2025 Macquarie Harbour Maugean skate population status and monitoring report

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Executive summary

The Maugean skate (*Zearaja maugeana*) is an endangered micro-endemic species, which has only been found in two estuaries on the west coast of Tasmania, Australia but is now likely restricted to a single estuary, Macquarie Harbour. Monitoring of the Maugean skate population in Macquarie Harbour is critical to ensure the most contemporary information is available to determine population viability and apply appropriate conservation actions as required. Results presented here cover all targeted (2012-2025) and non-targeted (2011) netting programs that have captured Maugean skate, with specific reference to the results from those years where a standardised monitoring protocol was employed (2021-2025) and a reference year (2014), which utilised a comparable standardised protocol.

Demographic structure and recruitment

Size composition data from 2014 and 2021-2025 showed significant interannual variation in total length. Female median and mean sizes in 2021-2025 were generally larger than the 2014 reference year, while male size differed significantly only in 2022 from the reference year. These patterns were largely driven by differences in the number of juveniles captured in each year. In contrast, median ages of both sexes remained largely stable over the study period (2014 and 2021-2025), with only a short-term shift toward older individuals in 2021 in comparison to the 2014 reference year associated with reduced juvenile captures.

Age-based cohort analysis revealed important insights into Maugean skate recruitment patterns through time by retrospectively assigning individuals to a birth year, which represents the year they hatched from their egg, using modelled ages from size at capture. Cohorts inferred to have originated from the 2006-2008 period were relatively consistent and represented the highest contributions detected across the time series. These cohorts primarily comprised individuals caught from 2011-2014, and the proportional contribution of each capture year remained consistent across cohorts A marked decline in inferred cohort representation from 2009 onward was evident in the sampled data and corresponded with a prolonged reduction in deep-water dissolved oxygen in Macquarie Harbour. As this decline in inferred births preceded 2014, the 2014 reference year should be interpreted as a comparative benchmark only, rather than a baseline for population recovery.

From 2018 to 2021, inferred hatchling numbers remained relatively stable, though at lower levels than those observed for the 2006-2008 cohorts. Systematic population monitoring commenced in 2021, and 2025 represents the first year in which individuals born during the 2021 to 2025 monitoring period were detectable as adults due to the size selectivity of the fishing gear. Additionally, based on back calculated ages, a high proportion of individuals captured as adults in 2025 were assigned to the 2020 cohort, further providing evidence that a proportion of recent juveniles have survived to maturity.

Recent recruitment dynamics are consistent with a pulse-recruitment model, characterised by interannual variability in cohort contributions. While encouraging, pulse-driven recruitment is inherently vulnerable to environmental disturbance. Multiple consecutive years of successful recruitment will likely be required before a persistent recovery trajectory can be confidently inferred.

Catch Per Unit Effort

Previous reporting demonstrated a pronounced decline in relative abundance, with lower catch per unit effort (CPUE) observed in 2021–2022 compared to 2014, defined as the number of Maugean skate caught per 50 m net per hour (N/net/hr). This decline coincided with a prolonged period of apparent suppressed recruitment beginning around 2009 and two high-impact environmental events in 2019. Catch per unit effort in 2014 was 0.17 ± 0.03 N/m/hr, declining to 0.06 ± 0.03 N/m/hr in 2021 and further to 0.04 ± 0.02 N/m/hr in 2022, representing point-to-point declines of 64% and 77%, respectively, relative to the 2014 reference year. In 2023 and 2024, raw mean CPUE increased to 0.11 ± 0.05 N/m/hr and 0.21 ± 0.07 N/m/hr respectively. This translates to a 35% decline in 2023 relative to

2014 and a 23% relative increase in 2024, but both years were statistically comparable to 2014 reference levels. The updated relative abundance data reported here shows that raw mean CPUE decreased to 0.15 ± 0.06 N/m/hr in 2025, representing a 12% decrease relative to 2014 and a 29% decrease relative to 2024 but remained statistically comparable to 2014 reference levels.

Since 2023, CPUE variability has increased substantially with pronounced seasonal and within year fluctuations. In this report, standardised CPUE modelling using generalised linear models identified significant spatial and temporal effects, with recent increases in harbour-wide CPUE largely driven by rising catch rates in the World Heritage Area, while catch rates in Table Head and Swan Basin were declining. Given the life history constraints of this species (time to maturity and short reproductive lifespan), these patterns cannot be attributed solely to population growth in the World Heritage Area alone, and therefore, likely reflect some redistribution of adult individuals among sites. These observations suggest that since 2023 there have been factors that can influence the catchability of Maugean skate, and which are likely to be site specific. Consequently, harbour-wide CPUE should not be interpreted as a simple proxy for abundance, instead, standardised CPUE estimates derived from models that explicitly account for spatial and temporal effects should be used. Future work should aim to further improve these models by better characterising additional sources of variability, including interannual environmental variation.

Implications for monitoring and management

New information presented in this report provides an updated understanding of recent demographic trends in the population, including evidence of recent recruitment pulses and continued stabilisation of relative abundance. However, the population remains highly vulnerable due to its restricted distribution, suspected pulse-driven recruitment dynamics, potential sensitivity to adverse environmental events and recruitment levels lower than those inferred during 2006-2008. Evidence of spatiotemporal variability, including within-year effects indicates that the sampling design should continue to retain site-specific coverage and a strict seasonal framework to characterise sources of uncertainty and environmental variability. Furthermore, distributing efforts across a broader range of observation opportunities (e.g., eight trips per year instead of four trips, with two per season) will improve statistical power and will help better characterise within year variability and seasonal effects, allowing these variables to be incorporated into the standardised CPUE model. Lastly, integration of complementary methods with standardised net monitoring including acoustic telemetry, genetic approaches and alternative survey technologies is essential to improve population assessments and inform effective conservation management. This recommendation aligns with conservation priorities emphasising that continued population monitoring of Maugean skate should include the development of robust, non-harmful, and logistically feasible sampling methods.

Female juvenile Maugean skate in Swan Basin. (Jane Ruckert IMAS).

Introduction

The Endangered, micro-endemic Maugean skate (*Zearaja maugeana*) is only known from two isolated estuarine systems located on the west coast of Tasmania, Australia, Bathurst and Macquarie Harbours, representing one of most restricted distributions of any elasmobranch (Last and Gledhill, 2007). A recent environmental DNA study (Moreno et al., 2022) has demonstrated that the vast majority, if not all, of the current population of Maugean skate live only in Macquarie Harbour. Monitoring the Maugean skate population in Macquarie Harbour is essential to ensure contemporary information is available for effective management and assessment of population viability.

Maugean skate were initially captured as part of a non-species-specific netting program conducted in Macquarie Harbour during 2011 to assess the impact of overnight gillnetting on survivorship of various catch species (Lyle et al., 2014; Bell and Lyle, 2016). This program continued through 2012 and into early 2013 under the same study framework; however, as Maugean skate were increasingly encountered during sampling, monitoring and tagging of individuals was initiated. Dedicated netting surveys targeting Maugean skate in Macquarie Harbour to address a range of research objectives were subsequently established in 2012 and continued through to 2019, funded by FRDC. Additional surveys were undertaken from 2021 to 2023 through the Sustainable Marine Research Collaboration Agreement (SMRCA), and from 2024 to November 2025 with support from the Department of Natural Resources and Environment Tasmania (NRET).

The most recent population status and monitoring report (Moreno et al., 2025), which analysed the full dataset from 2011 to 2024, found that size composition data collected from 2012 to 2021 showed a significant increase in the median size of female Maugean skate and a marked decline in juveniles and sub-adults, suggesting severely impaired recruitment for approximately eight years. A number of small juveniles were caught in 2022, with some individuals from that cohort also caught in 2023 and 2024, indicating that at least one significant recruitment event may have occurred since the monitoring program re-started in 2021. Despite the size composition data suggesting eight years of little to no recruitment between 2014 and 2021, examining abundance of skate individuals per birth year (which represents the year they hatched from their egg) from all catch data (2011-2024) based on back calculated ages, demonstrated that there had in fact been recruitment since 2014, noting that there is a time lag between when recruits appear and when they are captured in nets due to selectivity, but these recruitment levels are lower than those observed prior to the marked decline in recruitment in 2009 (Moreno et al., 2025), noting that the well documented decline in dissolved oxygen in the harbour commenced in approximately 2009 (Ross and Macleod, 2017). Moreno et al. (2025) also demonstrated a significant decline in CPUE between sampling in 2014 and in 2022 within the harbour. However, relative abundance stabilised in 2023 and 2024. This timing aligned with an improvement in environmental conditions within the harbour, in particular an increase in dissolved oxygen levels (EPA, 2024). While mean CPUE in 2024 was higher than in 2014, the standard errors were much larger than previous years, suggesting increased variability in catch rates, which may be linked to a change in behaviour, possibly driven by environmental changes and seasonality that affects catchability.

Here we report on the current data set, which includes data collected from 2025 monitoring surveys.

The objectives addressed in this report are:

- Assess the size and age (estimated) composition of Maugean skate catches as an indicator of population change, in particular recruitment variability.

- Compare catch per unit effort (CPUE) changes through time within Macquarie Harbour to describe changes in relative abundance.

- Test if the addition of 2025 tagging data can enable the use of more sophisticated CMR abundance models that are better suited for this population, but that were previously non converging due to the sparsity of recaptures.

- Use the current data for simulation-based power analysis to provide a prospective assessment of sampling and provide design recommendations.

Methods

Sampling methodology and historical data

Gillnet surveys were conducted in Macquarie Harbour, Tasmania at roughly three-month intervals between February 2021 and November 2025 to coincide with the austral seasons, noting that previous analysis demonstrated that CPUE was consistent across seasons (Bell et al., 2016). In 2024, sampling was only conducted between August and October due to a prioritisation for the research team to secure an ex-situ population of Maugean skate at the IMAS Taroona facility in the first half of the year, and then adverse weather through June and July.

Since 2021, surveys have primarily been conducted in three core areas within the harbour based on previous knowledge of skate movement and distribution as determined by acoustic telemetry, and areas of higher skate abundance identified through earlier systematic random sampling across the harbour (Bell and Lyle, 2016; Bell et al., 2016; Moreno et al., 2020, Treloar et al., 2016). Importantly, these three areas were comparable with the survey design employed in 2014 in terms of geographic location and fishing intensity, allowing 2014 to be considered a reference point for the post-2020 population monitoring program, noting that historical population baseline data that legitimately can be compared to 2021-2025 data (see Summary of available data section below for more details) does not exist prior to 2014. These sites were (A) the Table Head / Liberty Point area, (B) the World Heritage Area, and (C) the Swan Basin area, which includes Pine Cove, Long Bay and Swan Basin (Figure 1).

Maugean skate were captured using standard monofilament graball nets (50 m long by 33 mesh drops with a 114 mm stretched mesh size). In 2014, and from 2021 onwards, nets were only set during daytime and soak times were restricted to under 2 hrs (14 net deployments were excluded from subsequent analysis to avoid confounding effects of extended soak times). In general, all Maugean skate captured from 2021 onwards were measured for total length (TL), which was also used in age estimation (see back-calculated ages section), sexed, tagged, tissue samples collected, sexual maturity determined for females using ultrasonography and for males using clasper calcification, body condition assessed using a biometric impedance analysis (BIA) unit, 0.5 -1.0 ml of blood extracted via a caudal venipuncture, and a photograph taken before being released at the capture location. On some sampling occasions, however, procedures were limited to morphometric measurements, tagging and genetic tissue sampling due to unexpected circumstances such as high catch rates within a short period, rough weather, or poor holding conditions. Skate were submerged in a recirculating processing tub throughout the procedure (except for length measurement), with the dissolved oxygen levels kept at 100% through supplementary oxygen. The processing of each individual was completed in less than 10 minutes. Only one pre-landing skate mortality has occurred between 2021 to 2025, which was due to a ‘lice’ event while the animal was in the gillnet. Note that while the animals that cause these events are colloquially referred to as lice, they are in fact benthic amphipods of up to approximately 10 mm length that can ‘bore’ through skin and muscle while the skate is held unable to move in the net. As such, the current net deployment method has been designed to minimise lice interactions and mitigate risk of mortality by reducing set times and not netting at night.

All animal handling and processing procedures were conducted under University of Tasmania Animal Ethics approvals (A23857 and A30685). The research was conducted under NRET Permits to Take Threatened Fauna for Scientific Purposes: TFA20167; TFA22271; TFA23014; TFA24130; TFA23035;

TFA24272 and TFA25167 and NRET Living Marine Resources Management Act Scientific Research Permits: 20075; 21080; 22018; 22091; 24056; 23087; 24146 and 25177.

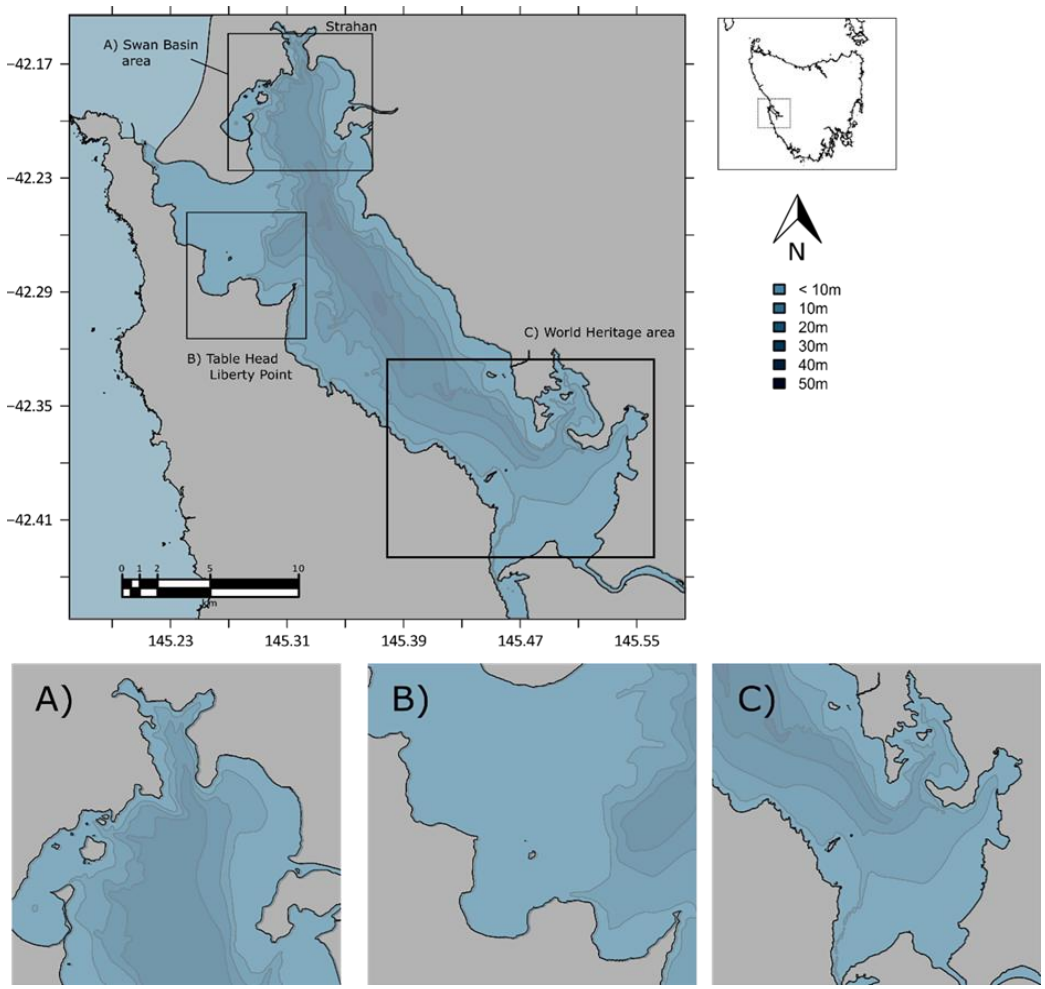

Figure 1. The primary sampling areas for Maugean skate population monitoring with inserts showing the complex bathymetry of (A) Swan Basin, (B) Table Head Liberty Point, and (C) the World Heritage Area.

Summary of available data

The sampling gear (gillnets) and broad methodology used throughout the current and previous studies (2011-2025) has not changed. However, the distribution, intensity and timing of sampling has varied due to different project objectives (i.e. assessing the impacts of gillnetting in the harbour, sampling to deploy acoustic tags, collection of skate for physiology experiments, etc.). An essential component of making CPUE a viable metric of population trends is to ensure that sampling biases are not introduced by changes to methodology that may affect catchability. For the Maugean skate dataset, this could only be ensured in years where a random stratified sampling design was implemented. All of the project objectives listed above involved targeted, repeated sampling, which does not meet this requirement of using a stratified random design and are therefore not suitable for use in population comparisons. Consequently, data collected under these specific project objectives should be excluded from CPUE estimates. Further, netting involves logistic and ethical considerations such as limiting deployment time to avoid bycatch or limiting risk of predation, which can affect catch rates independently of population abundance and must therefore be accounted for when comparing CPUE across years. Finally, all fishing methods are generally accompanied by size selective catchability bias.

Despite the caveats listed above, the time series available from the netting surveys provides the only currently available, longer-term, cost-effective option to assess changes in the population through time.

Specifically, this dataset can provide information on:

1. Changes in the size / age structure of the population
2. Retrospective estimation of birth cohorts through age back calculation
3. Changes to relative abundance through time
4. Changes in the relative importance of key skate habitat utilisation through time
5. Temporal and spatial patterns of health indices and reproductive maturity and fecundity (not included in this report)
6. Temporal patterns in population genomics, including indices of diversity and genomic health to inform conservation genomics (not included in this report)

2014 was selected as the reference year as it is the only pre 2020 dataset that exists which had both a comparable sampling design and temporal and spatial coverage (since the purpose of 2014 sampling was to provide a latitudinal assessment of Maugean skate abundance). To minimise the introduction of bias and ensure that results could be meaningfully compared through time post 2014, only catch data from sampling years that met the following criteria was included in the demographic structure and CPUE analysis:

1. Total fishing effort per year: > 300 net deployments to meet a sufficient sample size for comparison to the 2014 data (see Figure 10).
2. Spatial distribution of effort: Each sampling trip covered the three historical areas of high skate abundance informed by early broad fishing surveys and knowledge of movement and space use from acoustic tracking.
3. Temporal distribution of effort: The effort must be distributed across a comparable temporal distribution (more than three sampling trips spread across at least three different months and two different seasons).
4. Purpose of sampling: Random stratified design for purpose of estimating relative abundance.

Subsequently, sampling in 2021, 2022, 2023, 2024 and 2025 met the criteria listed above and were used for subsequent CPUE and length frequency analyses and comparison to the reference year (2014) data (Table 1).

Data analysis

Size and age structure

Maugean skate in Macquarie Harbour constitute a single homogeneous population (Weltz et al., 2018; Moreno et al., *In review*). There are no site-specific differences in sex or size frequency distribution (Bell et al., 2016). Therefore, analysis of size and age frequency data was conducted for total catch across the three reference sites combined for the years 2014 and 2021-2025. Sex specific size dimorphism is a feature of Maugean Skate (Bell et al., 2016), as such, size (TL) and age frequency composition was reported by sex. Individual skate with missing sex or size data were not included in size or age frequency composition analysis.

A Kruskal Wallace (KW) test was used to test for Rank (~median) differences in size by year. A post hoc planned comparison of rank differences (~median) using a Dunn test with a Bonferroni correction was used to test for significant differences for each sampling year from 2021 against the reference year (2014). A post hoc planned comparison of mean size differences using a Wilcox test with a Bonferroni correction was used to test for significant differences for each sampling year from 2021 against the reference year (2014). Significant differences in size frequency distribution for each sampling year from 2021 against the reference year (2014) were tested using a bootstrapped Kolmogorov-Smirnov test with a Bonferroni correction. Equivalent analyses were not conducted for age frequency data as ages were estimated from total length (see below), resulting in autocorrelation between age and length, making age-based results indistinguishable from the size frequency analysis.

The ages of 44 Maugean skate sampled from Macquarie Harbour between 2012 and 2019 were estimated using incremental banding in their vertebrae, with a multi-model inference framework and Akaike's information criterion to determine the best fit growth function based on length-at-age data (van Werven et al., 2025). The parameters of a Von Bertalanffy growth model (VBGM) were used to estimate the age of each animal caught based on their measured length. The yearly sex specific age structure of the catch is reported and compared to the reference year (2014).

Back calculated ages

To provide a retrospective analysis of recruitment, birth year was calculated based on the estimated age at capture for each individual in the non-standardised dataset (all individuals caught in Macquarie Harbour from 2011 to 2025 regardless of project objectives or sampling design). To avoid double counting individuals, for all recaptures only the first record was used. The number of individuals in each yearly cohort represents the observed or known recruits. The yearly index was plotted as a time series and individuals inside each yearly cohort were classified by year of capture.

Relative abundance and seasonality

To explore changes in relative abundance through time, data from 2014 and 2021-2025 were analysed from all areas combined and the three areas separately. Sampling in these years was restricted to daytime only and had a similar spatial and temporal design, meeting the criteria outlined above. There were 13 longer duration deployments in 2014, where soak times were greater than 5 and a half hours. These deployments were excluded from CPUE calculations to account for potential effects of being significantly longer than all other sets assessed. A further eight individual deployments (less than 0.1% of all net sets) that had incorrect or missing metadata records were excluded as it was not possible to calculate CPUE (2014 = 1, 2021 = 1, 2022 = 4, 2023 = 2). While these records were excluded from CPUE calculations, they were retained for analyses of size and age-frequency distributions and for back-calculation of individual ages, as these metrics are not influenced by deployment duration or missing effort metadata.

Observed CPUE was estimated as the number of Maugean skate caught per net (50 m) per hour (N/net/hr) for each of the included sampling years. Catch per unit effort metrics were calculated for the entire Harbour. Standard error was calculated based on CPUE per season to show within-year variability in catch rates.

Catch per unit effort was also modelled as a function of time using generalized linear models (GLMs), which are widely used for catch standardisation (Campbell, 2004). Catch data were standardised by calculating skates per net per hour (N/net/hr) for each individual deployment and modelling the resulting CPUE values. This is equivalent to modelling catches against an offset term for ln(effort), an approach commonly used in fisheries (Haddon et al., 2018). However, while both approaches produced identical patterns of CPUE per year, modelling CPUE directly allowed us to retain the same scale as the observed CPUE. Models were fitted using the R statistical environment (R core team, 2024). Given the large number of zero catches and resulting overdispersion in the data, negative binomial GLMs with a log link were fitted using the glm.nb function in the MASS package (Venables and Ripley, 2002), allowing the use of untransformed response data (Campbell, 2004). Considering recent increases in

interannual catch variability and changes in site-specific catches (Moreno et al., 2025), spatio-temporal effects were examined by including Year and Site as explanatory variables. Four candidate models were evaluated: a null (intercept-only) model, models including year or site as main effects, and a model including year, site, and their interaction ~Year, ~Site, and ~Year × Site) and model performance was compared using Akaike's Information Criterion (AIC). Standardised CPUE values were visualised using the VISREG package (Breheny and Burchett, 2017).

While previous analyses suggested that catch rates in 2014 and 2021 did not different significantly by season (Bell et al., 2016; Moreno et al., 2024), an observation of significantly increased catch rates in spring 2025 compared to winter and autumn 2025 prompted a more detailed examination of seasonality using one-way ANOVA. Catch per unit effort values were first analysed across all years combined (2014 and 2021–2025) to test whether mean CPUE differed significantly among seasons. Additionally, separate one-way ANOVAs were performed for individual years to assess intra-annual seasonal variation. For years where a significant effect of season was detected, post-hoc pairwise comparisons were performed using Tukey's HSD post-hoc tests. Prior to each ANOVA, residuals were examined for normality and homogeneity of variance to ensure the assumptions of the test were met.

Tagging and mark-recapture population estimation

From 2012 to 2014, 275 individuals were tagged with passive integrated transponder (PIT) tags placed under the skin and included in a Capture Mark-Recapture (CMR) model (Bell et al., 2016). Absolute population abundance based on the total number of tagged and recaptured skate was estimated at 3177 (CI: 1827 – 6247) (Bell et al., 2016). It was noted, however, that due to the low number of recaptures, many closed population models failed to converge. The best fitting closed model that did converge, based on the Akaike Information Criterion, was the Chao Lower Bound model (Mth), which accounts for both time-dependent variation in capture probability and behavioural responses to capture. Although this model converged, the resulting confidence intervals were broad, reflecting substantial uncertainty around the estimate, limiting its usefulness. It should be noted that these models were fitted due to their applicability to sparse recapture data, although some of the underlying assumptions of a closed population, e.g. no births, deaths, immigration, or emigration, are likely not to be appropriate for the Maugean skate. However, to date, available data has been insufficient to allow the use of more appropriate but more complex and data intensive approaches, such as open or state-space tag recapture models. Therefore, abundance estimates generated by these closed population models should be considered carefully and not be interpreted without the appropriate caveats.

More recently, CMR models were fitted to Maugean skate recapture data from 2021-2024 (Moreno et al., 2025). The models used failed to converge due to the scarcity of recapture records. A simplified Schnabel closed population model was successfully fitted to the data, however it did not account for time-dependent variation in capture probability and behavioural responses to capture and similarly produced a very wide estimate range (1602–15055), which, together with the assumptions associated with closed population models, limits its usefulness for population assessments.

Since 2021, 317 individuals have been tagged with dart tags, including 201 that were double-tagged in 2024 and 2025. Double-tagging has been introduced to determine tag loss, which is an important component of mark-recapture modelling. Further, genetic samples have been collected since 2021 to enable genetic identification of recaptures in cases where tag loss has occurred.

Since 2021 there have been an additional seven dart tag recaptures (Total, 2021=1, 2024=2, 2025 = 4) and three genetic recaptures (Total, 2023 = 1, 2024 = 2).

Despite the increase in recaptures, it is still not possible to fit more complex models that may produce an improved population estimate. Given that the converging models do not differ meaningfully from those reported previously, and no alternative models are yet working, no new population estimates are presented. As recapture numbers increase, we will continue to explore the use of alternate models that

may account for tag-loss (estimated through genetic identification), spatio-temporal variation in capture probabilities, recruitment and mortalities (open population), and behavioural responses.

Sampling design

The analyses presented in this report show significant spatio-temporal changes in CPUE as well as a recent increase in within-year variability. Whilst techniques like CPUE standardisation can help account for some of these changes, it is crucial that the sampling design accurately characterise these potential sources of uncertainty. As such, sampling design for the monitoring program needs to be periodically revisited to ensure that it is still fit for purpose. To this end, we conducted a series of power analyses and present recommendations for adjustments to the current sampling design.

To determine the influence of sample size (number of net deployments per year) on the statistical power to detect a significant effect in CPUE across years, power analysis was conducted on a simplified (no interactions) version of the GLM used to standardise CPUE (see above). The power analysis function from the ZIPowerAnalysis R package (Dennis et al., 2025) was modified to work with our model structure (i.e. negative binomial GLM without a zero inflated component). This method utilises a Monte Carlo simulation, where inside each iteration (N=1000), a dataset is simulated where the response variable is generated based on a mixture of Poisson and Bernoulli random variables, the original model coefficients (including theta) and simulated covariates. New models are fitted to the simulated datasets, and power is calculated based on the proportion of runs where the p value of the desired coefficient is below a pre-determined alpha (0.05). Upper and lower (95%) estimates for power were calculated from the binomial probability using the bayes method in the binom package in R. This was repeated for a series of sample size values ranging from 200 to 1000 to construct a power curve, with a standard power threshold of ≥ 0.8 (80%) considered sufficient to reliably detect an effect (Serdar et al., 2021).

Considering the evidence for recent increases in within-year variability in CPUE (Moreno et al., 2025), we performed a separate power analysis to explore the potential effect of changing how the sampling effort is spread across the year. Data was simplified by calculating the mean CPUE per season, so that the number of observations each year was equal to the number of unique sampling events. We simulated the effect of increasing the number of sampling events by modifying the number of observations within each year. A simplified model with a Poisson distribution was fitted to the data and power analysis was conducted using the SIMR package in R (simulation N=1000) (Green and MacLeod, 2016). Power analysis was built based on a fixed effect size equivalent to an approximate 10% change in CPUE between years (informed by the coefficients in the original model calculated using the fixef function), with a standard threshold of 0.8 used to define adequate power (Serdar et al., 2021).

Results and Discussion

Catch summary

A total of 851 individual skate have been captured since 2011 (female = 389, male = 402, not recorded=60; the number of juveniles (< 600mm) = 128, number of adults (≥ 600mm) = 671, not recorded=52). The smallest individual captured was 215 mm TL (2022) and the largest was 870 mm TL (2012) (Appendix 1).

Size and age structure

Size frequency

The size of females varied significantly across years (χ 2 = 11.023, p = 0.05) (Figure 2). Compared to the 2014 reference year, median female size was significantly larger in 2021 (D = 38.3, p = 0.02) but

not in subsequent years (Figure 2). Mean female size was significantly larger relative to the reference year in 2021 ($p < 0.01$), 2022 ($p = 0.03$) and 2024 ($p = 0.01$) but not in 2023 ($p = 0.14$) or 2025 ($p = 0.051$), with these years perhaps influenced by the capture of several juveniles which were absent in the net samples in the other years (Figure 2). Similarly, the overall size frequency distribution of females was significantly different from the 2014 reference year in 2021 ($D = 0.32$, $p = 0.02$), 2022 ($D = 0.34$, $p = 0.02$), 2024 ($D = 0.20$, $p = 0.03$) and 2025 ($D = 0.26$, $p = < 0.01$) but not in 2023 ($D = 0.20$, $p = 0.07$), reflecting the influence of juvenile captures in 2023 (Figure 2).

The size of males differed significantly by year ($\chi 2 = 9.11$, $p = 0.05$) (Figure 2). Compared to the 2014 reference year , median male size was significantly lower in 2022 ($D = -39.7$, $p = 0.04$), influenced by the capture of juveniles, but not in other years (Figure 2). Mean male size was significantly different to the reference year in 2022 ($p < 0.01$), again influenced by the capture of juveniles but not in other years (Figure 2). Size frequency distribution of males differed significantly from the reference year in 2021 ($D = 0.22$, $p = 0.05$), 2022 ($D = 0.36$, $p = 0.02$), 2023 ($D = 0.26$, $p = 0.04$), 2024 ($D = 0.18$, $p = 0.04$) and 2025 ($D = 0.24$, $p = < 0.01$) (Figure 2).

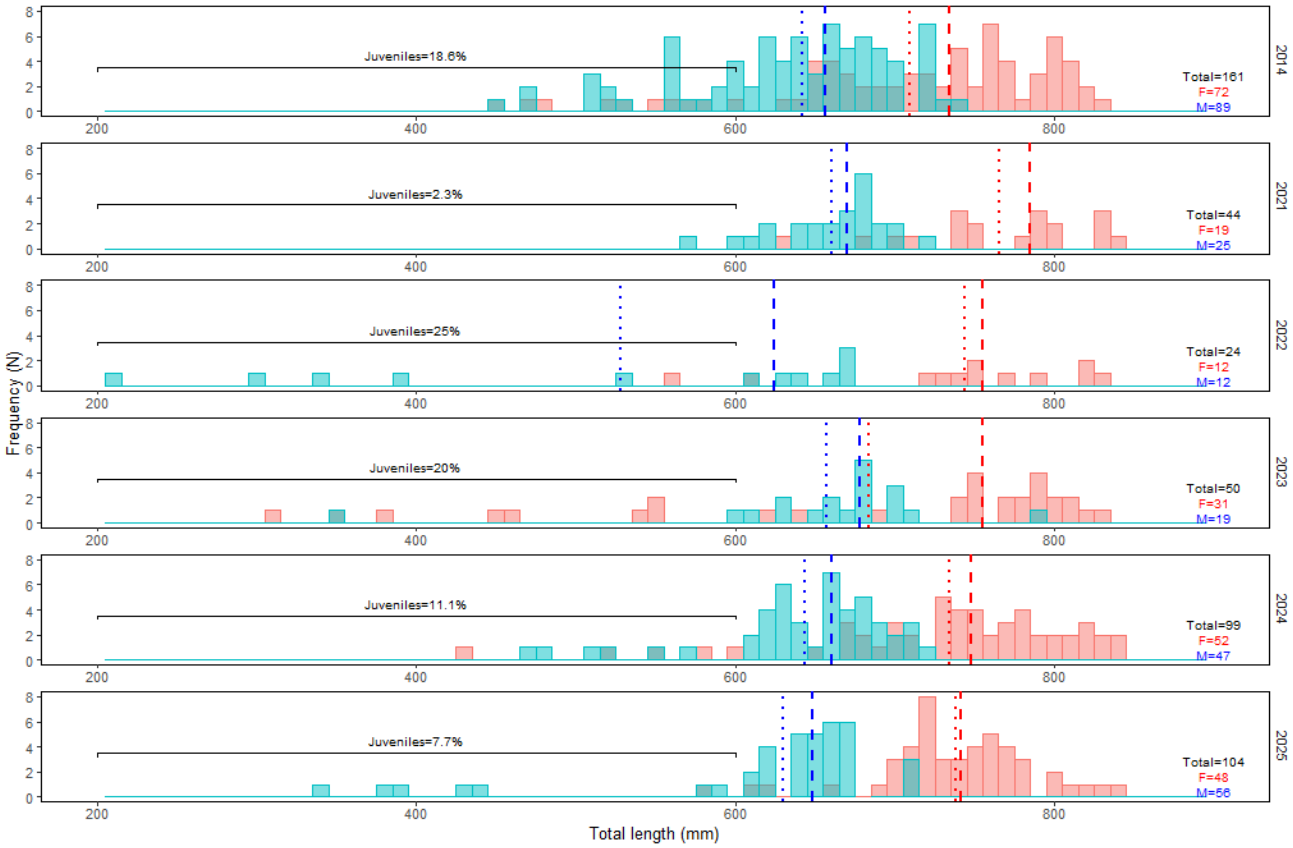

Figure 2. Size distribution of male (blue) and female (red) Maugean skate sampled from all sites during 2014 (top panel; FRDC 2013-008), 2021-2023 (second to fourth panels; SMRCA study), and 2024-2025 (fifth and sixth panels; current NRE study). Vertical dashed and dotted lines indicate median and mean lengths of males (blue) and females (red), respectively. Sample sizes are indicated (Total = sexes combined, M = males, F = females). Values below the horizontal bracket indicates the proportion of individuals < 600 mm (juveniles and sub-adults) present in each sample. Note some individuals were released without length measurements.

Proportion of juveniles

In 2021, juveniles/sub-adults comprised only 2.3% of total skate catches across the harbour, the lowest proportion recorded across all sampling years (Figure 2). While the net sampling method does impart a size selective bias with underrepresented smaller individuals (see Back-calculated ages section), this bias is consistent across years due to standardised gear and methodology. Therefore, the paucity of

juveniles raised concern regarding potential failures in hatching success, juvenile survival or recruitment between 2014 and 2021 (particularly when examined in conjunction with the decline in CPUE in this period – see catch, effort and seasonality section below).

During net sampling in 2022, multiple small sized juveniles (<400 mm TL) were observed for the first time since sampling in 2014 (see Moreno et al., 2024 for more details). Despite low overall catch numbers (and the lowest CPUE in the entire dataset - see Catch, effort and seasonality section below), recruits comprised 25% of sampled individuals in 2022, exceeding the proportion recorded in 2014 (18.6%) (Figure 2). Furthermore, in the three subsequent years, juveniles were caught in the size ranges that would be expected based on yearly growth of the juveniles captured in 2022, though at lower proportions than in 2022 (accounting for 20% in 2023, 11% in 2024, 7.7% in 2025 of the sampled population) (Figure 2). A new potential cohort of juveniles was also observed in 2025 (Figure 3).

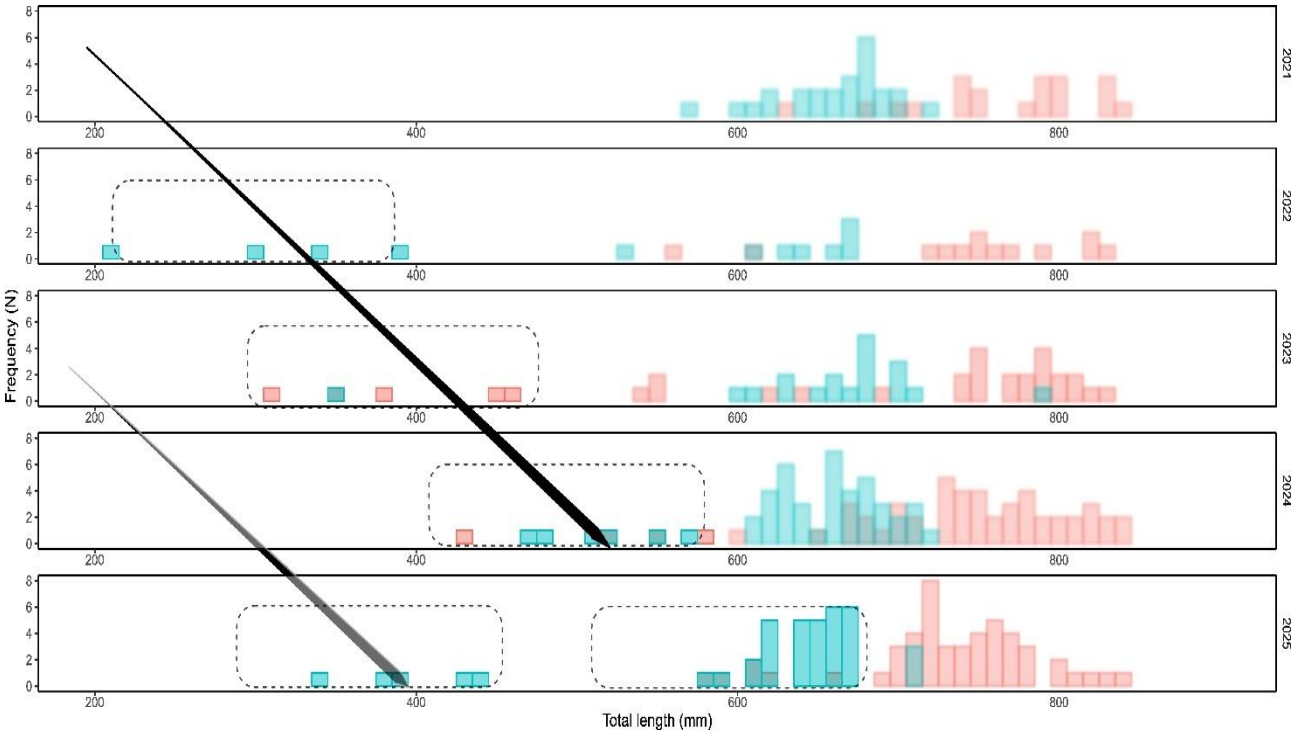

Figure 3. Size frequency distribution of catch from 2021-2025 showing observed similar sized clusters (cohorts) of juveniles (dotted boxes). Black arrows show the transition in size classes between years of suspected juvenile cohorts, corresponding with expected annual growth (arrows are not extended to the year where adult sizes are reached given that male and female growth rates diverge significantly at that point (Awruch et al., 2021)).

Age structure

The median estimated ages of both female and male skates have remained largely stable across the study period (Figure 4). The median female age has increased marginally from 2014 (7.30 years) to 2025 (7.34 years). Similarly, the median age of male skates showed almost no change from 2014 (5.55 years) to 2025 (5.51 years) (Figure 4). A slight shift towards older age groups is evident, in 2021 (female median age 7.58 years; male median age 5.62 years), due to the general lack of juvenile skate in that year.

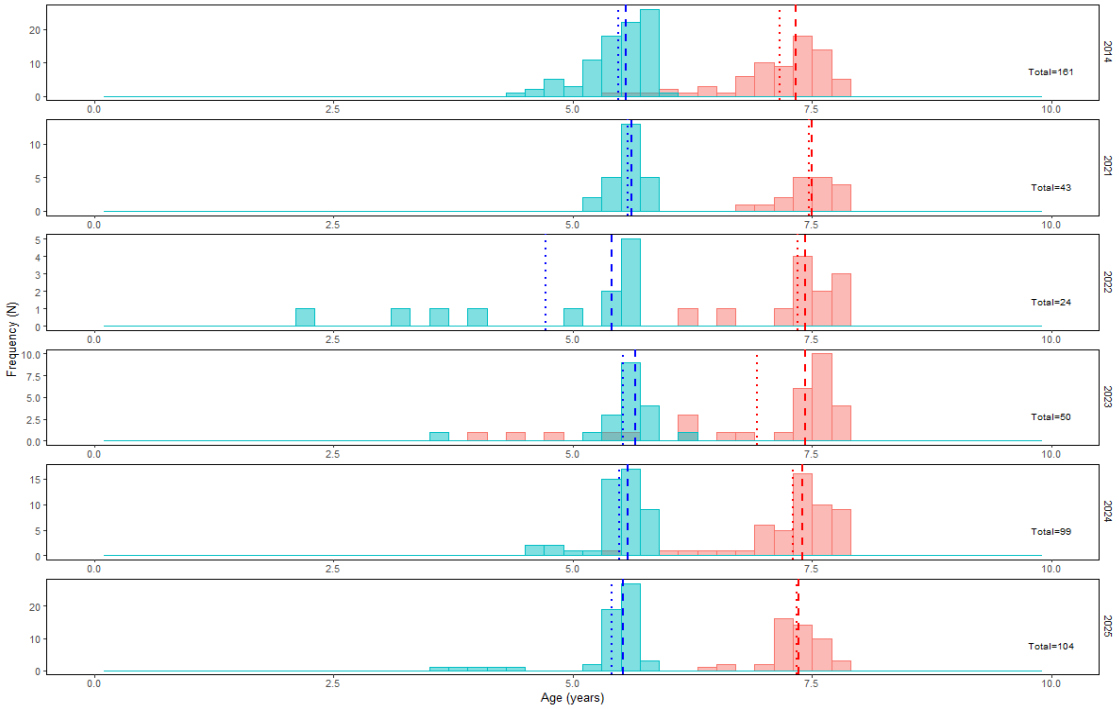

Figure 4. Calculated age frequency distribution of male (blue) and female (red) Maugean skate sampled from all sites during 2014 (top panel; FRDC 2013-008), 2021-2023 (second to fourth panels; SMRCA study), and 2024-2025 (fifth and sixth panels; current NRE study). Vertical dashed and dotted lines indicate median and mean ages of males (blue) and females (red), respectively. Sample sizes are indicated (Total = sexes combined). Note that there are differences in the scope of the y-axis, this is to highlight the relative variation in age structure rather than the magnitude of different age classes represented across sampling years.

Back-calculated ages

The derivation of modelled ages based on size at capture allowed for retrospective categorisation of all individuals into cohorts by year of birth. This approach provides some insight into birth and recruitment patterns despite the size selectivity bias of the sampling gear. However, individuals from each yearly cohort are assigned based on capture at different sizes and ages, and the size specific catchability coefficients are not known. Therefore, it was not possible to standardise these numbers by catch effort as per the CPUE calculations (next section). As a result, this analysis should not be interpreted as a straightforward hatching success index, but rather, should be considered as the minimum observed yearly births. Furthermore, given that most individuals are caught at adult sizes, this data also provides insight into the rate of juveniles reaching adulthood and joining the reproductive population each year.

Observation of the plotted back calculated ages showed that there is an approximate lag of 4-5 years between the time individuals are born and when they are caught in monitoring surveys (Figure 5). Note that the age lag is only an approximation, as selectivity is driven by size. This means that due to the under-representation of juveniles in the catch when using this capture methodology, there will be at least a 4-5-year hysteresis until individuals reach the size where they are better represented and the full extent of a potential recruitment event can be assessed. Likewise, individuals were not caught more than 9 years since the estimated date of birth, as the maximum modelled age of the species is 10 years (van Werven et al., 2025).

Births from the 2006 to 2008 cohorts were relatively consistent and represented the highest number of observed births detected across the time series (Figure 5). These cohorts primarily comprised individuals caught from 2011-2014, and the proportional contribution of each capture year remained consistent across cohorts (e.g. the number of individuals caught in 2014 seen in 2006, 2007, 2008, and 2009 was consistent) (Figure 5). From 2009 to 2011 there was a significant decline in yearly observed births compared to 2006-2008, even though fishing surveys occurred in 2014 and 2015, where individuals born in those years would be expected to occur and be large enough to be captured

consistently. This corresponds with a prolonged decline in dissolved oxygen concentrations in Macquarie Harbour, which began in 2009 (Ross and MacLeod, 2017). Therefore, it should be noted also that although size structure and CPUE from 2014 is used as a reference year for analytical and comparative purposes, analysis of skate abundance by back-calculated birth year indicates that a decline in hatchling production had already occurred prior to this time (Figure 5). Consequently, this reference year should not be interpreted as a target reference state or baseline for population rebuilding, but rather as a temporal benchmark within a broader period of reduced recruitment.

Between 2011 and 2021 there was a consistent pattern, where the proportion of individuals caught each year was allocated in similar proportions across 4-5 estimated birth years (Figure 5), albeit at lower numbers than 2006-2008. Two high-impact environmental events occurred in 2019 and were associated with significant inferred mortality of acoustically tagged skates (Moreno et al., 2020). These events may have reduced recruitment either by negatively affecting adult reproductive success or by decreasing juvenile survival, thereby reducing the number of individuals available to recruit into the adult population. The relative contribution of these cohorts will become clearer through continued monitoring as individuals from these hatching years become fully selected by the netting gear over the next several years.

Systemic population monitoring surveys began in 2021, and 2025 was the first year where individuals born inside that period would be detectable as adults. Indeed, the large number of individuals in the 2020-2021 cohorts that were caught in 2025 is consistent with juvenile catches from 2022-2024 (see age structure section). This shows for the first time evidence that at least a portion of the observed juveniles in these years survived into adulthood.

In recent years, the recruitment dynamics observed are consistent with a pulse-recruitment model, in which population recovery is likely driven by episodic periods of favourable environmental conditions. This is best shown by the variability in the proportion of individuals caught in 2024 and 2025 that are assigned to each birth year (Figure 5). For example, many individuals caught in 2025 were born in 2018 or 2020 (~50 individuals each), whereas fewer than a quarter were born in 2019. Notably, this pattern is absent from catches preceding 2021 and including the reference year 2014. Furthermore, evidence for pulse-driven recruitment is also apparent in interannual variation in size structure (Figure 3), with discrete cohort groups evident in some survey years. These patterns are consistent with episodic pulses of juveniles entering the population rather than steady annual recruitment. Successive pulses are not consistently replaced by new cohorts in subsequent years, resulting in gaps in size structure as cohorts progress through the population.

Recruitment under a pulse model is inherently year-to-year dependent, and the presence of one or a small number of strong cohorts may be insufficient to sustain population recovery. Multiple consecutive years of successful recruitment will likely be required before a persistent recovery trajectory can be confidently inferred. Furthermore, these dynamics mean that if a specific pulse of recruits is affected by adverse environmental events, this could rapidly suppress overall recruitment for multiple years and have downstream effects for the demographic structure of the population.

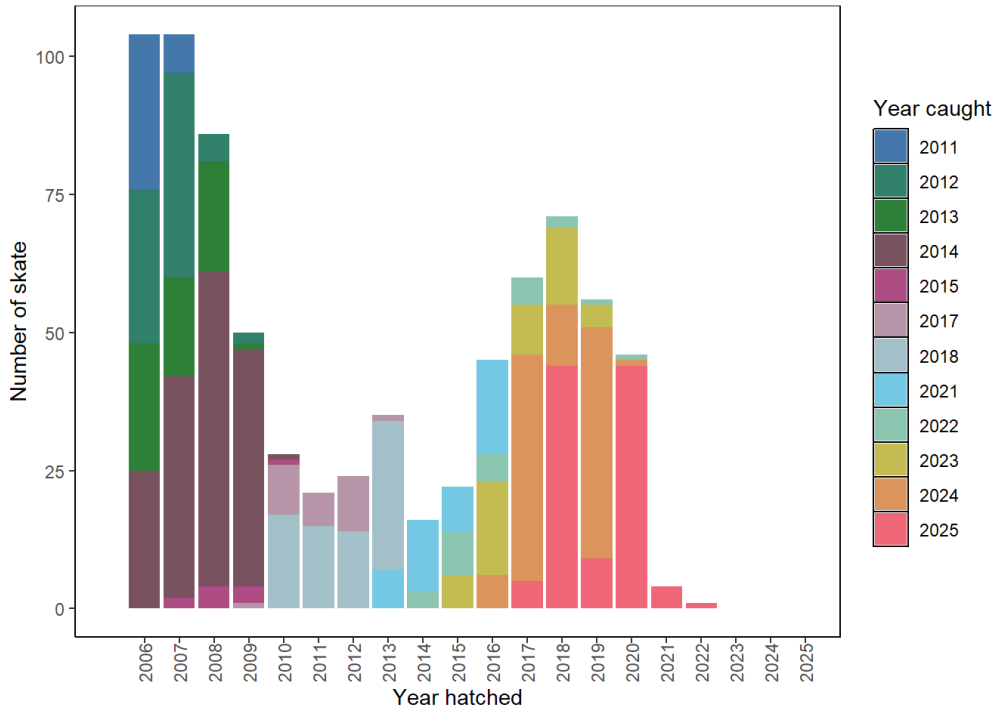

Figure 5. Abundance of skate individuals per birth year from all catch data based on back calculated ages. The colour indicates the year individuals in each birth cohort were caught. Individuals hatched in 2021 and possibly 2020 are likely not of a size where they are fully catchable by the nets, hence data from those years are possibly under-represented.

Catch, effort and seasonality

The filtered dataset analysed in this report for CPUE calculations based on the criteria outlined in the methods section (years 2014, 2021, 2022, 2023, 2024 and 2025) consisted of 2167 individual gillnet deployments in Macquarie Harbour and a total of 490 captured skate (female = 235, male = 250, not recorded = 5, < 600mm = 61, > 600mm = 416, not recorded = 13) (Table 1). The smallest individual captured was 215 mm TL (2022) and the largest was 845 mm TL (2025).

Table 1. Summary of gillnet survey effort by year within the 3 main survey areas (Table Head, Swan Basin and the World Heritage Area), excluding overnight and outlier deployments (see methods). Highlighted rows show the years included for CPUE analysis based on criteria defined in the methods section. Net deployments are the number of net deployments in the year that meet the defined criteria, Number of skate caught is the number of skate caught in those deployments inclusive of recaptures, and Areas is the number of areas sampled (3 key skate habitat areas are used for monitoring). The temporal distribution of effort (months and seasons) is also shown, as well as the number of skate tagged each year, and the number of recaptures, with the numbers in parentheses showing the days at liberty – in 2014 this is shown as the range of days at liberty.

Year	Net deployments	Mean soak time (hours)	Number of skate caught	Areas	Months	Seasons	Skate tagged	Recaptures
2012	135	2.6	59	3	3	3	84[1]	
2013	117	2.7	47	3	3	2	62[1]	
2014	328	2.7	158	3	4	4	151[1]	9[1](2-733), 1 (84*)
2015	74	2.6	10	3	1	1	9[1]	
2017	77	2.1	27	1	2	2	0	
2018	210	2.2	76	1	4	4	0	
2019	193	1.5	17	3	1	1	0	
2021	380	1.9	45	3	5	4	44[2]	1 (64)
2022	420	1.6	24	3	5	3	22[2]	
2023	300	1.8	57	3	4	3	50[2]	1 (533*)
2024	315	1.5	101	3	3	2	99[3]	4 (329, 469, 25*, 47*)
2025	424	1.6	105	3	5	3	102[3]	4 (744, 391, 377, 439)

[1] PIT tag
[2] Single Dart Tag
[3] Double Dart Tag
*Denotes genetic recapture

Catch per unit effort (CPUE)

In 2014, harbour-wide CPUE was 0.17 ± 0.03 N/m/hr (Figure 6). In 2021, CPUE declined to 0.06 ± 0.03 N/m/hr (Figure 6). In 2022, CPUE decreased further to 0.04 ± 0.02 N/m/hr (Figure 6). Given that within year variance and site-specific catch rates were consistent across these years, harbour wide CPUE patterns were directly compared. This translates to a point-to-point decline against the baseline year of 2014 to 2021 of 64%, and 77% in 2022. There were two documented impacts that could explain declines: a) limited hatching success and juvenile survival beginning in 2009/2010 (see Figure 5), and b) two high impact environmental events that occurred in 2019 (Moreno et al., 2020) and which likely resulted in widespread adult mortalities (inferred based on mortalities (~44%) of individuals being electronically tracked at the time (Moreno et al., 2020)).

In 2023 and 2024, CPUE increased to 0.11 ± 0.05 N/m/hr and 0.21 ± 0.07 N/m/hr respectively (Figure 6). This translates to a 35% decline in 2023 relative to 2014 and a 23% relative increase in 2024. With consideration of the standard errors, the declines in relative abundance reported in 2021 and 2022 were significantly different (lower) to the reference year (2014), while in 2023 and 2024, the relative abundance had increased to a point where the errors around the mean overlapped with the reference year (2014). This timing aligned with an improvement in environmental conditions within the harbour, in particular an increase in dissolved oxygen levels (EPA, 2024).

In 2025, CPUE decreased to 0.15 ± 0.06 N/m/hr (Figure 6), representing a 12% decrease relative to 2014 and a 29% decrease relative to 2024. However, the standard errors around the mean in 2025 still overlap with the 2014 reference year, consistent with patterns observed in 2023 and 2024.

Similarly to 2023 and 2024, the variability in CPUE in 2025 is considerably larger than during 2014-2022 (Figure 6), reflecting greater within year fluctuations in CPUE. During the period between 2014 and 2021, CPUE trends could be reasonably extrapolated to population-level changes as the variance remained relatively stable and catch data suggested no substantial changes in skate distribution or behaviour, independently supported by acoustic tracking data (Bell et al., 2016; Moreno et al., 2020). In contrast, the recent increase in CPUE variability may reflect changes in skate behaviour affecting catchability, such as shifts in depth distribution or altered movement patterns within the harbour both between and within years, which introduces uncertainty in the relationship between CPUE and population size.

Consequently, it is necessary that CPUE be standardised to account for these effects, as well as to reinitiate the collection of tracking data to help provide crucial context for the interpretation of Maugean skate population monitoring data.

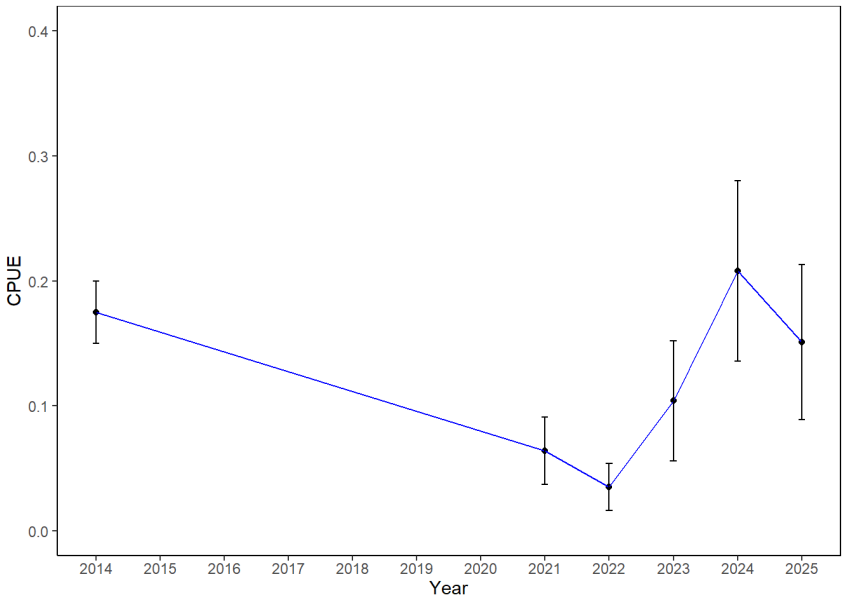

Figure 6. Raw Catch Per Unit of Effort (CPUE) of Maugean skate caught from the three reference sites in Macquarie Harbour. The error bars represent an approximate standard error calculated based on variability in skate per net hour caught between individual deployments. The blue line connecting 2014 and 2021 does not represent continuous CPUE estimates and is provided only as a visual representation of the difference in CPUE between these years.

Standardised CPUE

Catch per unit effort standardisation is an important component of using catch data for stock assessment and management, as it aims to account for factors that influence catchability across sites and through time (Hoyle et al., 2024). This is achieved by incorporating relevant covariates into statistical models so that standardised CPUE more closely reflects underlying abundance. In this study, CPUE was standardised using generalised linear models (GLMs), a widely applied approach for catch standardisation (Li et al., 2023). As the catch data contained a large number of zero observations resulting in overdispersion, models were fitted using a negative binomial distribution.

Model results indicated significant spatial (area) and temporal (year) effects on CPUE. The best-supported model, based on the lowest Akaike Information Criterion (AIC), included both covariates

and their interaction, indicating that temporal trends in CPUE differed among sites. Accordingly, site-specific standardised CPUE trends are presented (Figure 7). These models reveal a site-specific pattern, with the recent increase in harbour-wide CPUE largely driven by increasing catch rates in the World Heritage Area, despite declining trends in the Table Head and Swan Basin areas.

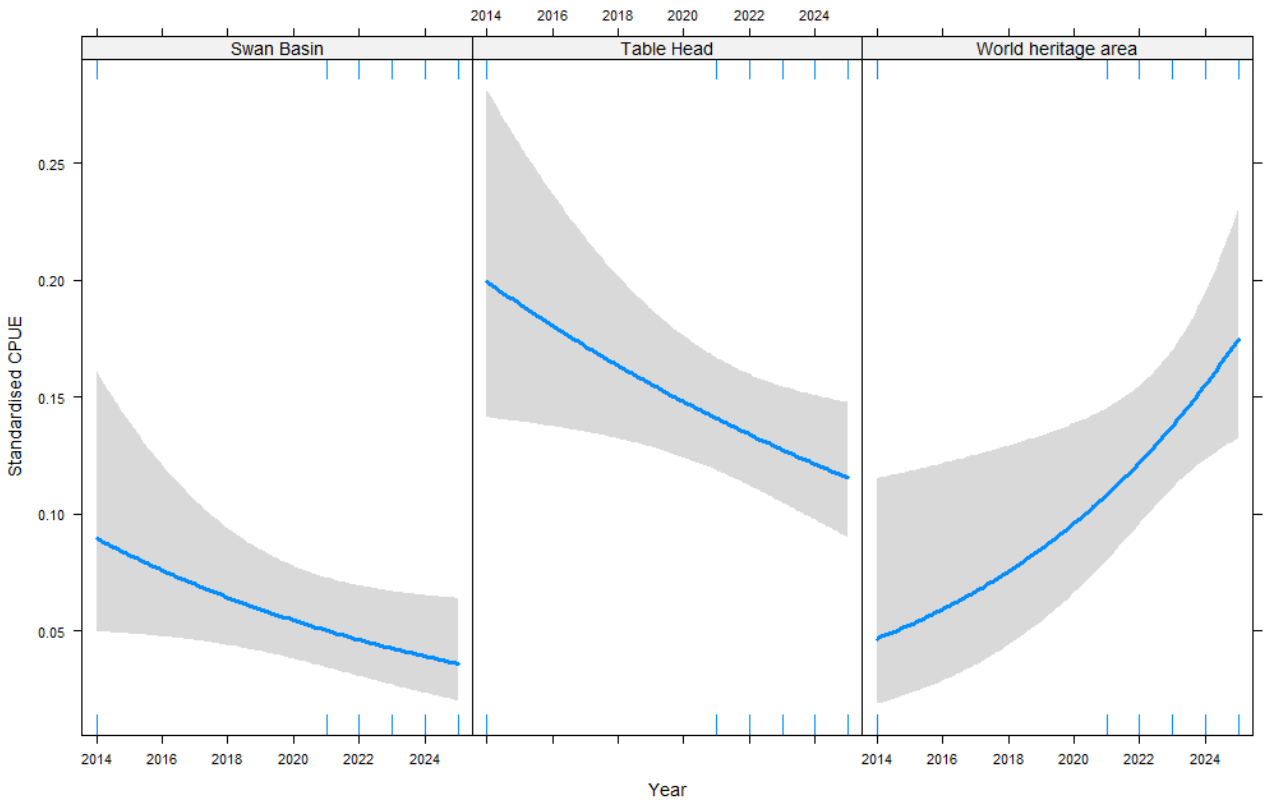

Figure 7. Standardised catch per unit effort (CPUE) (skates per net per hour)) of Maugean skate caught as a function of time from Swan Basin, Table Head and World heritage area using negative binomial GLMs. The grey shading indicates the variance around the model.

Based on the known life-history characteristics of the species (i.e. time to maturity and short reproductive lifespan) and observed size-structure data, much of the increase in CPUE at the World Heritage Area cannot be explained solely by population growth. Although some increase is likely attributable to recent recruitment events, the time between observed increases is insufficient for enough individuals to have been born and grown to sizes vulnerable to the sampling gear. Consequently, a portion of the observed CPUE increase is likely explained by movement of adult individuals among sites. Consistent with this interpretation, the World Heritage Area historically supported substantially lower skate catches than Table Head, as evident in 2014 and during 2021 when a high proportion of individuals were captured at Table Head (Figure 8). Since 2022, this pattern has changed, with the World Heritage Area contributing an increasing proportion of total catches relative to Table Head (Figure 8), further supporting the hypothesis that recent changes in CPUE at least in part reflect redistribution of adult individuals.

Figure 8. Proportion of Maugean skate catches by sampling year at the World Heritage Area (WHA; blue) and Table Head (TH; red), based on raw (unstandardised) catch counts. Points show the percentage contribution of each site to the total annual catch, with grey horizontal bars indicating the magnitude of the difference in proportional catch between sites within each year (longer bars indicate greater disparity). Total numbers of skates caught per year are shown on the right. Data are not standardised for sampling effort or catchability and therefore reflect relative spatial patterns in observed catches rather than true abundance.

Seasonal effects in catch rates

Significant seasonal variation in standardised CPUE was detected in all years analysed for CPUE except 2014, indicating that standardised CPUE varied across seasons in recent years (Figure 9). For 2014, ANOVA results showed no significant effect of season on CPUE, suggesting relatively stable catch rates throughout the year. In contrast, significant seasonal effects were observed in 2021-2025, highlighting pronounced intra-annual variability in skate catch rates, with CPUE differing markedly among seasons rather than remaining constant throughout the year. This suggests that, unlike 2014, catch rates in recent years were influenced by seasonal dynamics, potentially reflecting changes in environmental conditions and associated shifts in skate distribution or behaviour. Importantly, the nature of this seasonal effect was not consistent across years. For example, CPUE was highest in winter in 2023, but was highest in spring in 2024 and 2025. This lack of consistency suggests that recent seasonal patterns are unlikely to reflect a fixed or predictable cycle. Instead, within-year variability in catch rates is likely influenced not only by seasonally fluctuating variables, but also by freshwater inflow, prevailing weather patterns and their interactions (Ross and Macleod, 2017), resulting in distinct environmental conditions and catch dynamics among years.

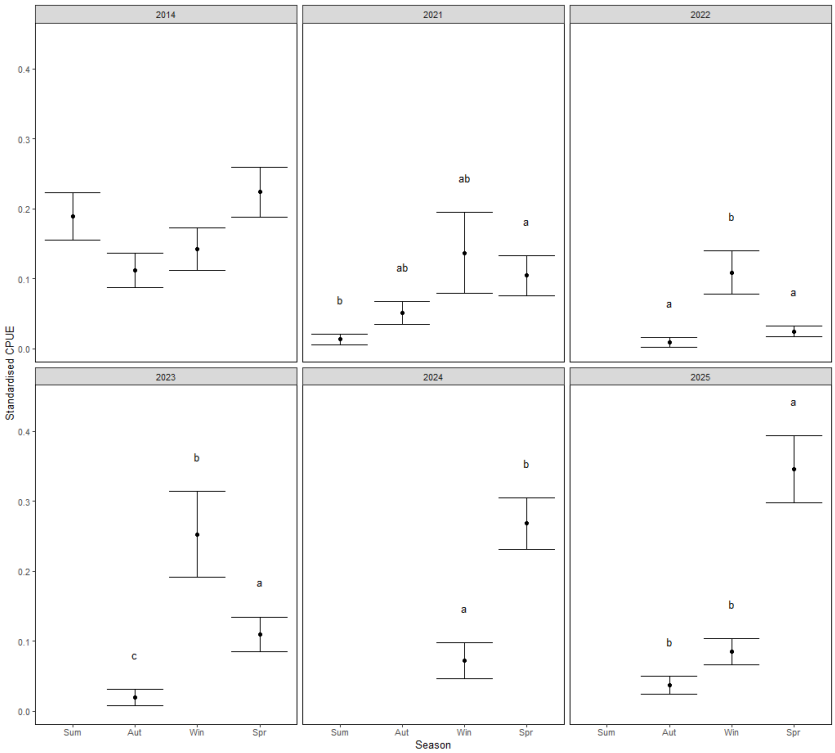

Figure 9. Seasonal variation in standardised catch per unit effort (CPUE) across years. Points show mean CPUE for each season (Summer, Autumn, Winter, Spring) within each year, with error bars representing ± standard error. Letters above points indicate results of Tukey HSD post-hoc comparisons within each year following one-way ANOVA; seasons sharing the same letter are not significantly different ($p > 0.05$). Only years with significant seasonal effects display letters.

Tag recaptures

A total of seven dart-tagged skate have been recaptured since 2021 (time at liberty > 1 day), as well as three confirmed genetic recaptures, where the skate had lost 1/1 or 2/2 (2 individuals) dart tags, for a total of 10 recaptures from 317 releases (see Table 1). All recaptured individuals were detected at the same site at which they were originally tagged. This current reporting period (2025) consisted of 4 recaptures and 7 same day recaptures (tagged and recaptured on the same day).

Despite recent recaptures in 2025, the available data remains insufficient to support a meaningful update to the population estimate. Previous population estimates have been derived using closed population models, as more complex open or state-space tag-recapture models have failed to converge given current sample sizes and recapture frequencies (Bell et al., 2016; Moreno et al., 2025). Although the inclusion of recent recaptures does not result in a significant change to the closed population point estimate, these models rely on strong assumptions of demographic and geographic closure (i.e. no births, deaths, immigration, or emigration), which are unlikely to be met across the temporally protracted time series of recaptures and limit biological relevance. While confidence intervals have continued to narrow marginally with additional recaptures, the changes are not statistically meaningful and do not warrant reporting an updated estimate. As further Maugean skate recaptures are obtained, continued model development and evaluation of more complex tag-recapture frameworks will be undertaken as data availability improves.

Sampling recommendations

Evidence of spatiotemporal variability, including within-year effects indicates that the sampling design should continue to retain site specific coverage and a strict seasonal framework (i.e. sampling all four seasons) to characterise sources of uncertainty and environmental variability.

Power analysis showed that additional net deployments offer diminishing returns for statistical power (Figure 10), with the current sampling effort (~350-400 net deployments per year from 2021 onwards) offering good power (using ≥ 0.8 as standard power threshold) to detect changes in CPUE when controlling for other covariates (Figure 10). This also supports removing years with less than 300 deployments from CPUE analysis as the power to detect change in these years is greatly reduced. However, there are other potential advantages to increased sampling effort, such as enhancing tagging and biological sampling datasets to support complementary objectives such as population abundance estimates through capture mark recapture (CMR) and close kin mark recapture (CKMR) models. As such, while increasing total effort is not strictly necessary to improve CPUE surveys (i.e. net deployments greater than 300 per year), it may be recommended (if logistically feasible) if there is a need to enhance alternative research objectives.

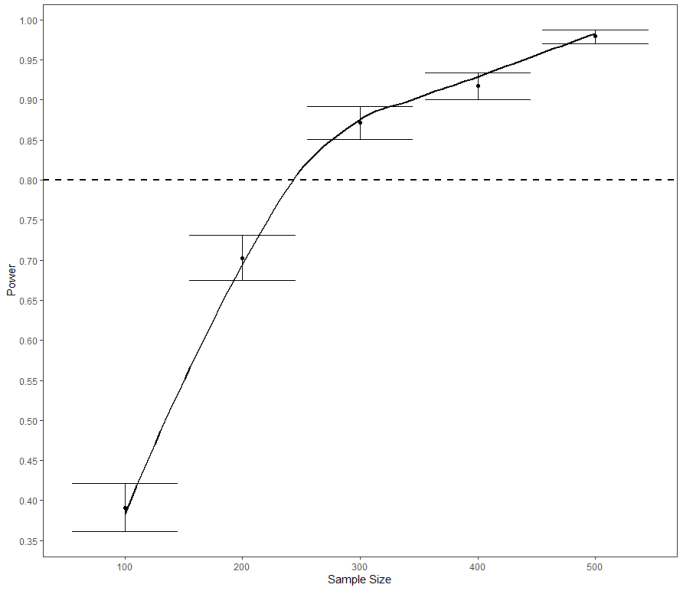

Figure 10. Power curve showing the influence of annual sample size (number of net deployments per year) on the statistical power to detect a significant effect in standardised CPUE by Year. Points represent mean power estimates at each sample size, with horizontal bars indicating the 95% binomial confidence intervals. The dashed horizontal line represents 80% statistical power.

By contrast, power analysis showed that distributing effort across a boarder range of observation opportunities can improve statistical power in a more effective way, even when total effort remains the same (Figure 11), i.e. eight trips per year. Additionally, this approach would help better characterize within year variability and seasonal effects, allowing these variables to be incorporated into the standardised CPUE model.

Alternative survey methods being developed such as video sonar (ARIS) should similarly incorporate these recommendations into a-priori power analysis and sampling design.

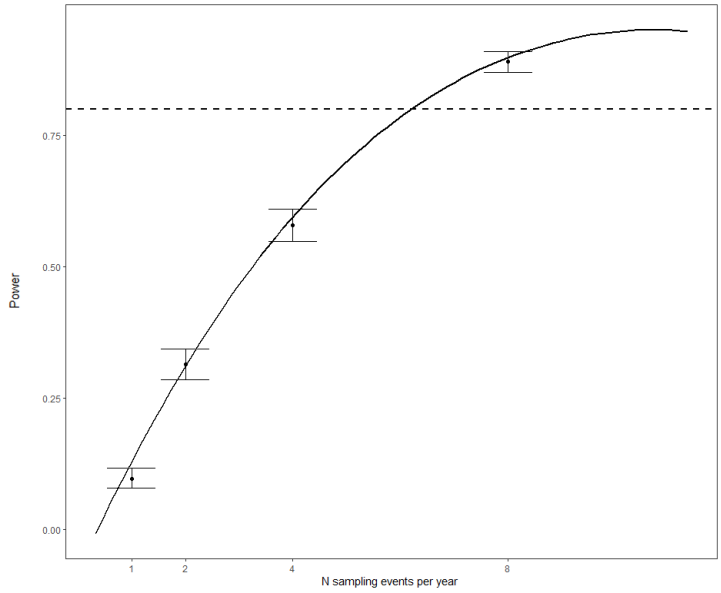

Figure 11. Power curve showing the effect of increasing the number of sampling events per year on the ability to detect interannual changes in CPUE. Power was estimated using simulations (N = 1000) based on a simplified Poisson GLM fitted to seasonal mean CPUE values, with the number of observations per year representing the number of discrete sampling events. The simulated effect size corresponds to an approximate 10% change in CPUE per year. Points represent mean power estimates with 95% confidence intervals. The dashed horizontal line represents 80% statistical power.

Conclusion

The 2025 Maugean skate monitoring program provides evidence of some individuals from the 2020–2021 cohort reaching adulthood, confirming that at least a portion of recent juveniles survived to maturity. Moreover, recent recruitment dynamics are consistent with a pulse-recruitment model, characterised by interannual variability in cohort contributions. While encouraging, recruitment under this pulse-driven model remains inherently vulnerable to acute environmental events, and due to the life history characteristics of the species (time to maturity and short reproductive lifespan), any single recruitment event may not be sufficient to allow sustained population growth unless its followed by further pulses. Multiple consecutive years of successful recruitment will likely be required before a sustained recovery trajectory can be confidently inferred.

Standardisation using negative binomial generalised linear models identified significant spatial and temporal effects on CPUE, with temporal effects differing significantly among sites. These findings underscore that harbour-wide CPUE should not be used as a simple proxy for abundance moving forward and highlight the importance of site-specific standardised CPUE estimates. Continued refinement of these models to incorporate additional sources of variability, including interannual environmental fluctuations, will be essential moving forward. To complement these efforts, an acoustic tracking program funded by NRE is currently underway in Macquarie Harbour to monitor Maugean skate movement and habitat use. This tracking data will help examine behavioural changes influencing catchability, assess whether individuals are redistributing among sites, and identify seasonal habitat shifts. Integrating movement information will provide critical context for interpreting standardised CPUE trends and strengthen population assessments. Likewise, as improved monitoring data becomes available, enabling more robust CPUE modelling, demographic analysis should be incorporated into future abundance modelling (Hostetler and Chandler, 2015), particularly in light of the potential changes to recruitment dynamics observed in this report.

Evidence of spatiotemporal variability indicates that sampling design should retain site-specific coverage and a strict seasonal framework (i.e. sampling all four seasons plus within season replication) to characterise sources of uncertainty and environmental effects. While additional net deployments offer diminishing returns for statistical power, distributing effort across a broader range of observation opportunities, such as targeting eight trips per year, may improve detection of temporal patterns. This approach would help better characterise within year variability and seasonal effects, allowing these variables to be incorporated into the standardised CPUE model. This work is consistent with the first listed priority, '*Continued population monitoring of the Maugean skate to support evaluating effectiveness of conservation actions and to underpin conservation planning decisions*', under the 'Urgent action' category of the 'Survey and monitoring priorities' in the *Zearaja maugeana* (Maugean Skate) Conservation Advice (DCCEEW, 2024), as well as Objective 5, '*Maintain and strengthen the knowledge base for Maugean skate recovery*' of the Conservation Action Plan for the Maugean skate (NRE Tas, 2024).

A novel alternative survey method, high-resolution imaging sonar (ARIS), is being developed to potentially complement and improve Maugean skate monitoring. Acoustic based imaging methods like ARIS are being investigated over traditional optical approaches (e.g. towed video) due to their ability to function in low visibility / high turbidity conditions, characteristic of the preferred Maugean skate habitats in Macquarie Harbour. To date, preliminary work has been successful in developing a method for towed deployment of ARIS, as well as validating the ability to identify and measure skate in a captive setting (Moreno et al. in prep). An initial in-situ pilot survey in Macquarie Harbour will be undertaken in the first quarter of 2026. If successful, ARIS would provide several advantages over the current netting-based approach for relative abundance estimation. Namely, it will be able to detect a broader range of sizes, including juveniles (nets have a size selectivity bias), it will be more robust to site specific catch efficiency bias (net efficiency is likely to vary depending on the physical characteristics of a site), and deployments will be more resource efficient, allowing for more effective and higher sampling effort surveys. It should be noted however, that even if ARIS is incorporated as a monitoring tool, some netting will still be necessary to collect biological samples and potentially maintain the only long-term data set currently available. Any ARIS based survey planning should similarly incorporate the design and power considerations presented here to enhance data quality and monitoring efficacy. This work aligns with the first listed priority, "*Continued population monitoring to include the development of a robust, non-harmful, and logistically feasible sampling method to effectively assess the Maugean skate population size, structure, status and trend…including through techniques such as video sonar (ARIS)*', under the 'Medium-term actions' category of the 'Survey and monitoring priorities' in the Zearaja maugeana (Maugean Skate) Conservation Advice (DCCEEW, 2024), as well as Objective 5, '*Maintain and strengthen the knowledge base for Maugean skate recovery*' of the Conservation Action Plan for the Maugean skate (NRE Tas, 2024).

Despite recent recaptures in 2025, current data remains insufficient to support a meaningful update to the population estimate. Closed-population models are limited by assumptions of demographic and geographic closure, while more complex open or state-space models have not converged due to low sample sizes and recapture frequencies. As further Maugean skate recaptures are obtained, continued model development and evaluation of more complex tag-recapture frameworks will be undertaken. A promising approach is the use of genetic mark-recapture, where individual DNA signatures confirm recaptures. Analyses have confirmed genetic recaptures of individuals previously tagged with PIT and dart tags, and a close-kin mark-recapture model (Bravington et al., 2016) is being applied by CSIRO to genetic samples collected by IMAS population monitoring between 2012 and 2024 to determine if this methodology can provide an estimate of absolute abundance.

Collectively, these findings highlight the continued need for carefully designed long-term population monitoring. Sustained recruitment, improved understanding of movement and catchability, and the integration of complementary survey and analytical approaches will be critical to increasing confidence in population assessments and guide effective conservation actions moving forward.

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Supplementary Material

Appendix 1:

Appendix 1. Summary of all gillnet survey effort by year. Deployments are the total number of net deployments in the year, Total caught is the number of skate caught in each year inclusive of recaptures, LF data is the number of skate used in size frequency analysis (showing how many skate were removed due to missing or incomplete data) and Areas is the number of areas sampled (3 key skate habitat areas are used for monitoring and any additional sampling areas that occur away from these 3 key areas are indicated by the + symbol). The temporal distribution of effort (months and seasons) is also shown, as well as the number of adults (≥600mm) and juveniles (< 600mm), the number of skate tagged each year, and the number of recaptures, with the numbers in parentheses showing the days at liberty – in 2014 this is shown as the range of days at liberty.

Year	Net deployments	Total caught	LF data	Areas	Months	Seasons	Total caught ≥600mm	Total caught <600mm	Skate tagged	Recaptures
2011	65	43	0	3	1	1	25	18	0	
2012	202	113	108	3 +1	3	3	84	28	84[1]	
2013	167	68	66	3 + 1	3	2	46	11	62[1]	
2014	342	163	161	3 + 2	4	4	130	26	151[1]	9[1](2-733), 1(84*)
2015	76	10	10	3	1	1	7	3	9[1]	
2017	77	27	27	1	1	1	23	4	0	
2018	210	76	73	1	2	2	70	3	0	
2019	212	17	0	3 +1	1	1	0	0	0	
2021	395	45	44	3 + 1	5	4	43	1	44[2]	1 (64)
2022	455	25	24	3 + 1	5	3	17	6	22[2]	
2023	303	57	50	3 + 1	4	3	40	10	50[2]	1 (533*)
2024	316	102	99	3 + 1	3	2	92	8	99[3]	4 (329, 469, 25*, 47*)
2025	424	105	104	3 + 1	5	3	96	8	102[3]	4 (744, 391, 377, 439)

[1] PIT tag
[2] Single Dart Tag
[3] Double Dart Tag
*Denotes genetic recapture