Population dynamics and implications for management of a Murray cod and golden perch recreational fishery in south-eastern Australia

by

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Certificate of authorship

I hereby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previously published or written by another person nor material which to a substantial extent has been accepted for the award of any other degree or diploma at Charles Sturt University or any other educational institution, except where due acknowledgment is made in the thesis. Any contribution made to the research by colleagues with whom I have worked at Charles Sturt University or elsewhere during my candidature is fully acknowledged.

I agree that this thesis be accessible for the purpose of study and research in accordance with the normal conditions established by the Executive Director, Library Services or nominee, for the care, loan and reproduction of theses.

Signature

Date 5/10/16
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Ethics approval

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Abstract

Providing sustainable fisheries and preventing species decline are goals for recreational fishery management. Research to identify the cause and effect of population declines is often used to inform management responses for fisheries mitigation, rehabilitation and recovery. Management efforts to recover declining fish species centre on strategies such as mitigation of factors causing stress, re-stocking, translocation, harvest regulations, protected zones, spawning closures, harvest quantification, and control of commercial fishing. Assessment of management strategies is vital to understand whether recovery, sustainability and fishing quality goals are being achieved. However, assessment of the relative success of mitigation strategies are not often performed, so the effectiveness of management efforts cannot be quantified. Such targeted research has been successfully used in management and recovery efforts for species in other countries, but has not been widely used for key Australian freshwater species Murray cod, *Maccullochella peelii*, and golden perch, *Macquaria ambigua*.

Murray cod and golden perch are popular recreational fishing species in Australia’s Murray-Darling Basin (MDB) but have declined in response to habitat degradation, river regulation, pollution and overfishing. These species are icons of Australia’s inland fisheries and efforts to rehabilitate and conserve populations through directed and informed management choices are vital to ensure that these fish continue to offer sustainable fisheries. Evaluating the effectiveness and efficiency of management initiatives such as fish stocking, size limits and recreational take was urgently needed to make evidence-based decisions about these fisheries. Therefore, the overarching objective of this thesis was to fill essential knowledge gaps to better understand these fisheries and also improve management strategies.

I used complemented fisher survey designs to quantify two major inland fisheries for the first time. I assessed the effectiveness of a closed season, which aimed to protect spawning Murray cod and found it was influenced by high fishing effort and deliberate bycatch during the closure period. I also found that boat-based fishers almost exclusively targeted Murray cod and harvested more and larger fish than shore-based anglers.
Recreational harvest and catch-and-release are often governed by length-based restrictions such as minimum length limits (MLL) or harvest slot limits. I set out to determine the effectiveness of MLLs and slot limits for protecting inland fisheries and found that Murray cod and golden perch had substantial differences in growth rate and onset of reproductive maturation between impoundments and rivers. My research suggested that because of these differences, that system-specific regulations may be needed to reduce overfishing risk and meet fishing quality objectives.

Stocking of Murray cod and golden perch had occurred for over 50 years, and is based on the perception that natural recruitment is insufficient to sustain the fishery. I quantified the effectiveness of stocking to optimise release strategies for future programs. My data indicated variable proportions of stocked Murray cod and golden perch among waterbodies, with more stocked fish surviving in impoundments than rivers. Stocked Murray cod had significantly steeper length-weight relationships (i.e. higher weight at a given length) to unmarked fish, indicating that hatchery-reared Murray cod may have an advantage over wild fish. The variable contributions of stocked and wild fish among waterbodies are critical information for the development of adaptive, location-specific stocking strategies.

The information from fisher surveys, size limit and stocking assessments; together with recommendations for changes to management strategies and fishing regulations should be implemented to ensure that management goals of sustainability and fishing quality are met. The findings and recommendations of this study are specific to Murray cod and golden perch; however input factors such as stocking, governing factors such as size limits and closed seasons, and outputs such as effort, catch-and-release and harvest that shape recreational fisheries, are common to fisheries throughout the world. Application of the research themes and tools used and developed in this study can be applied to inform management strategies for other global species that may be under threat.
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Chapter 1
General introduction

1.1 Background

Providing sustainable fisheries and preventing species decline are inherent aims of recreational fishery management. Fish population decline is a global concern for both marine and freshwater environments (Bottom et al. 2005; Hunt et al. 2011; Luckhurst and Trott 2009). Identifying the cause and effect of population declines are common research aspirations with results used to inform management responses for mitigation, rehabilitation and recovery of effected populations (Didham et al. 2007; Lintermans et al. 2005; Paragamian 2012)

Management efforts to recover declining fish species include mitigation of antagonistic factors, re-stocking, translocation, harvest regulations, protected zones, spawning closures, harvest quantification, and control of commercial fishing (e.g. Agostinho et al. 2010; Allen et al. 2008). These strategies are used individually, or in concert, with an aim to help rehabilitate and recover fish populations. However, more research into the effectiveness and efficiency of management initiatives including fish stocking, size limits and quantifying recreational fisheries are required so that evidence-based decisions can be made to protect and sustain fish populations. The absence of such data can mask decline and limit recovery efforts (Post et al. 2002). Targeted research on knowledge gaps are required to inform key management strategies. Such targeted research has been used in management and recovery efforts for iconic species such as striped bass, Morone saxatilis, in North America which experienced dramatic population decline (Stevens et al. 1985). Research that focused on this species biology was used to inform harvest restrictions such as closed seasons, bag and size limits, which ultimately led to a remarkable recovery (Richards and Rago 1999).

The knowledge to understand and manage global fish populations is far from comprehensive. Murray cod and golden perch are popular recreational fishing species in the Murray-Darling Basin (MDB) of south-eastern Australia (Figure 1; Hall et al. 2012; Rowland 2005).
Murray cod and golden perch are ideal species to examine in detail to help inform fisheries science, and demonstrate how additional knowledge about a fishery can assist understanding and management in general. Murray cod and golden perch were once widespread throughout the MDB, however factors such as habitat degradation, river regulation, pollution and overfishing, have reduced distribution and abundance of these species to an estimated 10% of pre-European levels (Murray-Darling Basin Commission 2003). For example, the annual catch of Murray cod from the commercial fishery reduced from 136 tonnes to less than 10 tonnes over a ten year period from the mid-1950s (Reid et al 1997). The effects of exploitation were compounded by the construction of over 10,000 weirs, dams and regulators that restrict spawning activities of Murray cod and golden perch (Baumgartner et al. 2014). However, evidence of recovery is apparent in some fisheries, but not others (Rowland 2005). The recovery of some fisheries was thought to be assisted by improved harvest regulations and stocking programs (Rowland 2005). Murray cod and golden perch are icons of Australia’s inland fisheries and efforts to rehabilitate, conserve and recover populations in decline and

**Figure 1:** Study sites for the Murray and Murrumbidgee rivers, Burrrinjuck and Copeton dams in New South Wales, Australia. The Murray-Darling Basin is represented by the shaded area.
enhance and expand fisheries through directed and informed management choices is vital for long term sustainability.

Fishery managers require relevant, accurate and up-to-date data to inform strategies for recovery of populations and to ensure sustainable fisheries (Richards and Rago 1999). For Murray cod and golden perch populations, research priorities include; (i) quantification of effort, catch and harvest to inform harvest restrictions, as very few recreational fisheries have been assessed for these species (Brown 2010; Hunt et al. 2011); (ii) update age- and length-at-maturity to inform size limit decisions, as existing size limit decisions were based on data collected three decades ago that did not include a variety of waterbody types (Allen et al. 2009; Rowland 1989); and (iii) assessment of the survival and growth of stocked fish to inform stocking strategies, as most stocking programs for these species persist without understanding of their effectiveness (Crook et al. 2015).

1.2 Thesis objectives

The thesis objective is to inform key management strategies for Murray cod and golden perch recreational fisheries, and to contribute knowledge that can be applied to a range of global fisheries where decline is evident, and recovery efforts are being undertaken. Four general knowledge gaps are investigated, with more specific questions under each as per below.

Q1  Recreational fishing effort, catch, and harvest for Murray cod and golden perch in the Murrumbidgee River, Australia (Chapter 3).
Q1.1  What are the Murray cod and golden perch recreational catch, harvest and effort in a popular recreational fishery that has previously experienced declines in native fish stocks?
Q1.2  What are the implications from fishing during the Murray cod closed season?

Q2  Quantifying the recreational fishery for large percichthyid fish in an impoundment on the Murray River, Australia (Chapter 4).
Q2.1  What fishery-dependent parameters characterise the Lake Mulwala recreational fishery?
Q2.2  How variable are these fishery-dependant parameters among recreational fisheries for Murray cod?
Q3  System-specific variability in Murray cod and golden perch maturation and growth influences fisheries management options (Chapter 5).
    Q3.1  How variable are Murray cod and golden perch maturation onset and growth among riverine and impounded waterbodies?
    Q3.2  Does maturation onset vary systematically between riverine and impounded populations, and how does this effect management decisions on size limits?
Q4  Assessment of stocking effectiveness for Murray cod, *Maccullochella peelii*, and golden perch, *Macquaria ambigu*a, in rivers and impoundments of south-eastern Australia (Chapter 6).
    Q4.1  What is the survival and contribution of stocked fish to population structure in a variety of riverine and impounded waters?
    Q4.2  Is stocking necessary to augment wild fisheries due to insufficient natural recruitment?
    Q4.3  Do stocked fish have an advantage over wild populations with regard to condition?

1.3 How chapters are linked

Incorporated in this thesis are a general introduction (Chapter 1), a literature review on species decline and management strategies for fishery recovery that focus on Murray cod and golden perch (Chapter 2), four research chapters (Chapters 3-6), and a general discussion (Chapter 7). The four research chapters use empirical data to advise decisions regarding the recreational fisheries for Murray cod and golden perch and provide suggestions for future improvement. Chapters 3, 5 and 6 have been published in refereed international journals (Forbes et al. 2015a; Forbes et al. 2015b; Forbes et al. 2015c).

Quantification of Murray cod and golden perch recreational fisheries are important to inform management strategies, such as harvest restrictions and fishing closures. The literature review identified a compelling need for fishery dependent surveys to inform management decisions regarding Murray cod and golden perch fisheries. Therefore, in the first research chapter (Chapter 3), I determine Murray cod and golden perch recreational catch, harvest and effort in the Murrumbidgee River (Q1.1), and investigate the effects of recreational fishing during the Murray cod closed season (Q1.2).
However, no data exist to assess variability that may exist among fisheries for Murray cod. Therefore in the second research chapter (Chapter 4), I quantified the fishery dependent characteristics collected from the Murray cod recreational fishery in Lake Mulwala (Q2.1). These characteristics were then used to assess variability that may exist among Murray cod fisheries in Lake Mulwala, the Murrumbidgee River (Chapter 3), and the Murray River (using data from Brown 2010; Q2.2).

The literature review identified that harvest restrictions are informed by data such as fishing effort, catch-and-release, and harvest, and that such regulations are a key strategy used by fishery managers to govern recreational fisheries. Murray cod and golden perch harvest restrictions include length-based limits, such as minimum legal lengths (MLLs), and harvest slot limits. The literature review also identified that length-based restrictions are usually set above length at maturity to reduce the risk of recruitment overfishing. However, little information exists on Murray cod and golden perch length- and age-at-maturity to assess variability within and among riverine and impounded populations to inform the setting of such restrictions. Thus, Chapter 5 quantifies variation in Murray cod and golden perch length- and age-at-maturity in a range of riverine and impounded systems (Q3.1); and evaluates whether length- and age-at-maturity for these species varies systematically between riverine and impounded populations (Q3.2). Stocking of hatchery-reared Murray cod and golden perch is another common management tool that is often used in conjunction with other harvest regulations, such as size limits, to enhance and recover recreational fisheries. The literature review identified that assessment of stocking program success is rarely performed. However, a study assessing golden perch stocking success (Crook et al. 2015) was used as justification for Chapter 6, where I expanded on this initial study to investigate survival and contribution of stocked Murray cod and golden perch to populations across a broader geographic scope (i.e. in two rivers and two impoundments; Q4.1). The outcome of this investigation was then used to assess whether stocking was necessary to supplement wild fisheries due to insufficient natural recruitment (Q4.2); and if stocked fish are in better condition than wild populations (Q4.3). The general discussion (Chapter 7) provides a summary and synthesis of the results from the four research chapters (Chapters 3-6). The thesis concludes with consideration of future research opportunities and management directions.
Chapter 2

Literature review

2.1 Management strategies to recover species in decline

Native species population declines are a global problem impacting a variety of environments including terrestrial, marine, and freshwater (Bouché et al. 2011; Hilborn et al. 2003; Witte et al. 1992). The impact of humans, introduced species, invasive species, and habitat alteration are all factors that can alter the often fragile niche that populations of native species occupy. Identifying the cause and effect of such declines are common research goals required to develop management responses for mitigation, rehabilitation and recovery of populations.

Fishery management strategies can help rehabilitate and recover fish populations (Agostinho et al. 2010; Allen et al. 2008; Paquet et al. 2011; Richards and Rago 1999). However, more research into the effectiveness and efficiency of several management initiatives including fish stocking, size limits and recreational take is required so that evidence-based decisions can be made. Research can inform this process but quite often many decisions are made on political or popular beliefs (Cowx 1999; Taylor 1998). The absence of scientific information does not necessarily lead to species decline, but understanding vital information such as; length- and age-at-maturity; recreational effort, catch and harvest; and effectiveness of stocking; can assist management outcomes (e.g. Agostinho et al. 2010; Richards and Rago 1999; Trippel 1995).

Species decline is a trigger for management intervention and subsequent recovery programs. Recovery efforts for species in decline can take many forms, but refer to actions that prevent further loss of a species and remove or reduce threats. Whilst the primary strategy should be to prevent species from becoming at risk in the first instance, the ultimate goals of recovery efforts are for wild species to persist in the natural habitat long-term and perhaps be restored to levels that existed prior to the introduction of negative impacts.
Management efforts to recover declining fish species centre on several strategies that are used individually, or in concert, to achieve the desired recovery outcomes. These strategies include:

- **Mitigation of stressors** – Involves the removal or lessening factors causing stress on a species, such as building a fishway to allow fish passage through a weir to access critical habitat.

- **Stocking** – Adult fish are collected from the wild to artificially spawn and rear juveniles in a hatchery, which are then released into waterways to increase fish numbers.

- **Translocation** – Involves relocation of adult or juvenile fish from their native waterway to a new waterway from which the species may not have existed, are extirpated, or spawning stocks are very low.

- **Harvest regulations**
  - **Size and bag limits** - Control size and number of fish that recreational and commercial fisherman can retain.
  - **Protection zones** – Areas closed to fishing, either for particular species or an imposed fishing moratorium.
  - **Spawning closures** – Fishing closures designed to protect species from harvest and fishing effort during their spawning period.
  - **Control of commercial fishing** – Regulations governing commercial harvest restricts the quantity of fish removed for sale, which effects what fish are available to recreational fisherman.

- **Recreational fishing** – Recreational fishing surveys gather information to quantify effort, harvest, catch-and-release, targeting behaviour, creel composition, how and where fishers utilise a recreational fishery. Such information is used to understand exploitation caused by recreational fishers, which subsequently informs harvest restrictions such as size and bag limits and spawning closures.

These rehabilitation and recovery strategies are applicable to a range of species globally, with fishery managers using this suite of tools to control fisheries. Large-scale changes such as removing barriers to migration, or reducing the impacts of cold water pollution are cost prohibitive, often outside fishery manager control, and are used less
often than policy-based decisions (Barrett 2008; Lugg and Copeland 2014). Thus, manipulation of harvest regulations are a common and effective strategy (Allen et al. 2009; Dotson et al. 2013). The conservation of fish populations can rely on all recovery strategies. However, this thesis is focussed on; quantification of recreational fisheries using fisher surveys (chapters 3 and 4); describing length- and age at maturity to inform length-based harvest regulations (chapter 5); and assessment of stocking effectiveness (chapter 6).

2.1.1 Quantification of recreational fisheries to inform management strategies

Recreational fisheries are often considerable and may have an impact on the sustainability of fisheries (Cooke and Cowx 2006). Fisher surveys of sound design and implementation are required to monitor effective outcomes of management initiated harvest regulation, stocking and habitat enhancement (Pollock et al. 1994). Surveys are traditionally used to estimate fishing effort and harvest in a waterway (Steffe et al. 2008), however they can also be used to evaluate fisher attitudes toward management programs such as fishing closures, bag limits, stocking, and habitat restoration (Wilberg 2009). Social and economic surveys assess the value of fishing to fishers and communities (Campbell and Murphy 2005; Henry and Lyle 2003), whilst surveys used in conjunction with other data can be used to answer biological questions such as the contribution of fishing to fish mortality (Kozfkay and Dillon 2010). In addition, surveys provide information on fish quality and quantity, and also how, when and where fishers are utilising the fishery (Smallwood et al. 2006).

Collecting survey information is a key component of fisheries management in that it allows comparisons of key indices over time to assess status, and can also be used to assess fishery quality among similar fisheries. Comparison over time allows assessment of fisheries in decline and whether management interventions are effective in arresting or reversing decline (Kozfkay and Dillon 2010). For example, white sturgeon, *Acipenser transmontanus*, have declined across most of its natural range (Kozfkay and Dillon 2010). Despite these declines, white sturgeon remain a popular recreational fishery with protection afforded by strict or nil harvest regulations, and allowances for catch-and-release angling (Pikitch et al. 2005). White sturgeon have been reported to live in excess of 100 years (Rien and Beamesderfer 1994). As such, based on catch-and-
release data from fisher surveys, it is estimated that a white sturgeon may be hooked several hundred times during its life span (Kozfkay and Dillon 2010). Catch-and-release fishing may contribute to increased mortality for fisheries protected by catch-and-release regulations (Arlinghaus et al. 2007), yet fishing for white sturgeon continues despite scant information being available on fishing intensity or catch-and-release mortality rates (Kozfkay and Dillon 2010). In order to fill this knowledge gap, fisher surveys were used to obtain estimates of white sturgeon fishing effort, catch and loss, and to measure frequency of catch and loss (Kozfkay and Dillon 2010). The fisher survey data identified that there were few small fish in the population, indicative of consistently poor recruitment (or that fishers were targeting large fish). The authors concluded that additional research into catch-and-release mortality for white sturgeon is required to bolster the annual angler survey data and provide more informed recommendations for management of the species.

This example highlights that without the collection of fishery dependant information the extent of catch-and-release would remain unknown. Coupling catch-and-release with post-release mortality allows fishery managers to understand the extent that such fishing practices affect a species, thus informing decisions for regulations that govern such practices. Fisher surveys are more commonly used to assess change over time (Pollock et al. 1994). However in the absence of historical data to make such comparisons, identification of recovery or decline can go undetected (Post et al. 2002). Under such circumstances, use of stock assessment tools such as empirical yield/abundance models and size-based indicators, can provide other key knowledge such as fisheries yield or fish abundance in relation to fishing effort, and qualitative and quantitative information on exploitation levels (Lorenzen et al. 2006; Lorenzen et al. 2016). These theoretical modelling tools are more commonly used for stock assessments in saltwater fisheries, but are being adapted for the data poor environment occupied by many freshwater fisheries (Lorenzen et al. 2016). The use of data collected from fisher surveys to populate theoretical models is a logical progression, and can provide greater insights than from the empirical data alone. However, before such modelling can be completed, information must be gathered from recreational fishers. On-site survey data is not often collected as they may require complex designs, are often expensive, and can require high labour input (Pollock et al. 1994). Therefore, optimising survey design to create comparable and generalizable results is a huge challenge.
2.1.2 Harvest regulations

Regulations governing recreational fishing are used to reduce harvest, provide equitable use of a fishery, and to manipulate fish populations (Cooke et al. 2013). The most common regulations include; creel or bag limits to regulate the number of fish that can be harvested; gear restrictions that dictate the type and amount of gear that can be used; size limits that direct the size of fish that can be harvested; and protected areas, seasonal or spawning closures that limit when and where anglers can fish (Cooke and Cowx 2006). These traditional regulatory options represent the standard in recreational fisheries management and have been successfully used to manage inland fisheries for hundreds of years. For example, catch limits were imposed in Massachusetts in 1652 because of overfishing by colonists at the time (Nielsen 1999). Size limits and fishery closures are common regulations currently used in fisheries worldwide to increase yield, safeguard against juvenile fish harvest, and to protect spawning fish (Allen et al. 2013; Arlinghaus et al. 2010; Cooke et al. 2013).

2.1.2.1 Size limits

Size limits, such as minimum length limits (MLL), are initiated to restrict harvest and also so that fish have the opportunity to reproduce before harvest eligibility (Winstanley 1992). The MLL sets the smallest length that a particular species can be legally harvested. It is largely used to lower fishing mortality, increase the number of large fish in the population, and improve angler catch relative to pre-MLL implementation (Allen and Pine 2000). By increasing the length at which fish are caught, the effective size of the spawning population also increases, which allows MLLs to assist in the control of both growth and recruitment overfishing (Hill 1990). MLLs also assist in the protection of immature fish, control of the number and size of fish landed, maximisation of marketing and economic benefits from commercial landings, and the promotion of aesthetic values of fish (Winstanley 1992). MLLs concentrate harvest on larger fish, which can maximise yield (Arlinghaus et al. 2010). But fish populations can also truncate in size at the MLL when fishing pressure in high (Nicol et al. 2004a). An alternative to MLLs are harvest slot limits, where intermediate lengths (ages) of fish are open to harvest, which are thought to maintain high yield and promote a more natural age-structure through the protection of large fish (i.e. longer than the upper length limit; Gwinn et al. 2015).
Determining the optimal MLL or harvest slot limit as part of a sustainable fishery management program is a challenging and imprecise process. Hindsight has shown that poor management of a fishery can lead to the collapse of the population, such as the decline of the Atlantic cod, *Gadus morhua* (Hilborn and Litzinger 2009; Hutchings and Myers 1994). The implementation of fishing regulations requires consideration of biological information, such as age- and length-at-maturity, in order to set appropriate size limits (Allen et al. 2009; Zukowski et al. 2012) in addition to understanding other stressors. The effectiveness of size limits, and the ability to assess current regulations, is dependent on the variable growth rates and recruitment exhibited by fish populations (Allen and Pine 2000). In addition, data from well-designed angler surveys offer detail about recreational effort, catch and harvest which inform on fishery status and how anglers use a fishery, that can be used to assist decisions regarding setting of harvest regulations (Buynak et al. 1991). Length-based harvest regulations are simple, and can be effective to recover fish populations in decline when set using scientific information such as the onset of first maturity (Richards and Rago 1999). However, the absence of science-based assessments makes it unclear whether length-based harvest restrictions are ensuring sustainable fisheries. Therefore, more research is required to collect information on maturation and growth that inform the setting of suitable size limits.

2.1.2.2 Spawning closures

Management of fisheries using spawning closures is driven by the need to increase fish abundance, either by protecting fish from harvest or through protection of spawning populations during reproductive activities (Zukowski et al. 2012). The impact of fishing may be exacerbated when fishing influences adult fish spawning behaviour and/or the survival of eggs (Butler and Rowland 2009). Closed seasons aim to protect breeding populations and if established across a well-defined season, can allow fish to breed without interference (Luckhurst and Trott 2009).

In the marine environment, establishment of closed seasons to protect spawning aggregations of fish may reverse species decline and provide sustainable fisheries (Luckhurst and Trott 2009). For example, after identification of two red hind, *Epinephelus guttatus*, spawning aggregation sites on the Bermuda reef platform in the early 1970s, legislation to close these two areas to all fishing for the four month (May-August) spawning period was enacted (Luckhurst and Trott 2009). Over a 30 year
period, the average length of red hind grew from 349 mm to 479 mm, whilst still supporting a commercial fishery.

Spawning closures have also been demonstrated to be effective on freshwater species, particularly those species with defined spawning periods in their life history, such as largemouth bass, *Micropterus salmoides* (Gwinn and Allen 2010). Largemouth bass in south-eastern USA spawn in April/May (spring), when males create a depression in the substrate of shallow areas of lakes and rivers. Spawning occurs at the nest site, following which males guard the eggs until the larvae become free swimming (Romero and Allan 1975). During this time, male largemouth bass demonstrate increased aggression (particularly larger males) and are more susceptible to angling (Suski and Philipp 2004). As large males guard bigger broods, protection of fish during the spawning period could increase survival of a greater percentage of juveniles due to decreased nest abandonment caused by fishing (Suski and Philipp 2004).

Further research to quantify the effects of recreational fishing on a range of species protected by closed seasons is necessary to understand the relative success of seasonal fishing closures. The collection of survey data (effort, catch-and-release and harvest) for fisheries protected by closed seasons can inform this process and provide an evidence-based assessment of the regulations effectiveness (Dunlop and Mann 2012).

### 2.1.3 Stocking

Stock enhancements are a set of management approaches that use artificially propagated fish to enhance or restore fisheries in natural ecosystems (Lorenzen 2014). Stock enhancement can be performed for the purposes of mitigating fishery declines, enhancing existing fisheries, restoration after a fish kill event, or for the creation of new fisheries (Cowx 1994)

The majority of stock enhancements are driven by the need (real or perceived) for enhancement stemming from angler feedback (whether anecdotal or from probability-based surveys) regarding fishery status (Cowx 1994). Assessment of population status may be significantly underestimated, which may stem from natural population fluctuations, or simply that estimates of the fishery potential are unrealistically high. If a fishery is already restricted in its potential, or reduced by natural population cycles, it is
unlikely that stocking will have a beneficial long-term effect (Cowx 1994). If a fishery is of a desired quality, there is little need for stocking, as this practice is less effective where there is adequate natural recruitment (Ellison and Franzin 1992). However, culture-based fisheries, where recruitment is largely or entirely based on hatchery releases, require ongoing stocking to sustain the fishery (Michaletz et al. 2008). An advantage of culture-based fisheries is that release and harvesting regimes may be designed to maximize production (Lorenzen 2005).

Stock enhancement has inherent risk associated with hatchery-reared fish introduction and the interactions between stocked and wild fish (Gillanders et al. 2006; Ingram et al. 2011; Nock et al. 2011). The risks associated with stocking may include loss of genetic integrity in wild fish (Rourke et al. 2010), and a shift in population structure from wild to hatchery fish (Jerry 1997). As such, fishery managers need to understand the relative contributions of stocked and wild fish in a population for understanding stocking program success, and the likelihood of fishery improvement and recovery.

Recent improvements in aquaculture technology have led to proliferation of global stocking programs, which has implications for the management and conservation of fisheries (Lorenzen 2014). For example, approximately one third of the Alaskan salmon harvest, *Oncorhynchus* spp, was found to originate from community-based aquaculture programs (Knapp et al. 2007), suggesting that the wild fishery may be imperilled. Many stocking programs fail or do ecological harm but persist regardless (Lorenzen 2005). Assessment of stocking programs is therefore vital to inform management of enhanced fisheries. A crucial advance in assessment of stocking programs is the development of population dynamics models. Using key biological parameters obtained from experimental studies in theoretical models, it is now possible to evaluate outcomes of stocking programs under various management scenarios (Lorenzen 2005; Lorenzen et al. 2006). Effectively, appraisals can be made prior to major investments being undertaken (Lorenzen 2014). However, despite the utility of such models, their accuracy relies on the collection of high-quality field-based data.
2.2 Conservation status, threats and reasons for decline of native fish species in the Murray-Darling Basin, Australia

In terms of abundance and diversity, native fish populations across the MDB are currently estimated to be about 10% of their pre-European settlement levels (Murray-Darling Basin Commission 2003). Without appropriate interventions, this level could decline further. However, given the continued need for human extraction of water resources, it would not be possible to return rivers and native fish to their pre-European settlement condition, and using all feasible rehabilitation options, a recovery level of 60% was seen as an achievable goal over a 50-year timeframe (Murray-Darling Basin Commission 2003).

Native fish decline was evident through changes in species composition and abundance between 1940 and 1992 for migratory fish using the fishway on the Murray River at Euston (Mallen-Cooper 1996). Major declines in Macquarie perch, *Maccullochella macquariensis*, (now absent), silver perch, *Bidyanus bidyanus*, (94% reduction) and Murray cod (96% reduction), and an increase in the proportion of introduced species from 1.3% to 62% (which related to the spread of carp, *Cyprinus carpio*, in the 1960s) were identified (Mallen-Cooper 1996).

The basin-wide decline of Murray cod in particular led to this species being listed in 1996 as critically endangered by the International Union for Conservation of Nature and Natural Resources (Wager 1996). Within Australia, Murray cod are listed as vulnerable under the Environment Protection and Biodiversity Conservation Act 1999 (Lintermans et al. 2005). A variety of factors have contributed to the decline in native fish abundance and distribution since European settlement. Eight major threats are recognised which include: flow regulation, habitat regulation, reduced water quality, in stream barriers, introduced species, disease, exploitation and translocation and stocking (Murray-Darling Basin Commission 2003).

2.2.1 Flow regulation

Regulation of most MDB rivers to provide water for irrigation and other purposes has greatly affected native fish populations (Cadwallader 1989; Lintermans et al. 2005;
The building of dams, storing and extraction of water and the regulation of flows is predicted to have the following effects; less water in the rivers; changes to normal seasonal flows (and possible reserving of seasonality); unnatural constant flows; reduced occurrence of small to medium sized floods; reduction in the size, rate of rise, and duration of floods; unseasonal irrigation releases reduce the occurrence of low flows; natural free-flowing rivers changed to sequences of pools because of weir construction; some wetlands and floodplains continually saturated, rather than the normal wetting and drying (see Lintermans 2007). These conditions affect water quality and flow regimes that may trigger fish migration, re-colonisation and spawning (Cadwallader 1989). River regulation also reduces habitat diversity and availability (Boys 2007), and the flows necessary for fish breeding and recruitment (Humphries et al. 1999; King et al. 2003; King et al. 2009). In addition, water extraction for irrigation removes larval and juvenile fish from the main channel into unsustainable habitats (Baumgartner and Boys 2012; Baumgartner et al. 2009; King and O’Connor 2007). The impacts of river regulation combine to decrease native fish species diversity and abundance.

2.2.2 Habitat degradation

The MDB contains a wide variety of aquatic habitat including large and small rivers and streams in upland and lowland environments, floodplains, wetlands, billabongs and lakes. Each of these systems contains a range of habitats such as weed beds, undercut banks, rocks, logs and riparian vegetation, which provide shelter, resting areas, spawning sites and food sources for fish (Lintermans 2007; Lintermans et al. 2004). These habitats and associated fish populations have been significantly impacted by factors such as de-snagging and removal of riverside vegetation (Koehn et al. 2004; Robertson and Crook 1999), channelization, realignment or reconstruction of river banks (Cadwallader 1989), the introduction of invasive plants such as willows (Erskine and Webb 2003; Zukowski and Gawne 2006), and siltation of rivers stemming from erosion of banks attributed to agricultural practices (Cadwallader 1989). Human-related impacts degrade fish habitat in a number of ways. For example:

- Increased sediment could smother spawning sites for adhesive egg laying fish such as Murray cod and Macquarie perch (see Ingram et al. 2000; Rowland 1983b), and also fills in deep holes reducing the amount of available habitat.
• Suspended silt increases turbidity that in turn alters heat radiation which may create unfavourable substrate conditions (Ellis 1936), which could subsequently alter the composition of benthic flora and fauna leading to negative effects on food chains utilised by fish (Cadwallader 1989)

• In stream woody habitat or ‘snags’ are important to native fish species for predator avoidance, shade, water velocity disruption, spawning sites, and to mark territories (Koehn et al. 2004; Robertson and Crook 1999). In addition, the presence of snags creates a range of habitats, such as deep pools and scour holes, along with variation in water velocity (Koehn et al. 2004), which is important to offer habitat for all size classes and species of fish (Nicol et al. 2004b; Robertson and Crook 1999). Removal of these structures has a significant impact on native fish.

• Loss of riparian vegetation results in less shade. In addition, the removal of large trees from the shoreline will result in fewer trees and tree limbs to augment in-stream habitat (Cadwallader 1989).

Habitat rehabilitation is a key factor toward the recovery of native fish species in the MDB, however full recovery of habitats to pre-European conditions is not believed to be attainable (Lintermans 2007).

2.2.3 Lowered water quality

Water quality impacts directly on native fish population health (King et al. 2012; Murray-Darling Basin Commission 2003). Factors that contribute to poor water quality include;

• Pollutants and poisons
• Drought, salinity and black water
• Discharges from industries and sewage treatment works
• Deep impoundments

The toxic effects of pollutants and poisons introduced to waterways have acute and long term effects on native species, including large scale fish kills (Cadwallader 1989). Similarly, the impact of drought and hypoxic black water events are shown to have significant impacts on fish populations through depletion of oxygen and increased toxins diffused into the water from leaf litter accumulations (Whitworth et al. 2012). Whilst many adult fish are demonstrated to be tolerant of high salinity, the early life stages of these same species are highly susceptible, resulting in diminished survival
under adverse saline conditions (James et al. 2003). Deep impoundments are problematic for native fish species that inhabit the river downstream, when water released from the dam is colder than surface layers. As many native fish species use increasing water temperature as a cue to commence spawning (Koehn and Harrington 2006; Rowland 1983a; Rowland 1983b), cold water releases during the breeding season can inhibit migration and spawning activities (Sherman et al. 2007; Todd et al. 2005). In addition, cold water slows growth rates of juvenile fish, exposing them to higher predation risk (Lyon et al. 2008).

2.2.4 Barriers to fish movement

The construction of dams, weirs and other barriers prevent fish moving from one part of a waterway to another. Fish movement is necessary for feeding, reproduction, colonisation of new habitat, or to escape unfavourable river conditions, such as a blackwater event (see Baumgartner 2007; King et al. 2012; Reynolds 1983). Native fish use different parts of a river system at various stages in their life cycle. For example, golden perch are able to live and grow effectively in still, impounded waters, but require access to flowing riverine environments for effective spawning and recruitment (Mallen-Cooper and Stuart 2003; Rowland 1983a). Without access to suitable riverine environments to reproduce, golden perch stocks must be supplemented by stocking to form put-and-take fisheries (Brown and Hall 2003). Large barriers such as weirs and dams significantly restrict fish movement and require construction of complex fishways and lifts to allow fish passage (Barrett 2008). In addition, smaller scale barriers such as regulators in lowland forests and levee banks restrict movement of fish between floodplains and rivers (Jones and Stuart 2008). Approximately 4,000 barriers exist across the MDB, with fishways constructed on about 55 of these barriers (Lintermans 2007). Many of these fishways are ineffective as they were built for strong swimming northern hemisphere species such as salmon, and offer only limited native naïve fish passage (Barrett and Mallen-Cooper 2006).

2.2.5 Introduced species

are considered pests (Nicol et al. 2004c), others such as the salmonids, form important recreational fisheries (Faragher et al. 2007), with fish such as redfin considered both a pest (Rowland 1989) as well as valued by recreational fishers (Henry and Lyle 2003). The abundance and attributes of some introduced fish are damaging to habitats and populations of native fish species. For example, as the suitability of habitat for native species has declined due to river regulation, it has improved for carp (Nicol et al. 2004c). As such, carp have been able to out-compete native species for food and other resources such as positions in the river that maximise feeding and refuge from flows and predation (Koehn et al. 2000). Redfin, trout and mosquito fish also impact on native fish through competition, predation or introduction of diseases and parasites (Cadwallader 1989). The control of introduced species already in the MDB is listed as a priority for current rehabilitation programs (Davies et al. 2010; Koehn and Lintermans 2012; Murray-Darling Basin Commission 2003), with the risk of further introductions from aquaria and other rivers outside the MDB highlighted as a consideration (Lintermans 2007).

2.2.6 Disease and parasites

In addition to the direct impacts of introduced species through competition and predation, alien fish have brought viral and parasitic pathogens that are able to infect native fish species. For example, redfin are the main host for Epizootic Haematopoietic Necrosis Virus (EHNV), which is a disease characterized by sudden high mortalities of fish displaying necrosis of the renal haematopoietic tissue, liver, spleen and pancreas (Langdon 1989). Murray cod infected under laboratory conditions with EHNV were found to be less susceptible than other experimentally infected freshwater fish species, but were potential carriers of the virus (Ingram et al. 2005).

Carp or redfin are suspected of introducing the parasitic copepod known as anchor worm Lernaea spp. into Australia (Lintermans 2007). Lernaea infestations may cause gill damage, haemorrhaging, muscle necrosis and an intense inflammatory response, sometimes associated with secondary bacterial infections (Marina et al. 2008). The parasite has been recorded on numerous large and small bodied species including; trout, carp, goldfish, Murray cod, golden perch, silver perch, Macquarie perch, river blackfish, Gadopsis marmoratus, catfish, Tandanus tandanus, southern pygmy perch, Nannoperca australis, and mountain galaxias, Galaxias olidus (Lintermans 2007). Whilst introduced
pathogens cause disease, endemic fungal, protozoan and parasitic organisms are also able to cause disease outbreaks with potentially devastating effects on native fish populations (Rowland and Ingram 1991).

2.2.7 Exploitation

Commercial and recreational overfishing have contributed to the decline of several freshwater native fish species including trout cod, *Maccullochella macquariensis*, Macquarie perch, golden perch and Murray cod (Allan et al. 2005; Allen et al. 2009; Cadwallader 1989; Hunt et al. 2011; Reid et al. 1997b). Overfishing, whether by the commercial or recreational sector, is not considered to be the primary reason for observed declines, however the added pressure may inhibit recovery (Lintermans et al. 2005). Management regulations such as spawning closures, bag and size limits, and gear restrictions are designed to prevent excessive harvest, but illegal fishing remains a threat to some species (Lintermans et al. 2005; Rowland 2005).

The annual Murray cod commercial harvest exhibited an increase between 1940/41 and 1955/56, however after the mid-1950s, the commercial fishery declined sharply (Rowland 1989; Rowland 2005). The decline in Murray cod abundance after 1955 was not restricted to New South Wales, with a reduction in commercial harvest of Murray cod in South Australia (Reynolds 1976). Commercial harvest remained low from 1980/81 to 1996/97 (Reid et al. 1997a). As a result of the decline, the Murray cod commercial fishery was ceased in New South Wales in 2001, Victoria in 2002 and South Australia in 2003. Pressure on Murray cod populations by recreational fisherman was further reduced by the introduction of closed seasons, bag and size limits in 1992, and the implementation of stocking programs (Lintermans et al. 2005).

Until 1951/52, Murray cod were shown to comprise 42–65% of the total annual catch from inland New South Wales, however after this time, golden perch replaced Murray cod as the predominant species in commercial harvests (Rowland 2005). Whilst the Murray cod commercial fishery in South Australia ceased in 2003, golden perch remain an important fishery in that state with harvest from both riverine and lake environments (Ye 2004). The stock status of the South Australian golden perch commercial fishery is considered fully exploited suggesting that current catches are close to optimum levels, but increases in harvest may lead to overfishing (Ye 2004). Golden perch stocks may
become increasingly vulnerable in South Australia if targeted effort increases in both commercial and recreational sectors (Ye 2004).

The cessation of the commercial fishery for Murray cod and golden perch in New South Wales and Victoria, and for Murray cod in South Australia, has reduced harvest and is thought to have contributed to species recovery in some areas (Rowland 2005; Ye 2004; Ye et al. 2000). However, harvest by recreational fishers can equal or exceed that from commercial fishers, and contribute to species decline (Cooke and Cowx 2004; Post et al. 2002). As such, assessment of harvest restrictions governing the recreational fishery for Murray cod and golden perch is important to protect the spawning stock from overfishing.

2.2.8 Stocking

The stocking of hatchery-reared fish has been a major management tool used in efforts to enhance and rehabilitate Murray cod and golden perch populations (Rowland and Tully 2004). Stocking is commonly used in waterways that are unable to sustain fish populations, such as impoundments without access to spawning habitat, or in rivers upstream of barriers to fish movement (Hunt et al. 2010).

The use of poor hatchery practices in the rearing of native fish can reduce genetic fitness, introduce disease and inadvertently translocate native or alien fish species to new waters (Rowland and Tully 2004). For example, closely related species and subspecies of Murray cod and golden perch have been identified in other drainages and discrete populations within the MDB. Inappropriate stocking with fish from outside these unique populations could reduce genetic fitness (Rourke et al. 2011). Similarly, wild fish populations are at risk from the release of genetically restricted individuals produced from hatcheries using limited numbers of broodstock (Rowland and Tully 2004). In addition, pathogens transferred on hatchery fish, may reduce survival and introduce diseases to new areas (Rowland and Ingram 1991; Rowland and Tully 2004).

Despite the potential negative effects of stocking, the practice continues often without assessment of whether management goals to enhance and recover fish populations are being achieved. Additional research to assess the effectiveness of stocking Murray cod
and golden perch is required to identify whether stocked fish are surviving, and the relative contributions that stocked fish make to existing populations.

### 2.3 Murray cod and golden perch

Recreational fishers in the MDB commonly target Murray cod and golden perch (Allen et al. 2009; Hunt et al. 2010). Both species were once widespread throughout the system, however factors such as habitat degradation, river regulation, pollution and overfishing, have impacted populations (Murray-Darling Basin Commission 2003). The threats, threatening processes and their applicability to Murray cod and golden perch are summarised in Table 1.

**Table 1:** Threats to native fish in the Murray-Darling Basin and the applicability of these threats to Murray cod and golden perch. Adapted from Murray-Darling Basin Commission (2003); Lintermans et al. (2005). One tick represents minimal impact, 2 ticks moderate impact and 3 ticks significant impact.

<table>
<thead>
<tr>
<th>Threat</th>
<th>Threatening process</th>
<th>Murray cod</th>
<th>Golden perch</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow regulation</td>
<td>Loss of water to other uses, critical low flows, loss of flow variation, loss of flow seasonality, loss of low to medium floods, permanent flooding and high water, increased periods of no flow</td>
<td>✓✓</td>
<td>✓✓✓</td>
</tr>
<tr>
<td>Habitat degradation</td>
<td>Damage to riparian zones, removal of in-stream habitats, sedimentation</td>
<td>✓✓✓</td>
<td>✓✓</td>
</tr>
<tr>
<td>Lowered water quality</td>
<td>Increased nutrients, turbidity, sedimentation, salinity, artificial changes in water temperature, pesticides, and other contaminants</td>
<td>✓✓</td>
<td>✓✓</td>
</tr>
<tr>
<td>Barriers</td>
<td>Impediments to fish passage resulting from the construction and operation of dams, weirs, levees, culverts, etc., and non-physical barriers such as increased velocities, reduced habitats, water quality and thermal pollution (changes in water temperature)</td>
<td>✓✓✓</td>
<td>✓✓✓</td>
</tr>
<tr>
<td>Introduced species</td>
<td>Competition with and/or predation by carp; mosquito fish; oriental weatherloach, <em>Misgurnus anguillicaudatus</em>; redfin and trout</td>
<td>✓✓</td>
<td>✓✓</td>
</tr>
<tr>
<td>Exploitation</td>
<td>Recreational and commercial fishing pressure on depleted stocks, illegal fishing</td>
<td>✓✓✓</td>
<td>✓✓</td>
</tr>
<tr>
<td>Diseases</td>
<td>Outbreak and spread of EHNV (Epizootic Haematopoietic Necrosis Virus) and other viruses, diseases and parasites</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Translocation and stocking</td>
<td>The loss of genetic integrity and fitness caused by inappropriate translocation and stocking of native species</td>
<td>✓✓</td>
<td>✓✓</td>
</tr>
</tbody>
</table>
The recreational value of these fish to rural and regional communities is high (Primary Industries and Regions South Australia 2009), particularly the Murray cod which is revered because of its size, strength and longevity. However, the prevalence and tenacity of golden perch has increased this species profile from primarily a fish for harvest, to a worthy sportfish through nationwide fishing tournaments and changing angler attitudes toward catch-and-release (Hall et al. 2012). Murray cod and golden perch are icons of Australia’s inland fisheries and efforts to rehabilitate, conserve and recover populations in decline, and enhance and expand fisheries through directed and informed management choices is vital to ensure that these fish continue to offer sustainable fisheries.

### 2.3.1 Identification

**Murray cod**

Australia’s largest wholly freshwater fish, recorded to 180 cm in length and 113.5 kg in weight. Murray cod are a large, elongated, deep bodied fish with small eyes and short, rounded, depressed snout with a distinctly concave profile. The mouth is large with a protruding bottom jaw (McDowall 1996). The underside is cream to white with a green mottled pattern on the body and head. The tail is rounded (Figure 2; Lintermans 2007).


**Golden perch**

Golden perch are a moderate to large fish with a deep, laterally compressed body. It has large eyes and dorsal profile of head is strongly convex, with a markedly arched nape, particularly in big fish. Recorded up to 76 cm and 23 kg, but fish up to 5 kg are
considered of good size (McDowall 1996). The body colour is generally olive green with a yellow or cream belly. The mouth is large with the lower jaw protruding slightly. The tail is rounded (Figure 3; Lintermans 2007).

Figure 3: Golden perch, *Macquaria ambigua*  

2.3.2 Biology

Murray cod

The species matures at 4–5 years of age and 50–60 cm in length (Rowland 1998b), and spawns in spring and early summer when water temperatures exceed about 15 °C. The eggs are large (3–3.5 mm in diameter), adhesive and usually deposited onto a hard surface such as logs, rocks or clay banks (Lintermans 2007). Fecundity ranges from 10,000–90,000 in females 2.5–23 kg in weight (McDowall 1996). The male guards the eggs during incubation and they hatch after 5–13 days. The larvae are about 5–8 mm long at hatching and have a large yolk sac. Larvae drift downstream for 5–7 days, particularly by night in spring and summer, once they leave the nest (Lintermans 2007).

Murray cod are a long lived species with the oldest aged fish reported at 48 years old, 140 cm in length and 47.3 kg (Rowland 1998a), however significant variation in length and weight can occur from similar aged fish (Rowland 1998a). Murray cod were once described as sedentary (Jones and Stuart 2007), however a recent study showed that this species undertakes complex movements which follow seasonal patterns (Koehn et al. 2009). Whilst there were sedentary periods with limited home ranges and high site fidelity, Murray cod also undertook larger movements (up to 130 km) coinciding with spawning (Koehn et al. 2009). These movements comprised a migration from a home reach in late winter / early spring to a new upstream position, followed by a rapid downstream migration typically back to the same river reach (Koehn et al. 2009).
Golden perch

Golden perch are a long lived species with the oldest fish recorded at 26 years old (Stuart 2006). Males mature at approximately 2 years of age and females at approximately 4 years, and are thought to generally spawn over spring and summer when water temperatures exceed about 20 °C, often in association with floods (McDowall 1996). However more recent research has shown that golden perch are able to spawn during relatively stable within bank flows (Gilligan and Schiller 2003). Females are able to hold ova at an advanced stage of development until conditions are suitable for spawning (McDowall 1996). Golden perch are highly fecund (Rowland 1983a) with a 2.5 kg female reported to hold over 500,000 eggs (McDowall 1996). The water hardened eggs are 3–4 mm in diameter, semi-buoyant and drift downstream. Hatching occurs after 1–2 days and newly hatched larvae are about 3.5 mm in length (Linternmans 2007).

Both adult and juvenile golden perch are migratory with upstream migrations in excess of 1,000 km recorded for some adult fish (Reynolds 1983). Migrations may be triggered by rises in flow (McDowall 1996) with most movement recorded in spring, summer and early autumn in association with spawning activity (Reynolds 1983). During non-spawning periods of the year, golden perch have also been shown to occupy relatively small home ranges, where they may spend weeks or months before relocating (Crook 2004). Downstream migration of adult fish is reported with large numbers of golden perch moving downstream during periods of minimal flow (Cadwallader 1977).

2.3.3 Habitat

Murray cod

Murray cod occupy a wide variety of habitats from small, clear, rocky streams to turbid, slow flowing rivers and creeks. Murray cod are generally found in or near deep holes and prefer habitats containing cover such as rocks, fallen trees, stumps, clay banks or overhanging vegetation (McDowall 1996). A high salinity tolerance allows this species to occupy a wide variety of water qualities (James et al. 2003).

Golden perch

Golden perch are predominantly a fish of warm, turbid, slow flowing inland rivers and their floodplain lakes and anabranches (McDowall 1996), and prefer deep pool habitats with structural woody habitat such as fallen trees, undercut banks or rocky ledges.
Golden perch are well adapted to the variable stream flow conditions of the MDB (Harris and Gehrke 1994), can tolerate water temperatures from 4–37 °C (McDowall 1996), and have high salinity tolerance (James et al. 2003).

2.3.4 Distribution

Murray cod

The Murray cod was once widespread and abundant throughout most of the MDB with the exception of high altitude reaches of some southern tributaries. Murray cod have been extensively translocated outside their historic natural range including the Yarra River, Wimmera River and lakes Charlegrark, Green, and Taylor in Victoria, the eastern drainages of the Coxs, Nepean and Wollondilly rivers, Cataract Dam, Mulwarree Ponds and Lake George in New South Wales, and also into Western Australia (McDowall 1996). In addition, the development of artificial rearing techniques (Rowland 1983b) have extended the range of Murray cod through stocking into numerous farm dams and other natural waterways with fish raised by private and government hatcheries (Cadwallader and Gooley 1984). Despite extensive human-assisted movement of this species within Australia, most translocations failed to result in sustainable populations (Cadwallader and Gooley 1984).

Golden perch

Golden perch exist throughout the MDB with the exception of higher altitudes. The species is also found in the Lake Eyre and Bulloo internal drainage systems of Queensland, New South Wales and South Australia, and the Dawson-Fitzroy river system in south-eastern Queensland (McDowall 1996). Golden perch have been translocated into various coastal basins and stocked into impoundments and rivers across the MDB to form recreational fisheries (Brown and Hall 2003; Hunt et al. 2010). The natural range and abundance of golden perch steadily declined since European settlement (Battaglene and Prokop 1987; Cadwallader 1977). The species is described as rare or absent from large areas of MDB tributaries and higher reaches of the larger rivers (McDowall 1996). Golden perch are abundant in the lower MDB rivers (Reynolds 1983; Ye 2004).

2.3.5 Historical abundance

The fish of the MDB were extremely important in the lives of indigenous Australians and the early explorers as they travelled inland. White settlers also utilised inland river
fish resources as they expanded inland from coastal settlements. Fish such as the Murray cod were entrenched in indigenous mythology and remain a strong part of Australian culture.

Murray cod, golden perch and other native fish were important foods for tribes living on inland rivers with evidence of the use of complex fishing methods in the lower Darling River around 25,000 years ago (Balme 1983). Indigenous Australians used a variety of methods to catch fish including spears, nets, poisons and different types of fish traps made of brush fences, stones or hollow logs (Balme 1983; Rowland 2005; Smith 1930).

The inland explorers and early settlers were astounded by the abundance, size and delicacy of the Murray cod (Rowland 1989). For example, the explorer John Oxley describes Murray cod in the following extract:

“If however the country itself is poor, the river is rich in the most excellent fish, procurable in the utmost abundance. One man in less than an hour caught eighteen large fish, one of which was a curiosity from its immense size and the beauty of its colours . . . It weighed an entire 70 pounds, . . . Most of the other fish taken this evening weighed from fifteen to thirty pounds each” (Oxley 1820).

Murray cod were held in such high regard by the early explorers and settlers that they were considered a species worthy of acclimatization in England (O'Connor 1897). During the mid to late 1800s a large, commercial fishery developed and was based mainly on the Murray and Murrumbidgee rivers. (Dannevig and Stead 1903; Rowland 1989; Rowland 2005). By 1883, the Murray River fishery formed a considerable part of the fish supply to Melbourne, other Victorian cities and towns, and South Australia. The scale of the fishery is highlighted by the report of more than 147 tons of fish sent to Melbourne from the port of Moama in one year alone (Rowland 1989). In 1900, Murray cod accounted for 75% and golden perch the remaining 25%, of river fish available at the Melbourne markets (Poole 1984). The overall commercial catch peaked in 1918 (Dakin and Kesteven 1938) and by the mid-1930s, the commercial fishery had declined to an unprofitable level for large-scale operators (Pollard and Scott 1966). Such accounts illustrate the historical abundance of Murray cod and golden perch, importance to people at this time, and size of the fishery.
2.3.6 Management of Murray cod and golden perch

The key threats to native fish recovery in the MDB (i.e. flow regulation, habitat degradation, lowered water quality, barriers to migration, introduced species, exploitation, disease and stocking) are the subject of many studies (e.g. Crook et al. 2015; Humphries et al. 2002; Hunt et al. 2014). However, the focus of this study is to inform knowledge gaps relating to exploitation and stocking, that specifically relate to quantification of recreational fisheries, the suitability of size limits, and the effectiveness of stocking.

Declines in freshwater fish stocks in the Murray River system during the late 1800s led to the introduction of management regulations (Dakin and Kesteven 1938; Rowland 1989). In 1902 and 1936, state conferences were held to discuss the Murray River fishery. Management outcomes from these conferences included seasonal closures to protect spawning fish, introduction of size limits, and artificial breeding of Murray cod for the purpose of re-stocking (Dannevig and Stead 1903; Isherwood 1939). These management strategies demonstrate a long term commitment to recovery and rehabilitation of a fishery in decline. The use of harvest regulations, spawning closures, and a re-stocking program are management actions started in 1902 that are still being used for recovery of fish populations in the MDB today (Barwick et al. 2014).

Murray cod and golden perch are protected by size limits, bag limits, and a closed season (Murray cod only) that vary between states (Table 2). Assessment of management regulations is important to evaluate potential impacts of current harvest policies on angling quality and sustainability (Allen et al. 2009). However, harvest regulations and stocking programs are often used simultaneously, making it difficult to assess the positive and negative effects of each strategy (Allen et al. 2009). Regardless, quantification of key Murray cod and golden perch fisheries, and assessment of size limits and stocking, will inform management decisions regarding recovery and sustainability strategies.
Table 2: State-based bag and size limits for Murray cod and golden perch. New South Wales = NSW, Victoria = VIC, Queensland = QLD, South Australia = SA. Minimum legal length = MLL.

<table>
<thead>
<tr>
<th>State</th>
<th>Size limit</th>
<th>Daily limit</th>
<th>Possession limit</th>
<th>Closed season</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>NSW</td>
<td>min 550 mm max 750 mm</td>
<td>2</td>
<td>4</td>
<td>1 Sep – 30 Nov</td>
<td>(New South Wales Department of Primary Industries 2015)</td>
</tr>
<tr>
<td>VIC</td>
<td>min 550 mm max 750 mm</td>
<td>1</td>
<td>1 (rivers) 2 (lakes)</td>
<td>1 Sep – 30 Nov</td>
<td>(Victorian Department of Environment and Primary Industries 2015)</td>
</tr>
<tr>
<td>QLD</td>
<td>min 600 mm max 1100 mm</td>
<td>2</td>
<td>2</td>
<td>1 Sep – 30 Nov</td>
<td>(Queensland Government - Agriculture Fisheries and Forestry 2011)</td>
</tr>
<tr>
<td>SA</td>
<td>catch-and-release only</td>
<td>0</td>
<td>0</td>
<td>1 Aug – 31 Dec</td>
<td>(Primary Industries and Regions South Australia 2012)</td>
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<table>
<thead>
<tr>
<th>State</th>
<th>Size limit</th>
<th>Daily limit</th>
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<tbody>
<tr>
<td>NSW</td>
<td>300 mm</td>
<td>5</td>
<td>10</td>
<td>-</td>
<td>(New South Wales Department of Primary Industries 2013a)</td>
</tr>
<tr>
<td>VIC</td>
<td>300 mm</td>
<td>5 (rivers) 10 (lakes)</td>
<td>5 (rivers) 10 (lakes)</td>
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<tr>
<td>QLD</td>
<td>300 mm</td>
<td>10</td>
<td>10</td>
<td>-</td>
<td>(Queensland Government - Agriculture Fisheries and Forestry 2011)</td>
</tr>
<tr>
<td>SA</td>
<td>330 mm</td>
<td>5</td>
<td>15 per boat</td>
<td>-</td>
<td>(Primary Industries and Regions South Australia 2012)</td>
</tr>
</tbody>
</table>

2.3.6.1 Use of recreational fisher surveys to inform harvest regulations

In Australia, fisher surveys have been used sparingly as a tool assisting in the management of freshwater native fish species, despite recognition that assessment of fishery sustainability is critical for management, particularly for endangered species or for species in decline (Hunt et al. 2011). For example, Macquarie perch are a freshwater fish endemic to the south-eastern reaches of the MDB. Significant reductions in Macquarie perch range and abundance by the 1970s, led the species to be listed as nationally endangered. In order to evaluate whether a significant decline in Macquarie perch population size occurred in Lake Dartmouth, fisher surveys investigating total catch, harvest and catch per unit effort were compared for surveys from the mid-1980s
and 2000 (Hunt et al. 2011). Catch rates from the most recent survey were considerably lower than the historic survey, indicative of decline in the Macquarie perch population in this impoundment (Hunt et al. 2011).

Soundly designed fisher surveys for Murray cod and golden perch fisheries have not informed the management of these species. The first significant attempt to quantify the recreational fishery for Murray cod was a national survey conducted during 2000/2001 as a joint initiative of Commonwealth and State governments (Henry and Lyle 2003). This survey required anglers to record fishing events in a diary, which was conveyed to researchers utilising a telephone survey. Results from the national survey indicated that 108,352 Murray cod were harvested annually, whilst 374,932 or 77.6% were released (Henry and Lyle 2003). The large percentage of Murray cod released was highlighted as a priority for additional research on post-release survival, and fisher behaviour leading up to fish release (Henry and Lyle 2003).

A more detailed study on sustainability of Murray cod recreational fisheries was conducted using fisher surveys on the Goulburn, Ovens and Murray Rivers from 2006–2008 (Brown 2010). This study used harvest estimates and released component from the fisher survey in combination to evaluate the relative impact of catch-and-release fishing to assist decisions regarding sustainable management of the fishery. The release rate for Murray cod was about 90%, with most releases compulsory as the fish were smaller than the MLL (Brown 2010).

Fisher survey data has also been used to assess and manage golden perch fisheries. For example, in 1981/1982 a fisher survey was conducted at Lake Keepit, NSW, which estimated that; fishing effort was highest in spring and lowest in winter; effort decreased with falling water levels; catch rates were greatest in spring and late autumn; and that golden perch constituted 94% of total catch (Battaglene 1985). In the Goulburn, Ovens and Murray Rivers, the golden perch fishery was found to be significantly smaller than that for Murray cod, except in some river reaches where cod catch was relatively low (Brown 2010).

By integrating the knowledge and data obtained from fisher surveys and from studies of Murray cod and golden perch biology and ecology (e.g. Rowland 1983a; Rowland
1983b; Rowland 1989; Rowland 1998a; Rowland 1998b; Rowland 2005) fishery managers will have better resources to make informed decisions on this fishery. Given that few fisher surveys have investigated Murray cod and golden perch populations, expansion of this methodology to new waterways will provide estimates of fishing effort, catch and harvest to inform management regulations (e.g. spawning closures) across a broader geographic scope than currently exists. The absence of such data can lead to population decline caused by recreational fishing that can go un-detected (Post et al. 2002).

A closure during the Murray cod spawning period (September-November) was introduced in NSW in 1992 as a result of base-line biological data demonstrating reproductive traits such as egg re-absorption following handling prior to spawning (Rowland 1988). The closure also increased protection for the species during this period when Murray cod may be more susceptible to fishing. Murray cod seasonal spawning closures are an important aspect of the management and recovery of this species, enabling fish to spawn with reduced human interference, with a goal to decrease harvest of brood fish and increase survival of eggs and larvae (Rowland 1988; Rowland 2005). However the effects of fishing (e.g. fishing related mortality associated with catch-and-release) during the closed season for Murray cod are not understood. Such information is important to validate the benefits of Murray cod spawning closures.

The recovery of Murray cod populations in some NSW waters between 1995 and 2005 (Rowland 2005) was assumed to stem from management initiatives such as the protection of this species during the breeding season, the increase in MLL allowing more fish to reach sexual maturity, stocking of hatchery-reared fingerlings, and an overall decrease in fishing mortality (Rowland 2005). In addition, there were good years of natural recruitment, lower numbers of redfin and carp, and improved environmental conditions (Rowland 2005). The Murray cod recovery was based on anecdotal evidence and unpublished fishery independent data. Analysis of such data, and the collection of fishery-dependant data, is required to understand the effectiveness of spawning closures, and to quantify the success of recovery efforts.
2.3.6.2 Changes in size limits

Existing Murray cod and golden perch harvest restrictions are based on the length where 50% of fish are sexually mature (LM50; Allen et al. 2009; Brown 2010; Rowland 2005). Murray cod MLLs have been increased from 500 to 600 mm TL since its introduction in 1992. These changes were based on data collected between 1978 and 1984, which estimated Murray cod LM50 at between 500 and 550 mm (Rowland 1998b). The assumption was that most Murray cod are mature at 600 mm and that this occurs consistently across the species range, therefore allowing a greater proportion of Murray cod the opportunity to reproduce before they are able to be harvested (Rowland 2005). More recently, implementation of a harvest slot limit (where regulations use minimum and maximum lengths), rather than an increase in MLL, has been suggested to reduce the risk of further Murray cod decline, and increase catch rates (Gwinn et al. 2015; Koehn and Todd 2012). Harvest slot limits for Murray cod have been adopted by all states, with the exception of South Australia, who have a catch-and-release only fishery for this species (Ye et al. 2000). Harvest slot limits for Murray cod in New South Wales and Victoria were adopted in December 2014 (New South Wales Department of Primary Industries 2015; Victorian Department of Environment and Primary Industries 2015).

A 300 mm MLL exists for golden perch in all states with the exception of South Australia where the limit is 330 mm. Golden perch differ from Murray cod in that they have not been shown to spawn and recruit in standing water and form significant put-and-take fisheries in impoundments (Hunt et al. 2010). The golden perch MLLs of 300 mm (New South Wales, Victoria, Queensland) and 330 mm (South Australia) are similar to, or well below the reported LM50 for males (325 mm) and females (397 mm) sampled from the Murray River (Mallen-Cooper and Stuart 2003). This would indicate a low level of protection for the golden perch spawning population at the current MLLs.

Combining biological parameters such as age, length, and maturity status from a sample population allows calculation of age-at-length, and length- and age-at-maturity, which can be used to assess MLLs (Rowland 1998b). Length-based harvest restrictions can be applied without knowledge of growth or onset of maturity, but this potentially exposes fish populations to overfishing if the MLL is not sufficiently conservative (van Poorten et al. 2013). Variation in the onset of reproductive maturation can be influenced by
factors such as increased fishing pressure, which can lower length-at-maturity (Lappalainen et al. 2016). Stocking can influence the onset of maturation if densities change (Lorenzen 2005). These factors are relevant to Murray cod and golden perch populations. Variability in the onset of maturation for these species could influence the effectiveness of length-based regulations to prevent overfishing (Allen et al. 2009), but so far no studies have quantified the degree of variation in length- and age-at-maturity among waterbodies or among system types (e.g. rivers versus impoundments). Collection of such data is vital to set appropriate size limits based upon differences in maturation that may exist among a range of waterbodies for protection of spawning populations.

2.3.6.3 Stocking programs

Toward the end of the 1950s, reports indicated that populations of Australia’s inland native fish species were declining, highlighting a need to both improve fish stocks and manage the important commercial and recreational fisheries (New South Wales State Fisheries 1979). Thus, research into the biology of Australia's inland fish began. Artificial breeding of native freshwater fish is reported as early at 1916 with breeding trials involving Murray cod at Berembed Weir (Brodie 1916). However it wasn’t until the late 1960s that research into the spawning requirements of Murray cod and golden perch, detailed the development of eggs and larvae of these species (Lake 1967b). The construction of a dedicated breeding and research facility at Narrandera NSW in 1962, allowed research expansion into the breeding biology of inland fish (New South Wales State Fisheries 1979). Hatchery breeding techniques were developed for native fish species including Murray cod (Rowland 1983b) and golden perch (Rowland 1983a). Since 1971/1972, 12.89 million Murray cod (Ingram et al. 2011), and over 32 million golden perch (Gillanders et al. 2006) produced at both government and private (commercial) hatcheries across several states, have been stocked into waterways and impoundments across the MDB to support recreational fishing (Ingram et al. 2011) and for conservation and recovery efforts (Lintermans et al. 2005).

River regulation is listed as a key threat to native fish populations with over 10,000 barriers limiting longitudinal and lateral connectivity throughout the MDB (Baumgartner et al. 2014; Baumgartner 2007; Jones and Stuart 2008). Stocking is used to enhance fish populations in areas that have experienced decline following the loss of
connectivity. For example, hatchery-bred Murray cod were released into the upper reaches of rivers on the NSW Northern Tablelands in the late 1970s (Rowland 2005). These stockings improved Murray cod abundance and distribution in the Gwydir River below Copeton Dam, where the species had declined significantly since the dam’s construction (Rowland 2005). Stocking plays a major role in the conservation and rehabilitation of Murray cod and golden perch populations, but assessment of the effectiveness of these stocking programs is required to optimise release strategies and support ecologically and economically defensible decision making. For example, despite closure of the New South Wales (NSW) Murray cod commercial fishery in 2001, and implementation of recreational harvest regulations such as bag and size limits in 1992 (Rowland 2005), stocking of this species continues based on the perception that natural recruitment is insufficient to sustain the fishery. Natural recruitment in the MDB is restricted by factors such as habitat degradation, river regulation and exploitation, and there is a compelling need to understand why natural recruitment is failing (Murray-Darling Basin Commission 2003). However, overlapping stocking programs and natural recruitment in many waterbodies create ambiguity as to the relative success or failure of either process. In recognition of the need for evidence regarding the outcomes of fish stocking, an initial study was undertaken between 2002 and 2007 to assess the effectiveness of stocking golden perch in the Edward River, Murrumbidgee River, and Billabong Creek (Crook et al. 2015). Hatchery-reared golden perch were marked with calcein (Billabong Creek) and alizarin complexone (Murrumbidgee and Edward Rivers), and demonstrated that stocked golden perch contributed 18–100% to populations (Crook et al. 2015). Other less extensive assessments of golden perch stocking in the MDB have been performed (see Harris 2002; Hunt et al. 2010), but the outcomes of Murray cod stocking have never been studied in detail, which presents a key knowledge gap.

2.4 Knowledge gaps to inform sustainable management of Murray cod and golden perch

Identification and analysis of population dynamics influencing the Murray cod and golden perch recreational fishery are essential for sound scientific advice to be gathered on aspects of the fishery that can be changed by managerial decision making. Current management decisions regarding the New South Wales Murray cod and golden perch
fishery are predominantly made using data collected two or more decades ago (see Cadwallader and Gooley 1985; Lake 1967b; Rowland 1983a; Rowland 1983b). These early studies focussed on captive Murray cod with research into the biology of wild Murray cod described for the first time in 1998 (Rowland 1998b). Relevant, accurate and up-to-date data to inform management strategies used to regulate Murray cod and golden perch fisheries are vital to conserve and improve these fisheries. New information collected should; (a) provide an understanding of stocking program success and the extent of natural recruitment where threats such as exploitation and habitat degradation have caused population decline; (b) quantify maturation onset in a range of waterbodies to inform the setting of size limits so that exploitation pressures do not lead to recruitment overfishing because of inappropriate size limits; and (c) obtain estimates of effort, catch and harvest from important recreational fisheries to inform harvest restrictions such as closed seasons, so that the interactions between anglers and the fishery do not lead to overfishing or inhibit spawning activities.

2.4.1 Recreational fisheries poorly understood

Few angler surveys exist that quantify the basic components of Australian freshwater recreational fisheries such as fishing effort, catch and harvest, despite the importance of such data to inform management regulations such as harvest restrictions and closed seasons (Hunt et al. 2011). An Australian national survey undertaken in 2000-01 estimated annual catch of 483,000 Murray cod, with 77.6% of these released (Henry and Lyle 2003). The same survey also estimated annual catch of 1,858,000 golden perch, with 44% of these released (Henry and Lyle 2003). Recently, a series of surveys focusing on Murray cod in the Murray, Goulburn, and Ovens rivers have provided annual estimates of recreational fishing effort, catch and harvest in these fisheries (Brown 2010). Despite this, few studies of riverine Murray cod and golden perch populations utilizing fisher survey data exist, which represents a key knowledge gap in the management of these important native species. Accordingly, Chapter 3 aims to: (1) provide estimates of harvest, catch, harvest rate, catch rate and effort for Murray cod and golden perch, and (2) investigate implications from fishing during the Murray cod closed season, in a popular recreational fishing reach of the Murrumbidgee River that has previously experienced declines in native fish stocks. Chapter 4 expands the knowledge gained from the riverine study in Chapter 3 by; (1) quantifying the fishery dependent characteristics in an impounded system (Lake Mulwala); and (2) assessing
variability in standardised measures of effort, catch, harvest, catch-rate and harvest-rate that may exist among fisheries for Murray cod in different systems.

2.4.2 Quantify maturation onset in a range of waterbodies

MLLs are typically set above a species length-at-maturity and aim to lower fishing mortality, increase the number of large fish in the population, and improve fisher catch rates compared to pre-limit conditions (Hill 1990). Variation in the onset of reproductive maturation for Murray cod and golden perch could influence the effectiveness of length-based regulations to prevent overfishing (Allen et al. 2009), but so far no studies have quantified the degree of variation in length- and age-at-maturity for these species among waterbodies or among system types (e.g. rivers versus impoundments). Therefore, Chapter 5 aims to quantify variation in Murray cod and golden perch length- and age-at-maturity, and growth parameters, within several MDB riverine and impounded systems, and investigated whether fish length is a suitable surrogate for age-at-maturity in these species. Understanding variability in the onset of reproductive maturation will inform fishery managers on the appropriateness of basin-wide size limits. Identifying whether length is a suitable predictor of maturity seeks to validate the management assumption that setting size limits based on length is reflective of reproductive maturation onset. Collection of such data is vital to set appropriate size limits based upon differences in maturation that may exist among a range of waterbodies, for protection of spawning populations.

2.4.3 Assessment of stocking effectiveness

Fishery managers must assess whether a particular fishery should be stocked, and with how many fish, and also at what time these fish should be added but these factors are rarely considered. Stocking is important for conservation and rehabilitation of Murray cod and golden perch populations (e.g. Brown and Hall 2003; Hunt et al. 2010; Ingram et al. 2011; Rowland 1995; Rowland and Tully 2004) with assessment of the effectiveness of these respective stocking programs required. For example, despite closure of the New South Wales Murray cod commercial fishery in 2001 and re-introduction of harvest regulations such as bag and size limits in 1992 (Rowland 2005), stocking of this species continues in rivers and impoundments based on the assumption that natural recruitment is insufficient to sustain the fishery.
A major study assessed the effectiveness of stocking golden perch, which showed that stocked fish survived to reach the minimum legal length in the Edward River, Murrumbidgee River, and the Billabong Creek (Crook et al. 2015). Stocked golden perch contributed 18–100% of age classes corresponding to stocking years in these rivers (Crook et al. 2015), and demonstrates that stocking has the potential to significantly alter population structure. Further research into the survival of stocked Murray cod and golden perch and the contribution of stocked fish to existing populations in a variety of rivers and impoundments is required to quantify the assumption that stocking is necessary to support the wild fishery due to insufficient natural recruitment. As such, Chapter 6 aims to build on existing research by Crook et al. (2015) through addition of Murray cod to the analysis, inclusion of additional riverine study sites, and extend stocking assessments to impoundments. The research will generate vital information on the survival, contribution, growth and population structure of stocked Murray cod and golden perch in both riverine and impounded systems.

2.5 Conclusion

Fishery management strategies can help rehabilitate and recover fish populations, however more research is required into the effectiveness and efficiency of several management initiatives including fish stocking, size limits and recreational catch, harvest and effort so that evidence-based decisions can be made. Identification of factors involved in shaping the fishery, and their impacts, are vital for informed decision making so that fishery health is preserved and where possible, enhanced. This will be achieved by utilising outcomes from each of the research objectives to build on historical knowledge to gain a clearer identity of the factors and their importance. The research will make management recommendations and strategies based on information collected in an effort to sustain and improve the New South Wales Murray cod and golden perch recreational fishery.
Chapter 3
Recreational fishing effort, catch, and harvest for Murray cod and golden perch in the Murrumbidgee River, Australia

Chapter 3 has been published as:

3.1 Abstract

The provision of sustainable fisheries and prevention of species decline are important aspects of recreational fisheries management. Overfishing is frequently suggested as a cause of historic fishery decline within the Murray-Darling Basin (MDB), Australia, but few quantitative surveys exist to provide fishery-dependent data to gauge status. Murray cod and golden perch are important target species of recreational fishers across the MDB. The fisheries are controlled by size and bag limits and gear restrictions (both species) as well as a closed season (Murray cod only). A complemented fisher survey design was used to assess the recreational fishery for both species in a 76 km reach of the Murrumbidgee River in 2012-13. Progressive counts were used to quantify boat- and shore-based fishing effort. Catch and harvest rate information was obtained from shore-based fishers via roving surveys and from boat-based fishers via bus route surveys. Murray cod catch rates (fish/angler-hour) were 0.228 ± 0.047 (mean ± SE; boat-based) and 0.092 ± 0.023 (shore-based), and harvest rates (fish/angler-hour) were 0.013 ± 0.006 (boat-based) and 0.003 ± 0.001 (shore-based). Golden perch catch rates were 0.018 ± 0.009 (shore-based) and 0.002 ± 0.001 (boat-based), and harvest rates were 0.006 ± 0.002 (shore-based) and 0.001 ± <0.001 (boat-based). The Murray cod fishery had maximal catch and harvest during the 5-month period after the closed season ended. The closed season aims to protect spawning Murray cod, but this strategy’s effectiveness may have been influenced by high fishing effort and deliberate bycatch during the closure period. To sustain and improve these MDB fisheries, I suggest quantification of catch-and-release impacts on spawning Murray cod, provision
of fish passage, re-stocking of golden perch, and education on fishing techniques that minimize Murray cod bycatch during the closed season.

### 3.2 Introduction

Fishery exploitation has become a global management issue with recreational and commercial fishers contributing to harvest in marine and inland waters. Commercial fishing is often suggested as a reason for fish population decline in marine fisheries (e.g. Fromentin and Powers 2005; Hilborn and Litzinger 2009). However, recreational fishing can also have a negative impact with an estimated 12% contribution to global fish harvest (Cooke and Cowx 2004). Recreational fishing may also contribute to declines in inland fish species, such as the rainbow trout, walleye, *Stizostedion vitreum*, Northern pike, *Esox lucius*, and lake trout, *Salvelinus namycush*, in Canada (Post et al. 2002), and the striped bass in North America (Boreman and Austin 1985). Measurement of angler impact within a fishery requires knowledge of recreational catch, harvest and fishing effort, but such data are often lacking (Cooke and Cowx 2004).

Recreational fisher surveys of sound design and implementation are essential tools for effective fisheries management (Pollock et al. 1994). Trends derived from long-term probability-based surveys are often used to assess exploited recreational fisheries and to determine whether management interventions are effectively rehabilitating populations (Kozfkay and Dillon 2010). Fisher surveys can also be used to evaluate angler attitudes toward fishing regulations (e.g. fishing closures and bag limits) and management programs (e.g. stocking and habitat restoration) thus providing information on the economic value of fishing to anglers and communities (Campbell and Murphy 2005; Henry and Lyle 2003; Wilberg 2009). If well designed, fisher surveys can also be used in conjunction with fishery-independent data to determine the contribution of recreational fishing to fish mortality (e.g. Douglas et al. 2010; Kozfkay and Dillon 2010).

The Murray-Darling Basin (MDB) is Australia’s largest catchment covering over 1 million km² (Barrett 2008). Fisheries within the MDB have declined markedly over the past 100 years, and native fish communities are estimated to be as low as 10% of pre-European settlement levels (Murray-Darling Basin Commission 2003). Commercial
exploitation and recreational exploitation may have been major contributors to declines in many large-bodied species (Murray-Darling Basin Commission 2003; Rowland 1989). However, recovery of some MDB fish populations is evident. For example, data collected from 28 long-term monitoring sites across New South Wales (NSW) indicated that the abundance of Murray cod increased by 740% between 1994 and 2011 (D. Gilligan, NSW Department of Primary Industries, unpublished data, cited in Rowland 2013).

Management strategies can assist recovery efforts include closure of commercial fisheries, recreational harvest restrictions, re-stocking programs, habitat restoration, restoration of fish passage, and closures to protect fish during their breeding season (see Allan et al. 2005; Barwick et al. 2014; Lintermans et al. 2005). Despite recognition that assessment of fishery sustainability is critical for management there have been few angler surveys on MDB recreational fisheries (Hunt et al. 2011). Fishing regulations (e.g. bag limits, minimum sizes or harvest slots) can be implemented without quantitative data to support decision making; however the absence of data from the recreational fishery makes it difficult to ascertain if the desired management outcomes have been achieved (Jones and Pollock 2013).

Fisheries in the MDB contribute an estimated AU$1.3 billion (i.e., $1.3 \times 10^9$) annually to the Australian economy (Ernst and Young 2011). The two main recreational target species are the Murray cod and golden perch (Allen et al. 2009; Brown 2010; Hunt et al. 2010; Rowland 2005). The Murray cod is listed by the Commonwealth as ‘vulnerable’ under the Environment Protection and Biodiversity Conservation Act of 1999; the species is also listed as ‘critically endangered’ according to the International Union for Conservation of Nature (Ingram et al. 2011; Wager 1996). Despite this, angling for Murray cod is permitted throughout the species range. Traditionally, fishers preferentially targeted Murray cod (Rowland 1989), but effort toward golden perch has recently increased via activities such as species-specific fishing tournaments and due to changing angler attitudes toward catch-and-release (Hall et al. 2012; Ye 2004).

An Australian national survey undertaken in 2000-01 estimated the annual catch of Murray cod at 483,000 fish, with 77.6% of those individuals released (Henry and Lyle 2003). The same survey also estimated the annual catch of Golden perch at 1,858,000
fish, with 44% of those individuals released (Henry and Lyle 2003). Recently, a series of surveys focusing on Murray cod in the Murray, Goulburn, and Ovens rivers has provided annual estimates of recreational fishing effort, catch and harvest (Brown 2010). Nevertheless, few studies have utilized fisher survey data for riverine Murray cod and golden perch populations; this represents a key knowledge gap in the management of these important native species. Accordingly, the objectives were to (1) provide estimates of recreational harvest, catch, harvest rate, catch rate and effort for Murray cod and golden perch; and (2) investigate the implications of fishing activity that occurs during the closed season for Murray cod within a river reach that is popular for recreational fishing and has previously experienced declines in native fish stocks.

3.3 Methods

3.3.1 Study site

The Murrumbidgee River is one of 23 sub-catchments comprising the MDB and extends for 1,690 km from the Great Dividing Range near Canberra to its confluence with the Murray River near Boundary Bend. Discharge in the Murrumbidgee River is regulated by two large impoundments and seven weirs (Baumgartner and Cameron 2012). The study area was selected because (1) it is popular for recreational fishing; (2) it is close to the regional centers of Wagga Wagga (population 60,000), Narrandera (population 4,000) and Leeton (population 7,000); and (3) biological survey data have determined declines in native fish stocks there (Gilligan 2012). The survey reach was declared an endangered ecological community in 2001 under the New South Wales Fisheries Management Act of 1994, and extends for 76 river kilometres between Berembed Weir (-34.8800 S 146.8370 E) and Yanco Weir (-34.7035 S 146.4167 E). The fisheries within these bounds were governed by size and bag limits for Murray cod and golden perch; gear restrictions; and a 3-month period (September–November) during which Murray cod harvest is prohibited (New South Wales Department of Primary Industries 2014).

3.3.2 Survey design and sampling protocols

Stratified random sampling methods were used, with day (calendar date) being the primary sampling unit for all strata. The survey year extended from May 1, 2012 to April 30, 2013, and was stratified into three periods: (1) a 4-month period
(May–August) prior to the implementation of the annual fishing closure for Murray cod (hereafter, ‘preclosure period’); (2) a 3-month period (September–November) when the fishery for Murray cod was closed (hereafter, ‘closed season’); and (3) a 5-month period (December–April) after the re-opening of the Murray cod fishery (hereafter, ‘postclosure’ period). The Murray cod closed season is not a complete fishing prohibition, as other species can be harvested during this period. However, fishing methods targeting Murray cod and the harvest of Murray cod are prohibited during this period (New South Wales Department of Primary Industries 2014). Day-type stratification (weekend and weekday strata) within each of the three survey periods was also used. Public holidays were included as part of the weekend day stratum.

Three separate survey components were used: (1) progressive counts (a roving survey method) were used to quantify shore- and boat-based fishing effort originating from all public and private access points within the study reach; (2) a roving survey was used to obtain catch rate and harvest rate information from shore-based fishers; and (3) an access point survey (i.e., two bus routes within the river reach) was used to obtain catch rate and harvest rate information from boat-based fishers. A complete census of all fishing effort was beyond scope, and therefore two weekdays and two weekend days were surveyed per month for each survey component. Each survey day covered the period sunrise to sunset.

The scheduling of progressive counts was done independent of sample day selection for the roving and access-point surveys. Progressive count and roving survey start locations and travel direction through the study reach were selected randomly, as recommended by Hoenig et al. (1993). Progressive counts and roving surveys were performed from a boat. The time required to travel through the fishery by boat (and between boat ramps by vehicle) was determined during a pilot study. Checkpoints were used to segment the study reach into seven sub-areas. Each sub-area was allocated 45 min for the roving boat clerk to travel and count or survey fishers. To ensure even temporal coverage of progressive counts or roving surveys throughout the day, clerks stopped the counts or surveys at the end of each sub-area for a designated time (which varied according to day length), before recommencing counts or surveys in the next sub-area. Boats that were traveling along the river and fishers that were moving along the shore were specifically
excluded from effort counts when it was not possible to determine their destination or their immediate intent to engage in any recreational fishing activity.

A bus route design (Jones and Pollock 2013; Pollock et al. 1994) was used to obtain unbiased estimates of the boat-based catch rates and harvest rates for the fishery. Two bus route surveys were used to cover the whole study reach. The first bus route encompassed four boat ramps, and the second bus route included two boat ramps. The wait time at each boat ramp (within a bus route) was assigned using estimates of expected usage that were based on the physical features of the site and prior fishery knowledge (Forbes and Asmus 2006; Forbes and Asmus 2007). Within bus route one each access point was assessed as high or low usage. The high and low usage access points were allocated 40% and 10% of the daily wait time respectively. For bus route two, the high usage access point received 70% of the daily wait time, whereas the low usage access point was allocated the remaining 30% for each survey day. The six major access points used in this study do not represent the entire boat-based fishery, as fishers also used private access points and public access with no formed boat ramp. It was not cost effective to assess all public and private access points in the study reach, and thus it was assumed that catch rates and harvest rates were similar between the major access points used in the bus routes and the other, infrequently used, access points. Starting site and travel direction between boat ramps for each respective bus route were selected at random. A boat ramp on the second bus route was damaged by floodwaters and closed from May 1, 2012 to January 11, 2013, during which time no sampling was performed at this access point. No adjustment for waiting time was made to the other access point on the second bus route during that period.

All fishing parties that were encountered during the roving and bus route surveys were asked to provide information about their fishing trip and catch. Fisher catch data was defined as the total of fish that were retained for harvest or discarded. Harvested fish were identified and measured (FL, mm) by creel clerks. Any refusal to provide information or to show harvested fish was recorded.
3.3.3 Estimation procedures

3.3.3.1 Fishing effort
The equations used are based on statistical methods and equations from Pollock et al. (1994), and are reproduced in Appendix 1. Fishing effort (angler-hours) was estimated separately for the boat-based fishery and the shore-based fishery (Pollock et al. 1994; Steffe and Chapman 2003). Daily progressive counts were multiplied by the length of the survey day to estimate the fishing effort for each survey day sampled. Fishing effort for each day-type stratum within each survey period (i.e., preclosure period, closed season, and postclosure period) was estimated by multiplying the number of possible sample days in each stratum by the mean of the daily effort estimates. Fishing effort estimates for each survey period were obtained by summing the effort estimates for the day-type strata. Annual estimates were calculated by summing the estimates for the survey periods. Variances were calculated by dividing the sample variance by the sample size and were additive when combining strata. Standard errors were calculated as the square root of the variance.

3.3.3.2 Catch and harvest rates
The mean-of-ratios estimator was used to estimate catch and harvest from interviews based on incomplete trips for the shore-based fishery (Hoenig et al. 1997; Jones et al. 1995; Pollock et al. 1997). The mean-of-ratios has a large variance when calculations include high harvest rates resulting from very short, incomplete fishing trips (Hoenig et al. 1997). The truncation of short incomplete trips is recommended to reduce the variance without inducing an appreciable bias (Hoenig et al. 1997). I examined the relationship between catch and harvest rates and the fishing trip duration for the shore-based interviews and found that the appropriate level of truncation was 20 party-minutes. Adoption of this truncation criterion resulted in the loss of 17 (11.0%) usable shore-based interviews from catch and harvest calculations.

The ratio-of-means estimator was used for estimating catch and harvest from interviews based on complete trips for the boat-based fishery (Jones et al. 1995; Pollock et al. 1997). The two bus routes were given equal weighting in the calculations of daily catch rates and harvest rates.
Catch and harvest rates and their variances for each survey period were calculated using weighted mean rates for each day-type stratum (this was done to compensate for size differences among the day-type strata). Similarly, weighted mean catch rates and harvest rates and the associated variances were calculated for the annual survey period by using weighted means for each of the three survey periods (Pollock et al. 1994).

3.3.3.3 Catch and harvest

Catch and harvest estimation for both boat- and shore-based fisheries were performed by multiplying fishing effort by an appropriate mean daily catch rate or harvest rate for each base-level stratum (Pollock et al. 1994; Steffe and Chapman 2003). Catch and harvest totals for each of the three survey periods were obtained by summing the appropriate estimates across day-type strata. Annual catch and harvest were estimated by summing the estimates from the survey periods. Variances were additive when combining strata, and standard errors were calculated as the square root of the variance.

3.3.3.4 Statistical comparison

Differences between boat- and shore-based fisheries and differences among survey periods were tested to determine whether the observed variability was statistically significant. I used the standard method described by Schenker and Gentleman (2001),

$$
\text{INT} = Q_1 - Q_2 \pm 1.96 \sqrt{SE_1^2 + SE_2^2}
$$

where, INT is the calculated interval; $Q_1$ and $Q_2$ are survey parameter estimates; and $SE_1$ and $SE_2$ are the standard errors of $Q_1$ and $Q_2$. Differences were considered significant (p < 0.05) when the INT did not contain zero.

3.4 Results

In total, 440 effort observations encompassing 941 fishers were made during the survey period. Roving surveys led to successful interviews of 155 shore-based fishing parties, comprising 318 fishers. Bus route surveyors successfully interviewed 81 boat-based fishing parties and 194 fishers: 27 boat-based fishing parties and 67 fishers from bus route 1; and 54 boat-based fishing parties and 127 fishers from bus route 2. One shore-based and one boat-based fishing party refused to be interviewed.
3.4.1 Fishing effort

We estimated that 77,267 ± 9,333 angler-hours (mean ± SE) of daytime recreational effort were expended in the study reach, with similar effort between boat-based (52%) and shore-based (48%) fishers. Effort distribution was not uniform across periods: effort was 9,645 ± 1,580 angler-hours during the preclosure period; 19,017 ± 2,819 angler-hours during the closed season for Murray cod; and 48,605 ± 8,756 angler-hours during the postclosure period (Table 3).

Table 3: Effort estimates for the Murrumbidgee River 2012–2013 (angler-hours; with standard errors) for boat- and shore-based fisheries between Berembed Weir and Yanco Weir.

<table>
<thead>
<tr>
<th>Day type</th>
<th>Boat-based Mean</th>
<th>SE</th>
<th>Shore-based Mean</th>
<th>SE</th>
<th>Total Mean</th>
<th>SE</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Effort, preclosure period</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>2,036</td>
<td>786</td>
<td>1,749</td>
<td>643</td>
<td>3,785</td>
<td>1,015</td>
</tr>
<tr>
<td>Weekend</td>
<td>3,809</td>
<td>903</td>
<td>2,051</td>
<td>806</td>
<td>5,860</td>
<td>1,211</td>
</tr>
<tr>
<td>Total</td>
<td>5,845</td>
<td>1,197</td>
<td>3,800</td>
<td>1,031</td>
<td>9,645</td>
<td>1,580</td>
</tr>
<tr>
<td><strong>Effort, closed season</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>1,791</td>
<td>526</td>
<td>8,264</td>
<td>2,249</td>
<td>10,055</td>
<td>2,310</td>
</tr>
<tr>
<td>Weekend</td>
<td>4,781</td>
<td>1,014</td>
<td>4,181</td>
<td>1,257</td>
<td>8,962</td>
<td>1,615</td>
</tr>
<tr>
<td>Total</td>
<td>6,572</td>
<td>1,142</td>
<td>12,445</td>
<td>2,577</td>
<td>19,017</td>
<td>2,819</td>
</tr>
<tr>
<td><strong>Effort, postclosure period</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>8,583</td>
<td>6,036</td>
<td>7,685</td>
<td>3,822</td>
<td>16,268</td>
<td>7,144</td>
</tr>
<tr>
<td>Weekend</td>
<td>19,093</td>
<td>2,765</td>
<td>13,244</td>
<td>4,240</td>
<td>32,337</td>
<td>5,062</td>
</tr>
<tr>
<td>Total</td>
<td>27,676</td>
<td>6,639</td>
<td>20,929</td>
<td>5,708</td>
<td>48,605</td>
<td>8,756</td>
</tr>
<tr>
<td><strong>Effort, annual</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>12,409</td>
<td>6,110</td>
<td>17,699</td>
<td>4,481</td>
<td>30,108</td>
<td>7,577</td>
</tr>
<tr>
<td>Weekend</td>
<td>27,683</td>
<td>3,080</td>
<td>19,476</td>
<td>4,495</td>
<td>47,159</td>
<td>5,449</td>
</tr>
<tr>
<td>Total</td>
<td>40,092</td>
<td>6,843</td>
<td>37,175</td>
<td>6,347</td>
<td>77,267</td>
<td>9,333</td>
</tr>
</tbody>
</table>

3.4.2 Catch rate

The annual Murray cod catch rate was significantly greater (standard method: INT = 0.033–0.239, p < 0.05) for boat-based fishers (mean ± SE = 0.228 ± 0.047 fish/angler-hour), than for shore-based fishers (0.092 ± 0.023 fish/angler-hour). There was no statistical difference between boat-based (0.068 ± 0.029 fish/angler-hour) and shore-based (0.071 ± 0.026 fish/angler-hour: standard method: INT = -0.079–0.073, p > 0.05).
Boat-based (0.490 ± 0.112 fish/angler-hour) and shore-based (0.147 ± 0.051 fish/angler-hour) catch rates for Murray cod during the postclosure period catch rates were greater than catch rates estimated for the closed season (Table 4). However, significant differences in Murray cod catch rates were only found; (1) between the boat- and shore-based fisheries (INT = 0.194–0.644], p < 0.05); and (2) between the closed season and postclosure period for the boat-based fishery (INT = 0.195–0.649, p < 0.05). The annual shore-based catch rate of golden perch of (0.018 ± 0.009 fish/angler-hour) was not significantly different from the boat-based catch rate (0.002 ± 0.001 fish/angler-hour; INT = -0.002–0.034, p > 0.05; Table 4).
Table 4: Catch-rate and harvest-rate estimates in the Murrumbidgee River 2012-13 (fish/angler-hour; with standard errors) for boat- and shore-based fishers between Berembed Weir and Yanco Weir.

<table>
<thead>
<tr>
<th>Day-type</th>
<th>Murray cod</th>
<th>Golden perch</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Boat</td>
<td>Shore</td>
</tr>
<tr>
<td></td>
<td>Mean  SE</td>
<td>Mean  SE</td>
</tr>
<tr>
<td>Catch rate, preclosure period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.017 0.014</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.049 0.018</td>
<td>0.134 0.049</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.026 0.011</td>
<td>0.038 0.014</td>
</tr>
<tr>
<td>Catch rate, closed season</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.030 0.035</td>
<td>0.051 0.033</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.157 0.056</td>
<td>0.119 0.039</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.068 0.029</td>
<td>0.071 0.026</td>
</tr>
<tr>
<td>Catch rate, postclosure period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.225 0.117</td>
<td>0.106 0.054</td>
</tr>
<tr>
<td>Weekend</td>
<td>1.010 0.238</td>
<td>0.229 0.109</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.490 0.112</td>
<td>0.147 0.051</td>
</tr>
<tr>
<td>Catch rate, annual</td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>0.228 0.047</td>
<td>0.092 0.023</td>
</tr>
<tr>
<td>Harvest rate, preclosure period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.004 0.003</td>
<td>0.011 0.008</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.001 0.001</td>
<td>0.003 0.002</td>
</tr>
<tr>
<td>Harvest rate, closed season</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Harvest rate, postclosure period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.024 0.019</td>
<td>0.001 0.001</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.043 0.025</td>
<td>0.012 0.006</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.031 0.015</td>
<td>0.005 0.002</td>
</tr>
<tr>
<td>Harvest rate, annual</td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>0.013 0.006</td>
<td>0.003 0.001</td>
</tr>
</tbody>
</table>
3.4.3 Harvest rate

For Murray cod, the annual boat-based harvest rate of 0.013 ± 0.006 fish/angler-hour (mean ± SE) was not significantly different from the shore-based harvest rate of 0.003 (± 0.001 SE) fish/angler-hour (INT = -0.002–0.022, p > 0.05). The harvest rate for Murray cod during the closed season was zero, as all captured fish were released (Table 4). For golden perch, the annual harvest rate was significantly greater for shore-based fishers (0.006 ± 0.002 fish/angler-hour) than for boat-based fishers (0.001 ± <0.001 fish/angler-hour; INT = 0.001–0.009, p < 0.05; Table 4).

3.4.4 Catch estimates

Estimated annual Murray cod catch was 27,276 ± 5,812 individuals (mean ± SE), and significantly greater catch was estimated for boat-based fishers (22,231 ± 5,525 fish), than for shore-based fishers (5,045 ± 1,814; INT = 5,788–28,584, p < 0.05). Murray cod closed season catch was 1,727 (± 473 SE) representing 6% of the total catch (Table 5). Annual Golden perch catch was 1,032 (± 420 SE) individuals. Golden perch shore-based catch (828 ± 405 SE), was not significantly different to that from the boat-based fishery (204 ± 108 SE; standard method: INT [-198, 1,446], p > 0.05; Table 5).
Table 5: Catch and harvest estimates in the Murrumbidgee River 2012-13 (number of fish; with standard errors) for boat- and shore-based fishers between Berembed Weir and Yanco Weir.

<table>
<thead>
<tr>
<th>Day type</th>
<th>Murray cod</th>
<th></th>
<th>Golden perch</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Boat</td>
<td>Shore</td>
<td>Total</td>
<td>Boat</td>
</tr>
<tr>
<td></td>
<td>Mean</td>
<td>SE</td>
<td>Mean</td>
<td>SE</td>
</tr>
<tr>
<td>Catch, preclosure period</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>35</td>
<td>29</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Weekend</td>
<td>186</td>
<td>81</td>
<td>276</td>
<td>143</td>
</tr>
<tr>
<td>Total</td>
<td>221</td>
<td>86</td>
<td>276</td>
<td>143</td>
</tr>
<tr>
<td>Catch, closed season</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>54</td>
<td>61</td>
<td>423</td>
<td>283</td>
</tr>
<tr>
<td>Weekend</td>
<td>751</td>
<td>306</td>
<td>499</td>
<td>215</td>
</tr>
<tr>
<td>Total</td>
<td>805</td>
<td>312</td>
<td>922</td>
<td>355</td>
</tr>
<tr>
<td>Catch, postclosure period</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>1,927</td>
<td>1,530</td>
<td>812</td>
<td>542</td>
</tr>
<tr>
<td>Weekend</td>
<td>19,278</td>
<td>5,299</td>
<td>3,035</td>
<td>1,677</td>
</tr>
<tr>
<td>Total</td>
<td>21,205</td>
<td>5,516</td>
<td>3,847</td>
<td>1,763</td>
</tr>
<tr>
<td>Harvest, preclosure period</td>
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<tr>
<td>Weekday</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Weekend</td>
<td>15</td>
<td>10</td>
<td>23</td>
<td>17</td>
</tr>
<tr>
<td>Total</td>
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<td>95</td>
</tr>
</tbody>
</table>

Harvest, preclosure period

Harvest, closed season

Harvest, postclosure period

Harvest, annual
### 3.4.5 Harvest estimates

Annual harvest of Murray cod was 1,230 ± 526 individuals (mean ± SE). For Murray cod, the boat-based harvest (1,045 ± 518 fish) was not significantly different from the shore-based harvest (185 ± 95; INT = -172–1,892, p > 0.05). No Murray cod were harvested during the closed season (Table 5). Annual harvest of golden perch was 401 ± 178 fish, with significantly greater harvest estimated for shore-based fishers (382 ± 177 fish) than for boat-based fishers (19 ± 17 fish; INT = 14–712, p < 0.05). Golden perch harvest from the shore-based fishery was highest during the closed season at 204 ± 126 fish; Table 5).

Eighteen Murray cod ranging from 350 to 740 mm FL were harvested; 78% of the harvest fish were above the current minimum legal length of 600 mm. Three of the undersize fish were harvested illegally by a single fishing party, and the fourth undersize fish was harvested by fishers who used an inaccurate measuring device. Sixteen golden perch ranging from 300 to 480 mm were harvested, and all were above the current minimum legal length of 300 mm. None of the fishers reached the existing daily bag limits for these species (2 fish/d for Murray cod; 5 fish/d for golden perch).

Fishing mortality rates for Murray cod are estimated at 2% under normal recreational fishing conditions (Douglas et al. 2010) and up to 15% during catch-and-release tournaments (Hall et al. 2012). Application of these catch-and-release associated mortality rates would add 521 to 3,907 fish to the estimates of Murray cod harvest, including mortality of 35 to 259 Murray cod during the closed season.

### 3.4.6 Released component

The annual percentage of Murray cod that were released was estimated at 95%. All Murray cod that were caught during the closed season were released. Murray cod release rates were 92% during the preclosure period and 95% during the postclosure period. The annual percentage of golden perch that were released was estimated at 61%, the release rates were 47% for the preclosure period, 60% for the closed season, and 62% for the postclosure period.
3.5 Discussion

An active Murray cod fishery and a poor golden perch fishery exist in this reach of the Murrumbidgee River. The Murray cod fishery had an annual boat-based catch rate of one fish every 4 h and an annual shore-based catch rate of one fish every 11 h. The Murray cod catch rates in this reach are approximately double those recorded from similar fisheries in the Murray River (boat-based fishery: 0.12 fish/angler-hour; shore-based fishery: 0.04 fish/angler-hour; Brown 2010), and are approximately four times greater than catch rates from the Goulburn and Ovens rivers (boat-based fishery: 0.07 fish/angler-hour; shore-based fishery 0.02 fish/angler-hour; Brown 2010). In contrast, the golden perch fishery in the study reach had an annual shore-based catch rate of one fish every 56 h and a boat-based catch rate of one fish every 500 h. Golden perch catch rates were an order of magnitude lower (shore-based) and two orders of magnitude (boat-based) than catch rates recorded from Lake Keepit, New South Wales (0.43 fish/angler-hour; Battaglene 1985). Golden perch are reportedly present in high abundance throughout the MDB (Ye 2004), but they did not contribute strongly to angler catches in the study reach. In this regard, consideration must be given to the interactions of fish and anglers as the behaviour of both groups are dependent on each other (Arlinghaus et al. 2013). For example, carp in Germany learn to avoid capture by recreational fishers (Klefoth et al. 2013). Should similar behaviour be exhibited by golden perch and Murray cod, the estimates of catch rate would lower as fishing effort increased. Murray cod and golden perch catch rates in the Murrumbidgee River are generally low compared to international fisheries such as the Tongariro River in New Zealand where 0.12 to 0.63 rainbow trout/angler-hour are captured (Dedual and Maheswaran 2016). That Australian anglers target Murray cod and golden perch when fish are infrequently caught by comparison, is testament to the high regard in which these fish are held, both for their sporting qualities and edibility, and suggests that high catch rates are not the primary motivation for cod and perch fishers.

The increased catch rates, and similar harvest rates of Murray cod in the Murrumbidgee compared with the Murray River, Goulburn and Ovens rivers (Brown 2010) suggests a more productive Murrumbidgee system. Survey observations from the study indicated that Murray cod releases were predominantly due to the fish being undersized, thus supporting the idea that the higher catch rates and similar harvest rates may be related to
greater numbers of fish below the minimum legal length in the Murrumbidgee River. The undersized fish may be the result of increased spawning and recruitment after cessation of a decade-long drought (Morrongiello et al. 2011), or the augmentation of fish numbers from stocking, as observed in North American salmonids (Moring 1993). Research on the source of Murray cod recruitment in the Murrumbidgee River is required to determine whether local populations are self-sustaining or augmented.

The closed season is intended to protect breeding Murray cod and was introduced as a result of biological data demonstrating that (1) the handling of Murray cod prior to spawning results in egg re-absorption (Rowland 1988), and (2) Murray cod are more susceptible to capture during the spawning period (Rowland 2005). I found that closed-season catch rates of Murray cod were moderate, which does not support increased capture susceptibility during the closure. However, there was still a considerable amount of fishing effort during the closed season, which could negatively impact Murray cod populations. Closed-season effort was less than half that estimated for the postclosure period. However, the closed season attracted more than double the effort associated with the preclosure period. Some closed-season effort could be attributed to fishers preferentially targeting golden perch and to an increase in fishing activity because of more favourable (warmer) climatic conditions. However, given the low catch rates of golden perch, the increased effort during the closed season was more likely related to fishers’ desire to go fishing and practice catch and release of Murray cod, and they were either unaware of or ignored the negative impacts of this fishing method. To reduce Murray cod bycatch during the closed season, some fishers may be changing to baits and lures better that are better suited to golden perch, but a corresponding effort reduction was not evident in this study. Rowland (1983b) suggested that male Murray cod guard their nests from spawning until hatch. Furthermore, when disturbed, males of the closely related Eastern cod, *Maccullochella ikei*, abandoned their nests prior to hatch; in some cases, the male did not return to the nest and the eggs were subsequently consumed by predators (Butler and Rowland 2009). Impaired reproductive success has also been shown in black bass, *Micropterus* spp, in North America where nest abandonment occurred following catch and subsequent release (Siepker et al. 2007). If similar behaviour is exhibited by Murray cod, deliberate bycatch (even with 100% catch-and-release), during the closed season could cause reductions in breeding and recruitment success. Given that levels of fishing effort in the survey reach were similar
between open and closed seasons, the catch of Murray cod is inevitable and could influence breeding success (Douglas et al. 2010; Hall et al. 2012) through nest abandonment. Accordingly, the effectiveness of the closed season for protecting Murray cod spawners was reduced; therefore, research to quantify the impacts of catch-and-release fishing on spawning Murray cod is of priority. In addition, targeted programs that educate fishers on the potential impacts of catch-and-release fishing on fish populations may positively influence spawning outcomes for affected species.

Fishing mortality associated with catch-and-release fishing requires consideration for improved management of species with high release rates (Douglas et al. 2010). The percentage of Murray cod released in this study (95%) was greater than the 77.6% release rate identified during a nationwide survey (Henry and Lyle 2003) and was also greater than the 90.0% release rate reported for the Murray, Goulburn and Ovens rivers (Brown 2010). Application of Murray cod fishing mortality rates demonstrated that catch-and-release fishing could generate additional mortality in excess of the harvest estimates. I identified no Murray cod harvest during the closed season, but based on previous work the postrelease mortality during this period could be up to 21% of the harvest estimate (Douglas et al. 2010; Hall et al. 2012). Postrelease mortality associated with recreational fishing is a global concern, with analogies drawn to bycatch discards in commercial fisheries (Cooke and Cowx 2004). Accordingly, it is important for fishery managers to consider this potential additional source of fishing-related mortality, particularly for fisheries with high release rates of target species, such as that observed for Murray cod in the current study. Future research could extend the current knowledge of fishing mortality rates by quantifying the effects of environmental factors, multiple instances of catch and release, and fish handling behaviours.

The golden perch is a migratory species (Reynolds 1983). These fish aggregate below in-stream barriers (Baumgartner 2007), where they are particularly targeted by shore-based fishers (e.g. Forbes and Asmus 2006; Forbes and Asmus 2007). The low estimates of golden perch catch in the present study could be due to a range of factors, such as poor recruitment, overfishing, or impediments to movement (e.g., the Berembed and Yanco weirs; Baumgartner 2007; Hall et al. 2012; Ye 2004). However, the low estimated catch of golden perch could also have resulted from anglers’ use of methods and equipment that were selective for Murray cod. Fishery-independent electrofishing
data shows that golden perch comprised 18% and Murray cod constituted 59% of the total number of large-bodied native fish species in the survey reach (J. P. Forbes, unpublished data), which indicates that golden perch are more abundant than catch estimates suggest. Despite this, golden perch are generally found in greater abundance than Murray cod throughout most catchments of the MDB (Gilligan 2005), thus supporting the view that the study area had a poor golden perch fishery. As such, consideration should be given to recovery initiatives such as re-stocking and provision of fish passage, in order to increase the golden perch population.

Based on the present results, I identified that the effectiveness of the closed season in protecting Murray cod spawners was reduced due to continued fishing effort for and bycatch of this species during this closure. In addition to providing effort, catch and harvest metrics, I have offered recommendations for sustaining and improving the Murray cod and golden perch fisheries. Specifically, I suggest (1) research to quantify the impacts of catch-and-release fishing on spawning Murray cod, (2) consideration of additional fishing-related mortality caused by catch-and-release fishing, (3) provision of fish passage in the study area, (4) re-stocking of golden perch, and (5) targeted education programs on fishing gear and techniques that minimise Murray cod bycatch during the closed season.

Quantification of Murray cod and golden perch fisheries are important to inform management strategies, such as harvest restrictions and fishing closures, as limited information is available to inform such interventions. Therefore, in Chapter 4, I will quantify the recreational fishery in Lake Mulwala, and use this information to assess variability that may exist among fisheries for Murray cod.
Chapter 4

Quantifying the recreational fishery for large percichthyid fish in an impoundment on the Murray River, Australia

4.1 Abstract

Quantifying recreational fisheries is critical to manage species under threat, and to understand the relative success or failure of recovery strategies. Angler surveys are commonly used to assess recreational fisheries around the world. However, few recreational fisheries for Murray cod, *Maccullochella peelii*, an important Australian inland target species, have been quantified. Lake Mulwala is an impoundment on the Murray River within Australia’s Murray-Darling Basin that contains a large Murray cod recreational fishery where no angler survey data currently exists. The objectives of the current study were to; (1) quantify boat- and shore-based fishing effort, harvest and discard within Lake Mulwala; (2) collect information from fishers regarding targeting preferences, fishing method and release habits; and (3) use these data to assess variability among different Murray cod fisheries. A complemented fisher survey design was used to assess the austral summer recreational fishery in Lake Mulwala. Boat- and shore-based daytime fishing effort were quantified using progressive counts. Catch- and harvest-rate information was collected from shore-based fishers using roving surveys, and from boat-based fishers using access point surveys. Boat-based fishers almost exclusively targeted Murray cod and harvested more and larger fish than shore-based anglers. The shore-based fishery was a more diverse, multi-species fishery. Most releases of Murray cod in Lake Mulwala were mandatory because fish were undersize, and few large cod were evident in the fishery, justifying the introduction of the 550-750 mm harvest slot. Standardised values of effort-, discard- and harvest density, relative stock density and harvest specificity illustrated variability among Murray cod fisheries in Lake Mulwala, the Murrumbidgee River and the Murray River; however boat-based fisheries in these waterbodies generally exhibited more desirable aspects of fishery quality (e.g. larger fish, higher harvest and discard) than shore-based fisheries.
4.2 Introduction

Global fishing trends suggest that recreational catches in freshwater fisheries are increasing (Cooke and Cowx 2006). The impact of recreational fishing on fish populations can be difficult to quantify as monitoring is often absent or insufficient (Post et al. 2002). An understanding of fishing effort, harvest, and catch-and-release (discard) in a recreational fishery is vital for managing potential impacts (Cooke and Cowx 2004; Pollock et al. 1994). Such information is collected from recreational fisheries around the world in response to fishery decline (Kozfkay and Dillon 2010), and to identify how recreational fishers use the resource (Beckley et al. 2008), which is subsequently used to inform management decisions (Kozfkay and Dillon 2010; Ward et al. 2013).

Assessment of recreational fisheries is critical for managing species under threat (Hunt et al. 2011). Management tools such as harvest restrictions and stocking can be implemented without assessments being completed (Forbes et al. 2015b). However, it is important that sound science inform the implementation and success of such programs so that management goals and objectives can be achieved (Arlinghaus et al. 2016; Cooke et al. 2015). Fishery dependant surveys are commonly used to inform the relative success or failure of management strategies (e.g. Hunt et al. 2011; Kozfkay and Dillon 2010). However, such surveys are seldom conducted for Australian freshwater recreational fisheries.

In Australia’s Murray-Darling Basin (MDB), the Murray cod, *Maccullochella peelii*, is an iconic recreational target species (Brown 2010; Forbes et al. 2015c; Hunt et al. 2010), because of its potential large size (up to 113 kg and 1.8 m TL) and edibility (Rowland 2005). Within the MDB, a 550-750 mm harvest slot for Murray cod replaced a 600 minimum length limit (MLL) for this species in the states of New South Wales and Victoria in 2014 (New South Wales Department of Primary Industries 2015; Victorian Department of Environment and Primary Industries 2015). The change was enacted in an effort to provide protection for large and small fish and to broaden the size structure (Gwinn et al. 2015). The recreational fishery for Murray cod is poorly understood, as little fishery-dependant data exists, and time-series data are generally not available. Murray cod populations have exhibited historic decline, caused in part by
recreational and commercial exploitation, but also from catchment development (Reid et al. 1997b; Rowland 1989). Despite such decline, recovery at the state jurisdictional level is apparent (Rowland 2013; West et al. 2016).

Within each state, Murray cod fisheries are often centred on tourist locations or key access points (Forbes and Asmus 2006). Many recreational fisheries exist for this species, with fishing tournaments held in waterbodies containing Murray cod attracting thousands of anglers (Hall et al. 2012). Lake Mulwala is an impoundment on the Murray River (the state border between New South Wales and Victoria) that is perceived to contain one of Australia’s premier recreational fisheries for Murray cod and hosts regular fishing tournaments (Howitt et al. 2004; Kearney and Kildea 2001; Koehn and Clunie 2010). However, regardless of its popularity as a destination to fish for Murray cod, the Lake Mulwala recreational fishery has not previously been quantified. The collection of such information is crucial to obtain base-level data for assessment of this fishery and to understand variability among Murray cod fisheries that may influence the effectiveness of existing harvest regulations.

The current study used a probability-based survey to quantify the Lake Mulwala fishery. Specifically, the aims of this study were to; (1) quantify the levels of daytime shore-based and boat-based fishing effort, harvest and discard in the recreational fishery within Lake Mulwala; (2) collect information from angling parties regarding their targeting practices, fishing method and reasons for releasing/discardng fish; and (3) use these data to assess variability that may exist among fisheries for Murray cod in Lake Mulwala and the Murrumbidgee and Murray rivers (Brown 2010; Forbes et al. 2015c).

4.3 Methods

4.3.1 Study site

Lake Mulwala (36.0039 S, 146.0676 E) is an impoundment on the Murray River formed in 1939 by the construction of Yarrawonga Weir. The lake forms part of the state border between New South Wales and Victoria, and is primarily used to provide irrigation water to these states (Howitt et al. 2004). Lake Mulwala covers an area of 3,561 ha, has a capacity of 118,000 ML, and is a popular site for water-skiing, swimming, boating and fishing (Murray-Darling Basin Commission 2004). Habitat within the lake includes
large amounts of dead standing and fallen timber, and remnant (inundated) river channels (Koehn 2009).

4.3.2 Survey design and sampling protocols

Stratified random sampling methods were used with day (calendar date) being the primary sampling unit (PSU) for all strata. Each survey day covered the period sunrise to sunset. Night fishing (sunset to sunrise) was considered but omitted from the sampling design because of limited resources. The temporal survey frame was the three month austral summer season; 1 December 2014 to 28 February 2015. The season was stratified into two periods: (1) the month of December during which the Murray cod fishery is opened after a three month fishing closure (termed summer-early period); and (2) a two month period (January and February; termed summer-late period) to accommodate the expected high fishing effort and catch resulting from the reopening of the Murray cod fishery in December. Day-type stratification (weekend and weekday strata) within each survey period was also used. Public holidays were included as part of the weekend day stratum. Thus, the base-level stratum was day-type (weekday, weekend), within period (summer-early, summer-late), within a season (summer). Four weekdays and four weekend days were sampled in summer-early, and three weekdays and three weekend days were sampled in summer-late. Sampling days were randomly selected within each survey period.

Progressive counts from a boat were used to quantify shore- and boat-based fishing effort originating from all public and private access points within the fishery. Progressive count start locations and travel direction through the fishery were randomly selected (Hoenig et al. 1993). A pilot study was used to determine the time required to travel through the fishery with three hours allocated to complete a progressive count. Each survey day was divided into four non-overlapping intervals and a progressive count was randomly allocated without replacement (within each base-level stratum) to one of the four intervals. Traveling boats, and fishers moving along the shore, were specifically excluded from effort counts when it was not possible to determine their destination or their immediate intent to engage in any recreational fishing activity.

A roving survey was used to obtain catch-and-release rate and harvest rate information from shore-based fishers. The roving surveys were done on the same days as the
progressive counts for fishing effort, but did not cover the interval of the progressive count. Roving survey travel direction for each survey day was randomly selected and the start location was the termination point of the progressive effort count. The roving surveys covered at least one complete circuit of the fishery during each survey day.

An access point survey was used to obtain catch-and-release rate and harvest rate information from boat-based fishers. Eleven access points (public boat ramps) were identified. Information about the physical features of the access points, prior knowledge of the fishery, and the expert opinions of maritime and fishery compliance officers, local fishing guides and tackle store owners were used to categorise each access point as either high or low usage. Unequal probability sampling was then used to allocate two access points for coverage on each survey day; with the exception of 6 December 2014, when three ramps were sampled to improve coverage of a large boat-based fishing tournament. The daily selection probability for each of the three high usage access points was 0.3 and the eight low usage access points were each assigned a daily selection probability of 0.0125. It was not cost effective to cover private access points during the survey. It was assumed that the behaviour of fishers using private and public access points were similar.

All fishing parties interviewed during the roving and access point surveys were asked to provide information about their fishing trip and catch. These data included; (a) trip duration; (b) primary target species of the fishing party; (c) the number and species that were caught-and-released; (d) the reason why those fish were released; and (e) fishing method. Harvested fish were identified and measured (FL, mm) by creel clerks. Any refusal to provide information or to show harvested fish was recorded.

4.3.3 Estimation procedures

The general form of the equations used can be found in Appendix 2, which is based on statistical methods and equations from Pollock et al. (1994). Additional information on estimation procedures can be found in Hoenig et al. (1993), Hoenig et al. (1997), and Pollock et al. (1997).
4.3.3.1 Fishing effort

Fishing effort (party hours) was estimated separately for boat- and shore-based fisheries. Daily progressive counts were multiplied by the length of the survey day to estimate the fishing effort for each day sampled. Fishing effort estimates for each base-level stratum were made by multiplying the number of possible sample days in that stratum with the mean daily effort estimate for that stratum. Fishing effort estimates for each survey period were obtained by summing the day-type stratum effort estimates. Seasonal estimates were calculated by summing the survey periods.

For comparative purposes with other studies, the estimates of fishing effort were converted from party hours to fisher hours. This was done separately for boat- and shore-based fisheries for each base-level stratum by multiplying the fishing effort (party hours) and the daily average of the mean number of fishers per fishing party. The equations used to calculate variances are provided in Appendix 2, Section A2.2. Variances were additive when combining strata. Standard errors (SE) were calculated as the square root of the variance.

4.3.3.2 Catch-and-release rates and harvest rates

The mean of ratios estimator was used for estimating shore-based catch-and-release rates and harvest rates as interviews were based on incomplete trips (Hoenig et al. 1997; Jones et al. 1995; Pollock et al. 1997). The mean of ratios have a large variance when high harvest rates resulting from very short, incomplete fishing trips are included in calculations (Hoenig et al. 1997). Plots of party-based catch-and-release rates, harvest rates, and the length of the incomplete trip were examined to identify an appropriate level of truncation for these shore-based interviews (Hoenig et al. 1997). Twenty party minutes was used as the truncation criterion, resulting in the removal of 6 (2.4%) shore-based interviews.

The ratio of means estimator was used for estimating catch-and-release rates and harvest rates for the boat-based fishery (Jones et al. 1995; Pollock et al. 1997). Each access point that was sampled on a survey day was given equal weighting when calculating daily catch-and-release rates and harvest rates.
Catch-and-release rates and harvest rates and their variances for each survey period were weighted to compensate for the different sizes in day-type strata. Similarly, weighted mean catch-and-release rates and harvest rates and their variances were calculated for the summer season by using weighted means that compensated for the different sizes of the two survey periods (Pollock et al. 1994). These weighting procedures were applied to data from both the shore- and boat-based fisheries.

4.3.3.3 Catch-and-release and harvest
Catch-and-release and harvest estimation for both boat- and shore-based fisheries were done by multiplying fishing effort (party hours) with an appropriate mean daily catch-and-release rate or harvest rate (fish/party-hour) for each base-level stratum (Pollock et al. 1994; Steffe and Chapman 2003). Catch-and-release and harvest totals for each survey period were obtained by summing the appropriate day-type stratum estimates together. Seasonal estimates of catch-and-release and harvest were made by summing the survey periods. Variances and SEs were calculated as described for fishing effort.

4.3.3.4 Targeting behaviour, fishing methods, and reasons for catch-and-release
Weighted frequency distributions were constructed to describe the nominated targeting preferences of anglers; reasons for their catch-and-release practices; and fishing method used. Weighted frequency distributions were initially done for each base-level stratum using data aggregated at the PSU level (i.e. day). Within each PSU the weighted response from each fishing party was given equal weighting. The fishing party response was derived by giving equal weighting to the responses of individual anglers within that party. Seasonal weighted frequency distributions were constructed by integrating the data from the base-level strata and weighting them to account for the different number of days in each base-level stratum. These weighted frequency distributions were created for each of the boat- and shore-based fisheries. The reasons why fishers practiced catch-and-release were categorised into whether released fish were undersized (below the MLL), oversized (above maximum length limit), legal voluntary (harvest eligible, but voluntarily released), or over bag limit (possession limits were exceeded). The fishing method used was categorised into bait, lures or a combination of bait and lures (where fishers used multiple methods).
4.3.3.5 Standardised parameters

In order to assess variability among fisheries for Murray cod, estimates of effort, harvest and discard were standardised per unit of surface area. Surface area (ha) of the survey area was calculated using ArcMap (Environmental Systems Resource Institute 2009). Boat-based fishers are able to effectively fish the entire surface area of Lake Mulwala, however shore-based fishers are restricted to the lake margins and the distance they can cast or wade from shore. Therefore, the surface area used to calculate effort, discard and harvest density for the shore-based fishery in Lake Mulwala was defined as the area extending between the shoreline and 50 m from the shore. The estimates of effort, discard and harvest were divided by the appropriate surface area (either boat- or shore-based) to obtain effort/ha, fish discarded/ha and fish harvested/ha.

The size structure of harvested Murray cod was standardised by calculating the relative stock density for each fishery (Neumann and Allen 2007). Relative stock density is the proportion of Murray cod that we deemed to provide a memorable fishing experience (i.e. > 700 mm) in the legally harvested population (i.e. > 550 mm – the lower bound of the existing harvest slot limit).

We propose a new standardised parameter of harvest specificity. Harvest specificity is the proportion of harvested Murray cod in the total harvest of all species combined and provides a relative measure of importance. Harvest specificity differs from targeting behaviour because it is independent of angler opinion, perceptions and attitudes.

Data from similar surveys of recreational fisheries conducted during 2012-2013 in the Murrumbidgee River between Berembed Weir and Yanco Weir (Forbes et al. 2015c); and from 2006-2008 in the Murray River downstream of Lake Mulwala to the South Australian border (Brown 2010); were re-analysed to obtain effort-, discard- and harvest-density, relative stock density, and harvest specificity in these systems equivalent to that calculated for Lake Mulwala. These standardised data allow comparisons among boat- and shore-based fisheries in each waterbody for the three month austral summer season.
4.4 Results

Roving surveys led to successful interviews of 253 shore-based fishing parties, comprising 493 fishers. Access point surveyors successfully interviewed 296 boat-based fishing parties and 627 fishers.

4.4.1 Boat-based fishery

Sixty five percent of summer fishing effort in Lake Mulwala was expended in the boat-based fishery (Table 6) and 98% of boat-based fishing parties were targeting Murray cod (Figure 4). Boat-based fishers predominantly used lures and a smaller proportion used bait (Figure 5). The summer boat-based catch-and-release rate of Murray cod in Lake Mulwala was 0.256 (± 0.064 SE) fish/fisher-hour and the harvest rate was 0.029 (± 0.011 SE) fish/fisher-hour (Table 7). Boat-based catch-and-release rates and harvest rates for species other than Murray cod were minimal (< 0.003 fish/fisher-hour; Table 7). Subsequently, boat-based catch-and-release and harvest estimates for species other than Murray cod were low (catch-and-release ≤ 200 fish; harvest ≤ 89 fish; Table 8). Boat-based catch-and-release for Murray cod in Lake Mulwala was 8,486 (± 2,769 SE) fish and harvest was 1,145 (± 454 SE) fish (Table 8).

Table 6: Effort estimates for Lake Mulwala 2014-15 (fisher-hr; with standard errors) for boat-based and shore-based fisheries.

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<td>mean SE</td>
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<td>51,775  24,046</td>
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<td>Total</td>
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<td>39,351  5,733</td>
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<tr>
<td><strong>Effort, summer total</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>17,259  4,954</td>
<td>17,316  3,924</td>
<td>34,575  6,320</td>
</tr>
<tr>
<td>Weekend</td>
<td>41,696 23,590</td>
<td>14,855  3,827</td>
<td>56,551  23,898</td>
</tr>
<tr>
<td>Total</td>
<td>58,955 24,104</td>
<td>32,171  5,481</td>
<td>91,126 24,719</td>
</tr>
</tbody>
</table>

Thirty three Murray cod ranging from 540-760 mm were measured during interviews with boat-based fishers in Lake Mulwala with 94% (31 fish) within the current harvest slot limit. One Murray cod was undersized (i.e., < 550 mm) and one was oversized (i.e.,
> 750 mm; Figure 6). Four carp were harvested from 500-700 mm, and one golden perch at 445 mm. The boat-based released component (i.e., the number of fish released as a percentage of fish harvested plus those caught-and-released) for Murray cod in Lake Mulwala was 88.1%. Most Murray cod releases in the boat-based fishery were mandatory as the fish were undersized (93.7%); however 6.2% of harvest eligible Murray cod were voluntarily released and 0.1% were released as they were oversized (Figure 7).

The boat-based fishery in Lake Mulwala had lower effort density (16.6 fisher-hours/ha), than the Murrumbidgee River (20.0 fisher-hours/ha) and the Murray River (26.4 fisher-hours/ha; Table 9). Discard density varied from 2.4 fish/ha in Lake Mulwala to 13.8 fish/ha in the Murrumbidgee River (Table 9). Harvest density varied from 0.2 fish/ha in the Murray River to 1.1 fish/ha in the Murrumbidgee River (Table 9). Boat-based harvest specificity in Lake Mulwala (92%) and the Murrumbidgee River (71%), were higher than that for the Murray River (16%; Table 9). The relative stock density for boat-based fisheries varied from 27 in the Murrumbidgee River, to 6 in Lake Mulwala (Table 9).

![Figure 4: Species-specific target preferences of fishers in Lake Mulwala 2014-15 for boat and shore-based fisheries. The total number of sampling days is represented by n.](image)
Figure 5: Weighted frequency distribution of fishing methods in Lake Mulwala used by boat- and shore-based fisheries during the 2014/15 survey period. The total number of sampling days is represented by n.
Table 7: Catch-and-release rates and harvest-rates in Lake Mulwala 2014-15 (fish/fisher-hr, with SE) for (a) boat-based, and (b) shore-based fisheries.

<table>
<thead>
<tr>
<th>Day-type</th>
<th>a) Boat-based fishery</th>
<th>b) Shore-based fishery</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Release rate</td>
<td>Harvest rate</td>
</tr>
<tr>
<td></td>
<td>mean SE</td>
<td>mean SE</td>
</tr>
<tr>
<td>Weekday</td>
<td>0.421 0.231</td>
<td>0.023 0.008</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.104 0.037</td>
<td>0.017 0.006</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.339 0.172</td>
<td>0.022 0.006</td>
</tr>
<tr>
<td><strong>Murray cod, summer-early period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.272 0.123</td>
<td>0.041 0.031</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.067 0.016</td>
<td>0.014 0.003</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.213 0.088</td>
<td>0.033 0.022</td>
</tr>
<tr>
<td><strong>Murray cod, summer total</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>0.256 0.064</td>
<td>0.029 0.011</td>
</tr>
<tr>
<td><strong>Carp, summer-early period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.001 0.001</td>
<td>0.001 0.001</td>
</tr>
<tr>
<td>Weekend</td>
<td>&lt;0.001 &lt;0.001</td>
<td>&lt;0.001 &lt;0.001</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.001 0.001</td>
<td>0.001 0.001</td>
</tr>
<tr>
<td><strong>Carp, summer-late period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.006 0.006</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0 0</td>
<td>0.006 0.003</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.005 0.005</td>
<td>0.002 0.001</td>
</tr>
<tr>
<td><strong>Carp, summer total</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>0.003 0.002</td>
<td>0.001 0.001</td>
</tr>
<tr>
<td><strong>Golden perch, summer-early period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0.003 0.002</td>
<td>0.001 0.001</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.004 0.002</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>0.003 0.002</td>
<td>0.001 0.001</td>
</tr>
<tr>
<td><strong>Golden perch, summer-late period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td><strong>Golden perch, summer total</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>0.001 &lt;0.001</td>
<td>&lt;0.001 &lt;0.001</td>
</tr>
<tr>
<td><strong>Trout cod, summer-early period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0.001 0.001</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>&lt;0.001 &lt;0.001</td>
<td>0 0</td>
</tr>
<tr>
<td><strong>Trout cod, summer-late period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td><strong>Trout cod, summer total</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>&lt;0.001 &lt;0.001</td>
<td>0 0</td>
</tr>
<tr>
<td><strong>Redfin, summer-early period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td><strong>Redfin, summer-late period</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>Weekend</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>weighted average</td>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td><strong>Redfin, summer total</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>weighted average</td>
<td>0 0</td>
<td>0 0</td>
</tr>
</tbody>
</table>
Table 8: Catch-and-release and harvest estimates in Lake Mulwala 2014-15 (number of fish per period; with standard errors) for Murray cod, carp golden, trout cod and redfin perch taken by boat-based and shore-based fishers.

<table>
<thead>
<tr>
<th>Day-type</th>
<th>a. Catch-and-release</th>
<th>b. Harvest</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Boat</td>
<td>Shore</td>
</tr>
<tr>
<td>Murray cod, summer-early period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>3,101</td>
<td>1,644</td>
</tr>
<tr>
<td>Weekend</td>
<td>3,029</td>
<td>2,137</td>
</tr>
<tr>
<td>Total</td>
<td>6,130</td>
<td>2,066</td>
</tr>
<tr>
<td>Murray cod, summer-late period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>1,423</td>
<td>1,705</td>
</tr>
<tr>
<td>Weekend</td>
<td>933</td>
<td>2,544</td>
</tr>
<tr>
<td>Total</td>
<td>2,356</td>
<td>4,149</td>
</tr>
<tr>
<td>Murray cod, summer total</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>4,524</td>
<td>1,742</td>
</tr>
<tr>
<td>Weekend</td>
<td>3,962</td>
<td>2,152</td>
</tr>
<tr>
<td>Total</td>
<td>8,486</td>
<td>3,935</td>
</tr>
<tr>
<td>Carp, summer period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>1,423</td>
<td>1,705</td>
</tr>
<tr>
<td>Weekend</td>
<td>933</td>
<td>2,544</td>
</tr>
<tr>
<td>Total</td>
<td>2,356</td>
<td>4,149</td>
</tr>
<tr>
<td>Golden perch, summer total</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>84</td>
<td>96</td>
</tr>
<tr>
<td>Weekend</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td>Total</td>
<td>99</td>
<td>111</td>
</tr>
<tr>
<td>Trout cod, summer period</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>29</td>
<td>20</td>
</tr>
<tr>
<td>Weekend</td>
<td>171</td>
<td>108</td>
</tr>
<tr>
<td>Total</td>
<td>200</td>
<td>110</td>
</tr>
<tr>
<td>Redfin, summer total</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weekday</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Weekend</td>
<td>171</td>
<td>108</td>
</tr>
<tr>
<td>Total</td>
<td>200</td>
<td>110</td>
</tr>
</tbody>
</table>

81
Figure 6: Length frequency distributions for Murray cod harvested in Lake Mulwala 2014-15 by boat- and shore-based fishers. The current 550-750 mm, TL harvest slot is boxed and pre-December 2014 MLL given as a dotted line.

Figure 7: Weighted frequency distribution of reasons for Murray cod catch-and-release in Lake Mulwala for boat- and shore-based fisheries during the 2014/15 survey period. The total number of sampling days is represented by n.
Table 9: Quantitative characteristics used to assess variability among Murray cod fisheries (boat- and shore-based) in Lake Mulwala, the Murrumbidgee River and the Murray River

<table>
<thead>
<tr>
<th></th>
<th>Mulwala boat-based</th>
<th>Mulwala shore-based</th>
<th>Murrumbidgee boat-based</th>
<th>Murrumbidgee shore-based</th>
<th>Murray boat-based</th>
<th>Murray shore-based</th>
</tr>
</thead>
<tbody>
<tr>
<td>Surface area (ha)</td>
<td>3,561</td>
<td>201</td>
<td>1,019</td>
<td>1,019</td>
<td>13,424</td>
<td>13,424</td>
</tr>
<tr>
<td>Harvest specificity</td>
<td>92%</td>
<td>19%</td>
<td>71%</td>
<td>3%</td>
<td>16%</td>
<td>3%</td>
</tr>
<tr>
<td>Relative stock density</td>
<td>6</td>
<td>0</td>
<td>27</td>
<td>25</td>
<td>21</td>
<td>0</td>
</tr>
<tr>
<td>Effort density (fisher-h/ha)</td>
<td>16.6</td>
<td>160.1</td>
<td>20.0</td>
<td>10.3</td>
<td>26.4</td>
<td>13.3</td>
</tr>
<tr>
<td>Discard density (Murray cod/ha)</td>
<td>2.4</td>
<td>18.8</td>
<td>13.8</td>
<td>2.3</td>
<td>3.2</td>
<td>0.6</td>
</tr>
<tr>
<td>Harvest density (Murray cod/ha)</td>
<td>0.3</td>
<td>1.8</td>
<td>1.1</td>
<td>0.1</td>
<td>0.2</td>
<td>0.1</td>
</tr>
</tbody>
</table>
4.4.2 Shore-based fishery

Thirty five percent of summer fishing effort in Lake Mulwala was expended in the shore-based fishery (Table 6). Shore-based fishing parties were more generalist in species targeted than the boat-based fishery with 64% targeting Murray cod, 25% with no preference (target species = ‘anything’), and 9% targeting carp (Figure 4). Ninety three percent of shore-based fishers used bait whereas 2% used lures (Figure 5).

The summer shore-based Murray cod catch-and-release rate in Lake Mulwala was 0.115 (± 0.019 SE) fish/fisher-hour. Catch-and-release rates for other species were ≤ 0.007 fish/fisher-hour (Table 7). The shore-based harvest rate of carp was 0.037 (± 0.007 SE) fish/fisher-hr and for Murray cod was 0.011 (± 0.003 SE) fish/fisher-hr. Harvest rates for other species in Lake Mulwala were zero. Shore-based catch-and-release of Murray cod in Lake Mulwala was estimated at 3,778 (± 941 SE) fish. Catch-and-release estimates for other species were low (≤ 96 fish; Table 8). The shore-based harvest estimate for carp was 1,496 (± 452 SE) and for Murray cod was 358 (± 125 SE). Harvest estimates for other species in Lake Mulwala were zero (Table 8).

Thirteen Murray cod ranging from 530-650 mm were measured during interviews with shore-based fishers in Lake Mulwala with 92% (12 fish) within the current harvest slot limit. One Murray cod was undersized (i.e., < 550 mm; Figure 6). Forty nine carp were harvested from 300-700 mm. The shore-based released component for Murray cod in Lake Mulwala was 91.3%. 99.8% of Murray cod released in the shore-based fishery were undersized, and 0.2% of harvest-eligible Murray cod were voluntarily released (Figure 7).

Shore-based effort density varied from 160.1 fisher-hours/ha in Lake Mulwala to 10.3 fisher-hours/ha in the Murrumbidgee River (Table 9). Discard density varied from 18.8 fish/ha in Lake Mulwala to 0.6 fish/ha in the Murray River (Table 9). Harvest density varied from 1.8 fish/ha in Lake Mulwala to < 0.1 fish/ha in the Murray River (Table 9). Shore-based harvest specificity was higher in Lake Mulwala (19%), than in the Murrumbidgee and Murray rivers (both 3%; Table 9). The relative stock density for shore-based fisheries varied from 25 in the Murrumbidgee River, to zero in Lake Mulwala and the Murray River (Table 9).
4.5 Discussion

The boat- and shore-based fisheries in Lake Mulwala are very different. The boat-based fishery almost exclusively targeted Murray cod and during the summer season accounted for most fishing effort. Most of the harvest and discard of Murray cod in Lake Mulwala was therefore from the boat-based fishery. In addition, boat-based fishers harvested larger Murray cod and released more harvest-eligible Murray cod than shore-based fishers during the survey period. In contrast, the shore-based fishery was a more diverse, multi-species fishery, characterised by low harvest specificity for Murray cod. The generally improved status of boat-based over shore-based fisheries for Murray cod may be related to the fishing methods used and habitat preferences of this species.

Differences in fishing preferences could be largely discussed in the context of fishing methods. Shore-based fishers almost exclusively used bait, whereas boat-based anglers mainly used lures. Bait fishing is an effective method to capture a range of MDB species (Brown 2010; Rowland 1989), whereas lure fishing can be a selective method for Murray cod (Forbes 2011). In addition to using selective methods for Murray cod, the habitat preferences of this species may favour boat-based fishers. Murray cod are often sedentary and prefer habitats that include structural woody debris, deeper water, and overhanging vegetation (Koehn 2009; Koehn et al. 2009). Such habitats are more accessible to mobile boat-based fishers, which may explain why shore-based fishers were less focused on Murray cod, and had increased harvest of less habitat-specific, generalist species such as carp.

Assessment of standardised parameters revealed variability amongst fisheries for Murray cod in Lake Mulwala, the Murrumbidgee River, and the Murray River. However, a general trend was identified where boat-based fisheries generally exhibited more desirable aspects of fishery quality (e.g. larger fish, higher harvest and discard) than for shore-based fisheries within each waterbody. These results could be caused by normal fluctuations in abundance (Post et al. 2002), by shifts in fishing effort (Arlinghaus and Cooke 2009), or as a result of different regulations governing each fishery at the time of survey. For example, the 550-750 mm harvest slot limit regulation
for Murray cod in New South Wales waters (New South Wales Department of Primary Industries 2015) was introduced just prior to the commencement of sampling for the Lake Mulwala fishery. The harvest slot replaced a 600 mm MLL, which governed the Murrumbidgee River fishery during this surveys 2012-2013 study period (Forbes et al. 2015c), whereas data on the Murray River fishery was collected during a transitional period (2006-2008) when the MLL increased from 500 mm to 550 mm (Brown 2010). It is therefore vital to continue monitoring recreational fisheries for Murray cod as populations adjust to changing harvest restrictions.

The harvest slot for Murray cod was introduced to allow a more natural age structure (Gwinn et al. 2013), as the previous minimum length regulations caused populations to truncate in length at the MLL (Nicol et al. 2004a). Our data illustrates that most Murray cod releases in Lake Mulwala were mandatory as fish length was below the existing lower slot bound, and also that there were few large fish in the population. Low numbers of large Murray cod and high incidence of undersized fish were also evident from surveys conducted in the Murrumbidgee River (Forbes et al. 2015c) and Murray River fisheries (Brown 2010), which suggests that this population structure may be common amongst recreational fisheries for this species. In addition, the abundance of larger Murray cod in the lower Goulburn River (a Murray River tributary) declined from surveys of recreational fishers conducted in 2006/07 and 2008/09 (Brown 2011). Therefore, the results of this and other surveys justify the requirement for a harvest slot to protect small and large Murray cod, and provide crucial baseline data upon which the success of the slot can be assessed.

The applicability of estimates obtained from surveys of recreational fisheries is largely dependent on the quantity and quality of data that is collected and used. For example, the number of Murray cod measured during our survey in Lake Mulwala was relatively low. More fish would have been measured by having higher levels of daily replication (i.e. more sampling days), and/or increased coverage of access points during the survey. Alternatively, if the fishery is based mainly on a single or iconic species and the peak season of fishing is known, it is possible to restrict sampling effort to that season thereby maximising replication when most fishing occurs. Such intensive sampling effort was used in this study, and is evident in other fisheries. For example, creel surveys of fisheries for Northern pike, *Esox lucius*, in North America were restricted to
the peak season and were successful in quantifying exploitation of this species (Pierce et al. 1995). In this regard, increased benefit (i.e. higher sample sizes and lower cost) can be gained from intensive studies of peak fishing seasons.

We identified that boat-based fisheries for Murray cod had better indicators of quality than shore-based fisheries, which may be resultant of differing fishing methods. Variability in standardised parameters was evident amongst fisheries in different waterbodies, which may have been caused by fluctuations in abundance, shifts in fishing effort, or from changing length-based harvest restrictions. Murray cod populations in Lake Mulwala (this study), the Murray River (Brown 2010), and the Murrumbidgee River (Forbes et al. 2015c), were comprised mostly of undersized fish with few large fish evident, which justifies the introduction of the harvest slot limit in an effort to increase protection for fish longer than the upper slot bound. The cost of a well-designed and replicated recreational fishing survey can be large and it is important that the survey data be used effectively to maximise cost-efficiency (Steffe et al. 2008). Fishing surveys can be viewed as ‘building blocks’ that provide important information used to assess relative changes among different fisheries and through time (e.g. Cass-Calay and Schmidt 2009; Hunt et al. 2011). In this sense, the collection of time-series data should be regarded as essential for fisheries managers to understand changes within a fishery and the effectiveness of interventions such as the harvest slot limit for Murray cod.
Chapter 5

System-specific variability in Murray cod and golden perch maturation and growth influences fisheries management options

Chapter 5 has been published as:

5.1 Abstract

The Murray cod and golden perch are important recreational species in Australia’s Murray-Darling Basin (MDB); both species have declined substantially, but recovery is evident in some areas. Minimum length limits (MLLs) – implemented to ensure fish could spawn at least once prior to harvest eligibility – have increased three times in the past decade. The variation in length at 50% maturity (LM50), age at 50% maturity (AM50), and von Bertalanffy growth parameters \(k = \text{Brody growth coefficient}; L_\infty = \text{asymptotic length}; t_0 = \text{theoretical age at zero length}\) was quantified for populations of these species within two rivers and two reservoirs of the MDB. This facilitated the investigation of whether fish length is a suitable surrogate for AM50 in setting MLLs. Between 2006 and 2013, 1,118 Murray cod and 1,742 golden perch were collected by electrofishing and gill netting. Values of \(k\) and \(L_\infty\) were greater for reservoir fish than for riverine fish. For both species, AM50 was generally greater in rivers than in reservoirs; for Murray cod, LM50 was greater in reservoirs than in rivers. A yield-per-recruit model demonstrated that smaller Murray cod MLLs for rivers and that an MLL at or below 600 mm (the existing MLL) across all populations could lead to overfishing in some systems. The differences in growth rate and the onset of reproductive maturation between riverine and reservoir populations suggest that system-specific regulations would be more effective at reducing overfishing risk and meeting fishing quality objectives.
5.2 Introduction

Length- and age-at-maturity can vary within and among fish populations (Köster et al. 2013; Trippel 1995), which has implications for length-based harvest restrictions. Length-based restrictions are simple and generally accepted by anglers as they are easy to understand, demonstrate active management and conservation ethics, and can be equitably applied across fisheries (Stewart 2008). Such regulations require the basic assumption that undersize fish are released alive, eventually reach sexual maturity, and then contribute to the population before they are harvested (Coggins et al. 2007). However, application of length-based regulations to fish populations with highly variable maturation can lead to overfishing if the minimum length limit (MLL) is not sufficiently conservative (van Poorten et al. 2013).

In a fisheries management context, maturity is often defined as the length where 50% of fish are sexually mature (LM50; Burr 1991; van Poorten et al. 2013; Walters and Martell 2004). However, age may be a more accurate estimate of maturity in some fish species (see Trippel 1995); if this is indeed the case, then it may be more appropriate to set length based harvest restrictions by using the age at which 50% of the fish are mature (AM50), instead of the LM50. Analysis of the correlation between AM50 and the von Bertalanffy length-age relationship (Bertalanffy 1938) allows for calculation of length at AM50, which can be used to set length-based harvest restrictions (e.g. Kennedy and Sutton 2007). The suitability of either LM50 or AM50 to set length-based harvest restrictions is important to ensure that sustainability and management objectives are based on the most accurate maturity parameter. Such knowledge is available for some popular recreational species such as striped bass (Berlinsky et al. 1995), but information on the utility of LM50 and AM50 is still lacking for many species.

Two main native fish species are targeted by recreational fishers in Australia’s Murray-Darling Basin (MDB): the Murray cod and the golden perch (Allen et al. 2009; Hunt et al. 2010). Murray cod are Australia’s largest wholly freshwater fish; individuals as large as 1,800 mm total length (TL) and 113.5 kg (McDowall 1996), and as old as 48 years (Rowland 1998a) have been recorded. Golden perch are also long lived (up to 26 years; Stuart 2006), and individuals as large as 760 mm TL and 23 kg have been observed (Lake 1967a). Commercial catch records dating back to 1883 suggest that stocks of
native fishes have declined substantially over the past century (Reid et al. 1997b). A sharp decline in the commercial catch of Murray cod and golden perch in the mid-1950s and 1960s (Rowland 2005) led to the complete closure of the commercial fisheries in 2001. There is a large recreational fishery for both species throughout the MDB (e.g. Chapters 3 and 4); both fisheries are governed by size and bag limits, and there is a closed season during the spawning period for Murray cod (Table 2). The fisheries are supported and enhanced by stocking programs to conserve and recover both species (Chapter 6: Crook et al. 2015; Forbes et al. 2015b).

Existing harvest restrictions for Murray cod and golden perch are based on LM50 estimates (Allen et al. 2009; Brown 2010; Rowland 2005). The Murray cod MLL was increased from 500 to 600 mm TL since the introduction of the MLL in 1992. These changes were based on data collected between 1978 and 1984, which estimated that the Murray cod LM50 at between 500 and 550 mm (Rowland 1998b). The underlying assumption was that most Murray cod are mature at 600 mm, and that this occurs consistently across the species’ range. More recently, implementation of a harvest slot limit (i.e., a regulation that uses minimum and maximum lengths), rather than an MLL increase for Murray cod, was suggested to reduce the risk of further Murray cod decline, and to increase catch rates (Gwinn et al. 2015; Koehn and Todd 2012).

Variation in the onset of reproductive maturation could influence the effectiveness of regulations to prevent overfishing, but so far no studies have quantified the degree of variation in Murray cod and golden perch LM50 and AM50 values among waterbodies or system types (e.g., rivers versus impoundments). Therefore, I aimed to quantify variation in length at maturity, age-at-maturity, and growth parameters, for Murray cod and golden perch within several riverine and impounded systems of the MDB, and I investigated whether fish length is a suitable surrogate for age at maturity in these species. An understanding of variability in the onset of reproductive maturation will inform fishery managers about the appropriateness of basin wide size limits and the need to consider system-specific size limits. The goal of identifying whether length is a suitable predictor of maturity seeks to validate the management assumption that setting MLLs based on length reflects the onset of reproductive maturation.
5.3 Methods

5.3.1 Study sites

Murray cod and golden perch were collected from the Murray and Murrumbidgee rivers, and from Burrinjuck and Copeton reservoirs within the MDB. Burrinjuck and Copeton reservoirs were included in the study to provide data from impounded populations. In addition, these reservoirs support popular recreational fisheries for Murray cod and golden perch (see Hall et al. 2012; Rowland 2005).

Within the Murray and Murrumbidgee rivers, the sections (reaches) selected for this study formed part of a long-term project that commenced prior to an increase in the Murray cod MLL (Crook et al. 2015), thus providing before-and-after data via a standardized collection method. In addition, these reaches contain wild and stocked Murray cod and golden perch that are targeted by popular recreational fisheries (e.g. Forbes and Asmus 2006; Forbes and Asmus 2007). The Murray River sampling reach extended downstream of Yarrawonga Weir (35.9785 S 145.9511 E) to Tocumwal (35.8146 S 145.4912 E). The Murrumbidgee sampling reach extended from Berembed Weir (34.8800 S 146.8370 E) to Euroley Bridge (34.6326 S 146.3627 E; Figure 1).

5.3.2 Study design

The Murray River, Murrumbidgee River and Copeton Reservoir were sampled from 2009 to 2013. Burrinjuck Reservoir was sampled from 2006 to 2013. All study sites were sampled over a 1-week period twice per year (in austral autumn and spring).

We used electrofishing and monofilament gill nets to collect Murray cod and golden perch in the impounded systems (up to 300 individuals.species\(^{-1}\).trip\(^{-1}\)). Electrofishing was conducted by using a boat fitted with a Smith-Root 7.5 GPP electrofisher that was operated at 1,000 V DC, 120 Hz, and a 10-30% duty cycle, producing 3-6 A. Electrofishing effort was twelve 90-s (power on) samples at each site. Gill nets were 30 m long, and comprised 5-m panels with mesh sizes of 27, 25, 50, 75, 100 and 150 mm. Each net was soaked for 2 h. Impoundment sampling was standardized across five sites to provide even spatial distribution across a range of habitats. When adverse environmental or hydrological conditions prevented us from collecting the maximum fish numbers by standardised sampling, additional electrofishing and gillnetting were
undertaken. All of the fish captured from a given impoundment were pooled for analysis.

To ensure a representative riverine sample, I collected fish from four sites in the Murray River and four sites in the Murrumbidgee River. Sites within river reaches were sampled during daylight hours by use of electrofishing only. Sampling effort was twenty 10-min (time elapsed) electrofishing samples. Fish collected at each site were pooled for each river reach.

Collected fish (from all sampling methods and all sites) were measured (TL, mm) and weighed (g), and then were euthanized in an ice slurry (Barker et al. 2002). Sagittal otoliths were removed and sectioned, and age was estimated using the methods validated by Anderson et al. (1992a), Anderson et al. (1992b), and Stuart (2006). Two independent otolith readers performed age estimation for each fish. Age discrepancies were settled with a concert read. Maturity was assessed by visual examination of the gonads as described by Rowland (1998b) for Murray cod, and Mallen-Cooper and Stuart (2003) for golden perch.

5.3.3 Data analysis

To identify the strength of predictive relationships, binary logistic regression models were initially fitted to the length-at-age data for Murray cod and golden perch. Length and age were the covariates, and maturity was the dependant variable. I used Akaike’s information criterion (AIC; Akaike 1998) as a model selection methodology to assess whether a model with waterbody as an explanatory variable or a similar model that also included the TL × waterbody was more parsimonious. If the waterbody was deemed important in the model, I used statistical hypothesis tests to compare the model parameters between waterbodies. Parameters derived from a logistic regression were used to plot probability of maturity based on each model;

\[
y = \frac{e^{(mx+b)}}{1+e^{(mx+b)}}
\]

where \(y\) is the probability of maturity (immature = 0, mature = 1); \(e\) is the base of natural logarithms; \(m\) is the slope; \(x\) is fish TL (mm) or age (years); and \(b\) is the intercept.
After testing model parameters, I compared the biological parameter of importance (either LM50 or AM50). The logistic equation was used to predict $x$ from $y$ (i.e. length or age from the probability of maturity) at the LM50 or AM50, and 95% confidence intervals (CIs) were compared between waterbodies. Non-overlapping 95% CIs were deemed significantly different; this method was used with the knowledge that it is conservative (Schenker and Gentleman 2001) and thus risks type II error.

The probability of maturity was used to assess the level of protection afforded by a range of MLLs (e.g. Hobday and Ryan 1997; Zukowski et al. 2012). Initially, I used the model to calculate the proportion of mature individuals (Murray cod or golden perch) at the existing MLL for each species. For Murray cod, I also calculated the proportion of mature fish at the previous MLL (500 mm TL), and at two potential upper bounds (700 and 800 mm TL) for the harvest slot.

The relationship between length and age was investigated by using the von Bertalanffy growth curve (VBGC; Bertalanffy 1938);

$$L = L_\infty \left(1 - e^{-k[t-t_0]}\right)$$

where; $L$ is TL (mm); $L_\infty$ is the theoretical asymptotic (maximum) length (mm); $k$ is the Brody growth coefficient, describing the rate at which $L$ approaches $L_\infty$; $t$ is fish age (years); and $t_0$ is the theoretical age at a length equal to zero.

Parametric tests of VBGC parameters are not appropriate (1) because; $L_\infty$ and $k$ are correlated; (2) these data are usually non-normal, and (3) the results depend on the degree of heterogeneity in the error variances and nonlinearity of the VBGC (Cerrato 1990). Thus, I used the AIC (Akaike 1998) as a guide to determine whether the VBGC parameters were improved by fitting waterbody-specific parameters, instead of using a set of parameters that were common to all waterbodies. First, values of $L_\infty$, $k$ and $t_0$ were fitted across all waterbodies (i.e. the common set of parameters); I then fitted models in which each of the three parameters was allowed to vary among waterbodies. When waterbody-specific parameters were identified as more parsimonious than the common parameters, I used a statistical hypothesis test to determine which waterbodies had significantly different parameters.
To compare the VBGC parameters between waterbodies, I used a permutation test (Forsythe and Frey 1970). I generated 2,000 permutations of length-at-age values, by randomizing fish within each age-group across two of the populations (i.e., pairwise comparisons of the four waterbodies). The difference in parameter estimates was calculated (1) between each pair of real curves and (2) between each pair in the permutation test. The difference in parameters for the real curves was deemed significant (p < 0.05) if it was at the extremes of the permutations (i.e., less than the 50th or greater than the 1,950th permuted difference).

Ages-at-length from the VBGC for Murray cod and golden perch were used to calculate the TL (mm) at the AM50 (e.g. Kennedy and Sutton 2007). Length at AM50 was then tabulated to LM50. The VBGC was also used to assess variability in the age at LM50 for each species and at the species-specific MLLs. Age-at-length (from the VBGC) was used to calculate the time taken by Murray cod to grow through 500–700 mm and 600–800 mm harvest slots. In addition, the difference between the AM50 and the age at the existing MLL was used to assess the number of potential spawning opportunities (i.e., years) for Murray cod and golden perch prior to attaining harvest eligibility.

To explore the effects of LM50 on the optimal choice of MLL, I used the Murray cod model and baseline parameters from Allen et al. (2009). The model was used to predict the spawning potential ratio (SPR, the percentage of the fecundity in the fished relative to the unfished condition) for a range of Murray cod MLLs at four levels of LM50. Stocks with SPRs below 35% are often considered to be at risk of recruitment overfishing. The exploitation rate in all simulations was set at 15%, the baseline from Allen et al. (2009). The model was used to explore how changes in Murray cod LM50 would be expected to influence the likelihood of recruitment overfishing.

### 5.4 Results

Overall, 1,118 Murray cod and 1,742 golden perch were collected for age estimation and maturity assignment. Murray cod TL varied from 59 to 1,270 mm; estimated age ranged from 0 to 37 years. Golden perch TL ranged from 85 to 640 mm, and estimated age from 0 to 27 years (Table 10, Figure 8). For both Murray cod and golden perch, the
best statistical model for investigating the LM50 and AM50 included TL, waterbody and the TL × waterbody interaction (Table 11).

**Table 10**: Number, length and age of Murray cod and golden perch sampled from two rivers and two impoundments in New South Wales, Australia. Length = TL.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Murray cod</th>
<th></th>
<th>Golden perch</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Length (mm)</td>
<td>Age (yr)</td>
<td>Length (mm)</td>
<td>Age (yr)</td>
</tr>
<tr>
<td></td>
<td>n</td>
<td>min</td>
<td>max</td>
<td>n</td>
</tr>
<tr>
<td>Murray River</td>
<td>323</td>
<td>59</td>
<td>1270</td>
<td>0</td>
</tr>
<tr>
<td>Murrumbidgee River</td>
<td>372</td>
<td>71</td>
<td>1170</td>
<td>0</td>
</tr>
<tr>
<td>Burrinjuck Reservoir</td>
<td>214</td>
<td>62</td>
<td>1130</td>
<td>0</td>
</tr>
<tr>
<td>Copeton Reservoir</td>
<td>209</td>
<td>130</td>
<td>1250</td>
<td>0</td>
</tr>
</tbody>
</table>

**Figure 8**: Length frequency histograms for immature and mature Murray cod and golden perch sampled from two rivers and two reservoirs in New South Wales, Australia (MLL = minimum length limit; length = TL, mm).
Table 11: Information criterion of models describing length and age at maturity of Murray cod and golden perch. Length at 50% maturity = LM50; Age at 50% maturity = AM50. Values in bold italics represent the best-performing models (lowest Akaike’s information criterion [AIC] values).

<table>
<thead>
<tr>
<th>Model</th>
<th>Murray Cod AIC</th>
<th>Golden Perch AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>LM50</td>
<td>AM50</td>
</tr>
<tr>
<td>TL + waterbody + (TL x waterbody)</td>
<td>1,037.5</td>
<td>1,140.2</td>
</tr>
<tr>
<td>TL + waterbody</td>
<td>1,038.9</td>
<td>1,170.5</td>
</tr>
<tr>
<td>TL</td>
<td>1,064.4</td>
<td>1,195.9</td>
</tr>
<tr>
<td>Null</td>
<td>1,496.4</td>
<td>1,652.7</td>
</tr>
</tbody>
</table>

5.4.1 Murray cod

The 95% CIs for the mean LM50 of Murray cod in the Murray River (mean LM50 = 479 mm TL) and Murrumbidgee River (481 mm TL) overlapped with each other, and the 95% CIs for the river populations were clearly lower than those for populations in Burrinjuck Reservoir (mean LM50 = 550 mm) and Copeton Reservoir (550 mm TL; Table 12). Similarly, overlapping 95% CIs were found for the mean AM50 of Murray cod in the Murray River (mean AM50 = 6.6 years) and Murrumbidgee River (5.6 years); these 95% CIs were clearly greater than the CIs for Murray cod in Burrinjuck Reservoir (mean AM50 = 4.6 years) and Copeton Reservoir (3.8 years; Table 12).

Table 12: Length and age at 50% maturity of Murray cod and golden perch. (LM50; AM50; with 95% confidence interval [CI] in parentheses) for fish sampled from two rivers and two impoundments in New South Wales. ‘Group’ indicates waterbodies with overlapping CIs; groups with different letters have significantly different LM50 or AM50 values.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>n</th>
<th>LM50 (mm)</th>
<th>Group</th>
<th>AM50 (yr)</th>
<th>Group</th>
</tr>
</thead>
<tbody>
<tr>
<td>Murray River</td>
<td>323</td>
<td>479 [449, 508]</td>
<td>A</td>
<td>6.6 [5.9, 7.2]</td>
<td>A</td>
</tr>
<tr>
<td>Murrumbidgee River</td>
<td>372</td>
<td>481 [459, 506]</td>
<td>A</td>
<td>5.6 [5.2, 6.1]</td>
<td>A</td>
</tr>
<tr>
<td>Burrinjuck Dam</td>
<td>214</td>
<td>550 [518, 588]</td>
<td>B</td>
<td>4.6 [4.2, 5.0]</td>
<td>B</td>
</tr>
<tr>
<td>Copeton Dam</td>
<td>209</td>
<td>550 [524, 586]</td>
<td>B</td>
<td>3.8 [3.4, 4.3]</td>
<td>B</td>
</tr>
</tbody>
</table>

Murray cod length-at-maturity as calculated from the truncated AM50 and VBGC predicted length-at-age (hereafter termed ‘AM50-VB’), was lower in the rivers (Murray River AM50-VB = 485 mm TL; Murrumbidgee River: 486 mm TL) than in the impoundments (Burrinjuck Reservoir AM50-VB = 519 mm TL; Copeton Reservoir 503 mm TL; Table 13). At an MLL of 500 mm, the proportion of Murray cod that were mature varied from 32% (Copeton Reservoir) to 59% (Murrumbidgee River), whereas
at an MLL of 600 mm, the proportion of fish that were mature increased, varying from 70% (Copeton Reservoir) to 81% (Murrumbidgee River; Table 13).

Murray cod in the Murray and Murrumbidgee rivers spend 5.2 and 4.6 years, respectively, within a potential 500-700 mm harvest slot; in contrast, the populations in Burrianjuck and Copeton Reservoir spend 2.4 years within this harvest slot. The time spent in the 600-800 mm harvest slot was longer for Murray cod sampled from the Murray and Murrumbidgee rivers (6.1 and 5.7 years, respectively) than for fish sampled from Burrianjuck and Copeton Reservoirs (2.7 and 2.8 years, respectively). The difference between the age of Murray cod at the 600-mm MLL (or lower bound of the harvest slot) and the AM50, was greater in the Murray River (2.1 years) and Murrumbidgee River (1.8 years) than in Burrianjuck Reservoir (0.3 years) and Copeton Reservoir (0.3 years). At the 500-mm MLL (or lower bound of the harvest slot), the Murray cod populations in the Murray and Murrumbidgee rivers took an additional 0.3 years to each the AM50, and the populations in Burrianjuck and Copeton reservoirs took an additional 0.8 years in to reach the AM50.
<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Treatment (mm)</th>
<th>TL % mature</th>
<th>Predicted age at length (yr)</th>
<th>Observed age range (yr)</th>
<th>Treatment (mm)</th>
<th>TL % mature</th>
<th>Predicted age at length (yr)</th>
<th>Observed age range (yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Murray River</td>
<td>LM50 479</td>
<td>50.0%</td>
<td>5.9</td>
<td>2-12</td>
<td>LM50 360</td>
<td>50.0%</td>
<td>3.8</td>
<td>3-8</td>
</tr>
<tr>
<td></td>
<td>AM50-VB 485</td>
<td>50.0%</td>
<td>6.0</td>
<td>2-12</td>
<td>AM50-VB 364</td>
<td>50.0%</td>
<td>4.0</td>
<td>3-8</td>
</tr>
<tr>
<td></td>
<td>pre 2007 MLL 500</td>
<td>51.4%</td>
<td>6.3</td>
<td>2-14</td>
<td>existing MLL 300</td>
<td>15.9%</td>
<td>1.7</td>
<td>2-6</td>
</tr>
<tr>
<td></td>
<td>existing MLL 600</td>
<td>72.3%</td>
<td>8.7</td>
<td>2-14</td>
<td>SA MLL 330</td>
<td>30.5%</td>
<td>2.7</td>
<td>2-8</td>
</tr>
<tr>
<td></td>
<td>upper slot limit 700</td>
<td>88.6%</td>
<td>11.5</td>
<td>8-12</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>upper slot limit 800</td>
<td>94.0%</td>
<td>14.8</td>
<td>8-15</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Murrumbidgee River</td>
<td>LM50 481</td>
<td>50.0%</td>
<td>4.9</td>
<td>3-8</td>
<td>LM50 371</td>
<td>50.0%</td>
<td>3.9</td>
<td>2-8</td>
</tr>
<tr>
<td></td>
<td>AM50-VB 486</td>
<td>50.0%</td>
<td>5.0</td>
<td>3-8</td>
<td>AM50-VB 375</td>
<td>50.0%</td>
<td>4.0</td>
<td>2-8</td>
</tr>
<tr>
<td></td>
<td>pre 2007 MLL 500</td>
<td>58.8%</td>
<td>5.3</td>
<td>3-10</td>
<td>existing MLL 300</td>
<td>13.8%</td>
<td>2.3</td>
<td>2-4</td>
</tr>
<tr>
<td></td>
<td>existing MLL 600</td>
<td>81.1%</td>
<td>7.4</td>
<td>4-11</td>
<td>SA MLL 330</td>
<td>27.0%</td>
<td>2.9</td>
<td>2-8</td>
</tr>
<tr>
<td></td>
<td>upper slot limit 700</td>
<td>91.2%</td>
<td>9.9</td>
<td>7-11</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>upper slot limit 800</td>
<td>97.5%</td>
<td>13.1</td>
<td>9-13</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burrinjuck Reservoir</td>
<td>LM50 550</td>
<td>50.0%</td>
<td>4.3</td>
<td>3-7</td>
<td>LM50 324</td>
<td>50.0%</td>
<td>4.1</td>
<td>2-6</td>
</tr>
<tr>
<td></td>
<td>AM50-VB 519</td>
<td>50.0%</td>
<td>4.0</td>
<td>3-7</td>
<td>AM50-VB 318</td>
<td>50.0%</td>
<td>4.0</td>
<td>2-6</td>
</tr>
<tr>
<td></td>
<td>pre 2007 MLL 500</td>
<td>40.7%</td>
<td>3.8</td>
<td>3-7</td>
<td>existing MLL 300</td>
<td>39.3%</td>
<td>3.5</td>
<td>2-6</td>
</tr>
<tr>
<td></td>
<td>existing MLL 600</td>
<td>69.5%</td>
<td>4.9</td>
<td>3-7</td>
<td>SA MLL 330</td>
<td>56.3%</td>
<td>4.3</td>
<td>2-6</td>
</tr>
<tr>
<td></td>
<td>upper slot limit 700</td>
<td>85.0%</td>
<td>6.2</td>
<td>5-8</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>upper slot limit 800</td>
<td>96.2%</td>
<td>7.6</td>
<td>6-8</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Copeton Reservoir</td>
<td>LM50 550</td>
<td>50.0%</td>
<td>3.5</td>
<td>2-8</td>
<td>LM50 318</td>
<td>50.0%</td>
<td>2.1</td>
<td>2-3</td>
</tr>
<tr>
<td></td>
<td>AM50-VB 503</td>
<td>50.0%</td>
<td>3.0</td>
<td>2-6</td>
<td>AM50-VB 312</td>
<td>50.0%</td>
<td>2.0</td>
<td>2-3</td>
</tr>
<tr>
<td></td>
<td>pre 2007 MLL 500</td>
<td>32.1%</td>
<td>3.0</td>
<td>2-6</td>
<td>existing MLL 300</td>
<td>40.1%</td>
<td>1.9</td>
<td>1-3</td>
</tr>
<tr>
<td></td>
<td>existing MLL 600</td>
<td>70.1%</td>
<td>4.1</td>
<td>3-6</td>
<td>SA MLL 330</td>
<td>52.7%</td>
<td>2.2</td>
<td>2-3</td>
</tr>
<tr>
<td></td>
<td>upper slot limit 700</td>
<td>91.5%</td>
<td>5.4</td>
<td>4-7</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>upper slot limit 800</td>
<td>98.3%</td>
<td>6.9</td>
<td>6-7</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
The models that included system-specific $L_\infty$, $k$ and $t_0$ for the Murray cod VBGCs were more parsimonious than the model with parameters that were common to all systems (Table 14). Murray cod grew relatively quickly for the first 10 years, followed by a general slowing in growth (Figure 9). Values of $k$ were significantly higher for Murray cod in Copeton Reservoir ($k = 0.1229$), than in the Murray River ($k = 0.0564$; $p < 0.005$) and Murrumbidgee River ($k = 0.0815$; $p < 0.0001$; Table 15). The $L_\infty$ was significantly larger for Murray cod in Burrinjuck Reservoir ($L_\infty = 1524$ mm TL), than for fish in the Murray River ($L_\infty = 1293$ mm TL; $p < 0.001$), or Copeton Reservoir ($L_\infty = 1283$ mm TL; $p < 0.0001$; Table 15).

**Table 14:** Information criterion of models for the selection of von Bertalanffy growth curve parameters for Murray cod and golden perch sampled from two rivers and two impoundments in New South Wales (subscripts: 1 = Murray River; 2 = Murrumbidgee River; 3 = Burrinjuck Reservoir; 4 = Copeton Reservoir). AIC = Akaike’s information criterion; $L_\infty$ = asymptotic length; $k$ = Brody growth coefficient; $t_0$ = theoretical age at zero length.

<table>
<thead>
<tr>
<th>Model</th>
<th>Parameters</th>
<th>Murray Cod AIC</th>
<th>Golden Perch AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unique $L_\infty$ per waterbody</td>
<td>$L_\infty_1, L_\infty_2, L_\infty_3, L_\infty_4, k, t_0$</td>
<td>9,185.1</td>
<td>10,979.0</td>
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<td>Unique $k$ per waterbody</td>
<td>$k_1, k_2, k_3, k_4, t_0$</td>
<td>9,205.5</td>
<td>11,388.4</td>
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<td>11,150.7</td>
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<tr>
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<td>$k, t_0$</td>
<td>9,629.9</td>
<td>11,455.4</td>
</tr>
</tbody>
</table>
Figure 9: Von Bertalanffy growth curves for Murray cod and golden perch sampled from two rivers and two reservoirs in New South Wales. Note that the x- and y-axis scales differ between the two species.
Table 15: Von Bertalanffy growth curve (VBGC) parameters and $R^2$ values for Murray cod and golden perch sampled from two rivers and two impoundments in New South Wales. Results are from pairwise comparison of VBGC parameters ($L_\infty$ [mm TL], $t_0$ [years], and $k$; defined in Table 14) between waterbodies (permutation test; $p < 0.05$; ns = nonsignificant comparison).

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Parameter</th>
<th>Value</th>
<th>Murray River</th>
<th>Murrumbidgee River</th>
<th>Burrinjuck Reservoir</th>
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</thead>
<tbody>
<tr>
<td>Murray River</td>
<td>$L_\infty$</td>
<td>1293</td>
<td>-</td>
<td>-</td>
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</tr>
<tr>
<td></td>
<td>$t_0$</td>
<td>-2.337</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>$k$</td>
<td>0.0564</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.7779</td>
<td>-</td>
<td>-</td>
<td>-</td>
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<tr>
<td>Murrumbidgee River</td>
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<td>1139</td>
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<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>$t_0$</td>
<td>-1.872</td>
<td>ns</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>$k$</td>
<td>0.0815</td>
<td>ns</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.5967</td>
<td>-</td>
<td>-</td>
<td>-</td>
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<tr>
<td>Burrinjuck Reservoir</td>
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<td></td>
<td>$t_0$</td>
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<tr>
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<tr>
<td></td>
<td>$R^2$</td>
<td>0.7995</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Copeton Reservoir</td>
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<td>ns</td>
<td>ns</td>
<td>$P &lt; 0.0001$</td>
</tr>
<tr>
<td></td>
<td>$t_0$</td>
<td>-1.05</td>
<td>$P &lt; 0.0005$</td>
<td>$P &lt; 0.005$</td>
<td>-</td>
</tr>
<tr>
<td></td>
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<td>0.1229</td>
<td>ns</td>
<td>$P &lt; 0.0001$</td>
<td>$P &lt; 0.005$</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.7852</td>
<td>-</td>
<td>-</td>
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</table>

Golden Perch

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Parameter</th>
<th>Value</th>
<th>Murray River</th>
<th>Murrumbidgee River</th>
<th>Burrinjuck Reservoir</th>
</tr>
</thead>
<tbody>
<tr>
<td>Murray River</td>
<td>$L_\infty$</td>
<td>528</td>
<td>-</td>
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<td></td>
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</tr>
<tr>
<td></td>
<td>$k$</td>
<td>0.1426</td>
<td>-</td>
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<tr>
<td></td>
<td>$R^2$</td>
<td>0.5742</td>
<td>-</td>
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<td>Murrumbidgee River</td>
<td>$L_\infty$</td>
<td>463</td>
<td>ns</td>
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<td>-</td>
</tr>
<tr>
<td></td>
<td>$t_0$</td>
<td>-0.5300</td>
<td>ns</td>
<td>-</td>
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</tr>
<tr>
<td></td>
<td>$k$</td>
<td>0.3669</td>
<td>ns</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.7172</td>
<td>-</td>
<td>-</td>
<td>-</td>
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<tr>
<td>Burrinjuck Reservoir</td>
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<td>534</td>
<td>ns</td>
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<td>-</td>
</tr>
<tr>
<td></td>
<td>$t_0$</td>
<td>-0.98</td>
<td>$P &lt; 0.0005$</td>
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</tr>
<tr>
<td></td>
<td>$k$</td>
<td>0.182</td>
<td>p&lt;0.05</td>
<td>$P &lt; 0.0005$</td>
<td>-</td>
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<tr>
<td></td>
<td>$R^2$</td>
<td>0.5708</td>
<td>-</td>
<td>-</td>
<td>-</td>
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<tr>
<td>Copeton Reservoir</td>
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<td>ns</td>
<td>$P &lt; 0.0001$</td>
<td>ns</td>
</tr>
<tr>
<td></td>
<td>$t_0$</td>
<td>-0.288</td>
<td>$P &lt; 0.0001$</td>
<td>ns</td>
<td>$P &lt; 0.0005$</td>
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<tr>
<td></td>
<td>$k$</td>
<td>0.3764</td>
<td>$P &lt; 0.0001$</td>
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</tr>
<tr>
<td></td>
<td>$R^2$</td>
<td>0.7826</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
5.4.2 Golden perch

The 95% CIs for the mean LM50 of golden perch populations in the Murray River (mean LM50 = 360 mm TL) and Murrumbidgee River (371 mm TL) overlapped with each other, and those 95% CIs were clearly greater than the CIs for populations in Burinjuck Reservoir (mean LM50 = 324 mm TL) and Copeton reservoirs (318 mm TL; Table 12). The 95% CIs for mean AM50 of golden perch in the Murray River (4.4 years), Murrumbidgee River (4.9 years), and Burinjuck Reservoir (4.7 years) overlapped with each other and were greater than the 95% CI for golden perch in Copeton Reservoir (mean AM50 = 2.6 years; Table 12).

The AM50-VB of golden perch was greater in the two rivers (Murray River: AM50-VB = 364 mm TL, Murrumbidgee River: 375 mm TL) than in the impoundments (Burinjuck Reservoir: AM50-VB = 318 mm TL; Copeton Reservoir: 312 mm TL; Table 13). The proportion of mature golden perch at the existing 300-mm MLL was lower in the rivers (Murrumbidgee River: 14%; Murray River: 16%) than in the impoundments (Burinjuck Reservoir: 39%; Copeton Reservoir: 40%; Table 13).

The model with system-specific $L_\infty$ for golden perch was more parsimonious than the models with system-specific $k$ or $t_0$; each of the system-specific models performed better than the model with common parameters for all waterbodies (Table 14). Growth rates for golden perch were variable during the first 10 years of life. For example, growth in Copeton Reservoir and the Murrumbidgee River slowed after age 6, but the growth of fish collected from the Murray River and Burinjuck Reservoir did not slow until age 10-12 (Figure 9). The $k$ value for golden perch in Burinjuck Reservoir ($k = 0.1820$) was significantly higher than the value for the Murray River population ($k = 0.1426; p < 0.05$) but was significantly lower than the value for Murrumbidgee River fish ($0.3669; p < 0.0005$; Table 15). The $L_\infty$ was significantly larger for golden perch collected from Burinjuck Reservoir ($L_\infty = 534$ mm TL) and Copeton Reservoir ($L_\infty = 541$ mm TL) than for fish from the Murrumbidgee River population ($L_\infty = 463$ mm TL; both $p < 0.0001$; Table 15).
5.4.3 Model example

The Murray cod model illustrated that the LM50 has strong implications for selecting the appropriate MLL to prevent recruitment overfishing. For example, a reduction in the MLL to 500 mm TL leads to SPR values ranging from 0.15 (for LM50 = 800 mm), to 0.27 (for LM50 = 500 mm; Figure 10). Thus, regardless of length at maturity, the Murray cod stock would be overfished if the MLL is 500 mm. However, the SPR is predicted to range from 0.34 (marginally overfished) to 0.47 (not overfished) if the MLL is 800 mm.

![Figure 10](image)

**Figure 10**: Results from a model predicting the spawning potential ratio (SPR) of Murray cod across a range of minimum length limits (x-axis), lengths at 50% maturity (LM50, mm TL; lines), and the 35% SPR threshold for recruitment overfishing (dotted line; Clark 1991). With the exception of the varying LM50 values, simulations were based on the model and parameters from Allen et al. (2009).

5.5 Discussion

The growth rate and the onset of reproductive maturation in Murray cod and golden perch exhibited substantial differences between impoundments and rivers. Fish in impoundments grew faster and larger than riverine fish. Murray cod length at maturity was lower in rivers than impoundments, whereas golden perch matured at a greater length in rivers than in impoundments. In addition, for both Murray cod and golden perch, the age variation at the existing MLLs was greater for riverine populations than for reservoir populations. Recreational fisheries within MDB impoundments and rivers
are managed differently across state boundaries; however length-based restrictions are applied uniformly to riverine and impounded waters. The observed differences in growth rate and the onset of reproductive maturation between fish in these waterbody types suggest that system-specific regulations would be more effective at reducing the risk of overfishing and meeting fishing quality objectives for Murray cod and golden perch.

Changing environmental conditions, exploitation pressures, food resources and density dependence leading to variable growth rates may explain changes in length at maturity for Murray cod and golden perch. For example, Anderson et al. (1992a) and Rowland (1998a) identified that Murray cod grew at different rates in different waterways; Mallen-Cooper and Stuart (2003) found similar variability in golden perch. Variability in growth is also found in other species; variable environmental conditions probably caused lower length at maturity for white sturgeon in impoundments than in rivers, but also led to variability in length at maturity between impoundments (Beamesderfer et al. 1995). Given that growth rate variability exists among Murray cod and golden perch populations, the use of LM50 from a given system to inform the MLL for one or more additional systems may undermine the basic assumption that most fish are mature at a length below the prescribed MLL, as LM50s may also differ among populations. The setting of MLLs also needs to consider fishing pressure and its associated effect on LM50. For example, overfishing is known to reduce length at maturity in some species (e.g. coastal pikeperch, Sander lucioperca, populations in the northern Baltic Sea; Lappalainen et al. 2016). The situation where length at maturity declines with increasing exploitation does not mean that a lower MLL can be set, as this would not allow recovery to the original state. The likelihood is that the population would adapt and become smaller individuals in general (Allan et al. 2005; Trippel 1995). The ongoing monitoring of LM50 is therefore critical to ensure that appropriate MLLs are applied.

The LM50 of riverine Murray cod populations was below the previous MLL of 500 mm; this would have allowed the majority of fish at least one spawning opportunity prior to their recruitment into the fishery (Rowland 2005). However, because LM50s for the reservoir populations of Murray cod were greater than 500 mm TL, an MLL of 500 mm would have hindered the goal of providing at least one spawning opportunity before fishery recruitment. Hence, these data support the current regulation of a 600-mm MLL.
for Murray cod. The change from a 500-mm MLL to a 600-mm MLL therefore increased protection for spawning populations of Murray cod in rivers (20.9% increase for the Murray River; 22.3% increase for the Murrumbidgee River) as well as impoundments (28.8% increase for Burrinjuck Reservoir; 38.0% increase for Copeton Reservoir). In contrast, the 300-mm MLL for golden perch in New South Wales (NSW) offers little protection for the spawning population. The 330-mm MLL in South Australian is considered effective in protecting the reproductive potential of golden perch (Ye 2004), but I found this MLL to be below the LM50s for riverine golden perch populations. An MLL lower than the LM50 potentially exposes the golden perch to recruitment overfishing (wherein harvest exceeds the population’s ability to replace itself; Allen et al. 2013; Coggins et al. 2007), and could contribute to population decline if recreational fishing becomes a major stressor in these systems.

The link between age, length and maturity in Murray cod and golden perch was demonstrated by the agreement between length-based maturity calculations (LM50) and age-based maturity calculations (AM50-VB). The results confirmed that despite variable growth rates, the LM50 is an accurate parameter from which to set the MLLs for Murray cod and golden perch; furthermore, the results validated the management assumption that length is appropriate for use in setting the MLLs for these species. In application to other species, setting regulations based on either the AM50 or the LM50 can be problematic because both contain uncertainty. For example, length was the only relevant determinant of maturation probability in Powan Coregonus lavaretus (Ficker et al. 2014), whereas orange roughy Hoplostethus atlanticus (Horn et al. 1998) and Atlantic cod (Beacham 1983) exhibited variable length and age at maturation. Fish length was found not to be a reliable determinant of maturity status in striped bass (Berlinsky et al. 1995). Further contributing to the uncertainty is the fact that the AM50 changes with growth rates and fishing levels (Köster et al. 2013).

The LM50s of Murray cod from the riverine sites were lower than values identified for riverine Murray cod in previous studies; 519 mm (Allen et al. 2009) and between 500 and 550 mm (Rowland 1998b), similar to the reservoir data. The modelling example demonstrated that as Murray cod length at maturity increased, the MLL that would be required to prevent overfishing also increased; thus, overall smaller MLLs could be applied to riverine than to impounded systems. However, a direct comparison of Murray
cod LM50 values from this study, with the values from Rowland (1998b) and Allen et al. (2009) was not possible, as Rowland (1998b) pooled samples from five separate rivers, and Allen et al. (2009) used unpublished data as a baseline. Despite the comparative limitations, the results indicate that a clear understanding of Murray cod length at maturity and growth rates is be needed for informed regulation choices, as application of a constant MLL across all populations could lead to overfishing in some systems if that MLL is at or below 600 mm. Accordingly, the observed variation in length at maturity across Murray cod populations would substantially influence the MLL required to prevent overfishing. Maturation, growth and natural mortality form part of an overall life history that is subject to established quantitative interrelations, such as the correlation between fish length and LM50 among Murray cod populations found in this study. However, in the modelling example for Murray cod, variable growth was not considered, but is likely to have an effect (Allen et al. 2013; Allen et al. 2009). For example, a fish that grows rapidly may be above the MLL, but not yet mature, and conversely, a slow growing fish may be sexually mature, but never attain the MLL.

Variability in LM50 was also evident for the golden perch populations examined within this study and other studies. For example, LM50 values of 350 mm (Reynolds 1983) and 361 mm (Mallen-Cooper and Stuart 2003) from prior studies of golden perch were similar to those identified at the riverine sites in this study, but a previous LM50 estimate of 225 mm (Roberts et al. 2008) was much lower than those I observed. Despite the greater LM50 variation in Murray cod than in golden perch, the inconsistent LM50 results between rivers and impoundments in this study and the differing results between the present work and previous studies demonstrate that the collection of time-series data from multiple locations is needed for accurate maturity assessment in both species. It is therefore important for MLLs to be set above known LM50 values (e.g. Allen et al. 2009; present study; Rowland 1998b). Accordingly, although I identified the 500-mm MLL as suitable for Murray cod in the Murray and Murrumbidgee rivers, consideration of the variability in age and length at maturity indicates that a 600-mm MLL is a more conservative approach—recognizing, however the model prediction that some populations may still be at risk of overfishing at this MLL.
Given that golden perch spawn in rivers and have not been shown to recruit in deep impoundments (King et al. 2003; Mallen-Cooper and Stuart 2003), a conservative approach would be to set MLLs based on LM50 values from riverine golden perch populations. Setting the appropriate MLLs based on LM50 may not be crucial for protecting golden perch spawners in ‘put-and-take’ reservoir fisheries, but length-based harvest restrictions are important for allowing fishery managers to control aspects of angling quality, such as population density and size structure, in these fisheries (Hansen et al. 2015). However, protection of wild spawning populations is a key consideration when setting MLLs; wild populations have formed the basis upon which length-based harvest restrictions are used to manage important recreational species, such as the shovelnose sturgeon *Scaphirhynchus platorynchus* and striped bass. In both cases, MLLs were set using data collected from the same system where the regulations were applied (see Berlinsky et al. 1995; Kennedy and Sutton 2007; Richards and Rago 1999; Trent and Hassler 1968). In particular, striped bass population recovery was achieved by using appropriate MLL’s (Richards and Rago 1999). Thus, it may be appropriate to set the golden perch MLL above the riverine LM50 range (225–371 mm TL) identified in this and other studies.

For exploited fisheries, a reduction in population size is typically correlated with a corresponding reduction in length and age at maturity, as was observed in Pacific halibut *Hippoglossus stenolepis* (Trippel 1995). However, growth rates generally improve in exploited populations (Botsford 1981), potentially leading to variability in length at maturity (Godo and Haug 1999). The predominantly wild riverine Murray cod fisheries sampled in this study demonstrate LM50 values below that identified by Rowland (1998b). In Rowland’s (1998c) study, Murray cod were collected during a period in which the populations were degraded (Rowland 2005); thus, I could expect Rowland’s (1998c) LM50 values to be lower than ours, but this was not the case. The variability in length at maturity demonstrated among the systems in this study, and the systems studied by Rowland (1998c, 2005) highlights the importance of continued maturity status assessment as fish populations adjust to fishing mortality levels. Accurate data describing the onset of maturity are important for establishing length-based restrictions, and the collection of such data should be an ongoing part of monitoring in fisheries management.
In conclusion, I found that variability in growth rate and the onset of reproductive maturation between riverine and reservoir populations of Murray cod and golden perch may necessitate system-specific MLLs to reduce the risk of overfishing and to achieve fishing quality goals. Both LM50 and AM50 were identified as suitable for use in setting length-based harvest restrictions for these species if the intended objective is to protect the age at first spawning. The 300-mm MLL for golden perch was below the LM50 and therefore did not allow a spawning opportunity prior to harvest eligibility; increasing the MLL to a level above the LM50 may be beneficial for this species. The modelling example with Murray cod illustrated that smaller MLLs would be required for riverine than for impounded systems and that the use of a constant MLL across all populations could lead to overfishing in some systems if that MLL (or the lower bound of the harvest slot) is 600 mm or less.

The sustainable management of Murray cod and golden perch recreational fisheries is underpinned by information such as; quantifying the influence anglers exert on a fishery (Chapters 3 and 4); and understanding maturation onset to inform size limits (Chapter 5). However, other strategies, such as stocking, are often used concurrently to assist sustainable manage of recreational fisheries for these species, and assessment of the relative success of stocking programs is required. Therefore in Chapter 6, I assess the effectiveness of stocking Murray cod and golden perch in a range of rivers and impoundments.
Chapter 6

Assessment of stocking effectiveness for Murray cod and golden perch in rivers and impoundments of south-eastern Australia

Chapter 6 has been published as:

6.1 Abstract

Stock enhancement is a management tool often used to assist fishery recovery worldwide, yet the success of many stocking programs remains unquantified. Murray cod and golden perch are important Australian recreational target species that have experienced widespread decline. Stocking of these species has been undertaken for decades, with limited assessment of effectiveness. I used a mark-recapture approach to assess stocked Murray cod and golden perch survival, contributions to wild fisheries, and condition. Stocked fish were marked with calcein. Marked fish were detected during surveys undertaken 3 years and 10 months from initial marking, and it is likely that marks will persist beyond this time. The proportion of calcein marked fish in the population sub-sample whose age was equal to, or less than, the number of years since release, varied from 7% to 94% for Murray cod, and 9% to 98% for golden perch. Higher proportions of marked fish were found in impoundments than rivers. Marked Murray cod had significantly steeper length-weight relationships (i.e. higher weight at a given length) to unmarked fish. The results show that application of methods for discriminating stocked and wild fish provides critical information for the development of adaptive, location-specific stocking strategies.
6.2 Introduction

Recovery initiatives to rehabilitate declining fish populations include harvest regulations, harvest quantification, mitigation of stressors, and stock enhancement (Cowx 1994; Halverson 2008; Molony et al. 2003). Despite the widespread use of stock enhancement as a fisheries management tool and the development of methods to distinguish stocked fish from wild fish (e.g. Mohler 1997; Munro et al. 2008), the success of many stocking programs remain unassessed (Lorenzen 2014). Additionally, some stocking programs have had little to no success (Cochran-Biederman et al. 2015). Therefore, quantifying stocking success is urgently needed to optimise such programs and to maximise outcomes.

In south-eastern Australia’s Murray-Darling Basin (MDB), stock enhancement has formed the basis of recovery and ongoing management of Murray cod and golden perch for several decades. Murray cod are Australia’s largest wholly freshwater fish, recorded to 1800 mm and 113.5 kg (Rowland 2005). Golden perch grow to a maximum of 760 mm and 23 kg (Lake 1967b). Both species are important recreational targets (Brown 2010; Rowland 2005). Fisheries within the MDB have declined over the past 100 years (Reid et al. 1997b) and native fish communities are estimated at 10% of pre-European settlement levels (Murray-Darling Basin Commission 2003). A number of factors have contributed to the decline in native fish abundance and distribution, including flow regulation, habitat modification, reduced water quality, in-stream barriers, introduced species, disease, exploitation, translocation and indiscriminate stocking (Murray-Darling Basin Commission 2003).

To prevent further Murray cod and golden perch decline, fishery managers have implemented a range of initiatives that include; closure of the commercial freshwater fishery in all states, except the South Australian golden perch fishery (Ye 2004), introduction of closed seasons and harvest restrictions such as bag and size limits, and implementation of stocking programs with hatchery-reared fish to conserve and recover both species (Lintermans et al. 2005). Since 1971, 12.89 million Murray cod (Ingram et al. 2011), and over 32 million golden perch (Gillanders et al. 2006), produced at both government and private (commercial) fish hatcheries across several states, have been
stocked into waterways and impoundments across the MDB to support recreational fishing (Ingram et al. 2011) and for conservation efforts (Lintermans et al. 2005).

If stocking is to play a major role in the ongoing conservation and rehabilitation of Murray cod and golden perch populations, then assessment of the effectiveness of these stocking programs is required to optimise release strategies and support ecologically and economically defensible decision making. For example, despite closure of the New South Wales (NSW) Murray cod commercial fishery in 2001, and implementation of recreational harvest regulations such as bag and size limits in 1992 (Rowland 2005), stocking of this species continues based on the perception that natural recruitment is insufficient to sustain the fishery. In recognition of the need for evidence regarding the outcomes of fish stocking, an initial study was undertaken between 2002 and 2007 to assess the effectiveness of stocking golden perch in the Edward River, Murrumbidgee River, and Billabong Creek (Crook et al. 2015). Crook et al. (2015) marked hatchery-reared golden perch with calcein (Billabong Creek) and alizarin complexone (Murrumbidgee and Edward Rivers), and demonstrated that stocked golden perch contributed 18–100% to populations. Other less extensive assessments of golden perch stocking in the MDB have been performed (see Harris 2002; Hunt et al. 2010), but the outcomes of Murray cod stocking have never been studied in detail. Long-term stocking success studies using chemical marking require a known degree of mark retention and detectability over time (e.g. Hill and Quesada 2010; Mohler 2003). Further research into the long-term retention of calcein marks was highlighted as a priority, so that the limitations of this technique can be understood (Crook et al. 2009).

Stocking effectiveness can be affected by hatchery practices that produce variable fish quality with regard to growth and condition. For example, hatchery-reared Atlantic salmon in Canada were shown to be smaller and in poorer condition than comparable wild fish leading to reduced stocked fish survival (McDonald et al. 1998). The release of hatchery fish in poor condition usually results in low survival and has the potential to undermine enhancement and recovery strategies (e.g. Cameron and Baumgartner 2012). Conversely, competition from hatchery-reared fish can lead to replacement of wild fish with stocked fish, particularly if they are released at a larger size than wild recruits of the same yearly cohort (Sweeting et al. 2003). Quantification of condition parameters
between stocked and wild fish is therefore important for determining the outcomes of fish stocking and to inform stock enhancement strategies.

The present study focusses on the stocking of Murray cod and golden perch in the MDB and aimed to (1) use a chemical marking technique to identify stocked fish and estimate their contributions to populations in several rivers and impoundments, (2) evaluate the necessity of stocking for fishery augmentation, (3) compare length-weight relationships and condition of stocked (marked) fish with unmarked fish, and (4) to assess the long-term retention of calcein marks in golden perch and Murray cod.

6.3 Methods

6.3.1 Study sites

Murray cod and golden perch were sampled from the Murray and Murrumbidgee rivers, Burrinjuck Dam (Murrumbidgee River catchment) and Copeton Dam (Gwydir River catchment) within the MDB (Figure 1). Reaches of the Murray and Murrumbidgee rivers were selected for inclusion in the study as they formed part of a pre-existing long-term project and both reaches were not previously stocked with calcein marked Murray cod and golden perch. Both reaches support popular recreational fisheries (Rowland 2005). The Murray River reach extends downstream of Yarrawonga Weir (35.9785 S 145.9511 E) to Tocumwal (35.8146 S 145.4912 E). The Murrumbidgee reach extends from Berembed Weir (34.8800 S 146.8370 E) to Euroley Bridge (34.6326 S 146.3627 E).

Burrinjuck and Copeton dams were selected for inclusion in the study to provide data for Murray cod and golden perch populations from impoundments. The extent of successful, wild recruitment of Murray cod spawned in large, deep impoundments, with extensive water level fluctuations is not known, whereas golden perch may not reproduce effectively in closed waters (King et al. 2009). Prior to commencing this study, Burrinjuck and Copeton dams did not contain calcein marked Murray cod and golden perch, but received regular stockings of unmarked hatchery-reared fish (Rowland 1995). Thus, unmarked fish in these dams could be of either stocked or wild origin. Both impoundments contain popular Murray cod and golden perch recreational fisheries (e.g. Hall et al. 2012; Rowland 2005).
6.3.2 Study design

To enable identification of stocked Murray cod and golden perch, hatchery-reared fingerlings (30-50 mm total length) were marked with calcein (2,4-bis-[N,N’-di(carbomethyl)-aminomethyl]fluorescein) using osmotic induction (Baumgartner et al. 2012; Crook et al. 2006; Crook et al. 2009). Marked Murray cod were usually released in December or January, and golden perch from February to March depending on hatchery availability. Calcein-marked hatchery-reared Murray cod and golden perch were released, in equal numbers, at the four sampling sites within each river reach, and released at various access points around Burrinjuck and Copeton dams (see Table 16 for numbers of fish released). The distance between impoundment release and sampling sites varied between 0 and 5 km. It was assumed that released fish dispersed throughout the existing population. Government and private native fish hatcheries produced fish for this study and all fish stocked into the study reach were calcein marked. However, a small number (< 1,515 per year) of unmarked Murray cod were inadvertently released in the Murray River in 2010 and 2012, and a larger number (between 7,374 and 25,111) of unmarked golden perch were released into the Murray and Murrumbidgee rivers, and Burrinjuck Dam in 2010 (Table 16). This study was limited by the possibility that some of these unmarked hatchery-reared fish were re-captured, which may have caused underestimates of stocked fish in the affected waterbodies.
Table 16: Murray cod and golden perch stocking releases into the Murray and Murrumbidgee rivers, Burrinjuck and Copeton dams. These data are summarised by year of spawning and whether fish were calcein marked.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Year of spawning</th>
<th>Murray cod calcein marked</th>
<th>Murray cod unmarked</th>
<th>Golden perch calcein marked</th>
<th>Golden perch unmarked</th>
</tr>
</thead>
<tbody>
<tr>
<td>Murray River</td>
<td>2009</td>
<td>22,364</td>
<td>1,515</td>
<td>23,924</td>
<td>25,211</td>
</tr>
<tr>
<td></td>
<td>2010</td>
<td>18,972</td>
<td></td>
<td>27,118</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2011</td>
<td>13,377</td>
<td>1,509</td>
<td>3,636</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2012</td>
<td>35,740</td>
<td></td>
<td>27,908</td>
<td></td>
</tr>
<tr>
<td>Murrumbidgee River</td>
<td>2009</td>
<td>20,143</td>
<td></td>
<td>65,552</td>
<td>7,374</td>
</tr>
<tr>
<td></td>
<td>2010</td>
<td>18,075</td>
<td></td>
<td>50,907</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2011</td>
<td>12,000</td>
<td></td>
<td>72,481</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2012</td>
<td>18,888</td>
<td></td>
<td>56,362</td>
<td></td>
</tr>
<tr>
<td>Burrinjuck Dam</td>
<td>2009</td>
<td>60,000</td>
<td></td>
<td>80,000</td>
<td>14,400</td>
</tr>
<tr>
<td></td>
<td>2010</td>
<td>60,000</td>
<td></td>
<td>66,000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2011</td>
<td>60,000</td>
<td></td>
<td>78,000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2012</td>
<td>36,000</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Copeton Dam</td>
<td>2009</td>
<td>60,000</td>
<td></td>
<td>90,000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2010</td>
<td>54,000</td>
<td></td>
<td>95,000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2011</td>
<td>54,000</td>
<td></td>
<td>89,727</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2012</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The Murray and Murrumbidgee River study sites are constrained by upstream migration barriers (weirs), however fishways along the Murray River (Barrett 2008), and flooding in both systems during 2010 and 2011 (Whitworth et al. 2012), provided opportunity for marked and unmarked fish to migrate in and out of both study reaches. Burrinjuck and Copeton dams offer no fish passage from downstream; however illegal stocking of unmarked Murray cod and golden perch has been reported in many MDB waterbodies (Anderson et al. 1992b; Faulks et al. 2010; Lintermans 2004), with such activity in the respective upper Copeton and Burrinjuck dam catchments potentially resulting in unmarked fish migrating to the impounded water.

The fish used in this study were also utilised for other research where destructive sampling was required to calculate age and length at maturity (Chapter 5; Forbes et al. 2015a). Although non-destructive external detection of calcein marks has been successfully applied for younger age classes (Baumgartner et al. 2012; Crook et al. 2012), field detection of calcein marks in the current study resulted in many false negatives. Non-destructive field detection methods for calcein can be unreliable (Ingram et al. 2015). As such, I only used otoliths from available collections for age and calcein assessment.
Fish were collected from all sites over one week, twice per year (austral seasons autumn and spring), with seasonal data pooled within each year to assess age structure on an annual basis. Fish were collected from the Murray River, Murrumbidgee River, Burrinjuck Dam and Copeton Dam between 2009 and 2013. Sampling generally commenced 1-4 months after initial fish release. The maximum time between fish release and sampling was 3 years 10 months. Boat electrofishing was used to collect a broad size range of fish from river and impoundment sites. Gill netting was used to increase fish numbers collected from impoundments. Electrofishing used a boat fitted with a Smith-Root GPP 7.5 electrofisher unit, operated at 1000 V DC, 120 Hz, 10-30% duty cycle and producing between three and six amps. Gill netting involved up to 15 nets from 25–55 m length with mesh sizes ranging from 40–300 mm, and multi-mesh panel nets (five metres of each mesh in the following order: 100 mm, 150 mm, 50 mm, 27 mm, 35 mm, 75 mm). Number, mesh size and net soak time were not consistent among sites, with nets cleared approximately every two hours.

All Murray cod and golden perch collected were measured (mm), weighed (g), then euthanized (Barker et al. 2002). Sagittal otoliths were removed, sectioned and aged (Anderson et al. 1992a; Anderson et al. 1992b; Stuart 2006), and assessed for the presence of a calcein mark using a fluorescence dissecting stereomicroscope (Model MZ165FC, Leica, Switzerland) and a ‘GFP3’ filter set (470 nm excitation filter with 40 nm half bandwidth, 525 nm band pass barrier filter with 50 nm half bandwidth).

6.3.3 Data analysis

6.3.3.1 Population structure
Calcein marked fish were allocated to year classes based on year of spawning to correlate with the age of natural recruits. For example, a fish spawned in October 2012, and released in January 2013, was deemed a member of the 2012 year class (Crook et al. 2010b). The Murray cod and golden perch sub-sample, whose age was equal to, or less than, the number of years since marked fish were released; I refer to hereafter as ‘calcein possible’. The proportion of marked fish in the calcein possible population was calculated by dividing the total number of marked fish by the total number of calcein
possible fish. These analyses were completed for each of species, waterbody and spawning year.

A log-linear model was used to test whether the proportion of marked fish in the calcein possible population for each species was different between waterbody and spawning year. When the log-linear models were performed using the complete dataset (which included some categories with low counts) poor model convergence was detected, and as such the models were refit including only categories where 10 or more fish were collected. Outputs and interpretations from both models were directly comparable and I present the complete model results.

6.3.3.2 Mark longevity
Maximum mark longevity was assessed by directly observing calcein marks within sectioned and mounted sagittal otoliths. Hatchery-reared Murray cod and golden perch were given the nominal birthdate of 1 October (Anderson et al. 1992b; Mallen-Cooper and Stuart 2003; Stuart 2006) to allow back calculation of age for marked fish at the time of capture. Marked fish were pooled for each species and waterbody by year to assess mark longevity. External calcein mark retention and minimum mark longevity were not assessed in this study.

6.3.3.3 Condition
Of the calcein possible Murray cod and golden perch populations, two indicators of fish condition were used to identify differences that may exist between marked and unmarked fish. Data were pooled for each waterbody and species. The two estimators were:

1) Condition factor; defined as Fulton’s K (e.g. Cameron and Baumgartner 2012; Harris 1987; Ingram 2009), is expressed as:
   \[ K = 100 \left( \frac{W}{L^3} \right) \]
   where \( W \) is fish weight (g), \( L \) is fish length (cm) and \( K \) is fish condition.

2) Logarithmic length-weight relationship;
   \[ \log_{10} W = b(\log_{10} L) + \log_{10} a \]
   where \( W \) equals fish weight (g), \( L \) is total fish length (mm), \( b \) is the slope (exponent), and \( a \) is the y-intercept determined from empirical data. The length-
weight relationship is considered isometric when \( b \) (regression slope) = 3 (Cameron and Baumgartner 2012).

Two way analysis of variance (ANOVA) was used to test for differences in condition factor between marked status (calcein marked or unmarked) and waterbody in the calcein possible population for each species. When the two-way interaction was not significant \( (p > 0.05) \), subsequent tests for differences in condition factor were performed independently for waterbody and marked status. Analysis of covariance (ANCOVA) was used to test for differences between waterbody, \( \log_{10}\text{length} \) and marked status on weight in the calcein possible population for each species. When the interaction of these three effects was significant, I used separate ANCOVA to test the two-way effects of; waterbody and \( \log_{10}\text{length} \); and marked status and \( \log_{10}\text{length} \). If the two-way interactions were not significant \( (p > 0.05) \), subsequent tests for waterbody, \( \log_{10}\text{length} \) and marked status were performed independently. The ANCOVA included \( \log_{10}\text{weight} \) as the response variable, \( \log_{10}\text{length} \) as the covariate and waterbody and marking status as fixed factors. When the ANCOVA or two-way ANOVA showed significant interactions with, or differences between effects, Tukey’s HSD test was used post hoc to identify groups that were significantly different.

Burrinjuck Dam Murray cod were excluded from condition factor and length-weight relationship analysis because of small sample sizes.

### 6.4 Results

A total of 1,093 Murray cod and 1,438 golden perch were collected across all sites. Of these, 343 Murray cod and 222 golden perch were calcein possible (fish whose age was equal to, or less than, the number of years since marked fish were released). Murray cod varied in length from 59–1,270 mm (59–630 mm for the calcein possible fish), whilst age spanned 0–37 years (0–4 years for the calcein possible fish). Golden perch varied in length from 85–640 mm (85–427 mm for the calcein possible fish), with an age span of 0–27 years (0–3 years for the calcein possible fish; Table 17).
Table 17: Number, length and age of Murray cod and golden perch sampled from the Murray and Murrumbidgee rivers, Burrinjuck and Copeton dams in New South Wales, Australia. Minimum and maximum length and age values are shown. Calcein possible are those fish whose age was equal to, or less than, the number of years since marked fish were released.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Dataset</th>
<th>Murray cod</th>
<th></th>
<th></th>
<th>Golden perch</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>n</td>
<td>length (mm)</td>
<td>Age (yr+)</td>
<td>n</td>
<td>length (mm)</td>
<td>Age (yr+)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>min</td>
<td>max</td>
<td>min</td>
<td>max</td>
<td>min</td>
<td>max</td>
</tr>
<tr>
<td>Murray River</td>
<td>all fish</td>
<td>323</td>
<td>59</td>
<td>1270</td>
<td>0</td>
<td>37</td>
<td>195</td>
</tr>
<tr>
<td></td>
<td>calcein possible</td>
<td>101</td>
<td>59</td>
<td>603</td>
<td>0</td>
<td>4</td>
<td>22</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1270</td>
<td>0</td>
<td>37</td>
<td>195</td>
<td>211</td>
<td>582</td>
</tr>
<tr>
<td></td>
<td></td>
<td>27</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Murrumbidgee River</td>
<td>all fish</td>
<td>373</td>
<td>71</td>
<td>1170</td>
<td>0</td>
<td>23</td>
<td>228</td>
</tr>
<tr>
<td></td>
<td>calcein possible</td>
<td>105</td>
<td>71</td>
<td>595</td>
<td>0</td>
<td>4</td>
<td>37</td>
</tr>
<tr>
<td></td>
<td></td>
<td>595</td>
<td>0</td>
<td>37</td>
<td>110</td>
<td>394</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>17</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burrinjuck Dam</td>
<td>all fish</td>
<td>178</td>
<td>119</td>
<td>1130</td>
<td>0</td>
<td>13</td>
<td>414</td>
</tr>
<tr>
<td></td>
<td>calcein possible</td>
<td>8</td>
<td>119</td>
<td>630</td>
<td>1</td>
<td>3</td>
<td>43</td>
</tr>
<tr>
<td></td>
<td></td>
<td>630</td>
<td>1</td>
<td>43</td>
<td>85</td>
<td>410</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Copeton Dam</td>
<td>all fish</td>
<td>219</td>
<td>107</td>
<td>1250</td>
<td>0</td>
<td>19</td>
<td>601</td>
</tr>
<tr>
<td></td>
<td>calcein possible</td>
<td>129</td>
<td>107</td>
<td>610</td>
<td>0</td>
<td>3</td>
<td>120</td>
</tr>
<tr>
<td></td>
<td></td>
<td>610</td>
<td>0</td>
<td>3</td>
<td>104</td>
<td>427</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>3</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

6.4.1 Murray cod

6.4.1.1 Population structure

The proportion of marked Murray cod in the calcein possible population was greater in Copeton Dam (94%), than in the Murray River (7%) or the Murrumbidgee River (15%; Table 18). No calcein marked Murray cod were collected from Burrinjuck Dam, and only eight unmarked calcein possible individuals were collected from this impoundment during the study (Table 18). The significant difference in the proportion of marked Murray cod between waterbodies (W) was dependent on spawning year (SY; \( \chi^2_{SY\times W} = 307.1, \text{df} = 15, p < 0.0001 \)). Of the spawning years where 10 or more fish were collected from Copeton Dam, the percentage of marked fish was always 80% or more, whereas in either river, spawning year classes where 10 or more fish were collected never had more than 17% marked (Table 18).

The maximum possible age that could be determined for marked Murray cod in the Murray and Murrumbidgee rivers and Burrinjuck and Copeton dams was 3 years 10 months (the study duration). Calcein marks were detected in Murray cod sagittal otoliths aged up to 3 years 10 months in the Murrumbidgee River and Copeton Dam, and in fish aged up to 3 years 5 months in the Murray River.
Table 18: Calcein marked and unmarked Murray cod and golden perch sampled from the Murray and Murrumbidgee rivers, Burrinjuck and Copeton dams in New South Wales, Australia. Only those fish aged equal to, or younger than the years since calcein marked fish were first stocked, are included in this table.

<table>
<thead>
<tr>
<th>Waterbody</th>
<th>Murray cod</th>
<th>Golden perch</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>n marked</td>
<td>unmarked</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Murray River</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>17</td>
<td>0</td>
</tr>
<tr>
<td>2010</td>
<td>6</td>
<td>1</td>
</tr>
<tr>
<td>2011</td>
<td>3</td>
<td>0</td>
</tr>
<tr>
<td>2012</td>
<td>58</td>
<td>6</td>
</tr>
<tr>
<td>2013</td>
<td>17</td>
<td>0</td>
</tr>
<tr>
<td>Murrumbidgee River</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>21</td>
<td>0</td>
</tr>
<tr>
<td>2010</td>
<td>61</td>
<td>10</td>
</tr>
<tr>
<td>2011</td>
<td>4</td>
<td>3</td>
</tr>
<tr>
<td>2012</td>
<td>12</td>
<td>2</td>
</tr>
<tr>
<td>2013</td>
<td>7</td>
<td>1</td>
</tr>
<tr>
<td>Burrinjuck Dam</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2010</td>
<td>8</td>
<td>0</td>
</tr>
<tr>
<td>2011</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2012</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2013</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Copeton Dam</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>10</td>
<td>8</td>
</tr>
<tr>
<td>2010</td>
<td>52</td>
<td>52</td>
</tr>
<tr>
<td>2011</td>
<td>65</td>
<td>61</td>
</tr>
<tr>
<td>2012</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>2013</td>
<td>1</td>
<td>0</td>
</tr>
</tbody>
</table>

6.4.1.2 Condition

There was no significant effect between marked and unmarked Murray cod condition factor, either on its own (F = 0.150, df = 1, 129, p = 0.70) or in the interaction with waterbody (F = 0.212, df = 2, 329, p = 0.81). Condition factor was significantly different between waterbodies (F = 21.2, df = 2, 332, p < 0.001). Mean Murray cod condition factor for the Murray River (1.25) and Murrumbidgee River (1.24) populations were significantly lower than in Copeton Dam (1.37; Figure 11).

The interaction of waterbody, marked status, and length was not significant for Murray cod length-weight relationships (F = 0.765, df = 2, 323 p = 0.47). However length-weight relationships were significantly different between marked and unmarked fish (F = 38.0, df = 1, 331, p < 0.0001). Murray cod length-weight relationships for marked (b = 3.17) and unmarked (b = 2.97) fish were classified as isometric, but the significantly
different regression slopes ($b$) indicate better condition (with respect to length-weight relationship) for marked than unmarked fish (Figure 12).

**Figure 11:** Box plot of condition factors (Fulton’s K) for Murray cod and golden perch sampled from the Murray and Murrumbidgee rivers and Burrinjuck and Copeton dams in New South Wales, Australia. Fish aged equal to, or younger than the years since calcein marked fish were first stocked, are included in this analysis. Box plots are shown with upper and lower quartiles with maximum and minimum outliers.
Figure 12: Scatter plots for log transformed length-weight relationships for Murray cod and golden perch sampled from the Murray and Murrumbidgee rivers and Burrinjuck and Copeton dams in New South Wales, Australia. Fish aged equal to, or younger than the years since calcein marked fish were first stocked, are included in this analysis.
6.4.2 Golden perch

6.4.2.1 Population structure
Copeton Dam had the largest proportion of marked golden perch (98%), compared to Burrinjuck Dam (23%), Murray River (9%) and the Murrumbidgee River (14%) populations (Table 18). The significant difference in the proportion of marked golden perch in the samples between waterbodies was dependent on spawning year ($\chi^2_{SY \times W} = 209.5$, df = 10, p < 0.0001). In the Murrumbidgee River, the proportion of marked golden perch was significantly different between spawning years ($\chi^2_{SY} = 15.2$, df = 2, p < 0.001). This appears to be because 3 out of 3 fish in 2012 were marked, compared to 2 out of 26, and 0 out of 8 fish, being marked in 2010 and 2011 respectively (Table 18).

There were no further significant differences in the proportion of marked fish between spawning year class in either of the Murray River, Burrinjuck and Copeton dams (p > 0.05). Nevertheless, the proportion of marked golden perch for each spawning year class in Copeton Dam was consistently high, varying between 91% (n = 23) for fish spawned in 2012, to 100% (n = 46) for the 2011 year class (Table 18).

Calcein marks were detected in golden perch sagittal otoliths aged up to 3 years 10 months in Burrinjuck and Copeton dams, and in fish aged up to 2 years 10 months in the Murray and Murrumbidgee rivers.

6.4.2.2 Condition
There was no significant effect between marked and unmarked golden perch condition factor, on its own (F = 0.34, df = 1, 213, p > 0.05), or in the interaction with waterbody (F = 1.62, df = 3, 213, p > 0.05). There was a significant difference in the condition factor for golden perch among waterbodies (F = 3.10, df = 3, 213, p < 0.05), with the Murrumbidgee River population having a significantly lower mean condition factor (1.44) than the Burrinjuck Dam population (1.60), but both were not different to the Murray River (1.50) and Copeton Dam (1.50) populations (Figure 11). The significant length-weight relationship for golden perch (F = 466.9, df = 1, 206, p < 0.0001) was not influenced by waterbody or marked status (all p > 0.05; Figure 12).
6.5 Discussion

The comparatively low proportion of marked Murray cod and golden perch in the riverine study sites suggests that these populations are primarily self-supporting through natural recruitment, and/or survival of marked fish is low into these waterbodies. In contrast, marked fish formed a larger proportion of the population within Burrinjuck (golden perch only) and Copeton dams. The virtual absence of unmarked golden perch in Copeton Dam supports observations that this species spawns and recruits in riverine environments (Harris and Gehrke 1994; King et al. 2003), rather than standing waters (King et al. 2009). I also identified few unmarked Murray cod in Copeton Dam, suggesting that natural recruitment of this species was limited. Murray cod have been observed to spawn in standing waters (Rowland 1983b), and larvae have been collected within weir pools (Koehn and Harrington 2005). However, the dominance of stocked fish in the impoundment populations suggests that fisheries in these systems are almost entirely dependent on stocking rather than natural recruitment. Declines in recruitment of golden perch and Murray cod were observed following the construction of several large impoundments throughout the Murray-Darling Basin during the 1950s and 1960’s (Rowland 1985), with populations established in the new reservoirs following successful stocking programs (Rowland 2005; Rowland 2013).

The variable contributions of marked fish demonstrate that adaptive, location-specific fishery enhancement strategies are required to maintain recreational fisheries. For example, the results suggest that in rivers with high levels of natural recruitment, stocking may be inefficient and potentially harmful. However, in some systems the conditions required for successful recruitment may not occur because of river regulation or drought conditions, and stocking may be effective in increasing the abundance of stocked species. Crook et al. (2015) for example, found little evidence of natural recruitment of golden perch in Billabong Creek, New South Wales, and stocking of golden perch fingerlings over successive years resulted in a four-fold increase in catch per unit effort (CPUE). Conversely, Davies et al. (1988) identified improved brown trout numbers and biomass followed stocking cessation in a Tasmanian river system. Optimal spawning conditions and high juvenile survival may have contributed to the sustainability of this fishery (Davies et al. 1988). Whether Murray cod and golden perch
populations would respond in a similar manner following removal of re-stocking programs is an important hypothesis warranting further investigation.

Riverine stocking reduction or cessation may have minimal effect because wild recruitment in rivers was high. However, river stocking may be important for management strategies designed to maintain sustainability and improve fishery quality. For example, stocking of fish into MDB rivers could assist recovery from fish kills (e.g. Gilligan 2005; Koehn 2005), to enhance recreational fisheries (see Gillanders et al. 2006), to re-introduce fish into degraded populations (e.g. Cadwallader and Gooley 1984; Crook et al. 2010b), or to overcome recruitment bottlenecks (Crook et al. 2010a). Quantifying the proportion of stocked fish to these populations is vital to measure the relative success of such recovery and enhancement programs. Such efforts are hindered when stocked fish are unable to be accurately identified within a population.

The current study was affected by the release of unmarked hatchery-reared fish in the Murray and Murrumbidgee rivers, and Burrinjuck Dam. As such, the proportion of stocked fish in these populations may be underrepresented. Although the numbers of unmarked hatchery-reared fish released were mostly low in comparison to total fish released, it is an important limitation to acknowledge, and strengthens the requirement to identify all stocked fish, whether by chemical marking or other means. To initiate a large-scale chemical marking study and gather enough data to make conclusive assessments would take several years, and even then it may deliver only information relating to a comparatively young portion of the population. Marking of all hatchery-origin fish, or the ability to discriminate stocked fish using alternative techniques such as genetic identification (e.g. Denson et al. 2012), would allow identification of natural recruitment in a population allowing adaptive decision making for fishery managers. Such programs are already in place for salmonid fisheries in North America (see Noakes et al. 2000; Rawding et al. 2012). Information collected from these programs identified that hatchery fish had replaced wild fish, with management intervention required to protect wild stocks (Noakes et al. 2000). A further limitation to acknowledge is that the quantification of marked fish did not separate among replacement and additive effects. For example, where a very high contribution of stocked fish was found, this could mean that natural recruitment was low, or that natural recruits were replaced by fitter stocked fishes. Future studies seeking to assess the effectiveness of stocking
should consider before-after-control-impact designs to clarify the interactions between stocked and naturally recruited fish. (e.g. Hühn et al. 2014).

We found marked Murray cod length-weight relationships were significantly steeper (i.e. higher weight at a given length) than unmarked fish, but whether this difference manifests into distinct biological outcomes is unknown. Should length-weight relationship be a key aspect to juvenile Murray cod survival, hatchery-reared fish may have an advantage over wild fish. Hatchery-reared Murray cod and golden perch are generally released as fingerlings (40-60 mm TL; Ingram et al. 2011), at which length they are past the high mortality associated with larval stages. Accordingly, fish stocked in impoundments may have an advantage over wild recruits, or it could be that impoundments offer better conditions for stocked fish survival. Adverse stocking effects may also be present, but less apparent, in riverine populations, with additional research required to evaluate the relationship between stocked and wild fish in both waterbody types.

Murray cod and golden perch have previously been aged to 48 and 26 years respectively (Rowland 1998a; Stuart 2006). Long term calcein mark retention is therefore important to allow assessment of the proportion of stocked fish for studies that collect data over many years, and also for future studies that aim to retrospectively assess the contribution of stocked fish in a particular population. Continued marking of stocked Murray cod and golden perch would enable this. However, should marks degrade over time; ambiguity in the ability to detect stocked and wild fish is introduced. Future studies using chemical marks should therefore validate longevity, so that compensation can be made for detection losses across the study duration. Such knowledge is vital for studies that span many years. Crook et al. (2009) detected golden perch calcein marks in otoliths over two years from the initial marking date, whereas Baumgartner et al. (2012) identified externally visible calcein marks in Murray cod for 57 days post marking. This study extends the known longevity of calcein marks for Murray cod and golden perch otoliths to 3 years 10 months. External calcein marks degrade with exposure to ultraviolet light (Hill and Quesada 2010), but have been detected for up to 1 year 7 months in golden perch (Crook et al. 2012), and for up to three years in salmonids (Mohler 2003; Negus and Tureson 2004). However, despite degradation of external marks, calcein can still be detected within internal bony structures (Honeyfield et al.
2008). For example, calcein marks have been detected in killifish, *Heterandria formosa*, bony structures for up to 6 years (Leips et al. 2001). Calcein marks in golden perch otoliths in the current study remained clearly detectable for 3 years 10 months post-marking, indicating that such marks in Murray cod and golden perch otoliths should persist well beyond the study duration. An alternative to destructive sampling for fish ageing and mark detection includes sectioned fish spines (e.g. Koenigs et al. 2015). Dorsal spine sections in eastern cod revealed no consistent annuli (Butler and Rowland 2008). However mark detection and age from Murray cod and golden perch spines should be further examined as this could potentially enable future studies to minimise lethal sampling of fish for otolith collection.

Our research provides data on the contribution of marked Murray cod and golden perch in riverine and impounded waters. Existing hatchery practices were also tested through comparison of condition between marked and unmarked fish. Marked golden perch have similar condition to unmarked fish, which is important for optimal stocking effectiveness; however marked Murray cod may have advantage over unmarked fish. Identification of variation in the proportion of marked and unmarked fish stresses the need for development of specific stocking strategies for each waterbody. Stocking programs are vital to support impoundment fisheries where natural recruitment was low, but may be of lesser importance to river fisheries where natural recruitment was high. This study suggests that marking of all hatchery-reared fish, whether by chemical marking or using non-destructive genetic techniques to distinguish stocked from wild fish (e.g. Denson et al. 2012), provides critical information for the evaluation of fish stocking outcomes, and for the development of future stocking strategies that can be adapted to suit individual water bodies. Improving the understanding of stocked fish survival and contribution to a population are important for fishery managers to make informed decisions regarding the allocation of hatchery-reared fish offering maximum benefit from a finite resource.
Chapter 7
General discussion

7.1 Synthesis

Effective fishery management requires relevant, accurate and up-to-date information to inform harvest and enhancement strategies with objectives to sustain and recover recreational fisheries. Decisions based on political or popular beliefs, or using old or inappropriate data, may undermine these objectives, and possibly lead to population decline. The absence of scientific information does not necessarily lead to species ongoing decline, but understanding vital information such as; length- and age-at-maturity; recreational effort, catch and harvest; and effectiveness of stocking; can assist management decisions (e.g. Agostinho et al. 2010; Richards and Rago 1999; Trippel 1995). Some of the key information sources for management of the Murray cod and golden perch fishery are not known. So therefore the aim of this thesis was to collect current, sound data to assist the management decisions regarding recreational fisheries for Murray cod and golden perch, and to contribute knowledge that can be applied to a range of global fisheries where decline is evident, and recovery efforts are being undertaken.

The concept where sound science informs management strategies to drive change within a fishery has been used for other recreationally important species, such as striped bass and snapper, Pagrus auratus (Christensen and Jackson 2014; Richards and Rago 1999). For example, striped bass fisheries on the mid-Atlantic USA coast started to decline in the late 1900s through until the mid-1980s. (Boreman and Austin 1985; Raney 1952; Stevens et al. 1985). Recovery efforts centred on seasonal spawning closures and increasing the MLL above age-at-maturity to allow prior spawning opportunity (Berlinsky et al. 1995; Richards and Rago 1999). In addition, hatchery-reared fish were stocked to supplement natural spawning and recruitment (Secor and Houde 1998). The strategy was successful, and by 1995, the species was thought to be recovered (Richards and Rago 1999). The snapper fishery in Shark Bay, Western Australia was on the brink of collapse in 1995. Research to quantify recreational exploitation and spawning behaviours was used to inform the setting of seasonal closures and a harvest slot limit.
The management intervention was successful and by 2011 the fishery was considered stable, and in 2012 daily bag limits were increased (Christensen and Jackson 2014).

Re-building of the striped bass and snapper fisheries hinged on re-establishment of spawning stocks and broadening of age structures, which occurred after strict management intervention. Similarity can be drawn to Murray cod and golden perch fisheries where historic declines have stimulated management responses such as the imposition of harvest regulations (Rowland 2005). However, inadequate, limited or no data to support decision making may be restricting recovery effectiveness. The new knowledge provided in this study on; quantification of Murrumbidgee River and Lake Mulwala recreational fisheries (Chapters 3 and 4); identification of maturation onset to inform size limits (Chapter 5); and assessment of stocking effectiveness (Chapter 6); will similarly inform decisions that influence the recovery and ongoing sustainability of Murray cod and golden perch fisheries. The key points obtained in this study from fisher surveys, biological data, and experiments highlighting links between methodology and management information that are important to increase the sustainability and quality of Murray cod and golden perch recreational fisheries are provided in Table 19.
Table 19: Summary of key points obtained from fishery dependant and independent data to link methodology and management information

<table>
<thead>
<tr>
<th>FISHERY DEPENDANT DATA</th>
<th>FISHERY INDEPENDENT DATA</th>
</tr>
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<tbody>
<tr>
<td><strong>MURRUMBIDGEE RIVER</strong></td>
<td><strong>LAKE MULWALA</strong></td>
</tr>
<tr>
<td>• Catch rate indicative this is a productive Murray cod fishery and poor golden perch fishery</td>
<td>• Boat and shore fisheries are different Murray cod; accounted for most fishing effort; and had most harvest and discard of Murray cod</td>
</tr>
<tr>
<td>• Most Murray cod caught were below the MLL</td>
<td>• Shore-based more diverse, multi-species fishery</td>
</tr>
<tr>
<td>• Reduced effectiveness of closed season to protect spawning Murray cod</td>
<td>• Most Murray cod were released as they were undersized</td>
</tr>
<tr>
<td>• High Murray cod release rate could add fishing mortality that exceeds harvest</td>
<td>• Assessment of standardised parameters revealed variability amongst fisheries for Murray cod in different systems</td>
</tr>
</tbody>
</table>

**FURTHER RESEARCH**

• Assess the impact of catch-and-release fishing on spawning Murray cod
• Research on the source of Murray Cod recruitment to determine whether populations are self-sustaining or augmented
• Repeat survey to assess change within the Murrumbidgee River fishery

• Assess the impact of catch-and-release fishing on spawning Murray cod
• Research on the source of Murray Cod recruitment to determine whether populations are self-sustaining or augmented
• Marks all hatchery-reared fish prior to release

**RECOMMENDATIONS**

• Consider fishing related mortality caused by catch-and-release fishing
• Provision of fish passage and stocking to increase golden perch numbers
• Stakeholder engagement program to reduce fishing impact to Murray cod during closed season
• Future surveys should increase sampling effort to maximise sample size
• System-specific regulations may be more effective at reducing the risk of overfishing and meeting fishing quality objectives

• Repeat survey to assess change within the Lake Mulwala fishery
• Continued monitoring to establish a long-term dataset to evaluate changes in maturation onset and the effects of any size limit change
• Assessment of stocking effectiveness in new waterbodies
• Investigate non-destructive methods to identify stocked from wild fish

• Eliminate the release of un-marked hatchery-reared fish in research areas
• Investigate whether hatchery-reared Murray cod growth difference results in biological benefit
The fisher survey data collected in Chapter 3, established a starting point to assess long-term trends and whether management interventions are effectively sustaining and rehabilitating populations. The survey results identified equal effort between open and closed seasons, thus Murray cod catch was inevitable and could influence spawning success through nest abandonment (Butler and Rowland 2009), egg re-absorption following handling prior to spawning (Rowland 1988), and fishing mortality related to catch-and-release fishing (Douglas et al. 2010; Hall et al. 2012). Seasonal closures were vital to the success of recovery efforts for snapper fisheries in Shark Bay, Western Australia (Christensen and Jackson 2014) and continued fishing effort during the closed season for Murray cod will likely reduce the effectiveness of this regulation and therefore hinder recovery efforts. Further research to quantify the effects of catch-and-release on spawning Murray cod is required because of such continued fishing effort during the closed season. In addition, angler engagement programs, where voluntary institutions and behaviours are encouraged as alternatives to formal regulations, can be used to promote alternate species and best-practice catch-and-release (Cooke et al. 2013). Such a program may reduce the impact to Murray cod during the closed season.

Understanding the magnitude and effects of catch-and-release on fish populations are important as fisheries management often relies on regulations requiring mandatory discards (i.e. closed seasons, size and bag limits; Arlinghaus et al. 2007). The results for Chapters 3 and 4 identified that most releases of Murray cod were because fish were below the MLL. Such high incidence of catch-and-release fishing could add fishing-related mortality that could exceed harvest. Most recreational fishing surveys collect information regarding the fish kept by anglers. However these results indicate that surveys in recreational fisheries where the prevalence of catch-and-release fishing is high, (such as largemouth bass fisheries in North America; Wilde 1997), should also consider fishing-related mortality, and its likely contribution to indirect harvest. The absence of such information may compromise recovery efforts in what can be unseen influences on a fishery (Post et al. 2002).

Golden perch do not feature in recreational fisheries in the Murrumbidgee River (Chapter 3) or Lake Mulwala (Chapter 4). Golden perch are a highly migratory species (Koehn and Nicol 2016), with barriers to movement contributing to decreased abundance in affected river reaches (Baumgartner et al. 2014). The provision of fish
passage has been successful in reconnecting populations of white sturgeon in North America, allowing fish to repopulate areas that were previously inaccessible (Jager et al. 2016). Similarly, construction of fishways across barriers along the Murray River has allowed movement of MDB fish species, including Murray cod and golden perch (Barrett 2008; Barrett and Mallen-Cooper 2006). However, existing barriers along the Murrumbidgee River restrict movement of native fish species (Baumgartner 2007). The provision of fish passage along this system would likely improve the ability of golden perch (and other species) to recolonise areas where their abundance may be low.

Surveys of recreational fisheries for Murray cod revealed variability amongst populations in Lake Mulwala, the Murrumbidgee River, and the Murray River. However, a general trend was identified where boat-based fisheries exhibited more desirable aspects of angling quality (e.g. larger fish, higher harvest and discard) than for shore-based fisheries within each waterbody (Chapters 3 and 4; Brown 2010; Forbes et al. 2015c). The recreational fishing data collected in Chapters 3 and 4 (i.e. effort, catch, harvest, and their rates, and size structure) should be further developed using stock assessment tools such as empirical yield/abundance models and size-based indicators (Lorenzen et al. 2016). The development of empirical yield/abundance models for Murray cod and golden perch populations would allow understanding of fisheries yield or fish abundance in relation to fishing effort (Lorenzen et al. 2006). Whereas data on size composition of harvested fish can be developed using theoretical models to provide qualitative and quantitative information on exploitation levels (Lorenzen et al. 2016).

Recreational harvest and catch-and-release identified in Chapters 3 and 4 are governed in many fisheries by closed seasons and length-based restrictions such as MLLs or harvest slot limits (Allen et al. 2013; Allen et al. 2009; Gwinn et al. 2015). For example, MLLs for Murray cod were implemented to allow fish the opportunity to spawn before attaining the minimum harvest length and have increased three times in the past decade; but no studies have assessed relationships between fish size and reproductive maturity to inform management decisions regarding size limits. Therefore Chapter 5 quantified variation in Murray cod and golden perch length- and age-at-maturity, and growth parameters, within several MDB riverine and impounded systems, and investigated whether length is a suitable surrogate for age-at-maturity. Chapter 5 also explored the
validity of previous assumptions that past and present size limits provide adequate protection for Murray cod and golden perch spawning populations.

In Chapter 5, I found that Murray cod and golden perch had substantial differences in growth rate and onset of reproductive maturation between impoundments and rivers. Impoundment fish grew faster and larger than river fish. Murray cod length-at-maturity was lower in rivers than impoundments, whereas golden perch matured at a greater length in rivers than impoundments. In addition, Murray cod and golden perch age variation at existing MLLs was greater for populations in rivers than impoundments. Length-based restrictions are applied uniformly to riverine and impounded waters. The differences in growth rate and the onset of reproductive maturation for Murray cod and golden perch between these waterbody types suggest that system-specific regulations may be more effective at reducing the risk of overfishing and meeting fishing quality objectives. Such system-specific regulations have been used to govern largemouth bass recreational fisheries in North America, where despite a state-wide 356 mm MLL, a stringent 406–609 mm protected slot limit was used to successfully create a trophy fishery in an impoundment (Chen et al. 2003). Although uniform regulations would simplify enforcement, a diverse management regime may benefit conservation efforts for Murray cod and golden perch populations as these species exhibit variability in maturation, growth and relative fishing quality among waterbodies (Chapters 4 and 5).

In Chapter 5, I confirmed that the increase in Murray cod MLL from 500 mm to 600 mm provided greater protection for spawning populations across both impounded and riverine waters. However, the Murray cod model provided in Chapter 5, illustrated that smaller length limits would be required in riverine compared to impounded systems and that a constant MLL across all populations could lead to overfishing in some systems if the MLL (or lower harvest slot bound) was at or below 600 mm. A 500 mm MLL offered little or no prior spawning opportunity in rivers or impoundments before harvest eligibility. In contrast, a 600 mm MLL is greater than Murray cod age- and length-at-maturity, which allows up to two seasons of possible spawning prior to attainment of harvest length, which satisfies the management goal of allowing spawning opportunity prior to harvest (Rowland 2005). The 300 mm (New South Wales) and 330 mm (South Australia) golden perch MLLs offer little protection for the spawning population. The 330 mm South Australian MLL is perceived as effective in protecting golden perch
reproductive potential in that state (Ye 2004), but this length was below LM50 for riverine populations in NSW. The situation where LM50 was below MLL potentially exposes these fish to recruitment overfishing where harvest exceeds the ability of the population to replace itself (Allen et al. 2013; Coggins et al. 2007), and could contribute to population decline should recreational fishing be a major stressor in these fisheries.

The low prevalence of anglers targeting, harvesting or discarding golden perch in Lake Mulwala and the Murrumbidgee River (Chapters 3 and 4) suggests that these fish are not readily available to the recreational fishery in these areas, and that their absence may not be directly attributable to current excessive recreational fishing, but by other factors such as barriers to fish passage (Baumgartner et al. 2014). Regardless, to ensure that overfishing is not occurring in these and other fisheries, and contributing to further decline, the MLL for golden perch should be set above LM50.

Murray cod feature strongly in the Lake Mulwala and the Murrumbidgee River recreational fisheries with size limits having a pronounced effect on harvest and discard in these systems, as most cod released were under the MLL (Chapters 3 and 4). Murray cod have been shown to truncate at the MLL in other waterbodies (Nicol et al. 2004a), and the introduction of a 550-750 mm harvest slot was designed to provide greater protection for this species and to ensure long term sustainability of cod stocks (New South Wales Department of Primary Industries 2013b; New South Wales Department of Primary Industries 2015). Harvest slots were successfully used to assist in the recovery of the snapper fishery in Shark Bay, Western Australia (Christensen and Jackson 2014), and should fisheries for Murray cod respond in a similar manner to the new regulation, recovery efforts for this species will be enhanced. However, monitoring of changes in size structure and maturation onset within key Murray cod fisheries is vital to ensure that the new regulation is achieving sustainability and recovery goals.

Another major tool used by fishery managers to sustain and enhance recreational fisheries is stocking using hatchery-reared fish (Cowx 1994). The need for further research on the outcomes of stocking was identified in a study by Crook et al. (2010b), that recommended further assessments be conducted in a variety of systems and that investigations be expanded to species other than golden perch. Therefore Chapter 6; quantified the proportion of stocked Murray cod and golden perch in several riverine
and impounded populations; evaluated the necessity of stocking for fishery augmentation; and compared length-weight relationships and condition of stocked (marked) fish with unmarked fish.

In Chapter 6, I found that the relatively low contributions of marked Murray cod and golden perch in rivers was because these populations are primarily self-supporting through natural recruitment, and/or survival of marked fish is low into these waterbodies. In contrast, marked fish formed a larger proportion of the population within impoundments. Very few unmarked golden perch were recovered from Copeton Dam which supports observations that this species is a river spawning specialist (Harris and Gehrke 1994; King et al. 2003; King et al. 2009). Few unmarked Murray cod were found in Copeton Dam, suggesting that natural recruitment of this species was limited. Murray cod are known to spawn in standing waters, and larvae have been collected from impounded waters (Koehn and Harrington 2005; Rowland 1983b), but few unmarked fish were evident in the impoundments used in this study. This observation suggests that impounded populations are more reliant on stocking than populations in rivers and may be considered a put-and-take fishery. Put-and-take fisheries are stocked to provide catchable-sized fish for rapid exploitation by anglers and do not take into account sustainability through natural recruitment (Cowx 1994). However, density-dependent processes resulting from fish stocking can have a significant impact on recreational put-and-take fishery performance. For example, stocking fewer fish in a put-and-take chinook salmon fishery in Victoria, Australia was associated with decreased catch rates, but heavier fish. Conversely, stocking more fish was associated with increased catch rates, but lighter fish (Hunt et al. 2014).

The variable contributions of marked fish demonstrate that adaptive, location-specific fishery enhancement strategies are required to maintain recreational fisheries. For example, in Chapter 6, the results suggest that in rivers with high levels of natural recruitment, stocking may be inefficient and potentially harmful. However, in some systems the conditions required for successful recruitment may not occur because of river regulation or drought conditions, and stocking may be effective in increasing the abundance of stocked species. For example, Crook et al. (2015) found little evidence of natural recruitment of golden perch in Billabong Creek, New South Wales, and stocking
of golden perch fingerlings over successive years resulted in a four-fold increase in catch-per-unit-effort.

The different contributions of stocked fish among waterbodies demonstrate that flexible fishery enhancement strategies are required to protect wild populations and maintain non-breeding recreational fisheries. Stocking of rivers has a role to play in management strategies designed to maintain sustainability and improve fishery quality. For example, stocking of fish into MDB rivers is thought to assist recovery from fish kills (e.g. Gilligan 2005; Koehn 2005), to enhance recreational fisheries (see Gillanders et al. 2006), and to re-introduce fish into degraded populations (e.g. Cadwallader and Gooley 1984; Crook et al. 2010b). Stocking under these circumstances is highly justified. However, quantifying the proportion of stocked fish to these populations is vital to measure the relative success of such recovery and enhancement programs. Marking of all hatchery-origin fish, or the ability to discriminate stocked fish using alternative techniques such as genetic identification (e.g. Denson et al. 2012), would allow identification of stocked fish contributions in a population allowing adaptive decision making for fishery managers. Such programs are already in place for salmonid fisheries in North America (Noakes et al. 2000; Rawding et al. 2012). Information collected from these programs identified that hatchery fish had replaced wild fish, with management intervention required to protect wild stocks (Noakes et al. 2000). Continued monitoring of riverine Murray cod and golden perch populations is therefore important to ensure sustainability of wild populations, and that wild fish are not replaced by stocked fish (whether by direct stocking, or indirectly through the progeny of previously stocked fish). Broad-scale marking/identification of stocked fish (i.e. MDB wide) is not available for Murray cod and golden perch, which limits the information available to fishery managers on the relative success of stocking programs. The ability to identify all stocked Murray cod and golden perch would extend assessments of stocking to new waterbodies and should be considered a priority.

The length-weight relationship for hatchery-reared Murray cod was greater than wild fish, but differences in golden perch condition and length-weight relationship between marked and un-marked fish were not evident (Chapter 6). Thus existing hatchery practices provide golden perch with similar condition and length-weight relationship characteristics to wild fish, but hatchery-reared Murray cod may have an advantage over
wild fish. These findings contrast to hatchery-reared salmonids, which generally do not perform as well on release compared to wild fish, with reduced survival and ultimately lower contribution to the spawning population (McDonald et al. 1998). Though stocked Murray cod appear to have an advantage, the significant difference was marginal, and may not lead to detrimental biological outcomes. Such negative outcomes can include replacement of wild fish with fish of hatchery origin as evident in North American populations of lake trout where stocking replaced indigenous populations with non-native stocks (Evans and Willox 1991). Should length-weight relationship be a key aspect to juvenile Murray cod survival, hatchery-reared fish may have an advantage over wild fish. Further research is required to investigate whether such a difference results in biological benefit.

Proven fishery research methods and techniques (i.e. creel survey analysis, maturation onset assessment and calcein marking of hatchery-reared fish) were used to answer chapter-specific research questions to inform strategies to recover and sustain Murray cod and golden perch recreational fisheries. The continued collection of relevant, accurate scientific data on Murray cod and golden perch fisheries will assist fishery managers make informed decisions to support the recovery.

7.2 Management recommendations

Through this research, consideration was given to management recommendations from my findings for Murray cod and golden perch recreational fisheries. These recommendations are summarised in Table 19, and are based on evidence from each of the research chapters.

- **Consideration of fishing related mortality caused by catch-and-release fishing**
  Application of Murray cod fishing mortality rates (Douglas et al. 2010; Hall et al. 2012) to discard estimates from the Murrumbidgee River demonstrated that catch-and-release fishing could add additional mortality that exceeds the harvest estimates (Chapter 3). Post-release mortality by recreational fishers has been highlighted as a global concern with analogies drawn to by-catch discards in commercial fisheries (Cooke and Cowx 2004). Accordingly, it is important for fishery managers to consider this potential additional fishing-related harvest, particularly for fisheries with high release rates of
target species such as seen in the Murrumbidgee River and Lake Mulwala (Chapters 3 and 4).

- **Provision of fish passage and stocking to increase golden perch numbers**
  The fishery for golden perch in the Murrumbidgee River fishery was poor, with an annual shore-based catch rate of a fish every 56 hours and a fish every 500 hours for boat-based fishers (Chapter 3). Opening of migration pathways for golden perch in the Murrumbidgee River would allow re-population of degraded populations (Baumgartner et al. 2014). However, building fishways to provide such passage is expensive (Barrett and Mallen-Cooper 2006), and a more immediate action to re-build populations of this species would be to stock river reaches that have low numbers of golden perch with hatchery-reared fish (e.g. the Murrumbidgee River reach studied in Chapter 3).

- **Implement a stakeholder engagement program to reduce fishing impact to Murray cod during the closed season**
  The effectiveness of the closed season to protect spawning Murray cod was reduced because of high levels of fishing effort and catch of Murray cod during this period (Chapter 3). Targeted education programs on fishing gear and techniques that minimise Murray cod by-catch during the closed season are required to increase protection for spawning fish. Such programs should seek to promote the use of gear such as circle hooks that reduce deep-hooking of fish and increase post-release survival (Cooke and Suski 2004). In addition, further information should be provided to educate fishers on baits and lures that reduce by-catch of Murray cod when they are not a target species, such as during the closed season. The extension of such programs will benefit Murray cod populations by reducing fishing-related mortality, which can exceed harvest in fisheries with high rates of release (e.g. the Murrumbidgee River and Lake Mulwala; Chapters 3 and 4).
• **Future fisher surveys should increase sampling effort to maximise sample size**

The number of Murray cod measured during the Lake Mulwala survey was relatively low (Chapter 4). More fish would have been measured by having higher levels of daily replication (i.e. more sampling days), and/or increased coverage of access points during the survey. The result of increased sampling effort (and more fish measured) are reductions in variance and increased accuracy of results (Pollock et al. 1994). Therefore, to further improve data quality and quantity, future surveys of Lake Mulwala include more sampling effort than that undertaken in Chapter 4. In this regard, improvements to the survey design used for the Murrumbidgee River (Chapter 3) could also be improved by; (1) increasing the number of sampling days; and (2) providing increased coverage (higher wait times) at all access points.

• **The use of system-specific regulations may be more effective at reducing the risk of overfishing and meeting fishing quality objectives**

In Chapter 5, I identified that Murray cod and golden perch had substantial differences in growth rate and onset of reproductive maturation between impoundments and rivers. Impoundment fish grew faster and larger than river fish. Murray cod length-at-maturity was lower in rivers than impoundments, whereas golden perch matured at a greater length in rivers than impoundments. Regulations governing recreational fisheries for Murray cod and golden perch vary among state-based jurisdictions, however length-based restrictions are applied uniformly to riverine and impounded waters (New South Wales Department of Primary Industries 2014; Primary Industries and Regions South Australia 2012; Queensland Government - Agriculture Fisheries and Forestry 2011; Victorian Department of Environment and Primary Industries 2015). System specific regulations may be more effective at reducing the effects of fishing and to control fishing quality objectives, given the differences in growth rate and maturation exhibited by Murray cod and golden perch between riverine and impounded waters.

• **The MLL for golden perch should be increased to a length above LM50**

Golden perch exhibited a broad range of LM50’s (225–371 mm), which were generally greater than the existing 300 mm MLL (Chapter 5). The situation where LM50 is below MLL potentially exposes these fish to recruitment overfishing where harvest exceeds
the ability of the population to replace itself (Allen et al. 2013; Coggins et al. 2007), and could lead to decline in the population if recreational harvest were a key stressor. Therefore, to reduce the risk of overfishing in golden perch fisheries, the MLL should be increased to a length greater than LM50.

- **All hatchery-reared fish should be marked prior to release**

  Marking of all hatchery-origin fish, or the ability to discriminate stocked fish using alternative techniques such as genetic identification (Denson et al. 2012), would allow rapid identification of natural recruitment in a population allowing adaptive decision making for fishery managers (e.g. Noakes et al. 2000; Rawding et al. 2012). Such information is critical for evaluation of fish stocking outcomes and the development of stocking strategies.

- **Stocking strategies should be waterbody-specific based on the relative contribution of stocked fish**

  The comparatively low proportion of marked Murray cod and golden perch in the riverine study sites suggests that these populations are primarily self-supporting through natural recruitment, and/or survival of marked fish is low into these waterbodies (Chapter 6). In contrast, marked fish formed a larger proportion of the population within impoundments, which suggests that reservoir fisheries are almost entirely dependent on stocking rather than natural recruitment (Chapter 6). The variable contributions of marked fish demonstrate that adaptive, location-specific fishery enhancement strategies are required to maintain recreational fisheries.

- **Eliminate the release of un-marked hatchery-reared fish in research areas**

  The results presented in Chapter 6 were impacted by the release of unmarked hatchery-reared fish in some study sites. Therefore, the proportion of stocked fish in these populations may be underrepresented. Although the numbers of unmarked hatchery-reared fish released were mostly low in comparison to total fish released, it is an important limitation to acknowledge, and strengthens the requirement to identify all stocked fish, whether by chemical marking or other means.
7.3 Further research

Essential information required for effective recreational fishery management includes harvest and catch-and-release (Barwick et al. 2014). Fishery dependent information collected from the Murrumbidgee River and Lake Mulwala addresses a knowledge gap on these popular recreational fisheries by quantifying previously non-existent parameters that included effort, catch, harvest and catch-and-release (Chapters 3 and 4). However, only by repeating these surveys can trends over time within each fishery be understood. For example, repetition of the Lake Mulwala survey would allow assessment of the Murray cod harvest slot limit and its effectiveness to increase numbers of large fish available to anglers. Brown (2010) also recommended survey repetition to establish a time-series of statistics on the Murray, Goulburn and Ovens River recreational fisheries that could be used to evaluate future management decisions. As such, future research should repeat the Murrumbidgee River and Lake Mulwala surveys to quantify change. Throughout the MDB there are numerous other important riverine and impounded recreational fisheries that have not been quantified. As such, future research should collect fishery dependant data from these other important recreational fisheries so the interactions between anglers and the fish populations on a broader spatial scale are better understood. These surveys should also collect information that enable differentiation of anglers into different psychological groups and skill levels to ascertain the influence that angler specialisation has on MDB fisheries (Cooke et al. 2015; Johnston et al. 2013).

Murray cod release after capture is predominantly because fish were undersize (Chapters 3 and 4). These undersized fish may be the result of increased spawning and recruitment following the cessation of a decade long drought (Morrongiello et al. 2011), or augmented fish numbers from stocking (Chapter 6). Further research on the recruitment source of Murray cod is therefore required to determine whether local populations are self-sustaining or augmented.

The high incidence of Murray cod catch-and-release identified during the closed season for this species has the potential to negatively impact spawning potential. The closely related eastern cod abandoned nests prior to hatching when males protecting the nest were caught and subsequently released. The male never returned and the eggs were
consumed by predators (Butler and Rowland 2009). If similar behaviour is exhibited by Murray cod, deliberate by-catch of this species (even with 100% catch-and-release), during the closed season would reduce spawning potential. Douglas et al. (2010) and Hall et al. (2012) both demonstrated fishing mortality associated with catch-and-release, which could exceed harvest where catch rates are high (Chapter 3). Given that fishing effort in the study reach was similar between open and closed seasons, Murray cod catch is inevitable and could influence spawning success, through nest abandonment, or increased fishing related mortality. Further research is required to quantify such risk, and to assess the impact of catch-and-release fishing on spawning Murray cod.

Continued population monitoring is important as length-based harvest restrictions have changed several times since the early 1990s, yet no program exists to monitor the success or failure of the revised regulations to achieve management goals. Murray cod length-based restrictions change from MLL to a harvest slot during the course of this study (New South Wales Department of Primary Industries 2015), and should golden perch MLL increase following the recommendation made in section 7.1, continued monitoring of fisheries for these species is imperative to identify changes in population structure and reproductive maturation onset, and the success of such management interventions. The monitoring program should include design aspects that allow collection of fish of all sizes. For example, the confidence intervals surrounding the proportion of immature fish that were calculated in Chapter 5 could have been reduced by including more, smaller fish in the analysis.

Understanding levels of natural recruitment, and identification of stocked fish in a population, are critical for effective fishery management (Barwick et al. 2014). I was able to provide stocked fish proportions for some riverine and impounded populations (Chapter 5), and Crook et al. (2015) provided estimates in the Murrumbidgee River, Edward River and Billabong Creek. However such data does not exist for the majority of rivers and impoundments within the MDB. Without knowledge of stocked fish proportion and the ability to identify stocked from wild fish it is impossible to measure the success of management strategies for recovery and enhancement. Thus, further research should extend stocking effectiveness assessments to more MDB waterbodies, enabling stocking strategies to be designed that are based on measured natural recruitment levels. Such research should also seek to validate the retention of applied
marks using double tagging, so that adjustments can be made any loss of marks over the study duration (Pine et al. 2003). Further assessments should also investigate non-destructive methods to identify stocked from wild fish. The external detection of calcein in Murray cod and golden perch is unreliable (Ingram et al. 2015). The use of other non-destructive techniques to distinguish stocked from wild fish (such as genetics) are established for other species (e.g. Valiquette et al. 2014), and further research is warranted to test whether such methods are suitable for Murray cod and golden perch.

I identified a significant difference in the length-weight relationship between hatchery-reared and wild Murray cod (Chapter 6). Research to ascertain whether this difference manifests as a biological difference between the two groups is important to inform whether hatchery-reared Murray cod have an advantage over wild fish.

7.4 Concluding remarks

Throughout this research, the aim was to contribute new knowledge to sustain and improve Murray cod and golden perch recreational fisheries. The research provided in this thesis provides knowledge that informs management strategies for Murray cod and golden perch recreational fisheries. The combination of fishery-dependant and fishery-independent data collection methods was vital to understanding the interactions between anglers and the fishery, and also how fish adapt in response to management strategies such as stocking and size limits. The isolated use of either fishery-dependant or fishery-independent data collection would limit the insight gained into fisheries for Murray cod and golden perch. For example, fisher surveys illustrated that undersized Murray cod comprise most of the fisheries in the Murrumbidgee River and Lake Mulwala (Chapters 3 and 4), that size limits governing these fisheries could lead to overfishing if the minimum bound were at or below 600 mm (Chapter 5), and that riverine Murray cod are primarily of wild origin (Chapter 6). The information from fisher surveys, maturation/size limit and stocking assessments; together with recommendations for changes to management strategies, fishing regulations and future research that expands the current study, should be implemented to ensure that management goals of sustainability and quality are met.
In this research, I have provided a review of factors that cause population decline, and created new knowledge that will help inform management strategies to mitigate Murray cod and golden perch decline, and enhance recovery. The findings and recommendations of this study are specific to Murray cod and golden perch; however input factors such as stocking, governing factors such as size limits and closed seasons, and outputs such as catch-and-release and harvest that work to shape recreational fisheries, are common to fisheries throughout the world. Ongoing collection of evidence to inform management decisions is vital to provide sustainable recreational fisheries and achieve recovery goals.
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Appendix 1: Equations used for effort, catch and harvest estimations in Chapter 3

A1.1 Basic notation

The following notation is used in defining the various estimation methods:

- $j$ denotes the stratum being considered ($j = 1, \ldots, J$);
- $J$ denotes the total number of strata;
- $i$ denotes the sample day unit within the stratum ($i = 1, \ldots, N_j$);
- $N_j$ is the total population size (all possible sampling days) in stratum $j$;
- $n_j$ is the sample size in stratum $j$;
- $x_{ij}$ denotes the value of the $i$th unit of stratum $j$;
- $\bar{x}_j$ is the sample mean for stratum $j$;
- $s_j^2 = \frac{\sum_{i=1}^{n_j} (x_{ij} - \bar{x}_j)^2}{n_j - 1}$ is the sample variance for stratum $j$.

A1.2 Effort estimation for boat- and shore-based fisheries

Step 1 - The progressive counts of boat- and shore-based fishers were calculated separately to estimate effort for each day of sampling.

$$\hat{e}_i = P_i \times T$$  \hspace{1cm} (A1.1)

where $\hat{e}_i$ is the estimate of fishing effort for the $i$th sample day; $P_i$ is the progressive count for the $i$th sample day (the number of fishers counted was used for both boat and shore-based fisheries); and $T$ is the length of the fishing day (we used the mean day length period in hours).

Step 2 - The daily effort estimates were then expanded for each day-type stratum type (weekend or weekday) by multiplying the number of possible sample days in each stratum with the mean of the daily estimates of effort.

$$\bar{e}_j = \frac{\sum \hat{e}_{ij}}{n_j}$$  \hspace{1cm} (A1.2)

where $\bar{e}_j$ is the estimated mean daily fishing effort (angler per day) for the $j$th day-type stratum within the season; $\hat{e}_{ij}$ is the estimate of fishing effort for the $i$th sample day in the $j$th day-type stratum within the season; $n_j$ is the number of days sampled in the $j$th day-type stratum within the season.
\[ \hat{E}_j = N_j \times \bar{e}_j \quad \text{(A1.3)} \]

where \( \hat{E}_j \) is the estimate of total effort (angler-hours) for the \( j \) th day-type stratum within the season.

**Step 3** – Calculate the precision of effort estimates by estimating variances and SEs for each stratum.

\[ \text{Var}(\bar{e}_j) = \frac{s_j^2}{n_j} \quad \text{(A1.4)} \]

where \( \text{Var}(\bar{e}_j) \) is the estimated variance of the mean daily fishing effort for the \( j \) th day-type stratum within the season; \( s_j^2 \) is the sample variance of the daily estimates of fishing effort for the \( j \) th day-type stratum within the season; \( n_j \) is the sample size as described for equation (A1.2). The SE of mean daily fishing effort was then estimated as

\[ \text{SE}(\bar{e}_j) = \sqrt{\text{Var}(\bar{e}_j)} \quad \text{(A1.5)} \]

where \( \text{SE}(\bar{e}_j) \) is the estimated standard error of the mean daily fishing effort; and \( \text{Var}(\bar{e}_j) \) is as described for equation (A1.4).

\[ \text{Var}(\hat{E}_j) = N_j^2 \times \text{Var}(\bar{e}_j) \quad \text{(A1.6)} \]

where \( \text{Var}(\hat{E}_j) \) is the estimated variance of total effort for a stratum and was calculated separately for each day-type (weekday or weekend). Finally, the SE of total effort was calculated,

\[ \text{SE}(\hat{E}_j) = \sqrt{\text{Var}(\hat{E}_j)} \quad \text{(A1.7)} \]

where \( \text{SE}(\hat{E}_j) \) is the estimated SE of total effort for a stratum; and \( \text{Var}(\hat{E}_j) \) is as described for equation (A1.5).

**Step 4** – Calculate total fishing effort separately for boat- and shore-based fisheries. This was done by adding the effort estimates for the strata to obtain seasonal totals,

\[ \hat{E}_{Tot} = \sum_{j=1}^{J} \hat{E}_j \quad \text{(A1.8)} \]

where \( \hat{E}_{Tot} \) is the total seasonal effort calculated by combining the effort estimates for day-type strata; and \( \hat{E}_j \) is the estimate of total effort for the \( j \) th stratum as defined for equation (A1.3).
Step 5 - The precision of effort estimates obtained by adding stratum totals was calculated by summing the estimated variances for all strata and calculating the SE,

$$Var(\hat{E}_{Tot}) = \sum_{j=1}^{J} Var(\hat{E}_j)$$  \hfill (A1.9)

where $$Var(\hat{E}_{Tot})$$ is the estimated total seasonal variance, calculated by combining the estimated effort variances for day-type strata. The SE was determined as

$$SE(\hat{E}_{Tot}) = \sqrt{Var(\hat{E}_{Tot})}$$  \hfill (A1.10)

where $$SE(\hat{E}_{Tot})$$ is the estimated SE for seasonal effort totals when adding day-type strata; and $$Var(\hat{E}_{Tot})$$ is as described for equation (A1.9).

**A1.3 Harvest rate estimator for the boat-based fishery**

The ‘ratio of means’ was used to estimate total harvest as the boat fishery data was based on completed trips (Jones et al. 1995; Pollock et al. 1997). This estimator is the ratio of mean harvest to mean effort; when applied to access point surveys, it has a statistical expectation equal to total harvest divided by total effort for the population of fishers (Pollock et al. 1997),

$$\bar{R}_1 = \frac{\sum_{k=1}^{n} H_k}{\sum_{k=1}^{n} L_k}$$  \hfill (A1.11)

where $$\bar{R}_1$$ is the ratio of means, the estimated daily harvest rate (fish/angler-hour) based on completed trips; $$H_k$$ is the complete harvest for the $$k$$th fishing party; $$L_k$$ is the complete trip length for the $$k$$th fishing party; and $$n$$ is the number of fishing parties in the daily sample.

The seasonal mean daily harvest rates ($$\bar{R}_1$$) for each day-type stratum were calculated. The estimated variances of the mean daily harvest rates ($$Var[\bar{R}_1]$$) were calculated using the general form of (A1.4), and the estimated SEs of the mean daily harvest rates ($$SE[\bar{R}_1]$$) were calculated using the general form of equation (A1.5).

**A1.4 Harvest rate estimator for the shore-based fishery**

The ‘mean of ratios’ was used to estimate total harvest from the shore-based fishery, as data were based on incomplete trips (Hoenig et al. 1997; Jones et al. 1995; Pollock et al. 1997). This estimator is the mean of the individual harvest rates for all fishers interviewed on a given day. When applied to incomplete trip interviews taken by roving
through the fishery, the mean-of-ratios estimator has a statistical expectation of total harvest divided by total effort for the population of fishing units provided short-duration trips are excluded (Hoenig et al. 1997). Given this expectation, the mean-of-ratios estimator ($\hat{R}_2$) used for incomplete trips (with short trip exclusion) provides an equivalent measure of fishing success to the ratio-of-means estimator ($\hat{R}_1$) used on complete trips (Hoenig et al. 1997; Pollock et al. 1997),

$$\hat{R}_2 = \frac{1}{n} \sum_{k=1}^{n} \frac{H_k}{L_k}$$  \hspace{1cm} (A1.12)

where $\hat{R}_2$ is the mean of ratios, the estimated shore-based fishery harvest rate (fish/angler-hour); $H_k$ is the harvest recorded at the time of interview for the incomplete trip for the $k$th fisher; $L_k$ is the length of the incomplete trip at the time of interview for the $k$th fisher; and $n$ is the number of anglers in the daily sample.

The seasonal mean daily harvest rates ($\bar{R}_2$) for each day-type stratum were calculated. The estimated variances of the mean daily harvest rates ($\text{Var}[\hat{R}_2]$) were determined by using the general form of equation (A1.4), and the estimated SEs of the mean daily harvest rates ($\text{SE}[\bar{R}_2]$) were calculated using the general form of equation (A1.5).

**A1.5 Harvest rate estimation for boat- and shore-based fisheries**

The contribution of each day-type stratum to the estimated seasonal harvest rate was apportioned by the relative size of each day-type stratum within the season (Pollock et al. 1994). Essentially greater weighting is given to the weekday stratum as there are more weekdays than weekend days in a season,

$$\bar{R}_{\text{Season}} = \left( \frac{N_{\text{wd}}}{N_{\text{season}}} \times \bar{r}_{\text{wd}} \right) + \left( \frac{N_{\text{we}}}{N_{\text{season}}} \times \bar{r}_{\text{we}} \right)$$  \hspace{1cm} (A1.13)

where $\bar{R}_{\text{Season}}$ is a stratified mean daily harvest rate (fish/angler-hour) for a season (the $\hat{R}_1$ estimator [equation A1.11] was used for the boat-based fishery; the $\hat{R}_2$ estimator [equation A1.12] was used for the shore-based fishery); $N_{\text{wd}}$ is the number of weekdays in the season; $N_{\text{we}}$ is the number of weekend days (includes public holidays) in the season; $N_{\text{season}}$ is the total number of days in the season; $\bar{r}_{\text{wd}}$ is the mean daily harvest rate for the weekday stratum; and $\bar{r}_{\text{we}}$ is the mean daily harvest rate for the weekend day stratum.
Estimates of variance for the stratified mean daily harvest rates were calculated as follows:

\[ \text{Var}(\bar{R}_{\text{season}}) = \left[ \left( \frac{N_{\text{wd}}}{N_{\text{season}}} \right)^2 \times \text{Var}(\bar{r}_{\text{wd}}) \right] + \left[ \left( \frac{N_{\text{we}}}{N_{\text{season}}} \right)^2 \times \text{Var}(\bar{r}_{\text{we}}) \right] \]  

(A1.14)

where \( \text{Var}(\bar{R}_{\text{season}}) \) is the estimated variance of the stratified mean daily harvest rate for the season; \( \text{Var}(\bar{r}_{\text{wd}}) \) is the estimated variance for the mean daily harvest rates for the weekday stratum in the season; and \( \text{Var}(\bar{r}_{\text{we}}) \) is the estimated variance for the mean daily harvest rates for the weekend day stratum in the season. Both \( \text{Var}(\bar{r}_{\text{wd}}) \) and \( \text{Var}(\bar{r}_{\text{we}}) \) can be calculated by using the general form of equation (A1.4).

Estimates of SE for the stratified mean daily harvest rates were obtained as

\[ \text{SE}(\bar{R}_{\text{season}}) = \sqrt{\text{Var}(\bar{R}_{\text{season}})} \]  

(A1.15)

where \( \text{SE}(\bar{R}_{\text{season}}) \) is the SE of the stratified mean daily harvest rate; and \( \text{Var}(\bar{R}_{\text{season}}) \) is as described for equation (A1.14).

### A1.6 Harvest estimation for boat- and shore-based fisheries

To estimate total harvest for boat- and shore-based fisheries, the independent effort estimate was multiplied by the appropriate harvest rate estimate (Hoenig et al. 1997; Pollock et al. 1997; Pollock et al. 1994):

\[ \bar{H}_{\text{Boat}} = \bar{E}_{\text{Boat}} \times \bar{R}_1 \]  

(A1.16)

where \( \bar{H}_{\text{Boat}} \) is the estimate of harvest (numbers of fish) for the boat-based fishery (estimation was for each day-type stratum within the season); \( \bar{E}_{\text{Boat}} \) is the estimate of effort (angler-hours) for the boat-based fishery; and \( \bar{R}_1 \) is the estimate of mean daily harvest rate (fish/angler-hour) as described for equation (A1.11). Likewise for the shore-based fishery,

\[ \bar{H}_{\text{Shore}} = \bar{E}_{\text{Shore}} \times \bar{R}_2 \]  

(A1.17)

where \( \bar{H}_{\text{Shore}} \) is the estimate of harvest (numbers of fish) for the shore-based fishery (estimation was for each day-type stratum within the season); \( \bar{E}_{\text{Shore}} \) is the estimate of effort (angler-hours) for the shore-based fishery; and \( \bar{R}_2 \) is the estimate of mean daily harvest rate (fish/angler-hour) as described for equation (A1.12).

Estimates of variance in harvest for the boat- and shore-based fisheries were calculated, and the base level of estimation was for a day-type stratum within the season:
\[ Var(\hat{H}) = \left[ \hat{E}^2 \times Var(\hat{R}) \right] + \left[ \hat{R}^2 \times Var(\hat{E}) \right] - \left[ Var(\hat{R}) \times Var(\hat{E}) \right] \] (A1.18)

where \( Var(\hat{H}) \) is the estimated variance of the boat-based fishery when using equivalent terms from equation (A1.16), and is the estimated variance for the shore-based fishery when using equivalent terms from equation (A1.17); \( \hat{R} \) is the estimate of mean daily harvest rate for a stratum (the \( \hat{R}_1 \) estimator [equation A1.11] was used for the boat-based fishery; the \( \hat{R}_2 \) estimator [equation A1.12] was used for the shore-based fishery); \( Var(\hat{R}) \) is the estimated variance of the mean daily harvest rate for a stratum and is calculated using the general form of equation (A1.4); \( \hat{E} \) is the total effort for a stratum and is described in equation (A1.3); and \( Var(\hat{E}) \) is the estimated variance of the total effort for a stratum, as described in equation (A1.6).

The estimated SE of the harvest was calculated as

\[ SE(\hat{H}) = \sqrt{Var(\hat{H})} \] (A1.19)

Where \( SE(\hat{H}) \) is the estimated SE of the harvest for the boat-based fishery when using equivalent terms from equation (A1.16), and is the estimated SE of the harvest for the shore-based fishery when using equivalent terms from equation (A1.17); and \( Var(\hat{R}) \) is the estimated variance of the harvest, as described in equation (A1.18).

Harvest estimates for weekday and weekend day strata were combined to obtain seasonal totals. The general forms of the equations used in the effort and harvest estimations and the associated variances and SEs were used for catch estimation.
Appendix 2: Equations used for effort, discard and harvest estimations in Chapter 4

A2.1 Basic notation

The notation used to describe the various estimation methods are as described in Appendix 1, Section A1.1.

A2.2 Effort estimation for boat- and shore-based fisheries

Step 1 - The progressive counts of boat- and shore-based fishers were calculated separately to estimate effort for each day of sampling.

\[ \hat{e}_i = P_i \times T \]  
(A2.1)

where \( \hat{e}_i \) is the estimate of fishing effort for the \( i \)th sample day; \( P_i \) is the progressive count for the \( i \)th sample day (the number of fishing parties counted was used for both boat and shore-based fisheries); and \( T \) is the length of the fishing day (we used the mean day length period in hours).

Step 2 - The daily effort estimates were then expanded for each day-type stratum type (weekend or weekday) by multiplying the number of possible sample days in each stratum with the mean of the daily estimates of effort.

\[ \bar{e}_j = \frac{\sum \hat{e}_{ij}}{n_j} \]  
(A2.2)

where \( \bar{e}_j \) is the estimated mean daily fishing effort (fishing parties per day) for the \( j \)th day-type stratum within the season; \( \hat{e}_{ij} \) is the estimate of fishing effort for the \( i \)th sample day in the \( j \)th day-type stratum within the season; \( n_j \) is the number of days sampled in the \( j \)th day-type stratum within the season.

\[ \hat{E}_j = N_j \times \bar{e}_j \]  
(A2.3)

where \( \hat{E}_j \) is the estimate of total effort (angler-hours) for the \( j \)th day-type stratum within the season.

Step 3 – Calculate the precision of effort estimates by estimating variances and SEs for each stratum.

\[ Var(\bar{e}_j) = \frac{s_j^2}{n_j} \]  
(A2.4)
where $\text{Var}(\bar{e}_j)$ is the estimated variance of the mean daily fishing effort for the $j$ th day-type stratum within the season; $s_j^2$ is the sample variance of the daily estimates of fishing effort for the $j$ th day-type stratum within the season; $n_j$ is the sample size as described for equation (A2.2). The SE of mean daily fishing effort was then estimated as

$$SE(\bar{e}_j) = \sqrt{\text{Var}(\bar{e}_j)}$$  \hspace{1cm} (A2.5)

where $SE(\bar{e}_j)$ is the estimated standard error of the mean daily fishing effort; and $\text{Var}(\bar{e}_j)$ is the estimated variance of the mean daily fishing effort as described for equation (A2.4). The variance for total effort was calculated as

$$\text{Var}(\hat{E}_j) = N_j^2 \times \text{Var}(\bar{e}_j)$$  \hspace{1cm} (A2.6)

where $\text{Var}(\hat{E}_j)$ is the estimated variance of total effort for a stratum, and was calculated separately for each day-type (weekday or weekend). SE was subsequently calculated as

$$SE(\hat{E}_j) = \sqrt{\text{Var}(\hat{E}_j)}$$  \hspace{1cm} (A2.7)

where $SE(\hat{E}_j)$ is the estimated SE of total effort for a stratum; and $\text{Var}(\hat{E}_j)$ is the estimated variance of total effort for a stratum as described for equation (A2.5).

**Step 4** – Calculate total fishing effort separately for boat- and shore-based fisheries. This was done by adding the effort estimates for the strata to obtain seasonal totals,

$$\hat{E}_{Tot} = \sum_{j=1}^{J} \hat{E}_j$$  \hspace{1cm} (A2.8)

where $\hat{E}_{Tot}$ is the total seasonal effort obtained by summing the effort estimates for day-type stratum; and $\hat{E}_j$ is the estimate of total effort for the $j$ th stratum as defined for equation (A2.3).

**Step 5** - The precision of effort estimates obtained by adding stratum totals was calculated by summing the estimated variances for all strata and calculating the SE,

$$\text{Var}(\hat{E}_{Tot}) = \sum_{j=1}^{J} \text{Var}(\hat{E}_j)$$  \hspace{1cm} (A2.9)

where $\text{Var}(\hat{E}_{Tot})$ is the estimated total seasonal variance, calculated by combining the estimated effort variances for day-type strata. The SE was determined as

$$SE(\hat{E}_{Tot}) = \sqrt{\text{Var}(\hat{E}_{Tot})}$$  \hspace{1cm} (A2.10)
where \( SE(\hat{E}_{Tot}) \) is the estimated SE for seasonal effort totals when adding day-type strata; and \( Var(\hat{E}_{Tot}) \) is the estimated total variance as described for equation (A2.9).

**Step 6** - The estimates of fishing effort were converted from party hours to fisher hours. This was done for each of the boat- and shore-based fisheries at the base-level stratum (i.e., day-type within a season) using the equation

\[
\hat{E}_{fisher\_h} = \hat{E}_{party\_h} \times \bar{f}
\]

where \( \hat{E}_{fisher\_h} \) is the new estimate in fisher hours; \( \hat{E}_{party\_h} \) is the old estimate in party hours; and \( \bar{f} \) is the mean number of fishers per party in that stratum.

**Step 7** - The precision of the new estimates of effort in fisher hours were determined by

\[
Var(\hat{E}_{fisher\_h}) = \left[ \hat{E}_{party\_h}^2 \times \text{Var}(\bar{f}) \right] + \left[ \bar{f}^2 \times \text{Var}(\hat{E}_{party\_h}) \right] - \left[ \text{Var}(\bar{f}) \times \text{Var}(\hat{E}_{party\_h}) \right]
\]

where \( \text{Var}(\hat{E}_{fisher\_h}) \) is the estimated variance for the new estimate of effort; \( \text{Var}(\bar{f}) \) was calculated using the general form of equation (A2.4); and \( \text{Var}(\hat{E}_{party\_h}) \) was calculated by using the general form of equation (A2.6). The SEs were determined as

\[
SE(\hat{E}_{fisher\_h}) = \sqrt{\text{Var}(\hat{E}_{fisher\_h})}
\]

where \( SE(\hat{E}_{fisher\_h}) \) is the estimated SE of the new effort estimate; and \( \text{Var}(\hat{E}_{fisher\_h}) \) is described by equation (A2.12).

**A2.3 Harvest rate estimators for boat- and shore-based fisheries**

The ‘ratio of means’ was used to estimate total harvest for the boat-based fishery (Jones et al. 1995; Pollock et al. 1997) as detailed in Appendix 1, Section A1.3. The ‘mean of ratios’ was used to estimate total harvest from the shore-based fishery (Hoenig et al. 1997; Jones et al. 1995; Pollock et al. 1997) as detailed in Appendix 1, Section A1.4.
A2.4 Harvest rate and harvest estimation for boat- and shore-based fisheries

Harvest rate estimations for boat- and shore-based fisheries were determined using the general form of the equations presented in Appendix 1, sections A1.3 to A1.5. Harvest estimations for boat- and shore-based fisheries were determined using the general form of the equations presented in Appendix 1, A1.6.