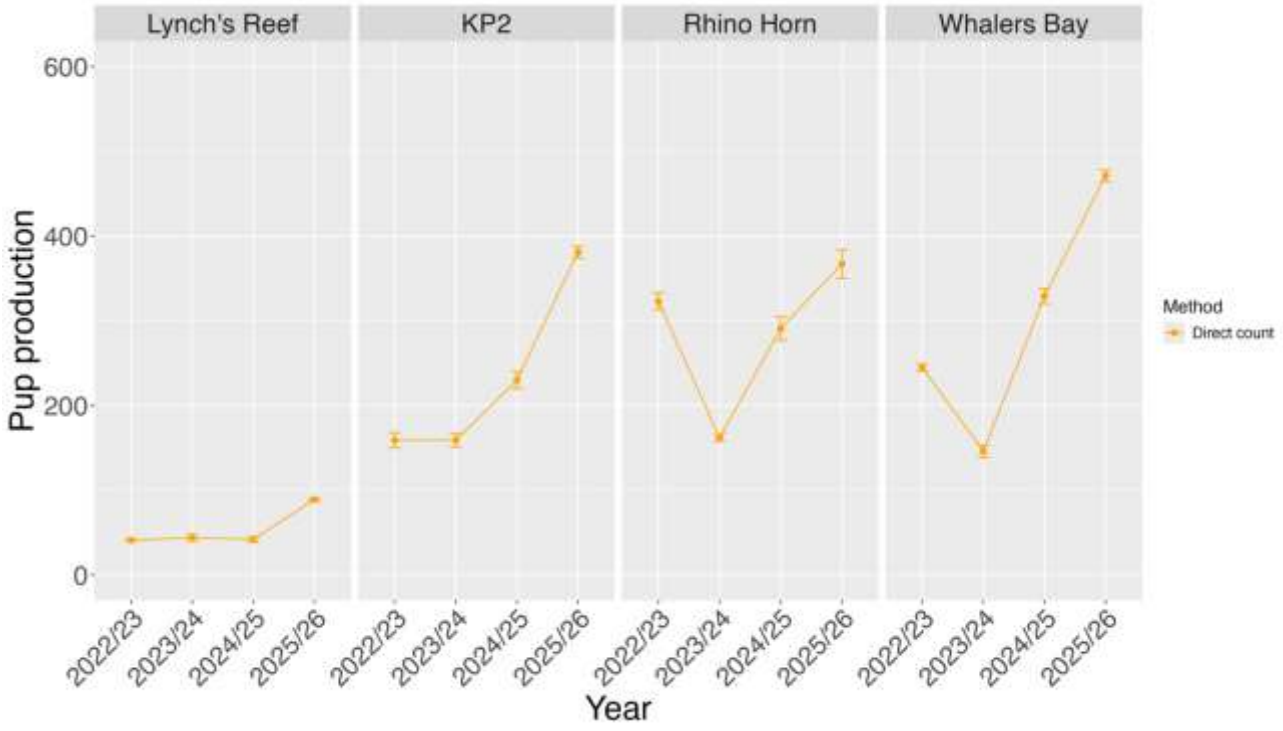

Figure 8 Kekeno pup production at the Kaikōura Peninsula colony (by sub-colony) from 2022/23 – 2025/26. Error bars represent standard errors.

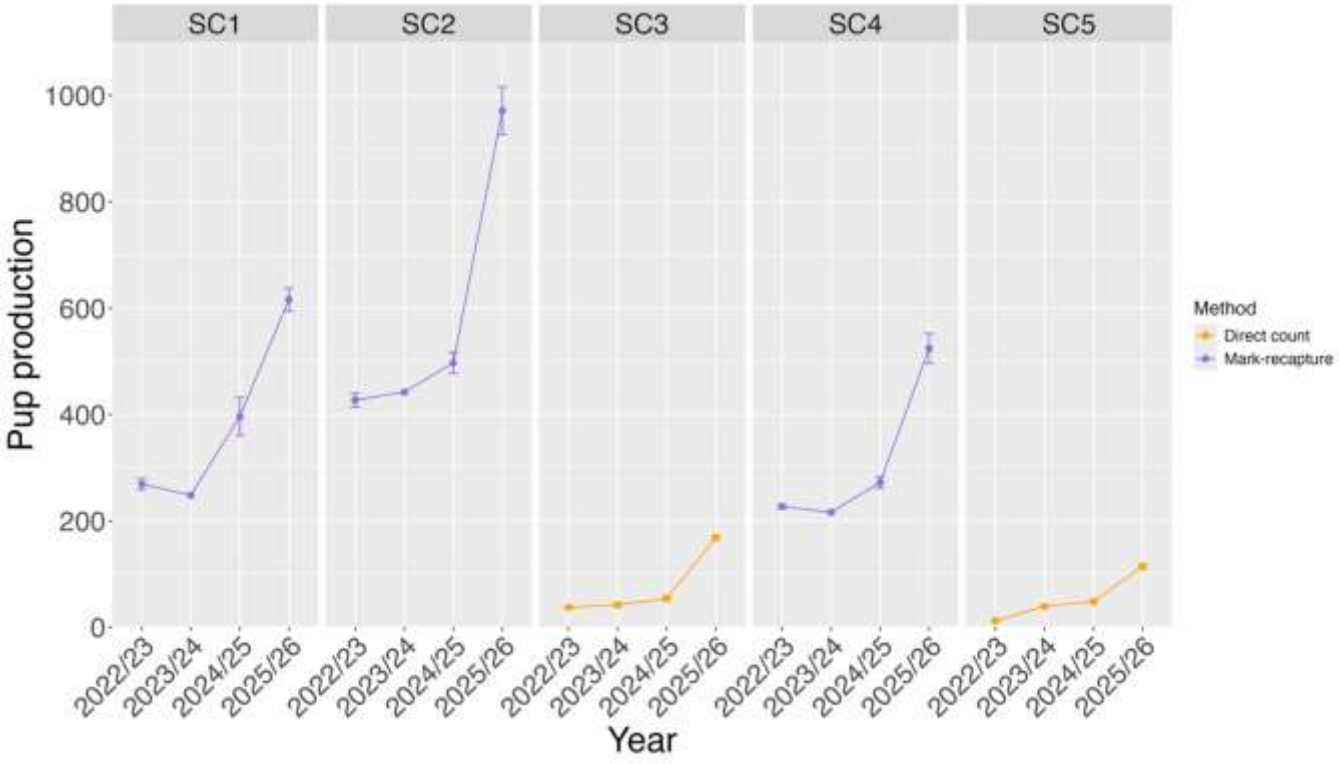

Figure 9 Kekeno pup production at the South Coast colony (by sub-colony) from 2022/23 – 2025/26. Error bars represent standard errors.

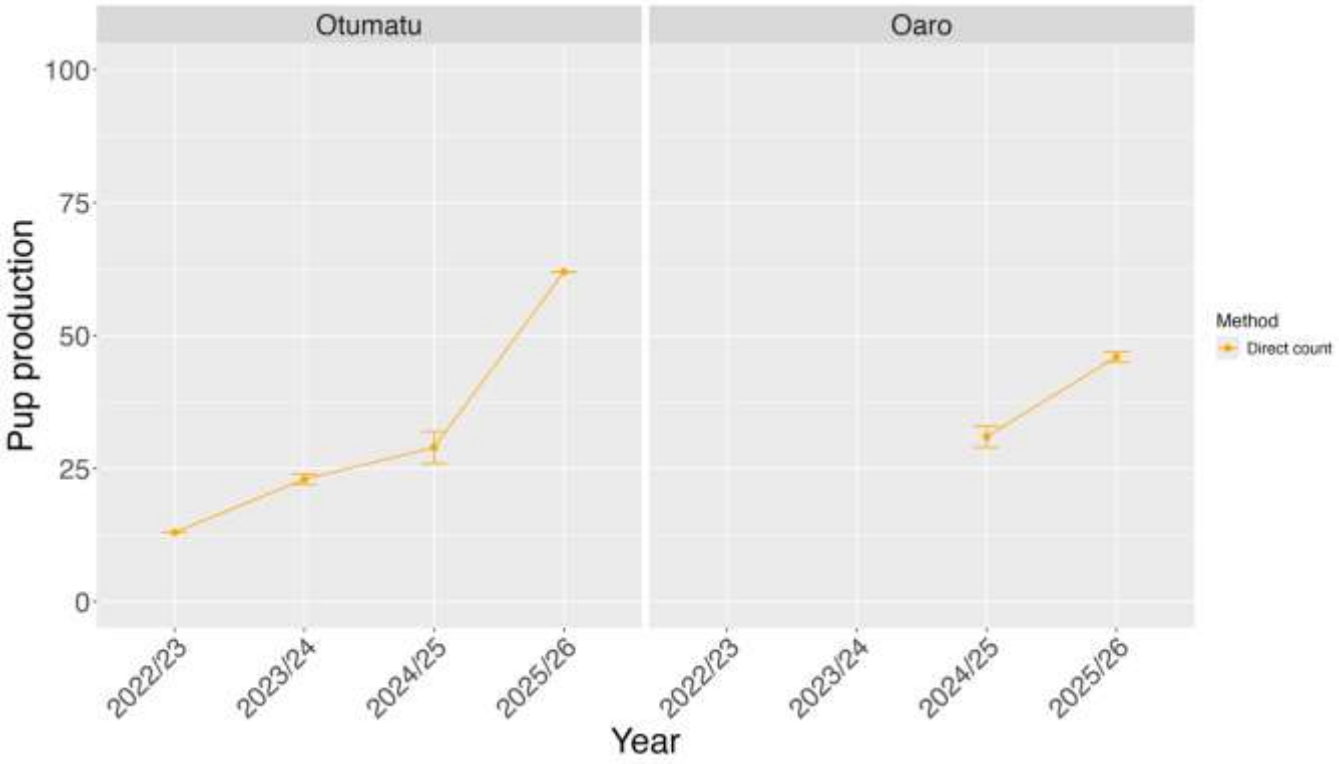

Figure 10 Kekeno pup production at the Otumatu and Oaro colonies from 2023 – 2026. Error bars represent standard errors.

On the WCSI, long term abundance and condition trends can be confidently identified due to the long time series of data available (Hall et al. 2026b). Recorded pup abundances on the WCSI study show sharp, episodic drops as well as long-term declines (Figure 5), pointing to both abrupt pup production declines and poor survival (Hall et al. 2026b). Associations between periods of low pup abundance and reduced pup condition suggest that foraging efficiency is among the drivers for the declines, given links between otariid female condition and both pup production (Boyd 1991, Geeson et al. 2022) and survival (Wall et al. 2023, Cruz-Vallejo et al. 2024). On the WCSI, kekeno prey species such as hoki have shifted into deeper, cooler waters (Devine et al. 2025), potentially forcing lactating female kekeno to exert more energy foraging, which could decrease pup production and negatively impact pup provisioning, as observed in other otariids (McKenzie et al. 2005, Gibbens et al. 2010). The sustained nature of the observed declines suggests that reduced female survival across multiple age classes may also be a driver (Hall et al. 2026b), as episodic pup production declines would be unlikely to produce this persistent trajectory. Female survival is of greater importance than male survival to population trajectories in polygynous otariid species (Hamilton et al. 2019, Iriarte & Winter 2025), as only certain adult males will mate. Thus, the loss of females can be especially impactful, particularly on small populations/sub-populations (Leung et al. 2012, Goldsworthy et al. 2022). Additional studies (described in Section 4) are required to determine what, if any, impact bycatch may be having on these declining colonies, so that appropriate management responses can be established. To facilitate this, colony monitoring would ideally revert to an annual basis.

The recent updates to the population estimates acquired in 2024 for the Bounty Islands and 2026 for Mātakitaki-a-Kupe/Cape Palliser provide single year snapshots. In the case of Mātakitaki-a-Kupe/Cape Palliser, the approximation of ~1,000 pups in the 2025/26 breeding season should be treated with caution, for the reasons given above. The only other recent assessments occurred in February 2022, when 335 pups were counted (Pavanato et al. 2023), and February 2023 when ~600 pups were counted (L. Angus, pers. comm.). As in 2026, the 2023 survey did not cover the whole colony (L. Angus, pers. comm.). The postponed mark recapture study scheduled for January 2027 will provide a reliable new baseline.

Prior to 2024, the most recent kekeno abundance assessment for the Bounty Islands was in 1994 (Taylor 1996), when ~4,380 pups were estimated from aerial photographs. This is similar to the 4,252 – 4,341 pups estimated in 2024, however direct comparisons are ill-advised due to methodological limitations that may impact the reliability of Taylor's (1996) estimate (Hall et al. 2026a). As such, the 2024 estimate should serve as a new baseline. The 2024 Bounty Islands update is particularly important in the wider context of New Zealand's sub-Antarctic, as kekeno abundances on most other sub-Antarctic islands have not been assessed since the 1980s or 1990s (Moore & Moffat 1990, Carey 1998). The only other known recent kekeno abundance data from the sub-Antarctic come from Reef Point on Antipodes Island, where ~193 pups were counted in January 2024, again from drone images (Mattern 2024). The ecological distinctiveness of the sub-Antarctic

(Selkirk 2007) means kekeno population trends from mainland New Zealand should not be transposed onto the sub-Antarctic.

2.6.2 The use of drones in kekeno monitoring

The 2024 Bounty Islands survey highlights how drones can facilitate multi-species assessments, and enable kekeno monitoring in difficult-to-access locations. A trial study is currently underway using drone images captured on the Kaikōura Peninsula in January 2026 to assess the possibility of using deep learning to identify and count pups. There are other known sites in New Zealand – such as the Nee Islets and Chalky Island in Fiordland (Chilvers 2021) – where drones could likely be used for kekeno population assessments, although the species' preference for complex habitats (Ryan et al. 1997, Bradshaw et al. 1999) means this technology will likely not be suitable for all colonies (Gooday et al. 2018).

2.6.3 Conclusions

The results described in Sections 2.2 – 2.5 provide several important insights. First, not all New Zealand kekeno colonies are growing or stable, as evidenced from the continued declines at the WCSI colonies (Hall et al. 2026b). Since the cessation of commercial sealing and formal protection of the species, the understanding is that kekeno abundance has been increasing and the species is re-establishing colonies in their pre-exploitation range (Lalas & Harcourt 1995, Lalas & Bradshaw 2001, Boren et al. 2006, Bouma et al. 2008, Emami-Khoyi et al. 2018, Fisheries New Zealand 2025). While this may be the current trend in some regions, it is clearly not a universal trajectory, and greater nuance is required when discussing kekeno abundance trends. This is particularly important in the context of fisheries' interactions, as the impacts and risks of bycatch are likely to be higher for particular colonies than at the national population level (Fisheries New Zealand 2025). The need for this nuance is reinforced by the lack of consistent kekeno abundance monitoring across most of New Zealand, and care should be taken not to extrapolate results from regularly monitored colonies/regions to others where data are not consistently collected.

For the WCSI colonies, where long-term declines are now confidently established, there is a pressing need to gather data on the drivers of these dynamics, including foraging and diving behaviour, diet composition, and survival and fecundity rates (see Section 4).

2.6.4 Recommendations for future kekeno monitoring at Kaikōura, WCSI, Mātakitaki-a-Kupe/Cape Palliser and the Bounty Islands

Kaikōura:

- The monitoring programme that has run from 2022/23 – 2025/26 should be continued annually to promote understandings of the population's trajectory and health.

Mātakitaki-a-Kupe/Cape Palliser:

- The survey postponed to January 2027 should form a baseline for future monitoring at the colony.

The WCSI colonies:

- The monitoring programme that has run at Wekakura Point, Cape Foulwind and Taumaka Island since 1991/1992 should continue and ideally revert to an annual basis.

The Bounty Islands:

- Pup counts should be conducted if future drone surveys include kekeno breeding areas.

3 An updated nationwide population estimate for kekeno

3.1 Introduction

Abundance estimates typically form part of best practice approaches to understanding and managing marine mammal bycatch, allowing the risk to a species' persistence to be quantitatively assessed (Moore et al. 2021, Iriarte & Winter 2025). There have been few empirical attempts to estimate the nationwide abundance of kekeno in New Zealand. An estimate of ~200,000 is mentioned in Emami-Khoyi et al. (2018), and is quoted elsewhere (Fisheries New Zealand 2025), originating from Goldsworthy and Gales (2008). The only attempt at a nationwide census of New Zealand's kekeno was undertaken by Wilson (1981) in the 1970s, which provided an estimate of 30 – 50,000 kekeno. However, not all colonies were assessed. Since then, subsequent nationwide abundance estimates have been calculated by combining irregular, localised pup counts as they become available with the figures derived by Wilson (1981) (Lalas & Bradshaw 2001).

Rather than relying on data from Wilson (1981), Hall et al. (2025b) trialled different approaches to modelling contemporary nationwide kekeno abundance. This involved collating available data on kekeno abundance from around New Zealand (see catalogue in Appendix 3), up to and including 2024, and forward projecting outdated figures. A stage-structured matrix model (Lefkovitch 1965, Caswell 2001), which included a measure of basic density-dependence, was adopted to forward project outdated estimates (Hall et al. 2025b). This was used in combination with unmodelled abundance estimates for colonies that did not meet the criteria for modelling (see below) to provide a total nationwide abundance estimate (Hall et al. 2025b). The model produced an estimate of between ~182 – 239,000 kekeno in New Zealand in 2024. This section adopts Hall et al.'s (2025b) model, incorporating data collected since it was run, to provide a nationwide abundance estimate for kekeno in New Zealand in 2026.

3.2 Methods

3.2.1 Hall et al.'s (2025b) kekeno nationwide population model

Of the approaches trialled by Hall et al. (2025b) the 'density-dependent projection model' was found to be most reliable. A full description of this model can be found in the original paper (Hall et al. 2025b), with a summary provided below. Modifications for the 2026 update are described in Section 3.2.2.

The model can be described as:

$$\boldsymbol{n}_{t+1} = \mathbf{A}\mathbf{n}_t$$

With $\mathbf{n}_t$ and $\mathbf{n}_{t+1}$ representing the number of individuals in each stage at time intervals *t* and *t*+1 respectively, and **A** representing the population matrix:

$$\mathbf{A} = \begin{pmatrix} 0 & 0 & \omega \\ \varphi_1 & \varphi_2\sigma & 0 \\ 0 & \varphi_2(1-\sigma) & \varphi_3 \end{pmatrix}$$

Table 4 Parameter description for the model used to estimate nationwide kekeno abundance

Symbol	Parameter
φ_1	Pup survival
φ_2	Juvenile survival
φ_3	Adult survival
ω	Adult fecundity
σ	Juvenile transition

Three stages were selected for use in the model: pups (0 – 1), juveniles (1 – 4) and adults (4+). Given the relative lack of kekeno survival and fecundity data from New Zealand (Mattlin 1978, Dickie & Dawson 2003), such data were also sourced from kekeno in Australia, and other otariid species. The vital rate data used in the matrix permutations are provided in the Supplementary Information associated with Hall et al. (2025b). The transition rate from the juvenile – adult stages was calculated as:

$$1 - \frac{1}{x_j - x_i}$$

Where x_i is the first age class in stage *i* and x_j is the first age class in stage i + 1 (Fujiwara & Diaz-Lopez 2017).

Hall et al. (2025b) used the following criteria to select colonies/regions for projection:

- The colony/region does not have an estimate from the last five years
- The colony/region has more than 100 pups at its last count
- The count is not listed as a "guesstimate" in the literature
- The count is not from a region/colony where the population is thought to have stabilised.

For colonies that did not meet these criteria ("unmodelled colonies"), the most recently available abundance data were used without forwards projection. For unmodelled colonies, if population abundance was provided in the literature these figures were used directly. If only pup counts were available, these were scaled up to population estimates using Goldsworthy and Page's (2007) pup to population multiplier (4.76).

Five matrix permutations were adopted using different combinations of the demographic rate data. These permutations were, 'Low', 'Average', 'High', 'High Survival, Low Fecundity' (HSLF) and 'High Fecundity, Low Survival' (HFLS). The *stoch.proj* (Stubben 2020) function from the *pop.bio* R package (Stubben & Milligan 2007) was used to prescribe a weighted probability vector for matrix selection across 1,000 model iterations for each colony/region projected. Expert opinion suggests that an 'average' vital rate matrix should occur more frequently than any other matrix, and thus it was assigned a greater probability weighting, with the remaining four matrices assigned equal weightings. To demonstrate the impacts of varying the weightings, Hall et al. (2025b) tested three weightings, and found that the "high difference" model (whereby the Average matrix had a 0.96 probability of selection, and all others had a 0.1 probability) performed best. As such, this weighting was adopted here. Pup counts were sourced from the literature, and the *stable.stage* function in *pop.bio* (Stubben & Milligan 2007) was used to calculate starting abundances for the juvenile and adult cohorts. A simple measure of density dependence was included using *stoch.proj* (Stubben 2020), whereby *nmax* (an upper ceiling the modelled estimates could not exceed) was set. For comparability with Hall et al. (2025b), the same *nmax* values were used. The sum of the modelled colonies was then added to the sum of the unmodelled colonies to produce the overall abundance estimate.

3.2.2 Modifications to the model for the 2026 update

Modelled colonies

Hall et al. (2025b) did not forward project any estimates derived from assessments that occurred within five years of their study, as recent estimates were deemed more reliable than those derived

from the model, given the input limitations (see below). For the 2026 update, this rule was extended so that any estimate collected since 2020 was used without forward projection. This modification impacted two colonies/regions: Admiralty Bay where 100 pups were estimated in 2021 (L. Boren, pers comm) and Lower Fiordland, where a regional population estimate of ~14,000 – 24,000 was derived in 2021 (Chilvers 2021). Thus, the colonies/regions with totals that were forward projected to 2026 are the same as those modelled in Hall et al. (2025b), with the exception of the Bounty Islands, given a new abundance estimate is now available (Hall et al. 2026a).

For the modelled colonies/regions, the projection interval used was equal to the number of years since the last population assessment and 2026.

Unmodelled colonies

Since Hall et al. (2025b), available updated regional/colony abundance estimates come from Kaikōura, Mātakitaki-a-Kupe/Cape Palliser, the WCSI colonies, The Bounty Islands (Sections 2.2 – 2.5), The Tasman region (Department of Conservation, unpublished data) and Reef Point on Antipodes Island (Mattern 2024) (Table 5). Boat-based counts of kekeno in the Tasman Region (Separation Point to Onetahuti, Tonga Island, Pinnacle Island, Adele Island and Fisherman Island) have occurred since 2022/23. The data from March 25th, 2026 were used here, when 168 pups were counted across all sites (Department of Conservation, unpublished data). Mattern (2024) estimated ~193 pups present at Reef Point on Antipodes Island in January 2024, and this figure was used in the updated nationwide abundance estimate.

The most recent pup estimates for both the modelled and unmodelled colonies are presented in Appendix 3. Table 5 summarises colonies/regions where pup production data have been collected since Hall et al. (2025b).

Table 5 Pup and population abundance estimates for unmodelled kekeno colonies surveyed since Hall et al. (2025b)

Colony/Region	Pup estimate (year)	Estimate source	Derived population estimate (pup estimate * 4.76)
Mātakitaki-a-Kupe/Cape Palliser	~1000 (2026)	Section 2.3	4,760
Tasman Region (Separation Point to Onetahuti, Tonga Island, Adele Island, Pinnacle Island, Fisherman Island)	168 (2026)	Department of Conservation (unpublished data)	800
Kaikōura*	10,413 – 11,023 (2026)	Section 2.2	49,566 – 52,469
Wekakura Point	178 – 194 (2025)	(Hall et al. 2026b)	847 – 923
Cape Foulwind	122 – 140 (2025)	(Hall et al. 2026b)	581 – 666
Taumaka Island	555 – 577 (2025)	(Hall et al. 2026b)	2,642 – 2,747
Reef Point (Antipodes Island)	~193 (2024)	(Mattern 2024)	919
Bounty Islands	4,252 – 4,341 (2024)	(Hall et al. 2026a)	20,237–20,663

* To be comparable with the nationwide abundance estimate in Hall et al. (2025b), the Kaikōura pup estimate provided here was calculated using the multipliers described in Hall et al. (2024), and therefore differs from the result presented in Section 2.2.2.1 and Appendix 2.2. The multipliers rendered the results from Kaikōura colonies/sub-colonies assessed via direct counts in the 2025/26 field season comparable with those assessed via mark-recapture. See Hall et al. (2024) for details of the multipliers.

3.3 Results

The total estimated pup abundance from the unmodelled colonies/regions was 27,337 – 33,451. When the pup estimates from these colonies/regions were subjected to Goldsworthy and Page's (2007) population multiplier, a total unmodelled kekeno abundance of 128,875 – 162,828 was estimated. For the modelled colonies, total estimated pup abundance was 20,494 – 23,485, and total estimated kekeno abundance was 79,661 – 88,816 (Table 6).

Table 6 Updated 2026 estimates for modelled kekeno colonies.

Colony/Region modelled	Most recent pup count and source	2026 modelled pup estimate (mean ± SD)	2026 modelled population estimate (mean ± SD)
Stephens Island	276 (1994)[a]	2,717 – 3,226	10,541 – 12,226
Black Reef	200 (2010)[b]	526 – 657	2,047 – 2,945
Charleston	476 (2010)[b]	1,646 – 1,835	5,665 – 6,993
Cascade Point	1,466 (2009)[c]	2,926 – 3,188	11,576 – 11,912
Long Reef, Martins Bay	163 (2009)[c]	537 – 685	2,084 – 2,607
Yates Point	1,228 (2009)[c]	2,923 – 3,188	11,552 – 11,927
Stewart Island Region	2,694 (2003)[d]	7,183 – 7,809	28,355 – 29,160
Campbell Island	126 (1988)[e]	2,036 – 2,897	7,841 – 11,046
Modelled colony total		**20,494 – 23,485**	**79,661 – 88,816**

[a] (Taylor et al. 1995), [b] (Baird 2011), [c](Baker et al. 2009), [d] (Watson et al. 2015),[e] (Moore & Moffat 1990)

Combined, the totals for the modelled and unmodelled colonies provided a total pup abundance estimate of 47,831 – 56,936 pups, and a total nationwide kekeno abundance estimate of 208,536 – 251,644 individuals in 2026.

3.4 Discussion

The estimates produced through this modelling approach provide the first empirical attempts to update understandings of kekeno abundance in New Zealand since the 1970s census (Wilson 1981). The 2026 nationwide kekeno abundance estimate provided here of 208,536 – 251,644 individuals is larger than the estimate of 181,646 – 239,473 provided by Hall et al. (2025b) for 2024. However, limitations in the available kekeno abundance and demographic data, and their impacts on the model, mean that the difference between these two figures should not be interpreted as an increase in the nationwide kekeno population size since 2024 (Hall et al. 2025b) (see below).

3.4.1 Study limitations

The most obvious impediment to estimating New Zealand's kekeno population size is the lack of consistent abundance data collection in parts of the country. This has three principal effects: (1) some colonies have never been formally assessed and thus could not be included in this estimate, (2) for many colonies, the available data are outdated and (3) different methodologies have been used to estimate pup production at different colonies over time, with different associated uncertainties. As discussed in Section 4 (below), future attempts to monitor kekeno in New Zealand should include nationwide surveys that would provide data on hitherto unassessed colonies. Certain areas in particular have received minimal or no survey effort – for example, the Chatham Islands and much of the North Island. The modelling approach sought to mitigate the issue of outdated data by forward projecting colonies with outdated abundance estimates. However, for the reasons given in the modelling criteria in Section 3.2.1, some colonies with outdated data could not be reliably modelled, meaning their most recent counts were used without projection. The available data also necessitated the use of counts derived from different methods (e.g. mark-recapture vs. direct counts). While the incorporation of counts derived from multiple methodologies is done in largescale monitoring programs for similar species overseas (for example, Australian fur seals (McIntosh et al. 2022)), there is variability in the precision of these methods which should be acknowledged (for example, mark-recapture is considered more precise at colonies with significant numbers of pups and/or colonies with cryptic terrain). The constraints of the available data, and particularly the incomplete nature of the current colony catalogue, means that the nationwide population figure produced here (208,536 – 251,644) may underestimate the true size of New Zealand's kekeno population. That said, given the observed declines at some kekeno colonies over the last 30 years – for example, the WCSI (Hall et al. 2026b), it is possible that some of the model projections overestimated true kekeno abundance.

Another impact of spatially inconsistent kekeno monitoring effort is that updated results from regularly monitored colonies/regions exert an outsized influence on the nationwide figure. For example, a substantial portion of the difference between the nationwide abundance estimate provided by Hall et al. (2025b) and the updated figure provided here reflects changes in the estimated size of the Kaikōura population in the intervening period. Hall et al. (2025b) used the most contemporary regional abundance estimate for Kaikōura available at the time of writing: 20,944 – 27,518 kekeno from the 2022/23 breeding season (Hall et al. 2024). This update used the Kaikōura pup abundance data collected in February 2026 (subjected to the same direct-count to mark-recapture multipliers as Hall et al. (2024)), giving a population estimate of 49,566 – 52,469

kekeno[3]. The spatial biases in kekeno monitoring effort, and the impacts of this on the nationwide abundance estimate, creates a risk that certain regions, such as Kaikōura, become unintended proxies for the national population and mask divergent trends elsewhere – such as on the WCSI. Understanding the causes of regional or colony level declines, such as on the WCSI (Hall et al. 2026b), is important, even if the nationwide kekeno abundance was found to be increasing (a trend that cannot currently be confirmed). In part, this is because anthropogenic impacts on kekeno, such as fisheries bycatch, are likely greater for particular colonies/regions than they are at the national population scale (Fisheries New Zealand 2025). Additionally, high trophic level predators such as otariids play important roles in their local marine ecosystems (Heithaus et al. 2008, Rupil et al. 2022). As such, kekeno could be used as important sentinels for monitoring local marine ecosystem health, as has been achieved with California sea lions (*Zalophus californianus*) (Elorriaga-Verplancken et al. 2016, Cruz-Vallejo et al. 2024).

Another important limitation to our approach is that abundance increases are assumed for the modelled colonies. While an upwards trajectory has been true for the vast majority of monitored kekeno colonies in both New Zealand and Australia for decades (Lalas & Harcourt 1995, Lalas & Bradshaw 2001, Boren et al. 2006, Campbell et al. 2014, MacDiarmid & Lalas 2014, Shaughnessy & Goldsworthy 2015, Watson et al. 2015, Hall et al. 2024, 2025a), the WCSI study, and data from Sandymount (Appendix 4) show that some colonies are declining (Hall et al. 2026b). Limited understanding of the drivers of the observed declines on the WCSI, and their impacts on demographic rates, means that this information could not be factored into the model. The model could be improved if comprehensive data on these parameters were collected for New Zealand's kekeno. Ideally, it would be possible to vary these demographic inputs, as well as their impacts on density dependence, for different colonies/regions, to better reflect how these parameters vary through space and time (Bradshaw et al. 2000a, Raithel et al. 2007, Warlick et al. 2023).

3.4.2 Recommendations for future nationwide kekeno abundance estimates

Implementing a coordinated, regular, nationwide monitoring programme for New Zealand's kekeno, building on the success of the WCSI programme in tracking the trajectory and population health of the subject colonies, is an important next step in improving our understandings of the species and the threats it faces. In particular, effort should be made to provide estimates for colonies that have never been assessed before and therefore could not be included in the estimate provided here.

For the WCSI colonies in particular, where a long-term data set exists, there is an opportunity to gain insights into the drivers of the observed declines, most importantly by investigating the

[3] Note - the use of multipliers from Hall et al. (2024) to render direct count results more comparable with mark-recapture results was done to facilitate comparability between this estimate and that from Hall et al. (2025b). In future, it is recommended that direct count – mark recapture multipliers are not adopted and a single method is consistently adopted for each site.

pressures kekeno face at sea. A future nationwide abundance monitoring programme should seek to incorporate data collection on these parameters, as discussed in Section 4.

4 A framework for future kekeno population monitoring

4.1 Introduction

This section presents a proposed framework for future kekeno monitoring, drawing on best practice from international pinniped monitoring programmes. The recommendations are informed by current knowledge of kekeno and highlight key research opportunities.

4.2 An overview of kekeno research and knowledge gaps

Substantial long-term effort has documented kekeno distribution, abundance and pup condition around New Zealand (Wilson 1981, Taylor 1982, 1992, 1996, Dix 1993, Lalas & Harcourt 1995, Taylor et al. 1995, Bradshaw et al. 2000a b, Boren et al. 2006, Best et al. 2008, Bouma et al. 2008, Watson et al. 2009, 2015, MacDiarmid & Lalas 2014, Chilvers 2021, Hall et al. 2024, 2025a, 2026b). While there have been spatial biases in where monitoring has taken place (Lalas & Bradshaw 2001, Boren et al. 2006), these studies have provided vital baseline knowledge and insights into population status, particularly on the WCSI, as described in Hall et al. (2026b). Further studies analysing kekeno marine distribution and foraging behaviour would provide valuable insights which would inform effective management and conservation strategies.

There have been two long-term abundance monitoring programmes for New Zealand's kekeno – the WCSI programme at Wekakura Point, Cape Foulwind and Taumaka Island (Hall et al. 2026b) and a mark-recapture study at the Sandymount colony on the Otago Peninsula (Seddon et al., unpublished data). The WCSI programme ran almost every year from 1991/1992 to 2016, and every 2 – 3 years since. The Sandymount programme ran consistently between 1995/1996 – 2022/2023 (Seddon et al., unpublished data). Additional colonies in south-eastern South Island were previously regularly assessed over multiple decades (MacDiarmid & Lalas 2014). At Sandymount, the maximum recorded pup production estimate was in 2005, when 163 pups were estimated (95% CI 141 – 197), with the most recent estimate (2023) being 29 (95% CI 28 – 31) (Seddon et al., unpublished data). As on the WCSI (Hall et al. 2026b), there have been periods of both increase and decrease in estimated Sandymount pup production, however, since 2009, the overall trend has been declining pup production (Seddon et al., unpublished data). The full-time series from the Sandymount mark-recapture study are reproduced with permission from P. Seddon in Appendix 4. Assessments in other locations have been less regular, and short-term or single-year studies have been common. For example, data on pup production at the Bounty Islands comes from studies in 1980 (Taylor 1982), 1994 (Taylor 1996) and 2024 (Hall et al. 2026a). On the WCSI, excluding the three regularly monitored colonies (Hall et al. 2026b), many of the most recent pup count estimates

come from an aerial survey in 2009 (Baker et al. 2009). Currently, there are spatial biases favouring central areas of the species' nationwide range, with comparatively few assessments in the North Island and the sub-Antarctic. The longitudinal WCSI programme (described in Hall et al. 2026b) provides a blueprint for how consistently executed colony monitoring can provide confidence in abundance trend identification. This approach could be implemented elsewhere in New Zealand to develop current understandings of population trajectories, and quantify the impacts of threats (Abraham et al. 2017, 2021).

Most of the currently available biologging data for New Zealand kekeno come from studies conducted in the 1990s on female kekeno in Otago (Harcourt et al. 1995, 2001, 2002) and the WCSI (Sinclair & Wilson 1994, Mattlin et al. 1998). Additionally, Boren (2005) used very high frequency (VHF) trackers to monitor female kekeno attendance patterns at Ōhau Point in 2004 and 2005, and data were also collected on the marine distribution and diving behaviour of kekeno at Ōhau Point (Kaikōura), Cape Foulwind (WCSI) and Tonga Island (Abel Tasman) between 2010 – 2012. The latter study is currently documented only in field reports (Meynier 2011, 2012), however work is underway to further analyse the collected data. Additional studies have been conducted on kekeno foraging and diving behaviour at colonies in Australia (Page et al. 2005b c, 2006, Baylis et al. 2005, 2008a b, 2012, Foo et al. 2019).

Boren (2010) and Baird (2011) provide comprehensive accounts of studies of kekeno diet in the late 20th century, but this subject has received less attention in recent decades. Important exceptions include scat and regurgitate analyses used to assess the relative importance of different prey species at two colonies on Banks Peninsula (Allum & Maddigan 2012) and at North East Island (The Snares) (Lalas & Webster 2014). Emami-Khoyi et al. (2016) used massive parallel sequencing to identify prey items from kekeno scats, and Galbraith and Chilvers (2025) analysed stable isotopes extracted from pup vibrissae to assess spatial variability in kekeno foraging. A valuable next step would be to conduct a multi-year, multi-site assessment of the biomass contributions of different prey species to kekeno diet (Emami-Khoyi et al. 2016). In other pinniped species, similar studies have been used to improve understandings of the potential for fisheries interactions (Wilson & Hammond 2019), and to elucidate changes in diet resulting from environmentally mediated effects on prey availability (Kliska et al. 2022, McHuron et al. 2025). There is evidence of shifts in kekeno prey distribution from the WCSI (Devine et al. 2025), potentially placing additional pressure on proximal colonies. Further study of kekeno diet has been conducted in Australia (Page et al. 2005a).

The collection of demographic data (e.g. survival and fecundity rates) has played an important role in pakake monitoring (Chilvers 2012, Chilvers & Meyer 2017), and in overseas pinniped monitoring programmes assessing the sustainability of bycatch rates (Goldsworthy et al. 2022), but there are currently little demographic data available for New Zealand's kekeno. Some information on pregnancy rates has been derived from kekeno bycaught in 1996 (Dickie & Dawson 2003), while Boren (2005) estimated fecundity at Ōhau Point based on sightings of females in 2003 – 2005. The

only published estimates of survival rates for New Zealand kekeno are for pups (Mattlin 1978, Lalas & Harcourt 1995, Bradshaw et al. 2003, Boren 2005), although survival data exist for adult female kekeno in Australia, along with additional reproductive rate data (McKenzie et al. 2005, 2007, McKenzie 2006).

4.3 Case studies of overseas pinniped monitoring programmes

The following case studies demonstrate coordinated approaches to pinniped monitoring for three different species.

4.3.1 Australian sea lions (*Neophoca cinerea*) in South Australia

The Australian sea lion (*Neophoca cinerea*; 'ASL') is an endangered otariid found in South and West Australia, with an estimated population size of between 10 – 15,000 (Goldsworthy et al. 2021). A major study of Australian sea lions in South Australia was implemented with the aim of assessing the risk from bycatch mortality in the Gillnet Hook and Trap (GHAT) fishery (Goldsworthy et al. 2010, 2022).

The project comprised four main components: (1) assessing ASL-fisheries interactions through an observer programme; (2) creating species distribution models through satellite telemetry data; (3) collating fishing effort spatial data; (4) creating sea lion population models from abundance surveys and demographic data to ascertain the impacts of varying levels of bycatch on ASL.

The size and location of ASL breeding sites in South Australia were based on pre-existing pup production data (Goldsworthy et al. 2009), combined with life-tables and matrix models developed from demographic analyses of the ASL population from one colony (Seal Bay) (McIntosh 2007, Goldsworthy et al. 2020). These demographic data provided age-specific survival rates for ASL based on resight effort from a long-term marking programme.

To understand the spatial distribution of ASL foraging, satellite telemetry data from a total of 210 individuals (157 females, 31 males and 22 juveniles) were used. Key areas used during foraging were determined by calculating the amount of time each sea lion spent in the cells of a grid overlay (Goldsworthy et al. 2009). The spatial distribution of ASL foraging effort was determined through statistical models described in Goldsworthy et al. (2020;2022) using a 1x1 km grid. ASL diving behaviours were used to estimate how much time individuals would be vulnerable to gillnets set on the seafloor, by determining how long animals spent in the bottom phase of their dives. These data were derived from time-depth recorder (TDR) deployments on 11 adult females and four adult males. This information was combined with the foraging distribution model to facilitate estimation of total ASL foraging effort during which animals were at risk from set net interactions.

Bycatch impact assessments were made through independent observers on 10 trips in the shark gillnet fishery between February 2006 – January 2008 (Hamer et al. 2013). Observations allowed for the inclusion of any ASL that dropped out of nets as they were hauled aboard, and records included the time and location of hauls, the presence of drowned ASL and, if possible, their sex and age class (Hamer et al. 2013). Net locations were plotted onto the species distribution model, and bycatch rates were estimated as per Goldsworthy et al. (2022). Population viability analyses (PVAs) were used to assess the potential impacts of varying bycatch levels on ASL subpopulations (Goldsworthy et al. 2009, 2022), using the demographic information from Seal Bay (McIntosh 2007).

The comparison of ASL marine distribution and fishing effort found an almost complete spatial overlap in ASL foraging and fishing effort, and the PVAs indicated that most subpopulations were subject to unsustainable levels of bycatch mortality (Goldsworthy et al. 2022). In response to the findings of this study, several actions were taken. These actions included: establishing an independent monitoring programme, permanent spatial gillnet closures around breeding sites and temporal closures triggered by area-specific bycatch trigger limits and incentives for gillnet fishers to switch to other gear types (Goldsworthy et al. 2022). In the following decade, estimated ASL bycatch in gillnets reduced by 98%, and the decline in sea lion pup abundances stabilised at some sites (Goldsworthy et al. 2022, McIntosh et al. 2022). There was an almost 100% transition in the fishery from gillnets to longlines, and catches returned to pre-management levels.

One area of weakness in ASL monitoring has been the collection of pup abundance data, which has been patchy, and investigators are calling for a more systematic monitoring strategy to address current gaps (Goldsworthy et al. 2021). The recent inclusion of aerial surveys has improved survey efficiency and capacity, facilitating a move away from focussing solely on a small number of indicator sites, which may not have represented broader trends (Goldsworthy et al. 2021).

4.3.2 Grey seals (*Halichoerus grypus*) in the United Kingdom

In the United Kingdom, grey seals (*Halichoerus grypus*) breed on coastlines in England, Scotland and Northern Ireland, with pup production monitored at some colonies since the 1950s (Russell et al. 2019). In 2022/23, it was estimated that 79,122 pups were born across all UK colonies (SCOS 2025).

Under national and supranational legislative requirements, most grey seal colonies were surveyed annually between 1984 – 2010, and at least biennially since – with pup production from these regularly monitored colonies accounting for ~90% of the total UK grey seal pup production (Russell et al. 2019). To ensure newly founded colonies are not missed, periodic scouting surveys take place to cover potential additional pupping sites, which may be added to regular monitoring. Both ground and aerial surveys are used, with the latter covering the majority of sites. Abundance estimation methodologies have been standardised at the colony level. By incorporating mark-recapture estimates from several intensively studied colonies, this long-term monitoring effort has facilitated

detailed population modelling to estimate the total UK grey seal population size, and trends therein, over multiple decades (Thomas et al. 2019). As a result, when decreasing pup production growth rates were detected in several study regions, a driver could be identified with relative confidence (Russell et al. 2019, Thomas et al. 2019).

Additional grey seal monitoring effort includes large spatial scale diet analysis through scat and regurgitate sampling, to determine the potential size of any competition with commercial fisheries – as well as other marine predators (Wilson & Hammond 2019) – and long-term marine space use monitoring via telemetry tags (Jones et al. 2015, Russell et al. 2017). The data derived from the telemetry studies were collected to inform recommendations for marine space use planning, for example the importance of using marine protected areas (MPAs) to connect haul-out sites with offshore foraging habitats.

4.3.3 Australian fur seal (*Arctocephalus pusillus doriferus*) in south-eastern Australia

The Australian fur seal ('AFS') is endemic to Australia and ranges from New South Wales to South Australia and Tasmania (McIntosh et al. 2022). It was estimated that there were 16,903 live AFS pups in 2017 (McIntosh et al. 2022).

Until 2002, AFS monitoring consisted of rare and sporadic counts until the late 1960s, followed by opportunistic and surveillance-style monitoring until 2002 (McIntosh et al. 2018). From then, five-yearly range wide AFS pup count monitoring has been adopted (Kirkwood et al. 2005; McIntosh et al. 2018), alongside annual pup surveys and focused research at selected sites (McIntosh et al. 2022). Pup production surveys included a mixture of ground surveys (direct counts or mark-recapture) and aerial surveys (Kirkwood et al. 2005, McIntosh et al. 2018). The choice of survey method was generally determined by pup production, accessibility and working conditions (McIntosh et al. 2018). As with UK grey seals, the emphasis has been on standardizing a methodology by site, rather than across the entire study range (McIntosh et al. 2018). The range-wide surveys aim to survey all recognized AFS breeding colonies, and ensure understandings of the broader health and trajectory of the meta-population (McIntosh et al. 2022).

The five yearly AFS censuses, combined with historic counts from single sites, were successful in tracking AFS population trends, including sustained declines in the meta-population following a period of growth (McIntosh et al. 2022). A power analysis of the trends in the AFS meta-population found that more reliable trends would be obtained by sampling more regularly than every five years (McIntosh et al. 2018), and a subsequent study documenting sustained reductions in the AFS population stated that the five-yearly census combined with more frequent assessments at selected sites should be a minimum requirement (McIntosh et al. 2022). The 2022 study identified key sites for annual monitoring in different regions, comprising large colonies with distinct trends as well as recently established and growing populations (McIntosh et al. 2022).

Alongside the coordinated abundance monitoring programme, studies have examined factors affecting AFS population dynamics. These have included analyses of AFS diet in relation to environmental variation using scat and regurgitate analysis (Kirkwood et al. 2008, Kliska et al. 2022), impacts of environmental change on pup body condition (Wall et al. 2023), and the links between pup condition, survival, and future population trajectories. Demographic studies have collected survival and reproductive rate data (Gibbens & Arnould 2009, Gibbens et al. 2010). Satellite telemetry studies have updated understandings of AFS marine space use and foraging behaviour (Salzeri et al. 2024, Bartes et al. 2024, 2025), and interactions between AFS and commercial fishing vessels have been observed to inform potential modifications to industry practices to reduce AFS bycatch (Tilzey et al. 2006, Hamer & Goldsworthy 2006).

4.3.4 Summary of international pinniped monitoring case studies

Several common practices emerge from the case studies described above:

(1) Large spatial scales – large spatial scale abundance monitoring prevents reliance on a few sites as proxies for the whole population/meta-population, and reduces the risk of distribution shifts being mistaken changes in abundance (Kirkman et al. 2011). Aerial surveys have facilitated large spatial scale coverage in all three case studies.

(2) Survey coordination – effort has been made to coordinate abundance surveys across these large spatial scales, facilitating long-term trend analysis (McIntosh et al. 2018, 2022, Russell et al. 2019).

(3) Methodological consistency – the above case studies demonstrate that one abundance estimation methodology need not be adopted for use at all colonies, with the emphasis instead on consistent methodologies at the colony/regional level.

(4) Identification of population drivers – in each case, effort was directed towards understanding population drivers, including foraging and diving, diet, and demographic rates (survival and fecundity).

4.4 Recommendations for kekeno monitoring in New Zealand

4.4.1 Monitoring programme objectives

Ecological monitoring programmes designed in conjunction with management typically have objectives that fall into two categories: (A) identifying current system characteristics, or (B)

monitoring system responses to management actions (Yoccoz et al. 2001). Given our current knowledge of kekeno and the opportunities to expand on this, the following objectives are suggested:

1. *Ascertain and track trends in New Zealand's kekeno population and understand the drivers of its status.*
2. *Investigate parameters that contribute to a reduction in fisheries-related deaths towards zero*[4].
3. *Define trigger points for management interventions when sustained negative population trends are identified, and determine the nature of these interventions.*

The second and third objectives ensure long-term abundance monitoring programmes maximise the value of invested resources (Reynolds et al. 2016). This is well illustrated on the WCSI, where substantial effort has identified declining colony trajectories (Hall et al. 2026b), with the next step being to develop understandings of what is driving the observed trends. Doing so will enable the evaluation and implementation of management initiatives.

4.4.2 Recommended data collection overview

From the case studies above, and drawing on a proposed monitoring framework for Cape fur seals in southern Africa (Kirkman et al. 2011), the below parameters are suggested to promote the framework objectives. Recommended methodologies are provided in Sections 4.4.3 and 4.4.4 and summarized in Table 8:

(i) Disease screening: to demonstrate the population impacts of disease, such as CDV (Wilson et al. 2025, Taylor et al. 2026) and the threat of HPAI (Ashley et al. 2026).

(ii) Pup abundance and condition data: to provide insights into the size, health and trajectories of colonies, and indicate their threat resilience (Wall et al. 2023, Krause et al. 2024, Hall et al. 2026b).

(iii) Diet and foraging data: to provide fisheries independent at-sea tracking data, map spatial and temporal overlaps with fisheries (Vincent et al. 2016, Bamford et al. 2021, Goldsworthy et al. 2022, Riaz et al. 2023) and elucidate the impacts of marine environmental change on foraging efficiency (Speakman et al. 2020, Cruz-Vallejo et al. 2024), which impacts pup survival, and thus population trends (Jeanniard-Du-dot et al. 2017, Wall et al. 2023).

(iv) Survival and reproductive rate data: to understand how threats can impact population trajectories via population viability assessments (Chilvers 2012, Goldsworthy et al. 2022).

[4] *Objective 12.2.1 of Te Mana o te Taiao, Aotearoa New Zealand Biodiversity Strategy 2020 (Department of Conservation 2020): 'The number of fishing-related deaths of protected marine species is decreasing towards zero for all species'.*

4.4.3 How and where to monitor abundance

Future kekeno monitoring could adopt a tiered approach – similar to AFS monitoring (McIntosh et al. 2018, 2022) – combining annual research at focal colonies (Table 7) with periodic nationwide surveys to counteract selection bias (Kirkman et al. 2011, McIntosh et al. 2022). Focal sites can be strategically selected to build on existing datasets, capture a range of ecological conditions, reflect exposure to key risks and ensure practical access (Lindenmayer et al. 2012, Battershill et al. 2016, Carvalho et al. 2016, Clairbaux et al. 2024).

Table 7 Suggested focal colonies for future kekeno monitoring

Site	Rationale for selection
Mātakitaki-a-Kupe/Cape Palliser	- High bycatch in the proximal Cook Strait - CDV detection - Likely the largest North Island kekeno colony
Kaikōura (Ōhau Point, Kaikōura Peninsula, South Coast)	- Pre-existing population and marine distribution data - Represents apparently growing regional population - Recent Unusual Mortality Event + CDV detection
West Coast of the South Island (Wekakura Point, Cape Foulwind, Taumaka Island)	- Pre-existing population and marine distribution data - Represents declining regional population - Bycatch concerns in proximal commercial fisheries
Sandymount (Otago Peninsula) *	- Pre-existing population and foraging data - Overlap with pakake/New Zealand sea lion (a kekeno predator)

* Long-term direct count data exist from other south-eastern South Island colonies (MacDiarmid & Lalas 2014). Counts at some of these colonies could also be re-established to complement mark-recapture at Sandymount.

4.4.4 Data collection recommendations

Disease screening

Routine disease screening (e.g. faecal sampling and seasonal necropsies) could usefully be integrated into existing monitoring to enable detection of diseases such as CDV or HPAI. Screening would ideally be conducted each season, with additional effort during periods of elevated mortality, and should consist of at least 30 faecal swab samples per colony, combined with swabs from dead animals and/or necropsies. Effective disease screening should be coordinated between agencies including DOC, the Ministry for Primary Industries and specialist wildlife veterinarians, as occurred during the investigations that identified CDV (Taylor et al. 2026).

Pup production and condition data collection

Pup production and condition data should be collected annually at focal sites. Abundance data collection protocols can follow established site-specific methodologies where they exist – for example, mark-recapture on the WCSI (Hall et al. 2026b). A subset of captured pups should be weighed and measured to assess body condition trends. Where possible, survey timing should be coordinated between colonies.

Five-yearly nationwide aerial surveys to counteract spatial bias and improve understandings of connectivity between colonies – for example by detecting emerging colonies and distribution shifts (Kirkman et al. 2011, McIntosh et al. 2018) – would complement the annual focal site assessments. Nationwide surveying would also address the current gaps in the surveying of kekeno colonies – for example, in the North Island and the Chatham Islands. Given the success of the Bounty Islands study (Hall et al. 2026a), drones may be leveraged to improve efficiency and coverage at suitable colonies, with calibrations to ground methods to promote continuity for trend analyses (McIntosh et al. 2018). Modern genetic techniques could also play a role in understanding colony connectivity (Dussex et al. 2016), although previous attempts using microsatellite markers found that low spatial genetic differentiation, potentially a result of the historic bottleneck the species experienced, meant less than one-third of individuals could be assigned to their colony of origin.

Foraging ecology and diet studies at the focal sites

Foraging ecology studies at focal sites would involve equipping lactating females with telemetry devices to record marine distribution and diving behaviour. For comparability with international studies assessing overlaps between pinnipeds and commercial fisheries, a minimum of 10 instrumented female kekeno per colony in both summer and winter is recommended, with the project running across at least two years (Cronin et al. 2016, Bamford et al. 2021, Riaz et al. 2023).

Scat and regurgitate sampling is a cheap, efficient and non-invasive method for assessing pinniped diets (Fea et al. 1999, Kirkman et al. 2011, Allum & Maddigan 2012, Emami-Khoyi et al. 2016). It is recommended that at least 30 scats and 30 regurgitate samples are collected twice per year (once in summer and once in winter to account for changes in diet through the year (Harcourt et al. 2002, Emami-Khoyi et al. 2016)) at the focal colonies. This will help to quantify resource competition with commercial species and detect changes in prey availability (Wilson & Hammond 2019, Kliska et al. 2022, Lyssikatos & Wenzel 2024). As scat sampling is inexpensive and can be combined with disease screening – annual sampling for at least 10 years is recommended to assess dietary responses to marine environmental variability over time (Kliska et al. 2022). Stable isotope analyses can also be conducted to improve understandings of kekeno diet and foraging behaviours in different regions, as has been achieved with pakake (Chilvers & Galbraith 2025).

Survival and fecundity data

Demographic data are integral to models assessing species population viability, which include assessments of risks to population trajectories (Hernández et al. 2008, Chilvers 2012, Goldsworthy et al. 2022, Luck et al. 2022).

Collecting these data would involve the establishment of long-term marking/resight programmes of pups and females at selected colonies. Depending on colony size, 100 – 200 pups/year would need to be tagged over at least five consecutive breeding seasons (Beauplet et al. 2005, Kirkman et al. 2011). The numbers and regularity of adult female tagging have varied substantially between studies seeking to quantify survival. Wege et al. (2014) tagged a minimum of 30 sub-Antarctic fur seal (*Arctocephalus tropicalis*) mother-pup pairs per year between 2006 – 2011 on Marion Island, 134 female pakake were branded on Enderby Island in 2000 (Chilvers & MacKenzie 2010), while a total of 240 female sub-Antarctic fur seals were tagged across five breeding seasons on Amsterdam Island (Beauplet et al. 2006). Systematic resighting effort would be key to the success of this programme, thus accessible colonies such as those at Kaikōura or Cape Foulwind should be prioritized.

4.4.5 Data collection timelines and summary

Together, the above methodologies provide a comprehensive, integrated monitoring framework that builds on earlier studies and strengthens population trend detection going forward. The methodologies suggested would also improve understanding of the ecological drivers of kekeno population trends – supporting targeted and effective management responses. These suggested analyses would form a new, robust baseline. Additional studies should be encouraged where they add further insight.

The planning and resources required to collect the data recommended above vary considerably. As such, the following time-scales are suggested:

Within the next two years:

1. Seasonal scat sampling initiated at focal colonies for disease screening and diet sampling.
2. Annual pup abundance and condition monitoring continued at Kaikōura, the WCSI and Sandymount, and commenced at Mātakitaki-a-Kupe/Cape Palliser.
3. Establishing a representative dataset of kekeno foraging activity and marine distribution through telemetry device deployment, prioritising sites with declining pup production.

Within the next five years:

4. Continuation of 1 and 2 (above)

5. Deployment of telemetry devices on lactating females at Mātakitaki-a-Kupe/Cape Palliser, Kaikōura and Sandymount.
6. Planning and execution of the first nationwide abundance aerial survey.
7. Implementation of the permanent mark-resight programme in Kaikōura and Cape Foulwind.

Table 8 Summary of the suggested future monitoring framework for kekeno in New Zealand.

Parameter	Suggested location	Method	Regularity	Sample size
Pup production	Nationwide	Aerial direct counts	Every 5 years	NA
	Focal sites	Mark-recapture or direct counts	Annually	Marking between 10 – 50% of expected pup production for colony
Pup body condition	Focal sites	Length and weight measurements	Annually	n = 100 pups (50 of each sex)
Diet	Focal sites	Scat and regurgitate analysis	Bi-annually	n ≥ 30 scats and regurgitates per colony per sampling period
Foraging behaviours	Focal sites	Satellite transmitters and time-depth recorders affixed to lactating females	Multi-year study Tags deployed in both summer and winter	Minimum of 10 lactating females/season/colony
Survival rates	Cape Foulwind and Kaikōura	Tagging and resights of pups and adult females	Multi-year study with systematic resight effort	100 – 200 tagged pups/year/colony + ~100 adult females/colony
Reproductive rates	Cape Foulwind and Kaikōura	Resights of known-age females	Multi-year study with systematic resight effort	Based on tagged numbers
Disease screening	Focal sites	Faecal swabs Tissue swabs from dead animals/necropsies	Seasonally + in the event of UMEs	n = 30/colony (faecal swabs) with swabs from dead animals/necropsies dependent on sample availability

5 Overall summary and conclusions

The monitoring studies described in Sections 2.2 – 2.5 provide important updates to our understandings of kekeno abundance in four distinct regions. The inferences that can be drawn for each colony/region vary due to the differing time series available, which range from a multi-decade study on the WCSI, to single year snapshots in the cases of Mātakitaki-a-Kupe/Cape Palliser and the Bounty Islands. The comprehensive WCSI study demonstrates the value of consistent, longitudinal colony monitoring for identifying kekeno abundance and health trends, and provides a template for data collection in other regions.

The updated regional abundance estimates demonstrate the variety of dynamics experienced by kekeno colonies in different regions. For example, the substantial drop in pup production between the 2022/23 – 2023/24 breeding seasons at Ōhau Point in Kaikōura, followed by increases in the 2024/25 and 2025/26 breeding seasons, represents a markedly different short-term trend to those observed at the WCSI study colonies, where previously observed pup abundance declines (Roberts & Neale 2016) have broadly continued from 2020 – 2025. These disparities reinforce the importance of a spatially explicit approach to managing and mitigating anthropogenic risk factors.

The longitudinal pup abundance and condition time series available on the WCSI provide an important opportunity to assess the drivers of the observed declines. Building an understanding of these drivers would require data collection on parameters known to impact kekeno population dynamics: including the availability of prey (through foraging and diet studies), the impacts of disease (through screening), and survival and fecundity rates (through long-term mark-resighting). Assessments of these parameters would improve current understandings of the population impacts of a variety of threats on kekeno, while positioning the species as a valuable sentinel for New Zealand's marine environment.

The approach used to generate the nationwide kekeno abundance estimate provided in this report (208,536 – 251,644), conducted within the limits of the currently available data, represents an empirically based update to our understanding of New Zealand's kekeno population size, the first since Wilson's (1981) 1970s census. It should be emphasised, however, that pup production data are not currently available for all known kekeno colonies, meaning these colonies could not be included in the estimate. Additionally, care should be taken to ensure that nationwide kekeno abundance estimates do not mask colony or regional-level trends, particularly in the context of spatially-explicit threats. Similarly, population trends from one region should not be extrapolated to others. Increasing the regularity with which high-quality kekeno population data are collected, and expanding regular data collection to colonies that have received little to no research effort would help to move away from using nationwide abundance, with the specified problems that entails.

The proposed framework for a coordinated future monitoring programme for New Zealand's kekeno draws on international best practices, and seeks to maximise the value of existing research while identifying opportunities for complementary data collection. The primary objectives of the proposed framework are to (1) ascertain and track trends in New Zealand's kekeno population and understand the drivers of its status (2) investigate parameters that contribute to a reduction in fisheries-related deaths towards zero and (3) define trigger points for management interventions when sustained negative population trends are identified, and determine the nature of these interventions. Tracking trends in the population requires consistent, coordinated approaches to abundance monitoring, and the framework recommends the use of focal sites for annual intensive studies combined with periodic, nationwide aerial surveys to mitigate selection biases and address current data gaps. To better understand the drivers of kekeno abundance trends, efforts should be directed toward assessing marine distribution and diving behaviours, diet, survival and reproductive rates and disease prevalence. These approaches collectively form a monitoring framework that would improve trend detection, advance understanding of the ecological drivers of population change, and inform more targeted management responses.

6 Recommendations for future research

Future assessments of the size of New Zealand's kekeno population, and its susceptibility to threats, would benefit from:

- Centralized and coordinated research to collect kekeno abundance data, with an emphasis on expanding the number of colonies regularly assessed. In particular, kekeno abundance data are lacking from colonies on the North Island, Chatham Islands, and some sub-Antarctic Islands.

- The establishment of spatially representative datasets of kekeno marine distribution, foraging behaviours and diet.

- Long-term resight studies of permanently marked individuals to provide age-specific survival and fecundity data.

- Examination of the prevalence and impacts of disease on kekeno colonies around New Zealand.

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Appendix 1 Kaikōura pup production methods

Appendix 1.1 Summary of methods used to assess kekeno pup production at Kaikōura colonies in the 2025/26 breeding season.

Site	Method	No. pups marked to estimate pup production
Ōhau Point	Mark-recapture	602
South Rakautara	Mark-recapture	65
Lynch's Reef (Kaikōura Peninsula)	Direct counts	-
KP2 (Kaikōura Peninsula)	Direct counts	-
Rhino Horn (Kaikōura Peninsula*)	Direct counts	-
Whalers Bay (Kaikōura Peninsula*)	Direct counts	-
SC1 (South Coast)	Mark-recapture	70
SC2 (South Coast)	Mark-recapture	100
SC3 (South Coast)	Direct counts	-
SC4 (South Coast)	Mark-recapture	60
SC5 (South Coast)	Direct counts	-
Otumatu	Direct counts	-
Oaro	Direct counts	-
Total marked		998*

*101 pups were marked on the Kaikōura Peninsula during morphometric assessments. This is included in the total marked.

Appendix 2 Kaikōura pup production from 2022/23 to 2025/26

Appendix 2.1 Kekeno pup production estimates from Kaikoura colonies and sub-colonies from the 2022/23 breeding season to the 2025/26 breeding season.

Colony/sub-colony (Method)		Pup production estimate ± SE			
		2022/23	2023/24	2024/25	2025/26
Ōhau Point (Mark-Recapture)		2,401 ± 99	1,261 ± 66	3,137 ± 87	5,346 ± 102
South Rakautara (Mark-Recapture)		285 ± 25	-	373 ± 20	591 ± 42
Kaikōura Peninsula Colony	Lynch's Reef (Direct Counts)	41 ± 1	44 ± 4	42 ± 3	89 ± 1
	KP2 (Direct Counts)	159 ± 9	159 ± 8	230 ± 10	381 ± 7
	Rhino Horn (Direct Counts)	323 ± 10	162 ± 4	291 ± 14	367 ± 17
	Whalers Bay (Direct Counts)	245 ± 4	146 ± 7	329 ± 9	471 ± 7
South Coast (SC) Colony	SC1 (Mark-Recapture)	269 ± 11	248	396 ± 36	616 ± 22
	SC2 (Mark-Recapture)	427 ± 13	442	497 ± 20	971 ± 45
	SC3 (Direct Counts)	37 ± 2	42 ± 3	54 ± 4	168 ± 5
	SC4 (Mark-Recapture)	227 ± 4	216	272 ± 11	524 ± 28
	SC5 (Direct Counts)	12 ± 0	39 ± 1	48 ± 3	114 ± 4
	Otumatu (Direct Counts)	13 ± 0	23 ± 1	29 ± 3	62 ± 0
	Oaro (Direct Counts)	-	-	31 ± 2	46 ± 1

Appendix 2.2 Kekeno pup production for Kaikōura colonies between the 2022/23 – 2025/26 breeding seasons.

Breeding season	Total pup production estimate
2022/23	4,400 – 4,478
2023/24*	2,688 – 2,876
2024/25**	5,507 – 5,951
2025/26**	9,465 – 10,027

*No data available for South Rakautara colony in 2023/24, and SC1, SC2 and SC4 do not have error ranges for mark-recapture estimates.

** Oaro colony added to surveys. This accounted for 31 (± 2 SE) pups in 2024/25 and 46 (± 1 SE) pups in 2025/26.

Appendix 3 Kekeno pup production estimates used in the nationwide abundance estimate calculation

Appendix 3.1 Kekeno colonies with estimates used in the nationwide abundance estimate. Updated from Hall et al. (2025b).

Colony/region	Year of most recent pup production estimate	Count method	Most recent pup production estimate	Source
Gannet Island & Albatross Point	2023	Boat direct count	15	C. Hansen (pers. comm.)
Sugarloaf Islands	2023	Boat direct count	25	Chaddy's Charters (pers. comm.)
Honeycomb Rock	1999	Direct count	10	(Baird, 2011)
Cape Palliser	2026	Direct count	~1000	This report
Stephens Island	1994	Direct count	276[A]	(Taylor et al., 1995)
Trio Islands	2007	Boat direct count	50	(Baird, 2011)
Admiralty Bay	2021	Direct count	100	L. Boren (pers. comm.)
Tasman Region (Separation Point to Onetahuti, Tonga Island, Adele Island, Pinnacle Island and Fisherman Island)	2026	Direct count	168	Department of Conservation (unpublished data)
Kahurangi to Otukoroiti	2009	Aerial direct count	63	(Baker et al. 2009)
Steep Point	2009	Aerial direct count	25	(Baker et al. 2009)
Wekakura Point	2025	Mark-recapture	178 – 194	(Hall et al. 2026b)
Black Reef	2010	Direct count	200[A]	(H. Best, *unpublished data*)
Cape Foulwind	2025	Mark-recapture	122 – 140	(Hall et al. 2026b)
Charleston	2010	Direct count	476[A]	(H. Best, *unpublished data*)

Seal Island	2009	Aerial direct count	6	(Baker et al. 2009)
Point Elizabeth	2003	Direct count	10	(Baird, 2011)
Needles Point	2023	Direct count	36	L. Boren (pers. comm.)
Kaikōura (Ōhau Point – Oaro)[B]	2026	Mark recapture and direct count	10,413 – 11,023	This report
Banks Peninsula	2024	Mark recapture and direct count	2,762 – 3,713	(Hall et al. 2025a)
West of Pukutuaro Cliff	2009	Aerial direct count	16	(Baker et al. 2009)
Gillespie's Point	2009	Aerial direct count	3	(Baker et al. 2009)
Hanata Island	2009	Direct count	66	(Baker et al. 2009)
Abbey Rocks	2009	Aerial Direct count	12	(Baker et al. 2009)
Island off Knights Point	2009	Aerial direct count	13	(Baker et al. 2009)
Taumaka Island	2025	Mark-recapture	555 – 577	(Hall et al. 2026b)
Popotai Island[C]	2009	Aerial direct count	118	(Baker et al. 2009)
Jackson Head	2009	Aerial direct count	21	(Baker et al., 2009)
Cascade Point	2009	Aerial direct count	1,466[A]	(Baker et al., 2009)
Long Reef, Martins Bay	2009	Aerial direct count	163[A]	(Baker et al., 2009)
Yates Point	2009	Aerial direct count	1,228[A]	(Baker et al. 2009)
Lower Fiordland (Doubtful/ Pateā, Dusky and Breaksea Sounds, and Chalky Inlet)	2021	Mark recapture and direct count	2,935 – 5,042	(Chilvers, 2021)
Otago and the Catlins Moeraki, Heyward Point,	2009	Direct count	2,993 – 5,294	(MacDiarmid and Lalas 2014)

Otago Peninsula, Green Island, Nugget Point, Tucks Cove, Penguin Bay, Cosgrove Island)				
Solander Island	2009	Aerial direct count	1,107	(Baker et al., 2009)
Rakiura/Stewart Island (and surrounding islands)	2003	Mark recapture and direct count	2,694[A]	(Watson et al. 2015)
Snares Islands	1997	Direct count	188	(Carey, 1998)
Bounty Islands	2024	Direct count	4,252 – 4,341	(Hall et al. 2026a)
Antipodes Islands	2024	Direct count	~193	Mattern 2024
Campbell Islands	1988	Direct count	126[A]	(Moore and Moffat 1990)

[A] Indicates sites that met the criteria to be modelled.
[B] To be comparable with the nationwide abundance estimate in Hall et al. (2025b), the Kaikōura pup estimate provided here was calculated using the multipliers described in Hall et al. (2024), and therefore differs from the result presented in Section 2.2.2.1 and Appendix 2.2. The multipliers rendered the results from Kaikōura colonies/sub-colonies assessed via direct counts in the 2025/26 field season comparable with those assessed via mark-recapture. See Hall et al. (2024) for details of the multipliers.
[C] As in Hall et al. (2025b), the 2009 estimate for Popotai Island was not used in any calculations due to uncertainty caused by declining pup numbers on nearby Taumaka Island, but is included here for completeness.

Appendix 4 Sandymount pup production from 1995/96 to 2022/23

Appendix 4.1 Kekeno pup production from the Sandymount colony on Otago Peninsula from 1995/96 to 2022/23. Reproduced here with permission from P. Seddon.

Breeding season	Mark-recapture estimate (95% CI)
1995/96	100 (96 – 104)
1996/97	133 (131 – 135)
1997/98	140 (138 – 142)
1998/99	128 (120 – 136)
1999/2000	98 (92 – 104)
2000/01	79 (72 – 86)
2001/02	86 (80 – 97)
2002/03	115 (106 – 126)

2003/04	123 (116 – 132)
2004/05	163 (141 – 197)
2005/06	113 (101 – 131)
2006/07	138 (121 – 163)
2007/08	117 (100 – 143)
2008/09	121 (101 – 141)
2009/10	88 (68 – 107)
2010/11	74 (64 – 85)
2011/12	69 (53 – 85)
2012/13	75 (66 – 83)
2013/14	67 (56 – 77)
2014/15	102 (78 – 126)
2015/16	67 (55 – 79)
2016/17	100 (76 – 125)
2017/18	60 (52 – 68)
2018/19	60 (48 – 73)
2019/20	49 (44 – 55)
2020/21	56 (47 – 67)
2021/22	55 (44 – 67)
2022/23	29 (28 – 31)