Comparing Murray Crayfish (*Euastacus armatus*) Population Parameters Between Recreationally Fished and Non-fished Areas

Sᴙʟᴠɪᴀ Zᴜᴋᴏᴡsᴋɪ,[1,2] Nɪᴄᴋ S. Wʜɪᴛᴇʀᴏᴅ [2,*] ᴀɴᴅ Rᴏʙʏɴ J. Wᴀᴛᴛs [1]

[1] Institute for Land, Water and Society, Charles Sturt University, PO Box 789, Albury, NSW 2640, Australia.
[2] Aquasave – Nature Glenelg Trust, 7 Kemp Street, Goolwa Beach, SA 5214, Australia.
*Corresponding Author.—*nick.whiterod@aquasave.com.au*

Abstract.— The implementation of fishing regulations becomes increasingly complex where the natural state of fisheries resources is unknown. Comparing populations in fished and non-fished areas can provide information that is vital for the management and protection of species. We conducted field surveys of *Euastacus armatus* in non-fished and fished reservoirs and provide comparisons with a heavily fished area of the River Murray. The non-fished population (Talbingo Reservoir) of *E. armatus* exhibited almost equal sex ratios, robust normally-distributed population structure and a high proportion of mature and berried females. The parameters defining two fished populations (Blowering Reservoir and the River Murray) deviated significantly, to varying degrees, from the benchmark population (Talbingo). These differences suggest that recreational fishing may impose a considerable impact on the population parameters of *E. armatus*. Comparison with the benchmark defined in the present study will allow tracking of the population recovery under the new fishing regulations for *E. armatus* in the southern Murray-Darling Basin. [*Keywords*.— *Euastacus armatus*; non-fished areas; population parameters; recreational fishing pressure].

INTRODUCTION

The sustainability of recreational fisheries is underpinned by sound fishing regulations, typically involving minimum legal limits (MLL), bag limits, area and seasonal closures and the protection of breeding females (King 2007; McPhee 2008). These fishing regulations are informed by knowledge of key parameters — abundance (Attwood and Bennett 1995), size frequencies and sex ratios (Horwitz 1991; Barker 1992), growth rates, reproduction, recruitment, fecundity (Rochet 1998), and the size of onset of sexual maturity (SOM) — that define the status of populations (Hobday and Ryan 1997; Rochet 1998; Gardner et al. 2006). As the status of natural populations is often unknown, fishing regulations are based on progressively degraded and unsustainable baseline information (e.g., shifting baseline syndrome: Pauly 1995), which has hampered the sustainable management of many recreational fisheries (Freeman 2008).

The investigation of natural populations can reveal important information on the ecology of recreationally important fish and crayfish populations (Russ and Alcala 2004; Gardner et al. 2006; Freeman 2008). It is acknowledged that the addition of recreational fishing pressure to a non-fished system can lead to major changes in many of the population parameters (Lewin et al. 2006). Indeed, where comparison between natural populations and those under fishing pressure is possible, impacts such as reductions in abundance, alteration of sex ratios, and increased egg dislodgment and infections associated with handling, are typically realised in the fished population (Adams et al. 2000; Gardner et al. 2006; Freeman 2008). Benchmarks can also be defined through this comparison, so that fishing regulations can reflect acceptable levels of deviation from the parameters characterising natural populations.

Several members of the *Euastacus* genus of freshwater crayfish have been exposed to recreational fishing (Furse and Coughran 2011) with the largest species of the genus, Murray crayfish *Euastacus armatus* (von Martens, 1866), supporting a popular recreational fishery in the southern Murray-Darling Basin (MDB), Australia. *Euastacus armatus* was once widespread across its range, but has experienced declines in population abundance and distribution over the past 50 years (Horwitz 1995; Furse and Coughran 2011). Population declines have been related to river regulation, pesticides and pollutants, habitat degradation and fishing pressure (Walker and Thoms 1993; Gilligan et al. 2007; Furse and Coughran 2011; McCarthy et al. 2013). The recreational *E. armatus* fishery is considerable (e.g., 160,000 individuals harvested annually: Henry and Lyle 2003) and the impacts widely acknowledged (Gilligan et al. 2007; Zukowski et al. 2011, 2012), yet no studies have compared the parameters defining fished and non-fished populations of the species.
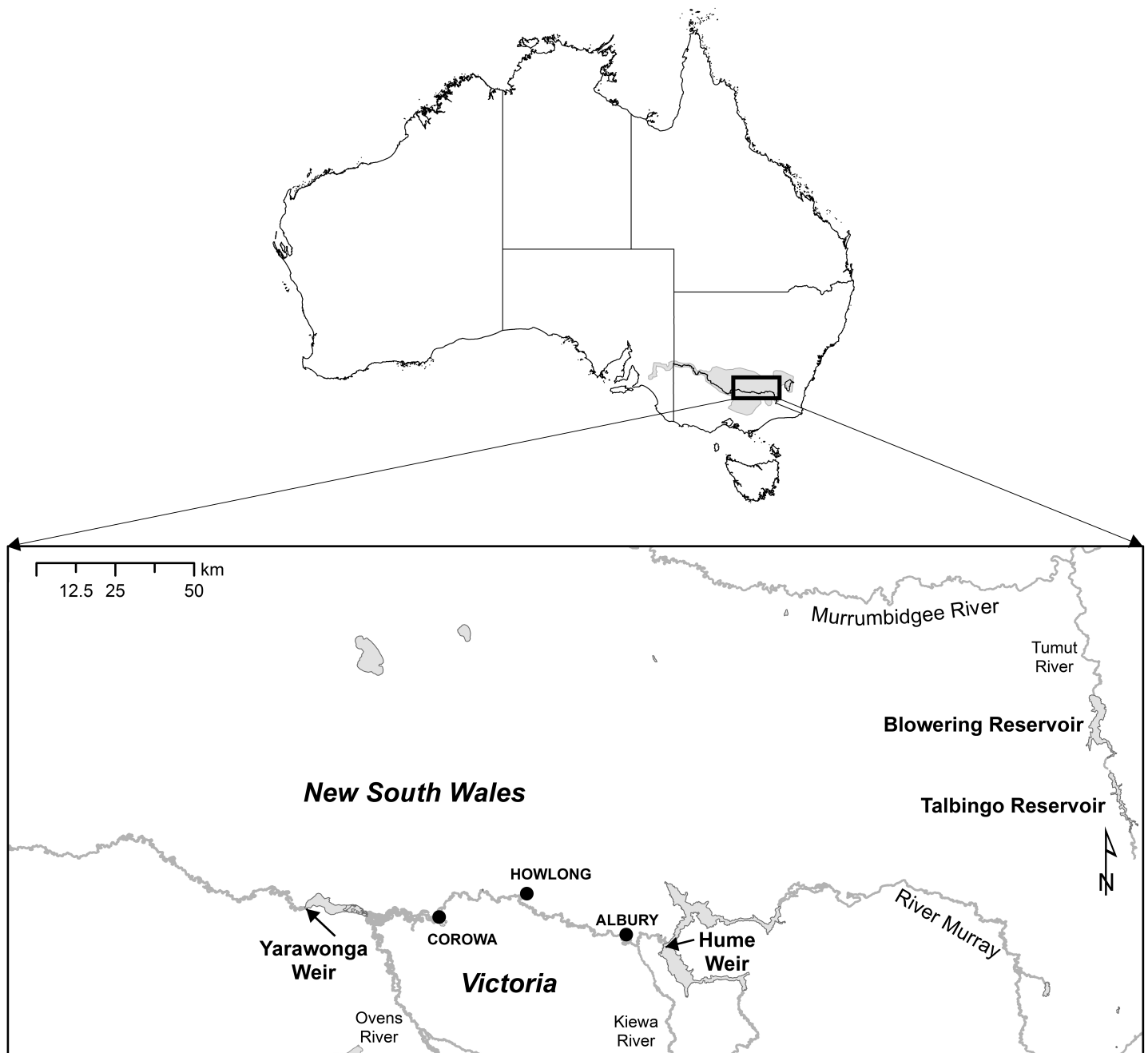
Figure 1. Location of the likely natural distribution of *Euastacus armatus* within Australia (grey shaded area) (Gilligan *et al.* 2007) and the location of the two reservoirs (Blowering and Talbingo) and the River Murray reach (Albury to Yarrawonga) where three sites were sampled at Albury, Howlong and Corowa (see Zukowski et al. 2011, 2012).

This deficiency largely stems from the limited number of non-fished populations, as most areas of the range of the species have been subjected to previous fishing pressure or are currently open to recreational fishing. Two adjacent reservoirs (Talbingo and Blowering Reservoirs) in south-eastern Australia provide a unique opportunity to compare the effects of recreational fishing on *E. armatus*, as they have been subjected to different fishing pressures for an extended period of time. Talbingo Reservoir has been considered a non-fished area as fishing has been banned for over 20 years and compliance appears to be high. In contrast, Blowering Reservoir has a history of heavy recreational fishing pressure. The impoundment was open to *E. armatus* fishing with

a bag limit of only five individuals until 2003 — yet catches of up to 100 individuals in one weekend were common (anonymous fisher 2009, *personal communication*) — before a rapid reduction in abundance prompted total closure between 2004 and 2006. The recreational fishery was briefly opened (2006), before a five year closure was implemented in 2008, and was recently continued (NSW DPI 2013).

Here, we aim to define the parameters (abundance, population structure, sex ratio and SOM) of a non-fished *E. armatus* population (Talbingo Reservoir) to define benchmark information for the species. This benchmark information is subsequently used

to assess the impacts that fishing pressure may impose on fished *E. armatus* populations (Blowering Reservoir). Comparison with a heavily fished River Murray population helps to provide additional assessment of the effects of recreational fishing on populations of this species.

METHODS

Study Location

Talbingo (-35.63332, 148.3016, GDA-94) and Blowering (-35.41987, 148.2553, GDA-94) Reservoirs occur within and adjacent to the Kosciuszko National Park just south of Tumut in the Murrumbidgee Valley on the Tumut River, southeastern Australia (Figure 1). The reservoirs were constructed in 1968 and are used to supply water for irrigation and industry, hydro-power and environmental flows and for flood mitigation (State Water Corporation 2009). The reservoirs are also used for recreational activities including waterskiing, sailing, boating, and fishing (State Water Corporation 2009).

Talbingo Reservoir has a surface area of 1,945 hectares (19.45 km²), a maximum depth of 110 m, a capacity of 950,500 ML at an elevation of 549 m, and is fed by the Tumut and Yarrangobilly Rivers, as well as Long and Middle Creeks. Blowering Reservoir has a surface area of 2,100 hectares (21 km²), a maximum depth of 91 m, a capacity of 1,628,000 ML at an elevation of 376 m, and is supplied by the Tumut River and the Sandy, Yellowin, Blowering and McGregors Creeks (State Water Corporation 2009).

Talbingo and Blowering Reservoirs are similar in location, size and depth and thus comparisons between these two environments would have been ideal. However, the low catch rates of *E. armatus* in Blowering Reservoir hindered robust statistical comparisons between the two populations of the *E. armatus* for all of the parameters examined. To strengthen predictive ability, *E. armatus* parameters from the reservoirs were compared with those from previous studies (Zukowski et al. 2011; Zukowski et al. 2012) that examined *E. armatus* population parameters in a 230 km recreationally fished reach of the River Murray between Hume Dam (-36.10774, 147.03075, GDA-94) and Yarrawonga Weir (-36.00857, 145.99956, GDA-94), New South Wales (NSW).

Crayfish Surveys

Crayfish surveys were carried out in Talbingo and Blowering Reservoirs in winter annually over three years (2008, 2009, and 2010). Seven randomly selected sites were sampled within each reservoir (14 sites in total) on each sampling trip on 14 consecutive days at 0900 h. Sampling of the three sites within the River Murray reach was conducted across 2009 (Zukowski et al. 2011). All sampling followed the standardised sampling protocol of Zukowski et al. (2012) as follows: single hoop nets of 700 mm diameter with a mesh size of 13 mm were baited with ox liver. The catch was recorded as catch per net per hour in order to standardise effort, with each net relocated after each haul. On each sampling day, at each site, twenty nets were set and checked hourly for a total of three hours (60 hoop net hauls per site). Data recorded from each net set comprised date, position (latitude and longitude), depth, distance from bank, time set and time retrieved. The catch data recorded comprised number of crayfish, occipital carapace length
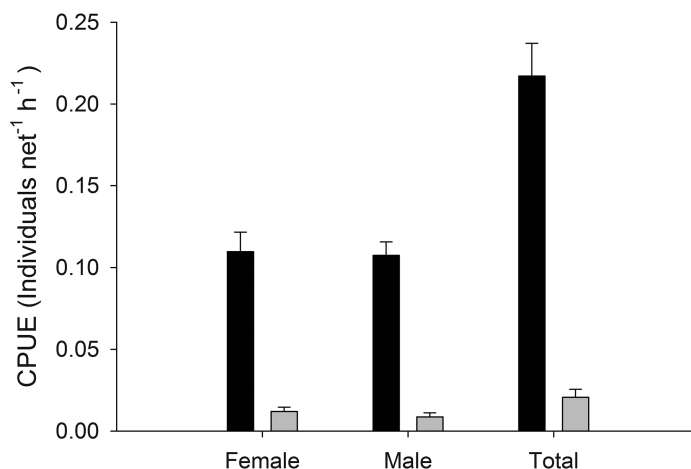
Figure 2. Mean (± standard error) catch-per-unit-effort (CPUE) abundance (individuals net⁻¹ hour⁻¹) of *Euastacus armatus* (females, males and total) from non-fished Talbingo Reservoir (black bar) and fished Blowering Reservoir (grey bar) in combined years (2008 – 2010).

(OCL, measured from the rear of the eye socket to the middle of the rear of the carapace) to the nearest 0.1 mm, sex, the maturity stage of adult females (stages 1 – 3) (Turvey and Merrick 1997), and whether females were in berry (carrying eggs).

Data Analysis

Abundance data was expressed as the number of individuals net⁻¹ h⁻¹ (Gilligan et al. 2007). Kruskal-Wallis tests were used to compare overall abundance and the number of females and males across the years in the Blowering and Talbingo populations. A Mann-Whitney U-test was used to compare overall abundance between the two populations. A G-test for goodness-of-fit was used to compare the number of sexually mature females caught each year and to compare the number of females with eggs captured each year in each reservoir.

A two sample Kolmogorov-Smirnoff test (KS-test) was used to test whether there was a significant difference in the length frequencies among years for each sex in Talbingo Reservoir to determine whether the three years of data could be pooled. Since no significant differences were detected between years, data were pooled to produce length frequencies for each sex in Talbingo Resevoir, which were compared using a two-sample KS-test. A one-way KS-test was then undertaken to test whether the length frequency differed significantly from normality (i.e., taken from normal distribution). Similar analyses were conducted on the River Murray population, but insufficient numbers precluded analysis of the Blowering Reservoir population.

A Chi-squared analysis was used to ascertain whether there was a difference in adult *E. armatus* sex ratios in Talbingo Reservoir among years. Chi-squared tests for the comparison of two proportions (from independent samples) were used to determine whether sex ratios differed between the three areas.

Annual sampling data from Talbingo Reservoir and the River Murray surveys were used to determine SOM. Data from sexually mature females, as ascertained by the presence of eggs or ovigerose stage 3 setae (Turvey and Merrick 1997), were grouped into 5 mm
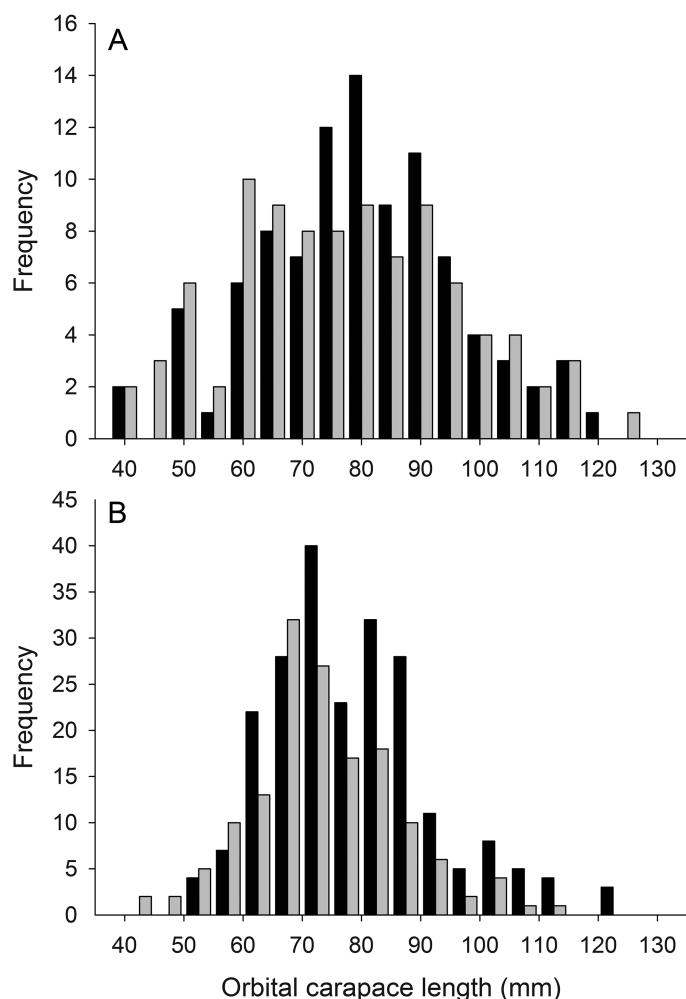
Figure 3. Population structure of male (grey bar) and female (black bar) *Euastacus armatus*, from (a) the non-fished Talbingo Reservoir (n = 188) pooled across years (2008 – 2010) and (b) the fished River Murray (n = 370) from 2009 (taken from Zukowski et al. 2011).
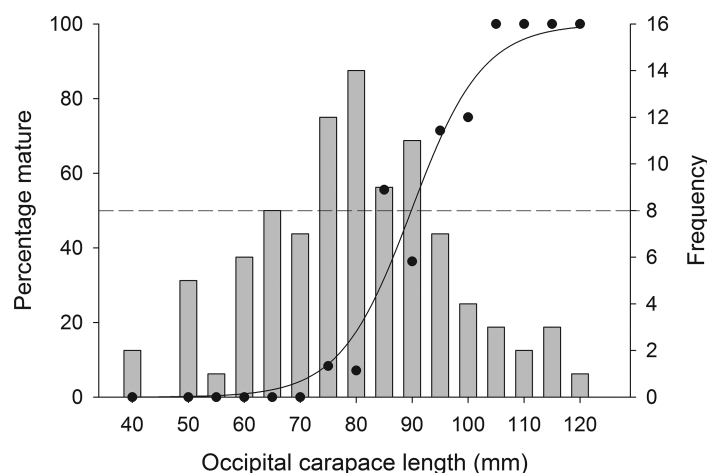
Figure 4. Occipital carapace length (OCL) size frequencies (grey bars), size of onset of sexual maturity (SOM) data (dots), fitted line (black line) and associated SOM_{50} values (where 50% of females are mature, dashed line) for female *Euastacus armatus* in non-fished Talbingo Reservoir (n = 95).

OCL size classes. A size structure analysis (length-frequency histogram) was developed to ascertain female *E. armatus* maturity stages. Following the methods of Hobday and Ryan (1997), the percentage of sexually mature females in a given size class (OCL) was determined, and then fitted by means of the logistic equation:

$$M = \frac{100}{[1 + (\frac{L}{SOM_{50}})^b]}$$

Where M is the percentage of females in a size class, L is the OCL (mm), SOM_{50} is the length at which 50% of females are mature (SOM), and b is a constant.

RESULTS

Catch Summary

Totals of 188 (95 females (13 berried), 93 males) and 19 (11 females (2 berried), 8 males) *E. armatus* were sampled during 866 and 921 fishing hours in annual sampling from 2008 to 2010 in Talbingo and Blowering Reservoirs, respectively. Abundance did not vary significantly between years in Talbingo (Kruskal-Wallis test, H = 0.080, d.f.= 1, P = 0.77) and Blowering Reservoir (Kruskal-Wallis test, H = 0.014, d.f.= 1, P = 0.90). Similarly, no significant difference was found in the number of males and females captured among the different years in either Talbingo (Kruskal-Wallis test, H = 0.079, d.f. = 1, P = 0.78) or Blowering Reservoirs (Kruskal-Wallis test, H = 0.039, d.f. = 1, P = 0.84). Therefore, abundance data for the three years for each sex in each reservoir were pooled (Figure 2). The overall mean abundance for the population from Talbingo Reservoir was 0.22 individuals net^{-1} h^{-1} (females = 0.11, males = 0.11 individuals net^{-1} h^{-1}) and 0.021 individuals net^{-1} h^{-1} for the population from Blowering Reservoir (females = 0.012, males = 0.009 individuals net^{-1} h^{-1}). A significant difference was found in the overall abundance between the two populations (Mann-Whitney U-test, H = 2.4, d.f. = 1, P < 0.05). Totals of 45 and 14 individuals ≥ 90 mm OCL were captured in Talbingo and Blowering Reservoirs, respectively (Table 1).

A total of 28 and 10 sexually mature female *E. armatus* were recorded in Talbingo and Blowering Reservoirs, respectively. No significant difference was found in the number of sexually mature female *E. armatus* captured among years in either Talbingo (G-test, G = 0.24, d.f. = 2, P = 0.88) or Blowering (G-test, G = 0.194, d.f. = 2, P = 0.91) Reservoirs. Totals of 13 and 2 berried female *E. armatus* were recorded in Talbingo and Blowering Reservoirs respectively. Thus, of the total number of sexually mature females caught, 46% were carrying eggs (in berry) in Talbingo Reservoir and 20% in Blowering Reservoir. No significant difference was found in the number of berried female *E. armatus* captured among years in Talbingo Reservoir (G-test, G = 1.058, d.f. = 2, P = 0.59).

Population Structure and Sex Ratios

Across the three years of the study, *E. armatus* individuals were collected between 39 – 120 mm OCL in non-fished Talbingo Reservoir and 88 – 152 mm OCL in fished Blowering Reservoir. In Talbingo Reservoir, there were no significant differences in the length-frequencies between years for either males (KS-test, D = 0.247, P = 0.27 (2008 vs. 2009), D = 0.161, P = 0.80 (2008 vs.

Table 1. Sex ratios (males, M; females, F) of *Euastacus armatus* in the River Murray, New South Wales, in 2009, and in Blowering and Talbingo reservoirs from 2008 – 2010.

Size (OCL)	Talbingo Reservoir			Blowering Reservoir			River Murray		
	M	F	Ratio (M:F)	M	F	Ratio (M:F)	M	F	Ratio (M:F)
All classes	93	95	0.98:1	8	11	0.73:1	173	248	0.70:1
< 90 mm	71	72	0.99:1	4	1	4.00:1	154	207	0.74:1
≥ 90 mm	22	23	0.96:1	4	10	0.44:1	19	41	0.46:1

2010), $D = 0.256$, $P = 0.20$ (2009 vs. 2010)) or females (KS-test, $D = 0.186$, $P = 0.58$ (2008 vs. 2009), $D = 0.300$, $P = 0.11$. (2008 vs. 2010), $D = 0.119$, $P = 0.97$ (2009 vs. 2010)), so data were pooled across years for each sex (Figure 3). Pooled length-frequencies did not differ significantly between sexes for all size classes (KS-test, $D = 0.158$, $P = 0.18$), size classes ≥ 90 mm OCL (KS-test, $D = 0.121$, $P = 0.10$), or size classes < 90 mm OCL (KS-test, $D = 0.203$, $P = 0.09$). Finally, the pooled length-frequencies for each sex did not vary significantly from normality (males, KS-test, $D = 0.152$, $P = 0.20$; females, KS-test, $D = 0.140$, $P = 0.21$). Similarly, no significant difference was found in the OCL size frequencies between total female and sexually mature females in the non-fished population (KS-test, $D = 0.506$, $P < 0.05$). The low catch in Blowering Reservoir prevented meaningful investigation of its population structure. However, the structure of the fished River Murray population varied significantly for both males (KS-test, $D = 0.088$, $P < 0.05$) and females (KS-test, $D = 0.073$, $P = 0.047$) (Zukowski et al. 2011) (Figure 3).

No significant skews were observed in the sex ratios in Talbingo Reservoir when all size classes were compared (0.98:1, $P = 0.94$) or when size classes ≥ 90 mm OCL were compared (0.96:1, $P = 1.0$) (Table 1). In Blowering Reservoir, male to female sex ratios of 0.73:1 and 0.44:1 were observed when all size classes were compared and when size classes ≥ 90 mm OLC were compared, respectively (Table 1). Significant differences were found in the sex ratios between Talbingo and Blowering reservoirs when all size classes were combined ($P < 0.05$) and in size groups ≥ 90 mm OCL ($P < 0.05$). Significant differences were found in the sex ratios between populations from Talbingo Reservoir and the River Murray data when all size classes were combined ($P < 0.05$) and in the crayfish size group ≥ 90 mm OCL ($P < 0.05$) (Chi-squared test for comparison of two proportions). No significant differences were found in the sex ratios found between Blowering Reservoir and the River Murray when all size classes were combined ($P = 0.98$) or in those ≥ 90 mm OCL ($P = 0.87$).

Size at Onset of Sexual Maturity (SOM)

In Talbingo and Blowering Reservoirs, sexually mature females ranged between 74 and 120 mm OCL (28 individuals) and 88 and 152 mm OCL (10 individuals), and no immature females were found ≥ 100 and 92 mm OCL, respectively. In Talbingo Reservoir, the mode of the length-frequency distribution of females peaked at 75 – 80 mm OCL (Figure 3). Size classes between 75 and 95 mm OCL were well represented, while those ≤ 70 and ≥ 100 mm OCL had lower numbers of individuals. The percentage of sexually mature females increased between 75 and 100 mm OCL.

For Talbingo Reservoir, the logistic model, fitted to the proportion of mature females in each 5 mm OCL size class, resulted in an estimated SOM_{50} of 90.83 mm OCL ($r^2 = 0.96$; $n = 16$ size classes) (Figure 4). At the MLL when the present study was conducted (90 mm OCL), 49% of female crayfish were sexually mature and 80% were sexually mature at 120 mm OCL. In the River Murray, a SOM_{50} of 91.77 mm OCL was estimated ($r^2 = 0.99$; $n = 15$ size classes). At 90 mm OCL, 39% of female *E. armatus* were sexually mature and 98% were sexually mature at 110 mm OCL.

DISCUSSION

The present study represents the first investigation of the parameters defining a non-fished population of *E. armatus* across its range over the southern Murray-Darling Basin (MDB). The parameters of two fished *E. armatus* populations varied considerably from this non-fished benchmark and, whilst other factors may have contributed, this comparison revealed useful insight into the potential impacts of fishing pressure on freshwater crayfish species. The findings of the present study are consistent with other studies of exploited recreational fisheries (Gardner et al. 2006; Lewin et al. 2006; Freeman 2008). The following sections provide specific comparison of the parameters defining non-fished and fished populations.

Abundance

The abundance of large and recreationally important crayfish populations is strongly influenced by fishing pressure (Lintermans and Rutzou 1991). Low abundances of *E. armatus* have been reported in known heavily fished areas, including the upper Murrumbidgee River (Lintermans and Rutzou 1991; Ryan 2005) and River Murray (McCarthy 2005). In the present study, significantly higher abundances were observed in the non-fished reservoir population compared to that of the fished reservoir population. Similar patterns are readily observed, such as in rock lobsters *Jasus edwardsii* (Hutton) populations off the coast of New Zealand (Freeman 2008) and Tasmania (Gardner et al. 2006), where a comparison between non-fished and fished populations is possible. In the fished reservoir, the very low abundance, coupled with slow population growth of the species (i.e., k-selected life history), indicate that recovery will be slow and the area closure presently in place may need to be continued indefinitely, particularly if non-compliance is occurring.

It must be noted that freshwater crayfish abundances are inherently variable and using only this parameter to investigate impacts of fishing can lead to misleading conclusions. For instance, the *E. armatus* abundance in Talbingo Reservoir was considerably

lower than the heavily fished site on the River Murray (Zukowski et al. 2012). These two areas are vastly different environments: the River Murray site is a warm lowland flowing river in the mid-range of the species, whereas, Talbingo reservoir is a cool and deep upland impoundment on the edge of the species' range. Thus, greater natural abundance at the River Murray site is confounding comparisons with the non-fished site. It is clearly important to compare similar sites (such as the two upland impoundments of the present study) when evaluating changes in abundance imposed by fishing pressure.

Population Structure

The non-fished Talbingo Reservoir featured a robust normally-distributed population structure with juveniles, sexually mature females (often in berry) and, importantly, large males (> 90 mm OCL) well represented. It is clear that strong and regular recruitment is occurring in this non-fished *E. armatus* population. In contrast, the population structure in the fished reservoir was unstable with no *E. armatus* below 88 mm OCL and few larger males. Whilst this finding may imply broad non-compliance to fishing regulations (e.g., taking of juveniles), it is more likely that this population has declined to such a degree that insufficient mature individuals (e.g., only two females in-berry observed in the present study) were present to facilitate recruitment. There were greater numbers of individuals at the fished River Murray site, but the population remained patchy, with a particular deficiency of larger individuals (Zukowski et al. 2011). Indeed, only 15% of sampled individuals from the fished River Murray site were above the MLL when the present study was conducted (90 mm OCL) compared to 37% of individuals in the non-fished population. The impacts to the population structure experienced in fished populations can largely be explained by fishing regulations in force when the present study was conducted (MLL and the protection of berried females) that impose differential pressure on larger males (Zukowski et al. 2012). This is commonly observed in recreational fisheries regulated by MLLs and may result in the truncation of the size range (Arlinghaus et al. 2010).

There was an expectation that a naturally condensed size range would occur in the cooler upland conditions of the study region (cf. Johnston et al. 2008), yet, the size range of sampled individuals in Talbingo Reservoir (39 – 120 mm OCL) was comparable with warm lowland locations (River Murray, 40 – 124 mm OCL) (Zukowski et al. 2012). Interestingly, in the fished Blowering Reservoir a truncated size range was not observed (88 – 152 mm OCL) but rather there was a skew toward larger individuals (including the largest individual of all three sites). This indicates that fishing pressure has not reduced the upper size range at this site, but rather recruitment failure was contributing to the observed population structure. However, low numbers in this impoundment may be confounding this observation.

Sex Ratios

In the non-fished Talbingo reservoir, sex ratios were almost in unity, which is consistent with the findings of other *E. armatus* populations in the Ovens River, where fishing closures (for up to seven years) were in place (Barker 1992). More broadly, other freshwater crayfish in non-fished areas exhibit equal sex ratios,

including Tasmanian giant freshwater crayfish, *Astacopsis gouldi* Clark (Horwitz 1991), noble crayfish, *Astacus astacus* (Linnaeus) in Lake Gailintas, Lithuania (Mackeviciene et al. 1999) and Swiss populations of slender-clawed crayfish, *Astacus leptodactylus* (Eschscholtz) (Stucki 1999). It is suggested that natural (non-fished) populations should exhibit sex ratios in unity due to equal sex ratios at birth (e.g., secondary sex ratio) and approximately even moulting frequencies, growth increments and survival between the sexes (Holdich 2002).

Euastacus armatus populations in the two fished areas in the present study exhibited sex ratios consistently skewed towards females in all size classes examined, and this skew was most extreme when examining size classes above the MLL for the species (up to 0.39:1). Skewed sex ratios have been observed in other fished populations of the *E. armatus* (McCarthy 2005; Zukowski 2012) and the closely-related Glenelg spiny crayfish *Euastacus bispinosus* (Clark) (Honan and Mitchell 1995; Johnston et al. 2008). The consequence of such skewed sex ratios in *E. armatus* is unclear but effective population size may be decreased, thus altering reproductive success and genetic diversity (Lewin et al. 2006).

Onset of Sexual Maturity (SOM) and Minimum Legal Length (MLL)

During the present study, the MLL was 90 mm OCL across the southern MDB, which appears broadly consistent in the non-fished area (SOM_{50} = 90.8 mm OCL) as 49% of females are mature at this size. In the fished River Murray area a SOM_{50} of 91.8 mm OCL was revealed, which indicates that only 39% of *E. armatus* females had reached sexual maturity at the current MLL (Zukowski et al. 2012). Similarly, comparisons of SOM_{50} in *J. edwardsii* between fished areas and nearby non-fished marine protected areas on the east coast of Tasmania revealed a small change in SOM_{50} between the areas despite large differences in abundance (Gardner et al. 2006). Since the present study, the MLL has been increased to 100 mm OCL across the range of the species (NSW DPI 2013; Victorian DEPI 2013), which will allow a greater number of individuals to reach sexual maturity, but could intensify the fishing pressure on individuals above 100 mm OCL (Zukowski et al. 2012).

Implications for Fisheries Management

Recent changes to the regulations for the *E. armatus* recreational fishery across the southern MDB, including area closures, an increase in the MLL to 100 mm and shortening of the open season (NSW DPI 2013; Victorian DEPI 2013), have been instigated to assist the recovery of the species. The present study informs management by providing benchmark information from a natural population of the species that will allow tracking of the recovery of the species. It is recommended that *E. armatus* populations be managed toward a robust, normally-distributed population structure (including juveniles and large males), sex ratios in unity and a high proportion of mature and berried females. Acceptable levels of deviations from these parameters must be established and routine monitoring undertaken to ensure the sustainability of the recreational fishery. It is recommended that benchmark information should be used to guide the management of other recreationally fished freshwater crayfish species.

LITERATURE CITED

Adams S, Mapstone BD, Russ GR and Davies CR (2000). Geographic variation in the sex ratio, sex specific size, and age structure of *Plectropomus leopardus* (Serranidae) between reefs open and closed to fishing on the Great Barrier Reef. *Canadian Journal of Fisheries and Aquatic Sciences* 57(7):1448–1458.

Arlinghaus R, Matsumura S and Dieckmann U (2010). The conservation and fishery benefits of protecting large pike (*Esox lucius* L.) by harvest regulations in recreational fishing. *Biological Conservation* 143(6):1444–1459.

Attwood CG and Bennett BA (1995). A procedure for setting daily bag limits on the recreational shore-fishery of the South-Western Cape, South Africa. *South African Journal of Marine Science* 15(1):241–251.

Barker J (1992). The spiny freshwater crayfish monitoring program 1990 Victoria. Department of Conservation and Environment Fisheries Management, Victoria, Australia.

Freeman D (2008). The ecology of spiny lobsters (*Jasus edwardsii*) on fished and unfished reefs. The University of Auckland, Auckland, New Zealand.

Furse JM and Coughran J (2011). An assessment of the distribution, biology, threatening processes and conservation status of the freshwater crayfish, genus *Euastacus* (Decapoda: Parastacidae) in continental Australia. III. Case studies and recommendations. *Crustaceana Monographs: New Frontiers in Crustacean Biology* 15:265–274.

Gardner C, Frusher SD, Barrett NS, Haddon M and Buxton CD (2006). Spatial variation in size at onset of maturity of female southern rock lobster *Jasus edwardsii* around Tasmania, Australia. *Scientia Marina* 70(3):423–430.

Gilligan D, Rolls R, Merrick J, Lintermans M, Duncan P and Koehn J (2007). Scoping the knowledge requirements for Murray crayfish (*Euastacus armatus*). Fisheries Final Report Series No 89. NSW Department of Primary Industries, Cronulla, Australia.

Henry GW and Lyle JM (2003). The National Recreational and Indigenous Fishing Survey. NSW Fisheries Final Report Series No. 48. NSW Fisheries, Cronulla, Australia.

Hobday DK and Ryan TJ (1997). Contrasting sizes at sexual maturity of southern rock lobsters (*Jasus edwardsii*) in two Victorian fishing zones: Implications for total egg production and management. *Marine and Freshwater Research* 48(8):1009–1014.

Holdich DM (2002). Biology of Freshwater Crayfish. Blackwell Science, Oxford, London, UK.

Honan JA and Mitchell BD (1995). Catch characteristics of the large freshwater crayfish, *Euastacus bispinosus* Clark (Decapoda: Paratacidae), and implications for management. *Freshwater Crayfish* 10:57–69.

Horwitz P (1991). On the distribution and exploitation of Tasmanian giant freshwater lobster *Astacopsis gouldi* Clark. Centre for Environmental Studies, University of Tasmania, Hobart, Australia.

Horwitz P (1995). The conservation status of Australian freshwater crayfish: Review and update. *Freshwater Crayfish* 10:70–80.

Johnston K, Robson BJ and Austin CM (2008). Population structure and life history characteristics of *Euastacus bispinosus* and *Cherax destructor* (Parastacidae) in the Grampians National Park, Australia. *Freshwater Crayfish* 16:165–173.

King M (2007). Fisheries biology, assessment and management. Wiley-Blackwell, Oxford, UK.

Lewin W-C, Arlinghaus R and Mehner T (2006). Documented and potential biological impacts of recreational fishing: Insights for management and conservation. *Reviews in Fisheries Science* 14(4):305–367.

Lintermans M and Rutzou T (1991). The status, distribution and management of the Murray crayfish *Euastacus armatus* in the Australian Capital Territory. ACT Parks and Conservation Service, Canberra, Australia.

Mackeviciene G, Mickeniene L, Burba A and Mazeika V (1999). Reproduction of noble crayfish *Astacus astacus* (L.) in semi-intensive culture. *Freshwater Crayfish* 12:462–470.

McCarthy B (2005). Distribution of Murray Crayfish (*Euastacus armatus*) in the Mallee Region 2004. Murray-Darling Freshwater Research Centre, Mildura, Australia.

McCarthy B, Zukowski S, Whiterod N, Villizi L, Beesley L and King A (2013). Hypoxic blackwater event severely impacts Murray crayfish (*Euastacus armatus*) populations in the Murray River, Australia. *Austral Ecology*. In Press.

McPhee D (2008). Fisheries management in Australia. The Federation Press, Sydney, Australia.

NSW DPI (2013). NSW Recreational Freshwater Fishing Guide 2013. New South Wales Department of Primary Industries, Cronulla, Australia.

Pauly D (1995). Anecdotes and the shifting baseline syndrome of fisheries. *Trends in Ecology and Evolution* 10(10):430.

Rochet MJ (1998). Short-term effects of fishing on life history traits of fishes. *Journal of Marine Science* 55(3):371–391.

Russ GR and Alcala AC (2004). Marine reserves: Long-term protection is required for full recovery of predatory fish populations. *Oecologia* 138(4):622–627.

Ryan K (2005). The home range behaviour of the Murray River crayfish, *Euastacus armatus* (Decopoda: Parastacidae), in the Murrumbidgee River, Australian Capital Territory. Honours thesis. University of Canberra, Canberra, Australia.

State Water Corporation (2009). Dam facts and figures: Blowering Dam. State Water Corporation, NSW, Australia.

Stucki TP (1999). Life cycle and life history of *Astacus leptodactylus* in Chatzensee Pond (Zurich) and Lake Ageri, Switzerland. *Freshwater Crayfish* 12:430–448.

Turvey P and Merrick JR (1997). Reproductive biology of the freshwater crayfish, *Euastacus spinifer* (Decapoda: Parastacidae), from the Sydney region, Australia. *Proceedings of the Linnean Society of New South Wales* 118:131–155.

Walker KF and Thoms MC (1993). Environmental effects of flow regulation on the River Murray, South Australia. *Regulated Rivers: Research and Management* 8(1-2):103–119.

Zukowski S, Curtis A and Watts RJ (2011). Using fisher local ecological knowledge to improve management: The Murray crayfish in Australia. *Fisheries Research* 110(1):120–127.

Zukowski S, Watts R and Curtis A (2012). Linking biology to fishing regulations: Australia's Murray crayfish (*Euastacus armatus*). *Ecological Management and Restoration* 13(2):183–190.