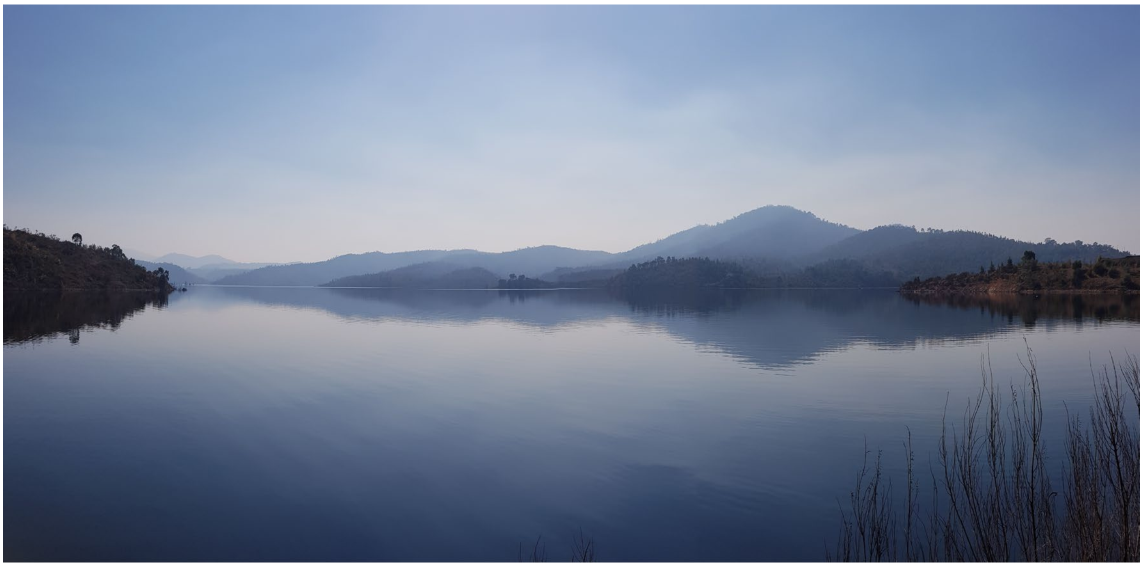
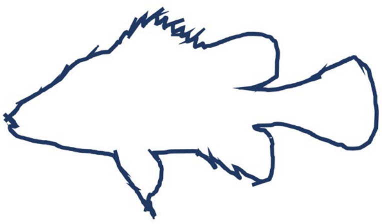
Enlarged Cotter Reservoir Ecological Monitoring Program

Technical Report 2025



Report to Icon Water

July 2025



Enlarged Cotter Reservoir Ecological Monitoring Program: Technical Report 2025

Report prepared for: Icon Water

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EXECUTIVE SUMMARY

Ongoing drought and threats to water security in the ACT led to the recommissioning and augmentation of Cotter Reservoir, expanding its capacity from approximately 4 GL to 76.2 GL—now known as the Enlarged Cotter Reservoir (ECR). The ECR and the Cotter River upstream to Bendora Dam are home to four threatened aquatic species. Of these, Macquarie perch (*Macquaria australasica*) and Two-spined blackfish (*Gadopsis bispinosa*) are most likely to be directly impacted by the ECR and are therefore the focus of associated research and mitigation efforts.

To address potential impacts from the construction, filling, and operation of the ECR, a comprehensive fish monitoring program was initiated. This program spans three management phases:

- Baseline (2010–2013)
- Filling (2014–2015)
- Operational (2016–present)

During the 2024–2025 monitoring year, flow in the Cotter River below Bendora Dam was dominated by regulated releases, with a notable peak in late November 2024. The ECR reached full supply level in August 2020 and has remained functionally full (within 1.0 m of FSL) since.

Macquarie Perch

Key changes in the Macquarie perch population across monitoring phases include shifts in adult abundance, body length, and recruitment rates. Since peak abundance in 2015 and early operational years (2016–2017), adult numbers have remained relatively stable with minor annual fluctuations. However, 2025 recorded the lowest adult abundance since monitoring began. High water temperatures likely reduced gill net capture rates, as fish occupied depths beyond net reach. May 2025 monitoring showed a rebound in capture rates, aligning with previous operational years.

Adult body length has increased during the operational phase, suggesting fewer smaller adults. Encouragingly, recruitment to the young-of-year stage was detected for the seventh consecutive year in 2025, albeit at low levels—possibly due to the November 2024 flow pulse disrupting early development stages.

Two-Spined Blackfish

This species remains rare in the ECR, with only a few individuals detected in newly inundated areas since filling began. While persistence is likely, there is no evidence of a recruiting population. Continued monitoring is essential to assess long-term viability. Low abundances in Bendora Reservoir also warrant further investigation.

Trout

Adult Rainbow trout captures were low in early 2025, likely due to elevated water temperatures. May sampling showed increased captures, supporting this hypothesis. Rainbow trout abundance and size have increased since the operational phase began, though the size increase (13.8 mm FL) is not

currently a significant threat to Macquarie perch. Brown trout abundance remained very low in 2025 (two captured), contrasting with high levels recorded between 2016–2021. Future monitoring will clarify whether this represents a sustained decline. No evidence of predation on Macquarie perch or Two-spined blackfish was found in trout stomachs in 2025.

Alien Species

Goldfish remain the most commonly detected small-bodied alien species. Their abundance surged during the filling phase but has remained low since 2017. While they pose minimal direct threat to native species, their decline could shift predator pressure (e.g., from cormorants and trout) onto Macquarie perch.

Piscivorous Birds

Monitoring from 2010 to 2025 identified Little Black, Little Pied, and Great Cormorants as the most abundant species. Seasonal fluctuations were observed, with Little Pied Cormorants showing increased summer presence, especially in sections 2 and 4. Community composition shifted significantly between baseline and operational phases, driven initially by Little Black Cormorants and later by broader changes across all three dominant species.

Macrophytes and Food Resources

Stable reservoir levels over the past four years have supported the establishment and expansion of four emergent macrophyte stands. Although small, these stands signal positive habitat development.

Macquarie perch food resources—primarily decapods and microcrustaceans—varied significantly by phase and season. Coarse pick samples revealed shifts in community composition, particularly between baseline and operational phases. Chironomidae dominated during the operational phase, while Cladocera and Copepoda were more prevalent during the baseline. Plankton tow samples showed consistent microcrustacean dominance across all phases, indicating stability in the pelagic food web.

RECOMMENDATIONS

Gill netting for adult Macquarie perch and trout in Cotter Reservoir to commence once water temperatures have dropped below 20°C.

A formal population estimate of adult Macquarie perch in Cotter Reservoir should be undertaken every 3 years to allow for timely adjustments to management practice. With the last population estimate provided in 2023, we recommend additional effort be employed in 2026 to undertake a commensurate population estimate of adult Macquarie perch in Cotter Reservoir. Namely, in addition to the standard survey effort, a recapture effort of;

- Two nights of gill netting (12 gill nets per night);
- 3 nights of fyke netting (20 fyke nets per night);
- Two days and two nights of boat electrofishing.

The failure to catch Two-spined blackfish in the Bendora Reservoir reference site is the first time in 30 years of sampling this reservoir with fyke nets in 2021 and then again in 2024, and the continuing low abundances in recent years warrants further investigation. If the site continues to return very low abundances of blackfish another reference site will need to be sampled (Corin Reservoir).

No other change to the current monitoring program or management actions are recommended at this stage. Continued close scrutiny of adult Macquarie perch size and abundance and the annual occurrence of recruitment to YOY is recommended alongside monitoring of pest fish species such as trout and Goldfish.

BACKGROUND

Ongoing drought and its threat to water security in the ACT resulted in the recommissioning and augmentation of Cotter Reservoir from ~4 GL to 76.2 GL capacity. The enlarged Cotter Reservoir (ECR) and Cotter River upstream to Bendora dam contain four threatened fish and crayfish species: Macquarie perch *Macquaria australasica*, Trout cod *Maccullochella macquariensis*, Two-spined blackfish *Gadopsis bispinosa* and Murray River crayfish *Euastacus armatus*. Trout cod are not present in the ECR, with Murray River crayfish only confirmed from a handful of occasions. Both species are rarely encountered in the Cotter River below Bendora dam. Consequently, the major focus for threatened fish research and mitigation projects associated with the ECR has been Macquarie perch and Two-spined blackfish. Potential impacts of the construction of the ECR have been well described and reviewed (Lintermans, 2012, Lintermans, 2005, ACTEW Corporation, 2009a, ACTEW Corporation, 2009b) and in response to these impacts, a range of projects including this monitoring program commenced (ACTEW Corporation, 2009b). This monitoring program encompasses three distinct management phases; baseline (2010-2013 (Lintermans et al., 2013)), filling of the ECR (2014 – 2015) and operational (2016 – current). The underlying sampling design and priority knowledge gaps for the filling and operational phases of the monitoring program were revised in 2013 and modified and now address ten management questions:

1. Has there been a significant change in the abundance and body condition of Macquarie perch in the enlarged Cotter Reservoir (Young-of-Year, juveniles and adults) as a result of filling and operation?
2. Has there been a significant change in the abundance, body condition and distribution of the Macquarie perch in the Cotter River above and below Vanitys Crossing as a result of the filling and operation of the ECR?
3. Have Two-spined blackfish established a reproducing population in the enlarged Cotter Reservoir and are they persisting in the newly inundated section of the Cotter River?
4. Has there been a significant change in the abundance, distribution and size composition of adult trout in the enlarged Cotter Reservoir as a result of filling and operation?
5. Has there been a significant change in the abundance and size composition of trout in the Cotter River upstream of the enlarged Cotter Reservoir as a result of the filling and operation of ECR?
6. Are Two-spined blackfish and Macquarie perch present in trout stomachs in the Cotter River?
7. Has there been a significant change in the abundance and distribution of non-native fish species) in the enlarged Cotter Reservoir as a result of filling and operation?
8. Has there been a significant change in the abundance, distribution and species composition of piscivorous birds in the vicinity of the enlarged Cotter Reservoir as a result of filling and operation?
9. Have macrophyte beds re-established in the enlarged Cotter Reservoir?
10. Are there adequate food resources (particularly decapods) for the Macquarie perch following the filling and operation of the enlarged Cotter Reservoir?

STUDY AREA

This monitoring program centres on the Cotter River catchment, to the west of Canberra, in the Australian capital territory (Figure 1). The integrated monitoring program has field activities with techniques employed at each site often addressing multiple questions (Table 1). Details of sites used are detailed for each question in the sections below.

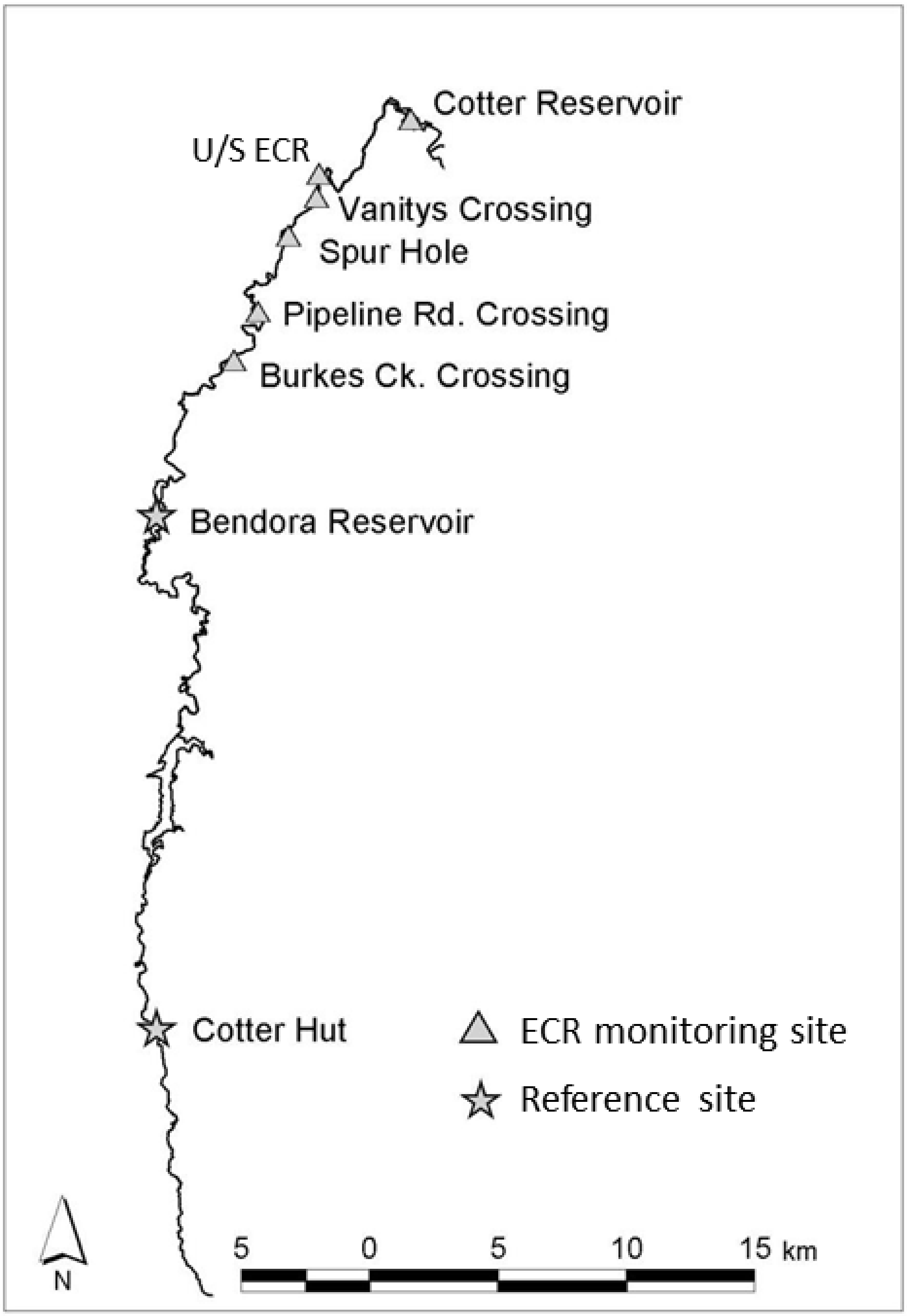

Figure 1. Location of sampling sites on the Cotter River. Note: Map does not include the reference site on the Murrumbidgee River (Kissops Flat).

Table 1. Monitoring questions to be addressed at each monitoring site (see Figure 1 for location of monitoring sites).

Site	Question addressed
Cotter Reservoir	1, 3, 4, 7–10
Bracks Hole*	2, 3, 5, 6
Downstream of Vanitys Crossing	2, 5, 6
Vanitys Crossing	2, 5, 6
Spur Hole	2, 5, 6
Pipeline Rd. Crossing	2, 5, 6
Burkes Ck. Crossing	2, 5, 6
Bendora Reservoir**	3, 4
Kissops Flat***	1, 2
Cotter Hut	5, 6

*Bracks Hole has been inundated. This site has been replaced by the Downstream of Vanitys Crossing site for questions based on riverine habitats (Questions 2, 5 and 6).
** Reference site for Questions 3 and 4.
***Reference site on the Murrumbidgee River for Questions 1 and 2.

HYDROLOGICAL SUMMARY

Following several very wet years (see Figure 2a), the last two monitoring years saw a return to river flows which were largely dominated by regulated flow releases (because of dry climatic conditions), except for a single short flow pulse associated with rainfall event, most notably the ~55 mm that fell 30 November / 1st December 2024 and resulted in a sharp rise in the Cotter River at Vanitys Crossing from mean hourly discharge of 33 to 715 ML Day-1 (Figure 2b and Figure 3). This rise was even more evident at the gauge downstream of Cotter Reservoir (Cotter River at Kiosk 410700) where the Cotter River rose from 36 ML Day-1 on the 30th November to ~1600 ML Day-1 on the 2nd December 2024 (largely from inflows from Condor Creek). Enlarged Cotter Reservoir was at or very close to full supply level for the entire monitoring year of 2024 – 2025 (Figure 4).

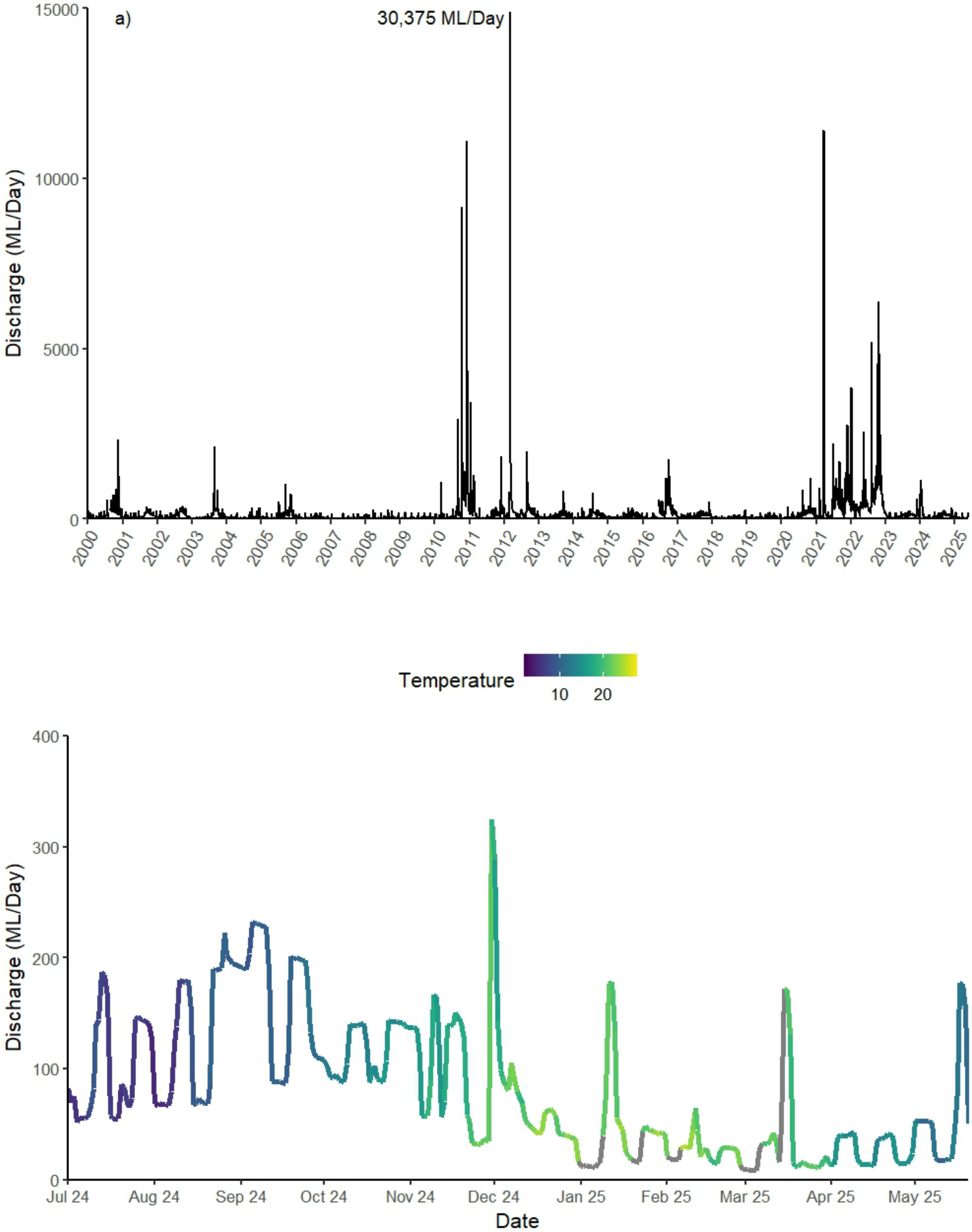

Figure 2. Mean daily discharge of the Cotter River at Vanitys Crossing from a) January 2000 until May 2025 and b) July 2024 – May 2025.

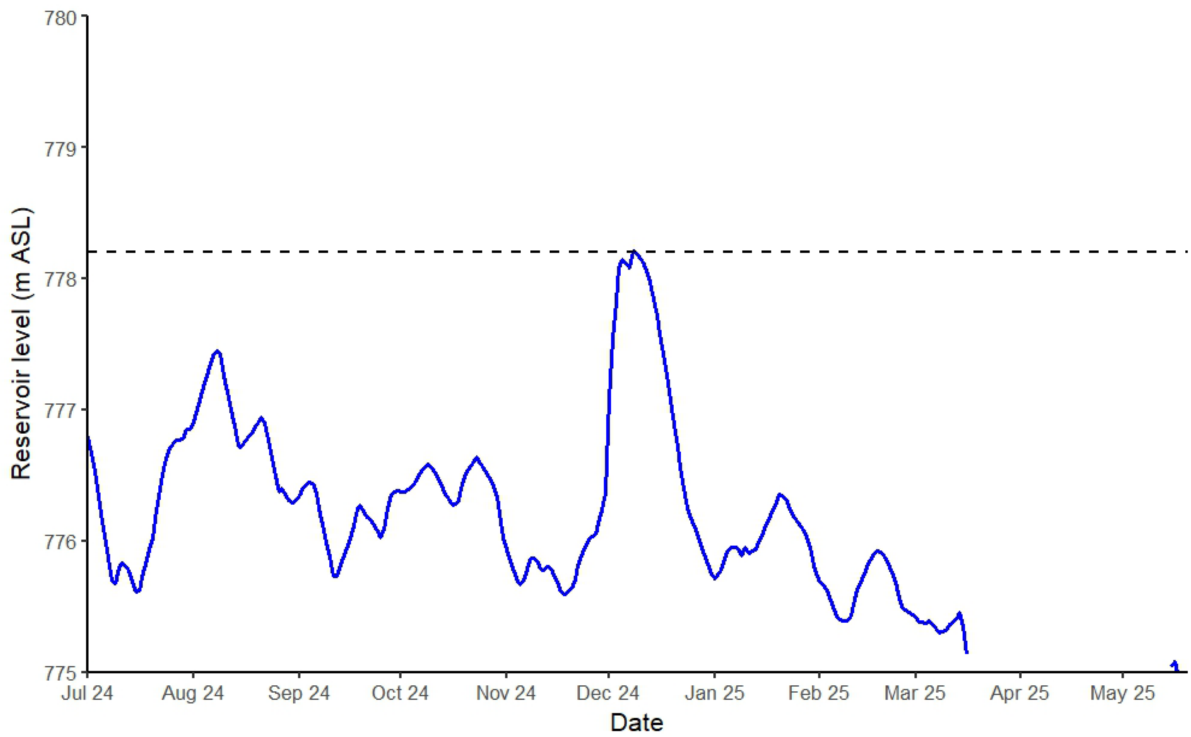

Figure 3. Water level (in metres above sea level) of Bendora Reservoir (blue line) from July 2024 – May 2025. Dashed line indicates Bendora Reservoir full supply level.

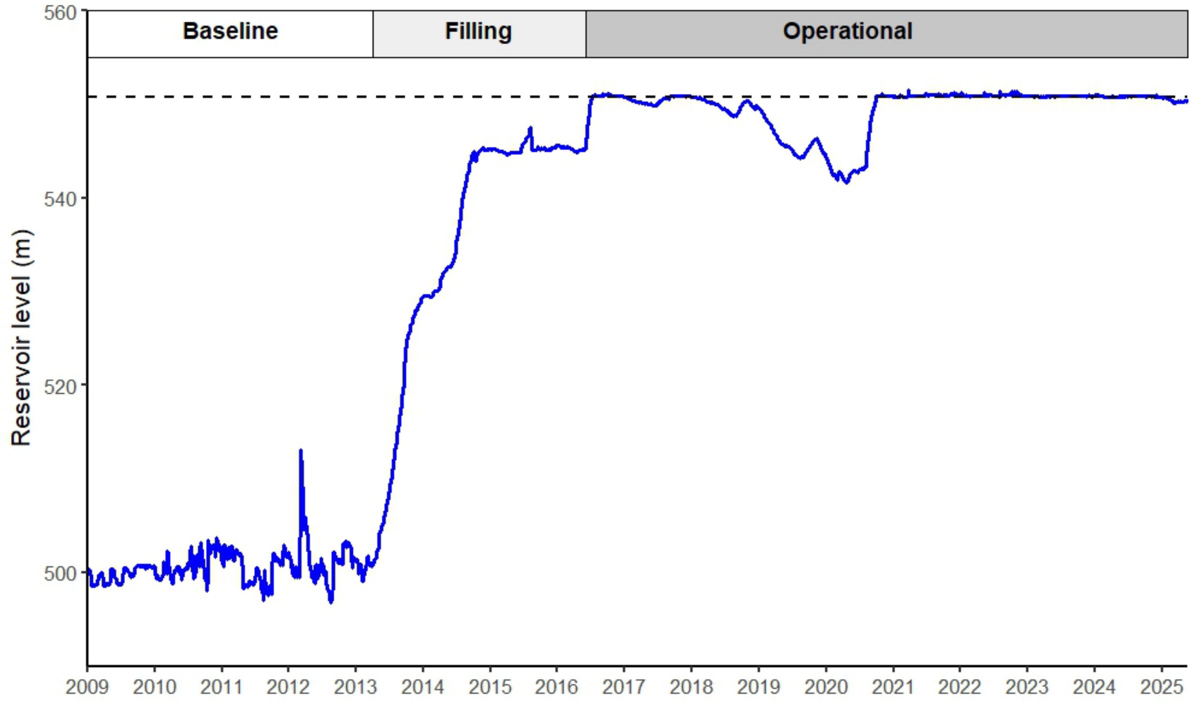

Figure 4. Water level (in metres above sea level) of Cotter Reservoir from January 2009 until May 2025. Rectangles indicate the three monitoring phases: white = Baseline, grey = Filling, hatched = Operational. Blue line indicates Cotter Reservoir level. Dashed line indicates enlarged Cotter Reservoir full supply level. Full supply level prior to enlargement was 500.8 m ASL.

MONITORING COMPONENTS

QUESTION 1: Has there been a significant change in the abundance and body condition of Macquarie perch in the enlarged Cotter Reservoir (young-of-year, juveniles and adults) as a result of filling and operation?

BACKGROUND

A range of potential threats such as loss of habitat, interactions with alien fish species, and predation by cormorants can impact the Macquarie perch population in the Cotter River and reservoir as a result of the filling and operation of the new ECR. In considering these potential ECR impacts, we must account for natural fluctuations in Macquarie perch abundance that can arise from interannual variations in climate, flow regime, stochastic extreme events (flood, drought, etc.) and other factors that influence rates of spawning, recruitment and mortality. Body condition of adult Macquarie perch is a key indicator of reproductive potential, and so monitoring changes in adult body condition can be a useful indicator of future recruitment events and overall population trajectories (Gray et al., 2000). Spawning of reservoir-resident Macquarie perch may be impacted by newly encountered riverine barriers in a filling reservoir or as reservoir levels fluctuate (Broadhurst et al., 2016a), so early detection of spawning success (via snorkelling for larvae) and how this relates to young-of-year (YOY) captured in the reservoir via netting will all contribute to the understanding of recruitment success or failure for a given year.

METHODS

The sampling design for Question 1 largely follows that of the baseline monitoring program Question 1 (Table 2; (Lintermans et al., 2013). An additional metric – the wet weight of individuals captured in gill nets - was used to calculate fish condition. Boat electrofishing was added as a complimentary sampling method to mitigate potential sampling inefficiencies via gill netting for adult Macquarie. To determine the likely contribution of young-of-year (YOY) recruitment to the reservoir population by reservoir adults, snorkelling of the river from immediately upstream of Cotter Reservoir (ECR) to Vanitys Crossing was undertaken.

Table 2. Outline of the sampling design for Question 1 of the ECR monitoring program.

Feature	Detail
Target species and life history phase	Macquarie perch *Macquaria australasica*. Adults (> 150 mm total length (TL)), Juveniles (85 - 150 mm TL) and young-of-year (< 85 mm TL). Larvae and early juveniles observed during snorkelling are likely to be 15 – 25 mm TL.
Sampling technique/s	Gill nets (10 (4 x 100 mm; 4 x 75 mm; 2x 125 mm stretch mesh)) per night for 5 nights, with an additional 2 x 125 mm gill nets per night to capture larger individuals) and fyke nets (12 mm stretch mesh, 20 per night for 3 nights in Cotter Reservoir; 12 per night for 2 nights at Kissops Flat). Boat-electrofishing 12 shots per shoreline per section for daytime and 6 shots per shoreline section for night-time sampling. Snorkelling (visual survey) of stream pools for larvae / early juveniles.
Timing	Netting and electrofishing conducted annually in February - April; snorkelling in Nov/Dec.
Number / location of sites	One impacted site: Enlarged Cotter Reservoir; one reference site: Kissops Flat (upper Murrumbidgee River). Fyke netting only undertaken at Kissops Flat.
Information to be collected	Number and total length for all Macquarie perch. Wet weight (g) for subadults/adults captured in gill nets. Number of larvae/early young-of-year per pool.
Data analysis	Catch-per-unit-effort (CPUE) of both gill and fyke netted Macquarie perch assessed between monitoring phases and years using generalised linear mixed models (GLMM). Length and body condition of individuals captured gill netting will be assessed between phases and years using linear mixed model (LMM) and GLMM, respectively.

Non-larval sampling targeted adult, juvenile and young-of-year Macquarie perch. Individuals were classed as adults if they were > 150 mm total length (TL), based on results from Ebner and Lintermans (2007) who found that males are sexually mature from this size. At the time of netting (i.e. autumn), Young-of-year are approximately 60 – 85 mm TL based on results of this monitoring program. Individuals were considered juvenile if they fell between 85 – 150 mm TL. Snorkelling surveys target larval and early young-of-year Macquarie perch that are ~ 2 – 4 weeks of age (~15 – 25 mm TL).

Sampling was conducted at two sites: ECR (impacted site) and Kissops Flat (reference sites for young-of-year and juveniles). Only fyke netting was employed at Kissops Flat (see below for details). In the ECR two sampling techniques were employed to capture a representative sample of the entire size-range of the Macquarie perch population. Both gill nets (free-floating, multi-filament) and fyke nets (12 mm stretch-mesh single-winged) were deployed, as the former is most effective for capturing adult Macquarie perch and the latter is most effective at capturing young-of-year and juvenile Macquarie perch (Ebner and Lintermans, 2007, Lintermans et al., 2013, Lintermans, 2016).

Gill nets were deployed as per the baseline and previous filling and operational monitoring. Specifically, 12 gill nets were set independently around the perimeter of the reservoir in between late-Feb and early-April 2025. The reservoir was divided into five longitudinal sections, with two gill nets set in each section. Gill netting was undertaken over five nights (based on power analysis conducted Robinson, 2009), though was not conducted for more than two consecutive nights at a time to avoid stress on adult Macquarie perch by multiple sequential re-captures. In 2022, an additional two 125 mm gill nets (1 x deep-drop (66 meshes deep)) have been added to the previous effort of 10 gill nets (75, 100, 125 mm stretch mesh) to capture the increasingly larger adult Macquarie perch in Cotter Reservoir (following recommnedations in Broadhurst et al., 2018). Gill nets were set for six hours soak time commencing at ~15:30hrs following the existing threatened species netting protocol to minimise potential issues with prolonged retention of threatened fish in gill nets.

Low capture rates of adult Macquarie perch in gill nets in 2025 prompted further investigation into mechanisms driving the low captures. It was noted that water temperatures were warm during sampling, further confirmed by analysing water temperature data from Cotter Reservoir loggers, which indicated that water temperatures during the time of sampling in 2025 were among, if not the highest since monitoring began in 2010 (Figure 5). Radio-tracking of adult Macquarie perch in Cotter Reservoir in 2008 – 2009 indicated that mean depth use increased to > 7 m during warmer months (i.e. over summer when water temperatures were > 20 °C) (Thiem et al., 2012), well deeper than our gill nets can reach (maximum of ~4 m). To test whether adult Macquarie perch captures would increase once water temperatures decreased, two supplementary nights of gill netting (employing the fleet of 12 gill nets described above) were conducted in May 2025 (6^{th} and 15^{th}) when water temperatures had dropped to below 20 °C (actual mean water temperatures during supplementary sampling were 16.7 – 17.6 °C taken at 1m depth). Results of the supplementary sampling will not be included in formal analysis but used for interpretation of the routine monitoring results.

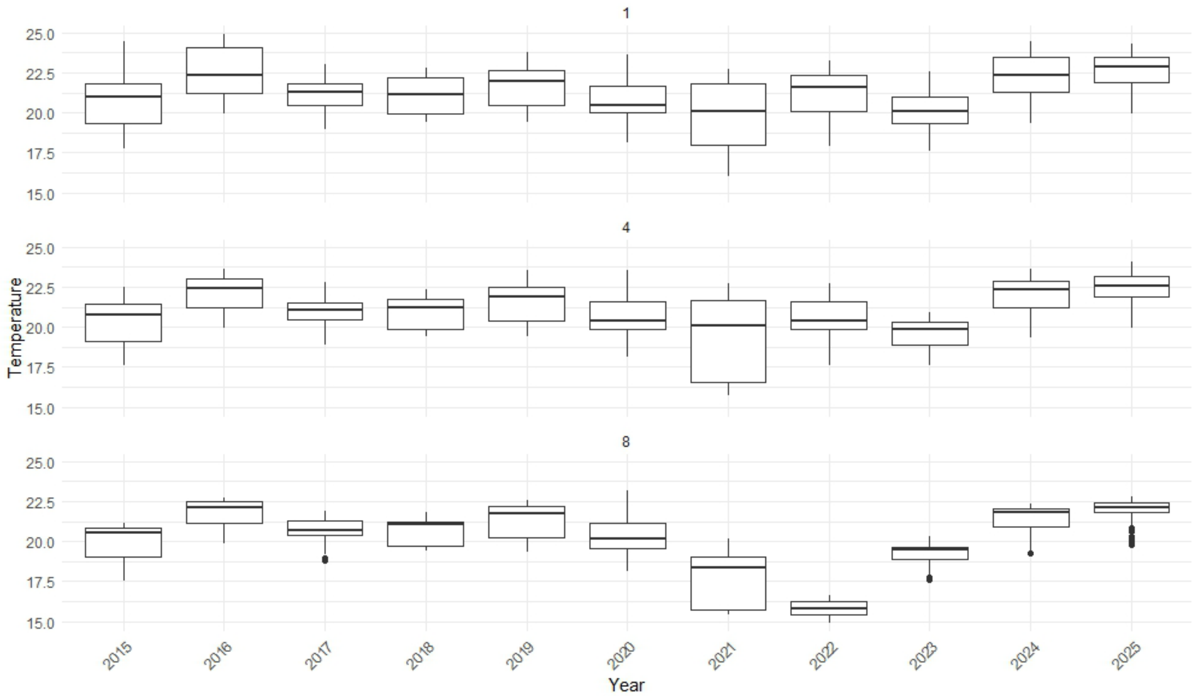

Figure 5. Box plots of daily water temperature in Cotter Reservoir (410704BC – Central Buoy) at 1, 4 and 8 m depths between late-February and mid-April, when gill netting is usually undertaken.

Twenty fyke nets were set singularly around the perimeter of the reservoir over three nights in between late-Feb and early-April 2025. Twelve fyke nets were set for two nights in the pool at Kissops Flat in February 2025. Fyke nets were set for ~16-hour soak time (existing fyke netting protocol) commencing at ~15:30-16:00 hrs.

Larval monitoring was not undertaken in December 2024 because of a high flow event in late November / early December increasing river flows and reducing water clarity.

Boat electrofishing occurred across multiple days with the reservoir divided into five longitudinal sections, with twelve 90-second "on time" electrofishing shots undertaken along each shoreline (left and right banks) of each section (10 replicates in total). Catches from boat electrofishing are compared with gill netting results from the same year to determine if catches of adult Macquarie perch follow the same patterns between techniques. Night-time electrofishing mimicked daytime, although at a reduced effort per shoreline section (six shots per sections instead of 12). As this was an increased survey aimed at capturing adult Macquarie perch, individuals less than 150 mm TL were observed only (not counted), although estimates of abundance were made for individuals < 85 mm TL (young-of-year) and Juveniles (85 > 150 mm TL) for each shot.

Abundance of Macquarie perch was standardised for effort applied during each sampling technique by calculating number of fish caught per hour (i.e., catch-per-unit-effort, or CPUE). Given the spatial ecology of Macquarie perch, CPUE of gill netted Macquarie perch was then scaled according to the shoreline length at the time of sampling, which varies with ECR water level. This was done by multiplying the CPUE for each net night by the proportional increase in shoreline according to the reservoir water level in each survey year (relative to the old Cotter Reservoir water level above sea level). In the case of fyke netting, where net effort was also increased from 2017 onwards, the increased shoreline was divided by the increased proportional net effort for these years. See below for scaled CPUE equation:

Scaled CPUE = CPUE / (Prefilling ECR shoreline / shoreline at time of sampling) / (baseline number of nets / current number of nets).

All analyses were performed using R V4.2 (R Development Core Team, 2022). To analyse Macquarie perch CPUE in gill nets (excluding the additional 125 mm gill nets used in 2022 from analyses), we employed a Generalized Linear Mixed Model (GLMM) with a negative binomial distribution to account for overdispersion in the count data (Zuur et al., 2009). The model included phase and year as fixed effects, and net-night as a random effect to account for repeated measures across sampling units. We also evaluated the need for a zero-inflated model, but zero-inflation was not found to be significant. The analysis was conducted in R using the glmmTMB package, which provides flexible modelling of count data with overdispersion and zero-inflation capabilities. Model diagnostics and overdispersion checks were performed using functions from the DHARMa package.

To assess changes in the catch per unit effort (CPUE) of Macquarie perch in fyke nets across different size classes and management phases, we fitted generalized linear mixed models (GLMMs) using a negative binomial distribution with a log link function. This approach was selected to account for overdispersion in the count data and the high frequency of zero catches. Separate models were fitted for the young-of-year (YOY) (<80 mm TL), juvenile (80 > 150 mm TL), and all size classes combined (sum of YOY, juvenile, and adult CPUE per net per night). Fixed effects included phase (baseline, filling, operational), site (Cotter Reservoir, Kissops Flat), and their interaction to evaluate site-specific temporal effects consistent with a Before-After-Control-Impact (BACI) design. A random intercept for year was included in each model to account for repeated sampling across years. Model fitting was conducted using the glmmTMB package in R, and model assumptions were evaluated using residual diagnostics. Estimated marginal means and pairwise contrasts were computed using the emmeans package to interpret differences in CPUE across phases within each site.

To assess the relative contribution of young-of-year (YOY) and juvenile Macquarie perch to total catch per unit effort (CPUE) across years, we conducted a correlation and multiple regression analysis using R. Annual mean CPUE values were calculated for each size class and for total CPUE (sum of YOY, juvenile, and adult CPUE). Pearson correlation coefficients were computed to evaluate the strength of association between total CPUE and each size class. A multiple linear regression model was then fitted using the lm() function, with total CPUE as the response variable and YOY and juvenile CPUE as predictors. Model coefficients and significance levels were used to interpret the relative influence of each size class on interannual variation in total CPUE.

To evaluate changes in adult Macquarie perch length over time, two complementary linear mixed-effects modelling approaches were used. First, a robust model was fitted using the nlme package, with Phase (baseline, filling, operational) as a fixed effect and Year as a random effect to account for repeated measures. A variance structure (varIdent) was specified to allow for heteroscedasticity across phases. Second, to enable direct comparisons between years, a separate model was fitted using the lmerTest package, treating Year as a fixed effect and Phase as a random effect. This model allowed for post-hoc Tukey-adjusted pairwise comparisons between years using the emmeans package.

Condition of adult Macquarie perch was analysed using Fulton's condition index, which is calculated as K = 100(weight/length^3) following (Ricker, 1975). To assess differences in the condition of Macquarie perch across monitoring phases (baseline, filling, and operational), a linear mixed-effects model (LMM) was initially fitted using Fulton's condition factor as the response variable. The model included Phase as a fixed effect and Year as a random intercept to account for repeated annual measurements from a single site. Model assumptions were evaluated through residual diagnostics and normality tests. Although residual plots suggested acceptable linearity and homoscedasticity, the Shapiro-Wilk test indicated a significant deviation from normality. To address this, a generalized linear mixed model (GLMM) was subsequently fitted using a Gamma distribution with a log link, which is more appropriate for continuous, positive, and potentially skewed data. The GLMM retained the same fixed and random effects structure and was evaluated using residual simulation diagnostics. ECR monitoring program body condition data (2014 onwards) was compared against historical data from 2009 (data from Lintermans et al., 2010).

RESULTS

Adult Macquarie Perch

A total of 11 Macquarie perch were captured using all gill nets in ECR in 2025, ranging from 230 – 415 mm TL (Figure 6). This was a lowest raw number of adult Macquarie perch captured since monitoring began (the next lowest years were 2012 with 18 individuals, and 2022 with 19 individuals). Adult Macquarie perch abundance was highest in 2015, more than double the next most abundant years (2014, 2016 and 2017, and roughly quadruple the abundances of all other years Figure 7). Macquarie perch CPUE was significantly different among years and phases (Table 3) and has shown a significant decline during the monitoring period, largely because of inflated CPUE during filling year of 2015 (Figure 7).

Table 3. Estimated effects of reservoir management phase and year on catch per unit effort (CPUE) of adult Macquarie perch captured in gill nets in Cotter Reservoir 2010 - 2025, based on generalized linear mixed models (GLMMs) fitted with a negative binomial distribution and log link. Estimates are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-1.52	0.42	-3.66	<0.001
Filling	2.45	0.29	8.33	<0.001
Operational	1.84	0.36	5.08	<0.001
Year	-0.07	0.03	-2.38	0.02

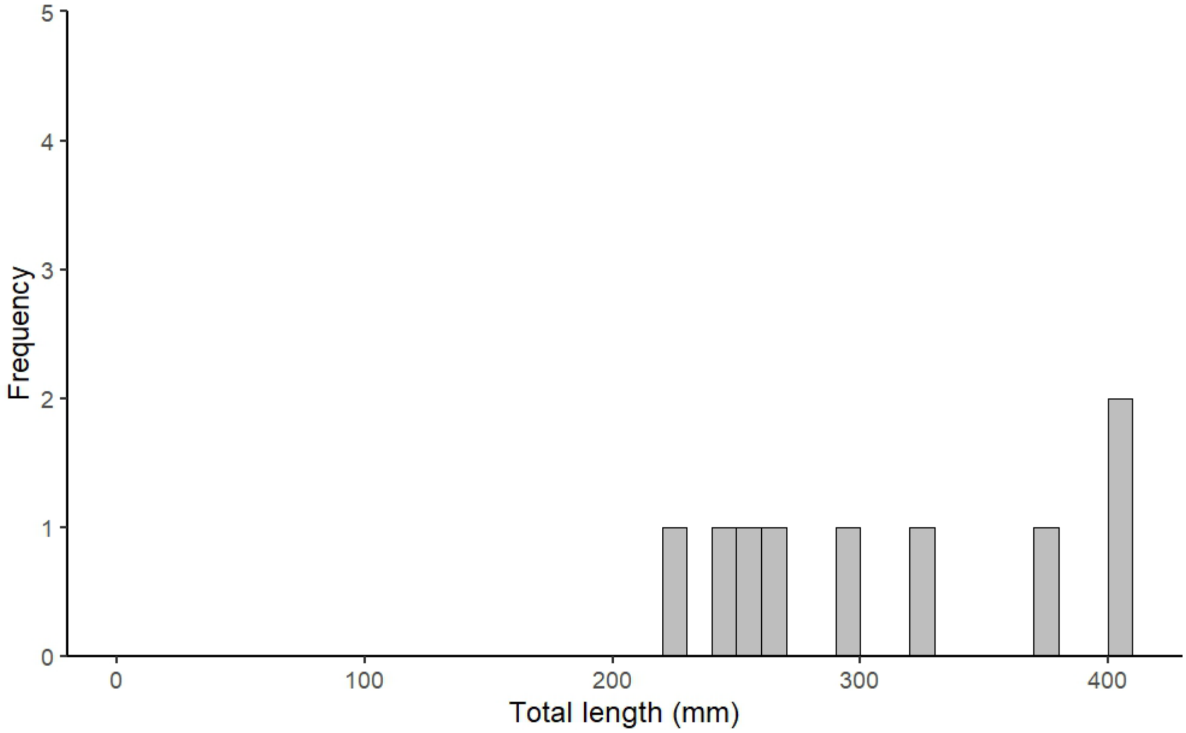

Figure 6. Length frequency of Macquarie perch captured from the Enlarged Cotter Reservoir in autumn 2025 using gill nets.

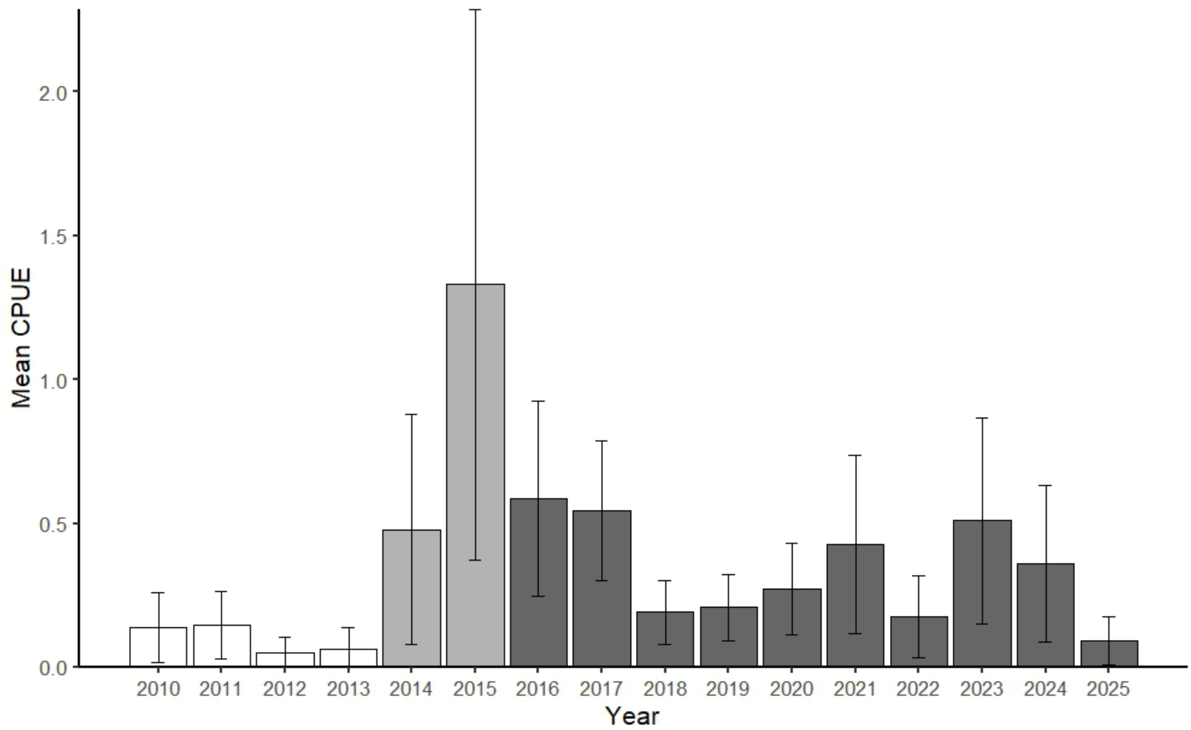

Figure 7. Relative abundance (displayed as mean CPUE ± 95% Confidence limits with Bonferroni correction and scaled to relative reservoir shoreline length at time of sampling) of adult Macquarie perch captured in Cotter Reservoir using gill nets between 2010 – 2025. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronological order from 2010 to 2025 from left to right on the x-axis.

Supplementary monitoring of adult Macquarie perch in May 2025 captured 12 adult Macquarie perch over two nights of gill netting. Lengths of Macquarie perch captured in the supplementary sampling ranged from 230 – 424 mm TL. Mean (+/- SE) CPUE (scaled for shoreline) of adult Macquarie perch captured in the supplementary monitoring was 0.3 +/- 0.11 fish per hour. This was more than three times the mean CPUE found for the standard monitoring in the months prior in 2025 (0.09 +/- 0.09) and was similar to mean CPUE of 2020 (0.27 +/- 0.16) and 2024 (0.36 +/- 0.27).

The robust mixed-effects model (year as random) revealed a significant increase in fish length during the operational phase compared to the baseline phase, while no significant difference was observed between the filling and baseline phases (Figure 8). The model also confirmed unequal residual variances across phases, with the filling phase exhibiting notably lower variability. In the fixed-effects model (year as fixed), several years showed significantly different length. Tukey-adjusted pairwise comparisons highlighted significant differences such as 2017 > 2021, 2019 > 2021, 2020 > 2023, and 2025 > 2023 (Figure 8).

There were no statistically significant differences in adult Macquarie perch condition between phases. However, the contrast between the baseline and filling phases approached significance (p = 0.075), suggesting a potential trend toward improved condition during the filling phase (Figure 9). Estimated marginal means on the log scale showed slightly higher condition values during the filling and operational phases compared to baseline, but these differences were not statistically robust.

Overall, the analyses suggest that while there may be subtle phase-related trends in condition, there is no strong evidence of significant changes across the monitoring phases.

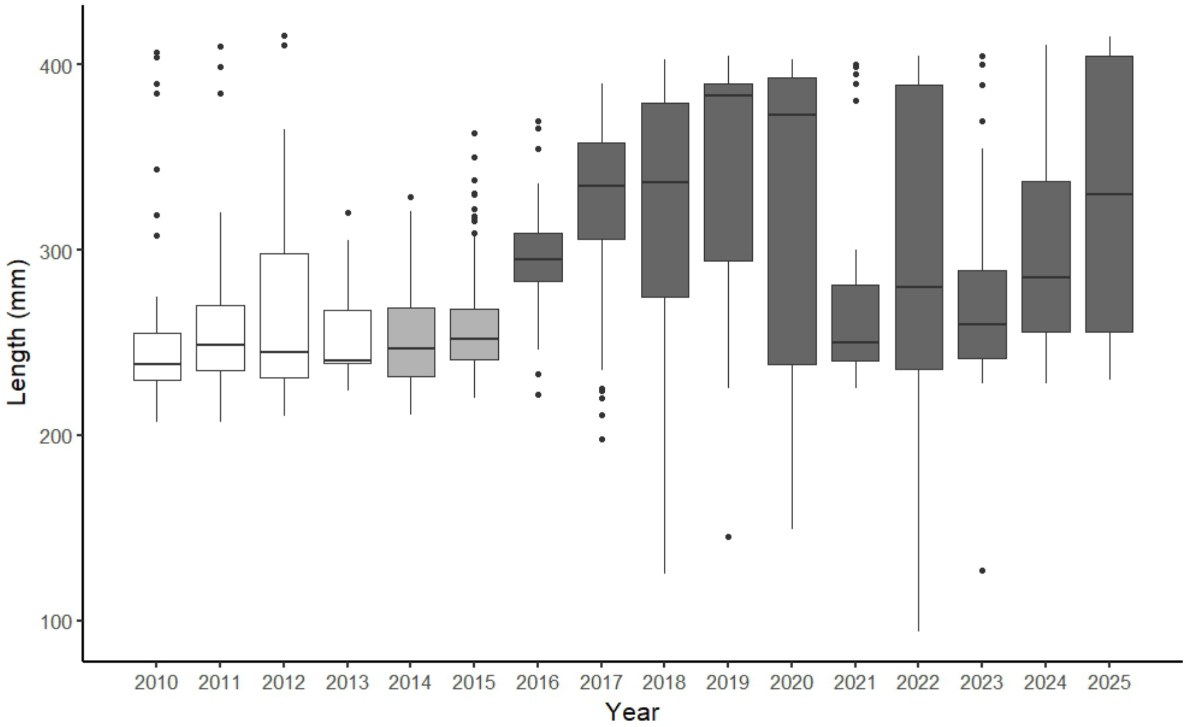

Figure 8. Box plot of length of adult Macquarie perch captured in gill nets in Cotter Reservoir from 2010 to 2025. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); plot arranged in chronological order from 2010 to 2025 from left to right on the x-axis.

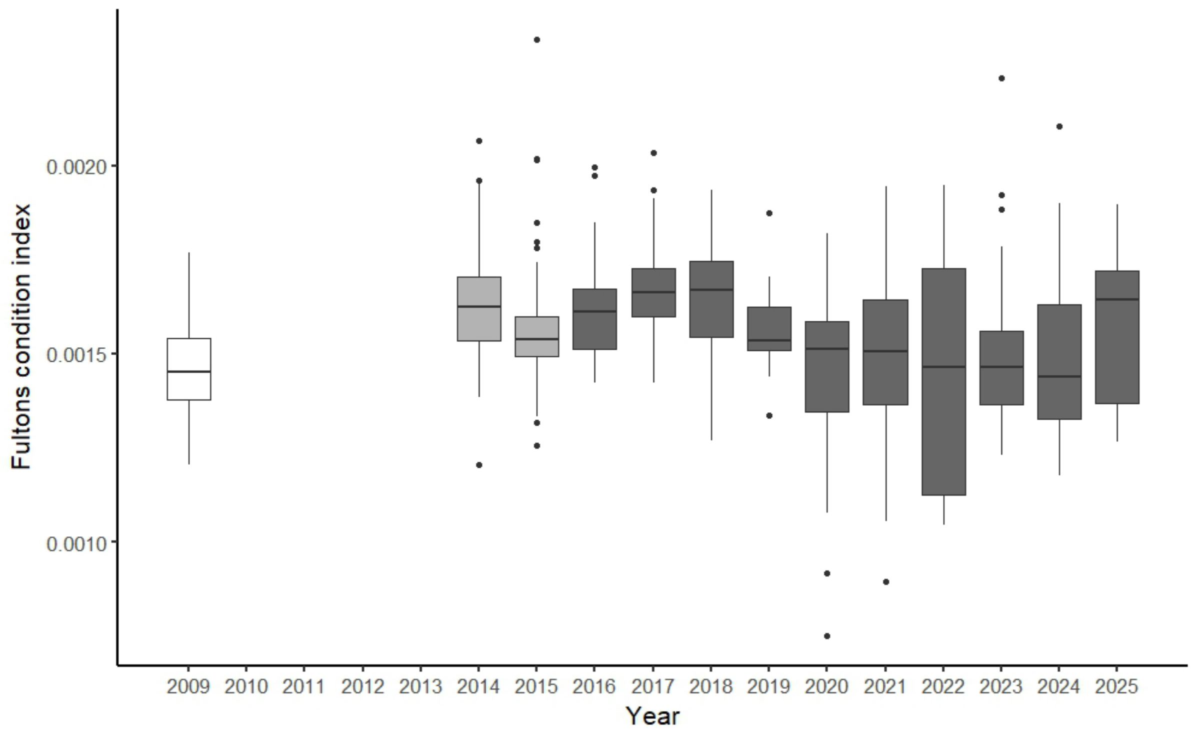

Figure 9. Box plot of condition (using Fulton's condition index) of adult Macquarie perch captured in gill nets in Cotter Reservoir from 2009 to 2025. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); plot arranged in chronological order from 2009 to 2025 from left to right on the x-axis.

Boat electrofishing

A total of 18 adult Macquarie perch were captured by boat electrofishing in 2025 ranging in lengths from 204 – 407 mm TL (Figure 12). Similar to gill netting captures, length frequency of individuals caught by electrofishing was dominated by individuals between 200 - 350 mm TL (Figure 12). Of these, 12 individuals were captured during the day and six individuals captured at night. Relative abundance of adult Macquarie perch captured by boat electrofishing in 2025 has rebounded slightly since the low capture rates in 2024 (Figure 13).

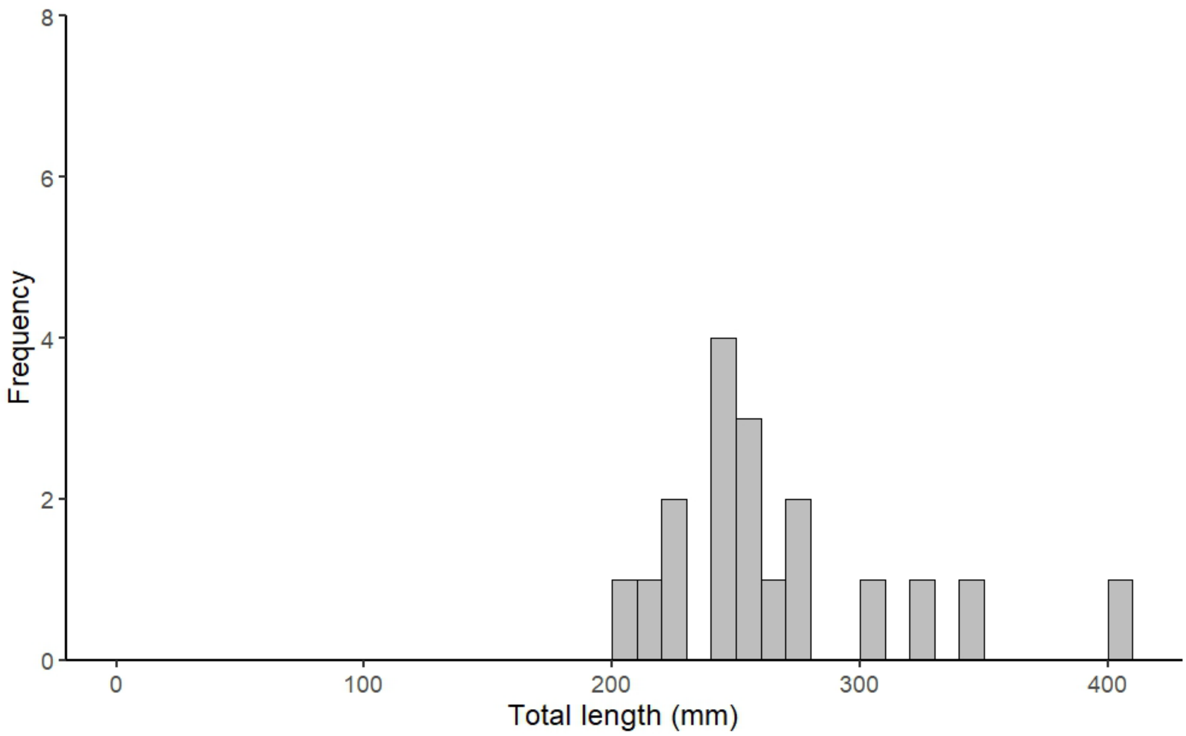

Figure 10. Length frequency of all Macquarie perch captured via boat electrofishing in Cotter Reservoir in 2025.

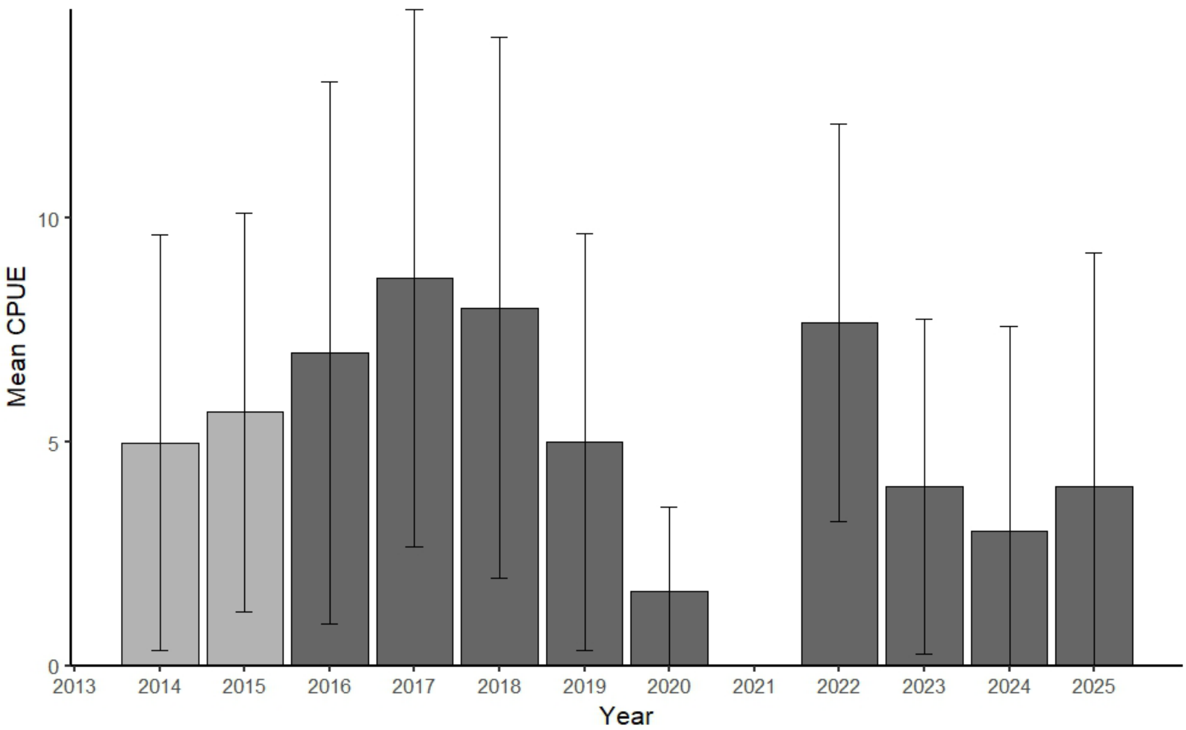

Figure 11. Relative abundance (displayed as mean CPUE ± 95% Confidence limits with Bonferroni correction) of adult Macquarie perch captured in Cotter Reservoir via boat electrofishing between 2014 – 2025. Light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronological order from 2014 to 2025 from left to right on the x-axis. Boat electrofishing not undertaken in 2021 due to high water turbidity.

Fyke netting - all age classes

A total of 158 Macquarie perch ranging from 50 – 385 mm TL were captured in 2025 from the ECR using fyke nets (Figure 12). When all size classes were combined, CPUE was significantly lower during both the filling and operational phases compared to baseline, and at the reference site compared to ECR (Table 4). A significant interaction between the filling phase and the reference site suggests that the decline in CPUE during the filling phase was again less pronounced at the reference site (Table 4). When CPUE was aggregated across all size classes, 2010 emerged as a strong baseline year, with the highest overall CPUE and variability (Figure 13). A sharp decline was observed during the filling phase, particularly in 2014 and 2015, which likely contributed to the significant phase effect detected in the combined model (Figure 13). While some recovery was evident in later years (e.g., 2021–2024), CPUE levels remained below baseline, suggesting a lasting impact of reservoir filling and operation on overall Macquarie perch abundance.

Table 4. Estimated effects of reservoir management phase and site on catch per unit effort (CPUE) of Macquarie perch (all size classes combined), captured using fyke nets in the ECR between 2010 and 2025. Estimates are derived from generalized linear mixed models (GLMMs) fitted with a negative binomial distribution and a log link function. Values are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-0.2672	0.1514	-1.765	0.078
Filling	-1.1482	0.309	-3.716	0.000
Operational	-0.8918	0.1776	-5.023	0.000
Reference site	-1.0947	0.2412	-4.538	0.000
Filling * Reference site	1.0131	0.4541	2.231	0.026
Operational * Reference	0.1105	0.314	0.352	0.725

Juvenile Macquarie perch

In 2025 a total of 77 Macquarie perch juveniles (85 > 150 mm TL) were captured in fyke nets. Juvenile Macquarie perch have been captured each year since monitoring began in 2010 but were particularly low in abundance during 2015 – 2017 as a result of successive years of recruitment failure over 2014 – 2016 (Figure 13). Analysis revealed a significant decline in juvenile CPUE at ECR during filling phase relative to the baseline (Table 5). The decline during the operational phase was not statistically significant (Table 5). At the reference site, CPUE was significantly lower than ECR during the baseline phase (Table 5). However, a significant positive interaction was observed during the filling phase, indicating that the reference site experienced a relative increase in juvenile CPUE compared to ECR during this period (Table 5). The interaction for the operational phase was not significant, suggesting no meaningful difference between sites during that phase (Table 5). These results suggest that the ECR experienced a significant decline in juvenile CPUE during the filling phase, while the reference site did not, supporting a site-specific impact at ECR during this period.

Table 5. Estimated effects of monitoring phase on catch per unit effort (CPUE) of juvenile Macquarie perch (80–150 mm total length), captured using fyke nets in Cotter Reservoir (ECR) and at the reference site (Kissops Flat). Estimates are derived from generalized linear mixed models (GLMMs) fitted with a negative binomial distribution and a log link function. Values are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-1.4195	0.3583	-3.961	<0.001
Filling	-2.2327	0.876	-2.549	0.0108
Operational	-0.6284	0.4146	-1.516	0.1296
Reference site	-2.8305	0.8599	-3.292	<0.001
Filling * Reference site	3.432	1.2602	2.723	0.0065
Operational * Reference	0.5995	1.0039	0.597	0.5504

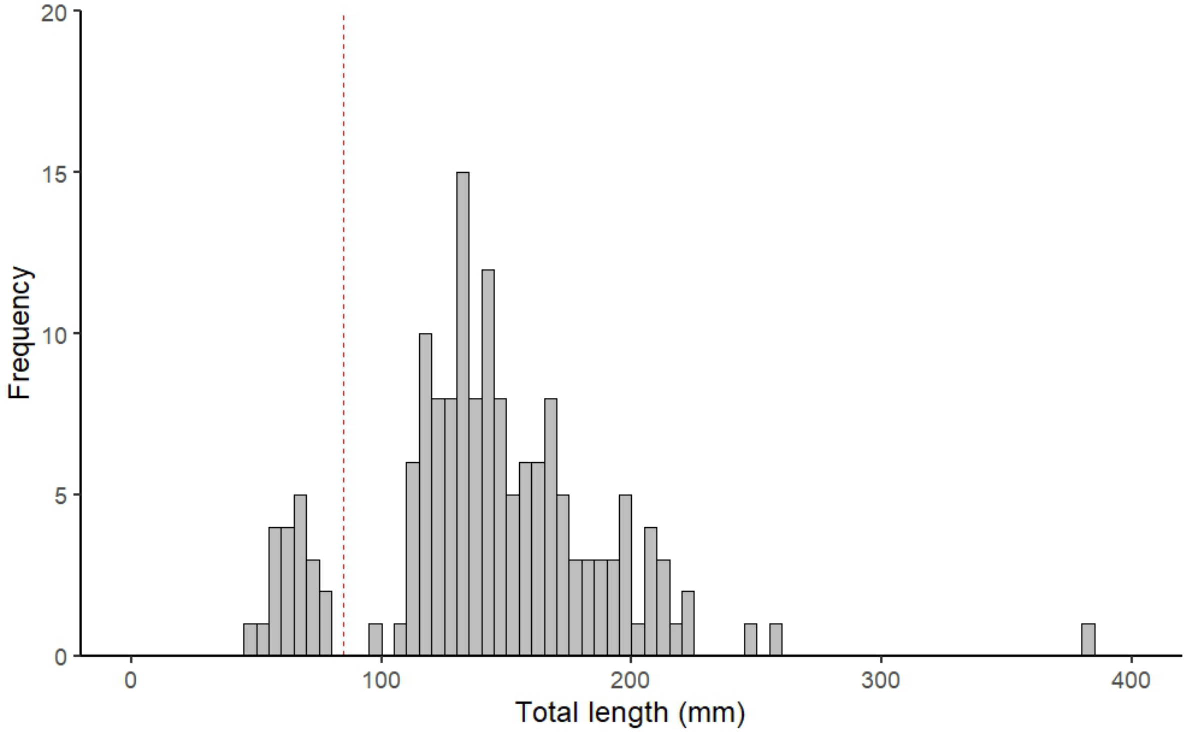

Figure 12. Length frequency of Macquarie perch captured from the ECR in autumn 2025 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

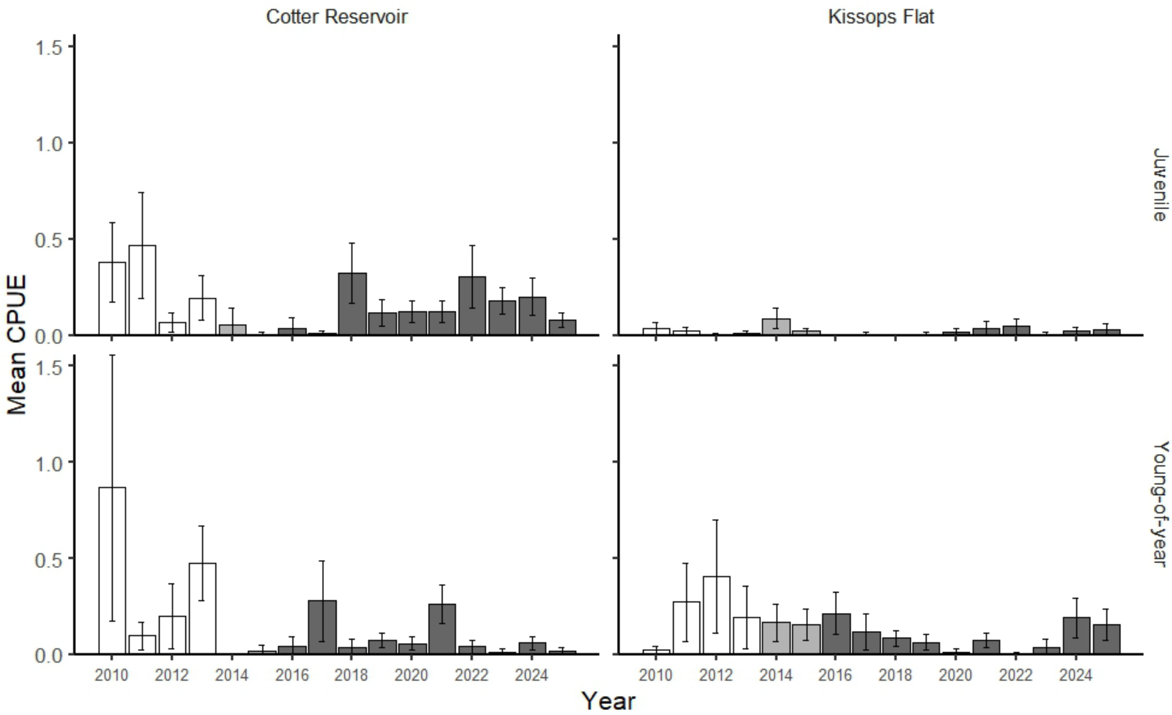

Figure 13. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections, scaled to relative net effort versus shoreline length at the time of sampling) of juvenile (>85 mm TL) and young-of-year (< 85 mm TL) Macquarie perch captured in Cotter Reservoir (pre and post enlargement) (impact site) and Kissops Flat (reference site) using fyke nets between 2010 and 2025. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronological order from 2010 to 2025 from left to right on the x-axis.

Young-of-year Macquarie perch

A total of 56 young-of-year (YOY) Macquarie perch were captured using fyke nets in the ECR in 2025. Analysis revealed a significant decline in YOY CPUE at the ECR during both the filling and operational phases relative to the baseline (Table 11). In contrast, the reference site, Kissops Flat, exhibited a smaller decline, with a significant positive interaction during the filling phase, indicating a relative increase in CPUE compared to Cotter during this phase (Table 11). The interaction for the operational phase at the reference site was non-significant (Table 11). These findings indicate that the ECR has experienced a site-specific and phase-dependent decline in YOY Macquarie perch abundance in both filling and operational phases.

Table 6. Estimated effects of monitoring phase on catch per unit effort (CPUE) of young-of-year Macquarie perch (<80 mm total length), captured using fyke nets in Cotter Reservoir and at the reference site (Kissops Flat). Estimates are derived from generalized linear mixed models (GLMMs) fitted with a negative binomial distribution and a log link function. Values are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-0.9414	0.2857	-3.295	0.001
Filling	-3.7419	1.2686	-2.95	0.0032
Operational	-1.5843	0.3535	-4.482	<0.001
Reference site	-0.6021	0.2796	-2.154	0.0313
Filling * Reference site	3.404	1.2754	2.669	0.0076
Operational * Reference	0.6473	0.3748	1.727	0.0842

Yearly trends in YOY CPUE revealed that 2010 had the highest mean CPUE, indicating a strong recruitment year during the baseline phase (Figure 13). In contrast, filling years of 2014 and 2015, showed marked declines in YOY CPUE (Figure 13), contributing significantly to the observed phase-related differences (Table 11). Although some variability was observed in later years (e.g., 2017), mean CPUE remained low throughout the operational phase, reinforcing the pattern of reduced recruitment following reservoir filling.

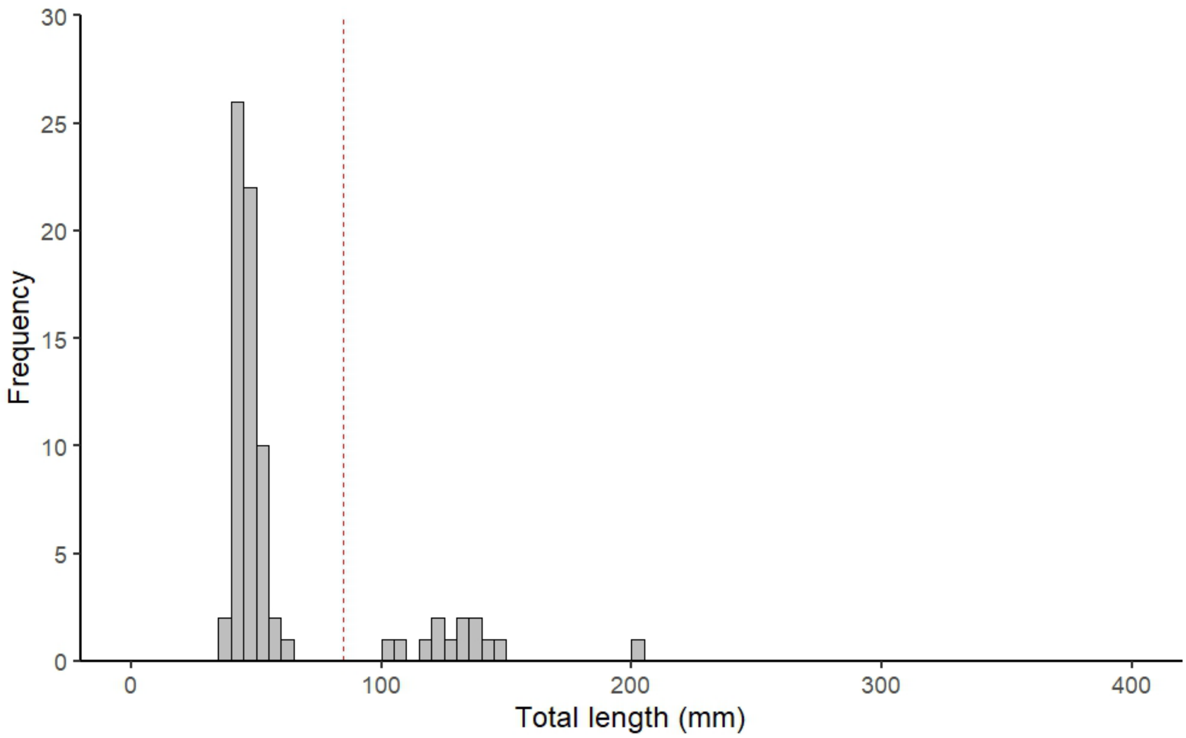

Figure 14. Length frequency of Macquarie perch captured from Kissops Flat (Reference site) on the upper Murrumbidgee River in 2025 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

DISCUSSION AND CONCLUSIONS

Annual abundances of Macquarie perch size classes in Cotter Reservoir have been highly variable since monitoring began, which is partly expected due to natural variations in recruitment and mortality arising from a range of environmental factors. However, the question is to what extent more recent changes in the size structure of fish can be attributed to shifts in the reservoir habitat conditions, and connectivity to upstream sections of the Cotter River.

Relative abundance (CPUE) of adult Macquarie perch (as captured in gill nets) in the ECR in 2025 was the lowest since filling commenced in 2013, and in terms of raw numbers was the lowest since monitoring began in 2010. Statistical analysis also revealed a decline in adult CPUE over time, though this was largely due to extraordinarily high abundance in 2015, and to a lesser extent 2014 and 2016. Water temperatures in the ECR throughout the monitoring period in 2025 were the highest since monitoring began, and it was hypothesised that this may have been driving the low captures of adult Macquarie perch in the gill nets because of adult movement behaviour. A movement study of adult Macquarie perch undertaken in 2008 – 2009 found that tagged fish occupied deeper water (average of 8 m) during summer than other seasons (Thiem et al., 2012). The longest gill nets in our fleet only have a drop of around 4 m, and fish moving deeper in the water column are not going to encounter them. Supplementary gill netting undertaken when water temperatures had cooled to ~17.5 °C saw an increase in the capture rates of adult Macquarie perch in the ECR in 2025, and provided some evidence to support our hypothesis that high water temperatures were contributing to low numbers of adult Macquarie perch captured. Future sampling as part of this program, especially during warmer than usually climatic conditions, should aim to commence sampling once water temperatures have dropped below 20 °C.

Estimates of actual population size of long-lived freshwater fish species requires a strategic balance between ecological relevance and resource efficiency. For species with lifespans exceeding a decade, such as the Macquarie perch (*Macquaria australasica*), undertaking population estimates every 3 years would strike this balance. This frequency allows for the detection of meaningful population trends while minimising disturbance to the population and logistical burden (Radinger et al. 2019). This frequency allows for tracking population trends over a meaningful portion of the lifespan of a Macquarie perch and provides a reasonable chance of detecting significant changes.

Catch per unit effort (CPUE) of young-of-year (YOY) Macquarie perch captures in fyke nets in the ECR have been relatively low over the past four years, with 2023 and 2025 being the lowest since filling was completed. The previous three years of low abundances of YOY captured in the ECR were largely attributed to flow pulses during spawning season impacting on egg and larval development. The spawning season of 2024 included a large flood pulse at the end of November / start of December, when larvae are generally at the peak in the river (Broadhurst et al., 2012a) . Tonkin et al. (2017) found that high discharges during spawning season was negatively associated with recruitment strength of Macquarie perch. The decrease in recruitment strength is likely driven by high water velocities washing out spawning sites and potentially damaging eggs and early larval phase individuals, or displacing them to less desirable habitat. These impacts are fundamentally outside the reach of reservoir operation, though their compounding effects have the potential to drive future ECR management priorities (i.e. if drawdown was to occur, management of Macquarie perch spawning on the back of a run of poor recruitment years would be of high priority).

Despite the low numbers of YOY over the past few years, juvenile Macquarie perch abundances have been relatively stable, and comparable to all years except the highest abundance years. Relative abundance of juveniles in 2025 was the lowest since the recruitment drought broke in 2017. The discordance between low numbers of young-of-year (YOY) and relatively sound numbers of juveniles may be due to compensatory density dependence, an ecological concept that suggests when population densities are low, such as following a poor recruitment year, individual survival rates can increase due to reduced competition for limited resources like food and shelter. In freshwater fish populations, this can result in a greater proportion of YOY surviving to older age classes, even if initial recruitment was low. This mechanism has been well-documented in fish population dynamics and is considered a key factor in buffering populations against environmental variability and recruitment failure (Rose et al., 2001).

RECOMMENDATIONS

Adult population

Current methods for surveying the majority of the adult Macquarie perch population size classes appear to be adequate, although there may be some size-based or behaviour-based bias in capture efficiency and some effects of water temperature on catchability. We recommend continuation of the increased effort to capture larger size classes of Macquarie perch which includes all the following:

1) Additional 125 mm 33 meshes deep 'shallow' gill net;
2) Deployment of 125 mm 66 meshes deep 'deep' gill net; and
3) Extra boat electrofishing effort (night-time boat electrofishing).

Based on the results of this year's gill netting (and supplementary gill netting) we recommend that future sampling for adult Macquarie perch using gill nets commence once water temperatures have fallen below 20 °C. Sampling at water temperatures above 20 °C are likely to result in artificially low estimates of adult abundance in the ECR.

A formal population estimate of adult Macquarie perch in Cotter Reservoir should be undertaken every 3 years to allow for timely adjustments to management practice. With the last population estimate provided in 2023, we recommend additional effort be employed in 2026 to undertake a commensurate population estimate of adult Macquarie perch in Cotter Reservoir. Namely, in addition to the standard survey effort, a recapture effort of;

- Two nights of gill netting (12 gill nets per night);
- 3 nights of fyke netting (20 fyke nets per night);
- Two days and two nights of boat electrofishing.

Juvenile population

The strong presence of 1+ and 2+ Macquarie perch in ECR indicates that conditions in the reservoir were suitable for survival and growth during the early life history of Macquarie perch over recent years. At this stage, no management intervention is recommended for juvenile Macquarie perch in Cotter Reservoir.

Young-of-year

Increased fyke net effort has reduced the variability in young-of-year captures between net nights and provides a comparable level of effort to that of baseline monitoring. We recommended continuation of the increased fyke netting effort in Cotter Reservoir.

The annual detection of a cohort of young-of-year since 2017 is heartening after the consecutive recruitment failures in 2014-2016. Continuing fyke net monitoring as the ECR moves further into operational phase (i.e. use for water supply and further fluctuations in water level) is essential to determine whether reservoir Macquarie perch can spawn and recruit during fluctuating and regulated conditions.

Larval monitoring

It is recommended that snorkelling continues as currently undertaken, as it is a well-tested method that can detect even low numbers of larvae in the Cotter catchment (Broadhurst et al., 2012a) and provides another important link when assessing recruitment of a given year.

QUESTION 2: Has there been a significant change in the abundance and distribution of Macquarie perch in the Cotter River above and below Vanitys Crossing as a result of the filling and operation of the ECR?

BACKGROUND

The construction of Vanitys Crossing fishway in 2001 (Ebner and Lintermans 2007) has allowed the Macquarie perch population to expand its distribution upstream of this road crossing (Broadhurst et al., 2012a, Broadhurst et al., 2013, Broadhurst et al., 2015, Broadhurst et al., 2016b). Remediation of the fish passage barrier at Pipeline Road Crossing (ACTEW Corporation, 2009b) was designed to open up the availability of further spawning habitat for the species. The remediation of Pipeline Road Crossing is an offset to compensate for the inundation of existing Macquarie perch spawning habitat by the ECR (ACTEW Corporation, 2009a). The successful expansion of the distribution of Macquarie perch past this upstream road crossing is largely reliant on the continued success of the Vanitys Crossing fishway, as otherwise reservoir fish are largely blocked from migrating up the river. Monitoring is required to determine the success of fish passage remediation at Vanitys Crossing and Pipeline Road Crossing and the effects of improved access to additional spawning habitat by the riverine Macquarie perch population. Enhancement of the distribution of riverine Macquarie perch will decrease the likelihood of localised extinctions associated with stochastic events.

SAMPLING DESIGN

Sampling design for Question 2 follows that of the baseline monitoring program Question 8 (Lintermans et al., 2013), with a few changes (Table 7). The site immediately above the old Cotter Reservoir (Bracks Hole) has been inundated and is no longer a riverine site, so a riverine site between ECR full supply level and Vanitys Crossing has been monitored as a substitute. The site immediately downstream of Bendora Dam has been dropped from the monitoring program as this site is unlikely to be directly affected by the operation of ECR.

Table 7. Outline of the sampling design for Question 2 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Macquarie perch. Sub-adults / adults (> 150 mm TL), Juveniles (80 > 150 mm TL) and young-of-year (< 80 mm TL).
Sampling technique/s	Fyke nets (12 per night; 3 nets per pool at four pools for 1 night); Backpack electro-fishing (4 x 30 m sections).
Timing	Conducted annually in late summer / early autumn.
Number / location of sites	5 sites on the Cotter River between full supply level and Burkes Creek Crossing (see Figure 1) and one reference site (Kissops Flat).
Information to be collected	Number and total length (mm) for all Macquarie perch.
Data analysis	Catch-per-unit-effort (CPUE) of fyke netting data assessed between years and sites using zero-inflated Poisson (ZIP) regression models for all sizes combined, juveniles and young of year.

TARGET SPECIES AND LIFE STAGE

Adult / sub-adult, juvenile and young-of-year Macquarie perch were targeted. Individuals were classed as adults if they were > 150 mm TL, based on results from Ebner and Lintermans (2007) who found that males are sexually mature from this size. At the time of net sampling (i.e. late summer-early autumn) young-of-year in the river will be generally 40 - 70 mm TL based on results of the data collected between 2010 and 2021 (Figure 15). Individuals were generally considered juvenile if they fell between 100 – 150 mm TL.

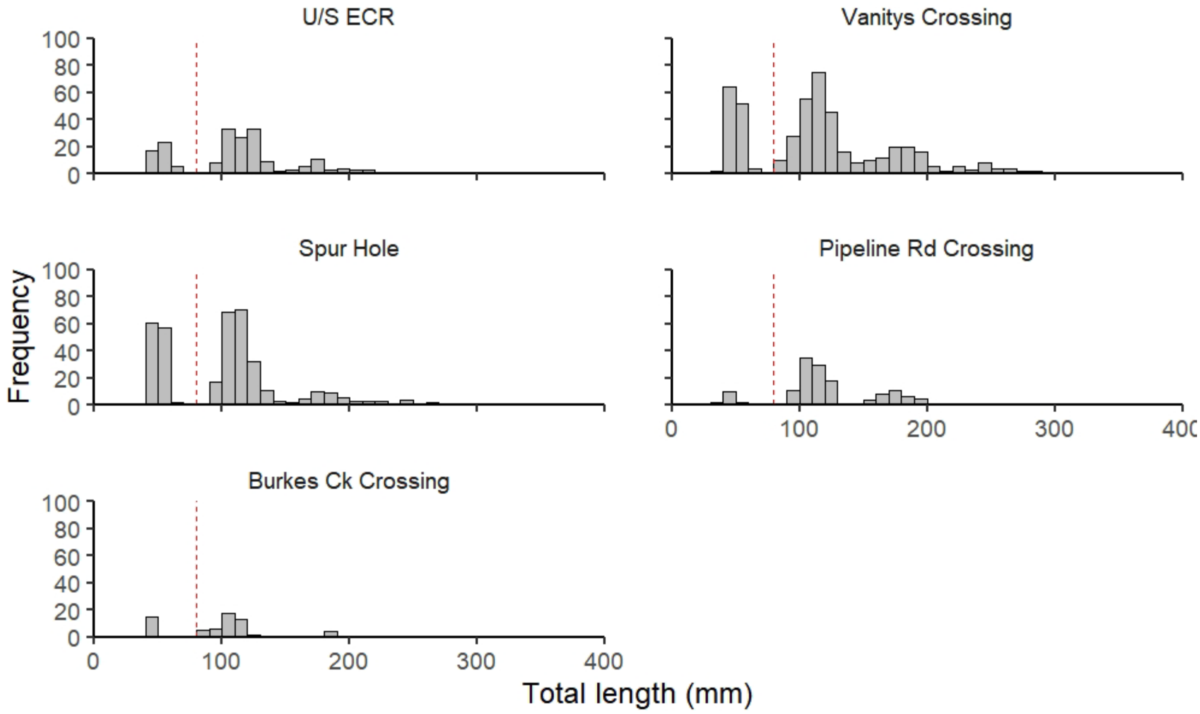

Figure 15. Length frequency of Macquarie perch captured in fyke nets and backpack electrofishing from Cotter River between 2010 and 2021 at sites; U/S ECR, Vanitys Crossing, Spur Hole, Pipeline Road Crossing and Burkes Creek Crossing (red dashed line indicates cut-off for length of young-of-year individuals < 80 mm TL).

SAMPLING METHODS AND NUMBER OF REPLICATES

Fyke netting (12 mm stretch mesh, single-winged) and backpack electrofishing were employed to monitor riverine sites for Macquarie perch. Twelve fyke nets were set in pools overnight (~16-hour soak time) per site (three nets per pool for four pools with the exception of Kissops Flat which was 12 fykes in one large pool). Backpack electrofishing (4 x 30 m sections) was conducted in wadeable (i.e. depths less than 0.8 m) sections of each site, except Kissops Flat as this sampling technique was dropped due to budget constraints and low capture rates using this method at this site.

TIMING

Sampling for this question was undertaken in February and March 2025 (so as to be comparable with sampling undertaken in the baseline monitoring program).

NUMBER AND LOCATION OF SITES

Five sites were monitored between Cotter Reservoir and Burkes Creek Crossing (see Figure 1) and one reference site on the upper Murrumbidgee River (Kissops Flat). Monitoring sites on the Cotter River are (from downstream to upstream) U/S ECR (approximately 150 – 750 m upstream of ECR full supply level), Vanitys Crossing, Spur Hole, Pipeline Road Crossing and Burkes Creek Crossing. U/S ECR replaces the now-inundated Bracks Hole in the Operational sampling design.

DATA ANALYSIS

To assess temporal and spatial variation in Macquarie perch catch per unit effort (CPUE) across different life stages, we fitted a series of zero-inflated Poisson (ZIP) regression models using the glmmTMB package in R. Separate models were constructed for young-of-year (YOY), juvenile, and total CPUE. For the YOY and juvenile datasets, we specified ZIP models with fixed effects for monitoring phase (baseline, filling, operational), site type (control vs. impact), and their interaction to capture potential BACI effects. Zero-inflation was modelled using an intercept-only term to account for excess zeros in the data. For total CPUE, we extended the model to include random intercepts for site and year to account for repeated measures and temporal autocorrelation. All models assumed a Poisson distribution with a log link function. Model outputs were used to evaluate whether CPUE differed between the baseline and subsequent phases, and whether these differences varied between control and impact sites.

RESULTS

General

A total of 85 Macquarie perch were captured by fyke nets in the Cotter River across the five sites in 2025, ranging in total length (TL) from 44 – 276 mm (Figure 16). Young-of-year (< 80 mm TL) and 1+ year old / juvenile (80 – 150 mm) individuals were captured at four and five riverine sites in 2025, respectively (Figure 16 & Figure 17). The GLMM indicated that there was a significant decline in CPUE during the operational phase at the control site, while no significant change was observed during the filling phase (Table 8). Although CPUE appeared lower at impact sites overall, this difference was not statistically significant (Table 8). Importantly, the interaction terms between phase and site type were also non-significant, indicating no strong evidence that CPUE trends differed between control and impact sites across phases (Table 8).

Table 8. Estimated effects of reservoir management phase and site on catch per unit effort (CPUE) of Macquarie perch (all size classes combined), captured using fyke nets in the Cotter River and Kissops Flat between 2010 and 2025. Estimates are derived from a zero-inflated Poisson (ZIP) regression model and are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-1.35201	0.56619	-2.388	0.01694
Filling	-0.13725	0.36778	-0.373	0.70902
Operational	-0.78273	0.28028	-2.793	0.00523
Impact sites	-1.00408	0.65793	-1.526	0.12698
Filling * Impact sites	0.08681	0.51747	0.168	0.86678
Operational * Impact sites	0.50035	0.39202	1.276	0.20184

Juveniles

Relative abundance of juvenile Macquarie perch at U/S ECR (formerly Bracks Hole) and Vanitys Crossing was highly variable through time, with peaks in CPUE in 2013 and 2012 at each of these sites, respectively (Figure 17). Juvenile Macquarie perch were detected from at least four of the five monitored sites in each monitoring year in the Cotter River, and at all five sites in 2010, 2011, 2014, 2018, 2019, 2020 and 2025 using fyke nets (Figure 17). Macquarie perch were not detected at Burkes Creek Crossing in 2012, 2013, 2016, 2023 or 2024 using fyke nets and at very low abundances at this site in other years, apart from 2020 which had relatively high abundances of individuals captured (Figure 17). The model for juvenile Macquarie perch CPUE did not identify any statistically significant effects of monitoring phase, site type, or their interaction. While the intercept was significantly negative, indicating low baseline CPUE for juveniles, neither the filling nor operational phases differed significantly from the baseline, and impact sites did not show a consistent difference from the control site. Interaction terms were also non-significant, suggesting no evidence of a differential response between control and impact sites over time. These results imply that juvenile CPUE remained relatively stable across phases and site types, or that variability in the data limited the ability to detect meaningful differences.

Table 9. Estimated effects of reservoir management phase and site on catch per unit effort (CPUE) of juvenile Macquarie perch (80 – 150 mm TL), captured using fyke nets in the Cotter River and Kissops Flat between 2010 and 2025. Estimates are derived from a zero-inflated Poisson (ZIP) regression model and are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-4.11821	0.835637	-4.928	0.0000
Filling	1.207112	1.03978	1.161	0.2460
Operational	-0.00365	0.977376	-0.004	0.9970
Impact sites	1.109717	0.896545	1.238	0.2160
Filling * Impact sites	-0.89633	1.144711	-0.783	0.4340
Operational * Impact sites	-0.06819	1.049321	-0.065	0.9480

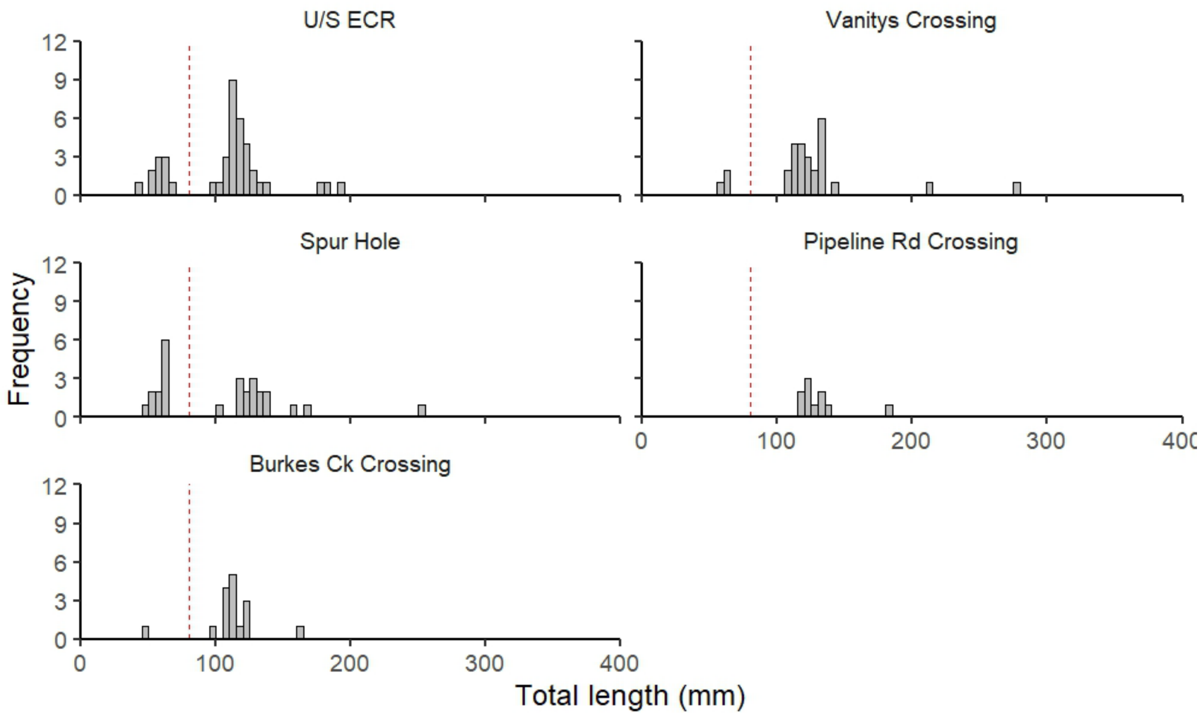

Figure 16. Length frequency of Macquarie perch captured in fyke nets and backpack electrofishing from Cotter River in 2025 at sites; U/S ECR, Vanitys Crossing, Spur Hole and Pipeline Road Crossing (red dashed line indicates cut-off for length of young-of-year individuals < 80 mm TL).

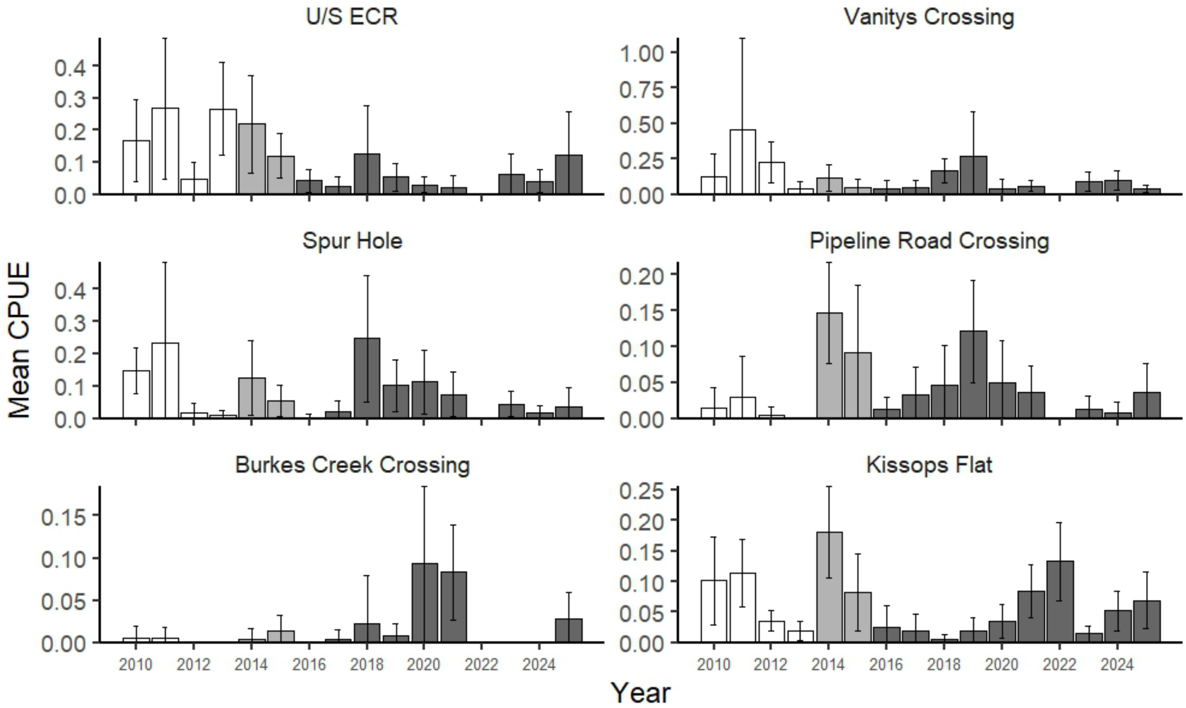

Figure 17. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of juvenile Macquarie perch (80 – 150 mm TL) captured in Cotter River and Kissops Flat (Murrumbidgee River) using fyke nets between 2010 and 2025. (Note that Bracks Hole (sampled from 2010 – 2013) was replaced by U/S ECR (sampled in 2014 – 2025) as the most downstream riverine site). White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates operational phase of monitoring program. Note: Cotter River sites were not able to be sampled in 2022 due to high river flows throughout the sampling period.

Relative abundance of Macquarie perch captured using backpack electrofishing was highly variable between sites and years (Figure 18). Backpack electrofishing captured a total of nine Macquarie perch (all 1+ year olds) at two of the five sites (Vanitys Crossing and Burkes Ck Crossing) in 2025. Macquarie perch were not captured at any site in 2011 and 2012 using backpack electrofishing (Figure 18). Excessive numbers of zero samples prevented statistical testing, which in itself, highlights the patchy presence of Macquarie perch above and below Vanitys Crossing over most years as detected by electrofishing.

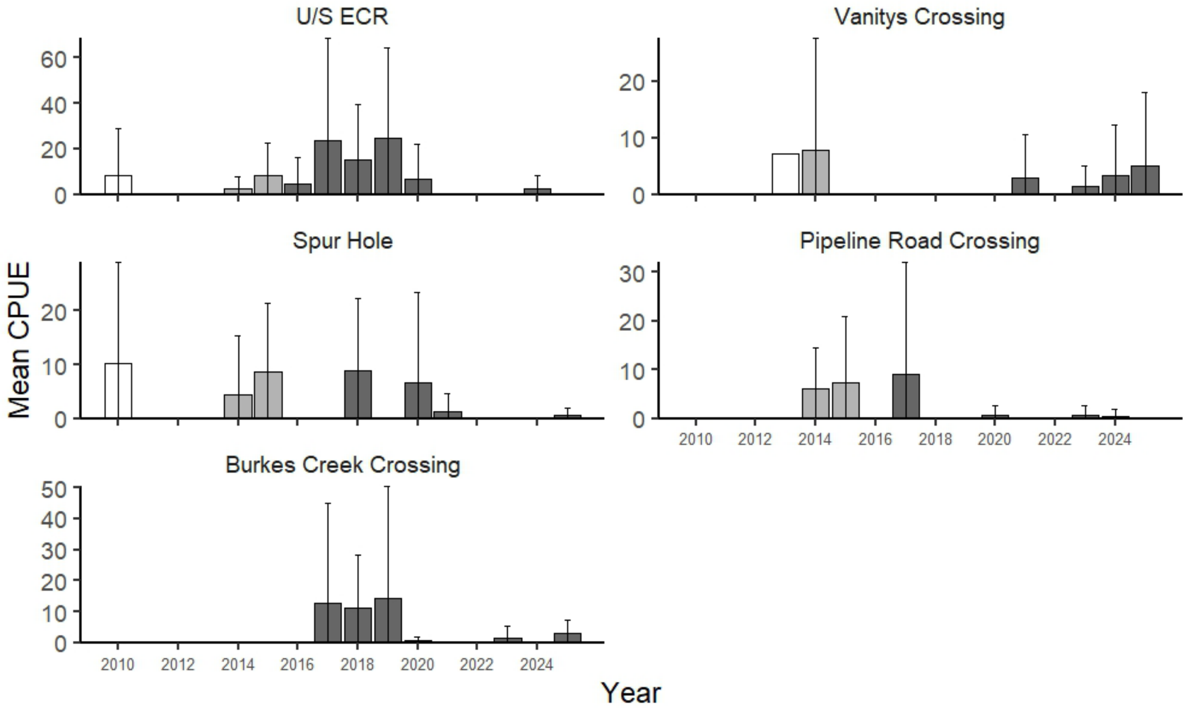

Figure 18. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Macquarie perch (all sizes pooled) captured in Cotter River by backpack electrofishing between 2010 and 2025. (Note that Bracks Hole (sampled from 2010 – 2013 was replaced by U/S ECR (sampled in 2014 – 2025) as the most downstream riverine site). White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates operational phase of monitoring program. Note: Cotter River sites were not able to be sampled in 2022 due to high river flows throughout the sampling period.

Young-of-year (YOY)

A total of 20 YOY Macquarie perch (< 80 mm TL) were captured using fyke nets in the Cotter River in 2025 (Figure 19 and Figure 16). Young-of-year were detected at the four of the five sites on the Cotter River in 2025 (excluding Pipeline Rd. Crossing). Analysis of CPUE for young-of-year (YOY) Macquarie perch revealed significant differences associated with both monitoring phase and site type. The operational phase was associated with a significant decrease in YOY CPUE compared to the baseline phase, indicating a decline in recruitment during this period. Additionally, impact sites exhibited significantly lower YOY CPUE overall compared to the control site. However, the interaction terms between phase and site type were not statistically significant, suggesting that the temporal changes in YOY CPUE were consistent across both control and impact sites. These findings indicate a general decline in YOY abundance or catchability over time, particularly during the operational phase, but do not provide strong evidence that this decline was more pronounced at impact sites relative to the control.

Table 10. Estimated effects of reservoir management phase and site on catch per unit effort (CPUE) of young of year Macquarie perch (< 80 mm TL), captured using fyke nets in the Cotter River and Kissops Flat between 2010 and 2025. Estimates are derived from a zero-inflated Poisson (ZIP) regression model and are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-1.4916	0.2247	-6.637	0.0000
Filling	-0.3332	0.4239	-0.786	0.4318
Operational	-0.848	0.3062	-2.77	0.0056
Impact sites	-2.2156	0.5125	-4.323	0.0000
Filling * Impact sites	-0.5082	1.086	-0.468	0.6398
Operational * Impact sites	0.8397	0.6181	1.359	0.1743

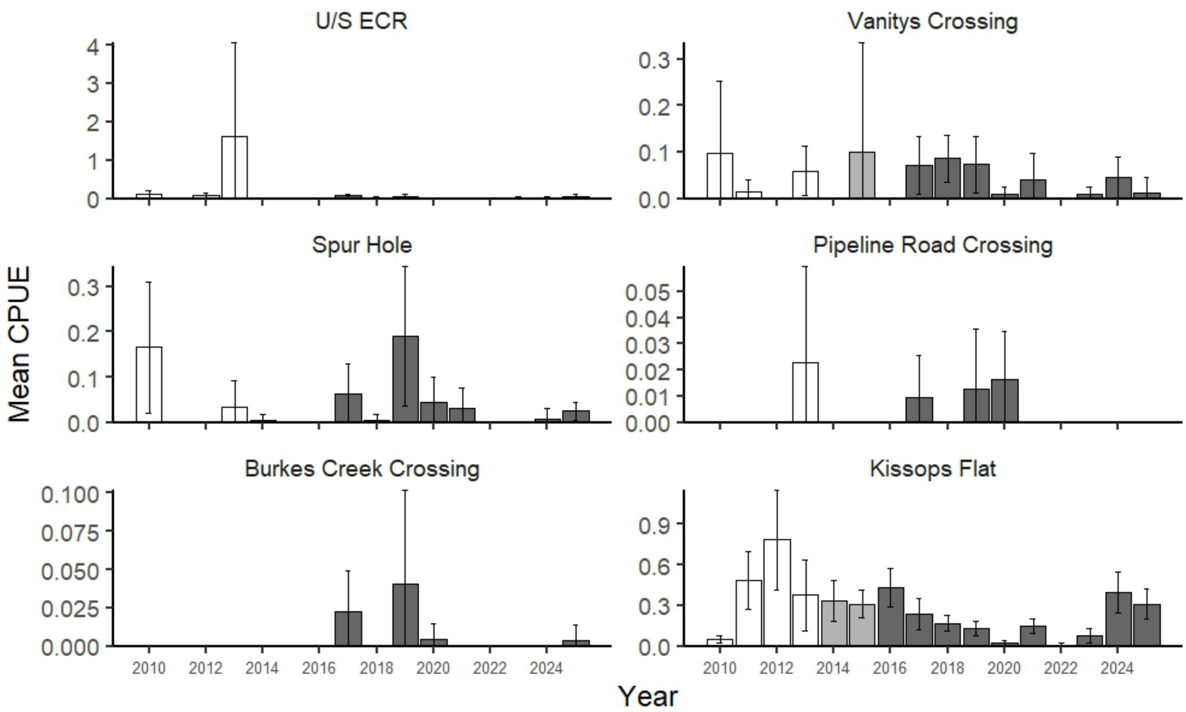

Figure 19. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Young-of-year Macquarie perch (< 80 mm TL) captured in Cotter River Kissops Flat (Murrumbidgee River) by fyke netting between 2010 and 2025. (Note that Bracks Hole (sampled from 2010 – 2013) and U/S ECR (sampled in 2014 – 2025) have been combined into the site U/S ECR to represent the site immediately u/s of the impounded waters). White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates operational phase of monitoring program. No sampling was undertaken in 2022 due to flooded river conditions during the monitoring period.

DISCUSSION AND CONCLUSIONS

Relative abundance

As has been the case since monitoring began in 2010, relative abundance of Macquarie perch in Cotter River in 2025 was variable between sites, as determined by both fyke netting and backpack electrofishing. Previously relative abundance generally decreased with distance upstream from Cotter Reservoir. These results are consistent with previous findings that this Macquarie perch population was restricted to Cotter Reservoir and the Cotter River downstream of Vanitys Crossing until the fishway was built in 2001, with the species taking considerable time in extending their population to newly accessible upstream river reaches (Broadhurst et al., 2012a, Lintermans, 2013).

One-year old individuals were present at all sites in 2025. This follows on from young-of-year being detected in the Cotter River 2024. These results suggest that the past year has been suitable for survival and growth of juvenile Macquarie perch in the Cotter River.

Young-of-year Macquarie perch were captured from four sites on the Cotter River in 2025 and comprised 23.5% of the total number of Macquarie perch captured. Recruitment in 2025 was relatively low compared to most other years and is in decline. Although the mechanism behind the continued low levels of Macquarie perch recruitment is not as clear as previous years (when multiple flood pulses most likely impacted on early development), a pulse in December 2024 may have contributed to a reduction in recruitment (see discussion in Question 1 above). The statistical decline of young-of-year Macquarie perch is likely because of sequential low levels of recruitment, attributed to unfavourable hydrological conditions (flow pulse during spawning and early develop), and is largely consistent with the trends observed at the reference site. To date, monitoring suggests that the Macquarie perch population is surviving but not thriving in the Cotter River.

Distribution

Macquarie perch were detected at the four most downstream sites in all years and at the fifth site in 10 of 14 years, indicating that their distribution is somewhat stable. Distribution differences between years is likely driven by the decreasing density (and potential patchy capture at low density sites) as one moves upstream from Cotter Reservoir and not a true change in the actual distribution of this population between years. The difficulties in detecting rare species are well documented (Maxwell and Jennings, 2005, Poos et al., 2007, Joseph et al., 2006, Lintermans, 2016). The stable distribution suggests that conditions in the Cotter River habitat and hydrology is suitable for survival, growth and even reproduction across sites and years.

RECOMMENDATIONS

Juveniles and adults

Methods for assessing the population of Macquarie perch < 150 mm TL appear to be adequate, given they have been tested across a range of natural variation in recruitment of this species for many years (Lintermans, 2013, Lintermans et al., 2013, Ebner and Lintermans, 2007, Lintermans, 2016). The limitations of fyke nets and backpack electrofishing in sampling adult Macquarie perch is well understood (Lintermans, 2013, Lintermans, 2016) and deployment of gill nets to sample adults in the river would involve significant additional cost, and pose significant risk of platypus bycatch. Provided the presence of young-of-year or juvenile individuals is readily detected, the presence of adults can be inferred. No change to monitoring recommended.

Juvenile Macquarie perch were detected at all sites in 2025. At this stage, no management intervention is recommended for juvenile Macquarie perch in Cotter River.

Young-of-year

Methods for assessing the YOY relative abundance appear to be adequate. No change to the monitoring program is recommended.

Young-of-year were detected at four of five riverine sites in 2025, suggesting conditions were suitable for recruitment in the catchment, albeit at a relatively low level compared to some other years. No management intervention is recommended at this stage.

QUESTION 3: Have Two-spined blackfish established a reproducing population in the enlarged Cotter Reservoir and are they persisting in the newly inundated section of the Cotter River?

BACKGROUND

Two-spined blackfish have long been absent from Cotter Reservoir (Ebner et al., 2008, Lintermans, 2002) (thought to be a result of excessive sedimentation smothering potential spawning sites) apart from a small number of individuals detected in 2012, possibly washed down from the river during flooding (Lintermans et al., 2013) (Figure 20). However, the species was present in the river reach inundated by the ECR (Ebner et al., 2008, Lintermans et al., 2013). Inundated habitats around the perimeter of the ECR should provide suitable spawning habitats for the species. The monitoring program will determine whether the species persists in the newly inundated river reach, and subsequently expands to colonise newly inundated habitats around the perimeter of the ECR.

METHODS

Sampling design for Question 3 follows a similar approach to the baseline monitoring program (Lintermans et al., 2013). One of the reference reservoirs from the baseline monitoring program (Corin Reservoir) was dropped from the subsequent (filling and operational) monitoring program to minimise costs.

Table 11. Outline of the sampling design for Question 3 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Two-spined blackfish; Adult (>150 mm TL); juveniles (80 – 150 mm) and young-of-year (<80 mm).
Sampling technique/s	Fyke nets (20 set on the first night around the entire perimeter as part of question 1; then the 8 most upstream nets from nights 2 and 3 of the 20 set as part of questions 1), 12 x 1 night in Bendora Reservoir. 10 x Bait traps (with light stick) set in the newly inundated section of the reservoir.
Timing	Conducted annually in late summer- early autumn.
Number / location of sites	3 sites; 1 around the entire ECR, 1 focussed in the newly inundated area and Bendora Reservoir (reference site).
Information to be collected	Number and total length (mm) for all Two-spined blackfish.
Data analysis	Catch-per-unit-effort (CPUE) assessed between years where possible using 95% (Bonferroni corrected) confidence limits.

Sampling targeted adult, juvenile and young-of-year Two-spined blackfish. Individuals were classed as adults if they are > 150 mm TL, juveniles if 80 – 150 mm TL; and young-of-year if <80 mm TL based on results of Lintermans (1998). At the time of sampling (i.e. late summer / early autumn) young-of-year will be approximately 50 – 79 mm TL based on results of the baseline data collected (Lintermans et al., 2013).

Overnight fyke netting (approx. 16 hours soak time) was used to capture Two-spined blackfish. For the reproduction component of the question all 20 nets from the first night of netting for question 1 was used. For the persistence in the inundation zone component, the eight most upstream nets from nights two and three of sampling undertaken as part of question 1 were used. Sampling for this question is undertaken annually in late summer-early autumn (to be comparable with sampling undertaken in the baseline monitoring program). Two sites within the reservoir were monitored, one around the entire ECR (to detect establishment and recruitment in the ECR), one in the newly inundated section of the ECR (upstream of Bracks Hole reach) and one reference site at Bendora Reservoir. Bait traps were not able to be employed in 2014 as it was not possible to get sufficient number of identical traps in time for sampling (same mesh size, shape, entrance size and colour).

Abundance was standardised as fish caught per net hour (represented as CPUE). Due to the predominance of zero catch data across most samples in Cotter Reservoir, formal statistical tests were not feasible for differences between years. Abundance between years was assessed in Bendora by comparing mean (fish per net hour) CPUE using 95% confidence limits (with Bonferroni correction) overlap.

RESULTS

Five Two-spined blackfish were captured in by fyke nets in the ECR in 2025 as part of the all reservoir netting component, with all five captured in the most upstream end of the ECR (Figure 20). Lengths of Two-spined blackfish captured from ECR in 2025 ranged from 156 – 226 mm (TL), meaning that all individuals were likely mature adults. There were no Two-spined blackfish captured in the bait traps set in the ECR in 2025. Only a single Two-spined blackfish was captured in Bendora Reservoir in 2025 (Figure 20). This continues a run of very low abundances of Two-spined blackfish in Bendora Reservoir and a population that has been in decline for some time (Figure 20).

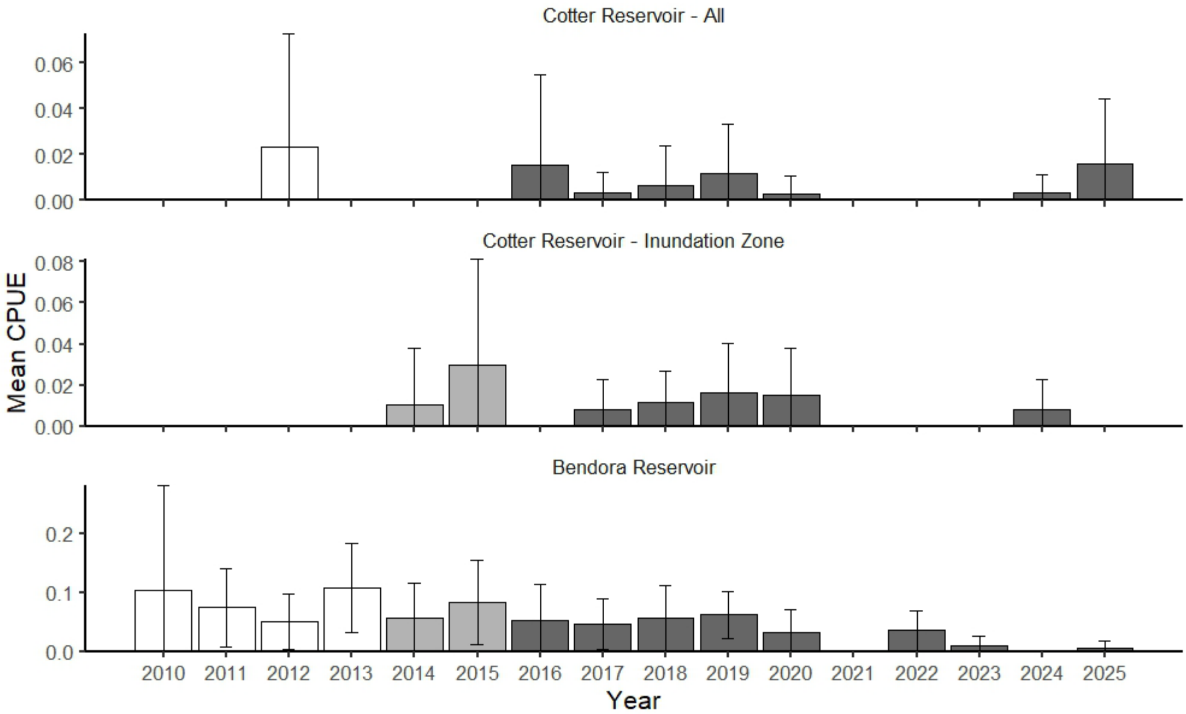

Figure 20. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Two-spined blackfish captured by fyke netting in Cotter Reservoir (both all around the reservoir and just the inundation zone) and Bendora Reservoir between 2010 and 2025. For Cotter Reservoir, white bars indicate baseline phase, grey bars indicate filling phase and white bars with diagonal stripes indicates operational phase of monitoring program.

DISCUSSION AND CONCLUSIONS

Two-spined blackfish have been in very low abundances in Cotter Reservoir in the 16 years of ECR monitoring. This result supports previous research that identified that the original Cotter Reservoir had sub-optimal habitat for Two-spined blackfish as a result of forestry and associated sedimentation of the reservoir smothering rocky substrate preferred by this species (Lintermans, 1998, Broadhurst et al., 2012b, Broadhurst et al., 2011, Ebner et al., 2008, Ebner and Lintermans, 2007).

The capture of Two-spined blackfish in the newly inundated upstream third of the enlarged Cotter Reservoir, suggest that newly inundated shoreline of the enlarged Cotter Reservoir may serve as suitable habitat for the species, though at low densities. To date, 40 Two-spined blackfish have been captured in fyke nets in the ECR, with only two smaller than 150 mm TL (size at sexual maturity). No recruitment of Two-spined blackfish has yet been detected in the ECR. It is likely that Two-spined blackfish in the ECR are larger adults that have colonised from the Cotter River upstream.

The continued very low abundances of Two-spined blackfish in Bendora Reservoir is concerning. The status of this population appears to be precarious and requires immediate further investigation.

Furthermore, this site appears to be less representative as a reference site for Two-spined blackfish, as the population appears to be in severe decline. We recommend that a second reservoir reference site be considered to accompany Bendora Reservoir until a robust investigation into the Bendora Reservoir Two-spined blackfish population has been undertaken.

RECOMMENDATIONS

Methods for assessing the population of Two-spined blackfish in reservoirs appear to be adequate. No change to monitoring recommended.

Captures of Two-spined blackfish in the Cotter Reservoir have been rare to this point. At this stage, no management intervention is recommended for Two-spined blackfish in Cotter Reservoir.

Continued low abundance of Two-spined blackfish in Bendora Reservoir is concerning. If the site continues to return very low abundances of blackfish another reference site will need to be sampled (Corin Reservoir). Further, investigation into the driver behind the continued low abundances is required.

QUESTION 4: Has there been a significant change in the abundance, distribution and size composition of adult trout in the enlarged Cotter Reservoir as a result of filling and operation?

BACKGROUND

Trout are a potential threat to Macquarie perch in the Cotter Reservoir due to their potential for significant predation of other fishes (Budy et al., 2013). An increased reservoir area and depth, and the inundation of terrestrial vegetation were predicted to drive a trophic upsurge that could increase food and/or habitat resources for the resident trout population to increase in abundance and biomass within the Cotter Reservoir (Lintermans, 2012). Increased food resources, thermal refuge habitat (increased depth), and improved habitat quality (increased dissolved oxygen as a result of changed destratification procedures) were expected to result in improved growth (and size) of trout individuals, based on their preferred resource requirements (Budy et al., 2013). Monitoring changes in the reservoir trout population is needed to give early warning of potential increases in predatory interactions with Macquarie perch.

METHODS

Sampling design for Question 4 is similar to the baseline monitoring program for Question 3 (Lintermans et al., 2013) (Table 12). One of the reference reservoirs from the baseline monitoring program (Corin Reservoir) was dropped from the subsequent (filling and operational) monitoring program to minimise costs.

Table 12. Outline of the sampling design for Question 4 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Rainbow and Brown trout; sub-adult and adult fish likely to be piscivorous (> 150 mm FL).
Sampling technique/s	10 Gill nets (fleet of mixed mesh sizes, approx. 6 hours soak time, 5 nights netting in Cotter Reservoir, 2 nights netting in Bendora Reservoir).
Timing	Conducted annually in early autumn.
Number / location of sites	Two sites; enlarged Cotter Reservoir (impact) and Bendora Reservoir (reference), with each site divided into 5 sections.
Information to be collected	Number, location and fork length (mm) for both Rainbow and Brown trout.
Data analysis	Catch-per-unit-effort (CPUE) and adult trout assessed between years and reservoir (baseline vs. impact) using GLMM, based on the first two nights of netting from each Reservoir. Size of adult trout was compared between years using LMM.

Sampling targeted sub-adult and adult Rainbow and Brown trout of a size considered to be piscivorous (individuals of >150 mm Fork Length, FL) with sufficient gape to ingest larval or early juvenile Macquarie perch and Two-spined blackfish (Ebner et al., 2007).

Gill netting (as covered in Question 1) was employed to capture trout species, with the exception of the two additional 125 mm gill nets deployed from 2022 that were excluded from analysis for this question. Sampling for this question is undertaken annually in early autumn (so as to be comparable with sampling undertaken in the baseline monitoring program). Two sites were assessed, the impact site (ECR) and a reference site (Bendora Reservoir).

Only Rainbow trout was used in the comparative analyses, as Brown trout does not occur in Bendora Reservoir. Catch-per-unit-effort (CPUE) data for Rainbow trout were analysed using a generalized linear mixed modelling (GLMM) framework to assess temporal and spatial changes in abundance associated with reservoir development. The study followed a Before-After-Control-Impact (BACI) design, with data collected annually from 2010 to 2025 at two sites (impact and control) across three monitoring phases: baseline, filling, and operational. CPUE was calculated as the number of trout captured per net-hour, with 10 nets set over two nights at each site each year. Due to convergence issues and high data sparsity, reservoir section variable was excluded from the final model. A zero-inflated negative binomial GLMM was fitted using the glmmTMB package in R, with fixed effects for site, phase, and their interaction, and random intercepts for year and sample nested within year to account for repeated measures. Zero inflation was modelled with a constant intercept. Model diagnostics were assessed using residual simulation and dispersion tests to evaluate fit and overdispersion.

To assess changes in adult Rainbow trout lengths over time, we analysed data collected annually from 2010 to 2025 at both a control (Bendora Reservoir) and impact site (Cotter Reservoir). We used a linear mixed-effects model to evaluate the effects of monitoring phase, site, and their interaction on trout length, with year included as a random effect to account for repeated measures across time. The model was fitted using restricted maximum likelihood (REML) with the lmer function from the lme4 package in R. Post hoc comparisons were conducted using the emmeans package to explore differences between phases and sites where relevant. Summary analysis of the Brown trout CPUE and lengths is also provided for Cotter Reservoir (as they are not present in Bendora Reservoir).

RESULTS

Abundance and distribution

Five Rainbow trout and one Brown trout were captured in gill nets in the ECR in 2025. The total number of trout captured was the equal lowest (equal to 2022) (Figure 21). Eight Rainbow trout were captured in Bendora Reservoir in 2025 (Brown trout are not present in this reservoir).

The model revealed a significant decrease in CPUE at the reference site (Bendora Reservoir) during both the filling and operational phases compared to baseline (Table 13). While the interaction between site and phase during the filling phase was not statistically significant, the operational phase showed a significant positive interaction, indicating a relative increase in CPUE at ECR during this phase (Table 13). These results suggest that Rainbow trout abundance at ECR increased during the operational phase relative to the reference site. Brown trout abundance was against very low, continuing a sudden drop in abundance of this species in Cotter Reservoir since 2021 (Figure 22).

Table 13. Estimated effects of reservoir management phase and site on catch per unit effort (CPUE) of adult Rainbow trout, captured using gill nets at control (Bendora Reservoir) and impact (Cotter Reservoir) sites between 2010 and 2025. Estimates are derived from a zero-inflated negative binomial generalized linear mixed model (GLMM) and are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-1.5761	0.2731	-5.772	0.0000
Site	-0.5222	0.4285	-1.219	0.2229
Filling	-1.7361	0.8671	-2.002	0.0453
Operational	-0.8585	0.3636	-2.361	0.0182
Site (Cotter Reservoir) * Filling	1.703	1.0455	1.629	0.1033
Site (Cotter Reservoir) * Operational	1.2892	0.524	2.46	0.0139

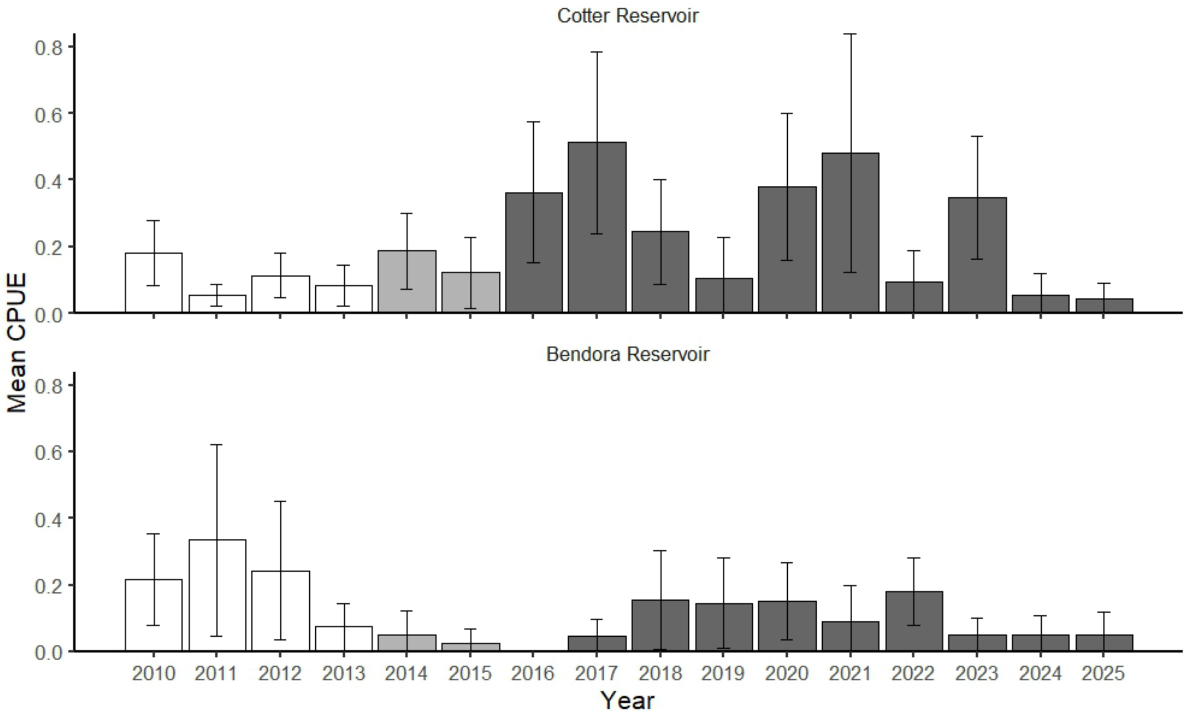

Figure 21. Mean catch-per-unit-effort (± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of adult Rainbow trout captured in Cotter Reservoir and Bendora Reservoir using gill nets each year from 2010 until 2025. White bars indicate baseline phase, light-grey bars indicate filling phase and dark-grey bars indicates operational phase of monitoring program.

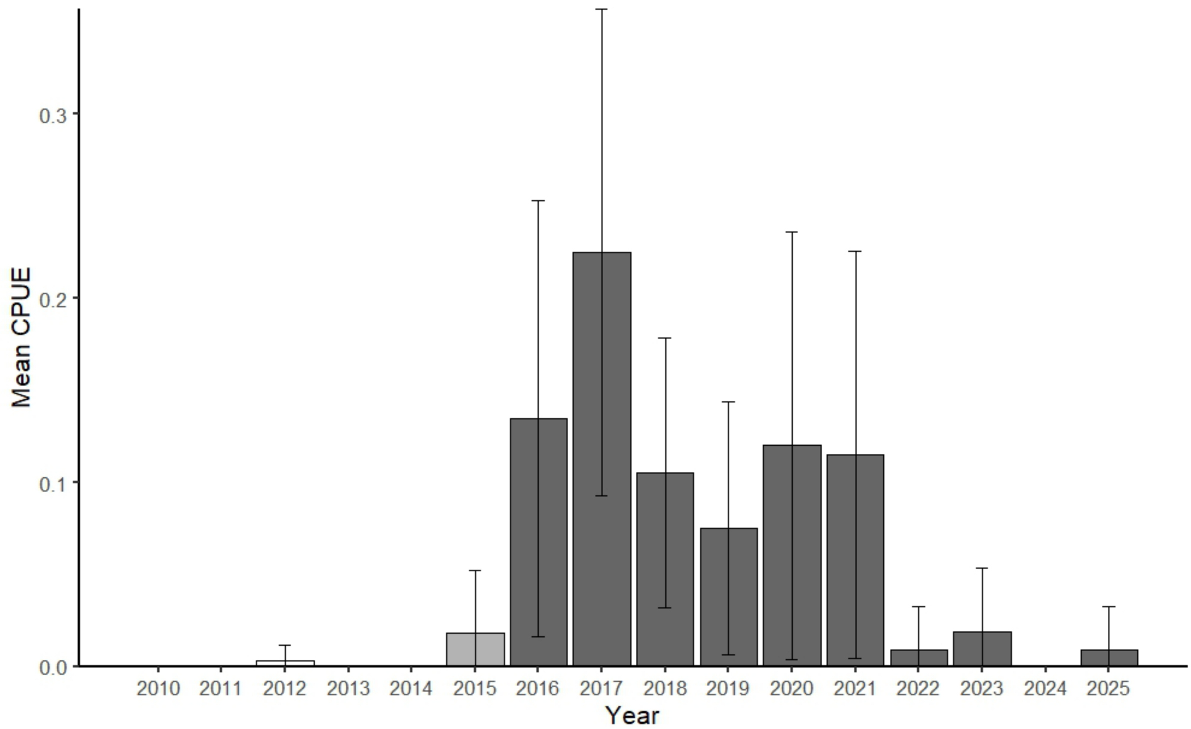

Figure 22. Mean catch-per-unit-effort (± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of adult Brown trout captured in Cotter Reservoir using gill nets each year from 2010 until 2025. White bars indicate baseline phase, light-grey bars indicate filling phase and dark-grey bars indicates operational phase of monitoring program.

Size composition

Linear mixed model analysis revealed a significant effect of monitoring phase on adult Rainbow trout length, with individuals being significantly longer (~37.5 mm) during the operational phase compared to the baseline across both site, though this was lessened at ECR being only ~14 mm longer during operational phase (Table 14). There was no significant difference in length during the filling phase. While the overall site effect was not significant, a significant interaction between phase and site was detected during the operational phase, indicating that the increase in trout length during the operational phase was less pronounced at the impact site (Cotter Reservoir) compared to the control (Bendora Reservoir). These results suggest that trout lengths have changed over time, particularly during the operational phase at Bendora Reservoir.

Table 14. Estimated effects of monitoring phase and site on the total length of Rainbow trout, based on a linear mixed model. The model revealed a significant increase in trout length during the operational phase compared to the baseline, with a smaller increase observed at the impact site (Cotter Reservoir) due to a significant interaction effect. Estimates are presented with standard errors (SE), degrees of freedom (df), t-values, and associated p-values.

Term	Estimate	Std. error	df	t value	Pr(>|z|)
(Intercept)	338.612	7.632	14.336	44.367	<0.001
Filling	11.989	21.394	90.711	0.56	0.577
Operational	37.498	9.941	19.999	3.772	0.001
Site (Cotter Reservoir)	13.847	7.568	649.112	1.83	0.068
Filling * Cotter Reservoir	-10.521	21.952	685.721	-0.479	0.632
Operational * Cotter Reservoir	-23.223	10.173	549.731	-2.283	0.023

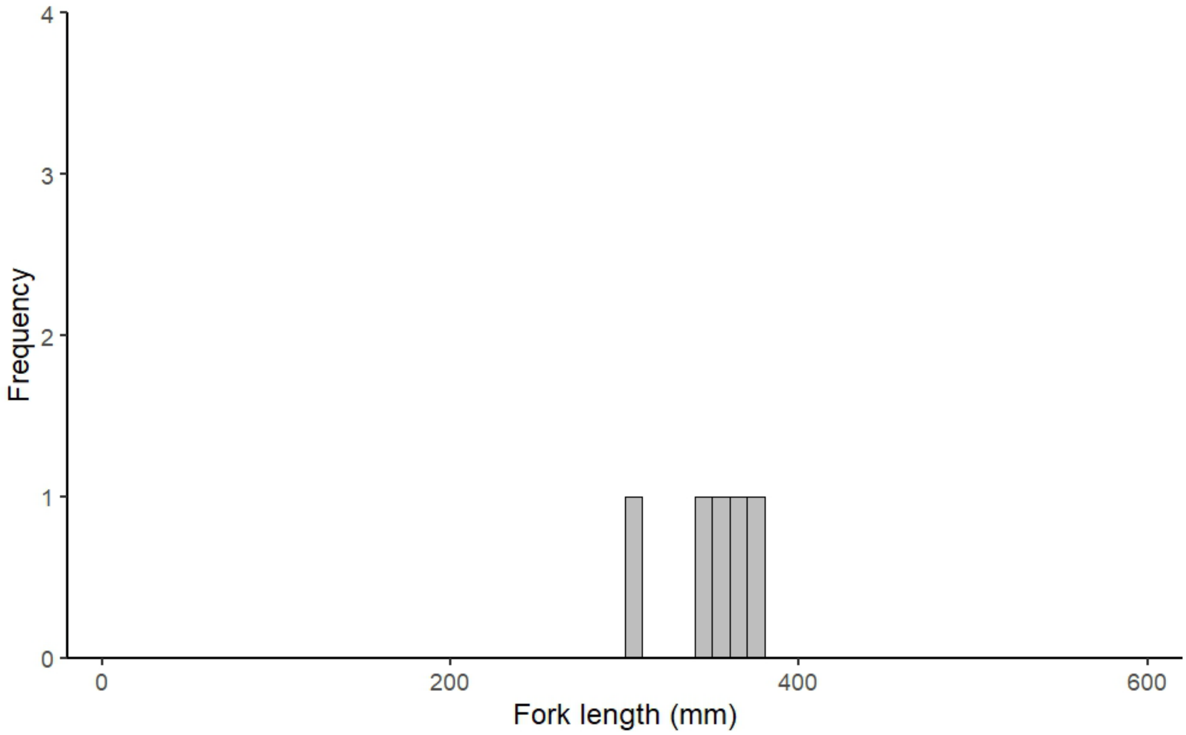

Figure 23. Length frequency of Rainbow trout (n = 5) captured from the ECR in autumn 2025 using gill nets.

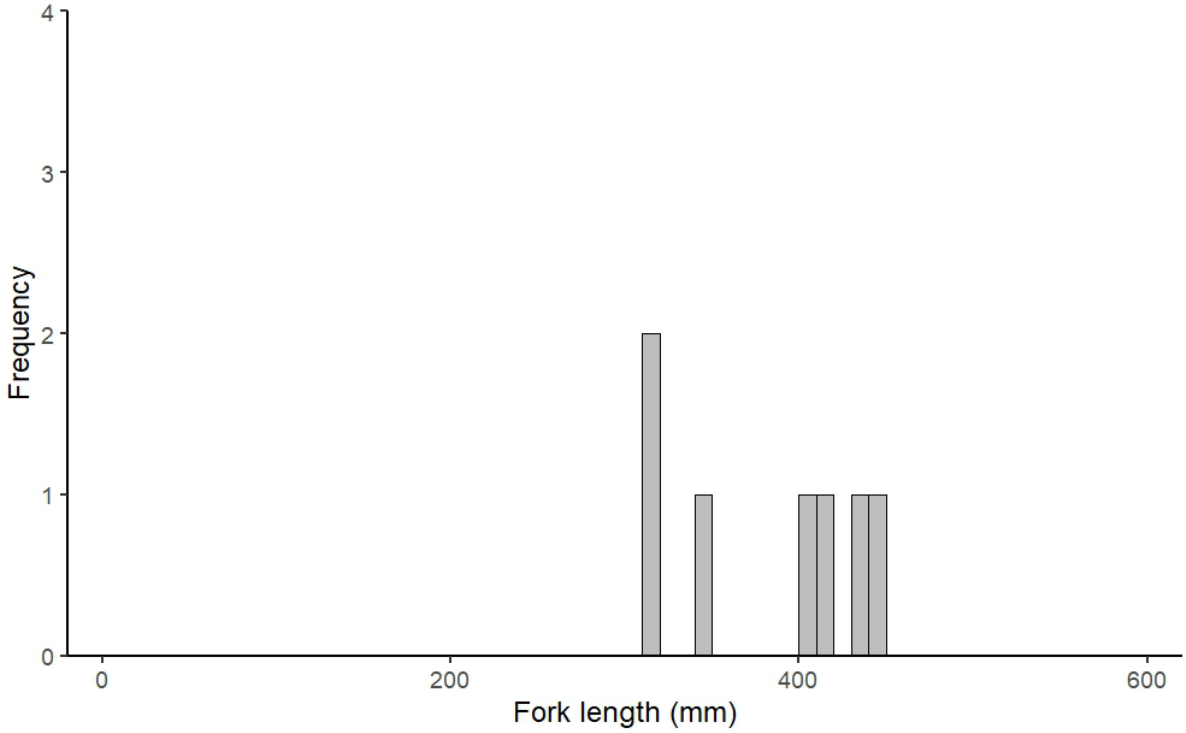

Figure 24. Length frequency of Rainbow trout (n = 6) captured from Bendora Reservoir in autumn 2025 using gill nets.

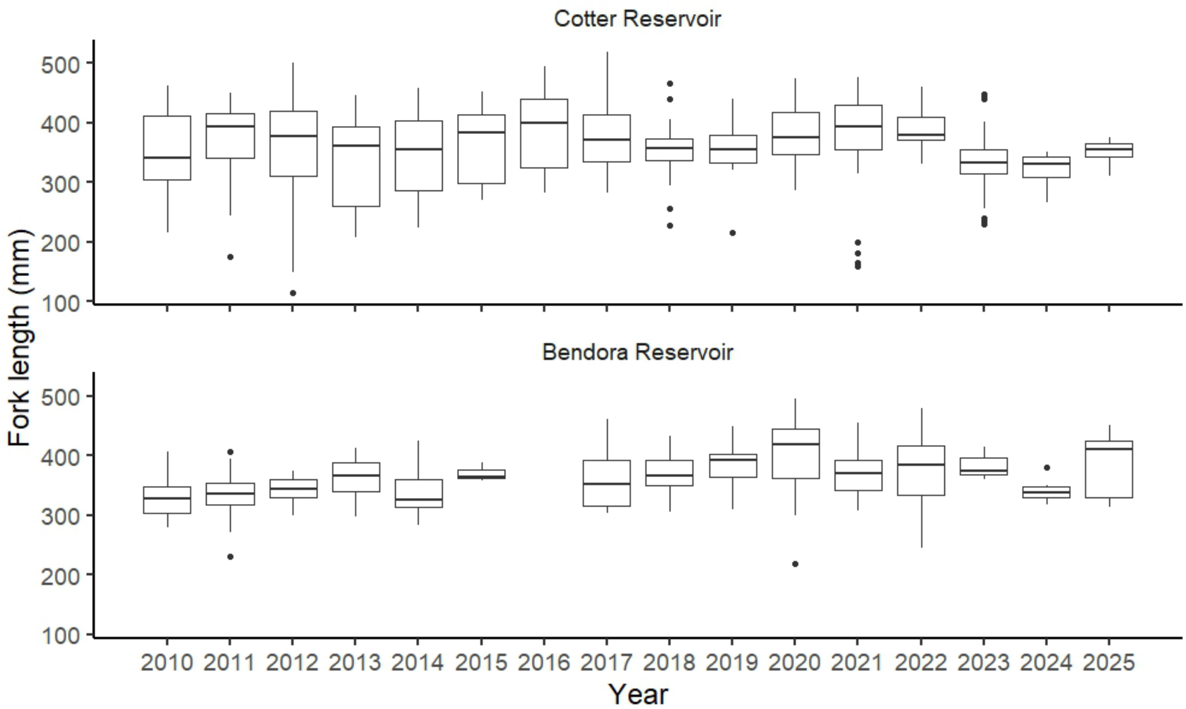

Figure 25. Boxplots of adult Rainbow trout length captured in gill nets each year from Cotter and Bendora Reservoirs from 2010 to 2025 (solid line = median, box represents 25 – 75th percentiles, bars represent minimum and maximum lengths and black circles represent outliers).

DISCUSSION AND CONCLUSIONS

Relative abundance of Rainbow trout in Cotter Reservoir in 2025 was the equal lowest since monitoring began in 2010. As discussed earlier for adult Macquarie perch, water temperature may have been a likely contributor to the reduced Rainbow trout captures in 2025. Low captures in gill nets during such warm water temperatures aligns with general ecological understanding that Rainbow trout, being cold-water fish, tend to avoid warmer surface waters during summer and instead occupy deeper, cooler depths (< 20 °C) when available (Barwick et al., 2004). In support of this theory, abundance of Rainbow trout captured in subsequent gill netting undertaken later in May when surface water temperature had dropped below 20 °C was 3.6 times higher than that captured during the routine monitoring in 2025. Based on these results, it is recommended that gill netting aiming at capturing Rainbow trout be undertaken once surface temperatures have dropped below 20 °C.

Abundances of Rainbow trout within the Cotter Reservoir have been somewhat variable over the past 16 years, with higher abundances observed during filling and operational years (especially years of 2016, 2017, 2020 and 2021). The abundance of Rainbow trout in Bendora Reservoir was highest in early baseline years, and has been relatively stable since 2018, albeit with a minor decrease since 2023. Capture of Rainbow trout in Bendora Reservoir appears to be generally related to temperature at the time of sampling, where a negative correlation between surface water temperature and number of Rainbow trout captured exists (Broadhurst et al., 2023). Lowest catches seem to occur when surface water temperature is > 17 °C, with the exception of 2022, which recorded greater than median number of Rainbow trout, even though water temperature was ~20 °C (Broadhurst et al., 2023).

The number of Brown trout captured in the ECR in the past four years was considerably lower than the preceding six years (which had recorded high abundances of Brown trout). This result indicates that the Brown trout population in Cotter Reservoir has decreased since its peak in early operation phase. Monitoring over the next few years will resolve whether this trend persists. This is a good result for Macquarie perch, as, anecdotally, Brown trout are considered more piscivorous and potentially more damaging to threatened fish populations than Rainbow trout (NSW Fisheries, 2003).

The increase in size of Rainbow trout during the operational phase compared to baseline, is largely attributed to a disproportionate increase in size at Bendora Reservoir. To date, there was no significant change in the size composition of adult Rainbow trout captured between years in Cotter Reservoir. It was expected that as the reservoir filled, food resources would increase and would lead to increases in size of adult trout (albeit with a time lag of a few years) (Kimmel and Groeger, 1986, Ploskey, 1986, O'Brien, 1990). Current operational conditions have led only a small increase in trout size during operational phase, which is unlikely to have changed the threat of this population to Macquarie perch.

RECOMMENDATIONS

Methods for assessing the population metrics of adult trout relative abundance, distribution and size appear to be adequate, on the proviso that monitoring is undertaken when surface temperatures have dropped below 20 °C. No other changes to monitoring are recommended.

No management response to the Rainbow trout in Cotter Reservoir is recommended at this time. However, it is still considered a risk that trout size and abundance may increase over time, and modelling has shown that trout predation can have significant impacts on blackfish in the Cotter River (Todd et al., 2017). These potential impacts indicate that there is still value in investigating potential trout control mechanisms so that management action could be deployed should an increased abundance or size be detected in subsequent monitoring (Lintermans, 2012, ACTEW Corporation, 2013). The reduction in abundance of Brown trout over the past four years is likely positive for Macquarie perch (reduce predation and competition). No management of this species is currently warranted, but a more rigorous investigation of the diet of Brown trout should be considered should abundance rebound from the low captures of the past three years.

QUESTION 5: Has there been a significant change in the abundance and size composition of trout in the Cotter River upstream of the enlarged Cotter Reservoir as a result of filling and operation?

BACKGROUND

If trout populations within the ECR increase as a result of expanded habitat availability and quality, and increased access to thermal refugia, it is probable that there will be an increase in trout abundance in the river upstream of the ECR driven by two factors: (i) density-dependent competitive exclusion of individuals (particularly smaller individuals) from the reservoir population and (ii) adult trout entering the river to spawn in flowing waters (Lintermans, 2012). Monitoring of changes in trout abundance and size distribution in the river will provide insight into potential increases in predatory or competitive interactions with Macquarie perch and Two-spined blackfish.

METHODS

The sampling design for Question 5 is similar to that of the baseline monitoring program Question 6 (Lintermans et al., 2013), with a few changes (Table 15). Sampling for this question is covered by sampling conducted for Question 2. As previously discussed, the site immediately above the old Cotter Reservoir (Bracks Hole) has been inundated and no longer represents a riverine site. Consequently, a replacement site (U/S ECR) approximately 1000 – 1500 m downstream of Vanitys Crossing has been substituted as the most downstream pool site. The site immediately downstream of Bendora Dam is no longer monitored as this site is unlikely to be directly affected by the operation of ECR.

Table 15. Outline of the sampling design for Question 5 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Rainbow and Brown trout, all size classes.
Sampling technique/s	Fyke nets (12 per night; 3 nets per pool at four pools for 1 night); Backpack electro-fishing (4 x 30 m sections and additional effort of up to 20 individuals or 1 km of stream).
Timing	Conducted annually in late summer / early autumn.
Number / location of sites	5 sites on the Cotter River between ECR full supply level and Burkes Creek Crossing (see Figure 1) and one reference site (Cotter Hut – upper Cotter River).
Information to be collected	Number, fork length (mm) for all trout species.
Data analysis	Catch-per-unit-effort (CPUE) assessed between monitoring phases and years using GLMM, graphical representations of the means (with 95% confidence limits with Bonferroni corrections).

Fyke netting (12 mm stretch mesh, single-winged) and backpack electrofishing methods similar to that employed in the baseline monitoring program were employed to monitor riverine sites for trout species. Twelve fyke nets were set overnight (~16-hour soak time) in four pools per site (3 nets per pool). Backpack electrofishing (4 x 30 m sections) was conducted in wadeable (i.e. depths less than 0.8 m) sections of each site (runs and riffles) as well as additional effort of either 20 individuals or 1 km of river (additional effort deployed in baseline phase and then since 2017 which is only used for length analysis at this stage; and to inform analysis of trout diet (see Q6)). Sampling targeted adult trout (either Brown or Rainbow) over 150 mm fork length (FL).

Sampling for this question is undertaken annually in late summer / early autumn (so as to be comparable with sampling undertaken in the baseline monitoring program). Five sites are usually monitored along the Cotter River between full supply level of ECR and Burkes Creek Crossing (see Figure 1) and one reference site in the upper Cotter (Cotter Hut). Monitoring sites on the Cotter River between ECR and Bendora Dam are (from downstream to upstream) U/S ECR (approximately 1000 - 1500 m downstream of Vanitys Crossing), Vanitys Crossing, Spur Hole, Pipeline Crossing and Burkes Creek Crossing. Cotter River sites between Bendora Reservoir and Cotter Reservoir were not able to be sampled in 2022 due to persistently high flows throughout the monitoring period. In 2020 and 2023 the Cotter Hut reference site could not be monitored as a result of the cessation of university fieldwork due to covid 19 risks and the lack of access from bushfire and road damage impacts.

Brown trout are only rarely captured in the river monitoring, so analysis of their abundance and size distribution was not conducted. Abundance of Rainbow trout was standardised for each technique as fish caught per net hour for fyke netting and fish caught per electrofishing shot on-time for electrofishing (represented as CPUE). To assess changes in CPUE over time and between site types (control vs impact), we fitted a zero-inflated negative binomial generalized linear mixed model (GLMM) using the glmmTMB package in R. The model included monitoring phase and site type as fixed effects and year as a random factor to account for repeated measures across time, and had a zero-inflation component to accommodate the high frequency of zero CPUE observations. The model used a log link function appropriate for count data. Model fit was evaluated using Akaike Information Criterion (AIC), and residual diagnostics were assessed using the DHARMa package. Statistical significance was evaluated at $\alpha = 0.05$. All analyses were conducted in R (version [insert version], R Core Team), and data visualizations and summaries were generated using ggplot2 and dplyr. Graphical presentations of site-level mean CPUE for each year (with 95% confidence limits with Bonferroni corrections) were used to explore pairwise variations in Rainbow trout among sites and years.

RESULTS

A total of 219 Rainbow trout were captured from six riverine sites on the Cotter River using fyke nets and backpack electrofishing in 2025 (Figure 26). Rainbow trout captured in 2025 from the Cotter River ranged in size from 65 – 260 mm FL (Figure 26). Rainbow trout were captured at every riverine site and numbers were relatively consistent between impact sites, ranging from 12 – 30 per site. 120 Rainbow trout were captured at the control site, four times as many trout as the highest impact site. The size composition of Rainbow trout in the Cotter River upstream of the ECR has been somewhat variable at the site level, though variability of medians is usually constrained to below 300 mm FL (Figure 26 and Figure 27). Lengths of captured trout in 2024 were bi-modal, with peaks around 100 mm and 250 mm FL (Figure 26 and Figure 27). The presence of Rainbow trout at each site is patchy over years, though the likelihood of detecting this species at a site generally increased with distance upstream of the ECR (Figure 28 and Figure 29). The analysis revealed a significant interaction between phase and site type, indicating a differential response between control and impact sites over time (Table 16). CPUE increased significantly during the operational phase at the control site, though this increase was significantly reduced at impact sites (Table 16). Although CPUE also appeared to increase during the filling phase at the control site, the interaction with site type was not statistically significant (Table 16). Impact sites exhibited higher CPUE than the control site during the baseline phase. These results suggest that while CPUE generally increased over time at the control site, the same trend was not observed at impact sites, particularly during the operational phase. Of the test sites, Pipeline Road Crossing and Burkes Creek Crossing had the most consistent frequency of detection of this species for both fyke netting and backpack electrofishing (Figure 28 and Figure 29) with Bracks Hole / U/S ECR recording lower relative abundances of Rainbow trout compared to Spur Hole, Pipeline Road Crossing, Burkes Creek Crossing and Cotter Hut (Figure 28 and Figure 29). Vanitys Crossing also had a lower relative abundance of Rainbow trout compared to Burkes Creek Crossing (Figure 28 and Figure 29). In general, backpack electrofishing was more likely to detect the presence of Rainbow trout than fyke netting (Figure 28 and Figure 29). Brown trout have been a rare capture in the standardised sampling with most caught towards the upstream end of the study reach (i.e. closer to Bendora Dam). There were four Brown trout captured in 2025 (all via backpack electrofishing); two each from Pipeline Road crossing (228 and 550 mm FL) and Spur Hole (231 and 248 mm FL).

Table 16. Estimated effects of reservoir management phase and site on catch per unit effort (CPUE) of Rainbow trout captured via backpack electrofishing in the Cotter River between 2010 and 2025. Estimates are derived from a zero-inflated negative binomial GLMM and are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	0.02523	0.93825	0.027	0.9785
Filling	2.29132	1.33332	1.719	0.0857
Operational	3.50723	1.00371	3.494	0.0005
Impact sites	2.04382	0.89569	2.282	0.0225
Filling * Impact sites	-1.30737	1.27199	-1.028	0.3040
Operational * Impacts	-2.25108	0.94929	-2.371	0.0177

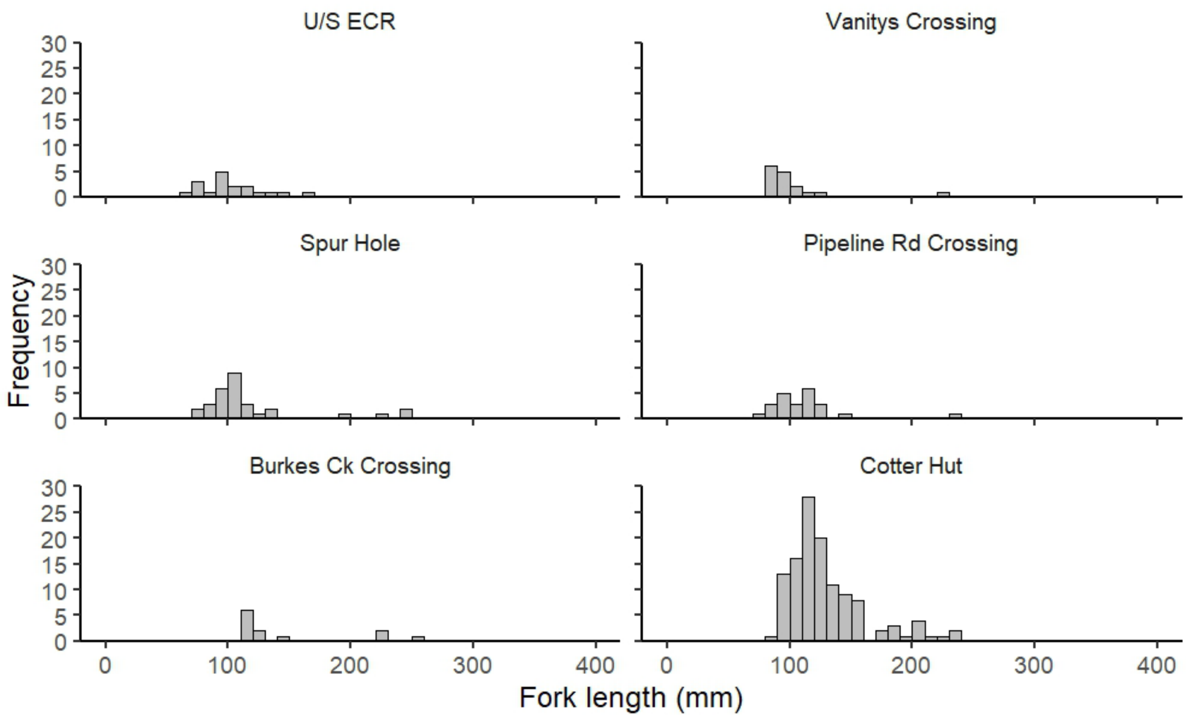

Figure 26. Length frequency of Rainbow trout captured from Cotter River sites in 2025 using fyke nets and backpack electrofishing (inclusive of additional backpack electrofishing effort).

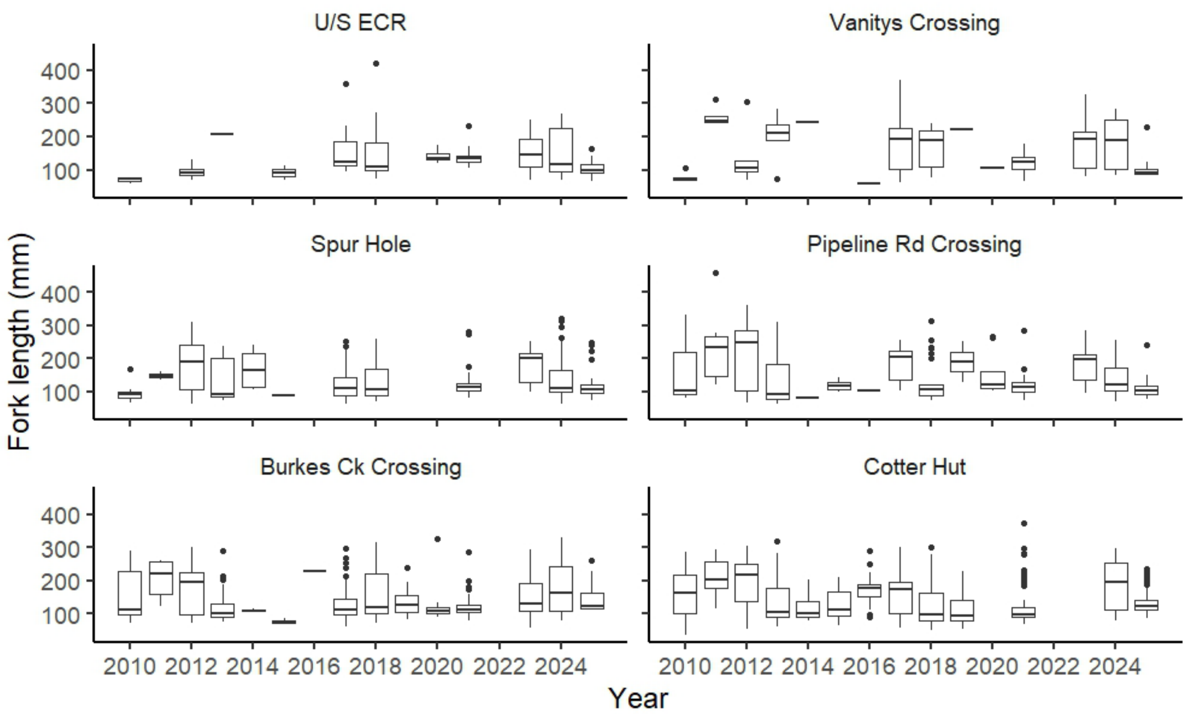

Figure 27. Boxplots of Rainbow trout captured in Cotter River from 2010 to 2025 (solid line = median, box represents 25 – 75th percentiles, bars represent minimum and maximum lengths and black circles represent outliers). Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2025). Note that the increased effort employed from 2017 onwards was used for this figure. Cotter Hut was not able to be sampled in 2020 and 2023 because of fire and COVID-19 restrictions. Sites between Cotter Reservoir and Bendora Reservoir were not able to be sampled in 2022 due to persistent high river flows.

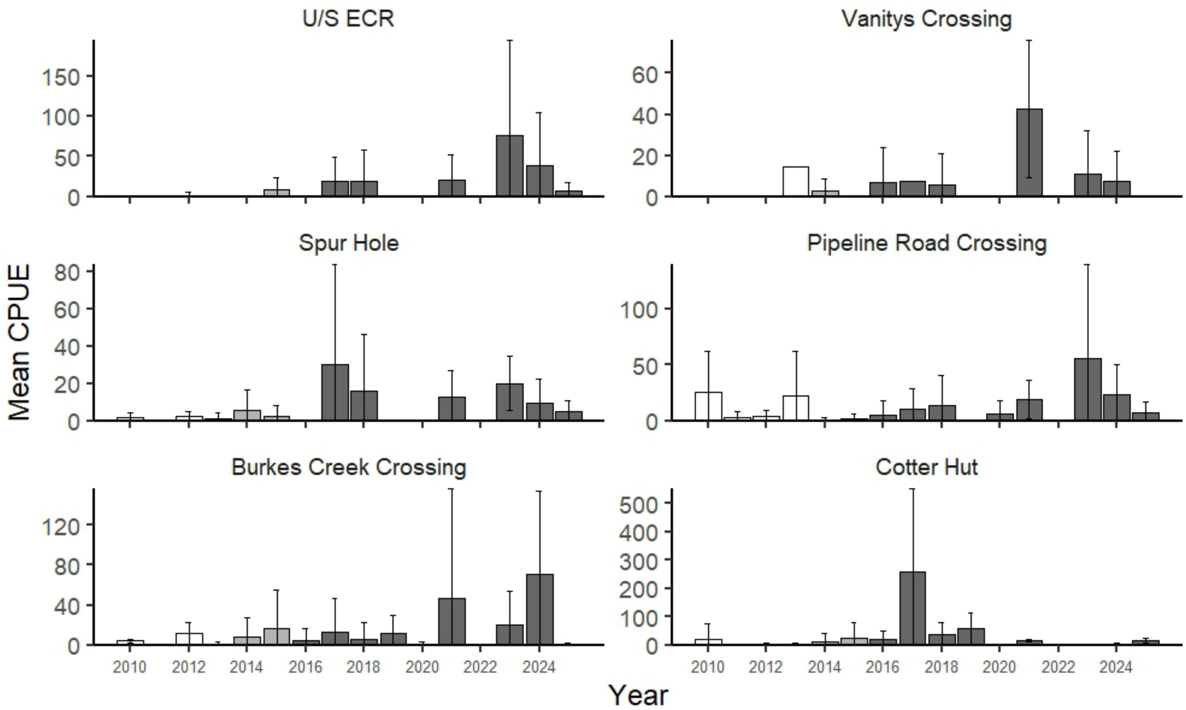

Figure 28. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Rainbow trout captured in Cotter River by backpack electrofishing (4 x 30 m shots) between 2010 and 2025. (Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2025). Cotter Hut was not able to be sampled in 2020 and 2023 because of fire and COVID-19 restrictions. Sites between Cotter Reservoir and Bendora Reservoir were not able to be sampled in 2022 due to persistent high river flows. White bars indicate baseline phase, grey bars indicate filling phase and white bars with diagonal stripes indicates operational phase of monitoring program.

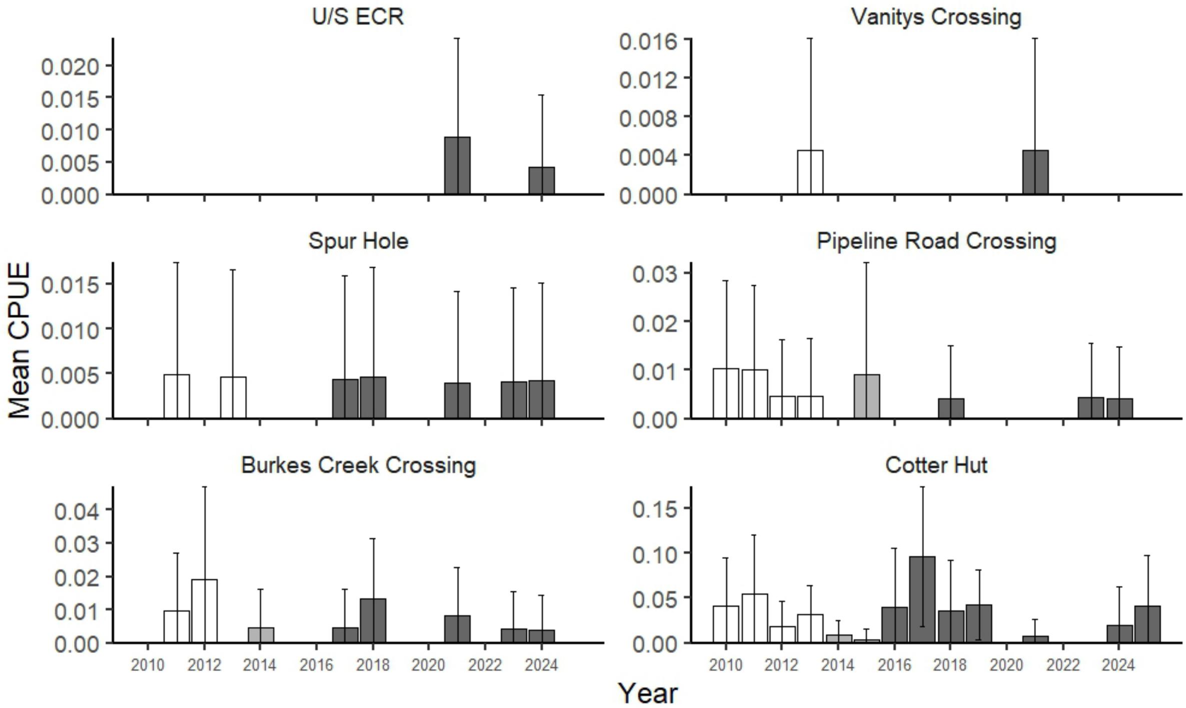

Figure 29. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Rainbow trout captured in Cotter River by fyke net between 2010 and 2025. (Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2025). Cotter Hut was not able to be sampled in 2020 and 2023 because of fire and COVID-19 restrictions. Sites between Cotter Reservoir and Bendora Reservoir were not able to be sampled in 2022 due to persistent high river flows.

DISCUSSION AND CONCLUSIONS

The increased electrofishing effort employed from 2017 onwards captured 2 – 6-fold the number of trout captured on average than the standard 4 x 30 m method. This no doubt provides a more representative sample of the length of trout in the Cotter River at each site and at all sites combined.

Based on data using the basic electrofishing methods (just using the 4 x 30 m sections), there has been no statistically significant change in Rainbow trout abundance in the Cotter River upstream of the ECR based on phase. Similarly, Brown trout abundance in the Cotter River remains extremely low. The low numbers of trout recorded per site using the basic sampling effort and the extremely high variability makes it difficult to detect statistically significant change in trout length. The increased sampling effort employed from 2017 – 2025 helps to reduce fish length variability (by increasing the sample size) and so will allow an increased chance of detecting significant trends in trout length in future years.

The high abundances of Rainbow trout in 2025 at the control site (Cotter Hut) appears to be attributable to a very successful spawning event and subsequent survival over winter / spring 2024. This successful recruitment event was driven by catchment scale effects (likely climate driven), and not as a result of the operation of the ECR. Initial impacts of increased trout abundances will likely be non-lethal (competition for resources – food and refuge), though may translate into increased predation of juvenile Two-spined blackfish and Macquarie perch as trout increase in size.

Monitoring results so far indicate that backpack electrofishing is more effective at obtaining relative abundances or documenting presence / absence of Rainbow trout in the river than fyke netting. The combination of techniques still provides greater confidence that if trout are present at a site, they will be detected. The fyke netting adds significant information on the abundance of Two-spined blackfish at each site, which is important when considering trout predation levels on this species (see Question 6).

Only 20 Brown trout have been collected in the standardised monitoring of the river over the 16 years of monitoring to date. This indicates that Brown trout are still in relatively low abundance when compared to Rainbow trout in the Cotter River. The continued presence of large Brown trout in the river, coupled with the increased abundance of this species in the ECR (see Question 4) increases the likelihood of future population expansion of this species. The threat of Brown trout to native species in the river at this stage is low because of their low abundance (but see concerns for reservoir in Question 4 recommendations above). Monitoring should continue to report on Brown trout numbers as the ECR water level fluctuates to determine if this threat changes.

RECOMMENDATIONS

The increase in raw numbers of trout captured with the revised method (increased electrofishing effort) over the past four years of implementation provides a more accurate representation of trout size in the Cotter River. No change to the current revised methods is recommended.

If detection of seasonal increases in riverine trout abundance is deemed desirable (e.g. spawning runs), then additional sampling is required during such periods (Late autumn for Brown trout; winter for Rainbow trout).

No change has been detected in the riverine Rainbow and Brown trout populations and / or size of individuals since filling has commenced. Distribution of trout in the river also remains similar since filling commenced, as expected. However, it is likely that trout abundance will increase over time (i.e. return to pre-monitoring levels) and so potential trout control mechanisms should be investigated and/or constructed so that they can be implemented rapidly should an increased abundance or size be detected in subsequent monitoring (Lintermans, 2012, ACTEW Corporation, 2013).

QUESTION 6: Are Two-spined blackfish and Macquarie perch present in trout stomachs in the Cotter River?

BACKGROUND

Trout are known to prey on Two-spined blackfish (Lintermans et al., 2013) and are reported to also be predators upon Macquarie perch (Cadwallader, 1978, Broadhurst et al., 2018, Broadhurst et al., 2019)(Lintermans and Kaminskas unpublished data). If the trout population in the ECR increases as a result of expanded habitat availability and increased access to thermal refugia, it is probable that there will be an increase in trout abundance in the river upstream of the ECR. Such increased abundance of trout in the Cotter River upstream of the ECR could also increase predation pressure upon Two-spined blackfish and Macquarie perch (Lintermans, 2012). Monitoring trout diet will allow early detection of changes in the predation of Two-spined blackfish and Macquarie perch.

METHODS

Sampling design for Question 6 is a refinement of that conducted in the baseline monitoring program (Lintermans et al., 2013) (Table 17). Sampling for this question is covered by sampling conducted for Question 2 and 5. This will be conducted in one season only (later summer / early autumn).

Table 17. Outline of the sampling design for Question 6 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Rainbow trout and Brown trout, sub-adults and adults (> 150 mm fork length).
Sampling technique/s	Backpack electro-fishing (4 x 30 m sections and additional effort of up to 20 individuals or 1 km of stream). Field visual processing of dietary items (primarily looking for presence of fish remains).
Timing	Conducted annually in late summer-early autumn
Number / location of sites	Five sites on the Cotter River between ECR full supply level and Burkes Creek Crossing (see Figure 1). One reference site in the upper Cotter (Cotter Hut).
Information to be collected	Number, fork length (mm) for all trout species and visual field identification of fish remains in stomachs.
Data analysis	Comparison of the instances of predation and the size of prey fish between years (baseline vs. impact).

Sampling targets sub-adult and adult Rainbow and Brown trout (> 150 mm fork length). Backpack electrofishing (4 x 30 m sections) was conducted in wadeable (i.e. depths less than 0.8 m) sections of each site (runs and riffles) as well as additional effort of either 20 individuals or 1 km of river. Sampling was undertaken annually in late summer –early autumn. Five sites were sampled along the Cotter River between full supply level of ECR and Burkes Creek Crossing (see Figure 1) and one reference site in the upper Cotter (Cotter Hut). Monitoring sites were (from downstream to upstream) U/S ECR (approximately 1000 – 1500 m downstream of Vanitys Crossing), Vanitys Crossing, Spur Hole, Pipeline Road Crossing and Burkes Creek Crossing. Fork length (FL) in mm was recorded for all captured trout. Stomach contents of trout > 150 mm FL were examined for remains of Two-spined blackfish or Macquarie perch. Non-target species captured during sampling were released at the site of capture unharmed.

RESULTS

A total of 30 Rainbow trout and four Brown trout 150 mm FL or greater were captured in 2025 (Table 18). Visual examination of stomach contents detected no Two-spined blackfish or Macquarie perch in any of the fish examined (Table 18).

Table 18. Details of trout captured and examined for fish and fish remains from the Cotter River in 2025 via backpack electrofishing

Site	Species	Fork Length (mm)	Fish remains in stomach
U/S ECR	Rainbow trout (n=1)	162	No
Vanitys Crossing	Rainbow trout (n=1)	230	No
Spur Hole	Rainbow trout (n=4)	198 – 250	No
	Brown trout (n=2)	231 – 248	No
Pipeline Road Crossing	Rainbow trout (n=1)	240	No
	Brown trout (n=2)	228 – 550	No
Burkes Creek Crossing	Rainbow trout (n=3)	223 – 260	No
Cotter Hut	Rainbow trout (20)	150 – 237	No*

The number of trout stomachs examined for evidence of predation on threatened fish increased significantly in 2017 – 2018, 2021, 2023 – 2025 with the addition of the extra sampling effort (20 individuals or 1 km of river) compared to the previous years (2014-2016), however there was a dramatic reduction in abundance of the targeted size range in 2019 - 2020 (Table 19). Numbers of trout examined in the impact reach was very low in 2025 (n = 10 across 5 sites).

Table 19. Comparison of number of trout stomachs examined from 2010 – 2025 (no shading = baseline, light grey = filling, dark grey = operational).

	2010	2011	2012	2014- 2016
No. examined	198	290	222	16
Rainbow Trout	190	288	216	14
Brown trout	8	2	6	2
Size range FL mm	90-513	148-460	150-500	170-290
Predation Rate	**1**	**1**	**0**	**0**

	2017	2018	2019	2020	2021	2023	2024	2025
No. examined	71	75	19	5	45	102	**91**	**34**
Rainbow Trout	68	70	16	4	44	101	**89**	**30**
Brown trout	3	5	3	1	1	1	**2**	**4**
Size range FL mm	150-371	156-445	155-451	175-485	150-274	155-560	**150-510**	**150-550**
Predation Rate	**1.4**	**2.7**	**10.5**	**0**	**0**	**0**	**0**	**0**

2013 samples collected but not analysed as a result of lack of funding.

DISCUSSION AND CONCLUSIONS

Two-spined blackfish and Macquarie perch were absent from any of the trout stomachs examined in the field in 2025. This is despite the increased sampling effort (20 individuals or 1 km of river) has greatly increased the catch rate of trout. Baseline data also suggests that predation rates of Two-spined blackfish were low (<6 found in over 700 trout stomachs examined) in the three years of baseline sampling and predation of Macquarie perch was not detected at all (Lintermans et al., 2013). This is despite the potential of larger adult trout (i.e. those greater than 350 mm FL) to theoretically predate upon Macquarie perch up to 180 mm total length (Ebner et al., 2007). In 2019, detected predation rates of threatened species by trout had increased to 10.5 % (2 from 19) of trout containing Two-spined blackfish. In contrast to this there was no evidence of predation from 2020 onwards.

Visual inspection of stomach contents is highly unlikely to be able to detect the presence of Macquarie perch larvae, and larvae are not present during the period (late summer/early autumn) that samples are collected. Consequently, no conclusions can be drawn about predation on Macquarie perch larvae from the current sampling. The first stage of development of a genetic test to detect Macquarie perch DNA in trout stomachs has been completed (MacDonald et al., 2014), but has not been progressed due to lack of funding. Further development of this technique requires laboratory feeding trials to confirm the validity of the test on partially digested material, and to establish the sensitivity of the test. Applying a refined test in the field would require sampling in late spring or early summer when Macquarie perch larvae are present.

RECOMMENDATIONS

The increased sampling of 1 km of river or 20 trout per site generally increases the likelihood of robustly assessing the incidence of trout predation on threatened fish. No further change to the monitoring methods for this question required at this time.

Visually detectable predation rates of post-larval native species by trout remain absent to low, but there is little confidence in this result for larval Macquarie perch (see Ebner et al., 2007). However, it is likely that trout abundance and species mix (i.e. increased relative abundance of Brown trout) will increase over time with concomitant increases in potential predation pressure on threatened riverine fish. Consequently potential trout control mechanisms should be investigated and/or constructed so that they can be operated rapidly should an increased predation rate of threatened fish by trout be detected in subsequent monitoring (ACTEW Corporation, 2013, Lintermans, 2012). Continued development of the genetic test would greatly enhance confidence in whether or not trout prey on larval Macquarie perch and could potentially be funded through a Masters scholarship.

QUESTION 7: Has there been a significant change in the abundance and distribution of non-native fish species in the enlarged Cotter Reservoir as a result of filling and operation?

BACKGROUND

The dynamics of the trout population in the ECR is addressed by Question 4. The other non-native species present in Cotter Reservoir are Goldfish *Carassius auratus*, Oriental weatherloach *Misgurnus anguillicaudatus* and Eastern gambusia *Gambusia holbrooki*, all of which have noted preferences for still-water or slow-flowing habitats (Lintermans, 2002, Lintermans, 2007). The enlargement of the reservoir provided a significant increase in habitat for these species as well as the trophic upsurge, and consequent increases in abundance were observed in the first few years since filling commenced. These species could competitively interact with Macquarie perch for resources (particularly food and shelter) but are not considered a predatory threat. Increased Gambusia abundance could lead to increased aggressive interactions between this and native fish species (Lintermans, 2007). Also, expansion of populations of Goldfish and Oriental weatherloach could facilitate the expansion of trout and cormorant populations, which are a potential predation threat to threatened fish populations. Both Goldfish and Oriental weatherloach have been recorded in trout diet from the reservoir, with Goldfish being particularly important (Ebner et al., 2007). Cormorant diet in the Cotter Reservoir has also been shown to contain significant numbers of Goldfish (Lintermans et al., 2011). Monitoring changes in status of non-native fish in the reservoir, along with monitoring of trout predation in the river (Question 6) will provide insights into the dynamics of the fish community in the reservoir. This monitoring will also facilitate early detection for non-native fish species not currently in the Cotter catchment upstream of Cotter Dam (i.e. Carp & Redfin perch).

METHODS

Sampling design for Question 7 is covered by sampling outline for Question 1 (fyke netting) and is similar to the baseline monitoring program (Lintermans et al., 2013) (Table 20). The changes from the baseline monitoring program are the removal of an urban lake reference site (where these non-native species are present/abundant). This is the seventh year of monitoring following the commencement of filling and the fifth year of monitoring since the ECR filled.

Table 20. Outline of the sampling design for Question 7 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Non-native species (other than trout); all sizes.
Sampling technique/s	Fyke nets (20 per night for 3 nights) and bait traps (10 traps for one night).
Timing	Conducted annually in late summer / early autumn.
Number / location of sites	1 site; ECR.
Information to be collected	Number and total length or fork length (mm) for all species.
Data analysis	Comparison of catch-per-unit-effort (CPUE) of Goldfish via a zero-inflated negative binomial generalized linear mixed model (ZINB GLMM). Graphical representations of the means are provided (with 95% confidence limits with Bonferroni corrections).

Sampling targeted all non-native fish species and life stages (other than trout). Fyke netting was used to monitor Oriental weatherloach and Goldfish. Specifically, 20 fyke nets were set around the entire ECR over three nights. 10 Bait traps were set for one night around the perimeter of the reservoir. Sampling for this question was undertaken in early autumn. Total length (TL) and/or fork length (FL) in mm to be recorded for all captured individuals.

Abundance of Goldfish was standardised for each technique as fish caught per hour (represented as catch per unit effort or CPUE). CPUE was scaled in relation to increases in shoreline length as the reservoir filled and as net effort (see question 1 for scaling equation). To evaluate whether Goldfish catch-per-unit-effort (CPUE) in Cotter Reservoir varied across different management phases, we employed a zero-inflated negative binomial generalized linear mixed model (ZINB GLMM). This model was selected due to the high proportion of zero CPUE values (52.4%) and evidence of overdispersion in the data. The fixed effect in the model was phase (baseline, filling, operational), while year was included as a random effect to account for repeated annual sampling from 2010 to 2025. Model comparison using Akaike Information Criterion (AIC) indicated that the ZINB model provided a better fit than a zero-inflated Poisson alternative, suggesting it more appropriately captured the structure of the data. Graphical presentations of site-level means with 95% confidence limits (with Bonferroni corrections applied) were then used to explore pairwise variations in Macquarie perch size classes among sites and years.

RESULTS

Goldfish

Over three nights of fyke netting in Cotter Reservoir in 2025, 93 Goldfish were captured ranging in length between 35 – 200 mm FL (Figure 30). The vast majority of these individuals were less than 100 mm FL, most likely corresponding to 0+ and 1+ year-old age class (Merrick and Schmida, 1984)(Figure 30). Goldfish relative abundance was significantly different among monitoring phases, with filling phase having significantly higher abundance of Goldfish compare to baseline (Figure 31). Relative abundance of Goldfish from 2017 – 2025 was not significantly different from any of the baseline monitoring years.

Table 21. Estimated effects of reservoir management phase catch per unit effort (CPUE) of Goldfish captured using fyke nets in the Cotter Reservoir between 2010 and 2025. Estimates are derived from a zero-inflated negative binomial GLMM and are presented with standard errors (SE), z-values, and associated p-values.

Term	Estimate	Std. error	Z value	Pr(>|z|)
(Intercept)	-2.8256	0.6233	-4.533	<0.001
Filling	3.6162	0.8903	4.062	<0.001
Operational	0.4873	0.694	0.702	0.483

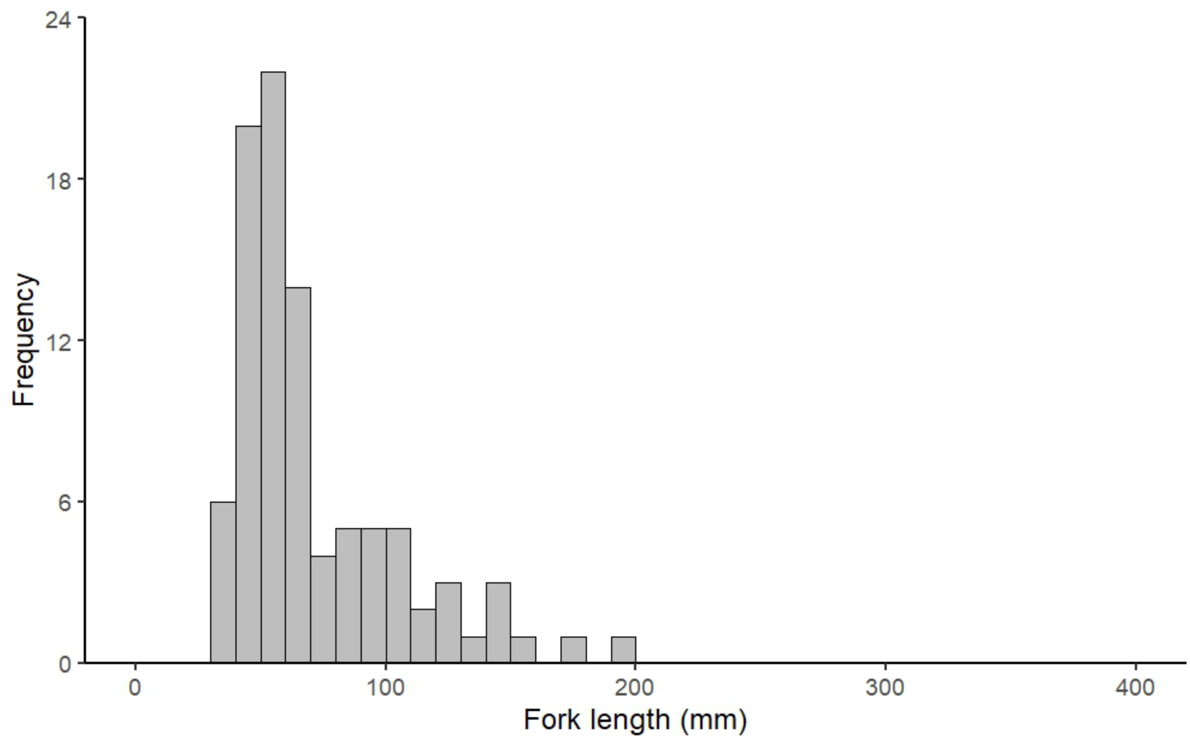

Figure 30. Length Frequency of Goldfish captured in the ECR in 2025 over three nights of fyke netting.

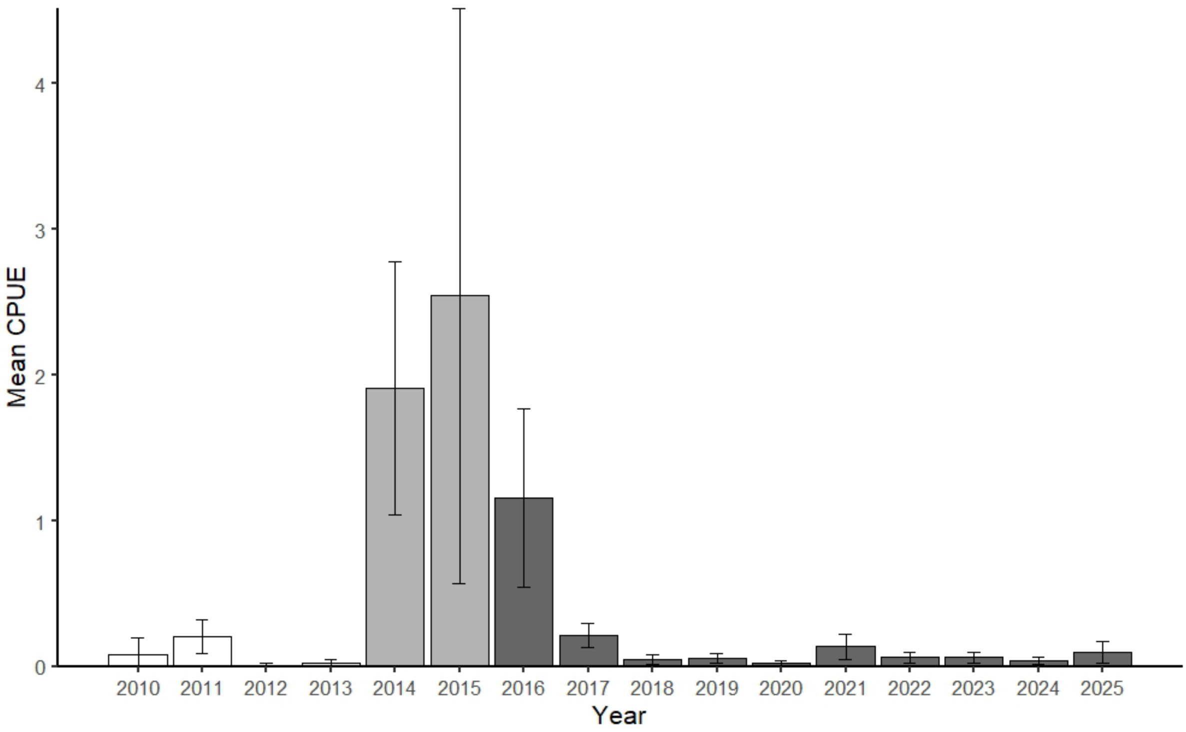

Figure 31. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of Goldfish captured in the ECR using fyke nets between 2010 and 2025. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey indicates operational phase of monitoring program.

Oriental weatherloach and Eastern gambusia

Both Oriental weatherloach and Eastern gambusia are rare captures in Cotter Reservoir in the monitoring undertaken to date. A total of seven Oriental weatherloach have been captured so far in fifteen years of monitoring in Cotter Reservoir (one each in 2010 and 2011; three in 2012 and two in 2017). In 2025, 57 Oriental weatherloach were observed while boat electrofishing, this highest number observed since boat electrofishing commenced in 2014. To date, 130 Oriental weatherloach have been observed whilst undertaking the boat electrofishing of Cotter Reservoir. Three Eastern gambusia were captured in 2025, two in bait traps and one in a fyke net. A total of 145 Eastern gambusia were observed whilst boat electrofishing in 2025, taking the overall tally to 1403 Eastern gambusia observed since 2014. Of note, from 2020 – 2024 inclusive, only one Eastern gambusia had been observed in Cotter Reservoir whilst electrofishing.

DISCUSSION AND CONCLUSIONS

Relative abundance of Goldfish has been relatively low since 2017, following on from very high abundances during filling and the first year of operational phase. The higher relative abundance of Goldfish in the initial stages of filling was likely to have been caused by an increase in availability of food resources associated with the filling phase (and refilling in the case of 2021) of the ECR (Kimmel and Groeger, 1986, Ploskey, 1986, O'Brien, 1990). As predicted, the increase in Goldfish observed in 2014 – 2016 was likely a significant factor to the commencement of cormorant breeding observed in the top end of the reservoir in February 2014 and in every year since (see Question 8). Goldfish abundances in operational years of 2017 – 2025 were not significantly different to baseline abundances, indicating that the boom in resources associated with filling and early operational phases has ceased.

The relatively low catch rates again of both Oriental weatherloach and Eastern gambusia is not considered to reflect population sizes of these two species, especially that of Eastern gambusia. The low capture rates are likely to be an artefact of the sampling method (fyke nets and bait traps) and the size or behaviour of (Eastern gambusia) or body-shape (narrow cylindrical, Oriental weatherloach) of these two species. The mesh size of the fyke nets used is too large to reliably capture either species. Eastern gambusia is regularly observed in large schools (sometime in excess of 100 individuals) at the boat ramp and other open shallow habitat (such as where existing roads run into the reservoir) at Cotter Reservoir during monitoring. Eastern gambusia are known to prefer shallow waters (Lintermans, 2007, Pyke, 2005, Macdonald and Tonkin, 2008) which may explain the congregations at these shallow open habitats. Certainly, many of these congregations were observed in 2015 and 2016 during the boat electrofishing surveys, but not as many in 2017 or in 2018. The lack of adequate representation of Oriental weatherloach and Eastern gambusia populations in the monitoring program is not of major concern, as the major target of this research question is Goldfish and their likely role in the expanding predator (trout and cormorant) populations.

RECOMMENDATIONS

Sampling methods appear to be adequate for monitoring abundances of Goldfish species in Cotter Reservoir. No change to monitoring regime for this species recommended at this time.

Both Oriental weatherloach and Eastern gambusia have low catch rates in the ECR monitoring program, and observation of localised schools of Eastern gambusia around shallow exposed areas of the reservoir indicate that this species can be numerous at a small spatial scale. Alternative sampling techniques are available for these two species (e.g. seine netting shallow habitats for Gambusia; backpack electrofishing of soft substrates for Oriental weatherloach), or eDNA for both species, but such sampling will require additional sampling days, for data currently considered to be of little consequence for management of threatened fish species. No management intervention required for these two species.

A significant increase in the abundance of Goldfish was detected in 2014 and 2015 and higher relative abundances persisted into 2016. Monitoring from 2017 – 2024 revealed a decrease in

Goldfish abundances indicating that the resources boom associated with the filling reservoir has ceased or slowed significantly. The decline in Goldfish abundance and the first detections of Macquarie perch predation by trout warrant close attention being paid to this potential shift in food webs, but at this stage, no management intervention specifically related to Goldfish is recommended.

QUESTION 8: Has there been a significant change in the abundance, distribution and species composition of piscivorous birds in the vicinity of the enlarged Cotter Reservoir as a result of filling and operation?

BACKGROUND

Piscivorous birds (predominantly cormorants) have been identified as a potential threat to Macquarie perch in the ECR (Lintermans, 2005). Predation of Macquarie perch by cormorants in Cotter Reservoir has been confirmed (Ebner and Lintermans, 2007, Lintermans et al., 2011, Lintermans, 2012), and a significant expansion of the piscivorous bird population following enlargement of the reservoir could have severe consequences on the small adult population size of Macquarie perch (Farrington et al., 2014). Assessment of population trend in piscivorous birds on Cotter Reservoir is required with monthly monitoring enabling early detection of significant changes in the abundance and distribution of cormorant species. A cormorant management plan has been included in the fish management plan version 4 (Icon Water Limited, 2019).

METHODS

Sampling design followed that exactly outlined in the baseline monitoring program (Lintermans et al., 2013) (Table 22).

Table 22. Outline of the proposed sampling design for Question 8 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Piscivorous bird species (incl. Great cormorants, Little black cormorants and Little pied cormorants, Darters and Pied cormorants).
Sampling technique/s	Visual survey of piscivorous birds per section (longitudinal fifth) of the ECR.
Timing	Monthly, year-round.
Number / location of sites	1 site; ECR.
Information to be collected	Species, abundance, abundance per section.
Data analysis	Comparison of abundance and distribution of each species of cormorants between years (baseline vs. impact) using Multivariate analysis PERMANOVA. Graphical representations of the means are provided (with 95% confidence limits with Bonferroni corrections).

Monthly visual surveys are undertaken of the entire ECR targeting piscivorous bird species including Great cormorant, Little black cormorant, Little pied cormorant, Pied cormorant, and Darter. The presence of nests of piscivorous birds was also noted, and if present the contents (eggs or chicks) noted (though this was not part of the monitoring program or analysis). Visual surveys were conducted from a boat using 10 x 40 mm binoculars. Location of each individual was recorded on a map. To determine distribution of piscivorous birds, the reservoir was divided longitudinally into five equal parts. Abundance and distribution can be assessed against trigger levels in the fish management plan: Appendix G (Icon Water Limited, 2019). Count data was analysed using PERMANOVA (Permutational Multivariate Analysis of Variance) to assess differences in community composition across phases and sections. The data was log-transformed (Log10(x + 1)) to stabilize variances, and a Bray-Curtis dissimilarity matrix was computed. PERMANOVA was performed with phase and section as fixed factors, and year nested within phase to account for temporal variability. Pairwise PERMANOVA comparisons were conducted to identify specific differences between phases and sections. SIMPER (Similarity Percentage) analysis was used to determine which species contributed most to the observed differences.

RESULTS

Great, Little black and Little pied cormorants were the most abundant species of piscivorous birds recorded on the ECR with much lower numbers of Darter and one Pied cormorant recorded in 2023-2025 (Figure 32). There have been only seven observations of Pied cormorant (all of single individuals) since monitoring began in 2010, though none in 2018, 2019, 2021, 2022 and 2023. Abundances of the three most common species were relatively consistent with expectations during the monitoring period with some seasonal fluctuations present (Figure 32). Since filling began, abundances of both Great cormorant and Little black cormorant have been stable, though with some definitive seasonal fluctuations (Figure 32). Abundances of Little pied cormorant during warmer months has been increasing annually since filling began, with these annual influxes concentrating in section 4 and as of 2018 section 2 of the reservoir (Figure 32 and Figure 33).

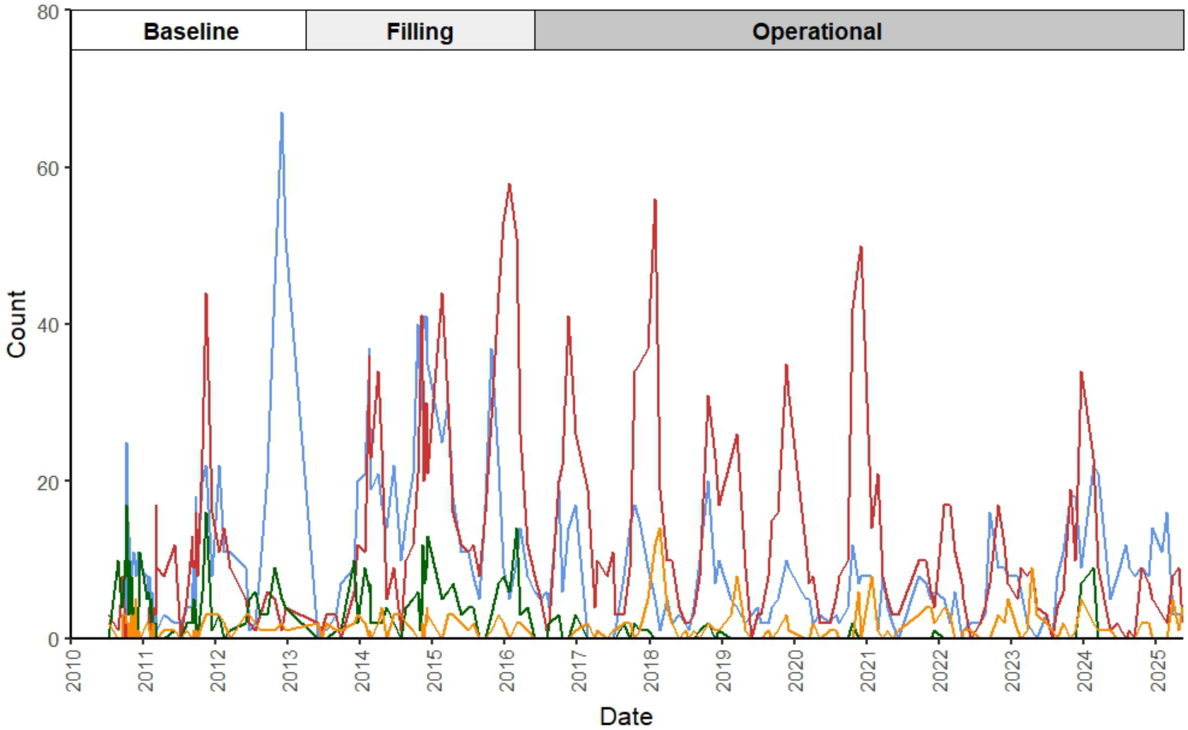

Figure 32. Monthly abundances of each common piscivorous bird species on the Cotter Reservoir between July 2010 and May 2025 (Great cormorant – blue line; Little black cormorant – dark green line; Little pied cormorant – red line; Darter – orange line).

There was considerable overlap in the composition of the piscivorous bird community (composition, abundance and distribution) among pre-filling and filling phases in Cotter Reservoir (Figure 33). PERMANOVA revealed significant differences in bird community composition across all tested factors and the interactions between Phase and Section (Table 23). These results suggest that both spatial and temporal factors significantly influence bird community structure.

Pairwise comparisons indicated that cormorant community composition changed significantly across all phases, with the largest difference observed between the baseline and operational phases. SIMPER analysis revealed that in the comparison of baseline vs filling phases, Little black cormorants were the only species with a statistically significant contribution, with higher average abundance during the baseline phase. The baseline vs operational contrast revealed significant contributions from all three species, with all three more abundant in the baseline phase. In the filling vs operational comparison, although no species showed statistically significant contributions, Great cormorants and Little pied cormorants had slightly higher average abundances during the filling phase. These patterns suggest that Little black cormorants played a prominent role in early community shifts, while broader changes involving all three species became more evident by the operational phase.

Table 23. Results of PERMANOVA analysis of piscivorous bird community composition in Cotter Reservoir from 2010 – 2025.

Term	DF	Sum of sq	R^2	F	Pr(>F)
Phase	2	11.781	0.09246	42.0499	0.001
Section	4	3.87	0.03037	6.9057	0.001
Year	13	10.56	0.08288	5.7986	0.001
Phase * Section	8	5.25	0.0412	4.6845	0.001
Residual	685	95.957	0.75309		
Total	712	127.417	1		

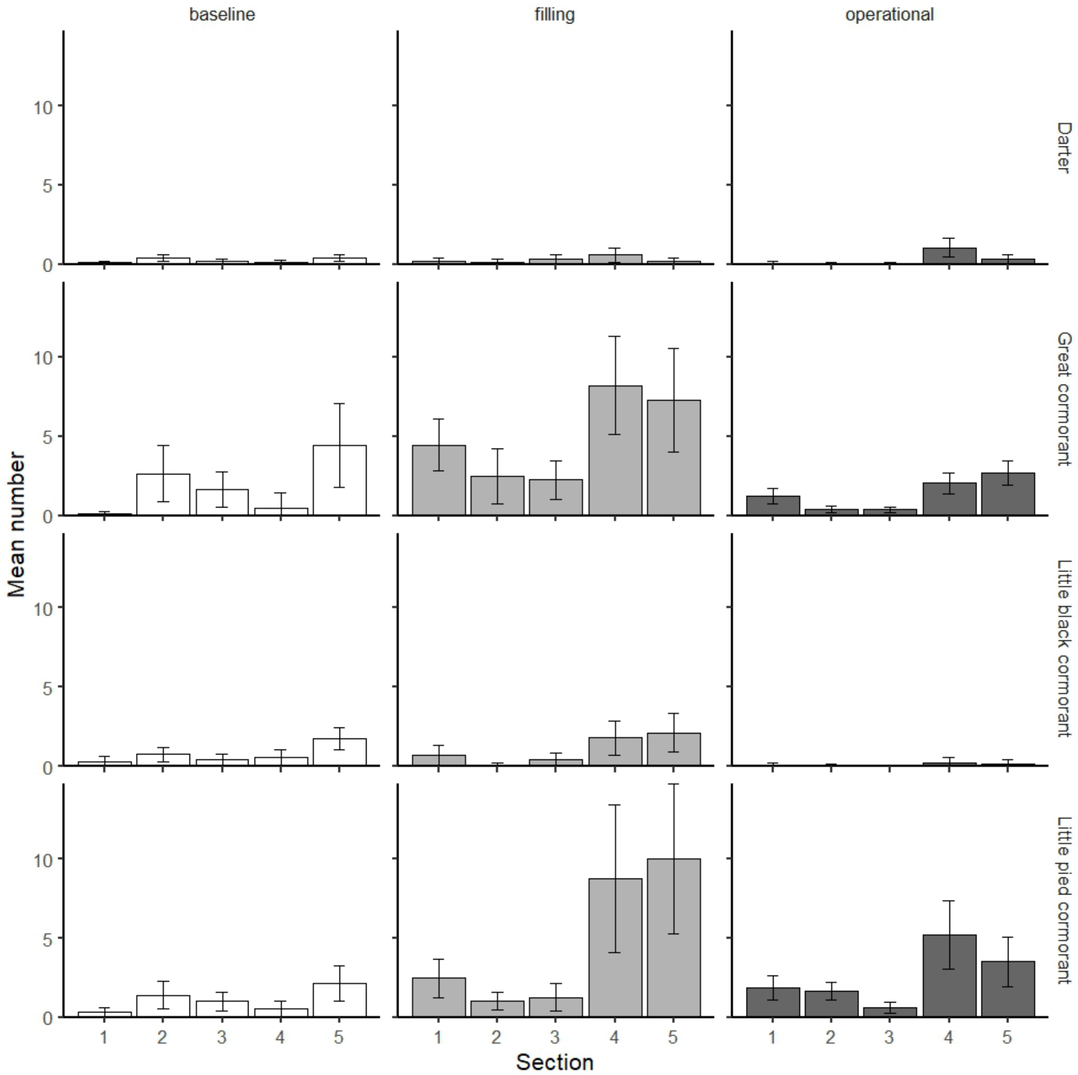

Figure 33. Mean monthly abundance (displayed as mean ± 95% confidence limits with Bonferroni correction) of each piscivorous bird species in each section of Cotter Reservoir for baseline phase (July 2010 – March 2013), filling phase (April 2013 – December 2015) and operational phase (January 2016 - May 2025) of monitoring program.

For the 11th consecutive year cormorants have established a breeding colony on the ECR. Nesting has occurred in the same reservoir section across years (in sections 4 and 5 of the reservoir approximately 200 m downstream of the Pierces Creek junction Figure 34), though a nesting site was found in 2018 in section 1. Many of the nests observed had chicks varying from just hatched to well-developed. It is believed that the bulk of these were Little pied cormorant as the adult birds were observed on the nests.

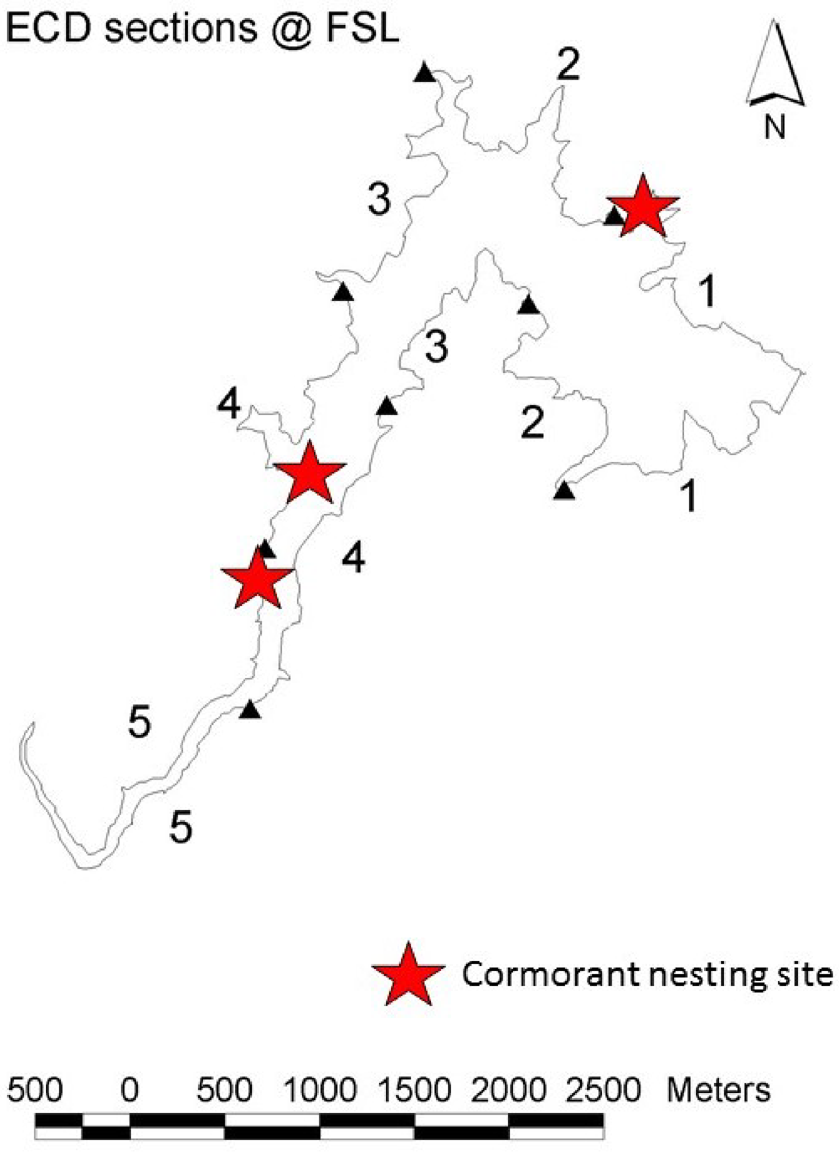

Figure 34. Map of the enlarged Cotter Reservoir shoreline sections with Cormorant nesting colony locations.

DISCUSSION AND CONCLUSIONS

Abundance

Peak abundances (see Figure 32) of Great cormorants occurred during the filling phase, though since the operational phase began have stabilised, though with some definite seasonal fluctuations.

Abundances of Little pied cormorant during warmer months have increased since filling began, with these annual influxes concentrating in section 4 of the reservoir. The seasonal increases in all three common species is most likely attributable to an increase in productivity and food resources (decapods and Goldfish) within the reservoir during the warmer months. During filling and early operational phases the enlarged reservoir has seen an increase in the abundance of Goldfish (see Question 7 above), a favoured prey item of cormorants in the Cotter Reservoir and elsewhere (Lintermans et al., 2011, Miller, 1979). The increase in prey abundance and the abundance of partially inundated larger trees (predominantly Eucalypts and pine trees) has provided suitable conditions for nesting to commence. Indeed, Little pied cormorant has bred in all years since filling began and its seasonal abundance peaks during the warmer months increasing until ~2018, though appear to be subsiding each year since ~2019. The establishment of a breeding colony of cormorants in the ECR is undesirable as the energy requirements of maintaining fledglings as well as adults would incur increased pressure on food resources (i.e. Goldfish and Macquarie perch) by cormorants in Cotter Reservoir (Lintermans et al., 2011). The potential early signs of a shift in predation pressure by trout (i.e. first records of predation of Macquarie perch) in 2018 potentially associated with declines in Goldfish abundance suggest that a re-examination of cormorant diet is required in the near future. This is especially so as the major avian predation threat to Macquarie perch (Great cormorant) were confirmed as breeding for the first time in 2022. If cormorants are also shifting their food preference from Goldfish to Macquarie perch then management of cormorant breeding colonies becomes critical.

Distribution

Distribution of all three common cormorant species has been relatively stable during baseline (2010 – 2013), filling and operational phases with a few exceptions. All three species have been most abundant in the two upstream sections of the reservoir. Previous research has found that cormorants commonly hunt in depths of less than 5 m (Dorfman and Kingsford, 2001, Ropert-Coudert et al., 2006) and this depth range is most prevalent in these two reservoir sections and provides the greatest area for which effective hunting can be conducted (Ryan, 2010, Ryan et al., 2013). The most upstream section is also where the greatest risk of predation is for Macquarie perch (Ryan, 2010, Lintermans et al., 2011, Ryan et al., 2013). Interestingly a change in the distribution has occurred between baseline and filling and operational phases, where section 4 has seen an increase in abundance of both Great cormorant (during filling only) and Little pied cormorant (both filling and operational). This is most likely attributable to the location of a nesting colony in section 4 (Figure 34) and associated roost that was not present in baseline monitoring.

RECOMMENDATIONS

The current monitoring regime appears to be adequate at monitoring abundances and distributions of cormorant species in Cotter Reservoir. No changes to the monitoring regime are recommended at this time.

An increase in cormorant abundance and multiple cormorant nesting events have been detected since filling. The increase in abundance of both Great cormorant and Little pied cormorant triggered management action under the ECR Cormorant Management Plan in 2016. The management trigger thresholds in the Cormorant Management Plan have been revised to reflect the likely normal increase in cormorant abundance with an increasing reservoir surface area and shoreline length.

Given the significant decline in the Goldfish population, and the continued presence of breeding colonies of cormorants (including Great cormorant in 2022), it may be that the increased cormorant abundance and presence of breeding colonies may need to be sustained by another fish species (i.e. Macquarie perch). This highlights that a re-examination of cormorant diet is required in the near future. If cormorants are also shifting their food preference from Goldfish to Macquarie perch then management of cormorant breeding colonies becomes critical.

QUESTION 9: Have macrophyte beds re-established in the enlarged Cotter Reservoir?

BACKGROUND

Existing macrophyte beds in Cotter Reservoir have been demonstrated to provide important daytime resting habitat for adult Macquarie perch (Ebner and Lintermans, 2007). It was known that existing macrophyte beds would be drowned by up to 50 m of water once the reservoir filled. Modelling indicated that the reservoir would remain within 3 m of full supply level for at least 73 percent of the time once the reservoir has filled, potentially allowing new macrophyte beds to establish. Such macrophyte beds could once again provide important cover habitat for threatened fish including Macquarie perch.

DOES COMPARABLE BASELINE DATA EXIST?

Partially. Surveys by Roberts (2006) and Ryan (2010) provide an indication of macrophyte extent and distribution in the current Cotter Reservoir.

SAMPLING DESIGN

Sampling design will be a standard on-ground survey (e.g. Roberts 2006) of the perimeter of the ECR for signs of establishment of macrophytes. It is considered unlikely macrophytes will establish during filling phase and so the project team recommends that surveys for macrophyte establishment do not commence until ECR has reached full supply level.

Table 24. Outline of the proposed sampling design for Question 9 of the fish monitoring program.

Feature	Detail
Target species and life history phase	The survey will target emergent macrophyte species that are likely to provide adult Macquarie perch with cover from cormorant predation (i.e. *Phragmites australis*).
Sampling technique/s	Visual on-ground survey of the perimeter of the ECR by boat.
Timing	Annual in late summer / autumn
Number / location of sites	1 site, the entire ECR.
Information to be collected	Location, extent (length / area covered) of each emergent macrophyte species.
Data analysis	Descriptive statistics length, width and area of each species and stand. Map of size and location of macrophyte stands will be derived.

The survey will target emergent macrophyte species that are likely to provide adult Macquarie perch with cover from cormorant predation (i.e. *Phragmites australis*). Visual surveys will be conducted around the entire perimeter of the ECR by boat. GPS points will be taken around the extent of the macrophyte bed to determine both its location and its extent (by area in square metre). Surveys will be conducted in the ECR from September to February as it is during the warmer months that emergent macrophytes will be growing and flowering. Location, extent (length / area covered) of each emergent macrophyte species will be recorded. When the monitoring program commenced in 2010 it was anticipated that monthly sampling in spring and autumn would occur, but a decade on this sampling frequency may need to be revisited. Descriptive statistics (length, width and area covered) of each emergent macrophyte species will be calculated. In addition, a GIS based map showing locations of each macrophyte stand location will be derived.

RESULTS

Three stands of emergent macrophytes, all of Cumbungi (*Typha domingensis*), were found along the shoreline of the Cotter Reservoir in March and May 2022 (Figure 27). The first, located on the western shore just downstream of Bracks Hole road, measured approximately 2 x 2 m (4m^2) in area (Figure 28). The other two were located next to each other near the old boat ramp road on the eastern shoreline of the reservoir (Figure 27). The largest of the two, measured approximately 9 x 4 m (36 m^2) and formed the densest of the three stands (Table 25, Figure 29). The third comprised two loosely grouped sparse stands of approximately 2 x 2 m (4 m^2) each (Table 25). An additional stand of Cumbingi was detected in May 2023, in the bay just south of Bracks Hole (Figure 27). This is a relatively small and sparsely vegetated stand, measuring approximately 2 x 4 m. All four stands increased in area and appeared thicker in composition, especially those located in the old boat ramp road bay (Table 25 and **Figure 35**).

Figure 35. Photograph of the largest Cumbungi stand located in the old boat ramp bay on the Eastern shoreline of the ECR taken in April 2025 (photo: Ben Broadhurst).

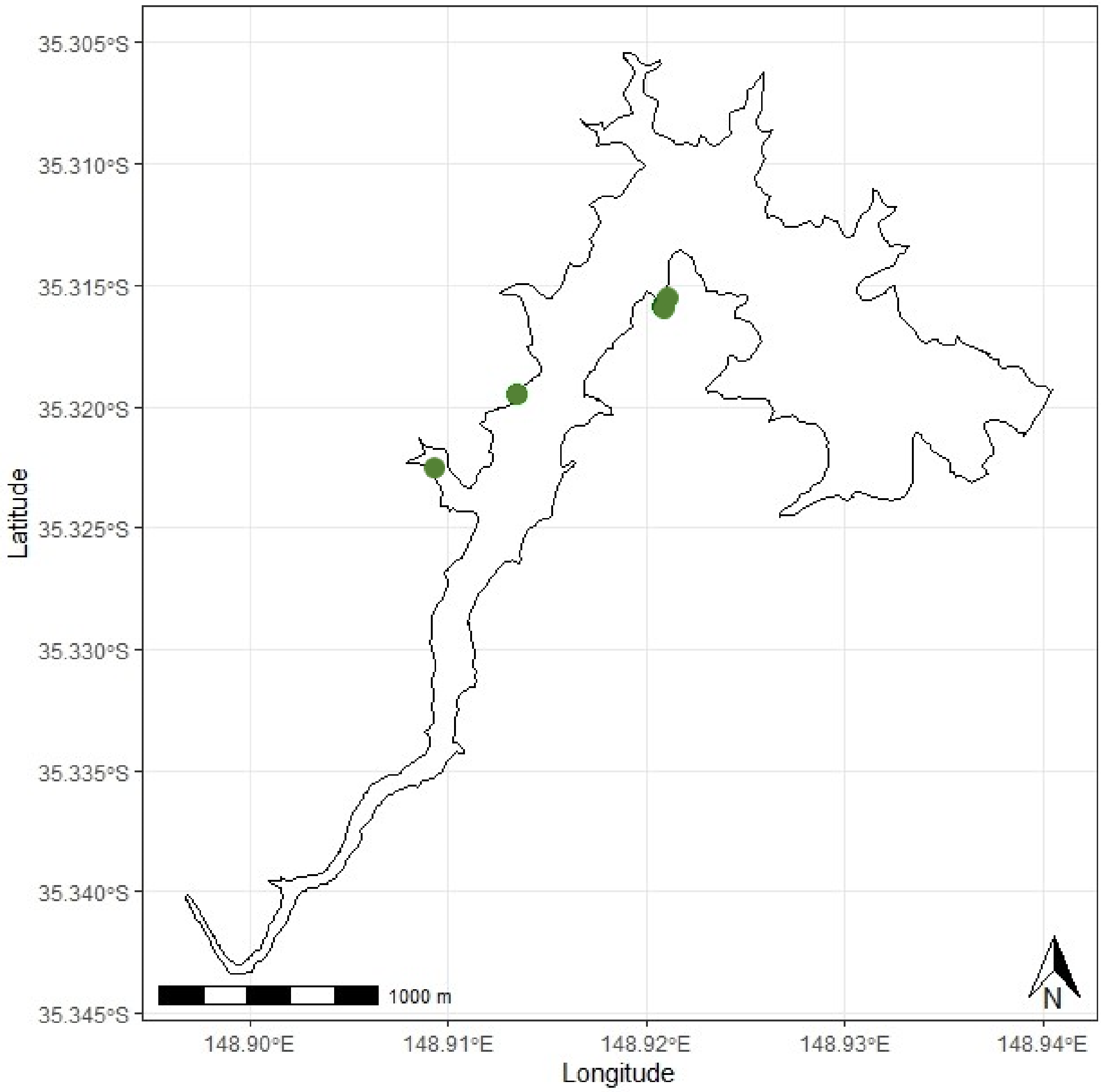

Figure 36. Map of the location of emergent macrophyte stands (green circles) along the shoreline of the Cotter Reservoir as monitored in March - May 2025.

Table 25. Details of each emergent macrophyte stand in the Cotter Reservoir as of May 2025 (including stand size change since last annual estimate).

Species	Location	Approx. size (m^2)	Annual increase
Cumbungi (*Typha domingensis*)	Western shoreline, shoreline near Bracks Hole	8 m^2	2 m^2
Cumbungi (*Typha domingensis*)	Eastern shoreline, inlet near old boat ramp road	54 m^2	4 m^2
Cumbungi (*Typha domingensis*)	Eastern shoreline, inlet near old boat ramp road	25 m^2	5 m^2
Cumbungi (*Typha domingensis*)	Western shoreline, in large bay south of Bracks Hole	12 m^2	2 m^2

DISCUSSION AND CONCLUSIONS

Since filling four emergent macrophyte stands have been established in the ECR (the first was detected in 2022). The four stands were all comprised of Cumbungi, which was a common emergent macrophyte in the Cotter Reservoir prior to enlargement (Roberts 2006). Establishment of emergent macrophytes, Cumbungi to date, in Cotter Reservoir is most likely attributable to very stable water level since September 2020 (Figure 4). Macrophytes provide a number of important services, including providing habitat and food for aquatic animals (e.g. fish, macroinvertebrates and birds) and stabilising sediments (Richardson et al., 2002, Madsen et al., 2001, Fleming et al., 2011). Emergent macrophytes were found to be crucially important refuge habitat for Macquarie perch in Cotter Reservoir prior to enlargement (Ebner and Lintermans 2007; Lintermans 2012). Whilst the current extent of emergent macrophytes in the Cotter Reservoir is very small (in comparison to other available habitats) and currently comprised of a single species it does indicate that stable water levels of the enlarged reservoir are suitable to macrophyte germination. The other formerly abundant emergent macrophyte Common Reed (*Phragmites australis*) has not yet re-established.

RECOMMENDATIONS

We propose that monitoring of emergent macrophytes continue to occur annually, during later summer / early autumn. The reduction in frequency from the original methods and timing of the survey reflects;

a) Emergent macrophyte extent is unlikely to alter detectably on the monthly scale

b) Late summer / early autumn is the time at which the macrophyte stands will be at their maximum extent.

<u>QUESTION 10:</u> Are there adequate food resources (particularly decapods) for the Macquarie perch following the filling and operation of the enlarged Cotter Reservoir?

BACKGROUND

It was expected that as the ECR filled and became operational the food resources of Macquarie perch were likely to change. The substantial beds of emergent macrophytes that fringed the old Cotter Reservoir have been submerged, and the fluctuating water levels of an operational reservoir may prevent their reestablishment (Lintermans, 2012). These reed beds supported significant densities of decapod crustaceans, particularly freshwater prawn (*Macrobrachium*) and shrimp (*Paratya*) which are favoured food items for Macquarie perch (Norris et al., 2012). During and following inundation a trophic upsurge was expected (and occurred) where food resources of Macquarie perch were plentiful and Macquarie perch had increased body condition (as experienced in other newly filled reservoirs such as Lake Dartmouth). As the reservoir ages, it was expected that the food resources of Macquarie perch may diminish and result in poorer body condition and this is likely to result in reduced fecundity and could lead to a negative impact on recruitment to the population.

METHODS

The sampling design follows that outlined in the Food Resources study of Norris *et al.* (2012) (the baseline samples) so comparisons with pre, during and post-filling can be made. All Macquarie perch food resources were targeted, with an emphasis on decapoda.

Table 26. Outline of the sampling design for Question 10 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Food resources of Macquarie perch (primarily decapods).
Sampling technique/s	Edge sampling of each major habitat (3 each of rocky shore, bare shore, woody habitat and macrophyte – where possible) and plankton tows (1 in each longitudinal third of the reservoir).
Timing	Bi-annually in spring and autumn.
Number / location of sites	Conducted in the ECR only.
Information to be collected	Relative abundance and composition of food resources.
Data analysis	Relative abundance of prey items (with particular focus on decapods) was compared between phase (baseline, filling and operational), season and habitat type using three-way PERMANOVA and principal components analysis.

Food resources sampling was undertaken in autumn and spring in the ECR and followed the sampling and processing protocols of Norris *et al.* (2012). Each sampling event involved taking three replicate invertebrate samples of each habitat type occurring in the reservoir (of bare shore, rocky shore, timber and macrophyte when available). Sampling locations were determined by dividing the reservoir into three equal sections and sampling each habitat type per section. Invertebrate samples were collected from edge habitats with a sweep net (250 µm mesh) over a 10 m transect. Samples were then preserved in 70% ethanol for later processing in the laboratory. In the laboratory, samples were rinsed through a 250 µm mesh sieve to remove fine sediment and ethanol and then placed in a large tray with water. Coarse scale invertebrate selection of entire edge habitat samples was performed using a magnifying lamp for one hour to calculate the numerical abundance of each invertebrate taxa. This method effectively captured information on the large quantities of abundant items such as decapods and generally resulted in the selection of larger invertebrates.

Fine scale invertebrate selection of 10% of the remaining sample under a stereomicroscope was then performed, and higher-powered magnification facilitated selection of smaller taxa. This was achieved by placing the remaining sample into a sub-sampler consisting of a box divided into 100 cells, 3 cm x 3 cm x 2.5 cm deep (Marchant, 1989). The box was agitated until the sample was distributed evenly across cells. A total of 10 cells out of the 100 were randomly selected using two ten-sided dice, and their contents were removed with a vacuum pump. This standardised sampling method allowed for calculation of numerical abundance of taxa. All invertebrate identification was to order for aquatic taxa or a terrestrial item category for terrestrial-occurring invertebrates for edge habitat food availability analyses.

Plankton tows to collect invertebrates from open water habitats were also undertaken in three replicate sections of the reservoir (downstream, middle and upstream thirds). A weighted, modified 250 µm mesh net with a circular opening (300 mm wide) was lowered into the water column 50 m away from shore at 1 m depth and pulled by a two-person crew in motorised boat along a 50 m transect (distance was determined using a rangefinder) for each tow (sampling 3.54 m^3 of open water habitat). Sampling was conducted bi-annually in spring and autumn at one site (the ECR). Open water invertebrates were identified from a 10% sub-sample using a 100 cell sub-sampler (Marchant, 1989) as described for fine scale invertebrate selection of edge habitat samples above.

PERMANOVA (Permutational Multivariate Analysis of Variance) to test for differences in community structure of both coarse pick and subsample data across different phases, seasons, and habitats. The analysis was performed on Log10(x+1) transformed data with Bray-Curtis resemblance matrix. Phase, Season, and Habitat were treated as fixed factors, with the highest interaction term removed. Following the PERMANOVA, pairwise comparisons were conducted to identify specific differences between phases. SIMPER (Similarity Percentage) analysis was used to determine which taxa contributed most to the observed differences between phases.

To assess differences in microcrustacean community composition across reservoir management phases and seasons, we conducted a PERMANOVA using the adonis2() function from the *vegan* package in R. Prior to analysis, the data were square-root transformed to reduce the influence of dominant taxa, and Bray–Curtis dissimilarities were calculated to quantify community differences. The PERMANOVA model included main effects for phase and season, as well as their interaction. Statistical significance was assessed using 999 unrestricted permutations.

To investigate the effects of season and phase on Decapoda abundance, a generalized linear mixed model (GLMM) with a negative binomial distribution was fitted using the glmmTMB package in R. The model included season, phase, and their interaction as fixed effects, with Year treated as a random effect to account for temporal variability. Model diagnostics using the DHARMa package indicated no significant issues with dispersion, outliers, or residual uniformity, although one group showed a minor deviation from uniformity.

RESULTS

Edge samples - Coarse pick

The PERMANOVA results indicate that phase and season are significant drivers of macroinvertebrate community structure in the coarse pick samples, while habitat and its interactions are not significant. The interaction between Phase and Season is also significant, suggesting that seasonal effects vary across different phases (Table 27). The pairwise PERMANOVA results show that all three Phase comparisons are statistically significant, even after Bonferroni correction. This indicates that all phases differ significantly in community composition, with the largest difference observed between baseline and operational phases. For the transition from baseline to filling phases, Decapoda and terrestrial items were the top contributors, both being higher in abundance during filling phase. For the transition from baseline to operational, Chironomidae was the most significant contributor, followed by Hemiptera and terrestrial items. The transition from filling to operational showed no significant individual taxa contributors, indicating a more subtle shift in community structure.

Table 27. Results of PERMANOVA analysis of coarse pick macroinvertebrate community composition in Cotter Reservoir from 2010 – 2025.

Term	df	SS	R2	F	P
Phase	2	3.4047	0.21476	13.5863	0.001
Season	1	0.9631	0.06075	7.6864	0.001
Habitat	2	0.1812	0.01143	0.7232	0.688
Phase x Season	2	0.5255	0.03315	2.0969	0.027
Phase x Habitat	4	0.3648	0.02301	0.7279	0.824
Season x Habitat	2	0.1393	0.00879	0.556	0.871
Residuals	82	10.2744	0.64811		

Analysis revealed a significant reduction in Decapoda abundance during spring compared to autumn, and a significant interaction between spring and the operational phase, suggesting that seasonal effects on abundance vary by phase (Table 27). The filling phase showed a marginal increase in abundance, while the operational phase alone was not significantly different from baseline monitoring phase (Table 27). Decapoda abundances in autumn generally increased between 2018 - 2022, though have been lower since then (Figure 37).

Table 28. Summary of fixed effects from the negative binomial generalized linear mixed model (GLMM) examining the influence of season and monitoring phase on Decapoda abundance in the Cotter Reservoir from 2009 – 2025.

Term	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	4.4264	0.5861	7.552	< 0.001
Spring	-2.8053	0.4522	-6.204	< 0.001
Filling	1.6164	0.9125	1.771	0.0765
Operational	0.6238	0.6718	0.929	0.3531
Spring × Filling	0.2164	0.7795	0.278	0.7813
Spring × Operational	1.467	0.5819	2.521	0.0117

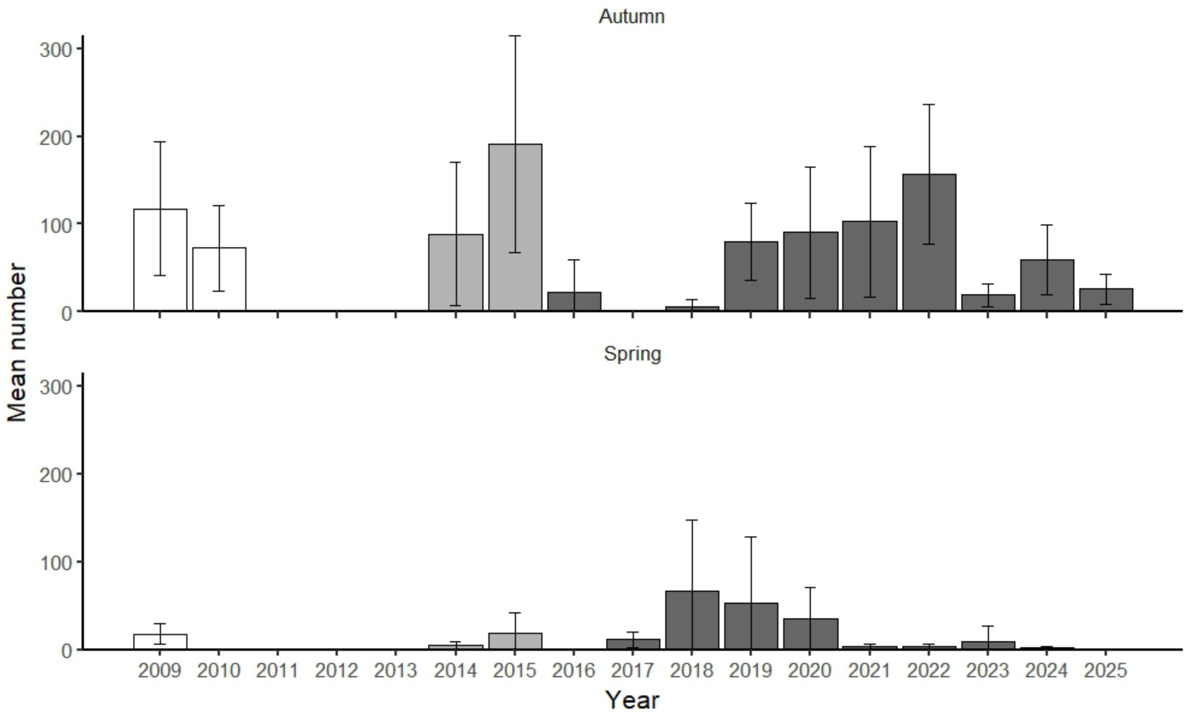

Figure 37. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of decapods collected from coarse pick edge samples taken from ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2025; dark grey bars) monitoring periods for autumn and spring. Note: Spring 2025 has not been sampled at the time of reporting.

Edge samples - 10% sub-sample

The PERMANOVA analysis revealed significant differences in community structure across the factors of phase, season, and habitat, with phase being the most influential factor (Table 29). A significant phase × season interaction suggests that seasonal changes influenced community structure differently across phases (Table 29). Pairwise comparisons between phases confirmed that all were significantly different from one another (p = 0.001), with the greatest dissimilarity observed between Baseline and Filling (R^2 = 0.161)

SIMPER analysis identified Chironomidae as the most influential taxon across all phase comparisons, with markedly higher abundances in the operational phase. Cladocera and Copepoda were more abundant in the Baseline phase. In the Filling phase, Cladocera remained relatively abundant, while Stratiomyidae and Hemiptera began to increase. Operational phase was characterized by higher abundances of Chironomidae, Diptera, and Ephemeroptera.

Table 29. Results of PERMANOVA analysis of 10% subsample macroinvertebrate community composition in Cotter Reservoir from 2009-2010 (baseline) & 2013 – 2015 (Filling) and 2016 – 2025 (Operational).

Source	df	SS	R^2	F	P
Phase	2	2.8373	0.17624	11.4389	0.001
Season	1	1.0582	0.06573	8.5325	0.001
Habitat	2	0.4845	0.03010	1.9535	0.021
Phase x Season	2	1.3950	0.08665	5.6242	0.001
Phase x Habitat	4	0.5686	0.03532	1.1461	0.273
Season x Habitat	2	0.0816	0.00507	0.3289	0.994
Residuals	78	9.6736	0.60089		
Total	91	16.0988	1.00000		

Tow samples

Microcrustaceans dominated the plankton tow samples from the baseline and the filling and operational phase monitoring, comprising 99.9%, 100% and 100% of samples, respectively. Although there are some interannual variations in the microcrustacean community, there was no significant effect of phase or season, or the interactions between the two (Table 30 & Figure 38).

Table 30. Results of PERMANOVA analysis of tow sample microcrustacean community composition in Cotter Reservoir from 2009-2010 (baseline) & 2013 – 2015 (Filling) and 2016 – 2025 (Operational) (bold text indicates statistically significant difference at the 0.05 level).

Source	df	SS	R^2	F	P
Phase	2	0.2832	0.01885	0.804	0.587
Season	1	0.2356	0.01568	1.338	0.250
Phase × Season	2	0.7106	0.04730	2.019	0.060
Residuals	69	13.7951	0.91817		
Total	74	15.0245	1.00000		

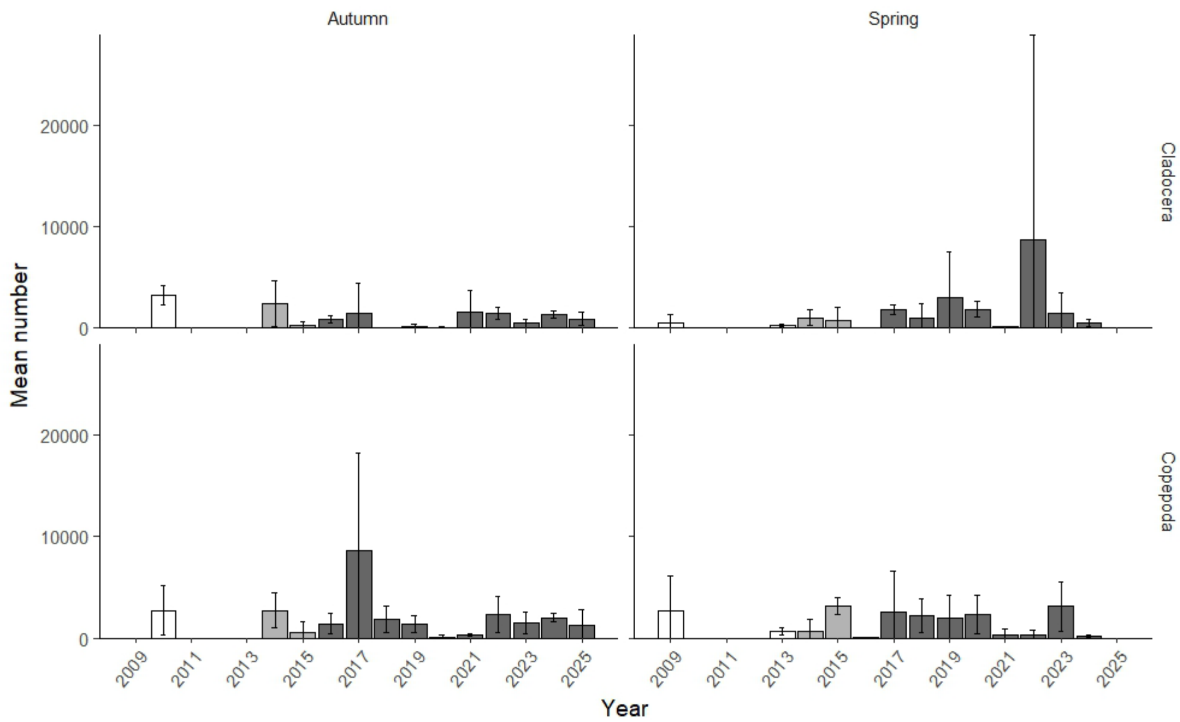

Figure 38. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of each microcrustacean taxa collected in autumn and spring of each phase of monitoring phase in ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2025; dark grey bars) monitoring periods. Note: Spring 2025 had not been collected at time of reporting.

DISCUSSION AND CONCLUSIONS

Decapod abundances in coarse pick edge samples are similar between phases. Decapod abundance in autumn during early operational phase monitoring was lower than that observed for autumn in baseline, filling and latter operational phase monitoring. Decapod abundance was much lower in spring than autumn. Despite being in very low abundances in early years of the operational phase, Decapod abundance was generally increasing between spring 2018 – 2022, though has reduced again 2023 (see Figure 37). Decapods haven been previously found to be an important food item of adult Macquarie perch and may be an important antecedent factor in spawning success as previous studies have found Macquarie perch fecundity to be positively related to body condition (Norris et al., 2012, Gray et al., 2000, Lintermans, 2006, Hatton, 2016). The reduction in Decapod abundance over the past few years is noteworthy, though hasn't resulted in a reduction in the body condition of adult Macquarie perch to date.

Terrestrial items were more abundant in the coarse pick samples in the filling and operational phase monitoring compared to the pre-filling/baseline study. As terrestrial habitats (earth and vegetation) became inundated, terrestrial invertebrates entered the water column. Also, whilst the reservoir is full, overhanging vegetation would provide a source of terrestrial insects to the reservoir. This would explain the increased abundance of terrestrial items in the filling and operational phase monitoring. Macquarie perch are an opportunistic feeder in Cotter Reservoir (Norris et al., 2012), and it is likely that they will take advantage of terrestrial items present during filling (Cadwallader and Douglas 1986). Indeed, data from stomach flushing showed that Macquarie perch were feeding on earthworms during spring 2013 in the Cotter Reservoir, but that this dietary item was not important in the following year (Hatton, 2016).

Tow net samples were numerically dominated by the microcrustaceans Cladocera and Copepoda in all phases. Abundances of both taxa varied, but there was no significant effect of phase or season on abundances. Cladocera were found to be an important dietary item of Macquarie perch in the Cotter Reservoir prior to filling (Norris et al., 2012) and also in another reservoir study of Macquarie perch in Lake Dartmouth (Cadwallader and Douglas, 1986), but had a reduced importance whilst the reservoir was filling (Hatton, 2016). Population abundance of Cladocera and Copepods are largely driven by temperature, turbidity, water residence time and predation (Bartrons et al., 2015, Dejen et al., 2004, Silva et al., 2014, Obertegger et al., 2007). Effect of the operation of Cotter Reservoir on microcrustacean abundances at this stage appears to be minimal.

RECOMMENDATIONS

Sampling for this question follows previously developed methods and appears to be adequate for detecting change. No change the monitoring approach is recommended.

The majority of the food resource differences between phases likely fall within natural annual and sampling variation. The main change of importance to the resident Macquarie perch population is the reduction in Decapod abundance during early operational phase, though this appears to be increasing in latter operational years. So far, this has not appeared to have a negative effect on adult condition, spawning or survival and growth of juveniles. No management intervention is recommended. It is now eight years since diet of Macquarie perch in the ECR was investigated, and Macquarie perch diet has not been assessed since the Cotter Reservoir has entered operational phase. A re-examination of Macquarie perch diet is recommended.

REFERENCES

Actew Corporation 2009a. Enlargement of Cotter Reservoir and associated works. Final public environment report 2008/4524. Canberra: ACTEW Corporation.

Actew Corporation 2009b. Enlargement of the Cotter Reservoir and associated works. Environmental Impact Statement. Canberra: ACTEW Corporation.

Actew Corporation 2013. *Alien fish management plan: Enlarged Cotter Dam,* Canberra, ACTEW Corporation.

Bartrons, M., Einarsson, Á., Nobre, R. L., Herren, C. M., Webert, K. C., Brucet, S., Ólafsdóttir, S. R. & Ives, A. R. 2015. Spatial patterns reveal strong abiotic and biotic drivers of zooplankton community composition in Lake Mývatn, Iceland. *Ecosphere,* 6**,** 1-20.

Barwick, D. H., Foltz, J. W. & Rankin, D. M. 2004. Summer habitat use by rainbow trout and brown trout in Jocassee Reservoir. *North American Journal of Fisheries Management,* 24**,** 735-740.

Broadhurst, B., Clear, R., Allan, H. & Lintermans, M. 2023. Enlarged Cotter Reservoir ecological monitoring program: technical report 2023. Canberra: Institute for Applied Ecology, University of Canberra.

Broadhurst, B., Clear, R. & Lintermans, M. 2016a. *Potential barriers to upstream migration of Macquarie perch to spawning areas in the Cotter River, ACT* Canberra, Institute for Applied Ecology, University of Canberra.

Broadhurst, B., Clear, R. C., Fulton, C. & Lintermans, M. 2018. Enlarged Cotter Reservoir ecological monitoring program: technical report 2018. Canberra: Institute for Applied Ecology, University of Canberra.

Broadhurst, B., Clear, R. C., Fulton, C. & Lintermans, M. 2019. Enlarged Cotter Reservoir ecological monitoring program: technical report 2019. Canberra: Institute for Applied Ecology, University of Canberra.

Broadhurst, B., Dyer, J., Ebner, B., Thiem, J. & Pridmore, P. 2011. Response of two-spined blackfish *Gadopsis bispinosus* to short-term flow fluctuations in an upland Australian stream. *hydrobiologia,* 673**,** 63-77.

Broadhurst, B., Ebner, B. & Clear, R. 2012a. A rock-ramp fishway expands nursery grounds of the endangered Macquarie perch *Macquaria australasica*. *Australian Journal of Zoology,* 60**,** 91–100.

Broadhurst, B. T., Clear, R. C., Fulton, C. & Lintermans, M. 2015. Enlarged Cotter Reservoir ecological monitoring program: technical report 2015. Canberra: Institute for Applied Ecology, University of Canberra.

Broadhurst, B. T., Clear, R. C., Fulton, C. & Lintermans, M. 2016b. Enlarged Cotter Reservoir ecological monitoring program: technical report 2016. Canberra: Institute for Applied Ecology, University of Canberra.

Broadhurst, B. T., Ebner, B. C., Lintermans, M., Thiem, J. D. & Clear, R. C. 2013. Jailbreak: A fishway releases the endangered Macquarie perch from confinement below an anthropogenic barrier. *Marine and Freshwater Research,* 64**,** 900–908.

Broadhurst, B. T., Lintermans, M., Thiem, J. D., Ebner, B. C., Wright, D. W. & Clear, R. C. 2012b. Spatial ecology and habitat use of two-spined blackfish *Gadopsis bispinosus* in an upland reservoir. *Aquatic Ecology,* 46**,** 297–309.

Budy, P., Thiede, G. P., Lobón-Cerviá, J., Fernandez, G. G., Mchugh, P., Mcintosh, A., Vøllestad, L. A., Becares, E. & Jellyman, P. 2013. Limitation and facilitation of one of the world's most invasive fish: an intercontinental comparison. *Ecology,* 94**,** 356-367.

Cadwallader, P. L. 1978. Some causes of the decline in range and abundance of native fish in the Murray-Darling River system. *Proceedings of the Royal Society of Victoria,* 90**,** 211–224.

Cadwallader, P. L. & Douglas, J. D. 1986. Changing food habits of Macquarie perch *Macquaria australasica* Cuvier (Pisces: Percichthyidae). during the initial filling phase of Lake Darmouth, Victoria. *Australian Journal of Marine and Freshwater Research,* 37**,** 647–657.

Dejen, E., Vijverberg, J., Nagelkerke, L. A. & Sibbing, F. A. 2004. Temporal and spatial distribution of microcrustacean zooplankton in relation to turbidity and other environmental factors in a large tropical lake (L. Tana, Ethiopia). *Hydrobiologia,* 513**,** 39-49.

Dorfman, E. J. & Kingsford, M. J. 2001. Environmental determinants of distribution and foraging behaviour of cormorants (Phalacrocorax spp.) in temperate estuarine habitats. . *Marine Biology,* 138**,** 1–10.

Ebner, B., Broadhurst, B., Lintermans, M. & Jekabsons, M. 2007. A possible false negative: lack of evidence for trout predation on a remnant population of the endangered Macquarie perch, *Macquaria australasica*, in Cotter Reservoir, Australia. *New Zealand Journal of Marine and Freshwater Research,* 41**,** 231–237.

Ebner, B. & Lintermans, M. 2007. Fish passage, movement requirements and habitat use for Macquarie perch. Final report to the Department of Agriculture, Fisheries and Forestry Australia. Canberra: Parks, Conservation and Lands.

Ebner, B., Thiem, J., Broadhurst, B., Clear, R. & Frawley, K. 2008. Delivering environmental flows to large biota. Final Report to the Department of the Environment, Water, Heritage and the Arts. Canberra: Parks, Conservation and Lands.

Farrington, L. W., Lintermans, M. & Ebner, B. C. 2014. Characterising genetic diversity and effective population size in one reservoir and two riverine populations of the threatened Macquarie perch. *Conservation Genetics***,** 1-10.

Fleming, J. P., Madsen, J. D. & Dibble, E. D. 2011. Macrophyte re-establishment for fish habitat in Little Bear Creek Reservoir, Alabama, USA. *Journal of Freshwater Ecology,* 26**,** 105-114.

Gray, S. C., De Silva, S. S., Ingram, B. A. & Gooley, G. J. 2000. Effects of river impoundment on body condition and reproductive performance of the Australian native fish, Macquarie perch (*Macquaria australasica*). *Lakes & Reservoirs: Research and Management,* 5**,** 281–291.

Hatton, S. 2016. *Ecological responses to the initial filling of an enlarged temperate reservoir: Cotter Reservoir (ACT, Australia).* PhD PhD thesis, University of Canberra.

Icon Water Limited 2019. Cotter Reservoir Fish Management Plan (FMP). Version 4. Canberra: Icon Water Limited.

Joseph, L. N., Field, S. A., Wilcox, C. & Possingham, H. P. 2006. Presence-absence versus abundance data for monitoring threatened species. *Conservation Biology,* 20**,** 1679-1687.

Kimmel, B. L. & Groeger, A. W. 1986. Limnological and ecological changes associated with reservoir aging. *In:* HALL, G. E. & VAN DEN AVYLE, M. J. (eds.) *Reservoir Fisheries Management: Strategies for the 80's.* Bethesday, Maryland: Reservoir Committee, South Division American Fisheries Society.

Lintermans, M. 1998. The ecology of the two-spined blackfish *Gadopsis bispinosus* (Pisces: Gadopsidae). Unpublished M. Sc. Thesis. Canberra: School of Botany and Zoology, Australian National University.

Lintermans, M. 2002. *Fish in the Upper Murrumbidgee Catchment: A Review of Current Knowledge,* Canberra, Environment ACT.

Lintermans, M. 2005. ACT future water options fish impact study: a review of potential impacts on fish and crayfish of future water supply options for the Australian Capital Territory: stage 1. Canberra: ACTEW Corporation.

Lintermans, M. 2006. The re-establishment of endangered Macquarie perch *Macquaria australasica* in the Queanbeyan River, New South Wales, with an examination of dietary overlap with alien trout. Canberra: Cooperative Research Centre for Freshwater Ecology.

Lintermans, M. 2007. *Fishes of the Murray-Darling Basin: an introductory guide,* Canberra, Murray-Darling Basin Commission.

Lintermans, M. 2012. Managing potential impacts of reservoir enlargement on threatened *Macquaria australasica* and *Gadopsis bispinosus* in southEastern Australia. *Endangered Species Research,* 16**,** 1–16.

Lintermans, M. 2013. The rise and fall of a translocated population of the endangered Macquarie perch *Macquaria australasica* in southEastern Australia. *Marine and Freshwater Research,* 64**,** 838–850.

Lintermans, M. 2016. Finding the needle in the haystack: comparing sampling methods for detecting an endangered freshwater fish. *Marine and Freshwater Research,* 67**,** 1740-1749.

Lintermans, M., Broadhurst, B. & Clear, R. 2011. *Characterising potential predation of Macquarie perch Macquaria australasica by cormorants in Cotter Reservoir.,* Canberra, Institute for Applied Ecology.

Lintermans, M., Broadhurst, B. & Clear, R. 2013. *Assessment of the potential impacts on threatened fish from the construction, filling and operation of the Enlarged Cotter Dam Phase 1 (2010-2012): Final Report. Report to ACTEW Corporation.,* Canberra, Institute for Applied Ecology, University of Canberra.

Lintermans, M., Broadhurst, B., Thiem, J., Ebner, B., Wright, D., Clear, R. & Norris, R. 2010. *Constructed homes for threatened fishes in the Cotter River catchment: Phase 2 final report. Report to ACTEW Corporation.,* Canberra, Institute for Applied Ecology, University of Canberra.

Macdonald, A. J., Young, M. J., Lintermans, M. & Sarre, S. D. 2014. Primers for detection of Macquarie perch from environmental and trace DNA samples. *Conservation genetics resources,* 6**,** 551-553.

Macdonald, J. & Tonkin, Z. 2008. A review of the impacts of Eastern gambusia on native fishes of the Murray–Darling Basin. *Murray–Darling Basin Authority Publication*.

Madsen, J. D., Chambers, P. A., James, W. F., Koch, E. W. & Westlake, D. F. 2001. The interaction between water movement, sediment dynamics and submersed macrophytes. *Hydrobiologia,* 444**,** 71-84.

Marchant, R. 1989. A sub-sampler for samples of benthic macroinvertebrates. . *Bulletin of the Australian Society for Limnology,* 12**,** 49–52.

Maxwell, D. & Jennings, S. 2005. Power of monitoring programmes to detect decline and recovery of rare and vulnerable fish. *Journal of Applied Ecology,* 42**,** 25-37.

Merrick, J. R. & Schmida, G. E. 1984. *Australian freshwater fishes: biology and management,* North Ryde, NSW, J. Merrick.

Miller, B. 1979. Ecology of the little black cormorant, *Phalocrocorax sulcirostris*, and the little pied cormorant, *P. melanoleucos*, in inland New South Wales. I. Food and feeding habits. *Australian Wildlife Research,* 6**,** 79–95.

Norris, R., Wright, D., Lintermans, M., Bourke, D. & Harrison, E. 2012. *Food resources for Macquarie perch in Cotter Reservoir. Final report to the Bulk Water Alliance.,* Canberra, Institute for Applied Ecology, University of Canberra.

Nsw Fisheries 2003. Environmental Impact Statement on Freshwater Fish Stocking in NSW. *Public Consultation Document. 3 volumes.* Cronulla: NSW Fisheries.

O'brien, W. J. 1990. Chapter 8. Perspecitives on fish in reservoir ecosystems. *In:* THORNTON, K. W., KIMMEL, B. L. & PAYNE, F. E. (eds.) *Reservoir limnology: Ecological perspectives.* New York: John Wiley & Sons Inc.

Obertegger, U., Flaim, G., Braioni, M. G., Sommaruga, R., Corradini, F. & Borsato, A. 2007. Water residence time as a driving force of zooplankton structure and succession. *Aquatic Sciences,* 69**,** 575-583.

Ploskey, G. R. 1986. Effects of water-level changes on reservoir ecosystems, with implications for fisheries management. *In:* HALL, G. E. & VAN DEN AVYLE, M. J. (eds.) *Reservoir fisheries management: strategies for the 80's.* . Bethesda, Maryland.: Reservoir Committee, Southern Division American Fisheries Society.

Poos, M., Mandrak, N. & Mclaughlin, R. 2007. The effectiveness of two common sampling methods for assessing imperilled freshwater fishes. *Journal of Fish Biology,* 70**,** 691-708.

Pyke, G. H. 2005. A Review of the Biology of *Gambusia affinis* and *G. holbrooki*. *Reviews in Fish Biology and Fisheries,* 15**,** 339-365.

R Development Core Team 2022. R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing.

Richardson, S. M., Hanson, J. M. & Locke, A. 2002. Effects of impoundment and water-level fluctuations on macrophyte and macroinvertebrate communities of a dammed tidal river. *Aquatic Ecology,* 36**,** 493-510.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations. *Bulletin of the Fisheries Research Board of Canada,* 191**,** 1-382.

Roberts, J. 2006. Wetland Vegetation around Cotter Reservoir. *Report JR 11/2006.* Canberra, ACT.

Robinson, W. 2009. Biometrical review of proposed fish monitoring programs for the Enlarged Cotter Dam project. *Consultancy report to the Bulk Water Alliance.* University of the Sunshine Coast.

Ropert-Coudert, Y., Gremillet, D. & Kato, A. 2006. Swim speeds of free-ranging great cormorants. *Marine Biology,* 149**,** 415-422.

Rose, K. A., Cowan Jr, J. H., Winemiller, K. O., Myers, R. A. & Hilborn, R. 2001. Compensatory density dependence in fish populations: importance, controversy, understanding and prognosis. *Fish and Fisheries,* 2**,** 293-327.

Ryan, K. A. 2010. *Macquarie perch under pressure: predation risk to an endangered fish population. Unpublished PhD thesis.* Institute for Applied Ecology, University of Canberra.

Ryan, K. A., Lintermans, M., Ebner, B. C. & Norris, R. 2013. Using fine-scale overlap in predator-prey distribution to assess avian predation risk to a reservoir population of threatened Macquarie Perch. *Freshwater Science,* 32**,** 1057-1072.
Silva, L. H., Huszar, V. L., Marinho, M. M., Rangel, L. M., Brasil, J., Domingues, C. D., Branco, C. C. & Roland, F. 2014. Drivers of phytoplankton, bacterioplankton, and zooplankton carbon biomass in tropical hydroelectric reservoirs. *Limnologica-Ecology and Management of Inland Waters,* 48**,** 1-10.
Thiem, J. D., Broadhurst, B. T., Lintermans, M., Ebner, B. C., Clear, R. C. & Wright, D. 2012. Seasonal differences in the diel movements of Macquarie perch (*Macquaria australasica*) in an upland reservoir. *Ecology of Freshwater Fish,* 22**,** 145–156.
Todd, C. R., Lintermans, M., Raymond, S. & Ryall, J. 2017. Assessing the impacts of reservoir expansion using a population model for a threatened riverine fish. *Ecological Indicators,* 80**,** 204-214.
Tonkin, Z., Kearns, J., Lyon, J., Balcombe, S. R., King, A. J. & Bond, N. R. 2017. Regional-scale extremes in river discharge and localised spawning stock abundance influence recruitment dynamics of a threatened freshwater fish. *Ecohydrology,* 10**,** e1842.
Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A. & Smith, G. M. 2009. *Mixed effects models and extensions in ecology with R*, Springer.