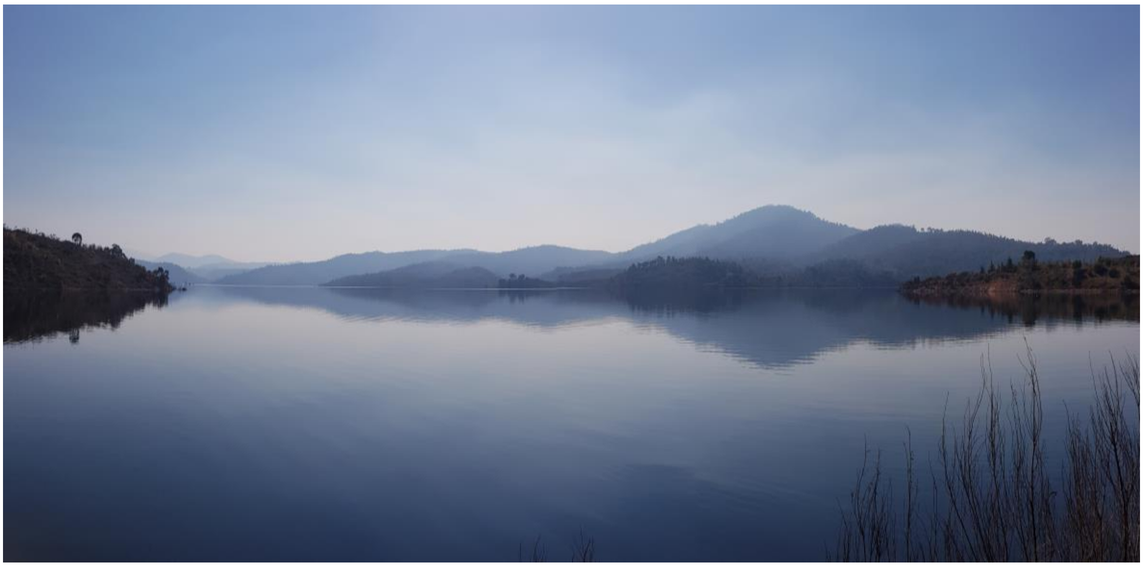
Enlarged Cotter Reservoir Ecological Monitoring Program

Technical Report 2020



Report to Icon Water

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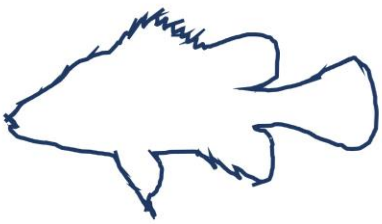

Enlarged Cotter Reservoir Ecological Monitoring Program: Technical Report 2020

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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 74 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 bispinosus* 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. A monitoring program commenced in 2010 with baseline monitoring (pre-filling) completed in 2013.

Since 2013, this ecological monitoring program centres on 10 management questions that aim to determine the impact of the filling and operation of the ECR on populations of the two focal species and potential threats (predators and competitors) in the ECR and river upstream. The post-2013 monitoring encompasses the ECR filling phase, since filling commenced in April 2013. This report addresses the 10 management questions by comparing baseline (2010 – 2013, (see Lintermans *et al.* 2013), and filling (2014 and 2015) (Broadhurst *et al.* 2014, Broadhurst *et al.* 2015) and operational (2016 – 2020)(Broadhurst *et al.* 2016b, Broadhurst *et al.* 2017, Broadhurst *et al.* 2018, 2019) monitoring data (where possible).

The 10 management questions that underpin the Enlarged Cotter Reservoir Ecological Monitoring Program since 2013 are:

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?
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 the 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 the 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?

The monitoring year of 2019 / 2020 was relatively dry and river flows below Bendora were largely regulated. There were several small peaks in flow associated with rainfall events, most notably during March 2020. The Enlarged Cotter Reservoir (ECR) water level was in steady decline from February 2018 and is now approximately 9 m below full supply level (FSL). 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 ben in decline to its lowest level in 2018, whilst adult 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. Body condition of adults was higher during filling and early operational phases, compared to baseline, though indications from 2019 and 2020 are that are that body condition is returning to a similar level to that seen in baseline monitoring. Encouragingly, successful recruitment to young-of-year stage was detected for the fourth consecutive year in 2020, which is a positive result given that this population had not successfully recruited during 2014, 2015 and 2016. Indeed, Macquarie perch recruitment was detected at all five riverine sites in 2020 (as was the case in 2019), indicating that conditions were suitable across the catchment for spawning and early development of young-of-year. Although the ECR was drawn down slightly during the spawning season of 2019 (approximately 6.55 – 4.47 m below FSL), it is still not clear whether the reservoir population of Macquarie perch will continue spawning when the reservoir is repeatedly or substantially drawn down during operation, particularly below the most upstream large barrier located at around 540 m ASL.

Abundance and distribution of Macquarie perch in the Cotter River upstream of the ECR remains relatively stable since monitoring began in 2010. Abundance of young-of-year Macquarie perch was different among years, and these differences were mixed between years of baseline and filling phases suggesting that recruitment of Macquarie perch in the Cotter River is somewhat sporadic and likely reflects the behaviour of the small resident riverine population.

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 six 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 Cotter Reservoir and Cotter River in 2020 was not significantly different to any other year of monitoring. Relative abundance of Brown trout captured in Cotter Reservoir has remained high in the past five years (an average of 15.4 each year over 2016 to 2020), which is around five times higher than annual captures in any other year (including three caught during the baseline period in 2010). This species had a dramatic reduction in abundance in the catchment during the millennium drought. Brown trout are more piscivorous than Rainbow trout and a change in the species composition of trout in Cotter Reservoir could lead to changes in predation upon Macquarie perch and Two-spined blackfish in the catchment.

Although increased backpack electrofishing effort resulted in a dramatic increase in the number of trout captured in Cotter River in 2020 was very low compared to previous years and predation by trout upon native fish was not detected. Due to the timing of sampling (autumn), the current examination process has no capacity to detect predation of larval Macquarie perch (ideally this would be undertaken in early summer when larvae are present). Some caution around the lack of predation of Macquarie perch by trout should be exercised, difficulty in visually detecting larval Macquarie perch may lead to a false negative predation detection. One Macquarie perch (145 mm TL) was found in the stomach of a Brown trout (484 mm FL) captured in Cotter Reservoir, which continues the chain of consecutive years where predation of Macquarie perch has been detect by brown trout in the Cotter Reservoir, after never being detected in the diet previously.

Small-bodied alien species other than trout continue to be detected in the ECR, with Goldfish accounting for the vast majority of captures. Goldfish abundance had increased since filling commenced, most likely in response to increased availability of food resources. However, Goldfish abundance has been decreasing over the past four annual assessments such that captures since 2018 are now similar to baseline. The decrease in Goldfish abundances likely reflect a slowing of the productivity of the newly filled reservoir. Although Goldfish probably pose little direct threat to Macquarie perch and Two-spined blackfish, there is potential for wider effects from drops 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. It is interesting to note that that the decline in goldfish abundance and the increase in Brown trout abundance in the ECR since 2017 coincides with the first records of trout predation on Macquarie perch. This may represent the first signs of prey switching by trout or increased predation risk to Macquarie perch, as was predicted.

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.

Monitoring macrophyte bed re-established in the ECR has not yet formally commenced as the reservoir was filling and no macrophytes have been observed whilst conducting other fieldwork around the perimeter of the reservoir. Macrophytes may establish now that the reservoir has filled, although this I likely influenced by the frequency and level of drawdown and subsequent inundation.

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 operational phase monitoring, which is of concern as this is an important dietary item of Macquarie perch in the ECR. Monitoring in spring 2019 and autumn

2020 suggest that decapod abundances are returning to baseline. Microcrustaceans revealed 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

No change to the 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 79.4 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 bispinosus* 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 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 a fish monitoring program commenced (ACTEW Corporation 2009b).

The broad scope of the potential impacts of the ECR are summarised below.

The main threats to the Macquarie perch population in the Cotter Reservoir as a result of the ECR are related to:

- loss of adult shelter habitat (fringing emergent reedbeds)
- alteration to primary food resources associated with fringing reedbeds,
- increased predation from cormorants and trout,
- loss of riverine spawning habitat through inundation of existing habitat and restricted access to alternative habitat,
- impacts associated with competition, predation and disease transmission from existing alien fish species, and
- invasion by two additional alien fish species (Redfin perch *Perca fluviatilis* and Carp *Cyprinus carpio*).

The anticipated trophic upsurge that occurred in the newly filled ECR was considered to likely result in enhanced populations of trout within the ECR, whose impacts then spill over into the river as trout move into the river to spawn (Lintermans 2012, Todd *et al.* 2017). Threats to the riverine Macquarie perch and Two-spined blackfish populations between the ECR and Bendora dam are:

- increased predation from trout
- loss of riverine spawning habitat through inundation of existing habitat and restricted access to alternative habitat, and
- invasion upstream by two additional alien fish species (Redfin perch and Carp) should these species establish in the reservoir (Lintermans 2012).

As well as enhancing trout populations, the trophic upsurge in the ECR was considered likely to benefit Macquarie perch in the reservoir, both in terms of individual fish condition and population size, potentially providing a window of opportunity for the establishment of additional populations of this species outside the lower Cotter catchment (Lintermans 2013b, Todd and Lintermans 2015). As the reservoir has filled and has entered the operational phase, the window of trophic upsurge is now considered to have largely closed.

Consequently, information on the condition and size of alien and native fish populations (including a range of life history phases), cormorants, and habitat conditions in the ECR and the river upstream is an essential requirement to adaptively manage the aquatic resources of the lower Cotter catchment.

The baseline phase of the ECR monitoring program (2010-2013) has been completed (Lintermans *et al.* 2013), which along with other available datasets provides a pre-filling baseline of threatened and alien fish species abundance and occurrence both in the impoundment and the river upstream. The filling phase of the monitoring program was conducted between 2013 – 2015 and the operational phase monitoring commenced in 2016. The underlying sampling design and priority knowledge gaps for the filling and operational phases of the monitoring program were revised 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?

The integrated monitoring program has field activities often addressing multiple questions (Table 1).

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.

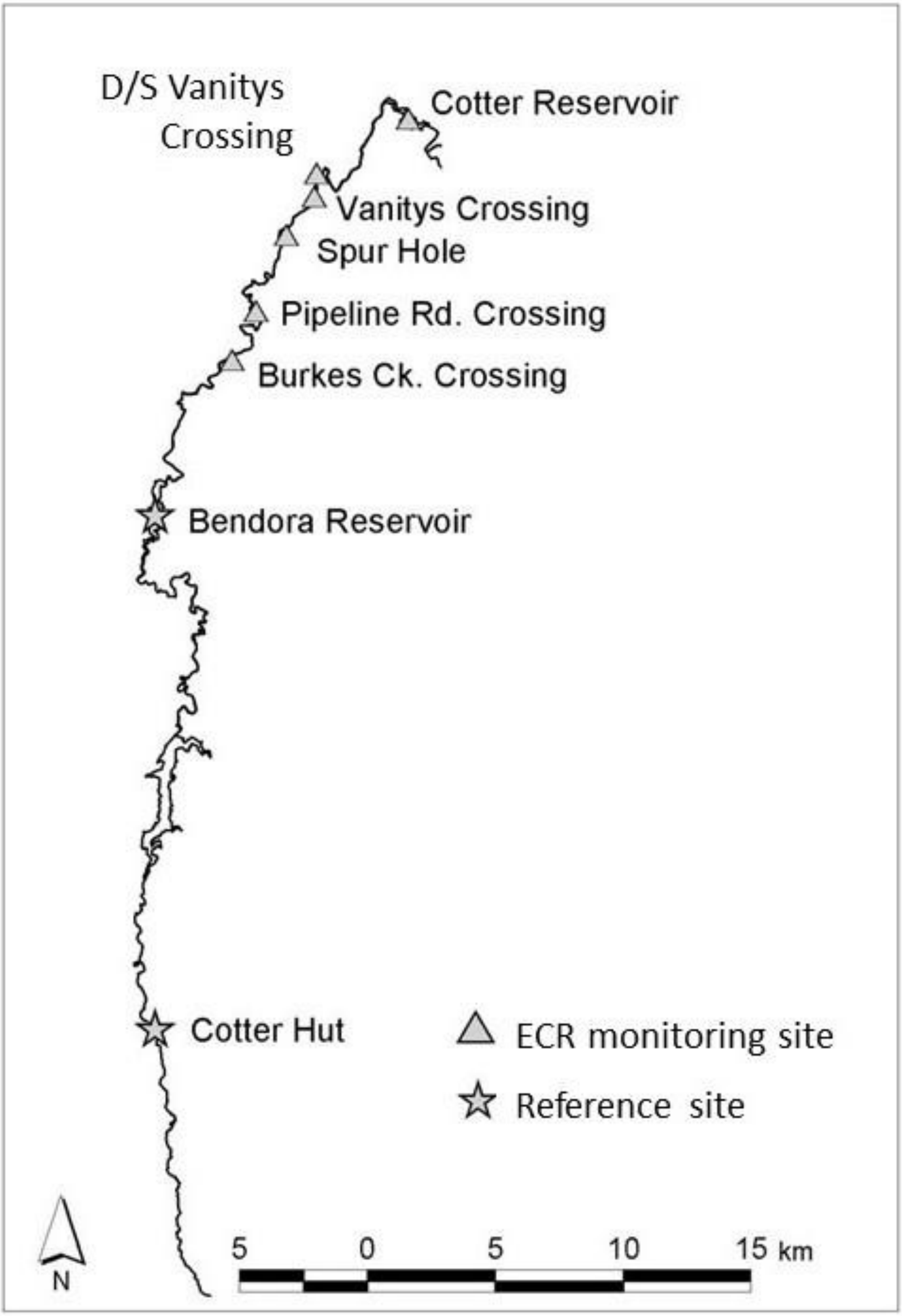

Figure 1. Location of sampling sites on the Cotter River. Note: Map does not include the reference site on the Murrumbidgee River (Kissops flat).

The filling and operational monitoring program effectively utilised the methods and sites from the baseline monitoring program with the following changes:

- Some reference sites (Lake Ginninderra, Micalong Creek and Corin Reservoir) were excluded as a cost-saving measure, as requested by Icon Water.
- The data collected for Questions 1 and 2 has been expanded to include weight, body depth and body width of adult Macquarie perch captured (for body condition estimates).
- Snorkelling of the river immediately upstream of the ECR to Vanitys Crossing has been added to determine the recruitment input from the reservoir population of the ECR (as opposed to potential supplementation from riverine reaches further upstream)
- Boat electrofishing is being trialled to determine its effectiveness in capturing adult Macquarie perch compared to gill netting
- The site immediately downstream of Bendora Dam has been removed from the program due to budget restrictions.
- For questions 2, 5 and 6, the site at Bracks Hole (immediately upstream of the old Cotter Reservoir full supply level) has been replaced by another riverine site immediately upstream of the full supply level of the ECR (but downstream of Vanitys Crossing). This replacement was necessary as Bracks Hole has already been inundated and no longer represents a riverine site. A riverine monitoring site immediately upstream of the full supply level of the ECR is required as this is the most likely area where impacts of the operation of the ECR will be greatest.
- Bait traps have been added to the sampling techniques for Question 3 and Question 7. The addition of bait traps will increase the likelihood of capture of juvenile Two-spined blackfish (detection of recruitment) and also increase the likelihood of capture of small and juvenile non-native species associated with Question 7.
- The intensity of sampling effort for characterisation of trout diet associated with Question 6 has changed, as has the way in which the stomach contents of trout are processed, as requested by Icon Water.
- The Phase 2 monitoring program has proposed methods for assessing the establishment of macrophytes that were not covered in the baseline monitoring program.
- Additional fyke netting and gill netting effort in Cotter Reservoir.
- Additional boat electrofishing at night to be undertaken in Cotter Reservoir (2019 onwards)

The rationale and results from each of the 10 management questions are presented in the following sections.

HYDROLOGICAL SUMMARY

The Cotter River experienced prolonged drought through the late 1990-2000s (van Dijk *et al.* 2013), with the phenomenon worsening from 2006 until the latter half of 2010 when significant rains resulted in flooding (Figure 2). This flooding caused the original Cotter Reservoir to rise by 2 – 4 m from mid-October 2010 to Mid-April 2011 (Figure 3). A significant single rainfall event and associated large-scale flooding also occurred in early 2012, which led to water levels in the under-construction Enlarged Cotter Reservoir prematurely increasing by about 10 – 12 m in February 2012 (Figure 3). In terms of effects on monitoring results, 2010/11 fish monitoring reflects the previous year that was dry and the ending of an extended extreme drought. Monitoring in 2011/12 and 2012/13, and to some extent 2013/14, were years where the preceding year had an average of high rainfall and discharge, when compared to recent history. Monitoring in 2014/15 and again in 2015/2016 follows relatively dry conditions as it appeared the area was moving towards another period of lower than average rainfall. Monitoring in 2016/2017 followed a wetter than average winter and spring where all three reservoirs on the Cotter River filled in winter 2016, and remained full throughout the Macquarie perch spawning season (September – December) resulting in the Cotter River between Bendora Dam and the Enlarged Cotter Reservoir operating as largely unregulated (Figure 2). The monitoring year of 2019 / 2020 saw a return to flows which were dominated by regulated flow releases (because of dry climatic conditions), except for a few short, high flow pulses associated with rainfall events (notably early March 2020) (Figure 2). Because of these low inflows and abstraction for water supply, the Enlarged Cotter Reservoir level has receded over the past year to its current level of approximately 9 m below full supply level, as of May 2020 (Figure 3).

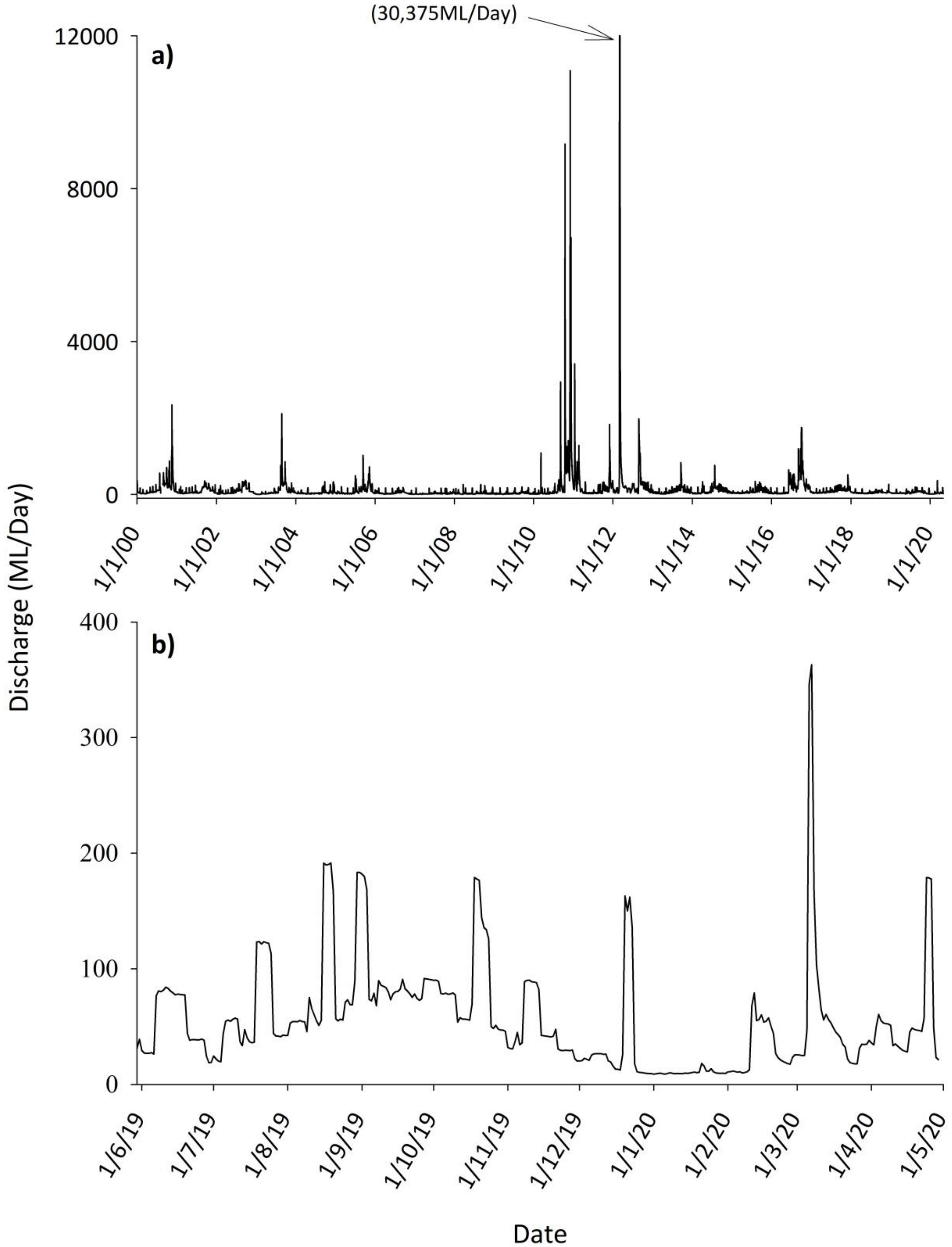

Figure 2. Daily discharge of the Cotter River at Vanitys Crossing from a) January 2000 until May 2020 and b) May 2019 – May 2020.

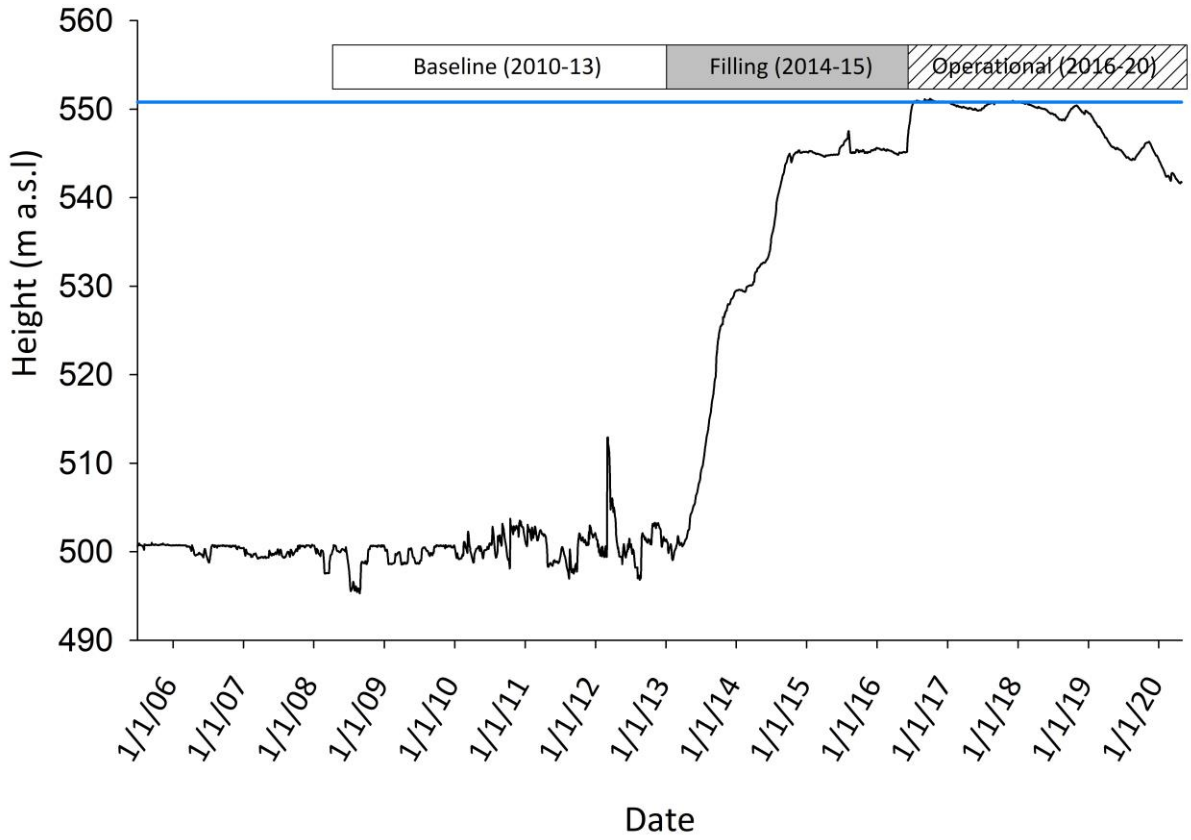

Figure 3. Water level (in metres above sea level) of Cotter Reservoir from May 2005 until May 2020 (blue lines indicates full supply level of the Enlarged Cotter Reservoir). Rectangles indicate the three monitoring phases: white = Baseline, grey = Filling, hatched = Operational. Blue line indicates enlarged Cotter Reservoir full supply level. Full supply level prior to enlargement was 500.5 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 influencing 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 Macquarie perch may be impacted by new barriers in a filling reservoir (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. Previous monitoring conducted during elevated reservoir levels indicated that sampling for adult Macquarie perch in a filling reservoir may be difficult using gill nets and that a complementary technique (boat electrofishing) required exploration (Lintermans *et al.* 2013).

METHODS

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 sampling method to mitigate potential sampling inefficiencies via gill netting for adult Macquarie perch during ECR filling (see below for details). 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 (100 - 150 mm TL) and young-of-year (< 100 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 was conducted annually in March - April; snorkelling in Nov/Dec.
Number / location of sites	One impacted site: enlarged Cotter Reservoir (and river immediately upstream); one reference site: Kissops Flat (upper Murrumbidgee River). No snorkelling 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 – 99 mm TL based on results of the baseline data collection (Lintermans *et al.* 2013). Individuals were considered juvenile if they fell between 100 – 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 and April 2020. 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 2019 and again in 2020, an additional two 125 mm gill nets (1 x deep-drop) 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 and April 2020. Twelve fyke nets were set for two nights in the pool at Kissops Flat in March 2020. Fyke nets were set for ~16-hour soak time (existing fyke netting protocol) commencing at ~15:30-16:00 hrs.

The baseline monitoring report suggested that an alternative sampling technique for adult Macquarie perch was required during the filling phase as gill nets failed to capture any adult Macquarie perch during high water levels in 2012, most likely due to gill nets being set on partially-submerged vegetation further from the shoreline than usual (Lintermans *et al.* 2013). 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. To attempt to increase captures of large adult Macquarie perch (as per recommendations from Broadhurst *et al.* 2018), a trial of night-time boat electrofishing commenced in 2019. 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 enumerated), although broad estimates of abundance were made for individuals < 100 mm TL (young-of-year) and Juveniles (100 > 150 mm TL) for each shot.

A snorkelling visual survey of pools along the Cotter River immediately upstream of the ECR was not undertaken in 2019 because of staff illness.

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 habitat 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 in 2017 and 2018, 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 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 33 Macquarie perch were captured using gill nets in ECR in 2020, which ranged from 149 – 403 mm TL, although the majority (>90%) of individuals were 220 - 400 mm TL (Figure 4). Adult Macquarie perch have been captured by gill nets in every year since monitoring began in 2010. 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, including 2020 (Figure 5). Macquarie perch CPUE was significantly different among years (Global R = 0.009, $p < 0.01$), 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 = 0.181, p = 0.109) However, there appears to have been a general decline in adult abundance since 2015, with 2018 and 2019 CPUE being significantly lower than the preceding 3 years (Figure 5).

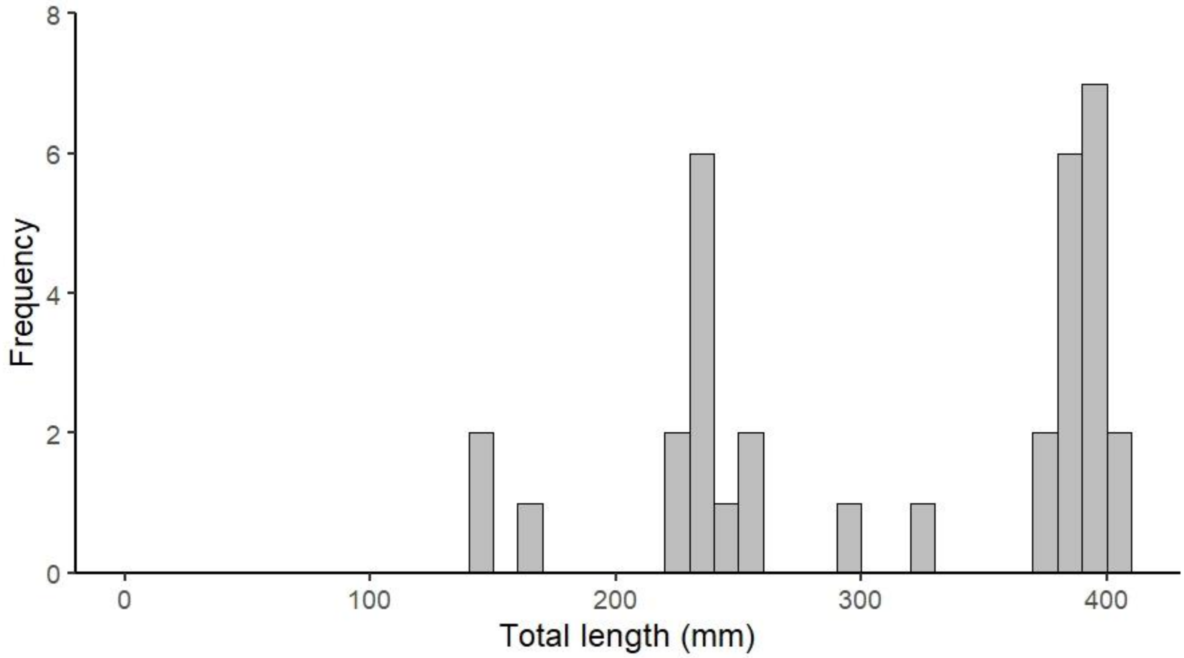

Figure 4. Length frequency of Macquarie perch captured from the Enlarged Cotter Reservoir in autumn 2020 using gill nets.

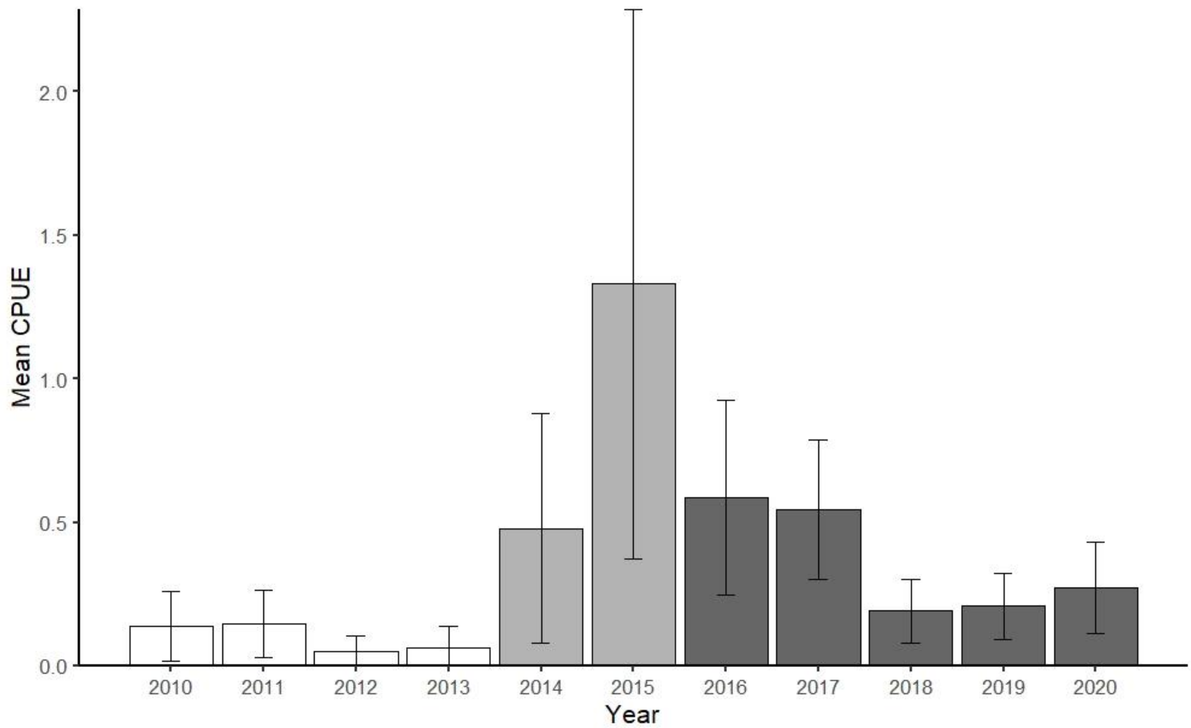

Figure 5. Relative abundance (displayed as mean CPUE ± 95% Confidence limits with Bonferroni correction, and scaled to relative reservoir shoreline length at time of sampling) of adult Macquarie perch captured in Cotter Reservoir using gill nets between 2010 – 2020. White bars indicate baseline phase, light grey bars indicate filling phase and dark grey bars indicates the operational phase of the Enlarged Cotter Reservoir (ECR); bars are arranged in chronology from 2010 to 2020 from left to right on the x-axis.

Length of adult Macquarie perch 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_6 = 106.032, $P < 0.01$) (Figure 6). The most recent year (2020) was the anomaly amongst the operational years as it was the only significantly year different to 2010 and 2014. The reason for the more similar lengths between 2020 and baseline and filling phase years is likely to be the return of the 200 – 300 mm TL size class (albeit in relatively small abundances), which was in low abundance during the other operational years (Table 3, Figure 4 and Figure 6). Adult Macquarie perch condition was significantly higher during the filling phase (2014 – 2015), compared to operational (2016– 2020) and baseline (2007 – 2009) phases (H_2= 65.842, $P < 0.01$). Mean Fulton's condition index has been declining since 2017, though there is no significant difference between the last two operational years (2019 & 2020) and any other year.

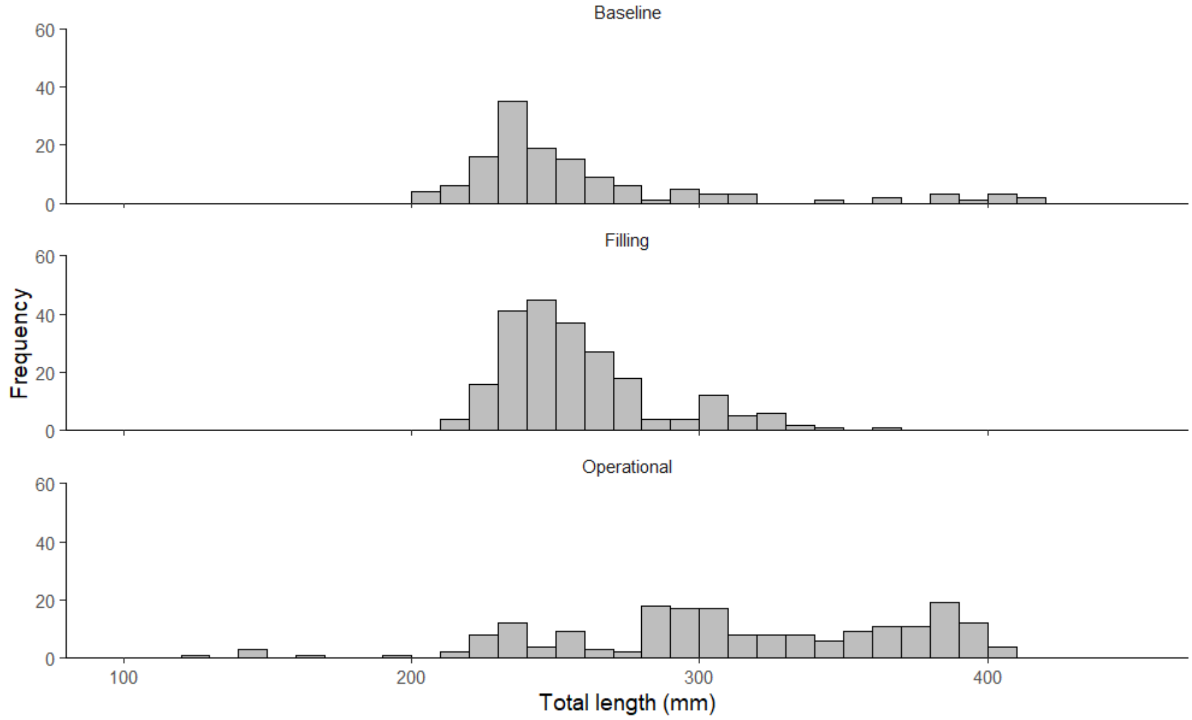

Figure 6. Length frequency of all Macquarie perch captured in gill nets in Cotter Reservoir for each monitoring phase from 2010 to 2020.

Table 3. Raw numbers of Macquarie perch in each size class captured in Cotter Reservoir using gill nets each year over the period 2010 – 2020. Monitoring phases are indicated by shading: none = Baseline; blue = Filling; green = operational.

Size classes (TL)	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
200 > 299 mm	39	41	13	18	66	120	23	10	8	5	12
300 > 349 mm	2	6	1	2	5	26	33	21	1	2	1
350 >369 mm	1	0	0	0	0	1	1	11	0	0	2
>370 mm	4	3	4	0	0	1	2	14	10	14	15
All sizes total	**46**	**50**	**18**	**20**	**71**	**149**	**59**	**56**	**20**	**21**	**33**
% of 200-300mm	*85*	*82*	*72*	*90*	*93*	*81*	*39*	*18*	*40*	*24*	*36*

A total of five Macquarie perch were captured by daytime boat electrofishing in the ECR in 2020, ranging from 378 – 396 mm TL. In contrast to what occurred in 2014 – 2016, the majority of Macquarie perch captured via electrofishing in 2017 – 2020 did not come from submerged constructed rock reefs, they came from elsewhere around the perimeter of the reservoir. This matches with captures made by gill netting where individuals were captured from a range of shoreline habitat types. Relative abundance of Macquarie perch captured between years was similar across all survey periods, though was relatively low in 2020 (Figure 7).

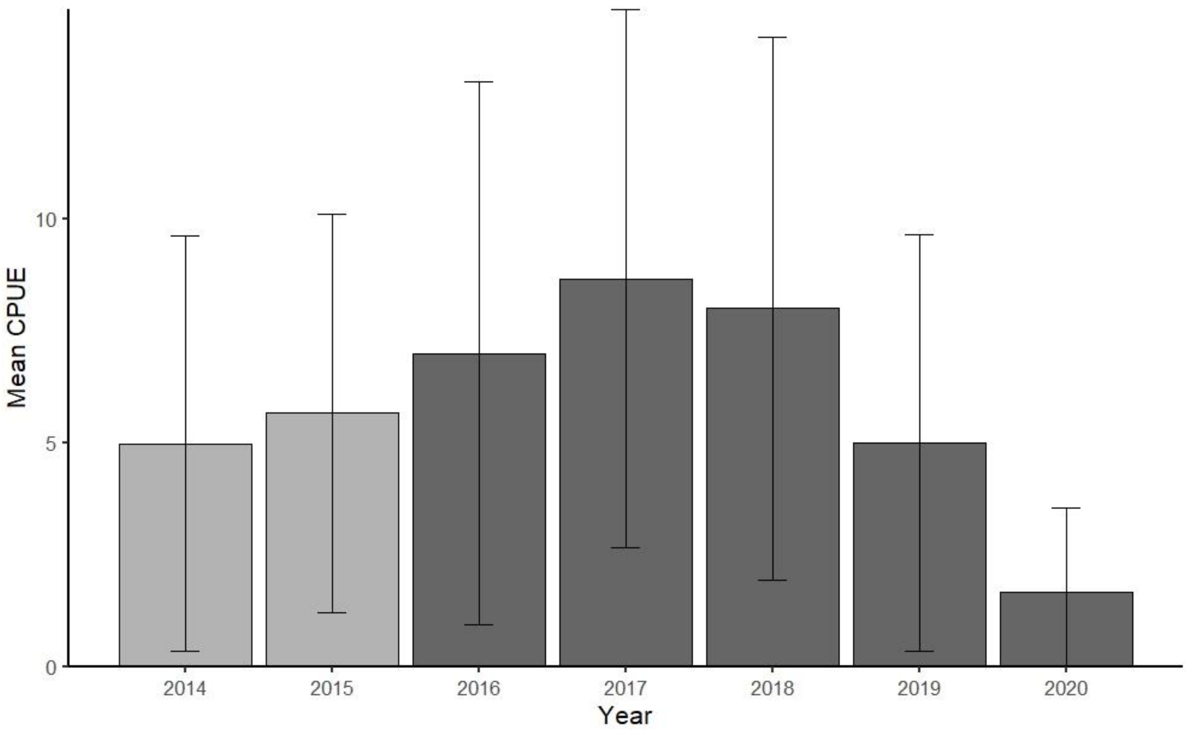

Figure 7. Relative abundance (displayed as mean CPUE ± 95% Confidence limits with Bonferroni correction) of Macquarie perch captured in Cotter Reservoir using boat electrofishing between 2014 and 2019. Light grey bars indicate filling phase, dark grey bars indicate operational phase.

Fyke netting - all age classes

A total of 295 Macquarie perch ranging from 56 – 393 mm TL were captured in 2020 from the ECR using fyke nets (Figure 8). 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 4). The significant site by year interaction is largely driven by the lack of young-of-year at Cotter Reservoir during 2014 and 2015 (Figure 9).

Table 4. 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 2020 (bold text indicates significant effects at the P(perm) 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Site	**1**	**0.5891**	**0.5891**	**20.054**	**0.0001**	**9948**
Phase	**2**	**0.76043**	**0.38021**	**5.1459**	**0.0141**	**8674**
Year(Phase)	**8**	**0.64714**	**0.080892**	**2.7536**	**0.0008**	**9909**
Site x Phase	**2**	**0.1775**	**0.088752**	**3.0212**	**0.0172**	**9943**
Residuals	678	19.917	0.029376			
Total	691	22.072				

Juvenile Macquarie perch

In 2020 a total of 234 Macquarie perch juveniles / sub adults (> 100 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 no significant difference in the relative abundance of juvenile Macquarie perch between sites (Global R = -0.036, p = 1) though there is weak evidence to suggest an effect of year (Global R = 0.012, p = 0.053) (Figure 9).

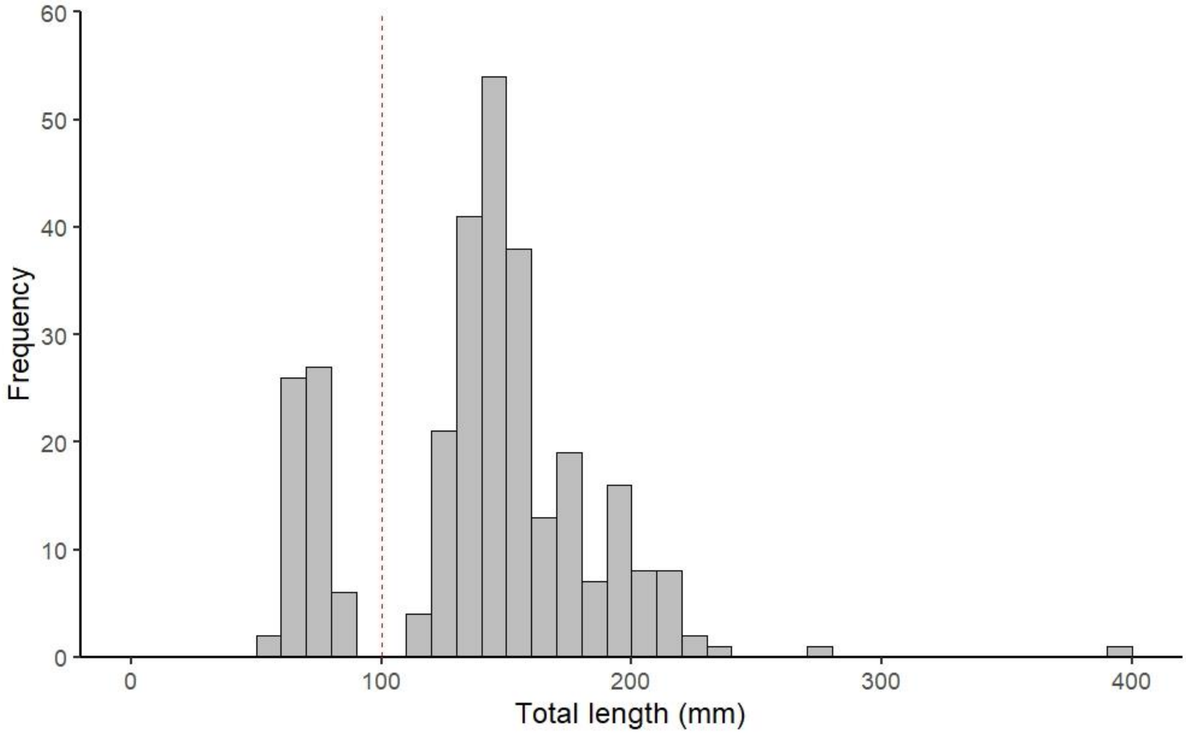

Figure 8. Length frequency of Macquarie perch captured from the ECR in autumn 2020 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

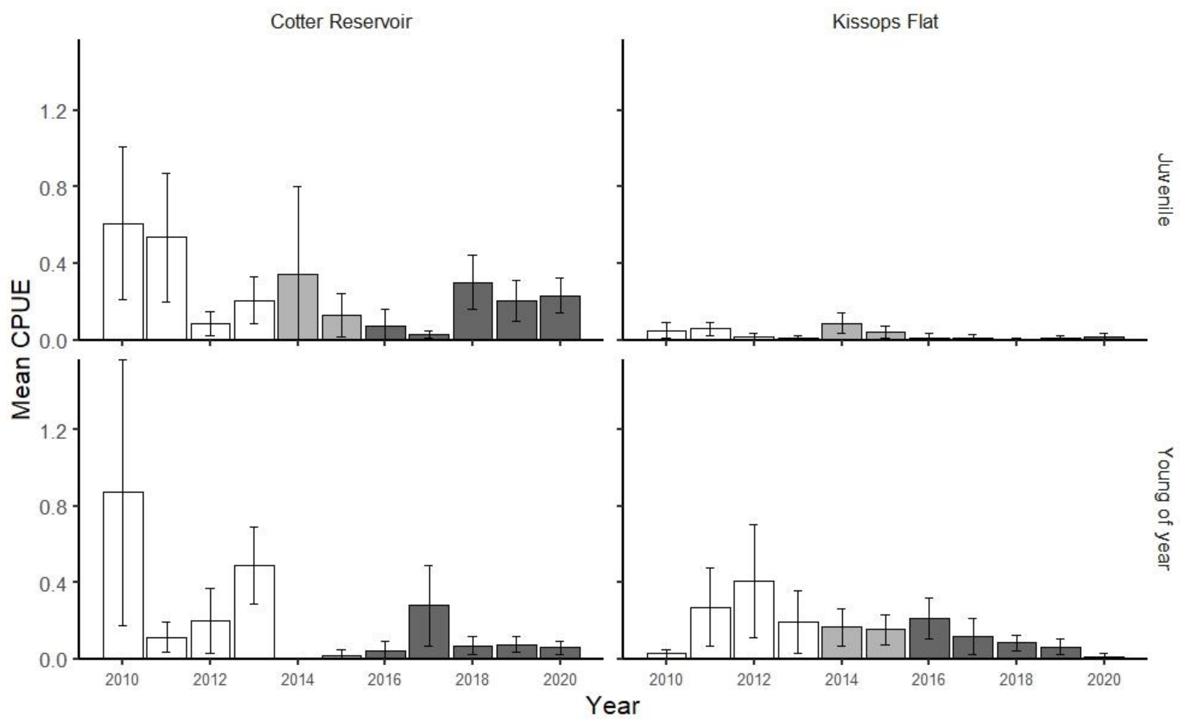

Figure 9. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections, scaled to relative net effort versus shoreline length at the time of sampling) of juvenile (>100 mm TL) and young-of-year (< 100 mm TL) Macquarie perch captured in Cotter Reservoir (impact site) and Kissops Flat (reference site) using fyke nets between 2010 and 2020. 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 chronology from 2010 to 2020 from left to right on the x-axis.

Young-of-year Macquarie perch

A total of 61 young-of-year (YOY) Macquarie perch were captured using fyke nets in the ECR in 2020. Operational years 2017 – 2020 show an improvement in CPUE of YOY compared to filling 2014 – 2015 and early operational year 2016 when extremely low abundances of YOY were captured (Figure 9). There was no significant difference in the CPUE of YOY Macquarie perch between sites (Global R = -0.002, p = 0.559), but there was a significant difference between years (Global R = 0.039, p = 0.0001). 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 2020 were significantly less than those of prefilling years 2010, 2012 and 2013 (which were exceptionally high) but not different to the prefilling year of 2011 (Figure 9). YOY Macquarie perch were detected in all years of monitoring at the reference site (Kissops Flat, upper Murrumbidgee River) using fyke nets, though abundances of this size class were recorded at their lowest during 2020 in comparison to all other years since monitoring began in 2010 (Figure 9 and Figure 10).

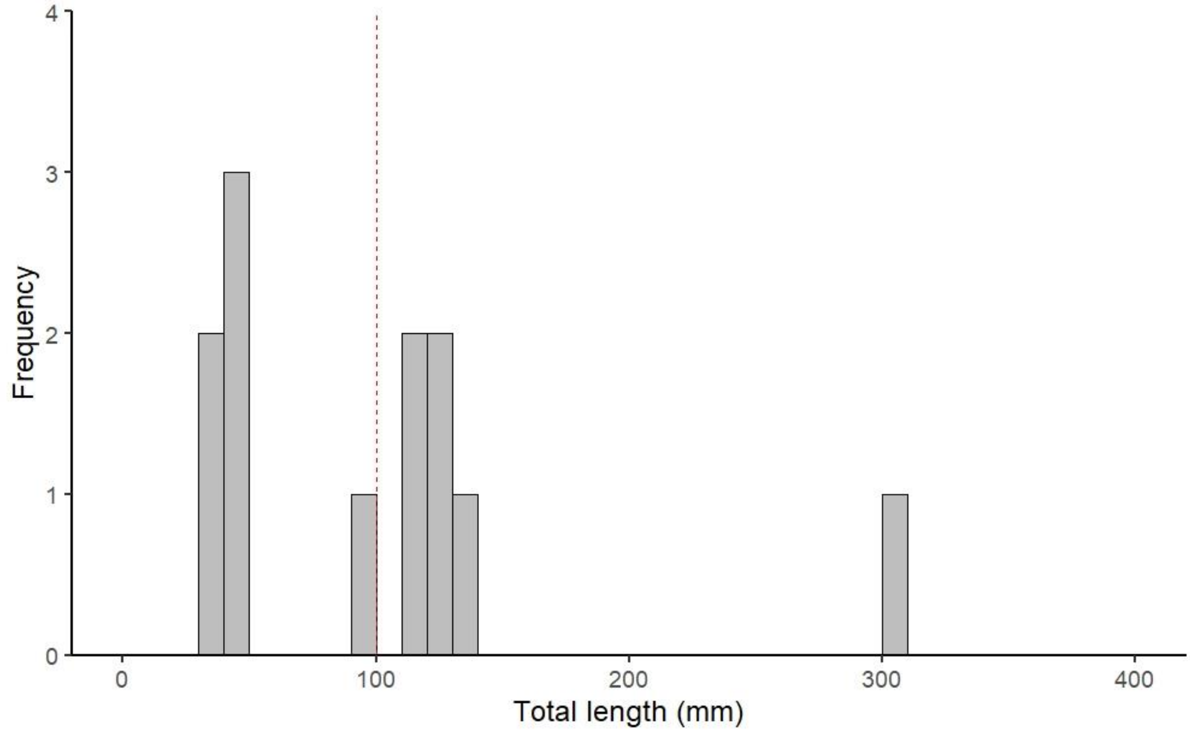

Figure 10. Length frequency of Macquarie perch captured from Kissops Flat on the upper Murrumbidgee River in autumn 2020 using fyke nets (red dashed line indicates cut-off for length of young-of-year individuals).

DISCUSSION AND CONCLUSIONS

Annual abundances of Macquarie perch size classes 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. Most recently, 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 – 2020 has indicated successful spawning and recruitment to young-of-year, 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 three years (2018 – 2020), suggesting good annual recruitment conditions through to 1 – 3-year old individuals. This is extremely positive for this population, especially seeing as though the reservoir was drawn down 4 – 6 m during the spawning season in 2019.

Concerns remain from the previous assessments that adult Macquarie perch CPUE (in gill nets) shows ongoing signs of decrease since 2015, with significantly lower captures in between 2018 and 2020 compared to filling and early operational years, albeit similar to that of baseline monitoring (which would be the bare minimum to be aimed for as this is an endangered species population). Although present levels of adult abundance appear to be sufficient to allow successful spawning and recruitment, any further declines or lack of recovery will warrant further investigation. Raw numbers captured in 2018 and 2019 were the lowest since monitoring began in 2010, although these appear to be increasing (ever so slightly) in 2020. As discussed in Broadhurst *et. al.* (2018) there could be a number of factors underpinning this decline in adult abundance since filling phase, but most likely these are:

1) Actual reduction in adult abundance (i.e. mortality); and/or
2) Recruitment shadow from lack of recruitment in 2014-2016
3) Reduction in capture efficiency related to either:
    a. Gear specifications, and/or
    b. Fish behaviour (spatial ecology).

To reduce uncertainty around these factors (and based on recommendations from Broadhurst *et al.* 2018), increased effort targeted at larger adult Macquarie perch was employed in 2019 and 2020, namely increased larger-mesh gill netting effort, and trials of night-time boat electrofishing (2019 only). The additional effort has achieved modest results, with a small number of adults captured via each additional effort method. The main difference between adult captures between 2018 / 2019 and 2020 is that the majority of adult Macquarie perch captured via gill netting in 2018 / 2019 were large in size, with very few captured in the 200 – 300 mm TL size range (aged 3 – 5 years old, Battaglene 1988, Douglas 2002) (Table 3). The gap in this size range (which may be the most prone to capture by the techniques employed in this monitoring program (Broadhurst *et al.* 2018) in the last 2 – 3 years is likely a gap from the very low recruitment cohorts from the failed 2013 – 2015 spawning seasons (where the filling reservoir had drowned out existing spawning grounds, and barriers to upstream movement had prevented access to new spawning grounds).

Numbers of the largest cohort of individuals (> 370 mm TL) captured in 2020 were the equal highest since monitoring began in 2010 (Table 3), indicating that this cohort from ~2012 is still present in relatively high abundance. Tonkin *et al.* (2018) found that the modal age of spawning Macquarie perch was 5 – 10 years old with a modal length of around 357 – 400 mm TL in Dartmouth Reservoir. Once fish reach approximately 10 years of age, there is relatively constant low-level mortality (Todd and Lintermans 2015) and fecundity increases with size and age and is predicted to plateau at around 15+ years of age (Todd and Lintermans 2015). Macquarie perch are known to live up to 30 years of age (Tonkin et al 2018). Our monitoring suggests that relative abundances of individuals in this large size range (>370 mm) are as high as they have been since monitoring began and that if the spawning stock size structure in the ECR is similar to Dartmouth, then the spawning stock of larger Macquarie perch in ECR is healthy. Monitoring in 2020 has seen a modest return of the 200 – 300 mm TL size class, which is an early indication that recruitment to the adult population may be starting once again.

The breaking of the 3-year recruitment drought occurred in the 2016 spawning season when the ECR reached full supply level, with the third highest relative abundance of young-of-year detected in 2017 fyke netting. These individuals are now 3+ years old and around 200 – 250 mm TL and as such are starting to enter the size class of individuals that gill netting targets. Indeed captures of this size range have increased in 2020, compared to the past two years, and we expect them to continue to increase as strong recruitment cohorts (currently captured in fyke nets) grow into the targeted size for gill nets. Following the current assessment of adult abundance, we do not have any evidence of further decline, and do not recommend any change to management or monitoring practices at this stage.

Adult Macquarie perch captured in both filling years displayed significantly higher body condition relative to individuals captured in baseline and operational phases. Condition of adult Macquarie perch in 2020 had decreased, but were not different to any other year (baseline, filling or operational). Body condition was not measured during baseline monitoring from 2010 – 2013. This higher body condition was expected as rising water levels of Cotter Reservoir have inundated banks and vegetation, resulting in a trophic upsurge from the increased amount of organic matter available to drive productivity up through the food chain (Kimmel and Groeger 1986, Ploskey 1986, O'Brien 1990, Lintermans 2012, Hatton 2016). Increased submerged habitat area, due to inundation of terrestrial environment, has introduced another food source in the form of displaced terrestrial invertebrates (Hatton 2016). Prior to this time Cotter Reservoir had relatively stable water levels and drought inflows that are likely to be associated with relatively low inputs of organic carbon and/or terrestrial dietary items (Blanchet *et al.* 2008, Winemiller *et al.* 2010). Our data continues to support predictions that Cotter Reservoir would undergo an upsurge in productivity and food resources used by adult Macquarie perch, resulting in increased body condition. This increased condition compared to baseline fish condition prior to 2010 persisted until 2017, where it has decreased since. The latest information on condition of adult Macquarie perch indicates that increased body condition associated with increased resources because of reservoir enlargement has expired and condition is returning to that of prefilling phases.

Utilising boat electrofishing to further assess the reduction in relative abundances of adult Macquarie perch in the ECR indicated that the abundance of individuals > 350 mm did not decrease (as per gill net captures) in 2018 compared to 2017, and indeed were higher than all other years since boat electrofishing was employed in 2014 (Table 5). Captures of Macquarie perch in 2020 by boat electrofishing were very low and comprised only of very large individuals. This is at odd with the gill netting data which indicates that a cohort of smaller (200 – 300 mm TL) individuals is present. It is not clear what is driving the low abundance of Macquarie perch captured by boat electrofishing in 2020, though it does highlight the benefit of utilising multiple methods (along with gill netting) to make assessments of the adult population in Cotter Reservoir.

Table 5. Total numbers of fish and per size class for Macquarie perch captured within the ECR using boat electrofishing between 2015 – 2020.

	Total	**Large (300-350 mm)**	**Very large (> 350 mm)**
2015	17	3	0
2016	21	16	2
2017	26	14	11
2018	24	12	11
2019	15	0	1
2020	5	0	5

Abundance of juvenile Macquarie perch in Cotter Reservoir has been relatively low since 2012, albeit with high levels of inter-annual variability (2014, 2018 – 2020 were highest). Relative abundances of young-of-year Macquarie perch were particularly low in 2011 – 2012 and 2014 – 2016, resulting in the reduced recruitment to the juvenile size classes over the following years (i.e. 2012/2013 and 2015 – 2017, respectively). Strong young-of-year abundances in 2017 saw a large 1+ year old cohort present within the 2018 catches and 2 + cohort in 2019, suggesting the ECR provided suitable conditions for early survival and growth of Macquarie perch recruits between autumn 2017 and autumn 2019 (Figure 11). Juvenile abundances continued to be strong in 2020, with similar abundances to both 2018 and 2019, though the modal size was slightly larger in 2020 compared to those years (around 150 mm TL). We expect that the strong year class from the 2016 spawning, now passed its third year of survival and growth, and starting to attain a size not well represented in fyke net captures, would contribute significantly to the adult population abundance (and spawning and recruitment) over the next few years.

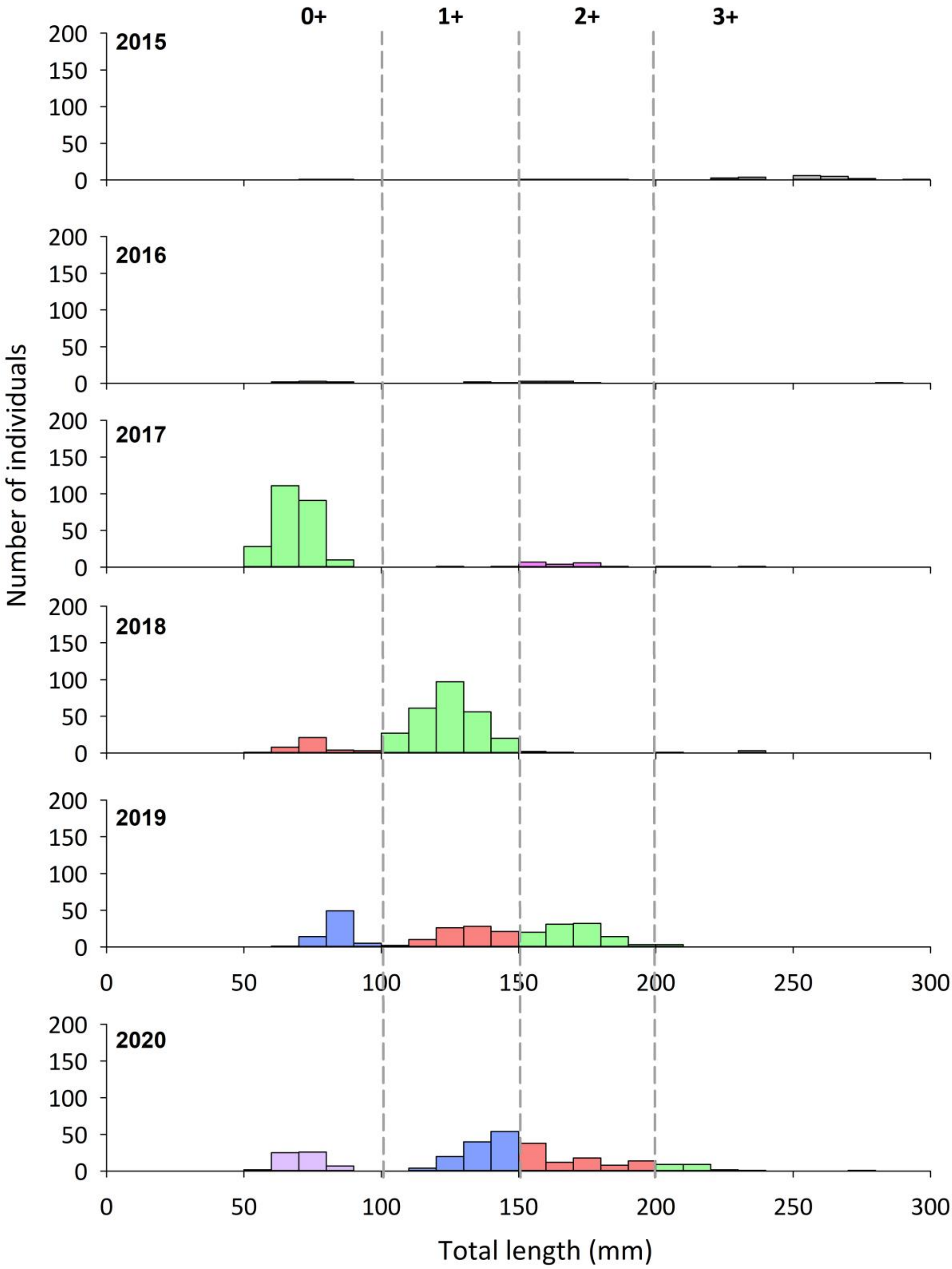

Figure 11. Length frequency (and estimated age class) of Macquarie perch captured from the ECR in fyke nets from 2015 – 2020.

Fyke netting in 2020 revealed regular captures (per fyke net) of young-of-year Macquarie perch in ECR, though significantly lower than the baseline monitoring years of 2010, 2012 and 2013 and the bumper operational year of 2017. The presence of young-of-year in the reservoir suggests that access to suitable spawning habitat was achieved in the spawning season of 2019, despite the reservoir being drawn down between 4 – 6 m from full supply level. This is a similar reservoir water level to that of the spawning season of 2015, which saw comparable numbers of young-of-year present in fyke netting in autumn 2016, though very low numbers of juveniles in autumn 2017. We hope to see a continuation of strong recruitment of the young-of-year cohort to juveniles in fyke net captures of autumn 2021, though expect some decrease as the strong cohort of 2016 grows beyond the size captured by fyke nets.

There was considerable concern arising from the lack of young-of-year during 2014 and very low numbers in 2015 and 2016, which is likely to have arisen during the filling phase for a range of reasons largely revolving around lack of access to spawning habitat (see Broadhurst *et al.* (2015) for discussion of previous years recruitment failure). Significant rainfall in the Cotter catchment meant that all three impoundments filled and the Cotter River was behaving as unregulated during the entire 2016 Macquarie perch spawning season. Importantly, a wet spring meant that river discharges between Bendora and Cotter were higher than those of the required environmental flow releases, which provided a strong natural cue (including temperature from dam overflow) for migration and spawning while facilitating higher levels of fish passage past potential instream barriers. By comparison, the hydrology of the 2017 – 2019 spawning seasons was vastly different to 2016 spawning season, in that river discharge during October 2017, 2018 and 2019 was largely regulated. The detection of young-of-year in 2020, albeit in lower abundances than those detected in 2017, suggests that spawning and early recruitment can be achieved with the ECR below full supply level and the Cotter River operating under regulated flow conditions. It is still unknown whether this ongoing low level of recruitment would be adequate to sustain the ECR population in the long term. It also remains to be resolved whether regulated flows can facilitate successful spawning and subsequent recruitment of young-of-year when the reservoir is drawn down below the most upstream of the large barriers to passage (situated at approximately 540 m ASL) (Broadhurst *et al.* 2016a).

Young-of-year abundance at the reference site (Kissops Flat) were at their lowest in 2020 since monitoring began. This continues a decrease in relative abundance of this size class since 2016. It is possible that bushfires in the catchment over the summer of 2019 / 2020 may have impacted on recruitment of Macquarie perch through sedimentation (because of run off) or potentially short-term declines in water quality.

RECOMMENDATIONS FOR 2020-21 MONITORING PERIOD

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 bias in capture efficiency. Based on low captures of large adults in gill nets but increased captures by boat electrofishing in 2018 (compared to previous years), capture efficiency of larger adults (> 350 mm) in gill nets appears to be reduced, compared with smaller adults. This may be associated with a reduced level of effort regarding the larger mesh gill nets (only shallow drop nets used in 5" mesh size prior to 2019). We recommend continuation of the trial of 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 nets;
2) Deployment of 125 mm 66 meshes deep 'deep' gill nets; 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 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. 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 good captures of young-of-year Macquarie perch in 2017 and the detection of a cohort of young-of-year since (2018 – 2020) is heartening after the consecutive recruitment failures since filling began. This is especially pertinent in 2020 as the reservoir was operating between 4 – 6 m below full supply level during spawning (the first time since the reservoir has filled that this has occurred). 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).

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 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 6). 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 6. 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 (100 - 150 mm TL) and young-of-year (< 100 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) 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 sampled. 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 will be approximately 60 – 99 mm TL based on results of the baseline data collected (Lintermans *et al.* 2013). Individuals were considered juvenile if they fell between 100 – 150 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 2020 (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

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 (<100 mm, >100mm 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 (<100 mm, >100mm 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 101 Macquarie perch were captured by fyke nets in the Cotter River across the five sites in 2020, ranging in total length (TL) from 39 – 225 mm (Figure 12). Young-of-year (<100 mm) and 1+ year old / juvenile (100 – 150 mm) individuals were captured at all riverine sites. CPUE of Macquarie perch (all sizes pooled) was not significantly different across sites and phases, but was significantly different among years within each operational phase and a significant site by year interaction (Table 7).

Table 7. Results of PERMANOVA analysis of fyke net catch-per-unit of Macquarie perch (all sizes combined) from Cotter River and Kissops Flat from 2010 – 2020 (bold text indicates significant result).

Source	df	SS	MS	Pseudo-F	P(perm)	perms
Site	4	0.57906	0.14476	1.1728	0.246	9932
Phase	2	0.4155	0.20775	1.4113	0.258	1646
Year (phase)	**8**	**1.1767**	**0.14709**	**5.8037**	**<0.01**	**9917**
Site x Phase	8	1.023	0.12787	1.0373	0.286	9924
Site x Year (phase)	**32**	**3.9524**	**0.12351**	**4.8734**	**<0.01**	**9846**
Res	599	15.181	0.025344			
Total	653	20.769				

Juveniles and adults/ sub-adults

There was no significant difference in the CPUE of Macquarie perch > 100 mm TL among sites (Global R = 0.001, p =0.357) or monitoring phases (Global R = 0.001, p =0.461). 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 13). In congruence, CPUE of Macquarie perch > 100 mm TL was also variable at the Kissops Flat reference site through time (Figure 13). 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 13). Macquarie perch were not detected at Burkes Creek Crossing in 2012, 2013 or 2016 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 13).

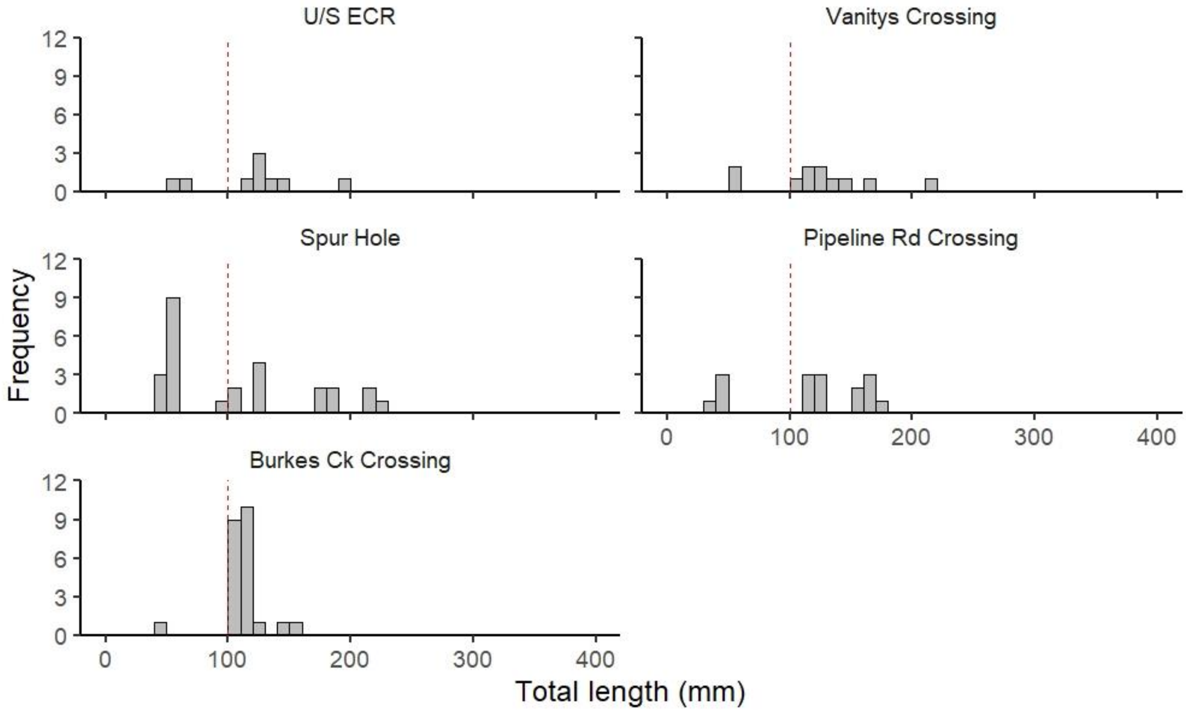

Figure 12. Length frequency of Macquarie perch captured in fyke nets and backpack electrofishing from Cotter River in 2020 at sites; U/S ECR, Vanitys Crossing, Spur Hole, Pipeline Road Crossing and Burkes Creek Crossing (red dashed line indicates cut-off for length of young-of-year individuals).

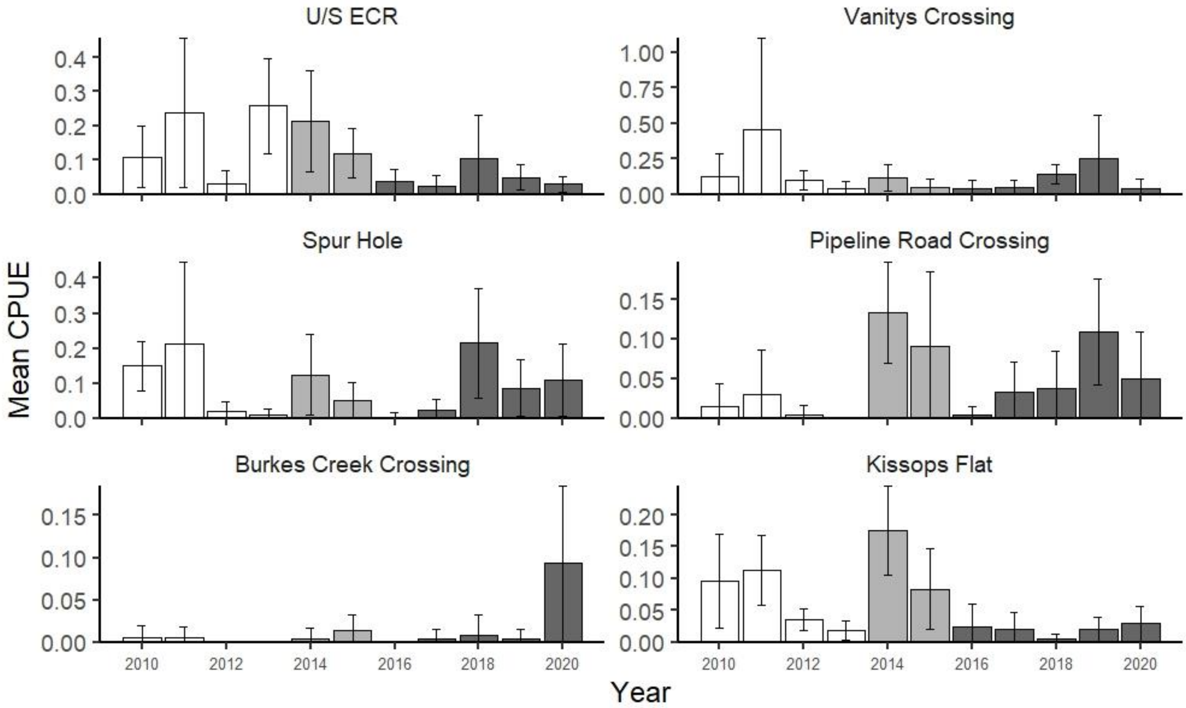

Figure 13. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Macquarie perch (greater than 100 mm TL) captured in Cotter River using fyke nets between 2010 and 2020. (Note that Bracks Hole (sampled from 2010 – 2013) was replaced by U/S ECR (sampled in 2014 – 2020) 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.

Relative abundance of Macquarie perch using backpack electrofishing was highly variable between sites and years (Figure 14). Backpack electrofishing captured a total of six Macquarie perch (one of which was YOY) at four of the five sites (not captured at Vanitys Crossing) in 2020. Macquarie perch were not captured at any site in 2011 and 2012 using backpack electrofishing (Figure 14). 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.

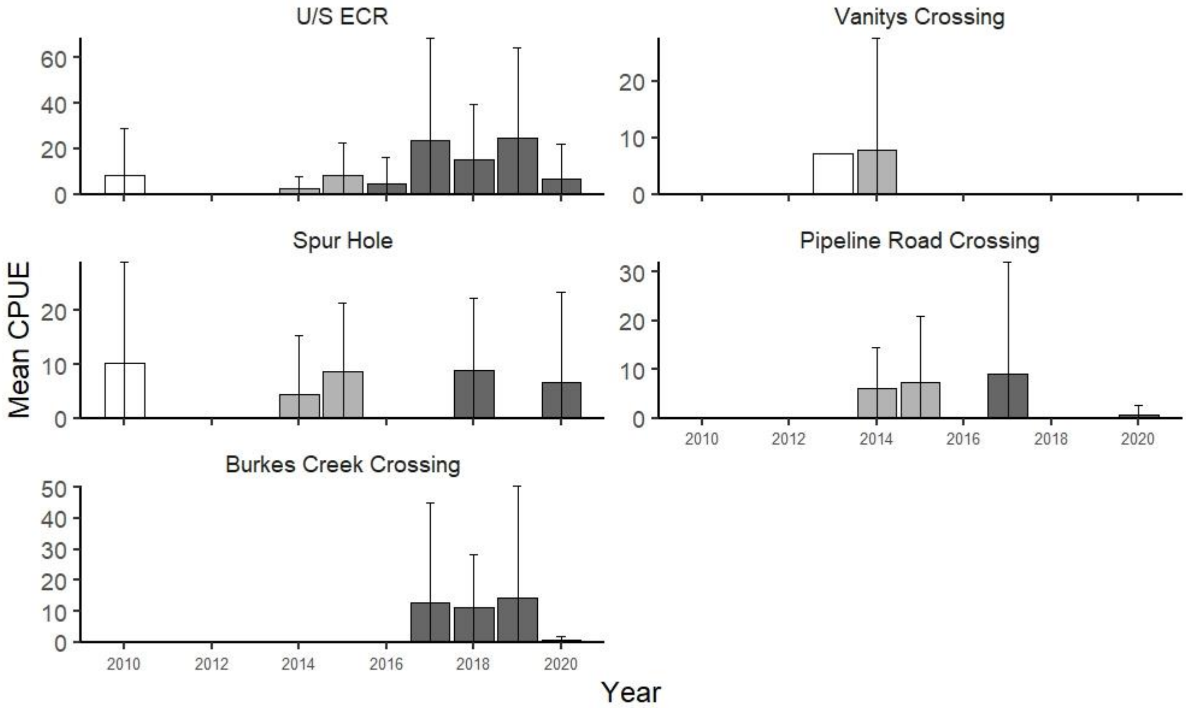

Figure 14. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Macquarie perch (all sizes pooled) captured in Cotter River by backpack electrofishing between 2010 and 2020. (Note that Bracks Hole (sampled from 2010 – 2013 was replaced by U/S ECR (sampled in 2014 – 2020) 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.

Young-of-year (YOY)

A total of 22 YOY Macquarie perch (< 100 mm TL) were captured using fyke nets and one collected using backpack electrofishing in 2020 (Figure 15 and Figure 12). Young of year were detected at every site. There was a significant difference in the CPUE of YOY Macquarie perch among sites with fyke netting (Global R = 0.002, p = <0.01), but no significant difference among phases (Global R = 0.004, p = 0.123).

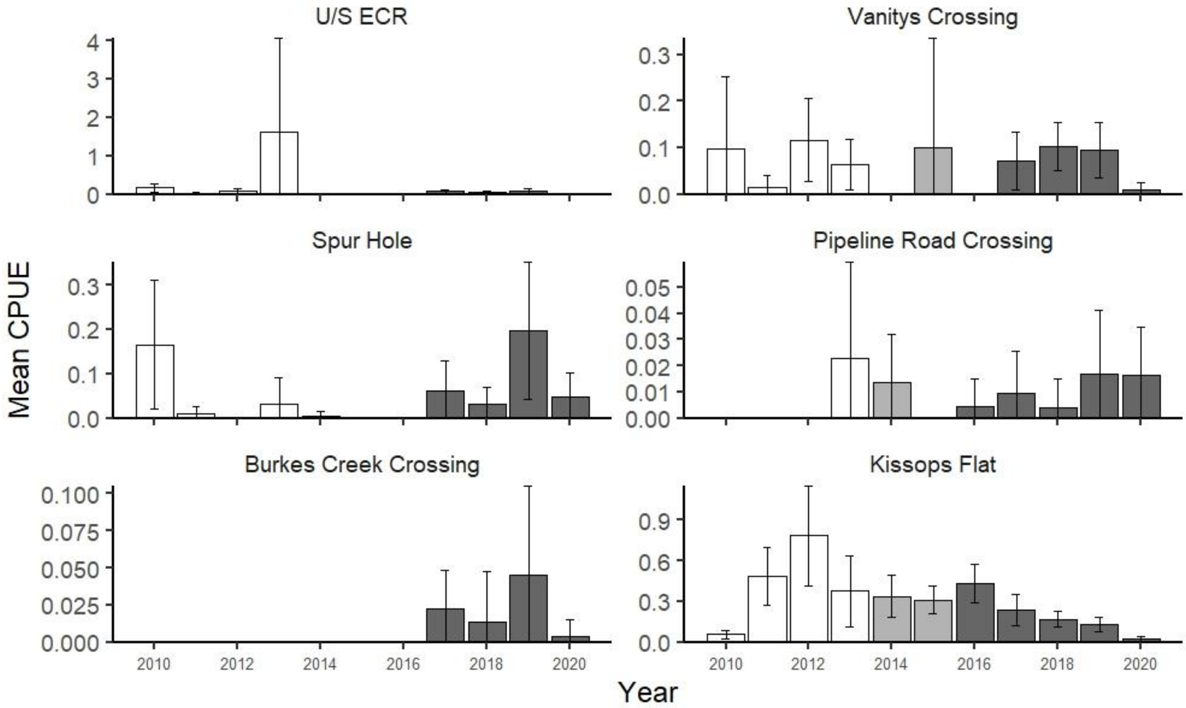

Figure 15. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni corrections) of Young-of-year Macquarie perch (< 100 mm TL) captured in Cotter River by fyke netting between 2010 and 2020. (Note that Bracks Hole (sampled from 2010 – 2013) and U/S ECR (sampled in 2014 – 2020) 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.

DISCUSSION AND CONCLUSIONS

Relative abundance

As has been the case since monitoring began in 2010, relative abundance of Macquarie perch in Cotter River in 2020 was highly variable between sites, as determined by both fyke netting and backpack electrofishing. Relative abundance generally decreased with distance upstream from Cotter Reservoir. The only exception to this trend was for the most upstream site (Burkes Ck Crossing), which recorded the highest abundance of Macquarie perch in 2020. Apart from the high numbers captured of Macquarie perch at Burkes Creek Crossing, 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 2013a).

One-year old individuals were present at each site in 2020. This follows on from 2019, where young-of-year were detected at all sites. These results suggest that the past year have been suitable for survival and growth of juveniles Macquarie perch in the Cotter River.

Young-of-year Macquarie perch were captured from every site in 2020 and comprised 20% of the total number of Macquarie perch captured. The presence of both young-of-year and high abundance of 1+ year class present at most sites is a positive result following three years of low abundances of recruits in the catchment from 2014 – 2016. It appears as though predominantly regulated conditions were suitable for Macquarie perch spawning at multiple sites, and survival and growth of individuals spawned between 2016 – 2019.

Distribution

Macquarie perch were detected at the four most downstream sites in all years and at the fifth site in eight of nine years, indicating that their distribution is relatively stable. Distribution differences between years is likely driven by the decreasing density (and potential patchy capture at low density sites) as you move 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 2013a, Lintermans *et al.* 2013, Lintermans 2016). The limitations of fyke nets and backpack electrofishing in sampling adult Macquarie perch is well understood (Lintermans 2013a, Lintermans 2016) and deployment of gill nets to sample adults in the river would involve significant additional cost, and pose significant risk of platypus bycatch. Provided the presence of young-of-year or juvenile individuals is readily detected, the presence of adults can be inferred. No change to monitoring recommended.

Juvenile Macquarie perch were detected at all sites in 2020. 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 all riverine sites in 2020, suggesting suitable conditions for wide-spread 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 16). 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 8. 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

Monitoring by fyke nets in the ECR in 2020 captured five Two-spined blackfish, four of which were captured in the newly inundated river reach and the other in the all-of-reservoir sampling. No young-of-year or juvenile Two-spined blackfish were captured in the ECR in 2020. Over six years of monitoring, 2012, 2017, 2018, and 2020 were the only years where Two-spined blackfish was captured in Cotter Reservoir downstream of the newly inundated zone. There was no Two-spined blackfish captured in the bait traps set in the Cotter Reservoir in 2020. Relative abundance of Two-spined blackfish was stable in the reference site over the monitoring period, with no significant differences apparent across years within Bendora Reservoir (Figure 16). Of the six Two-spined blackfish captured in Bendora Reservoir in 2020, none were likely to be young-of-year (<80 mm TL), and one was possibly a 1+ year old (80 – 150 mm).

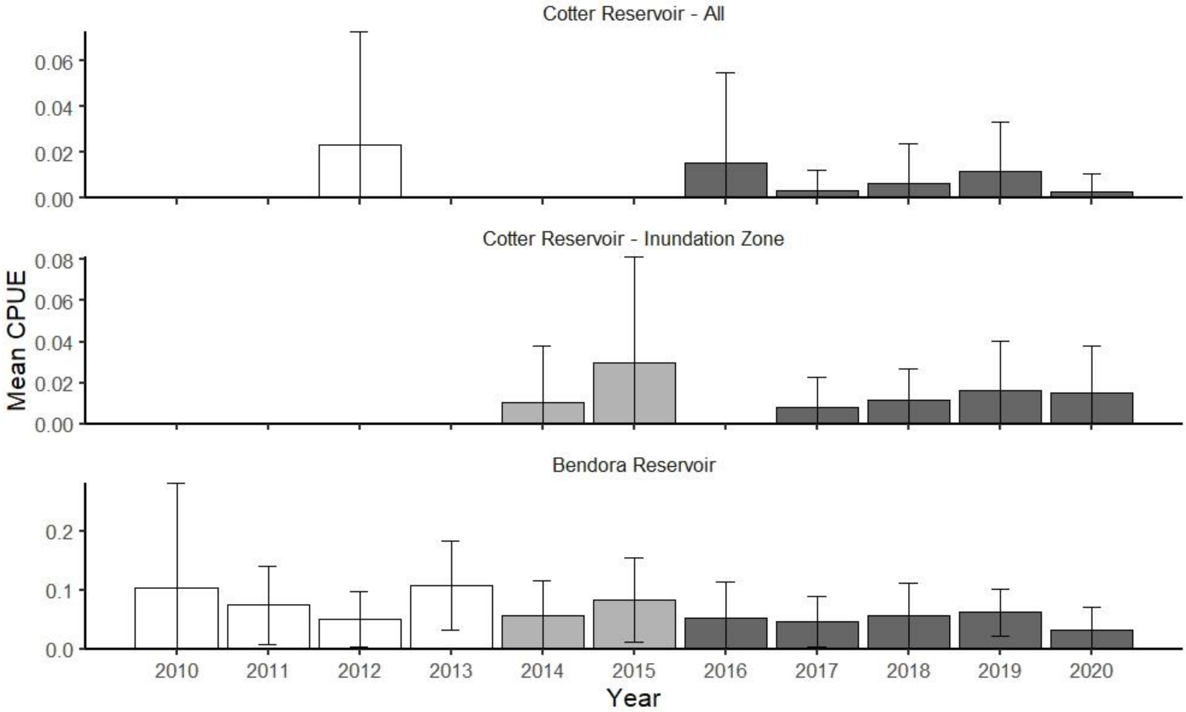

Figure 16. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Two-spined blackfish captured by fyke netting in Cotter Reservoir (both all around the reservoir and just the inundation zone) and Bendora Reservoirs between 2010 and 2020. For Cotter Reservoir, 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 densities in Cotter Reservoir in the 10 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 individuals captured in 2012 were at small experimental reefs associated with the *Constructed Homes* project (see Lintermans *et al.* 2010). As previously noted, fyke net sampling of the ECR in April 2016 for the Macquarie perch translocation project, one blackfish was caught on one night and another two were captured on a subsequent night (Lintermans 2017). All individuals were large adults and are not considered to be recaptures (Lintermans unpubl. data). Two of the individuals were captured adjacent to constructed rock reefs on Pryors Road. Translocation sampling with fyke nets in 2019 also captured a large (267 mm TL) blackfish in the non-inundation zone of the reservoir adjacent to constructed rock reefs (Lintermans unpubl. data). In light of this, the large-scale rock reef deployment for adult Macquarie perch has the potential to provide suitable habitat for colonisation by Two-spined blackfish. Monitoring since filling indicates that this is yet to occur on a significant scale and to date no evidence of breeding has been detected in Cotter Reservoir.

The capture of Two-spined blackfish in the newly inundated upstream third of the enlarged Cotter Reservoir across the past three years (and those captured by boat electrofishing in previous years) suggest that newly inundated shoreline of the enlarged Cotter Reservoir may serve as suitable habitat for the species, as is the case in Bendora Reservoir. No recruitment of Two-spined blackfish has yet been detected in the ECR. Monitoring over the coming years will provide further clarification of the reservoir's suitability longer-term.

RECOMMENDATIONS

Methods for assessing the population of Two-spined blackfish 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 juvenile and sub-adult Two-spined blackfish in Cotter Reservoir.

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 9). 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 9. 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) because of 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 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 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 resemblance matrix 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 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.

RESULTS

Abundance and distribution

Fifty-seven trout were captured in the ECR in 2020, comprising 43 Rainbow trout and 14 Brown trout. Twenty-one Rainbow trout were captured in Bendora Reservoir in 2020 (Brown trout are not present in this reservoir). There was no significant effect of reservoir, phase, year or section on the relative abundance of Rainbow trout captured in the ECR (Table 10), though there was a significant reservoir by year interaction (Figure 17). The latter was likely driven by the scarcity of Rainbow trout in Bendora Reservoir in 2016 and low abundances again in 2017. The number of Brown trout (n = 14; Figure 20) captured in the ECR in 2020 continues the trend of high relative abundances of this species over the past four years.

Table 10. 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 – 2020 (bolded text indicates statistically significant difference at the P(perm) 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	perms
Reservoir	1	0.14483	0.14483	0.76583	0.406	9824
Phase	2	0.17671	.08354	0.91579	0.423	9960
Year (phase)	8	0.55842	0.069803	1.0312	0.415	9927
Section(Reservoir)	8	0.8932	0.11165	1.6493	0.106	9929
Reservoir x Phase	2	0.66341	0.3317	1.4989	0.087	9907
Reservoir x Year(phase)	**8**	**1.3269**	**0.16586**	**2.4502**	**0.012**	**9953**
Phase x Section(Reservoir)	16	1.6095	0.1006	1.486	0.107	9908
Residuals	394	26.671	0.067694			
Total	439	32.596				

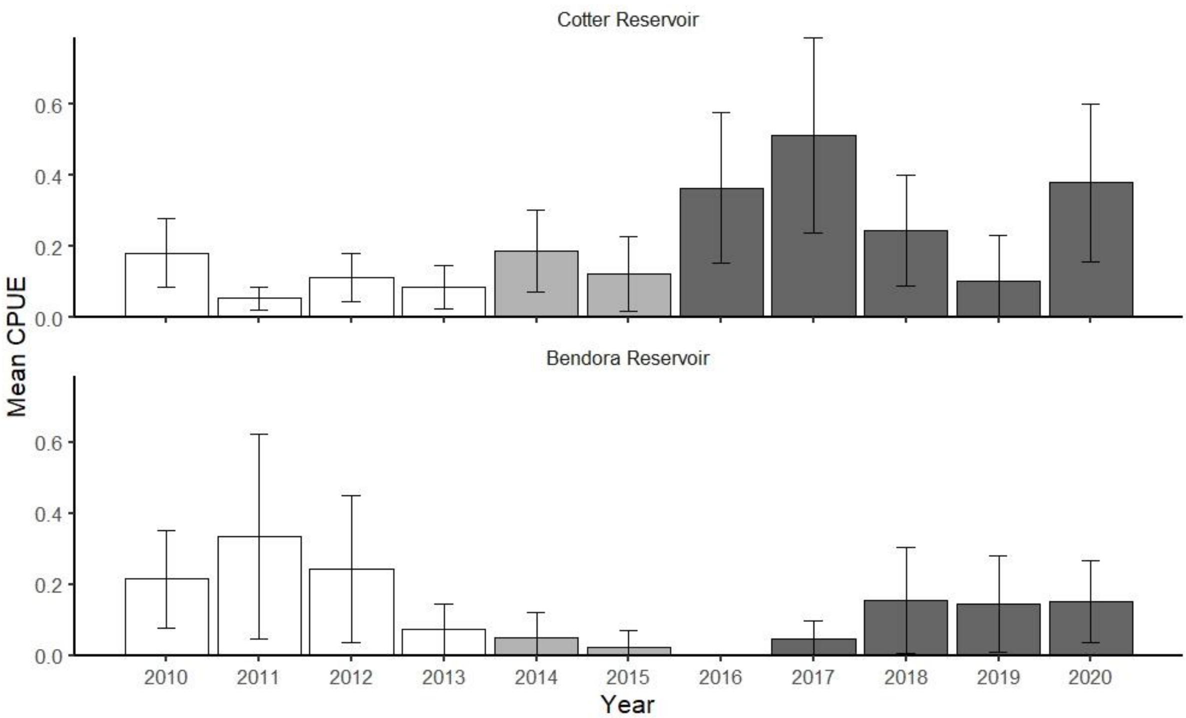

Figure 17. Mean catch-per-unit-effort (± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of adult Rainbow trout captured in Cotter Reservoir and Bendora Reservoir using gill nets each year from 2010 until 2020. White bars 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 2020 ranged from 285 – 473 mm Fork Length (FL) (Figure 18). Mean size of adult trout in the ECR was similar between all years (Figure 19). Size of adult Rainbow trout captured in Bendora Reservoir during 2019 ranged from 218 – 495 mm Fork Length (FL). Brown trout captured in gill nets in Cotter Reservoir in 2020 ranged in length from 434 – 560 mm (FL)(Figure 20). Brown trout abundances remain higher in operational years compared to baseline and filling years (Figure 21).

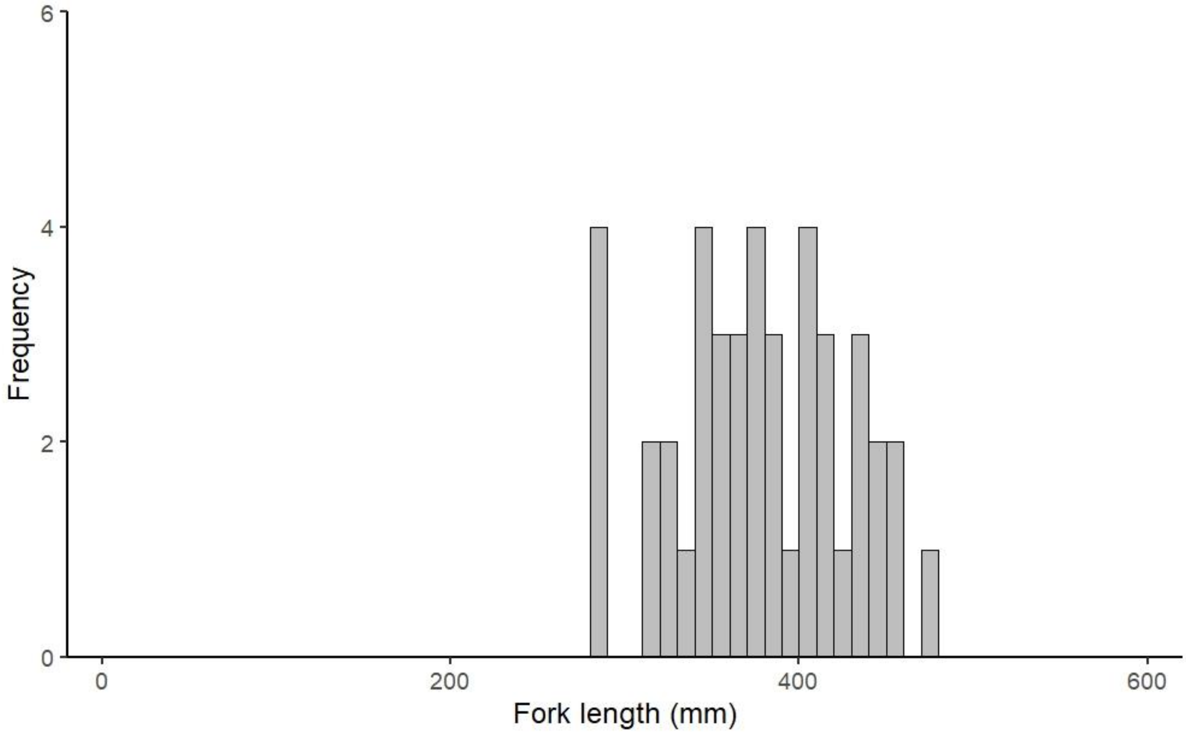

Figure 18. Length frequency of Rainbow trout (n = 43) captured from the ECR in autumn 2020 using gill nets.

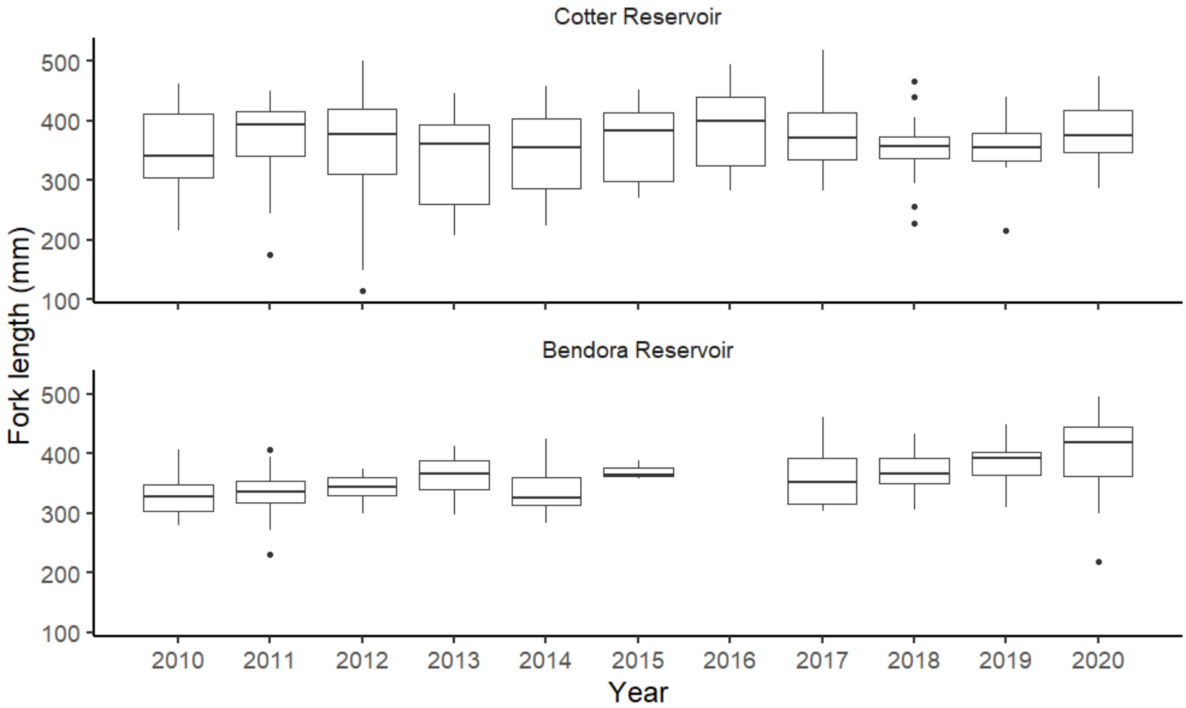

Figure 19. Boxplots of adult Rainbow trout captured in gill nets each year from Cotter and Bendora Reservoirs from 2010 to 2020 (solid line = median, box represents 25 – 75th percentiles, bars represent minimum and maximum lengths and black circles represent outliers).

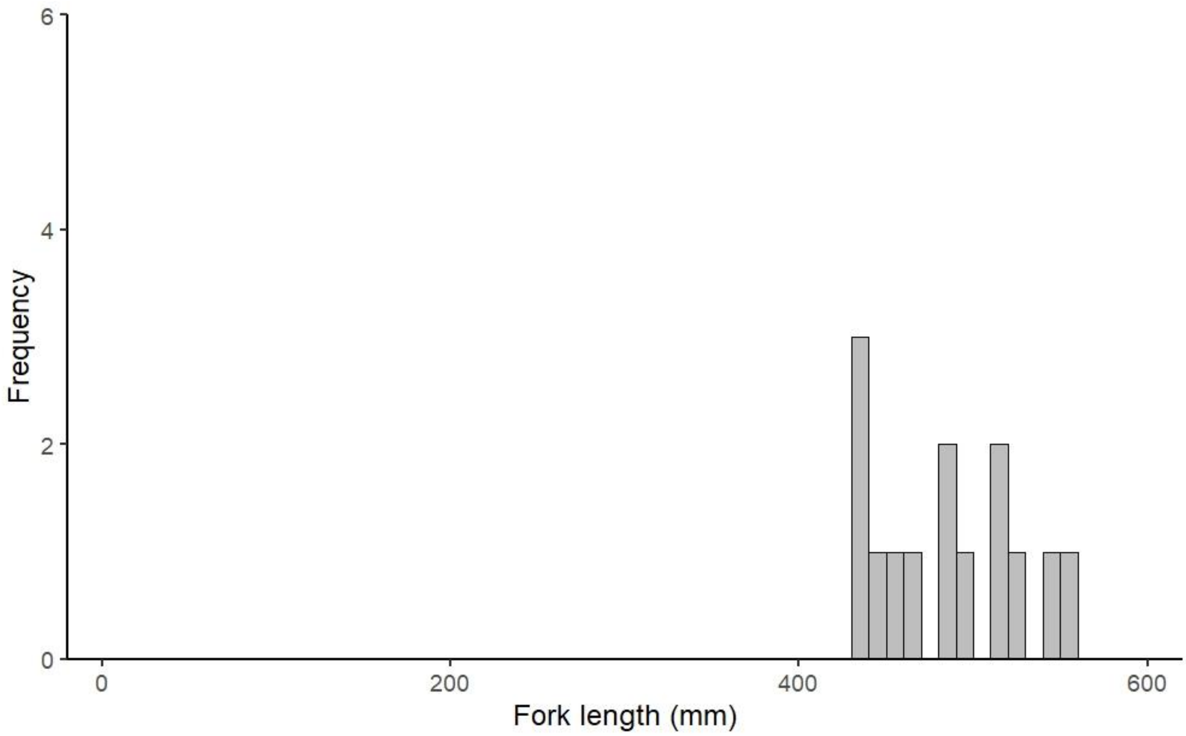

Figure 20. Length frequency of Brown trout (n = 14) captured from the ECR in autumn 2020 using gill nets.

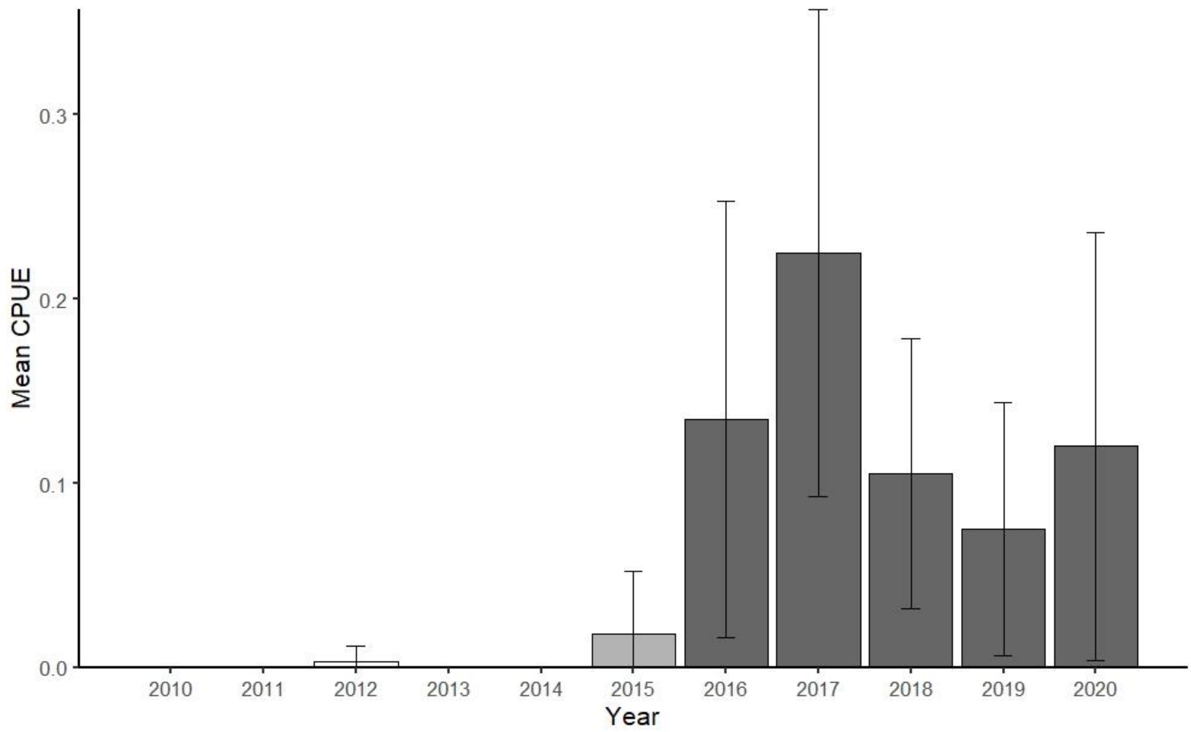

Figure 21. Mean catch-per-unit-effort (± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of adult Brown trout captured in Cotter Reservoir using gill nets each year from 2010 until 2020. White bars indicate baseline phase, light-grey bars indicate filling phase and dark-grey bars indicates operational phase of monitoring program.

DISCUSSION AND CONCLUSIONS

Abundances of Rainbow trout within the Cotter Reservoir have been relatively stable since monitoring began with the exception of higher abundances observed in 2010, 2017 and 2020 and lower abundances observed in 2011 and 2015. The abundance of Rainbow trout in Bendora Reservoir is variable, and appears to be related to temperature at the time of sampling, where a negative correlation between surface water temperature and number of Rainbow trout captured exists (Correlation co-efficient = - 0.7713343, p = 0.01) (Table 11). Lowest catches seem to occur when surface water temperature is > 17°C. The increased captures of Rainbow trout since 2018 in Bendora Reservoir indicate that a sizable adult population of trout remain in the reservoir, despite very low catches over the preceding four-year period.

Table 11. Details of Rainbow trout captures by gill nets and associated surface water temperatures in Bendora Reservoir 2010 – 2020. NR indicates temperature was not recorded.

Year	Date sampled	Water Temperature	No. Rainbow trout
2020	20/5/20; 21/5/20	12.5°C	21
2019	29/4/19; 30/4/19	16.2°C	20
2018	5/4/18; 19/4/18	15.0°C	22
2017	1/5/17; 4/5/17	14.2°C	6
2016	29/2/16; 7/3/16	23.7°C	0
2015	10/3/15; 16/3/15	19.7°C	3
2014	13/3/14; 17/3/14	NR	7
2013	4/4/13; 8/4/13	18.3°C	11
2012	19/4/12; 30/4/12	14.9°C	33
2011	9/5/11; 11/5/11	11.2°C	44
2010	20/5/10; 27/5/10	NR	28

The number of Brown trout captured in the ECR in 2020 continued to be relatively high, as has been the case since 2016 (noting that Brown trout do not occur in Bendora Reservoir). Prior to 2016, there had only been three individuals caught in the six years of monitoring combined. It appears that the Brown trout population is recovering in the Cotter system below Bendora since the Millennium Drought in the 2000's decimated the population in the reservoir, and the lower reaches of the Cotter River. Previous sampling of the reservoir in the 1990s and early 2000s regularly caught Brown trout (Lintermans unpubl. data; Ebner and Lintermans 2007). The Increased food resources (e.g. Goldfish, see Question 7), availability of thermal refuge (increased depth), and improved habitat quality (increased dissolved oxygen because of changed destratification procedures) since filling commenced in the ECR may also be assisting Brown trout recovery, with the reservoir providing a stable refuge for individuals (compared to the river; see Question 5). Anecdotally, Brown trout are considered more piscivorous and potentially more damaging to threatened fish populations than Rainbow trout (NSW Fisheries 2003). With respect to the hardier Rainbow trout, the relatively stable abundance of this species between years in Cotter Reservoir suggests that filling of the ECR has not driven a significant population increase of adult Rainbow trout in this water body.

To date, there was no difference in the size composition of adult Rainbow trout captured between years in Cotter Reservoir. It was expected that as the reservoir fills, food resources would increase and would lead to increases in size of adult trout. There was likely to be a time-lag before we see any increases in body length in the resident trout population. Such a change in length would be more likely to occur over multiple years (2 – 5) as the trout would likely first increase their body condition by taking advantage of increased food resources (Kimmel and Groeger 1986, Ploskey 1986, O'Brien 1990), which would over time result in increased growth and length of adult trout. Visual ad hoc examination of trout indicates that body condition is improving. An increase in the average length of trout (and the maximum length) will likely change predation dynamics in the reservoir as larger trout (with larger gape-range) seek larger food items that tend to be resident prey fishes (Jonsson *et al.* 1999, Ebner *et al.* 2007). So far, this has not materialised, though a scenario of considerable conservation concern would be the growth of extremely large trout that have the capacity to consume a wide range of sizes of Macquarie perch.

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 relatively stable rainbow trout abundance in Cotter Reservoir is recommended at this time. However, it is still considered a risk that trout size and abundance may increase over, 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 recovery of the Brown trout population in the Cotter catchment is of some concern, with this species being highly piscivorous once they attain length of > 250 mm FL (all of the 2017 – 2020 captures were above this size - Figure 20). When considered alongside the evidence for failed recruitment of young Macquarie perch, the increasing abundance of Brown trout (and particularly large brown trout) needs to be monitored closely and appropriate management action taken if the current trend continues. No management of this species is currently warranted.

<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 12). 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 12. 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 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. In 2020 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 impacts.

Brown trout were only rarely captured, 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 resemblance matrix constructed with modified Gower (base 2) dissimilarity measure. Cotter River site and year as 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

Number of Rainbow trout captured in Cotter River in 2020 was very low. A total of 25 Rainbow trout were captured from four riverine sites on the Cotter River using fyke nets and backpack electrofishing in 2020, all captured via electrofishing (Figure 22). Rainbow trout captured in 2020 from the Cotter River ranged in size from 90 – 325 mm FL (Figure 22). Rainbow trout were captured at all riverine sites downstream of Bendora Dam except for Spur Hole. The size composition of Rainbow trout in the Cotter River upstream of the ECR has been relatively stable at each site during the nine years of monitoring (Figure 23). Rainbow trout captured in 2020 were largely within the normal size range of those captured prior to filling commencing.

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 24 and Figure 25). There was a significant difference in the relative abundance of Rainbow trout between sites and between years (Table 13). 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 24 and Figure 25) 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 24 and Figure 25). Vanitys Crossing also had a lower relative abundance of Rainbow trout compared to Burkes Creek Crossing (Figure 24 and Figure 25). In general, backpack electrofishing was more likely to detect the presence of Rainbow trout (4 of 6 sites) than fyke netting (1 of 6 sites) (Figure 24 and Figure 25). 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 was one Brown trout captured in 2020 (via electrofishing); a 485 mm FL individual from U/S ECR.

Table 13. Results of PERMANOVA analysis of trout relative abundance (determined by 4 x 30 m backpack electrofishing CPUE) in Cotter River from 2010 – 2020 (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**	**33.222**	**6.6444**	**5.8738**	**<0.001**	**9938**
Year	**10**	**31.66**	**3.166**	**2.7988**	**0.002**	**9924**
Site x Year	47	70.149	1.4925	1.3194	<0.01	9874
Residuals	186	210.4	1.1312			
Total	248	349.1				

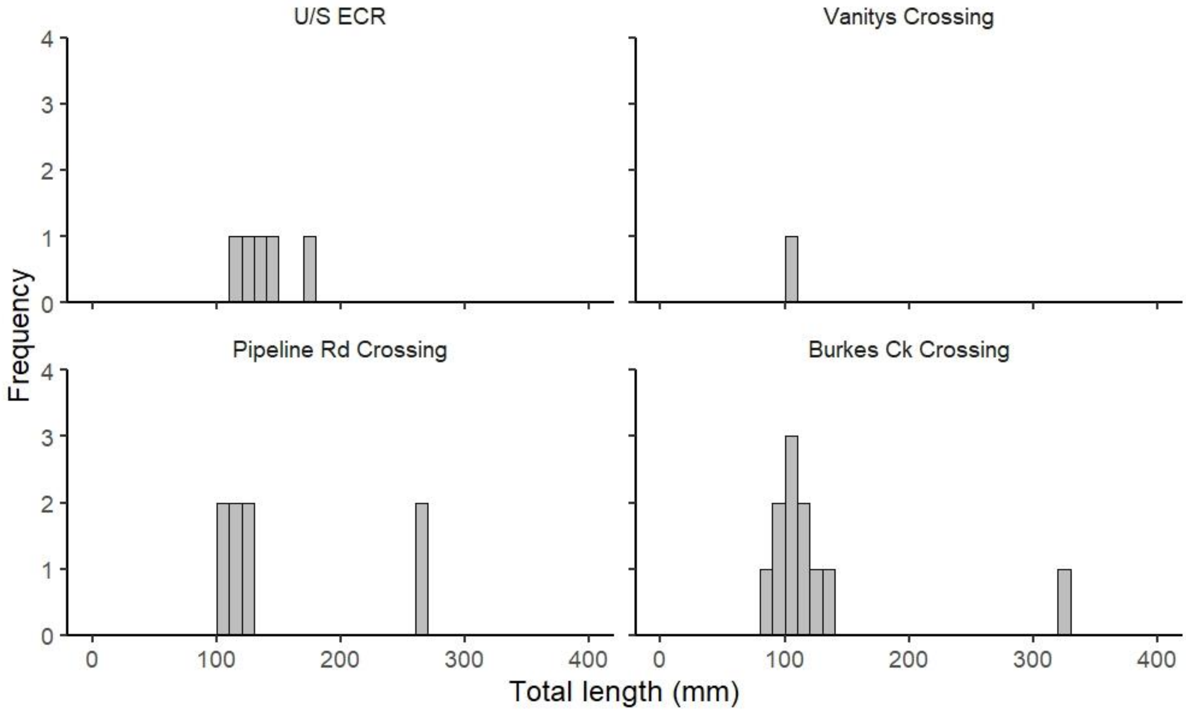

Figure 22. Length frequency of Rainbow trout captured from Cotter River at U/S ECR; Vanitys Crossing; Spur Hole; Pipeline Road Crossing and Burkes Creek Crossing in 2020 using fyke nets and backpack electrofishing (inclusive of additional backpack electrofishing effort).

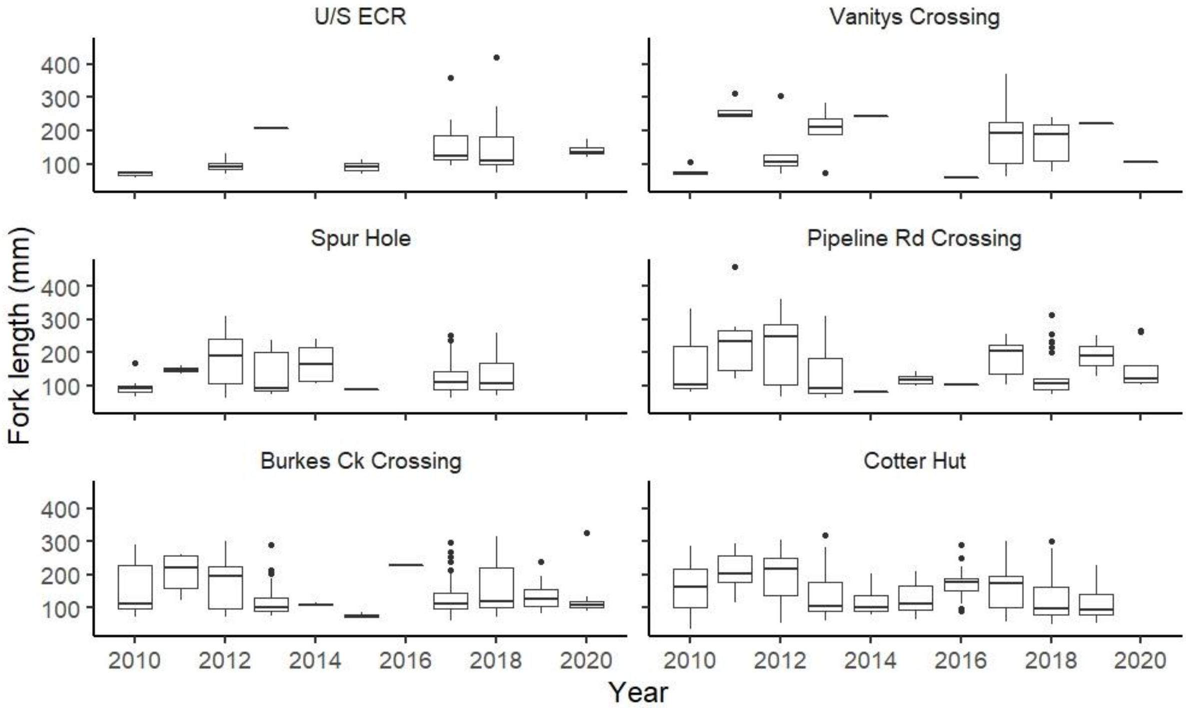

Figure 23. Mean lengths (± 95% confidence limits with Bonferroni correction) of Rainbow trout captured from Cotter River using both fyke nets and backpack electrofishing from 2010 – 2019. Red dots indicate maximum size of trout captured per year and blue dots represent the smallest trout captured per year. Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2020). Note that the increased effort employed from 2017 onwards was used for this figure. Cotter Hut was not able to be sampled in 2020 because of fire and COVID-19 restrictions.

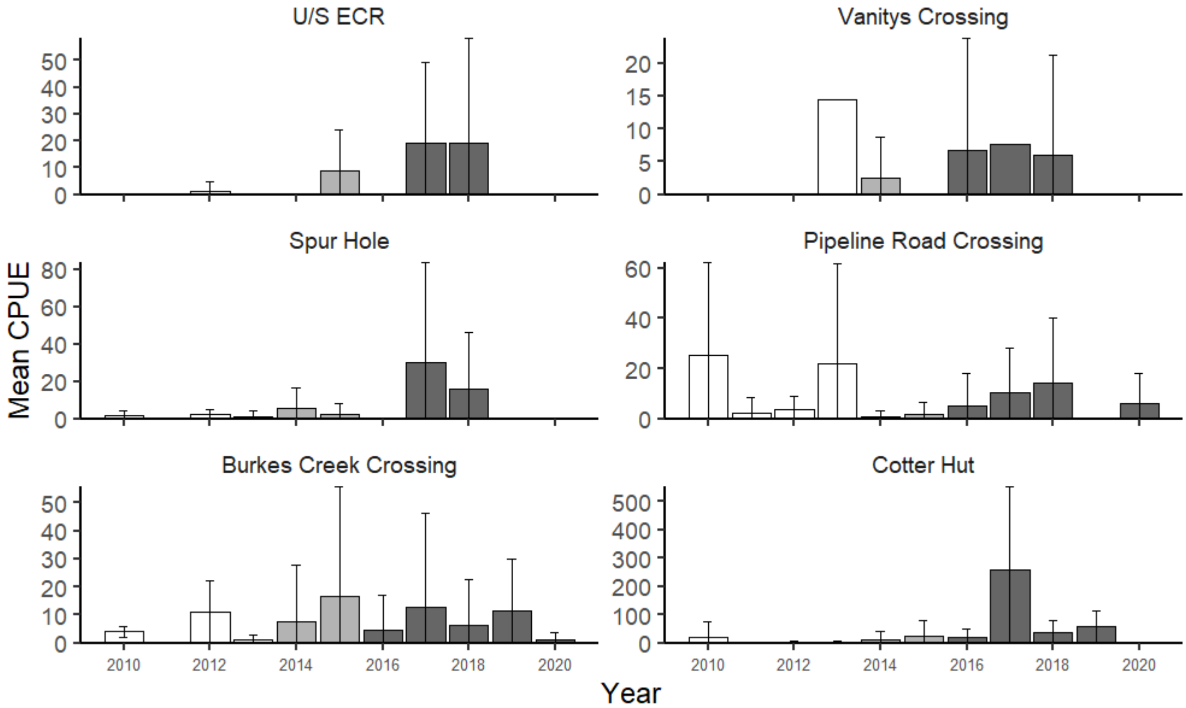

Figure 24. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Rainbow trout captured in Cotter River by backpack electrofishing (4 x 30 m shots) between 2010 and 2020. (Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2020). Cotter Hut was not able to be sampled in 2020 because of fire and COVID-19 restrictions. White bars indicate baseline phase, grey bars indicate filling phase and white bars with diagonal stripes indicates operational phase of monitoring program.

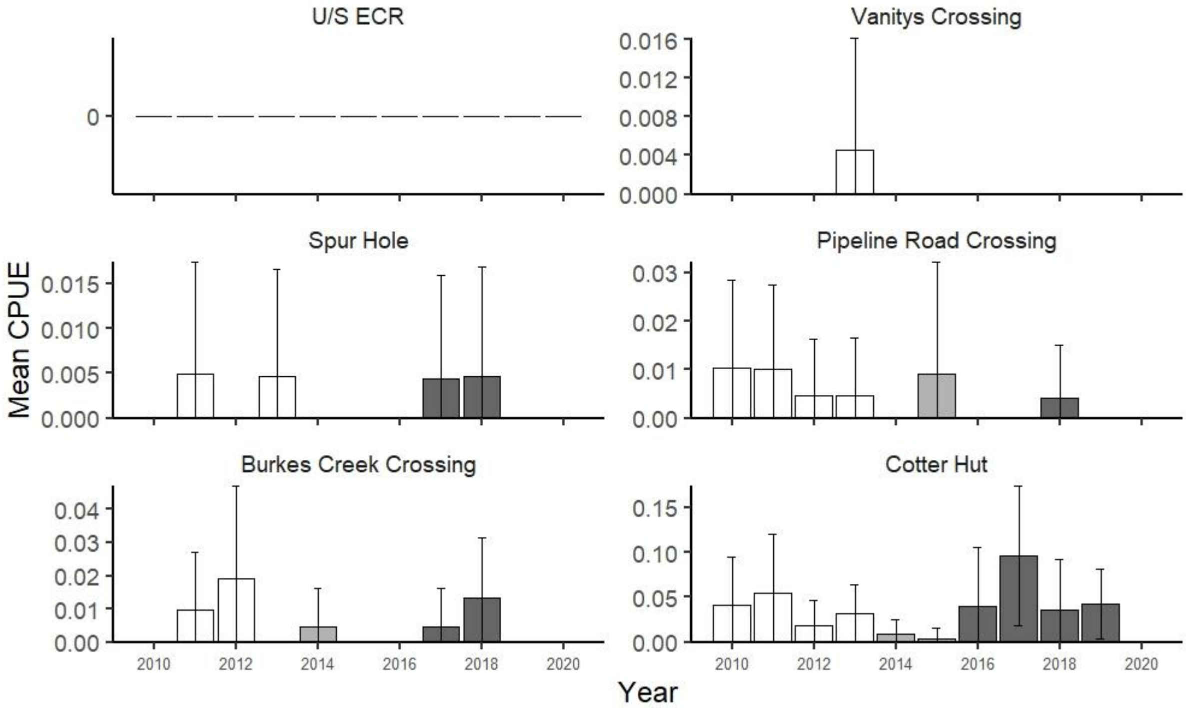

Figure 25. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction) of Rainbow trout captured in Cotter River by fyke net between 2010 and 2020. (Note that Bracks Hole (sampled from 2010 – 2013) has been replaced by U/S ECR (sampled in 2014 – 2020). Cotter Hut was not able to be sampled in 2020 because of fire and COVID-19 restrictions.

DISCUSSION AND CONCLUSIONS

The increased electrofishing effort employed from 2017 onwards captured 10-fold the number of trout captured on average than the previous methodology. 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. If current capture trends continue, a fourth year of sampling at the increased backpack electrofishing effort rate will allow robust comparisons of abundance across years and sites.

Based on data using the previous 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 during the monitoring period to date. Similarly, Brown trout abundance in the Cotter River remains extremely low. The low numbers of trout recorded per site using the previous sampling effort and the extremely high variability made it difficult to detect statistically significant change in trout length. The increased sampling effort employed from 2017 – 2020 helps to reduce fish length variability, and so will allow an increased chance of detecting significant trends in trout length in future years. However, we still need to catch an adequate sample of trout, which was difficult in 2020. 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 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). As mentioned earlier in Question 4, a lag time likely exists between the increase in resources in the ECR and any change in the Rainbow trout population, most likely in the order of 2 – 5 years based on this species is an annual spawner and reaches maturity at 2 – 3 years of age in the Cotter system (Lintermans and Rutzou 1990).

It is still expected that filling of the ECR and the resultant increase of food resources (Kimmel and Groeger 1986, Ploskey 1986, O'Brien 1990) would lead to increased growth and size of trout in the reservoir (Jonsson *et al.* 1999), which could potentially spill into the river upstream due to density–dependent competition (i.e. insufficient space in reservoir leading to increased competition between trout for space) and/or increased numbers of spawning adults entering the river. Assuming logistic growth of the trout populations in response to an increase in the carrying capacity of the reservoir, such effects would take 2 – 5 years post-filling to appear; given this species is an annual spawner and time taken to convert increased food to increased body condition and size (including increasing gape). Monitoring of trout abundance and size may show changes in future years to that documented in the filling and operational phase monitoring from 2014 – 2020 (Lintermans 2012, Hatton 2016).

Monitoring results so far indicate that backpack electrofishing is more effective at obtaining relative abundances 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 11 Brown trout have been collected in the standardised monitoring of the river over the 11 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. The threat of Brown trout to native species 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

Despite low captures of trout in 2020, the 5-fold increase in raw numbers of trout captured with the revised method (increased electrofishing effort) over the past three years of implementation provides a more accurate representation of trout size in the Cotter River. No change to the current 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 also remains similar since filling commenced, as expected. However, it is likely that trout abundance will increase over time 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).

<u>QUESTION 6:</u> 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 14). 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 14. 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 over 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 four Rainbow trout and one Brown trout over 150 mm FL were captured in 2020 (Table 15). Visual examination of stomach contents detected no Two-spined blackfish or Macquarie perch in of any of the fish examined (Table 15). During sampling in Cotter Reservoir, one Brown trout was captured, which was 484 mm FL and 1440 g, during gill netting on the 26th March 2020 that had one Macquarie perch in its stomach (145 mm TL).

Table 15. Details of trout captured and examined for fish and fish remains from the Cotter River in 2020.

Site	Species	Fork Length (mm)	Capture technique	Fish remains in stomach
U/S ECR	Rainbow trout (n=1)	175	Backpack electrofishing	No
	Brown trout (n=1)	485		No
Pipeline Road Crossing	Rainbow trout (n=2)	262 -266	Backpack electrofishing	No
Burkes Creek Crossing	Rainbow trout (n=1)	325	Backpack electrofishing	No

The number of trout stomachs examined for evidence of predation on threatened fish increased significantly in 2017 - 2018 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 has been a dramatic reduction in 2019 - 2020 (Table 16).

Table 16. Comparison of number of trout stomachs examined from 2010 - 2020.

	2010	2011	2012	2014-2016	2017	2018	2019	2020
No. of fish examined	198	290	222	16	71	75	19	5
No. of Rainbow Trout	190	288	216	14	68	70	16	4
No. of Brown trout	8	2	6	2	3	5	3	1
Size range of trout examined (Fork Length) (mm)	90-513	148-460	150-500	170-290	150-371	156-445	155-451	175-485

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 five trout stomachs examined in the field in 2020. The increased sampling effort (20 individuals or 1 km of river) had greatly increased the catch rate of trout with 71 and 75 trout captured in 2017 and 2018, respectively, compared to 16 fish for the previous three years combined (2014 – 2016). The number of trout captured in 2019 was very low compared to the previous two years with 2020 having a further ~75% reduced catch rate compared to 2019. Other electrofishing sampling in the Cotter River between Bendora and Cotter reservoirs in March 2019 also failed to capture trout (Lintermans unpubl. data). The explanation for the very low trout abundance in 2020 is unknown. Body length to gape relationships of trout (taken from Ebner *et al.* 2007) suggest that trout over 150 mm in length have a gape sufficient to ingest early juvenile (< 30 mm) Two-spined blackfish. Ebner *et al*. (2007) also noted that trout moved to piscivory at approximately 250 mm FL. Baseline data also suggests that predation rates of Two-spined blackfish were low (<6 found in over 700 trout stomachs examined) in the 3 years of baseline sampling and predation of Macquarie perch was not detected (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 in 2020. As mentioned previously in this report, an increase in food resources in the ECR is likely to result in increased growth, size and abundance of trout in the ECR and this increased abundance is likely to extend into the Cotter River (Jonsson *et al.* 1999, Lintermans 2012). No evidence of a change in trout size or abundance has been detected (see Question 5) thus far, so a change or increase in the predation rate of native species by trout would not be expected. Continuation of monitoring of both trout size and abundance and predation rates will provide an early indicator of change and could lead to early management action.

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.

As noted in Broadhurst *et al.* (2018), a contractor provided evidence of trout predation upon Macquarie perch in the ECR. Despite the almost complete overlap in distribution between trout and Macquarie perch, verified records of predation of Macquarie perch by trout have been extremely rare (Ebner *et al.* 2007). The 2018 record was the first verified (i.e. with photographic evidence) case of predation of Macquarie perch by trout in the Cotter Catchment, with the 2019 records from the ECR confirming that trout predation on Macquarie perch is continuing. Another Macquarie perch (145 mm TL) was detected in 2020 in the stomach of a Brown trout (484 mm FL) in Cotter Reservoir providing more evidence that this predation is occurring. Whilst in isolation these records represent a small proportion of Macquarie perch have been predated upon, it exhibits the potential predation pressure that trout could present if a change in diet to include more Macquarie perch occurred, and particularly if the brown trout population in the ECR and the Cotter River continues to increase.

RECOMMENDATIONS

The increased sampling of 1 km 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 (ie 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 17). 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 17. 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 resemblance matrix 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 2020, 26 Goldfish were captured ranging in length between 36 – 193 mm FL (Figure 26). The vast majority of these individuals were between 50 – 150 mm FL, most likely corresponding to 0+ and 1+ year-old age class (Merrick and Schmida 1984)(Figure 26). Goldfish relative abundance was significantly different among years (Global R = 0.145, $p < 0.001$), 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 – 2020) (Figure 27). Relative abundance of Goldfish from 2017 – 2020 was not significantly different from any of the baseline monitoring years.

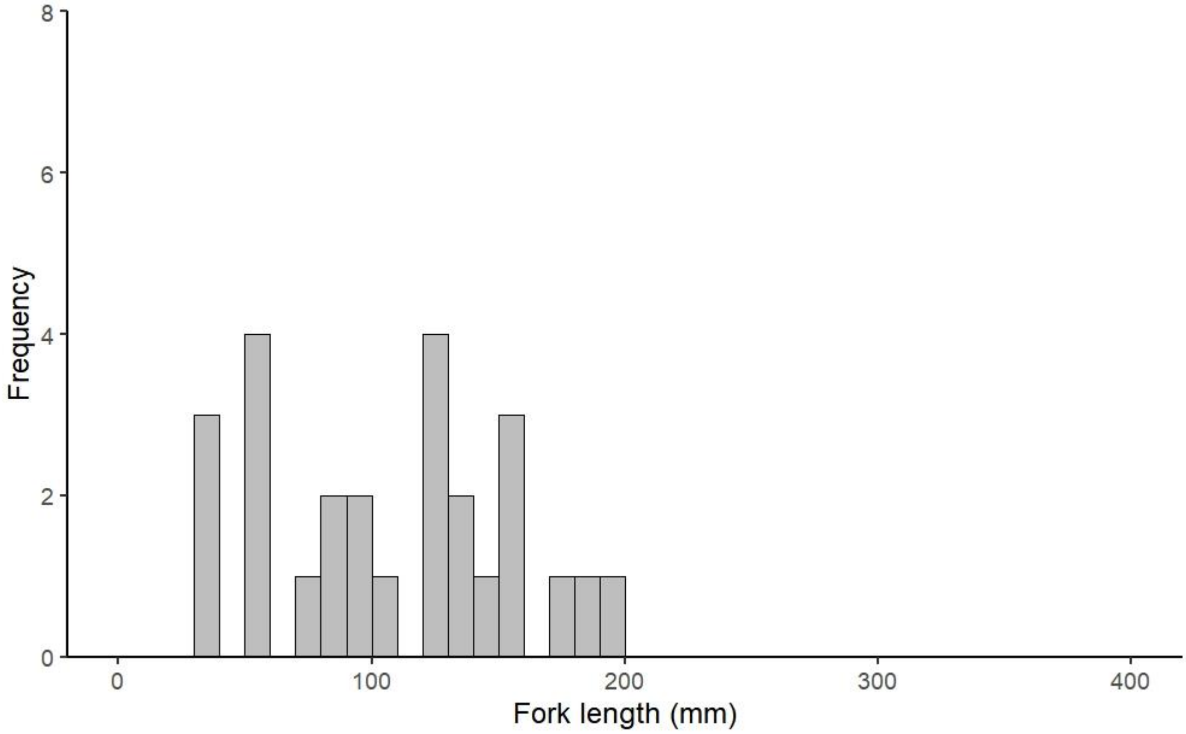

Figure 26. Length Frequency of Goldfish captured in the ECR in 2020 over three nights of fyke netting.

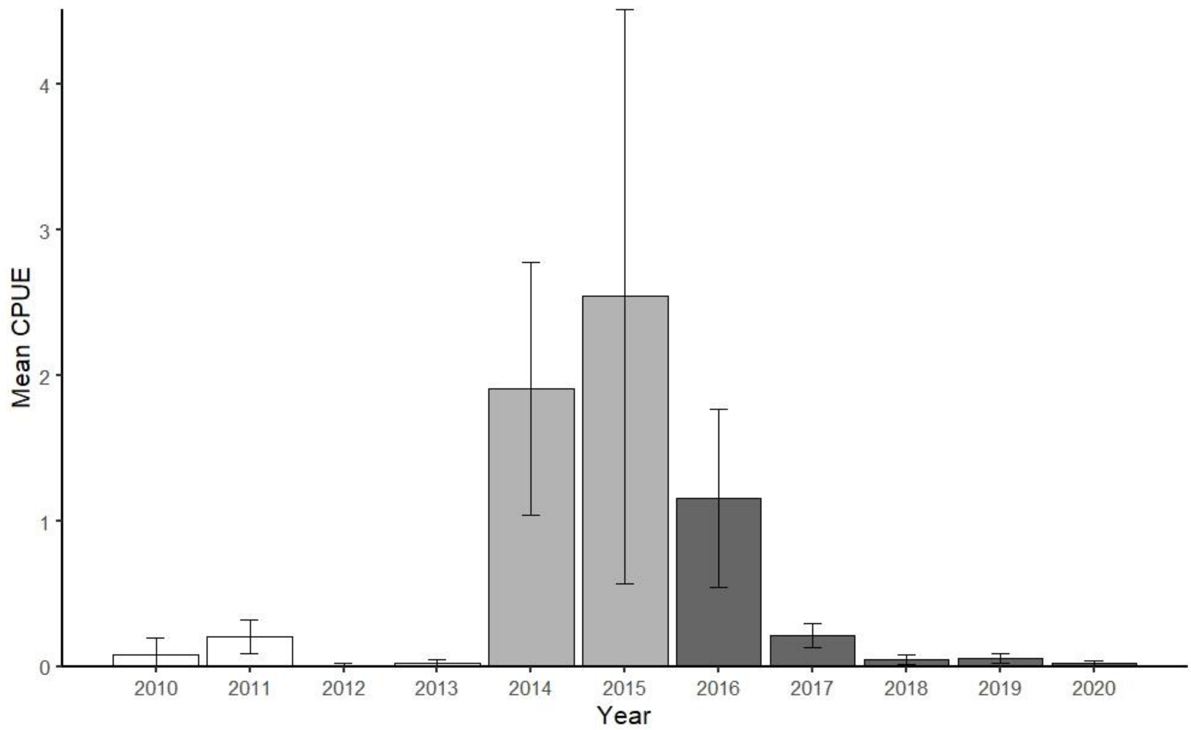

Figure 27. Relative abundance (displayed as mean CPUE ± 95% confidence limits with Bonferroni correction, scaled for relative net effort versus shoreline length at the time of sampling) of Goldfish captured in the ECR using fyke nets between 2010 and 2020. White bars indicate baseline phase, light grey 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 ten years of monitoring in Cotter Reservoir (one each in 2010 and 2011; three in 2012 and two in 2017). Thirty-seven Oriental weatherloach were observed whilst undertaking the boat electrofishing of Cotter Reservoir (30 in 2016 and four in 2017). No Oriental weatherloach were observed or captured in 2020 via any of the sampling methods. Eastern gambusia have only been captured in fyke nets in three years, 2014, where 13 individuals were captured, all from one fyke net and 2015 and 2017 where one individual was captured, respectively. Eight Eastern gambusia (ranging from 18 – 33 mm TL) were captured in bait traps in 2015 and a further seven captured in bait traps in 2016. No fish were captured in bait traps in 2017 – 2020. It must be noted that large numbers of Eastern gambusia were observed whilst undertaking the boat electrofishing during three years of monitoring (2014 – 2016). However, only 29 were observed while boat electrofishing in 2017 and only 1 observed in 2018. No Eastern gambusia were observed during the day in 2020. Small numbers of Eastern gambusia are also regularly observed in the reservoir at night during gill netting operations.

DISCUSSION AND CONCLUSIONS

After two years of very high relative abundance of Goldfish during the first two years of filling (2014 and 2015), there was a reduction in relative abundance from 2016 – 2018, then stabilising in 2019 – 2020. The higher relative abundance of Goldfish in the initial stages of filling is likely to have been caused by an increase in availability of food resources associated with the filling phase of the ECR (Kimmel and Groeger 1986, Ploskey 1986, O'Brien 1990). An increase in Goldfish abundance could have a number of effects on the native fish species (primarily Macquarie perch) in the ECR. Goldfish are a known prey item of both trout and cormorant species in the Cotter Reservoir (Ebner *et al.* 2007, Lintermans *et al.* 2011). An increase in the abundance of Goldfish could lead to increased abundance of Goldfish predators, which are very likely to increase predation upon other fishes like Macquarie perch, especially in prey-switching scenarios, i.e. if Goldfish abundances suddenly drop, piscivores switch to the next most abundant fish prey (e.g. Beukers-Stewart and Jones 2004, Baker and Sheaves 2005). Increased piscivory has been shown to lead to an increase in growth rates and size of trout species (Jonsson *et al.* 1999). A change in the size composition of trout present in the reservoir could change predation and competition interactions with Macquarie perch both in the reservoir and in the river upstream. At the commencement of the monitoring program it was hypothesised that an increase in food availability for cormorants could also lead to a significant change in abundance or the establishment of a breeding colony in the reservoir (Lintermans 2012, Lintermans *et al.* 2013), which would increase the energy demands of the predators by an order of magnitude (Chip Weseloh and Ewins 1994). As predicted, the increase in Goldfish observed in 2014 and 2015 is 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 – 2020 were not significantly different to baseline abundances, suggesting that the boom in resources associated with filling and early operational phases has ceased. It is interesting to note that that the decline in goldfish abundance and the increase in Brown trout abundance in the ECR since 2017 coincides with the first records of trout predation on Macquarie perch. This may represent the first signs of prey switching by trout or increased predation risk to Macquarie perch, as was predicted. Monitoring of Goldfish, cormorant and Macquarie perch abundance along with trout size and abundance, in the reservoir and river upstream is essential in determining causal relationships between the ECR filling and operation and changes in predator—prey dynamics.

The relatively low catch rates again of both Oriental weatherloach and Eastern gambusia is not considered to reflect population sizes of these two species. 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 much in 2017 or in 2018. The lack of adequate representation of Oriental weatherloach and Gambusia populations 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), 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 – 2020 has revealed a further 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.

<u>QUESTION 8:</u> 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 18).

Table 18. 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 resemblance matrix constructed with modified Gower (base 2) dissimilarity measure. For PERMANOVA analysis, monitoring phase and section as 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 Pied cormorant recorded in 2020 (Figure 28). There have been only six observations of Pied cormorant (all of single individuals) since monitoring began in 2010, though none in 2018 and 2019. Abundances of the three most common species were relatively consistent with expectations during the monitoring period with some seasonal fluctuations present (Figure 28). Since filling began, abundances of both Great cormorant and Little black cormorant have been stable, though with some definitive seasonal fluctuations (Figure 28). 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 28 and Figure 29).

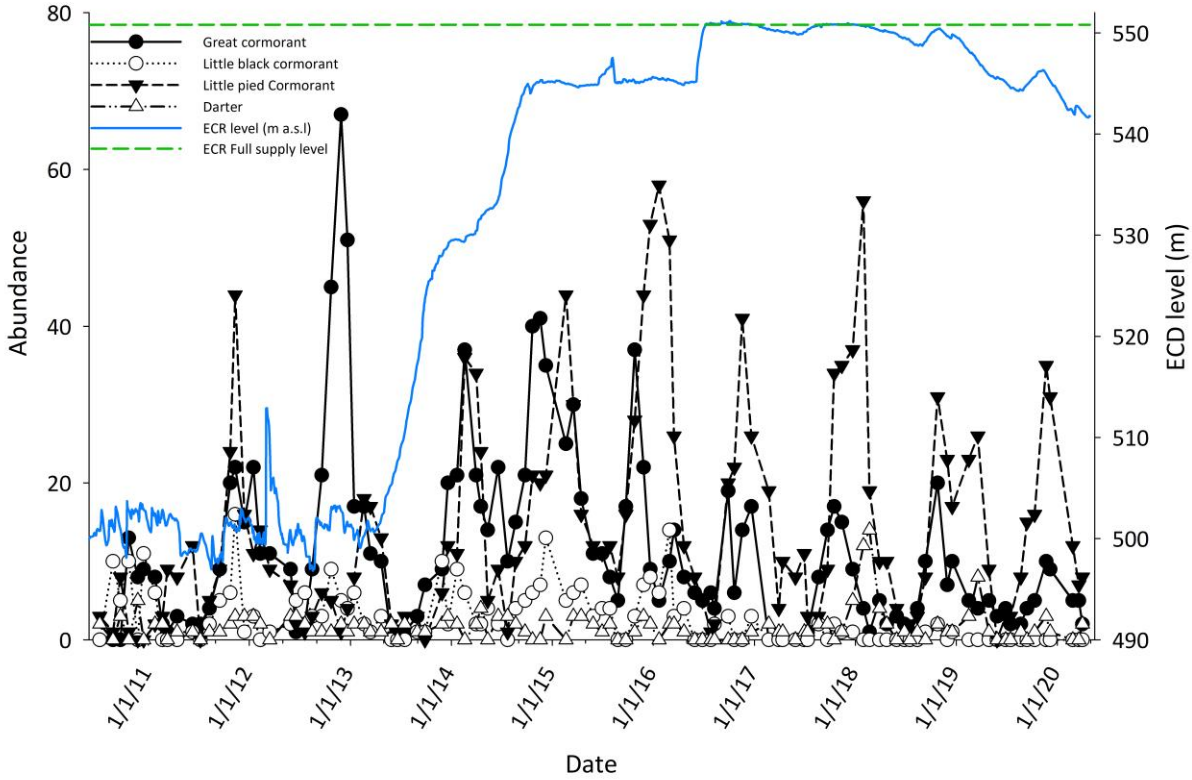

Figure 28. Monthly abundances of each common piscivorous bird species on the Cotter Reservoir between July 2010 and April 2020 with ECR reservoir level (m a.s.l) shown in blue and ECR full supply level shown in dashed green.

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 19). Pairwise comparisons indicate these significant phase x section interaction differences are among baseline versus filling and operational phase for section 1 and section 4 ($p < 0.05$).

Table 19. Results of PERMANOVA analysis of piscivorous bird community composition in Cotter Reservoir from 2010 – 2020 (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**	**47.404**	**23.702**	**15.107**	**< 0.001**	**9939**
Section	**4**	**37.068**	**9.2669**	**15.884**	**<0.001**	**9940**
Year (within Phase)	**8**	**13.141**	**1.6427**	**2.8157**	**<0.001**	**9919**
Section x Phase	**8**	**48.33**	**6.0413**	**10.355**	**<0.001**	**9920**
Residuals	722	421.21	0.5834			
Total	744	577.77				

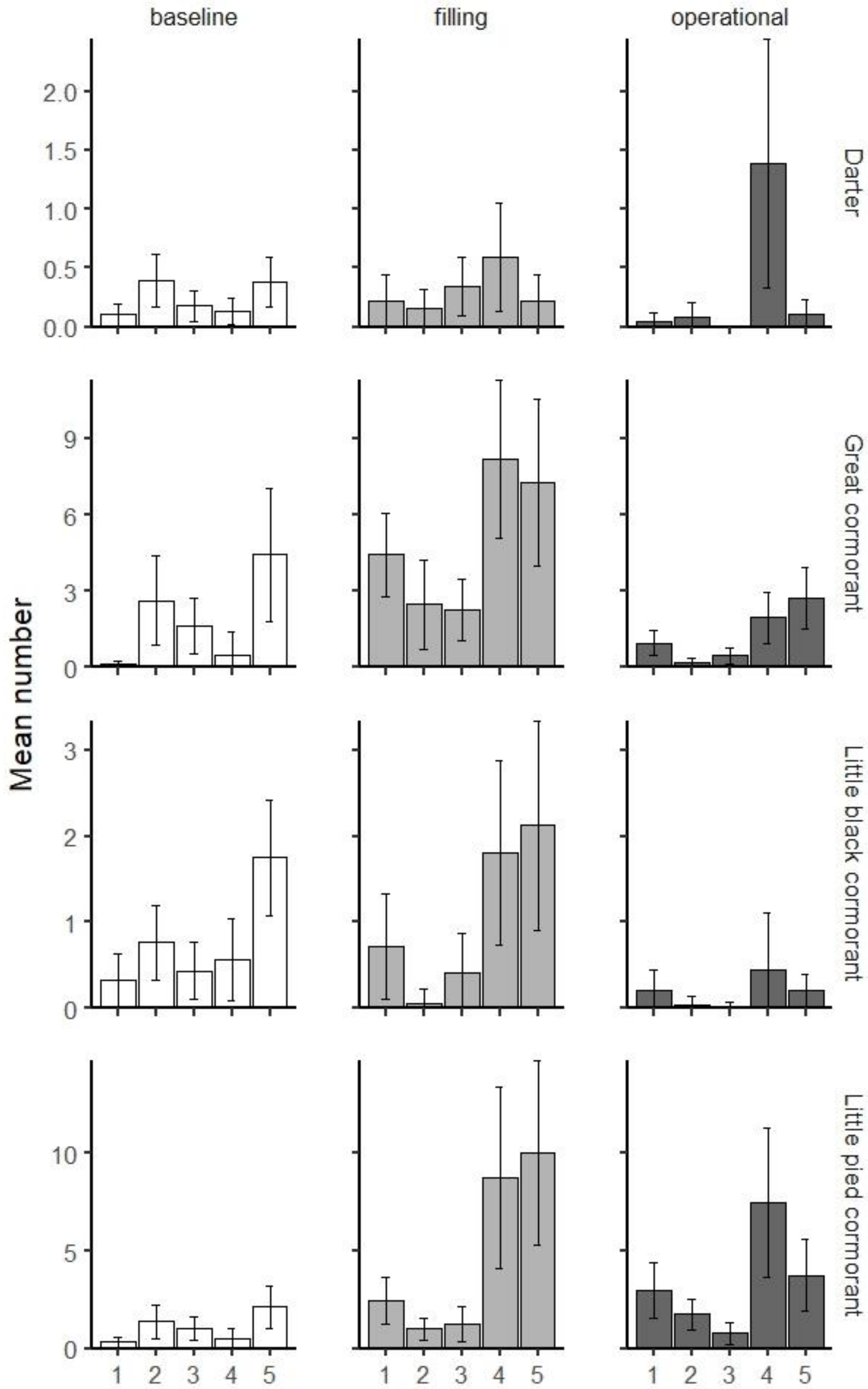

Figure 29. Mean monthly abundance (displayed as mean ± 95% confidence limits with Bonferroni correction) of each piscivorous bird species in each section of Cotter Reservoir for baseline phase (July 2010 – March 2013), filling phase (April 2013 – December 2015) and operational phase (January 2016 - April 2020) of monitoring program.

For the seventh 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 30), though a new 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, though the colonies also contained Darters.

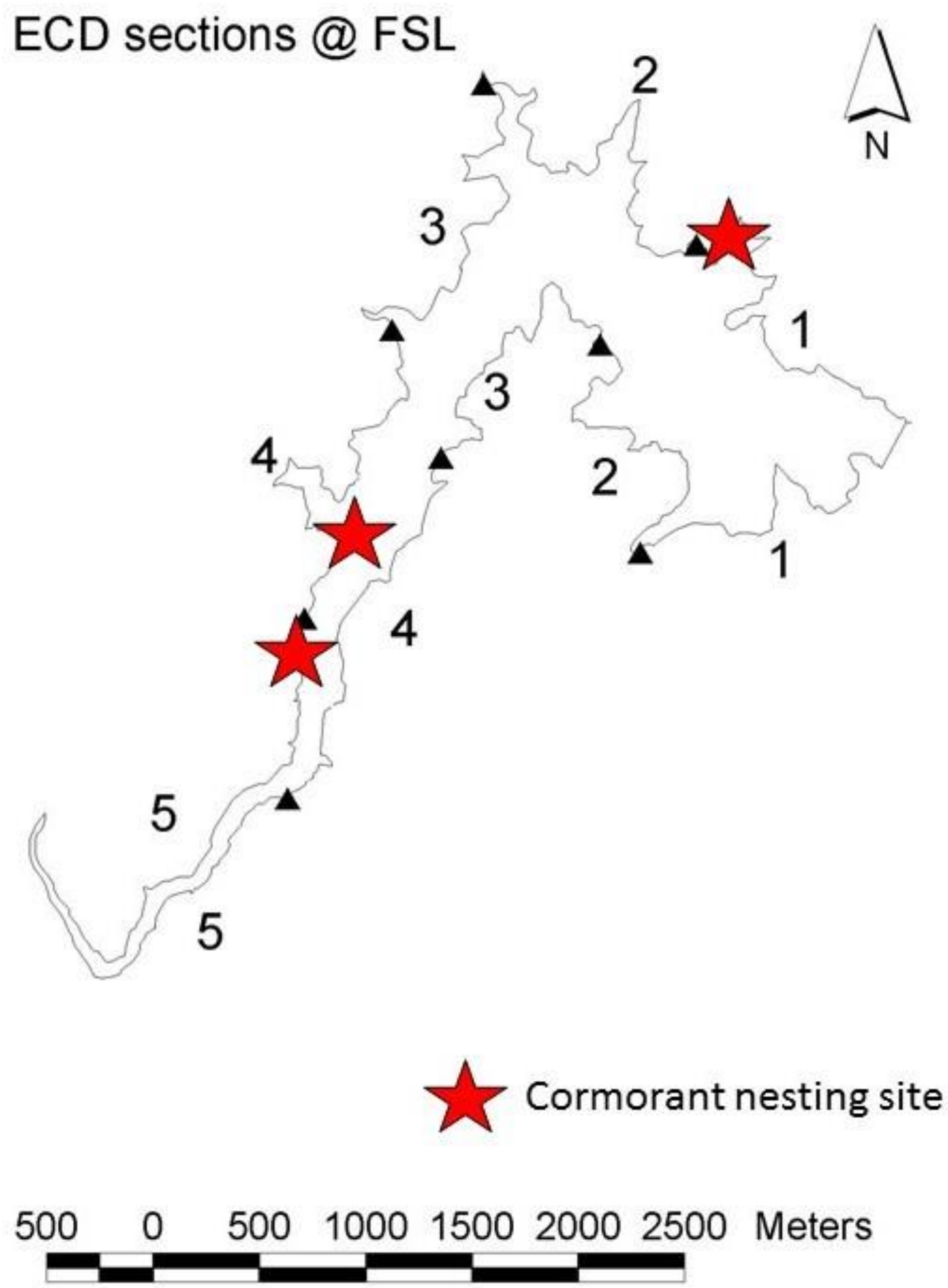

Figure 30. Map of the enlarged Cotter Reservoir shoreline sections with Cormorant nesting colony locations.

DISCUSSION AND CONCLUSIONS

Abundance

Since filling began, peak abundances (see Figure 32) of both Great cormorant and Little black cormorant have been stable, though with some definite seasonal fluctuations. Abundances of Little pied cormorant during warmer months has been increasing annually 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 operation 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 seven years since filling began and its seasonal peaks during the warmer months were increasing, though appear to be subsiding each year since 2017. Cormorants are opportunistic and nomadic, responding to 'boom' conditions, and will breed if resources are sufficient (Kingsford *et al.* 1999, Dorfman and Kingsford 2001b). A 'boom' in food resources in the ECR and the presence of emergent flooded trees has resulted in the establishment of a breeding population of cormorants. 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) potentially associated with declines in Goldfish abundance suggest 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.

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 2001a, 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 30) 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, 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?

NOTE: Formal monitoring for this question has not yet commenced as the reservoir only filled for the first time in July 2016 and no macrophytes have been observed whilst conducting other work around the perimeter of the reservoir.

Below are the proposed monitoring methods as outlined in the monitoring proposal.

BACKGROUND

Existing macrophyte beds in Cotter Reservoir have been demonstrated to provide important resting habitat for adult Macquarie perch (Ebner and Lintermans 2007). It is certain that existing macrophyte beds will be drowned by up to 50 m of water once the reservoir has filled. Modelling indicates that the reservoir will 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 (unpublished data) 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 20. 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	Conducted monthly during spring and summer.
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). Monthly 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. 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

N/A

DISCUSSION AND CONCLUSIONS

Continued draw down and fluctuating water levels (as experienced since February 2018) are unlikely to facilitate the establishment of macrophytes at this stage.

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

BACKGROUND

It was expected that as the ECR filled and became operational the food resources of Macquarie perch were likely to change. The substantial beds of emergent macrophytes that fringed the old Cotter Reservoir have been submerged, and the fluctuating water levels of an operational reservoir may prevent their reestablishment (Lintermans 2012). These reed beds supported significant densities of decapod crustaceans, particularly freshwater prawn (*Macrobrachium*) and shrimp (*Paratya*) which are favoured food items for Macquarie perch (Norris *et al.* 2012). During and following inundation a trophic upsurge is expected where food resources of Macquarie perch are plentiful and Macquarie perch will have increased body condition (as experienced in other newly filled reservoirs such as Lake Dartmouth). As the reservoir ages, it is 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 21. 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). Principal Component analysis ordinations (PCO) of square-root 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 square root transformed and a Bray-Curtis measure used for resemblance matrix. PERMANOVA analysis consisted of Phase, season and habitat as fixed factors. Highest interaction term 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.

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 22). All sampling phases were significantly different from each other in coarse pick edge sample compositions. PCO revealed that this difference between phases of coarse pick samples was largely driven by higher abundances of terrestrial items, Hemitera and Diptera in filling and operational phases (Figure 31). Decapod abundances have increased since Spring 2018 after being low during early operational years (Figure 32).

Table 22. Results of PERMANOVA analysis of coarse pick macroinvertebrate community composition in Cotter Reservoir from 2010 – 2020 (bold text indicates statistically significant difference at the 0.05 level).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Phase	**2**	**22847**	**11424**	**8.5117**	**<0.001**	**9929**
Season	**1**	**5872.9**	**5872.9**	**7.2472**	**<0.001**	**60**
Habitat (season)	4	2733.7	683.43	0.50921	0.943	9922
Phase x Season	**2**	**6376.3**	**3188.2**	**2.3755**	**0.02**	**9937**
Residuals	58	77843	1342.1			
Total	67	116730				

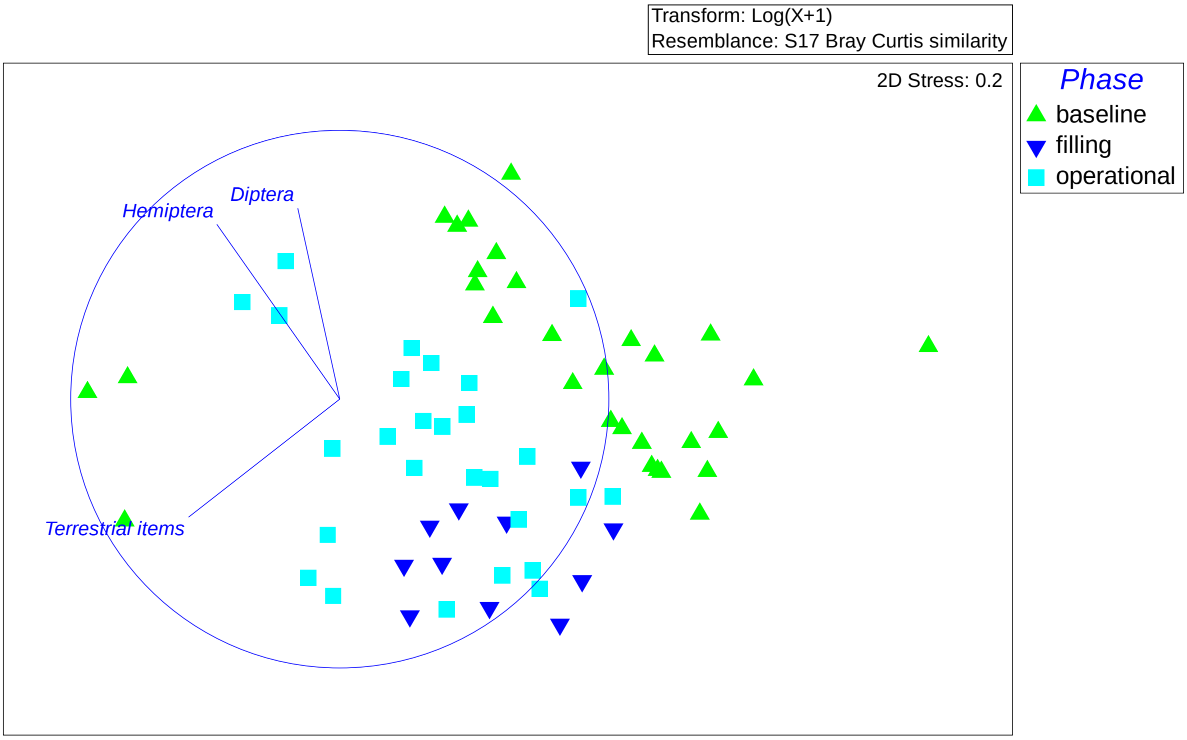

Figure 31. Graphical representation of a principal component analysis ordination of invertebrates from the coarse pick from spring and autumn monitoring in pre-filling (baseline) (data from Norris et al. 2012), filling phase (2013–2015) and operational phase (2016–2020).

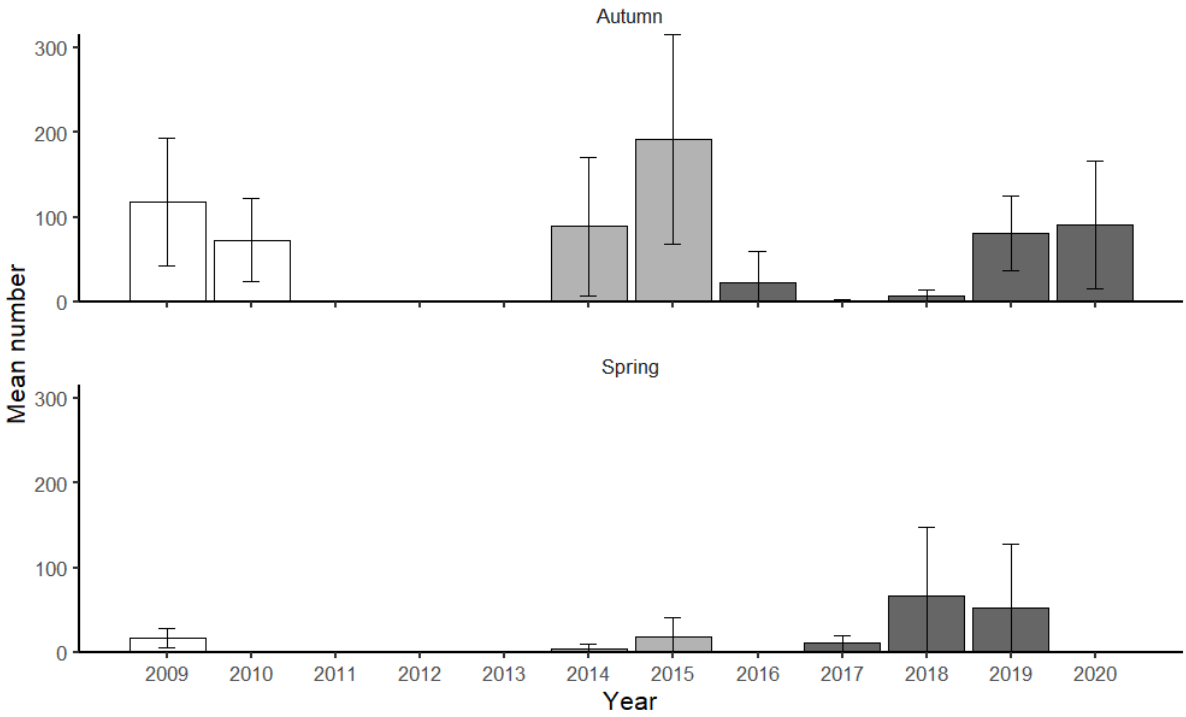

Figure 32. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of decapods collected from coarse pick edge samples taken from ECR during baseline (Pre-filling, 2009 / 2010; white bars) (Norris et al. 2012), filling (2013 – 2015; light grey bars) and operational (2016 – 2020; dark grey bars) monitoring periods for autumn and spring. Note: Spring 2020 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, but no significant effect of habitat and a significant phase x season interaction (Table 23). 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 Decapods (especially 2010) and Oligochaetes in the baseline samples (especially 2010) (Figure 33).

Table 23. Results of PERMANOVA analysis of 10% subsample macroinvertebrate community composition in Cotter Reservoir from 2009-2010 (baseline) & 2013 – 2015 (Filling) and 2016 – 2020 (Operational) (bold text indicates statistically significant difference at the 0.05 level).

Phase	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
Phase	**2**	**24158**	**12079**	**8.0818**	**<0.001**	**9937**
Season	**1**	**13578**	**13578**	**11.813**	**<0.001**	**360**
Habitat	4	4322.8	1080.7	0.72305	0.867	9864
Phase x Season	**2**	**13739**	**6869.4**	**4.5961**	**<0.001**	**9922**
Res	65	100140	1494.6			
Total	74	154650				

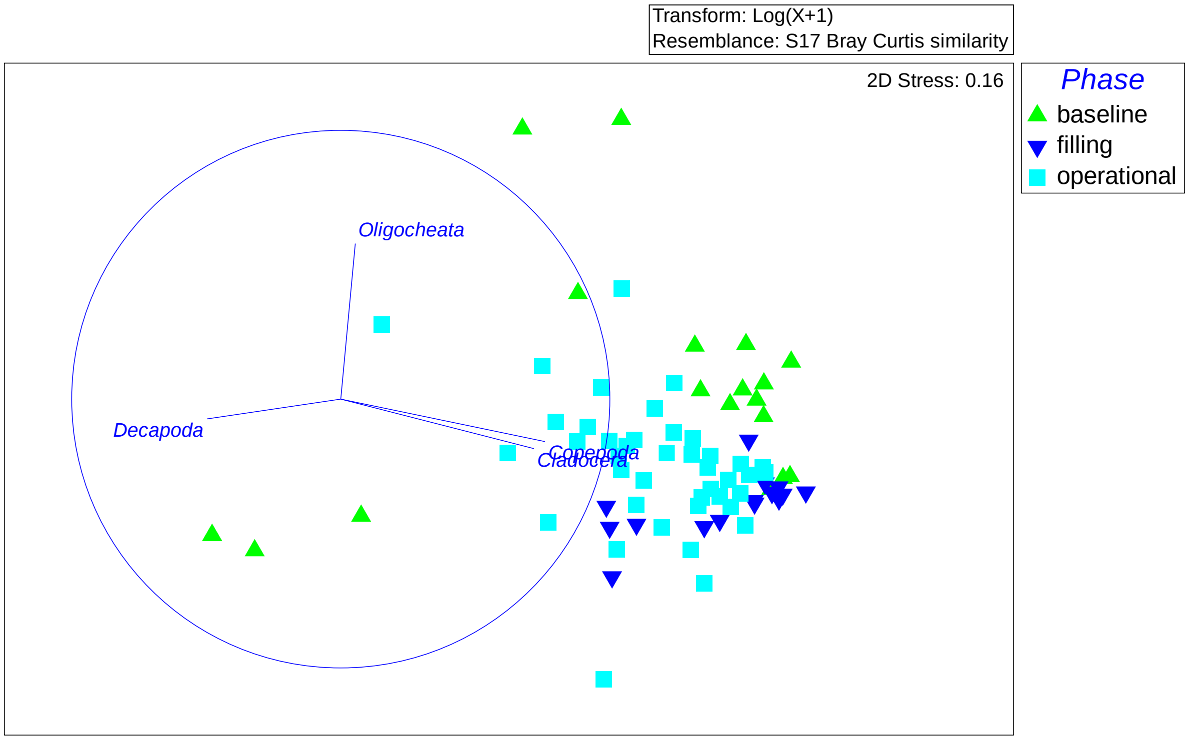

Figure 33. Graphical representation of a principal component analysis ordination of invertebrates from the 10% sub-sample processing from spring and autumn monitoring in baseline (data from Norris et al. 2012), filling phase (2013–2015) and operational phase (2016–2020).

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. There has been slightly contrasting dynamics between taxa between phases and seasons. Cladocerans have slightly increased across phases in spring, but have decreased across phases in autumn, though these differences are not significant due to large confidence intervals (Figure 34). For example, Cladocera were lowest in autumn 2016 and spring 2017, with mean abundances (± SE) of 29 ± 10 and 76 ± 4, respectively. In contrast mean abundances in the same phase were as high as 1800 ± 182 in spring 2017. Mean Copepoda abundance has decreased slightly in spring across monitoring phase, but was highest in operational phase monitoring for autumn (Figure 34).

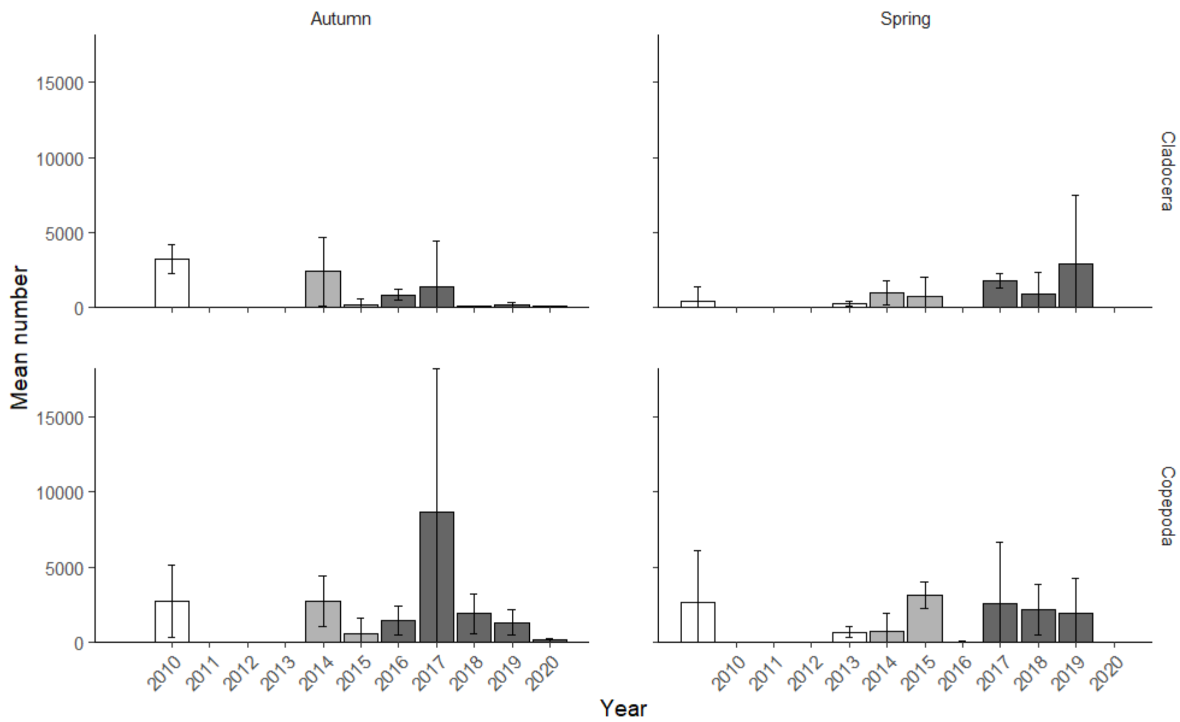

Figure 34. Relative abundance (mean ± 95% confidence limits with Bonferroni corrections) of each microcrustacean taxa collected in autumn and spring of each phase of monitoring phase in Cotter Reservoir using tow nets. White bars indicates baseline phase, light grey bars indicate filling phase and dark grey bars indicate operational phase. Note: Spring 2020 had not been collected at time of reporting.

DISCUSSION AND CONCLUSIONS

Decapod abundances in coarse pick edge samples are similar between baseline and filling phase monitoring, and across seasons. Decapod abundance in autumn during operational phase monitoring was much lower than that observed for autumn in baseline and filling phase monitoring. Decapod abundance was much lower in spring than autumn during both baseline and filling phase monitoring but was similar for operational phase monitoring. Despite being in very low abundances in early years of the operational phase, decapod abundance has been increasing since spring 2018, with decapod abundances now similar to that of baseline and filling phase monitoring (see Figure 32). 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).

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) become inundated, terrestrial invertebrates will enter 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 appeared to be a general decrease in Cladocerans abundance during autumn sampling (but not during spring sampling), mainly driven by exceedingly low abundances in autumn 2018, 2019 and 2020. 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 a reservoir was filling (Hatton 2016). Population abundance of Cladocerans 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). As the reservoir filled in 2016 and has been slightly receding from early 2018 onwards, resource availability (nutrients and associated algal biomass, the primary food of Cladocerans) is likely to have also reduced which may have had a negative effect on Cladoceran abundance. The same processes have not appeared to affect the relatively stable population of Copepods in the reservoir. As resources associated with filling reside further over the coming years, it is likely that reductions in Cladocerans and potentially Copepods may occur.

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, 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 or juveniles. No management intervention is recommended.

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