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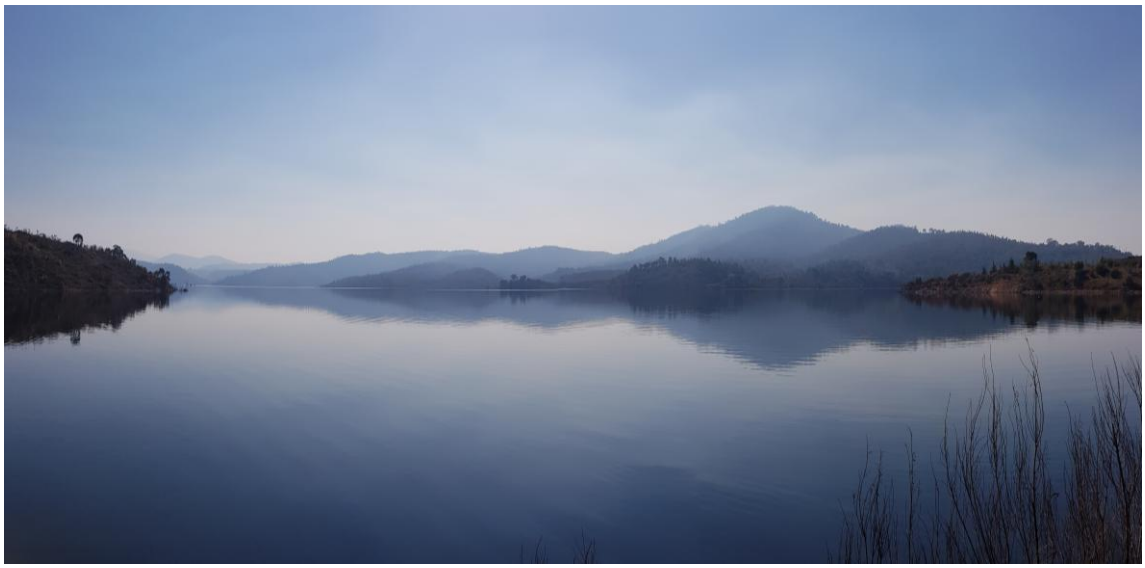
Enlarged Cotter Reservoir Ecological Monitoring Program

Technical Report 2026



Report to Icon Water

June 2026

Enlarged Cotter Reservoir Ecological Monitoring Program: Technical Report 2026

Report prepared for: Icon Water

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Cite this report as: Broadhurst, B. T., Clear, R. C., Allan, H., Lhendup, U. and Lintermans, M. (2026). *Enlarged Cotter Reservoir ecological monitoring program: technical report 2026*. Centre for Applied Water Science, University of Canberra, Canberra.

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EXECUTIVE SUMMARY

The Enlarged Cotter Reservoir (ECR) monitoring program assesses ecological responses to reservoir enlargement and operation. The 2026 monitoring year is the eleventh year of the operational phase and the seventeenth year of the program overall, providing an extensive long-term dataset spanning baseline (2010–2013), filling (2014–2015), and operational (2016–2026) phases against which current conditions are assessed.

The analytical framework underpinning the monitoring program has been strengthened in this reporting period. Both previous and current analyses used the same underlying field measurements and shoreline data; what has changed is how those data are handled statistically. Previously, catch rates were calculated by dividing raw catches by effort and applying a shoreline correction factor before analysis. In 2026, these corrections are incorporated directly inside the statistical models, which produces more reliable confidence intervals and avoids known problems with pre-processed ratio data. Models now use flexible curve-fitting approaches (Generalised Additive Models) that better describe the nonlinear trajectories characteristic of a long-term reservoir filling and operation dataset, and abundance estimates are expressed on a consistent pre-enlargement reference scale to make phase comparisons directly interpretable. These refinements do not change the broad ecological conclusions of previous years but do sharpen two specific findings, one more positive than previously reported and one identifying a spatially targeted management concern not previously detectable, and provide greater confidence that reported trends reflect genuine ecological responses rather than analytical artefacts.

Overall, the 2026 monitoring year was characterised by:

- Adult Macquarie perch abundance in Cotter Reservoir consistent with the recent operational trend and, when correctly standardised for habitat expansion, not significantly different from pre-enlargement baseline conditions
- Recruitment of Macquarie perch (young-of-year and juveniles) remaining persistently and significantly below pre-enlargement levels in both the reservoir and river, continuing the multi-year pattern with no recovery trend evident in 2026
- Rainbow and Brown trout abundance in both reservoirs continuing the declining trend observed since 2021, with catches in 2026 among the lowest of the operational phase
- Rainbow trout abundance in the Cotter River declining significantly from the 2023–2025 peak but remaining elevated above baseline at reservoir-adjacent sites, maintaining an ongoing predation risk at the spawning interface
- Two-spined blackfish remaining at very low abundance in Cotter Reservoir and undetected at Bendora Reservoir for the second consecutive year
- Ecosystem conditions broadly stable, with most indicators continuing within the operational range established over recent years

The central management concern remains the persistent suppression of Macquarie perch recruitment. In 2026 this pattern continued unchanged, with young-of-year near-absent at both reservoir and river monitoring sites and no indication of improvement relative to the 2023–2025 period. Adults remain in good condition and at stable abundance, but without improved recruitment the long-term trajectory of the population warrants continued close attention.

Macquarie Perch – Reservoir Population

Adult Macquarie perch abundance in 2026 was low but consistent with the recent operational trend and within the range observed since 2018. When correctly standardised for the fourfold expansion of reservoir shoreline since enlargement, adult density is not significantly different from pre-enlargement baseline conditions, a more positive assessment than previously reported. Fish

condition in 2026 was stable and consistent with recent years, indicating the reservoir continues to provide good foraging conditions. A formal mark-recapture population estimate is scheduled for 2027 and will provide the most direct assessment of whether total adult numbers have changed since enlargement.

Recruitment remained severely suppressed, representing the primary ongoing risk to population persistence. Young-of-year catches were among the lowest of the operational phase and approximately 12–17-fold below pre-enlargement baseline levels across recent monitoring windows. There was no improvement relative to 2023–2025. Juvenile catches were also significantly below baseline, consistent with the progressive depletion of pre-enlargement cohorts. Only occasional years during the operational phase have produced catches approaching baseline levels and no sustained recovery trend has been detected. The divergence between stable, well-conditioned adults and persistently scarce young fish points to a constraint acting during spawning or the earliest life stages rather than on adult survival.

Macquarie Perch – River Population

Macquarie perch were recorded at most Cotter River monitoring sites in 2026, confirming continued population persistence across the study area. Young-of-year were near-absent at impact sites in 2026, continuing the pattern of the previous two years. Juvenile abundance at impact sites has declined significantly below baseline in recent monitoring windows, indicating that pre-enlargement cohorts that previously buffered juvenile catches have been progressively exhausted. The Kissops Flat reference site maintained young-of-year catches at or above baseline in 2026, confirming that regional spawning capacity remains intact and that the suppression at Cotter River impact sites is localised rather than driven by broader environmental conditions.

Two-spined Blackfish

Four Two-spined blackfish were detected in Cotter Reservoir in 2026, all large adults in the upstream inundation zone, consistent with recent years. No juveniles were recorded. At Bendora Reservoir, no Two-spined blackfish were detected for the second consecutive year, continuing a statistically confirmed long-term declining trend. A December 2025 survey of Corin Reservoir confirmed a robust, self-sustaining population with broad age-class structure and catch rates well above historical baseline, and Corin Reservoir is formally recommended as the replacement reference site for future monitoring.

Trout – Reservoirs

Rainbow trout abundance in Cotter Reservoir in 2026 was low, continuing the declining trend evident since 2021 and among the lowest catches of the operational phase. Bendora Reservoir showed a similar pattern. The previously reported finding that Cotter Reservoir trout increased relative to Bendora during the operational phase does not hold under the revised analysis; both reservoirs declined in parallel, consistent with trophic upsurge attenuation rather than a reservoir-specific effect. Brown trout abundance continued its decline toward near-zero, a positive outcome for native species. Overall the reservoir trout situation in 2026 is reassuring, with no indication of a resurgence.

Trout – Cotter River

Rainbow trout were captured at all Cotter River sites in 2026. System-wide abundance declined significantly from the 2023–2025 peak, including at the unregulated Cotter Hut reference site, consistent with a regional rather than reservoir-specific driver. Despite this decline, catch rates at U/S ECR and Vanitys Crossing, the two sites closest to the reservoir, remain elevated relative to pre-enlargement baseline and above the levels at more distant sites, maintaining the spatial gradient identified in the 2026 analysis. This concentration of trout at the reservoir-river interface, which

spatially overlaps with the primary Macquarie perch spawning reach during October to January, remains the most important current management concern from a predation risk perspective. Monitoring this gradient in 2027 will help determine whether the pattern is persisting or resolving as regional abundance declines.

Non-native Fish

Goldfish abundance in Cotter Reservoir in 2026 was low and consistent with recent years, remaining well below the filling-phase peak. Catches in 2026 were at their lowest recorded level, significantly below even pre-enlargement baseline conditions. The filling-phase trophic upsurge response has fully attenuated. The continuing low Goldfish abundance is noted in the context of cormorant foraging dynamics, as Goldfish represent an important alternative prey source for piscivorous birds.

Piscivorous Birds

Cormorant abundance and community composition in 2026 were broadly consistent with the recent operational trend. Great cormorant breeding activity was confirmed for the fourth consecutive year. Little pied cormorant abundance remains above baseline while Little black cormorant has continued its operational-phase decline. The combination of an active Great cormorant breeding colony and declining Goldfish availability as alternative prey continues to be monitored as a potential pathway for increased predation pressure on Macquarie perch. Re-examination of cormorant diet composition is recommended.

Macrophytes and Food Resources

The four Cumbungi stands established since 2022 remained present in 2026 but all showed signs of deterioration in April 2026, a new observation not recorded in previous years. The cause warrants investigation given the value of macrophyte beds as refuge habitat. Macroinvertebrate communities continue to differ from baseline, with decapod abundance now formally confirmed as significantly below pre-enlargement levels. Adult Macquarie perch condition remains good, indicating food availability is not currently limiting, but the longer-term trajectory of the prey base warrants continued monitoring.

Synthesis and Implications

The 2026 monitoring year continued the pattern established over the past several years, with most indicators stable within the operational range. The adult Macquarie perch population remains in better condition than early analyses suggested, trout abundance in the reservoirs is declining, and the broader ecosystem has stabilised following the pronounced post-filling changes of 2014–2018. These are positive signals.

The outstanding concern is the persistent suppression of Macquarie perch recruitment, which showed no improvement in 2026 relative to recent years. This pattern has now been consistently documented throughout the operational phase. Adults persist and are healthy; the concern is whether sufficient recruitment is occurring to sustain the population over the longer term. The 2027 population estimate and continued monitoring of the spawning period will be important in assessing this risk.

Key Recommendations

1. Proceed with the 2027 mark-recapture population estimate for Macquarie perch in Cotter Reservoir to confirm whether total adult population size has changed since enlargement, as current indicators suggest but cannot confirm.
2. Prioritise investigation of the Macquarie perch recruitment bottleneck, with particular focus on flow conditions during the October to December spawning window, spawning habitat availability and riffle condition, and larval survival at the reservoir-river interface. This remains the most important conservation action identified by the program.
3. Conduct targeted Rainbow trout control and diet assessment at headwaters of the ECR and U/S ECR during December to January, using visual stomach content examination and eDNA metabarcoding, to determine whether trout at these sites are actively preying on Macquarie perch larvae during the peak larval and juvenile development period. This, in conjunction with managed river flows, will likely support early survival of Macquarie perch recruits.
4. Consider proposed trout trigger levels in Cotter Reservoir (see recommendations for Question 4).
5. Monitor the spatial gradient in river trout abundance in 2027 to determine whether the reservoir-adjacent elevation is persisting or resolving as regional abundance continues to decline following the 2023–2025 peak. This will help distinguish reservoir-specific from regional drivers and inform the urgency of targeted management.
6. Formally adopt Corin Reservoir as the primary reference site for Two-spined blackfish monitoring before the next monitoring cycle, given the confirmed collapse of the Bendora reference population.
7. Re-examine cormorant diet composition, with particular focus on Great cormorant given confirmed breeding for the fourth consecutive year and the declined availability of Goldfish as alternative prey.
8. Maintain the current monitoring program with continued emphasis on recruitment dynamics. If flow management interventions do not produce measurable improvement in recruitment at Cotter River impact sites within 3–5 years, initiate planning for translocation of reservoir-origin juveniles to upper river reaches as a contingency measure.

BACKGROUND

The Enlarged Cotter Reservoir (ECR) was commissioned to improve water security for the Australian Capital Territory, increasing storage capacity from approximately 4 GL to 76.2 GL. While this augmentation provides substantial benefits for water supply, it represents a major modification of an existing reservoir–river system, primarily through the expansion and deepening of inundated habitats rather than the construction of a new impoundment. Such reservoir enlargement increases the extent of lentic habitat, inundates additional riverine reaches, and modifies connectivity between the reservoir and upstream environments, with potential implications for habitat availability, ecological processes, and species persistence (Lintermans, 2012).

Reservoir impoundment and enlargement are widely recognised to influence aquatic ecosystems through multiple interacting pathways. These include modification of natural flow regimes, inundation of terrestrial and riverine habitats, disruption of longitudinal connectivity, and changes to thermal and physicochemical conditions (Poff et al., 1997, Poff et al., 2007, Dudgeon et al., 2006). Such changes can alter habitat availability, species interactions, and reproductive processes, particularly for species dependent on flowing water (Agostinho et al., 2008).

A critical but comparatively less well understood phase of reservoir development is the initial filling period, during which terrestrial ecosystems are rapidly transformed into aquatic environments. This transition is characterised by unstable environmental conditions and rapid ecological change, as previously unflooded vegetation and soils are inundated, releasing nutrients and organic matter into the water column (Grimard and Jones, 1982, Gunkel et al., 2003). This process drives a short-term increase in system productivity, commonly referred to as trophic upsurge, where nutrient enrichment stimulates primary production and propagates through higher trophic levels, influencing invertebrate and fish communities.

The magnitude and duration of trophic upsurge vary among systems, but it is typically followed by a stabilisation phase as nutrient inputs decline and the system approaches a new equilibrium (Grimard and Jones, 1982, Straskraba et al., 1993). In temperate reservoirs, trophic upsurge is generally less intense and shorter in duration than in tropical systems, reflecting lower biomass in inundated vegetation and lower nutrient availability in catchments (Gunkel et al., 2003, Hatton, 2016). Consequently, while short-term increases in productivity may enhance growth and condition of some species, these effects are often transient and may not translate into sustained recruitment or long-term population persistence (Hendrickson and Power, 1999, Irz et al., 2006).

The ecological responses to reservoir filling extend beyond nutrient dynamics to include broader changes in food webs and species interactions. Inundation of terrestrial habitats can temporarily increase the availability of allochthonous food resources, leading to shifts in trophic pathways and feeding behaviour (Cadwallader and Douglas, 1986, Albrecht et al., 2009, Hatton, 2016). At the same time, changes in habitat structure and water level regimes influence the distribution and abundance of aquatic organisms, particularly in littoral zones where habitat is rapidly restructured during filling (Baxter, 1977, Schmude et al., 1998, Furey et al., 2006). These processes can result in rapid but dynamic reorganisation of reservoir ecosystems as they transition from riverine to lentic conditions (Voshell Jr and Simmons Jr, 1984, Jorcin and Nogueira, 2008, Gunkel et al., 2003).

These mechanisms are particularly relevant in the Cotter system, where reservoir enlargement was predicted to inundate additional riverine habitat, increase depth, and further modify connectivity between the reservoir and upstream environments (Lintermans, 2012). Key risks identified for native fish populations include loss of refuge habitat, reduced access to spawning areas, increased predation pressure, and altered food resources. Species reliant on lotic habitats, such as Macquarie perch (*Macquaria australasica*) and Two-spined blackfish (*Gadopsis bispinosa*), were identified as especially vulnerable due to their dependence on flowing water for reproduction and early life stages.

The Cotter River and associated reservoirs support several threatened aquatic species, including Macquarie perch, Two-spined blackfish, Trout cod (*Maccullochella macquariensis*), and Murray River crayfish (*Euastacus armatus*). Of these, Macquarie perch and Two-spined blackfish are the primary focus of this monitoring program due to their confirmed presence within the system and their sensitivity to habitat alteration associated with flow regulation and impoundment (Lintermans, 2023, Koehn et al., 2020).

In light of these predicted risks and ecological processes, a structured long-term monitoring program was established to assess the response of aquatic ecosystems to reservoir enlargement. The program encompasses three management phases:

- **Baseline (2010–2013):** Pre-enlargement conditions
- **Filling (2014–2015):** Reservoir impoundment and transition
- **Operational (2016–present):** Post-filling system behaviour

The program is designed around ten key management questions addressing population dynamics, distribution, recruitment, predator–prey interactions, and ecosystem condition across multiple trophic levels. A Before–After Control–Impact (BACI) framework is applied where possible, incorporating both impact sites (Cotter Reservoir and River) and reference sites (e.g. Bendora Reservoir and Kissops Flat) (Underwood, 1994, Downes et al., 2002).

The central objective of the program is to determine whether observed ecological changes can be attributed to reservoir enlargement and operation, and to provide an evidence base for adaptive management of threatened species, particularly Macquarie perch. This includes evaluating key predicted risks such as recruitment limitation, altered trophic dynamics, and habitat constraints, as well as assessing whether ecological processes have stabilised under operational conditions.

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.

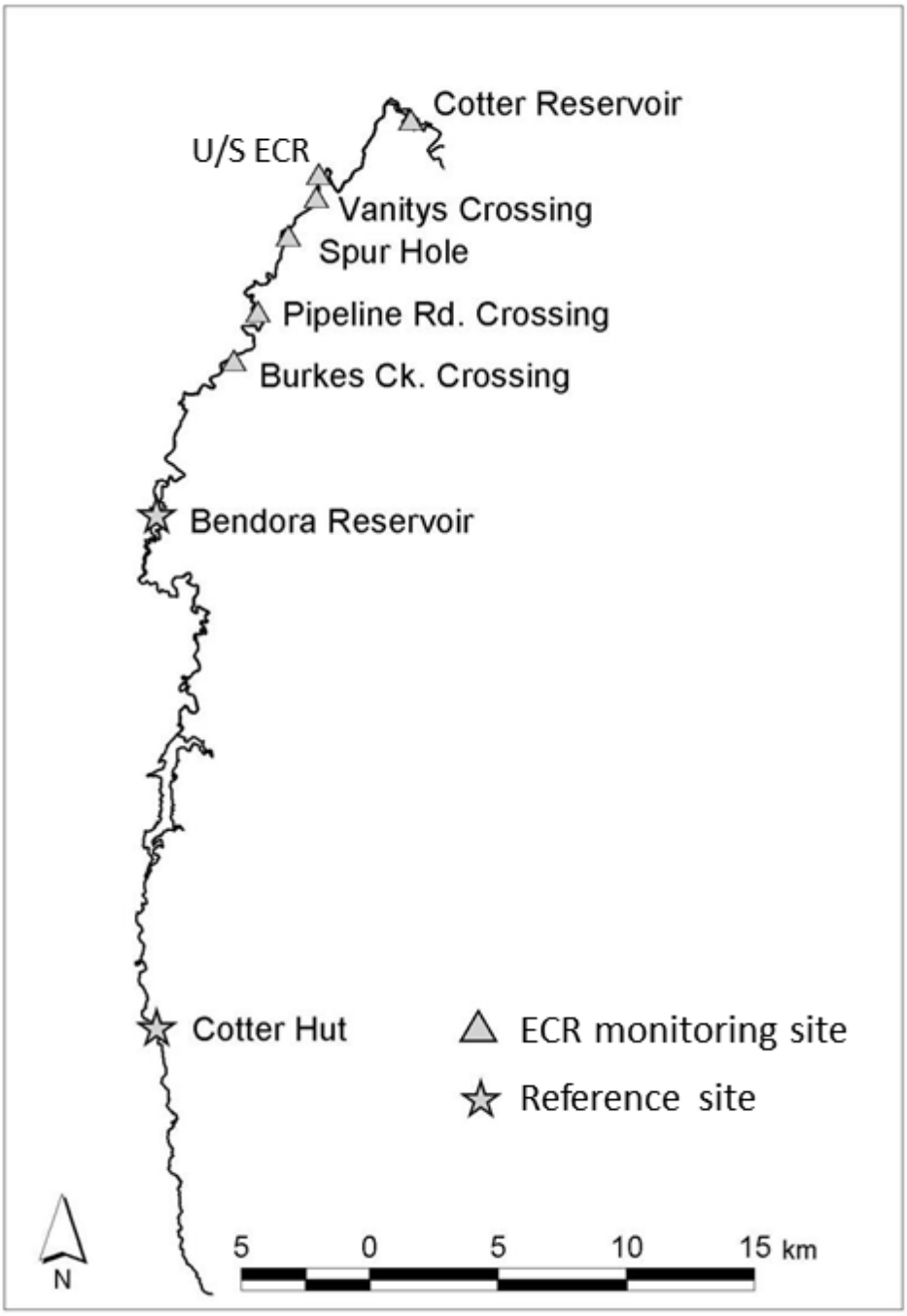

Figure 1. Location of sampling sites on the Cotter River. Note: Map does not include the reference site on the Murrumbidgee River (Kissops Flat).

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
U/S ECR	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 U/S ECR 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 three monitoring years saw a return to river flows which were largely dominated by regulated flow releases (because of dry climatic conditions), (Figure 2b and Figure 3). Cotter River was largely low and stable flows during November & December (key spawning and early development time for Macquarie perch), though a sharp spike in discharge occurred in mid-late November 2025 (Figure 2). Bendora Reservoir was at or just above full supply level (FSL) from mid-September to end of October 2025 but otherwise was below full supply level (Figure 3). Cotter Reservoir was at or just above FSL from July 2025 until the start of January 2026, then declined steadily to nearly 1 m below FSL by April 2026 (Figure 4).

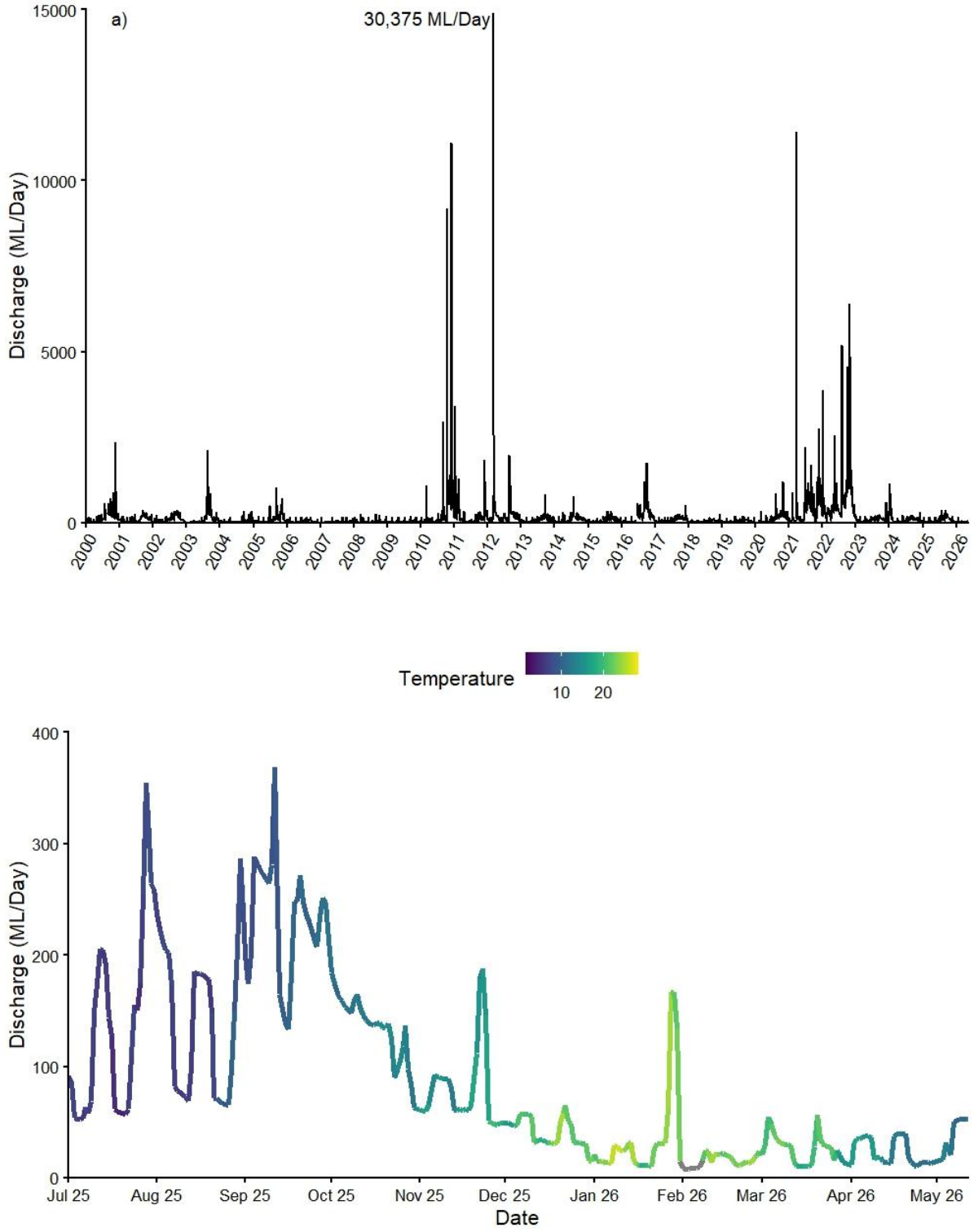

Figure 2. Mean daily discharge of the Cotter River at Vanitys Crossing from a) January 2000 until May 2026 and b) July 2025 – May 2026.

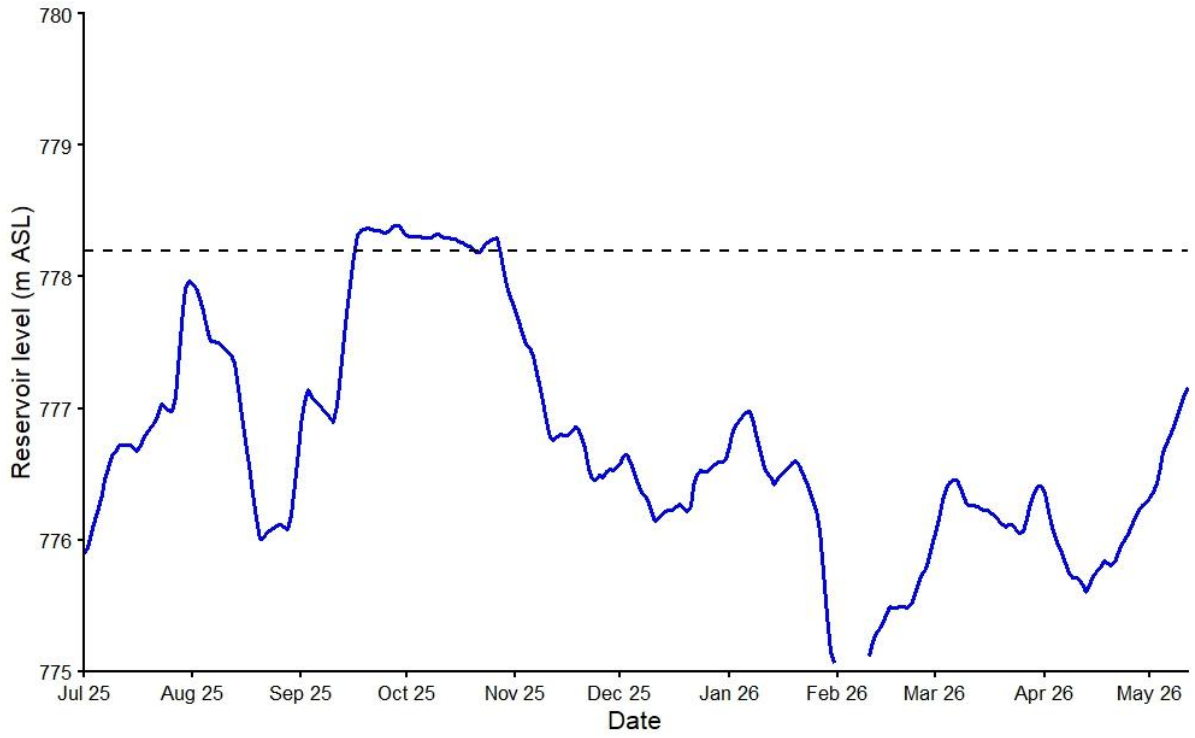

Figure 3. Water level (in metres above sea level) of Bendora Reservoir (blue line) from July 2025 – May 2026. Dashed line indicates Bendora Reservoir full supply level.

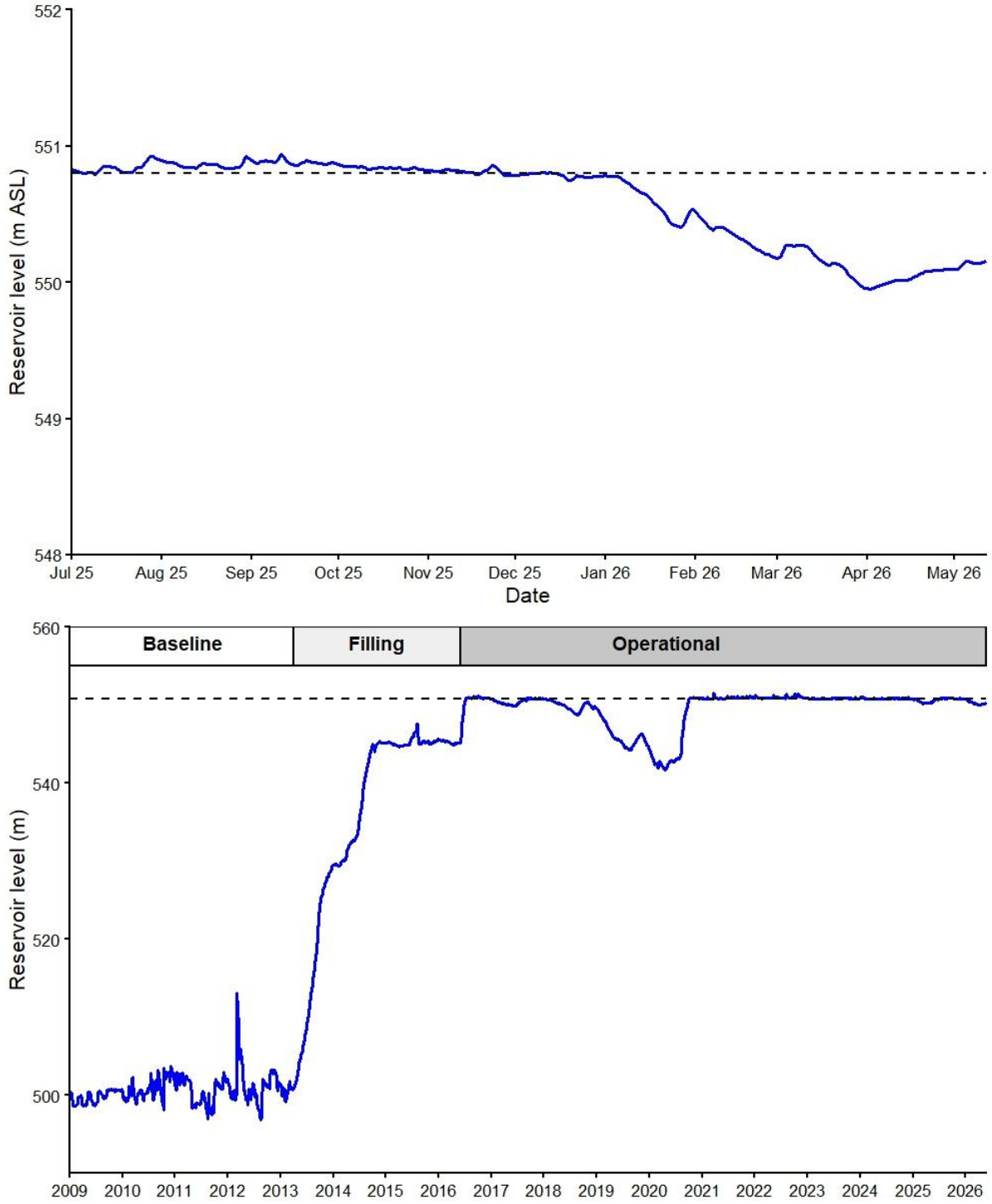

Figure 4. Water level (in metres above sea level) of Cotter Reservoir for the monitoring year (July 2025 – May 2026 (top panel) and long term from January 2009 until May 2026 (bottom panel) - rectangles on bottom panel indicate the three monitoring phases: white = Baseline, grey = Filling, hatched = 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 habitat loss, interactions with alien fish species, and predation by cormorants, can impact the Macquarie perch (*Macquaria australasica*) population in the Cotter River and reservoir during the filling and operation of the new Enlarged Cotter Reservoir (ECR). When evaluating these potential ECR impacts, monitoring must account for natural fluctuations in population abundance. These variations often arise from interannual climate shifts, flow regimes, stochastic extreme events (e.g., floods, droughts), and other factors influencing rates of spawning, recruitment, and mortality.

The body condition of adult Macquarie perch serves as a key indicator of reproductive potential; thus, tracking changes in adult condition provides valuable insights into future recruitment events and overall population trajectories (Gray et al., 2000). Spawning of reservoir-resident populations may be impacted by newly encountered riverine barriers in a filling reservoir or by fluctuating water levels (Broadhurst et al., 2016a). Consequently, the early detection of spawning success (via snorkelling for larvae) and its relationship to young-of-year (YOY) captured via reservoir netting are critical to understanding annual recruitment success or failure.

METHODS

The sampling design for Question 1 largely follows the baseline monitoring program framework outlined in Table 2 (Lintermans et al., 2013). To calculate fish condition, an additional metric—the wet weight of individuals captured in gill nets—was incorporated. Boat electrofishing was also introduced as a complementary sampling method to mitigate potential sampling inefficiencies associated with gill netting for adult Macquarie perch. Additionally, to determine the likely contribution of reservoir adults to YOY recruitment, snorkelling surveys were conducted in the river from immediately upstream of the ECR to Vanitys Crossing.

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 typically 15–25 mm TL.
Sampling technique/s	Gill nets: 10 nets per night for 5 nights (4 × 100 mm; 4 × 75 mm; 2 × 125 mm stretch mesh), plus an additional 2 × 125 mm nets per night to target larger individuals. 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 section for daytime and 6 shots per shoreline section for nighttime sampling. Snorkelling: Visual surveys of stream pools for larvae / early juveniles.
Timing	Netting and electrofishing conducted annually in February–April; snorkelling in November/December.
Number / location of sites	One impacted site: Enlarged Cotter Reservoir (ECR). One reference site: Kissops Flat (upper Murrumbidgee River; fyke netting only).
Information to be collected	Abundance and TL for all captured Macquarie perch. Wet weight (g) for subadults/adults captured in gill nets. Number of larvae/early YOY per pool during snorkelling.
Data analysis	Length and condition of adults were modelled across monitoring phases using Gaussian GLMMs with a random effect for year to account for temporal autocorrelation. Adult abundance (gill net data) was analysed using two complementary negative binomial GAMs fitted to raw count data with soak duration as a log-offset: a smooth-only model (s(year)) for temporal trend testing, and a phase-only model for phase-level mean estimation via coefficient back-transformation (delta method). Raw CPUE was interpreted in the context of reservoir shoreline expansion to assess population-level change. Young-of-year and juvenile abundance (fyke net data) were modelled within a Before–After Control–Impact (BACI) framework comparing Cotter Reservoir (impact) and Kissops Flat (reference) using Tweedie GAMs with site-specific penalised regression splines and a log-offset for soak duration. Additional two-level GAMs compared recent monitoring windows (2026, 2023–2025, 2024–2026) against baseline for both gill net and fyke net data. Pairwise contrasts were evaluated using estimated marginal means with Tukey adjustment.

Non-larval sampling targeted adult, juvenile, and YOY life stages. Based on findings by Ebner and Lintermans (2007) indicating that males reach sexual maturity at this threshold, individuals > 150 mm TL were classified as adults. During the autumn netting periods, YOY typically measure 60–85 mm TL, while juveniles are defined as those falling between 85–150 mm TL. Snorkelling surveys specifically targeted larval and early YOY individuals estimated to be 2–4 weeks old (~15–25 mm TL).

Sampling was conducted at two primary locations: the ECR (impacted site) and Kissops Flat (reference site for YOY and juveniles). While only fyke netting was deployed at the reference site, sampling within the ECR utilised two complementary netting techniques to secure a representative sample across the population's entire size range. Multi-filament, free-floating gill nets were used to optimize the capture of adult Macquarie perch, while single-winged fyke nets (12 mm stretch-mesh) were deployed to target YOY and juveniles (Ebner and Lintermans, 2007, Lintermans et al., 2013, Lintermans, 2016).

Gill nets were deployed in accordance with baseline and historical operational monitoring protocols. Twelve gill nets were set independently around the reservoir perimeter between late February and early April 2026. The reservoir was divided into five longitudinal sections, with two gill nets allocated per section. To prevent excessive stress on adult fish from sequential recaptures, netting was conducted over five nights (based on power analysis conducted by (Robinson, 2009)) but restricted to a maximum of two consecutive nights. Following recommendations by Broadhurst et al. (2018) to account for increasingly large adult fish, two 125 mm gill nets (including one 66-mesh deep-drop net) were added in 2022 to supplement the original 10-net array. Soak times were restricted to six hours, commencing at approximately 15:30 hrs, matching established threatened species protocols to minimize prolonged fish retention.

Twenty fyke nets were set singularly around the ECR perimeter over three nights between late February and early April 2026. For reference sampling, 12 fyke nets were deployed for two nights in the pool at Kissops Flat in February 2026. All fyke nets followed a standard ~16-hour soak time, commencing between 15:30 and 16:00 hrs. Snorkelling was undertaken on the 5th of December 2025, to look for early juvenile Macquarie perch. The first four pools upstream of the headwater of Cotter Reservoir were surveyed (as well as ad hoc survey of the most upstream ~100m of the reservoir)

Boat electrofishing was conducted across multiple days. The reservoir was divided into five longitudinal sections, with twelve 90-second "on-time" electrofishing shots completed along both the left and right shorelines of each section (totalling 10 replicates). Catches from boat electrofishing are compared against concurrent gill netting data to evaluate if adult abundance trends remain consistent across different gear types. Nighttime electrofishing mirrored daytime parameters but operated at a reduced effort of six shots per shoreline section. Because this expanded survey specifically targeted adult Macquarie perch, individuals under 150 mm TL were observed but not handled or counted; however, visual abundance estimates were recorded for YOY (< 85 mm TL) and juveniles (85–150 mm TL) for every shot.

Macquarie perch abundance data were standardised by calculating catch-per-unit-effort (CPUE) as the number of fish caught per net-hour. For gill net sampling, adult CPUE was analysed directly using raw catch counts with soak duration incorporated as a log-offset term within statistical models, rather than as a pre-calculated rate. This approach retains the count structure of the data and avoids assumptions inherent in pre-divided CPUE metrics. The effect of reservoir shoreline expansion on catch rates is interpreted at the inference stage: observed CPUE during the operational phase was similar to baseline despite an approximately four-fold expansion in available shoreline, consistent with a stable or growing adult population redistributed across a larger habitat. For fyke net sampling,

sampling effort was scaled proportionally with reservoir expansion to maintain representative coverage of available littoral habitats throughout the monitoring period.

Statistical analyses evaluating Macquarie perch abundance, body length, and condition across three river management phases (Baseline, Filling, and Operational) were conducted in R. Condition of adult Macquarie perch was analysed using Fulton's condition index, which is calculated as K = 100(weight/length^3) following (Ricker, 1975). ECR monitoring program body condition data (2014 onwards) was compared against historical data from 2009 (data from Lintermans et al., 2010). Adult abundance (gill net data) was modelled using negative binomial Generalised Additive Models (GAMs) incorporating a smooth term for year and a random effect for net deployment, with monitoring phase assessed via a complementary phase-only model and estimated marginal means. Young-of-year and juvenile abundance (fyke net data) were analysed within a Before–After Control–Impact (BACI) framework using Tweedie GAMs to accommodate zero-inflated data, with a significant site × phase interaction providing causal inference attributable to reservoir enlargement. Concurrently, changes in body length and condition were assessed using Gaussian Generalized Linear Mixed Models (GLMMs) incorporating an autoregressive [AR(1)] structure to account for temporal autocorrelation across sampling years Post-hoc pairwise contrasts for all models were evaluated using estimated marginal means with Tukey's adjustment. A comprehensive description of the statistical modelling, assumptions, and diagnostic procedures is provided in Appendix 1.

RESULTS

Adult Macquarie Perch

A total of 22 Macquarie perch were captured using all gill nets in ECR in 2026, ranging from 167 – 434 mm TL (Figure 5). Adult CPUE varied across the monitoring period, with a significant nonlinear temporal trend identified by the generalised additive model (s(year): edf = 2.87, χ^2 = 21.43, $p < 0.001$; Figure 6). Observed mean CPUE was 10.0 fish per 100 net-hours during the baseline period, rising to 34.1 during the filling phase, before returning to 10.0 during the operational phase. Adult CPUE was significantly elevated during the filling phase, approximately 3.15-fold above baseline levels (coefficient = 1.146 ± 0.248 SE, $p < 0.0001$), consistent with concentration of fish from inundating upstream riverine habitats into the expanding impoundment. In contrast, operational-phase CPUE was statistically indistinguishable from baseline (coefficient = −0.072 ± 0.181 SE, $p = 0.689$), indicating no overall change in catch rate relative to pre-enlargement conditions across the operational period as a whole.

Targeted comparisons of recent monitoring windows against baseline confirmed this pattern. CPUE in 2026 alone was approximately 0.76-fold that of baseline (i.e. marginally lower), though this difference was not statistically significant ($p = 0.548$). Similarly, CPUE during 2023–2025 was similar to baseline (1.09-fold; $p = 0.762$), and the most recent three-year window (2024–2026) showed no significant departure from baseline conditions (0.77-fold; $p = 0.400$) (Figure 6). Collectively, these comparisons indicate that adult abundance in the most recent monitoring years is broadly consistent with pre-enlargement catch rates.

The near-identical raw CPUE between baseline and operational phases, despite an approximately four-fold expansion in available reservoir shoreline, is consistent with a dilution effect in which a stable or growing adult population has redistributed across a substantially larger habitat. A total

abundance index, observed CPUE multiplied by contemporary shoreline length, was approximately 65 during baseline, 620 during filling, and 259 during the operational phase, suggesting that Cotter Reservoir may now support a larger adult population overall than under pre-enlargement conditions.

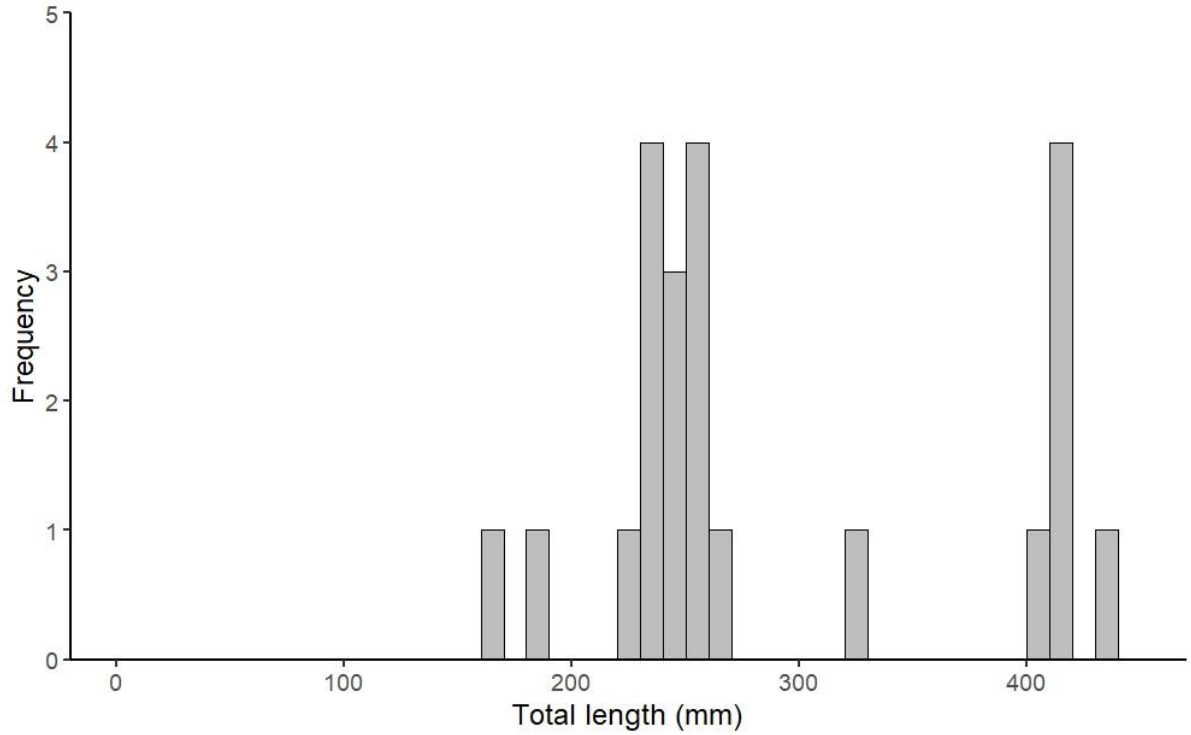

Figure 5. Length frequency of Macquarie perch captured from the Enlarged Cotter Reservoir in autumn 2026 using gill nets.

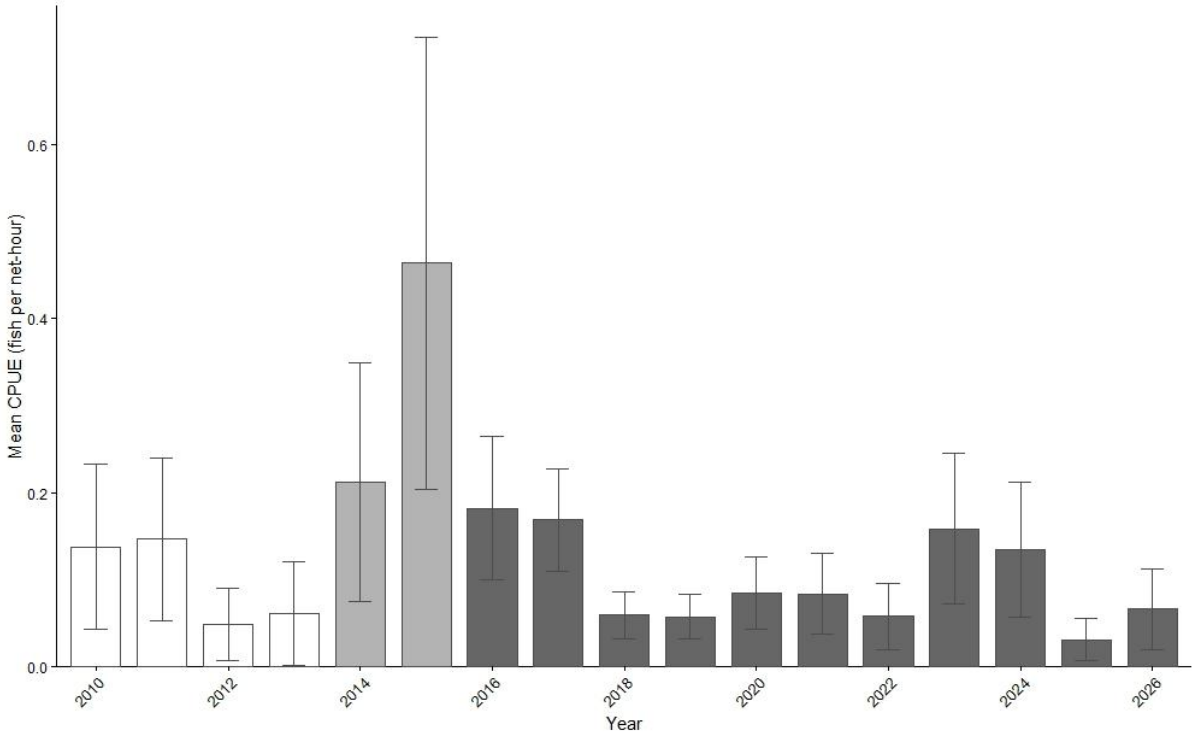

Figure 6. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of adult Macquarie perch captured using gill nets in Cotter Reservoir, 2010–2026. CPUE is expressed as fish per net-hour, standardised for variation in soak duration among net deployments. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Error bars represent 95% confidence intervals derived from among-net variation within each sampling year.

Autoregressive linear mixed models indicated that management phase was an important predictor of fish length, driven primarily by differences between the filling and operational periods. Average fish length was lowest during the filling phase, though this direct drop was not statistically significant after adjusting for multiple comparisons relative to the baseline phase ($p = 0.127$). Differences between the operational and baseline phases were also not statistically significant, with average lengths remaining highly stable between the two periods ($p = 0.997$). However, a statistically significant difference was detected between the filling phase and the operational phase ($p = 0.044$), demonstrating that Macquarie perch caught during the operational phase were significantly larger on average than those handled during the filling phase.

When evaluating finer annual scales, the year-specific model indicated that individual sampling year had a highly significant effect on body length (AIC = 6899.0), driven by distinct year-to-year fluctuations. The smallest average fish sizes were observed in filling phase years of 2014 and 2015, which were significantly smaller than the historical 2009 baseline ($p < 0.001$ for both years). Following these lows, average length steadily increased, peaking significantly in 2019, making fish in 2019 significantly larger on average than those caught in 2009 ($p < 0.001$), 2014, 2015, 2021, and 2023. Significant, sharp drops in average length were also observed later in the time series during 2021 and 2023, indicating periods where smaller individuals again dominated the catch (Figure 7).

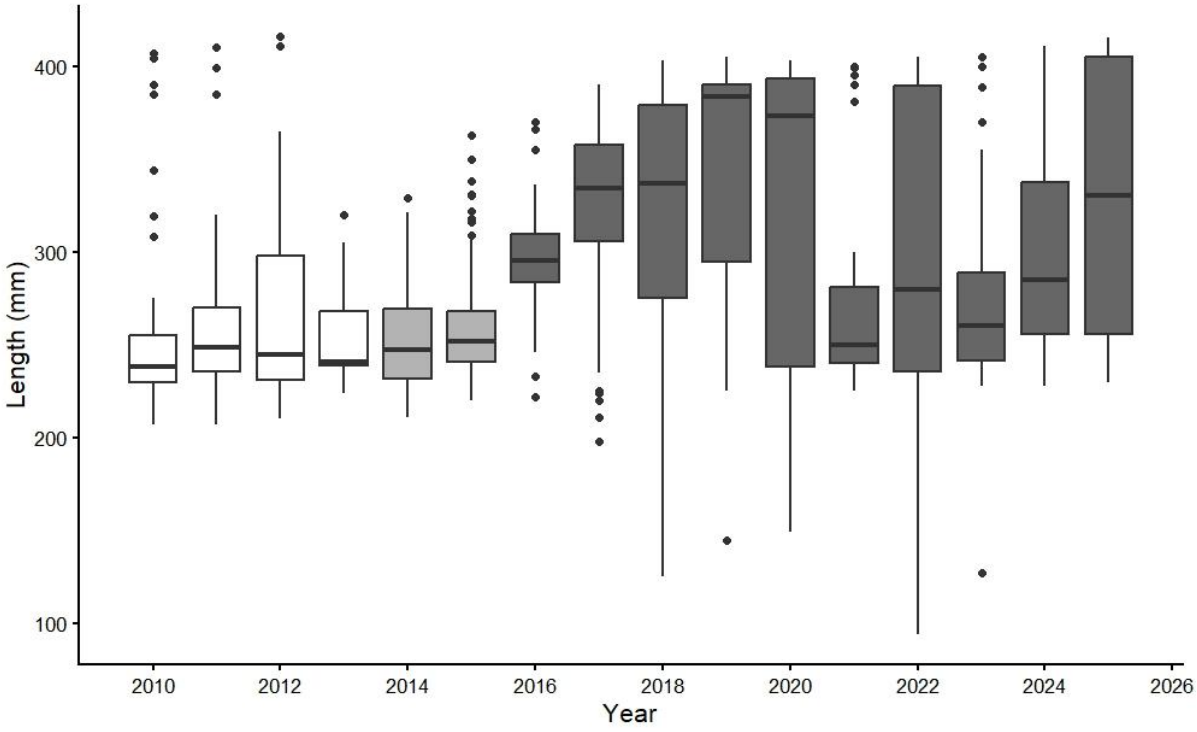

Figure 7. Box plot of length of adult Macquarie perch captured in gill nets in Cotter Reservoir from 2010 to 2026. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); plot arranged in chronological order from 2010 to 2026 from left to right on the x-axis.

Autoregressive linear mixed models indicated that management phase was an important predictor of fish condition, with Macquarie perch scoring significantly higher in Fulton's condition factor (K) during the filling phase relative to the baseline phase ($p = 0.044$) (Figure 8). In contrast, after adjusting for multiple comparisons, differences between the operational and baseline phases were not statistically significant ($p = 0.099$). No significant difference was detected between the filling and operational phases ($p = 0.732$), indicating the improved condition established during filling was maintained into the operational period. The model revealed an exceptionally high degree of temporal autocorrelation between consecutive sampling years (rho = 0.95), and accounting for this annual dependency significantly improved model fit over a standard random-intercept structure (delta AIC = 4.30).

When evaluating finer annual changes, the year-specific model showed highly significant fluctuations in fish condition factor across the time series (AIC = -9041.1). Average fish condition factor peaked significantly in 2017, which was significantly higher than the 2009 baseline ($p < 0.001$) as well as the 2014 and 2015 periods (Figure 8). Following this peak, a stark and significant population-wide decline in body condition occurred in 2020, dropping to an all-time monitoring low. This 2020 drop represented a significant decrease relative to the 2009 baseline ($p = 0.029$) and established 2020 as having significantly lower fish condition than 2014, 2015, 2016, and 2017 (Figure 8).

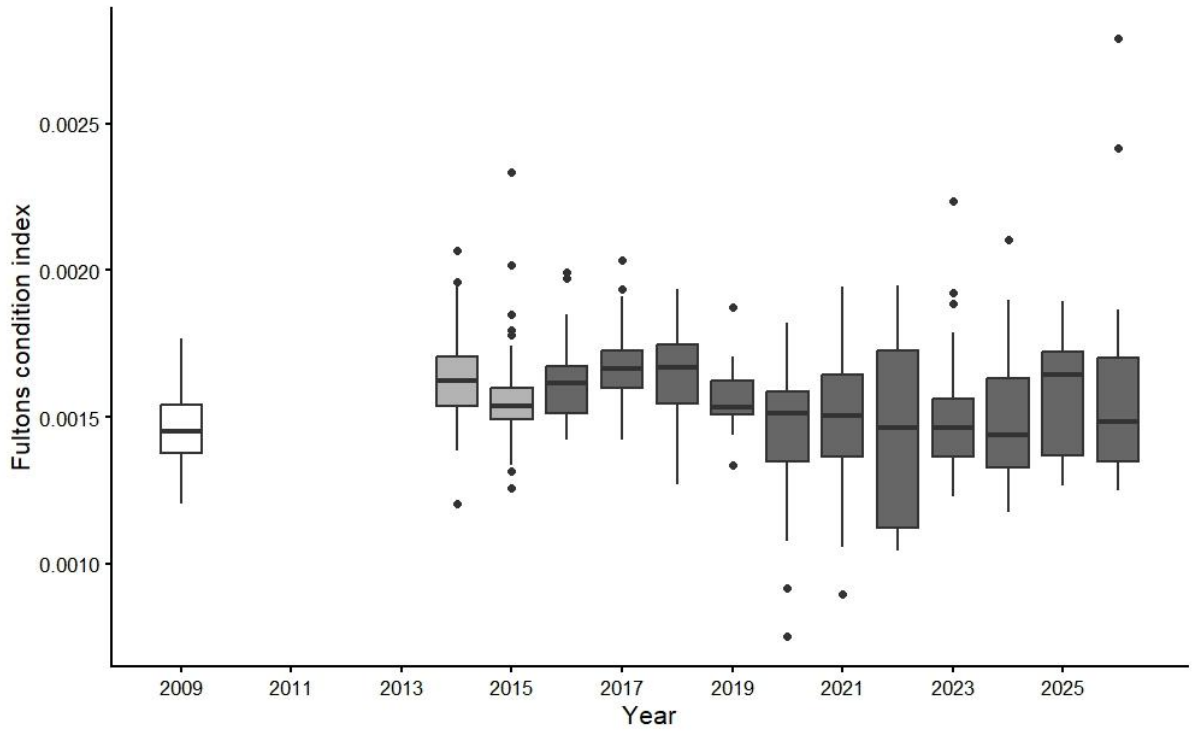

Figure 8. Box plot of condition (using Fulton's condition index) of adult Macquarie perch captured in gill nets in Cotter Reservoir from 2009 to 2026. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); plot arranged in chronological order from 2009 to 2026 from left to right on the x-axis.

Boat electrofishing

A total of 16 adult Macquarie perch were captured by boat electrofishing in 2026 ranging in lengths from 199 – 422 mm TL (Figure 9). Length frequency of individuals caught by electrofishing was dominated by individuals between 200 - 350 mm TL (Figure 9). Of these, five individuals were captured during the day and 11 individuals captured at night. Relative abundance of adult Macquarie perch captured by daytime boat electrofishing in 2026 was very low (Figure 10).

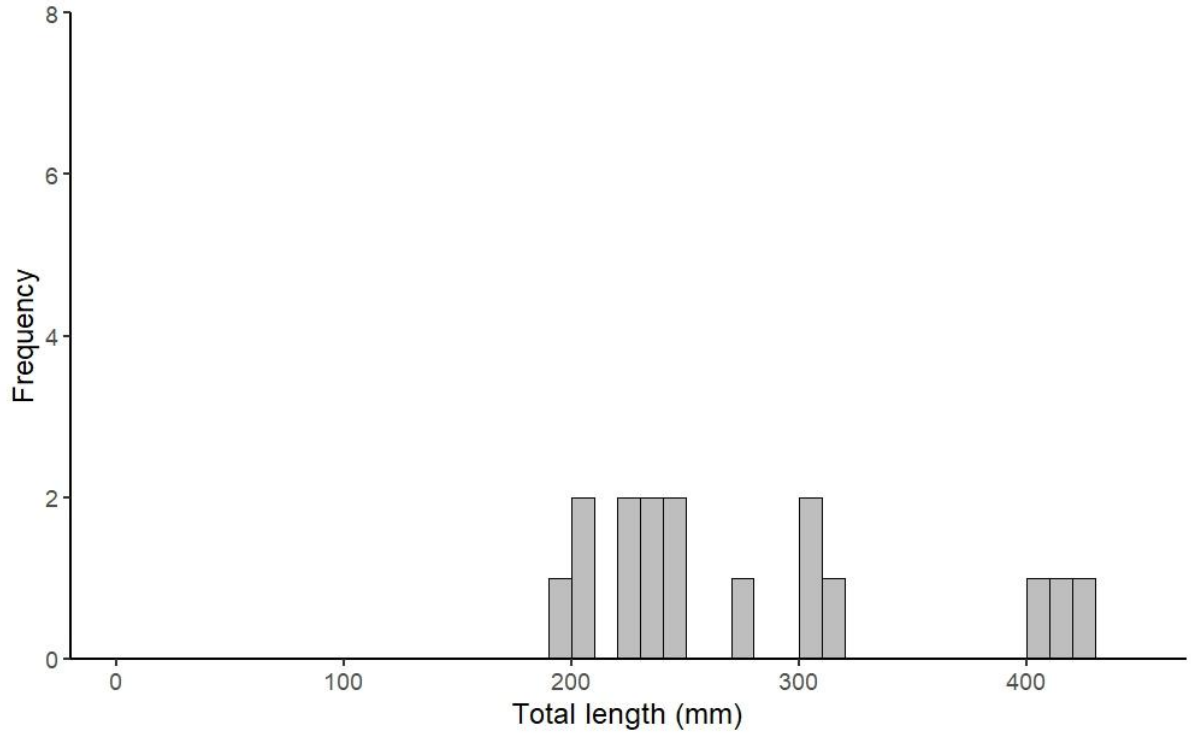

Figure 9. Length frequency of all Macquarie perch captured via boat electrofishing in Cotter Reservoir in 2026.

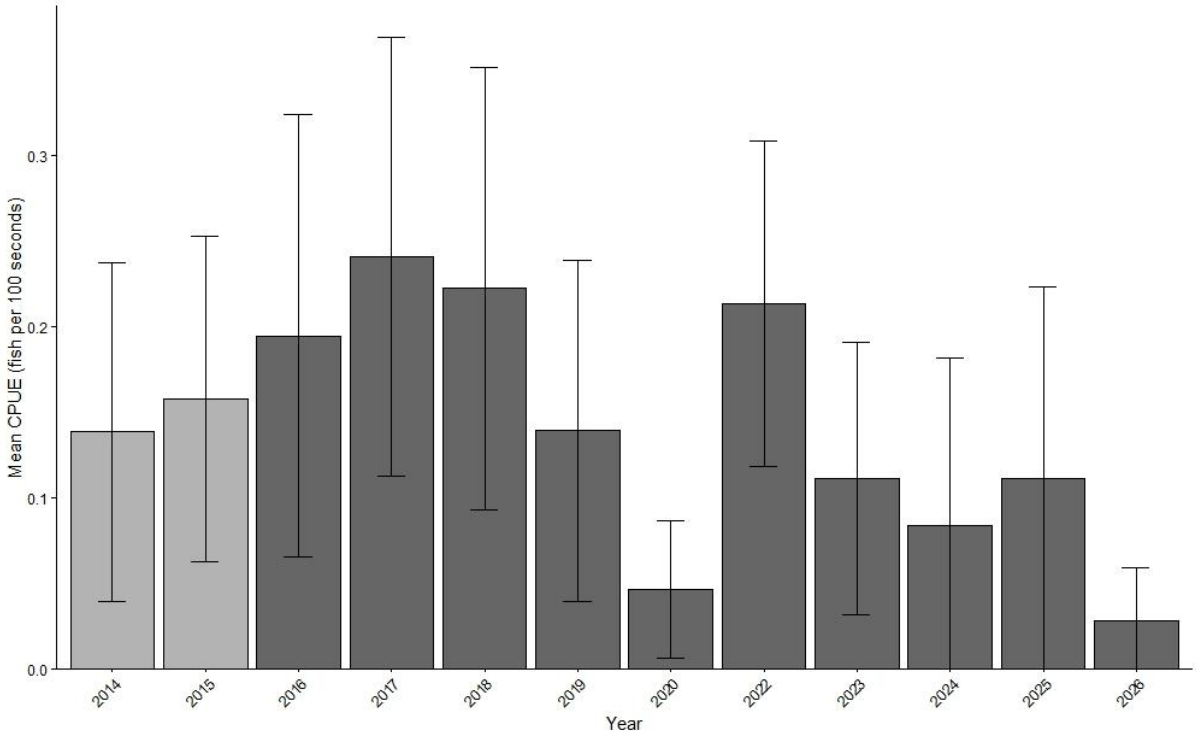

Figure 10. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Macquarie perch (*Macquaria australasica*) captured by boat electrofishing in Cotter Reservoir, 2014–2026. CPUE is expressed as fish per 100 seconds of electrofishing on-time, calculated from raw catch counts standardised for variation in effort among sampling runs across five reservoir sections. Sampling commenced in 2014 coinciding with reservoir filling; no baseline data are available for this method. No sampling was conducted in 2021. Light grey bars indicate the filling phase (2014–2015) and dark grey bars indicate the operational phase (2016–2026).

Fyke netting

A total of 208 Macquarie perch ranging from 41 – 320 mm TL were captured in 2026 from the ECR using fyke nets, comprising 51 YOY (< 85 mm), 7 juveniles (85 > 150 mm) and 150 sub-adults/ adults (> 150 mm) (Figure 11). Young-of-year and juvenile Macquarie perch abundance at Cotter Reservoir showed highly significant declines relative to baseline conditions, consistent across all recent monitoring windows (Figure 12). The BACI models identified significant site × phase interactions for both life stages, confirming that observed declines at Cotter Reservoir were attributable to reservoir enlargement rather than regional environmental variability.

For YOY abundance, pairwise BACI contrasts indicated that the ratio of Cotter to Kissops CPUE was significantly lower during the operational phase relative to baseline (ratio = 31,250,623; $p < 0.0001$), demonstrating persistent and severe suppression of recruitment at the impact site (Figure 12). Targeted comparisons of recent monitoring windows confirmed this pattern: YOY CPUE in 2026 alone was approximately 0.08-fold that of baseline (i.e., ~12.5-fold lower; $p < 0.0001$), CPUE during 2023–2025 was approximately 0.06-fold baseline (~16.5-fold lower; $p < 0.0001$), and the most recent three-year window (2024–2026) remained at approximately 0.07-fold baseline (~14.3-fold lower; $p <$

0.0001). No significant change in YOY abundance relative to baseline was detected at the Kissops reference site in 2026 (p = 0.198) or over 2024–2026 (p = 0.978), confirming the suppression at Cotter Reservoir is site-specific rather than a regional signal.

For juvenile abundance, suppression at Cotter Reservoir was similarly persistent, though somewhat less severe than for YOY, consistent with the legacy cohort buffering effect in which individuals recruited prior to reservoir filling continue to contribute to juvenile catches in early operational years. CPUE in 2026 was approximately 0.22-fold baseline (~4.5-fold lower; $p < 0.0001$), CPUE during 2023–2025 was 0.45-fold baseline (~2.2-fold lower; $p < 0.0001$), and the 2024–2026 window was 0.34-fold baseline (~2.97-fold lower; $p < 0.0001$). The less severe but still highly significant juvenile suppression is consistent with the expected temporal lag between YOY recruitment failure and its propagation through to the juvenile size class. Collectively, these results demonstrate that recruitment in the Cotter Reservoir Macquarie perch population has remained severely and persistently limited throughout the operational phase, with no evidence of recovery in any recent monitoring window.

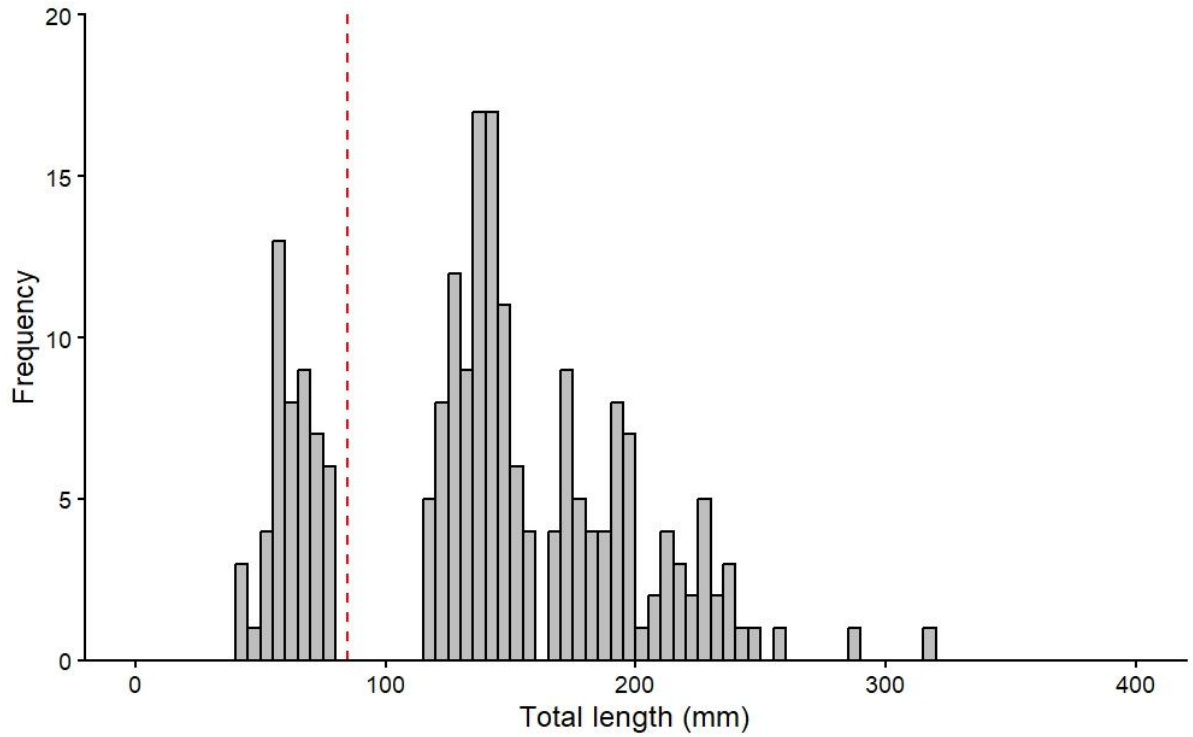

Figure 11. Length frequency of Macquarie perch captured from the ECR in autumn 2026 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

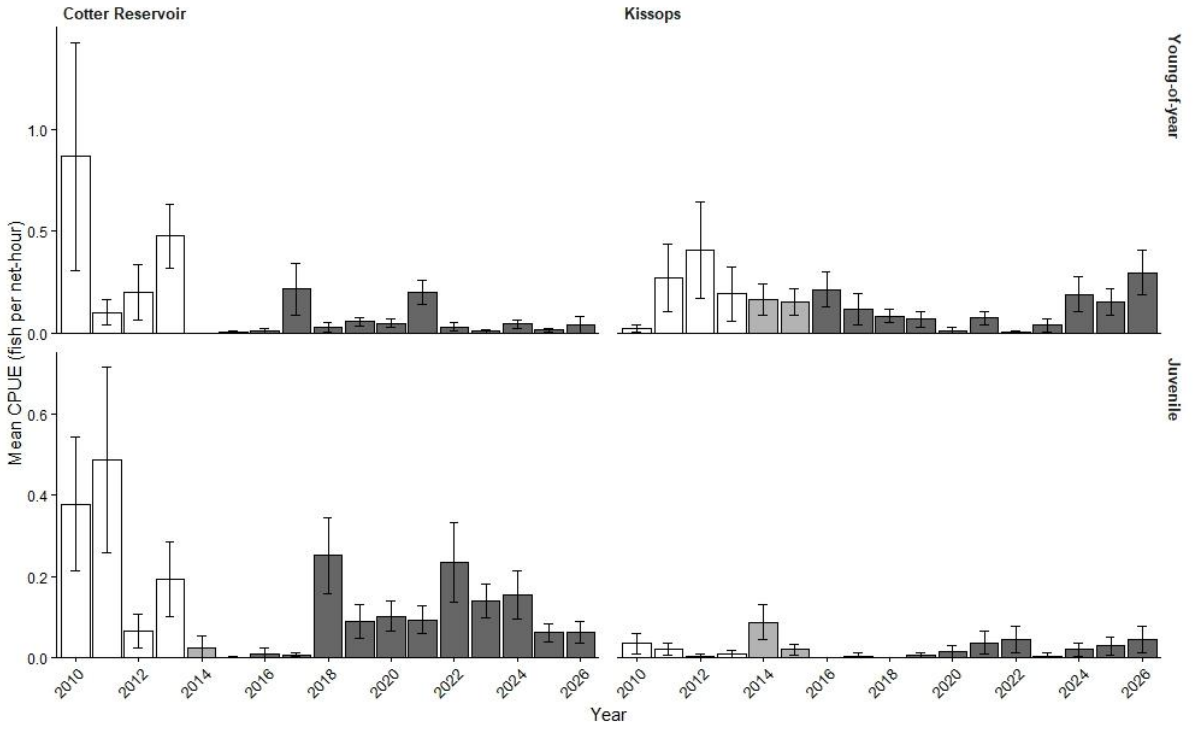

Figure 12. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of juvenile (≥ 85 mm TL; upper panels) and young-of-year (< 85 mm TL; lower panels) Macquarie perch captured using fyke nets at Cotter Reservoir (impact site; left panels) and Kissops Flat (reference site; right panels), 2010–2026. CPUE is expressed as fish per net-hour, standardised for variation in soak duration among net deployments. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Error bars represent 95% confidence intervals derived from among-net variation within each sampling year.

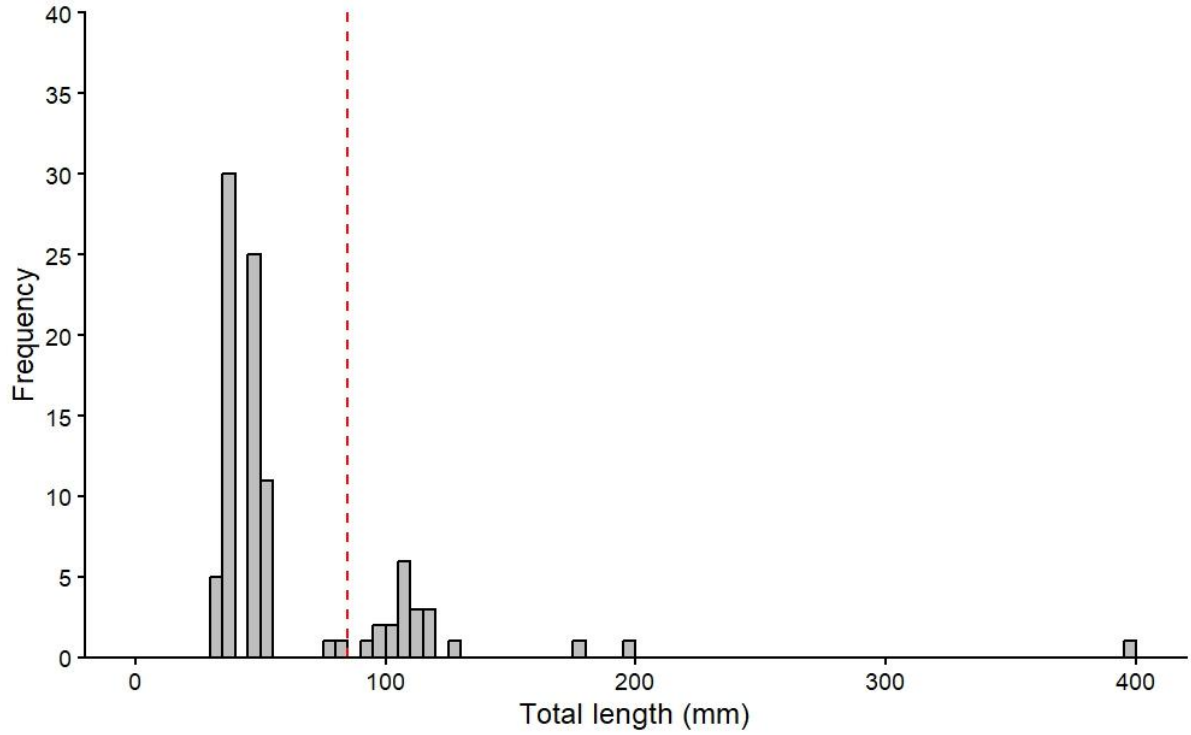

Figure 13. Length frequency of Macquarie perch captured from Kissops Flat (Reference site) on the upper Murrumbidgee River in 2026 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

Snorkelling

No juvenile Macquarie perch were observed in the four pools, however, groups of 5-10 were common across a 75 m stretch at the headwaters of the dam.

DISCUSSION AND CONCLUSIONS

A size-structured pattern emerged across models evaluating the Macquarie perch population in the Enlarged Cotter Reservoir (ECR). Strong Before–After Control–Impact (BACI) effects were evident in early life stages, with recruitment declining during the filling phase and a subsequent lagged reduction in juvenile abundance. These changes were initially masked in aggregate CPUE, as adult persistence buffered declines in younger cohorts. This demonstrates that aggregate metrics can obscure ecologically important shifts, reinforcing the value of size-structured analyses.

Recruitment (YOY) has remained low in most years since filling, with recent monitoring (2024–2026) confirming sustained suppression relative to baseline. However, the occurrence of several years of moderate recruitment indicates that the system retains functional reproductive capacity. This suggests a conditional recruitment bottleneck, where recruitment succeeds intermittently rather than being uniformly constrained.

While stochastic events such as high-discharge pulses likely contribute to recruitment failure by displacing eggs and larvae (Tonkin et al., 2017), these do not fully explain the long-term pattern. Instead, recruitment appears governed by interacting constraints that limit success to years when conditions align. Macquarie perch are obligate lotic spawners requiring flowing water and clean cobble substrates (Koehn et al., 2020), and reservoir fish typically migrate only short distances upstream (< 1 km; Broadhurst et al., 2019b, Broadhurst et al., 2020). Although reservoir enlargement greatly expanded foraging habitat, the extent of suitable riverine spawning habitat has remained unchanged. Consequently, the data supports a spatial and functional mismatch, where an expanded adult population relies on a limited and environmentally sensitive spawning reach. Rather than a fixed habitat ceiling, the effective availability of spawning and early-rearing habitat is variable, depending on flow regime, reservoir level, and connectivity. Recruitment is therefore best described as a dynamic early-life-stage bottleneck, with successful cohorts produced only under favourable conditions.

In contrast, adult Macquarie perch have remained abundant and in good body condition throughout the monitoring period. Raw CPUE during the operational phase was similar to pre-enlargement baseline levels; however, given the approximately four-fold expansion of available reservoir shoreline following enlargement (~6.5 km to ~25.9 km), this stable catch rate is most parsimoniously interpreted as a dilution effect rather than stasis in total numbers. A total abundance index, observed CPUE multiplied by contemporary shoreline length, was approximately four-fold higher during the operational phase than at baseline, suggesting the reservoir now likely supports a larger adult population in total than the pre-enlargement system. Stable or improved body condition throughout the operational phase confirms that the enlarged reservoir provides a favourable environment for adult fish.

The divergence between strong adult performance and persistent recruitment failure indicates that adult fecundity is unlikely to be limiting population recovery; instead, processes affecting early life stages dominate population dynamics. Adults are thriving in the expanded lentic environment while the lotic spawning habitats required for successful reproduction remain spatially constrained and hydrologically variable upstream of the dam. While occasional successful recruitment cohorts - evident as episodic YOY detections in some operational years - reduce immediate extinction risk, the overall trajectory may become consistent with demographic buffering: a population that persists through the longevity of its adults while failing to replace them. If recruitment remains insufficient over the long term, this pattern is consistent with the concept of extinction debt, whereby apparent population stability masks an underlying trajectory toward decline (Kuussaari et al., 2009, Lackmann et al., 2024).

The divergence between adult and juvenile trajectories also highlights the importance of complementary sampling approaches. The low adult CPUE observed in 2025 was likely an artifact of elevated water temperatures (> 20 °C) reducing catchability, consistent with Thiem et al. (2012). The rebound in 2026, supported by deep-drop gill nets, reinforces the need to prioritise sampling under appropriate thermal conditions. Given the importance of early life stages, population assessment should extend beyond abundance to include genetic considerations. Previous work has identified reduced genetic diversity and signs of inbreeding (Farrington et al., 2014, Pavlova et al., 2024). These risks are likely exacerbated by variable recruitment and a limited effective population size associated with the constrained spawning domain. Routine genomic monitoring is therefore important to

evaluate the long-term success of genetic management interventions under these conditions. Finally, triennial intensive mark–recapture population estimates remain a cornerstone of the monitoring framework, ensuring that a sufficient breeding population is maintained while providing robust detection of demographic change with minimal cumulative stress on adults (Cantin and Post, 2018).

The Macquarie perch population in the ECR is characterised by high adult abundance and condition, juxtaposed against suppressed and highly variable recruitment. While the system retains the capacity for successful recruitment, this occurs infrequently, indicating a condition-dependent bottleneck affecting early life stages.

Reservoir enlargement has substantially increased foraging habitat without increasing the reliability of spawning and early-rearing habitat, creating a mismatch that limits the frequency of successful recruitment events. While adult longevity currently buffers overall population metrics, continued reliance on intermittent recruitment may pose a long-term risk if favourable recruitment conditions occur infrequently.

The ACT Water Resources Environmental Flow Guidelines 2026 (DI2026–11) include a short-term ecological outcome indicator for Macquarie perch in Cotter Reservoir: a minimum total catch of three Macquarie perch per fyke net night per year, comprised of greater than 50% individuals less than 150 mm TL. This indicator is used here as a contextual reference point against which monitoring program findings are situated. Performance in relation to this indicator across the full 17-year monitoring period is presented in the inset table.

Inset table: Macquarie perch fyke net catch at Cotter Reservoir (2010–2026) in relation to the ACT Environmental Flow Guidelines 2026 short-term ecological outcome indicator (≥3 fish per net night, >50% individuals <150 mm TL). Green shading indicates the indicator is met; red indicates it is not met. Filling phase years (2014–2015) are shown in red as the indicator was not met, noting that low catches during active reservoir inundation reflect filling dynamics rather than stable operational conditions.

Sampling year	Phase	Mac. perch captured	Net nights	Fish per net night	% <150 mm	EFG indicator met
2010	Baseline	525	22	23.86	82.6%	**Yes**
2011	Baseline	249	22	11.32	86.7%	**Yes**
2012	Baseline	105	22	4.77	91.7%	**Yes**
2013	Baseline	257	22	11.68	96.1%	**Yes**
2014	Filling	93	36	2.58	15.1%	**No**
2015	Filling	32	36	0.89	15.6%	**No**
2016	Operational	22	36	0.61	68.2%	**No**
2017	Operational	266	60	4.43	93.2%	**Yes**
2018	Operational	306	60	5.10	98.0%	**Yes**
2019	Operational	262	60	4.37	64.9%	**Yes**
2020	Operational	295	60	4.92	59.7%	**Yes**
2021	Operational	517	60	8.62	68.1%	**Yes**
2022	Operational	364	60	6.07	86.8%	**Yes**
2023	Operational	269	60	4.48	68.0%	**Yes**
2024	Operational	316	60	5.27	75.6%	**Yes**
2025	Operational	158	60	2.63	60.8%	**No**
2026	Operational	208	60	3.47	60.6%	**Yes**

[1] EFG ecological outcome indicator: ≥3 fish per net night AND >50% individuals <150 mm TL (ACT EFG Guidelines 2026, Table 4). Used here as a contextual reference point, not a formal compliance determination. Note that low catches in filling phase years (2014–2015) reflect reservoir inundation dynamics rather than stable operational conditions.

Catch rates were high during the baseline phase (mean 12.91 fish per net night across 22 net nights) before collapsing during reservoir filling, when active inundation disrupted spawning habitat across three consecutive spawning seasons (2013/14–2015/16). Recovery from 2017 onwards coincides with reservoir stabilisation at full supply level, though operational-phase catch rates (mean 5.06 fish per net night across 60 net nights) remain well below baseline despite nearly three times the sampling effort. The EFG indicator of ≥3 fish per net night is met in most operational years but sits at only 23% of mean baseline catch rates, and the two most recent years (2025: 2.63; 2026: 3.47) are

approaching the threshold from below, a trajectory the BACI analysis confirms reflects persistent recruitment suppression rather than normal inter-annual variation.

The monitoring program findings suggest that the EFG ecological outcome indicator of ≥3 fish per net night, while useful as a minimum detection threshold, does not capture the magnitude of the decline in reservoir population productivity relative to pre-enlargement conditions. The current indicator sits at approximately 23% of mean baseline catch rates (12.91 fish per net night) and is therefore more accurately characterised as a minimum viability signal than a meaningful recovery benchmark. Baseline fyke net catch rates regularly exceeded 10 fish per net night during productive years, and an indicator calibrated to ecologically meaningful productivity would more likely sit in the range of 8–10 fish per net night. Under an indicator of this magnitude, the reservoir population would not have met the threshold in any operational-phase year with the exception of 2021, a considerably more accurate reflection of population trajectory than the current indicator implies.

These observations are offered as contextual information to inform the next EFG Guidelines review cycle, in which the monitoring program dataset, now spanning 17 years with a robust BACI framework, provides a strong empirical basis for revisiting the ecological outcome indicators for Macquarie perch in Cotter Reservoir. It is suggested that the ACT Threatened Species Recovery Team and EPA be engaged in any such review.

We recommend that future management shift from passive monitoring to a recruitment-focused strategy aimed at increasing the frequency of successful recruitment years. This includes identifying and supporting conditions that enhance spawning success and larval survival within the reservoir–river ecotone, alongside maintaining routine genomic surveillance to evaluate population resilience. By integrating genetic monitoring with existing mark–recapture approaches, management can ensure that the population retains sufficient adaptive capacity while navigating ongoing recruitment variability. Ultimately, long-term persistence will depend on actively supporting key life-history processes rather than relying on adult population stability alone.

RECOMMENDATIONS

Monitoring and Methodological

Standardised sampling protocols: Maintain the current enhanced effort for adult monitoring, including the extra 125mm gill nets, alongside nocturnal boat electrofishing.

Temperature-Dependent timing: Future adult gill netting must commence only after water temperatures consistently fall below 20 °C to minimise sampling bias.

Continued juvenile and larval monitoring: Maintain high-effort fyke netting for YOY and snorkel surveys for larval detection, as primary indicators of annual recruitment success.

Triennial population estimates: Conduct formal, intensive adult population estimates every three years. The previous estimate was undertaken in 2023, so this is required next monitoring cycle (i.e. autumn 2027) This includes an extra two nights of gill netting, three nights of fyke netting, and two days/nights of boat electrofishing for recapture of marked individuals (captured and marked during routine monitoring effort).

Integrated genomic surveillance: Implement routine genetic health assessments (e.g. five-year cycle). Building on established baselines (Farrington et al., 2014, Pavlova et al., 2024), these assessments are essential to monitor the success of ongoing genetic rescue translocations and detect the resurgence of inbreeding depression.

Management and mitigation

Prioritising recruitment through flow and operational stability
Macquarie perch recruitment remains severely suppressed, representing the primary ongoing risk to population persistence. The primary objective is to increase the frequency of successful recruitment events by providing stable, high-quality conditions for spawning and early development. A two-stage flow management strategy is proposed targeted at the key spawning reach immediately upstream of the ECR:

- Pre-spawning habitat maintenance (September):
  Implement targeted high-flow pulses to "clean" key riffle habitats by removing fine sediments and periphyton, maintaining the clean cobble substrates required for egg deposition and development.

- Recruitment-window stability (October–December):
  Following habitat preparation, flows should transition to a stable regime during the critical spawning and early life-history period. Flows must balance provision of adequate discharge for passage and spawning cues with minimal variability to avoid scouring or dewatering of spawning habitats and to reduce displacement of eggs and larvae.

Operational connectivity
Reservoir levels should be managed to maximise connectivity between the reservoir and upstream spawning habitats. In particular, water levels should avoid exposing natural instream barriers within the historical river channel during the spawning period, ensuring the greatest possible extent of accessible riverine habitat.

Invasive predator mitigation (targeted recruitment protection)

Persistent recruitment failure represents the primary threat to the long-term viability of the Cotter Reservoir Macquarie perch subpopulation. While flow management during the December–January spawning and early development window addresses the abiotic constraint on recruitment success, predation by resident salmonids in the reservoir headwaters and immediately upstream river reach represents a concurrent biotic risk that warrants direct management action.

A three-year precautionary predator removal trial is recommended, targeting Rainbow and Brown trout within the reservoir headwaters and the first kilometre of Cotter River immediately upstream during December–January each year. This zone encompasses the critical early settlement and development habitat and, being within the existing recreational fishing closure, supports no

recreational angling pressure — meaning resident trout in this area experience no natural offtake and represent a standing predation risk to Macquarie perch during the most vulnerable life-history window.

Removal will be conducted via targeted angling, which offers high selectivity for salmonids with negligible disturbance to adult Macquarie perch, spawning habitat, or other non-target species. Effort should be standardised to a defined number of angler-hours per survey occasion to allow consistent comparison across years. All captured trout should be examined for stomach contents by visual inspection in the field, with stomach material retained for eDNA metabarcoding to detect highly digested early-stage fish remains not identifiable by visual means alone.

The stomach content and eDNA program operates as the evaluative arm of the trial rather than a prerequisite for action — providing evidence to refine spatial and temporal targeting over the three-year period and to assess the degree to which predation on early life stages is occurring within the removal zone. Outcomes of the trial should be assessed in conjunction with YOY fyke net CPUE at Cotter Reservoir and flow conditions during the December–January window, recognising that recruitment in Macquarie perch is episodic and that three years of data across varying hydrological conditions will be required before meaningful conclusions can be drawn.

This trial directly complements the flow management strategy recommended under Question 2, together representing a two-pronged approach to maximising recruitment potential in the Cotter Reservoir subpopulation during the period of greatest ecological leverage.

Contingency for genetic supplementation

If recruitment remains insufficient to sustain effective population size, continuation of an adaptive genetic supplementation (genetic rescue) program is recommended. Introducing genetically diverse juveniles will help maintain genetic diversity and mitigate long-term risks associated with low recruitment frequency, while underlying habitat and recruitment constraints are addressed.

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 the Vanitys Crossing fishway in 2001 (Ebner and Lintermans, 2007) enabled the Macquarie perch (*Macquaria australasica*) population to expand its distribution upstream of the road crossing (Broadhurst et al., 2012, Broadhurst et al., 2013, Broadhurst et al., 2015, Broadhurst et al., 2016b). Subsequently, remediation of the fish passage barrier at Pipeline Road Crossing (ACTEW Corporation, 2009) was designed to open up additional upstream spawning habitat. This remediation served as an environmental offset to compensate for the inundation of existing Macquarie perch spawning habitat caused by the Enlarged Cotter Reservoir (ECR) (ACTEW Corporation, 2009). The successful, long-term expansion of the Macquarie perch population past Pipeline Road Crossing relies heavily on the continued efficacy of the Vanitys Crossing fishway; without it, reservoir fish are largely blocked from upstream riverine migration. Ongoing monitoring is required to determine the success of fish passage remediation at both Vanitys and Pipeline Road crossings, and to assess how improved habitat access affects the riverine population. Ultimately, expanding the distribution of riverine Macquarie perch decreases the population's vulnerability to localised extinctions driven by 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 3). 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.

The sampling design for Question 2 adapts the baseline monitoring framework established for Question 8 (Lintermans et al., 2013), with two structural modifications:

- **Site Substitution:** The "Bracks Hole" site, located immediately above the old Cotter Reservoir, has been inundated by the ECR and is no longer a riverine environment. A new riverine site located between the ECR full supply level and Vanitys Crossing ("U/S ECR") was monitored as a substitute.
- **Site Omission:** The site immediately downstream of Bendora Dam was removed from the monitoring program, as it is unlikely to be directly affected by ECR operations.

Table 3. Outline of the sampling design for Question 2 of the fish monitoring program.

Feature	Detail
Target species and life history phase	Macquarie perch. Adults / sub-adults (> 150 mm TL), Juveniles (80 to 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 the ECR 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 captured Macquarie perch.
Data analysis	YOY, juvenile, and total Macquarie perch abundance (fyke net raw counts + log[soak duration] offset) were analysed within a BACI framework (Cotter River impact sites vs Kissops Flat reference) using Tweedie GAMs with site-type × phase interaction, site-specific temporal smooths, and random effects for site and net deployment. U/S ECR was excluded from primary models due to a three-phase sampling confound (inflated baseline, barrier-suppressed filling, net-relocation-deflated operational). Additional two-level GAMs compared recent windows (2026, 2023–2025, 2024–2026) against baseline. Sensitivity analysis including U/S ECR confirmed primary results are conservative.

TARGET SPECIES AND LIFE STAGE

Adult / sub-adult, juvenile, and young-of-year Macquarie perch were targeted. Individuals were classified into cohorts based on total length (TL) to maintain consistency with long-term monitoring benchmarks:

- **Adult / sub-adult (> 150 mm TL):** Based on results from Ebner and Lintermans (2007) who found that males are sexually mature from this size.
- **Juveniles (80 to 150 mm TL).**
- **Young-of-year (< 80 mm TL):** Based on late summer/early autumn river collection data compiled between 2010 and 2021.

SAMPLING METHODS AND NUMBER OF REPLICATES

Fyke netting and backpack electrofishing were employed to monitor riverine sites for Macquarie perch:

- **Fyke Netting:** Single-winged fyke nets (12 mm stretch mesh) were set in pools overnight (~16-hour soak time). Twelve fyke nets were deployed per site (three nets per pool across four pools), with the exception of Vanitys crossing and Kissops Flat where all 12 fyke nets were set within a single large pool.
- **Backpack Electrofishing:** Electrofishing was conducted across four 30 m sections in wadeable portions of each site (depths less than 0.8 m). This technique was omitted at the Kissops Flat reference site due to budget constraints and historically low capture rates using this method at that location.

TIMING

Sampling for this question was undertaken in February and March 2026 (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

Macquarie perch abundance in the Cotter River was assessed within a Before–After Control–Impact (BACI) framework comparing multiple Cotter River impact sites against a reference site (Kissops Flat, upper Murrumbidgee River) across three monitoring phases: baseline (2010–2013), filling (2014–2015), and operational (2016–2026). Analyses were conducted separately for young-of-year (YOY; <85 mm TL), juvenile (85–150 mm TL), and total combined catch. Raw fyke net counts were modelled using Tweedie GAMs with a log-offset for soak duration, with site type, phase, and their interaction as fixed effects, site-specific temporal smooths, and random effects for monitoring site and net deployment. The U/S ECR site was excluded from primary BACI models due to a three-phase sampling confound — baseline catches were inflated by proximity to the old reservoir spawning ecotone, filling-phase catches were suppressed by natural rock barriers preventing adult egress, and operational catches were deflated by net relocation outside the active spawning reach — rendering pre- and post-filling CPUE at this site non-comparable. A sensitivity analysis including U/S ECR confirmed the primary analysis is conservative. Additional models compared recent monitoring

windows (2026, 2023–2025, 2024–2026) against baseline. A comprehensive description of model structure, assumptions, and diagnostics is provided in Appendix 1.

RESULTS

General

A total of 109 Macquarie perch were captured using both fyke netting and backpack electrofishing (includes the extended effort applied for capturing trout) ranging from 43 – 200 mm TL (Figure 14). Young-of-year and juveniles were detected at three and five of the five monitored sites, respectively (Figure 14).

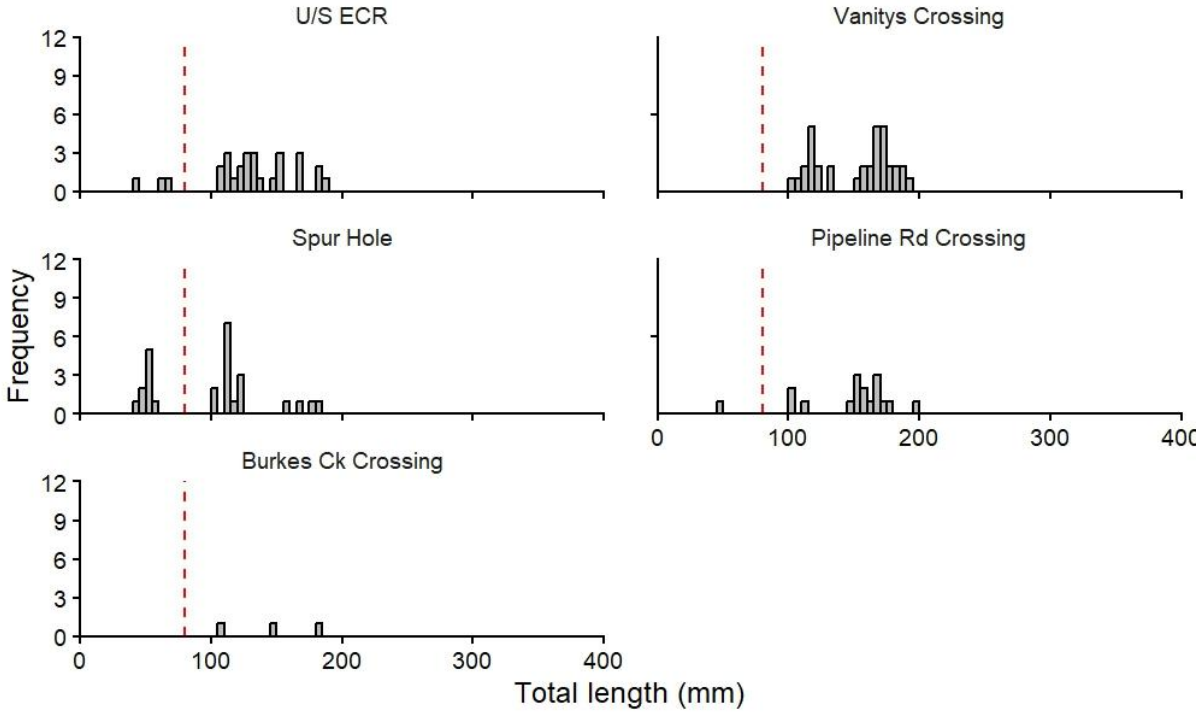

Figure 14. Length frequency of Macquarie perch captured in fyke nets and backpack electrofishing (including extended effort) from Cotter River in 2026 at sites; U/S ECR, Vanitys Crossing, Spur Hole, Pipeline Road Crossing and Burkes Creek Crossing (red dashed line indicates cut-off for length of young-of-year individuals < 80 mm TL).

Fyke net captures – All Sizes Combined

Total CPUE (YOY and juvenile combined) exhibited significant spatial and temporal variation, with the BACI analysis identifying a significant interaction during the filling phase relative to baseline (ratio = 3.40, t = 1.98, p = 0.048) and a highly significant filling vs operational contrast (ratio = 0.108, t = −3.47, p < 0.001), indicating continuing divergence between impact and reference sites through the operational period. The baseline vs operational contrast was not significant (p = 0.292),

reflecting the offsetting effects of elevated juvenile catches against suppressed YOY catches at impact sites when both stages are combined.

Recent-window comparisons showed consistent and significant suppression of total CPUE at Cotter River impact sites relative to baseline: 1.7-fold lower in 2026 ($p = 0.041$), 2.0-fold lower over 2023–2025 ($p < 0.001$), and 1.9-fold lower over 2024–2026 ($p < 0.001$). Observed total CPUE at impact sites declined from 0.105 fish per net-hour at baseline to 0.057 in 2026. Kissops Flat total catches were significantly lower than baseline over 2023–2025 ($p = 0.002$), consistent with the lower YOY catches at the reference site in that period, but were not significantly different from baseline in 2026 ($p = 0.184$) or over 2024–2026 ($p = 0.819$).

Fyke net captures – Young-of-year

Young-of-year Macquarie perch CPUE was characterised by frequent zero catches and high interannual variability across all sites, with episodic recruitment events rather than consistent annual production (Figure 15). Significant nonlinear temporal trends were detected at both Cotter River impact sites and the Kissops Flat reference site (s(year): $p < 0.0001$ for both groups; deviance explained = 50.5%).

The BACI analysis revealed highly significant site_type × phase interactions, indicating that temporal change at Cotter River impact sites diverged substantially from the reference site across both the filling and operational phases (baseline/filling: ratio = 2.94×10^{-4}, $t = -4.26$, $p < 0.0001$; baseline/operational: ratio = 1.77×10^{-6}, $t = -4.81$, $p < 0.0001$). In practical terms, the ratio of impact-site to reference-site YOY catch declined by several orders of magnitude relative to baseline conditions during both post-filling phases, demonstrating that Cotter River impact sites experienced a disproportionate reduction in YOY abundance relative to the reference. This result was robust to the inclusion of the U/S ECR site in sensitivity analysis, where BACI contrasts strengthened rather than weakened, confirming the primary analysis is conservative.

Observed mean YOY CPUE at impact sites declined from 0.024 fish per net-hour during baseline to 0.002 in 2026. Targeted comparisons against baseline confirmed persistent suppression at impact sites: YOY CPUE in 2026 was approximately 18-fold lower than baseline ($p = 0.002$); over 2023–2025 it was approximately 2.5-fold lower ($p = 0.005$); and over 2024–2026 approximately 2.7-fold lower ($p = 0.003$). No significant decline relative to baseline was detected at Kissops Flat in 2026 ($p = 0.231$) or over 2024–2026 ($p = 0.811$), and YOY catches at Kissops Flat remained at or above baseline levels in 2026 (0.291 fish per net-hour vs 0.228 at baseline). Note that Kissops Flat YOY CPUE was significantly lower than its own baseline over 2023–2025 ($p = 0.004$), suggesting some regional environmental variation in that period; the concurrent suppression at impact sites was nevertheless of markedly greater magnitude.

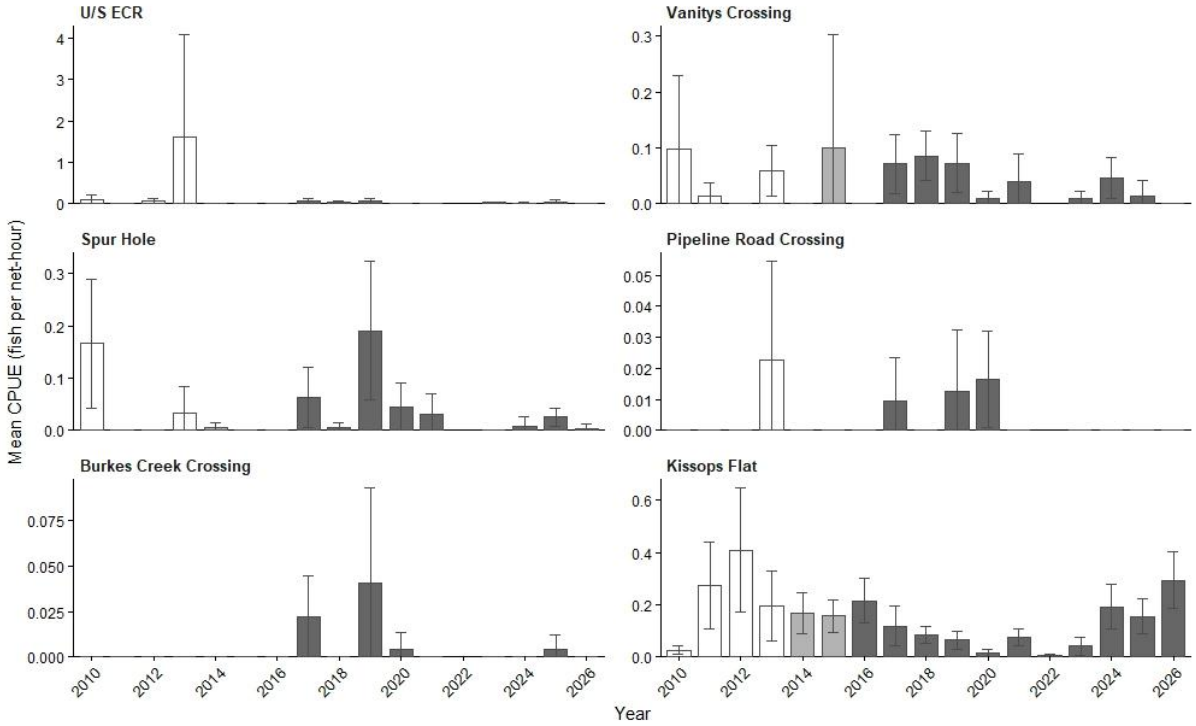

Figure 15. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of young-of-year Macquarie perch (<85 mm TL) captured using fyke nets at six monitoring sites in the Cotter River and Kissops Flat (upper Murrumbidgee River), 2010–2026. CPUE is expressed as fish per net-hour, standardised for variation in soak duration among net deployments. Note that Bracks Hole (sampled 2010–2013) was replaced by U/S ECR (sampled 2014–2026) as the most downstream riverine site. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Bars are arranged chronologically from left to right within each panel. Error bars represent 95% confidence intervals derived from among-net variation within each sampling year. Cotter River sites were not sampled in 2022 due to high river flows.

Fyke net captures – Juveniles

The BACI interaction for juveniles showed a different pattern from YOY. Impact sites had significantly higher juvenile catch rates than the reference site relative to baseline during both filling (ratio = 94.3, t = 3.14, p = 0.002) and operational phases (ratio = 1025, t = 3.08, p = 0.002) (Figure 16). This reflects the very low juvenile baseline at Kissops Flat (0.016 fish per net-hour), which was more variable and occasionally near-zero during the operational phase, while Cotter River impact sites maintained sporadic juvenile catches driven by pre-filling cohorts propagating through successive size classes. The filling vs operational contrast was not significant (p = 0.129), suggesting juvenile catch rates at impact sites relative to reference did not diverge further during the operational period beyond the initial shift.

Comparisons against the impact sites' own baseline conditions reveal an emerging decline. Juvenile CPUE at impact sites was not significantly different from baseline in 2026 alone (p = 0.156), but was significantly lower over 2023–2025 (~1.9-fold lower; p = 0.006) and over 2024–2026 (~1.6-fold lower; p = 0.029), with observed mean CPUE declining from 0.049 fish per net-hour at baseline to

0.030 in 2026. Kissops Flat juvenile catches were significantly higher than baseline in 2026 (approximately 3-fold higher; p = 0.024), consistent with the elevated YOY production at that site in recent years beginning to propagate into the juvenile size class.

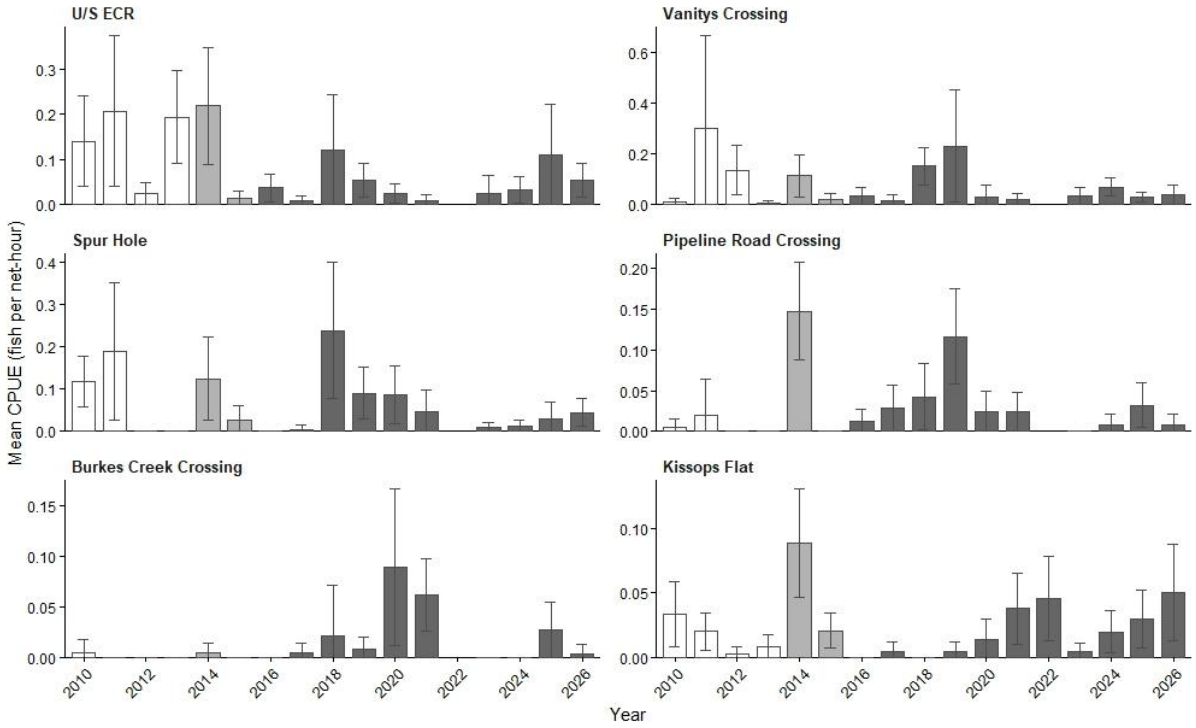

Figure 16. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of juvenile Macquarie perch (≥85 mm TL) captured using fyke nets at six monitoring sites in the Cotter River and Kissops Flat (upper Murrumbidgee River), 2010–2026. CPUE is expressed as fish per net-hour, standardised for variation in soak duration among net deployments. Note that Bracks Hole (sampled 2010–2013) was replaced by U/S ECR (sampled 2014–2026) as the most downstream riverine site; these are treated as a continuous record for the site panel. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Bars are arranged chronologically from left to right within each panel. Error bars represent 95% confidence intervals derived from among-net variation within each sampling year. Cotter River sites were not sampled in 2022 due to high river flows.

Backpack electrofishing

Relative abundance of Macquarie perch captured using backpack electrofishing was highly variable between sites and years (Figure 17). Backpack electrofishing captured a total of five Macquarie perch ranging from 49 – 183 mm TL at two of the five sites (Vanitys Crossing and Pipeline Road Crossing) in 2026. Macquarie perch were not captured at any site in 2011 and 2012 using backpack electrofishing (Figure 17). 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.

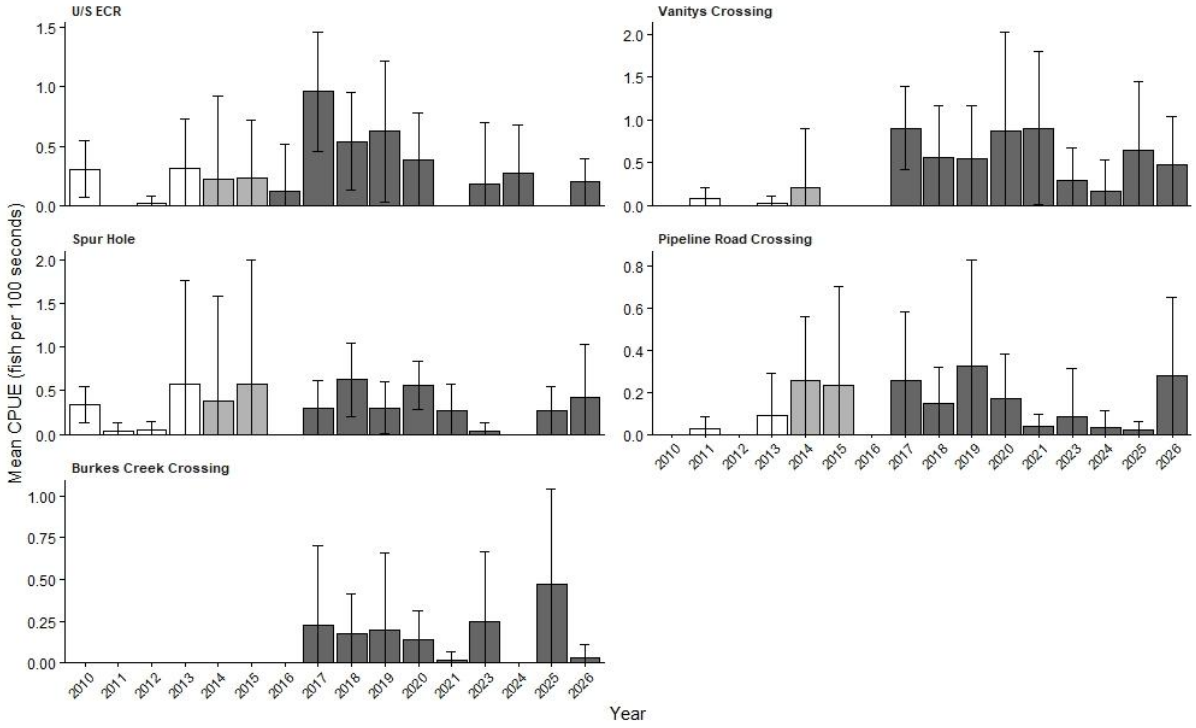

Figure 17. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Macquarie perch captured by backpack electrofishing (all effort) at five impact sites in the Cotter River, 2010–2026. CPUE is expressed as fish per 100 seconds of electrofishing on-time, calculated from raw catch counts standardised for variation in effort among sampling runs. Sites are ordered by increasing distance from the Enlarged Cotter Reservoir. No sampling was conducted in 2022. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026).

DISCUSSION AND CONCLUSIONS

A significant and persistent decline in young-of-year recruitment and total Macquarie perch abundance has been detected across Cotter River monitoring sites throughout the filling and operational phases of the Enlarged Cotter Reservoir. The BACI analysis provides robust evidence that these declines at impact sites exceeded concurrent changes at the Kissops Flat reference site, satisfying the causal inference criteria of the BACI design (Underwood, 1994). Critically, results were robust to the exclusion of the U/S ECR site, which carries a known three-phase sampling confound, BACI contrasts for YOY actually strengthened when this site was removed, confirming the primary inference is conservative rather than artefactual.

The primary mechanism driving YOY suppression at Cotter River sites is most likely the hydrological sensitivity of this small, confined channel to spring flow variation rather than direct reservoir operations. The Cotter River lacks the extensive low-velocity refuges characteristic of larger regulated rivers; spring discharge events during the spawning and early larval window impose disproportionately high shear stress on eggs and larvae, causing physical displacement and mortality consistent with the recruitment models of Tonkin et al. (2017). The episodic nature of YOY catches at

impact sites, with occasional recruitment pulses in 2017–2019 followed by near-zero catches in subsequent years, is consistent with stochastic year-to-year variation in spring flows interacting with a narrow spawning window. The concurrent elevation of YOY catches at Kissops Flat in 2024 and 2026, indicating good regional recruitment in those years, strengthens this interpretation: if dam operations were the primary suppressor, Kissops Flat abundance would not diverge positively from impact sites in the same years that impact site catches are near-zero. Notably, Kissops Flat YOY catches were lower than baseline during 2023–2025, suggesting even the reference site experiences periodic recruitment shortfalls — consistent with regional hydrological variability affecting the entire upper Murrumbidgee catchment, within which the Cotter impact is an additional localised stressor.

The juvenile pattern provides complementary evidence of gradual cohort depletion rather than acute population collapse. The BACI interaction for juveniles was significant, but in the direction of impact sites maintaining higher juvenile catch rates relative to the reference throughout the operational phase, reflecting the contribution of pre-filling cohorts still propagating through the juvenile size class against a backdrop of low and variable Kissops Flat juvenile production. The recent-window comparisons tell the more ecologically relevant story: juvenile CPUE at impact sites is now significantly below its own baseline over both 2023–2025 and 2024–2026, indicating that pre-filling cohorts are progressively depleted and that episodic YOY recruitment events in the operational phase have been insufficient to maintain juvenile abundance at historical levels.

The distribution of the population across the Cotter River study area remains broadly intact, with abundance declining predictably with distance upstream from the reservoir interface. At the most upstream sites, Pipeline Road Crossing and Burkes Creek Crossing, chronically low catch rates throughout all phases, including baseline, reflect genuine low ambient density rather than a phase effect; statistical inference at these sites is limited by severe zero-inflation and the well-documented difficulty of detecting rare species at low-density locations (Maxwell and Jennings, 2005, Joseph et al., 2006, Lintermans, 2016). While fishway remediations have successfully maintained upstream connectivity (Broadhurst et al., 2012), current recruitment levels in the river have not offset the sustained declines in YOY and total abundance observed across mid-river sites. The combined evidence indicates the Cotter River population is persisting but not recovering: the primary constraint on long-term viability is the frequency of favourable low-flow conditions during the spring spawning window, and targeted flow management during this period represents the most tractable intervention available to support population recovery.

The ACT Water Resources Environmental Flow Guidelines 2026 (DI2026–11) include a short-term ecological outcome indicator for the Cotter River reach between Bendora Dam and Cotter Reservoir: Macquarie perch recruitment detected at ≥80% of monitoring sites, with a minimum capture of one Macquarie perch per net night per site. Performance in relation to this indicator across the monitoring period is presented in the inset table.

Inset Table. Macquarie perch fyke net catch at Cotter River monitoring sites (2010–2026) in relation to the ACT Environmental Flow Guidelines 2026 short-term ecological outcome indicator (≥1 fish per net night at ≥80% of sites). Green shading indicates the indicator is met; red indicates it is not met. 2022 not sampled (grey) due to persistent high flows.

Sampling year	Phase	Mac. perch captured	Sites sampled	% sites with recruits	% sites meeting EFG (≥1/net)	EFG indicator met	% <150 mm
2010	Baseline	160	5	100.0%	60.0%	**No**	77.4%
2011	Baseline	230	5	100.0%	60.0%	**No**	72.9%
2012	Baseline	75	5	80.0%	40.0%	**No**	62.7%
2013	Baseline	238	5	80.0%	40.0%	**No**	92.4%
2014	Filling	133	5	100.0%	80.0%	**Yes**	100.0%
2015	Filling	94	5	100.0%	80.0%	**Yes**	38.3%
2016	Operational	24	5	80.0%	0.0%	**No**	83.3%
2017	Operational	83	5	100.0%	60.0%	**No**	81.9%
2018	Operational	169	5	100.0%	60.0%	**No**	95.9%
2019	Operational	224	5	100.0%	100.0%	**Yes**	94.2%
2020	Operational	101	5	100.0%	60.0%	**No**	83.2%
2021	Operational	85	5	100.0%	60.0%	**No**	69.4%
2022	Not sampled	—	—	—	—	**—**	—
2023	Operational	59	5	80.0%	40.0%	**No**	40.7%
2024	Operational	56	5	80.0%	40.0%	**No**	82.8%
2025	Operational	88	5	100.0%	60.0%	**No**	90.6%
2026	Operational	71	5	100.0%	60.0%	**No**	55.9%

The indicator has been met in only three of fifteen sampled years and not in any year of the 2023–2026 window, with 40–60% of sites meeting the per-site threshold in recent years — consistently 20 percentage points below the 80% requirement. Only U/S ECR and Vanitys Crossing perform reliably in the recent window, a spatial pattern inconsistent with a recovering population and consistent with the BACI finding that current recruitment levels in the river are insufficient to offset sustained declines. As with the reservoir indicator, the current threshold of ≥1 fish per net night is a minimum detection floor rather than a meaningful recovery benchmark; baseline catch rates at productive sites regularly exceeded 5–10 fish per net night, and these observations are offered to inform the next EFG Guidelines review cycle.

RECOMMENDATIONS

Monitoring program and methodology

Consistency in monitoring: Current fyke net methods for assessing YOY and juvenile relative abundance remain appropriate for long-term trend detection and no changes to standard field protocols are recommended. The monitoring program should be maintained at all current sites, including the low-density upstream sites (Pipeline Road Crossing and Burkes Creek Crossing), to ensure any future upstream range expansion or recovery is captured.

Refining data interpretation: Future reporting should explicitly note that the U/S ECR site carries a three-phase sampling confound — baseline catches inflated by proximity to the old reservoir spawning ecotone, filling-phase catches suppressed by natural rock barriers preventing adult egress, and operational catches deflated by post-2014 net relocation — rendering direct phase comparisons at this site unreliable. Formal BACI inference should continue to exclude U/S ECR from primary analysis, with sensitivity results reported alongside. This does not preclude continued monitoring at the site, which retains value as a qualitative indicator of activity at the reservoir–river interface.

Statistical constraints at low-density sites: The absence of statistically significant phase effects at Pipeline Road Crossing and Burkes Creek Crossing reflects chronically low catch rates and zero-inflation throughout all monitoring phases rather than population stability. These sites should be retained in the program with the explicit recognition that conventional significance thresholds will remain difficult to achieve at ambient densities, and that trend interpretation should rely on qualitative patterns and multi-year aggregated data rather than annual significance testing alone.

Management and intervention

The weight of evidence points to spring flow variation during the spawning and early larval window as the primary proximate driver of year-to-year recruitment success in the Cotter River. The most tractable management intervention is therefore targeted flow stabilisation during the critical October–December spawning period, minimising high-discharge pulses that impose shear stress on eggs and larvae in this hydraulically confined, refuge-limited channel. Importantly, this recommendation does not apply to winter flow pulses, which fall outside the larval vulnerability window and provide positive habitat benefits by scouring fine sediment and algae from cobble spawning substrates; winter flushing events should not be constrained.

Given that YOY suppression has persisted for over a decade with no evidence of recovery at mid-river sites, future monitoring reports should assess outcomes against explicit recovery benchmarks, for example, detection of statistically significant YOY CPUE at Spur Hole or Vanitys Crossing in at least two of five consecutive years, rather than describing results purely in relation to prior-year status. This will allow adaptive management decisions to be anchored to defined population-level targets rather than year-to-year fluctuations.

If flow management interventions prove operationally infeasible or insufficient to generate measurable recruitment recovery within a defined timeframe (suggested three to five years of post-intervention monitoring), consideration should be given to targeted translocation of juveniles or

adults from the reservoir population into upper-river reaches to supplement river population density and genetic connectivity, building on established translocation protocols for the species in the ACT.

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 (*Gadopsis bispinosa*) have been historically absent from the Cotter Reservoir, primarily due to excessive sedimentation smothering spawning habitats (Ebner et al., 2008, Lintermans, 2002). Exceptions include a small number of individuals detected in 2012, likely displaced by flooding (Lintermans et al., 2013). Conversely, the species was known to inhabit the river reach subsequently inundated by the ECR (Ebner et al., 2008, Lintermans et al., 2013). This study determines whether the species persists in this newly inundated reach and whether it has expanded to colonize the broader ECR perimeter.

METHODS

Sampling design follows the baseline monitoring program (Lintermans et al., 2013). To balance logistical requirements, Corin Reservoir was excluded from the filling and operational phases but was re-surveyed in 2025 to evaluate its suitability as a replacement control site.

Table 4. 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) was assessed using descriptive comparisons where data were sufficient. In Bendora Reservoir, temporal variation was evaluated using 95% confidence intervals (with Bonferroni correction) and modelled using Tweedie GLMMs and GAMs. In Cotter Reservoir, analyses were limited to descriptive summaries due to predominance of zero-catch data.

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). To explore whether Corin Reservoir may act as a suitable control site replacement for Bendora Reservoir (due to its decline to nearly undetectable levels), a night of fyke netting was conducted in December 2025, comprising 20 fyke nets set in a stratified random position around the perimeter of the reservoir.

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. Temporal trends in the relative abundance (CPUE) of Two-spined blackfish at Bendora Reservoir were evaluated using a Tweedie-distributed model to account for the high proportion of zero-catch data. We compared a log-link Generalized Linear Mixed Model (GLMM) with a linear year term against a Generalized Additive Model (GAM) incorporating a penalised regression spline on year. Both models included net identity as a random effect to account for spatial variance. Models were compared using Akaike's Information Criterion (AIC), with model adequacy confirmed via basis dimension checks and residual diagnostics. Analyses were performed in R (v4.4.1).

RESULTS

Four Two-spined blackfish were captured by fyke nets in the ECR in 2026, as part of the all reservoir netting component, with all four captured in the most upstream end of the ECR (Figure 18). Lengths of Two-spined blackfish captured from ECR in 2026 ranged from 205 – 268 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 2026. For the third time in the past six years, fyke netting failed to capture a Two-spined blackfish in Bendora Reservoir in 2026 (Figure 18). 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 18).

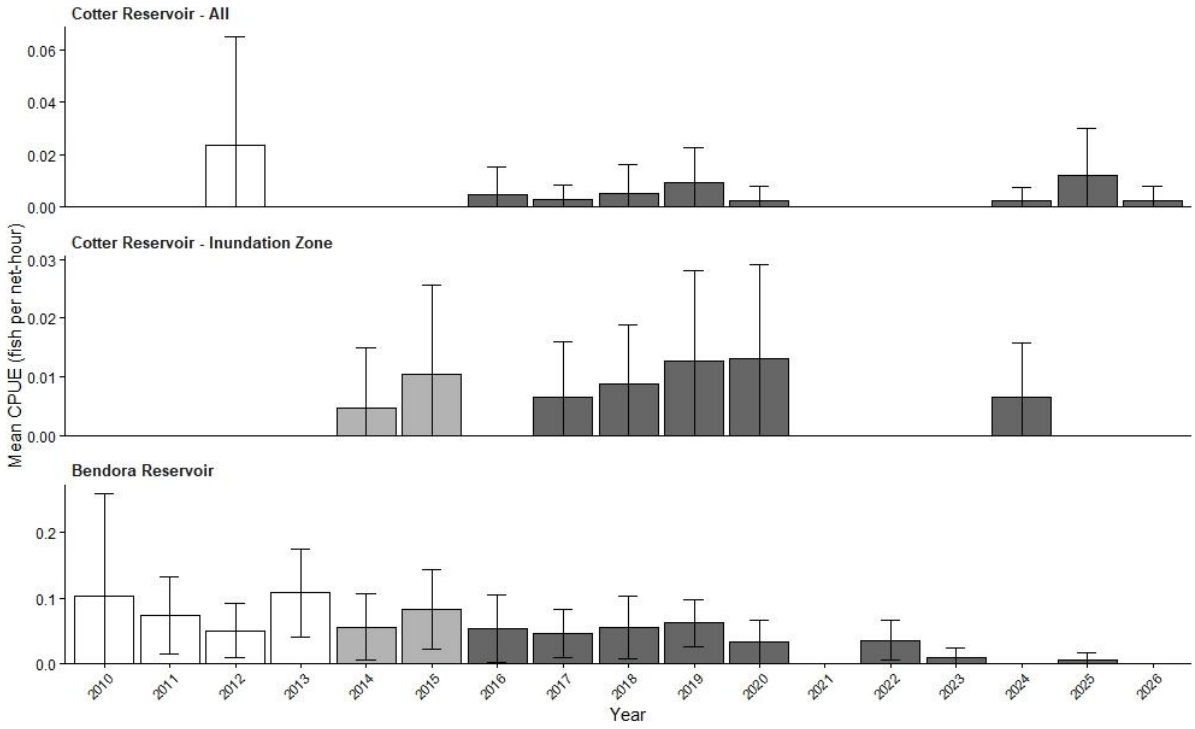

Figure 18. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Two-spined blackfish captured using fyke nets in Cotter Reservoir (all nets; inundation zone nets only) and Bendora Reservoir, 2010–2026. CPUE is expressed as fish per net-hour, calculated from raw catch counts standardised for variation in soak duration among net deployments. The inundation zone panel includes only nets deployed in the newly inundated upstream section of Cotter Reservoir. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026).

CPUE of Two-spined blackfish at Bendora Reservoir declined significantly over the 17-year monitoring period. The Tweedie GAM provided a better fit to the data than the log-linear GLMM (delta AIC = 7.9), with the significant smooth term for year (edf = 2.75, F = 12.35, $p < 0.001$) indicating a non-linear decline. CPUE was relatively stable from 2010 to approximately 2017 before declining sharply, with catches approaching zero from 2023 onwards and the species effectively undetectable by 2026 (Figure 19). The log-linear GLMM corroborated the direction and significance of the trend (beta = -0.69 +/- 0.10 per SD year, z = -7.12, $p < 0.001$).

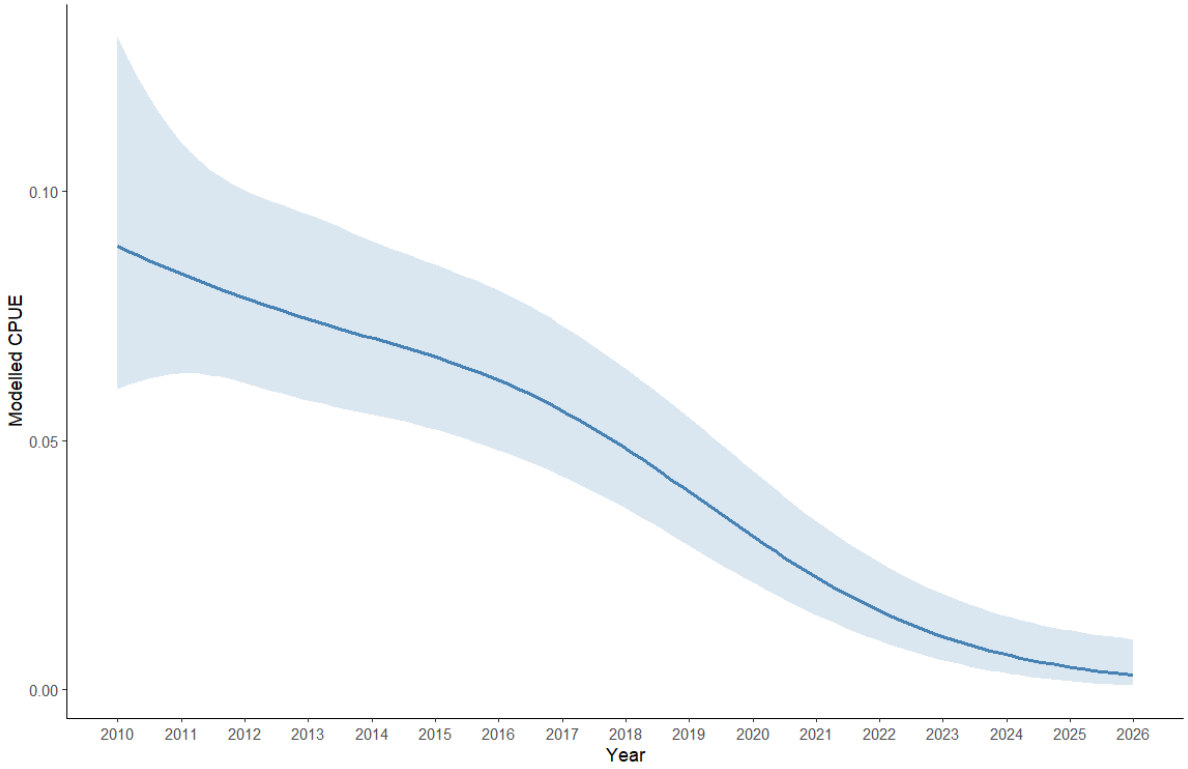

Figure 19. Predicted CPUE of Two-spined blackfish at Bendora Reservoir from 2010 to 2026. The solid line represents the fitted trend from a Tweedie generalised additive model with a penalised spline on year; the shaded band represents the 95% confidence interval.

Corin Reservoir – alternate control site

A total of 82 Two-spined blackfish were captured across 20 fyke nets during a single overnight survey at Corin Reservoir in 2025. Mean CPUE was 0.224 ± 0.034 fish per net per hour (mean ± SE), with captures recorded in 17 of 20 nets. Measured specimens (n = 55) ranged from 82 mm to 306 mm in total length (mean ± SE = 153.1 ± 5.3 mm), indicating a broad size range representative of multiple age classes (Figure 20). To evaluate the change in relative abundance, a Generalized Linear Model (GLM) with a Tweedie error distribution was employed to compare CPUE data between the historical (2010–2013) and contemporary (2025) sampling periods. The analysis confirms a statistically significant increase in mean CPUE in 2025 compared to the historical baseline ($\beta = 0.874$, $p < 0.0015$), with modelling indicating that current catch rates are approximately 2.4 times higher than historical levels (Figure 21). This significant contrast, alongside the documented demographic breadth, suggests that Corin Reservoir supports a robust and self-sustaining population of Two-spined blackfish.

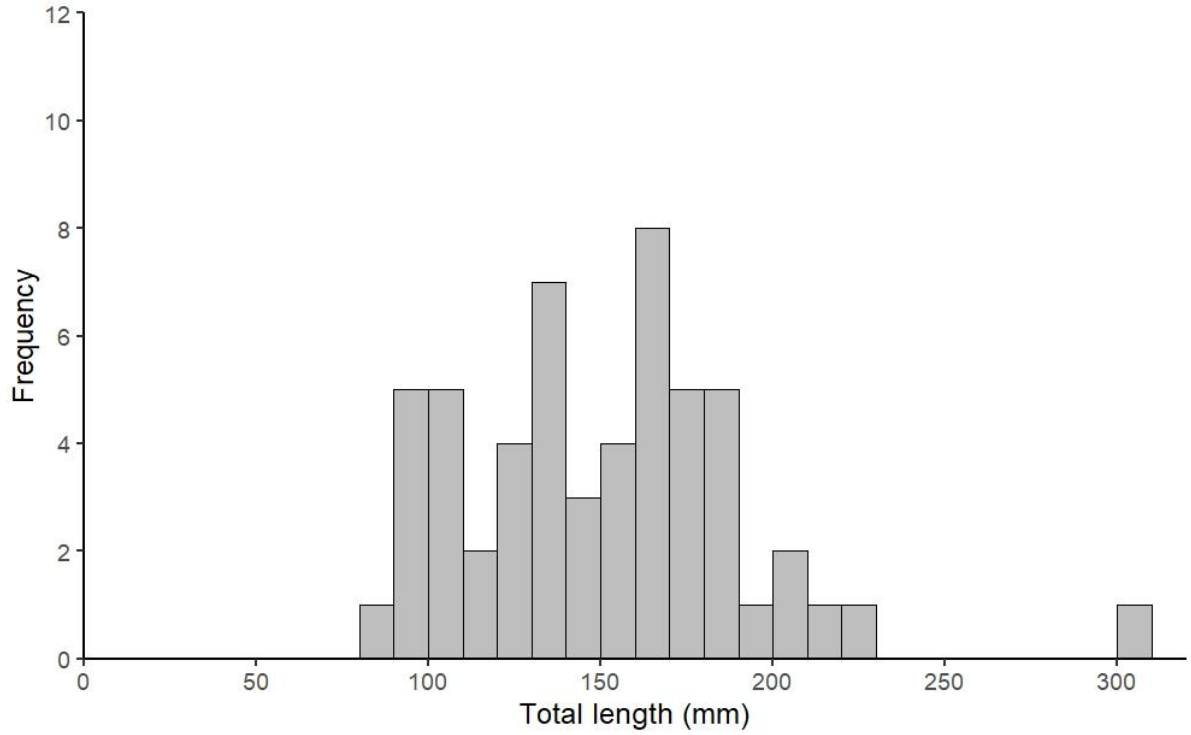

Figure 20. Length frequency distribution of Two-spined blackfish captured at Corin Reservoir in a single night of fyke netting in 2025 (n = 55 fish measured of 82 captured).

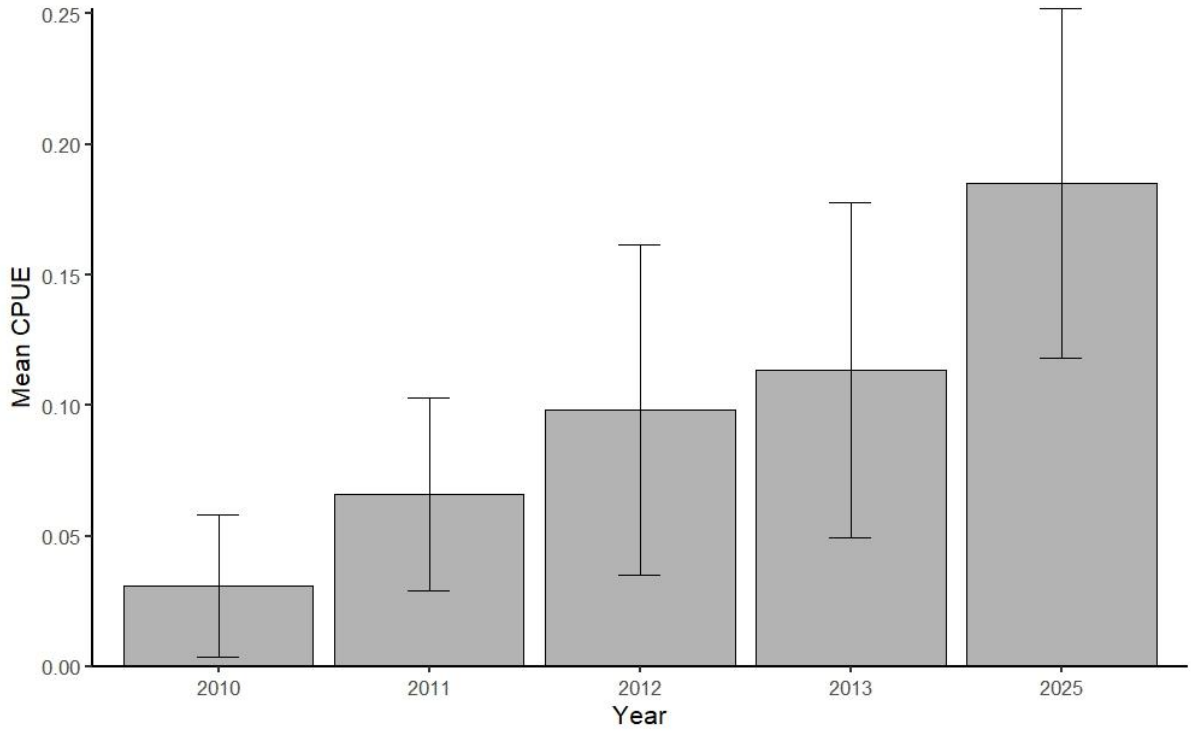

Figure 21. Mean catch per unit effort (CPUE, fish per net per hour ± 95% confidence interval) of Two-spined blackfish captured during fyke net surveys at Corin Reservoir in 2010 – 2013, and 2025. Surveys comprised 10 fyke nets set overnight, with the exception of 2025 which used 20 nets.

DISCUSSION AND CONCLUSIONS

The low abundance of Two-spined blackfish in the Cotter Reservoir reaffirms that the original reservoir environment was sub-optimal due to historical forestry-related sedimentation (Lintermans, 1998, Ebner et al., 2008). While the newly inundated upstream reaches provide suitable habitat, captures remain rare, and no recruitment has been detected; current captures are likely mature adults colonising from upstream river reaches.

The precarious status of the Bendora Reservoir population is concerning and necessitates further investigation into the drivers of this decline. Given Bendora's diminished utility as a reference site, the integration of Corin Reservoir is essential. Corin provides a robust, high-performing "healthy system" benchmark. Transitioning to this site mitigates the bias associated with failing reference points and provides the statistical stability required to discern meaningful population trends across our experimental sites, ensuring our monitoring remains sensitive to ecological change.

RECOMMENDATIONS

In light of the observed population trends at the Cotter and Bendora reservoirs, we provide the following recommendations to guide future monitoring efforts;

- Maintain current ECR sampling methodology.
- No active management intervention is required for the ECR population.
- Formally adopt Corin Reservoir as the primary reference site for future monitoring, given the continued decline at Bendora.
- Conduct a targeted investigation into the drivers of the population collapse at Bendora Reservoir.

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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 Cotter Reservoir due to their capacity for piscivory and associated predation pressure on native fishes (Budy et al., 2013). Enlargement of the reservoir was predicted to promote a trophic upsurge response through the inundation of terrestrial vegetation and increased reservoir area and depth, potentially increasing food availability and habitat resources for trout (Lintermans, 2012). These changes were expected to support increases in trout abundance and biomass during the early operational phase.

In addition to increased productivity, the deeper reservoir was expected to provide improved thermal refuge and modified habitat conditions, including changes to dissolved oxygen regimes associated with altered destratification practices. Together, these factors were predicted to enhance growth and size of trout individuals by improving access to key resources (Budy et al., 2013). Monitoring changes in trout abundance and size structure is therefore critical to provide early warning of any increase in predatory interactions with Macquarie perch.

METHODS

Sampling design for Question 4 followed the baseline monitoring framework described in Lintermans et al. (2013), with minor modifications to reduce costs. Corin Reservoir was removed from the post-filling monitoring program, resulting in two sites: Cotter Reservoir (impact) and Bendora Reservoir (reference).

Table 5. 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) of Rainbow trout was analysed between reservoirs (impact vs. reference) and management phases using generalized additive mixed models (GAMMs) with a negative binomial error structure. Trout fork length was analysed using Gaussian GAMMs with an equivalent model structure. Both analyses followed a Before–After–Control–Impact (BACI) framework and incorporated temporal trends.

Sampling targeted sub-adult and adult Rainbow and Brown trout of a size considered to be piscivorous (>150 mm fork length, FL), with sufficient gape to consume larval or early juvenile Macquarie perch and Two-spined blackfish (Ebner et al., 2007). Trout were sampled using gill nets as described for Question 1, with the exception that two additional 125 mm mesh nets introduced in 2022 were excluded from analyses to maintain consistency through time. Sampling was conducted annually in early autumn to align with the baseline monitoring program. Two sites were assessed: the impact site (Cotter Reservoir) and a reference site (Bendora Reservoir).

Rainbow trout abundance in Cotter and Bendora reservoirs was assessed within a BACI framework comparing Cotter Reservoir (impact) against Bendora Reservoir (reference) across baseline (2010–2013), filling (2014–2015), and operational (2016–2026) phases. Raw gill net counts were modelled using two complementary negative binomial GAMs with a log-offset for soak duration. The primary smooth model included site-specific nonlinear temporal trends (s(year, by = site, k = 5)) for trend significance testing. A secondary phase model (site × phase fixed effects, net deployment random effect) provided phase-level BACI contrasts and phase-level estimated means via coefficient back-transformation using the delta method. Three additional two-level GAMs compared recent monitoring windows (2026, 2023–2025, 2024–2026) against baseline by site. Model adequacy was assessed using DHARMa simulation-based residual diagnostics. Full model specifications are provided in Appendix 1. Post hoc comparisons were used to interpret differences between sites and phases where relevant, and model performance was assessed using standard diagnostic procedures. Full details of model structure, implementation, and diagnostics are provided in Appendix 1.

As Brown trout are not present in Bendora Reservoir, formal BACI modelling was not undertaken for this species, and temporal trends in CPUE and fork length are presented descriptively for Cotter Reservoir only.

RESULTS

Abundance

Twelve Rainbow trout and one Brown trout were captured in gill nets in Cotter Reservoir in 2026 (Figure 24). Four Rainbow trout were captured in Bendora Reservoir in 2026 (Figure 25). Rainbow trout CPUE exhibited significant nonlinear temporal trends at both Cotter Reservoir (s(year): edf = 1.00, $\chi^2 = 17.17$, $p < 0.001$) and Bendora Reservoir (s(year): edf = 3.60, $\chi^2 = 41.70$, $p < 0.001$), with both sites showing elevated baseline CPUE followed by marked declines during the filling phase and partial recovery during the operational phase (Figure 22). Observed mean CPUE was 21.2 and 13.3 fish per 100 net-hours at Bendora and Cotter reservoirs during baseline, declining to 4.0 and 5.0 during filling, and recovering partially to 8.9 and 6.3 during the operational phase.

The BACI analysis revealed no significant site × phase interaction for any phase contrast (baseline/filling: ratio = 2.12, $p = 0.215$; baseline/operational: ratio = 1.17, $p = 0.652$; filling/operational: ratio = 0.55, $p = 0.298$), indicating that temporal changes in Rainbow trout CPUE did not differ significantly between Cotter and Bendora reservoirs across any monitoring phase. Both reservoirs followed broadly similar trajectories, consistent with system-wide drivers rather than reservoir-enlargement-specific effects.

Targeted comparisons of recent monitoring windows against baseline confirmed significant declines at both sites. At Bendora Reservoir, CPUE was approximately 7.3-fold lower than baseline in 2026 ($p = 0.002$), 4.1-fold lower over 2023–2025 ($p < 0.001$), and 4.9-fold lower over 2024–2026 ($p < 0.001$). At Cotter Reservoir, declines were of similar magnitude: approximately 4.2-fold lower in 2026 ($p = 0.026$), 2.6-fold lower over 2023–2025 ($p = 0.007$), and 6.1-fold lower over 2024–2026 ($p < 0.001$). The parallel declines at both reservoirs, without significant site × phase divergence, indicate that the reduction in Rainbow trout abundance over recent years reflects a system-wide trend rather than a differential response to Cotter Reservoir enlargement.

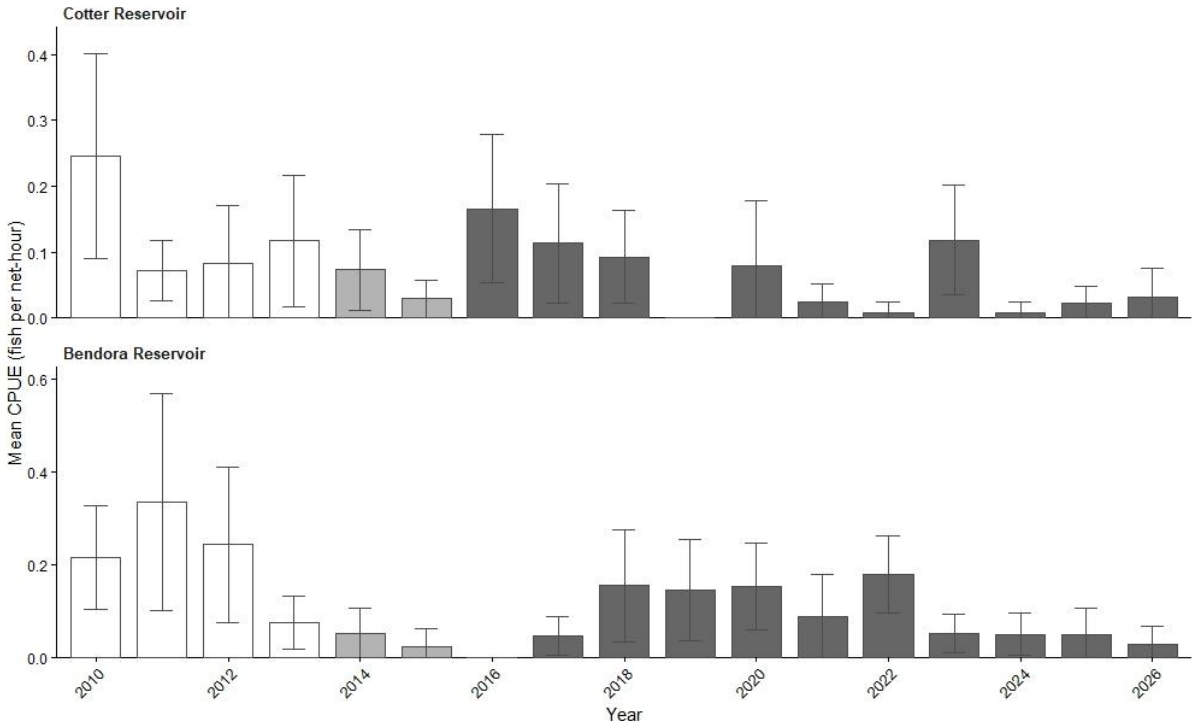

Figure 22. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Rainbow trout captured using multi-mesh gill nets at Cotter Reservoir (upper panel) and Bendora Reservoir (lower panel), 2010–2026. CPUE is expressed as fish per net-hour, standardised for variation in soak duration among net deployments. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Error bars represent 95% confidence intervals derived from among-net variation within each sampling year.

Mean CPUE of Brown trout in Cotter Reservoir was low prior to reservoir filling, with negligible or zero captures recorded up to 2014 (Figure 23). Following commencement of filling, CPUE increased markedly, peaking between 2016 and 2017 (approximately 0.15–0.25 fish per net-hour), and remaining moderately elevated through to around 2020. However, since 2021, CPUE has declined sharply, with consistently low values recorded in recent years (generally <0.05 fish per net-hour), and several years with very low or near-zero captures (Figure 23). Overall, Brown trout abundance exhibited a short-term increase during the early operational phase followed by a sustained decline to low levels in the most recent monitoring period.

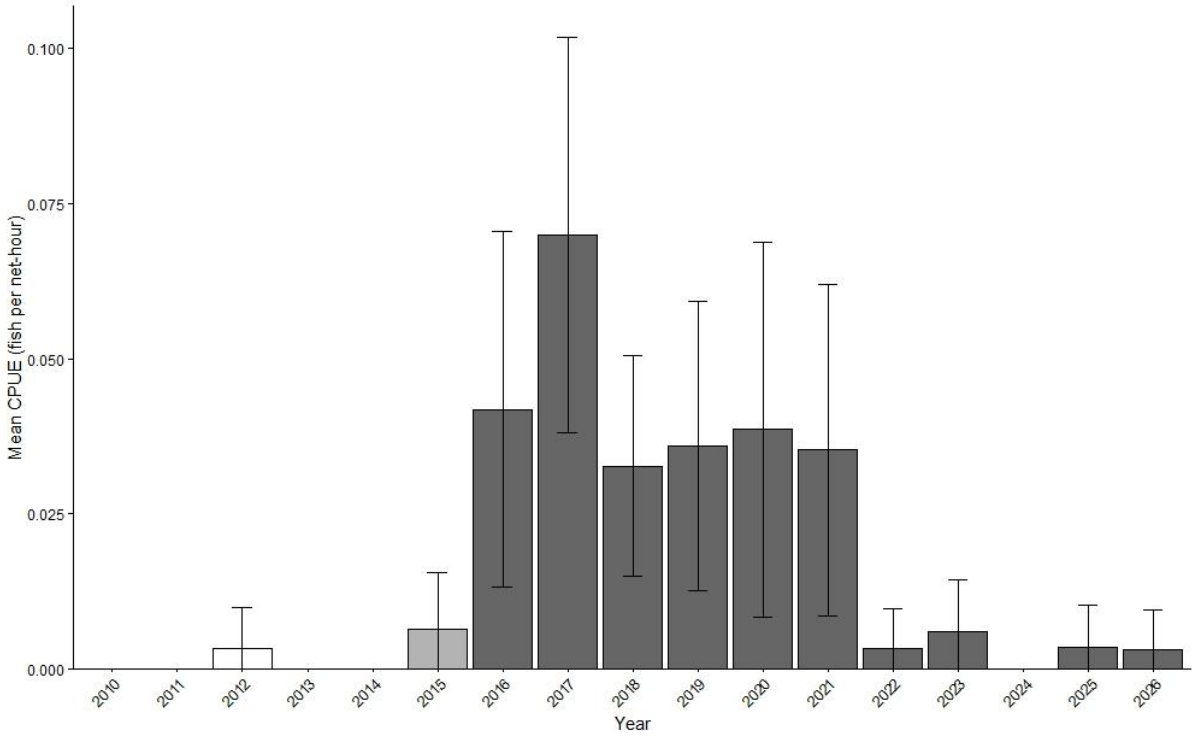

Figure 23. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Brown trout captured using gill nets in Cotter Reservoir, 2010–2026. CPUE is expressed as fish per net-hour, calculated from raw catch counts standardised for variation in soak duration among net deployments. Nets 11 and 12 (5-inch mesh panels) are excluded from all analyses. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026).

Size composition

Rainbow trout fork length increased significantly through time in both reservoirs, with a strong effect of monitoring phase (Operational vs Baseline: +60.6 mm, SE = 13.5, $p < 0.001$). A significant interaction between reservoir and phase ($p = 0.019$) indicated that temporal changes in fish size differed between sites. In Bendora Reservoir, fish were approximately 61 mm larger in the operational phase compared to baseline ($p < 0.001$), whereas in Cotter Reservoir the increase was smaller, at approximately 36.9 mm ($p = 0.006$). Although no significant differences in mean fork length were detected between reservoirs within any single phase ($p > 0.09$), the differing magnitude of increase confirms a site-specific response. A weak but significant linear temporal trend in fork length was also detected ($F = 5.24$, $p = 0.043$).

Comparisons of recent periods (2023–2025 and 2026) to baseline conditions did not detect statistically significant differences in fork length following adjustment for multiple comparisons ($p > 0.05$). However, effect sizes were consistent with the broader GAMM results, indicating increased fish size through time in both reservoirs, with a more moderate response in Cotter Reservoir. This pattern is also evident in annual length distributions (Figure 26), which show a steady increase in fish

size in Bendora Reservoir compared to a more moderate increase followed by stabilisation and increased variability in Cotter Reservoir.

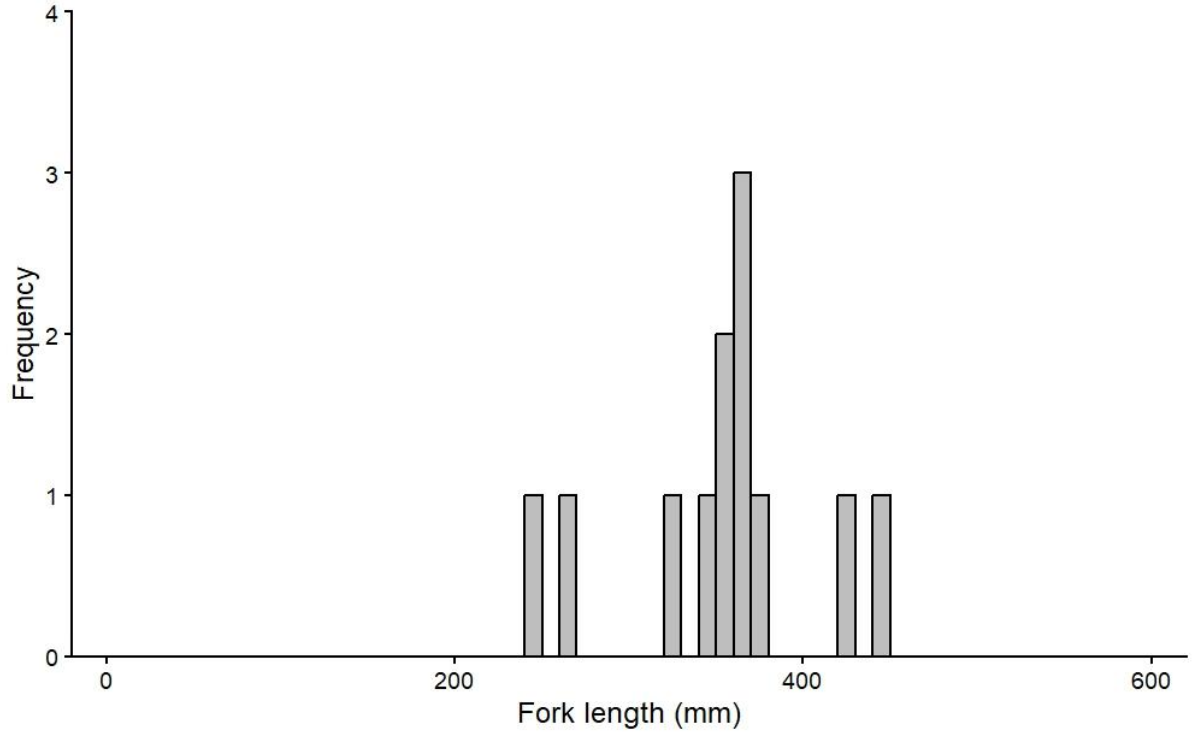

Figure 24. Length frequency of Rainbow trout (n = 12) captured from the ECR in autumn 2026 using gill nets.

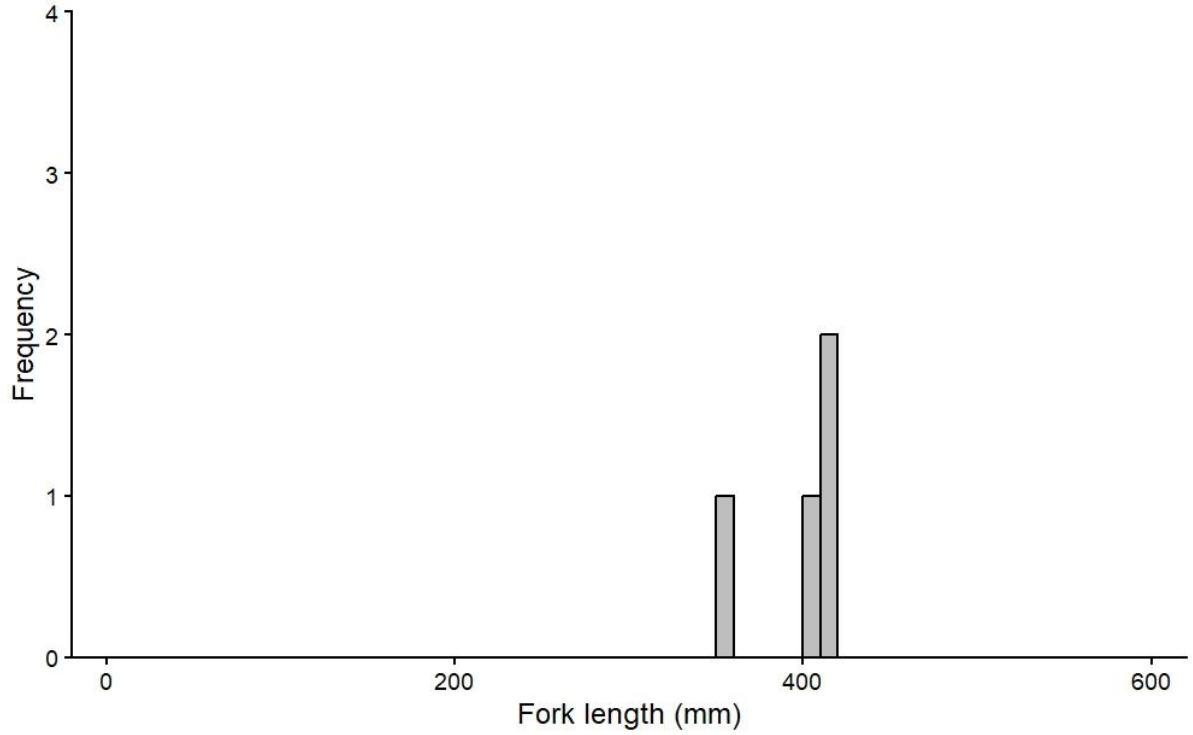

Figure 25. Length frequency of Rainbow trout (n = 4) captured from Bendora Reservoir in autumn 2025 using gill nets.

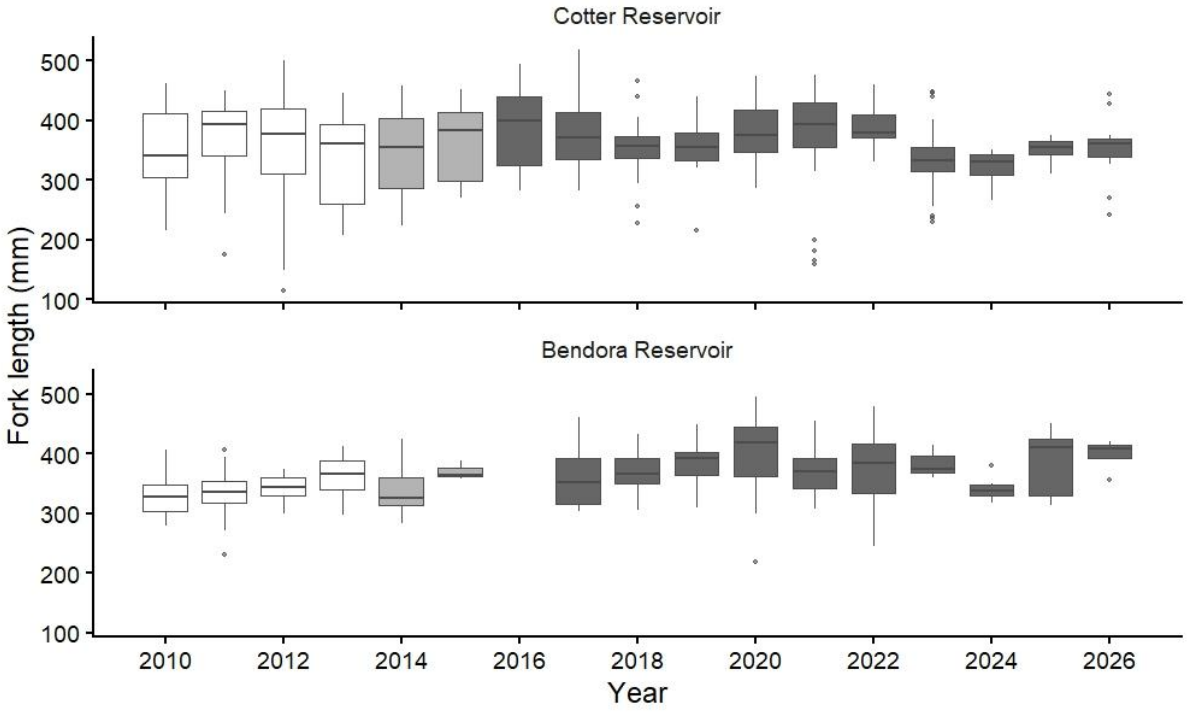

Figure 26. Fork length (mm) of adult Rainbow trout (≥ 150 mm FL) captured using gill nets in Cotter Reservoir (impact) and Bendora Reservoir (reference) from 2010 to 2026. Boxes represent the interquartile range (IQR), horizontal lines indicate median values, whiskers extend to 1.5 × IQR, and points represent outliers. Shading indicates monitoring phases: baseline (white; 2010–2013), filling (light grey; 2014–2015), and operational (dark grey; 2016–2026). Fork length increased through time in both reservoirs, with a more consistent increase observed in Bendora Reservoir, while Cotter Reservoir exhibited a more moderate increase followed by stabilisation or increased variability in recent years.

DISCUSSION AND CONCLUSIONS

The initial increase in Rainbow trout abundance and body size in Cotter Reservoir following enlargement is consistent with a trophic upsurge response, whereby nutrient release from inundated terrestrial vegetation elevates primary and secondary productivity and cascades through to higher trophic levels (Grimard and Jones, 1982, Kimmel and Groeger, 1986, Hatton, 2016). However, a broadly similar temporal pattern was evident at Bendora Reservoir over the same period, despite Bendora not undergoing enlargement. This parallel trajectory resulted in no significant site × phase divergence in the BACI analysis, indicating that the early operational-phase peak at Cotter Reservoir was not a reservoir-specific response to enlargement but reflects a shared regional signal, most likely driven by catchment-wide hydrological variability and associated productivity dynamics (Poff et al., 1997, Elliott and Elliott, 2010). The synchronous post-2021 decline across both reservoirs, and across all Cotter River sites (Question 5), reinforces this interpretation.

Patterns in fish size provide further insight. Although Rainbow trout fork length increased significantly through time in both reservoirs, the magnitude of increase was greater at Bendora

Reservoir than at Cotter Reservoir. The drivers of this difference are not clear from available data but likely reflect differences in baseline fish density, habitat structure, or food availability between the two systems rather than a differential response to reservoir enlargement.

Taken together, these results paint a reassuring picture from a native species management perspective. Trout abundance in both reservoirs peaked in the early-to-mid operational phase and has since declined to near-baseline levels, with no significant divergence between sites and no evidence that ECR enlargement specifically elevated reservoir trout populations above what regional conditions would have produced regardless. Current catch rates at Cotter Reservoir are among the lowest recorded across the entire monitoring period. Continued monitoring remains important to detect any future resurgence that would warrant reassessment of predation risk to Macquarie perch, and explicit trigger thresholds for management intervention should be defined before the next monitoring cycle.

The number of Brown trout captured in the ECR over the past five years was considerably lower than during the preceding six years, which recorded comparatively high abundances. This pattern indicates that the Brown trout population in Cotter Reservoir has declined since its peak in the early operational phase. Similar to Rainbow trout, this temporal pattern is consistent with a trophic upsurge response following reservoir filling, whereby increased productivity initially supports elevated fish abundance before stabilising or declining as the system matures. Monitoring over the next few years will determine whether this decline persists or stabilises at current low levels. This represents a positive outcome for Macquarie perch, as Brown trout are considered more strongly piscivorous and potentially more impactful on threatened fish populations than Rainbow trout (NSW Fisheries, 2003), particularly during periods of elevated abundance associated with early post-impoundment productivity.

RECOMMENDATIONS

Monitoring and assessment

Current methods for assessing adult trout abundance, distribution, and size are appropriate and should be maintained, provided monitoring continues when surface temperatures are below 20 °C. Continued long-term monitoring is important to detect changes associated with reservoir ageing.

Both Rainbow and Brown trout abundance have declined substantially from their early operational-phase peaks, with current catch rates at Cotter Reservoir among the lowest recorded across the monitoring period. The parallel decline at the unenlarged Bendora Reservoir indicates this trajectory reflects regional rather than reservoir-specific dynamics. The recent reduction in Brown trout abundance is a positive outcome for Macquarie perch.

Management

Trout nevertheless remain a potential long-term risk to native species (Todd et al., 2017), and formalisation of explicit trigger thresholds for management intervention is recommended so that actions can be implemented promptly if future increases in abundance or size occur (Lintermans, 2012, ACTEW Corporation, 2013). Any such response should be coordinated with the targeted

predation assessment framework established under Question 1, which addresses predation risk during the December–January larval and early juvenile window, the period of greatest vulnerability for Macquarie perch and the life-history stage least accessible to conventional monitoring methods.

Proposed salmonid management trigger framework

The 2018 Cotter Reservoir Alien Fish Management Plan (Icon Water) committed to developing management trigger thresholds for salmonid abundance once sufficient monitoring data were available. With 17 years of standardised gill net data now available from Cotter Reservoir and the parallel reference site at Bendora Reservoir (2010–2026), a robust empirical basis for trigger-setting now exists. This section proposes a formal trigger framework, assesses its statistical power, and recommends adoption for ratification by Icon Water and the ACT Government.

Dataset

All analyses use gill net catch data from the standard 10-net configuration (nets 1–10 only). Nets 11 and 12, which deploy larger 125 mm mesh panels targeting adult Macquarie perch, are excluded to ensure sampling consistency across the full monitoring period. Sampling comprises 5 overnight sets at Cotter Reservoir (50 net-nights per year) and 2 overnight sets at Bendora Reservoir (20 net-nights per year). CPUE is expressed as mean catch per net per night within each sampling year. Raw CPUE at Cotter Reservoir is subject to water level-dependent variation in littoral habitat accessibility; trigger assessments in years of atypically high or low reservoir level should therefore be interpreted with appropriate caution, and it is recommended that a water level qualifier be incorporated in the final ratified protocol.

The annual CPUE dataset for Rainbow trout at both sites, the Cotter:Bendora ratio, and Brown trout CPUE at Cotter Reservoir across all monitoring phases is presented in Table 6. Orange highlighting in the ratio column indicates single-year exceedance of the Trigger B threshold; orange in the Cotter CPUE column indicates single-year exceedance of the Trigger A fallback threshold; red in the Brown trout column indicates single-year exceedance of the Trigger C threshold.

Table 6. Annual gill net CPUE (fish per net night, nets 1–10 only) for Rainbow trout (RT) at Cotter and Bendora reservoirs, the Cotter:Bendora ratio, and Brown trout (BT) CPUE at Cotter Reservoir, 2010–2026. Net nights = distinct net × sampling night combinations. C:B = Cotter:Bendora ratio. Blue shading = baseline; amber = filling phase; white = operational. Bold values indicate exceedance of a trigger threshold.

Year	Phase	RT Cotter total	Net nights	RT Cotter CPUE	RT Bendora total	Net nights	RT Bendora CPUE	C:B ratio	BT Cotter CPUE
2010	Baseline	62	50	1.240	28	20	1.400	0.886	—
2011	Baseline	18	50	0.360	42	20	2.100	0.171	—
2012	Baseline	38	50	0.760	33	20	1.650	0.461	—
2013	Baseline	27	50	0.540	11	20	0.550	0.982	—
2014	Filling	27	48	0.562	7	20	0.350	1.607	—
2015	Filling	14	50	0.280	3	20	0.150	1.867	—
2016	Operational	39	50	0.780	0	20	0.000	N/A	0.26
2017	Operational	53	50	1.060	6	20	0.300	**3.533**	**0.48**
2018	Operational	25	50	0.500	22	20	1.100	0.455	0.22
2019	Operational	9	50	0.180	20	20	1.000	0.180	0.22
2020	Operational	35	50	0.700	21	20	1.050	0.667	0.26
2021	Operational	34	50	0.680	11	20	0.550	1.236	0.22
2022	Operational	6	50	0.120	24	20	1.200	0.100	0.02
2023	Operational	36	50	0.720	7	20	0.350	**2.057**	0.04
2024	Operational	5	50	0.100	6	20	0.300	0.333	0.00
2025	Operational	5	50	0.100	7	20	0.350	0.286	0.02
2026	Operational	12	50	0.240	4	20	0.200	1.200	0.02

Proposed trigger framework

Three triggers are proposed. All apply a two-consecutive-year activation rule, consistent with the activation logic established in the ECR Cormorant Management Plan (Icon Water, 2018), ensuring that single-year CPUE spikes, common in gill net data and frequently reflecting water level variation or sampling noise rather than genuine population change, do not generate unnecessary management responses.

Trigger B (primary — Rainbow trout reference site ratio)

Trigger B is the recommended primary signal for Rainbow trout management. It uses the ratio of Cotter Reservoir to Bendora Reservoir Rainbow trout CPUE, consistent with the BACI framework used throughout this monitoring program. A reference site ratio approach is preferred over an absolute threshold because it controls for regional variation in salmonid abundance — a Cotter-specific increase above regional trends is the ecologically relevant management signal. Trigger B activates when the Cotter:Bendora CPUE ratio is ≥2.0 AND Cotter mean CPUE is ≥0.5 fish per net night, in two consecutive years. The minimum CPUE floor of 0.5 fish per net night prevents activation from low-count artefacts where a ratio ≥2.0 could arise from trivially small catches on both sides. In years where Bendora CPUE equals zero and the ratio cannot be calculated, Trigger A (fallback) applies for that year's contribution to the two-consecutive-year assessment.

Trigger A (fallback — Rainbow trout absolute CPUE)

Trigger A is not an independent parallel trigger but a defined fallback applied only in years where Bendora CPUE equals zero, making the Trigger B ratio incalculable. In such years, Trigger A substitutes as that year's contribution: Cotter mean Rainbow trout CPUE ≥1.5 fish per net night. This threshold is derived from the baseline mean plus two standard deviations (0.725 + 2 × 0.380 = 1.49, rounded to 1.5), representing a departure exceeding the upper range of pre-enlargement variability. Bendora CPUE equalled zero in 2016 only across the monitoring period.

Trigger C (Brown trout — absolute CPUE)

Brown trout warrant a separate and lower threshold given their effectively zero pre-enlargement baseline and greater piscivorous tendency relative to Rainbow trout. As Brown trout are absent from Bendora Reservoir, no reference site ratio is available. Trigger C activates when Cotter mean Brown trout CPUE is ≥0.3 fish per net night in two consecutive years, anchored to the operational-phase peak of 0.48 fish per net night observed in 2017. Brown trout CPUE has been near zero since 2022.

Management response

On activation of any trigger, a formal management review should be convened in consultation with the ACT Threatened Species Recovery Team to determine whether active removal is warranted in addition to any standing precautionary measures in place. None of the three triggers have been activated across the 11-year operational phase to date, consistent with the observed regional decline in salmonid abundance (Figure 27).

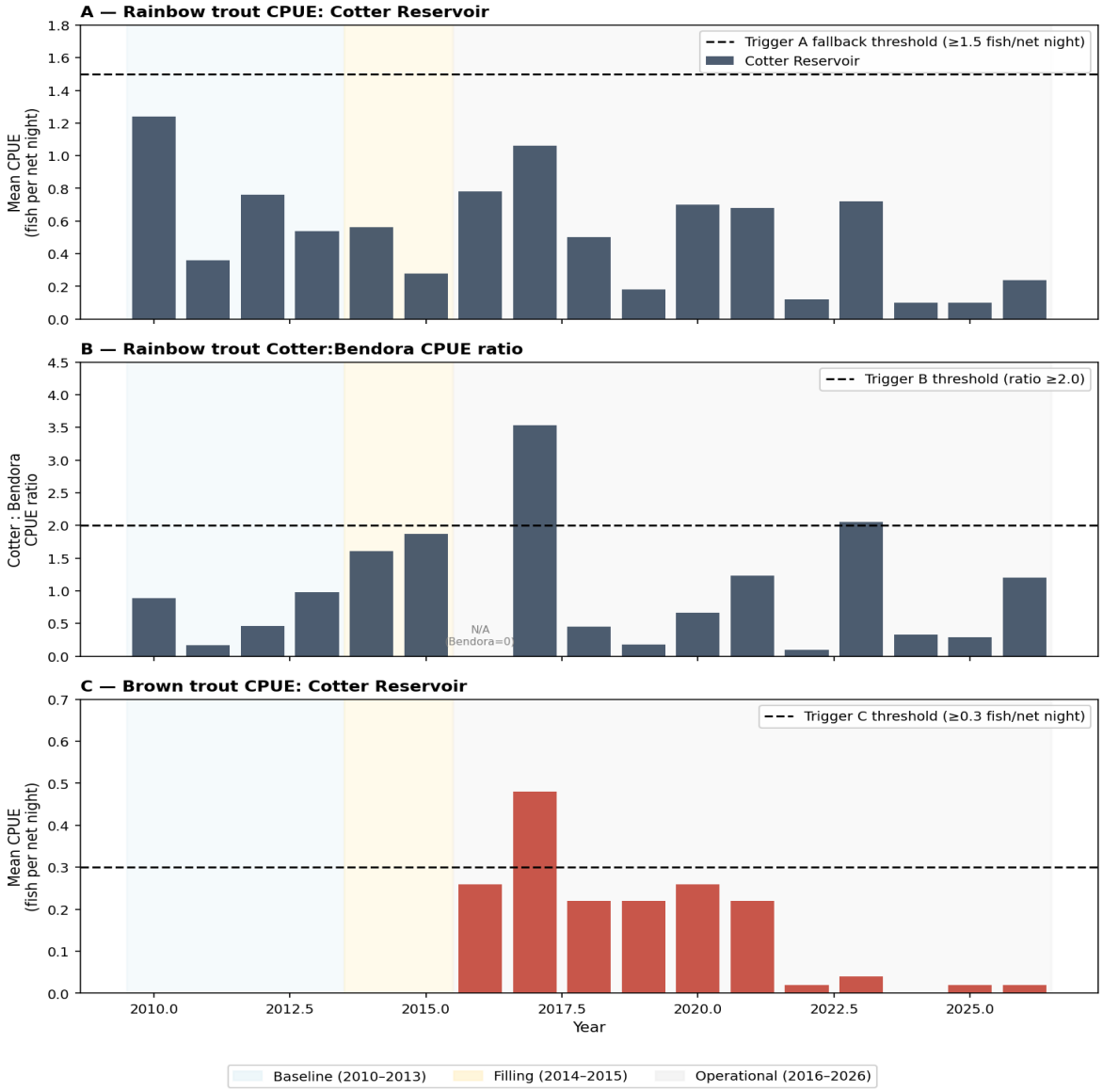

Figure 27. Annual gill net CPUE for Rainbow trout at Cotter Reservoir (A), the Cotter:Bendora CPUE ratio with Trigger B threshold (B), and Brown trout CPUE at Cotter Reservoir with Trigger C threshold (C), 2010–2026. Dashed lines indicate proposed trigger thresholds. Trigger A fallback threshold (1.5 fish per net night) shown in Panel A. Blue shading = baseline; amber = filling; grey = operational. N/A in Panel B = 2016 where Bendora CPUE = 0 and the ratio is incalculable.

Power analysis

A bootstrap power analysis was conducted to assess the probability that each trigger would activate as a function of the true underlying population mean. Per-net catches from the operational phase were resampled to simulate the distribution of annual means under varying true mean multipliers, with two-consecutive-year power estimated as the square of single-year detection probability. Figure 28 presents power curves for all three triggers across both single-year and two-consecutive-year activation. Table 7 summarises detection probabilities at key effect sizes.

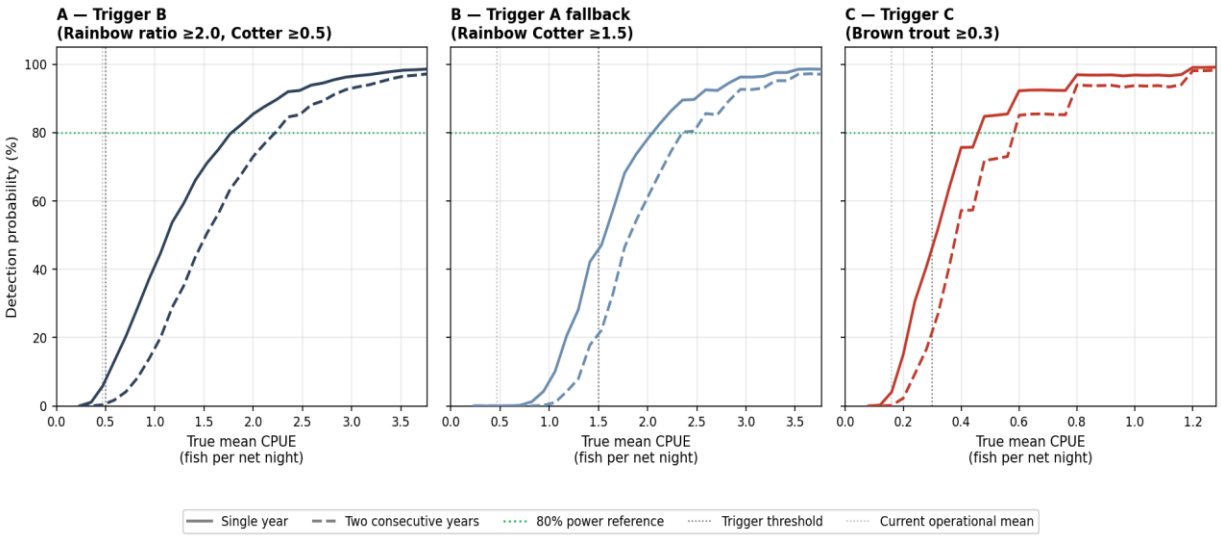

Figure 28. Bootstrap power curves for the three proposed triggers (A: Trigger B, B: Trigger A fallback, C: Trigger C). Solid lines = single-year detection probability; dashed lines = two-consecutive-year detection probability. Green dotted line = 80% power reference. Vertical dotted lines show the trigger threshold and current operational mean CPUE.

Table 7. Bootstrap detection probabilities (%) for each trigger at key effect sizes (multiples of current operational mean CPUE). ★ denotes the recommended activation rule. CPUE at 80% (2-yr) = true mean CPUE at which two-consecutive-year power reaches 80%. BT = Brown trout.

Trigger	2× mean	3× mean	4× mean	5× mean	CPUE at 80% (2-yr)	80% achieved?
B — Single year (ratio ≥2.0)	37%	54%	83%	92%	~1.88	**Yes (4×)**
B — Two consec. (ratio ≥2.0) ★	14%	29%	69%	84%	~2.24	**Yes (4.75×)**
A fallback — Single year (≥1.5 CPUE)	4%	21%	74%	90%	~2.12	**Yes (4.5×)**
A fallback — Two consec. (≥1.5 CPUE)	<1%	4%	54%	81%	~2.35	**Yes (5×)**
C — Single year (BT ≥0.3)	52%	85%	92%	97%	~0.48	**Yes (3×)**
C — Two consec. (BT ≥0.3) ★	27%	72%	85%	94%	~0.60	**Yes (3.75×)**

The power analysis reveals a fundamental constraint of the annual gill net program: the high inter-annual variability in salmonid CPUE means that moderate true population increases are difficult to reliably detect within two consecutive annual sampling events. Trigger B achieves 80% two-

consecutive-year power when Cotter CPUE reaches approximately 4.75 times the current operational mean (~2.24 fish per net night). Trigger C is better calibrated, reaching 80% power at approximately 3.75 times the current Brown trout mean (~0.60 fish per net night). Trigger A fallback requires approximately 5 times the current mean (~2.35 fish per net night) for 80% two-consecutive-year power and is by design a conservative safety net for extreme resurgence rather than a sensitive detector of moderate increases.

The two-consecutive-year rule is appropriate under the current downward trajectory of salmonid abundance, where the primary risk is false activation rather than missed detection. The framework is therefore best characterised as a conservative resurgence detector rather than a sensitive early-warning signal — a deliberate design choice reflecting the management cost of unnecessary intervention.

Recommendation

It is recommended that the proposed trigger framework, Trigger B as primary, Trigger A as defined fallback, and Trigger C for Brown trout, be ratified by Icon Water and the ACT Government and formalised in the next revision of the Cotter Reservoir Alien Fish Management Plan, fulfilling the commitment made in the 2018 plan. Trigger thresholds should be reviewed at five-year intervals as the monitoring dataset grows, using the bootstrap power framework described here to assess whether thresholds remain appropriate.

Bootstrap power analysis: statistical methods

Data preparation

All analyses used per-net-per-night raw catch counts from gill nets 1–10 only. The annual mean CPUE for each site and year was calculated as the arithmetic mean of all per-net-per-night catches within the sampling event (50 net-nights for Cotter Reservoir; 20 net-nights for Bendora Reservoir).

Bootstrap simulation

For each trigger and each candidate true mean multiplier:

- Observed per-net catches from the operational phase (2016–2026) were scaled by the multiplier to represent a scenario where the true population mean had increased by that factor relative to the current operational mean.
- From the scaled distribution, 50,000 simulated annual means were generated by bootstrap resampling with replacement at the appropriate sample size.
- Single-year detection probability was estimated as the proportion of simulated means exceeding the trigger threshold.
- Two-consecutive-year detection probability was estimated as the square of single-year probability, assuming independence between consecutive years (conservative, as positive autocorrelation would increase two-consecutive-year power).

For Trigger B, Cotter and Bendora annual means were simulated independently in each iteration; the ratio was computed as Cotter mean divided by Bendora mean, with iterations where Bendora mean equalled zero excluded. The joint probability of (ratio ≥2.0) AND (Cotter mean ≥0.5) was estimated from the same simulation. Power curves were computed at 0.25× intervals from 0.5× to 8.0× the current operational mean, using a fixed random seed (42) for reproducibility.

Limitations

Power estimates assume the operational-phase variance structure is representative of future sampling conditions. Water level-dependent variation in gill net catchability at Cotter Reservoir contributes substantially to inter-annual CPUE variability and is not explicitly modelled; to the extent that this source of variability could be controlled for (as in the main program's GAMM analyses), true power would be higher than estimated here. The independence assumption for two-consecutive-year power is conservative if true abundance is positively autocorrelated between 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 6). 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 8. 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	Raw electrofishing count data from five Cotter River impact sites and one unregulated reference site (Cotter Hut) were analysed within a BACI framework using Tweedie GAMs with a log-offset for electrofishing on-time, site-type × phase interaction, site-type-specific temporal smooths, and random effects for site and run. A secondary distance attenuation model (impact sites only) tested whether phase-related changes in abundance varied with distance from the reservoir headwaters (phase × dist_km interaction). Three recent-window comparisons (baseline vs 2026; baseline vs 2024–2026; 2023–2025 vs 2026) were evaluated by site type using the same modelling framework.

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. Rainbow trout abundance was assessed using a BACI design comparing five Cotter River impact sites against an unregulated reference site (Cotter Hut, ~62 km upstream past Bendora and Corin reservoirs) across baseline (2010–2013), filling (2014–2015), and operational (2016–2026) phases. Fish were sampled by backpack electrofishing; all available data were included with electrofishing on-time incorporated as a log-offset to account for variable effort across years. GAMs were fitted in mgcv using a Tweedie error distribution and log link, with site type, phase, and their interaction as fixed effects, site-type-specific temporal smooths, and random effects for monitoring site and electrofishing run. A secondary model tested whether the magnitude of phase-related change varied with distance from the reservoir headwaters (impact sites only). Three additional targeted comparisons evaluated baseline versus 2026, baseline versus 2024–2026, and 2023–2025 versus 2026. Full analytical details are provided in Appendix 1.

RESULTS

A total of 114 Rainbow trout were captured from six riverine sites on the Cotter River using fyke nets and backpack electrofishing in 2026 (Figure 27). Rainbow trout captured in 2026 from the Cotter River ranged in size from 61 – 350 mm FL (Figure 27). Rainbow trout were captured at every riverine site and numbers were relatively consistent between impact sites, with the exception of Spur Hole (n=2) and Burke Creek Crossing (n= 9) which were much lower than the other sites. 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 27 and Figure 28). Lengths of captured trout in 2026 were largely smaller fish (< 150 mm FL) which were most likely recruits from the previous spawning seasons (Figure 27 and Figure 28).

Rainbow trout abundance across the Cotter River catchment increased substantially relative to pre-enlargement baseline conditions, with significant nonlinear temporal trends detected at both Cotter River impact sites (s(year): edf = 12.0, $p < 0.0001$) and the Cotter Hut reference site (s(year): edf = 9.4, $p < 0.0001$; Figure 29). Observed CPUE at grouped impact sites increased from 0.55 fish per 100 seconds at baseline to 0.78 in 2026 and 1.08 over 2024–2026, while Cotter Hut increased from 1.03 at baseline to 1.20 in 2026 and 2.34 over 2024–2026 (Figure 29). The BACI analysis found no significant site_type × phase interaction for any phase contrast (baseline/filling: ratio = 12.5, $p = 0.197$; baseline/operational: ratio = 3.8, $p = 0.633$; filling/operational: ratio = 0.31, $p = 0.547$), indicating that impact and reference sites followed broadly similar temporal trajectories and that changes in trout abundance cannot be attributed specifically to reservoir enlargement at the grouped level.

Targeted comparisons against baseline confirmed significant system-wide increases at both site types. Over the 2024–2026 window, CPUE at impact sites was approximately 2.1-fold higher than baseline ($p < 0.001$) and at Cotter Hut approximately 2.3-fold higher ($p < 0.001$), consistent with a regional productivity response rather than a reservoir-specific effect. In 2026 alone, impact site CPUE was approximately 1.5-fold above baseline ($p = 0.041$) while Cotter Hut showed no significant

difference from baseline (p = 0.771). Notably, 2026 CPUE was significantly lower than the preceding 2023–2025 period at both impact sites (~2.2-fold lower, p < 0.001) and Cotter Hut (~2.8-fold lower, p = 0.004), indicating that 2023–2025 represented an exceptional peak in abundance system-wide, with 2026 representing a return toward more typical operational-phase levels.

Examination of site-level observed catch rates revealed a spatial pattern consistent with proximity to the reservoir influencing the magnitude of increase (Figure 29). Operational-phase CPUE was most elevated relative to baseline at the two sites closest to the reservoir — U/S ECR (0.45 km; 4.3-fold increase) and Vanitys Crossing (2.5 km; 5.5-fold increase) — with more moderate increases at Spur Hole (4.3 km; 1.9-fold) and Pipeline Road Crossing (9.8 km; broadly stable). The distance attenuation model confirmed this pattern statistically: at baseline, trout catch rates showed a significant positive relationship with distance from the reservoir headwaters (coefficient = 0.103 ± 0.025 SE, p < 0.001), reflecting the natural upstream-biased distribution of trout in an unimpacted river (Figure 30). This gradient attenuated significantly during the operational phase (phase × distance interaction: −0.065 ± 0.022 SE, p = 0.004), with the baseline slope flattening as sites adjacent to the reservoir gained proportionally more than those further upstream (Figure 30). The filling-phase attenuation was of similar magnitude but non-significant (p = 0.308), consistent with the effect developing progressively as the enlarged reservoir matured. Taken together, these results indicate that Rainbow trout abundance has increased across the Cotter River system since reservoir enlargement, with the strongest relative increases at sites closest to the reservoir — consistent with a trophic subsidy from the enlarged reservoir influencing adjacent riverine reaches and compressing the natural upstream-biased spatial distribution of trout toward the reservoir interface.

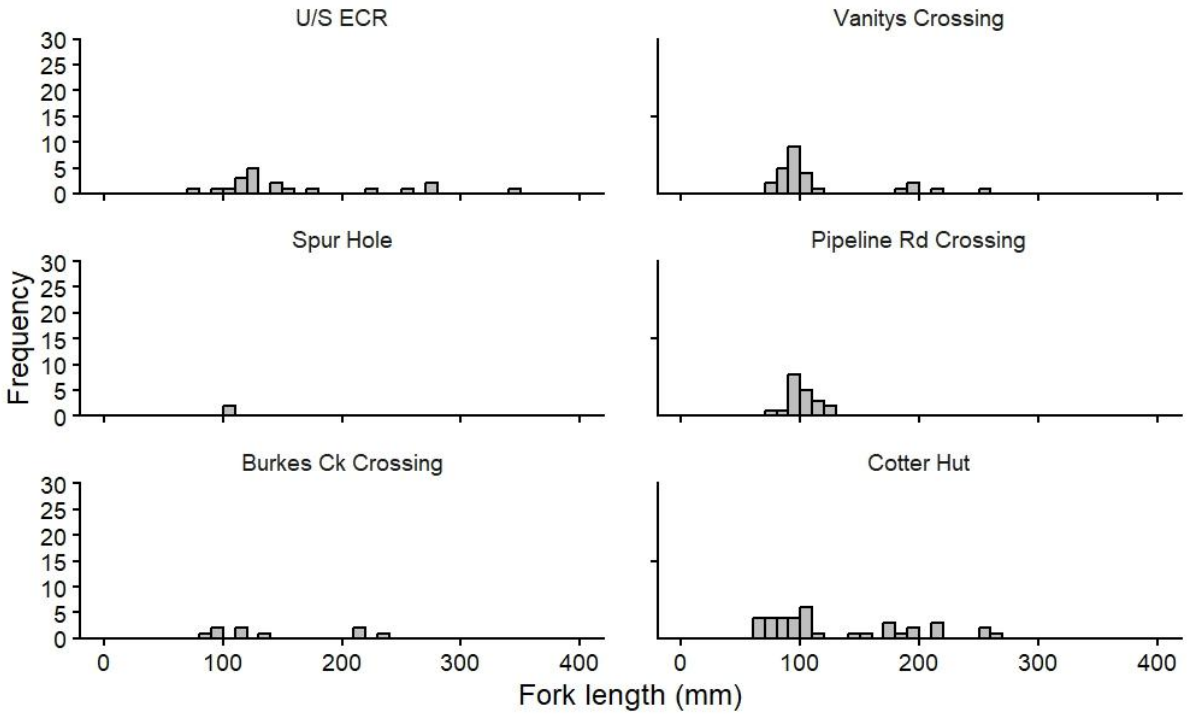

Figure 29. Length frequency of Rainbow trout captured from Cotter River sites in 2026 using fyke nets and backpack electrofishing (inclusive of additional backpack electrofishing effort).

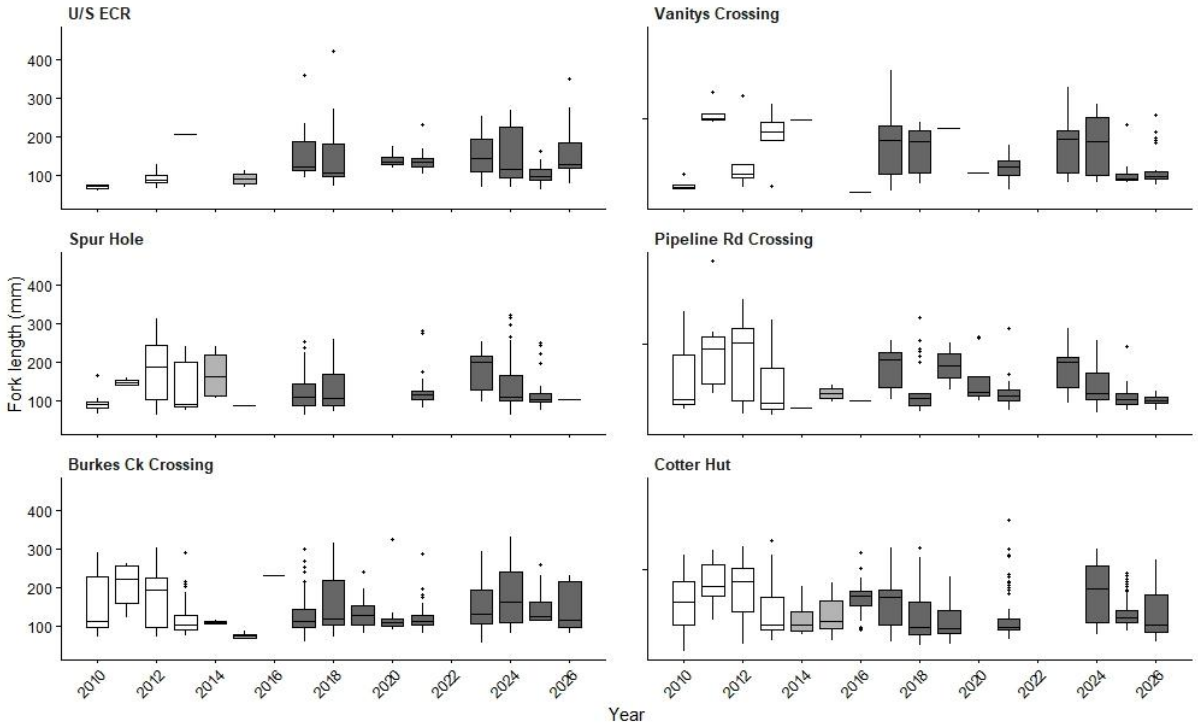

Figure 30. Boxplots of Rainbow trout captured in Cotter River from 2010 to 2026 (solid line = median, box represents 25 – 75th percentiles, bars represent minimum and maximum lengths and black circles represent outliers). Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2026). 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.

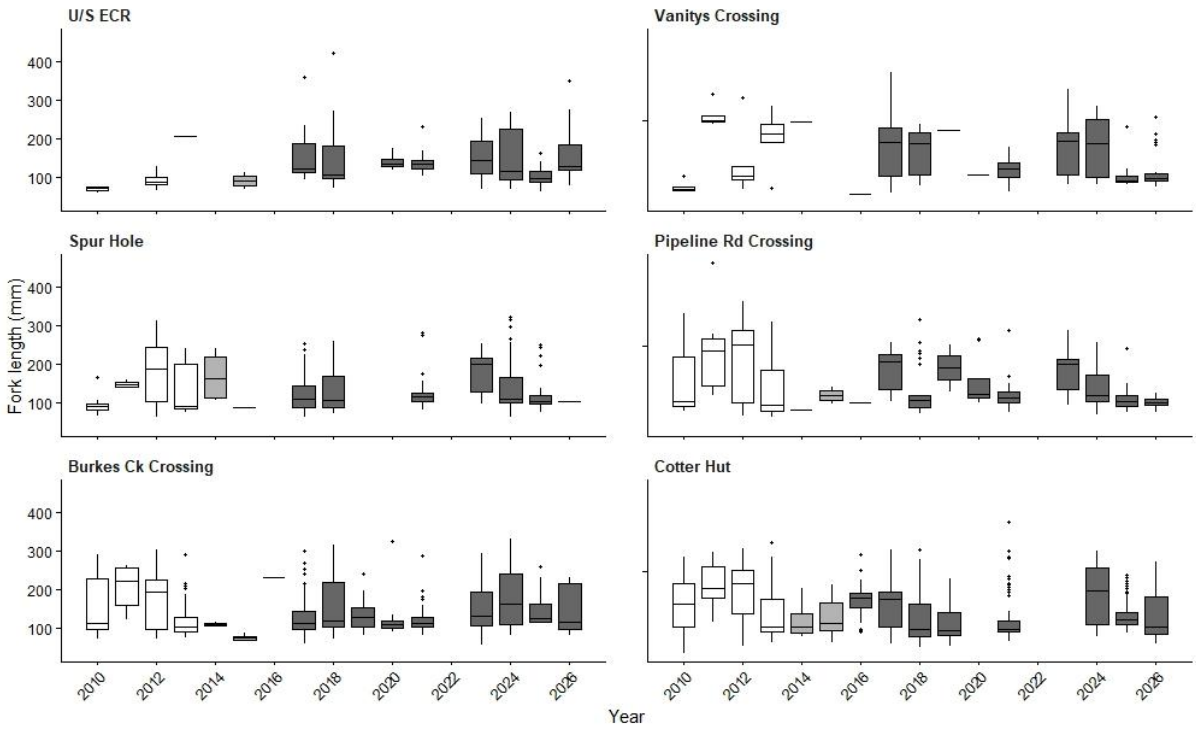

Figure 31. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Rainbow trout captured by backpack electrofishing (all effort) at five Cotter River impact sites (ordered by distance from Cotter Reservoir headwaters, closest to furthest) and Cotter Hut reference site (upper Cotter River, unregulated, ~62 km upstream of reservoir headwaters past Bendora and Corin reservoirs), 2010–2026. CPUE is expressed as fish per 100 seconds of electrofishing on-time, standardised for variation in sampling effort among runs. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Error bars represent 95% confidence intervals derived from among-run variation within each sampling year. Site panel labels include distance from Cotter Reservoir headwaters in kilometres. Cotter River sites were not sampled in 2020 (Cotter Hut only) or 2022 (all sites).

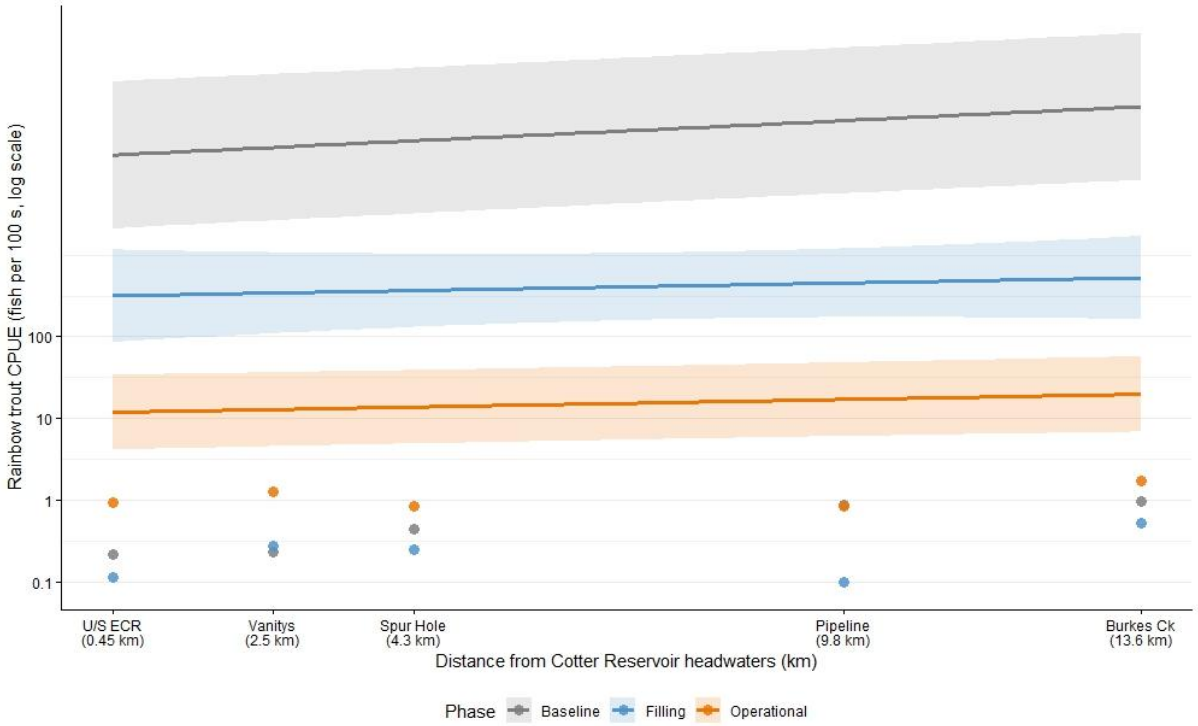

Figure 32. Modelled relationship between Rainbow trout catch-per-unit-effort (CPUE) and distance from Cotter Reservoir headwaters across three monitoring phases, derived from a Tweedie GAM fitted to impact site electrofishing count data with a log-offset for sampling effort (n = 587 runs; Cotter Hut excluded as it is not on the same spatial transect). Lines show model-predicted CPUE on a log scale at mean year and reference effort (273 seconds of on-time), with 95% confidence ribbons; site and run random effects are marginalised and line heights should be interpreted as relative indices rather than absolute abundance estimates. Points show observed mean CPUE (fish per 100 seconds) aggregated across all years within each phase for each site.

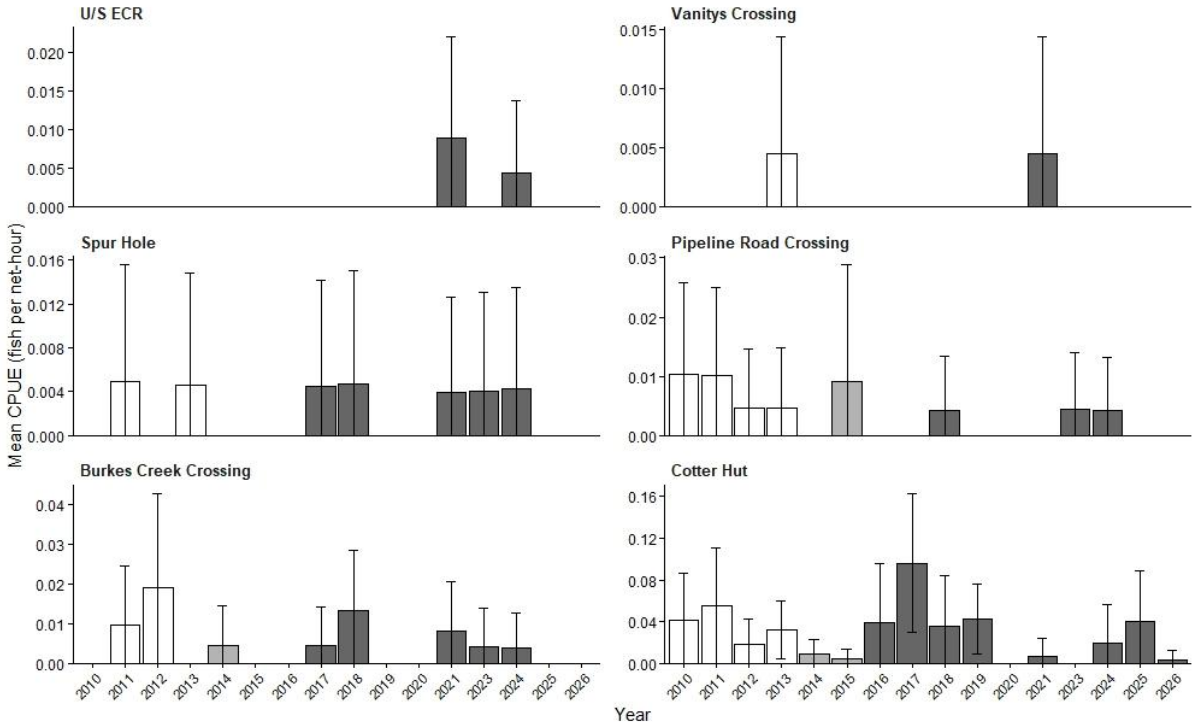

Figure 33. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Rainbow trout captured using fyke nets at five impact sites and one reference site (Cotter Hut) in the Cotter River, 2010–2026. CPUE is expressed as fish per net-hour, calculated from raw catch counts standardised for variation in soak duration among net deployments. Sites are ordered by increasing distance from the Enlarged Cotter Reservoir. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). 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

Rainbow trout are monitored in the Cotter River system as a recognised threat to native fish, particularly Macquarie perch and Two-spined blackfish, both listed as threatened species in the ACT. Alien salmonids are widely documented as a primary driver of native freshwater fish decline in south-eastern Australia, operating through predation, competition for food resources, and suppression of recruitment (Kaminskas, 2022). In this context, the substantial increase in Rainbow trout abundance relative to pre-enlargement baseline conditions observed across the Cotter River system represents an ecologically adverse outcome warranting careful interpretation and management consideration.

The BACI analysis found no significant divergence between impact and reference sites across phase transitions, indicating that the system-wide increase in trout abundance cannot be attributed exclusively to Cotter Reservoir enlargement. The synchronous peak across all sites in 2021, spanning locations from 0.45 km to 62 km from the reservoir, points to a regional recruitment event, most plausibly associated with improved hydrological conditions following the prolonged drought of 2014–2020. Complete population wipeouts recorded at multiple impact sites during drought years

(zero catches across all runs at Spur Hole in 2016, 2019, and 2020; U/S ECR in 2016 and 2019) confirm that the Cotter River Rainbow trout population is highly sensitive to low-flow conditions, with episodic crashes followed by rapid recovery when conditions improve. This boom-bust dynamic is consistent with documented responses of salmonid populations to hydrological variability (Poff et al., 1997, Elliott and Elliott, 2010).

Nevertheless, the distance attenuation model identified a significant spatial signal that warrants careful interpretation. At baseline, Rainbow trout catch rates increased naturally with distance from the reservoir. During the operational phase this gradient flattened significantly, with sites immediately adjacent to the reservoir showing the largest relative increases in abundance (U/S ECR: 4.3-fold; Vanitys Crossing: 5.5-fold), while U/S ECR and Vanitys Crossing maintained elevated catches through 2023–2024 as more distant sites declined from the 2021 regional peak. Two mechanisms may explain this pattern and are not mutually exclusive. First, the gradient flattening is consistent with the trophic subsidy hypothesis predicted by Todd et al. (2017) for the enlarged Cotter Reservoir, in which nutrient release during inundation was forecast to benefit trout growth rates and population expansion in adjacent riverine reaches. Šmejkal et al. (2023) further demonstrated that reservoirs act as source habitats for generalist predators that move into adjacent riverine reaches, with edge effects typically greatest at the immediate reservoir–river interface and attenuating with distance upstream. Second, density-dependent redistribution from upstream reaches at or near carrying capacity represents a plausible alternative mechanism. The baseline data show that Burkes Creek (0.978 fish per 100 seconds) and Pipeline Road (0.873) already supported the highest catch rates of any impact site, suggesting these reaches were approaching saturation prior to the operational phase. Following the 2021 system-wide recruitment peak, surplus individuals displaced from saturated upstream habitat may have dispersed downstream into lower-density reaches adjacent to the reservoir, a well-documented process in stream salmonids where intraspecific competition for food and space drives subordinate individuals into suboptimal downstream habitat (Grossman and Simon, 2020, Elliott, 1994). Distinguishing between these mechanisms is not possible from the current monitoring data and would require individual movement tracking or concurrent habitat capacity assessment; both processes may have operated simultaneously. Regardless of mechanism, the result, elevated trout abundance at sites immediately adjacent to the reservoir throughout the operational phase, has the same management implications.

The management implications of elevated trout abundance at the reservoir–river interface are significant. This zone coincides spatially with the primary Macquarie perch spawning reach and larval development corridor, and with habitat occupied by Two-spined blackfish throughout their life history. The elevated trout abundance documented at reservoir-adjacent sites relative to pre-enlargement baseline conditions, combined with the spatial overlap with the primary Macquarie perch larval development reach during December–January and habitat occupied by Two-spined blackfish throughout their life history, represents a substantially elevated predation risk that warrants targeted investigation during this critical window. Todd et al. (2017) using population modelling that even low levels of trout predation on Two-spined blackfish are sufficient to drive population decline. While Ebner et al. (2007) found no direct evidence of trout predation on Macquarie perch within Cotter Reservoir itself, post-enlargement trout abundance in the river reaches immediately upstream is substantially higher than at the time of that study, warranting reassessment of predation risk in the riverine context. The potential for Rainbow trout to impose

additional mortality on already constrained Macquarie perch recruitment is therefore a priority management concern.

RECOMMENDATIONS

Retain the current full-effort electrofishing approach (~1000 m per site) and continue to include all available data in analyses. The distance attenuation model introduced in this assessment provides ecologically informative spatial context and should be retained as a standard analytical component in future reporting cycles. Continued monitoring at all sites is essential given Rainbow trout's status as a recognised threat species, and longitudinal data will progressively allow density-dependent redistribution and reservoir trophic subsidy hypotheses to be distinguished as the system matures.

The elevated Rainbow trout abundance at reservoir-adjacent sites, combined with spatial overlap with key Macquarie perch spawning and larval development habitat, is identified as a priority concern. Targeted diet assessment of Rainbow trout at U/S ECR and Vanitys Crossing during December–January, as outlined in the predation assessment framework under Question 1, will directly address this risk. If predation on Macquarie perch larvae or Two-spined blackfish juveniles is confirmed at these sites, removal effort should be concentrated here given the combination of elevated trout abundance and proximity to critical spawning and nursery habitat.

If predation on Macquarie perch larvae or Two-spined blackfish juveniles is confirmed at reservoir-adjacent sites, targeted trout removal within key spawning and nursery reaches during December–January should be implemented. Removal effort should be concentrated at U/S ECR and Vanitys Crossing, where the combination of elevated trout abundance and proximity to key Macquarie perch spawning habitat presents the greatest risk. This approach is consistent with the staged predation management framework outlined in Chapter 1 and maximises selectivity by targeting the most vulnerable life-history window.

The current operational-phase CPUE at reservoir-adjacent sites (U/S ECR ~1.0–2.8 fish per 100 seconds; Vanitys Crossing ~1.1–4.9 fish per 100 seconds) is substantially elevated relative to baseline (~0.1–0.5 fish per 100 seconds) and represents the current reference point for threat assessment. Future surveys recording CPUE consistently above the operational-phase mean at these sites, combined with confirmed predation on native species, should constitute a trigger for active management intervention. Explicit thresholds should be defined prior to the next monitoring cycle in consultation with the ACT Government. Notably, if continued monitoring shows reservoir-adjacent CPUE declining toward baseline as upstream reaches recover from the 2021 peak, consistent with the density-dependent redistribution hypothesis, the management urgency would be reduced; this trajectory itself constitutes useful diagnostic information.

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

BACKGROUND

Trout are known predators of Two-spined blackfish (Lintermans et al., 2013) and are also reported to prey on Macquarie perch (Cadwallader, 1978, Broadhurst et al., 2018, 2019a)(Lintermans and Kaminskas unpublished data). Increases in trout abundance associated with reservoir enlargement and trophic upsurge may lead to greater downstream dispersal and elevated predation pressure in the Cotter River. Monitoring trout diet provides an important means of detecting changes in predation pressure on Two-spined blackfish and Macquarie perch, particularly as trout populations respond to post-impoundment conditions.

METHODS

Sampling design for Question 6 is a refinement of that conducted in the baseline monitoring program (Lintermans et al., 2013) (Table 7). 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 9. 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 27 Rainbow trout (≥150 mm FL) and no Brown trout were captured in 2026 (Table 8). Visual examination of stomach contents detected no Two-spined blackfish or Macquarie perch in any individuals sampled. Sampling effort increased substantially from 2017 onwards, resulting in higher numbers of trout examined compared to earlier years. However, trout abundance within the target size range declined markedly during 2019–2020 before increasing again in more recent years.

Table 10. Details of trout captured and examined for fish and fish remains from the Cotter River in 2026 via backpack electrofishing

Site	Species	Fork Length (mm)	Fish remains in stomach
U/S ECR	Rainbow trout (n=7)	158 – 350	No
Vanitys Crossing	Rainbow trout (n=5)	185 – 255	No
Burkes Creek Crossing	Rainbow trout (n=3)	215 – 231	No
Cotter Hut	Rainbow trout (12)	178 – 270	No

DISCUSSION AND CONCLUSIONS

Two-spined blackfish and Macquarie perch were absent from all trout stomachs examined in 2026, consistent with results from recent years despite increased sampling effort. Baseline data similarly indicated low predation rates on Two-spined blackfish and no detected predation on Macquarie perch (Lintermans et al., 2013). Although larger trout (>350 mm FL) are theoretically capable of preying on Macquarie perch up to approximately 180 mm total length (Ebner et al., 2007), such predation appears to be rare within the sampled population.

A temporary increase in predation on Two-spined blackfish was observed in 2019 (10.5%; 2 of 19 individuals), but no further predation events have been detected since 2020. This suggests that predation on post-larval native fish by trout is episodic and currently low under prevailing conditions. However, the current sampling approach is unlikely to detect predation on Macquarie perch larvae. Sampling occurs outside the larval drift period (late spring–early summer), and visual stomach content analysis is not sensitive to highly digested or small prey items. As a result, no conclusions can be drawn regarding predation on early life stages of Macquarie perch. This limitation is particularly important given evidence of recruitment suppression in Macquarie perch populations.

Development of molecular techniques to detect Macquarie perch DNA in trout stomach contents (e.g. eDNA metabarcoding; (MacDonald et al., 2014)) would substantially improve detection sensitivity. Application of such methods during the larval drift period would provide a more robust assessment of predation risk at the most vulnerable life stage. Overall, while predation on larger life stages of native fish appears to be low, uncertainty remains regarding predation during early developmental stages, particularly under conditions where trout abundance or size structure may change in response to reservoir dynamics.

RECOMMENDATIONS

Current sampling effort (up to 1 km of river or 20 trout per site) is appropriate and improves the ability to detect predation events. No changes to the existing monitoring methods are recommended at this time.

Visually detectable predation on post-larval native fish remains low; however, confidence in detecting predation on Macquarie perch larvae is limited. Given the likelihood that early life stages represent the most vulnerable period, further development of genetic techniques to detect Macquarie perch DNA in trout stomach contents is recommended. Pending results of predation on larvae at the reservoir–river interface (see Chapter 1 recommendations), targeted sampling during the larval drift and early juvenile settlement period (December–January) should be considered to better resolve predation risk at critical life stages within the Cotter River.

Although current predation rates are low, there remains potential for increased predation pressure under changing trout abundance or species composition (e.g. increased Brown trout). Accordingly, investigation and development of trout control mechanisms should continue so that targeted management actions can be implemented if elevated predation is detected in future monitoring (ACTEW Corporation, 2013, Lintermans, 2012).

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 trout populations in Cotter Reservoir are addressed in Question 4. Other non-native fish species present include Goldfish *Carassius auratus*, Oriental weatherloach *Misgurnus anguillicaudatus*, and Eastern gambusia *Gambusia holbrooki*, all of which favour still or slow-flowing habitats (Lintermans, 2002, 2007). Expansion of the reservoir increased the availability of suitable habitat for these species, and their abundances increased during the initial filling phase, consistent with trophic upsurge processes.

These species may interact competitively with native fishes, including Macquarie perch, through overlap in resource use (e.g. food and shelter), although they are not considered major predatory threats. Eastern gambusia may also exhibit aggressive behaviour towards native species (Lintermans, 2007). In contrast, increases in Goldfish and Oriental weatherloach abundance may indirectly influence higher trophic levels by supporting populations of predators such as trout and cormorants. Both species have been recorded in trout diets in Cotter Reservoir, with Goldfish comprising an important component (Ebner et al., 2007), and Goldfish are also a significant prey item in cormorant diets (Lintermans et al., 2011).

Ongoing monitoring of non-native fish populations, together with assessment of trout predation in the river (Question 6), provides important insight into fish community dynamics within the reservoir. Monitoring also enables early detection of additional non-native species not currently present upstream of Cotter Dam (e.g. Common Carp and 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 9). The changes from the baseline monitoring program are the removal of an urban lake reference site (where these non-native species are present/abundant).

Table 11. 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	Goldfish CPUE (raw fyke net counts, log[soak duration] offset) analysed using negative binomial GAMs (REML) within a before–after framework. Smooth temporal trend model tested for significant nonlinear change; phase model provided phase-level estimated means (delta method). Pulse model (2014–2016) retained for comparison; model selection by AIC. Three recent-window comparisons (2026, 2023–2025, 2024–2026) versus baseline. Diagnostics via DHARMa.

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.

Goldfish abundance in Cotter Reservoir was assessed using fyke net count data modelled within a before–after framework (baseline 2010–2013; filling 2014–2015; operational 2016–2026). Raw catch counts were analysed using negative binomial GAMs with log(soak duration) as an offset, a net deployment random effect, and REML estimation. Two primary models were fitted: a smooth temporal trend model (s(year)) to test for significant nonlinear change over time, and a phase model to quantify phase-level estimated means and contrasts via delta-method back-transformation. A pulse model (2014–2016 defined as pulse years) was retained for comparison with the previous analytical approach and model selection evaluated by AIC. Three additional models compared recent monitoring windows (2026, 2023–2025, 2024–2026) against baseline. Model fit was assessed using DHARMa simulation-based diagnostics.

RESULTS

Goldfish

Over three nights of fyke netting in Cotter Reservoir in 2026, 28 Goldfish were captured ranging from 40–156 mm FL (Figure 32). The majority of individuals were below 120 mm FL, most likely corresponding to 0+ and 1+ year-old age classes (Merrick and Schmida, 1984) (Figure 32). Goldfish CPUE differed markedly across the monitoring period, with observed mean raw CPUE of 0.081 fish per net-hour during baseline, 0.864 during filling, and 0.078 during the operational phase — a

pattern consistent with a pronounced but transient trophic upsurge response to reservoir inundation.

The smooth temporal trend model confirmed a highly significant nonlinear trajectory in Goldfish abundance across the 17-year monitoring period (s(year): edf = 4.82, χ^2 = 216.4, p < 0.0001; deviance explained = 33.4%), characterised by a sharp elevation during reservoir filling followed by a return to near-baseline levels throughout the operational phase (Figure 33). The phase model quantified the magnitude of this response: CPUE during filling was approximately 10.5-fold above baseline (p < 0.0001), while operational-phase CPUE was statistically indistinguishable from pre-enlargement conditions (p = 0.798). A complementary pulse model — defining 2014–2016 as a disturbance pulse coinciding with reservoir filling and the first operational year — provided an equivalent fit to the smooth model (AIC: smooth = 2736.2, pulse = 2736.8, phase = 2786.9), confirming that the temporal signal is best characterised as a short-term pulse rather than a sustained phase-level shift. CPUE during pulse years was approximately 11.8-fold above non-pulse years (p < 0.0001), with non-pulse CPUE not significantly different from baseline conditions (Figure 33).

Targeted comparisons of recent monitoring windows against baseline confirmed that Goldfish abundance has not only returned to pre-enlargement levels but has fallen below them in the most recent monitoring period (Figure 33). CPUE in 2026 was significantly lower than baseline (0.27-fold; 95% CI: 0.13–0.57; p = 0.0007), as was the 2024–2026 window (0.54-fold; 95% CI: 0.31–0.95; p = 0.033), while the 2023–2025 period did not differ significantly from baseline (0.66-fold; p = 0.146). Collectively, these results confirm a pronounced but transient increase in Goldfish abundance associated with reservoir filling, followed by a sustained return to, and in the most recent years a decline below, pre-enlargement catch rates.

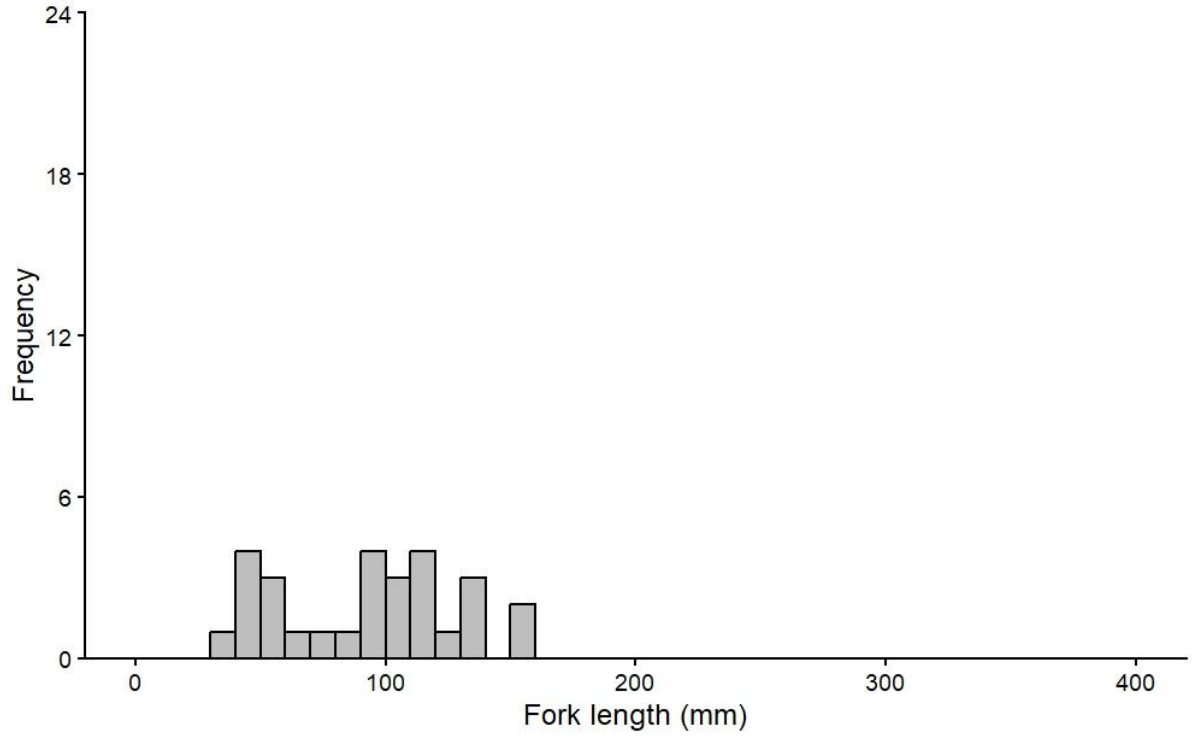

Figure 34. Length Frequency of Goldfish captured in the ECR in 2026 over three nights of fyke netting.

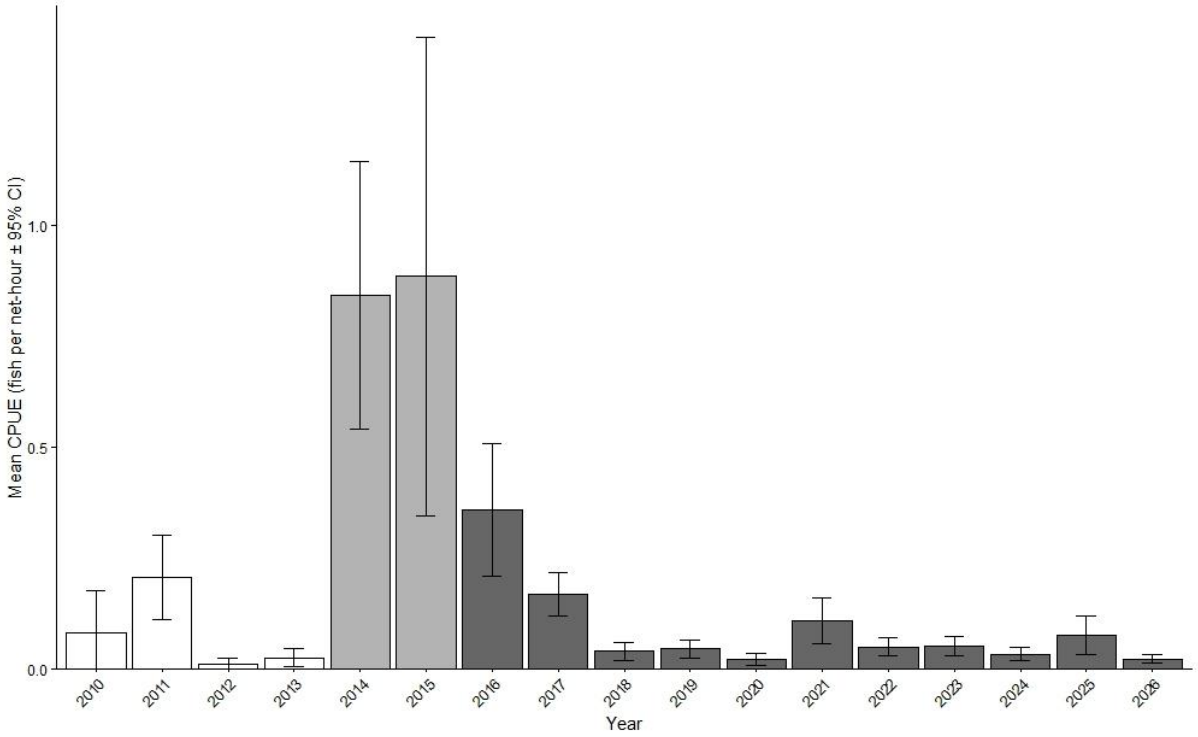

Figure 35. Mean annual catch-per-unit-effort (CPUE ± 95% confidence intervals) of Goldfish captured using fyke nets in Cotter Reservoir, 2010–2026. CPUE is expressed as fish per net-hour, standardised for variation in soak duration among net deployments. White bars indicate the baseline phase (2010–2013), light grey bars indicate the filling phase (2014–2015), and dark grey bars indicate the operational phase (2016–2026). Error bars represent 95% confidence intervals derived from among-net variation within each sampling year.

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 2026, 17 Oriental weatherloach were observed while boat electrofishing, this is the highest number observed since boat electrofishing commenced in 2014. To date, 147 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 three Eastern gambusia were observed whilst boat electrofishing in 2026, taking the overall tally to 1406 Eastern gambusia observed since 2014.

DISCUSSION AND CONCLUSIONS

Goldfish abundance remained low during operational years following a pronounced increase during reservoir filling and the early operational phase. This initial increase is consistent with a short-term response to enhanced resource availability associated with reservoir filling, likely driven by increased productivity and inundation of littoral habitats (Kimmel and Groeger, 1986, Ploskey, 1986, O'Brien, 1990). Such short-lived increases in fish abundance are characteristic of trophic upsurge responses in newly filled reservoirs.

The elevated abundances observed during 2014–2016 were not sustained, with CPUE during the operational phase as a whole not significantly different from baseline levels. However, targeted comparisons of the most recent monitoring windows indicate that Goldfish abundance has in fact declined below pre-enlargement baseline conditions in 2026 and over 2024–2026, suggesting a continuing downward trend within the operational phase rather than simple stabilisation. This indicates that the initial productivity pulse associated with filling has fully attenuated and the system has transitioned to a condition of relatively low, and recently declining, Goldfish abundance. As noted previously, the temporary increase in Goldfish abundance during filling likely contributed to the initiation of cormorant breeding within the reservoir, which has persisted since 2014 (see Question 8). The longer-term implications of sustained low Goldfish abundance for cormorant foraging dynamics and potential dietary shifts toward native species warrant continued attention.

The increase in Goldfish abundance during filling coincided with similar increases in Brown Trout and Rainbow Trout (Chapter 4), indicating a system-wide response to reservoir development. This pattern is consistent with a trophic upsurge, where increased productivity during inundation drives concurrent increases across multiple trophic levels. While trout responses are primarily attributable to enhanced resource availability, elevated Goldfish abundance during this period may have contributed additional prey resources, either directly or indirectly through increased secondary production. This suggests that trout and Goldfish responses were driven by shared environmental processes, with potential trophic linkages reinforcing increased fish abundance during the filling phase.

Low catch rates of Oriental weatherloach and Eastern gambusia are unlikely to reflect true population sizes, particularly for Eastern gambusia. Instead, these low capture rates are attributed to limitations of the sampling methods (fyke nets and bait traps), which are not well suited to small-bodied or elongated species. Eastern gambusia were frequently observed in large schools in shallow habitats, particularly around the boat ramp and inundated roadways, consistent with known habitat preferences (Lintermans, 2007, Pyke, 2005, Macdonald and Tonkin, 2008). Although these species are likely locally abundant, their limited representation in monitoring data is not considered critical, as the primary focus of this assessment is Goldfish and their role in supporting higher trophic levels.

RECOMMENDATIONS

Sampling methods are considered adequate for monitoring Goldfish abundance in Cotter Reservoir, and no changes are recommended at this time.

Low catch rates for Oriental weatherloach and Eastern gambusia reflect methodological limitations rather than true abundance. Alternative sampling approaches (e.g. seine netting for gambusia, backpack electrofishing for weatherloach, or eDNA) could improve detection; however, given their limited management relevance, additional targeted sampling is not recommended at this stage.

The increase in Goldfish abundance during 2014–2016, followed by a decline from 2017 onwards, indicates that the resource-driven boom associated with reservoir filling has largely ceased. While the reduction in Goldfish abundance, together with initial evidence of Macquarie perch predation by trout, highlights potential shifts in food web dynamics, no management intervention specific to Goldfish is currently warranted. Continued monitoring is recommended to track longer-term trends and potential trophic interactions.

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 10).

Table 12. 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	Cormorant abundance and distribution were analysed using generalised linear mixed models (GLMMs; negative binomial distribution) to test effects of management phase, reservoir section, and species identity, including interactions and seasonal variation. Changes in community composition were assessed using PERMANOVA.

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 (Icon Water Limited, 2019). Cormorant abundance data were analysed using generalised linear mixed-effects models (GLMMs) with a negative binomial error distribution to test for effects of reservoir phase, spatial section, and species identity, including their interactions. Survey identity was included as a random effect to account for repeated monthly sampling, and month was included as a fixed effect to control for seasonality. Differences among phases and periods were examined using estimated marginal means. Changes in community composition were assessed using permutational multivariate analysis of variance (PERMANOVA) based on Bray–Curtis dissimilarities.

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 34). 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 34). Since filling began, abundances of both Great cormorant and Little black cormorant have been stable, though with some definitive seasonal fluctuations (Figure 34). 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 34 and Figure 35).

Cormorant abundance varied substantially across reservoir phases, with strong species-specific responses and clear spatial restructuring across the reservoir (Figure 35). Visual inspection of temporal trends highlights pronounced increases in abundance during the filling phase for some species, followed by divergent trajectories under operational conditions. Great and Little pied cormorants exhibited significant increases in abundance during the filling phase relative to baseline conditions (both $p < 0.001$; Figure 35). Following this peak, Great cormorant abundance declined to levels not significantly different from baseline ($p = 0.70$), whereas Little pied cormorants remained significantly elevated under operational conditions ($p < 0.001$). In contrast, Little black cormorants showed no difference between baseline and filling phases ($p = 0.87$) but declined markedly during the operational phase relative to both baseline and filling periods (both $p < 0.001$; Figure 35).

Strong evidence of spatial redistribution was detected, with highly significant phase × section interactions ($p < 0.001$). As illustrated in Figure 35, sections that supported relatively high abundances during the baseline period frequently declined in importance following reservoir filling, particularly during the operational phase. These patterns indicate substantial reorganisation of habitat use across the reservoir. Species-specific redistribution was evident, with significant higher-order interactions indicating that changes in spatial patterns differed among species. Little pied cormorants expanded across multiple sections during and after filling, whereas Little black cormorants declined broadly across sections under operational conditions (Figure 35).

Comparisons between baseline and recent conditions (2023–2026) further demonstrated that the system has not returned to its pre-enlargement state. Great cormorant abundance was significantly lower in recent years compared to baseline (p = 0.015), while Little pied cormorants did not differ significantly (p = 0.11), suggesting persistence at elevated levels. Little black cormorant abundance in recent years was low and highly variable, reflecting a sustained decline relative to earlier phases. At the community level, reservoir phase had a significant effect on assemblage structure (PERMANOVA: F = 10.59, R^2 = 0.0766, p = 0.001), indicating consistent but moderate shifts in species composition associated with reservoir enlargement.

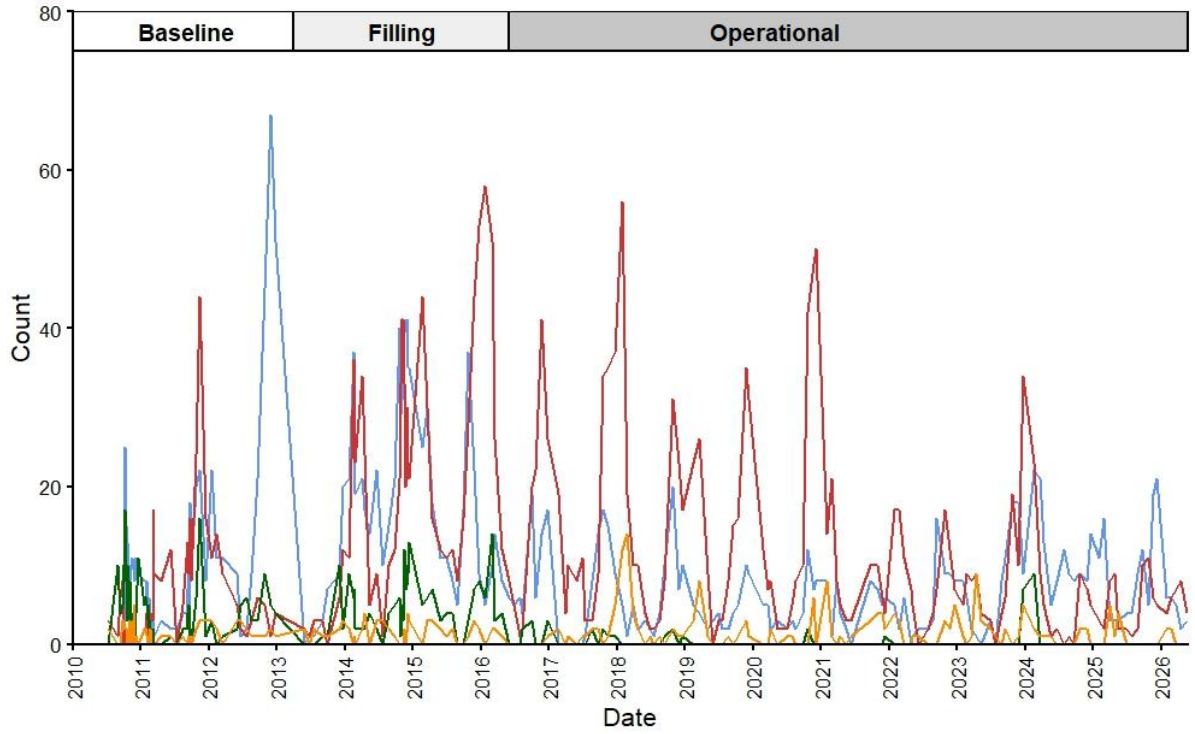

Figure 36. Monthly abundances of each common piscivorous bird species on the Cotter Reservoir between July 2010 and May 2026 (Great cormorant – blue line; Little black cormorant – dark green line; Little pied cormorant – red line; Darter – orange line).

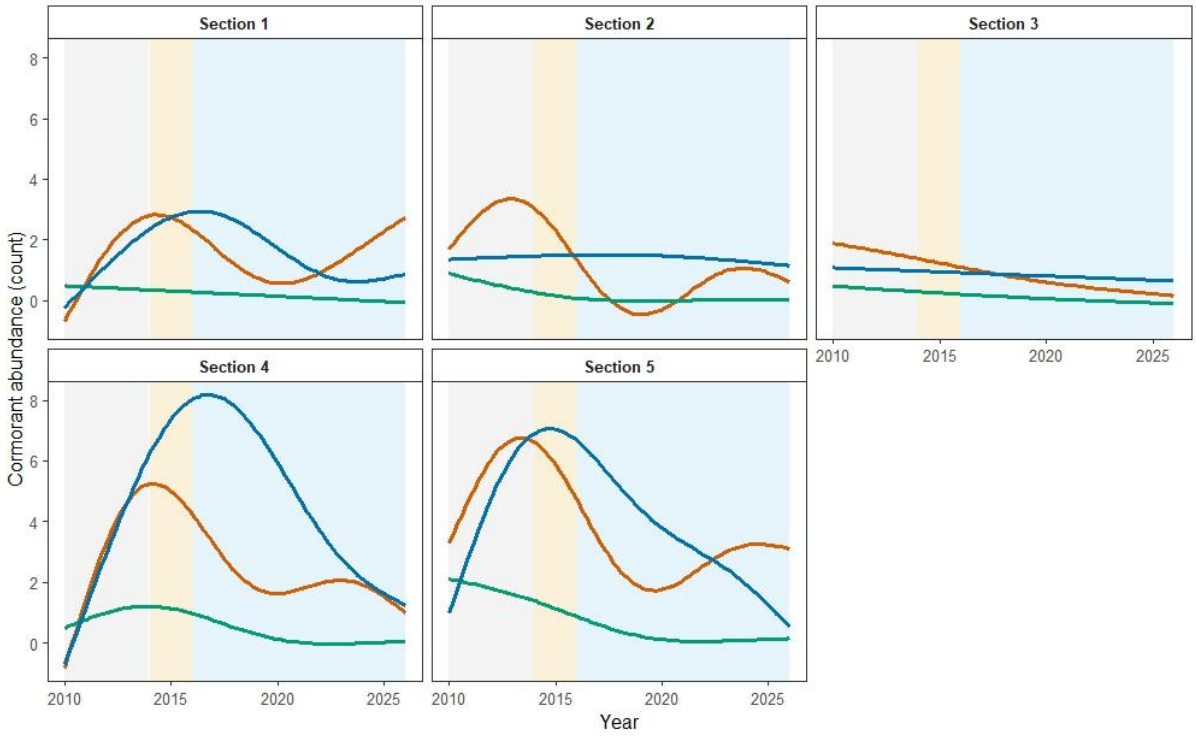

Figure 37. Temporal trends in cormorant abundance across five sections of Cotter Reservoir from 2010 to 2026. Lines represent smoothed trends (GAM) for Great cormorant (orange), Little black cormorant (green), and Little pied cormorant (blue). Shaded background regions indicate reservoir management phases: baseline (2010–2013), filling (2014–2015), and operational (2016 onwards).

For the 12th 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 36), 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.

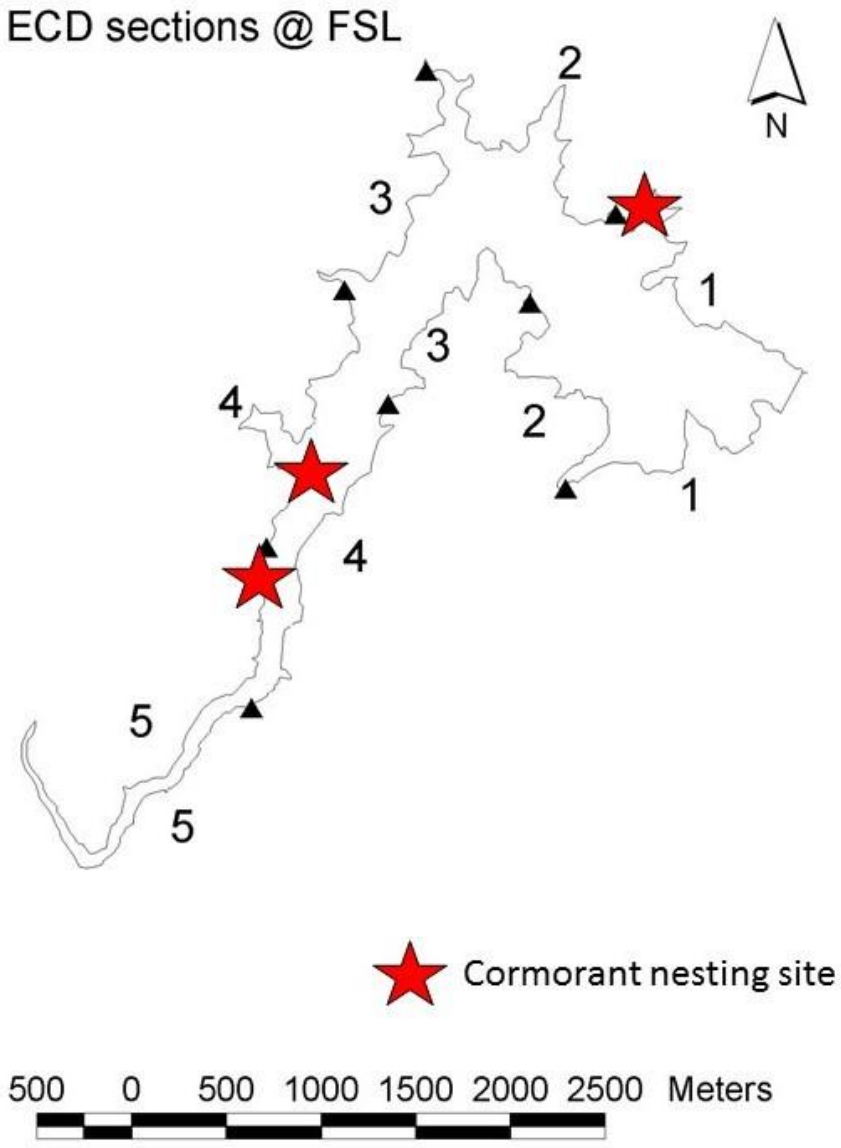

Figure 38. Map of the enlarged Cotter Reservoir shoreline sections with Cormorant nesting colony locations.

DISCUSSION AND CONCLUSIONS

Reservoir enlargement produced substantial and persistent changes in cormorant abundance, distribution, and assemblage structure, consistent with well-documented ecological effects of impoundment (Kimmel and Groeger, 1986, Thornton et al., 1990). The marked increases in Great and Little pied cormorants during the filling phase are consistent with the trophic upsurge hypothesis, whereby nutrient release from flooded terrestrial vegetation elevates primary and secondary productivity and prey availability (Grimard and Jones, 1982, Kimmel and Groeger, 1986). Inundation of large trees also provided roosting and nesting substrates not available under baseline conditions, contributing to the establishment of breeding colonies, most notably Little pied cormorant, which has bred at the reservoir in every year since filling commenced.

Following this peak, species exhibited divergent trajectories. Little pied cormorant abundance remained significantly elevated under operational conditions, with seasonal influxes concentrated in upper reservoir sections and a persistent breeding colony, despite apparent declines in peak abundance since approximately 2019. Little black cormorants declined substantially under operational conditions, likely reflecting reduced foraging efficiency in deeper, structurally altered water. Great cormorant abundance has declined to levels below baseline in recent years (2023–2026), yet the species was confirmed breeding for the first time in 2022, ecologically significant given that Great cormorant is recognised as the primary avian predation threat to Macquarie perch (Ryan, 2010, Lintermans et al., 2011).

Strong phase × section interactions indicate fundamental reorganisation of spatial habitat use following enlargement. All species have historically concentrated in upper, shallower reservoir sections where depths less than 5 m predominate (Ropert-Coudert et al., 2006, Dorfman and Kingsford, 2001), the same areas posing the greatest predation risk to Macquarie perch (Ryan, 2010, Ryan et al., 2013). The expansion of Little pied cormorants into additional sections during and after filling likely reflects the location of nesting colonies and associated roost sites established post-inundation.

A critical contextual consideration is the substantial decline in Goldfish abundance since the filling phase (see Question 7 above). Goldfish are a favoured cormorant prey item at Cotter Reservoir (Lintermans et al., 2011, Miller, 1979), and their decline may be driving dietary shifts toward Macquarie perch, particularly given the energetic demands of active breeding colonies of both Little pied and Great cormorants. This represents a plausible and concerning pathway for elevated predation pressure on a threatened species, and re-examination of cormorant diet composition is considered a priority.

At the community level, PERMANOVA confirmed a significant effect of reservoir phase on assemblage structure. Comparisons between baseline and recent conditions (2023–2026) demonstrate that the system has not returned to its pre-enlargement state after approximately a decade of operation, consistent with evidence that large-scale hydrological modification can produce persistent shifts in ecosystem structure (Poff et al., 2007). The cormorant assemblage at Cotter Reservoir appears to have stabilised in an alternative state relative to pre-enlargement conditions.

Collectively, the persistence of altered abundance patterns, confirmed breeding activity, declining alternative prey availability, and spatial concentration of foraging effort in high-risk areas highlight the need for continued vigilance and adaptive management of cormorant–fish interactions.

RECOMMENDATIONS

The current monitoring program is considered adequate. The long-term monthly dataset provides strong temporal resolution and has successfully detected both short-term trophic responses and longer-term assemblage shifts. Continuing this framework without modification is recommended, with emphasis on maintaining consistency in methods and spatial coverage.

The confirmed establishment of a Great cormorant breeding colony, combined with declining Goldfish availability and the continued breeding presence of Little pied cormorant, warrants a more cautious posture than in previous reporting periods. While overall abundance has broadly stabilised, the convergence of reduced alternative prey and active breeding colonies of the species posing greatest risk to Macquarie perch represents a credible pathway for increased predation pressure.

Re-examination of cormorant diet composition is recommended as a priority. If dietary analysis confirms a shift toward Macquarie perch, targeted management of breeding colonies consistent with the ECR Cormorant Management Plan should be considered. In the interim, a precautionary adaptive management approach is recommended, with management trigger thresholds reviewed in light of the Great cormorant breeding confirmation and the changing prey landscape. Active control measures are not recommended at this stage in the absence of direct evidence of elevated predation pressure.

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 13. 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 38). The first, located on the western shore just downstream of Bracks Hole road, measured approximately 2 x 2 m (4m^2) in area (Figure 38). The other two were located next to each other near the old boat ramp road on the eastern shoreline of the reservoir (Figure 38). The largest of the two, measured approximately 9 x 4 m (36 m^2) and formed the densest of the three stands (Table 12, Figure 38). The third comprised two loosely grouped sparse stands of approximately 2 x 2 m (4 m^2) each (Table 12). An additional stand of Cumbingi was detected in May 2023, in the bay just south of Bracks Hole (Figure 38). This is a relatively small and sparsely vegetated stand, measuring approximately 2 x 4 m. All four stands showed signs of stress (browning) in 2026, especially those located in the old boat ramp road bay (Figure 37).

Figure 39. Photograph of the browning Cumbungi stand located in the old boat ramp bay on the Eastern shoreline of the ECR taken in April 2026 (photo: Ben Broadhurst).

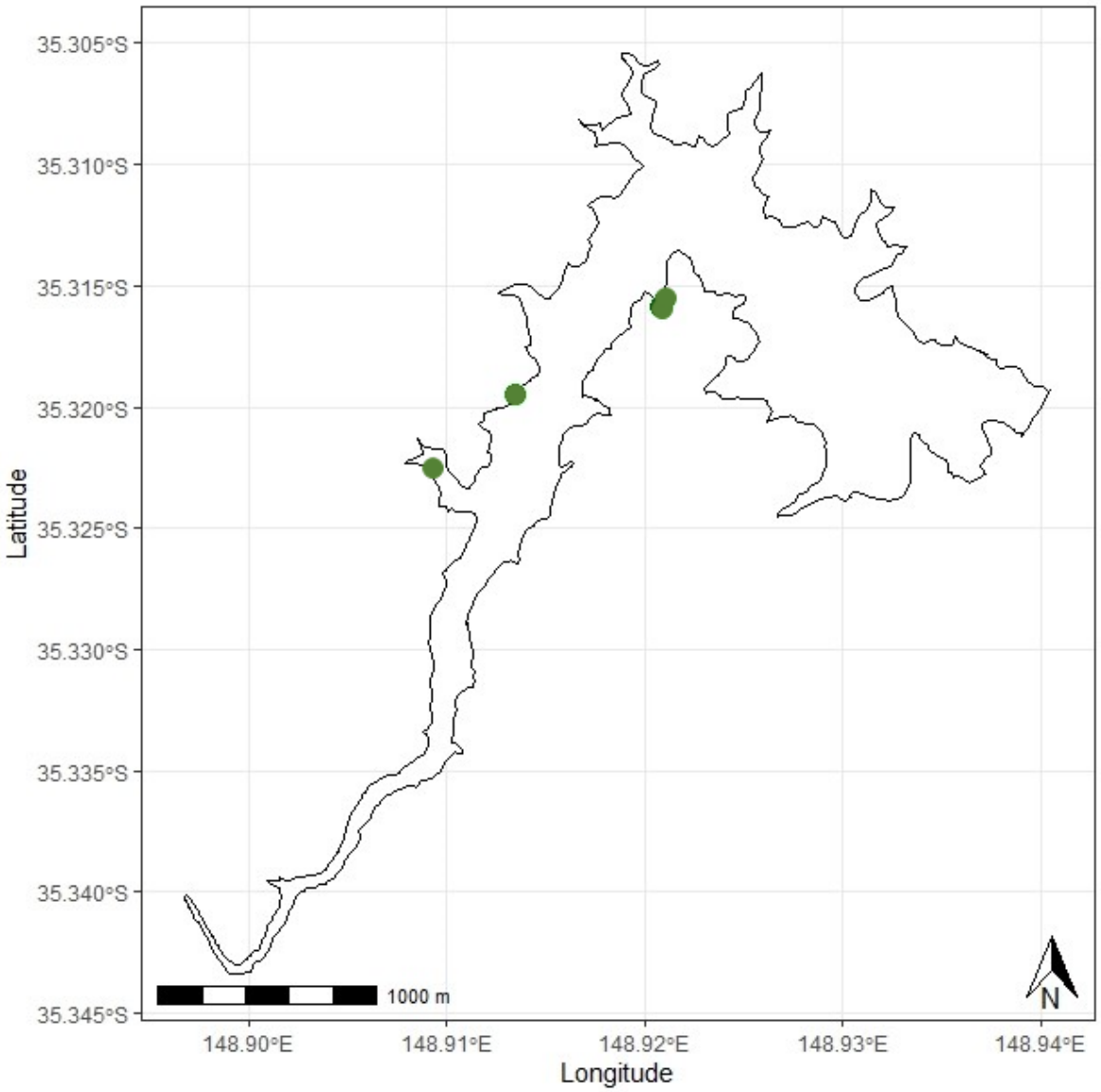

Figure 40. Map of the location of emergent macrophyte stands (green circles) along the shoreline of the Cotter Reservoir as monitored in March - May 2026.

Table 14. Details of each emergent macrophyte stand in the Cotter Reservoir as of May 2026 (including stand size change since last annual estimate).

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

QUESTION 10: 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

Food resource sampling was conducted biannually in autumn and spring in the enlarged Cotter Reservoir (ECR) following Norris et al. (2012). Three replicate samples of each available edge habitat type (bare shoreline, rocky shoreline, timber, and macrophytes) were collected from each of three reservoir sections (downstream, mid, upstream). Samples were taken along 10 m transects using a 250 µm mesh sweep net and preserved in 70% ethanol.

In the laboratory, samples were rinsed and processed using a two-stage approach. Larger invertebrates were first selected by eye from whole samples under magnification to quantify coarse-scale abundance. The remaining material was then subsampled (10%) using a 100-cell grid subsampler (Marchant, 1989), with randomly selected cells examined under a stereomicroscope to quantify smaller taxa. This approach enabled standardised estimation of abundance across both large and small-bodied invertebrates. Taxa were identified to order level for aquatic groups, with terrestrial material classified into a general category.

Open-water invertebrates were sampled using plankton tows in each reservoir section. A weighted 250 µm mesh net (300 mm diameter) was towed for 50 m at approximately 1 m depth, sampling 3.54 m^3 per tow. Samples were processed using the same subsampling approach described above.

Table 15. 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	Macroinvertebrate community composition was analysed using a hierarchical, size-structured approach and compared among reservoir phases, seasons, and habitats using PERMANOVA and ordination methods. Focal prey groups, including Decapoda (large-bodied) and microinvertebrates (Cladocera and Copepoda), were analysed using linear models, with temporal comparisons used to assess current (2026) conditions relative to baseline and recent periods.

Macroinvertebrate communities and associated food resources were analysed using a hierarchical, size-structured approach to assess responses across trophic levels. Macroinvertebrates were separated into coarse and subsample fractions, and community composition was analysed using PERMANOVA and ordination methods. Decapoda were analysed as a focal large-bodied prey taxon, and microinvertebrates (Cladocera and Copepoda) were analysed as indicators of lower trophic level dynamics using linear models. Temporal comparisons of decapod abundance were also undertaken to evaluate the most recent monitoring period relative to baseline and recent conditions (see Appendix 1 for detailed analytical methods).

RESULTS

Edge samples - Coarse pick and subsample

Macroinvertebrate community composition differed significantly among reservoir phases and seasons in both coarse pick and subsample fractions, although the strength and contributing taxa differed between fractions. In the coarse pick samples, PERMANOVA identified significant effects of Phase ($F_{2,78}$ = 13.30, p = 0.001) and Season ($F_{1,78}$ = 7.53, p = 0.001), with a significant Phase × Season interaction (p = 0.031). Phase accounted for the largest proportion of variation in community composition (21.5%), compared with 6.1% explained by season. Multivariate dispersion differed among phases ($F_{2,93}$ = 3.93, p = 0.023), and pairwise comparisons confirmed significant differences among all phases, with the greatest separation between baseline and operational conditions. SIMPER analysis indicated that differences between baseline and filling phases were primarily driven by Decapoda and terrestrial invertebrates, whereas differences between baseline and operational

phases were associated with increases in Chironomidae and changes in Hemiptera, Diptera, and Ephemeroptera. Ordination supported these patterns, with clear separation of baseline communities from filling and operational phases. In the subsample fraction, community composition also differed significantly among Phase, Season, and Habitat, with phase again explaining the largest proportion of variation. Differences were driven primarily by smaller-bodied and planktonic taxa, including Copepoda, Cladocera, Chironomidae, and Oligochaeta.

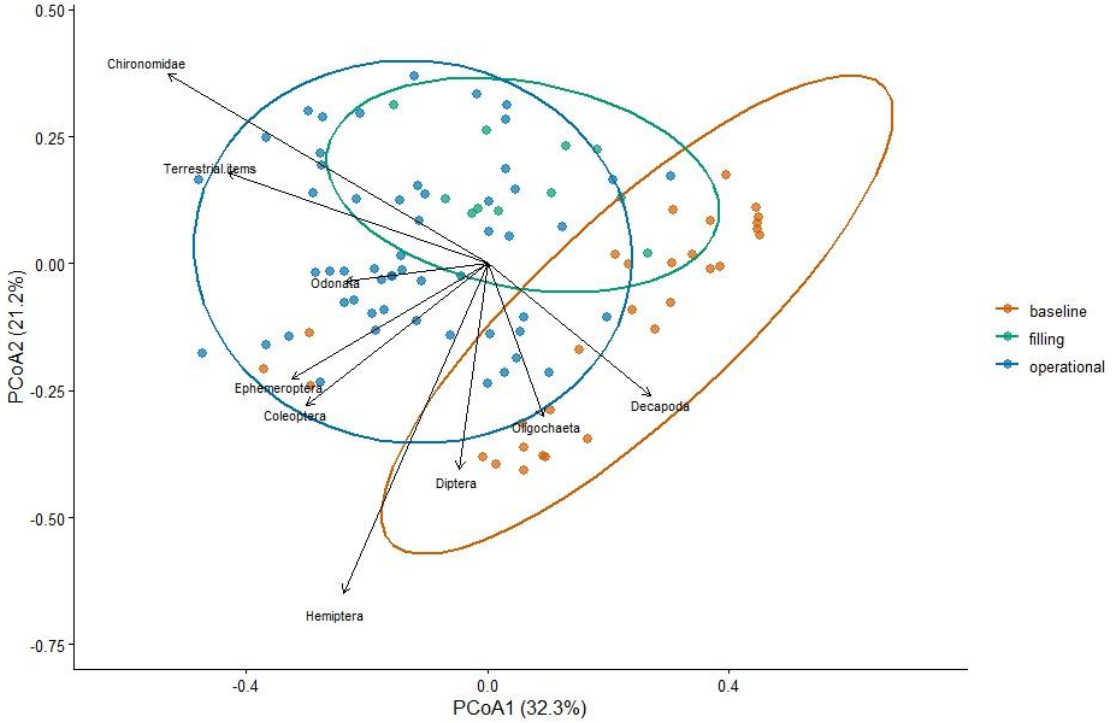

Figure 41. Principal coordinates analysis (PCoA) ordination of macroinvertebrate communities based on coarse pick across reservoir phases based on Bray–Curtis dissimilarities. Points represent samples coloured by phase, with ellipses showing group dispersion. Arrows indicate taxa significantly associated with community variation (envfit, $p < 0.05$). Axes explain 32.3% (PCoA1) and 21.2% (PCoA2) of the variation.

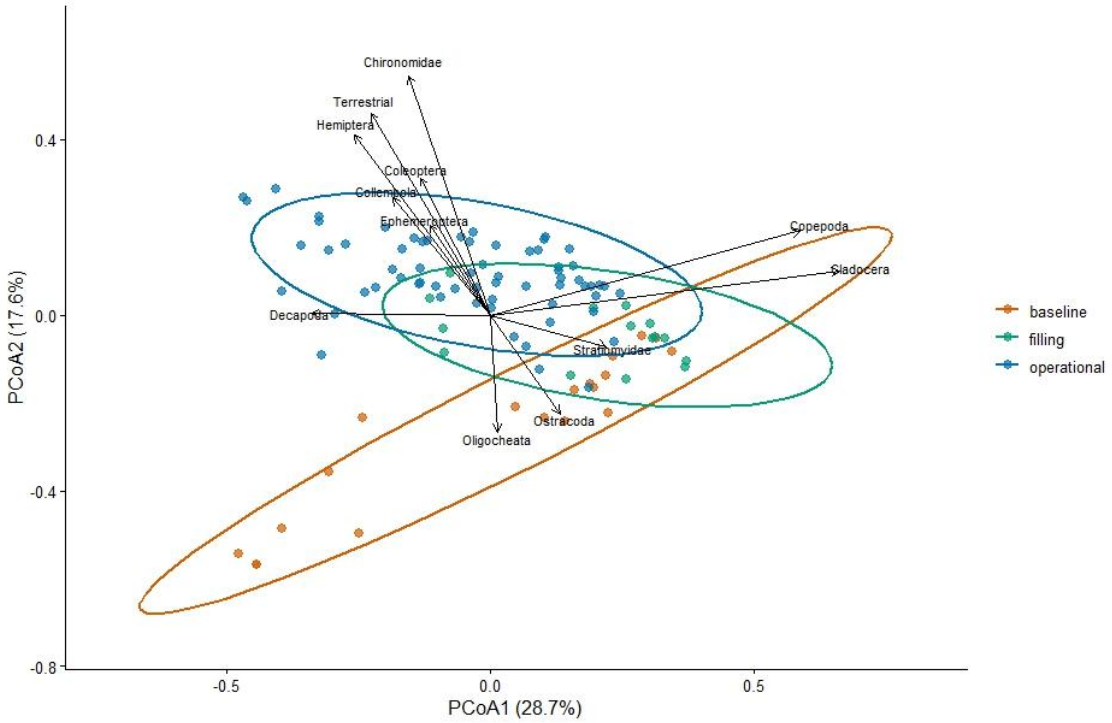

Figure 42. Principal coordinates analysis (PCoA) of macroinvertebrate community composition from the subsample fraction across reservoir phases, based on Bray–Curtis dissimilarities of $\log_{10}(x + 1)$-transformed abundance data. Points represent individual samples coloured by phase (baseline, filling, operational), with ellipses indicating group dispersion (±1 SD). Vectors represent taxa significantly correlated with ordination axes (envfit, $p < 0.05$), with arrow length proportional to the strength of correlation. The first two axes explained 28.7% and 17.6% of the variation, respectively.

Despite strong phase effects at the community level, Decapoda abundance did not vary significantly among phases or habitats, but showed a strong seasonal effect, with substantially higher abundance in autumn than in spring ($p < 0.001$) (Figure 41). The proportional contribution of Decapoda similarly declined in spring. Temporal analysis of decapod abundance indicated that the current monitoring period (2026) was significantly lower than baseline conditions ($F_{1,31} = 7.29$, $p = 0.011$), indicating a reduction in this key prey resource relative to historical reference conditions (Figure 41). Comparisons with more recent monitoring periods showed weaker and non-significant differences, although abundance tended to be lower in the current period relative to both the preceding three years ($p = 0.079$) and the broader recent period ($p = 0.057$). Across all comparisons, season remained a strong predictor of abundance, with no effect of habitat detected.

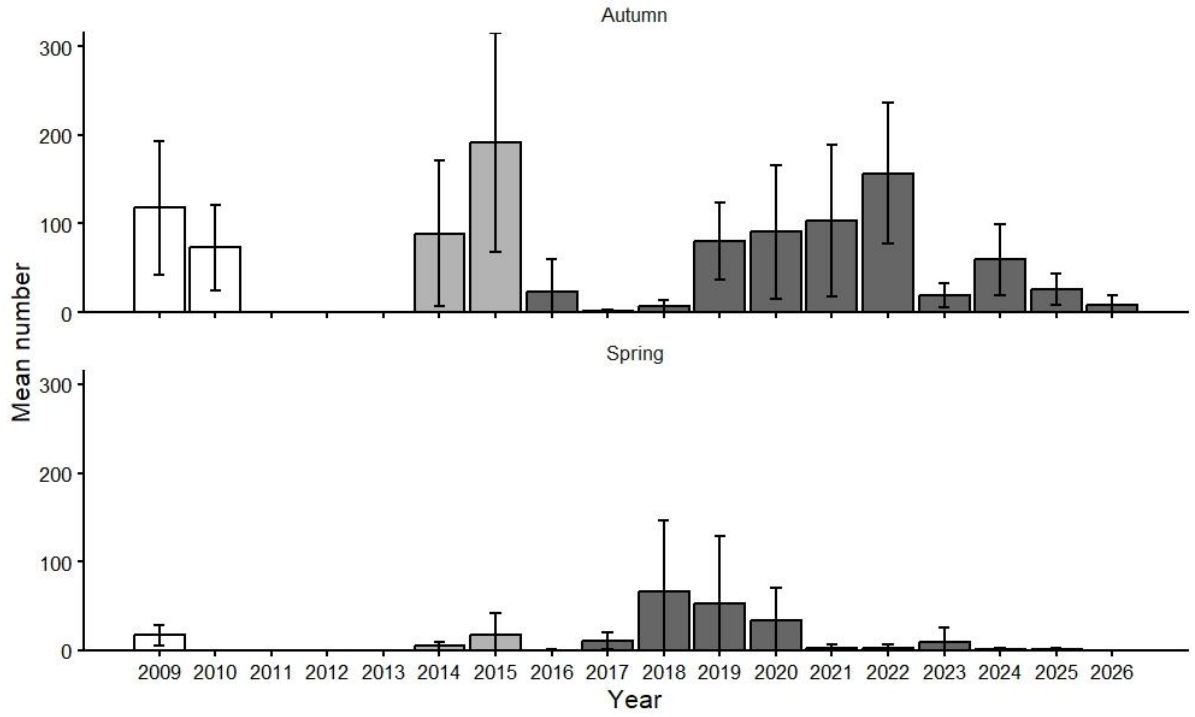

Figure 43. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of decapods collected from coarse pick edge samples taken from ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2026; dark grey bars) monitoring periods for autumn and spring. Note: Spring 2026 has not been sampled at the time of reporting.

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. Microinvertebrate abundance exhibited weak and inconsistent patterns, with no significant effects of phase, season, or their interaction for Copepoda or total zooplankton abundance, and only marginal trends for Cladocera. Overall, microinvertebrate abundance and composition showed limited response to reservoir phase (Figure 42).

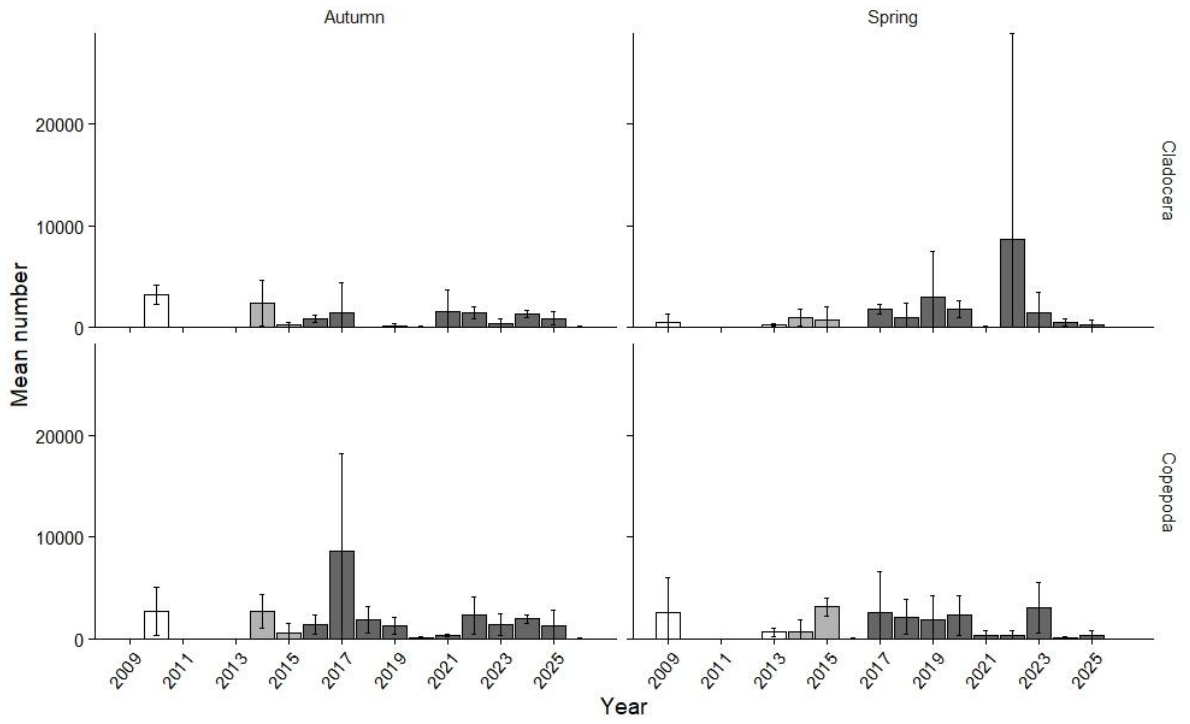

Figure 44. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of each microcrustacean taxa collected in autumn and spring of each phase of monitoring phase in ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2026; dark grey bars) monitoring periods. Note: Spring 2026 had not been collected at time of reporting.

DISCUSSION AND CONCLUSIONS

Reservoir phase exerted a strong influence on macroinvertebrate community composition, with consistent restructuring observed across both coarse and subsample fractions. These changes reflect shifts in both taxonomic composition and size structure of the macroinvertebrate assemblage, indicating that reservoir processes alter the distribution and availability of prey resources across multiple trophic pathways. Larger-bodied taxa dominated patterns in the coarse fraction, whereas smaller-bodied and planktonic taxa were the primary drivers in the subsample fraction, highlighting size-structured responses within the community.

Such restructuring is consistent with ecological responses to reservoir impoundment, where inundation of terrestrial habitats releases nutrients and organic matter that stimulate production across lower trophic levels (Grimard and Jones, 1982, Ostrofsky and Duthie, 1980, Gunkel, 2009, Boulton et al., 2014). These changes can alter benthic and littoral invertebrate communities both through enhanced productivity and shifts in habitat structure (Boulton et al., 2014). In the Cotter system, changes in the relative contribution of taxa such as Chironomidae, Copepoda, Cladocera, and Oligochaeta suggest a transition toward assemblages more characteristic of lacustrine conditions, consistent with patterns observed in newly impounded systems (Grimard and Jones,

1982, Straskraba et al., 1993). This reflects both a reorganisation of habitat use and adaptation to altered hydrological and productivity regimes.

Despite strong community-level responses, Decapoda, representing a key large-bodied prey resource, did not vary significantly among reservoir phases and instead exhibited strong seasonal control. This apparent stability suggests that larger-bodied benthic taxa may be relatively resilient to short-term environmental changes associated with reservoir filling, potentially due to longer life cycles, greater mobility, and broader habitat tolerance compared to smaller invertebrates (Boulton et al., 2014). However, temporal analysis revealed that decapod abundance in the current monitoring period was significantly lower than baseline conditions, representing the first time a statistically significant long-term decline in this key prey resource has been detected across the monitoring program. This is an ecologically meaningful finding: while decapod abundance has not varied significantly among reservoir phases or relative to recent monitoring periods, the significant departure from pre-enlargement baseline confirms that a genuine longer-term shift has occurred, even if year-to-year variability is high and short-term trends remain difficult to resolve. This finding strengthens the case for a reassessment of Macquarie perch diet, which has not been undertaken in over nine years, to determine whether observed shifts in prey community structure are being reflected in dietary composition and whether the decline in a historically important prey group is influencing trophic pathways for native fish in the reservoir.

This pattern highlights an important distinction between short-term and long-term responses in reservoir systems. Initial increases in productivity following impoundment are often followed by stabilisation or decline as nutrient subsidies diminish and systems transition toward equilibrium (Grimard and Jones, 1982, Straskraba et al., 1993, Gunkel, 2009). While decapods appear buffered against short-term variability, the detected decline relative to baseline conditions suggests that longer-term processes associated with reservoir ageing may influence their abundance. Given the importance of Decapoda in supporting higher trophic levels, including fish populations (Hatton, 2016), such changes may influence the efficiency of energy transfer through benthic pathways.

At lower trophic levels, microinvertebrates (Copepoda and Cladocera) showed weak and highly variable responses to reservoir phase, with no consistent patterns detected across analyses. This contrasts with expectations of trophic upsurge, where increased nutrient availability is predicted to stimulate phytoplankton and subsequently zooplankton production (Rodhe, 1964, Ostrofsky and Duthie, 1980). The absence of a strong signal in zooplankton abundance suggests that responses at this trophic level may be transient or strongly regulated by other processes, including grazing pressure, water column mixing, and dilution effects in reservoirs (Campbell et al., 1998, Boulton et al., 2014). These processes can dampen or obscure simple bottom-up responses, particularly in temperate systems with relatively low nutrient inputs and rapid turnover.

The contrasting responses among trophic levels indicate that reservoir phase effects are not uniformly propagated through the food web. Strong restructuring in macroinvertebrate communities, combined with weak responses in zooplankton and mixed temporal patterns in key prey taxa, suggest a degree of decoupling between benthic and pelagic pathways. Similar decoupling has been observed in reservoir ecosystems where physical and biological processes differentially influence trophic compartments (Straskraba et al., 1993, Gunkel, 2009).

Collectively, these findings reinforce the importance of adopting a multi-level analytical approach when assessing ecological responses to reservoir impoundment. Integration of community-level analyses with focal taxon and temporal assessments provides a more complete understanding of how reservoir processes influence energy pathways and prey availability. In the Cotter Reservoir, this approach indicates that while trophic upsurge has likely driven initial system responses, its effects are expressed unevenly across trophic levels and continue to evolve as the reservoir transitions toward a more stable state.

RECOMMENDATIONS

Sampling for this question continues to follow established methods and is adequate for detecting change across trophic levels. No modifications to the current monitoring approach are recommended.

Macroinvertebrate community analyses indicate clear and consistent differences among reservoir phases, demonstrating that reservoir processes influence prey composition and size structure. In contrast, Decapoda, a key large-bodied prey resource, show strong seasonal patterns and do not vary with reservoir phase, but temporal analyses indicate that abundance in the current monitoring period is lower than baseline conditions. Differences between the current period and recent years are weaker and more variable, suggesting that longer-term changes may be occurring alongside substantial interannual variability.

These findings indicate that while short-term variation in food resources is influenced primarily by seasonal dynamics, longer-term shifts relative to baseline conditions may be occurring, particularly for key prey taxa. Despite these changes, there is currently no evidence that reduced decapod abundance has negatively affected Macquarie perch condition, recruitment, or population dynamics. Accordingly, no direct management intervention is recommended at this stage.

However, it has now been more than nine years since Macquarie perch diet was assessed in the enlarged Cotter Reservoir. Given the detected changes in prey community structure and potential longer-term shifts in key prey availability, a reassessment of Macquarie perch diet is recommended. This would provide important insight into whether shifts in prey availability are being reflected in dietary composition and trophic pathways.

Continued monitoring is recommended to determine whether the observed reduction in decapod abundance represents natural variability or a longer-term directional change associated with reservoir ageing.

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APPENDIX 1: DATA ANALYSIS

Overview

All analyses were conducted in R (Team, 2024) using packages specified per question below. Data spanned 2009–2026 and were evaluated against three predefined reservoir management phases: Baseline (2010–2013), Filling (2014–2015), and Operational (2016–2026).

Question 1: Macquarie perch — Cotter Reservoir

Adult abundance (gill net CPUE)

Adult Macquarie perch abundance from standardised gill net sampling was modelled using two complementary Generalised Additive Models (GAMs) fitted via restricted maximum likelihood (REML) in the mgcv package (Wood, 2017). A negative binomial error distribution with a log link was used to accommodate overdispersion in individual net catch counts (Zuur et al., 2009). Rather than standardising catch as CPUE prior to analysis, raw count data were modelled directly with soak duration incorporated as a log-offset term (log[hours]), retaining the integer count structure of the data and avoiding assumptions inherent in pre-divided CPUE metrics (Maunder and Punt, 2004). The effect of reservoir shoreline expansion on catch rates is addressed at the inference stage rather than within the model, as incorporating shoreline length as a time-varying offset introduced confounding between the temporal scaling correction and the biological trend estimated by the smooth (see Results).

The primary model included a smooth function of year (thin-plate regression spline, k = 5) and a random effect for net deployment identity to account for within-survey clustering, but no categorical phase predictor, to avoid collinearity between phase step-changes and the continuous temporal smooth on a single-site time series (Zuur et al., 2009). A secondary model replaced the smooth with monitoring phase as a fixed categorical effect (baseline, filling, operational), retaining the net random effect, and was used exclusively to generate phase-level estimated means via back-transformation of model coefficients (delta method) and pairwise phase contrasts. Additional two-level GAMs (baseline versus each recent window) compared CPUE in 2026 alone, 2023–2025, and 2024–2026 against baseline conditions. Model adequacy was assessed using mgcv diagnostic functions and simulation-based residual diagnostics (dispersion, zero-inflation, and residual uniformity) implemented in the DHARMa package (Hartig, 2022).

YOY and juvenile abundance (fyke net CPUE)

Changes in fyke net CPUE for young-of-year (YOY; <85 mm TL) and juvenile (85–150 mm TL) Macquarie perch were analysed within a Before–After Control–Impact (BACI) framework comparing Cotter Reservoir (impact site) with Kissops Flat on the upper Murrumbidgee River (reference site). Raw count data were modelled directly using a Tweedie error distribution with a log link (Dunn and Smyth, 2005), which accommodates zero-inflated continuous data typical of fyke net catch records. Soak duration was incorporated as a log-offset term (log[hours]) within each model. Models

included fixed effects for site, monitoring phase, and their interaction (the BACI term), with site-specific nonlinear temporal trends modelled using penalised regression splines (s(year, by = site)) estimated via REML (Wood, 2017). Net deployment was included as a random effect (s(op, bs = "re")) to account for within-survey clustering. A significant site × phase interaction indicates that temporal change at the impact site exceeded that at the reference site beyond what can be attributed to shared regional environmental drivers, providing causal inference consistent with a reservoir enlargement effect (Underwood, 1994). The smooth basis dimension (k = 15) was evaluated using the k-index from gam.check() and confirmed adequate for the temporal resolution of the dataset.

YOY and juvenile size classes were treated as separate analytical units because they carry fundamentally different information about recruitment dynamics. YOY fish represent a single annual cohort and provide a temporally precise index of spawning and early survival success in the preceding spring, whereas juvenile fish integrate individuals from multiple year classes simultaneously. As a consequence, juvenile CPUE is an inherently lower-resolution signal expected to lag behind and be statistically more attenuated than equivalent changes in YOY CPUE; this expectation was incorporated explicitly into the interpretation of results.

Additional two-level GAMs were fitted to compare CPUE at the Cotter Reservoir impact site in recent monitoring windows (2026 alone, 2023–2025, and 2024–2026) against baseline conditions, using the same Tweedie offset structure. Model adequacy was assessed using mgcv diagnostic functions and DHARMa simulation-based residual diagnostics. Post-hoc pairwise contrasts for all models were evaluated using estimated marginal means via the emmeans package (Lenth, 2024) with Tukey adjustment for multiple comparisons.

Body length and condition

Gill net data for body length and Fulton's condition factor (K) were analysed using Generalised Linear Mixed Models (GLMMs) fitted with a Gaussian error distribution via the glmmTMB package (Brooks et al., 2017). To account for temporal autocorrelation arising from cohort effects and shared environmental conditions, an autoregressive AR(1) term was applied to the random effect of Year, grouped by a population-level vector. The AR(1) structure outperformed a standard random intercept model based on AIC. Separate linear fixed-effects models evaluated finer-scale annual dynamics. P-values for management phase contrasts were adjusted using Tukey's HSD method; annual trend contrasts used the Šidák adjustment to control Type I error.

Question 2: Macquarie perch — Cotter River

Macquarie perch abundance in the Cotter River was assessed within a Before–After Control–Impact (BACI) framework comparing multiple impact sites along the Cotter River with a single reference site (Kissops Flat) on the upper Murrumbidgee River. Data spanned 2010–2026 and were categorised into three monitoring phases: baseline (2010–2013), filling (2014–2015), and operational (2016–2026). Analyses were conducted separately for young-of-year (YOY; <85 mm TL), juvenile (85–150 mm TL), and total (YOY + juvenile combined) catch per unit effort.

Raw count data were modelled directly using Generalised Additive Models (GAMs) fitted via restricted maximum likelihood (REML) in the mgcv package (Wood, 2017), with a Tweedie error distribution and log link to accommodate zero-inflated count data (Dunn and Smyth, 2005). Soak duration was incorporated as a log-offset term (log[hours]) rather than pre-dividing catch by effort, retaining the integer count structure of the data (Maunder and Punt, 2004). Primary models included fixed effects for site type (impact vs reference), monitoring phase, and their interaction (the BACI term), site-specific nonlinear temporal trends modelled using penalised regression splines (s(year, by = site_type, k = 10)), a random effect for monitoring site identity (s(site, bs = "re")) to account for among-site variation within the impact group, and a random effect for net deployment (s(op, bs = "re")) to account for within-survey clustering. The site_type × phase interaction provided the primary causal test of whether temporal change at impact sites exceeded that at the reference site beyond any shared regional environmental signal (Underwood, 1994).

The U/S ECR site (sampled as "Bracks Hole" during 2010–2013) was excluded from primary BACI models due to a three-phase sampling confound rendering pre- and post-filling catch rates non-comparable. During the baseline period, all nets were positioned within approximately 1 km of the original reservoir margin within the active spawning ecotone, inflating YOY and juvenile catches. During the filling phase, natural rock barriers prevented adult egress from the expanding reservoir, physically suppressing catches at upstream sites independent of population status. During the operational phase, nets were relocated and split across more than 1 km, with a proportion positioned outside the active spawning reach, deflating catches relative to the original configuration. Because these changes coincided exactly with the phase transitions under investigation, including this site would generate a spurious BACI signal indistinguishable from a true biological effect. A sensitivity analysis including U/S ECR is reported alongside primary results; BACI contrasts weakened or reversed when this site was included, confirming the primary analysis is conservative.

Additional two-level GAMs compared catch rates at impact sites for three recent monitoring windows — 2026 alone, 2023–2025, and 2024–2026 — against baseline conditions (2010–2013), using the same Tweedie offset structure with site_type as an interacting fixed effect. Pairwise contrasts were evaluated using estimated marginal means via the emmeans package (Lenth, 2024) with Tukey adjustment. Model adequacy was assessed using mgcv diagnostic functions and DHARMa simulation-based residual diagnostics (Hartig, 2022).

Question 3: Two-spined blackfish

Annual CPUE (fish per net-night) of Two-spined blackfish at Bendora Reservoir was modelled as a function of year to assess temporal trends. Given the continuous, non-negative, and zero-inflated nature of the data, a Tweedie distribution was used (Jørgensen, 1987, Shono, 2008). Two candidate models were fitted and compared using AIC: (1) a Tweedie GLMM with a log link and linear year term fitted via glmmTMB (Brooks et al., 2017); and (2) a Tweedie GAM with a penalised regression spline on year (k = 6) fitted via mgcv (Wood, 2011). Both models included net identity as a random effect to account for variation among net positions. Basis dimension adequacy was confirmed using the k-index (k-index = 0.90, p = 0.275). GAM smooth estimates and confidence intervals were extracted using the gratia package (Simpson, 2024).

Question 4: Rainbow trout — Cotter Reservoir

Temporal variation in Rainbow trout CPUE in Cotter and Bendora reservoirs was analysed using Generalised Additive Models (GAMs) fitted via restricted maximum likelihood (REML) in the mgcv package (Wood, 2017), structured around a Before–After Control–Impact (BACI) design comparing Cotter Reservoir (impact) and Bendora Reservoir (reference) across baseline (2010–2013), filling (2014–2015), and operational (2016–2026) phases. Rather than standardising catch as CPUE prior to modelling, raw gill net count data were analysed directly with soak duration incorporated as a log-offset term (log[hours]), retaining the integer count structure of the data (Maunder and Punt, 2004). A negative binomial error distribution with a log link was used to accommodate overdispersion in individual net catch counts (Zuur et al., 2009).

Two complementary GAMs were fitted. A primary smooth model included site-specific nonlinear temporal trends (s(year, by = site, k = 5)) and a random effect for net deployment identity (s(netnight, bs = "re")), and was used for temporal trend significance testing. A secondary phase model replaced the temporal smooth with reservoir, monitoring phase, and their interaction as fixed categorical effects, retaining the net random effect, and was used to generate phase-level estimated means via coefficient back-transformation (delta method) and pairwise BACI contrasts. Additional two-level GAMs compared recent monitoring windows (2026 alone, 2023–2025, and 2024–2026) against baseline conditions by reservoir. Model adequacy was assessed using mgcv diagnostic functions and DHARMa simulation-based residual checks (Hartig, 2022). Fork length was analysed using a Gaussian GAMM with the same fixed-effect structure; sampling event was included as a random effect to account for non-independence of fish within occasions.

Question 5: Rainbow trout — Cotter River

Rainbow trout abundance in the Cotter River was assessed within a Before–After Control–Impact (BACI) framework comparing five Cotter River impact sites against a single reference site (Cotter Hut, upper Cotter River, unregulated, approximately 62 km upstream of the reservoir headwaters past Bendora and Corin reservoirs) across baseline (2010–2013), filling (2014–2015), and operational (2016–2026) monitoring phases. Fish were sampled annually by backpack electrofishing; sampling effort varied over time, with surveys in 2010–2013 and 2017 onwards encompassing approximately 1000 m of stream per site and surveys in 2014–2016 reduced to approximately 120 m. Rather than restricting analyses to a standardised effort subset, all available data were included with soak duration incorporated as a log-offset term (log[seconds of electrofishing on-time]), retaining the integer count structure of the data and accounting for variable effort without pre-dividing catch by effort (Maunder and Punt, 2004).

Two complementary GAMs were fitted in mgcv (Wood, 2017) using a Tweedie error distribution and log link (Dunn and Smyth, 2005) to accommodate zero-inflated, positively skewed count data. A primary BACI model included site type (impact vs reference), monitoring phase, and their interaction as fixed effects — the site_type × phase interaction providing the formal causal test — together with site-type-specific nonlinear temporal smooths (s(year, by = site_type, k = 15)), a random effect for monitoring site identity (s(site, bs = "re")) to account for among-site variation within the impact group, and a random effect for electrofishing run (s(op, bs = "re")) for within-survey clustering. A

secondary distance attenuation model (impact sites only; Cotter Hut excluded as it is not on the same spatial transect) incorporated monitoring phase, distance from the reservoir headwaters as a continuous covariate (km), and their interaction (phase × dist_km) to test whether the magnitude of phase-related change in trout abundance varied with proximity to the reservoir. Three additional two-level GAMs compared catch rates at impact and reference sites for recent monitoring windows against baseline: (i) baseline versus 2026; (ii) baseline versus 2024–2026; and (iii) 2023–2025 versus 2026. All smoothing parameters were estimated via REML and model adequacy was assessed using DHARMa simulation-based residual diagnostics (Hartig, 2022).

Question 7: Goldfish

Goldfish (*Carassius auratus*) abundance in Cotter Reservoir was assessed using fyke net data collected concurrently with Macquarie perch sampling (Question 1), with all nets deployed identically across the same nights and locations. Abundance data were analysed using a before–after framework spanning 2010–2026, with years grouped into baseline (2010–2013), filling (2014–2015), and operational (2016–2026) phases. Raw fyke net count data were modelled directly using Generalised Additive Models (GAMs) fitted via restricted maximum likelihood (REML) in the mgcv package (Wood, 2017), with a negative binomial error distribution and log link to accommodate overdispersion in count data (Zuur et al., 2009). Soak duration was incorporated as a log-offset term (log[hours]) rather than pre-dividing catch by effort, retaining the integer count structure of the data and ensuring that variation in soak time across years was correctly propagated through the model likelihood (Maunder and Punt, 2004). A net deployment identifier was included as a random effect (penalised regression spline, bs = "re") to account for spatial clustering of catches within each sampling occasion.

Two complementary models were fitted. A smooth-only model (s(year, k = 6)) tested for the presence and shape of a significant nonlinear temporal trend across the full 17-year record, providing the primary test of whether abundance changed significantly over time. A phase model (with baseline, filling, and operational as fixed categorical effects) provided phase-level estimated means and pairwise contrasts, with means and 95% confidence intervals derived by back-transformation of model coefficients using the delta method (Ver Hoef and Boveng, 2007). A third model explicitly tested for a short-term disturbance pulse by classifying 2014–2016 as pulse years and all other years as non-pulse, capturing the filling phase and the first operational year during which trophic upsurge responses were anticipated to be strongest. Model selection among the three approaches was evaluated using Akaike's Information Criterion (AIC) (Burnham and Anderson, 2002). Model adequacy was assessed using simulation-based residual diagnostics implemented in the DHARMa package (Hartig, 2022), including tests for overdispersion and zero-inflation. To contextualise current Goldfish abundance relative to pre-enlargement conditions, three additional two-level GAMs compared recent monitoring windows — 2026 alone, 2023–2025, and 2024–2026 — against the baseline period, using the same negative binomial offset structure with a binary period factor (baseline vs recent) as the sole fixed effect.

Question 8: Piscivorous birds

Monthly counts of Great cormorant (*Phalacrocorax carbo*), Little pied cormorant (*Microcarbo melanoleucos*), and Little black cormorant (*Phalacrocorax sulcirostris*) from five reservoir sections (July 2010–May 2026) were analysed across three management phases: Baseline (July 2010–December 2013), Filling (January 2014–December 2015), and Operational (January 2016 onwards). Data were structured in long format with species identity, month (for seasonal adjustment), and survey identity (random intercept for repeated observations within monthly events) as model terms.

Species abundance was analysed using GLMMs with a negative binomial error distribution (Bolker et al., 2009), implemented via glmmTMB (Brooks et al., 2017). The full model included fixed effects for reservoir phase, section, species identity, and all interactions (phase × section × species), enabling simultaneous evaluation of abundance responses, spatial redistribution, and species-specific trajectories. Significance of model terms was assessed via likelihood ratio tests (Zuur et al., 2009), with post-hoc Tukey-adjusted pairwise contrasts using emmeans (Lenth, 2024). A separate subset analysis compared baseline (2010–2013) with recent conditions (2023–2026) using equivalent GLMMs.

Community composition was assessed using PERMANOVA (Anderson, 2001) based on Bray–Curtis dissimilarities (Bray and Curtis, 1957, Legendre and Legendre, 2012), implemented in vegan (Oksanen et al., 2023) with 999 permutations. Samples with zero individuals were excluded. All analyses used the tidyverse (Wickham et al., 2019), glmmTMB, emmeans, and vegan packages in R (Team, 2024).

Question 10: Food resources

Food resource samples were collected in autumn and spring from the Enlarged Cotter Reservoir following protocols of Norris et al. (2012). A two-stage subsampling approach was used: large invertebrates were first selected by eye (coarse fraction), with remaining material subsampled and examined under magnification (subsample fraction). Both fractions were analysed independently to account for size-structured variation.

Taxon abundance data were $\log_{10}(x + 1)$ transformed to reduce dominance of highly abundant taxa while retaining information on rare species (Legendre and Legendre, 2012). Bray–Curtis dissimilarities (Bray and Curtis, 1957) were used to quantify compositional differences. PERMANOVA (Anderson, 2001) tested for effects of Phase, Season, and Habitat using the adonis2() function in vegan (Oksanen et al., 2023) with 999 permutations. Homogeneity of multivariate dispersion was tested using betadisper() (Anderson, 2006) to distinguish centroid shifts from dispersion differences. Where phase effects were significant, pairwise PERMANOVA comparisons applied Bonferroni-adjusted p-values. Taxa contributing most to compositional differences were identified via SIMPER analysis (Clarke, 1993), and community patterns were visualised using principal coordinates analysis (PCoA) with taxon–gradient relationships assessed via envfit().

Decapod abundance (a key prey taxon) was $\log_{10}(x + 1)$ transformed and analysed using linear models with Phase, Season, and Habitat as fixed factors. Proportional decapod abundance was log-transformed following addition of a small constant (0.001) to accommodate zeros, following Warton

& Hui (2011). Monitoring-year groupings (spring of the preceding year paired with autumn of the current year) enabled targeted temporal comparisons across baseline, current (most recent monitoring year), recent (last three years), and recent-prior (three years before recent) periods.

Microinvertebrate abundances (Cladocera, Copepoda, and total) were $\log_{10}(x + 1)$ transformed and analysed using linear models with Phase and Season as fixed factors. All analyses were conducted in R (Team, 2024) using vegan (Oksanen et al., 2023), dplyr, and ggplot2 (Wickham et al., 2023, Wickham, 2016).