Impacts of River Regulation, Drought and Exploitation on the Fish in a Degraded Australian Estuary, with Particular Reference to the Life-history of the Sciaenid, *Argyrosomus japonicus*

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Mulloway *Argyrosomus japonicus* (Photograph courtesy of Brian Kowald, South Australian Research and Development Institute - Aquatic Sciences).

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THESIS ABSTRACT

The overall aims of this thesis were to (i) investigate the impacts of river regulation, drought and exploitation on fish assemblages in the lower Murray River system and to identify populations that may be vulnerable (Chapter 2), then for one such population of the sciaenid *Argyrosomus japonicus*, to further investigate the spatial structure of stocks (Chapter 3), the role of environmental flows in the life-history (Chapter 4), and age, growth and reproduction (Chapter 5). Chapter 5 also considers implications for sustainable management of *A. japonicus*.

Annual catch and effort from a small multi-species fishery in the lower Murray River system were stable for 25 years but proportional contribution from each of freshwater, estuarine and adjacent marine habitats, and the species within them, varied (Chapter 2). Fish assemblages generally differed between subsequent 5-year periods and species richness declined steeply in freshwater and estuarine habitats. Species with rapid growth and early maturation (opportunistic strategists), increasingly dominated catches while species with slow growth and late maturation (periodic strategists) declined. Truncated population age structures suggested longevity overfishing of three periodic strategists, and one intermediate strategist species, with a population of the sciaenid *A. japonicus* identified as particularly vulnerable.

Shape and trace element composition of otoliths were evaluated for their ability to delineate stock structure of *A. japonicus* in South Australia (Chapter 3). Low mean concentrations of Ba:Ca in otoliths from the western coast contrasted strongly with high levels in otoliths from the eastern coast, indicating sub-structuring of the stock. Constrained Canonical Analysis of Principal Coordinates (CAP), of elemental concentrations was used to allocate individual otoliths to western, central and eastern coasts with respectively, 100, 100, and 87% success. Otolith shape (elliptical Fourier descriptors) validated these results although classification success was lower with 78, 59 and 70% of otoliths allocated to western, central and eastern coasts respectively. Successful use of otolith shape to differentiate between *A. japonicus* from an aquaculture facility and wild stocks may be useful for determining the success of possible stock enhancement in future.

For the population of *A. japonicus* centred about the Murray River system, age distributions were dominated by the 1993 age class, which comprised 35% and 41% of 2001 and 2002 catches, respectively. In 1993 annual freshwater inflow was 2.5 times the 25 year average. Freshwater inflow explained 28% and 35% of the variability in year class strength in the nearshore marine fishery in 2001 and 2002, respectively. These results suggest that this population is estuarine-

dependent, the estuary provides important refuge for juveniles, and strong year classes, or their absence, may be related to freshwater inflow to this environment.

Validated, otolith-based growth rates for *A. japonicus* in South Australia were the lowest reported world wide and were lower (p ≤ 0.001) on the east coast (L_{inf} = 1,406.18, K = 0.136, t_0 = -0.252, n= 561) than the west coast (L_{inf} = 1,419.83, K = 0.182, t_0 = -0.098, n=157) which also suggested that they are separate populations (Chapter 4). On the east coast, juveniles (2-6 years) utilise habitat within the Murray River estuary and sub-adults/adults (7-25 years) occur exclusively in adjacent nearshore waters. Size at maturity (SAM_{50}) was 811 and 812 mm TL for males and females respectively, and was 57% of L_{inf} in eastern South Australia.

Setting restoration targets for depleted populations of *A. japonicus* and other vulnerable large-bodied native species should initially be based on conserving remnant age structures, and then rebuilding them. The population of *A. japonicus* centred about the Murray River estuary in eastern South Australia (i) has high size/age at maturity, (ii) the lowest reported growth rates in the world, (iii) is dependent on estuarine habitat for juveniles, and also (iv) on flood pulses to establish a strong year class. This population is depleted due to a combination of habitat degradation, loss of environmental flows, and longevity overfishing, and would benefit from management measures that aim to preserve capacity for egg production, allow recruits to enter the adult population, and maintain long-tailed age structures. Measures that would contribute to these aims for *A. japonicus* in eastern South Australia include (i) legal minimum size commensurate with at least SAM_{50} (i.e. > 810 mm TL), (ii) protection of juveniles in estuaries and shallow nearshore waters, (iii) maintenance of environmental flows, and (iv) protection of spawning/feeding aggregations.

In the Murray River estuary, age distributions of adult *A. japonicus* from the nearshore environment may provide an indicator of environmental health because its reproductive success may be dependent on freshwater inflows, and because it is the apex predator in the Murray River estuary. Such species-based indicators could also be developed for golden perch (*Macquaria ambigua*) and black bream (*Acanthopagrus butcheri*) with improved knowledge of stock structure and life-history.

Thesis Preface

Note on Chapter Style

This thesis begins with a General Introduction (Chapter One) to estuaries, and anthropogenic impacts on fish communities and populations. Information available on multi-species estuarine fisheries is highlighted. Here the broad research objectives of the thesis are established. The General Introduction is followed by four research chapters (Chapters Two to Five) addressing the outlined research objectives. Finally, the environmental, biological and management implications of the findings are addressed in the General Discussion (Chapter Six).

Chapters are presented in logical order, rather than in the order in which the work was done. Sampling opportunities for age/size information on the case study species, *Argyrosomus japonicus*, and also golden perch, black bream and greenback flounder were limited due to severe drought conditions that began in the same year as this project.

Each chapter of this thesis that presents original data (Chapters 2-5) has been written in a style suitable for publication in a scientific journal and can be read as a separate study. Chapter 4 was published in an international, peer-reviewed journal in 2008 and is presented as published (see Appendix 1 for permissions from publisher). Chapter 2 has also been submitted to an international, peer-reviewed journal. Each chapter is preceded by a statement outlining the contributions of all co-authors to the research therein. Tables and figures appear within the text and all references cited in this thesis are compiled at the end of the thesis and not at the end of each chapter.

Satellite photograph of lower Murray River region showing Lakes Alexandrina (centre left), Albert (centre right), remnant estuary, nearshore marine environment (surf zone) and Murray River mouth (bottom left). Photograph courtesy of CSIRO.

Chapter 1

General Introduction

Estuaries

River systems around the world are under threat from water abstraction for irrigation, and diversions by dams and levees (Benson 1981; Whitfield and Bruton 1989; Whitfield 1994; Cullen 1999). Because the effects of river regulation on flow regimes may be cumulative (Walker 2006) the lower part of river systems, including freshwater, estuarine and adjacent nearshore marine environments, may be strongly affected.

Estuaries may be defined as partly enclosed coastal water bodies with one or more inflowing rivers or streams, with, or without a free connection to the open sea (Pritchard 1967; Day 1981). This definition can be extended to include adjacent shallow, coastal waters that also experience lowered salinities from river flows (Blaber 1997).

The ecological functioning of estuaries requires adequate, seasonal inflows of freshwater (Benson 1981; Whitfield and Bruton 1989). Seasonal and inter-annual cycles of temperature, rainfall and river flows drive recurrent changes of primary and secondary productivity (Kennish et al. 2008). In particular, flooding events scour accumulated sediments, provide nutrient inputs to increase estuarine phyto- and zooplankton production, create salinity gradients that contribute to habitat complexity and increase habitat and species diversity, and maintain tidal inlets that prevent salinity and temperature extremes and facilitate larval exchange and fish migrations (Schlacher and Wooldridge 1996).

Estuaries have been the focal points for human colonization for centuries and anthropogenic impacts include pollution, altered hydrological regimes, sediment input, and turbidity which result in habitat loss and alteration (Kennish et al. 2008) as well as overfishing. They have been pushed far from their historical baseline of rich, diverse and productive ecosystems (Lotze et al. 2006) and are now among the most degraded habitats on earth (Edgar et al. 2000).

Fish in estuaries

Estuaries often support a greater abundance of fish life than the surrounding ocean (Haedrich and Hall 1975; Whitfield 1993). This may be because of increased food availability from higher primary and secondary productivity (Kennish et al. 2008) and/or protection from predation by stenohaline species or larger individuals of the same species. Understanding the importance of estuaries for fishes has been strongly influenced by two views: first, that estuaries are important nurseries and second, that many species are estuarine dependent (Able 2005).

Estuaries throughout the world are considered as nursery areas for a wide variety of organisms because they often support high densities of juveniles (Day et al. 1989; Blaber et al. 2000). Beck et al. (2001) proposed that a habitat is a nursery for juveniles of a particular species if its contribution per unit area to the production of individuals that recruit to adult populations is greater than production from other habitats in which juveniles occur. Greater contributions to adult recruitment can be provided by any combination of density, growth, survival of juveniles and movement to adult habitat (Beck et al. 2001). These authors identified studies of movement patterns of fishes as a vital missing link in the understanding of nurseries.

Fishes are often assumed to be estuary dependent because they use estuaries as juvenile habitat. However, assessing the estuarine dependence of fish is difficult because many species found in estuaries spawn in marine habitat, and/or juveniles are found in both estuaries and alternative nearshore-marine habitats (Lenanton 1982). Also, there are many examples of populations of marine species that are considered estuarine dependent for only part of their life-history, e.g. early juveniles of *Argyrosomus japonicus* (dusky kob, or mulloway) in South Africa (Griffiths 1996).

Because estuaries have well-defined roles as nursery and feeding areas and as areas of refuge and migration, Elliott et al. (2007) proposed a standardised approach whereby species are assigned to functional groups, or guilds, based on estuarine use, reproductive modes, or feeding modes. For example, the estuarine-use functional-group has a category for 'marine migrant' species which includes all species that spawn at sea but may enter estuaries in large numbers (Table 1). Marine migrants may be further subdivided into 'marine-estuarine opportunists' which occur in large numbers in estuaries but also use alternative nearshore marine habitats, and 'marine- estuarine dependent' species which require sheltered estuarine habitat along coasts with no such alternative habitat.

Understanding the relationship between estuarine habitat and fish is impeded by lack of knowledge of the life-history of many species and important questions remain: (i) is recruitment dependent on environmental conditions within the estuary; (ii) does a population that uses estuarine habitat experience greater mortality in years of low flow; (iii) is estuarine use facultative in some parts of the species range, and obligate in others?

Anthropogenic and climatic impacts on estuaries

Reduced river flows, drought and climate change

Worldwide, reduction of inflows from river regulation has resulted in decreased area of estuarine habitat, increased salinity, siltation and in many cases closure of the mouths of estuaries (Marais 1988). Reviews of the impacts of over-allocation of water resources on estuaries have highlighted the false perception that water is lost when allowed to flow into estuaries and the marine environment (Whitfield and Bruton 1989; Gillanders and Kingsford 2002; Whitfield 2005). Reductions in river flows have a predominantly negative effect on fish species using estuarine and associated nearshore habitats (Drinkwater and Frank 1994; Grimes and Kingsford 1996).

Drought may compound the existing impacts of excessive impoundment and abstraction of water (Whitfield 1999; Baptista et al. 2010). For example, drought affected estuaries on regulated rivers in South Africa showed declining biotic diversity (Whitfield and Bruton 1989; Whitfield and Kok 1992). Drought may also impact fish populations by reduction of suitable juvenile habitat (Whitfield and Kok 1992) and reduction of olfactory cues used by larvae to locate these habitats (Whitfield 1998). Closure of river mouths impedes juvenile movement from marine spawning areas into juvenile habitat in estuaries (Vivier et al. 2010). The impacts of drought may be particularly severe on exploited fish populations (Gillson et al. 2009; Baptista et al. 2010).

Climate change may impact fisheries by affecting primary and secondary production, and thus food supply, growth and survival (Brander 2010). Extreme climatic events, such as droughts and floods, are increasing in frequency worldwide (Mirza 2003) and river discharge into many estuaries has been affected (Gleick 2003). Climate change predictions indicate that this trend of increasing frequency of droughts may continue and that droughts may also become more severe (Hughes 2000; Hughes 2003; Chiew et al. 2009).

Fisheries

Fisheries are an integral part of human activities in the coastal zone and most global fish production, except near major upwelling areas, comes from near the mouths of major rivers (Walters and Ahrens 2009). However, while estuaries are particularly important for fisheries production most estuarine fisheries are either fully exploited or overexploited (Lenanton and Potter 1987; Blaber et al. 2000; Elliott and Hemingway 2002).

Fishing may have a large number of ecosystem effects (Blaber et al. 2000; Elliott and Hemingway 2002) and effects on fish populations include decreases in abundance, changes in population age structure and size composition, and changes in overall species composition (Blaber et al. 2000). Fishing may selectively remove long-lived, slow growing fish in favour of those with high turnover rates (Jennings and Kaiser 1998; Pauly et al. 1998; Pitcher 2001) or cause mortality of species or size classes discarded as bycatch. Overfishing has the potential for loss of species diversity (Greenstreet and Hall 1996), and can lead to imbalances in biotic community structure and ecosystem function (Jennings and Kaiser 1998).

Although several studies have addressed the effects of fishing on fish assemblages in marine environments (see review in Hall 1999), few have focussed on those in temperate estuaries (Blaber et al. 2000; Elliott and Hemingway 2002). One of the main challenges is to separate the effects of fishing, decreased flows, drought, industrial, agricultural and other impacts in these environments (Blaber et al. 2000).

Fish as indicators of ecosystem health

Fish assemblages provide an appropriate indicator for estuarine health because they are directly and indirectly coupled with human impacts and usually include species that represent a variety of trophic levels (Whitfield 1997; Whitfield and Elliott 2002). Fish abundance and species richness and diversity can provide an indication of the health of a particular system (Whitfield 1996; Soto-Galera et al. 1998; Whitfield and Elliott 2002) and as such are an important component of the national environment reporting system for Australian estuaries (Ward et al. 1998).

Populations of single species may also provide an indication of estuarine health. In the northern hemisphere sciaenid species have been suggested as environmental indicators because of their habitat preferences and sensitivity to environmental variables (Burke et al. 1993; Bortone et al. 2005; Eklund 2005). For example the sciaenid spotted seatrout *Cynoscion nebulosus*, has been

suggested as an indicator of estuarine health for estuaries in North America (Bortone 2003; Bortone et al. 2005).

Successful use of biological or ecological indicators based on fishes may assist in the formulation of environmental and ecological quality objectives and in the setting of these standards for aquatic systems (Whitfield and Elliott 2002). Such indicators can be used for setting restoration targets for species when populations have declined (Patton et al. 1998; Maki et al. 2006).

Detailed life-history information is unlikely to be available for all species in a fish assemblage therefore a combination of methods may be required to assess the health of an estuary. This may be achieved by investigating: (i) changes in fish assemblages; and (ii) by developing sufficient knowledge of populations of potential 'indicator species' by investigating life-history and geographic population structure.

The need for research on estuaries and fish assemblages

Fish communities and populations may be impacted by anthropogenic effects such as habitat degradation and exploitation. Such impacts may be particularly severe in end-river environments that are close to human population centres and may be compounded during drought. Drought is predicted to occur with increasing severity and frequency in future. It is important to be able to separate the impacts on fish communities of river flows, drought and exploitation, to identify species and populations that may be vulnerable to local extinction.

For species that have been identified as vulnerable, knowledge of the geographic range of the population is required to identify potential sources of recruitment. Combined with knowledge of demography and life-history, this information may contribute to improved understanding of the environmental influences on recruitment and abundance in these populations. Lack of such information is a significant impediment to sustainable management of fish stocks and natural populations because it is required to assess the risk of local depletion and to set restoration targets for depleted populations.

Study area and fishery

The lower Murray River system comprises the original estuary of the Murray River. Prior to 1940, the estuary covered 660 km^2, and included Lakes Alexandrina and Albert and the region around the Murray River mouth as well as the north and south Coorong lagoons (Walker 2006). In 1940, five barrages were constructed between the lakes and lagoons and habitats above and

below the barrages are now respectively, freshwater and estuarine. The remnant estuary comprising the area around the Murray River mouth and the Coorong lagoons is now only 11% of its original size. Since river irrigation began early in the 20th century, water abstraction has reduced mean annual flow into the estuary by 71% of the long-term average (1895-2006) (CSIRO 2008). The extent and frequency of the natural spring floods has been reduced, and flow at the river mouth now ceases for 40% of the time, compared to only 15% under natural conditions (CSIRO 2008). Since 2001 there has been a major decline in the condition of the lower River Murray system due to the combination of past management practises and prolonged drought (Lester and Fairweather 2009). The mouth of the Murray River has been dredged since 2001 to avoid closure caused by depleted flow volumes and accumulated sediments.

The remnant estuary of the Murray River is the largest estuarine habitat in temperate Australia, is recognised internationally as an important breeding and feeding ground for waterbirds, and supports significant populations of several species of fish and invertebrates. This ecosystem also supports recreational and commercial fisheries for a number of species, with the 'icon' species mulloway (*Argyrosomus japonicus*) being particularly important (Henry and Lyle 2003; Jones 2009, Chapter 4). The commercial fishery is a small (39 licences), multi-species, multi-gear fishery which operates in estuarine, freshwater and adjacent marine habitats and had a production value of $6,500,000 in 2008-09 (Knight and Tsolos 2010).

Thesis aims

The aim of this study was to determine how river regulation, and associated habitat degradation combined with drought and fishing impact on fish assemblages and several populations in habitats at the lower end of a major river system.

The specific objectives were to:

- assess the fish assemblages in several habitats of the lower Murray River and to further investigate the demography and life-history of several fish populations that may be most vulnerable to the effects of river regulation, drought and fishing (Chapter 2);

- assess the impacts of river regulation, drought and fishing on a particularly vulnerable population of *Argyrosomus japonicus* centred about the Murray River system by investigating;

  o the stock structure in South Australian waters (Chapter 3),

  o the impact of variability of freshwater inflows to estuarine habitat on the population centred about the remnant estuary of the Murray River (Chapter 4),

  o age, growth and reproduction in South Australian waters (Chapter 5), and to

  o discuss the implications for sustainable management of *A. japonicus*.

Table 1. Estuarine use functional groups (Elliott et al. 2007).

Functional group	Description
Marine stragglers	Species that spawn at sea and typically enter estuaries only in low numbers, occurring most frequently in the lower reaches where salinities are approximately 35 PSU. These species are often stenohaline and associated with coastal marine waters
Marine Migrants	Species that spawn at sea and often enter estuaries in large numbers, particularly as juveniles. Some of these species are highly euryhaline and move throughout the full length of the estuary. This category can be subdivided into:
	1. Marine estuarine-opportunist: marine species that regularly enter estuaries in substantial numbers, particularly as juveniles, but use, to varying degrees, nearshore marine waters as an alternative habitat;
	2. Marine estuarine dependent: marine species that require sheltered estuarine habitats as juveniles but live along coasts where there are no such habitats and these species are thus dependent on the habitats of that type that are present in estuaries);
Estuarine species	1. Estuarine residents: estuarine species capable of completing their entire life cycle within the estuarine environment;
	2. Estuarine migrants: Estuarine species that have larval stages of their life cycle completed outside the estuary or are also represented by discrete marine or freshwater populations
Anadromous	Species that undergo their greatest growth at sea and which, prior to the attainment of maturity, migrate into rivers where spawning subsequently occurs;
Semi-anadromous	Species whose spawning run from the sea extends only as far as the upper estuary rather than going into freshwater;
Catadromous	Species that spend all of their trophic life in freshwater and which subsequently migrate out to sea to spawn;
Semi-catadromous	Species whose spawning run extends only to estuarine areas rather than the marine environment;
Amphidromous	Species which migrate between the sea and freshwater and in which the migration in neither direction is related to reproduction;
Freshwater migrants	Freshwater migrants (FM) Freshwater species found regularly and in moderate numbers in estuaries and whose distribution can extend beyond the oligohaline sections of these systems;
Freshwater stragglers	Freshwater species found in low numbers in estuaries and whose distribution is usually limited to the low salinity, upper reaches of estuaries.

Clockwise from top left, golden perch, (*Macquaria ambigua*), pipi (*Donax deltoides*), yellow-eye mullet (*Aldrichetta fosteri*), and mulloway (*Argyrosomus japonicus*)

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Statement of authorship

In this chapter, Gregory Ferguson performed all analyses on the data. Timothy Ward, Bronwyn Gillanders, and Mike Geddes provided advice on the manuscript.

For mulloway all collection of samples, laboratory measurements, and age estimates from otolith ring counts was performed by Gregory Ferguson.

Ring counts for golden perch, greenback flounder, and black bream were provided by Dr Qifeng Ye. All allocation of ring counts to age classes and subsequent analyses were done by Greg Ferguson.

Certification that the statement of contribution is accurate.

Signed   ████████   Date   30.9.2010

Gregory J Ferguson (*Candidate*)

Certification that the statement of contribution is accurate and permission is given for the inclusion of the paper in the thesis.

████████

Bronwyn M Gillanders   Date   30.9.2010

Timothy W Ward   Date   30.9.10.

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Chapter 2

Impacts of drought, flow regime and fishing on the fish assemblage in southern Australia's largest temperate estuary

Abstract

We analysed a 25 year time-series of fishery catch and effort data, and age/size information for four large-bodied, native fish species to investigate the hypotheses, that under conditions of reduced freshwater inflows and high fishing pressure; (i) the structure of fish assemblages in the lower River Murray system have changed, (ii) species diversity of fishes has declined, and (iii) population age structures of large-bodied, late-maturing, native fish have been reduced. Annual catch and effort in the lower Murray River system were stable for 25 years but proportional contribution from each of freshwater, estuarine and adjacent marine habitats, and the species within them, varied. Fish assemblages generally differed between` subsequent 5-year periods, with the exception of 1989-93 when floods occurred in 4 out of 5 years, and the following 5-year period (1994-98). Species richness declined steeply over 25 years in freshwater and estuarine habitat and species diversity (Hill's H_2) also declined after 2001 in estuarine habitat. Species with rapid growth and early maturation (opportunistic strategists), increasingly dominated catches while species with slow growth and late maturation (periodic strategists) declined. Truncated population age structures suggested longevity overfishing of three periodic strategist species; golden perch (*Macquaria ambigua*), black bream (*Acanthopagrus butcheri*), mulloway (*Argyrosomus japonicus*), and a fourth species with an intermediate strategy, greenback flounder (*Rhombosolea tapirina*). This has implications for management because loss of older/larger individuals suggests reduced capacity to withstand, or recover from deteriorating environmental conditions as the lower Murray River system experiences historically extreme drought. Management of these species should seek to preserve the remnant population age structures, then to rebuild age structures by allowing recruits to become established in the adult population. We recommend that assessment of multi-species fisheries, in changeable environments, such as occur in estuaries and other end-river environments requires a suite of indicators that address changes in fish assemblages and populations.

Introduction

Many cities are located where rivers meet the sea and estuaries are among the most degraded habitats on earth (Blaber et al. 2000; Edgar et al. 2000). Fish assemblages in freshwater, estuarine and near-shore, marine environments are commonly exposed to the combined impacts of reductions in river flows resulting from droughts and/or excessive water abstraction and intensive fishing pressure (Gillanders and Kingsford 2002; Lotze et al. 2006; Bauchbaum and Powell 2008). Changes to the scale, frequency and seasonality of flows have particularly severe impacts on species that require flood pulses for successful reproduction or larval survival (Drinkwater and Frank 1994; Griffiths 1996; Whitfield and Marais 1999; Chapter 4).

Fishing tends to remove long-lived, slow growing species and to favour those with high turnover rates (Jennings and Kaiser 1998; Pauly et al. 1998; Jackson et al. 2001). The combination of reductions in freshwater inflows and intensive fishing pressure have been implicated as causal factors in changes to the structure of estuarine fish assemblages, including reductions in species diversity and in abundance of long-lived, slow growing species worldwide (Musick et al. 2001; Lotze et al. 2006; Bauchbaum and Powell 2008; Ecoutin et al. 2010).

Large, long-lived fishes with delayed maturation have been classified as periodic strategists (Winemiller and Rose 1992). An extended lifespan allows periodic strategists to survive long periods of sub-optimal environmental conditions and large body size, with correspondingly high fecundity. Periodic strategists also produce large numbers of progeny when environmental conditions are conducive to successful reproduction. Such species that require flood pulses for successful reproduction may be particularly vulnerable to the combined impacts of reduced flows and the impacts of fishing (Griffiths 1996; Rowell et al. 2005; Rowell et al. 2008a; Chapter 4). The removal of large/old, highly fecund individuals from the population through fishing during unfavourable environmental conditions, can reduce the capacity of periodic strategists to produce strong year classes when environmental conditions are favourable (Laë 1995; Jennings and Kaiser 1998; Beamish et al. 2006).

In contrast to periodic strategists, opportunistic strategists are characterised by reduced longevity, small body size, early maturation, small eggs, small clutches and continuous spawning (Winemiller and Rose 1992). This suite of traits increases the potential reproductive capacity (intrinsic rate of population growth) of these species and provides resilience to high rates of adult mortality due to predation or fishing pressure (Jennings and Kaiser 1998). In situations where predator populations are large, the population size of opportunistic strategists is often controlled

by predation (May et al. 1979; Pimm and Hyman 1987; Blaber et al. 2000). When predation levels are reduced by fishing, the population size of opportunistic strategists commonly increases (Jennings and Kaiser 1998; Pauly et al. 1998).

Mixed-species fisheries pose a particular threat to species with low productivity i.e. periodic strategists, because the persistent abundance of highly productive species may continue to stimulate fishing activity, even though the abundance of species with low productivity has been reduced below the level where targeted fishing is economically viable (Musick 1999). Even in mixed-species fisheries, stock assessments have tended to focus on a single species (Gulland 1987; Greenstreet and Hall 1996; Pitcher 2001; Pikitch et al. 2004). Progress towards a multi-species approach to stock assessments requires knowledge of the interacting impacts of by-catch and discarding, habitat degradation and climate change on the sustainability of exploited species along with information on the life-history of individual species (May et al. 1979; Fowler 1999; King and McFarlane 2003). Although progress has been made towards providing information for multi-species stock assessments in estuaries (Gray et al. 2005; Rotherham et al. 2006), such assessments are rare.

There have been several studies of the temporal change in fish assemblages in Australia's temperate estuaries (Loneragan and Potter 1990; Gray et al. 1996; Jackson and Jones 1999). All were based on fishery-independent data and done over a relatively short time period (<9 years). Long-term datasets compiled to monitor commercial fisheries may provide valuable insights into changes in the structure of estuarine fish assemblages and provide a baseline against which to assess the current status of estuarine systems and plan future management (Gulland 1987; Claro et al. 2009). In many cases, fisheries data provide the only information to identify species that have been impacted and to set restoration targets (Patton et al. 1998; Maki et al. 2006).

The remnant Murray River estuary is the largest estuarine system in temperate Australia and supports a fish assemblage that includes species with a wide range of life-history characteristics. Since 2002, freshwater flows into the estuary have been reduced by a severe drought combined with excessive water abstraction (Lester and Fairweather 2009). A small multi-gear, multi-species, commercial fishery targets finfish in freshwater and estuarine habitats and a bivalve species in the adjacent marine habitat. The area is also subject to significant recreational fishing because it is close to the major metropolitan centre of Adelaide (Jones 2009).

Chapter 1 investigates the influence of freshwater inflows and fishing pressure on the fish assemblages of the lower Murray River system. Analyses are based on: (i) modelled freshwater

flow data; (ii) a 25-year time-series of catch and effort data from the commercial fishery; and (iii) age/size information for four large-bodied, native fish species. These data are used to investigate the hypotheses that under conditions of reduced freshwater inflows and high fishing pressure: (i) fish assemblages in the lower River Murray system have changed; (ii) species diversity of fishes has declined; and (iii) population age structures of large-bodied, late-maturing, native fish have been reduced. Implications for stock assessment of multi-species fisheries in end-of-river environments are discussed and options for mitigating the impacts of fishing are identified.

Materials and methods

Study area

The original estuary of the Murray River covered 660 km^2, and included the brackish Lakes Alexandrina and Albert, and the north and south Coorong lagoons (Figure 1, 35º 32' S, 138º 52' E). In 1940, five barrages were constructed between the lakes and lagoons such that habitats above and below the barrages are now respectively, freshwater and estuarine. The remnant estuary comprising the Coorong lagoons is now only 11% of its original size. Since construction of the barrages, water abstraction for irrigation has reduced mean annual flow into the estuary to 20% of natural levels, and the extent and frequency of the natural spring floods has been reduced so that in some years they do not occur (Thomas 1999). In addition, the frequency with which flow ceases at the river mouth has increased from one year in twenty to approximately one year in two (Close 1990). Marine habitat adjacent to the river mouth is high energy ocean beach.
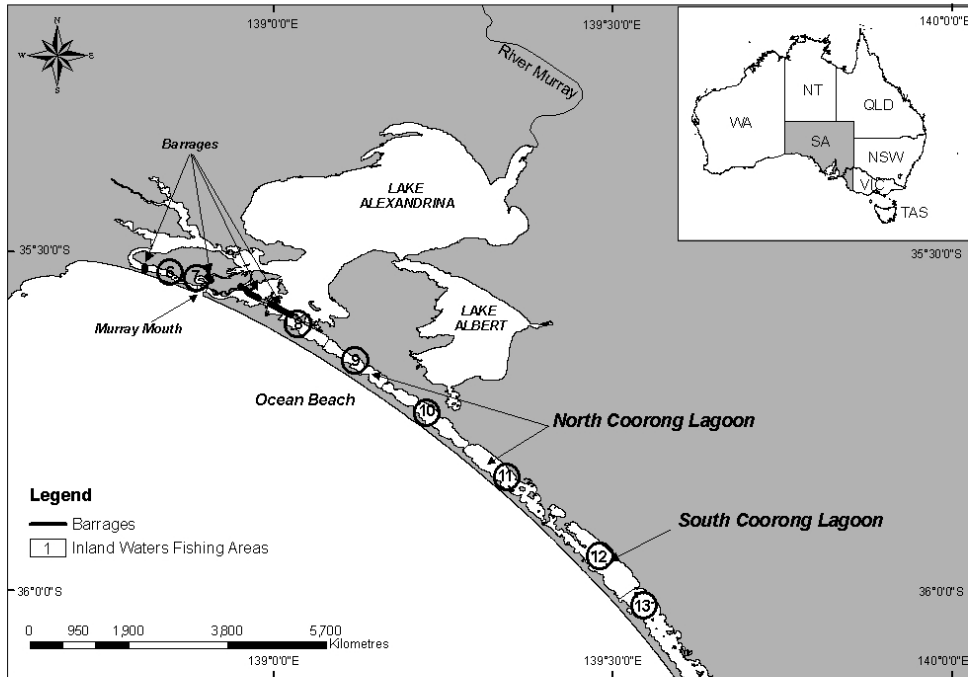
Figure 1. Map of lower River Murray system showing Lakes Alexandrina and Albert (freshwater habitat), remnant estuary of the Murray River comprising the Coorong Lagoons (estuarine habitat), and the marine environment (adjacent marine habitat). (Inset map of Australia showing South Australia and the lower Murray River system, circled numbers are fishery catch and effort reporting areas).

The Lakes and Coorong Fishery

The Lakes and Coorong Fishery (LCF) is a multi-species, multi-method fishery which has had access to resources in freshwater, estuarine and adjacent marine habitats in the lower Murray River system in South Australia since 1846 (Olsen and Evans 1991). Gill nets are the primary gear used in the fishery (>95% of all catches). In freshwater, the LCF uses large mesh gill nets (>115 to ≤150 mm mesh size) to target golden perch (*Macquaria ambigua*), bony bream (*Nematalosa erebi*), and populations of the introduced species; European carp, (*Cyprinus carpio*) and redfin perch (*Perca fluviatilis*). Within the remnant estuary, large mesh gill nets accounted for 65% of effort (25 year average) and were used to target juvenile mulloway (*Argyrosomus japonicus*), black bream (*Acanthopagrus australis*), and greenback flounder (*Rhombosolea taparina*). Small mesh gill nets (>50 to ≤64 mm mesh size), which accounted for the remaining estuarine effort were used to target yellow-eye mullet (*Aldrichetta forsteri*). In the adjacent marine habitat adult mulloway were targeted using large mesh gill nets and pipi (*Donax deltoides*), a bivalve mollusc was collected manually using rakes.

Recent management of the fishery has been through a mixture of input and output controls. Since the early 1980's input controls include limited entry (36 licences in 2006), with gear entitlements and owner-operator provisions applied to licences. Also, since the early 1980's gear restrictions apply to numbers of nets, net dimensions and mesh sizes. Output controls include legal minimum lengths (LML) for most targeted species and the fishery for pipi has been subject to quota management since 2007, but quota has failed to constrain catches (Ferguson 2010a).

Data

Freshwater inflows

In this paper "inflow" refers to freshwater from the Murray River entering the estuary through the barrage system. Estimates of mean monthly freshwater inflow to the Murray River estuary were obtained from the regression-based Murray hydrological model (MSM-Bigmod, Murray-Darling Basin Commission) for financial years beginning 1962 to 2008. This suite of models uses inputs based on inflows (from rainfall and tributaries), storage volumes and outflows (including diversions and losses) and provides estimates of daily flows.

Flows were aggregated into financial years (July to June) because the highest monthly flows occur during late spring (August-November) and summer (December-February). The unit of freshwater flow is gigalitre.year^{-1} (GL.y^{-1}).

Fishery catches

Since 1984, fishers have been required to provide daily catch and effort data, submitted to the South Australian Research and Development Institute on a monthly basis. Fishery data available from 1984 to 2008 were: location (Figure 1); species; catch (kg); gear; and effort (fishing days). Data were aggregated into financial years to capture spring-summer seasonality patterns i.e. 1984 refers to 1984-85. Data were also aggregated into three types of habitat to allow comparison. These were freshwater (Lakes Alexandrina and Albert), estuarine (Coorong lagoons) and adjacent marine (marine waters within 5km either side of the mouth of the Murray River).

Analyses

Temporal trends in catch structure

To determine if the species structure of catches had changed over time the quantitative comparison of catch species compositions among five, arbitrary, 5-year time periods was done using multivariate techniques. Data were aggregated over 5 years to capture periods of high, medium and low freshwater inflows. Variables were first transformed to a Euclidean distance dissimilarity matrix. Canonical Analysis of Principal Coordinates (CAP) was used to ordinate the axes (Anderson and Willis 2003). Permutational multivariate analysis of variance (PERMANOVA) (Anderson et al. 2005) was used to determine differences in species composition between 5-year periods. For all CAP and PERMANOVA tests 4,999 unrestricted random permutations of the raw data were used (Anderson 2001).

Temporal trends in species richness and diversity

To determine if the species diversity of fishes in the lower River Murray system had decreased over time, indices of richness and diversity were estimated from annual catch data. Hill's suite of numbers provide appropriate estimates of species richness and diversity for investigating fishery impacts on fish assemblages (Rice 2000). Species richness (S, Hill's H_0), was estimated, and two univariate measures of diversity: Hill's N_1 and N_2 (Hill 1973) were estimated for freshwater; estuarine and marine habitats; and for all habitats combined. Hill's N_1, the exponential of the Shannon-Wiener function (exp H') is most sensitive to changes in rare species and Hill's N_2, the

reciprocal of Simpson's index (1/D), is most sensitive to changes in the abundant species (Krebs 1989).

Temporal trends in fishery catch and effort

For each of freshwater, estuarine and adjacent marine habitats, estimates of total annual catches and effort, proportional contribution of key species and catch per unit effort (CPUE) of key species were calculated. Catch per unit effort was estimated as total annual catch (kg)/total annual effort (fisher days). Estimates of catches and CPUE for mulloway were made separately for estuarine and marine habitats. Annual catches of species that were <1 t were aggregated into an 'other' category. CPUE was estimated from annual catch (kg)/targeted effort (fisher days).

Trends in annual CPUE for each species were compared with trends in annual freshwater inflow using linear regression. Time lags between annual catches and inflows were determined from the modes in population age structures for golden perch, mulloway, black bream, and greenback flounder. For other species appropriate lag periods were estimated from life-history information sourced from the literature searches.

Population life-history and demography

Age and size data were available from commercial catches of four large-bodied, native species; golden perch, (2006-07, n=98), mulloway, (2001-02, $n_{juveniles} = 260$, $n_{adults} = 73$), greenback flounder, (2007-08, n = 85) and black bream, (2006-07, n = 138). Catches were intercepted at the point of landing and the total length (TL) of each fish was measured and sagittae removed via a cut through the ventral, ex-occipital region of the skull. Sagittae were cleaned, dried and stored in labelled plastic bags. Sagittae from mulloway, greenback flounder and black bream were embedded in fibreglass resin, and a longitudinal section cut with a diamond blade mounted on a Gemmasta 6" (150 mm) bench top saw. Each 500 μm thick section was cut so as to incorporate the otolith centre and then mounted on a glass microscope slide using Cyano-Acrylate glue. The mounts were examined using a Leica MZ-16 dissecting microscope with incidental light and the ages of the fish were determined from counts of opaque zones. Ages of golden perch were estimated from opaque zones in otoliths that had been broken and burned. The pattern of deposition for otoliths suggests that the opaque zone is completed in November-December, and that one annulus is deposited each year, for golden perch, black bream and mulloway in southern Australia (Anderson et al. 1992; Sarre and Potter 1999). For greenback flounder the annulus is completed in September-November (J Earle, 2010 pers. com.).

Literature searches were conducted to locate estimates of maximum age/size, age/size at maturity, and growth rates. Searches were first directed at peer-reviewed articles in journals (Web of Science, December 2009), then second at grey literature (e.g. theses and technical reports).

Results

Temporal trends in freshwater flows

Annual freshwater flows greater than 11,000 GL occurred in four years 1989, 1990, 1992, and 1993 within a single 5-year period (Figure 2A). After 1993, annual flows were generally low (<1,000 GL) with only four years experiencing flows greater than 1,000 GL.y^{-1}.
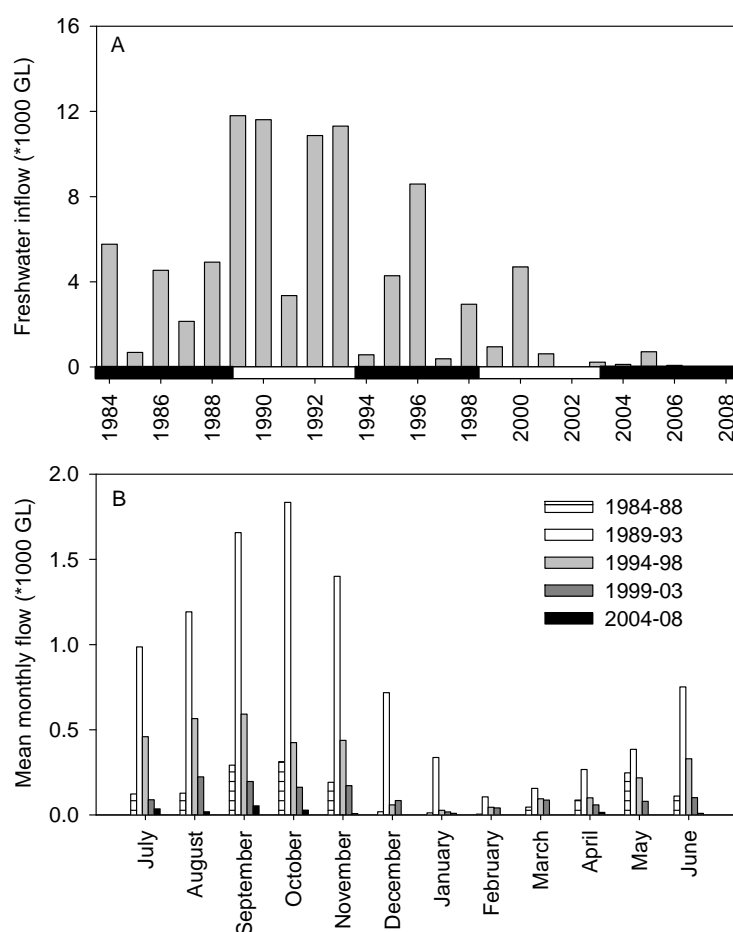
Figure 2. (A) Mean annual freshwater inflows to the Lower Lakes and Murray River estuary (black and white bars show five-year periods), and (B) mean monthly freshwater inflows for each five-year period.

When annual freshwater inflows were aggregated by five-year periods there was one period with high freshwater inflow (1989-93), and two periods with intermediate freshwater inflow (1984-88 and 1994-88). Flows were low for the two most recent five-year periods (1999-2003 and 2004-08). Flows were highly seasonal (Figure 2B) with peak flows in September - October for each five-year period.

Temporal trends in species structure of catches

Species structure of catches changed over time. Overall species composition differed among 5-year periods (PERMANOVA, Pseudo-$F_{4,24}$ = 8.3463, $p<0.001$). Pairwise comparison showed that catch composition differed among all combinations of 5-year periods (PERMANOVA, $p<0.05$), except for 1989-93 and 1994-98 ($p>0.05$). The species composition could be allocated to five-year time periods with an overall classification success of 76% (Figure 3A, CAP analysis).

Overall species composition differed between high, medium and low freshwater inflow (PERMANOVA, Pseudo-$F_{2,24}$ = 2.4733, $p< 0.036$). Pairwise comparison showed that catch composition differed between the low-flow years and medium, (3,000 – 6,000 GL.y^{-1}, $p<0.05$) and high-flow years (>6,000 GL.y^{-1}, $p<0.05$) but did not differ between medium and high flow-years ($p>0.1$). CAP analysis, constrained by high, medium, and low freshwater inflow, rather than the 5-year periods, provided less clear separation of groups (Figure 3B). Classification success for the low-flow group (< 3,000 GL.y^{-1}) was 100% but overall classification success was only 56%.
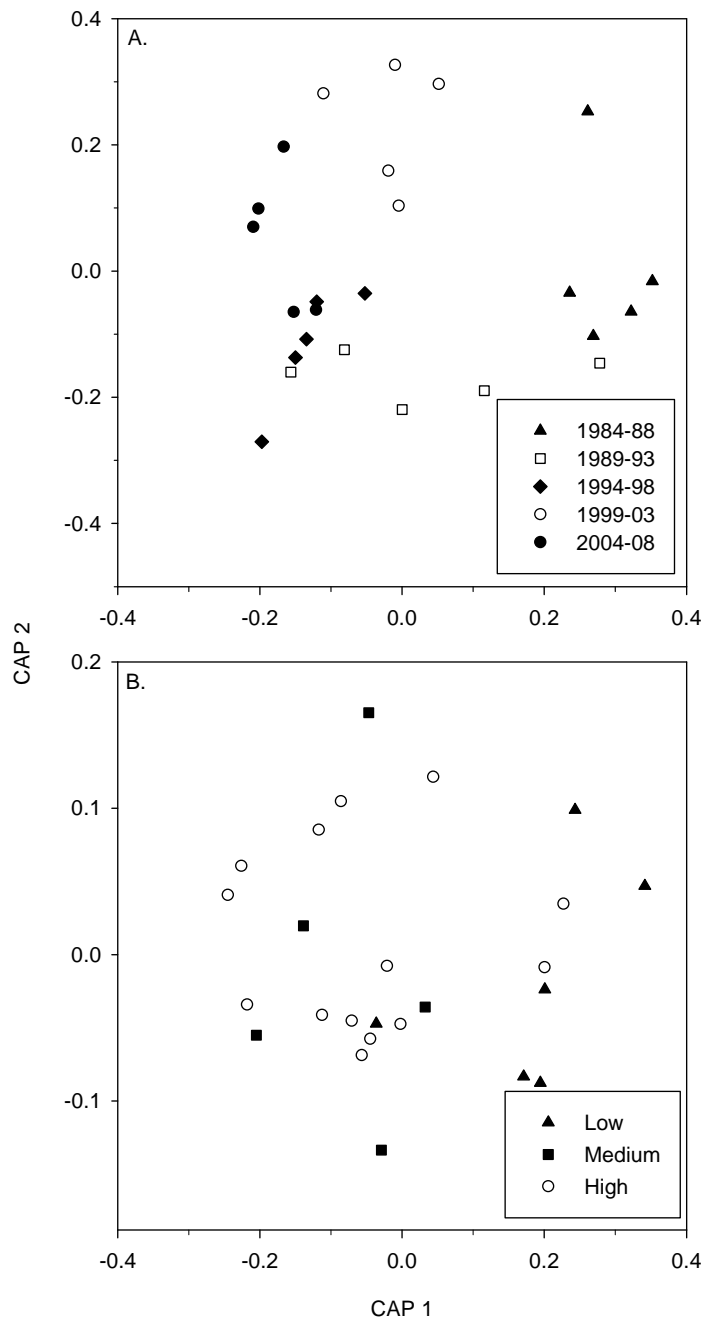
Figure 3. Canonical analysis of principal coordinates of annual catches from the Lakes and Coorong Fishery between 1984 and 2008; (A) for five 5-year periods, and (B) for low (< 3,000 GL.y^{-1}), medium (3,000 to 6,000 GL.y^{-1}), and high (> 6,000 GL.y^{-1}) annual freshwater inflows.

Temporal trends in species richness and diversity

The number of species (N_0) reported in all catches from all habitats combined from each year, declined linearly from 29 species in 1985 to 16 in 2008 (y = -1.483x + 45.322, r^2 = 0.59, $F_{1,24}$ = 32.432, p<0.001) (Figure 4). This reflected declines of species richness in both freshwater (y = -2.791x + 31.647, r^2 = 0.58, $F_{1,24}$ = 32.296, p<0.001) and estuarine habitats (y = -1.570x + 30.839, r^2 = 0.33, $F_{1,24}$ = 11.503, p<0.01) but not for the marine habitat (r^2 = 0.00). Higher species richness in freshwater habitat prior to 1996 was partly due to the presence of two exotic species, rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*), which are no longer found. Within the estuary the strongest decline occurred in the southern section which comprised the north and south Coorong lagoons (Figure 1, Areas 9-13), but excluded the region adjacent to the river mouth (y = -1.358x + 30.839, r^2 = 0.42, $F_{1,24}$ = 17.006, p<0.01).

Diversity of rare species across all habitats (Hill's N_1) changed little over time (Figure 4). However, diversity of the most abundant species (Hill's N_2) varied between years in freshwater and estuarine habitats, but not for the marine habitat. In the freshwater and estuarine habitats, diversity of abundant species increased from 1988 to 1992, then remained stable until 2008 in freshwater habitat, but declined steeply in the estuarine habitat.
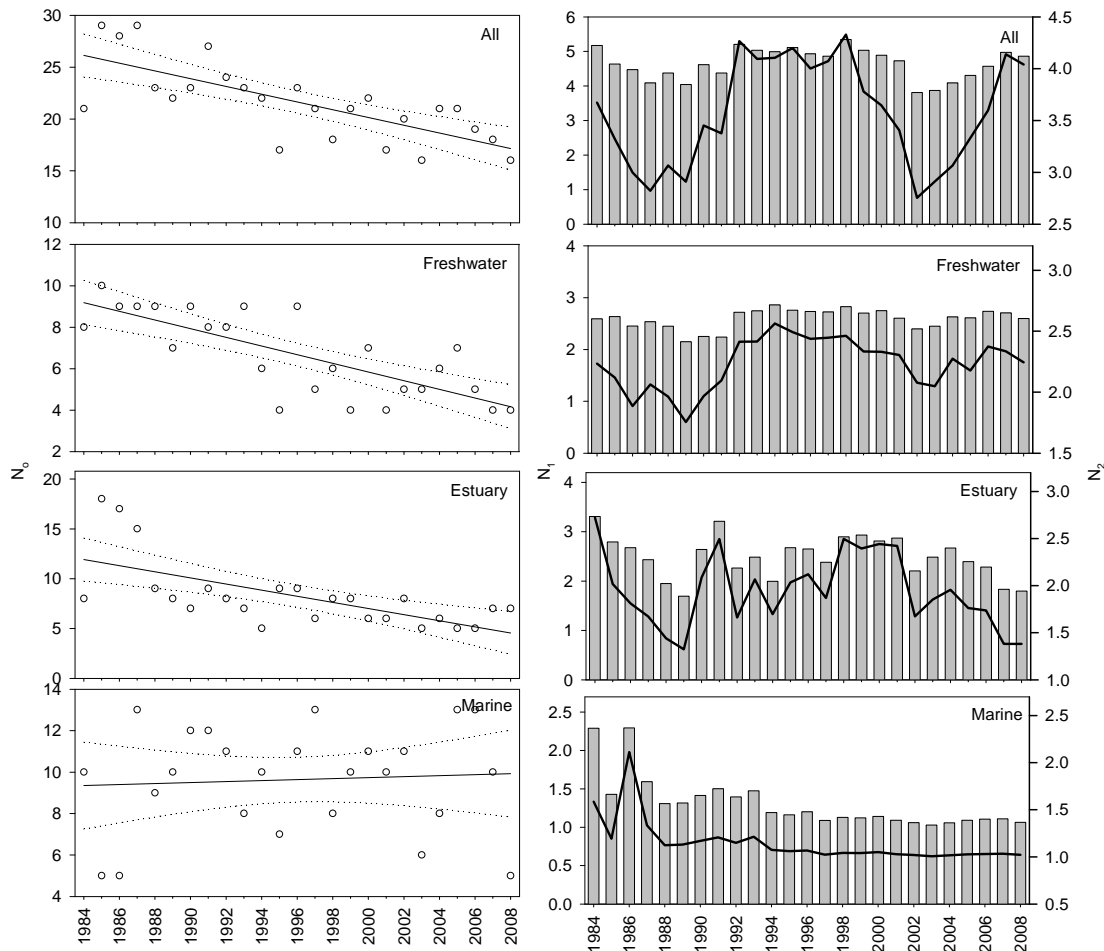
Figure 4. Hill's numbers for species richness (H_0, open circles), diversity (N_1, bars) and evenness (N_2, line) estimated from historical catch data from three habitats in the lower Murray River system. Species richness: solid line is linear regression and dotted lines are 95% confidence intervals.

Temporal and spatial trends in fishery catch and effort

All habitats

Total annual catches in the Lakes and Coorong Fishery increased from a minimum of 1,142 t in 1984 then were between ~2,100 t and ~2,700 t from 1989 to 2008, with an average total annual catch of 2,114 t.y^{-1} (±SE 79.3) (Figure 5A). The contribution from freshwater, estuarine, and marine habitats changed over time. Freshwater habitat contributed 70-80% to the total catch in the mid to late 1980's but this declined to 33.2% in 2001, before increasing again to 64.3% in 2008. Catches from the estuary were relatively stable with an average of 276 t.y^{-1} (±SE 17.8), over 25 years, contributing ~14% of the total catch. Marine habitat contributed 2.5% of the total

annual catch in 1984, which increased to a peak of 55.4% in 2002, before declining to 22.8% in 2008.

Total annual effort was consistent among years and ranged from 7,343 to 10,110 days with an average of 8,643 days.y^{-1} (±SE 168.4), over 25 years (Figure 5B). The years of lowest annual effort occurred from 2001 to 2008. The contribution from marine, estuarine, and freshwater habitats changed over time. Effort in freshwater habitat ranged from 68.8% of total effort in 1997 to 36.2% in 2002, with years of highest effort between 1993 and 1998. Effort in the estuary declined over 25 years from 53.0% of the total in 1984 to 29.6% in 2008 (LR: y = -0.007x + 34.933, r^2=0.56, $F_{1,24}$ =32.414, p<0.001). As effort increased in the freshwater habitat (1987-1990, 1992-99, and 2002-08) there was a corresponding decline in the estuarine habitat. Effort in the marine environment increased steadily from 1% of total effort in 1984 to peak at 23% in 2006 (LR: y = 0.14x + 0.76, r^2=0.94, $F_{1,24}$ =170.268, p<0.001).
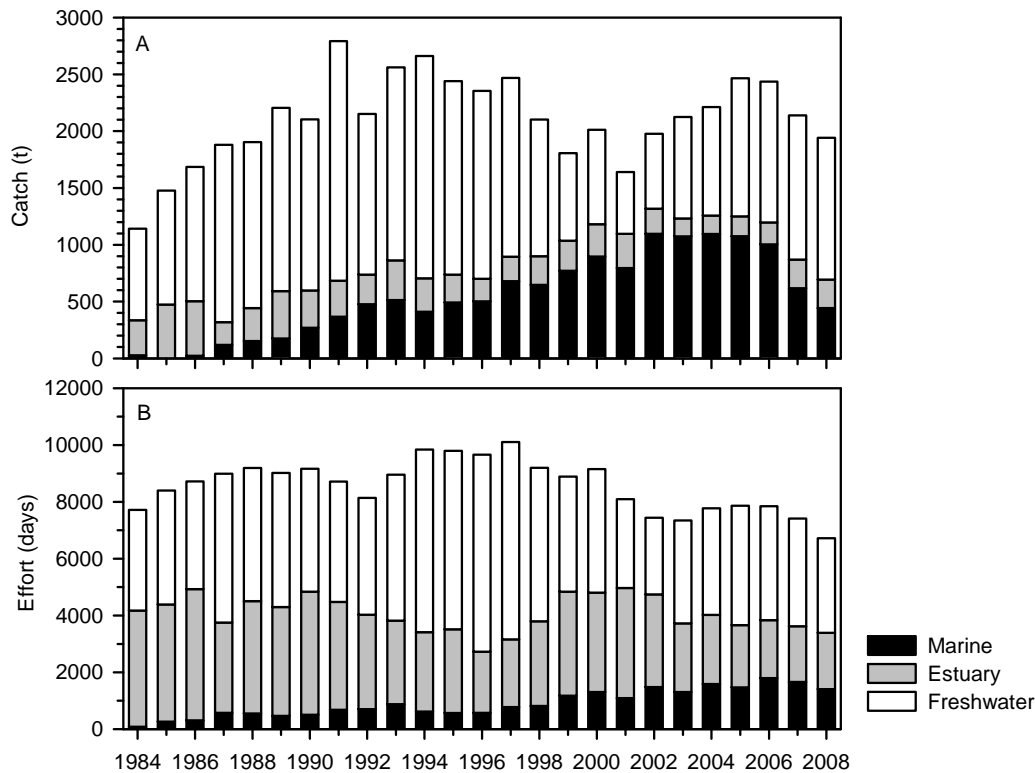
Figure 5. Catch (A) and effort (B) associated with commercial fishing in marine, estuarine and freshwater habitats in the lower Murray River system.

Freshwater

In freshwater habitat total annual catches comprised mostly bony bream (*Nematalosa erebi*) and European carp (*Cyprinus carpio*) which together contributed >88% of the catches in all years (Figure 6A). The contribution of European carp to total annual freshwater catch increased from 30 to 57% between 1990 and 2008, and total annual catch across all habitats was 15 to 38%. The contribution to total annual freshwater catch of bony bream declined from >65% in the mid 1980's to 34% in 2008. Golden perch (*Macquaria ambigua*) contributed 7 to 10% of total annual freshwater catches.

Estuary and marine

For catches from the Murray River estuary, the contribution from the small-bodied, native yellow-eye mullet (*Aldrichetta forsteri*), increased from 50 to 84% of catches from 1998 and 2008. Juvenile mulloway (*Argyrosomus japonicus*) contributed 18-35% of the total estuarine catch from

1993-2001, which then declined to 10% in 2007-08 (Figure 6B). For the marine habitat, annual catches were dominated by pipi (*Donax deltoides*, >95%) after 1986 (Figure 6C).
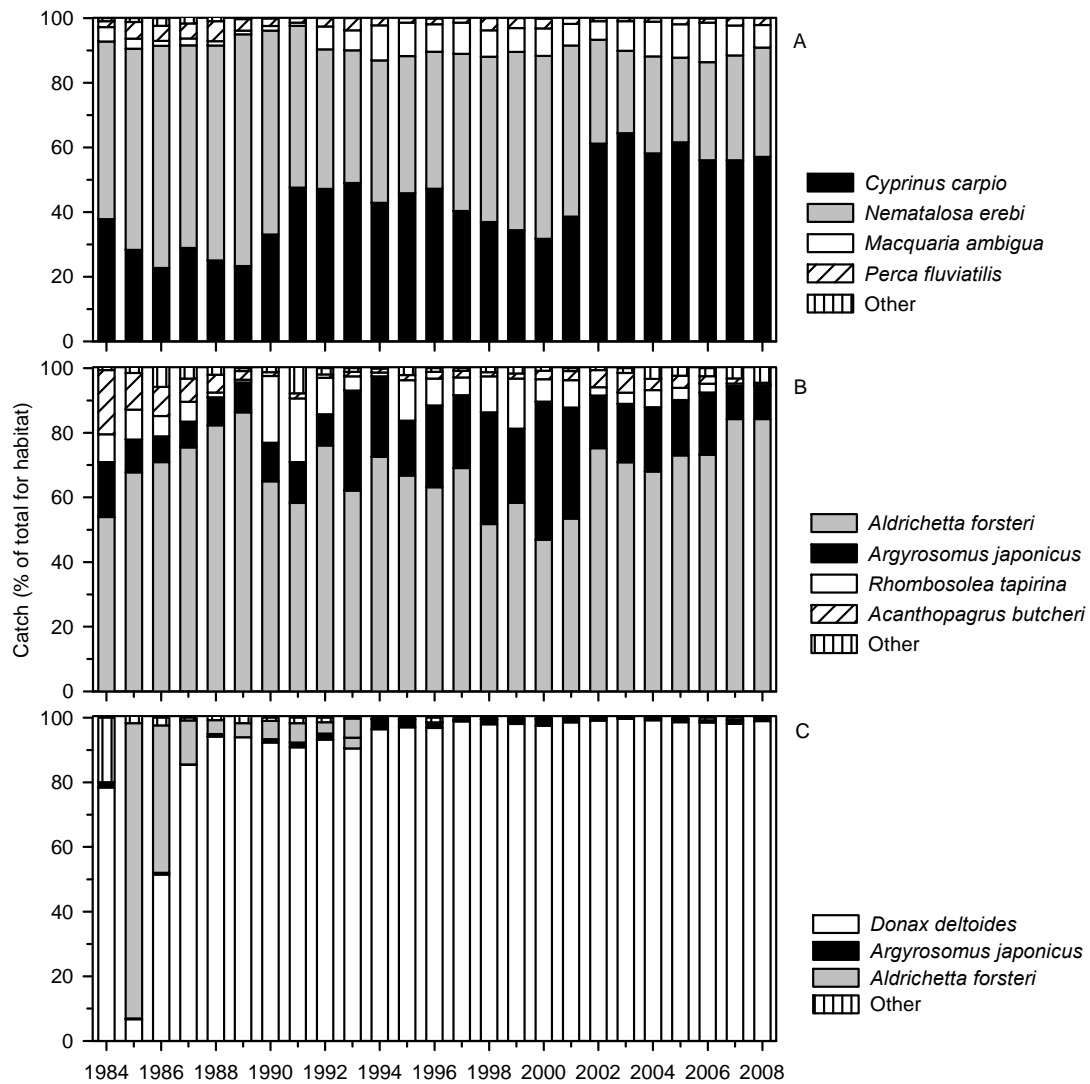
Figure 6. Catch species composition from (A) freshwater, (B) estuarine, and (C) marine habitat in the Murray River system.

Population demography, life-history and impact of freshwater flows

High growth rates and early maturation indicated that European carp is an opportunistic strategist (Table 1). Relative abundance of European carp increased following high freshwater inflows with variability in annual freshwater inflow explaining 12% of variability in relative abundance (Table 2). Relative abundance increased steeply in the early 1990's following historically high freshwater inflows in the late 1980's (Figure 7).

An estimate of growth rate was not available for bony bream but low maximum age and early age of maturity also indicated an opportunistic life history strategy (Table 1). Although relative abundance of bony bream increased following flooding in the late 1980's there was little inter-annual variability in relative abundance in later years (Table 2, Figure 7).

Slow growth rate, large maximum size and age, combined with high age at maturity indicated that the perchichthyid golden perch had a periodic strategist life history (Table 1). Although a strong increase in relative abundance occurred in 1993 following flooding, there was no relationship between annual freshwater inflows and relative abundance (Figure 7, Table 2). Ages ranged from 3 to 10 years with a mode of 5 years, corresponding to a year class originating from 2003 (Figure 8). The maximum age of fish was 10 years suggesting that the age distribution has been truncated given this species may live to 26 years (Table 1).

The life-history of the opportunistic strategist yellow-eye mullet is characterised by rapid growth, high fecundity, early maturation and a maximum age of 4-7 years indicating an opportunistic life-history strategy (Table 1). It is likely that relative abundance of yellow-eye mullet from the Murray River estuary has increased because annual CPUE has increased whilst catches have remained stable over 25 years (Figure 6, 7). Variability in annual freshwater inflow to the estuary explained 45% of variability in relative abundance (Table 2), with high annual freshwater inflow associated with low CPUE (Figure 9).

The life-history of the periodic strategist mulloway is characterised by slow growth, late maturation and high maximum age (Table 1). Mean annual flow explained 16% of the variability in CPUE for adults of this species from marine habitat (Table 2). Increasing CPUE of juvenile mulloway from 2001 should be interpreted with caution because the area fished contracted with decreasing freshwater inflow (Figure 9). Consequently, CPUE may remain high at the small spatial scales of the area fished, while abundance at the level of the overall stock declines i.e. CPUE may be hyper-stable (Hilborn and Walters 1992; Maunder et al. 2006). The ages of

juveniles in the estuary ranged from 0 to 6 years with a mode of 3 years (Figure 8). Age structures of adults from marine habitat were truncated because few individuals (4%) were >16 years old, which was 60% below the maximum age of 41 years (Figure 8, Table 1).

The life history of the periodic strategist black bream (*Acanthapagrus butcheri*) is characterised by slow growth, intermediate age of maturity and high fecundity (Table 1). Although CPUE increased over 20 years (Figure 7) this should be interpreted with caution because proportional catches: (i) declined precipitously in the early 1980's and subsequently remained low (Figure 6); and (ii) appeared to contract spatially with decreasing freshwater inflows such that CPUE may be hyper-stable (Figure 9). Age structures for black bream ranged from 3 to 18 years with modes at 3 and 9 years corresponding to year classes from 2003 and 1998 (Figure 8). A maximum age of 29 years (Table 1), and presence of few individuals >11 years old (<7%) indicated that the age distribution was truncated. The legal minimum length (LML) for black bream is 46% above size of maturity (SOM).

Little is known of the life-history of the pleuronectid greenback flounder in the Murray River estuary. This species is possibly intermediate between opportunist and periodic strategist because individuals are thought to grow quickly, to mature at approximately one year old, and to be highly fecund, but may attain >6 years age (Table 1). Catches and relative abundance declined steeply from 2000 (Figures 6, 7). Age structures from commercial catches were dominated by one and two year old females (>93%) although the maximum age is likely >6 years (Figure 8, Table 1).

The r-strategist (MacArthur and Wilson 1967) life-history of pipi is characterised by fast growth, early maturation, and a short life span of 3-5 years (Table 1). Variability in freshwater flows explained 45% of variability in relative abundance of pipi (Table 2).

Table 1. Life history parameters for species in three habitats in the lower River Murray; F = freshwater, E = estuarine, M = adjacent marine, AOM = age of maturity, SOM = size of maturity. Max size is the von Bertalanffy growth parameter L_{inf}, where available.

Species	Habitat	Max Age (y)	Max size (mm)	K	AOM (y)	SOM (mm)	Source
Golden perch *Macquaria ambigua*	F	26	-	0.25 - 0.45	4-5	F/M 400	(Anderson et al. 1992)
European carp *Cyrinus carpio)*	F	32	1,200	0.38 - 0.48	1.4-2.7	F 273-328 M 287-307	(Vilizzi and Walker 1999; Smith 2005)
Bony bream *Nematalosa erebi*	F	~3	480	-	1-2	F 108 M 121	(Puckridge and Walker 1990)
Murray cod *Maccullochella peelii peelii*	F	48	1,200	0.11	M 3-4 F 4-6	F 500 M 400	(Anderson et al. 1992; Gooley et al. 1995)
Mulloway *Argyrosomus japonicus*	M & E	41	1,600	0.16 - 0.14	5	F/M 812	This study (Chapter 4).
Black bream *Acanthopagrus butcheri*	E	29	540	0.04 - 0.08	1.9 - 4.3	F/M 129-169	(Coutin et al. 1997; Morison et al. 1998; Norriss et al. 2002)
Greenback flounder *Rhombosolea tapirina*	E	6	-	-	~1	F 219 M 190	(Kurth 1957; Crawford 1986; Stevens et al. 2005)
Yellow-eye mullet *Aldrichetta forsteri*	M & E	4-7	500	-	2	F/M 220-230	(Harris 1968)
Pipi *Donax deltoides*	M	4	-		1	M/F 36	(King 1970)

Table 2. Linear regressions performed on CPUE for species in three habitats in the Lakes and Coorong Fishery. For *Argyrosomus japonicus* (a = adults, j = juveniles). For the independent variable the subscript (L) is the time-lag in years. The coefficients of determination (r^2) are shown with their *P* values (*significant, $\alpha=0.05$, [ns]non significant).

Habitat	Species	Independent variable	b (±SE)	$F_{1,24}$	r^2	*p*
					Regression statistics	
Freshwater	*Cyprinus carpio*	$flow_{L-2}$	4.51 (2.571)	3.079	0.12	0.073[ns]
	Macquaria ambigua	$flow_{L-5}$	-0.03 (0.311)	0.013	0.00	0.910[ns]
	Nematalosa erebi	flow	-1.04 (0.997)	1.106	0.05	0.304[ns]
Estuary	*Acanthopagrus butcheri*	$flow_{L-3}$	-0.65 (0.234)	7.732	0.25	0.011*
Rho	*mbosolea tapirina*	$flow_{L-1}$	0.19 (0.414)	0.215	0.01	0.647[ns]
	Argyrosomus japonicus (j)	$flow_{L-3}$	0.12 (0.627)	0.040	0.00	0.844[ns]
Aldrichett	*forsteri*	$flow_{L-1}$	-4.89 (1.135)	18.563	0.45	0.000**
Marine	*Argyrosomus japonicus* (a)	$flow_{L-5}$	2.35 (1.142)	4.250	0.16	0.051[ns]
	Donax deltoides	flow	60.39 (13.802)	19.149	0.45	0.000**
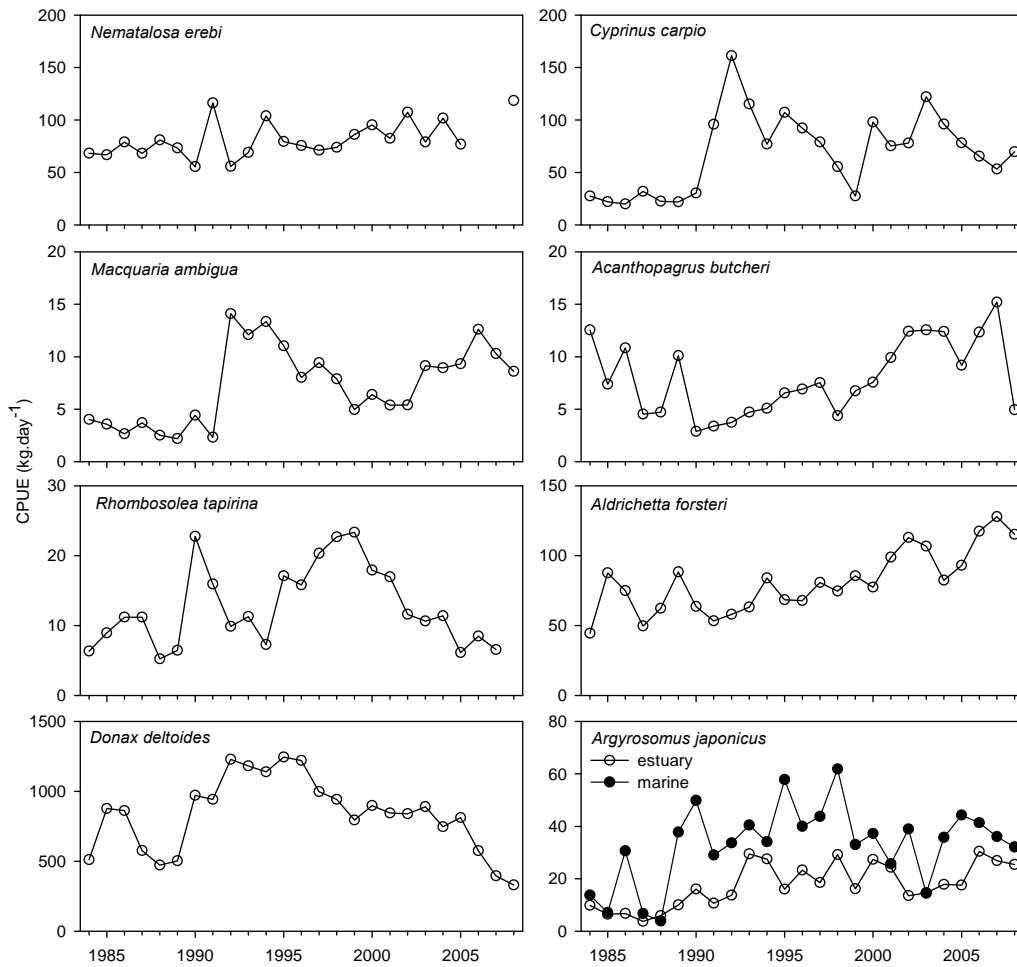
Figure 7. Relative abundance of exploited species in lower Murray River system: *Nematalosa erebi*, *Cyprinus carpio* and *Macquaria ambigua* in freshwater habitat; *Acanthopagrus butcheri*,.*Rhombosolea tapirina*, *Aldrichetta forsteri*, and juvenile *Argyrosomus japonicus* in estuarine habitat; and *Donax deltoides* and adult *A. japonicus* in adjacent marine habitat.
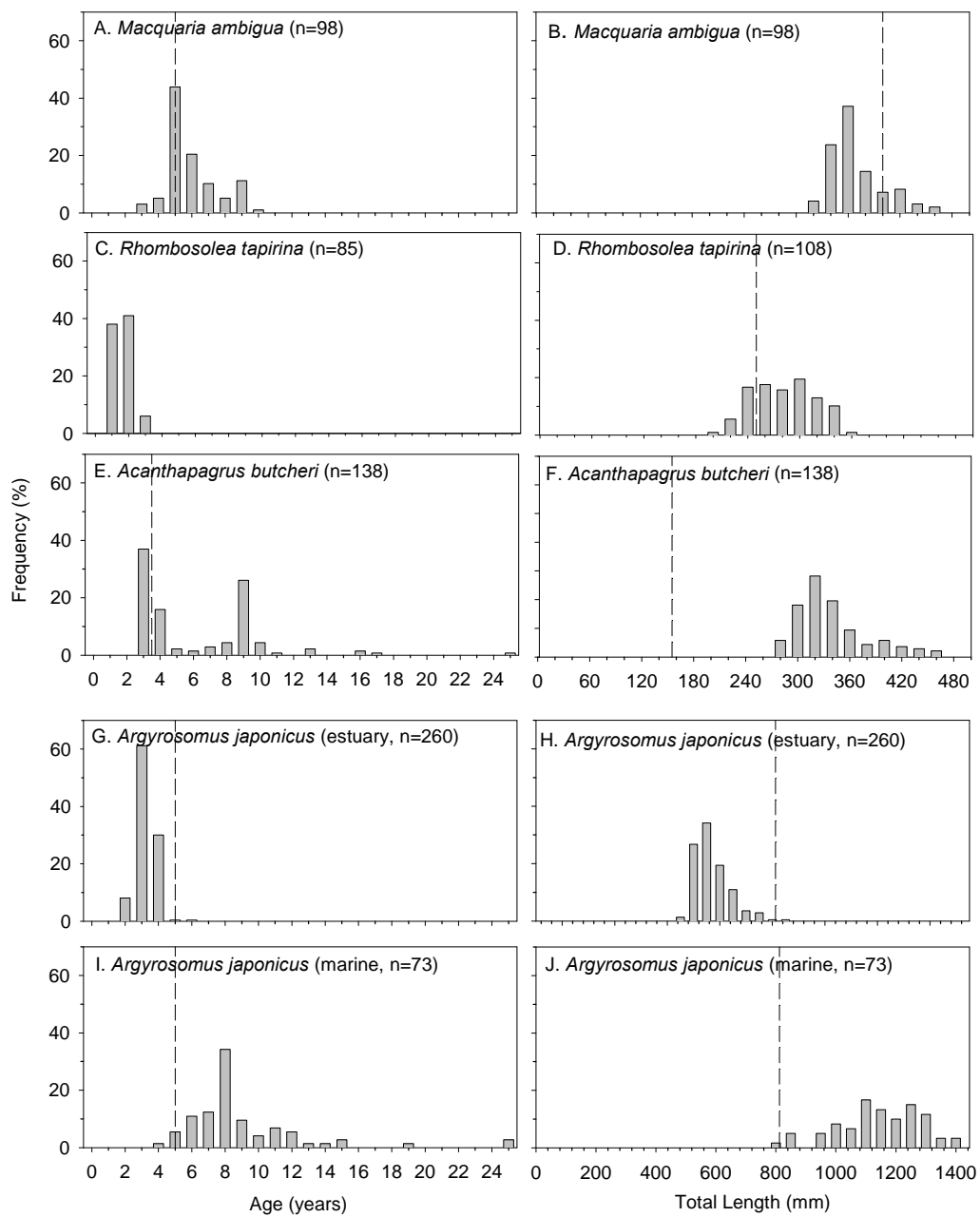
Figure 8. Age (left) and size structures (right) for commercial catches from freshwater (A, B); estuarine (C - H), and nearshore marine (I, J) habitats. On left hand panels vertical dashed line represents age at maturity, /size. On right hand panels vertical dashed line represents legal minimum length.

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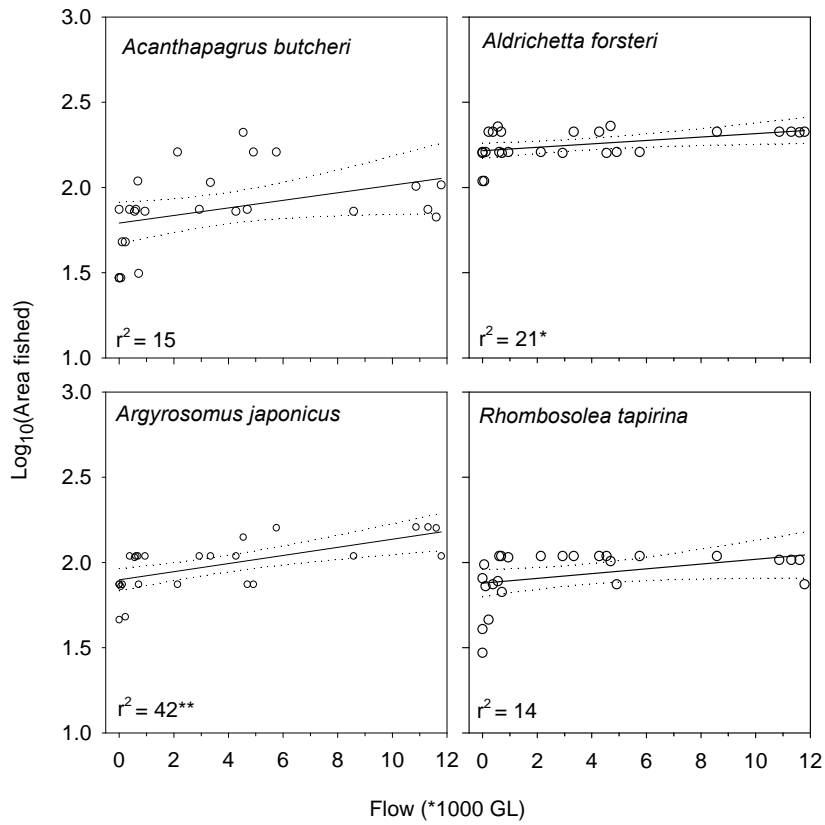
Figure 9. Relationships between annual freshwater flows and area of Murray River estuary that was fished in that year. Significant correlations are marked * (p <0.05), or ** (*p*<0.005).

Discussion

Changes in the species composition of catches in the Lakes and Coorong Fishery suggest that the structure of the fish assemblage in the lower River Murray system has changed in the 25 years from 1984 to 2008. In freshwater and estuarine habitats, species richness has declined and the abundances of species with opportunistic life history strategies and generalist habitat requirements has increased. The truncation of the age structures of four species of long-lived, large-bodied, native fishes suggests environmentally-induced recruitment failure, longevity over-fishing and compromised egg production. These fish species are all periodic strategists which require strong freshwater flows for successful reproduction and it is important that the remnant populations be protected from over-exploitation until environmental conditions conducive to successful reproduction are restored to the system.

Temporal changes in the species composition of catches in the Lakes and Coorong Fishery were due to impacts of reduced freshwater flows, fishing pressure, or a combination of both factors. However, separating the effects of such factors in freshwater and estuarine environments is difficult (Laë 1995; Cabral et al. 2001), particularly when major changes to population and assemblage structure, and collapse of populations occur prior to data collection (Dayton et al. 1995; Jennings and Kaiser 1998). In the lower Murray River system fish populations have been exploited for more than 100 years and freshwater flows have been restricted for almost 70 years. For example, prior to construction of the barrages in the early 1940's, catches of mulloway were an order of magnitude larger than in the last 25 years (Chapter 4). Similarly, records from fish processors indicate a steep decline in the catches of Murray cod from 140 to 2 $t.y^{-1}$ between 1954 and 1975, almost a decade before collection of fishery catch and effort information (Ye et al. 2000).

The similarity in species composition of catches from the Lakes and Coorong Fishery in the five-year periods of 1989-93 and 1994-98, and high diversity of abundant species (Hills N_2), may reflect successful recruitment of flood-dependent species during 1989-93 when floods (>10,000 $GL.y^{-1}$) occurred in four of five years and the persistence of strong age classes through 1994-98. The potential for multiple flood years to have cumulative, persistent effects on the structure of fish assemblages has been identified for freshwater species (Puckridge et al. 2000; Leigh et al. 2010). It is notable that the 1993 age class still dominated the mulloway population in 2002.

After 1993, the frequency of strong flows into the lower Murray River system diminished and there were no years when flows exceeded 10,000 $GL.y^{-1}$ (our definition of a flood). This

reduction may explain the difference between the species composition of catches in 1989-93 and 1994-98 and subsequent 5-year periods, when declines in both species richness and diversity of abundant species occurred in freshwater and estuarine habitats. Reductions in flow may have negatively influenced spawning and recruitment, and thus reduced the abundance of species, such as golden perch, black bream and mulloway that require strong flows for successful recruitment. The change in species composition of catches over time may also reflect the reductions in abundance of exploited species due to fishing (Blaber et al. 2000), particularly for periodic strategists, such as golden perch, black bream and mulloway.

In freshwater habitat, catches prior to this study were dominated by the periodic strategist Murray cod (Rowland 1989; Ye et al. 2000). However, during the present study, the periodic strategists, including golden perch, contributed less than 10% of the total annual catch whilst opportunistic species dominated (>90%). Prior to the current drought, catches were dominated by bony bream, which is the only large-bodied, native species that has not declined since river regulation in the 1940's. The successful recruitment of this species is not dependent on river flows (Puckridge and Walker 1990). Since the current drought there have been significant increases in catches of the exotic European carp. Although this species has been established in the lower River Murray for decades, and is now the most abundant, large-bodied fish species in the region (Gehrke et al. 1995; Smith 2005), its recent success may be because carp are generalists that can perform well in anthropogenically degraded habitats (Cadwallader 1978; Koehn 2004). Furthermore, successful recruitment is not dependent on strong flows (Ye et al. 2008). The recent, low abundance of golden perch may be due to its dependence on elevated water temperatures and flooding for successful recruitment (Mallen-Cooper and Stuart 2003; Ye 2004; Ye et al. 2008). The truncated age structures of this species, with the oldest individuals only 62% of the maximum age, reflect the effects of fishing which removes older, larger individuals from populations (Hilborn and Walters 1992; Planque et al. 2010; Walsh et al. 2010).

In the estuarine habitat, the combined catches of the periodic strategists, mulloway and black bream and the intermediate strategist greenback flounder decreased from ~52% in 2000 to ~12% in 2008. During this period, catches were increasingly dominated by the opportunistic strategist yellow-eye mullet. The apparent increase in the relative abundance of yellow-eye mullet may be due to its: (i) tolerance of a wide range of salinities (Young and Potter 2002); (ii) ability to recruit successfully each year regardless of freshwater inflows; and (iii) reduced predation by mulloway (Ferguson, unpublished data, Deegan et al 2010). Although the population of yellow-eye mullet in the estuary is thought to be self- recruiting, it is a marine species that uses estuarine

habitat opportunistically (Harris 1968; Lenanton and Potter 1987), and there is potential for recruitment from the marine environment.

Environmental limitation of populations of periodic strategists in the estuarine habitat, i.e. mulloway, and black bream, partly explain their declining abundance. Lack of strong freshwater inflows since 1993 may have resulted in poor recruitment of mulloway, with possible recruitment failure in recent drought years (Hall 1986; Griffiths 1996; Chapter 4). Black bream may also depend on freshwater inflows and other environmental cues for successful spawning and recruitment (Hobday and Moran 1983; Norriss et al. 2002; Nicholson et al. 2008). The truncated age structures of black bream and mulloway with the oldest individuals captured being significantly younger than the recorded maximum ages suggest that fishing has also impacted these populations (Hilborn and Walters 1992; Planque et al. 2010; Walsh et al. 2010). Age structures of greenback flounder in the estuary appears to be particularly truncated as only one and two year old females comprised >95% of the population, while longevity is >6 years (Stevens et al. 2005).

In the marine habitat, species composition of catches was similar between years, and after 1998, >90% was contributed by the r-strategist pipi. Although the relative abundance of pipi appeared to be positively related to freshwater flows, this finding should be interpreted with caution because the long-term decline in relative abundance, combined with high effort and catch (Ferguson and Mayfield 2006), suggest that severe overfishing occurred contemporaneously with the recent drought.

Information available for assessment of fish in ecosystems commonly includes fishery catches, estimates of relative abundance from CPUE, and species composition. Total catch (biomass) provides a poor indicator of the performance of a multi-species fishery. Although total catch can be partitioned spatially i.e. by habitat, it may fail to capture temporal changes in catch composition. In this study, total annual effort and catches were relatively consistent, but the proportional contribution from each of three fished habitats, and the species within them, varied among years. An overall decline in catches of finfish was compensated for by relocation of effort to a population of bivalve in marine habitat. This is consistent with other studies of multi-species fisheries that have reported stable total annual catches whilst catch species composition changed (May et al. 1979; Laë 1995). For example, in the North Sea, the catch of herring and mackerel declined over 10 years but total catches remained constant due to increasing catches of gadoids such as Norway pout (May et al. 1979).

At the population level, CPUE often provides the only available estimate of relative abundance although its use over extended periods may be problematic. CPUE may not be proportional to abundance over the entire exploitation history of a given population (Maunder 1998) and requires interpretation in the context of each species' life history and likely response to environmental factors such as drought (King and McFarlane 2003). CPUE may also provide an inconsistent index of relative abundance in highly changeable environments such as in end-river systems. For example, CPUE may over-estimate relative abundance of mulloway and black bream because the area of estuarine habitat that is available, contracts as freshwater inflow decreases. Other studies have suggested that changes in freshwater inflows affect catchability of estuarine fish because of effects on migration and schooling caused by associated salinity fluctuations which may alter habitat availability (Loneragan and Bunn 1999; Gillson et al. 2009). CPUE may also overestimate relative abundance when fish are targeted as they aggregate to spawn, which may be the case for black bream and greenback flounder in the estuary and mulloway in the marine habitat.

Species composition of catches may provide a useful indicator of spatial and temporal changes in fish assemblages. In this study, we found that fish assemblages changed over time, and that flood, drought and fishing were likely causal factors. Of most concern was: (i) the temporal trend of increasing contribution to catches from shorter-lived; early-maturing, habitat generalists that have the ability to recruit successfully each year; and (ii) reduced contributions of long-lived, late maturing, environmentally-limited species.

A suite of indicators may provide the most robust estimation of the relative impacts of anthropogenic environmental effects and fishing on ecosystems. Together, time series of biomass (total catches), catch composition and species diversity indices provide a broad indicator of ecosystem health (Whitfield 1996; Soto-Galera et al. 1998). However, understanding the relative impacts of environmental change and fishing and the potential for rebuilding of populations requires supplementary information including detailed life-history information and time-series of age structure data.

Management of multi-species fisheries needs to be tailored to the most sensitive, rather than the most robust species (Myers and Worm 2005). Consideration of life-history strategies is fundamental to assessments of resource status (King and McFarlane 2003) and should recognise the vulnerability of periodic strategists to longevity over-fishing, especially when successful recruitment is infrequent (Beamish et al. 2006). In the Murray River system, populations of golden perch, mulloway and black bream rely on the establishment of one, or two, strong year

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classes at irregular intervals to maintain their populations. The truncated age distributions of these species provide evidence that they are longevity overfished (Beamish et al. 2006). Continued over-exploitation will reduce their potential egg production and capacity to produce strong age classes when or if, environmental conditions again become favourable (Myers and Worm 2005; Hsieh et al. 2006; Hsieh et al. 2010). The first step to rebuilding the populations of large-bodied, native species in the Murray River system is to protect existing populations from further over-exploitation.

Management options for preventing further over-exploitation are to: (i) increase size limits to protect immature fish; (ii) reduce by-catch of juveniles of target species; (iii) establish an upper size limit to protect large, highly fecund females; (iv) protect spawning aggregations from fishing; and (v) restrict gear-types used to target periodic strategists. For periodic strategists, setting a legal minimum length (LML) that protects immature and young adult fish is critically important. While the size at which 50% of females mature (SAM_{50}) is commonly used for opportunistic species, more conservative LMLs may be needed for periodic strategists, especially in habitats where conditions favouring successful recruitment are infrequent. It is notable that the LML for black bream (280 mm TL) is 80% higher than the best available estimate of SAM_{50} (Norriss et al. 2002) and although this species' age structure is truncated it is the only large-bodied, native fish with more than one strong year class in the population. In contrast, the LML of 460 mm TL for mulloway in the estuary is 44% below SAM_{50} (Chapter 5), and approximately 90% of the commercial catch is comprised of sexually immature individuals (Ferguson and Ward 2003). Allowing recruits to become established in the adult population is essential if age structures are to rebuild. However, the bycatch of estuarine fisheries typically includes juveniles of the target species (Gray 2002; Gray et al. 2004). Many recruits do not reach LML when by-catch mortality is high. A recent study showed that the number of sub-legal sized mulloway taken incidentally in gillnets in the Murray River estuary is equivalent to the number harvested, with low rates of survival (23% alive at capture) (Ferguson 2010b).

The results of this study have implications for the management of fish populations in a changing climate. The decline in species richness over 25 years in both freshwater and estuarine habitats and decline in diversity of abundant species since 2002 in estuarine habitat may contribute to reductions of ecosystem resilience and increase sensitivity to climate change (Worm et al. 2006). In addition, under predicted climate change scenarios, populations of long-lived species with fisheries-induced, truncated age structures, may be more prone to collapse (Planque et al. 2010). This is particularly important for environmentally-limited populations of golden perch,

mulloway, black bream and greenback flounder due to current impacts of drought on critical habitat and climate change predictions that indicate further reduction to freshwater inflows of 15% by 2030 and up to 35% by the 2050's (Hughes 2003).

Conclusions

Species composition of catches provides valuable information about fish assemblages and populations. In combination with information on demography and species' life-histories catch composition information may provide; (i) indications of changes in environmental health, (ii) identify vulnerable populations, and (iii) inform management of multi-species fisheries in highly changeable environments such as estuaries.

Management of periodic strategist species, in end-river and estuarine habitats should seek to conserve age structures, because populations of these species depend on healthy age/size structures to withstand environmental variability. Management of commercially-exploited, large-bodied, native species in drought-affected areas should seek not only to preserve remnant age structures but also to rebuild them.
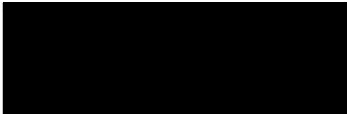
Twelve month old *Argyrosomus japonicus* in on-land aquaculture facility (Photograph courtesy of South Australian Research and Development Institute - Aquatic Sciences).

Statement of authorship

In this chapter, Gregory Ferguson performed all analyses on the data. Timothy Ward and Bronwyn Gillanders provided advice on the manuscript

Certification that the statement of contribution is accurate.

Signed ████████████████ Date 30 · 9 · 2010

Gregory J Ferguson (*Candidate*)

Certification that the statement of contribution is accurate and permission is given for the inclusion of the paper in the thesis.

Bronwyn M Gillanders ████████████ Date 30. 9. 2010

Timothy W Ward ████████████ Date 3.0.9.10.

Chapter 3

Population structure of a large sciaenid, *Argyrosomus japonicus*, in southern Australia

Abstract

The stock structure of *Argyrosomus japonicus* in South Australia was investigated by comparing elemental signatures and shape characteristics (otolith meristics and overall shape) of sagittal otoliths of fish from three regions. Mean concentrations of Sr:Ca, Ba:Ca, and Mg:Ca were examined in otolith edges from western, central and eastern regions of South Australia. Low levels of Ba:Ca in otoliths from the western region contrasted with high levels of Ba:Ca in otoliths from the eastern region and intermediate levels were observed from the central region. Constrained Canonical Analysis of Principal Coordinates (CAP) of elemental concentrations was used to allocate individual otoliths to geographic regions. Allocation success was 100, 100, and 87% for western, central and eastern South Australia respectively. Although analysis of otolith shape (elliptical Fourier descriptors) had poorer classification success, results supported those of the elemental study with 78, 59, and 70% of otoliths allocated to western, central and eastern regions respectively. Shape analysis also successfully discriminated otoliths of *A. japonicus* reared in aquaculture from those of wild caught fish in nearby waters with 100% allocation success. Shape analysis was then used to investigate the origin of individuals caught in marine waters but suspected of being from an aquaculture facility. The two stock discrimination methods were complementary because trace-element analysis of the otolith edge provides very high classification success and gives a snapshot of differences between groups from different geographic areas, while shape analysis indicates that these discrete groups of fish experienced different environmental conditions over a long period of time. Management of *A. japonicus* in South Australia currently assumes a single stock. Sub-structuring of the stock suggested by this study may mean that that potential for recruitment between these populations is limited.

Introduction

Understanding stock structure is important because different stocks may respond differently to exploitation and rebuilding (Begg and Waldman 1999). While stock has traditionally been defined as an intra-specific group of randomly mating individuals with temporal and spatial integrity (Ihssen et al. 1981), fish populations are more likely to be only partially reproductively isolated and comprise a series of sub-populations (Shaklee and Currens, 2003).

Methods that have been used to delineate stocks of fish include genetic, meristic, morphometric and mark-recapture techniques, as well as comparison of life-history parameters (Cadrin et al. 2005). Of these methods, those based on features of otoliths are regarded the best for stock discrimination because otoliths are considered metabolically inert and grow throughout the life of the fish (Campana and Neilson 1985). The shape of otoliths may be influenced by environmental and genetic factors, although environmental effects are generally thought to be greater (Campana and Casselman 1993). Successful stock delineation has been achieved using the external outline of otoliths (Petursdottir et al. 2006; Pothin et al. 2006; Mérigot et al. 2007) and also trace element composition of otoliths (Campana 1999; Campana 2005). Several studies have recommended that results from more than one stock discrimination method be integrated into a 'holistic' stock identification approach to maximise the likelihood of correctly delineating stocks (Begg and Waldman 1999). Using a combination of approaches and comparing between several of these is therefore important (Waldman 1999).

The sciaenid *Argyrosomus japonicus* has an Indo-Pacific distribution between latitudes of 40ºS and 40ºN and occurs in estuarine and nearshore Pacific Ocean and Indian Ocean waters surrounding Australia, Africa, India, Pakistan, China, Korea and Japan (Silberschneider and Gray 2008). This species supports commercial and recreational fisheries in Australia and South Africa (Griffiths 1997; Henry and Lyle 2003; Chapter 4), and has been in aquaculture production in South Australia since 2002 (96.5 t in 2007-08). Significant concerns regarding sustainability of fisheries for this species have been made in Australia and South Africa (Griffiths 1996; Griffiths 1997; Silberschneider et al. 2008; Chapter 4). Stock enhancement of populations of *A. japonicus* with hatchery produced fish has been done in Australia (Fielder and Bardsley 1999), and is being considered in South Africa (Palmer and Snowball 2009). The ability to differentiate hatchery reared *A. japonicus* from wild stocks is important because estimating the survival and contribution of hatchery reared fish to the fishery is a requirement of responsible stock enhancement (Taylor et al. 2005).

Attempts to understand the population-structure of *A. japonicus* in Australia have included studies based on tag-recapture (Hall 1986; Silberschneider and Gray 2008) and genetic techniques (Dixon 1988; Dixon 1990; Archangi 2008; Farmer 2008). Tagging of juveniles in estuarine habitat resulted in low recapture rates with few individuals moving more than 10 km (Hall 1986; Silberschneider and Gray 2008). This was likely due to high site fidelity of juveniles which tend to remain in the natal estuary (Griffiths 1996; Taylor et al. 2006). A recent study of genetic variability based on mitochondrial DNA and DNA microsatellites suggested that *A. japonicus* in Australia is a single panmictic population, but this was based mostly on samples from the east coast of Australia (Archangi 2008). A contemporary study using mitochondrial DNA indicated that the genetic composition of populations on the west and east coasts of Australia were significantly different, however this study mostly considered samples from Western Australia (Farmer 2008). However, high variability in life-history traits of *A. japonicus* between New South Wales, South Australia and Western Australia suggests sub-structuring of the population in Australia (Hall 1986; Farmer 2008; Silberschneider and Gray 2008; Chapter 4).

In this study the stock structure of *A. japonicus* in South Australia was investigated by comparing trace element signatures and shape characteristics (otolith meristics and overall shape) of sagittal otoliths. Specific objectives were to investigate population structure of *A. japonicus* in South Australia by examining variation (i) in elemental composition of otoliths among three geographic regions, and (ii) otolith morphology using meristic and elliptic Fourier analysis among these same regions. Further, we examine (iii) the potential for using otolith shape descriptors to allocate individuals of unknown origin to aquaculture or wild populations.

Materials and methods

Study population

Argyrosomus japonicus is widely distributed in estuaries and nearshore coastal waters in the Pacific and Indian Oceans (Griffiths and Heemstra 1995). Known as mulloway in Australia, this species occurs in sub-tropical and temperate waters south of the Burnett River (25°20 S) in southern Queensland and North West Cape (21°53 S) in Western Australia (Figure 1) (Kailola et al. 1993; Silberschneider and Gray 2008; Silberschneider et al. 2008).

Within Australia the largest commercial fishery for this species occurs in South Australia with further catches from Western Australia and New South Wales. Within South Australia most commercial and recreational catches are taken on the eastern coast with smaller catches from central and western coasts (Ferguson and Ward 2003; Henry and Lyle 2003; Jones 2009). Aquaculture production of this species occurs in central South Australia.

Sample collection

Otoliths of *A. japonicus* were collected from commercial and recreational fishers on western, central and eastern coasts of South Australia. Collection was from the point of landing and from the Adelaide fish market. All fish were measured (total length, TL). Sizes ranged from 790 to 1350 mm TL. Sagittal otoliths were extracted using fine forceps, cleaned, weighed and stored in plastic bags.

Elemental chemistry of otoliths

To determine if the structure of the stock of *A. japonicus* in South Australia differed, concentrations of trace elements in otoliths were compared among geographic regions. Prior to analysis otoliths were cleaned in de-ionised water, ensuring no tissue remained, then allowed to dry in a laminar flow cabinet for 24 hours. Otoliths were embedded in EpoFix® resin (Struers, Denmark) that was spiked with Indium (In) at 30 μg g^{-1} to provide a resin indicator. A transverse section of approximately 500 μm thickness was taken through the primordium of each otolith using a low-speed Isomet saw (Buehler Ltd), continuously lubricated with Milli-Q water. Sections were polished with 9-μm aluminium oxide lapping film followed by 3-μm lapping film. Because otoliths of *A. japonicus* are particularly brittle, polishing was minimised to avoid damage to the edges.

The sections were rinsed with Milli-Q water between polishing stages and then dried. Sections were mounted onto glass microscope slides using indium-spiked (30 μg g) Crystal Bond 509® thermoplastic cement. Sample slides were then cleaned in an ultrasonic water bath for one minute to remove any surface impurities, dried in a laminar flow hood for 24 hours and stored individually in sealed plastic bags until analysed.

Concentrations of elements in the otoliths were determined using a New Wave UP-213 high-performance (Nd:YAG) ultraviolet laser microprobe with an ablation spot size of 30 μm. Ablations occurred in a sealed chamber, with sample gases extracted to an Agilent 7500cs inductively coupled plasma mass spectrometer (ICP-MS). During analysis the chamber was flooded with a helium atmosphere (0.82 L.min^{-1}). Prior to activation of the laser the elemental isotopes were measured in the blank sample gas for ~30 s. This allowed the sample chamber to stabilise and provided average background counts of the analysed isotopes, which were then subtracted from the sample counts for each ablation. Between samples the ablation chamber was purged with Argon gas for 2 min at a rate of 0.1 L.min^{-1} to eliminate background gases that may have contained contaminants after each sample.

The ablation site was approximately 35 μm from the edge of the otolith because the otolith edge provides the best estimate of the immediate environment (Campana et al. 2000). The site was adjacent to the sulcus and located by comparing features on a pre-prepared, high resolution photograph of the sample and an image of the ablation chamber viewed via a microscope objective lens connected to a video camera and computer monitor.

The elemental isotopes ^{23}Na, ^{24}Mg, ^{88}Sr, ^{115}In, ^{138}Ba were chosen for analysis, as well as ^{43}Ca which provided the internal standard. To allow for instrument drift, samples were analysed in random order and a reference standard (National Institute of Standards and Technology, NIST 612) was analysed after every 12 samples. All analyses involved collecting background data for 30 seconds, followed by collection of otolith data for 120 seconds. Mean detection limits determined from blank ablations for each element exceeded the detection limit of the instrument by three times for all samples (^{23}Na, 0.10; ^{24}Mg, 0.24; ^{43}Ca, 307.80; ^{88}Sr, 0.11; ^{115}In, 6.34; and ^{188}Ba, 0.10 ppm). Precision (CV) was <0.5 for all elements (^{23}Na, 0.019; ^{24}Mg, 0.016 ^{43}Ca, 0.000; ^{88}Sr, 0.015; ^{115}In, 0.012 and ^{188}Ba, 0.16 ppm) (Campana et al. 1997).

Background measurements were subtracted and mass count data converted to concentrations (ppm) using Glitter software (http://www.glitter-gemoc.com/). Elemental concentrations were

then converted to molar concentrations and standardised to ^{43}Ca. All further data analyses were carried out on the element:Ca data.

Otolith Shape

Image acquisition

Whole otoliths were digitised using a high resolution camera mounted on a Leica MZ16 binocular microscope at 5x magnification with transmitted light. Otoliths were positioned in the microscope stage with the convex side up, showing the sulcal groove, and with the rostrum pointing to the left. The left otolith was used in most cases but when damaged the right otolith was substituted. In these cases the image was re-orientated to the left by flipping the image horizontally using standard image analysis techniques. Image capture was done using Leica Application suite 2.7.0 R1.

Shape descriptors

To determine if the shape of otoliths varied between regions the shape of each otolith was described using; (i) elliptical Fourier analysis (EFA), and (ii) morphometric indices of otolith shape. Elliptical Fourier analysis was done by estimating coefficients (EFCs) for each otolith using the program Shape v. 1.3 to perform elliptical Fourier analysis (EFA) on otolith images (Iwata and Ukai 2002). This method is considered more flexible and gives more consistent results for otolith shape analysis than the commonly used Fourier transform (Mérigot et al. 2007). EFA fits a closed curve to an ordered set of data points in a two-dimensional plane, then decomposes the contour into a sum of harmonically related ellipses (Kuhl and Giardina 1982). The number of harmonics required was estimated from the average Fourier power spectrum. The Fourier power of a harmonic is proportional to its amplitude and provides a measure of the amount of shape information described by this harmonic (Pothin et al. 2006; Mérigot et al. 2007). For the nth harmonic, the Fourier power (PF_n) is expressed as:

$$PF_n = (A_n^2 + B_n^2 + C_n^2 + D_n^2)/2$$

$$FP_c = \sum_1^n FP_n$$

where A_n, B_n, C_n, and D_n are the Fourier coefficients of the nth harmonic. Then the cumulated power percentage (PF_c) is defined by: To determine the appropriate number of Fourier descriptors, 30 otoliths were randomly selected and both the Fourier series and the Fourier power were calculated (Pothin et al. 2006). The first seven harmonics reached >99.99% of the

cumulative power, indicating that the otolith shape could be adequately summarised by 28 Fourier coefficients (Figure 2) harmonics x 4 coefficients. The elliptical Fourier harmonics for each otolith were normalised to the first harmonic in the SHAPE program and were thus invariant to otolith size (Kuhl and Giardina 1982). As a consequence of the normalisation the first three coefficients (A_1, A_2, and A_3) were constant for all outlines, reducing the number of Fourier descriptors to 25.

Morphometric measurements were made from each otolith using the program ImageJ v. 1.43f (Rasband 2009). Measurements of perimeter p, area a, and length were used to calculate five otolith shape indices (Table 1). Form factor, estimates surface area irregularity, and has a value of 1 when it is a perfect circle and values between 0 and 1 when it is irregular; roundness and circularity compare otolith shape to a perfect circle at a value of 1 and 4*Pi (~12.6) respectively; rectangularity describes length and width variations with respect to area; and ellipticity indicates proportionality of the axes (Tuset et al. 2003).

The effect of otolith length on the magnitude of each morphometric index was determined using linear regression. Where otolith morphometric indices were significantly correlated with otolith length, they were corrected for otolith length by removing the common between group slope (b) using the following formula:

$$V_{adjusted} = V_i - b*\text{Otolith Length}$$

Where V_i is the variable analysed (Cardinale et al. 2004). The effect of otolith length was removed from the variables instead of fish length because: (i) otolith width and total length were closely related (LR: $r^2 = 0.84$, $F_{1,145} = 785.540$, $P < 0.001$); and (ii) otolith width is unaffected by inter-sample differences in shrinkage and distortion (Campana and Casselman 1993). Where significant relationships existed between morphometric indices the redundant indices were excluded from the analysis.

Examination of otolith shape descriptors to discriminate fish from aquaculture and wild sources

The utility of otolith shape for classifying *A. japonicus* individuals as having originated from wild or aquaculture populations was examined for several sites within the central coast. Otoliths of small juveniles (<400 mm TL) from recreational catches and trawl by-catch taken in Spencer Gulf (n=14) were investigated. These were suspected to have originated from an aquaculture farm also located in Spencer Gulf since fish of this size are rarely observed in the area and were therefore treated as having unknown origin. The unknown groups of otoliths were compared

with three groups of otoliths of known origin. The first two groups comprised juvenile *A. japonicus* from aquaculture farms in Gulf St Vincent (Farm 1, n=15, 265-354 mm TL) and Spencer Gulf (Farm 2, n=15, 284-390 mm TL), with both groups from spawning of the same broodstock (n=12) in 2002 and 2003. The third group comprised wild caught fish from Gulf St Vincent (n=8, 785 - 1154 mm TL).

Statistical analyses

Elemental composition of otoliths

Variables were transformed to a Euclidean distance dissimilarity matrix. Canonical Analysis of Principal Coordinates (CAP) was then used to ordinate the matrices (Anderson and Willis 2003). Permutational multivariate analysis of variance (PERMANOVA) (Anderson et al. 2005) was used as a probabilistic test to assess differences in otolith shape between regions. For all CAP and PERMANOVA tests we used 4999 unrestricted random permutations of the raw data (Anderson 2001).

To determine the level of misclassification between sampling regions, the method of 'leave-one-out' (Anderson and Willis 2003) was applied to variables in canonical space. Because CAP analysis is a constrained ordination with region known *a priori*, the 'leave-one-out' approach gives a reasonable and unbiased measure of the distinctness of groups in multivariate space (Anderson and Willis 2003).

Otolith shape

In the analysis comparing otolith shape among geographic regions, several morphometric indices were excluded from further analysis: (i) indices derived from combinations of area and perimeter i.e. coefficient of form and circularity, had bimodal distributions and thus could not be adjusted for otolith length; and (ii) ellipticity was closely related to roundness. Thus, the morphometric indices roundness and rectangularity were retained for further analysis. For the analysis comparing otolith shape among aquaculture, wild and unknown groups, the morphometric indices roundness, rectangularity and circularity were included in the analysis.

Otolith shape (EFA, morphometric indices, and both combined) was compared among regions using CAP and PERMANOVA. This followed the same approach as that used in the analysis of elemental composition of otoliths. CAP of otoliths of unknown origin was then used to determine which group (aquaculture vs wild) the unknown otoliths were from.
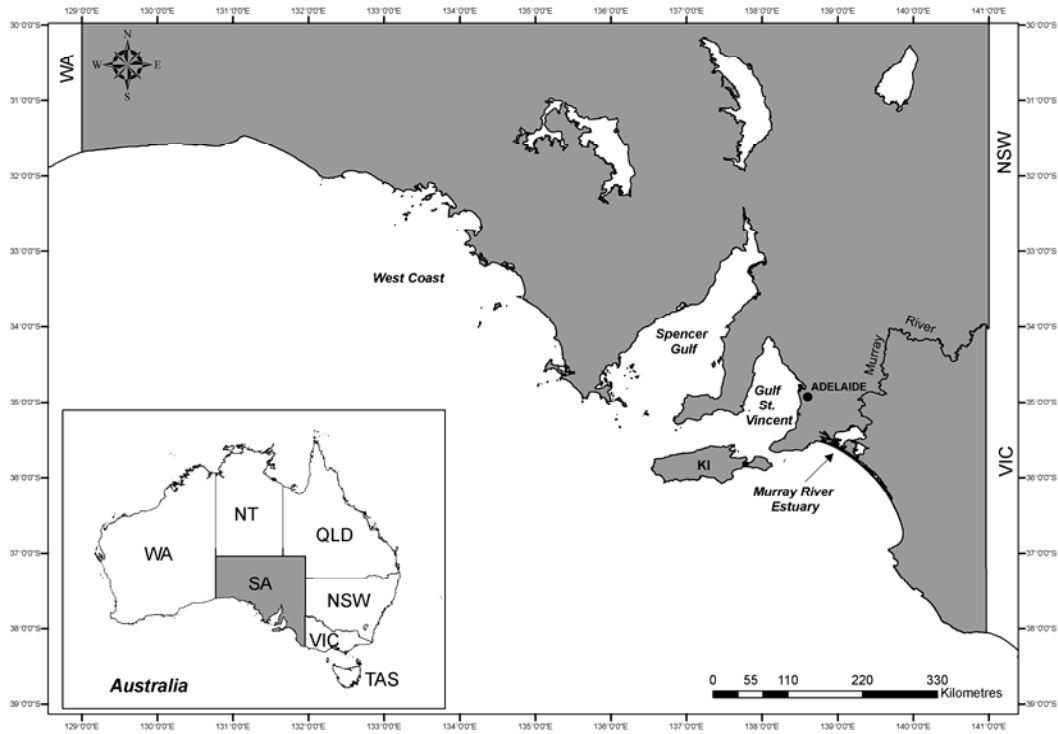
Figure 1. Map of Australia (inset) showing western, central (Spencer Gulf and Gulf St Vincent) and eastern coasts (east of the Murray River Mouth) of South Australia.
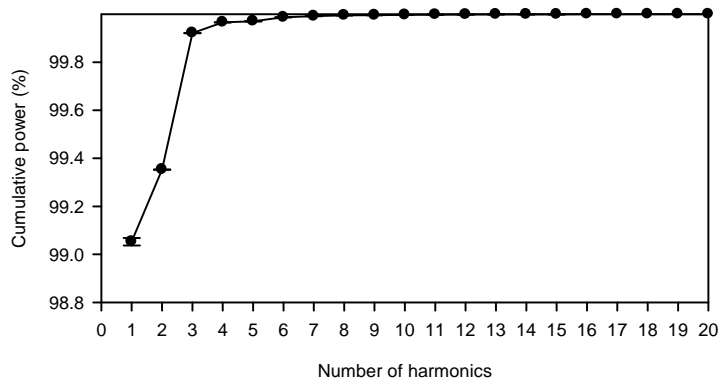
Figure 2. Variations in the mean cumulative Fourier power in relation to the number of harmonics used to describe the shape of *Argyrosomus japonicus* otoliths. Vertical bars indicate minimum and maximum range of the cumulative power.

Table 1. Otolith morphometric indices (from Tuset et al. 2003).

Morphometric	Formulae
Form factor	$(4\pi A)/P^2$
Roundness	$(4 A)/ (\pi L^2)$
Circularity	P^2 / A
Rectangularity	$A / (L \times W)$
Ellipticity	$(L - W) / (L + W)$

Results

Comparison of elemental composition of otoliths among regions

Single element analysis

Concentrations of Sr:Ca differed among regions (Pseudo- $F_{2,69}$= 64.305, P = 0.0002) with the highest in otoliths from the eastern coast (Figure 3). Pairwise comparison showed that the concentration of Sr:Ca differed between western and eastern (P = 0.0002), and central and eastern coasts (P = 0.0002), but was similar between western and central (P = 0.0858) coasts.

Concentrations of Ba:Ca differed among regions (Figure 3A, Pseudo- $F_{2,69}$ = 52.878, P = 0.0002). Pairwise comparison showed that concentrations of Ba:Ca were different between western and central (P = 0.0002), and western and eastern (P = 0.0002) coasts but similar between central and eastern coasts (P = 0.1276). Concentrations of Ba:Ca in otoliths from the eastern coast ranged from 0.0005 to 0.0020 μmol.mol^{-1}, and were up to 4 times the levels observed in otoliths from the western coast (Figure 3A). Similar concentrations of Mg:Ca (Pseudo- $F_{2,69}$ = 0.7299, P = 0. 4980) and Na:Ca (Pseudo-$F_{2,69}$ = 1.1227, P = 0. 3254) occurred among all regions (Figures 3B, C).

Multi-elemental analysis

Multi-elemental concentrations of otoliths differed among regions (Table 2A). Pairwise comparisons showed significant differences occurred between western and eastern, and central and eastern coasts, but not between western and central coast, similar to the trend observed for concentrations of Sr:Ca.

Fish from the three regions could be classified to their collection location with a high degree of accuracy (Figure 4, Table 3A, 94% correct classification). Classification success between regions was 100% for western and central coasts and 87% for eastern South Australia. The lower classification success for otoliths from the eastern coast group was due to misclassification of three individuals to the western, and one to the central coast.
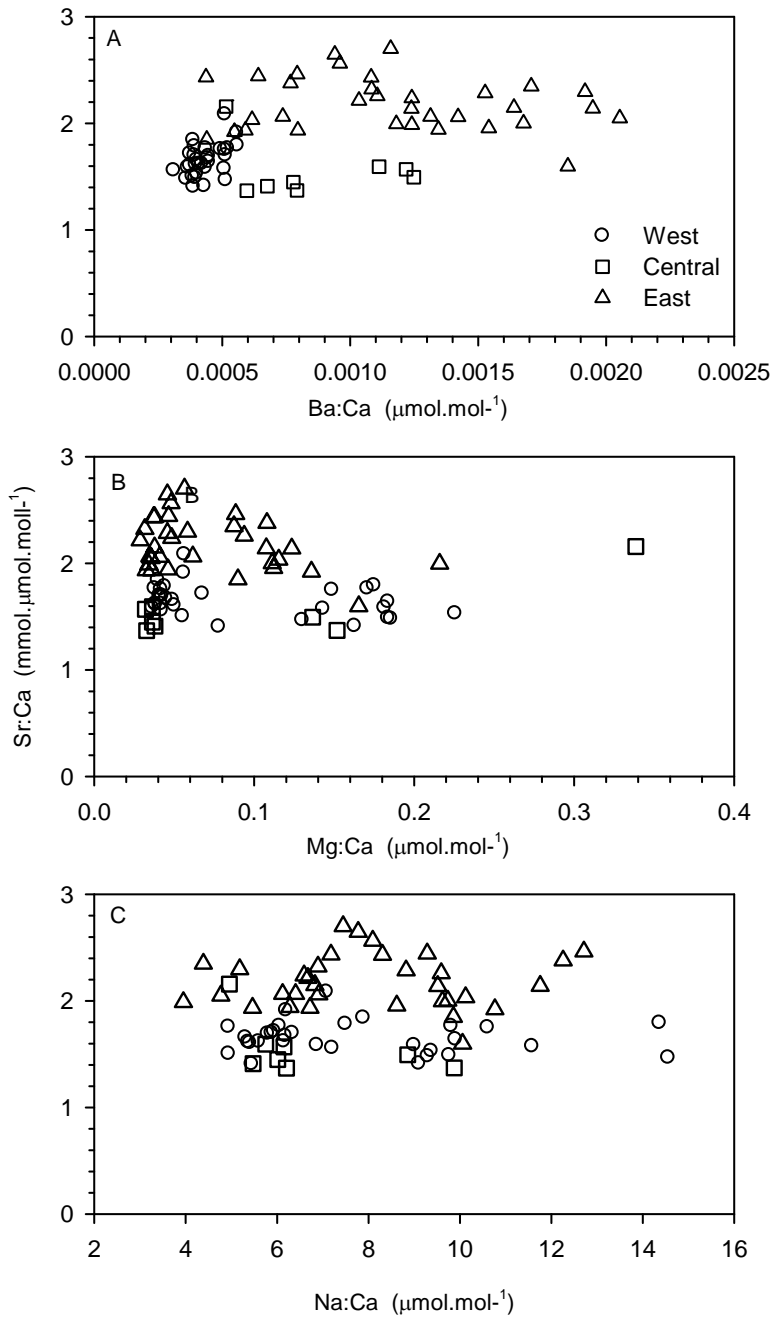
Figure 3. Elemental concentration of Sr:Ca vs (A) Ba:Ca, (B) Mg:Ca, and (C) Na:Ca trace element concentrations in edges of *Argyrosomus japonicus* otoliths from three regions of South Australia.

Table 2. PERMANOVA comparison of otoliths of *Argyrosomus japonicus* from three regions in South Australia. Shown are overall comparison among regions for (A) elemental composition of otoliths and (B) combined otolith shape (combined elliptical Fourier coefficients, and morphometric indices, roundness and rectangularity) including pairwise comparisons between regions.

Variables	Regions	df	Pseudo-*F*	*p*
(A) Trace element concentration	All	2,64	5.8045	0.0020*
	central, eastern	37		0.0056*
	central, western	33		0.4416
	eastern, western	58		0.0030*
(B) Combined Fourier coefficients and Morphometric Indices	All	2, 116	9.8962	0.0001**
	central, eastern	76		0.1480
	central, western	71		0.0001**
	eastern, western	85		0.0001**
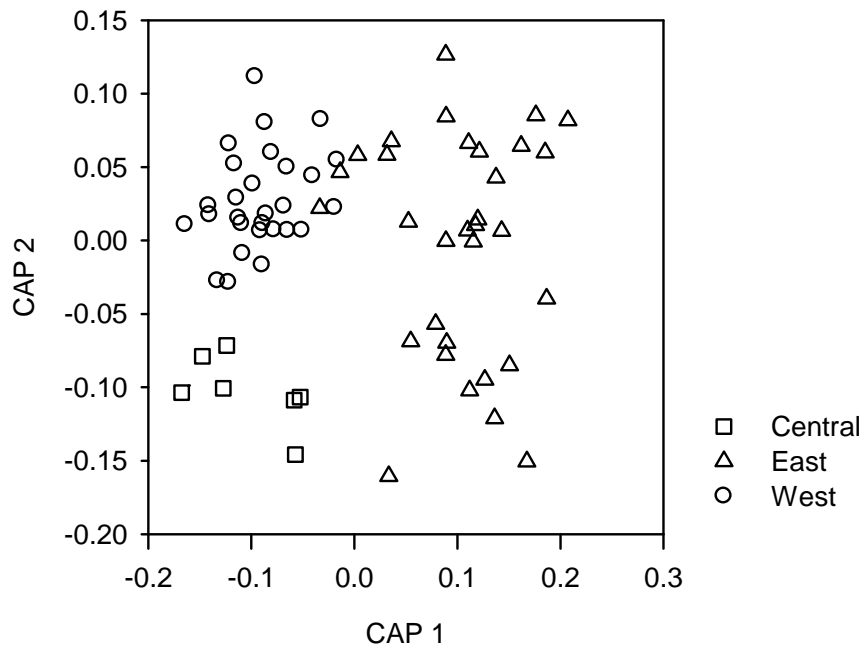
Figure 4. CAP analysis of mean concentrations of trace elements in the edges of otoliths of *Argyrosomus japonicus* from western, central and eastern coasts of South Australia.

Table 3. Canonical analysis of principal co-ordinates (CAP) examining differences in (A) elemental composition, and (B) shape of otoliths of *Argyrosomus japonicus* among western, central and eastern regions in South Australia. %Var: percentage of total variation explained by the first (m) principal co-ordinate axes; allocation success is the percentage of fish correctly allocated to their respective group.

	Factor	m	%Var	Allocation success				p
				western	central	eastern	Total	
(A)	Trace element	4	37.31	100.00	100.00	87.50	94.03	<0.001
(B)	Shape (elliptical Fourier analysis)	13	45.38	78.05	59.37	69.56	69.75	<0.001
	Morphometric indices	1	-	65.85	53.3	26.08	47.05	<0.001
	Shape and morphometric indices	26	47.90	85.36	50.00	67.39	68.91	<0.001

Comparison of otolith shape and morphometric indices among regions

Shape of otoliths based on elliptical Fourier coefficients did not differ among years for otoliths from central South Australia (PERMANOVA, $F_{4,13} = 1.3272$, $p = 0.21$; CAP, Figure 5). Otoliths from this region were subsequently pooled to increase the sample size (n = 14).

Significant differences in otolith shape (EFCs) among regions were found (PERMANOVA, Table 2B).   Pairwise comparison detected differences between west and central coasts and west and east coasts ($p < 0.0001$), but not between central and east coasts ($p = 0.148$). Otoliths were successfully allocated to regions of capture on the basis of shape variables (EFC's) (Table 3B). Combined shape (EFC) and morphometric indices explained a higher proportion of the variance. Overall classification success (CAP, Table 3B, Figure 6) was 69%.

To determine the nature of the differences in otolith shape between regions PCO analysis of morphological indices was done. Two PCO axes explained 92.6% of the variance with roundness explaining most of the variance (Pearson correlation coefficient, PCC: $r = 0.76$), compared to circularity and ($r = 0.459$) and rectangularity ($r = 0.276$). Mean roundness (where 1 is a perfect circle) of otoliths from the western coast (0.60 ±0.003) was higher than for central (0.56 ±0.002) and eastern coasts (0.57, ±0.004). Similarly, rectangularity was highest for otoliths from the western coast (0.83 ±0.003), compared with central (0.81 ±0.002) and eastern coasts (0.82 ±0.003), but similar for otoliths from east and central coasts.
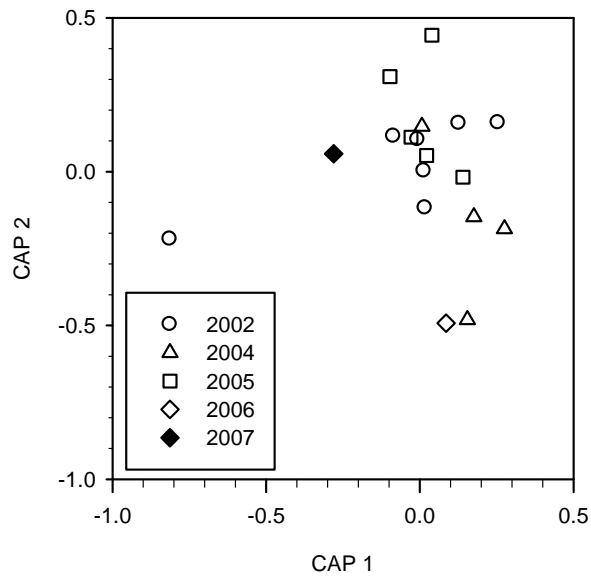
Figure 5. CAP analysis of temporal stability of otolith shape (elliptic Fourier coefficients) for *Argyrosomus japonicus* from the central coast of South Australia.
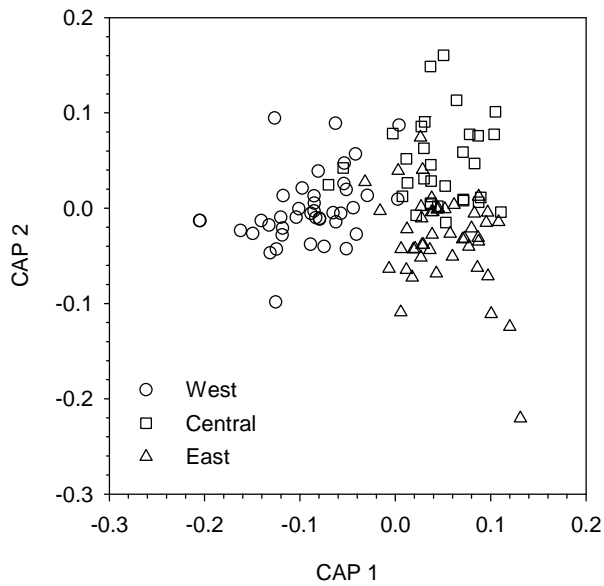
Figure 6. CAP analysis of otolith shape (combined elliptic Fourier coefficients and morphometric indices) from *Argyrosomus japonicus* from western, central and eastern South Australia.

Allocation of otoliths of unknown origin to known groups using shape and morphometric indices

Significant differences between the two aquaculture farms and wild-caught fish from within the central coast were found for shape, morphometric indices and both combined (PERMANOVA: $F_{3,45} = 8.4109$, $P = 0.0003$). Pairwise comparison showed that no detectable difference occurred between the unknown group and Farm 2 ($p = 0.1852$) or the wild group from Gulf St Vincent ($p = 0.1847$). Differences were detected between the unknown group and Farm 1 ($P = 0.0029$).

The collection location of fish could be accurately predicted using shape (EFA; 100% classification success) and a combination of shape and morphometrics (100% classification success (Table 4; Figure 7). All otoliths of unknown origin were allocated to the aquaculture farms. Ten (78.57%) unknown otoliths were allocated to Farm 1 in Spencer Gulf, with the remainder (21.43%) allocated to the group from Farm 2 in Gulf St Vincent (Table 4; Figure 7). When morphometrics only were used classification success was 47%.

TABLE 4. Canonical analysis of principal co-ordinates (CAP) examining the differences in otolith shape of *Argyrosomus japonicus* between three known groups (Farm 1, Farm 2 and wild caught) and one unknown group. %Var: percentage of total variation explained by the first (m) principal co-ordinate axes; allocation success is the percentage of points correctly allocated to their respective group.

Factor	m	%Var	Allocation success				p
			Farm 1 (n=15)	Farm 2 (n=15)	Wild (n=8)	Unknown (n=14)	
Shape (EFA)	14	100.00	100.00	100.00	100.00	Farm 1 (4) Farm 2 (10)	0.0001**
Morphometric indices	1	80.00	73.33	100.00	52.17	Farm 1 (4) Farm 2 (10)	0.0001**
Shape and morphometric indices	19	100.00	100.00	100.00	100.00	Farm 1 (3) Farm 2 (11)	0.0001**
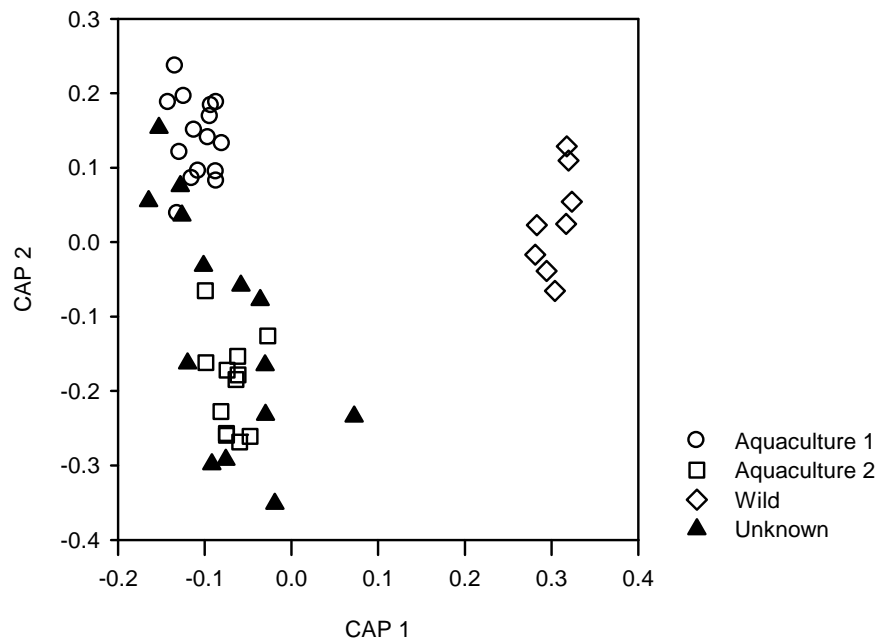
Figure 7. Comparison of otolith shape (combined elliptic Fourier coefficients and morphometric indices) among *Argyrosomus japonicus* from one sample of unknown origin from Spencer Gulf, and three samples of known origin (Aquaculture Farm 1 (Gulf St Vincent), Aquaculture Farm 2 (Spencer Gulf), and Wild (Gulf St Vincent).

Discussion

Otolith morphometrics and chemistry indicated that two separate sub-populations of *A. japonicus* occur in South Australia, with the possibility of further sub-structuring. Differences in life-history characteristics (Chapter 4) and growth rates (Chapter 4) have also been found between western and eastern South Australia.

Comparison of elemental composition of otoliths among regions

Differences among regions were mainly due to differences in concentrations of Ba:Ca and Sr:Ca. Concentrations of both elements were highest in eastern South Australia, and similar between central and western coasts. The major influence of Sr or Ba uptake into otoliths is likely to be the ambient concentration of these elements in sea water (Bath et al. 2003; Elsdon and Gillanders 2003b). Generally, higher Sr:Ca concentrations are associated with higher salinity water, and higher Ba:Ca concentrations with freshwater (Campana 1999; Elsdon and Gillanders 2003a; Elsdon et al. 2008). However, other factors that may also influence Sr:Ca and Ba:Ca ratios are temperature, salinity and biological factors such as growth rate (Campana 1999). Higher Sr:Ca ratios in eastern South Australia may reflect higher ambient Sr concentrations, although at present there are no data available for Sr and Ba concentrations in South Australian waters.

Concentrations of Ba:Ca were high in otoliths from the eastern coast, intermediate in those from the central coast and low in those from the western coast. Further sub-structuring of the population within eastern South Australia may also occur because concentrations of Ba:Ca in otoliths from this region were distributed bimodally with approximately 50% of individuals (n ~ 15), having Ba:Ca levels double those of other fish from the same region.

Estuaries may provide potential sources of Ba because this element is closely bound to fluvial sediments (Li and Chan 1979; Coffey et al. 1997). Lower salinities such as occur in estuaries may also increase bioavailability of Ba (Turner et al. 1981). In eastern South Australia *A. japonicus* are associated with the Murray River which is the largest estuary in South Australia and provides protected estuarine habitat for juveniles up to 5 years old (Chapter 4). Additionally, spawning/feeding aggregations of sexually mature adults occur near the mouth of the River during spring-summer particularly in years of high freshwater outflow (Chapter 4). High levels of Ba:Ca observed in otoliths of *A. japonicus* from eastern South Australia may be due to association with the Murray River estuary compared to the western coast which has no estuaries and little freshwater input. The source of intermediate concentrations of Ba observed in otoliths from the

central coast may be freshwater outflows from several small rivers that enter the sea along that section of coast.

An alternative source of Ba may be from upwelling of oceanic waters (Lea et al. 1989). Several annual upwellings of oceanic water occur along eastern and western coasts of South Australia (Ward and Staunton-Smith 2002; Kämpf et al. 2004). These cool upwelled waters are relatively high in nutrients and may also support higher concentrations of Ba and/or Sr (Fowler et al. 2005). Upwelled waters may reach nearshore environments in eastern South Australia where the continental shelf is narrow (<20km), but this is less likely on the western coast where it is much wider (>60 km). Consequently, the Bonney upwelling may have contributed to the high concentrations of Ba:Ca observed in otoliths from eastern South Australia.

Comparison of otolith shape among regions

Multivariate analysis of otolith shape detected differences between western and eastern coasts and generally supported results from the analysis of elemental composition of otoliths. Morphological data have been combined with elliptical Fourier descriptors in multivariate analyses (Tracey et al. 2006) and traditional Fourier series (Jónsdóttir et al. 2006; Petursdottir et al. 2006; Burke and King 2008) in an attempt to improve discriminating power. We found morphometric indices to be less useful than EFC's for discriminating between groups of otoliths partly because: (i) several indices were bimodally distributed and thus could not be normalised for allometric variation between samples; and (ii) because several indices were correlated thus precluding their use in the multivariate analysis. Although we found that allocation success was marginally improved (δ^2 and trace) by inclusion of morphometric indices with the EFC's in multivariate analyses they were more valuable for explaining which characteristics of the otolith varied between groups.

The morphometric indices roundness and rectangularity had high values for otoliths from the western South Australia but similarly low values among otoliths from eastern and central coasts. Such intra-specific differences in otolith shape have been attributed to differences in growth rate of cod (*Gadus morhua*) (Campana and Casselman 1993) and this is likely to be the case for *A. japonicus* in South Australia because growth rates differ between western and eastern coasts (Chapter 4).

Otolith shape characteristics also provided a tool for investigating the origin of *A. japonicus* that had been collected from Spencer Gulf and were suspected to have originated from a nearby

aquaculture farm. All otoliths from unknown origin were allocated to one of two aquaculture farms with most allocated to an aquaculture farm close to the collection site in Spencer Gulf. No fish were allocated to the wild caught group.

Multi-method approach to comparison of otoliths among regions

Few studies have combined otolith shape and otolith chemistry to investigate stock structure (Turan 2006), but there were several advantages to this approach. High classification success (100%) using elemental concentrations of otoliths indicated that *A. japonicus* in western and eastern South Australia were geographically separate during the recent life of the individual fish. Classification success was lower for otolith shape compared to elemental composition. However classification based on otolith shape has the advantage of indicating differences that may have occurred over the whole life of the individual fish that are unaffected by short-term changes in fish condition (Campana and Casselman 1993). This implies that significant periods of the fishes' lives were spent in different environments (Campana 1999). This was further supported by temporal stability of shape of otoliths from the central coast, as has been reported for other species e.g. *Gadus morhua* (Jónsdóttir et al. 2006). Assays of whole dissolved otoliths may also provide an indication of differences that have occurred over the whole life of an individual fish, or alternatively profiles across the entire otoliths may be analysed (Fowler et al. 2005; Elsdon et al. 2008; Steer et al. 2009). The latter may be subject to constraints associated with the physical size of the otoliths especially in a species such as *A. japonicus* which have large otoliths.

While differences were consistently detected in elemental composition and shape of otoliths of *A. japonicus* between western and eastern South Australia the relationship between these regions and the central coast was less clear. Multivariate analysis of elemental composition and univariate analysis of Sr:Ca suggested that *A. japonicus* from western and central coasts may be similar, while analyses of Ba:Ca concentration and otolith shape suggested that central and eastern coasts may be similar. However, *A. japonicus* from central and eastern parts of South Australia may be most similar because: (i) high concentrations of Ba:Ca occur in otoliths from the eastern coast and intermediate concentrations in those from the central coast may reflect terrestrial input from rivers and streams that do not occur in the western coast; and (ii) differences in otolith shape may be influenced by differences in growth rates (Campana and Casselman 1993) and these two regions are relatively close geographically, thus growth rates may be similar.

Overall, high classification success of trace element composition of otoliths, combined with differences in otolith shape and life-history parameters (Chapter 4), suggest that at least two populations of *A. japonicus* occur in South Australia with the possibility of further population sub-structuring.

Implications for management

The existence of two distinct groups of *A. japonicus* within South Australia has management implications. Given the clear differences in otolith elemental concentrations and shape, combined with differences in growth rates (Chapter 5) between western and eastern coasts, it is likely that these two regions are replenished by independent recruitment events and are relatively self sustaining.

Populations of *A. japonicus* are usually associated with estuaries (Griffiths 1996) and the population of *A. japonicus* in eastern South Australia is thought to be associated with the Murray River estuary (Chapter 4). The population located in western South Australia likely utilises alternative, near-shore reef habitat for protection of juveniles. Life-history profiles of elemental concentrations across the otoliths of juvenile and adult *A. japonicus* from both regions may provide further information on habitat use for these populations. Molecular techniques such as microsatellite and mtDNA have been developed (Archangi 2008; Farmer 2008; Archangi et al. 2009). They may provide further information on the stability of the detected phenotypic variation and should be investigated as part of a study of this species across its range in Australia. Such a study is currently being undertaken (B Gillanders 2010, pers. comm.).

Separate management for western and eastern stocks of *A. japonicus* may be necessary. Large, late maturing species such as *A. japonicus* are vulnerable to the impacts of overexploitation. The eastern population of *A. japonicus* in South Australia, is environmentally limited (Chapter 4), has been overfished, and truncated age distributions suggest compromised egg production (Chapters 2, 5). The likelihood of recruitment to this population from western or central coasts of South Australia is low, and recruitment from other areas (e.g. eastern Australia) is unknown.

Conclusions

Lack of recognition of stock sub-structure can result in loss of stock richness, which in turn may lead to declines in genetic diversity and reproductive potential of the stock. In many cases, management units contain stock complexes or meta-populations with several spawning components as may be the case for *A. japonicus* in South Australia. Because spawning

components are typically difficult to define, emphasis on stock complexity is necessary if management is to comply with a 'precautionary approach' (Stephenson 1999). Such diversity may also contribute to the resilience of fish populations to the impacts of climate change (Brander 2010).

Although comparison of elemental composition of otoliths provided superior allocation success in this study, comparison of otolith shape using elliptical Fourier analysis provides a potential, cost-effective, and relatively quick tool for discrimination among populations and sub-populations of *A. japonicus* and other species. This tool also has potential to discriminate individuals that originate from hatcheries and to indicate the success of stock enhancement.

Aerial view of the mouth of the Murray River in 2002, prior to commencement of dredging
(Photograph courtesy of Murray Darling Basin Commission)

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Statement of authorship

In this chapter, Gregory Ferguson performed all analyses on the data. Timothy Ward, Michael Geddes, and Bronwyn Gillanders provided advice on the manuscript.

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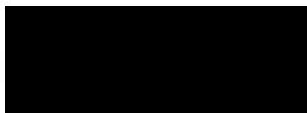
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Gregory J Ferguson (*Candidate*)

Certification that the statement of contribution is accurate and permission is given for the inclusion of the paper in the thesis.
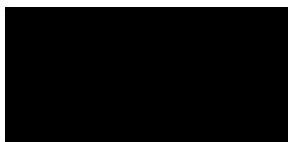
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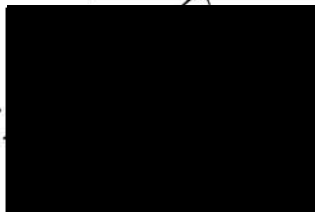
Michael C Geddes. ████████████ Date 30/9/2010

Chapter 4

Do recent age structures and historical catches of mulloway, *Argyrosomus japonicus* (Temminck & Schlegel, 1843), reflect freshwater inflows in the remnant estuary of the Murray River, South Australia?

Abstract

Patterns of annual freshwater flows in the Murray River and recruitment of mulloway, *Argyrosomus japonicus* were reviewed in terms of recent age structures and historical catches and CPUE. Age distributions from the nearshore marine fishery were dominated by the 1993 age class which comprised 35% and 41% of 2001 and 2002 catches, respectively. In 1993 annual freshwater inflow was 2.5 times the 25 year average. Freshwater inflow explained 28% and 35% of the variability in year class strength in the nearshore marine fishery in 2001 and 2002, respectively.

Over 80% of the current South Australian commercial catch of *A. japonicus* comprises juveniles taken from the remnant estuary of the Murray River. Our results suggest that recent low levels of recruitment in South Australia's fisheries for *A. japonicus* may reflect low fresh water inflows since 1993. Since 2000, southern Australia has experienced the worst drought in recorded history and management strategies for this fishery must take into account effects of both environmental factors and fishing mortality on this vulnerable sciaenid population. We suggest that the population of *A. japonicus* located about the Murray River system is estuarine dependent, that the estuary provides important refuge for juveniles, and that strong year classes, or their absence, may be related to freshwater inflow to this environment. We also suggest that age distributions of this apex predator may provide an indicator of environmental health for the Murray River estuary.

Introduction

Estuarine habitats are critically important for many species, including waterbirds, fishes and invertebrates (Blaber 1980; Lenanton 1982; Beck et al. 2001). The abundance of several estuarine-associated finfish species has been shown to be associated with environmental flows of freshwater (see review in Gillanders and Kingsford 2002) with a number of examples from tropical Australian estuaries (Loneragan and Bunn 1999; Robbins et al. 2005; Meynecke et al. 2006).

The Murray-Darling is Australia's largest river system and, in terms of catchment area and length, ranks approximately twentieth amongst the worlds great rivers (Newman 2000). The original Murray River estuary covered 660 km^2, and included the brackish Lakes Alexandrina and Albert, and the north and south Coorong lagoons (Figure 1). In 1940, a series of barrages were constructed between the Lakes and lagoons, reducing the estuary to 11% of its original size. Since then, water abstraction for irrigation has reduced mean annual flow to 20% of natural levels (Thomas 1999). The extent and frequency of the natural spring floods has been reduced, and in some years they do not occur at all. The frequency with which flow ceases at the river mouth has increased from one year in twenty to approximately one year in two (Close 1990). From 2001 to 2008, the Murray River system has experienced the worst drought in recorded history.

The remnant estuary of the Murray River is recognised internationally as an important breeding and feeding ground for waterbirds, and supports significant populations of several species of fish and invertebrates. The Murray River ecosystem is the largest estuarine habitat in temperate Australia and is the geographical centre of the Australian distribution of mulloway, *Argyrosomus japonicus* (Perciformes: Sciaenidae), a species known to be associated with estuaries in Australia (Hall 1986; Lenanton and Potter 1987; Gray and McDonall 1993) and South Africa (Griffiths 1996).

Australia's largest commercial fishery for *A. japonicus* operates in the Murray River estuary and nearby marine environments (Ferguson and Ward 2003, Figure 1) and has two multi-species sectors: the Lakes and Coorong Fishery (LCF) and the Marine Scale Fishery (MSF). The LCF operates within, and adjacent to the Murray River estuary, while the MSF operates in nearshore marine waters along the entire South Australian coastline. Fishers in the LCF harvest >90% of the South Australian commercial catch of *A. japonicus* by using gill nets in two broad habitats: the estuarine environment of the Murray River and the nearshore marine environment near the

river-mouth. The minimum legal harvest size of *A. japonicus* within the Murray River estuary is 46 cm TL but is 75 cm TL in all other waters in South Australia. The catch from the estuarine fishery is comprised almost exclusively of juveniles whilst the catch from the nearshore marine fishery comprises reproductively mature and sub-mature adults that aggregate at the interface of the Murray River plume during the spring-summer (November to March) spawning period (Ferguson and Ward 2003). Overall, approximately >80% of the South Australian commercial catch of *A. japonicus* are juveniles (< 75 cm TL) harvested from the Murray River estuary (Ferguson and Ward 2003).
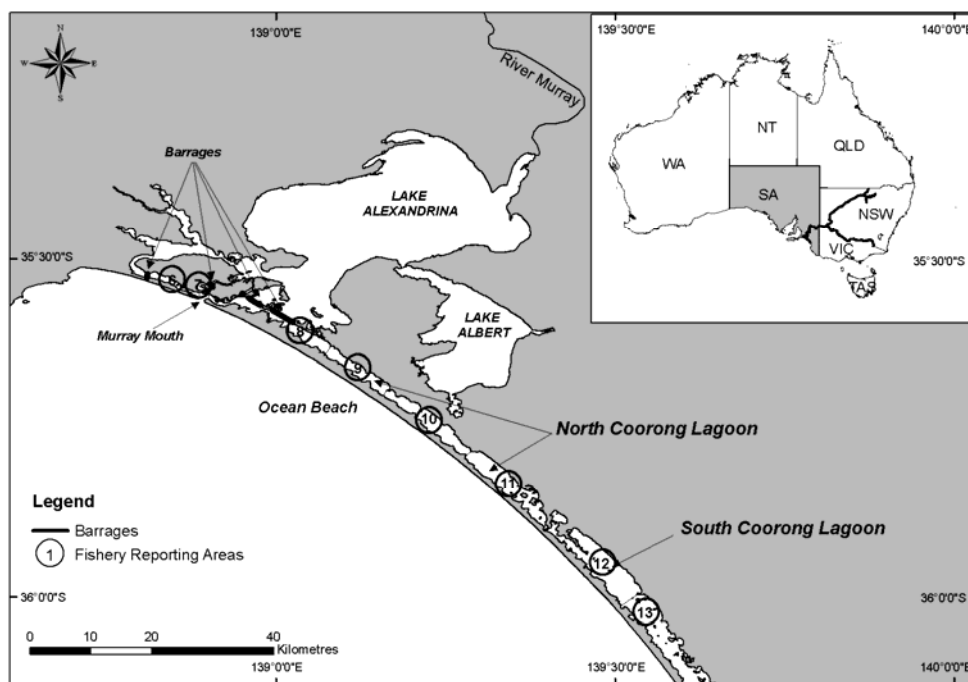
Figure 1. Map of Australia showing South Australia (SA) and the Murray River (inset) and the remnant estuary of the Murray River comprising the Coorong lagoons. Also shown are Lakes Alexandrina and Albert which were part of the original estuary. Circled numbers are fishery reporting areas.

This area is also fished by recreational line fishers who regard *A. japonicus* as an "icon" species. In 2000, recreational fishers harvested 38% of the combined commercial and recreational catch for South Australia (Henry and Lyle 2003). Like the commercial fishery, most of the catch is taken near the mouth of the Murray River, but in contrast to the commercial fishery, most of the

fish are adults from the pre-spawning/spawning aggregation in nearshore marine waters (Jones and Doonan 2005).

In southern Australia, fishers and others (Hall 1984; Anon. 1985; Hall 1986) have attributed variability in the annual catch of *A. japonicus* from the Murray River estuary, to variability in freshwater inflows. This study aims to examine relationships between: (i) relative age class strength of *A. japonicus* and freshwater inflow; (ii) historical catch and effort data from the fisheries for *A. japonicus* and freshwater inflow; and (iii) and the spatial distribution of *A. japonicus* within the Murray River estuary.

Materials and methods

Freshwater inflow and fishery data

In this chapter, "inflow" refers to freshwater from the Murray River entering the estuary through the barrage system. Estimated mean monthly freshwater inflow to the Murray River estuary were obtained from the regression based Murray hydrological model (MSM BIGMOD, Murray-Darling Basin Commission) for financial years beginning 1962 to 2005. Mean monthly inflow, in megalitres, were converted to SI units ($m^3.s^{-1}$). Highest monthly flows occurred during late spring (August-November) and summer (December-February) and were aggregated into financial years to align with the seasonality of catch and effort in the fishery. In this chapter all freshwater inflow, age and catch data are reported in financial years, i.e. 1962 represents the period from July 1, 1962 to June 30, 1963. Mean annual catch per unit effort (CPUE) was estimated for the estuarine and nearshore marine fisheries by dividing the annual catch by the annual effort in terms of fisher days. Data available for linear regression analyses are shown in Table 1.

Relationships between freshwater inflow to the Murray River estuary and age distributions

The LCF uses different types of gill nets to target *A. japonicus* in the estuary and the nearshore-marine environments. In the estuary, gill nets (>115 mm mesh) were set at dusk at 90 degrees to the edge of the main channels and retrieved at dawn. In the nearshore marine fishery, gill nets (245 mm mesh) were operated as "swinger nets". One end of the net was anchored to a motor vehicle (typically a 5 tonne truck) located on the beach. The free end of the net was then drifted 500-700 m out to sea and then drifted through the surf zone with the longshore current (for up to 5 km), while the anchoring vehicle moved along the beach. Recreational fishers targeted *A. japonicus* in the nearshore marine environment with rod and line.

Sagittal otoliths were sampled from commercial catches from the Murray River estuary and the nearshore marine environment adjacent to the River mouth in 2001 and 2002. This was done at either the point of landing or at the Adelaide Central Fish Market. In 2001 otoliths were also sampled from recreational catches taken in nearshore marine environment. The total length (TL) of each fish was measured and sagittae removed via a cut through the ventral ex-occipital region of the skull. Sagittae were cleaned, dried and stored in labelled plastic bags. The left sagitta from each pair was embedded in fibreglass resin, and a longitudinal section cut with a diamond blade mounted on a Gemmasta 6" (150 mm) bench top saw. The 500 μm thick section was cut so as to incorporate the otolith centre and the sections were mounted on glass microscope slides using Cyano-Acrylate glue. The mounts were examined using a Leica MZ-16 dissector microscope with incidental light and ages estimated from counts of opaque zones. Annual deposition of opaque rings has been validated for *A. japonicus* in South Africa and Western Australia (Griffiths and Hecht 1995; Farmer 2003). The pattern of deposition for otoliths from South Australia suggests that the opaque zone is deposited in November-December (Chapter 5).

All statistics were performed using SPSS 14.0® and all data tested for departure from normality prior to performing analyses where a normal distribution was assumed. Age distributions were compared using the Kolmogorov-Smirnov 2-sample goodness of fit test. Relative year class strength (percentage of total sample for year) from age distributions from the nearshore marine environment in 2000 and 2002 were related to freshwater inflow using linear regression.

Relationships between freshwater inflow to the Murray estuary and commercial catches and CPUE

Freshwater inflow to the Murray River estuary (independent variable) was compared with CPUE (kg.fisher day^{-1}) from the commercial estuarine and nearshore marine fisheries from 1984 to 2005 (dependent variable) using linear regression (α=0.05). Time lags were estimated from modes in the age structures and were respectively, 3 and 6 years for estuarine and nearshore-marine catches. These time lags were supported by cross correlations of flow data with a range of time lags (Pearson Correlation Coefficient, CC). CPUE from the estuarine and nearshore marine fisheries were also compared.

Relationship between freshwater inflow to the Murray estuary and spatial distribution of commercial catches

Catches from the Murray River estuary were harvested from 10 fishery reporting areas. Each fishery reporting area represented a 10-15 km long section of the lagoons which comprise the estuary (Figure 1). In years of higher freshwater inflow a salinity gradient is established along a greater proportion of the lagoons (Geddes 1987), therefore, because juveniles are known to prefer lower salinities, they would be expected to occur along an increasingly greater proportion of the lagoons in years of higher inflow. Thus, in the absence of longitudinal salinity data for each year, the number of spatial reporting areas from which catch was reported was used as an index of the proportion of the estuarine habitat that was used by juveniles.

Results

Relationship between freshwater inflow to the Murray estuary and age distributions

Age distributions of *A. japonicus* from the commercial estuarine fishery (2001, n=260; 2002, n = 91) had a modal age of 3 years and a range of 2 to 6 years (Figure 2). Age distributions from recreational (n=26) and commercial catches (n=73) from the nearshore marine fishery in 2001 had a statistically identical distribution (K-S D = 0.555, P = 0.918) and were combined (n = 99). The combined samples were dominated by the 1993 year class (8 year olds, 35% of sample, Figure 3A). Secondary modes occurred at year classes 1990 (11 years old, 8% of sample) and 1989 (12 years old, 4 % of sample). The dominant 1993 year class persisted in commercial samples from 2002 (41% of sample, 9 years old, Figure 3B). A smaller mode comprising the 1990 year class (8% of sample, 12 years old) was also present. The range of ages was 4 to 25 years in 2001 and 5 to 24 years in 2002 which represented year classes over 25 years from 1976 to 2001. The freshwater inflow in 1993 was 2.4 times greater than the 25 year average annual flow from 1976 to 2001 and in 1990 and 1989 was more than 2 times greater than the average for this period (Figure 3C).
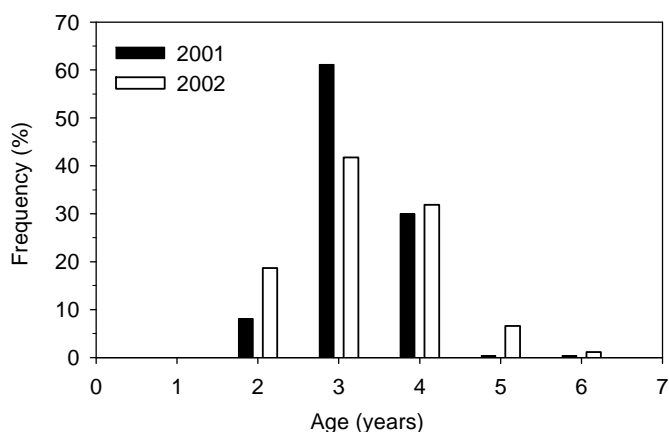
Figure 2. Age structures of *Argyrosomus japonicus* from the Murray River estuary from 2001 and 2002 (n = 351).

Freshwater inflow explained 28% of the variability in year class strength in combined samples from the nearshore environment in 2001 (LR (linear regression): $r^2=0.28$, $F_{1,24}=9.255$, $P = 0.006$) and 35 % in 2002 (LR: $r^2=0.35$, $F_{1,24}=13.002$, $P = 0.001$).
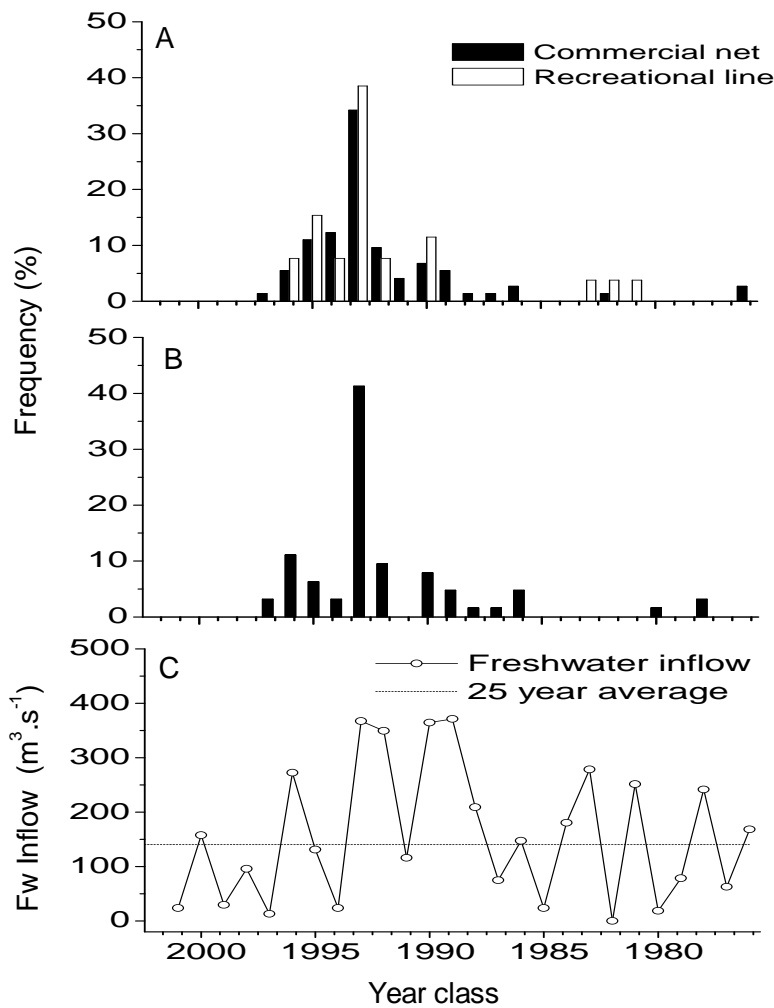
Figure 3. (A) Age distributions for commercial net (n=73) and recreational line catches (n=26) of *Argyrosomus japonicus* from the nearshore marine fishery in 2001, (B) commercial net catch from the nearshore marine fishery in 2002 (n=63), and (C) freshwater inflow to the Murray River estuary. Note that all x-axis scales are reversed to represent the year in which each year class was spawned.

Relationship between freshwater inflow to the Murray estuary and commercial catches and CPUE

South Australian total annual catches of *A. japonicus* from 1936 to 2005 are shown in relation to major hydrological events in Figure 4. The annual catch declined from a historic peak of 609 t in 1938 to 83% of this in 1941, coinciding with an 89% reduction in estuarine area associated with construction of the barrages in 1940. Catches continued to decline over the following 5 years, which included the 1943 drought, and in 1945 the catch was approximately 5% of the 1938 level. During the following 30 years three peaks in catches occurred (1957, 1966 and 1976) each preceded by a significant flood event (1955, 1963 and 1973 to 1974).
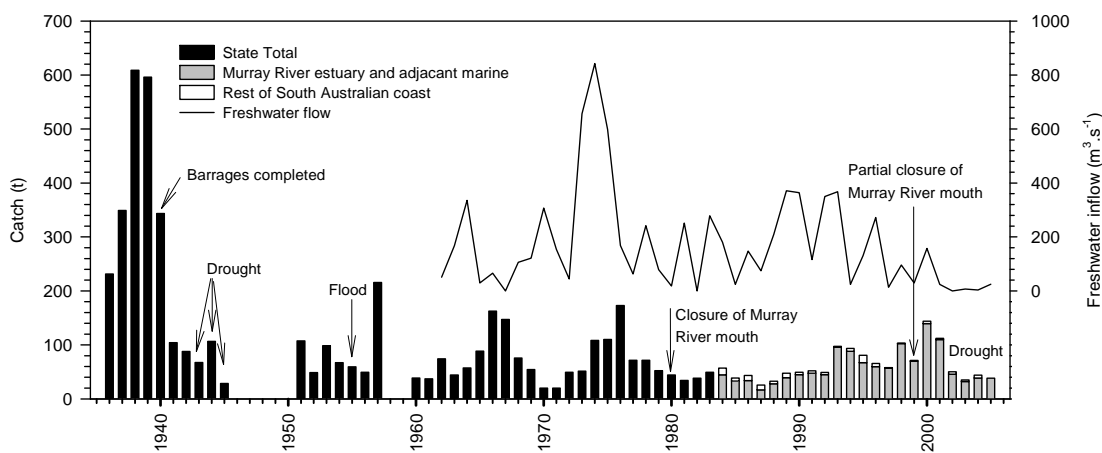
Figure 4. South Australian total catches of *Argyrosomus japonicus* from 1936 to 2005 with key hydrological events annotated. From 1984 to 2005 catches are divided into those from the Murray River mouth/estuary and those from the rest of the State. Freshwater inflow to the Murray River estuary from 1961 to 2005 is also shown (time scale is in financial years).

The Murray River mouth was closed due to drought in 1980 and catches remained below 100 t until 2001, when the catch was 145 t (Ferguson and Ward 2003). The mouth almost closed again in 1999 and drought conditions have prevailed since then. During this period the Murray River mouth has been kept open by a large-scale dredging program. Catches have continued to decline after 2001 and were 44 t in 2005 (Ferguson 2006). In the nearshore marine fishery, catches were strongly correlated with targeted effort, however there was a weak relationship between catch and targeted effort (fisher days) in the estuarine fishery (Table 1).

Freshwater inflows were generally a poor indicator of CPUE. For the period from 1984 to 2005, freshwater inflow (t_{-6} years, CC = 0.33) explained 16% of the variability in CPUE in the

nearshore marine fishery, (Table 1; Figure 5A). There was no relationship between freshwater inflow and CPUE in the estuarine fishery from 1984 to 2005 (Table 1; Figure 5B). CPUE (t-3 years, CC = 0.72) from the estuarine fishery was positively related to CPUE from the nearshore marine fishery (Table 1; Figure 6).

Table 1. Regressions performed on catch, effort and CPUE data for *Argyrosomus japonicus* caught by the Lakes and Coorong Fishery (e = estuarine fishery, nm = nearshore marine fishery).

Period data available	Fishery	Independent variable	Dependent variable	Regression statistics
1984-2005	e	effort	catch	r^2=0.28, $F_{1,20}$ = 7.730, P=0.012
1984-2005	nm	effort	catch	r^2=0.90, $F_{1,20}$ = 191.670, P=0.000
1987-2005	e, nm	$CPUE_{(t-3 \text{ years})}$ (e)	CPUE (nm)	r^2=0.52, $F_{1,17}$ = 18.443, P=0.000
1986-2005	e	fw. inflow	number of fishery reporting areas	r^2=0.43, $F_{1,18}$ = 13.777, P=0.002
1984-2005	e	fw. inflow $_{(t-3 \text{ years})}$	CPUE	r^2=0.00, $F_{1,20}$ = 0.046, P=0.832
1984-2005	nm	fw. inflow $_{(t-6 \text{ years})}$	CPUE	r^2=0.16, $F_{1,20}$ = 3.826, P=0.061
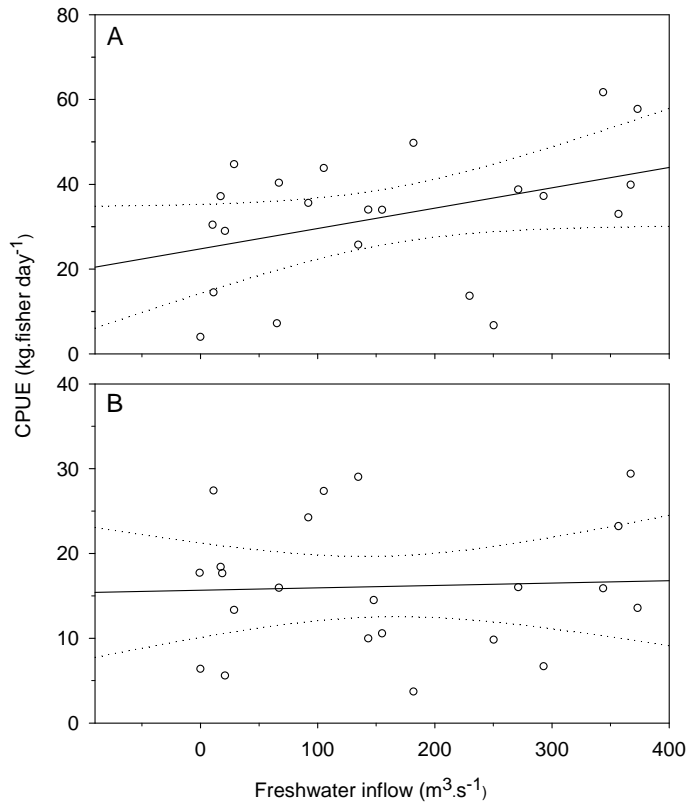
Figure 5. Relationship between freshwater inflow to Murray River estuary and CPUE from (A) the nearshore marine (t_{-6} years) and (B) estuarine (t_{-3} years) fisheries (95% confidence intervals shown as dotted lines).
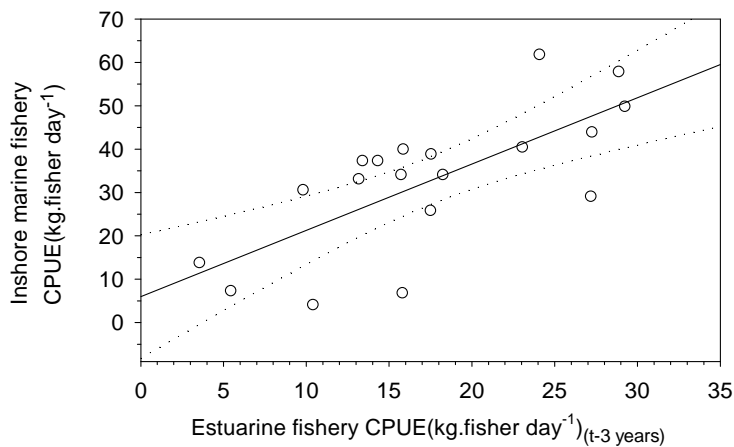
Figure 6. Relationship between CPUE estimates from the Murray River estuary fishery and nearshore marine fishery (95% confidence intervals shown as dotted lines).

Relationship between freshwater inflow to the Murray estuary and spatial distribution of mulloway catches

Mean annual inflows explained 43% of the variability in the number of spatial reporting areas in the Murray River estuary where *A. japonicus* were harvested by the LCF (Table 1; Figure 7).
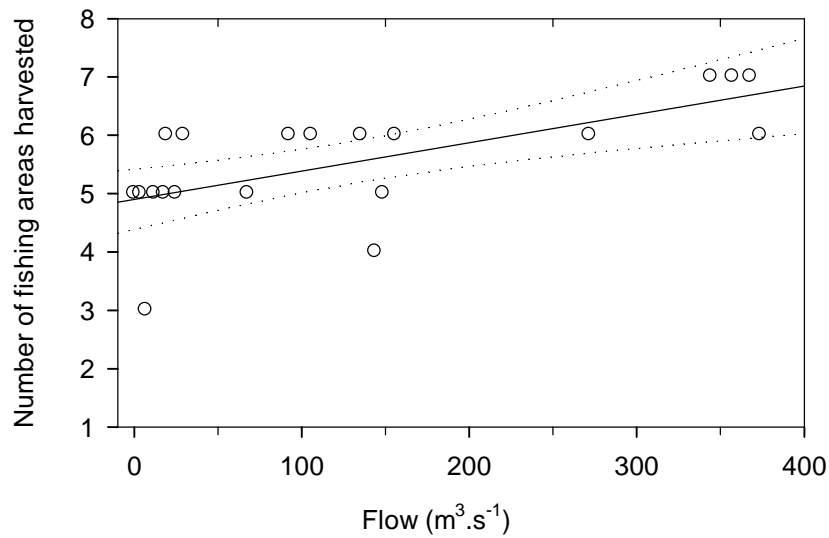
Figure 7. Relationship between freshwater inflow to the Murray River estuary and the number of estuarine fishery reporting areas from which juvenile mulloway were harvested (95% confidence intervals shown as dotted lines).

Discussion

In this study, the modes in age distributions of *Argyrosomus japonicus* from the nearshore marine environment suggested that a relationship may exist between freshwater inflow to protected nursery habitat in the Murray River estuary and subsequent recruitment to the fishery. The dominant year class from 1993 was present in samples from the nearshore marine environment in 2001 and persisted during 2002. This year class originated from the most recent year of relatively high freshwater inflow (>350 $m^3.s^{-1}$). Variability in annual freshwater inflows explained 28% and 35% of the variability in year class strength in 2001 and 2002 respectively.

The age distributions appear to be truncated with few individuals from year classes prior to 1991 even though 40 and 42 year old *A. japonicus* have been recorded in South Australia (Ferguson, unpublished data) and South Africa respectively (Griffiths and Hecht 1993). The absence of strong year classes prior to 1993 may have been caused by a combination of environmental impacts on recruitment and high levels of fishing mortality of both adults and juveniles (Ferguson and Ward 2003). The range of ages present was not related to gear selectivity because the age distributions from the nearshore marine net and line fisheries were statistically identical.

Truncation of the age distribution may reduce the capacity of the South Australian population of *A. japonicus* to withstand environmental events (Hsieh et al. 2006). This hypothesis is supported by the high inter-annual variability in CPUE that characterises the fishery for *A. japonicus* (Ferguson and Ward 2003). Populations of long-lived species are typically buffered from recruitment-induced fluctuations in biomass by the presence of many age classes (McGlennon 2000; Scharf 2000). However, the survival of strong cohorts is critical for the maintenance of fish populations during periods of poor recruitment, when only weak year classes are produced. Fishing mortality of *A. japonicus* obviously reduces the survival of these strong cohorts.

Recruits of *A. japonicus* enter the estuarine fishery as 3 year old juveniles. They enter the nearshore marine fishery as sub-adults at 5-6 years old. CPUE in the estuarine fishery provided a good indicator of CPUE in the nearshore marine fishery 3 years later which suggests that individuals caught in the nearshore fishery originated from the Murray River estuary and that juvenile and adult *A. japonicus* from the Murray River system may comprise a discrete sub-population within South Australia.

CPUE in the nearshore fishery generally increased approximately 6 years after increases in annual freshwater inflows. No detectable relationship was found between freshwater inflow and CPUE

in the estuarine fishery which may have been partly attributable to changes in catchability in response to environmental conditions within the estuary. For example, prolonged periods of drought i.e. early 1980's and 2000's, may have caused hyper-stability of CPUE due to aggregation of juveniles in response to reduced area of protected, low salinity habitat (Ferguson 2008).

The life-history of *A. japonicus* is characterised by high maximum age with large size and high age at maturity, and may have evolved under low rates of natural mortality (Griffiths 1996). Juvenile *A. japonicus* show a preference for turbid estuaries with relatively high freshwater input in South Africa (Marais 1988; Griffiths 1996) and southern and south-eastern Australia (Hall 1986; Gray and McDonall 1993). This may reduce predation rates, particularly by con-specific adults, and increase growth rates (Griffiths 1996). It is likely that *A. japonicus* spawn in the nearshore environment in South Africa and South Australia (Hall 1986; Griffiths 1996) and that their adaption of a river-discharge spawning relationship enhances recruitment of juveniles to protected estuarine habitat.

We propose that that the remnant estuary of the Murray River provides important habitat for juvenile *A. japonicus* (0 to 5 years old) and that the population centred about the Murray River system is estuarine dependent, like populations of this species in South Africa (Griffiths 1996). The Murray River estuary may be particularly important habitat for *A. japonicus* because it is the largest area of protected estuarine habitat in southern Australia.

The population of *A. japonicus* in South Australia may be environmentally limited because high quality protected habitat is only available when there are prolonged, seasonal freshwater inflows into the Murray River estuary. The high turbidity and longitudinal salinity gradient preferred by juvenile *A. japonicus* (Marais 1988; Griffiths 1996) only occurs in years of prolonged freshwater inflows (Geddes 1987). Hence, conditions of high freshwater inflow may increase the area of estuarine habitat available to juveniles as suggested by the spatial breakdown of fishery catches i.e. catches are taken from higher numbers of fishery reporting areas in years when there is freshwater inflow to the estuary. Conversely, loss of estuarine habitat in drought years may lead to poor recruitment and possibly recruitment failure.

Failure of *A. japonicus* to recruit to the Murray River estuary may have occurred at least once in the previous three decades. Based on the absence of one year old *A. japonicus* in seine netting surveys in 1982, Hall (1986) postulated that the 1981 year class failed to recruit to the Murray River estuary. This coincided with drought conditions from 1979 to 1981, when the estuary became moderately hyper-marine, culminating in complete closure of the mouth of the Murray River in 1981 (Geddes 1987; Walker and Jessup 1992). Recruitment failure in 1981 is supported

by; (i) the poor representation of individuals from year classes 1980 to 1985, (ii) low catches from the estuarine fishery from 1984 to 1990, and possibly (iii) by the lack of a relationship between freshwater inflows and CPUE from 1984 to 2005.

Populations of *A. japonicus* are overfished in South Africa (Griffiths 1997) and new South Wales (Silberschneider and Gray 2008) and the population in South Australia may be vulnerable to a combination of habitat degradation and overfishing. Habitat degradation combined with high fishing mortality has been implicated in the decline of several populations of sciaenids, including totoaba *Totoaba macdonaldii,* and the Chinese bahaba *Bahaba taipinhensis,* (Cisneros-Mata et al. 1995; Sadovy and Cheung 2003). The totoaba in the Gulf of Mexico was placed on the Convention on International Trade in Endangered Species (CITES) list after the population declined from a combination of juvenile habitat loss following damming of the Colorado River, high levels of juvenile mortality resulting in recruitment overfishing, and targeting of the annual spawning aggregations (Barrera-Guevara 1990; Cisneros-Mata et al. 1995).

Our findings have implications for management of the fishery of *A. japonicus* in and near the Murray River estuary. In South Australia, *A. japonicus* occurs at the southernmost part of their global distribution where winter water temperatures are close to the lowest temperatures (12 - 28°C) in which they occur (Bernatzeder and Britz 2007). The truncated age distribution, in combination with targeting of juveniles in a key nursery habitat (growth overfishing) and adults in a spawning/pre-spawning aggregation (recruitment overfishing) is likely to reduce the capacity of the population to withstand or recover from periods of prolonged habitat degradation through reduced freshwater flows. The relative rarity of sustained, seasonal freshwater inflows into the Murray River estuary has made this population extremely dependent on the presence of a few strong year classes resulting from years of above average flows. Future management of this fishery needs to consider the effects of environmental factors on the recruitment and survival of *A. japonicus* and the vulnerability of this species to increases in juvenile mortality and the targeting of pre-spawning aggregations.

The age distributions of *A. japonicus* from the nearshore fishery may also be a good indicator of the ecological status of the Murray River, because its reproductive success may be dependent on freshwater inflows, and because it is the apex predator in the Murray River estuary. In the northern hemisphere other sciaenid species have been suggested as environmental indicators because of their habitat preferences and sensitivity to environmental variables. For example spotted seatrout *Cynoscion nebulosus*, has been suggested as an indicator of estuarine health for estuaries in North America (Bortone 2003; Bortone et al. 2005).
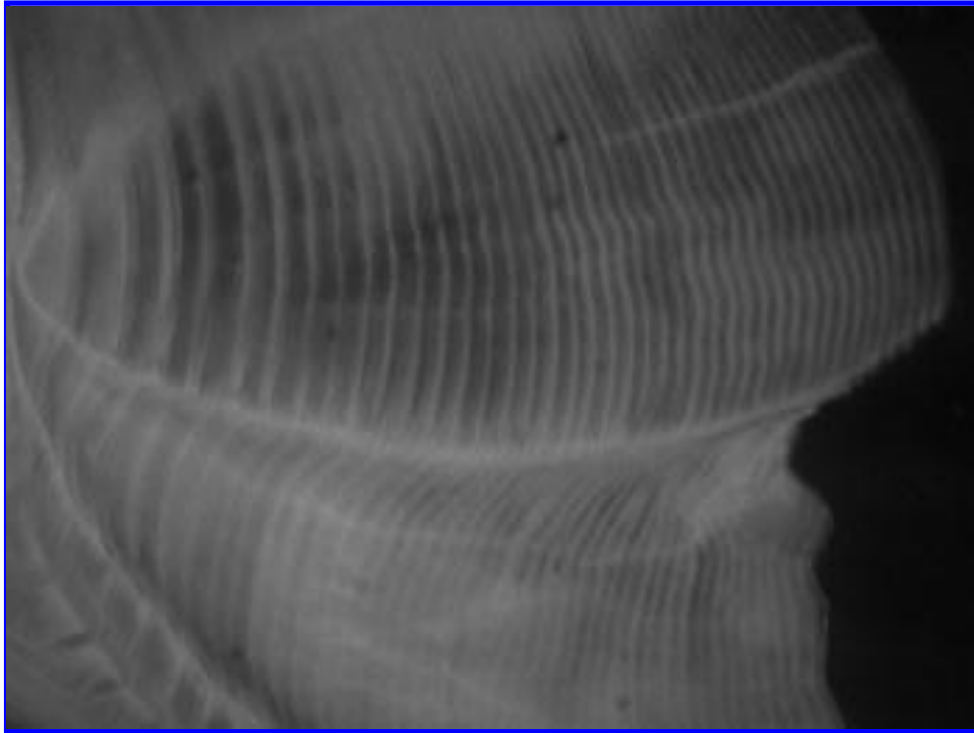
Transverse section of otolith from 41 year old *Argyrosomus japonicus*.

Statement of authorship
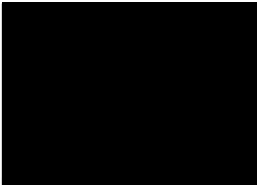
In this chapter, Gregory Ferguson performed all analyses on the data. Timothy Ward and Michael Geddes provided advice on the manuscript

Certification that the statement of contribution is accurate.

Signed ███████████████   Date   30 ·9· 2010
Gregory J Ferguson (*Candidate*)

Certification that the statement of contribution is accurate and permission is given for the inclusion of the paper in the thesis.

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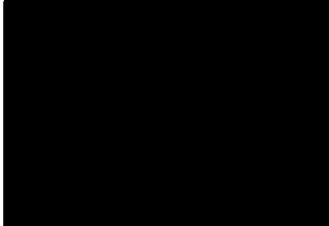
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Michael C Geddes.   Date   30/9/2010

Chapter 5

Age, growth and size at maturity of a large sciaenid at the southern part of its global distribution: implications for fisheries management

Abstract

The estuarine association of sciaenid fishes, combined with their late maturity and high maximum ages, make them vulnerable to combined impacts of habitat loss, flow regulation, drought and overfishing. In eastern (35° - 34°S) and western (33° - 29° S) South Australia, *Argyrosomus japonicus* occurs at the southernmost part of the species' global distribution. Aspects of the life-history of this large sciaenid were investigated to provide a basis for reviewing fisheries management. Validated, otolith-based growth rates were the lowest reported for this species and were lower ($P \leq 0.001$) in eastern ($L_{inf} = 1,406.18$, $K = 0.136$, $t_0 = -0.252$, n= 561) than western South Australia ($L_{inf} = 1,419.83$, $K = 0.182$, $t_0 = -0.098$, n=157), suggesting they may belong to separate populations. In eastern South Australia, juveniles (2-6 years) utilise estuarine habitats of the Murray River estuary and sub-adults/adults (7-25 years) occur in adjacent nearshore marine waters. There are no estuaries on the western South Australian coast and all age classes co-occur in nearshore waters. In eastern South Australia, size at maturity (SAM_{50}) was 811 and 812 mm TL for males and females respectively, and was 57% of L_{inf}. Also in this region, *A. japonicus* forms spawning/feeding aggregations in nearshore waters associated with spring-summer flows from the Murray River. Commercial and recreational fishers target juveniles in the Murray River estuary and adults in the spawning/feeding aggregations. Truncation of the age structures and reductions in catches suggest that the population of *A. japonicus* off the eastern coast of South Australia has been depleted by the combined effects of habitat loss, flow regulation, drought and overfishing. Additional environmental flows and fisheries management interventions, including protection of juveniles and spawning aggregations, are needed to prevent further depletion of this species.

Introduction

Estuaries are arguably the most anthropogenically-degraded habitats on earth (Edgar et al. 2000) and are subject to impacts including regulation of seasonal flows, excessive water abstraction, pollution, siltation and over-fishing (Whitfield and Bruton 1989; Gillanders and Kingsford 2002). Estuarine habitats are also critically important for many species of invertebrates, fishes and waterbirds (Blaber 1980; Lenanton 1982; Beck et al. 2001).

Community structure has been suggested as an indicator of estuarine health (Whitfield and Elliott 2002; Jordan and Smith 2003), but it is difficult and expensive to measure and monitor. Long-lived, apex-predators, that are estuarine-dependent and have life history characteristics that make them vulnerable to anthropogenic impacts, have been identified as potentially useful indicators of estuarine health (see review in Bortone 2003). Fishes with these life-history characteristics and for which a time-series of biological data are available (Bortone et al. 2005), as is the case for some fished species, may be most useful as ecological indicators.

Sciaenids are strongly associated with estuaries, with juveniles commonly utilising estuarine habitats as nursery areas and adults using habitats in and around estuaries for spawning. Sciaenids are potentially useful indicators of estuarine health because they can be particularly sensitive to anthropogenic effects (Flanagan and Hendrickson 1976; Murphy and Taylor 1990; Cisneros-Mata et al. 1995; Griffiths and Hecht 1995; Griffiths 1996). For example, the recruitment success of sciaenids is often linked to fresh water inflows into estuaries (Cisneros-Mata et al. 1995; Griffiths 1996; Rowell et al. 2005; Rowell et al. 2008b; Chapter 4). In addition, the delayed maturity of some sciaenid species may make them particularly vulnerable to overfishing (Musick 1999; Musick et al. 2001). Recruitment overfishing resulting from the combined effects of targeting juveniles in estuaries and adults in spawning aggregations, has been referred to as the "one-two punch" approach (Rowell et al. 2008b) and has been implicated in the declines of several sciaenid species. One such species, the spotted seatrout, *Cynoscion nebulosus*, has been suggested as a potential indicator of estuarine health in the Gulf of Mexico (Bortone 2003; Bortone et al. 2005).

Worldwide, many sciaenid populations are in serious decline including five species in the west Pacific (Sadovy and Cheung 2003; Liu and Sadovey de Mitcheson 2008), one species in Europe (Quéméner 2002) and seven species in North America (Musick et al. 2001). Estuarine dependence, slow growth rates and high age at maturity have been identified as factors

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contributing to the decline of *Sciaenops ocellatus* and *Totoaba macdonaldi* in the Gulf of Mexico (Murphy and Crabtree 2001; Lercari and Chávez 2007) and *Bahaba taipingensis* and *Larimichthyes crocea* in China (Sadovy and Cheung 2003; Liu and Sadovey de Mitcheson 2008). This suite of life-history characteristics has also been used to identify situations where the risk of local extinction may be high, e.g. *Argyrosomus japonicus* and *T. macdonaldi* (Musick et al. 2001; Cheung et al. 2005; Froese and Pauly 2009).

Argyrosomus japonicus (Temminck & Schlegel, 1843) is a large, predatory sciaenid that is widely distributed in estuaries and nearshore coastal (<100 m depth) waters in the Pacific and Indian Oceans (Griffiths and Heemstra 1995). Known as mulloway in Australia, this species occurs in sub-tropical and temperate waters south of the Burnett River (25°20 S) in southern Queensland and North West Cape (21°53 S) in Western Australia, where it is exposed to a wide range of environmental conditions, including variable ambient water temperatures (Kailola et al. 1993; Silberschneider and Gray 2008; Silberschneider et al. 2008). Like other sciaenids, *A. japonicus* has life-history characteristics that make it vulnerable to anthropogenic impacts, including estuarine association (Griffiths 1996; Chapter 4), slow growth rates, high age/size at maturity and high maximum age (Griffiths and Hecht 1995; Griffiths 1996; Farmer 2008). The population in South Australia may be particularly vulnerable because it is near the southernmost part of the global distribution of *A. japonicus*. Winter water temperatures (~14° C) in eastern South Australia (35-38°S) are close to the lowest experienced by this species (Bernatzeder and Britz 2007) while those on the western coast (32°S) are 1-2 ° C degrees higher.

Fishery management depends on understanding the dynamics of the variation in population abundance and determining the impact of fishing on the natural population processes (Hilborn and Walters 1992). This involves determining how the life history and demographic processes of egg production, recruitment, growth and mortality influence the biomass of the fish population. Age structures provide the basic information needed for understanding poulation rates and processes (Hilborn and Walters 1992). The most successful method for providing estimates of fish age is from interpretation of structures in otoliths, although validation of such interpretation is essential for providing unbiased estimates (Campana 2001; Campana and Thorrold 2001).

This paper examines the hypothesis that the life history characteristics of *A. japonicus* in South Australia, may make it particularly vulnerable to anthropogenic impacts such as habitat loss, flow regulation, drought and overfishing. To test this hypothesis we compare the catch history, habitat usage, age structures, growth rates and age/size at maturity of *A. japonicus* in two South

Australian locations. Findings are discussed in the context of populations in other locations and used to identify options for improving the management of this population.

Materials and methods

Fishery catches

A small commercial fishery targets *A. japonicus* in South Australia. The South Australian Lakes and Coorong Fishery operates within the Murray River estuary and adjacent nearshore marine environment and the South Australian Marine Scale Fishery operates in marine waters along the entire South Australian coastline. Annual commercial catches of *A. japonicus* from these fisheries were available from South Australian Research and Development Institute for 1984 to 2008. An estimate of annual recreational catch was provided by National Recreational and Indigenous Fishing Survey (Henry and Lyle 2003; Jones and Doonan 2005; Jones 2009).

Samples

Biological samples from *Argyrosomus japonicus* were collected from commercial and recreational catches from eastern South Australia between 2001 and 2005 and from western South Australia between 1997 and 2007 (Figure 1). Samples from eastern South Australia were collected from commercial gill net (>115 mm mesh) catches in the Murray River estuary (139° 22' 40.34" E, 35° 54' 24.88" S) in 2001 and 2002. Additional samples from the estuary were also collected from a research program which used multi-panel gill nets (45 m long, 40, 50, 70, 113 and 153 mm). This provided information on sub-legal size fish (<450 mm TL within the estuary). In eastern South Australia, samples were also collected from commercial and recreational catches in the nearshore marine environment in 2001 and 2002. These catches were taken from aggregations that occurred near the mouth of the Murray River in spring/summer of those years. Recreational fishers targeted these aggregations using rod and line and commercial fishers used modified gill nets (~245 mm mesh). The nets were operated as "swinger" nets where one end of the net was anchored to a motor vehicle, located on the beach, and the other was drifted out to a distance of 500-700 m into the surf zone. The net was then drifted through the surf zone, with the anchoring vehicle following, for a distance of up to 5 km. Further samples were obtained from commercial net and recreational line catches in Gulf St Vincent (135° 47' 25.75" E, 34° 59' 11.33"S) and western South Australia (131 ° 11' 51.12" E, 31° 34' 57.89" S) between 1997 and 2007.

For all samples, each fish was measured for total and standard lengths (TL, SL, nearest mm) and weighed to the nearest 100g. Gonads were dissected, weighed, sexed and staged according to a published method (Hunter and Macewiecz 1985) (Table 1). Briefly, macroscopic staging of ovaries (n=72) was to one of 5 developmental stages based on size, colour and visibility of oocytes and was validated by microscopic examination of histological preparations. For these ovaries (n=27), a segment was removed from the centre of one lobe and preserved in a fixative of formalin, acetic acid and calcium chloride (FAACC). Testes (n=90) were classified into 3 stages (Table 1). Sagittae were removed via a cut through the ventral ex-occipital region of the skull and were cleaned, dried, weighed and stored in labelled plastic bags.

Age and size

Laboratory preparation of otoliths

The left sagitta from each pair of otoliths was embedded in polyester casting resin, and a 500 μm thick longitudinal section was cut with a diamond blade mounted on a Gemmasta 6" (150 mm) bench top saw. Serial sections were cut and the section incorporating the otolith centre mounted on a glass microscope slide using Cyano-Acrylate glue. Mounts were ground to improve visibility of the opaque bands using silicon carbide polishing paper (grades 1200 and 900) and were examined on a black background under reflected light using a Leica MZ-16 dissecting microscope at 5x magnification.

Validation of ageing method

The pattern of deposition for otoliths was validated directly and indirectly. Direct validation of annuli was achieved by monthly sampling of captive fish of known birth date. Indirect validation was achieved by marginal increment analysis of wild caught fish.

Otoliths for direct validation were obtained from juvenile *A. japonicus* spawned at SARDI Aquatic Sciences, Adelaide from 10-18th December 2000. The juveniles were held in a 10,000 litre tank with flow-through seawater pumped from the nearby gulf and with water temperature and natural light reflecting the normal seasonal cycle. The fish were fed once per day on cockles (*Donax deltoides,* Bivalvia: Donacidae) at a rate of 5% of the total weight of live fish, and later on dry fish pellets at a rate of 2% of total weight per day. Each month, for 12 months, 10-12 fish were sampled, and the otoliths removed. Otoliths were sectioned, mounted and examined with a binocular microscope under reflected light. The third annulus was also validated directly from samples from the same spawning. These fish were kept in sea cages at an aquaculture facility in

Spencer Gulf. Fish (n = 10 fish per sample) were sampled at 34 months (October 2002), 36 months (December 2002) and 37 months January (2002).

Marginal increment analysis was also conducted on monthly samples of otoliths from small *A. japonicus* (1–2 opaque zones) caught in multi-panel research gill nets and older juveniles (3-4 opaque zones) from commercial gill nets. Measurements of the marginal increment of longitudinal sections of otoliths were made using digital imaging software (Image Pro 5.1™, Media Cybernetics). The marginal increment of each otolith was defined as the distance between the outer edge of the single, or outermost opaque zone, and the edge of the otolith. When only one opaque zone was present this was expressed as a proportion of the distance between the primordium and the outer edge of the opaque zone. When two or more opaque zones were present this was a proportion of the distance between the outer edges of the two outermost opaque zones. Each measurement was made perpendicular to the opaque zone(s) without knowledge of the date of capture of the fish and recorded to the nearest 0.01 mm. The values for marginal increments were separated into groups according to the number of opaque zones in the otoliths. For otoliths with 1 and 2 opaque zones, samples were pooled into corresponding months from 1 January 2001 to 31st December 2002. Otoliths with 3 and 4 opaque zones were similarly pooled but included additional samples from 1 January 2003 to 31 December 2003 to increase sample sizes.

Reading otoliths

Otoliths were examined under reflected light and ages estimated from counts of opaque zones. Opaque zones were counted along the ventral axis of the sulcal groove which has been reported as the best area for interpreting otoliths from *A. japonicus* (Griffiths and Hecht 1995; Griffiths 1996; Farmer 2003). The mean birth date of 1st January was estimated from the mid-point of the spawning season. This, in combination with the time at which the annulus becomes delineated on the otolith, was used to determine the age of the individual fish on their date of capture.

The relative precision of age estimates between multiple readings by two readers was calculated using an index of the average percentage error (IAPE) for a subset of 360 otoliths (Beamish and Fournier 1981). All otoliths were read twice by the author and those that did not agree on the number of annuli were re-read by a second reader. If the third reading did not agree the sample was removed from the analysis.

For eastern South Australia, separate age and length distributions were generated for the commercial and research data from estuarine habitat in 2001 (n_{ages} = 612, n_{TL} = 1327), recreational and commercial data from nearshore marine environment in 2001 (n_{ages} = 99, n_{TL} = 115), commercial data from the nearshore marine environment in 2002 (n_{ages} = 63, n_{TL} = 78). Ages from research netting in estuarine habitat (n= 98, <1 year olds) were estimated from otolith weights using the linear regression Age = 4.027*(otolith wt) + 0.174. The estimated ages were validated by reading a subset of sectioned otoliths from these samples (n = 30). Age and size distributions for western South Australia were prepared from samples from commercial net (n_{age} = 62, n_{TL} = 20) and recreational line (n_{age} = 20, n_{TL} = 7) catches.

Growth

Age structures from commercial and recreational catches in 2001 were compared using the Kolmogorov-Smirnov 2-sample goodness of fit test. This and all other statistical tests, estimates of model parameters, and confidence bounds were done using PASW Statistics® 18.

Von Bertalanffy growth curves (VBGF) were fitted to the lengths at age for male and female fish:

$$L_t = L_{\text{inf}} \left(1 - \exp^{-K[t-t_0]} \right)$$

where, L_{inf} is the mean asymptotic maximum length predicted by the equation, K is the growth coefficient and t_0 is the hypothetical age at which fish would have zero length if growth had followed that predicted by the equation. The growth equations were fitted to the data by minimising the sums of squares using a non-linear curve fitting routine.

Von Bertalanffy growth curves were compared using likelihood ratio tests (Kimura 1980; Haddon 2001). VBGF parameters were compared between: (i) males and females from eastern South Australia, and (ii) western and eastern populations in South Australia (males, females and unknown sex combined). Because there were no age/length data for individuals <4 years old from western South Australia the comparison of VBGF parameters between western and eastern populations was done using a sub-set of the data from eastern South Australia (individuals >4 years only).

Reproduction

Estimates for reproductive development were made from males and females that were collected from eastern South Australia in 2001 and 2002 and were >900 mm TL. Monthly gonadosomatic indices (GSI's, (gonad weight)/(total weight-gonad weight)*100 were prepared for sexually mature individuals from all months in 2001 where samples were available. Size at maturity (SAM) was estimated from males (stage III) and females (stages III, IV, V) collected between November and March. Due to small sample sizes, samples from 2001 and 2002 were pooled, and size classes of 100 mm were used. The logistic curve was fitted to the proportion (P) of sexually mature individuals by length class mid point, using the equation:

$$P = \frac{1}{\left(1 + \exp\left[-r\left(L - L_m\right)\right]\right)}$$

Where, r is the slope of the curve and L_m is the mean length of sexual maturity. In addition a qualitative assessment of age at maturity was made from these samples.
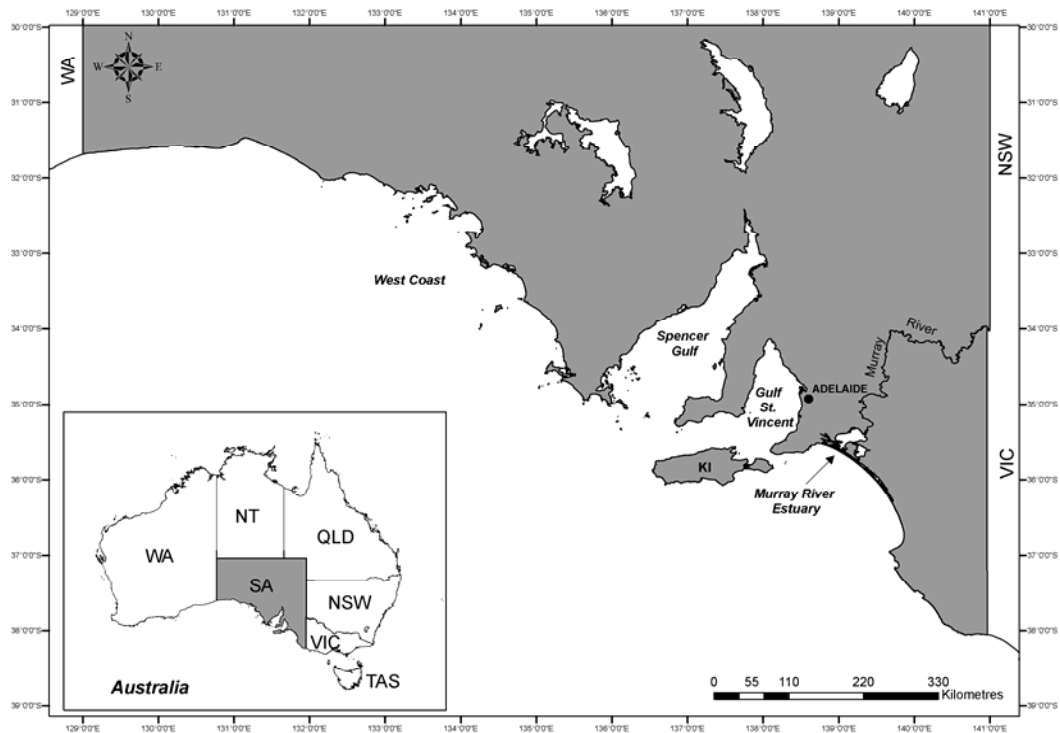
Figure 1. Map of Australia (inset) showing South Australia (SA), New South Wales (NSW) and Western Australia (WA). Main map shows western and eastern coasts of South Australia, and Murray River estuary.

Table 1. Stages of development used for macroscopic classification of the ovaries of *Argyrosomus japonicus* adapted from Hunter and Macewitz (1985).

Ovary Stage	Macroscopic Appearance	Microscopic Appearance
I Immature	Small, clear to translucent, jelly-like thread, grey to pink in colour. Close to posterior vertebral column. Oocytes invisible to naked eye	Unyolked, non atretic oocytes only
II Developing	Small to medium, translucent to yellow/brown. Ovary lobe may appear short, relative to stage III. Oocytes invisible to naked eye.	Mainly unyolked, few partially yolked oocytes
III Developed	Large, yellow to orange. Granular appearance due to visible individual oocytes. Ovary large, reaching forward into anterior gut cavity.	Dominated by advanced yolked oocytes, unyolked with partially yolked oocytes present in low numbers. Lumen large and obvious.
IV Gravid or running ripe	Very large orange, reaching anteriorly to/above stomach. Clear hydrated oocytes visible among opaque oocytes.	Oocytes of all stages present from unyolked to hydrated. Lumen large, obvious.
V Regressing or resting	Ovaries medium in size, brown to reddish-brown, Opaque. More flaccid than other stages.	Atretic oocytes present. Partially yolked oocytes present. Greater number of unyolked oocytes relative to yolked than in stage III/IV ovaries.
Testes Stage		
I Immature	Appear as black threads.	
II Developing	Thickened thread, black, no sperm or small amount of sperm when squeezed.	
III Developed	Swollen, grey, running ripe.	

Results

Fishery statistics

From 1984 to 2007, commercial catches of *Argyrosomus japonicus* ranged from approximately 40 t to a peak of 145 t in 2000-01 then declined to less than 35 t per year in 2007 (Figure 2). The long term average catch for eastern South Australia comprised mostly (86%) juveniles from the Murray River estuary. Most other commercial catches were from the adjacent nearshore environment with small annual catches from central and western South Australia (<4 t).

In 2000, recreational fishers caught 90 t most of which came from the nearshore marine environment adjacent the mouth of the Murray River (Henry and Lyle 2003; Jones and Doonan 2005). In 2007 the recreational catch was 62 t for all South Australian coastal waters (Jones 2009).
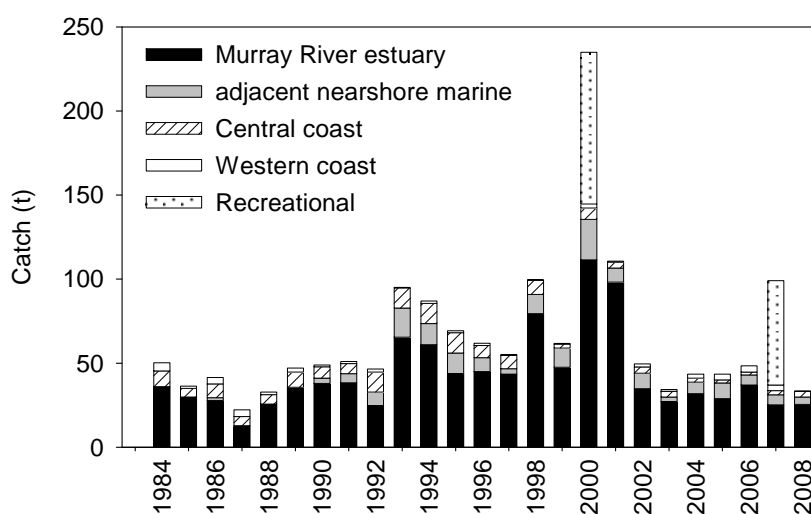
Figure 2. South Australian commercial catches of *Argyrosomus japonicus* for financial years 1984 to 2008. Commercial catches are sub-divided into those from the eastern (Murray River estuary and adjacent nearshore marine), central and western coasts. Annual recreational catch is available for 2000 and 2007 only.

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Age and size

Validation of ages

Sectioned otoliths clearly showed annuli as opaque zones that appeared lighter than the adjacent translucent zones. The first of these opaque zones were generally wider and less distinct than subsequent zones but relatively easy to define. The IAPE value was 4.9% suggesting a high level of reading precision. Opaque zones of growth increments were formed in spring-summer in otoliths from captive and wild *A. japonicus*. Otoliths from captive spawned fish showed that the first opaque edges appeared in July with one complete opaque zone and a clear edge present on all otoliths except one by October (Figure 3). Additional otoliths from the same cohort (not shown in Figure 3) revealed a complete third opaque band in 20, 80 and 100% of otoliths, sampled at 34 (October 2003), 36 (December 2003) and 37 months (January 2004) old respectively.
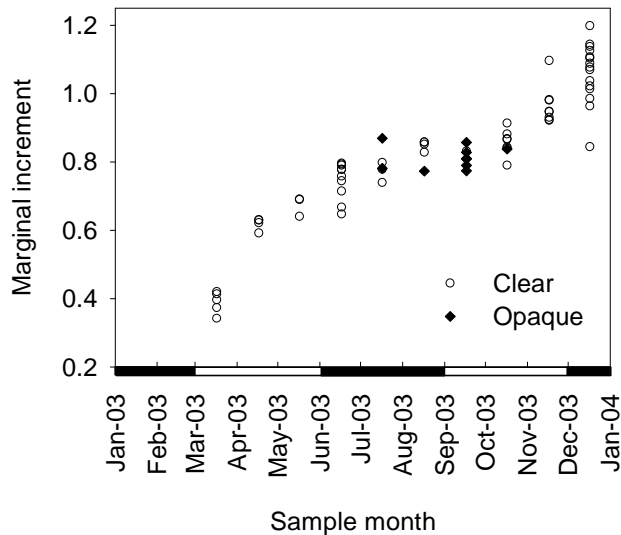
Figure 3. Marginal increment and edge type of sectioned sagittal otoliths from captive spawned *Argyrosomus japonicus*. On the x-axis, black rectangles represent summer and winter months and white represents spring and autumn months.

Indirect validation of annuli formation from analysis of marginal increments in otoliths with one and two opaque zones showed that marginal increments increased from January to September-October, then declined in November (Figure 4). In otoliths with 3 and 4 opaque zones the

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marginal increment increased from January to September, then declined precipitously in October.
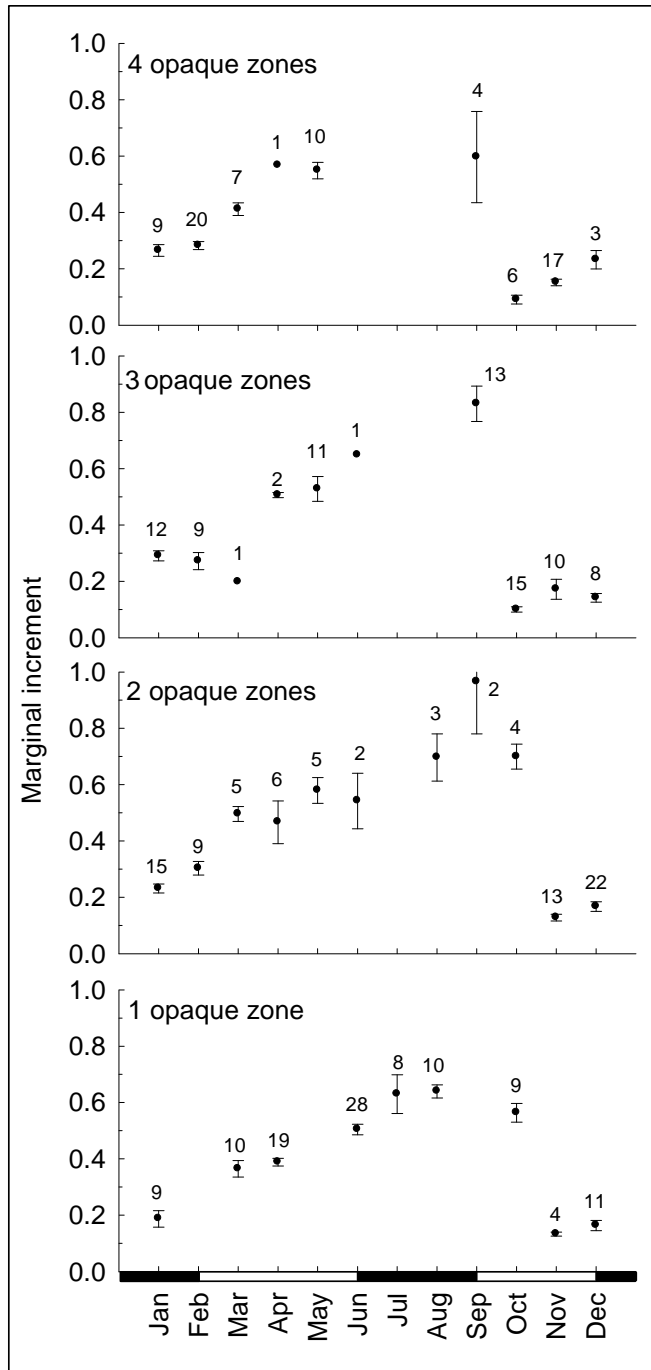
Figure 4. Marginal increments (± 1 SE) from sectioned sagittal otoliths of wild caught *Argyrosomus japonicus*. On the x-axis, black rectangles represent summer and winter months and white represents spring and autumn months.

Age/size structures

Clear spatial segregation was observed in age and size distributions between estuarine and nearshore marine habitats in eastern South Australia. Smaller, younger fish were found within the Murray River estuary, whereas larger/older fish were found exclusively in the nearshore marine environment (Figure 5). For fish from the Murray River estuary, two obvious modes were present in age and size structures representing 2+ (research nets) and 4+ individuals (commercial nets) (Figures 5A, B). Ages from research netting ranged from several months to 6 years. Ages from commercial nets ranged from 2 to 6 years, with a single mode at 3 years (Figure 5A). Sizes were < 750 mm TL, with a modal size of 500 mm TL.

For age distributions from the nearshore marine environment, adjacent to the Murray River mouth in 2001, recreational and commercial catches had a similar distribution (K-S $D = 0.555$, $P = 0.918$) and were therefore combined (n = 99). Ages ranged from 4 to 25 years with 8 year olds dominating the distribution (1993 year class, 35% of sample, Figure 5C). Secondary modes occurred at 11 (1990 year class, 8% of sample) and 12 years (1989 year class, 4% of sample). For the commercial sample from 2002, ages ranged from 5 to 24 years and the 1993 year class persisted as 9 year olds (41% of sample, Figure 5E) with a smaller mode of 12 year olds (1990 year class, 8% of sample) also present. Sizes were > 750 mm TL with a modal size of 850 mm TL (Figures 5D, F).

Ages from commercial net catches from western South Australia ranged from 3 to 12 years with 98% of individuals less than 9 years old (Figure 5G). Ages from recreational line catches ranged from 7 to 23 years with 30% of individuals younger than 9 years. The range of ages present in the recreational line fishery from western South Australia was similar to that from eastern South Australia. Statistical comparison of age structures between eastern and western South Australia was not possible due to the small sample sizes from the west.
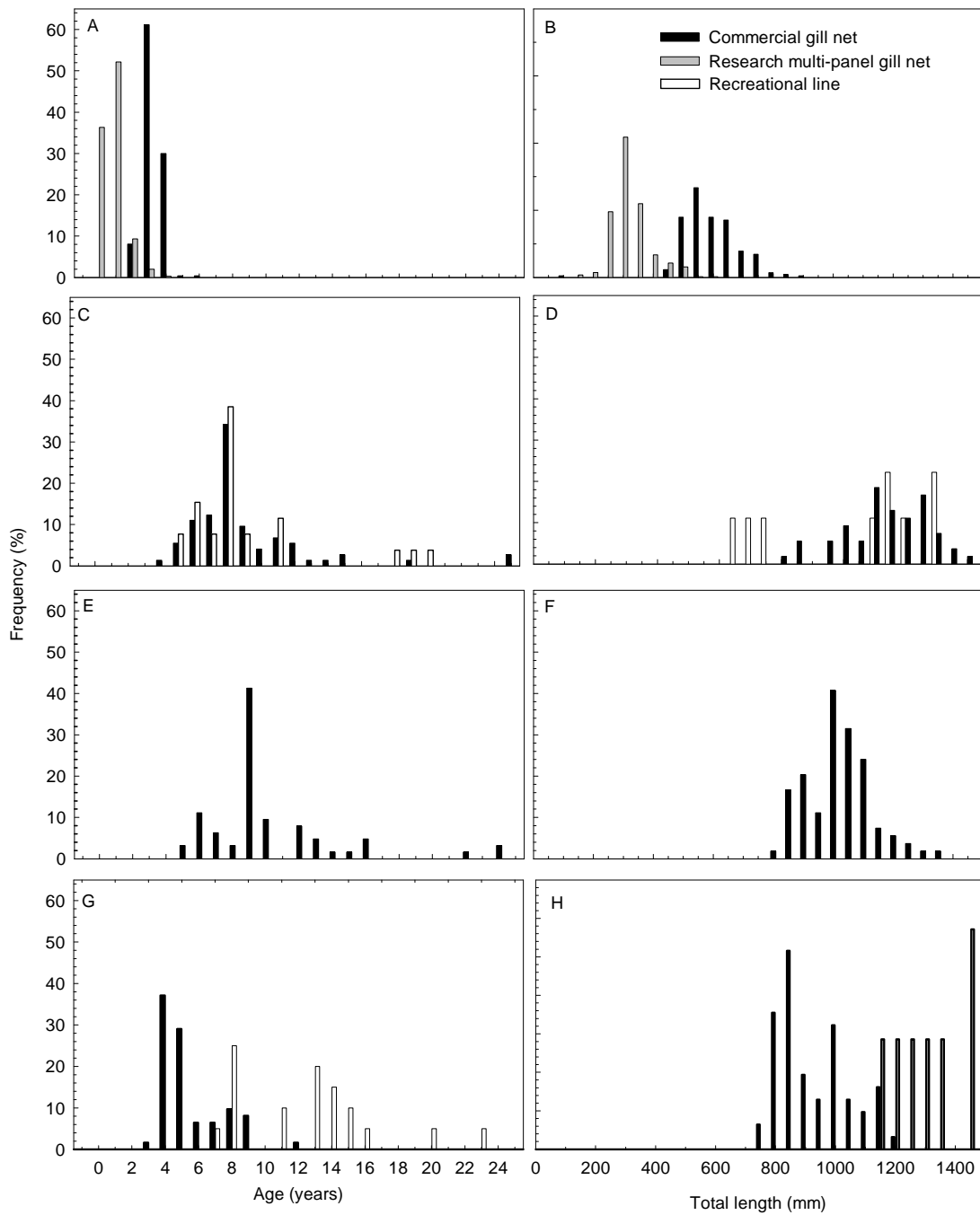
Figure 5. Age-length distributions of *Argyrosomus japonicus* from eastern South Australia. Figures on right hand side show age and length distributions from eastern South Australia: (A, B) the Murray River estuary in 2001; (C, D) nearshore marine environment in 2001; (E, F) nearshore marine environment in 2002. Bottom panels (G, H) show age and length of *A. japonicus* from the nearshore marine environment in western South Australia in 2003.

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Growth

The asymptotic size of female *A. japonicus* from eastern South Australia was higher than for males (L_{inf}, Kimura Likelihood Ratio Test: $P < 0.001$), although the growth rate (K, $P < 0.001$) was lower (Table 2, Figure 6). Age-length data from western South Australia were pooled (male, female, unknown sex) because sex was unknown for most samples (Table 2, Figure 6). Comparison of growth parameters between *A. japonicus* from western and eastern South Australia (pooled female, male, and unknown sex) indicated that asymptotic length (L_{inf}, $P < 0.001$) and growth rate (K, $P < 0.001$) were highest on the western coast.

Table 2. Von Bertalanffy growth parameters for *Argyrosomus japonicus* from eastern and western South Australia (95% confidence intervals in brackets).

Region	L_{inf} (mm)	K (y^{-1})	t_0 (y)	n
Eastern coast (Males)	1,356.23 (1,275.42 – 1,437.04)	0.159 (0.143 – 0.175)	0.00 (0.0 – 0.0)	185
Eastern coast (Females)	1,430.52 (1,353.53 – 1,507.51)	0.137 0.118 – 0.156)	-0.303 (-0.502 – -0.103)	209
Eastern coast (pooled sexes)	1,377.82	0.146	-0.112	561
Western coast (pooled sexes)	1,419.83	0.182	-0.098	157
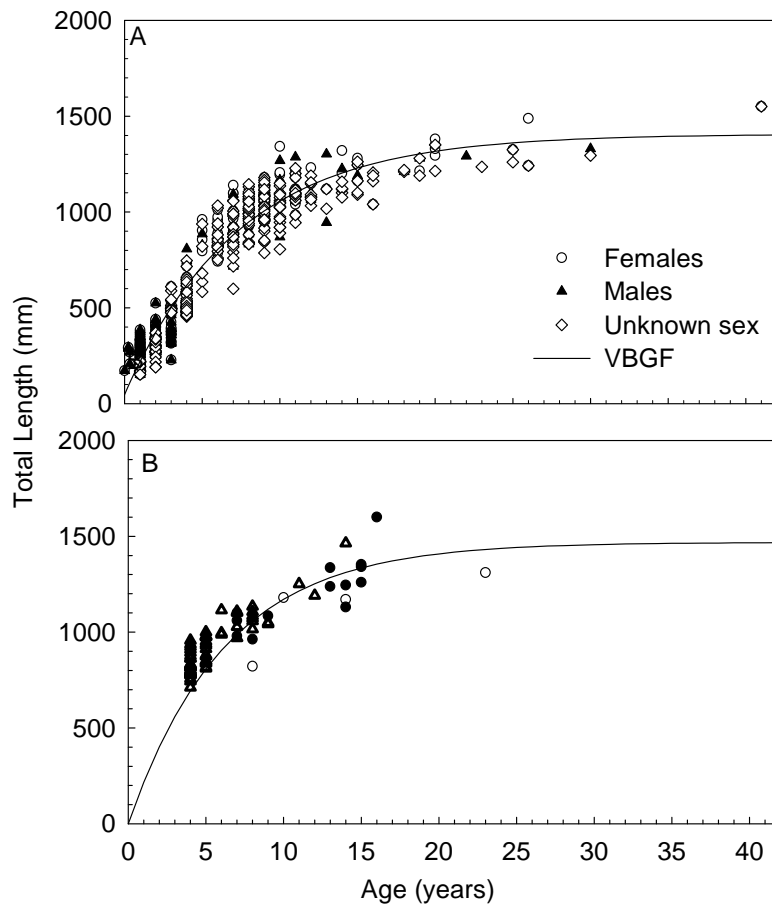
Figure 6. Von Bertalanffy growth function fitted to age-length data for *Argyrosomus japonicu*s from (A) eastern and (B) western South Australia.

Reproduction

GSI's for male and female *A. japonicus* from eastern South Australia increased from October to peak in December and declined to February (Figure 7A). Peak GSI for females occurred in November (n=39) and was 1% of body weight. However, a wide range of GSI's was observed (from <1 to 8%) with GSI's above 7% observed in 15% of females.

Macroscopic examination of ovaries (Figure 7B) showed that in October, November, and December, 23, 74 and 100% of females had developed ovaries, respectively. In January 55% of ovaries were yolked and 45% were recovering from spawning (stage V), with all ovaries spent/recovering from March onwards. Months where >30% of ovaries had yolked oocytes were November to January. Males with running ripe (stage III) testes were present from October to March.
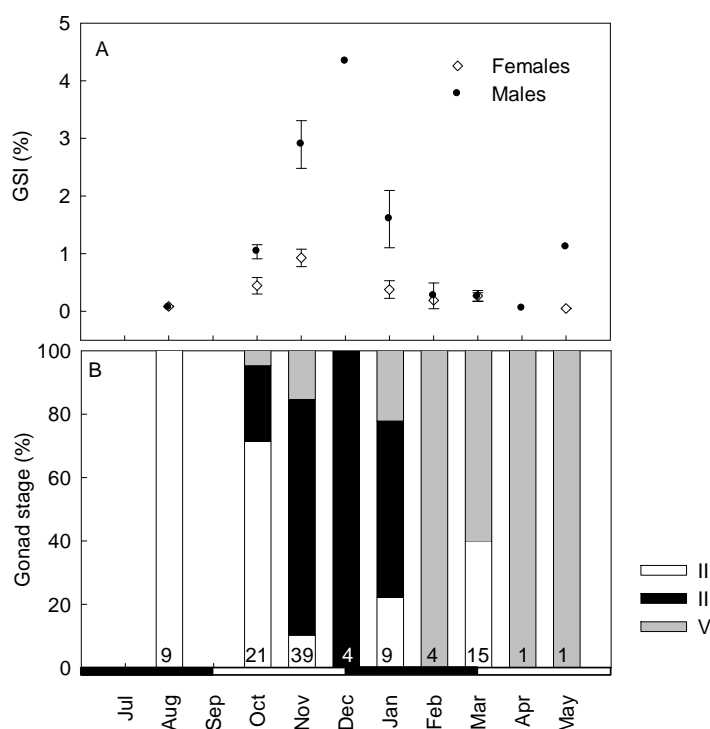
Figure 7. Female and male *Argyrosomus japonicus* from eastern South Australia: (A) Gonadosomatic indices; and (B) Ovarian developmental stages by month (Numbers on bars are sample sizes). On the x-axis, black rectangles represent summer and winter months and white represents spring and autumn months.

For males and females from eastern South Australia coast, the youngest individual with developed gonads was 5 years old. Age at maturity (>50% mature) was 5 years for males and 6 years for females although the age at which 100% of individuals had mature gonads was 8 and 9 years respectively. The size at maturity (SAM_{50}) was 811.0 (95% Confidence Bounds CB 731.29, 828.52) mm TL for males and 812 (95% CB 732.53, 974.75) mm TL for females from eastern South Australia (Figure 8).

For *A. japonicus* from western South Australia, maturity ogives were not constructed due to small sample sizes. However, gonad condition was recorded for 11 females (980 – 1,463 mm TL, 6-20 years) and 3 males (942 - 1,180 mm TL, 6-13 years) sampled from recreational line catches. All had mature, developed gonads except for the two smallest, youngest individuals which were 6 years old (female 980 mm TL, male 942 mm TL). Hydrated ovaries were observed in one 20 year old female (1,132 mm TL).

Figure 8. Size of maturity for (A) female and (B) male *Argyrosomus japonicus* from eastern South Australia.
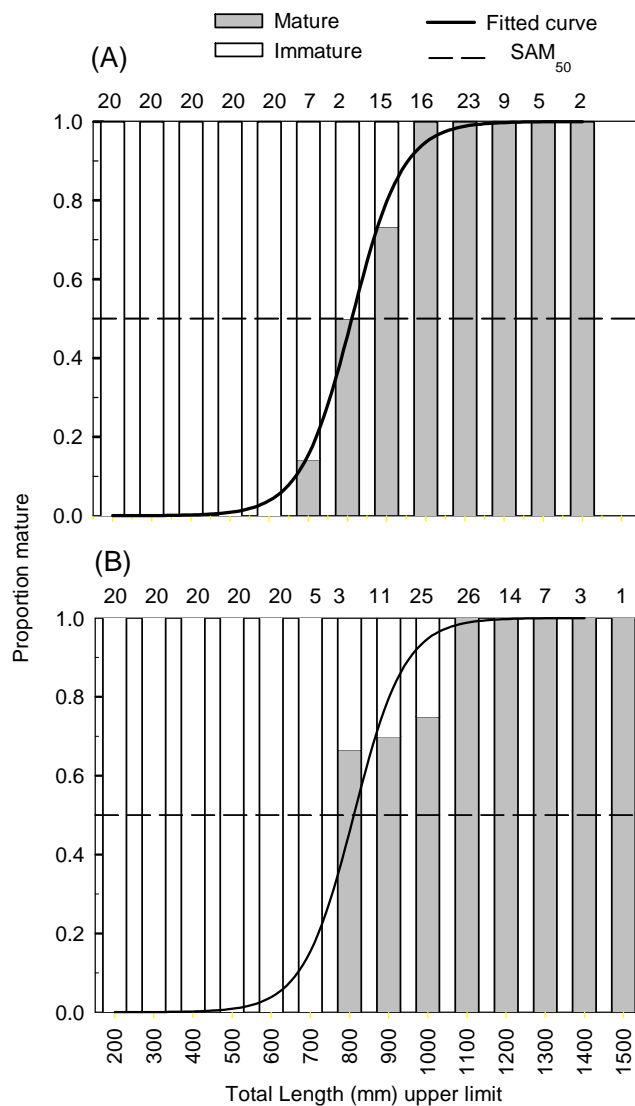
Numbers at top of bars indicate sample size.

Discussion

Results suggest that the population of *Argyrosomus japonicus* in South Australia may be particularly vulnerable to anthropogenic impacts due to its life history. Growth rates, based on validated ages from otoliths were the lowest reported across the range of the species. Maturity occurred at 5-6 years and at a size (SAM_{50}) of 812 mm TL, which is significantly above the legal minimum size.

Age and size

Juvenile *A. japonicus* in the Murray River estuary are spatially separated from adults in the nearshore marine environment. No individuals greater than 6 years old were collected from the estuary and reproductively mature adults were only obtained from the near-shore marine environment. Juveniles may enter the estuary shortly after spawning in spring-summer as has been suggested for this species in South Africa (Whitfield 1989; Griffiths 1996). Early juveniles (0+, >150 mm TL) occur in research nets from December onwards (Noell et al. 2009) and continue to live in the estuary until they are 4-5 years old. This apparent separation of juvenile and adult habitat is similar to the situation in South Africa (Griffiths and Hecht 1995; Griffiths 1996), but contrasts with New South Wales where size structures from estuarine and nearshore marine habitat are similar (Silberschneider et al. 2008). Griffiths (1996) suggested that the estuaries may provide protection from conspecific predation. This may be the case for the population associated with the Murray River estuary because small juveniles (100-150 mm TL) have been found in the stomachs of sub-adults (G. Ferguson unpublished results).

The maximum age of *A. japonicus* in eastern South Australia (41 years) is similar to the maximum age of 42 years reported for South Africa (Griffiths and Hecht 1995; Griffiths and Attwood 2005). Maximum ages recorded in New South Wales and Western Australia were 24 and 32 years respectively (Farmer 2008; Silberschneider et al. 2008). The low maximum age for New South Wales may in part reflect the high levels of fishing effort, however this explanation is less likely in Western Australia where fishing effort is lower (Farmer 2008; Silberschneider and Gray 2008; Silberschneider et al. 2008).

Despite the potential to reach a maximum age of 41 years in eastern South Australia few individuals older than 16 years were observed. This truncation of age classes suggests that the population may have been weakened, presumably by a combination of habitat degradation and over-fishing. Habitat reduction and subsequent degradation since the 1940s is known to have

dramatically reduced abundance of *A. japonicus* in eastern South Australia (Chapter 4). Since 1980, the increased frequency of low inflow years, and closure of the Murray River mouth, has likely resulted in the absence of a number of year classes (Chapter 4). For example, recruitment failure may have occurred in 1981when the Murray River mouth closed because ~1 year old *A. japonicus* were not observed in seine netting surveys in 1982 (Hall 1986). Additionally, individuals from year classes 1980 to 1985 are poorly represented in age structures.

In western South Australia juvenile *A. japonicus* may use shallow nearshore environments and embayments (Lenanton 1982; Farmer 2008). In this study, age and size of *A. japonicus* in catches from a shallow embayment (Fowlers Bay, 31° 59' 18.43" S, 132° 26' 20.22" E) were smaller than for those from an exposed section of coastline (Yalata, 31° 41' 26.68" S, 131° 41' 44.99" E). This suggests that in western South Australia smaller *A. japonicus* may prefer shallow sheltered environments, although small sample sizes and differences in fishing gear may also account for these differences.

Growth

Growth rates of *A. japonicus* in eastern South Australia are the lowest reported for this species (Griffiths and Hecht 1995; Farmer 2003; Silberschneider and Gray 2005; Farmer 2008) and were also lower than the growth rate for (5 to 15 year olds) western South Australia. Growth rates in lower latitude populations in Western Australia and New South Wales are much higher than those recorded in both eastern and western South Australia (Farmer 2008; Silberschneider et al. 2008). In South Africa, which is the only other region where this species' range extends to 35° S, growth rates are similar to those recorded in western South Australia (Griffiths and Hecht 1995; Farmer 2003) but higher than those recorded in eastern South Australia. The significant differences in the growth characteristics of *A. japonicus* between eastern and western South Australia suggests sub-structuring of the population(s), as has been found in a recent otolith-based study of stock structure (Chapter 3), and a previous genetic-based study (Dixon 1988).

Reproduction

In eastern South Australia, male and female *A. japonicus* reach sexual maturity at 5 and 6 years, respectively, which is similar to populations in South Africa and on the west coast of Western Australia (Griffiths 1996; Farmer 2008). In contrast, populations on the south coast of Western Australia and New South Wales reach sexual maturity at a ages of 3-5 years and 2-3 years, respectively (Farmer 2008; Silberschneider and Gray 2008). This suggests that populations on

the west coast of Western Australia and in South Australia and South Africa may have a longer pre-reproductive period than those from southern Western Australia and New South Wales.

The size at which 50% of individuals were mature (SAM$_{50}$, 812 mm TL) for male and female *A. japonicus* from eastern South Australia is smaller than for populations in Western Australia (F, 903; M, 873 mm TL) and South Africa (F, 1070; M, 920 mm TL), perhaps reflecting the lower growth rates in South Australia. Size at maturity for females in eastern South Australia is 57% of L$_{inf}$ which is lower than for populations of *A. japonicus* in Western Australia (77%) (Farmer 2008) and South Africa (73%) (Griffiths and Hecht 1995; Griffiths 1996) but higher than in New South Wales (49%) (Silberschneider et al. 2008). Although *A. japonicus* in South Australia matures at a smaller size than the population in South Africa, maturity occurs at the same age, suggesting that age-specific and lifetime egg production may be lower in South Australia than elsewhere.

Data obtained in the present study suggest that the spawning season for *A. japonicus*, in eastern South Australia extends from October to January and peaks in November. Almost all individuals caught near the mouth of Murray River during this period in 2001 and 2002 were in advanced stages of reproductive development or spent/recovering , suggesting these may be spawning aggregations. Previous studies have also suggested that *A. japonicus,* aggregates in spring-summer near the mouth of the Murray River to spawn (Hall 1986; Chapter 4). Similar seasonal spawning aggregations near river mouths have been reported for South Africa (Griffiths 1996). Alternatively, the low mean GSI in November (~1%) compared to South Africa and Western Australia (~8%), relatively small number of samples (15%) with GSI's of 7-8%, and the absence of hydrated oocytes in any females collected could suggest that spawning occurs elsewhere. However, developed ovaries in females from these aggregations and the presence of 0+ juveniles in catches from research nets in the Coorong lagoons from March onwards in 2001 suggests that spawning occurs close to the Murray River Mouth during spring-summer and that juveniles enter the estuary shortly afterwards.

The formation of spawning aggregations of *A. japonicus* near river mouths during peak flow periods in South Africa has been used to suggest the existence of a river-discharge spawning relationship (Griffiths 1996). Data from Chapter 4 suggested that spawning and/or recruitment in eastern South Australia is dependent on such flows. Such a relationship between spawning and environmental cues may be important in the life-history of populations of species that use protected juvenile habitat in estuaries. Because river flows may provide olfactory cues for postlarvae the release of eggs near the mouths of rivers may maximise the location of protected

nursery habitat in estuaries (Whitfield 1999; James et al. 2008). This may be critical for the population of *A. japonicus* located about the Murray River estuary due the apparent estuarine-dependence of juveniles (Chapter 4). Anecdotal information suggests that aggregations of *A. japonicus* at the Murray River mouth have failed to form in recent years, apparently due to the lack of freshwater flows.

Conclusions and Implications for management

Estuarine net fisheries for *A. japonicus* in Australian waters mainly target juveniles (Silberschneider et al. 2008; Chapter 4). More than 85% of the commercial catch of *A. japonicus* in South Australia is taken from the Murray River estuary. The legal minimum size in the Murray River estuary is 450 mm TL, which is 56% of the size of maturity (SAM_{50}, 812 mm TL). Similar size limits apply in estuaries in other parts of Australia and South Africa (450 – 550 mm TL) (Griffiths 1997; Ferguson and Ward 2003; Farmer 2008; Silberschneider et al. 2008). Yield-per-recruit analysis has demonstrated that exploitation of juveniles has resulted in growth overfishing in New South Wales and South Africa (Griffiths 1997; Silberschneider et al. 2008) and this is also likely to be the case in the Murray River estuary. Most of the remaining 15% of the commercial catch of mulloway in South Australia is taken from spawning/feeding aggregations in nearshore marine waters adjacent to the Murray River estuary.

The legal minimum length of 750 mm TL for *A. japonicus* in marine waters of South Australia is set at 92% of the size of 50% maturity and is unlikely to protect against recruitment overfishing. For long-lived species with slow growth rates and delayed maturity, allowing individuals to spawn at least once prior to capture is critical for sustainable management. An increase in the legal minimum size to at least SAM_{50} (812 mm TL) is needed to prevent further depletion of the *A. japonicus* sub-population in eastern South Australia.

Bycatch of juveniles of the target species is common in estuarine fisheries (Gray et al. 2001; Ueno 2001; Gray 2002; Gray et al. 2004). In the Murray River estuary, *A. japonicus* are discarded from gill net catches because they are below legal minimum size (Ferguson 2010b). Catch rates (number per net) of discarded *A. japonicus* are higher than catch rates of retained fish and survival rates are low (Ferguson 2010b). Discarding of *A. japonicus* from gill nets in the Murray River estuary was also described in a study in the early 1980's suggesting that this may have occurred for more than 25 years (Hall 1986). Discarding of sub-legal sized *A. japonicus* in the Murray River estuary is particularly concerning because catches comprise sexually immature fish. Consequently, this population is growth overfished and almost certainly recruitment overfished.

Mortality of juveniles from fishery discarding is particularly deleterious for long-lived, late-maturing species such as those sciaenids that depend on low mortality rates of juveniles in protected habitats in estuaries throughout a protracted juvenile phase (Griffiths 1996). For example, high levels of discarding of juveniles from trawling were implicated in the decline of the long-lived, late-maturing sciaenid, *Totoaba macdonaldi* (Cisneros-Mata et al. 1997). Similarly, in New South Wales where *A. japonicus* are considered recruitment overfished, sub-legal sized individuals occur as bycatch in estuarine trawling (Broadhurst and Kennelly 1994).

Aggregations of *A. japonicus* near the Murray River mouth in spring-summer are vulnerable to recreational fishing because they occur in nearshore marine environment and also because this area is close to a major city. Recreational fishers caught 38 and 62% of the combined commercial and recreational catch in 2000 and 2007 respectively (Henry and Lyle 2003; Jones and Doonan 2005; Jones 2009), which are the only years for which recreational catch information is available. There is a need to protect aggregations of mulloway at the Murray River mouth from targeting by fishers. This may be achieved with spatial and/or temporal closures to fishing. Such protection of spawning aggregations has been recommended for several species and has contributed to the successful recovery of the white seabass *Atractoscion nobilis*, off the Southern California Bight (Pondella and Allen 2008; Sadovy de Mitchensen et al. 2008).

The relative rarity of sustained, seasonal freshwater inflows into the Murray River estuary has made the population of *A. japonicus* associated with this system dependent on the presence of a few strong year classes resulting from years of above average flows (Chapter 4). While protection of juveniles in the Murray River estuary, and of seasonal aggregations of the estuary mouth is needed to help rebuild this population there is also a need for sufficient environmental flows to allow successful recruitment and establishment of strong year classes (Chapter 4).

Because *A. japonicus* is the apex predator in the Murray River estuary (Hall 1986; Deegan et al. 2010) age structures from the nearshore fishery may also be a good indicator of the ecological status of the Murray River. In the northern hemisphere other sciaenid species have been suggested as environmental indicators because of their habitat preferences and sensitivity to environmental variables. For example spotted seatrout *Cynoscion nebulosus*, has been suggested as an indicator for the health of estuaries in North America (Burke et al. 1993; Bortone et al. 2005). Regular (annual) monitoring of the age structures of such species, for which life-history and habitat association is well understood may provide such an indicator. Such monitoring of age structures from the nearshore environment associated with the Murray River may also provide the best indicator for the success of rebuilding this population (Chapter 2). Establishment of

new age classes in the nearshore population may be an appropriate restoration target for rebuilding the population of *A. japonicus* in eastern South Australia.

Sustainable management of sciaenids requires knowledge of the life history of each species, the structure of the stock, habitat association and the demography of individual populations. Many sciaenids are vulnerable to anthropogenic environmental effects because they require estuarine and nearshore marine habitats that often occur close to population centres and are vulnerable to exploitation as juveniles in estuarine habitats and as adults in spawning aggregations. The population of *A. japonicus* in eastern South Australia is depleted due to a combination of habitat degradation, loss of environmental flows (Chapter 4), and longevity overfishing (*sensu* Beamish et al. 2006). The population would benefit from management measures that aim to preserve capacity for egg production, allow recruits to enter the adult population, and maintain long tailed age structures. Measures that would contribute to these aims for *A. japonicus* in eastern South Australia include: (i) legal minimum size commensurate with at least SAM_{50} (i.e. > 810 mm TL); (ii) protection of juveniles in estuaries and shallow nearshore waters; (iii) maintenance of environmental flows (Chapter 4); and (iv) protection of spawning/feeding aggregations.

Mulloway *Argyrosomus japonicus* (Photograph courtesy of Brian Kowald, South
Australian Research and Development Institute - Aquatic Sciences).

Chapter 6

General Discussion

Introduction

This thesis provides quantitative information that enables impacts of river regulation, drought, and fishing pressure on exploited fish assemblages in a large, degraded temperate estuary to be investigated. Species composition of catches from a multi-species fishery, combined with information on life-history and demography were used to identify species that may be vulnerable to these impacts (Chapter 2). Results from this study indicated that large-bodied, native fish with periodic life histories were vulnerable to anthropogenic impacts and that one such species, the sciaenid *Argyrosomus japonicus*, was particularly vulnerable. To further understand the impacts of anthropogenic environmental factors and fishing on this population the spatial structure of stocks (Chapter 3), the role of environmental flows in the life-history (Chapter 4), and age, growth, and reproduction (Chapter 5) were investigated.

The present chapter provides a summary of the key findings presented in this thesis. For expanded discussion of the points made here and for further details that have been omitted the reader should consult the relevant text in Chapters 2 through 5.

River regulation, drought and fishery impacts on fish communities in estuaries

Information on (i) fish assemblages, (ii) life-histories, and (iii) age structures, demonstrated that fish have been impacted by reduced river flows from river regulation and drought. This is the first study to investigate changes in the fish assemblages of the lower Murray River over sufficient years to include flooding (late 1980's) and extreme drought (2002 onwards), although several studies of fish assemblages commenced during the current drought (Noell et al. 2009). Historical fisheries data enabled a longer term perspective to be obtained than that normally possible from short-term studies.

Temporal stability of fish assemblages only occurred during and immediately following a period of sustained annual flooding (1989-93, <10,000 GL.y^{-1}). This is significant because a sequence of flood years, rather than a single flood event, may be required for successful recruitment of large bodied native species. This was suggested by (i) relatively high abundances of *Macquaria ambigua*, *Acanthopagrus butcheri*, and *A. japonicus* during, and immediately after 1989-93, and (ii) the importance of the age class originating from 1993 in recent age structures of *A. japonicus* (2001

and 2002). Changes in fish assemblages following the high flow period reflected declining abundances of these large-bodied, native species due to fishing (Blaber et al. 2000). While several studies have linked high abundance of freshwater species with sequential flood years (Puckridge et al. 2000; Leigh et al. 2010) this study suggests such events may be equally important for estuarine associated species such as *A. japonicus*, and *A. butcheri*.

Age structures of exploited fish species interpreted in the context of each species' life-history provided insight into the effects of fishing. Truncation of the age structures of the periodic strategist species *M. ambigua*, *A. butcheri*, and *A. japonicus* indicated over-exploitation. The age structure of *Rhombosolea tapirina,* which has a life-history intermediate between periodic and opportunistic strategists, provided an extreme example with almost all catches comprising females from the one-year-old age class. Removal of older, larger fish, or 'longevity overfishing' (Beamish et al. 2006), has been shown to compromise egg production (Palumbi 2004) and to contribute to variability in abundance (Hsieh et al. 2006). Truncated age structures, and decreasing catches of periodic strategist species in the freshwater and estuarine habitats of the lower Murray River suggest that these populations have been over-exploited and their capacity to establish a strong year class when environmental conditions become favourable to successful recruitment is compromised.

Stock structure of A*rgyrosomus japonicus* in South Australia

Understanding stock structure is fundamental to management of fisheries and endangered species (Begg and Waldman 1999). For example, knowledge of stock structure may be important for recognition of potential sources of recruitment or for protection of nursery and spawning areas. Prior to this study genetic based investigation of the population structure of *A. japonicus* in South Australia was inconclusive (Dixon 1988; Dixon 1990), and results from studies in New South Wales and Western Australia also showed contrasting results (Archangi 2008; Farmer 2008).

Few studies combine analysis of otolith trace element composition and otolith shape to discriminate between fish stocks (Turan 2006). Results from this study indicated that the two methods were complementary because trace-element analysis of the otolith edge provides very high classification success and gives a snapshot of differences between groups from different geographic areas, while shape analysis indicates that these discrete groups of fish experienced different environmental conditions over a long period of time (Campana and Casselman 1993). Importantly, the use of two methods also provides corroboration.

Differences in trace element composition and overall shape of otoliths of *A. japonicus* from western and eastern South Australia indicated sub-structuring of the population (Chapter 3). In particular, concentrations of Ba:Ca were lowest in the edges of otoliths from western South Australia, and highest in those from the eastern coast, with intermediate concentrations in the central region. These results were supported by differences in life-history between western and eastern populations in South Australia. Life history information also suggested that separate populations between western and eastern South Australia occur. Growth rates differed between these regions (Chapter 5) with the western population having higher growth rates than the eastern population.. Sub-structuring of *A. japonicus* in South Australia implies that the likelihood of recruitment between depleted populations on either western or eastern coasts is low. Fidelity of juveniles to the natal estuary has been suggested in a study of *A. japonicus* in South Africa and this may also be the case for the population in eastern South Australia (Griffiths 1996; Cowley et al. 2008).

Analysis of otolith shape may also provide a relatively quick, cost-effective method to identify *A. japonicus* from aquaculture production, as occurs in South Australia, or from stock enhancement of populations with hatchery-produced fish as occurs in New South Wales (Fielder and Bardsley 1999), and potentially South Africa (Palmer and Snowball 2009) and South Australia (Chapters 2 and 5). Identification of such fish is important because estimating the survival and contribution of hatchery reared fish to the fishery is a requirement of responsible stock enhancement (Taylor et al. 2005; Lorenzen et al. 2010). Tools to differentiate hatchery reared fish from aquaculture or stock enhancement programs from wild populations are also important to address potential negative impacts including deleterious genetic effects, and disease (Blankenship and Leber 1995).

Importance of estuarine habitat and river flows for *Argyrosomus japonicus* in South Australia

Use of estuarine habitat by marine migrants may range from facultative to obligate (Elliott et al. 2007). Juveniles of marine estuarine-opportunists enter estuaries in large numbers but also use alternative nearshore marine habitats, in contrast to marine estuarine-dependent species where the juveniles do not have access to alternative protected habitat. However, it is also important to recognise that estuarine dependence may not be characteristic of an entire population or stock and divergent migration behaviours or habitat use may also occur within populations, i.e. contingents (Secor 1999).

Argyrosomus japonicus are marine migrants in estuaries in South Africa and New South Wales (Gray and McDonall 1993; Griffiths 1996; Silberschneider et al. 2008). However, they are more common in nearshore waters in Western Australia because there are numerous inshore reefs and few permanently open estuaries (Farmer 2008). In South Australia, discrete populations of *A. japonicus* (Chapter 3) occur on the western and eastern coasts. In eastern South Australia, *A. japonicus* may be classified as marine migrants with obligate use of estuarine habitat by juveniles because: (i) adults and juveniles are separated spatially (Chapter 5); (ii) there is a significant relationship between relative abundance of juveniles in estuarine waters and subsequent abundance of adults in nearshore marine habitat (Chapter 4); (iii) the dominant year class of adults was from the year of highest freshwater inflow to the estuary (Chapter 4); (iv) freshwater inflows to the estuary explained 28 and 35% of the variation in year class strength in samples from 2001 and 2002, respectively (Chapter 4). These results are supported by a study in South Africa which suggested that this species is estuarine dependent for at least the juvenile stage (Griffiths 1996). Low growth rates of this southern population also may contribute to the dependence on estuarine habitat (Chapter 5).

Little is known about juvenile habitat of *A. japonicus* on the west coast of South Australia where there are no estuaries (Pritchard 1967; Day 1981; Blaber 1997) although sheltered embayments do occur. It is not known if different habitats available to the western population, compared to the east, make them more vulnerable to exploitation. However, the western population is subject to lower fishing pressure, and has higher growth rates than the eastern population (Chapter 5).

Griffiths (1996) postulated that *A. japonicus* evolved its life-history strategy, including delayed reproduction, large size at first maturity, and long lifespan, as a result of lower rates of natural

mortality. Predation of juvenile *A. japonicus* by adults commonly occurs (Griffiths 1996). It is likely that juvenile habitat in the remnant estuary of the Murray River provides protection from higher levels of con-specific predation that may occur in the nearshore marine environments (Griffiths 1996).

While differences in age and size distributions of *A. japonicus* in estuarine and nearshore marine habitats in eastern South Australia provide indirect evidence of connectivity between juvenile and adult populations, further study using methods such as artificial or natural tags is needed to provide direct evidence (Gillanders et al. 2003).

Adults of the eastern population form spring-summer spawning/feeding aggregations at the mouth of the Murray River when spring-summer outflows occur. These aggregations are vulnerable to capture by line and net fishing (Chapters 4, 5). Traditional tagging e.g. t-bar tags, of older juveniles (3-4 years, ~450 - 550 mm TL) from within the estuary may subsequently identify adult fish that originated from the estuary. However, traditional tagging studies only indicate the point of tagging and recapture and do not provide information on where individual fish may have moved to in the intervening time. For example, Hall (1986) used t-bar tags to mark 6,000 *A. japonicus* inside the Murray River estuary. Almost all of the 180 (3%) recaptures were made in the estuary, within one year of release. Only 7 individuals (0.01%) moved a significant distance from the estuary. Telemetry studies may provide similar information with the advantage of being able to track and determine the location of individuals in real time (Gillanders et al. 2003; Taylor et al. 2006). Such methods do not require recapture of the tagged fish but may require monitoring and maintenance of equipment (Gillanders et al. 2003). Development of archival tags that log location and depth then upload data in situ via satellite links may provide the most promising approach.

Stable isotopes and elemental signatures of otoliths can be used to provide a natural tag to determine connectivity between estuarine and coastal habitats (Gillanders 2005) and overcome the disadvantages of artificial tags including tag shedding, mortality of tagged fish or the requirement to recapture specific fish. Natural tags would also mean that all fish are effectively tagged therefore every fish provides a recapture unlike traditional tagging techniques. Because *A. japonicus* show strong fidelity to the natal estuary (Griffiths 1996), analysis of the otolith centre of adults, corresponding to when they were juveniles(Gillanders and Kingsford 1996), may be used to indicate the proportion of individuals originating from the Murray River estuary and potentially the alternative nearshore marine habitats. Alternatively, life-time chronologies of salinity, or trace elements, from microprobe analysis of otoliths patterns may provide further

information on contingent behaviours of estuarine habitat use by *A. japonicus* (Secor 1999). Before such an approach could be utilised, further work on the temporal and spatial variability in elemental signatures would be required (Gillanders 2002; Elsdon and Gillanders 2003b).

Patterns in concentrations of oxygen isotopes have been used to investigate use of estuarine habitat by the sciaenids *Totoaba macdonadi* and *Cynoscion othonopterus* (Rowell et al. 2005; Rowell et al. 2008a; Rowell et al. 2008b). Declines in flows from the Colorado River, and subsequent reduction in area of estuarine habitat were related to declines in fishery production of *Cynoscion othonopterus* (Rowell et al. 2005). Such an approach may be useful for comparing estuarine habitat usage by *A. japonicus*:(i) when the original Murray River estuary was available prior to construction of barrages (pre-1940's), when freshwater flows were stronger and more regular; with (ii) current conditions in the remnant estuary (recent samples). Otoliths of *A. japonicus* from prior to construction of barrages may potentially be available from indigenous middens in the Murray River region (Disspain 2009). Similarly, for the western region, otoliths of *A. japonicus* from indigenous middens could be also compared with recent samples.

Demography and life-history of *Argyrosomus japonicus* in South Australia

Several key life-history parameters of *A. japonicus* were estimated in this study. This study validated the periodicity of annuli in otoliths of *A. japonicus* in South Australia for one to four year old fish. Growth rates were the lowest reported for this species. A possible explanation is that the South Australian coastline occurs at the southern-most part of the species' global range in water temperatures close to the environmental limit for the species (Bernatzeder and Britz 2007). Within South Australia, growth rates were lower in eastern than western South Australia, possibly also due to differences in water temperature associated with lower latitudes on the west coast.

The maximum age of 41 years for an individual *A. japonicus* from eastern South Australia suggests high longevity in this population, and is supported by a similar maximum age of 42 years in South Africa (Griffiths 1996). This population also has delayed maturity (SAM$_{50}$ ~810 mm TL) which occurs at 5 years of age and is 57% of L_{inf}. Differences in age structures indicated that juvenile and adult habitats were separated spatially because juveniles (2-6 years) were found in the Murray River estuary while sub-adults/adults (7-25 years) occurred exclusively in adjacent nearshore waters.

Implications for management of long-lived fish species in changeable environments

Human impacts do not act in isolation, and in 45% of species depletions, multiple human impacts were involved, commonly exploitation and habitat loss (Lotze et al. 2006). While separation of the various anthropogenic effects is difficult (Laë 1995; Blaber et al. 2000), this study identified long-term changes to fish assemblages and decline in species richness of fishes occurred with continued fishing and reduced freshwater flows. This is significant because protecting the biodiversity of the Coorong National Park is listed as a Matter of National Environmental Significance under the federal *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC Act).

Management of multi-species fisheries needs to be tailored to the most sensitive, rather than the most robust species (Myers and Worm 2005). Consideration of life-history strategies should be fundamental to assessment of resource status (King and McFarlane 2003) and naturally evolved mechanisms such as long tailed age structures should be protected (Beamish et al. 2006). In this study, periodic life-history strategists were particularly susceptible to long-term drought because: (i) they were dependent on environmental flows for establishment of a strong year class at least once in their life-time (Chapter 2, 4); (ii) they were exploited species with high market value; and (iii) their age structures had been severely truncated by fishing resulting in compromised egg production (Chapter 2). The consequences of age class truncation for species with highly episodic recruitment may be catastrophic (Longhurst 2002) and this may be particularly important for periodic strategists in highly variable estuarine and freshwater habitats such as those associated with the lower Murray River.

Commonly used fishery performance indicators such as catch, effort and CPUE within the limits that are set for the particular historical data-set may be unreliable under changing environmental conditions such as drought (Ferguson 2010a). Consequently, age structures may be the most valuable available information to assess population health. This information must be interpreted within the context of life-history and stock structure, information which may not be available for all exploited species in a fishery (Chapters 2, 4, 5).

Sustainable management of sciaenids also requires habitat association and the demography of individual populations. Many sciaenids are vulnerable to anthropogenic environmental effects because they require estuarine and nearshore marine habitats that often occur close to population centres and are vulnerable to exploitation as juveniles in estuarine habitats and as

adults in spawning aggregations. The population of *A. japonicus* in eastern South Australia is depleted due to a combination of habitat degradation, loss of environmental flows (Chapter 4), and longevity overfishing (*sensu* Beamish et al. 2006). The population would benefit from management measures that aim to preserve capacity for egg production, allow recruits to enter the adult population, and maintain long tailed age structures. Such management measures include legal minimum size commensurate with at least SAM_{50}, (i.e. > 810 mm TL) (ii) protection of juveniles in estuaries and shallow nearshore waters, (iii) maintenance of environmental flows, and (iv) protection of spawning/feeding aggregations.

Setting restoration targets for populations of *A. japonicus* and other vulnerable large-bodied native species should initially be based on preserving remnant age structures, and then rebuilding them. Age structures presented here provide a baseline for rebuilding capacity for egg production, although other options for baselines may be age structures obtained from indigenous middens (Disspain 2009). Restoration targets should allow for establishment of new age classes in population age structures. Options for establishment of new age classes include securing freshwater flows to end-river and estuarine habitats and protection of juveniles in estuarine habitat and reproductively mature adults in spawning/pre-spawning aggregations. Age structures should be monitored to determine the success of rebuilding of these populations.

Rebuilding age structures becomes especially important in the context of climate change where droughts are predicted to increase in severity and frequency and will severely impact estuarine and associated nearshore environments (Hughes 2000; Gleick 2003; Hughes 2003). Climate change has the potential to impact local populations along range boundaries (Hughes 2000) and populations such as *A. japonicus* in South Australia, which occurs at the southern most part of the global distribution, may be particularly affected.

Sub-structuring of the stock of *A. japonicus* also has implications for management. Differences in growth rates and habitat use indicate that these populations in western and eastern South Australia should be managed differently. Although the population on the west coast has higher growth rates it does not have access to protected habitat in estuaries. However, juveniles in the population on the west coast do not experience levels of fishing mortality as high as those in the Murray River estuary because the legal minimum size for *A. japonicus* in marine waters of South Australia is 750 mm TL.

The key management question should be "what is the best management decision that can be made with limited data" rather than "what data do we need to make a sound management

decision" (Johannes 1998). As implementation of Ecosystem Based Fishery Management increasingly occurs invocation of the Precautionary Principle (FAO 1996) may be required for protection of fishes in highly degraded, and changeable environments such as estuaries.

Fishes as indicators of estuarine health

Results from this study suggest that an effective fish based indicator of estuarine health may be provided by a combination of historical fishery catch data (assemblage), and detailed knowledge of an appropriate indicator species (population). Historical catches may have the advantage of providing longer term data than research-fishing based projects, and thus may be useful for setting restoration targets for exploited species.

Commercial catch ranked by species value is recommended for Australian state of the environment reporting of ecosystem condition (Ward et al. 1998). However such an indicator based on fish catches does not take into account the impacts of changes in fishery management and assumes that where management controls are effective, the mixture and rank of species will be relatively stable. In this study, fish catches provided an effective indicator of long-term change (25 years) in fish assemblages with the caveat that significant decline had occurred in at least one periodic strategist (Murray cod, *Maccullochella peelii peelii*) prior to collection of data.

Demographic information from populations of long-lived species may also provide an indicator for the health of habitats in end-river environments. For such a species knowledge of the geographic structure of the stock, habitat association, and life-history is needed. In the Murray River estuary, age distributions of adult *A. japonicus* from the nearshore environment may provide an indicator of environmental health because its reproductive success may be dependent on freshwater inflows (Chapter 4), and because it is the apex predator in the Murray River estuary (Deegan et al. 2010). Such species-based indicators could also be developed for golden perch (*Macquaria ambigua*) and black bream (*Acanthopagrus butcheri*) with improved knowledge of stock structure and life-history.

Future research

To prioritise management action, vulnerable species or populations need to be identified, and for each species there is a need for knowledge of stock structure, population demographics and population dynamics.

Results from this study highlight a number of specific areas that require future research.

- There is a need for better understanding of the population structure of *A. japonicus* across the entire geographic range in Australia, to inform management of this species. Inconsistent results from previous genetic studies may have been due to small sample sizes from limited geographic ranges (Dixon 1990; Farmer et al. 2005; Archangi 2008). Consequently a study across Australia with appropriate regional sub-sampling is needed. Of the genetic studies to date, limited sampling has been done in South Australia has been done and none in Victoria. However, it is possible that biogeographic barriers to dispersal may occur in these regions. Otolith based methods (shape, elemental composition), combined with recently developed molecular genetic markers (Archangi et al. 2009) may provide the best approach. Molecular genetic techniques combined with recent statistical approaches such as Bayesian clustering techniques may also provide indications of demographic connectivity among populations.

- Establishing the 'nursery' role of the Murray River estuary for *A. japonicus* requires: (i) an understanding of movement of juveniles from the estuary into the adult population in the marine habitat; and (ii) improved understanding of rates of growth and mortality of juveniles in protected estuarine habitat and alternative habitat in the adjacent nearshore marine environment. Such a study could include: (i) quantifying the contribution of juveniles from estuarine habitat in the Murray River estuary; to the adult population in the marine environment; (ii) comparison of growth rates of juveniles in estuarine and alternative nearshore marine habitats; (iii) comparison of rates of juvenile growth and mortality in western and eastern coast populations; (iv) comparison of a time series of age structures of adults from western and eastern South Australia to better understand recruitment in these two populations, and/or comparison of growth increments estimated from the internal structure of otoliths between samples from indigenous middens and recent catches.

- Early juveniles (< 12 cm) of *A. japonicus* have not been observed in South Australia and the location of larval and early juvenile habitat is poorly understood. Plankton tows conducted adjacent to the Murray River mouth during spring-summer when spawning aggregations of adult *A. japonicus* are thought to occur may improve knowledge of the location of spawning and early juvenile habitat. Alternatively a fine mesh beam trawl could be used to sample the Murray River estuary following this period.

- Improved understanding of use of estuarine and nearshore marine habitat by juvenile *A. japonicus* is needed and includes (i) the proportion of time that juveniles spend in

estuarine, and alternative nearshore marine habitat, (ii) age/size at which individuals move from estuarine habitat and use nearshore marine habitat exclusively, and (iii) whether these factors change in response to environmental conditions within the estuary. A telemetry based study, similar to that of Cowley (2008) may help to address these questions.

- Improved understanding of estuarine habitat use by *A. japonicus* may be achieved by comparing concentrations of oxygen isotopes (which may indicate environmental temperature and salinity) between (i) historical samples (from indigenous middens) with recent samples (from catches) for both eastern and western populations.

- There is a need for a reliable estimate of age/size at maturity for *A. japonicus* on the western coast. If age at maturity is the same for *A. japonicus* on western and eastern coasts, size at maturity may be higher for the west coast population because growth rates are higher. Consequently, an appropriate Legal Minimum Size may be higher than required for the eastern population.

Conclusions

Results from this study highlight the importance of long-term data for investigating environmental impacts on fish assemblages and populations in changeable environments. While river regulation, drought and overfishing impact fish assemblages, species with life-histories that include slow growth, delayed maturity and longevity are particularly vulnerable. While fishery statistics may suggest declining abundance of exploited species, population age structures are key to understanding the extent of impacts of (i) sequential years of poor recruitment due to river regulation and sustained drought, and (ii) compromised egg production from fishing. Such age structures also provide baseline information for setting restoration targets for these species.

Establishing baselines for assessing anthropogenic impacts on fish populations and their habitats and for setting restoration targets is difficult because significant changes may have occurred prior to collection of information. Consequently a suite of information such as fish assemblages from historical fishery catch data, published life-history information, and demographic information targeted at likely vulnerable species may provide a baseline indicator of estuarine environmental health for restoration of degraded end-river habitats. Results of this study highlight the importance of regularly updated demographic information for vulnerable species.

The population of *A. japonicus* in South Australia provided a case study for anthropogenic impacts on a fish species with a periodic life-history strategy. In this study several knowledge gaps were addressed. Otolith-based investigation of stock structure indicated that several populations of *A. japonicus* may occur in South Australia. The methods used to determine stock structure may also provide a rapid, cost effective method with the potential to describe population sub-structure for other species, and to assist in identifying hatchery reared fish that may have recruited to wild populations. An investigation of the annual freshwater flows in the Murray River and recruitment of *A. japonicus* indicated that estuarine habitat in the Murray River system was particularly important for the population in eastern South Australia. This population may be dependent on years of high freshwater inflow to the Murray River estuary for establishment of a strong year class, with years of low freshwater flows and closure of the estuary mouth associated with recruitment failure. Finally, the first validated, otolith-based estimates of growth and size at maturity for *A. japonicus* in South Australia indicated that growth rates are the lowest reported for this species, age/size at maturity was significantly higher than previously thought, and that changes to management of this species in South Australia are required.

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