Assessing Murray Crayfish populations in Northern Victoria

Comparing population parameters in 2016 and 2023

S. Raymond, N. Whiterod,
S. Zukowski and B. Fanson

December 2023

Arthur Rylah Institute for Environmental Research
Unpublished Client Report

Arthur Rylah Institute for environmental research

Victorian Fisheries AUTHORITY

VICTORIA State Government | Energy, Environment and Climate Action

Acknowledgment

We acknowledge and respect Victorian Traditional Owners as the original custodians of Victoria's land and waters, their unique ability to care for Country and deep spiritual connection to it. We honour Elders past and present whose knowledge and wisdom has ensured the continuation of culture and traditional practices.

We are committed to genuinely partner, and meaningfully engage, with Victoria's Traditional Owners and Aboriginal communities to support the protection of Country, the maintenance of spiritual and cultural practices and their broader aspirations in the 21st century and beyond.

Arthur Rylah Institute for Environmental Research
Department of Energy, Environment and Climate Action
PO Box 137
Heidelberg, Victoria 3084
Phone (03) 9450 8600
Website: www.ari.vic.gov.au

Citation: Raymond, S., Whiterod, N., Zukowski, S. and Fanson, B (2023). Assessing Murray Crayfish populations in Northern Victoria. Unpublished Client Report for the Victorian Fisheries Authority. Arthur Rylah Institute for Environmental Research, Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

Front cover photo: Lower Goulburn River (Nick Whiterod)

Assessing Murray Crayfish populations in Northern Victoria

Comparing population parameters in 2016 and 2023

Scott Raymond[1], Nick Whiterod[2], Sylvia Zukowski[2] and Ben Fanson[1]

[1]Arthur Rylah Institute for Environmental Research
123 Brown Street, Heidelberg, Victoria 3084

[2]Nature Glenelg Trust, Victor Harbor, South Australia 5211

Arthur Rylah Institute for Environmental Research
Unpublished Client Report for Victorian Fisheries Authority, Department of Energy, Environment and Climate Action

Arthur Rylah Institute for Environmental Research
Department of Energy, Environment and Climate Action
Heidelberg, Victoria

Acknowledgements

The Victorian Fisheries Authority is thanked for making staff available to assist with queries arising during sampling and for financially supporting Victorian Murray Crayfish research. We would also like to thank Parks Victoria providing access to field sites. Zeb Tonkin (Arthur Rylah Institute, ARI) is thanked for reviewing an earlier draft of this report. Scott Huntley and Cory Young from Nature Glenelg Trust are thanked for their assistance with the data collection.

Contents

Tables

Figures

Summary

Context:

In 2014-2016, the Nature Glenelg Trust (NGT) undertook an assessment of Murray Crayfish (*Euastacus armatus*) populations within five river systems across Northern Victoria.

In 2023, the Victorian Fisheries Authority (VFA) commissioned the Arthur Rylah Institute (ARI) to repeat surveys of Murray Crayfish populations at 17 locations in North-eastern Victorian waterways. An analysis of crayfish relative abundance, size structure and sex ratios were undertaken to assess changes in these population parameters over the 7-9-year study.

Aims:

This report aimed to re-sample 17 of 41 sites outlined in Whiterod et al. (2017) for Murray Crayfish, and to compare population parameters (crayfish abundance, size structure and sex ratios) from data collected in 2014-2016 with 2023 sample data.

Methods:

Seventeen sites (each 2 km in length) were sampled for Murray Crayfish using baited drop pots. Twenty pots were set approximately 100 m apart at each site, with pots checked twice, at hourly intervals.

Results:

Murray Crayfish abundance, size structure and sex ratios were variable across sites and years, with no statistically significant difference in parameters between survey years.

Conclusions and implications:

Our findings highlight the variability in assessing Murray Crayfish populations across broad spatial scales (10s to 100s of km), suggesting a need for (1) targeted sampling of individual populations over relevant spatial scales (sites of 200-500 m in river length), and (2) projects to consider more robust sampling (e.g. capture-mark-recapture studies) designed to provide estimates of crayfish densities, catch rates, reproductive value, and movement patterns. Long-term (10+ years) monitoring of key populations is recommended to provide baseline data, and establish population trends, to assess anthropogenic (e.g. harvest), and environmentally driven effects (e.g. drought, blackwater, habitat modification) on crayfish populations.

1 Introduction

Freshwater crayfish are imperilled globally, with almost one third of species classified as threatened and at risk of extinction (Richman et al., 2015). In Australia, crayfish are exposed to a range of threats driven by the regulation of river systems for water security, energy production, agricultural pursuits (Walker & Thoms, 1993; Maheshwari et al., 1995; Vorosmarty, 2010) and from harvest. Many crayfish fisheries are vulnerable to over-exploitation, with concomitant adverse effects on freshwater ecosystems and on the persistence of populations (Gilligan et al., 2007). Harvest effects within and among crayfish populations are often expressed through changes in crayfish abundance, size structure and sex ratio (Todd et al., 2018; Raymond et al., 2023). To effectively manage and conserve freshwater crayfish, it is critical to 1) employ population indices that adequately reflect harvest related effects and recovery from impact, 2) monitor populations over suitable temporal-spatial scales, and 3) communicate findings to waterway managers, regulatory bodies, fishers and to the public. In this study, we assessed the status of Murray Crayfish (*Euastacus armatus* von Martens, 1866) at 17 sites across five rivers in north-eastern Victoria over a 7-9-year period (Whiterod et al., 2017), using estimates of relative abundance, combined with data on crayfish size-structure and sex ratios.

Threats

In south-eastern Australia, crayfish face a range of threats driven by the anthropogenic regulation of river systems that support an extensive irrigation network for crops and livestock (Maheshwari et al., 1995; Dudgeon et al., 2006). These highly regulated systems have largely reversed historically high winter/ spring flows (now stored in impoundments) and low summer flows, to meet the increasing demands of irrigators. Modified flow regimes have had concomitant adverse effects on water temperature, stream productivity and habitat availability (Walker & Thoms, 1993), all which crayfish are dependent on for movement, breeding, and survival. Further, the removal of woody debris (snags) and channelisation of rivers to promote navigation, replacement of vast tracts of lowland forests with grasslands, and modified riparian zones has led to increasing runoff, erosion and sedimentation of habitats supporting freshwater crayfish. Further, climate change is predicted to increase the frequency and duration of extreme weather events (e.g. droughts, fires, and floods), placing additional pressure on crayfish (among other aquatic biota) in freshwater habitats (van Dijk et al., 2013). For crayfish, these threats are occurring on a backdrop of harvest, reported to alter crayfish distribution, density, size structure and population sex ratio (Gilligan et al., 2007; Zukowski et al., 2012; Whiterod et al., 2018, 2020; Raymond & Todd 2020).

Indices and harvest effects

Effective management of fisheries resources relies heavily on the use of indices that adequately reflect population trends (Downes et al., 2002). While abundance data has been useful for estimating and comparing population sizes over time and among studies (Pollock et al., 1991; Meynecke & Richards 2014), indices that explore the structure of populations (e.g. size structure and sex ratios) provide a more informative assessment of harvest effects through highlighting the mechanisms driving population dynamics (Raymond et al., 2023). For example, many harvest strategies focus on the selective removal of individuals based on their size and sex, with over-exploitation leading to increasing disparity in the structure and functioning of populations (Sato & Goshima 2006, 2007; Sato & Yoseda 2010). Disparity in size structure and sex ratios are linked with population truncation, modified behaviour of individuals, declining reproductive success, and population extirpation, with potentially severe evolutionary consequences (Burgman et al., 1993; MacDiarmid & Butler, 1999; Allendorf & Hard, 2009; Yorisue et al., 2020). In the absence of harvest, Murray Crayfish populations displayed a 1 to 1 sex ratio and a uniform size frequency distribution (Zukowski et al., 2013).

Harvest regulations

Concerns over the long-term sustainability of Murray Crayfish populations have led to the development and implementation of strict harvest regulations (New South Wales Department of Primary Industries 2022; Victorian Fisheries Authority, 2022). Harvest regulations include regional and seasonal closures (1 September to 31 May), bag limit of 2/person/day (maximum of 4 in possession), gear limits (five drop-pots/person/day), a ban on the removal of brooding females (females with eggs), and a Harvestable Slot Legal Limit (HSLL, 100-120mm Occipital Carapace Length, OCL). However, there is no limitation on the number of anglers participating in the fishery and no obligation to report catch data, so the numbers of fishers and the number of crayfish harvested is unknown (McPhee et al., 2002).

Murray Crayfish

Murray Crayfish is the second largest freshwater crayfish in the world (Riek, 1971), endemic to the freshwaters of the Murray-Darling Basin of south-eastern Australia. The species has high ecological, social, and cultural value (Noble et al., 2018; Todd et al., 2018) and despite their ‘threatened’ listing, remain subject to harvest for human consumption (Coughran & Furse, 2012; Yorisue et al., 2020). The species is slow growing, late maturing (8-9 years, Todd et al., 2018), has low fecundity (between ~300 and ~1495 eggs (Johnson and Barlow 1982), and limited mobility (Ryan, 2005; Whiterod et al., 2017). These life-history traits make the species vulnerable to harvest effects and are typically slow to recover from population declines driven by episodic events (King et al., 2012; Whiterod et al. 2018) and from harvest over-exploitation (Noble & Fulton 2017; Raymond et al., 2023).

Ecological impacts

Crayfish are integral components of many aquatic ecosystems, playing an important role in the structure and functioning of freshwater environments through their feeding and movement habits. Crayfish mediate the transfer of energy through aquatic food webs via their polytrophic feeding habitats that include the consumption of decaying wood and leaf litter, invertebrates, and small fish (Usio & Townsend, 2004) and are regarded as ‘ecosystem engineers’ through their actions of walking, burrowing, and moving instream wood (Jones et al., 1994; Creed & Reed, 2004). Murray Crayfish has suffered large declines in range and abundance since the 1950’s, with further declines expected to increase the species’ risk of extinction and disrupt key ecological processes, fundamental to lentic morphology, and community form and function.

Study objectives

Here, we compare the abundance, size structure and sex ratio of 17 Murray Crayfish populations between surveys conducted in 2014-2016 (hereafter ‘2016’) with surveys undertaken in 2023. Based on the biology (e.g. slow growth, low fecundity) and ecology (e.g. poor dispersal) of the species, continuing harvest pressure and reported harvest effects (declining abundance, disparate size structure and sex ratios) and the location of river reaches sampled (previously fished), we hypothesize that:

H1: the abundance of Murray Crayfish will decline from 2016 to 2023,

H2: the size structure will become increasingly disparate from 2016 to 2023, and

H3: sex ratios will become increasingly disparate from 2016 to 2023.

2 Methods

2.1 Site locations and experimental design

Seventeen sites across the Victorian lowland range of the species were sampled for Murray Crayfish. Sites were a subset of those sampled by Whiterod et al. (2017) in 2016 (Figure 1, Table 1). Sample sites were approximately 2 km in length, and 20 drop pots were deployed at 100 m intervals throughout a reach. Drop pots (single 800mm diameter, 13mm stretch mesh size, 0.3m drop baited) were baited with ox liver (~300 g), set 5-10 m from the bank, and checked three times at hourly intervals (60 net retrievals at each site). Drop pots were identified as per VFA regulations (Victorian Fisheries Authority, 2022). Crayfish sampled in the first two pot checks were marked (Whiterod et al., 2017) to detect re-captures. All Murray Crayfish populations surveyed were located within zones open to harvest.

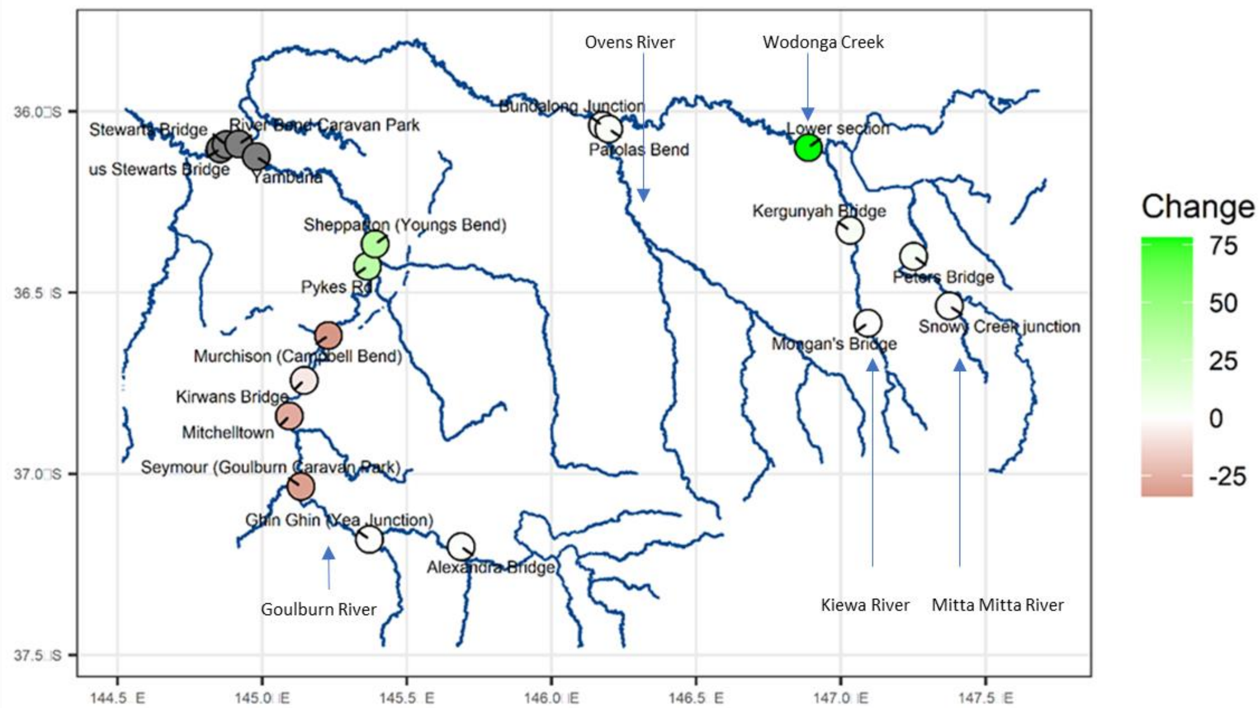

Figure 1. Map of Murray Crayfish survey sites sampled in 2014-2016 and 2023. Colours depict the percentage change (+/-) in the total crayfish catch at each site between survey years. Grey dots indicate sites not sampled in both years.

Table 1. Summary of survey location and experimental design.

River	Site	Number of lifts	Number of pots	Number of crayfish	
				2016	2023
Goulburn River	Alexandra Bridge	3	20	0	2
	Ghin Ghin (Yea Junction)	3	20	2	3
	Kirwans Bridge	3	20	15	8
	Mitchelltown	3	20	62	35
	Murchison (Campbell Bend)	3	20	66	32
	Pykes Rd	3	20	1	35
	Seymour (Goulburn Caravan Park)	3	20	44	13
	Shepparton (Youngs Bend)	3	20	4	40
	Yambuna	3	20	0	NS
	Riverbend Caravan Park	3	20	0	NS
	Stewart's Bridge	3	20	NS	0
Kiewa River	Kergunyah Bridge	3	20	4	8
	Mongan's Bridge	3	20	1	1
Mitta Mitta River	Peters Bridge	3	20	2	6
	Snowy Creek junction	3	20	1	0
Ovens River	Bundalong Junction	3	20	37	37
	Parolas Bend	3	20	58	61
Wodonga Creek	Lower section	3	20	34	112
Total catch				**331**	**393**

*NS =sites not sampled in both survey years.

2.2 Data analysis

Data analyses were performed using sample data from 15 sites; data from two of the 17 sites were excluded from the analysis as site locations varied between survey years. All analyses were performed using R V4.2 (R Core Team, 2022) using rstanarm (Goodrich et al., 2022) package. Default priors were used for all models. Models were checked for convergence using Gelman-Rubin convergence diagnostic ($\hat{r}<1.05$; see Appendix A.1). Posterior checks assessed model fit with simulated data (see Appendix A.2). We define significant differences based on if the 95% credible interval (95%CrI) overlaps with zero on the link scale.

For Catch Per Unit Effort (CPUE, hereafter 'relative abundance') data, we ran a Bayesian Generalised Mixed Model (bGLMM) with a negative binomial distribution. For the data we calculated the total catch per site per survey. We then included year and river as fixed effects. Site was included as a random effect.

The shape of length frequency (LF) curves provides information to assess changes in size-structuring of populations (with sex and year, Figure 4A), and may be useful for inferring factors (e.g. selective harvest, recovery from harvest) responsible for these changes. For length distribution, we ran two models: one comparing mean OCLs and second comparing proportion >120mm. For comparing mean OCL we ran a Gaussian bGLMM that included year and river as fixed effects and site as a random effect. For comparing proportions, we ran a binomial bGLMM with the same model structure as the mean OCL model.

For sex ratio, we modelled the proportion of catch that was female. We ran a binomial bGLMM. We broke up crayfish length (Occipital Carapace Length, OCL) into three size categories (<100mm, 100-120mm, >120mm) to represent crayfish below, within and above the HSLL, and then summarised the total number of females and males caught per OCL group per site per year. We included year and OCL group as well as the interaction. We included river as a fixed effect and site as a random effect.

3 Results

3.1 Catch

A total of 393 Murray Crayfish were captured in 2023, compared with the capture of 331 individuals in 2014-2016, representing a 15% increase in the total catch between the two survey periods. Half of all sample sites recorded<10 individuals. The mean length of male and female crayfish increased from 2016 to 2023 (Table 2, Figure 3). Sex ratios varied with size class. Relative abundance and size structure showed considerable variability within and among populations over the 7–9-year study. Sex ratios became increasingly disparate with increasing size, consistent across survey years.

Table 2. Mean, standard error (S.E.) and range of lengths (occipital carapace length, OCL) of Murray Crayfish sampled in 2014-2016 and 2023.

Year	Crayfish sex	Number sampled	OCL		
			Mean (mm)	S.E.	Range (mm)
2014-2016	Males	117	81.4	1.6	35-118
	Females with eggs	112	107.3	1.3	75-140
	Females without eggs	102	74.5	1.4	19-128
	Female total	214	91.7	1.5	19-140
	Total	**331**	**88.0**	**1.1**	**19-140**
2023	Males	151	87.6	1.1	54-135
	Females with eggs	116	108	1.3	76-149
	Females without eggs	126	81.2	0.8	59-110
	Female total	242	94.1	1.1	59-149
	Total	**393**	**91.6**	**0.8**	**54-149**

3.2 Relative abundance

Comparisons in relative abundance of crayfish between the sampling periods varied considerably across the sample sites (Figure 1, Table 2). Eight of sites showed no change in relative abundance, abundance increased in three sites and four sites declined (Figure 1). Of the eight sites where no change in relative abundance was detected, six were in the upper reaches of rivers. All four of the sites with declining relative abundance were in the mid-reaches of the Goulburn River. Populations with increased relative abundance were in the mid- to lower-reaches of the Goulburn and Ovens rivers. Very low numbers (n <10) of crayfish sampled from the Kiewa and Mitta Mitta rivers, across survey years, limited our capacity to draw useful trends using relative abundance data.

In 2016, the mean relative abundance across all sites was 9 (95%:1.9, 31.2) crayfish/survey compared to 13.1 (95%:3.3, 44.7) in 2023. There was not strong evidence for a decline in crayfish abundance from 2016 to 2023 (difference=48% (-39%,236%); 95%CrI overlapping 0%). However, the large credible interval indicates that there could be large changes hidden in the variation (Figure 2).

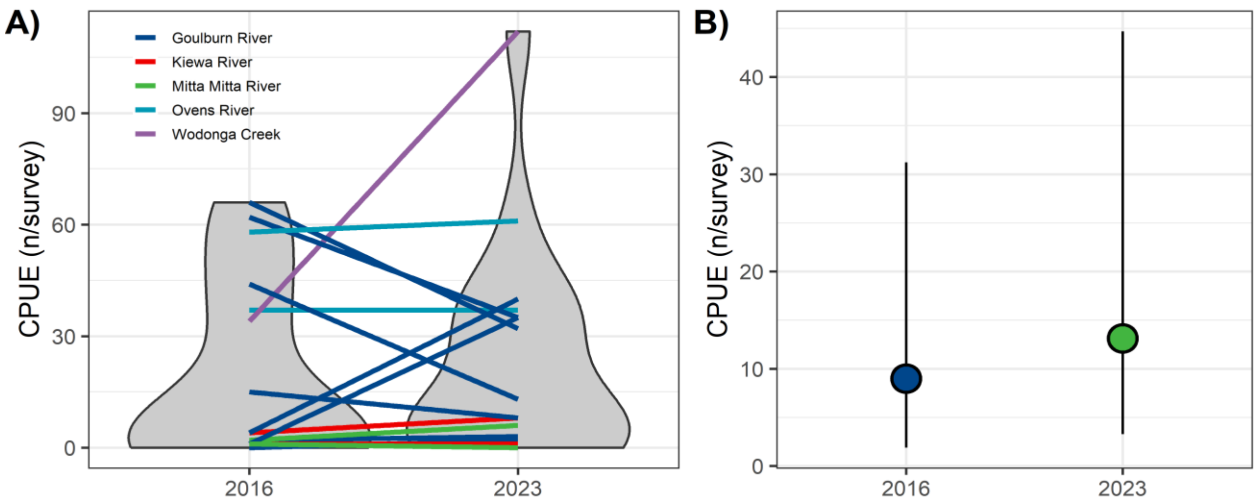

Figure 2.A) Violin plot showing the distribution of CPUEs (n crayfish/site). Overlaid lines show 2016 and 2023 CPUE data for each site with the colour of line indicating river. Thus, an increasing line indicates higher CPUE in 2023. B) CPUE estimates from model.

3.3 Length

Differences in length frequencies between study periods varied between sex and sites. Goulburn River length frequency curves show that females were larger than males in 2016, the mean length of males increased from 2016 to 2023, and that the size structure of males in 2023 sharply declined for individuals entering the HSLL (>100mm OCL). In contrast, length frequency curves of male and female crayfish in the Ovens River shifted to the right from 2016 to 2023, and that the proportion of large females and males increased to 2023, albeit at a greater rate in females. In Wodonga Creek, LF curves showed a clear increase in the proportion of females and males below the HSLL and a decline in smaller and larger individuals of both sexes, across survey years.

Average female length was 85.3mm (95%:78.9, 91.9) in 2016 and 89.5mm (95%:82.8, 95.5) in 2023 with 2023 females being significantly larger (diff=4.2mm (95%:0.8, 7.9)) (Figure 4B). Similarly, average male length was significantly larger (diff=7.7mm (95%:3.2, 12.1)) in 2023 (mean=83.7mm (95%:76.7, 90.1)) than in 2016 (mean =76.0mm (95%:68.9, 82.4); Figure 4B).

For both sexes, we found no changes in proportion of crays in 100-120mm (female diff=-4.1% (95%:-39.3%, 51%); male diff=13.2% (95%:-43.9%, 121%)) and >120mm (female diff=-6.7% (95%:-52.1%, 82%); male diff=320.8% (95%:-60.8%, 10 540%) ; Figure 4).

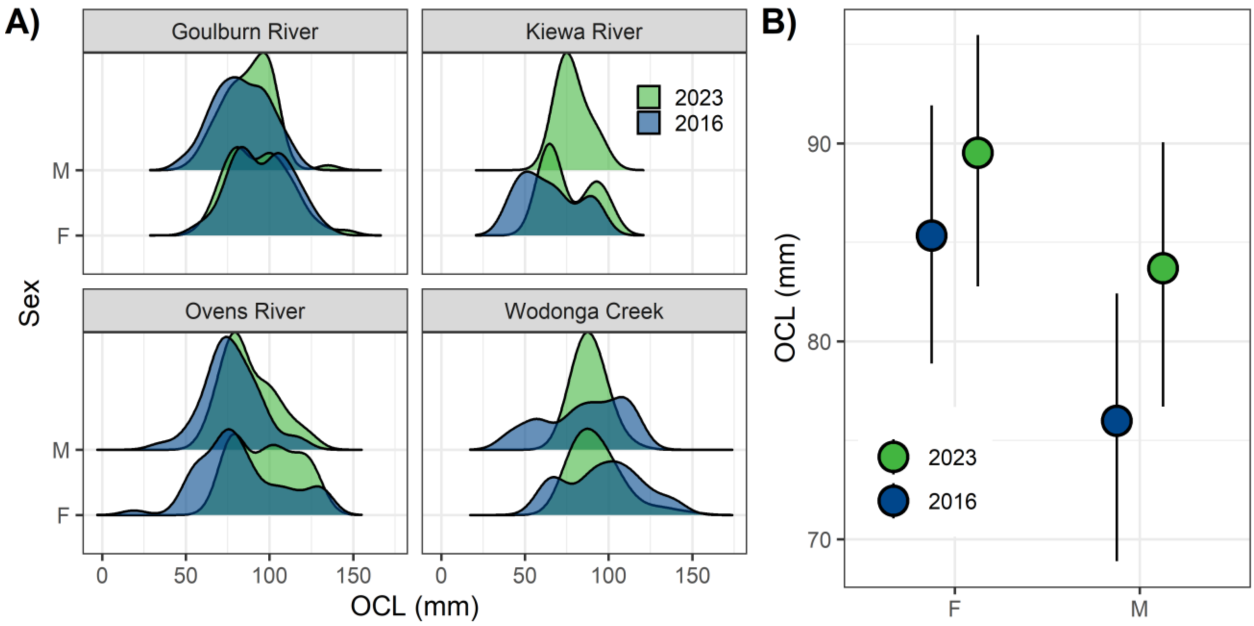

Figure 3. A) Density plots showing Occipital Carapace Length (OCL, mm) distributions for male and female crayfish broken up by river and year. B) Mean OCL estimates from model for 2016 and 2023. Error bars are 95%CrI.

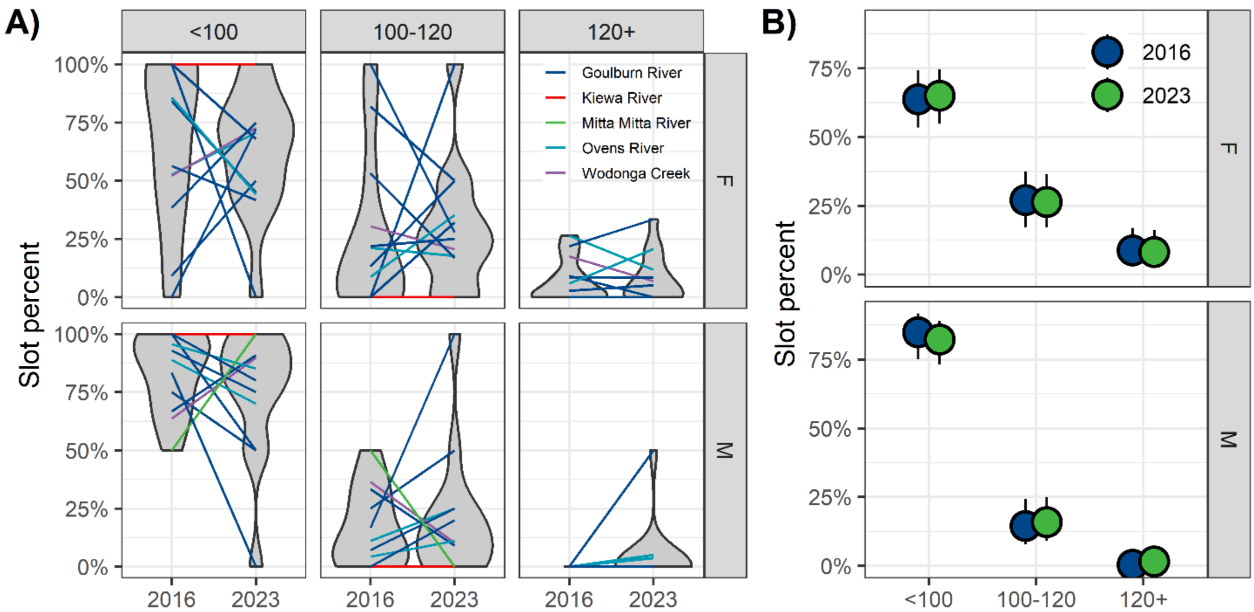

Figure 4. A) Violin plots showing percentage of cray in each slot group broken up by sex and year. Overlaid lines show 2016 and 2023 proportion data for each site with the colour of line indicating river. B) Proportion crayfish in each slot group. Error bars are 95%CrI.

3.4 Sex ratio

The proportion of the total catch that was female did not differ between 2016 and 2023 (averaged diff=-8% (95%:-34%, 26%), Figure 4). However, the proportion of the catch being female increased with OCL group length (Figure 4B). In general, sex ratios remained constant between years (total site catch) with high site variability. In the absence of exceptional circumstances (e.g. parthenogenesis) the ratio of females to males is expected to be 1:1 (Fisher, 1930; Zukowski et al. 2012).

Table 3. The sex ratio (female to male) of Murray Crayfish across survey years and size. Size cohorts represent crayfish below (<100mm Occipital Carapace Length, OCL), within (100-120mm) and above (>120mm) the Harvestable Slot Legal Limit.

Year	Number sampled			OCL		
	Female	Male	Total	<100mm	100-120mm	>120mm
2014-2016	214	117	1.8			
	129	99		1.3		
	61	18			3.4	
	24	0				>10
2023	242	151	1.6			
	156	124		1.3		
	64	24			2.7	
	22	3				7.3

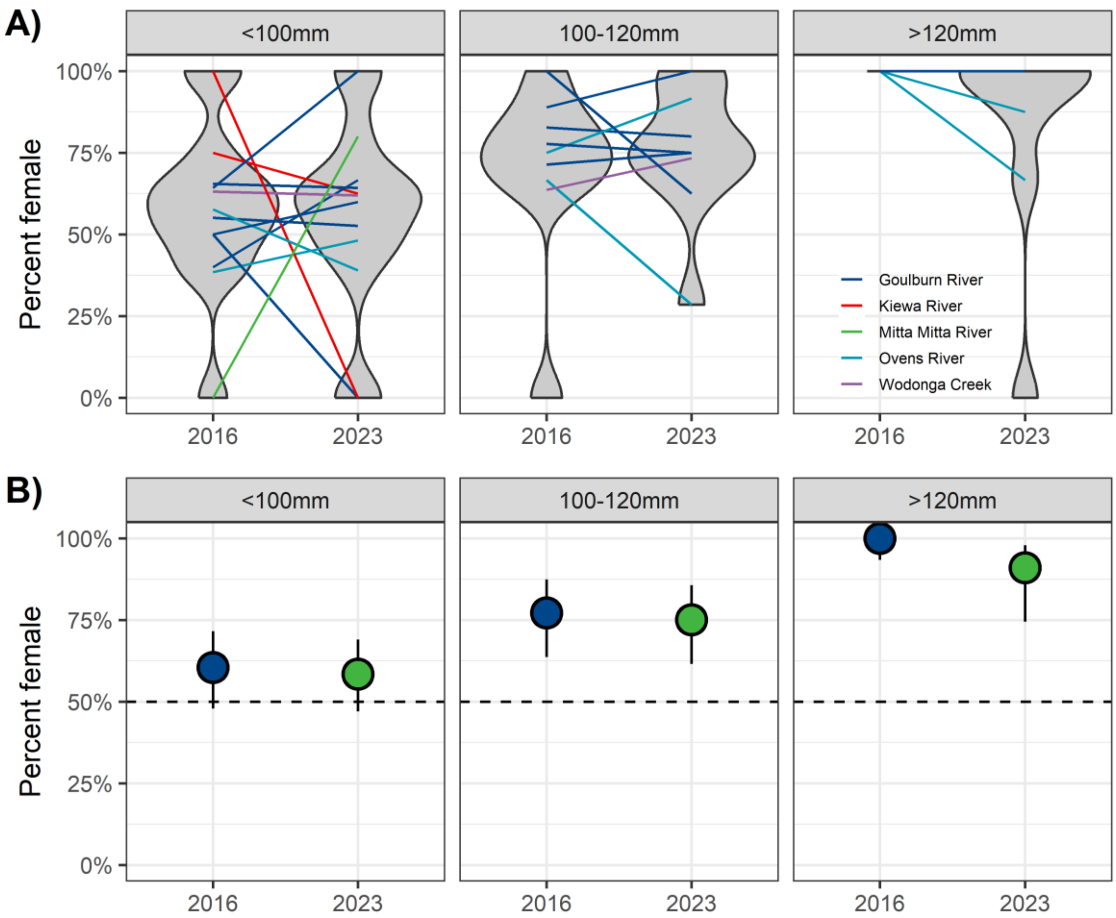

Figure 5. A) Violin plots showing the distribution of female proportion for each site. Overlaid lines show 2016 and 2023 proportion data for each site with line colour indicating river. Panels show results for each Occipital Carapace Length (OCL) group. B) Proportion of female estimates for each of the study years.

3.5 Berried females

Overall, longer females had higher odds of being berried (carrying eggs) (slope = 7% (95%:5%, 8%) increase in odds/mm; Figure 6), with the size at which half of the females were ‘berried’ (size at functional reproduction, SFR_{50}, Raymond & Todd 2020) calculated at ~98mm OCL.

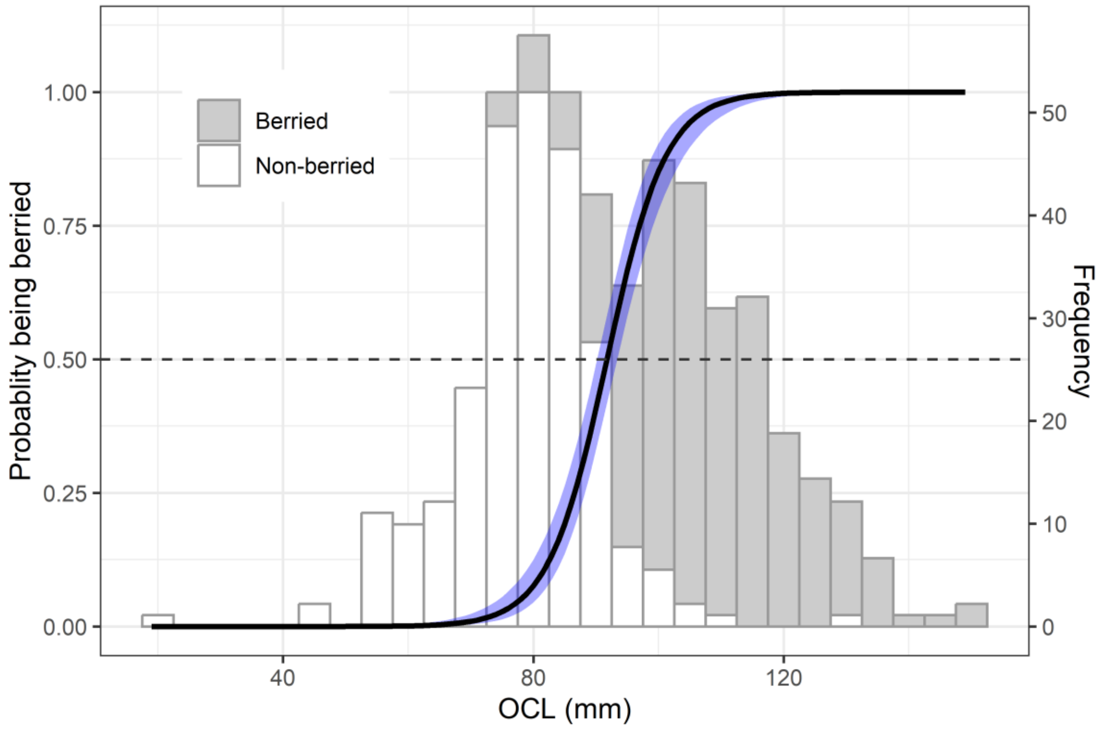

Figure 6. Probability of female Murray Crayfish being ‘berried’ (carrying eggs) in relation to Occipital Carapace Length. Curve shows predicted probability of being berried with blue shading indicating 95% confidence interval. Distribution of female size classes with fill indicating number of females ‘berried’.

4 Discussion

Comparison of Murray Crayfish surveys undertaken across north-eastern Victoria from 2016 to 2023, did not detect any significant difference in the relative abundance, size structure or sex ratios. While our three hypotheses were refuted, comparison of population parameters between the two survey periods spanning 7-9 years provided some useful information for assessing populations. However, large credible intervals around mean abundance, size, and sex data (within and across survey years) suggest that population-level changes were likely masked by highly variable catch rates, highlighting the need for more targeted research.

The 15% increase in total crayfish abundance over the 7-9-year study, although not significant, was driven by three populations where a > three-fold increases in individuals sampled was observed. While these three populations showed considerable improvement in catch rates, stable abundances in eight populations and declining abundances in four populations, reflected the variable nature of population responses across broad spatial scales (10s to 100s of km's). Further, low (< 10 individuals) catch rates in half of all surveyed populations adversely affected statistical power to detect change reliably and accurately. Inferences made from biased data, often associated with low catch rates limits any conclusions on the current state of a population or species (Lyon et al., 2014). While relative abundance data can be useful for assessing the effects of catastrophic episodic disturbance events (e.g. blackwater) over broad spatial scales (McCarthy et al., 2014; Whiterod et al. 2018), relative abundance data was limited in its ability to detect multiple population-level responses at broad spatial scales in the current study. At the population level, changes in abundance over the 7-9-year study may have been related to environmental conditions (e.g. flows, temperature, available habitat) and/or changes in harvest pressure, likely to be variable within and among river systems. In the absence of reported fisher catch data and reach scale habitat assessments, it was difficult to determine the potential factors responsible for the observed changes in crayfish populations.

The size structure (and changes therein) of Murray Crayfish populations, although not significant, was useful for inferring harvest related effects with indications of continuing harvest and of populations recovering from harvest. As expected, females were on average larger than males, likely driven by sex-selective harvest of males and the protection of brooding females (Victorian Fisheries Authority, 2022). While mean size was constant between study years, the declining disparity between crayfish size and sex suggests that harvest pressure may have moderated over the study. If harvest pressure had remained constant or increased, we would have expected to record continued declines in mean male size and an increasing disparity between male and female size, as more males would have been removed from the system. While there was no evidence to support a change in overall mean size (assessed using total catch data) between study years, mean size varied among sites. While some sites showed signs of recovery, signs of continuing harvest were also recorded, particularly for populations in larger river systems. For example, a sharp decline in males entering the HSLL (100-120mm OCL) in the Ovens River, suggest that harvest effects were greater in this mid- to lowland reach compared with the upper river reaches of the Mitta Mitta, Kiewa and Wodonga Creek systems. This is likely the result of lower river reaches supporting greater numbers and larger crayfish than upland reaches due to differences in reach characteristics (e.g. habitat availability, river size, water temperature). The preference of Murray Crayfish for river reaches with clay banks, complex structural habitat (e.g. large woody debris, deeper holes), and more reliable and larger flows (Whiterod et al., 2016) is more characteristic of larger systems than shallower, rocky upland systems. Consequently, we would expect that harvest pressure was greater in large than small systems, as fishers are more likely to catch legal sized crayfish in larger systems and are therefore more likely to target crayfish in lowland than upland rivers. In contrast, we would expect that the effects of harvest would be most pronounced in small populations, where the removal of small numbers of crayfish can cause disproportionately greater adverse effects.

The overall ratio of female to male crayfish was consistent across study years. As sex ratios are often used as an indicator of harvest (Solberg et al., 2005; Zukowski et al., 2013) consistency in this index suggests that harvest effects remained stable over the course of the study. However, disparity in sex ratio between size classes, representing crayfish below, within and above the HSLL, suggest that harvest effects were noticeable across study years. If harvest had declined, we would have expected that crayfish below the HSLL (equal sex ratio) in 2016 would have grown and entered the HSLL in 2023, with consistency in sex ratios between the two groups. Instead, we observed that the sex ratio of crayfish below and within the HSLL in 2023 was equivalent with 2016, suggesting that harvest effects were consistent across survey years and that harvest of select populations was ongoing. Further, the disparity in sex ratio between crayfish within (2:1 female:male) and above the HSLL (>7:1 female:male) is concerning as these ratios are indicative of populations subjected to intensive harvest pressure (Raymond et al., 2023b), with associated changes in behavior, reported to adversely affect sexual selection and increase aggressive encounters among conspecifics (Sato and Yoseda, 2010). The large variation in sex ratios within and among populations was likely influenced by low abundances of individuals captured, supporting the need for more robust studies focused at the population level.

The probability of females being ‘in berry’ (SFR_{50}) was estimated at approximately 98mm OCL, 9mm smaller than that reported by Raymond et. al. (2023), and 6mm greater than the size at onset of sexual maturity (SOM_{50}), on which the current HSLL was based (Zukowski et al., 2012). The HSLL was established to provide females at least one opportunity to produce offspring prior to entering the HSLL; a provision that was violated in this and other studies. This is concerning as large proportions (30-50%) of females within the HSLL have been recorded without eggs (Tonkin & Rourke, 2009; Raymond & Todd, 2020), indicating that the current HSLL may not provide all females with adequate protection from harvest, thereby inhibiting their opportunity to contribute offspring to the next generation. Given the unknown consequences of declining reproductive output on future populations, modelling the effects from declining reproductive output on population persistence is recommended.

Our study has highlighted several factors that may have adversely affected our ability to detect accurate changes in abundance, size-structure, and sex ratios of multiple Murray Crayfish populations over 100s of river km’s. Firstly, we were bound to using drop-pots to capture crayfish in 2023 to replicate capture methods used in 2016. Drop pots are reportedly biased toward the capture of larger individuals that dominate access to baits by forcing smaller individuals from pots and through restricting smaller individuals from walking onto pots (Fulton et al., 2017). Further, short occupancy rates (mean 387 s) and low sampling efficiency of hoop nets potentially bias capture data (Fulton et al., 2012). One solution is to use Munyana crab pots to sample crayfish. Unlike drop pots, Munyana nets are an enclosed structure with entry holes that largely restrict crayfish from escaping (dependent on mesh size), thereby increasing entrainment and capture rates. Once captured, Munyana nets provide crayfish with multiple locations to rest. Further, Munyana nets can be set overnight, when crayfish are most active, with higher catch rates (compared with daytime sets, Raymond et al., 2017), more representative of populations than low catch rates. Munyana nets must be set as per Raymond et al., (2023b) to avoid the entrainment of air-breathing mammals. Secondly, we assessed multiple populations over very broad temporal-spatial scales (10s to 100s of km’s, and 7-9-years). Considering the large variability in credible intervals, life-history characteristics of Murray Crayfish (e.g. slow growth, limited mobility), inherent variability between river systems and unreported harvest, we advocate for study designs targeted at the population level (see Raymond et al., 2023b) and for annual monitoring of target populations. Furthermore, we support the use of indices (e.g. density estimates in place of abundance data) and models (e.g. spatially explicit capture recapture, SECR models, see Efford, 2004, 2011, 2014) that 1) better represent population level impacts, 2) are more capable of detecting event driven change, and 3) provide additional (e.g. movement patterns) data, useful for assessing populations.

Future studies of targeted populations are crucial for managing and conserving Murray Crayfish and for assessing the factors (e.g. environmental, and anthropogenic) and mechanisms (e.g. recruitment failure) responsible for change. A comparison of gear types (e.g. drop pots and Munyana nets) and their relative catch rates would be informative for future studies. Further, modelling the effects of climate change (including its influence on the frequency and magnitude of disturbance events), harvest and handling on Murray Crayfish would provide further insight into the dynamics of populations. This would provide managers and conservationists with tools to assess change and the potential to mitigate change through regulatory modification, and through enhancing fisher, scientific and community education.

4.1 Conclusion

The current study provided useful insights into the general trends of Murray Crayfish populations, demonstrating there were not significant changes in relative abundance, size-structure, and sex ratios among populations between 2016 and 2023 across the 15 sites surveyed. Rather there was variable findings across the multiple indices assessed: some populations showing signs of recovery, but others no change or decline. Overall, the Murray Crayfish population across Victorian remains at risk, and sound management is necessary. Our findings highlight the need to design studies focusing on 1) appropriate spatial (1-10 km), and temporal (e.g. annual monitoring) scales, 2) the use of robust sampling gear (e.g. Munyana nets) to promote high catch rates, 3) study designs that provide estimates of density (e.g. capture-mark-recapture) and 4) employ mathematical models (e.g. SECR models) capable of accurately reflecting the state of populations. Implementation of these studies will provide more robust data to better inform the management of Murray Crayfish in Victoria.

References

Allendorf, F. W., & J. J. Hard. 2009. Human-induced evolution caused by unnatural selection through harvest of wild animals. Proceedings of the National Academy of Sciences **106,** 9987–9994. https://doi.org/10.1073/pnas.09010 69106

Baumgartner, L. J. 2007. Diet and feeding habits of predatory fishes upstream and downstream of a low-level weir. Journal of Fish Biology **70,** 879–894. https://doi.org/10. 1111/j.1095-8649.2007.01352.x

Burgman, M. A., Ferson. S., & H. R. Akçakaya. 1993. Risk Assessment in Conservation Biology, Chapman and Hall, London.

Coltman, D. W., O'Donoghue, P., Jorgenson, J. T., Hogg, J. T., Strobeck, C., & M. Festa-Bianchet. 2003. Undesirable evolutionary consequences of trophy hunting. Nature **426,** 655–658.

Coughran, J., & J. M. Furse. 2012. Conservation of freshwater crayfish in Australia. Crustacean Research 25–34. https://www.jstage.jst.go.jp/article/crust acea/Speci al2012/7/Special2012_KJ00008520117/_article.

Creed, R. P., & J. M. Reed. 2004. Ecosystem engineering by crayfish in a headwater stream community. Journal of the North American Benthological Society **23,** 224–236.

Downes, B., Barmuta, L., Fairweather, P., Faith, D., Keough, M., Lake, P., Mapstone, B.D., & G. Quinn. 2002. Monitoring Ecological Impacts: Concepts and Practice in Flowing Waters. Cambridge: Cambridge University Press. doi:10.1017/CBO9780511542015

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A., Soto, D., Stiassny, M. L. J., & C. A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological reviews **81,** 163-182. https://doi.org/10.1017/S1464793105006950

Efford, M. 2004. Density estimation in live-trapping studies. Oikos **106,** 598–610.

Efford, M. G. 2011. Estimation of population density by spatially explicit capture–recapture analysis of data from area searches. Ecology **92,** 2202–2207.

Efford, M. G. 2014. Bias from heterogeneous usage of space in spatially explicit capture-recapture analyses. Methods in Ecology and Evolution **5,** 599–602.

Fisher, R.A. 1930.The Genetical Theory of Natural Selection. Clarendon Press, Oxford.

Fulton, C. J., Starrs, D., Ruibal, M., & B. C. Ebner. 2012. Counting crayfish: active searching and baited cameras trump conventional hoop netting in detecting Murray Crayfish. Endangered species research **19,** 39-45. doi: 10.3354/esr00460

Gehlot, H. S., Adhikari, T., & K. Vipul. 2021. The perilous plight of Great Indian bustard (*Ardeotis nigriceps*) in the Thar Desert of Rajasthan. Applied Ecology and Environmental Sciences **9,** 410–418.

Gilligan, D., Rolls, R., Merrick, J., Lintermans, M., Duncan, P., & J. Kohen. 2007. Scoping the knowledge requirements for Murray crayfish. NSW Department of Primary Industries Aquatic Ecosystems Research Narrandera Fisheries Centre, 107. https://www.dpi.nsw.gov.au/__data/assets/pdf_fle/0011/155963 (Accessed 6/12/2023).

Goodrich, B., Gabry, J., Ali, I., & S. Brilleman. 2022. Rstanarm: Bayesian applied regression modeling via Stan.

Jones, C. G., Lawton, J. H., & M. Shachak. 1994. Organisms as ecosystem engineers. Oikos **69,** 373.

King, A. J., Tonkin, Z., & J. Lieshcke. 2012. Short-term effects of a prolonged blackwater event on aquatic fauna in the Murray River, Australia: considerations for future events. Marine and Freshwater Research **63,** 576.

Lyon, J.P., Bird, T., Nicol, S., Kearns, K., O'Mahony, J., Todd, C.R., Cowx, I.G., & C. J. A. Bradshaw. 2014. Efficiency of electrofishing in turbid lowland rivers: implications for measuring temporal change in fish populations. Canadian Journal of Aquatic Science **71,** 878–886.

MacDiarmid, A. B. & M. J. Butler IV. 1999. Sperm economy and limitation in spiny lobsters. Behavioral Ecology and Sociobiology **46,** 14–24.

McCarthy, B., Zukowski, S., Whiterod, N., Vilizzi, L., Beesley, L., & A. King. 2014. Hypoxic blackwater event severely impacts Murray crayfish (*Euastacus armatus*) populations in the Murray River, Australia: Hypoxic Blackwater Impacts Murray Crayfish. Austral Ecology **39,** 491–500.

McPhee, D., Leadbitter, D., & G. A. Skilleter. 2002. Swallowing the bait: is recreational fishing in Australia ecologically sustainable? Pacific Conservation Biology **8,** 40

Maheshwari, B. L., Walker, K. F., & T. A. McMahon. 1995. Effects of regulation on the flow regime of the river Murray, Australia. Regulated Rivers: Research & Management **10,** 15–38.

Meynecke, J. O., & R. G. Richards. 2014. A full life cycle and spatially explicit individual-based model for the giant mud crab (*Scylla serrata*): a case study from a marine protected area. ICES Journal of Marine Science **71,** 484–498.

NSW DPI (New South Wales Department of Primary Industries), 2022. NSW Recreational freshwater fishing guide. NSW DPI, Nowra. https://www.dpi.nsw.gov.au/fishing/recreational/fishing-rules-and-regs/freshwater-recreational-fishing-guide (Accessed 6/12/2023).

Noble, M., Fulton, C.J., & J. Pittock. 2018. Looking beyond fishing: Conservation of keystone freshwater species to support a diversity of socio-economic values. Aquatic Conservation: Marine and Freshwater Ecosystems **27,** 90–102.

Pinya, S., Tavecchia, G. & V. Pérez-Mellado. 2017. Population model of an endangered amphibian: implications for conservation management. Endangered Species Research **34,** 123–130.

Pollock, K. H. 1991. Review papers: modelling capture, recapture, and removal statistics for estimation of demographic parameters for fish and wildlife populations: past, present, and future. Journal of the American Statistical Association **86,** 225–238.

R Core Team. 2022. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.

Raymond, S., Duncan, M., Tonkin, Z., and W. Robinson. 2023a. Barmah-Millewa Fish Condition Monitoring (2007-2023). Unpublished Client Report for the Murray-Darling Basin Authority. Arthur Rylah Institute for Environmental Research, Department of Energy, Environment, and Climate Action, Heidelberg, Victoria.

Raymond, S. Tonkin, Z. Duncan, M., & W. Robinson. 2017. Barmah-Millewa Fish Condition Monitoring: 2006 to 2017. Arthur Rylah Institute for Environmental Research Unpublished Client Report for the Murray Darling Basin Authority. Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

Raymond, S. M. C. & C. R. Todd. 2020. Assessing risks to threatened crayfish populations from sex-based harvesting and differential encounter rates: a new indicator for reproductive state. Ecological Indicators **118,** 106661

Raymond, S., Todd, C., Ryall, J., Fanson, B., Koehn, J., Tonkin, Z., Hackett, G., O'Mahony, J., Berry, K., Lieschke, J., Rourke, M., & K. Ward. 2023b. Using density estimates, sex ratios and size structure to assess the status of a threatened Australian freshwater crayfish (*Euastacus armatus*) population. Hydrobiologia **850,** 4181-4194. https://doi.org/10.1007/s10750-023-05289-1

Richman, N. I., M. Böhm, S. B. Adams, F. Alvarez, E. A. Bergey, J. J. S. Bunn, Q. Burnham, J. Cordeiro, J. Coughran, K. A. Crandall, K. L. Dawkins, R. J. DiStefano, N. E. Doran, L. Edsman, A. G. Eversole, L. Füreder, J. M. Furse, F. Gherardi, P. Hamr, D. M. Holdich, P. Horwitz, K. Johnston, C. M. Jones, J. P. G. Jones, R. L. Jones, T. G. Jones, T. Kawai, S. Lawler, M. López-Mejía, R. M. Miller, C. Pedraza-Lara, J. D. Reynolds, A. M. M. Richardson, M. B. Schultz, G. A. Schuster, P. J. Sibley, C. Souty-Grosset, C. A. Taylor, R. F. Thoma, J. Walls, T. S. Walsh & B. Collen. 2015. Multiple drivers of decline in the global status of freshwater crayfish (Decapoda: Astacidea). Philosophical Transactions of the Royal Society b: Biological Sciences **370,** 20140060

Ryan, K. 2005. The home range behaviour of the Murray River crayfish, Murray Crayfish (Decapoda: Parastacidae), in the Murrumbidgee River, Australian Capital Territory. 84. Hons. Thesis

Sato, T., & S. Goshima. 2006. Impacts of male-only fishing and sperm limitation in manipulated populations of an unfshed crab, *Hapalogaster dentata*. Marine Ecology Progress Series **313,** 193–204.

Sato, T., & S. Goshima. 2007. Female choice in response to risk of sperm limitation by the stone crab, *Hapalogaster dentata*. Animal Behaviour **73,** 331–338.

Sato, T., & K. Yoseda. 2010. Infuence of size- and sex-biased harvesting on reproduction of the coconut crab *Birgus latro*. Marine Ecology Progress Series **402,** 171–178

Solberg, E. J., Loison, A., Ringsby, T. H. Sæther, B.-E. & M. Heim. 2002. Biased adult sex ratio can affect fecundity in primiparous moose *Alces alces*. Wildlife Biology **8,** 117–128.

Solberg, E. J., Grøtan, V., Rolandsen, C. M., Brøseth, H. & S. Brainerd. 2005. Change-in-sex ratio as an estimator of population size for Norwegian moose *Alces alces*. Wildlife Biology **11,** 163–172.

Sørdalen, T. K., Halvorsen, K. T., Harrison, H. B., Ellis, C. D., Vøllestad, L. A., Knutsen, H., Moland, E. & E. M. Olsen. 2018. Harvesting changes mating behaviour in European lobster. Evolutionary Applications **11,** 963–977.

Székely, T., Weissing, F. J. & J. Komdeur. 2014. Adult sex ratio variation: implications for breeding system evolution. Journal of Evolutionary Biology **27,** 1500–1512.

Todd, C. R., Whiterod, N., Raymond, S. M. C., Zukowski, S., Asmus, M., & M. J. Todd. 2018. Integrating fishing and conservation in a risk framework: a stochastic population model to guide the proactive management of a threatened freshwater crayfish. Aquatic Conservation: Marine and Freshwater Ecosystems **28,** 954–968.

Tonkin, Z., & M., Rourke. 2009. Barmah-Millewa Fish Condition Monitoring - 2008 annual summary and refuge habitat report. Unpublished report submitted to the Murray-Darling Basin Authority. Arthur Rylah Institute for Environmental Research Department of Sustainability and Environment and New South Wales Department Primary Industries.

Usio, N., & C. R. Townsend. 2004. Roles of crayfish: of predation and bioturbation for stream invertebrates. Ecology **85,** 807–822.

van Dijk, A. I. J. M., Beck, H. E., Crosbie, R. S., de Jeu, R. A. M., Liu, Y. Y., Podger, G. M., Timbal B., & N. R. Viney. 2013. The millennium drought in southeast Australia (2001–2009): Natural and human causes and implications for water resources, ecosystems, economy, and society. Water Resources Research **49,** 1040–1057.

Victorian Fisheries Authority. 2022. Victorian recreational fishing guide 2022: a guide to fishing rules and practices. www.vfa.vic.gov.au (accessed 6/12/2023)

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Liermann, C. R., & P. M. Davies. 2010. Global threats to human water security and river biodiversity. Nature **467**, 555-561.

Walker, K. F., & M. C. Thoms. 1993. Environmental effects of flow regulation on the lower river Murray, Australia. Regulated Rivers: Research & Management **8,** 103–119.

Whiterod, N. S., Zukowski, S., Asmus, M., Todd, C. R., & D. C. Gwinn. 2018. Take the long way home: Minimal recovery in a K-selected freshwater crayfish impacted by significant population loss. Ecological Indicators **89,** 622–630.

Whiterod, N., & S. Zukowski. 2017. The status of the Murray crayfish recreational fishery in Victoria. A report funded by the Victorian Government using recreational fishing Licence fees. Goolwa Beach: Aquasave–Nature Glenelg Trust

Yorisue, T., Iguchi, A., Yasuda, N., Yoshioka, Y., Sato, T., & Y. Fujita. 2020. Evaluating the efect of overharvesting on genetic diversity and genetic population structure of the coconut crab. Scientifc Reports **10,** 1–9.

Zukowski, S., Curtis, A., & R. J., Watts. 2011. Using fisher local ecological knowledge to improve management: The Murray crayfish in Australia. Fisheries Resources **110**, 120–127.

Zukowski, S., Watts, R., Curtis, A. 2012. Linking biology to fishing regulations: Australia's Murray crayfish (*Euastacus armatus*). Ecological Management and Restoration **13,** 183–190.

Zukowski, S., Whiterod, N.S., & R. J. Watts. 2013. Comparing Murray crayfish (*Euastacus armatus*) population parameters between recreationally fished and non-fished areas. Freshwater Crayfish **19,** 153–160

www.deeca.vic.gov.au

www.ari.vic.gov.au