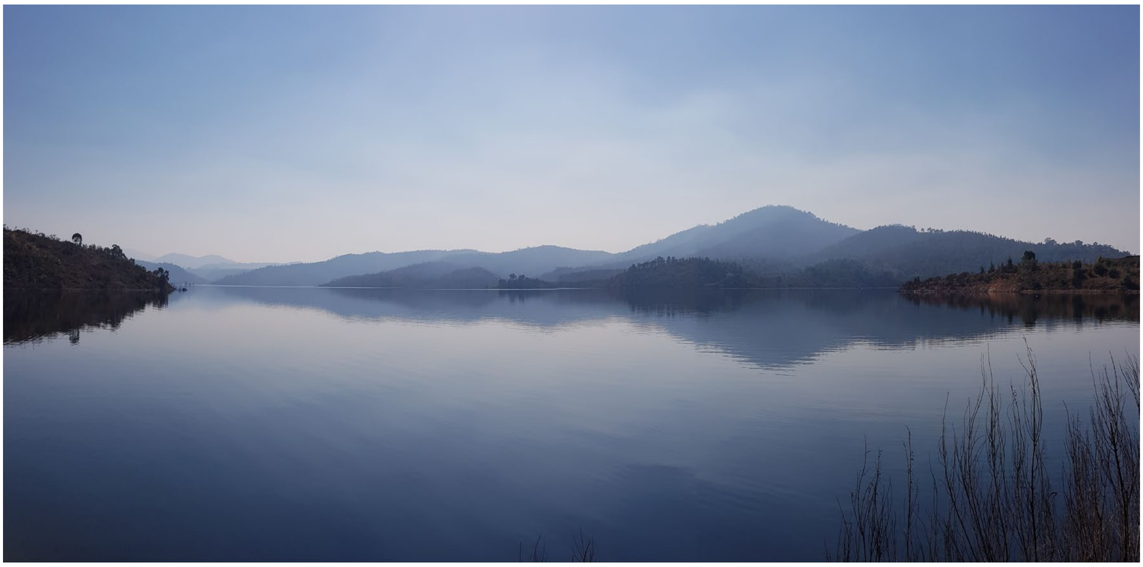
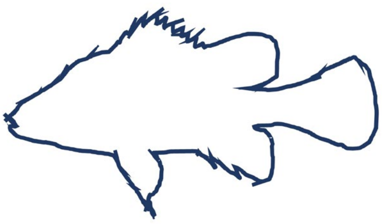
Enlarged Cotter Reservoir Ecological Monitoring Program

Technical Report 2024



Report to Icon Water

June 2024



Enlarged Cotter Reservoir Ecological Monitoring Program: Technical Report 2024

Report prepared for: Icon Water

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EXECUTIVE SUMMARY

Ongoing drought and its threat to water security in the ACT resulted in the recommissioning and augmentation of Cotter Reservoir from ~4 GL to 76.2 GL capacity, known as the Enlarged Cotter Reservoir (ECR). The ECR and Cotter River upstream to Bendora Dam contain four threatened fish and crayfish species, though only Macquarie perch *Macquaria australasica* and Two-spined blackfish *Gadopsis bispinosa* are likely to be directly impacted by the ECR and consequently are the focal species for research and mitigation projects associated with the ECR. Potential impacts of the construction, filling and operation of ECR have been well-described and in response to these impacts a range of projects including this fish monitoring program have been undertaken. This monitoring program encompasses three distinct management phases; baseline (2010-2013), filling of the ECR (2014 – 2015) and operational (2016 – current).

The monitoring year of 2023 / 2024 saw a return to largely regulated flows in the Cotter River below Bendora Dam. There were several peaks in flow associated with rainfall events, notably in early December 2023 and four large weekly events from late December 2023 to mid-January 2024. The Enlarged Cotter Reservoir (ECR) reached full supply level in August 2020 and has remained functionally full (within 1.0 m of FSL) since. The ECR has now been in the 'operational' phase (i.e. it has filled and is now fluctuating in level with changing inflows and river management) since 2016.

The main changes detected in the population of Macquarie perch in the ECR between the different monitoring phases (baseline, filling, operational) relate to adult abundance and body condition, and abundance of young-of-year recruits. Since peak abundances in 2015, adult relative abundance was in decline to its lowest level in 2018, whilst adult body lengths have been increasing. It appears that at least some of this trend may be due to changes in capture efficiency across size classes of Macquarie perch with changes in the ECR phases, with the gill nets deployed more effectively sampling smaller adults. Adult numbers as determined by gill netting in 2024 have continued to rebound from very low captures in 2022. The increase in catch was predominantly due to an increase in smaller sized adults (250 – 350 mm TL). This cohort, most likely from the spawning years of 2016 and 2017, are now large enough to be consistently captured in our standard gill net fleet. Body condition of adults was higher during filling and early operational phases, compared to baseline. Encouragingly, successful recruitment to young-of-year stage was detected for the sixth consecutive year in 2024, albeit at a low level.

Two-spined blackfish continued to be rare in the ECR, with only a few individuals being detected in the newly inundated section of the reservoir in the 10 years following the commencement of filling and operational phase to date. It is likely that this species is persisting in the newly-inundated section of the reservoir, though there is no evidence to suggest that a recruiting population has yet established in the ECR. Continuation of targeted monitoring over the coming years will provide further insight into these aspects of the population of Two-spined blackfish in the ECR.

Although some other annual differences are present, the abundance and size of Rainbow trout in the ECR in 2024 was not significantly different to any other year of monitoring. For the second consecutive year, relative abundance of Brown trout was very low in 2024 (zero captures), in contrast to the period between 2016 - 2021 which had recorded very high abundances. Monitoring in the coming years will provide clarification if this is a true decrease in Brown trout abundance in Cotter Reservoir.

Small-bodied alien species other than trout continue to be detected in the ECR, with Goldfish accounting for most captures. Goldfish abundance had increased since filling commenced, most likely in response to increased availability of food resources. However, Goldfish abundance has been low since 2017, following a boom in abundance during filling and the first year of operational phase. Although Goldfish probably pose little direct threat to Macquarie perch and Two-spined blackfish, there is potential for wider effects from declines in Goldfish abundance. For instance, the loss of Goldfish within the ECR food web could see a high abundance of potential predators (cormorants and trout) needing to switch their prey consumption to Macquarie perch.

Piscivorous birds have been relatively stable in their species composition and abundance in the ECR since filling commenced, though some subtle differences in distribution have occurred. There has been an increased number of Great cormorants and Little pied cormorants in sections (primarily section 4) that contain nesting sites and associated roosts. Breeding colonies of cormorants have far higher energy requirements than non-breeding colonies and the establishment of a breeding colony of cormorants in the ECR could increase predation pressure on adult and juvenile Macquarie perch. Cormorant management activities were undertaken as part of the Cormorant management strategy in 2014 and 2015, with mixed results. Cormorant thresholds have been revised (raised) to better reflect the increase in shoreline of the ECR.

The stable reservoir levels over the past three years have provided conditions suitable for both establishment and enlargement of the four emergent macrophyte stands detected to date. Whilst these stands, due to their relatively small size, are not likely to offer significant cover for adult Macquarie perch, it is heartening that establishment of emergent macrophytes is occurring in the enlarged reservoir.

Food resources of Macquarie perch (primarily decapods and microcrustaceans) showed small differences between baseline, filling and operational phases. Decapods were in low abundance in spring in both baseline and filling phases. However, there was no discernible difference in autumn decapod abundance between baseline and filling phase. There was, however, a sharp decrease in decapod abundance in autumn during the early operational phase monitoring, which was of concern as this is an important dietary item of Macquarie perch in the ECR. Monitoring since 2018 suggests that decapod abundances are returning to that observed in the baseline phase (albeit with some annual seasonal variation). Microcrustaceans demonstrate varying patterns through season and phase, though were in very low abundances in the latest samples. Operational phase monitoring has detected a downward trend in relative abundance of Cladocera, which have been shown to be part of Macquarie perch diet. The mechanism underpinning the reduction in Cladocera relative abundance may be related to a reduction in available resources (food and habitat) compared to baseline and filling phase.

RECOMMENDATIONS

We recommend that an adult population estimate be conducted every 3 – 5 years to provide a complimentary measure of adult Macquarie perch population health to the current monitoring program. An increase in effort to that applied in the current study would provide a more refined estimate of actual abundance.

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

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

BACKGROUND

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

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

STUDY AREA

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

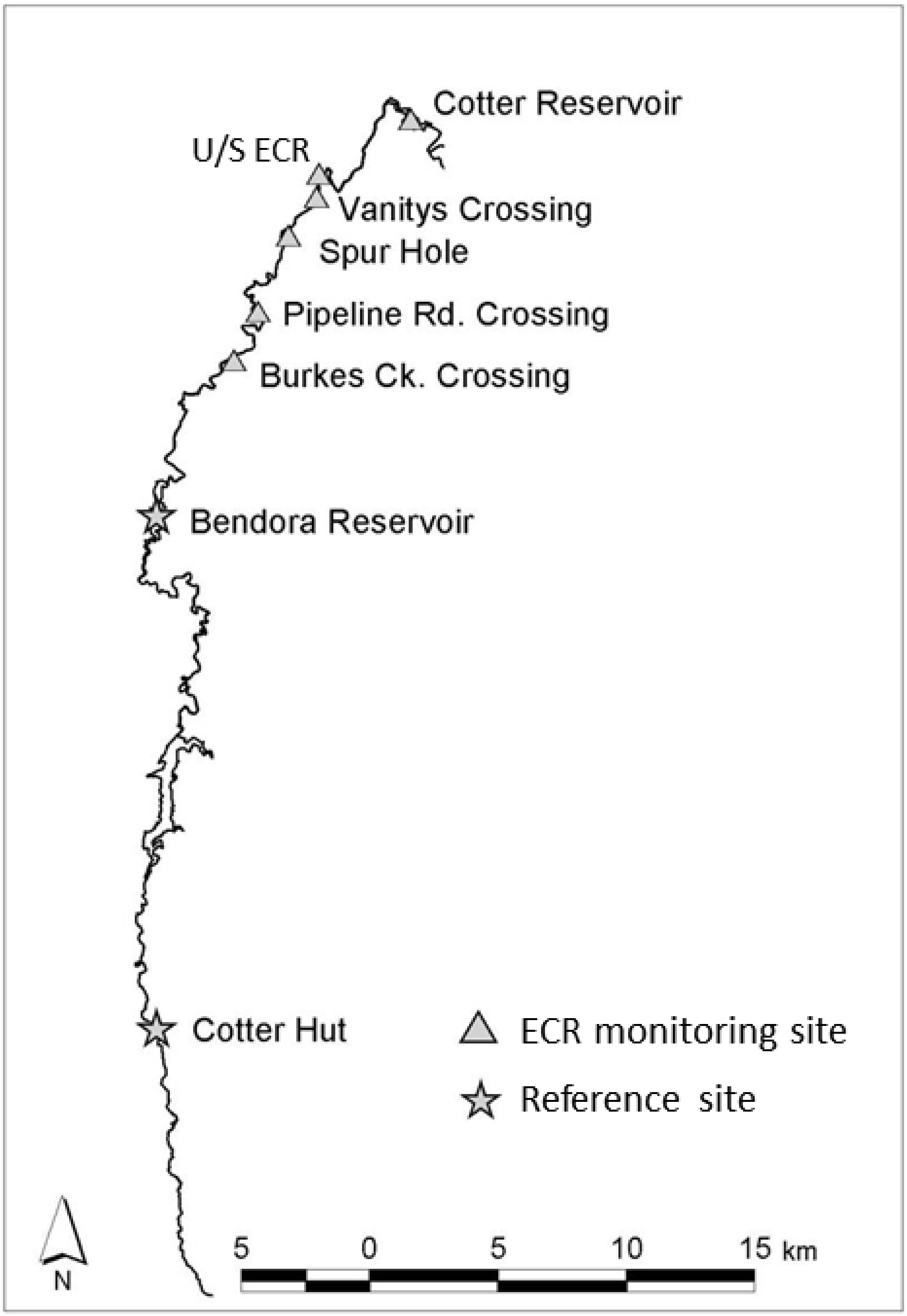


Figure 1. Location of sampling sites on the Cotter River. Note: Map does not include the reference site on the Murrumbidgee River (Kissops Flat).

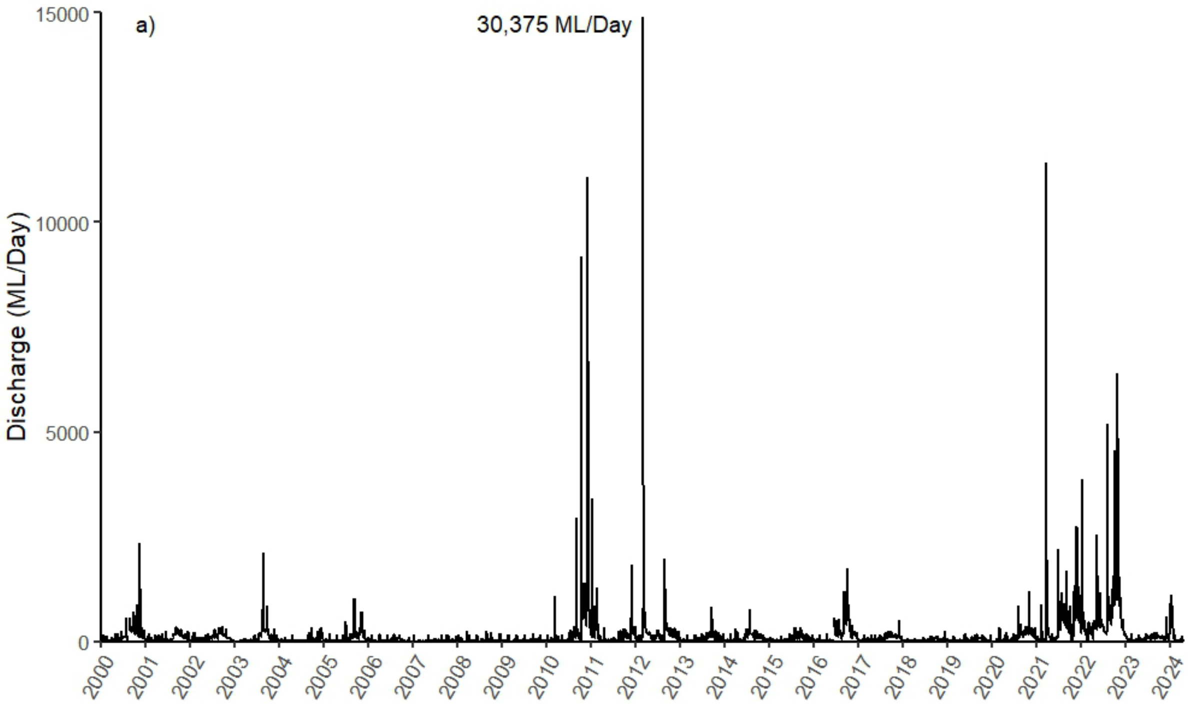
Table 1. Monitoring questions to be addressed at each monitoring site (see Figure 1 for location of monitoring sites).

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

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

HYDROLOGICAL SUMMARY

Following several very wet years (see Figure 2a), the monitoring year of 2023 / 2024 saw a return to river flows which were largely dominated by regulated flow releases (because of dry climatic conditions), except for a few short, high flow pulses associated with rainfall events (notably early December 2023, and four rainfall events of greater than 50 mm falling weekly from late December 2023 to mid-January 2024) overtopping a full Bendora Reservoir at these times mimicking an unregulated river (Figure 2b and Figure 3). Enlarged Cotter Reservoir was at or very close to full supply level for the entire monitoring year of 2023 – 2024 (Figure 4).



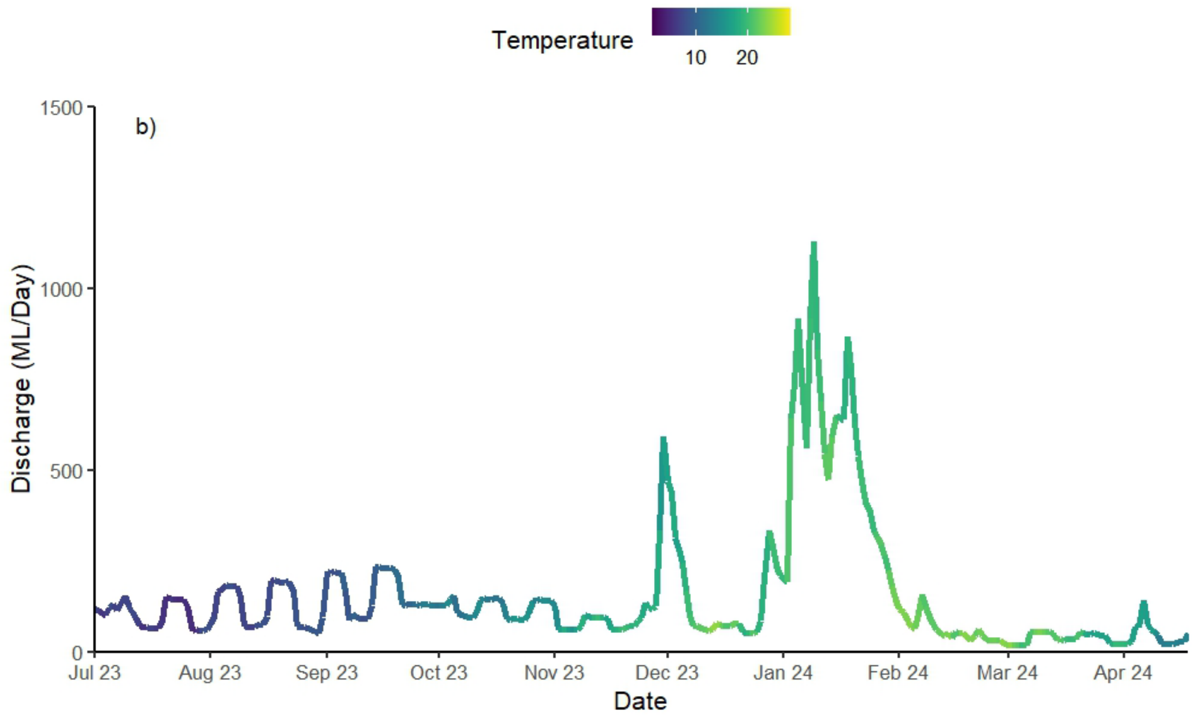


Figure 2. Daily discharge of the Cotter River at Vanitys Crossing from a) January 2000 until May 2023 and b) July 2023 – May 2024.

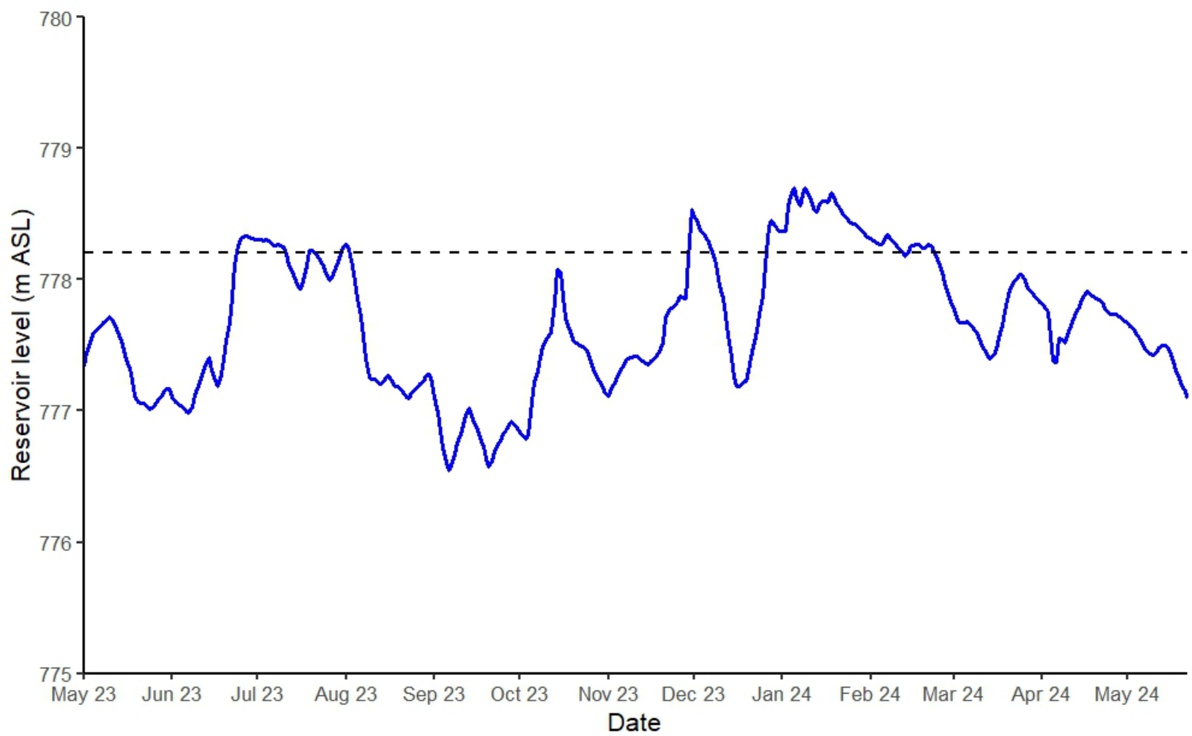


Figure 3. Water level (in metres above sea level) of Bendora Reservoir (blue line) from May 2023 – May 2024. Dashed line indicates Bendora Reservoir full supply level.

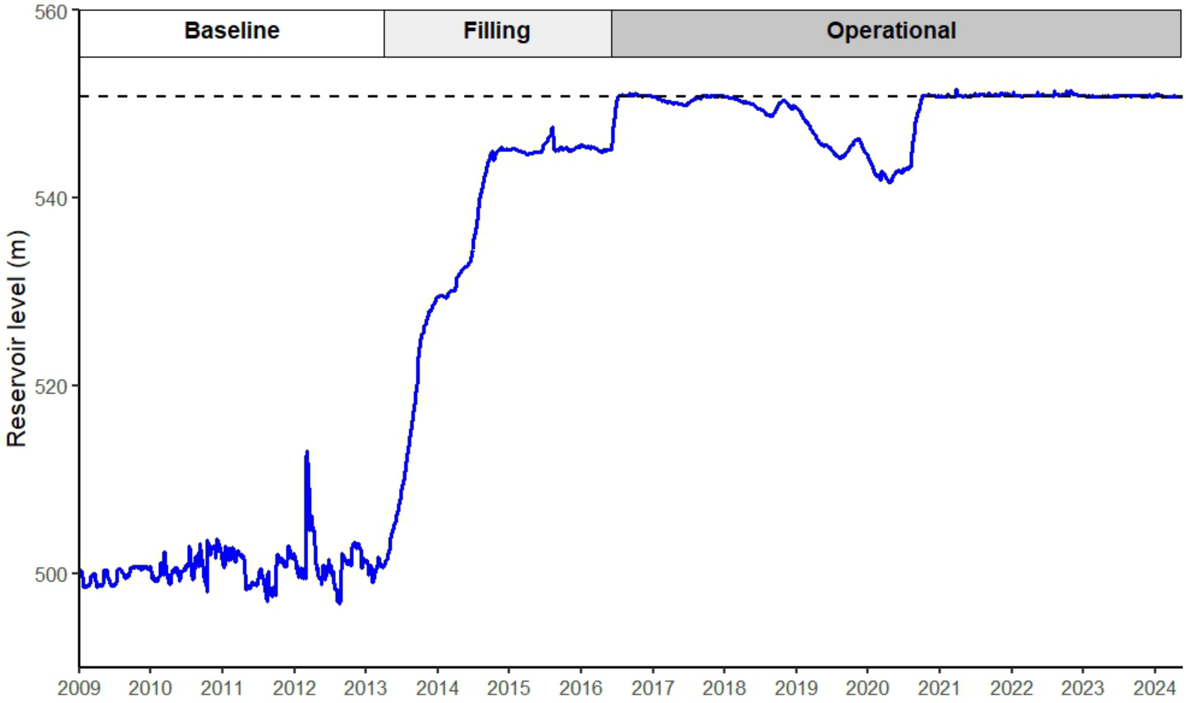


Figure 4. Water level (in metres above sea level) of Cotter Reservoir from January 2009 until May 2024. Rectangles indicate the three monitoring phases: white = Baseline, grey = Filling, hatched = Operational. Blue line indicates Cotter Reservoir level. Dashed line indicates enlarged Cotter Reservoir full supply level. Full supply level prior to enlargement was 500.8 m ASL.

MONITORING METHODS, RESULTS AND DISCUSSION

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

BACKGROUND

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

METHODS

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

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

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

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

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

Gill nets were deployed as per the baseline and previous filling and operational monitoring. Specifically, 12 gill nets were set independently around the perimeter of the reservoir in March 2024. The reservoir was divided into five longitudinal sections, with two gill nets set in each section. Gill netting was undertaken over five nights (based on power analysis conducted Robinson 2009), though was not conducted for more than two consecutive nights at a time to avoid stress on adult Macquarie perch by multiple sequential re-captures. In 2022, an additional two 125 mm gill nets (1 x deep-drop (66 meshes deep)) have been added to the previous effort of 10 gill nets (75, 100, 125 mm stretch mesh) to capture the increasingly larger adult Macquarie perch in Cotter Reservoir (following recommnedations in Broadhurst *et al.* 2018). Gill nets were set for six hours soak time commencing at ~15:30hrs following the existing threatened species netting protocol to minimise potential issues with prolonged retention of threatened fish in gill nets.

Twenty fyke nets were set singularly around the perimeter of the reservoir over three nights in March 2024. Twelve fyke nets were set for two nights in the pool at Kissops Flat in February 2024. Fyke nets were set for ~16-hour soak time (existing fyke netting protocol) commencing at ~15:30-16:00 hrs.

Larval monitoring was undertaken in December 2023.

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

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

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

Analysis of Macquarie perch CPUE in gill nets (excluding the additional 125 mm gill nets used in 2022 from analyses) was assessed between years using analysis of similarity (ANOSIM) with phase as a fixed factor and year as a random factor nested within phase. Gill netting data was Log10(x+1) transformed and fyke netting data was Log10(x+1) transformed to deal with skew, and then a resemblance matrix was constructed using the modified Gower (base 2) dissimilarity measure for gill netting data and a modified Gower base 2 (+ dummy variable to deal with double-zeros across sample pairs) for fyke netting data. Tests were run with a maximum of 9999 permutations. For fyke net data, size classes (<100 TL, >100mm TL) were included as variables, with site and phase as fixed factors, and a random factor of year nested within phase for a maximum of 9999 permutations. To test between differences in Macquarie perch CPUE in fyke nets for each size class (<100 mm TL and >100 mm TL), PERMANOVA using Type III sum of squares in a repeated measures design was employed and used for pairwise tests (site and year as fixed factors) (following Anderson et al. (2008). This approach allowed for an unbalanced design arising from the different number of samples collected across years. Significant interactions were interpreted using threshold metric MDS performed on group centroids for site by year. Graphical presentations of site-level means with 95% confidence limits (with Bonferroni corrections applied for n = x sampling years) were then used to the magnitude of pairwise variations in CPUE of Macquarie perch size classes among sites and years. Condition of adult Macquarie perch was analysed using Fulton's condition index, which is calculated as $K = 100(\text{weight}/\text{length}^3)$ following (Ricker 1975). Size (TL) and body condition of the adult population was analysed using Kruskal-Wallis ANOVA tests to determine if a significant change occurred through time. Pairwise comparisons were then undertaken using Dunn's method. ECR monitoring program body condition data was compared against historical data from 2007 – 2009 (data from Lintermans *et al.* 2010).

RESULTS

Adult Macquarie Perch

A total of 39 Macquarie perch were captured using all gill nets in ECR in 2024, which ranged from 228 – 411 mm TL (Figure 5). Adult Macquarie perch abundance was highest in 2015, more than double the next most abundant years (2014, 2016 and 2017, and roughly quadruple the abundances of all other years Figure 6). Macquarie perch CPUE was significantly different among years (Global $R = 0.008$, $p < 0.01$), with 2015, 2016 and 2017 having significantly higher captures of adult Macquarie perch compared to all other years. There was no significant difference in CPUE among monitoring phases (Global $R = 0.049$, $p = 0.375$).

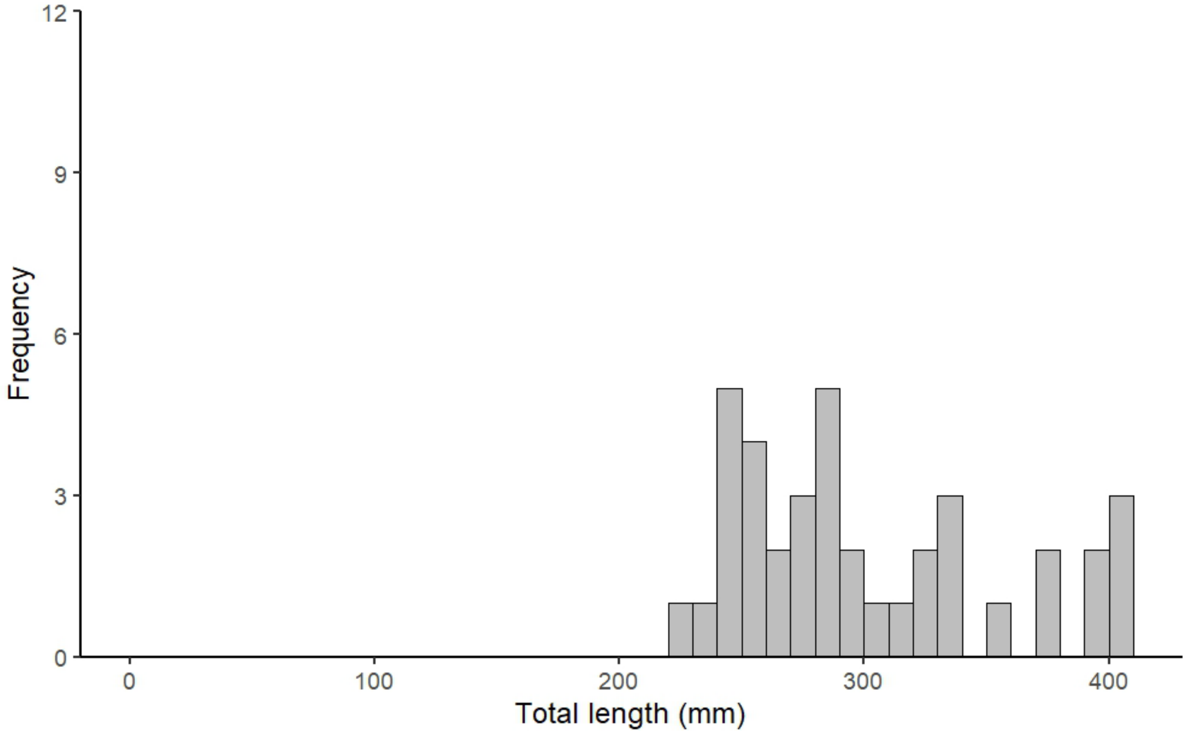


Figure 5. Length frequency of Macquarie perch captured from the Enlarged Cotter Reservoir in autumn 2024 using gill nets.

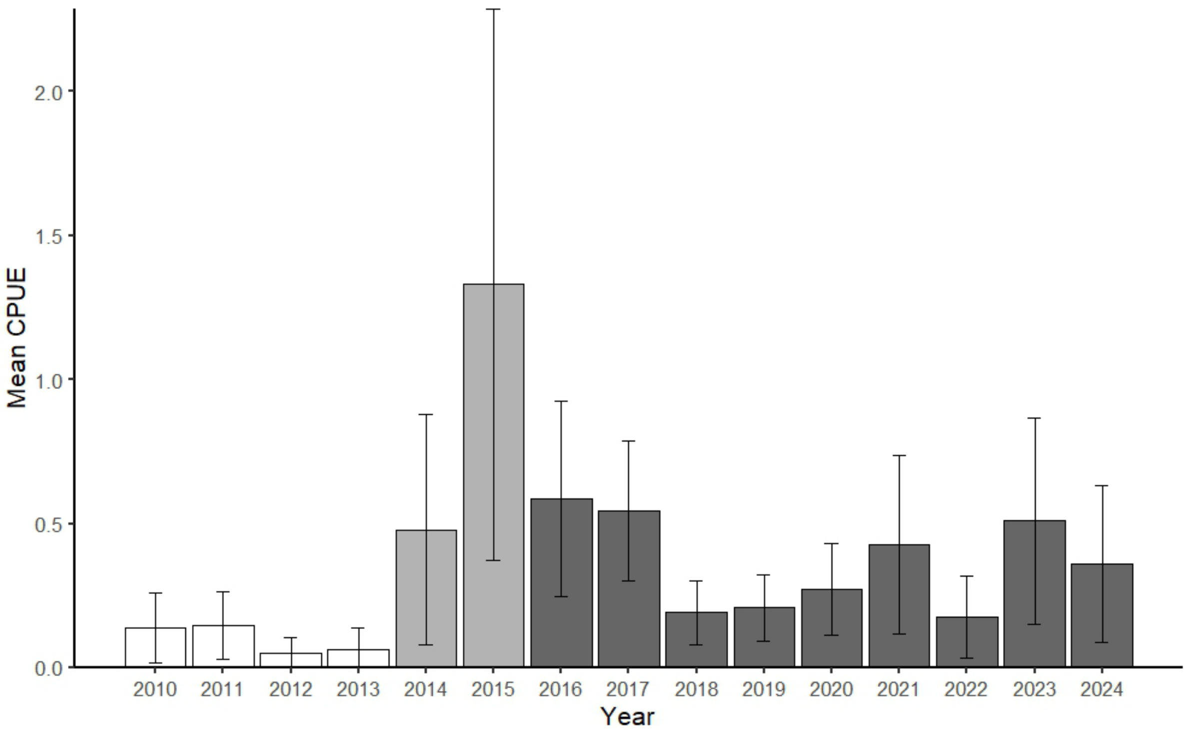


Figure 6. Relative abundance (displayed as mean CPUE ± 95% Confidence limits with Bonferroni correction and scaled to relative reservoir shoreline length at time of sampling) of adult Macquarie perch captured in Cotter Reservoir using gill nets between 2010 – 2024. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronological order from 2010 to 2024 from left to right on the x-axis.

Length of adult Macquarie perch captured in gill nets was significantly different between years. For the most part, individuals captured in baseline and filling phases were significantly smaller than those captured in operational phase ($H_2 = 76.514$, $P < 0.01$), though length frequency of fish captured in 2023 and 2024 was more similar to baseline and filling years than pervious operational years (Figure 7). Length of Macquarie perch captured in gill nets in 2024 was larger than baseline years 2010, 2012 and 2013 and filling years 2014 and 2015, but was not different to any operational years, but was not different to any of the baseline or filling years monitored (Figure 5 and Figure 7). Length of adult Macquarie perch captured in gill nets in 2024 was dominated by individuals of between 250 – 350 mm TL, and a range of large adults between 380 – 410 mm TL (Figure 5). Condition of adult Macquarie perch captured in gill nets was significantly higher during the filling phase (2014 – 2015) and operational phase (2016 – 2023) compared to baseline (2007 – 2009) phase ($H_2 = 41.7$, $P < 0.01$). Mean Fulton's condition index declined between 2017 and 2020, though has stabilised in the years since (Figure 8).

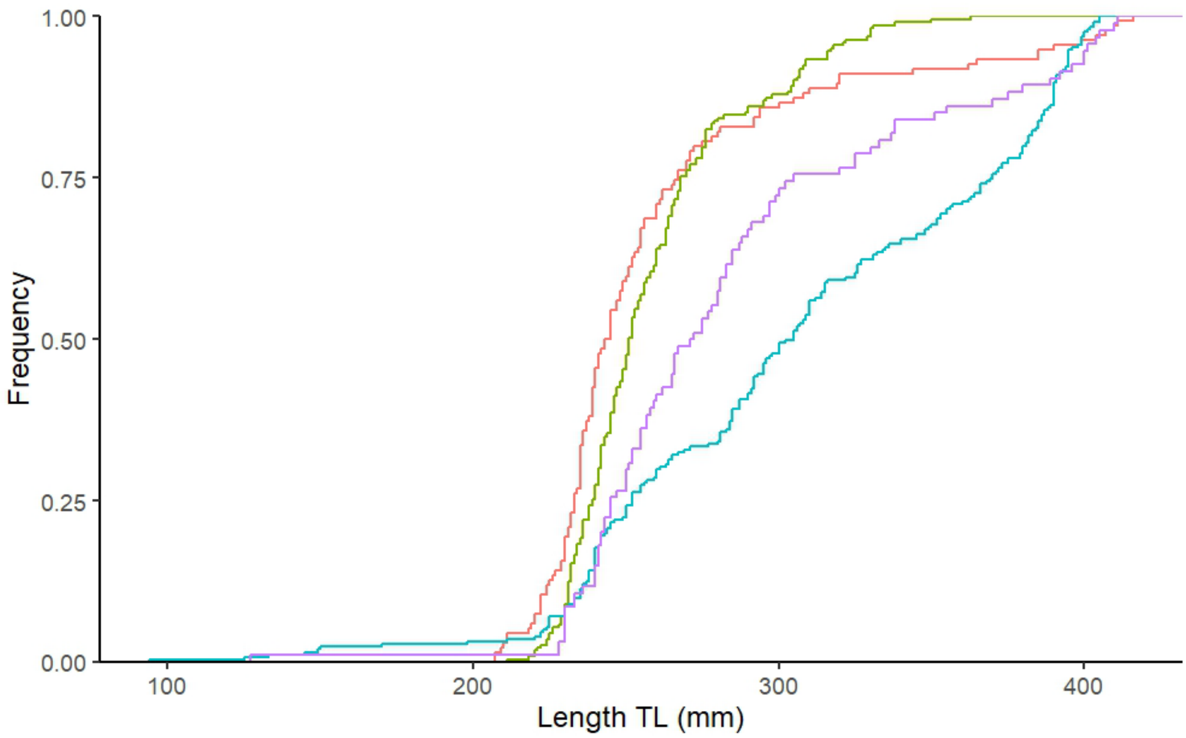


Figure 7. Cumulative length frequency plot of Macquarie perch captured in gill nets in Cotter Reservoir by year group. Year groups: 2010 – 2013 red line, 2014 – 2015 green line, 2016 – 2022 blue line, 2023 & 2024 purple line.

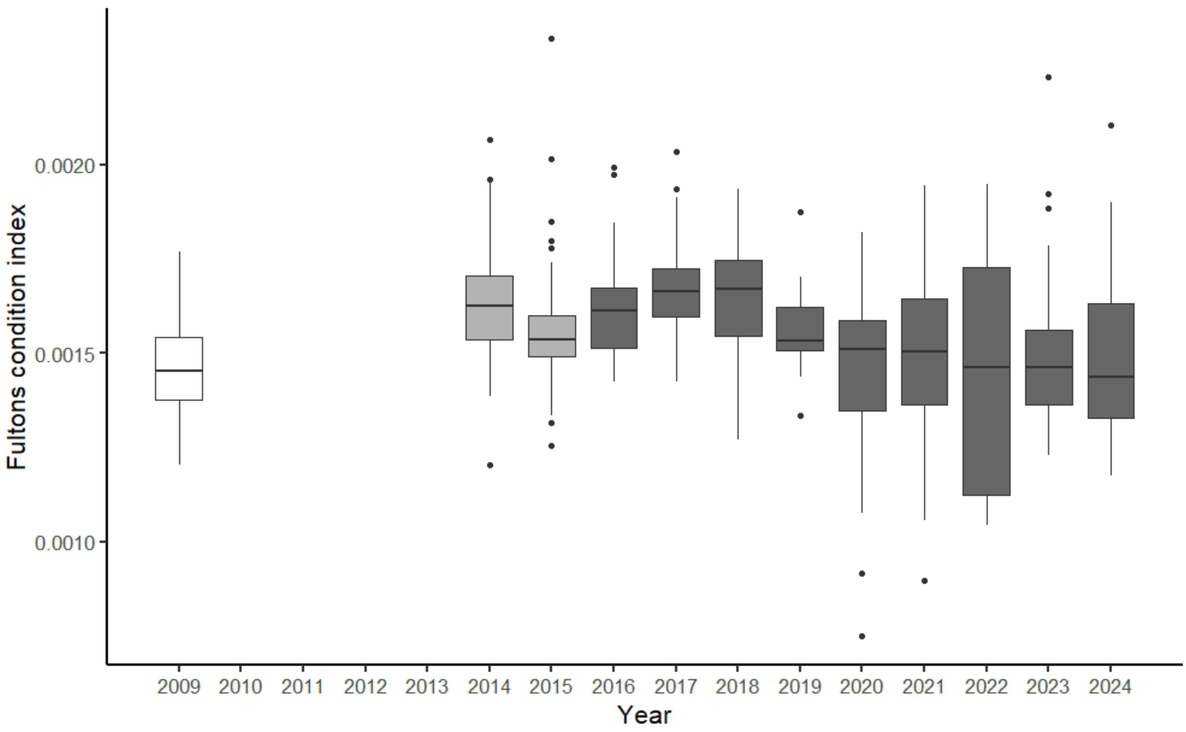


Figure 8. Box plot of condition (using Fulton's condition index) of adult Macquarie perch captured in gill nets in Cotter Reservoir for from 2009 to 2024. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); plot arranged in chronological order from 2009 to 2024 from left to right on the x-axis.

Boat electrofishing

A total of 31 Macquarie perch were captured by boat electrofishing in 2024 ranging in lengths from 200 – 402 mm TL (Figure 11). Similar to gill netting captures, length frequency of individuals caught by electrofishing was dominated by individuals between 200 - 350 mm TL (Figure 11). Of these, nine individuals were captured during the day and 22 individuals captured at night. Relative abundance of adult Macquarie perch captured by boat electrofishing in 2024 was the second lowest since monitoring began in 2014, but was very close to moderate abundance years of 2014, 2019 and 2023 (Figure 12).

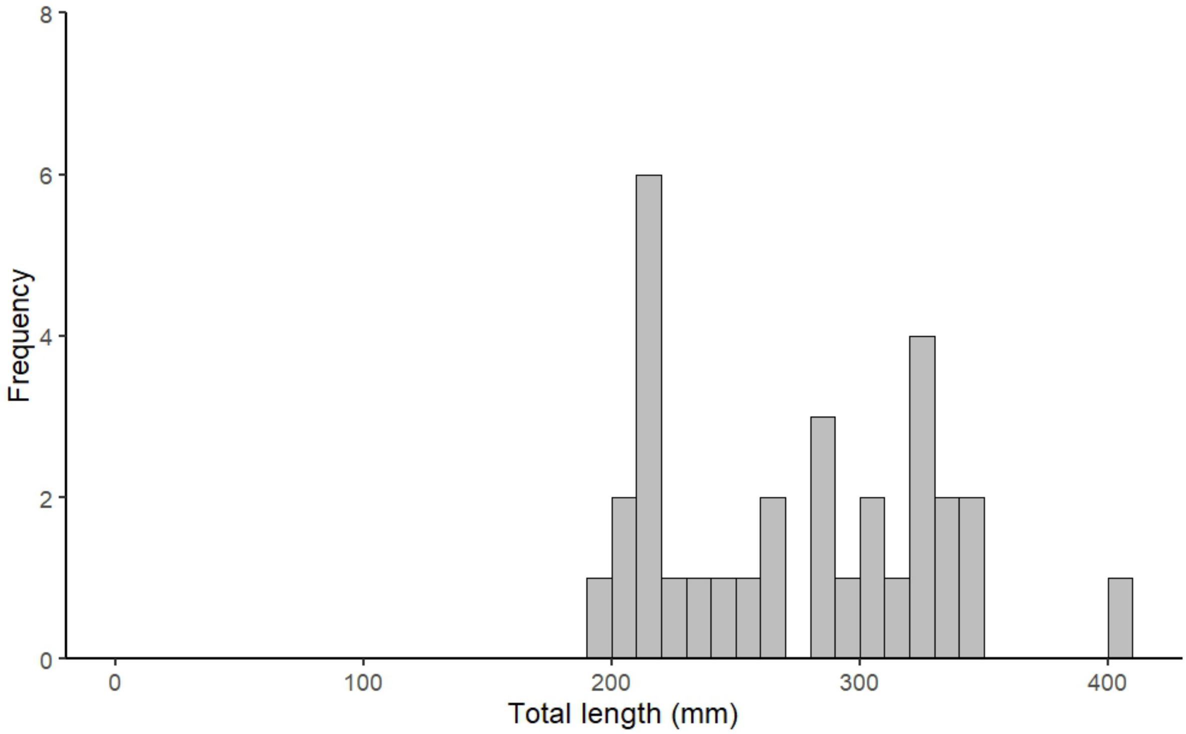


Figure 9. Length frequency of all Macquarie perch captured via boat electrofishing in Cotter Reservoir in 2024.

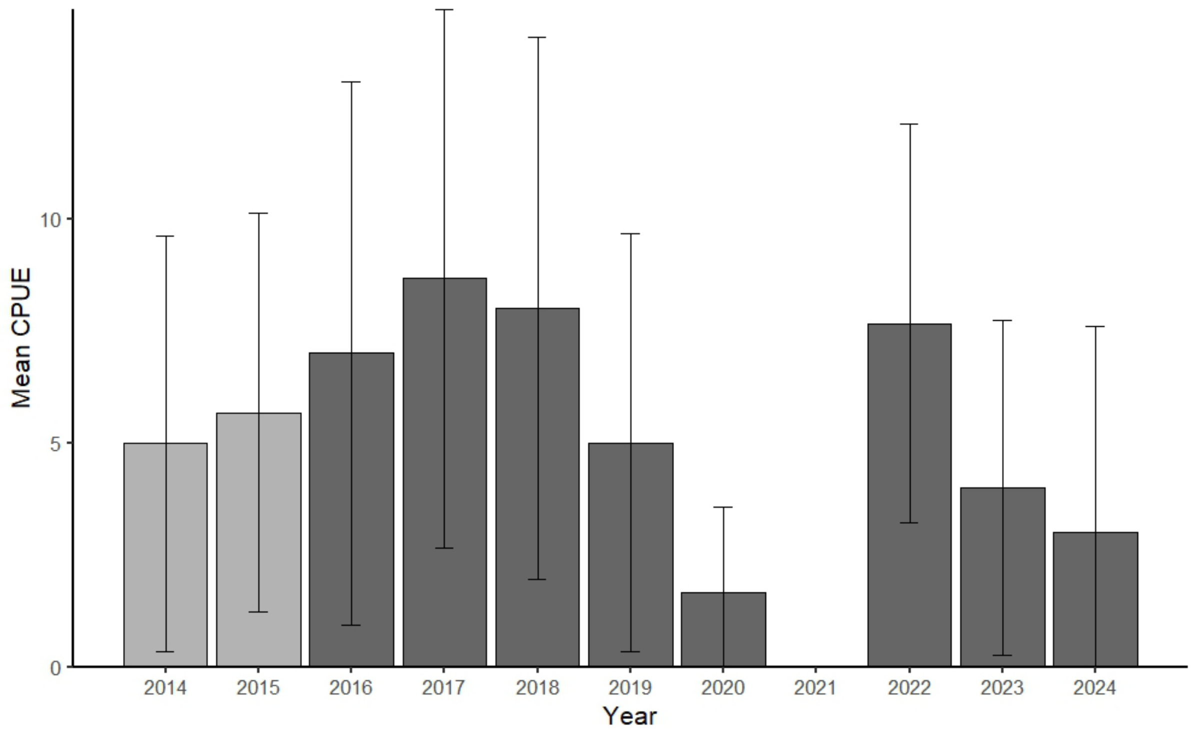


Figure 10. Relative abundance (displayed as mean CPUE ± 95% Confidence limits with Bonferroni correction) of adult Macquarie perch captured in Cotter Reservoir via boat electrofishing between 2014 – 2024. Light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronological order from 2014 to 2024 from left to right on the x-axis. Boat electrofishing not undertaken in 2021 due to high water turbidity.

Fyke netting - all age classes

A total of 316 Macquarie perch ranging from 43 – 370 mm TL were captured in 2024 from the ECR using fyke nets (Figure 11). CPUE of Macquarie perch (all sizes combined) captured by fyke netting were significantly different between the sites, years and phases, with a significant site by phase interaction (Table 3). The significant site by year interaction is largely driven by the lack of young-of-year at Cotter Reservoir during 2014 – 2016 (Figure 12).

Table 3. Results of PERMANOVA comparison of catch-per-unit-effort of Macquarie perch (all sizes combined) in fyke nets deployed in Cotter Reservoir and Kissops Flat each year over 2010 to 2024 (bold text indicates significant effects at the P(perm) 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Site	**1**	**0.37305**	**0.37305**	**16.029**	**0.0001**	**9955**
Phase	**2**	**0.91163**	**0.45582**	**8.2396**	**0.0023**	**9876**
Year(Phase)	**12**	**0.78141**	**0.065118**	**2.7979**	**0.0001**	**9878**
Site x Phase	**2**	**0.26062**	**0.13031**	**5.5988**	**0.0002**	**9956**
Residuals	1010	23.507	0.023274			
Total	1027	25.838				

Juvenile Macquarie perch

In 2024 a total of 185 Macquarie perch juveniles (85 > 150 mm TL) were captured in fyke nets. Juvenile Macquarie perch have been captured each year since monitoring began in 2010 but were particularly low in abundance during 2015 – 2017 as a result of successive years of recruitment failure over 2014 – 2016. There was a significant difference in the relative abundance of juvenile Macquarie perch between years (Global R = 0.035, $p < 0.01$) but not sites (Global R = -0.034, p = 0.999) (Figure 12). Pairwise comparisons among years suggests a mixture of differences, including those between pairs of recent pre- and post-filling years. Most differences lay between the baseline years of 2010 and 2011, which had significantly higher abundances of juvenile Macquarie perch compared to all filling and operation years (with the exception of 2018 and 2022 where there was no significant difference). Captures of juvenile Macquarie perch in 2024 were not significantly different to other years except 2010, 2011 and 2013, which had significantly high abundance compared to most other years (Figure 12).

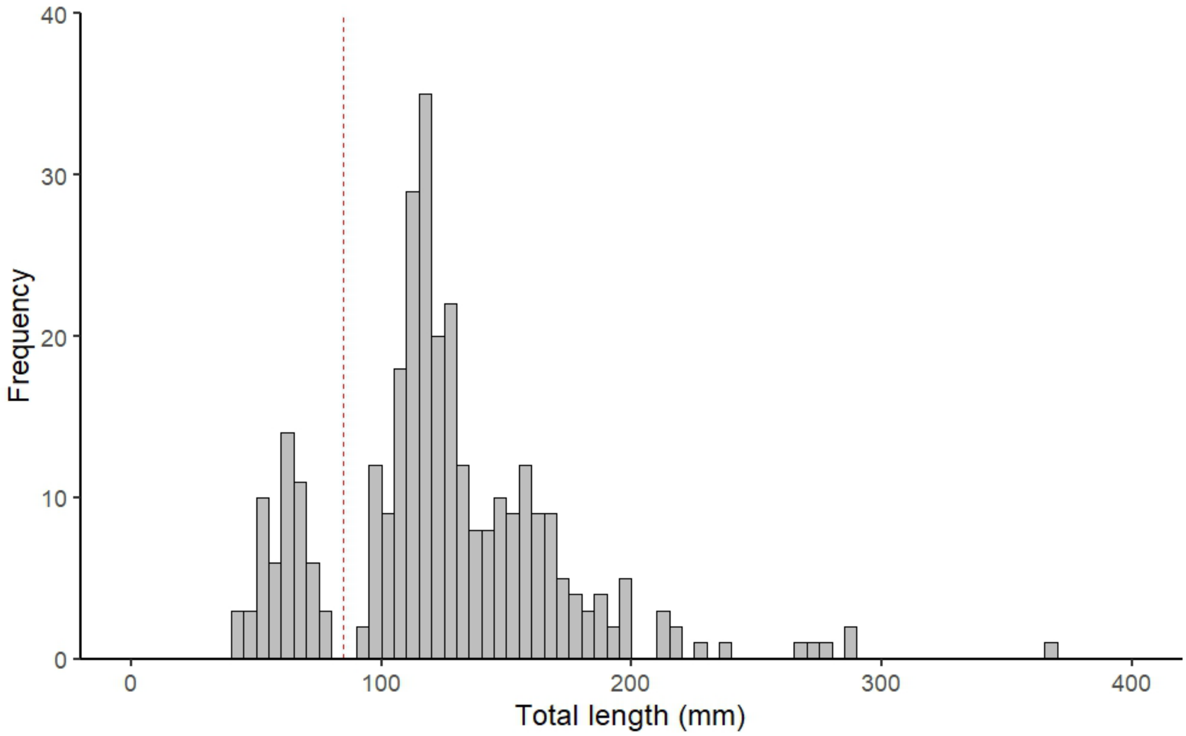


Figure 11. Length frequency of Macquarie perch captured from the ECR in autumn 2024 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

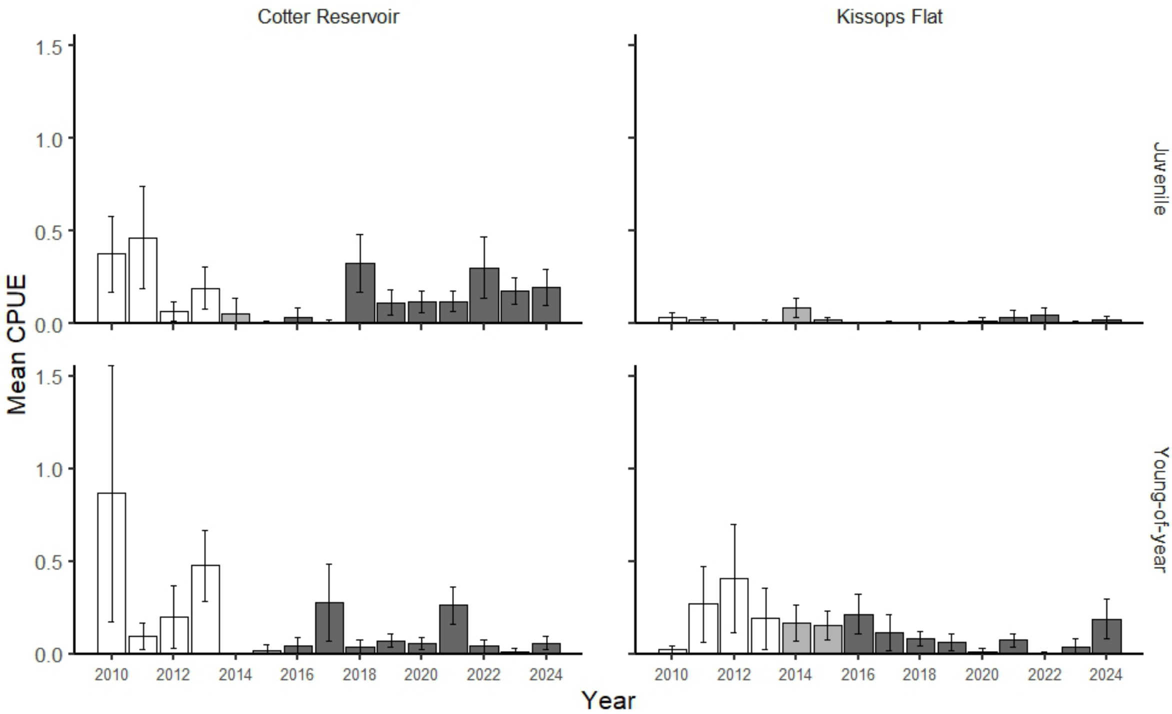


Figure 12. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections, scaled to relative net effort versus shoreline length at the time of sampling) of juvenile (>85 mm TL) and young-of-year (< 85 mm TL) Macquarie perch captured in Cotter Reservoir (pre and post enlargement) (impact site) and Kissops Flat (reference site) using fyke nets between 2010 and 2024. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronological order from 2010 to 2024 from left to right on the x-axis.

Young-of-year Macquarie perch

Snorkelling surveys undertaken in mid-December 2023 detected larval Macquarie perch at three of the four pools surveyed immediately upstream of the enlarged Cotter Reservoir. Abundances per pool were generally low compared to previous surveys where larvae have been observed (total of 25 individuals across three pools). A total of 56 young-of-year (YOY) Macquarie perch were captured using fyke nets in the ECR in 2024. There was no significant difference in the CPUE of YOY Macquarie perch between sites (Global R = -0.005, p = 0.756), but there was a significant difference between years (Global R = 0.026, p = <0.01) (Figure 12). Pairwise comparisons among years suggests a mixture of differences, including those between pairs of recent pre- and post-filling years. Captures of YOY Macquarie perch in the reservoir in 2024 were the same as all other years except baselines years of 2010, 2012 and 2013 and operational year of 2017. which had significantly high abundance compared to most other years (Figure 12). YOY Macquarie perch were detected in all years of monitoring at the reference site (Kissops Flat, upper Murrumbidgee River), and catches in 2024 reversed a trend of general decline observed since 2012 (Figure 12 and Figure 13).

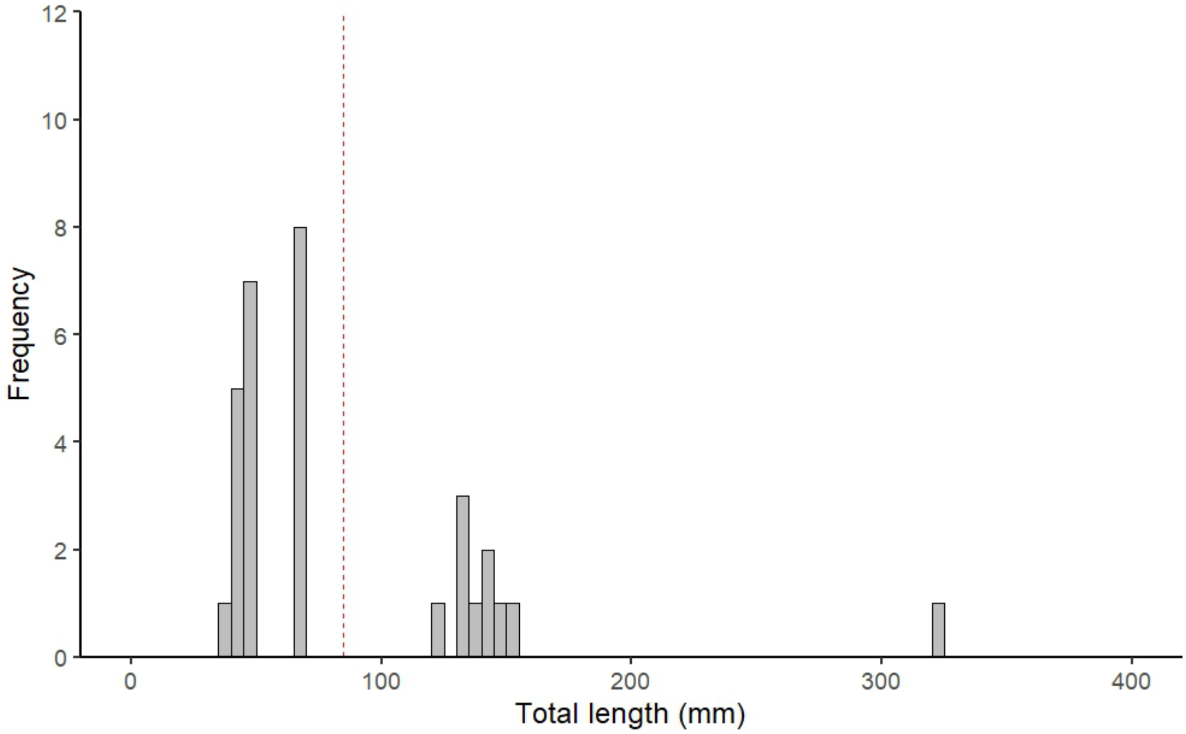


Figure 13. Length frequency of Macquarie perch captured from Kissops Flat on the upper Murrumbidgee River in autumn 2024 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

DISCUSSION AND CONCLUSIONS

Annual abundances of Macquarie perch size classes in Cotter Reservoir are highly variable since monitoring began, which is partly expected due to natural variations in recruitment and mortality arising from a range of environmental factors. However, the question is to what extent more recent changes in the size structure of fish can be attributed to shifts in the reservoir habitat conditions, and connectivity to upstream sections of the Cotter River. Concerns were raised about the failure of Macquarie perch recruitment over the 2014 – 2016 period since filling of the ECR began, as indicated by very low or nil catches of YOY Macquarie perch in the ECR over that period, and very low juvenile abundance in 2017. More recent monitoring from 2017 – 2024 has indicated successful spawning and recruitment to YOY, with captures of this size class higher than some of the baseline monitoring years prior to the commencement of filling. Notably, there is a strong class of 1+ year old (juvenile) fish captured in the last six years (2018 – 2024), suggesting good annual recruitment conditions through to 1 – 3+ year old individuals.

Relative abundance of adult Macquarie perch captured in 2024 was around the median value since entering the operation phase. Length frequency composition of gill net captured Macquarie perch in

2024 revealed that the majority of captures were of individuals < 300 mm TL, more closely resembling that of baseline and early filling. The good captures of younger adult individuals in 2024 is heartening and indicates that recruits spawned during the operational phase have entered adulthood, and of size catchable via our gill net fleet. This result and that of 2023 should substantially allay concerns expressed about potential declines of adult numbers in the reservoir following low captures in 2018, 2019 and 2022.

Adult Macquarie perch captured in both filling and early operational phases displayed significantly higher body condition relative to individuals captured in the baseline phase. Condition of adult Macquarie perch has declined since early filling phase years, though seems to have stabilised since 2020. The reservoir has been largely full now for nearly six years, and it appears as though the trophic upsurge has passed, and food resources have stabilised and body condition of adults reflect that of a stable reservoir environment (similar to baseline phase).

Abundance of juvenile Macquarie perch in Cotter Reservoir has been relatively stable since 2018, following on from some relatively poor years in late filling / early operation of the enlarged Cotter Reservoir. This has been largely due to a strong contingent of 1+ and subsequently 2 + year old individuals. The conversion of 0+ individuals into strong 1+ and 2+ individuals (and into adulthood – see above) suggests the ECR is providing suitable conditions for early survival and growth of Macquarie perch recruits since autumn 2017.

Abundances of young-of-year Macquarie perch have picked up a little since the previous monitoring year, but are still a way off that of more productive operational years of 2017 and 2021. Since Cotter Reservoir was enlarged, Macquarie perch young-of-year recruitment has been strongest when reservoirs on Cotter River have been full and the river running largely unregulated during spawning season (October – December) (e.g. 2016 and 2020) and without multiple large flow peaks during spawning time (as was the case in 2021 and 2022, which likely had a negative impact on spawning success (Broadhurst *et al.* 2022, 2023).

RECOMMENDATIONS

Adult population

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

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

Other than this potential bias in capture efficiency in gill netting, the adult population, especially the largest size classes, appears to be adequately protected. No management intervention is recommended for adult Macquarie perch in the ECR.

Juvenile population

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

Young-of-year

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

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

Larval monitoring

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

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

BACKGROUND

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

SAMPLING DESIGN

Sampling design for Question 2 follows that of the baseline monitoring program Question 8 (Lintermans *et al.* 2013), with a few changes (Table 4). The site immediately above the old Cotter Reservoir (Bracks Hole) has been inundated and is no longer a riverine site, so a riverine site between ECR full supply level and Vanitys Crossing has been monitored as a substitute. The site immediately downstream of Bendora Dam has been dropped from the monitoring program as this site is unlikely to be directly affected by the operation of ECR.

Table 4. Outline of the sampling design for Question 2 of the fish monitoring program.

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

TARGET SPECIES AND LIFE STAGE

Adult / sub-adult, juvenile and young-of-year Macquarie perch were targeted. Individuals were classed as adults if they were > 150 mm TL, based on results from Ebner and Lintermans (2007) who found that males are sexually mature from this size. At the time of net sampling (i.e. late summer-early autumn) young-of-year in the river will be generally 40 - 70 mm TL based on results of the data collected between 2010 and 2021 (Figure 14). Individuals were generally considered juvenile if they fell between 100 – 150 mm TL.

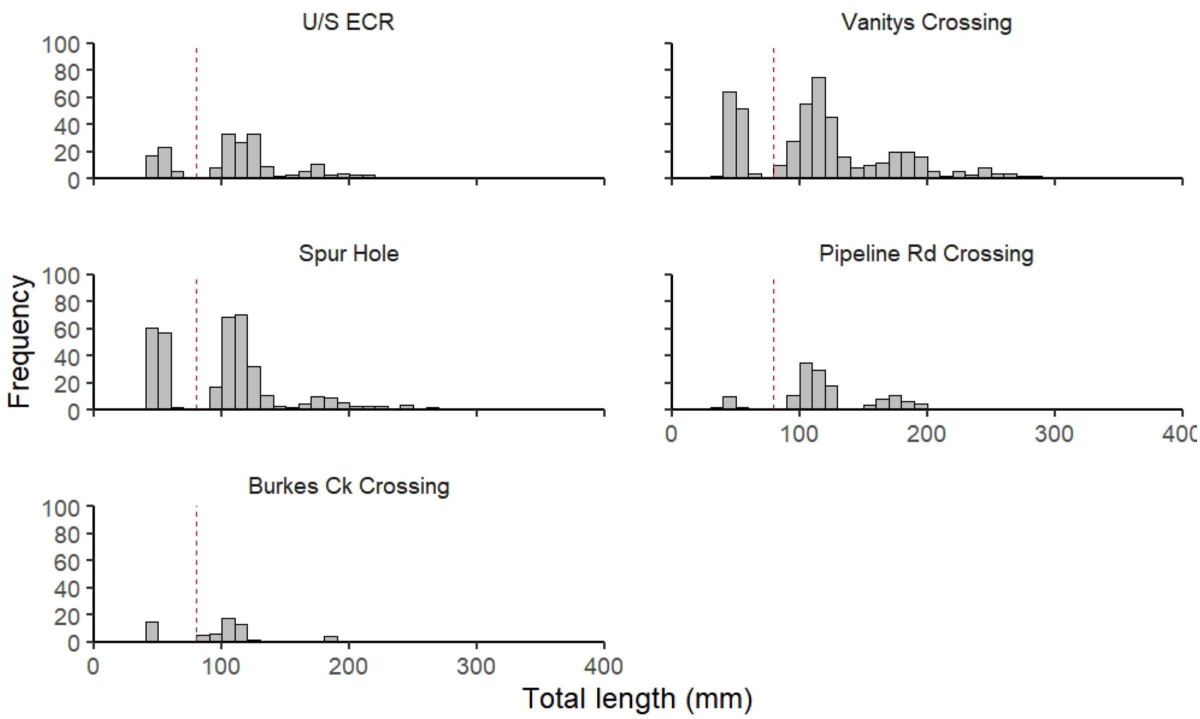


Figure 14. Length frequency of Macquarie perch captured in fyke nets and backpack electrofishing from Cotter River between 2010 and 2021 at sites; U/S ECR, Vanitys Crossing, Spur Hole, Pipeline Road Crossing and Burkes Creek Crossing (red dashed line indicates cut-off for length of young-of-year individuals < 80 mm TL).

SAMPLING METHODS AND NUMBER OF REPLICATES

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

TIMING

Sampling for this question was undertaken in March 2024 (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

Electrofishing data was excluded from formal analysis because of the high number of 0 data. Abundance was standardised for each sample as fish caught per unit effort (CPUE), with effort defined as hour per deployment of equipment. Unbalanced permutational analysis of variance (PERMANOVA) in a repeated measures design (highest interaction term excluded from model) following Anderson et al. (2008). It is unbalanced because of the different number of pools and samples across sites and years, explaining the use of Type III sum of squares. Data was Log10(x+1) transformed then resemblance matrix constructed with modified Gower (base 2) dissimilarity measure. Size classes (<80 mm, >80mm TL) included as variables. Site and phase as fixed factors, with random factor of year nested within phase. Tests were run with 9999 permutations of residuals under a reduced model. Pairwise comparisons for the significant site x year interaction indicated a mixture of significant and non-significant differences, and these do not seem to be consistent among the treatment (Cotter River) and reference (Kissops Flat) sites for each year group. Effects at a range of size classes were examined by performing separate ANOSIM (site and phase as fixed factors. Data was Log10(x+1) transformed then resemblance matrix constructed with modified Gower (base 2) dissimilarity measure. Size classes (<80 mm, >80mm TL) included as variables. Tests were run with 9999 permutations of residuals under a reduced model. Graphical presentations of site-level means with 95% confidence limits were used for pairwise comparisons of Macquarie perch mean CPUE among sites and years

RESULTS

General
A total of 58 Macquarie perch were captured by fyke nets in the Cotter River across the five sites in 2024, ranging in total length (TL) from 46 – 250 mm (Figure 15). Young-of-year (< 80 mm TL) and 1+ year old / juvenile (80 – 150 mm) individuals were captured at all riverine sites in 2024, except for Burkes Ck Crossing and Pipeline Rd Crossing (the two most upstream sites. CPUE of Macquarie perch (all sizes pooled) was not significantly different across phases or sites, but was significantly different among years within each operational phase (Table 5).

Table 5. Results of PERMANOVA analysis of fyke net catch-per-unit of Macquarie perch (all sizes combined) from Cotter River and Kissops Flat from 2010 – 2024 (bold text indicates significant result).

Source	df	SS	MS	Pseudo-F	P(perm)	perms
Site	4	0.032675	0.0081688	2.0325	0.0662	9931
Phase	2	0.010449	0.0052247	0.73719	0.4892	1383
Year (phase)	**11**	**0.080162**	**0.0072874**	**1.8132**	**0.0123**	**9889**
Site x Phase	7	0.033651	0.0048073	1.1961	0.27	9938
Res	767	3.0826	0.0040191			
Total	791	3.2331				

Juveniles and adults/ sub-adults

There was no significant difference in the CPUE of juvenile Macquarie perch among sites (Global R = 0.000, p = 0.382) or monitoring phases (Global R = 0.004, p =0.230). Relative abundance of Macquarie perch at U/S ECR (formerly Bracks Hole) and Vanitys Crossing was highly variable through time, with peaks in CPUE in 2013 and 2012 at each of these sites, respectively (Figure 16). In congruence, CPUE of juvenile or sub-adult Macquarie perch was also variable at the Kissops Flat reference site through time (Figure 16). Macquarie perch were detected from at least four of the five monitored sites in each monitoring year, and at all five sites in 2010, 2011, 2014, 2018,2019 and 2020 using fyke nets (Figure 16). Macquarie perch were not detected at Burkes Creek Crossing in 2012, 2013, 2016, 2023 or 2024 using fyke nets and at very low abundances at this site in other years, apart from 2020 which had relatively high abundances of individuals captured (Figure 16).

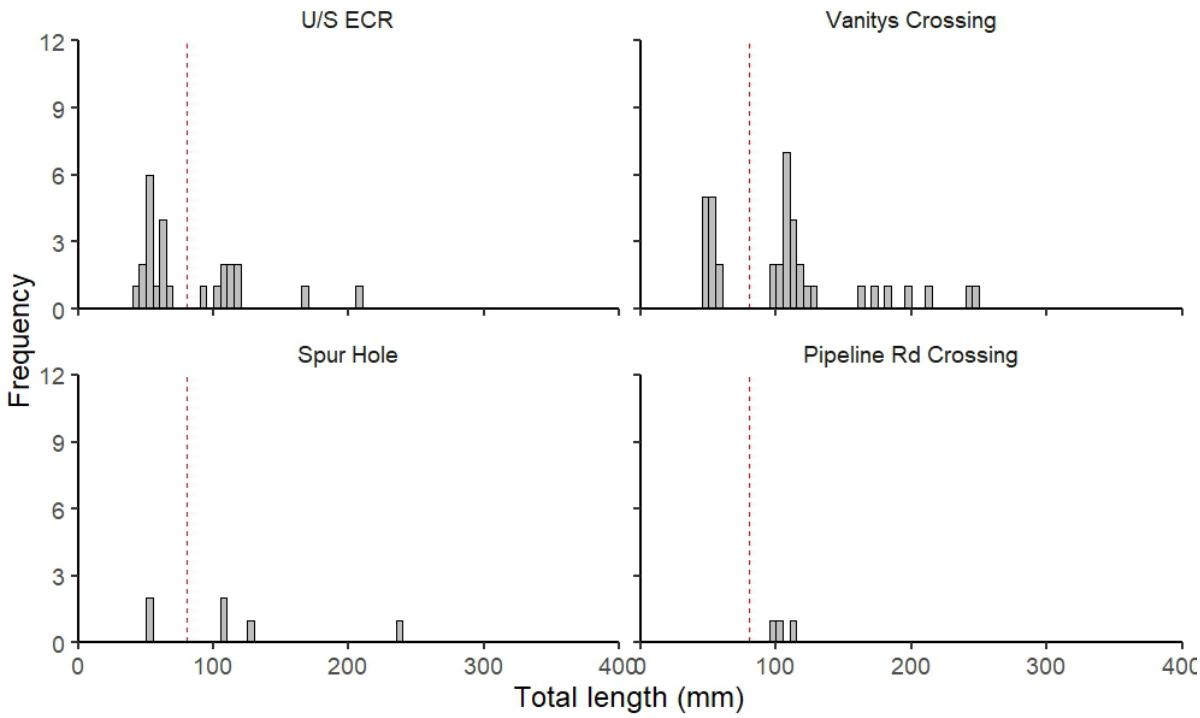


Figure 15. Length frequency of Macquarie perch captured in fyke nets and backpack electrofishing from Cotter River in 2024 at sites; U/S ECR, Vanitys Crossing, Spur Hole and Pipeline Road Crossing (red dashed line indicates cut-off for length of young-of-year individuals < 80 mm TL).

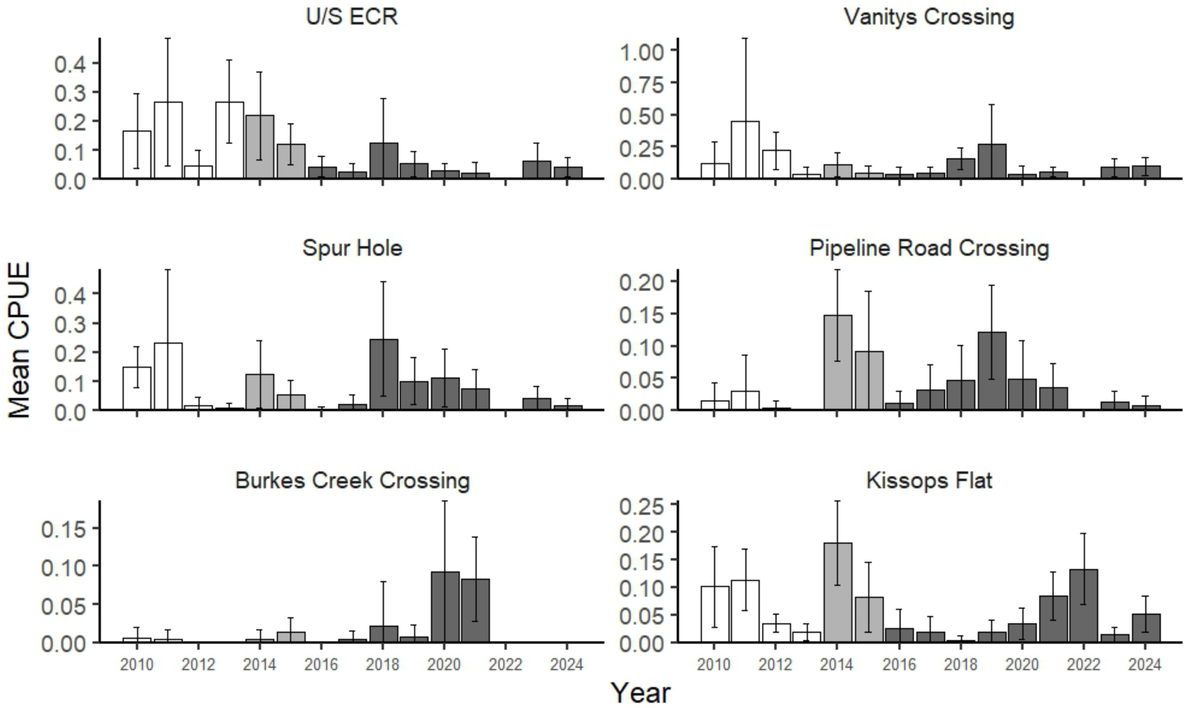


Figure 16. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of juvenile Macquarie perch (80 – 150 mm TL) captured in Cotter River using fyke nets between 2010 and 2024. (Note that Bracks Hole (sampled from 2010 – 2013) was replaced by U/S ECR (sampled in 2014 – 2024) as the most downstream riverine site). White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates operational phase of monitoring program. Note: Cotter River sites were not able to be sampled in 2022 due to high river flows throughout the sampling period.

Relative abundance of Macquarie perch captured using backpack electrofishing was highly variable between sites and years (Figure 17). Backpack electrofishing captured a total of three Macquarie perch (two 0+ and one 1+) at two of the five sites (U/S ECR and Vanitys Crossing) in 2024. 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.

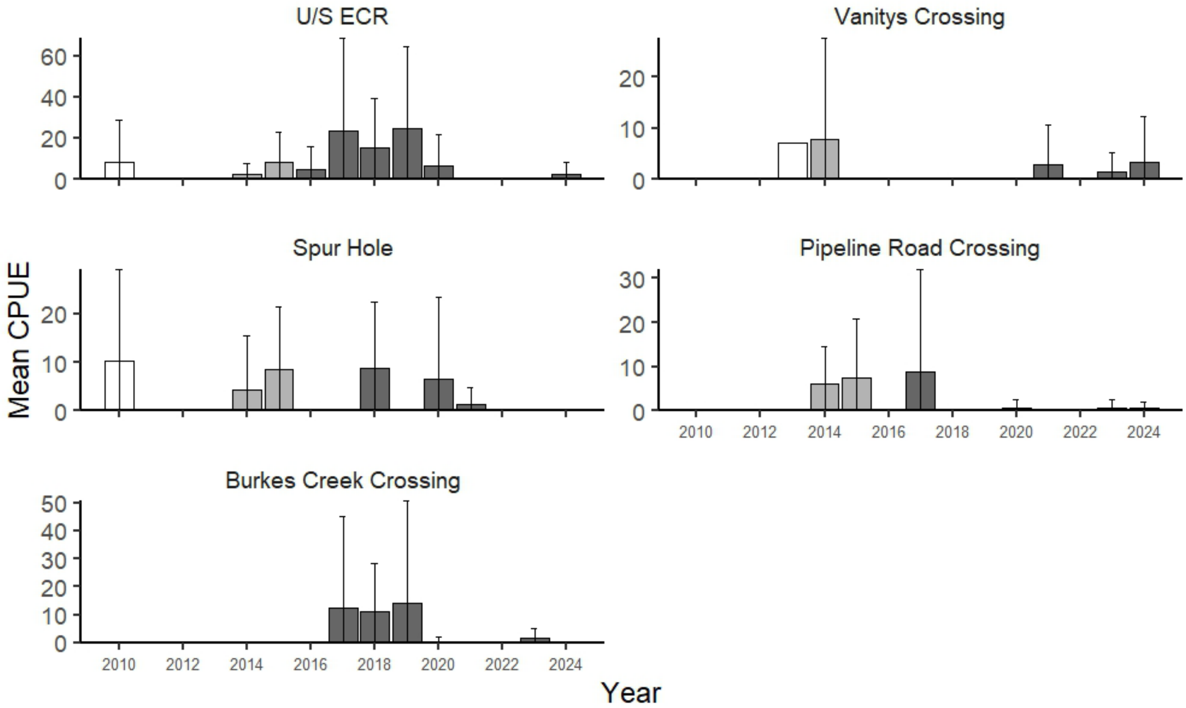


Figure 17. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Macquarie perch (all sizes pooled) captured in Cotter River by backpack electrofishing between 2010 and 2024. (Note that Bracks Hole (sampled from 2010 – 2013 was replaced by U/S ECR (sampled in 2014 – 2024) as the most downstream riverine site). White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates operational phase of monitoring program. Note: Cotter River sites were not able to be sampled in 2022 due to high river flows throughout the sampling period.

Young-of-year (YOY)

A total of 18 YOY Macquarie perch (< 80 mm TL) were captured using fyke nets in the Cotter River in 2024 (Figure 18 and Figure 15). Young-of-year were detected at the three most downstream sites on the Cotter River in 2024 (U/S ECD, Vanitys Crossing and Spur hole). There was no significant difference in the CPUE of YOY Macquarie perch captured in fyke among sites (Global R = 0.000, p = 1.0) or phases (Global R = 0.003, p = 0.254).

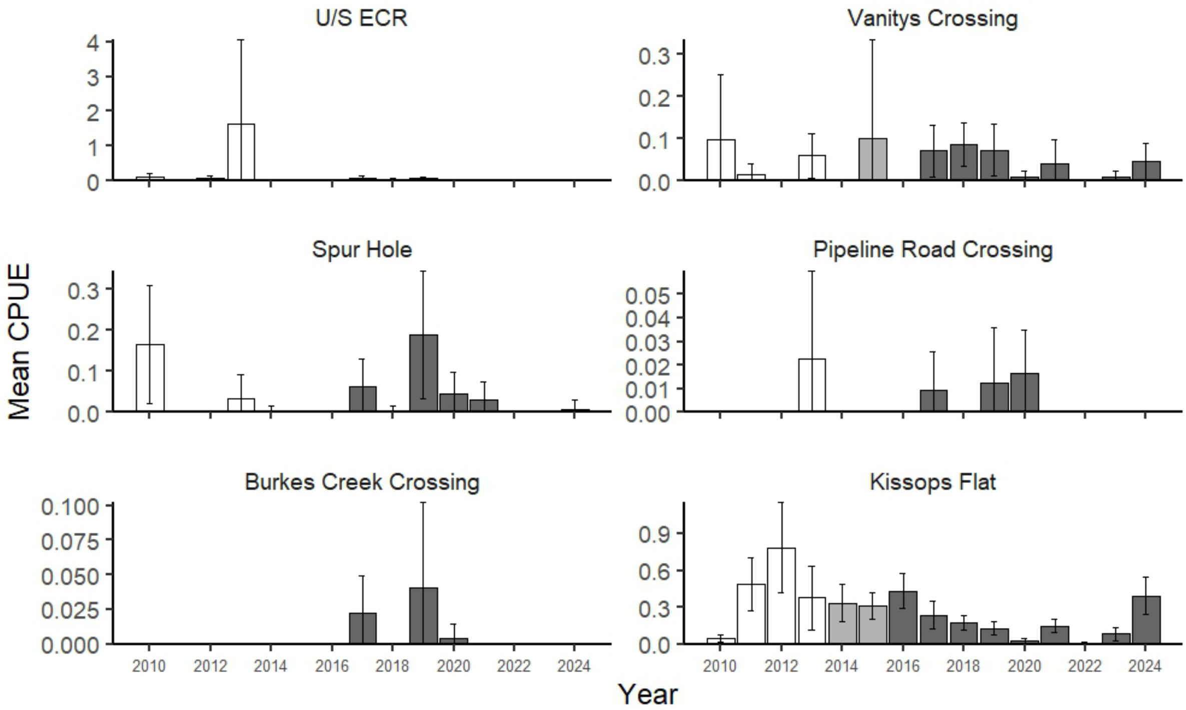


Figure 18. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Young-of-year Macquarie perch (< 80 mm TL) captured in Cotter River by fyke netting between 2010 and 2024. (Note that Bracks Hole (sampled from 2010 – 2013) and U/S ECR (sampled in 2014 – 2021) have been combined into the site U/S ECR to represent the site immediately u/s of the impounded waters). White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates operational phase of monitoring program. No sampling was undertaken in 2022 due to flooded river conditions during the monitoring period.

DISCUSSION AND CONCLUSIONS

Relative abundance

As has been the case since monitoring began in 2010, relative abundance of Macquarie perch in Cotter River in 2024 was variable between sites, as determined by both fyke netting and backpack electrofishing. Previously relative abundance generally decreased with distance upstream from Cotter Reservoir. These results are consistent with previous findings that this Macquarie perch population was restricted to Cotter Reservoir and the Cotter River downstream of Vanitys Crossing until the fishway was built in 2001, with the species taking considerable time in extending their population to newly accessible upstream river reaches (Broadhurst *et al.* 2012a, Lintermans 2013).

One-year old individuals were present at all sites except one (Burkes Creek Crossing) in 2024. This follows on from young-of-year being detected in the Cotter River 2023. These results suggest that the past year has been suitable for survival and growth of juvenile Macquarie perch in the Cotter River.

Young-of-year Macquarie perch were captured from three sites on the Cotter River in 2024 and comprised 30.5% of the total number of Macquarie perch captured. Recruitment in 2024 was relatively low compared to most other years. The mechanism behind the continued low levels of Macquarie perch recruitment is not as clear as previous years (when flood pulses most likely impacted on early development). To date, monitoring suggests that the Macquarie perch population is surviving, but not thriving in the Cotter River.

Distribution

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

RECOMMENDATIONS

Juveniles and adults

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

Juvenile Macquarie perch were detected at most sites in 2024. At this stage, no management intervention is recommended for juvenile Macquarie perch in Cotter River.

Young-of-year

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

Young-of-year were detected at three of five riverine sites in 2024, suggesting conditions were suitable for recruitment in the catchment. No management intervention is recommended at this stage.

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

BACKGROUND

Two-spined blackfish have long been absent from Cotter Reservoir (Lintermans 2002, Ebner *et al.* 2008) (thought to be a result of excessive sedimentation smothering potential spawning sites) apart from a small number of individuals detected in 2012, possibly washed down from the river during flooding (Lintermans *et al.* 2013) (Figure 19). However, the species was present in the river reach inundated by the ECR (Ebner *et al.* 2008, Lintermans *et al.* 2013). Inundated habitats around the perimeter of the ECR should provide suitable spawning habitats for the species. The monitoring program will determine whether the species persists in the newly inundated river reach, and subsequently expands to colonise newly inundated habitats around the perimeter of the ECR.

METHODS

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

Table 6. Outline of the sampling design for Question 3 of the fish monitoring program.

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

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

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

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

RESULTS

Three Two-spined blackfish were captured in by fyke nets in the ECR in 2024 as part of the all reservoir netting component, with two captured in the inundation zone (Figure 19). This is the first time since 2020 that Two-spined blackfish have been captured in the ECR (Figure 19). Lengths of Two-spined blackfish captured from ECR in 2024 ranged from 163 – 250 mm (TL), meaning that all individuals were mature adults. There was no Two-spined blackfish captured in the bait traps set in the ECR in 2024. For only the second time since the program began in 2010, no Two-spined blackfish were captured in Bendora Reservoir (the other being 2021 (Figure 19). 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.

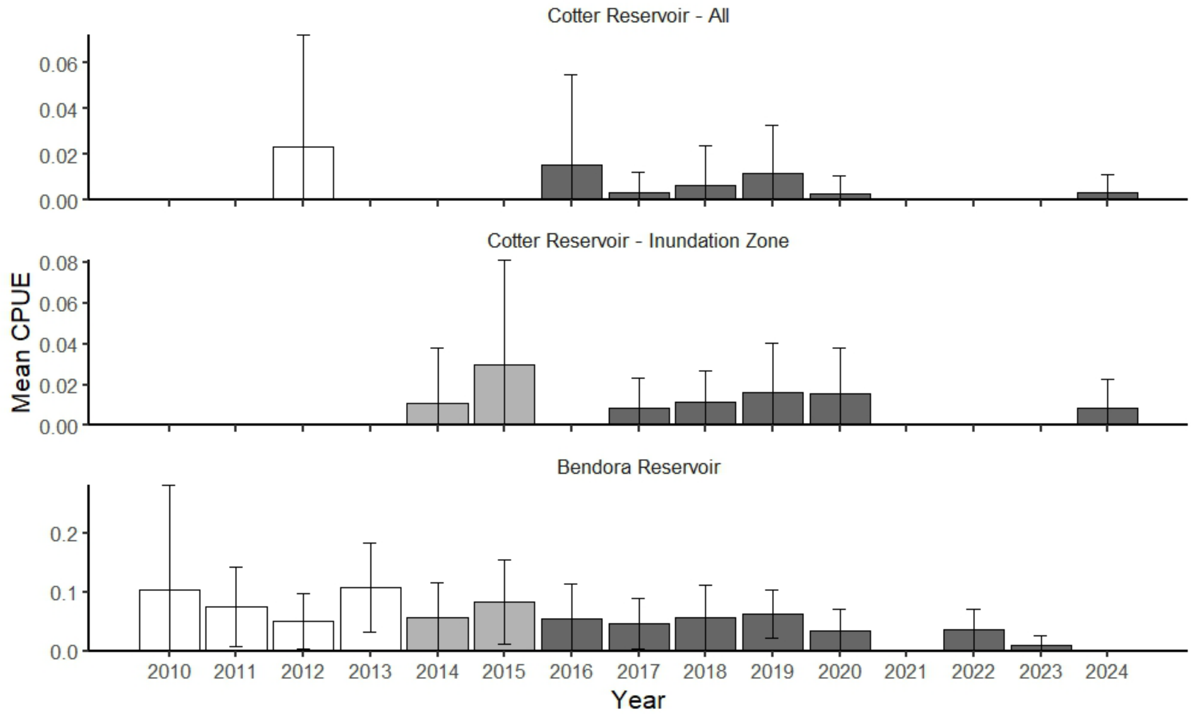


Figure 19. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Two-spined blackfish captured by fyke netting in Cotter Reservoir (both all around the reservoir and just the inundation zone) and Bendora Reservoir between 2010 and 2024. For Cotter Reservoir, white bars indicate baseline phase, grey bars indicate filling phase and white bars with diagonal stripes indicates operational phase of monitoring program.

DISCUSSION AND CONCLUSIONS

Two-spined blackfish have been in very low abundances in Cotter Reservoir in the 15 years of ECR monitoring. This result supports previous research that identified that the original Cotter Reservoir had sub-optimal habitat for Two-spined blackfish as a result of forestry and associated sedimentation of the reservoir smothering rocky substrate preferred by this species (Lintermans 1998, Ebner and Lintermans 2007, Ebner *et al.* 2008, Broadhurst *et al.* 2011, Broadhurst *et al.* 2012b).

The capture of Two-spined blackfish in the newly inundated upstream third of the enlarged Cotter Reservoir, suggest that newly inundated shoreline of the enlarged Cotter Reservoir may serve as suitable habitat for the species, though at low densities. To date, 35 Two-spined blackfish have been captured in fyke nets in the ECR, with only two smaller than 150 mm TL (size at sexual maturity). No recruitment of Two-spined blackfish has yet been detected in the ECR. It is likely that Two-spined blackfish in the ECR are larger adults that have colonised from the Cotter River upstream.

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

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

RECOMMENDATIONS

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

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

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

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

BACKGROUND

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

METHODS

Sampling design for Question 4 is similar to the baseline monitoring program for Question 3 (Lintermans *et al.* 2013) (Table 7). One of the reference reservoirs from the baseline monitoring program (Corin Reservoir) was dropped from the subsequent (filling and operational) monitoring program to minimise costs.

Table 7. Outline of the sampling design for Question 4 of the fish monitoring program.

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

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

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

Only Rainbow trout was used in the comparative analyses, as Brown trout does not occur in Bendora Reservoir. CPUE was then scaled to shoreline length at the time of sampling. This was done by multiplying the CPUE for each net night by the proportional change in shoreline as the reservoir filled for a given year. CPUE of trout was compared using a multivariate Permutational analysis of variance (PERMANOVA) in a repeated measures design (highest interaction terms excluded from model) following Anderson et al. (2008). Data was Log10(x+1) transformed then a resemblance matrix was constructed with modified Gower (base 2) dissimilarity measure. Reservoir and phase were treated as fixed factors, and section nested within reservoir and year nested within phase were treated as random factors. Tests were run with 9999 permutations of residuals under a reduced model with Type III sum of squares. Graphical presentations of mean CPUE within each reservoir section (five in total), with 95% confidence limits (with Bonferroni corrections), were used to explore pairwise differences in trout abundance. Size (fork length) variation between years was explored using non-parametric Kruskal-Wallis ANOVA due to severe violations of the data (principally kurtosis) that could be not rectified by data transformation. Summary analysis of the Brown trout CPUE and lengths is also provided for Cotter Reservoir (as they are not present in Bendora Reservoir).

RESULTS

Abundance and distribution

Seven Rainbow trout and 0 Brown trout were captured in gill nets in the ECR in 2024. The total number of trout captured was the second lowest (lowest was five in 2022) (Figure 20). Six Rainbow trout were captured in Bendora Reservoir in 2024 (Brown trout are not present in this reservoir). There was no significant effect of reservoir, phase or year on the relative abundance of Rainbow trout captured in the ECR (Table 8), though there was a significant effect of reservoir by year interaction (Figure 20). The latter was likely driven by the scarcity of Rainbow trout in Bendora Reservoir in 2016 and low abundances again in 2017. For the first time since entering the operational phase, no Brown trout were captured in gill nets, and continues a sudden drop in abundance of this species in Cotter Reservoir since 2021 (Figure 21).

Table 8. Results of PERMANOVA analysis of gill net catch-per-unit-effort (scaled to relative net effort versus shoreline length at the time of sampling) of Rainbow trout captured in Cotter Reservoir and Bendora Reservoir from 2010 – 2024 (bolded text indicates statistically significant difference at the P(perm) 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	perms
Reservoir	1	0.033465	0.033465	0.69902	0.4193	9840
Phase	2	0.061424	0.030712	0.87726	0.436	9958
Section(Reservoir)	8	0.20254	0.025318	1.6575	0.1119	9924
Year (phase)	12	0.32066	0.026722	1.7494	0.0522	9923
Reservoir x Phase	2	0.1283	0.064149	1.1329	0.1462	9910
Reservoir x Year(phase)	**12**	**0.53289**	**0.044407**	**2.9073**	**0.0008**	**9918**
Phase x Section(Reservoir)	16	0.41118	0.025698	1.6825	0.0524	9923
Residuals	546	8.3398	0.015274			
Total	599	10.071				

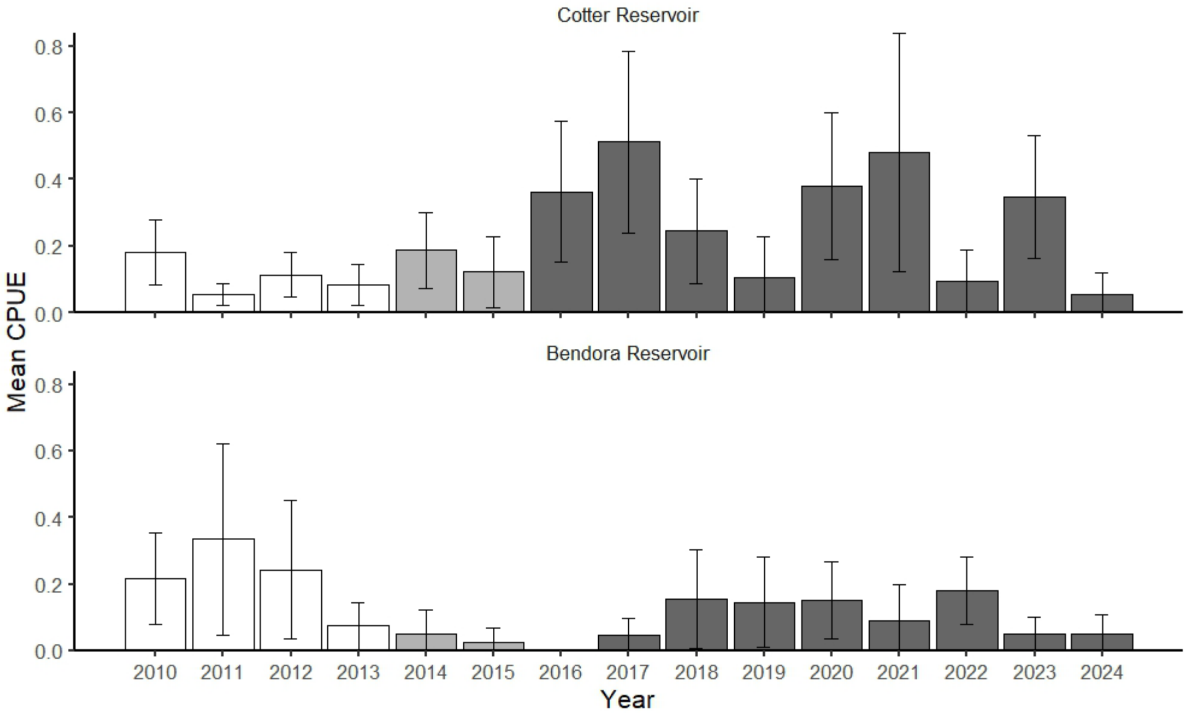


Figure 20. Mean catch-per-unit-effort (± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of adult Rainbow trout captured in Cotter Reservoir and Bendora Reservoir using gill nets each year from 2010 until 2024. White bars indicate baseline phase, light-grey bars indicate filling phase and dark-grey bars indicates operational phase of monitoring program.

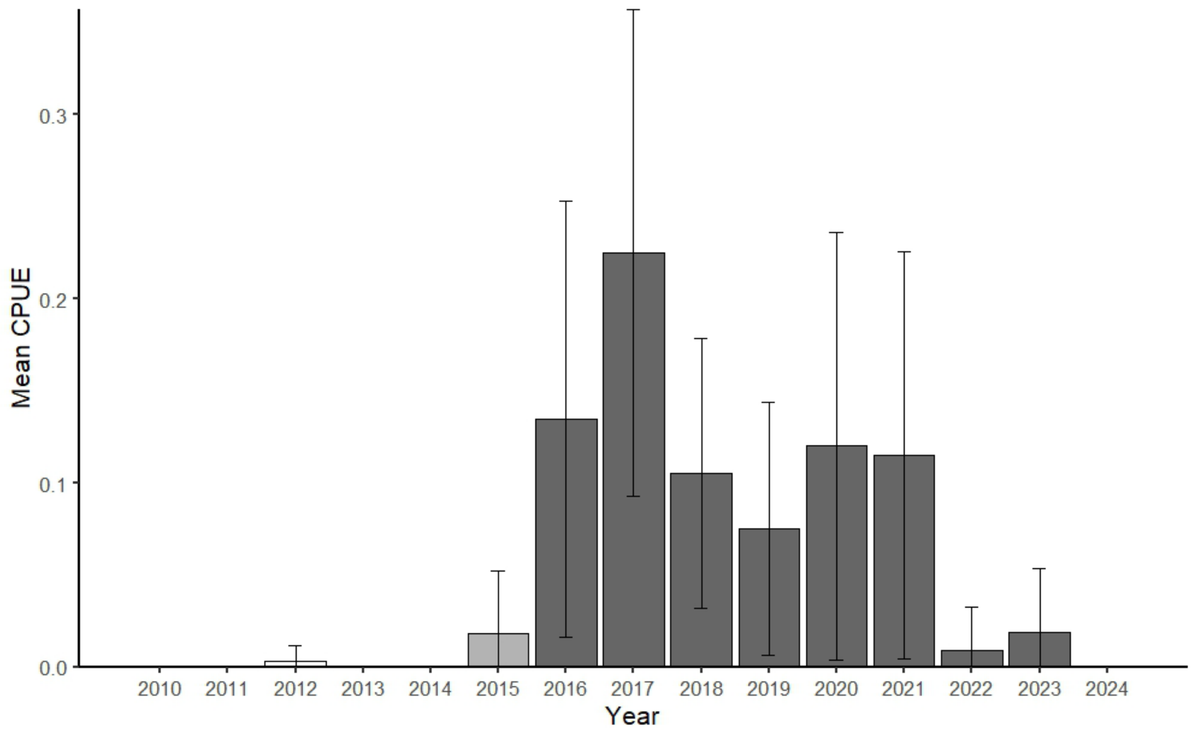


Figure 21. Mean catch-per-unit-effort (± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of adult Brown trout captured in Cotter Reservoir using gill nets each year from 2010 until 2024. White bars indicate baseline phase, light-grey bars indicate filling phase and dark-grey bars indicates operational phase of monitoring program.

Size composition

Size composition of captured Rainbow trout in Cotter Reservoir has been stable since monitoring commenced. Size of adult Rainbow trout captured in the ECR during 2024 ranged from 265 – 350 mm Fork Length (FL) (Figure 22). Median size of adult trout in the ECR was similar between all years (Figure 24). Size of adult Rainbow trout captured in Bendora Reservoir during 2024 was very uniform, ranging from 318 – 381 mm Fork Length (FL) (Figure 23).

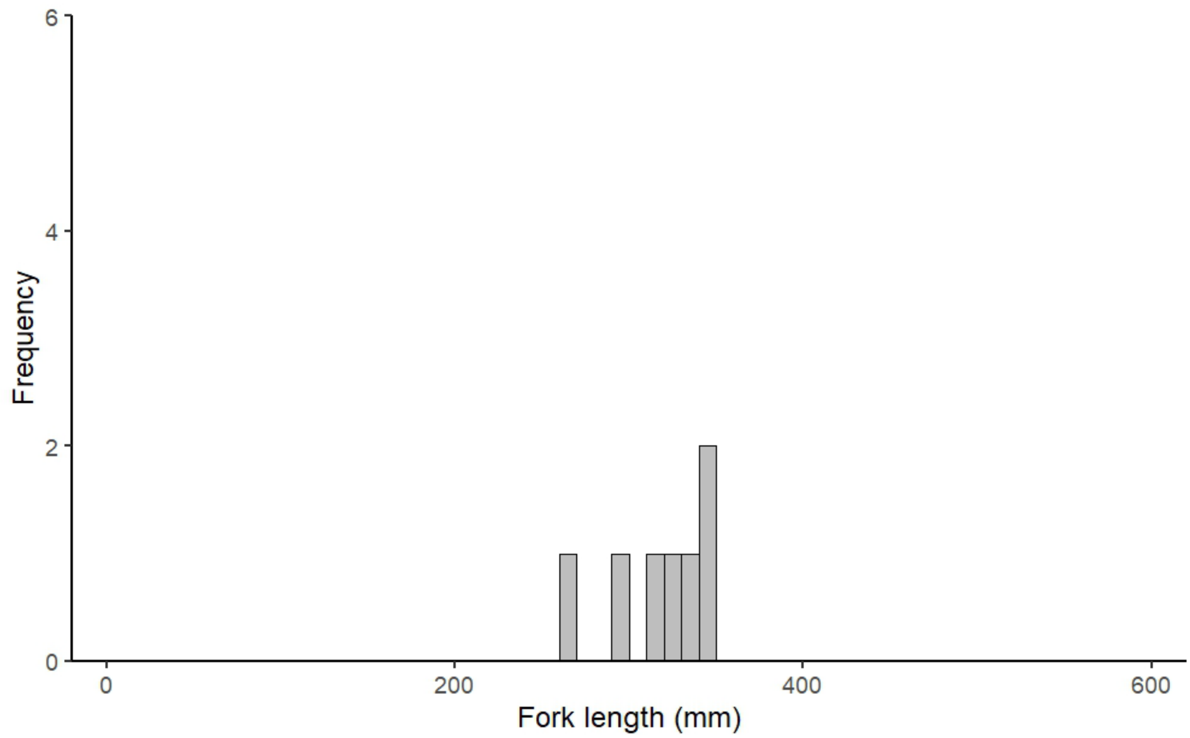


Figure 22. Length frequency of Rainbow trout (n = 7) captured from the ECR in autumn 2024 using gill nets.

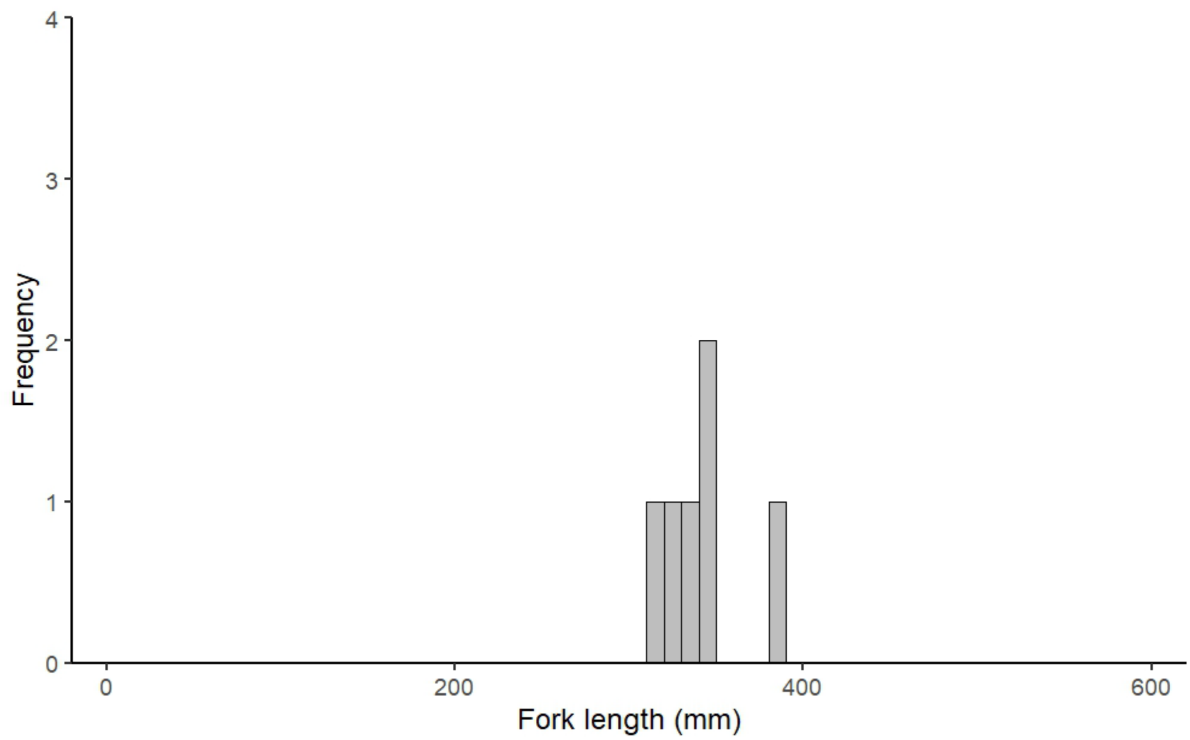


Figure 23. Length frequency of Rainbow trout (n = 6) captured from Bendora Reservoir in autumn 2024 using gill nets.

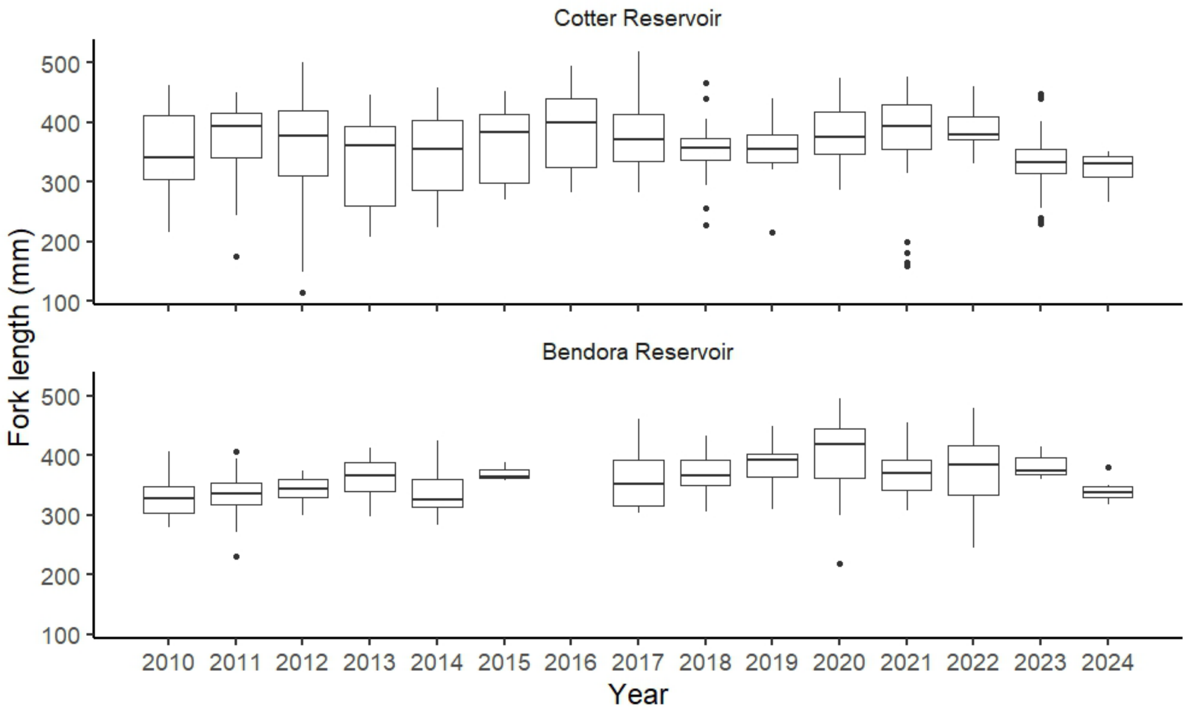


Figure 24. Boxplots of adult Rainbow trout length captured in gill nets each year from Cotter and Bendora Reservoirs from 2010 to 2024 (solid line = median, box represents 25 – 75th percentiles, bars represent minimum and maximum lengths and black circles represent outliers).

DISCUSSION AND CONCLUSIONS

Abundances of Rainbow trout within the Cotter Reservoir have been somewhat variable over the past 15 years, with higher abundances observed during filling and operational years (especially years of 2016, 2017, 2020 and 2021). The abundance of Rainbow trout in Bendora Reservoir was highest in early baseline years, and has been relatively stable since 2018, albeit with a minor decrease in 2023 and 2024. Capture of Rainbow trout in Bendora appears to be generally related to temperature at the time of sampling, where a negative correlation between surface water temperature and number of Rainbow trout captured exists (Broadhurst *et al.* 2023). Lowest catches seem to occur when surface water temperature is > 17 °C, with the exception of 2022, which recorded greater than median number of Rainbow trout, even though water temperature was ~20 °C (Broadhurst *et al.* 2023).

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

To date, there was no significant change in the size composition of adult Rainbow trout captured between years in Cotter Reservoir. It was expected that as the reservoir filled, food resources would increase and would lead to increases in size of adult trout (albeit with a time lag of a few years) (Kimmel and Groeger 1986, Ploskey 1986, O'Brien 1990). Current operational conditions have not led to a change in the size of trout in the ECR, and increased threat of predation due to increase size has not occurred thus far.

RECOMMENDATIONS

Methods for assessing the population metrics of adult trout relative abundance, distribution and size appear to be adequate. No changes to monitoring are recommended.

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

<u>QUESTION 5:</u> 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 10). 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 9. Outline of the sampling design for Question 5 of the fish monitoring program.

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

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

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

Brown trout are only rarely captured in the river monitoring, so analysis of their abundance and size distribution was not conducted. Abundance of Rainbow trout was standardised for each technique as fish caught per net hour for fyke netting and fish caught per electrofishing shot on-time for electrofishing (represented as CPUE). Unbalanced permutational analysis of variance (PERMANOVA) on trout e-fish CPUE (shots as replicates) using the data from the 4 x 30 m shots only (see Anderson *et al.* 2008). Data was Log10(x+1) transformed then a resemblance matrix was constructed with modified Gower (base 2) dissimilarity measure. Cotter River site and year are fixed factors. To test between differences CPUE between sites and years separately, PERMANOVAs were conducted using Type III sum of squares in a repeated measures design (site and year as fixed factors). Graphical presentations of site-level mean CPUE for each year (with 95% confidence limits with Bonferroni corrections) were used to explore pairwise variations in Rainbow trout among sites and years.

RESULTS

The number of Rainbow trout captured in Cotter River in 2024 was higher than most years (with exception of 2021 which was extremely high). A total of 224 Rainbow trout were captured from six riverine sites on the Cotter River using fyke nets and backpack electrofishing in 2024 (Figure 25). Rainbow trout captured in 2024 from the Cotter River ranged in size from 63 – 330 mm FL (Figure 25). Rainbow trout were captured at every riverine site and numbers were relatively consistent between sites, ranging from 24 – 45 per site. The size composition of Rainbow trout in the Cotter River upstream of the ECR has been somewhat variable at the site level, though variability of medians is usually constrained to below 300 mm FL (Figure 25 and Figure 26). Lengths of captured trout in 2024 were bi-modal, with peaks around 100 mm and 250 mm FL (Figure 25 and Figure 26).

The presence of Rainbow trout at each site is patchy over years, though the likelihood of detecting this species at a site generally increased with distance upstream of the ECR (Figure 27 and Figure 28). There was a significant difference in the relative abundance of Rainbow trout caught by electrofishing between sites and between years (Table 11). The most downstream site (U/S ECR) had significantly lower Rainbow trout abundance than the most upstream three sites (including the reference site). Of the test sites, Pipeline Road Crossing and Burkes Creek Crossing had the most consistent frequency of detection of this species for both fyke netting and backpack electrofishing (Figure 27 and Figure 28) with Bracks Hole / U/S ECR recording lower relative abundances of Rainbow trout compared to Spur Hole, Pipeline Road Crossing, Burkes Creek Crossing and Cotter Hut (Figure 27 and Figure 28). Vanitys Crossing also had a lower relative abundance of Rainbow trout compared to Burkes Creek Crossing (Figure 27 and Figure 28). In general, backpack electrofishing was more likely to detect the presence of Rainbow trout than fyke netting (Figure 27 and Figure 28). Brown trout have been a rare capture in the standardised sampling with most caught towards the upstream end of the study reach (i.e. closer to Bendora Dam). There were three Brown trout captured in 2024 (all via backpack electrofishing); one each from Vanitys Crossing (225 mm FL), Spur hole (120 mm FL) and Pipeline Road crossing (510 mm TL).

Table 10. Results of PERMANOVA analysis of trout relative abundance (determined by 4 x 30 m backpack electrofishing CPUE) in Cotter River from 2010 – 2024 (bold text indicates statistically significant difference at the P(perm) 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Site	**5**	**4.5295**	**0.9059**	**3.1609**	**0.0089**	**9947**
Phase	2	8.8737	4.4369	2.1261	0.1608	9956
Year(phase)	**11**	**23.944**	**2.1767**	**7.5952**	**0.0001**	**9918**
Site x Phase	10	4.7001	0.47001	1.64	0.0912	9931
Residuals	288	82.539	0.28659			
Total	316	125.82				

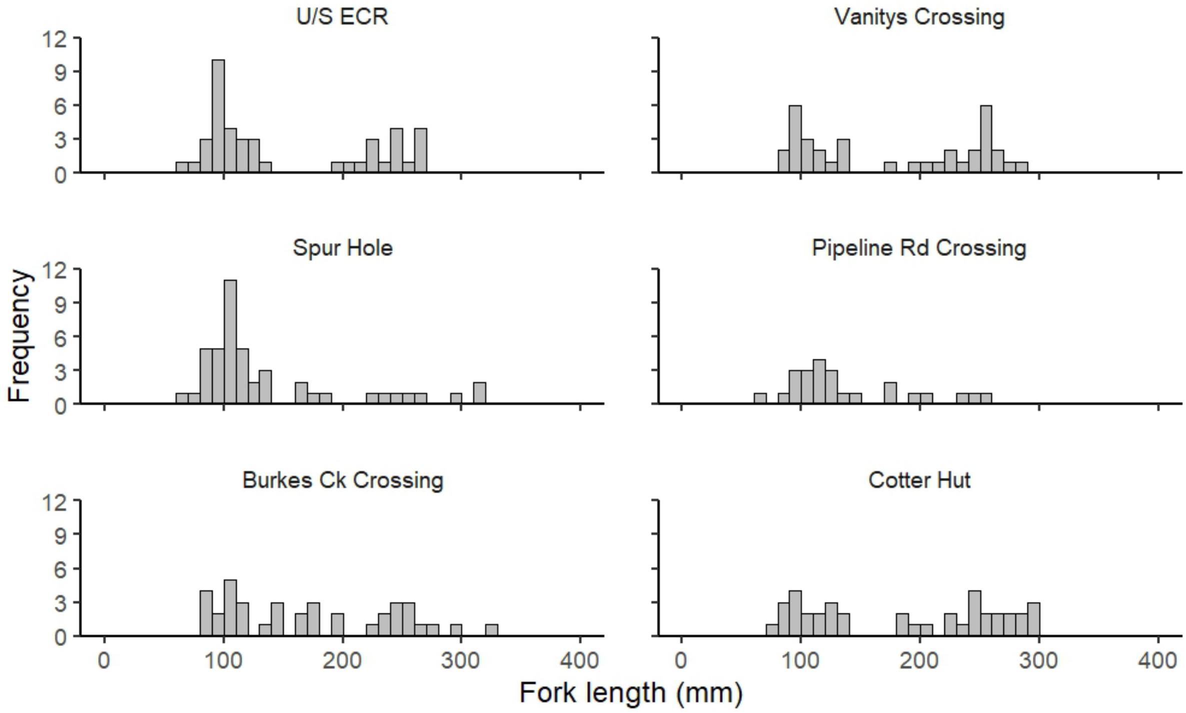


Figure 25. Length frequency of Rainbow trout captured from Cotter River at U/S ECR; Vanitys Crossing; Spur Hole; Pipeline Road Crossing and Burkes Creek Crossing in 2024 using fyke nets and backpack electrofishing (inclusive of additional backpack electrofishing effort).

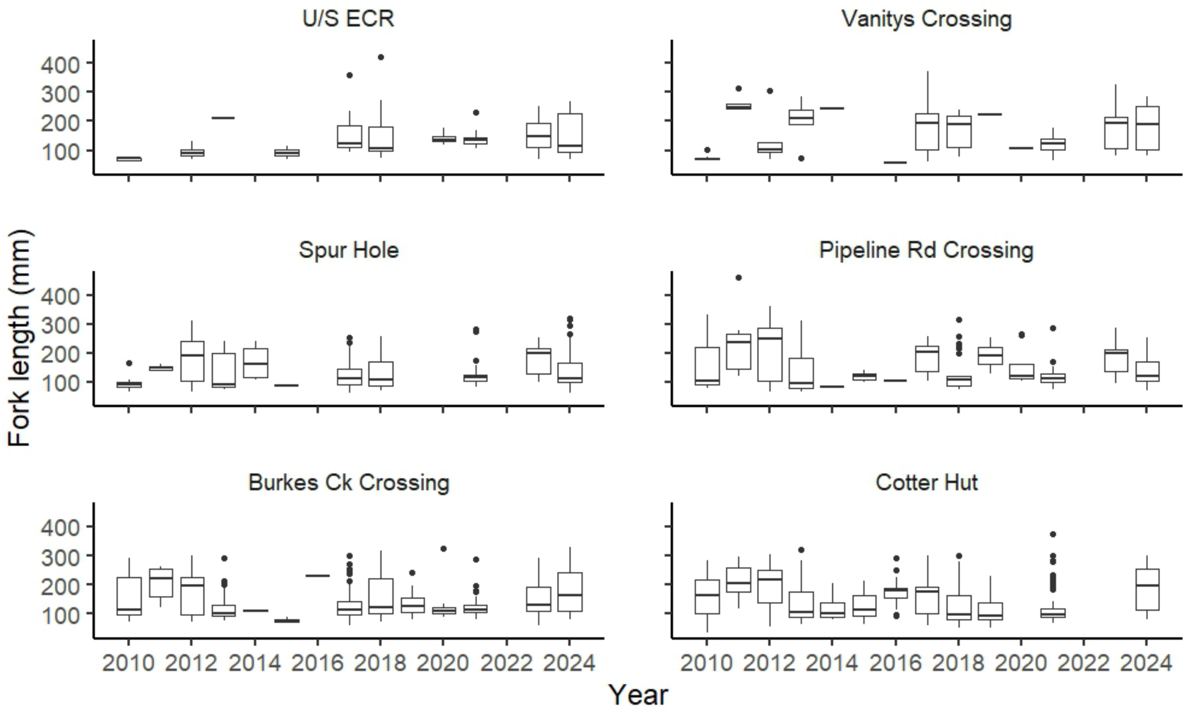


Figure 26. Boxplots of Rainbow trout captured in Cotter River from 2010 to 2024 (solid line = median, box represents 25 – 75th percentiles, bars represent minimum and maximum lengths and black circles represent outliers). Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2024). 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.

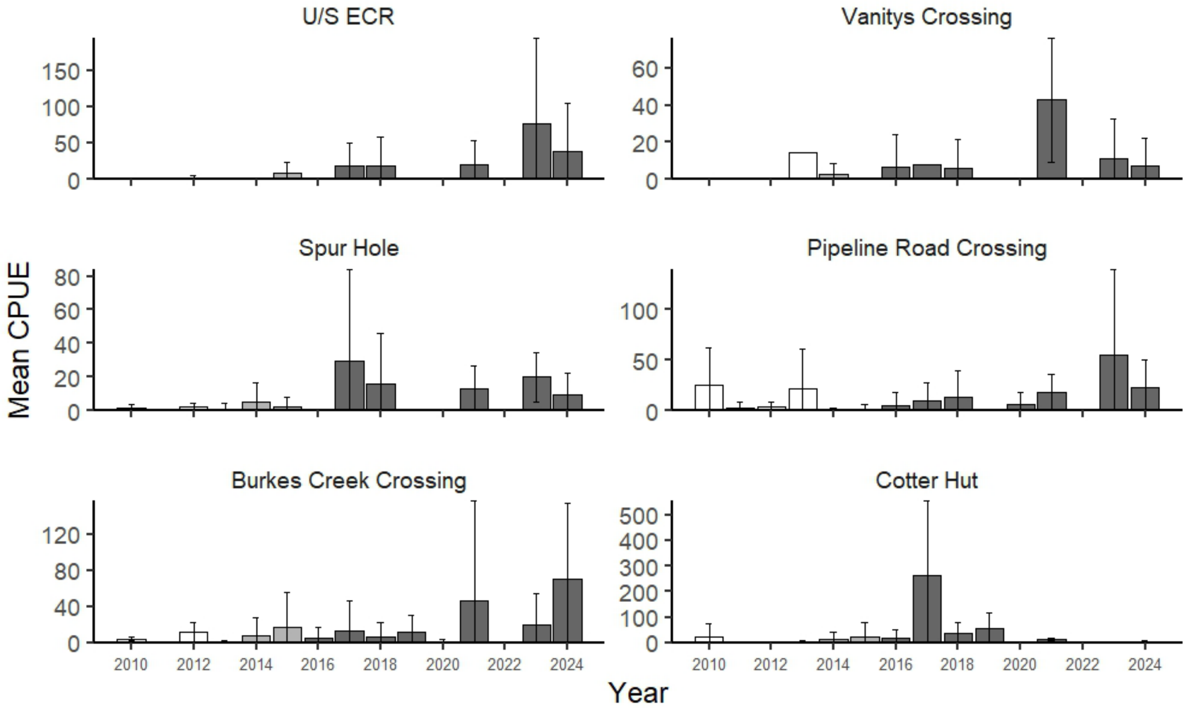


Figure 27. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Rainbow trout captured in Cotter River by backpack electrofishing (4 x 30 m shots) between 2010 and 2024. (Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2024). Cotter Hut was not able to be sampled in 2020 and 2023 because of fire and COVID-19 restrictions. Sites between Cotter Reservoir and Bendora Reservoir were not able to be sampled in 2022 due to persistent high river flows. White bars indicate baseline phase, grey bars indicate filling phase and white bars with diagonal stripes indicates operational phase of monitoring program.

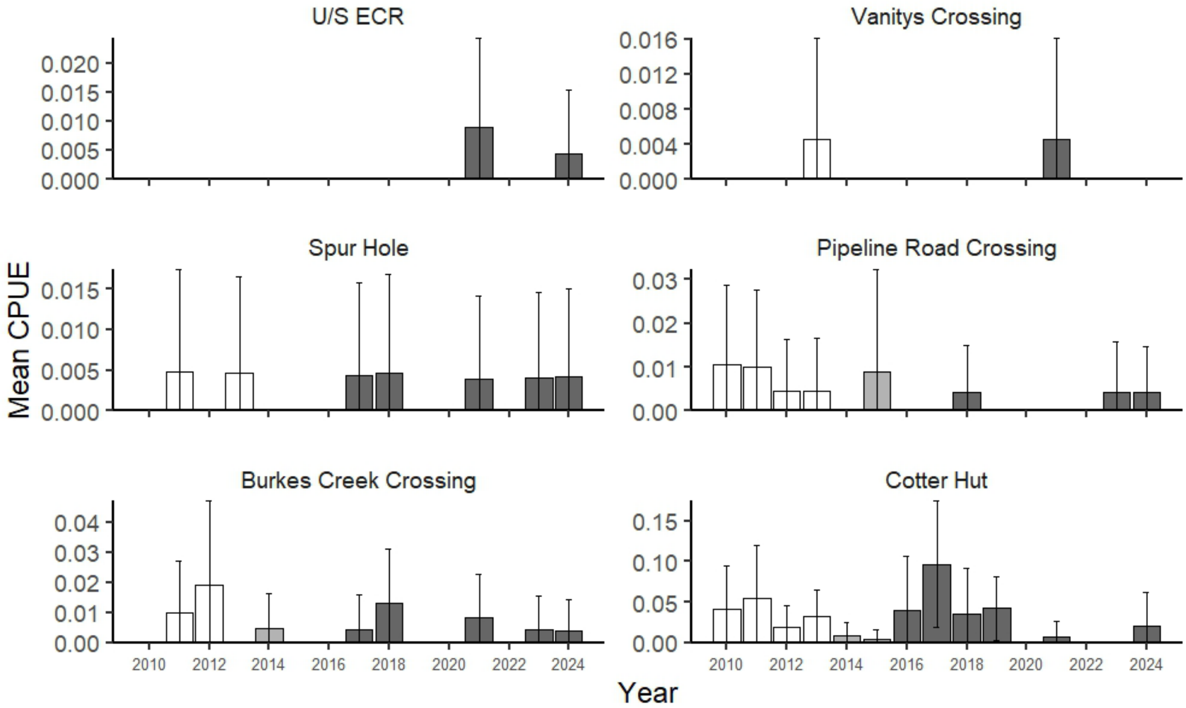


Figure 28. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Rainbow trout captured in Cotter River by fyke net between 2010 and 2024. (Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2024). Cotter Hut was not able to be sampled in 2020 and 2023 because of fire and COVID-19 restrictions. Sites between Cotter Reservoir and Bendora Reservoir were not able to be sampled in 2022 due to persistent high river flows.

DISCUSSION AND CONCLUSIONS

The increased electrofishing effort employed from 2017 onwards captured 2 – 6-fold the number of trout captured on average than the standard 4 x 30 m method (Table 12). This no doubt provides a more representative sample of the length of trout in the Cotter River at each site and at all sites combined. We recommend from 2025 that this data be used for analysis, instead of the 4 x 30 m data. This means that no analysis will be possible for filling and the first year of operation phase.

Table 11. Raw numbers of Rainbow trout captured in the Cotter River (all sites combined) via backpack electrofishing using both the standard 4 x 30m section method and the additional sampling method from 2010 – 2024.

		No. trout captured	
Year	**4x30 m**	**Additional effort**	**Total**
2010	17	77	**94**
2011	4	34	**38**
2012	20	91	**111**
2013	22	99	**121**
2014	10	-	**10**
2015	15	-	**15**
2016	14	-	**14**
2017	55	183	**238**
2018	22	197	**219**
2019	12	62	**74**
2020	6	19	**25**
2021	89	340	**429**
2023	82	123	**205**
2024	88	136	**224**

Based on data using the basic electrofishing methods (just using the 4 x 30 m sections), there has been no statistically significant change in Rainbow trout abundance in the Cotter River upstream of the ECR based on phase. Similarly, Brown trout abundance in the Cotter River remains extremely low. The low numbers of trout recorded per site using the basic sampling effort and the extremely high variability makes it difficult to detect statistically significant change in trout length. The increased sampling effort employed from 2017 – 2024 helps to reduce fish length variability (by increasing the sample size), and so will allow an increased chance of detecting significant trends in trout length in future years. Because of a predicted increase of food resources it was expected that the population of trout that reside in the ECR will increase in abundance, and likely spill over into the upstream Cotter River (Lintermans 2012). Whether the likely increase in riverine trout abundance is permanent (i.e. increase in resident riverine trout) or seasonal (i.e. increased spawning-run abundance) is unknown. The current monitoring program will not detect spawning run increases in riverine trout abundance, as sampling is not conducted in late autumn/winter when trout spawning occurs. Should the Rainbow trout population (and length of individuals) in the Cotter River increase, this may cause declines in the Macquarie perch and Two-spined blackfish present in the river through competition for resources (food and potentially shelter) and potentially by predation, particularly upon Macquarie perch larvae and juveniles and all size classes of Two-spined blackfish. The current monitoring results suggest that such a permanent increase in riverine trout abundance has yet to occur or cannot be detected (based on relative abundance and size composition of riverine trout in years since filling).

The high abundances of Rainbow trout in 2024 appears to be attributable to a very successful spawning event and subsequent survival over winter / spring 2023. This was observed at both the test sites (Cotter River between Bendora Dam and Cotter Reservoir) and also at the reference site (Cotter Hut in 2022 only), which indicates that this successful recruitment event was driven by catchment scale effects (likely climate driven), and not as a result of the operation of the ECR. Initial impacts of increased trout abundances will likely be non-lethal (competition for resources – food and refuge), though may translate into increased predation of juvenile Two-spined blackfish and Macquarie perch as trout increase in size.

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

Only 16 Brown trout have been collected in the standardised monitoring of the river over the 15 years of monitoring to date. This indicates that Brown trout are still in relatively low abundance when compared to Rainbow trout in the Cotter River. The continued presence of large Brown trout in the river, coupled with the increased abundance of this species in the ECR (see Question 4) increases the likelihood of future population expansion of this species. The threat of Brown trout to native species in the river at this stage is low because of their low abundance (but see concerns for reservoir in Question 4 recommendations above). Monitoring should continue to report on Brown trout numbers as the ECR water level fluctuates to determine if this threat changes.

RECOMMENDATIONS

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

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

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

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

BACKGROUND

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

METHODS

Sampling design for Question 6 is a refinement of that conducted in the baseline monitoring program (Lintermans *et al.* 2013) (Table 13). 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 12. 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 89 Rainbow trout and two Brown trout 150 mm FL or greater were captured in 2024 (Table 14). Visual examination of stomach contents detected no Two-spined blackfish or Macquarie perch in any of the fish examined (Table 14).

Table 13. Details of trout captured and examined for fish and fish remains from the Cotter River in 2024 via backpack electrofishing

Site	Species	Fork Length (mm)	Fish remains in stomach
U/S ECR	Rainbow trout (n=15)	200 – 269	No
Vanitys Crossing	Rainbow trout (n=19)	177 – 285	No
	Brown trout (n=1)	225	No
Spur Hole	Rainbow trout (n=11)	165 – 320	No
Pipeline Road Crossing	Rainbow trout (n=5)	175 – 255	No
	Brown trout (n=1)	510	No
Burkes Creek Crossing	Rainbow trout (n=20)	150 – 330	No
Cotter Hut	Rainbow trout (19)	185 - 296	No*

*A frog (*Crinia signifera*) was found in the stomach of a 284 mm FL Rainbow trout from Cotter Hut.

The number of trout stomachs examined for evidence of predation on threatened fish increased significantly in 2017 – 2018, 2021, 2023 and 2024 with the addition of the extra sampling effort (20 individuals or 1 km of river) compared to the previous three years (2014-2016), however there was a dramatic reduction in abundance of the targeted size range in 2019 - 2020 (Table 15).

Table 14. Comparison of number of trout stomachs examined from 2010 - 2023.

	2010	2011	2012	2014- 2016	2017	2018	2019	2020	2021	2023	2024
No. examined	198	290	222	16	71	75	19	5	45	102	**91**
Rainbow Trout	190	288	216	14	68	70	16	4	44	101	**89**
Brown trout	8	2	6	2	3	5	3	1	1	1	**2**
Size range FL mm	90-513	148-460	150-500	170-290	150-371	156-445	155-451	175-485	150-274	155-560	**150-510**
Predation Rate	**1**	**1**	**0**	**0**	**1.4**	**2.7**	**10.5**	**0**	**0**	**0**	**0**

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

DISCUSSION AND CONCLUSIONS

Two-spined blackfish and Macquarie perch were absent from any of the trout stomachs examined in the field in 2024. This is despite the increased sampling effort (20 individuals or 1 km of river) has greatly increased the catch rate of trout. Baseline data also suggests that predation rates of Two-spined blackfish were low (<6 found in over 700 trout stomachs examined) in the three years of baseline sampling and predation of Macquarie perch was not detected at all (Lintermans *et al.* 2013). This is despite the potential of larger adult trout (i.e. those greater than 350 mm FL) to theoretically predate upon Macquarie perch up to 180 mm total length (Ebner *et al.* 2007). In 2019, detected predation rates of threatened species by trout had increased to 10.5 % (2 from 19) of trout containing Two-spined blackfish. In contrast to this there was no evidence of predation from 2020 onwards.

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

RECOMMENDATIONS

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

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

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

BACKGROUND

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

METHODS

Sampling design for Question 7 is covered by sampling outline for Question 1 (fyke netting) and is similar to the baseline monitoring program (Lintermans *et al.* 2013) (Table 16). The changes from the baseline monitoring program are the removal of an urban lake reference site (where these non-native species are present/abundant). This is the seventh year of monitoring following the commencement of filling and the fifth year of monitoring since the ECR filled.

Table 15. Outline of the sampling design for Question 7 of the fish monitoring program.

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

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

Abundance of Goldfish was standardised for each technique as fish caught per hour (represented as catch per unit effort or CPUE). CPUE was scaled in relation to increases in shoreline length as the reservoir filled and as net effort (see question 1 for scaling equation). CPUE of Goldfish captured in fyke nets was assessed between years using analysis of similarity (ANOSIM) with year as fixed factor. Data was Log10(x+1) transformed then a resemblance matrix was constructed with modified Gower (base 2) dissimilarity measure transformed to meet the assumptions of sphericity and homoscedascity of variances. Graphical presentations of site-level means with 95% confidence limits (with Bonferroni corrections applied) were then used to explore pairwise variations in Macquarie perch size classes among sites and years.

RESULTS

Goldfish

Over three nights of fyke netting in Cotter Reservoir in 2024, 39 Goldfish were captured ranging in length between 32 – 175 mm FL (Figure 29). The vast majority of these individuals were less than 100 mm FL, most likely corresponding to 0+ and 1+ year-old age class (Merrick and Schmida 1984)(Figure 29). Goldfish relative abundance was significantly different among years (Global R = 0.395, $p < 0.01$), with relative abundance in filling years of 2014 and 2015 and operational year 2016 significantly higher than in baseline years 2010, 2012 and 2013 and the most recent operating phase years (2017 – 2024) (Figure 30). Relative abundance of Goldfish from 2017 – 2024 was not significantly different from any of the baseline monitoring years.

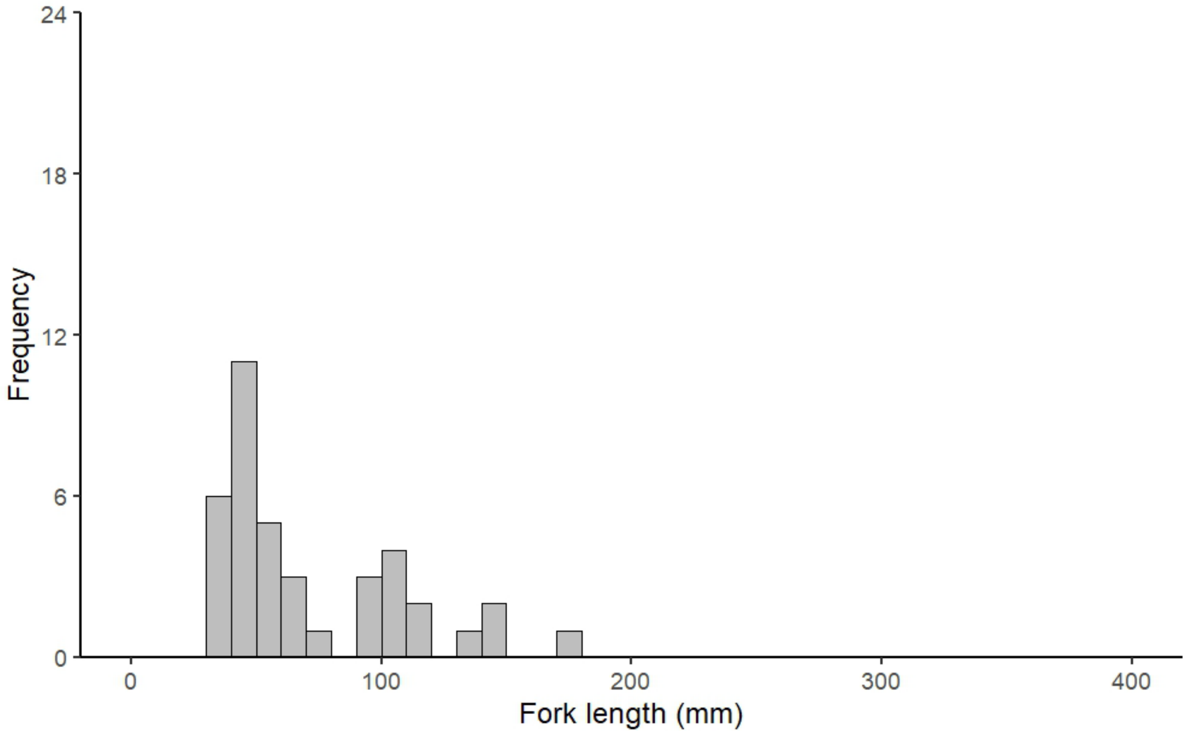


Figure 29. Length Frequency of Goldfish captured in the ECR in 2024 over three nights of fyke netting.

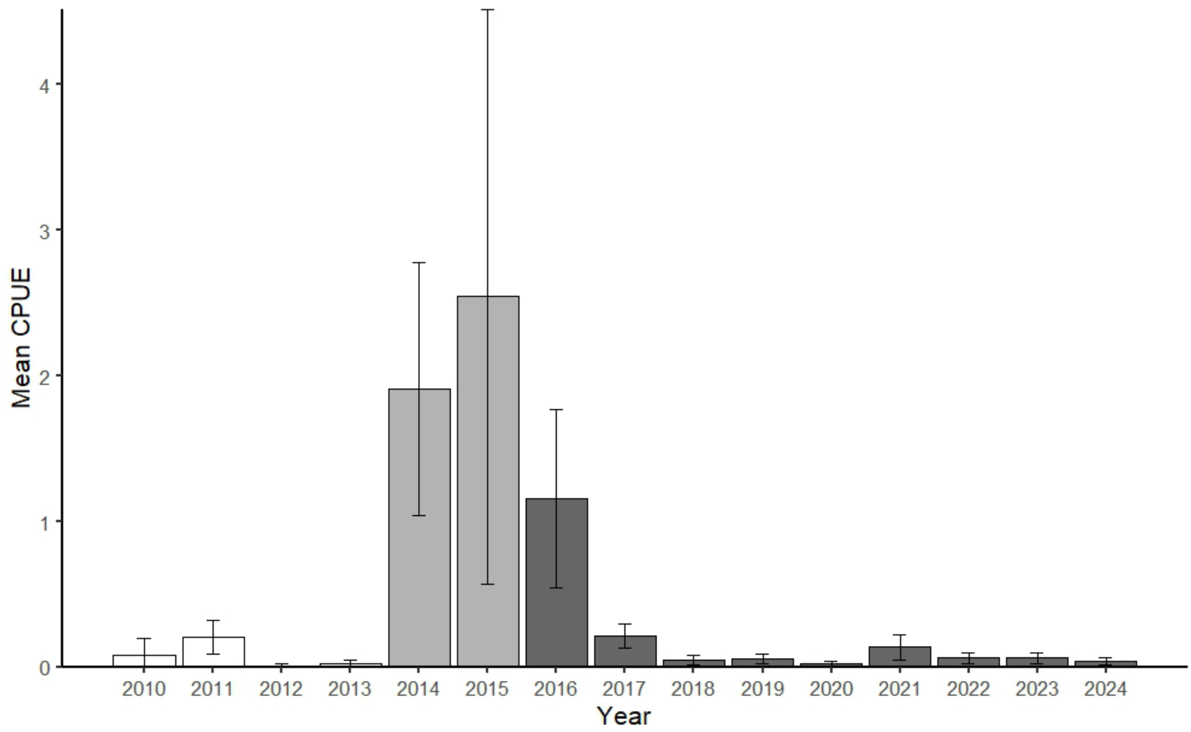


Figure 30. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of Goldfish captured in the ECR using fyke nets between 2010 and 2024. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey indicates operational phase of monitoring program.

Oriental weatherloach and Eastern gambusia

Both Oriental weatherloach and Eastern gambusia are rare catches 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). Seventy-three Oriental weatherloach have been observed whilst undertaking the boat electrofishing of Cotter Reservoir, including three observed in 2024. Three eastern gambusia were captured in 2024, all in bait traps. No eastern gambusia were observed whilst undertaking any other survey methods in 2024.

DISCUSSION AND CONCLUSIONS

Relative abundance of Goldfish has been relatively low since 2017, following on from very high abundances during filling and the first year of operational phase. The higher relative abundance of Goldfish in the initial stages of filling was likely to have been caused by an increase in availability of food resources associated with the filling phase (and refilling in the case of 2021) of the ECR (Kimmel and Groeger 1986, Ploskey 1986, O'Brien 1990). As predicted, the increase in Goldfish observed in 2014 – 2016 was likely a significant factor to the commencement of cormorant breeding observed in the top end of the reservoir in February 2014 and in every year since (see Question 8). Goldfish abundances in operational years of 2017 – 2024 were not significantly different to baseline abundances, indicating that the boom in resources associated with filling and early operational phases has ceased.

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

RECOMMENDATIONS

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

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

A significant increase in the abundance of Goldfish was detected in 2014 and 2015 and higher relative abundances persisted into 2016. Monitoring from 2017 – 2024 revealed a decrease in Goldfish abundances indicating that the resources boom associated with the filling reservoir has ceased or slowed significantly. The decline in Goldfish abundance and the first detections of Macquarie perch predation by trout warrant close attention being paid to this potential shift in food webs, but at this stage, no management intervention specifically related to Goldfish is recommended.

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

BACKGROUND

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

METHODS

Sampling design followed that exactly outlined in the baseline monitoring program (Lintermans *et al.* 2013) (Table 17).

Table 16. Outline of the proposed sampling design for Question 8 of the fish monitoring program.

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

Monthly visual surveys are undertaken of the entire ECR targeting piscivorous bird species including Great cormorant, Little black cormorant, Little pied cormorant, Pied cormorant, and Darter. The presence of nests of piscivorous birds was also noted, and if present the contents (eggs or chicks) noted (though this was not part of the monitoring program or analysis). Visual surveys were conducted from a boat using 10 x 40 mm binoculars. Location of each individual was recorded on a map. To determine distribution of piscivorous birds, the reservoir was divided longitudinally into five equal parts. Abundance and distribution can be assessed against trigger levels in the fish management plan: Appendix G (Icon Water Limited 2019). Comparison of abundance and distribution of each species of cormorants between the three phases (baseline, filling, operational) is undertaken using multivariate analysis (PERMANOVA) to explore overarching structure in the cormorant community. Unbalanced permutational analysis of variance (PERMANOVA) was conducted on cormorant abundances. Data was Log10(x+1) transformed then a resemblance matrix was constructed with modified Gower (base 2) dissimilarity measure. For PERMANOVA analysis, monitoring phase and section were fixed factors, with year nested within phase (Anderson *et al.* 2008). Highest interaction term removed for repeated measures design. Type III Sum of Squares used to account for unbalanced (years across phase) design and the three species of cormorant used as variables.

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 2023-2024 (Figure 31). 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 31). Since filling began, abundances of both Great cormorant and Little black cormorant have been stable, though with some definitive seasonal fluctuations (Figure 31). 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 31 and Figure 32).

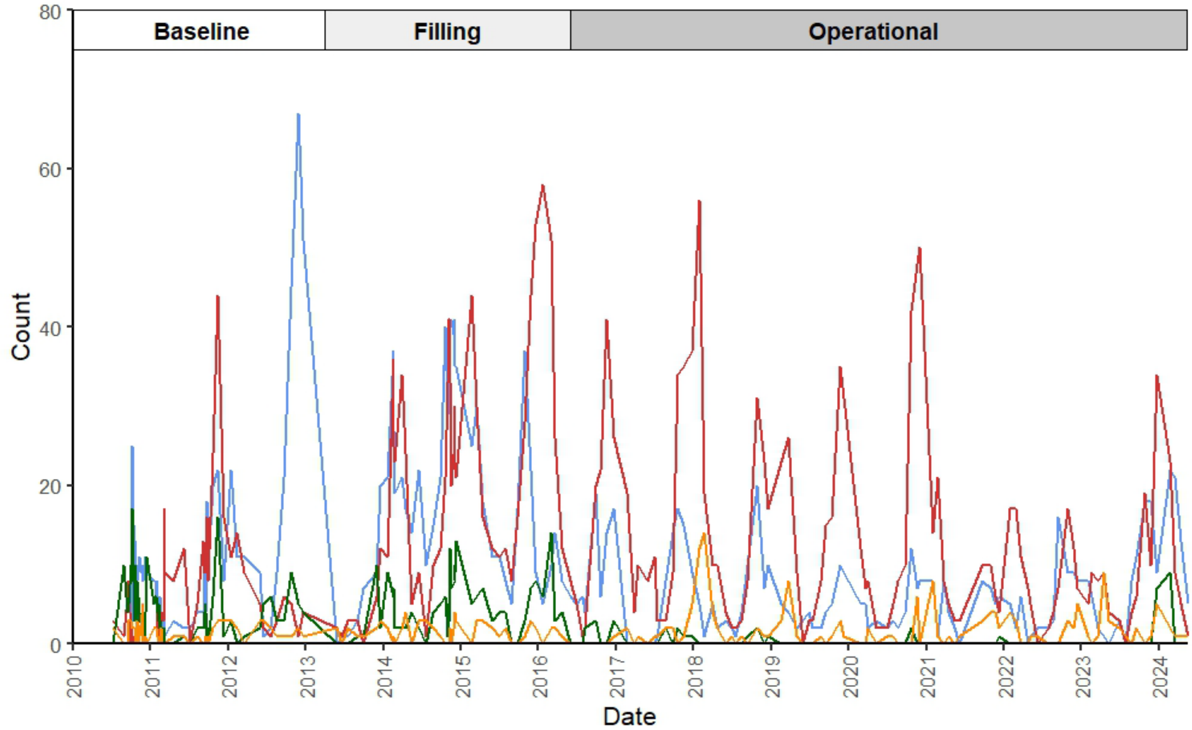


Figure 31. Monthly abundances of each common piscivorous bird species on the Cotter Reservoir between July 2010 and May 2024 (Great cormorant – blue line; Little black cormorant – dark green line; Little pied cormorant – red line; Darter – orange line).

There was considerable overlap in the composition of the piscivorous bird community (composition, abundance and distribution) among pre-filling and filling phases in Cotter Reservoir. There was a significant difference among phase, section, year and section by phase interaction in terms of piscivorous bird community composition (Table 18). Pairwise comparisons indicate these significant phase x section interaction differences are among baseline and filling versus operational phase for all sections ($p < 0.05$).

Table 17. Results of PERMANOVA analysis of piscivorous bird community composition in Cotter Reservoir from 2010 – 2024 (bold text indicates statistically significant difference at the P(perm) 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Phase	**2**	**25.71**	**12.855**	**16.471**	**0.0001**	**9941**
Section	**4**	**23.602**	**5.9004**	**24.857**	**0.0001**	**9937**
Year (within Phase)	**12**	**8.9772**	**0.7481**	**3.1515**	**0.0001**	**9890**
Section x Phase	**8**	**23.753**	**2.9692**	**12.508**	**0.0001**	**9908**
Residuals	948	225.03	0.23738			
Total	974	314.42				

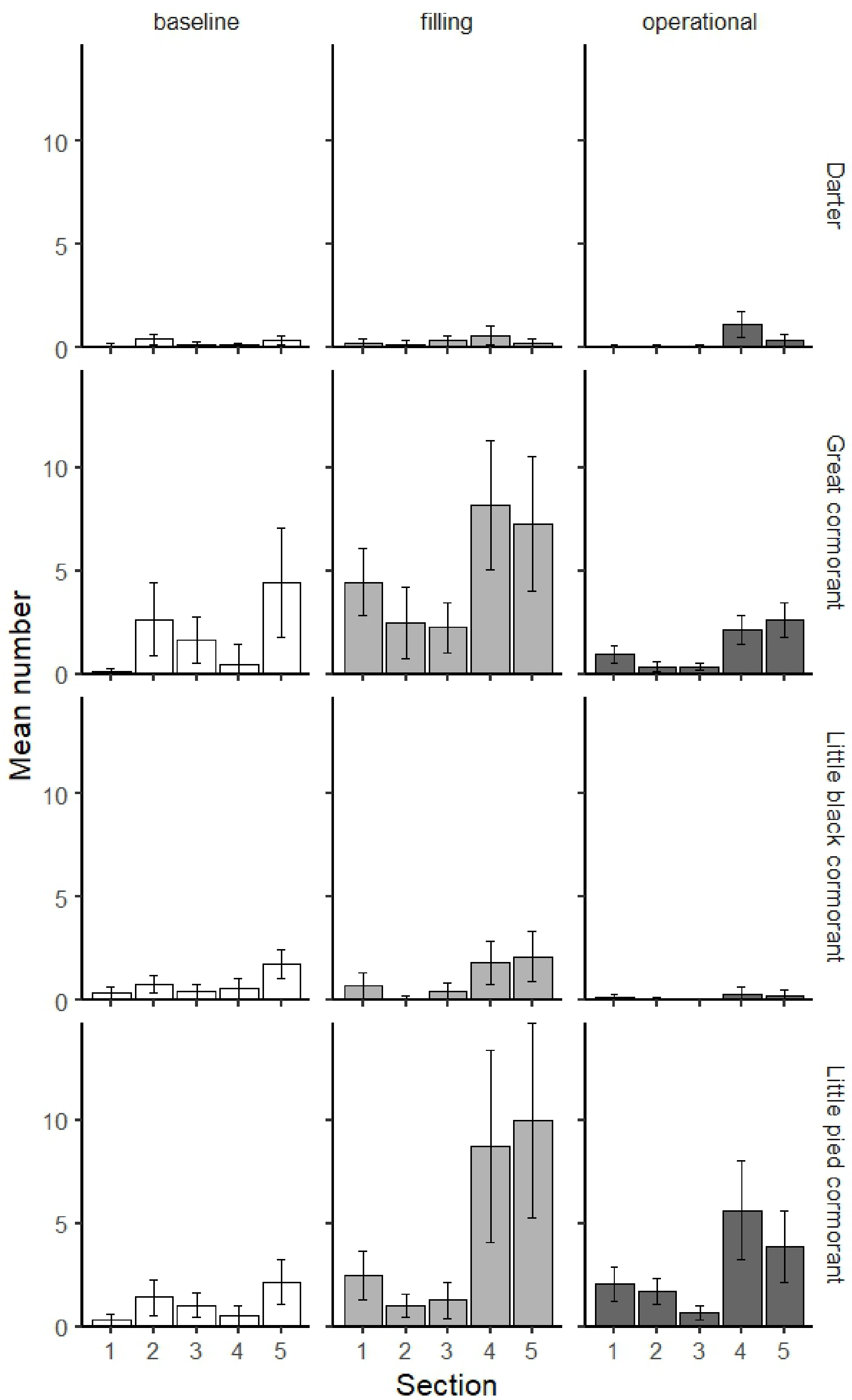


Figure 32. Mean monthly abundance (displayed as mean ± 95% confidence limits with Bonferroni correction) of each piscivorous bird species in each section of Cotter Reservoir for baseline phase (July 2010 – March 2013), filling phase (April 2013 – December 2015) and operational phase (January 2016 - May 2024) of monitoring program.

For the tenth 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 33), 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.

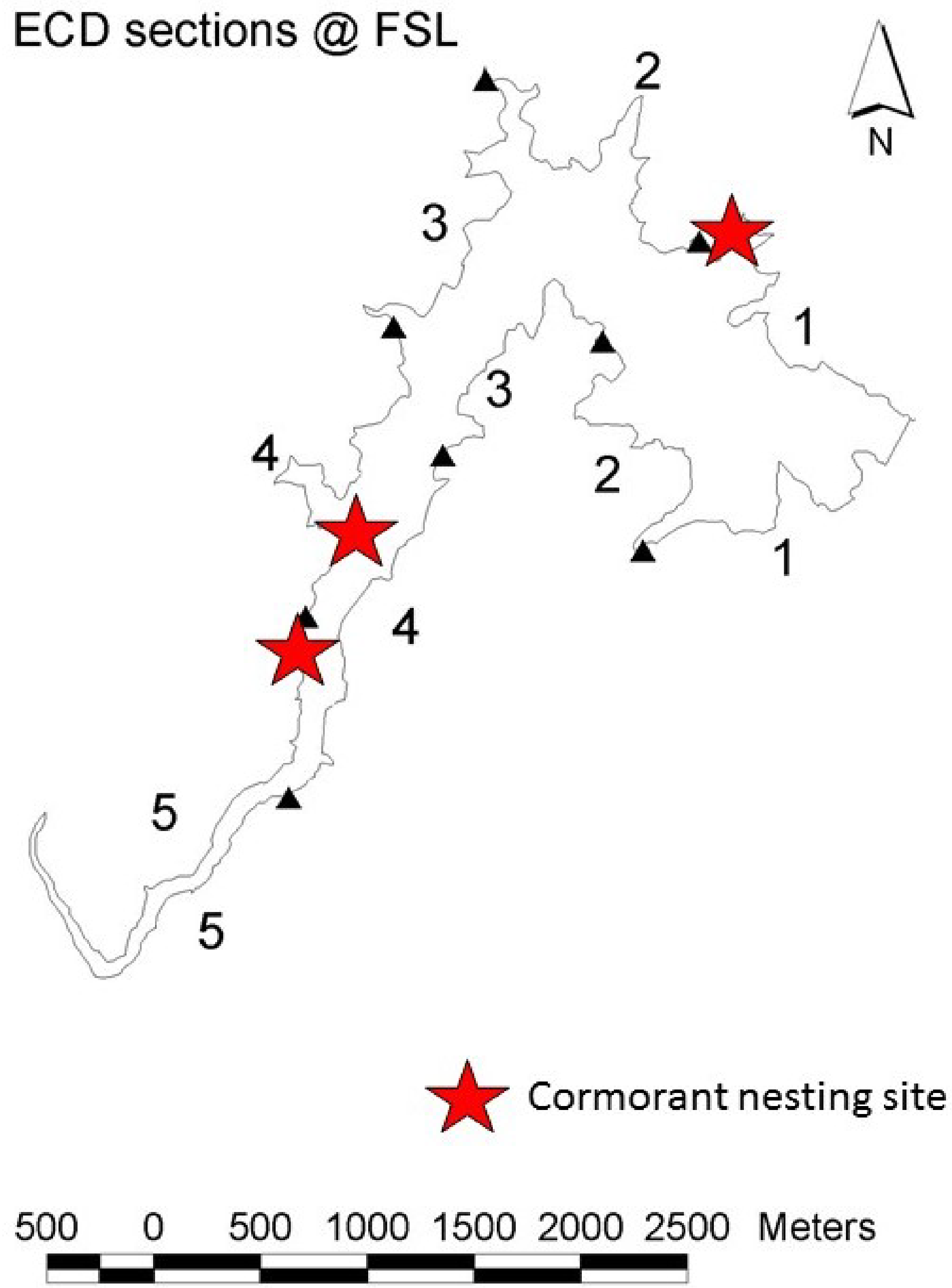


Figure 33. Map of the enlarged Cotter Reservoir shoreline sections with Cormorant nesting colony locations.

DISCUSSION AND CONCLUSIONS

Abundance

Peak abundances (see Figure 29) of Great cormorants occurred during the filling phase, though since the operational phase began have stabilised, though with some definite seasonal fluctuations. Abundances of Little pied cormorant during warmer months have increased since filling began, with these annual influxes concentrating in section 4 of the reservoir. The seasonal increases in all three common species is most likely attributable to an increase in productivity and food resources (decapods and Goldfish) within the reservoir during the warmer months. During filling and early operational phases the enlarged reservoir has seen an increase in the abundance of Goldfish (see Question 7 above), a favoured prey item of cormorants in the Cotter Reservoir and elsewhere (Miller 1979, Lintermans *et al.* 2011). The increase in prey abundance and the abundance of partially inundated larger trees (predominantly Eucalypts and pine trees) has provided suitable conditions for nesting to commence. Indeed, Little pied cormorant has bred in all years since filling began and its seasonal abundance peaks during the warmer months increasing until ~2018, though appear to be subsiding each year since ~2019. The establishment of a breeding colony of cormorants in the ECR is undesirable as the energy requirements of maintaining fledglings as well as adults would incur increased pressure on food resources (i.e. Goldfish and Macquarie perch) by cormorants in Cotter Reservoir (Lintermans *et al.* 2011). The potential early signs of a shift in predation pressure by trout (i.e. first records of predation of Macquarie perch) in 2018 potentially associated with declines in Goldfish abundance suggest that a re-examination of cormorant diet is required in the near future. This is especially so as the major avian predation threat to Macquarie perch (Great cormorant) were confirmed as breeding for the first time in 2022. If cormorants are also shifting their food preference from Goldfish to Macquarie perch then management of cormorant breeding colonies becomes critical.

Distribution

Distribution of all three common cormorant species has been relatively stable during baseline (2010 – 2013), filling and operational phases with a few exceptions. All three species have been most abundant in the two upstream sections of the reservoir. Previous research has found that cormorants commonly hunt in depths of less than 5 m (Dorfman and Kingsford 2001, Ropert-Coudert *et al.* 2006) and this depth range is most prevalent in these two reservoir sections and provides the greatest area for which effective hunting can be conducted (Ryan 2010, Ryan *et al.* 2013). The most upstream section is also where the greatest risk of predation is for Macquarie perch (Ryan 2010, Lintermans *et al.* 2011, Ryan *et al.* 2013). Interestingly a change in the distribution has occurred between baseline and filling and operational phases, where section 4 has seen an increase in abundance of both Great cormorant (during filling only) and Little pied cormorant (both filling and operational). This is most likely attributable to the location of a nesting colony in section 4 (Figure 33) and associated roost that was not present in baseline monitoring.

RECOMMENDATIONS

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

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

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

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

BACKGROUND

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

DOES COMPARABLE BASELINE DATA EXIST?

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

SAMPLING DESIGN

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

Table 18. Outline of the proposed sampling design for Question 9 of the fish monitoring program.

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

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

RESULTS

Three stands of emergent macrophytes, all of Cumbungi (*Typha domingensis*), were found along the shoreline of the Cotter Reservoir in March and May 2022 (Figure 27). The first, located on the western shore just downstream of Bracks Hole road, measured approximately 2 x 2 m (4m^2) in area (Figure 28). The other two were located next to each other near the old boat ramp road on the eastern shoreline of the reservoir (Figure 27). The largest of the two, measured approximately 9 x 4 m (36 m^2) and formed the densest of the three stands (Table 19, Figure 29). The third comprised two loosely grouped sparse stands of approximately 2 x 2 m (4 m^2) each (Table 19). An additional stand of Cumbingi was detected in May 2023, in the bay just south of Bracks Hole (Figure 27). This is a relatively small and sparsely vegetated stand, measuring approximately 2 x 4 m. All four stands increased in area and appeared thicker in composition, especially those located in the old boat ramp road bay (Table 19 and Figure 34).

Figure 34. Photograph of the largest Cumbungi stand located in the old boat ramp bay on the eastern shoreline of the ECR taken in May 2024 (photo: Ben Broadhurst).

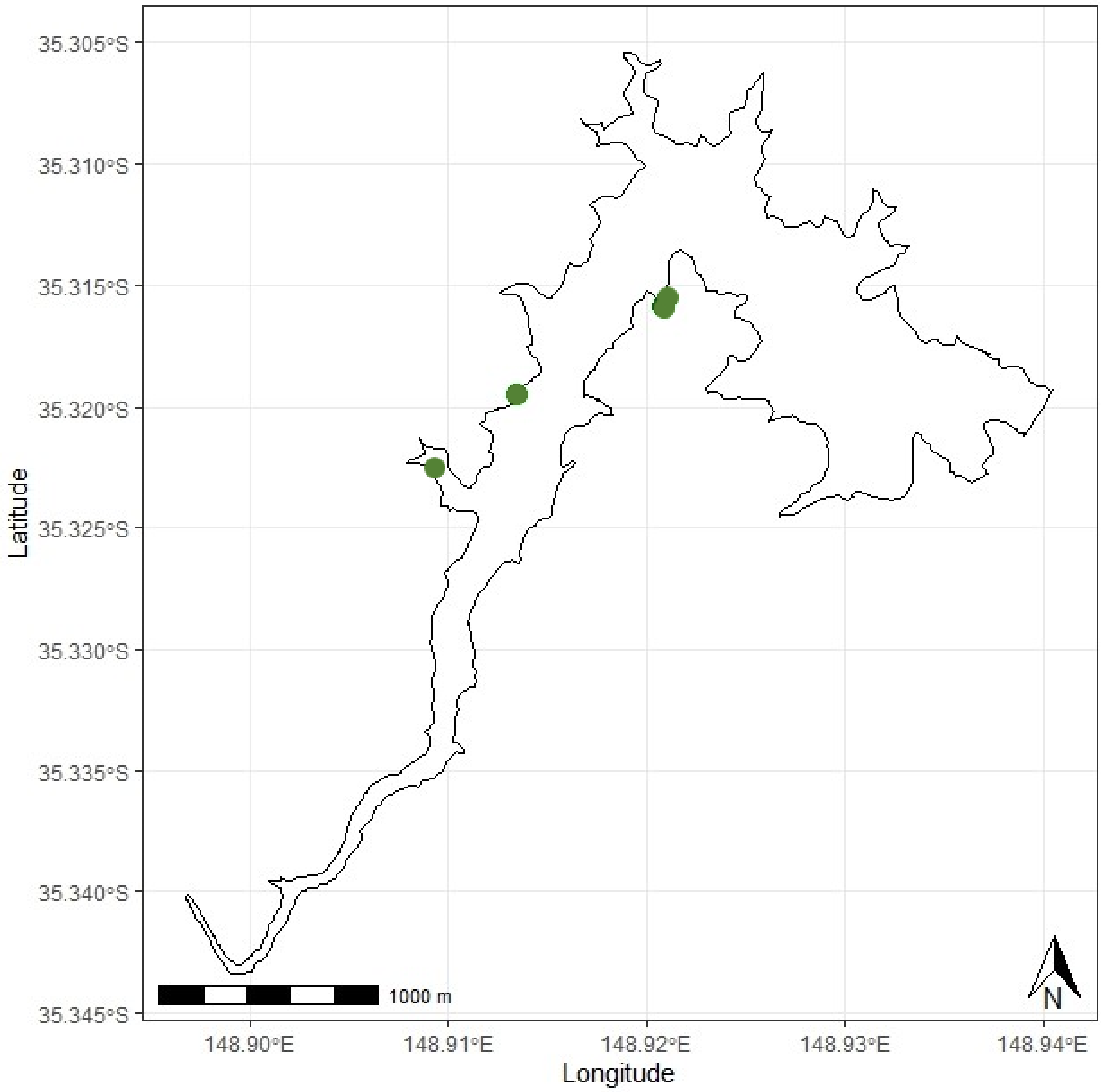


Figure 35. Map of the location of emergent macrophyte stands (green circles) along the shoreline of the Cotter Reservoir as monitored in March - May 2024.

Table 19. Details of each emergent macrophyte stand in the Cotter Reservoir as of May 2024 (including stand size change since last annual estimate).

Species	Location	Approx. size (m^2)	Annual increase
Cumbungi (*Typha domingensis*)	Western shoreline, shoreline near Bracks Hole	6 m^2	2 m^2
Cumbungi (*Typha domingensis*)	Eastern shoreline, inlet near old boat ramp road	50 m^2	14 m^2
Cumbungi (*Typha domingensis*)	Eastern shoreline, inlet near old boat ramp road	20 m^2	12 m^2
Cumbungi (*Typha domingensis*)	Western shoreline, in large bay south of Bracks Hole	10 m^2	2 m^2

DISCUSSION AND CONCLUSIONS

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

RECOMMENDATIONS

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

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

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

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

BACKGROUND

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

METHODS

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

Table 20. Outline of the sampling design for Question 10 of the fish monitoring program.

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

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

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

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

To analyse differences between the baseline samples of Norris *et al.* (2012) study and filling and operational edge samples, principal component analysis ordination and PERMANOVA analyses were conducted for each processing type (coarse pick and 10% subsample) as well as for the plankton tow samples. Principal Component analysis ordinations (PCO) of log10 transformed data were arranged into resemblance matrices using the Bray-Curtis Similarity measure. Vectors are the raw Pearson's correlations for the taxa that are most ($r > 0.4$) correlated with each of the PCO axes. Unbalanced permutational analysis of variance (PERMANOVA) was conducted on coarse pick data and subsample data separately. Data was log10 and a Bray-Curtis measure used for resemblance matrix. PERMANOVA analysis consisted of Phase, season and habitat as fixed factors. Highest interaction term was removed for repeated measures design. Type III Sum of Squares used to account for unbalanced (years across phase) design and the counts of each macroinvertebrate taxa used as variables. Decapod abundances derived from the course pick samples were analysed separately using ANOSIM with phase and season as factors.

RESULTS

Edge samples - Coarse pick

There was a significant difference in the coarse pick samples based on sampling phase, season and the phase x season interaction but no significant effect of habitat (Table 21). All phases were significantly different between season, except for between baseline and filling in spring, which was non-significant. PCO revealed that this difference between phases of coarse pick samples was largely driven by lower abundances of Coleoptera, Hemiptera and Diptera and higher abundances of Chironomids and terrestrial items in filling and operational phases compared to baseline monitoring (Figure 36). Decapod abundances were affected by season (Global R = 0.171, $p < 0.001$), with abundances being significantly higher in autumn compared to spring. Decapod abundances were not affected by phase (Global R = -0.063, p = 0.995). Decapod abundances in autumn have generally increased since 2018, apart from low abundances in autumn 2023 (Figure 37).

Table 21. Results of PERMANOVA analysis of coarse pick macroinvertebrate community composition in Cotter Reservoir from 2010 – 2024 (bold text indicates statistically significant difference at the 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Phase	**2**	**29954**	**14977**	**11.646**	**0.0001**	**9936**
Season	**1**	**7797.6**	**7797.6**	**6.0633**	**0.0002**	**9940**
Habitat	2	953.1	476.55	0.37057	0.9493	9947
Phase x Season	**2**	**5926.1**	**2963.1**	**2.3041**	**0.0154**	**9921**
Phase x Habitat	4	3223.4	805.86	0.62663	0.8891	9918
Season x Habitat	2	1671.5	835.77	0.64989	0.7684	9922
Residuals	76	97737	1286			
Total	89	1.4956E+05				

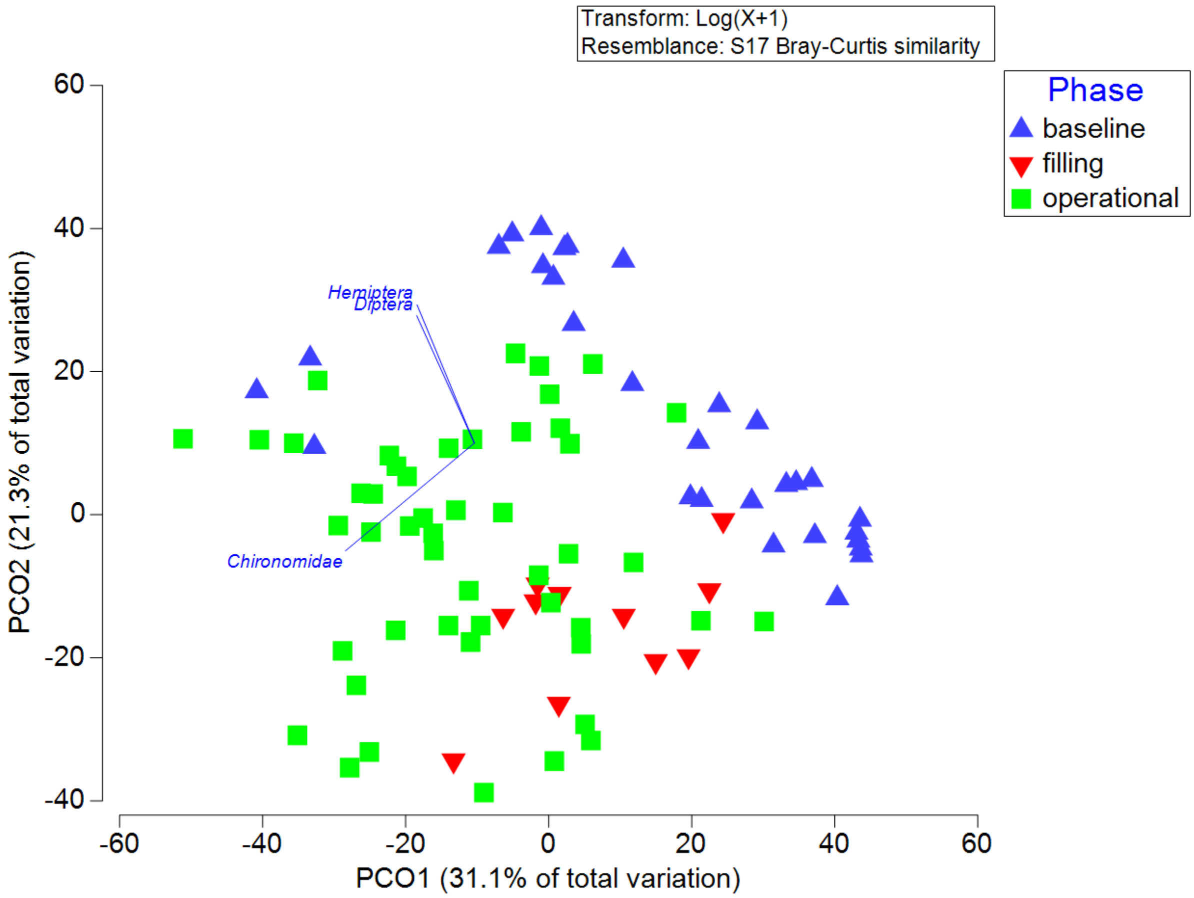


Figure 36. Graphical representation of a principal component analysis ordination of invertebrates from the coarse pick from spring and autumn monitoring in pre-filling (baseline) (data from Norris et al. 2012), filling phase (2013–2015) and operational phase (2016–2024).

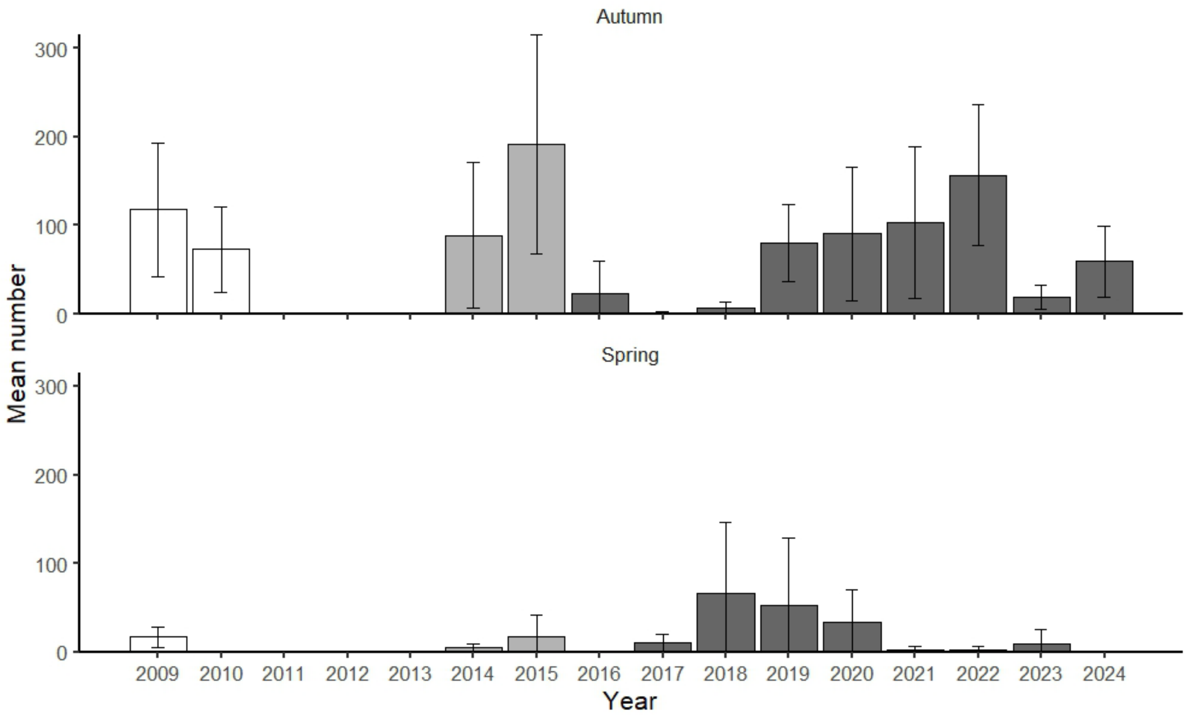


Figure 37. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of decapods collected from coarse pick edge samples taken from ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2024; dark grey bars) monitoring periods for autumn and spring. Note: Spring 2024 has not been sampled at the time of reporting.

Edge samples - 10% sub-sample

As for the coarse pick, there was a significant difference in the 10% sub samples based on sampling phase and season, and a significant phase x season interaction (Table 22). All phases were significantly different between seasons except for between operational and filling in autumn, which was non-significant. All phases were significantly different from each other in the composition of the 10% subsamples taken from edge habitat. The difference between baseline and filling and operational phase 10% sub-samples in spring was largely driven by the higher abundances of Diptera and in the baseline samples (especially in 2010), and higher abundances of terrestrial items during filling and operational phases (Figure 38).

Table 22. Results of PERMANOVA analysis of 10% subsample macroinvertebrate community composition in Cotter Reservoir from 2009-2010 (baseline) & 2013 – 2015 (Filling) and 2016 – 2024 (Operational) (bold text indicates statistically significant difference at the 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Phase	**2**	**28249**	**14125**	**14.403**	**0.0001**	**9925**
Season	**1**	**13517**	**13517**	**13.784**	**0.0001**	**9948**
Habitat	2	1837.1	918.55	0.93664	0.5098	9909
Phase x Season	**2**	**16734**	**8366.8**	**8.5316**	**0.0001**	**9920**
Phase x Habitat	4	4142	1035.5	1.0559	0.388	9898
Season x Habitat	2	784.33	392.17	0.39989	0.9625	9918
Residuals	74	72571	980.68			
Total	87	1.3591E+05				

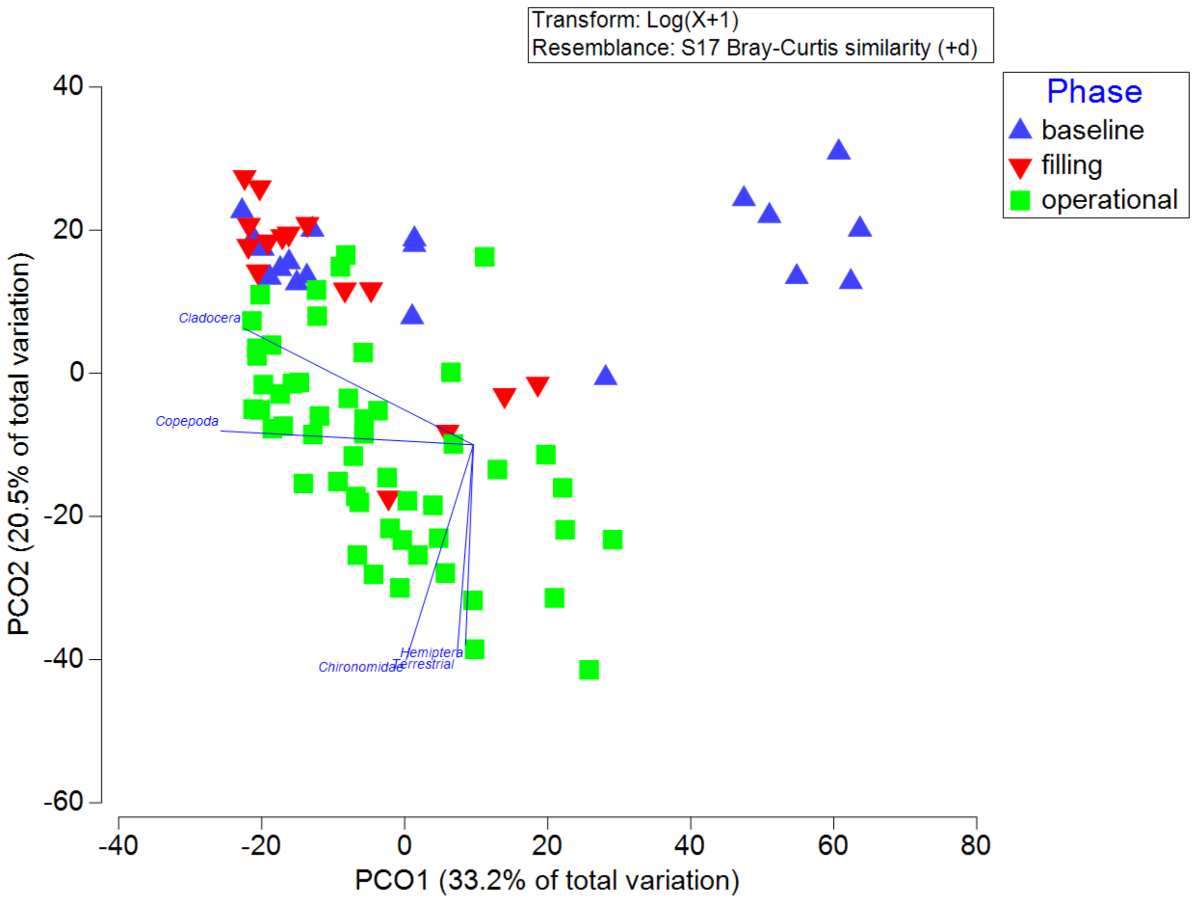


Figure 38. Graphical representation of a principal component analysis ordination of invertebrates from the 10% sub-sample processing from spring and autumn monitoring in baseline (data from Norris et al. 2012), filling phase (2013–2015) and operational phase (2016–2024).

Tow samples

Microcrustaceans dominated the plankton tow samples from the baseline and the filling and operational phase monitoring, comprising 99.9%, 100% and 100% of samples, respectively. Although there are some interannual variations in the microcrustacean community, there was no significant effect of phase or season (Table 23 & Figure 39).

Table 23. Results of PERMANOVA analysis of tow sample microcrustacean community composition in Cotter Reservoir from 2009-2010 (baseline) & 2013 – 2015 (Filling) and 2016 – 2024 (Operational) (bold text indicates statistically significant difference at the 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Phase	2	294.63	147.31	0.85992	0.4015	9964
Season	1	177.51	177.51	1.0362	0.2792	9940
Phase x Season	2	302.87	151.43	0.88396	0.3896	9937
Residuals	57	9764.8	171.31			
Total	62	10326				

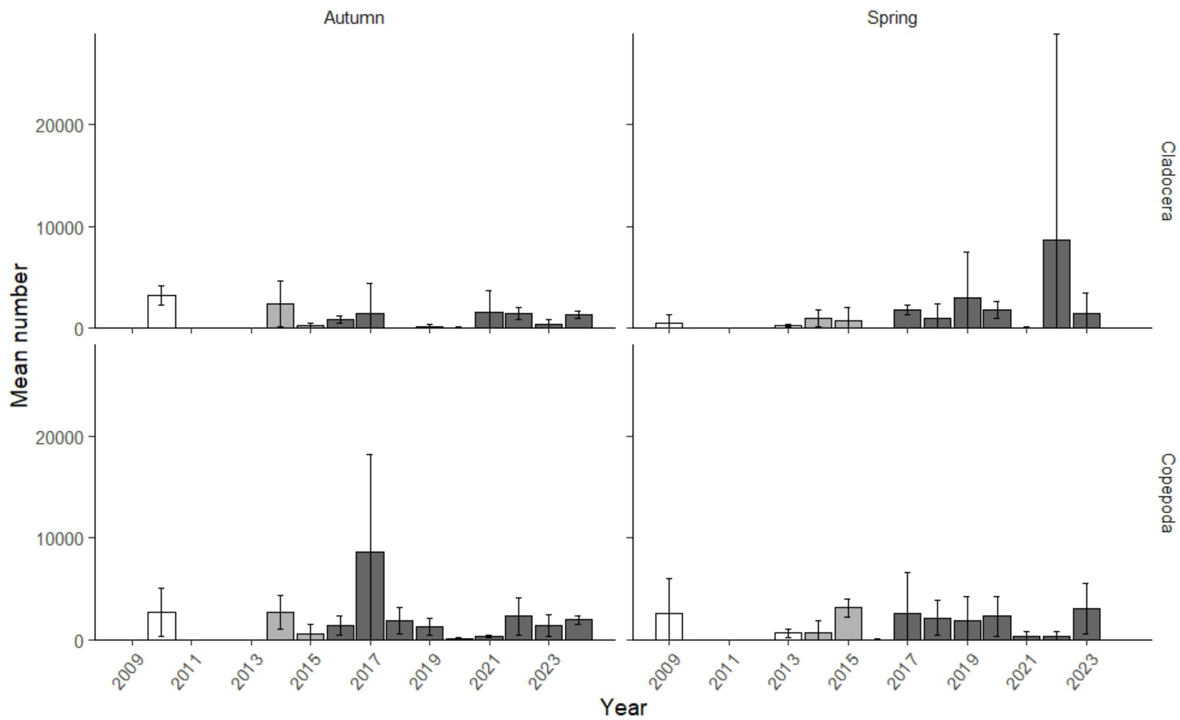


Figure 39. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of each microcrustacean taxa collected in autumn and spring of each phase of monitoring phase in ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2024; dark grey bars) monitoring periods. Note: Spring 2024 had not been collected at time of reporting.

DISCUSSION AND CONCLUSIONS

Decapod abundances in coarse pick edge samples are similar between phases. Decapod abundance in autumn during early operational phase monitoring was lower than that observed for autumn in baseline, filling and latter operational phase monitoring. Decapod abundance was much lower in spring than autumn. Despite being in very low abundances in early years of the operational phase, decapod abundance has been generally increasing since spring 2018 (see Figure 37). Decapods haven been previously found to be an important food item of adult Macquarie perch and may be an important antecedent factor in spawning success as previous studies have found Macquarie perch fecundity to be positively related to body condition (Gray *et al.* 2000, Lintermans 2006, Norris *et al.* 2012, Hatton 2016). Decapod abundances have stabilised in the latter years of operational phase, and it appears as though the operation of Cotter Reservoir is largely having minimal impact since 2018 on this important food resource of Macquarie perch.

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

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

RECOMMENDATIONS

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

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

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