Carp Exclusion Screens on wetland inlets: their value for control of common carp (*Cyprinus carpio* L.) and effects on offstream movements by other fish species in the River Murray, Australia

Karl Aaron Hillyard

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Contents

Abstract

Carp screens are used in the Murray-Darling Basin (MDB) to prevent invasive alien common carp, *Cyprinus carpio* L. from entering wetlands, minimising their ecological impacts and denying them access to spawning habitat. The effectiveness of existing screen designs has not been evaluated, however, and little is known of their incidental effects on the lateral (instream-offstream) movements of other fish and aquatic fauna.

This study investigates new screen designs that are based on carp morphology, with allowances for the lateral movements of carp and other species. The aims were (1) to determine the spatial and (2) temporal nature of lateral fish movements in the River Murray, South Australia, (3) to describe the location and design of existing carp screens across the Murray-Darling Basin, (4) develop new designs optimised to prevent the passage of sexually-mature carp, and (5) to compare and evaluate the new and existing designs.

The spatial movements of fish between the Murray and six perennially-inundated wetlands were monitored using directional fyke nets set in wetland inlets from August to November 2006. Some 210,000 juvenile and adult fish from 18 species (14 native, four alien) were recorded over 13 weeks. The spatial movements of fish varied among wetlands, despite the proximity of the wetlands to each other, and showed no consistent directionality. This may have reflected the prevailing low-flow conditions (hence the absence of flow-related cues for movements), the virtually permanent connections between the wetlands and channel (maintained by weirs and other regulating structures) and the predominance of 'generalist' species with broad habitat requirements. Temporal movements generally were haphazard, but several small-bodied species increased in abundance over the 13 weeks, co-incidentally in response to increasing water temperature and day length. For most species, the balance of directional movements was from, rather than to, wetlands, possibly in response to falling water levels.

Existing carp screens are concentrated along the Murray in South Australia, but are used throughout the Murray-Darling Basin. In a Basin-wide survey, 54 carp screens were located, including eight mesh designs and varied dimensions. Morphometric data (from fish captured in the lateral-movement study) were used to develop two new designs to exclude sexually-mature carp (44-mm square grid mesh, 'jail bar' mesh with 31.4-mm gaps). Up to 92% of carp captured in the lateral-movement study would have been excluded by either new design.

The new screen designs and the most common existing design (Alu-Tread® walkway mesh) were trialled using directional fyke nets at 12 wetlands on the Lower Murray. The new designs allowed

the passage of more small species (native species <250 mm total length) than the existing design, and excluded all adult common carp and turtles (Chelidae).

The new designs proposed here could be applied to, or adapted for, other carp-control technologies such as carp-separation cages. The ‘jail bars’ design especially is promising, but requires testing *in situ*, during higher flow conditions. In further development, extended monitoring of lateral movements could identify key time periods when carp screens need to be operated. Trials are also recommended to demonstrate the likely improvements in wetland water quality gained from use of carp screens.

Declaration

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1. Chapter One - GENERAL INTRODUCTION

1.1. Common carp - an invasive species

The common carp, *Cyprinus carpio* L. (hereafter 'carp'), originally from Europe and Asia, was domesticated as a source of food in Europe some 2000 years ago, and more recently as ornamental 'Koi' carp in Japan (Balon 1995a; Balon 1995b; Balon 2004). Today, it is part of worldwide aquaculture industries yielding about 33 million tonnes annually (FAO 2010), and the basis of commercial and recreational fisheries in many regions (e.g. Arlinghaus 2007). It is established in 91 of 121 countries where it has been introduced (Casal 2006), including parts of Europe (Copp *et al.* 2005), North and South America (Zambrano *et al.* 2006), Africa (Moreau and Costa-Pierce 1997) and Australasia (Koehn 2004; Tempero *et al.* 2006; Rowe 2007). In most of these regions it occurs as self-sustaining populations (Lever 1994). Indeed, carp now occur on every continent except Antarctica.

Carp are members of the Cyprinidae, allied to goldfish *Carassius auratus*, roach *Rutilus rutilus* and tench *Tinca tinca*. They are ecological 'generalists', able to live in a wide range of habitats (Koehn *et al.* 2000), and have a broad tolerance to environmental variations (Koehn 2004). They can survive temperatures between 1 and 35°C and tolerate a pH range from 5 to 9 (Billard 1995). Adults survive in water of 12.5 g L^{-1} (part per thousand) salinity (Geddes 1979), but salinities above 8.3 g L^{-1} affect sperm motility, hence reproduction (Whiterod and Walker 2006). Carp have a broad, wide ranging diet, feeding mostly on aquatic invertebrates, detritus and plant material (Vilizzi 1998b; Khan 2003). As carp develop beyond the planktivorous larval stage they assume a benthivorous filter feeding habit (Vilizzi 1998b), using their pharyngeal teeth to grind food items (Sibbing 1988). Carp reach maturity quickly (median males 1.1 years, 307 mm; females 2.7 years, 328 mm Fork Length (FL): Brown *et al.* 2005), especially in warm (i.e. tropical) water (Davies and Hanyu 1986); are highly fecund (0.163 million eggs per kilogram body weight: Sivakumaran *et al.* 2003) and are able to spawn multiple times each breeding season (Smith and Walker 2004). The eggs are spawned over aquatic vegetation in warm, shallow offstream areas such as wetlands (Koehn *et al.* 2000; Stuart and Jones 2006a).

Carp were initially released into Australian waters in the 1860s by 'acclimatisation' societies in Victoria and New South Wales, although the success of these releases is unclear (Koehn *et al.* 2000). The first indisputably successful introduction was *circa* 1900, into Prospect Reservoir near Sydney (hence, the 'Prospect' strain (genetic variant): Davis *et al.* 1999). In the mid 20th Century, a further introduction to the Murrumbidgee Irrigation Area, New South Wales, established the 'Yanco' strain (Haynes *et al.* 2009). In the 1960s, carp were illegally imported from Germany by Boolarra Fish Farm in Gippsland, Victoria, introducing the 'Boolara' (or, more correctly, 'Boolarra') strain. Boolarra strain carp were later introduced into Lake Hawthorn, on the River Murray floodplain in north-western Victoria, where they gained access to the Murray-Darling Basin (MDB). Flooding in 1974-75 assisted their dispersal throughout the MDB (Koehn *et al.* 2000). The Koi strain also has been released in the MDB (Davis *et al.* 1999). DNA analyses show that carp in the River Murray now are dominated by two of the four strains in the MDB, namely the Boolarra and Yanco strains, although the Prospect strain also occurs (Haynes *et al.* 2009). Sterile hybrids of carp and goldfish occur, but these are rare (Shearer and Mulley 1978; Hume *et al.* 1983a).

Carp are among the most abundant large-bodied fish in south-eastern Australia (Koehn 2004). Indeed, carp account for 58% of the fish biomass in the rivers of the MDB (Davies *et al.* 2008). It is this high biomass which, coupled with its adverse environmental impacts, has seen carp labelled a pest in Australia. As such, carp fit the description of an ‘invasive’ species, typically those that exist outside their native range and have an adverse impact on the environment or other biota (Mack *et al.* 2000; Mooney 2005). Approximately 80% of carp introductions result in viable populations, of which approximately 85% cause environmental damage (García-Berthou *et al.* 2005).

Carp have invaded most of the MDB, and their impacts on the environment have been documented (Table 1.1) (Koehn 2004). The invasion reflects its biological attributes, typical of invasive species. These include high reproductive capacity, long breeding and growth season, wide environmental tolerance, broad diet, lack of predators as adults, longevity and a gregarious nature (Koehn 2004). The invasion has been assisted by the effects of river regulation (Gehrke *et al.* 1995; Gehrke and Harris 2001). River levels have been stabilised by weirs, maintaining permanently flooded wetlands that are ideal for feeding, growth, spawning and nursery habitat for carp (Driver *et al.* 2005b; Stuart and Jones 2006a). It is in these warm, shallow habitats that the adverse impacts of carp are most pronounced (Gehrke *et al.* 1995; Koehn *et al.* 2000).

Table 1.1. Evidence to demonstrate the negative impacts of carp in Australia, in lentic and lotic habitats. The asterisks represent: * anecdotal evidence only, ** survey and/or dietary studies, *** artificial and/or tank experiments , **** field experimental studies (adapted from King 1995; Koehn *et al.* 2000).

Impact	Habitat type	
	Lentic	Lotic
Turbidity	****	**
Macrophytes	****	**
Macroinvertebrates	****	*
Phytoplankton	****	*
Interactions with native fish	**	**
Bank erosion	*	*
Nutrient concentrations	****	***

Carp are often implicated in adverse environmental impacts (Casal 2006), though clear evidence exists for only a limited number (Table 1.1). Carp modify their environment by ‘top-down’ and ‘bottom-up’ processes (Weber and Brown 2009) (Fig. 1.1), which has seen them described as ‘ecosystem engineers’ (Matsuzaki *et al.* 2009). Their filter feeding behaviour (‘mumbling’) (Sibbing 1988) can damage shallow-rooted, soft-leaved submerged macrophytes by undermining roots, and through direct herbivory, reported in the MDB (Fletcher *et al.* 1985; Roberts *et al.* 1995) and internationally (e.g. Sidorkewicj *et al.* 1998). Carp feeding behaviour has also been shown to increase turbidity both in the MDB (King *et al.* 1997) and in other parts of the world (e.g. Breukelaar *et al.* 1994), and this may suppress the growth of aquatic macrophytes by inhibiting photosynthesis (Zambrano *et al.* 2001). Liberation of nutrients into the water column by carp excretion and re-suspension by way of feeding has been shown in the MDB (Driver *et al.* 2005a) and internationally (Matsuzaki *et al.* 2007), potentially increasing phytoplankton blooms (MDB: Gehrke and Harris 1994; internationally: Roozen *et al.* 2007).

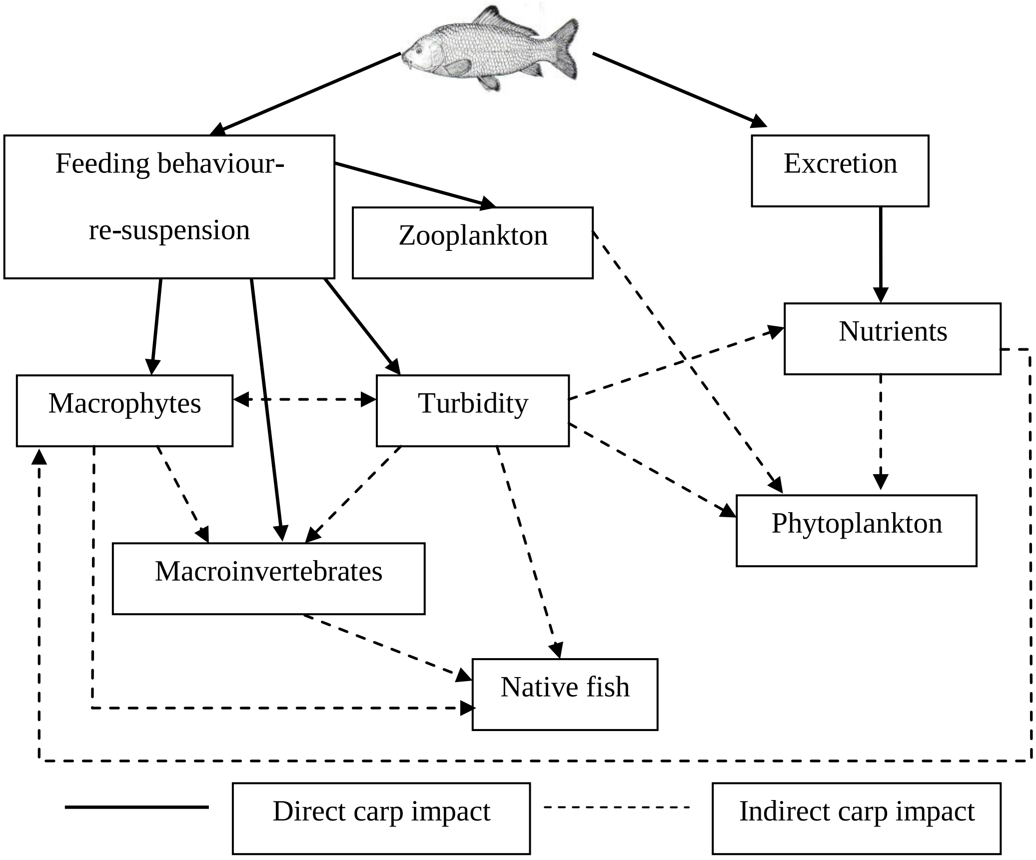

Figure 1.1. Schematic of carp mediated interactions and impacts in the aquatic environment (adapted from King 1995; Koehn *et al.* 2000).

The feeding behaviour and diet of carp may impact on native fauna. For example, it may have contributed to the decline of freshwater catfish *Tandanus tandanus* in the MDB due to similarities in diet and mode of feeding (Clunie and Koehn 2001). In the MDB carp also prey on macroinvertebrates (Khan 2003), and they adversely affect the density and behaviour of decapod crayfish in other parts of the world (Hinojosa-Garro and Zambrano 2004; Bartsch *et al.* 2005). In experimental ponds in North America, carp suppress the growth rate of juvenile fish by competing for food, and may affect larval fish foraging by increasing turbidity (Wolfe *et al.* 2009).

1.2. Wetlands in the MDB and the impacts of carp

Wetlands in the MDB are offstream or floodplain habitats (ephemeral or permanent) such as billabongs, swamps, backwaters, flood-runners or small, shallow natural lakes (Phillips 2005). Wetlands have environmental, economic and social value. Their diverse morphology provides a variety of habitats for fauna and flora (Kingsford 2000). In general, wetlands are also important sites of primary production and decomposition (Brinson *et al.* 1981), and help control floods, act to purify water, stabilise sediments and are a habitat for commercially important species (Woodward and Wui 2001). Socially and economically, wetlands are important for recreation and tourism (Wall 1998), because they are visually pleasing and provide a site for recreational pursuits such as fishing and boating.

Some 35% of the wetland area across the MDB, and almost 70% of the wetland area in the South Australian MDB (Pressey 1986), is permanently connected and inundated, providing ideal carp habitat. Large numbers of carp aggregate in these perennial wetland habitats (Smith *et al.* 2009b), concentrating their impacts. Estimated carp biomass in wetlands of the MDB is typically 100-1500 kg ha^{-1} (Fletcher *et al.* 1985), and as high as 3144 kg ha^{-1} (Harris and Gehrke 1997). Substantial impacts by carp occur at relatively low (*c*. 100 kg ha^{-1}) biomass densities (Matsuzaki *et al.* 2009).

Impacts of carp within permanently inundated wetlands in the MDB have been recorded at Little Duck Lagoon in the South Australian Riverland and at Websters Lagoon on Lindsay Island in north-west Victoria (Jensen 2002; Meredith *et al.* 2006; Wallace *et al.* 2009). The impacts include high turbidity levels and reduction in submerged macrophytes. Improvement in wetland condition has been achieved at these sites by drying the wetland and installing carp screens (Jensen 2002; Wallace *et al.* 2009). There is a need for effective management techniques, especially methods that operate at the wetland scale, to address the impacts of carp.

1.3. Options for carp management

1.3.1. Carp management overview

Efforts to control carp in the MDB "have been sporadic, short lived, and rarely successful" (Stuart *et al.* 2006a). To address the shortcomings of *ad hoc* carp control, integrated management approaches utilising a variety of management technologies are required. Combined approaches focussing on carp spawning habitat, spawning migrations and the effects of river regulation and agriculture may be effective in managing carp (Harris and Gehrke 1997). Integrated management programs in Tasmania, utilising 'Judas' carp (see below), fish screens, blocking nets and traditional fishing methods, have been successful in this way (Diggle *et al.* 2004; Inland Fisheries Service 2004). Further, integrated carp management programs, such as the Invasive Animals Cooperative Research Centre's carp management demonstration program in the Lachlan catchment, NSW, are also being implemented. Effective, scientifically valid control technologies are key to the success of integrated management programs.

Present control options are applicable at the individual wetland or river reach scale rather than a basin-wide scale. Options include ‘Judas’ carp (Diggle *et al.* 2004), Williams’ carp separation cages (Stuart *et al.* 2006a), pushing traps (Thwaites *et al.* 2010), water level manipulation to expose spawned eggs to desiccation (Sheilds 1958), electrical barriers (Verrill and Berry 1995; Smith-Root Inc. undated), non-specific chemical piscicides (e.g. Marking 1992) and carp exclusion screens (French *et al.* 1999; Hillyard *et al.* 2010). The scientific validity and effectiveness of these technologies vary and some would benefit from further development, as explored in greater detail below.

Other, more ‘speculative’ control methods are currently in development. Genetic manipulation to skew gender ratios (e.g. 'daughterless' carp: Thresher 2008) has been considered for carp management, in line with modelling of carp population dynamics (Brown and Walker 2004). Other novel control measures being developed include biological control (e.g. Koi herpes virus: McColl *et al.* 2007), pheromone attractants (Sorensen and Stacey 2004), environmental attractants (Elkins *et al.* 2009) and acoustic attractants/repellents (Leigh Thwaites, SARDI Aquatic Sciences, pers. comm.). A variety of fish ‘barriers’ are also proposed, or are in development, using sound (Popper and Carlson 1998; Bullen and Carlson 2003; Maes *et al.* 2004; Taylor *et al.* 2005; Sonny *et al.* 2006), bubble curtains (Champion *et al.* 2001) and light (Lines and Frost 1999). Field applications of these control methods are years from realisation, owing to the complexity of their development, the cost of research and delivery, and public acceptance. These control methods are not considered further in this thesis. A summary of current or previously used control options that operate on a wetland scale follows.

1.3.2. Existing carp management options

1.3.2.1. Judas carp and traditional fishing

The use of Judas carp involves the release of radio-tagged (typically male) carp that are tracked to summer spawning aggregations, or wintering aggregations. Once identified (3 or more radio-tagged fish found in the same location), aggregations are fished with traditional methods (i.e. netting, electrofishing) (Diggle *et al.* 2004). This method has been successfully applied in Tasmania (e.g. Lakes Crescent and Sorell: Inland Fisheries Service 2004). However, considerable labour is required to successfully control carp using this option, especially removal of the last few individuals. The method has been successfully applied only in closed (e.g. lake) carp populations, limiting its broad-scale feasibility throughout the MDB.

1.3.2.2. Water level drawdown

Carp reproduction can be disrupted by lowering water levels to desiccate eggs spawned in shallow water (Wilson *et al.* 2001). Though successful in the United States (Sheilds 1958), the feasibility of water level drawdown in individual wetlands in the MDB is questionable given that carp spawn up to three batches of eggs over nine months per annum (August to April: Smith and Walker 2004). Suitable wetlands for drawdown are scarce: sites require a flow control structure and either a water level greater than the adjacent river or pumps to achieve a water level reduction. While water level drawdown is also possible at a river weir pool scale, this option is impractical due to numerous irrigation offtakes and the inconvenience for river users (Wilson *et al.* 2001).

1.3.2.3. Piscicides

Numerous piscicides are commercially available (e.g. rotenone, antimycin, AQUI-S®), though not necessarily registered for use in Australia. Piscicides have been successful in controlling carp (Marking 1992; Meronek *et al.* 1996), but their use is complicated by the scale of application, impact on native fauna (e.g. other fish, invertebrates) and potential harm to humans (Clearwater *et al.* 2008).

A non-selective, rotenone-based piscicide (Prentox Prenfish, by Prentiss Inc.) designed for control of carp is available (Ling 2002). To target carp, non-toxic, floating ‘trainer’ baits are used to acclimate carp to feeding from a bait station. After the acclimation period has been completed the non-toxic baits are substituted for piscicide laced baits. Testing of Prentox Prenfish in billabongs, ponds and tanks in the MDB showed little success at controlling carp and resulted in native fish deaths (Gehrke 2003). Broad-scale use of piscicides to manage carp is impractical unless an effective carp-specific toxicant or delivery method is available. Until then, piscicide use (e.g. by spray application of liquid rotenone) will remain limited to small water bodies where the benefits of carp control outweigh impacts on native species.

1.3.3. Wetland inlet carp management options

Carp move between shallow, well vegetated offstream areas where they aggregate to spawn in summer and deeper riverine habitat where they overwinter (cf. Penne and Pierce 2008). These movements occur in the mid-River Murray where wetland access is controlled by river level rises associated with irrigation releases (e.g. Barmah-Millewa forest: Stuart and Jones 2006a; Jones and Stuart 2008a) and also in the Lower Murray (Vilizzi 1998a; Smith and Walker 2004), where carp have yearround access to spawning habitats. The high abundance of carp during spawning tends to amplify their adverse impacts (Gehrke *et al.* 1995; Koehn *et al.* 2000).

The movement of carp between river and offstream sites, through sometimes narrow wetland inlets, provides an ideal opportunity for management. Controlling carp access to wetlands can provide two benefits. First, it reduces carp access to their desired spawning grounds (Stuart and Jones 2006a) and second, it can mitigate localised damage associated with their presence (Jensen 2002). Carp access to wetlands can be controlled at inlets using electrical barriers, traps and screens.

1.3.3.1. Electrical barriers

Electrical barriers use electrical currents to prevent fish movement past a series of electrodes (Smith-Root Inc. undated). Field application of electrical barriers has seen mixed success at preventing entry of fish into lakes in North America. For example, the abundance of carp was lower in two lakes in Minnesota (North and South Heron Lakes) following use of electrical barriers and lake drawdown (Verrill and Berry 1995). In contrast, a costly electrical barrier (*c*. US $29m+: Jackson and Pringle 2010) was unsuccessful in preventing movements of alien Asian carp (bighead carp *Hypophthalmichthys nobilis* and silver carp *H. molitrix*) between the Mississippi river system and the Great Lakes (Stokstad 2003; Rahel 2007; Stokstad 2010). The latter example demonstrates the extreme cost and potential for failure which, along with infrastructure and access requirements at often isolated wetlands, makes electrical barriers a non-viable option for the MDB.

1.3.3.2. Wetland carp separation cages

Jumping traps (carp separation cages (CSCs): Stuart *et al.* 2006a; Stuart *et al.* 2006b) and pushing traps (Thwaites *et al.* 2010) exploit the innate jumping and pushing behaviours of carp. Carp separation cages are currently deployed in fishways along the River Murray (Barrett and Mallen-Cooper 2006; Barrett 2008) and are being adapted for use at regulated offstream wetlands (Smith *et al.* 2009a; Thwaites and Smith 2010). While some success has been demonstrated in the field (Anthony Conallin, The University of Adelaide, pers. comm.), the applicability of carp traps on wetland inlets is dependent on the wetland and inlet morphology, nature of water movement through the wetland, whether the wetland is ephemeral or permanent, logistics of trap installation, ability to dispose of trapped carp, and the ongoing need for trap maintenance (Smith *et al.* 2009a). CSCs catch few native species as jumping has not been observed in large-bodied native fish (Stuart *et al.* 2006a), but their reaction to a pushing trap is unknown (Thwaites *et al.* 2010). Wetland carp separation cages are a promising but still experimental technology.

1.3.3.3. Carp exclusion screens

Carp exclusion screens (hereafter ‘carp screens’) are metal mesh screens placed in wetland inlets, typically in flow control structures such as box- or pipe-culverts (Fig 1.2). Carp screens block carp from moving between river and wetland habitats (cf. Meredith *et al.* 2006). Carp screens are currently deployed in North America (French *et al.* 1999; Lougheed and Chow-Fraser 2001) and

Australia (Jensen 2002), but with little scientific validation (Smith *et al.* 2009b). Specifically, there is little information regarding screen design and the effects on native fishes.

Figure 1.2. Carp screen located at Websters Lagoon, north-west Victoria.

Other physical fish screen designs exist that could operate at wetland inlets. Considerable effort has been made to design appropriate screens to prevent fish entrainment to water intakes for irrigation or on power stations (e.g. Ehrler and Raifsnider 2000; Bestgen *et al.* 2004; McMichael *et al.* 2004; Gale *et al.* 2008; Rose *et al.* 2008; Ho *et al.* 2010). Preventing carp emigration from reservoirs has seen both simple and complex screens devised. Simple mesh screens, akin to carp screen designs discussed herein, albeit with a finer mesh (i.e. 1.1 mm square), have been used to prevent carp and their eggs from leaving Lake Crescent, Tasmania, to the Clyde River (Diggle *et al.* 2004; Inland Fisheries Service 2004). More complex Continuous Deflection Separation (CDS) screens (a rotating barrel screen) have been installed on Rocklands Reservoir, Victoria (http://www.ghcma.vic.gov.au/water/rivers/carp-cyprinus-carpio/; accessed 02 October 2010), to prevent carp from entering the Glenelg River (Glenelg Hopkins Catchment Management Authority 2003). These rotating screens are designed to prevent passage of all carp, including their larval stage, and destroy eggs. However, they also prevent the movement of native fish between the reservoir and river. The disadvantages of these screening technologies over the carp screen designs proposed herein include cost, complexity, need for frequent maintenance to prevent clogging and the exclusion of native fish. The loss of native fish passage caused by CDS and Lake Crescent type screens is offset by preventing carp from establishing in rivers they do not otherwise occupy. The Lake Crescent and Rocklands Reservoir examples contrast with the situation in the MDB where carp are already widespread. Consequently, these screen technologies are not considered further. A discussion follows of the benefits, disadvantages and knowledge gaps regarding 'traditional' (cf. French *et al.* 1999) carp screens.

1.3.3.3.1. Potential advantages of carp exclusion screens

The principal benefit of using carp screens at wetland inlets is the ability to limit carp access to potential spawning habitat (Smith *et al.* 2009a; Hillyard *et al.* 2010), thereby reducing reproduction and their impacts on water quality and macrophytes (Lougheed *et al.* 1998; Lougheed and Chow-Fraser 2001; Lougheed *et al.* 2004). Carp screens also extirpate populations of adult carp *via* stranding during wetland drying (Jensen 2002). Carp screens are a simpler technology than other fish 'barriers' based on sound (Popper and Carlson 1998; Bullen and Carlson 2003; Maes *et al.* 2004; Taylor *et al.* 2005; Sonny *et al.* 2006), electricity (Verrill and Berry 1995; Stokstad 2003), bubbles (Champion *et al.* 2001) or light (Lines and Frost 1999), and can be designed to exploit morphological differences that selectively exclude carp while allowing other species to pass (French *et al.* 1999).

1.3.3.3.2. Disadvantages of carp screens

Screens exclude any aquatic animals (e.g. native fish and turtles) that are too large, unable or unwilling to pass. Further, flow through a screen may cause small-bodied fish to contact the mesh, resulting in injury (Swanson *et al.* 2005), and large-bodied native species that enter as juveniles can grow and become stranded during drying (assuming the screens are not removed or other actions are not undertaken to rescue the fish). Clearly, screens are also not effective during over bank floods, they will not exclude carp that are smaller than the mesh aperture (Navarro and Johnson 1992), and can also become fouled with debris. Debris-fouled screens may also result in increased current velocities, which can be a barrier to fish movement (velocities >0.15 m.s-1 are a barrier to small fishes such as carp gudgeons *Hypseleotris* spp.: Mallen-Cooper 2001).

1.3.3.3.3. Carp screens - current considerations and knowledge gaps

Evidence demonstrating the actual benefits of using carp screens for remediating wetlands (Recknagel *et al.* 1998; Chow-Fraser 2005) and managing carp (Nichols and Gilligan 2003; Smith *et al.* 2009b) is scant. This limitation may relate to the installation of screens at inappropriate locations, or a lack of appropriate operating procedures. For example, carp screens have been fitted to wetlands that have no prospect of drying due to the location of irrigation pumps or underlying porous substrates that maintain wetland water levels *via* seepage (Chapter 4; Hillyard *et al.* 2010). Without the ability to dry a wetland, carp are concentrated and their impacts become pronounced. Integration of pushing elements (one-way gates: Smith *et al.* 2009a; Thwaites *et al.* 2010) into carp screens may prevent the concentration of adult carp in wetlands by allowing them to exit for overwintering in deeper river water (cf. Penne and Pierce 2008).

Recently published guidelines (Smith *et al.* 2009a) will improve future decisions on whether carp screens are a valid management option at a particular wetland, and how they are operated, but several questions remain regarding carp screen design. Only scattered reports exist that detail the distribution and design of carp screens in the MDB (e.g. Nichols and Gilligan 2003), hindering assessment of carp screen efficiency and impacts.

No scientific rationale is available to justify the various carp screen designs currently in use across the MDB. Reasoning for screen designs in current guidelines is limited. For example, the current South Australian wetland flow control structure guidelines suggest screen design should relate to the size of fish that require passage (Allder 2008), but offers no detail relating fish size to screen dimension. Carp screen designs that are underpinned by analysis of both carp and native species morphology promise accurate, length-based exclusion of carp and an understanding of the size of native fish that are likely to be restricted.

Carp screen designs can be customised for use in the MDB using the method described by French *et al.* (1999), who designed a carp screen to target a specific carp size (47 cm TL) and maintain passage of desirable fish. French *et al.* (1999) improved carp screens used at wetland inlets adjacent to the North American Great Lakes by analysing the morphology of carp and the native northern pike *Esox lucius*. This method requires detailed morphological information on carp and other large-bodied species likely to be impacted by carp screens, data which were unavailable prior to the current study (see Chapter 4).

By addressing these three knowledge gaps: (1) inadequate knowledge of current carp screen design, (2) limited scientific rationale to support current screens designs and (3) a lack of morphometric data for carp and large-bodied native fish in Australia; a standard protocol for carp screen design can be implemented across the MDB to prevent movement of carp based on their size at sexual maturity, while having an understanding of the size of native fish that may also be restricted. However, improved knowledge of the fish species that make lateral (between instream and offstream habitats) movements, and morphological data, are required to improve carp screen designs. Improved understanding of the timing of lateral carp movements may also allow discrete temporal application of carp screens or traps.

1.3.4. Potential impact of carp screens on lateral fish movements

1.3.4.1. Overview of lateral movements

Maintenance of hydraulic connectivity within river systems is vital for many riverine fish to complete their life-cycles (Ward and Stanford 1995; Amoros and Bornette 2002; Bunn and Arthington 2002). Fish move among hydraulically connected habitats to access refuges, and for feeding and spawning (e.g. Dodson 1997; Lucas and Baras 2001). While disrupting lateral hydraulic connectivity for carp is the principle purpose of carp screens, movements of other riverine fauna of a size similar to carp (e.g. large-bodied native fish and turtles) are also likely to be affected by carp screens. However, how carp screens impact movements of native fish is unclear due to a poor understanding of lateral fish movements.

Studies assessing lateral fish movements in the MDB and Lower Murray have relied on traditional fishing methods (e.g. fyke nets set in wetland inlets) and tagging *via* external tags or radio telemetry, as described below to establish movement between instream and offstream habitats, though sound evidence to support lateral movements has not necessarily been established. For example, large numbers of fish need to be tagged and significant effort is required to recapture sufficient tagged individuals to demonstrate movements effectively (Gillanders *et al.* 2003). Radio telemetric methods are effective in demonstrating lateral movements but are limited to fish of a sufficient size to be fitted with a transmitter, currently limiting radio telemetry studies to large-bodied species (e.g. golden perch and carp: Crook 2004). The following sections describe lateral movements in the MDB and the Lower Murray. Where present, any limitations (e.g. study method or description) associated with existing descriptions of lateral fish movements are indicated.

1.3.4.2. Lateral fish movements in the MDB

Descriptions of fish movements in the MDB are dominated by reports of longitudinal movements by large-bodied (adult total length (TL) >250 mm), commercially or recreationally popular species (e.g. golden perch *Macquaria ambigua*: Reynolds 1983; Crook 2004; O'Connor *et al.* 2005). Few peer-reviewed papers have been published describing lateral movements of fish in the MDB (Humphries *et al.* 1999).

1.3.4.2.1. Alien species

The lateral movements of carp, of all alien fish in the MDB, have received most attention due to their impact on wetlands. In the mid-River Murray (between Wentworth and Lake Mulwala), radio-telemetry has confirmed that carp move into Barmah-Millewa Forest wetlands seasonally, when the floodplain is inundated by river level rises associated with irrigation flows (Jones and Stuart 2007; Jones and Stuart 2008a). Carp maximise their time on the Barmah floodplain by entering the offstream habitat as soon as water levels allow, and delay their return to the river until the floodplain is nearly disconnected as river level falls (‘first on-last off’: Jones and Stuart 2008a). Carp also respond to seasonal wetland inundation in the upper River Murray (upstream of Lake Mulwala) where they make similar lateral movements to and from wetlands (Lyon *et al.* 2010). Lyon *et al.* (2010) used double-winged (directional) fyke nets set back to back in wetland inlets to capture all species moving laterally (cf. Chapters 2-4), contrasting with methods employed in the Lower Murray which failed to capture all laterally moving fish (cf. Nichols and Gilligan 2003).

Alien species other than carp are also known to utilise floodplain habitats (e.g. McNeil and Closs 2007; Smith *et al.* 2009b) but their lateral movements in the MDB have received little attention. Lyon *et al.* (2010) also reported lateral movements made by other large-bodied alien species (goldfish, redfin perch *Perca fluviatilis*, oriental weatherloach *Misgurnus anguillicaudatus*) and large numbers (*c*. 6000) of the small-bodied (<250 mm TL, and generally 15-100 mm TL) eastern gambusia *Gambusia holbrooki*, again in response to seasonal water level fluctuation.

1.3.4.2.2. Native species

Large-bodied (>*c*. 250 mm TL) native fish inhabit offstream habitats (eg. Jones and Stuart 2008b; Smith *et al.* 2009b), but their movements between the river and offstream sites are not well described. A radio-tracking study of eight juvenile Murray cod *Maccullochella peelii* (formerly *Maccullochella peelii peelii*: Nock *et al.* 2010) in the mid-Murray (Barmah-Millewa Forest) showed that they did not move into floodplain habitat (Jones and Stuart 2007). Conversely, three large-bodied native fish species (Murray cod (contrasting with Jones and Stuart 2007, though age not stated), golden perch, river blackfish *Gadopsis marmoratus*) were captured in directional fyke nets moving laterally in response to river level variation in the upper-Murray (Lyon *et al.* 2010). Three further large-bodied native species (silver perch *Bidyanus bidyanus*, trout cod *Maccullochella macquariensis*, bony herring *Nematalosa erebi*), were captured using nets, stranded behind a floodplain regulator, suggesting that they make lateral movements (Jones and Stuart 2008b).

Lateral movements of small-bodied native fish are poorly understood despite their numerical dominance in many wetlands (e.g. Smith *et al.* 2009b) and the MDB generally (more than three quarters of all fish: Lintermans 2007), and recognition of the importance of wetland habitats for completion of their life-cycle (e.g. King *et al.* 2003). Lyon *et al.* (2010) reported that five small-bodied native species, representing *c.* 96% of the total catch, dominated by carp gudgeons (*c.* 175 000), moved laterally in response to water-level variation.

1.3.4.3. *Lateral fish movements in the Lower Murray*

Lateral fish movements in the Lower Murray (downstream of the Murray-Darling confluence at Wentworth) warrant special mention owing to the high density of carp screens in this region (Chapter 4; Hillyard *et al.* 2010). The fish fauna of the Lower Murray includes at least 35 native and seven alien freshwater species (Hammer and Walker 2004), plus diadromous species whose local range and abundance depend on connectivity between the lowermost reaches of the River Murray, through Lake Alexandrina to the Southern Ocean. The fish community in the Lower Murray channel (Davies *et al.* 2008) and its adjacent wetlands (Smith *et al.* 2009b) have been documented, but lateral fish movements in the Lower Murray (Mallen-Cooper 2001) have received little attention until this study (Conallin *et al.* 2010; Chapter 2 and Chapter 3).

1.3.4.3.1. Alien species

No studies have focussed on movements of carp alone in the Lower Murray, despite the efforts made to limit their access to offstream sites. Reynolds (1983) reported carp's propensity to make lateral movements based on a small number (*n* = 12) of tagged fish released in the river and re-captured in an offstream wetland in the Riverland region (Fig. 1.3). Carp, eastern gambusia, goldfish, redfin perch and carp X goldfish hybrids are also reported to make lateral movements based on a netting survey, also in the Riverland region (Nichols and Gilligan 2003). Carp and eastern gambusia were the dominant species in Nichols and Gilligan's (2003) study, although their results do not allow quantification of the abundance of fish making lateral movements, as catch data were pooled among nets set in wetland, inlet and river habitats. A further weakness of Nichols and Gilligan's (2003) study lies in their netting method, where nets did not completely cover the wetland inlet (cf. Lyon *et al.* 2010; Hillyard *et al*. 2010), preventing quantification of all fish moving laterally. Nichols and Gilligan (2003) also tagged carp (and other alien species) to demonstrate lateral movements, though the few individuals tagged (*n* = 188) and recaptured (*n* = 2) limits the usefulness of these data.

1.3.4.3.2. Native species

Small-bodied Australian smelt *Retropinna semoni*, carp gudgeons and un-specked hardyheads *Craterocephalus stercusmuscarum fulvus*, and large-bodied bony herring are the most abundant native fish making lateral movements in the Lower Murray (Nichols and Gilligan 2003). Most other common native fish also move between instream and offstream habitats (Nichols and Gilligan 2003). Like the data available for carp, the details of lateral native fish movements provided by Nichols and Gilligan (2003) were limited by the study method and reporting.

1.3.4.4. Lateral movements and knowledge gaps

Few peer-reviewed papers concerning lateral movements have been published since Humphries *et al.* (1999) observed that lateral movements were poorly understood. Lateral movements have typically been described where movements appear cued by seasonal river-level fluctuations. In contrast, this thesis describes lateral movements in the Lower Murray where water level fluctuations are limited relative to elsewhere in the MDB, and where carp screens are concentrated (Jensen 2002; Nichols and Gilligan 2003; Meredith *et al.* 2006; Hillyard *et al.* 2010). Prior to this thesis, lateral fish movements in the Lower Murray had been poorly described owing to the cursory treatment of lateral movements in Reynolds (1983) study and the inadequate experimental design and reporting by Nichols and Gilligan (2003).

Improved understanding of lateral carp movements will allow carp screens to be used more efficiently to target distinct periods of carp movement, should they exist, allowing carp screens to be applied only when carp are moving laterally, thereby limiting impacts on native species. Determining any spatial and temporal differences in carp movements, and any variation in wetland or inlet morphology which may explain these different movement patterns, will allow better investment in carp control.

Details of species other than carp (both native and alien) that move between instream and offstream habitats are largely unknown. Identifying these species and describing the nature of their movements is necessary to determine the potential non-target impacts of carp screens (both existing and new designs). Similarly, monitoring lateral movement patterns of riverine fauna provides an ideal opportunity to collect morphological data from carp and other large-bodied fish to determine new ‘optimised’ designs for carp screens. This thesis addresses key knowledge gaps regarding lateral fish movements particularly of native species which have implications for the use of carp screens.

1.4. Summary of knowledge gaps, thesis aims and structure

1.4.1. Summary of knowledge gaps

Several issues relating to the use of carp screens in the MDB have been identified:

1. no scientific rationale exists for the varied designs of existing carp screens;
2. there is no concise record of where carp screens are used and their designs, which complicates the assessment of the screens’ efficiency and impacts;

3. no attempt has been made in Australia to design or test carp screens based on the morphology of carp, or to target a specific size class (or size classes), despite the availability of a suitable method in peer-reviewed literature;
4. few data exist describing the morphology and population size structure of carp and native fish that move through wetland inlets, hindering the design of screens that target a specific carp size class, and preventing assessment of the impact of carp screens (and flow control structures and wetland carp separation cages) on native species; and
5. the spatial and temporal movement patterns of fish through wetland inlets, particularly in the Lower Murray, has received little attention. Failure to understand these movement patterns, which are likely to be impacted by carp screens (and other flow control structures), complicates recommendations for appropriate location and timing of carp screen use.

1.4.2. Thesis structure

The aim of this thesis was to review the design and application of carp screens at wetland inlets in the MDB. This was achieved by two broad components. The first aim was to describe directional and, to a limited degree, spatial and temporal variation in lateral movement patterns of fish in the Lower River Murray. Following this, the second aim was to create and test new carp screen designs based on the morphology of carp and native species. These two broad aims are addressed in the four data chapters (Chapters 2-5). Chapter 6, the General Discussion, reviews the aims and results of each chapter, highlights key outcomes, and discusses areas requiring further research. An outline of each data chapter is given below.

1.4.2.1. Offstream Movements of Fish during Drought in a Regulated Lowland River (Chapter 2)

Directional and spatial variation in movement patterns of fish through wetland inlets in the Lower Murray, following installation of carp screens (and other flow control structures), have received little attention (cf. Reynolds 1983; Nichols and Gilligan 2003). Failure to understand these movement patterns complicates making recommendations for appropriate location of carp screens. The aim of this chapter was to describe lateral movement patterns of fish, specifically directional movements and spatial variation, between the river channel and six wetland inlets.

1.4.2.2. Connections without Directions: Lateral Movements of Fish in a Drought-Affected Regulated River (Chapter 3)

Where Chapter 2 focuses on spatial variation in fish movements, Chapter 3 describes the temporal and directional variation in fish movements during spring. Understanding temporal variation in carp movements may reveal distinct movement periods, and cues that trigger these movements, allowing screens to be used over shorter time spans, reducing impacts on native fauna. This chapter was originally planned as a comparative seasonal study that aimed to identify river-wetland movements of carp in spring and wetland-river movements of carp in autumn (cf. Penne and Pierce 2008), but low river levels over summer (due to drought) prevented monitoring during autumn. This chapter therefore focuses on river-wetland movements of carp and native species in spring only.

1.4.2.3. Optimising Exclusion Screens to Control Exotic Carp in an Australian Lowland River (Chapter 4)

Carp screens are viewed as a valid carp management tool (e.g. Jensen 2002; Meredith *et al.* 2006) and have been used in the MDB for almost 20 years (Nichols and Gilligan 2003), but have been applied without scientific validation or coordination in their design. Chapter 4 has two broad objectives: to document existing carp screens, and create new carp screen designs. Specifically, Chapter 4 aims to document the current locations and designs of the flow control structures fitted with carp screens in the MDB, and the designs and dimensions of the carp screen mesh in order to determine carp screen designs suitable for later ‘optimisation’, and enable an assessment of the suitability and potential impacts of existing designs. The second aim was to use data on the morphology of carp collected during lateral movement sampling (Chapters 2 and 3) to construct carp screen designs that target sexually mature carp while reducing impacts on native fish. Morphological data from other large-bodied fishes using wetland inlets in the Lower Murray are used to determine the likely size of native and other alien fish that would be excluded by the new carp screen designs.

1.4.2.4. Testing Exclusion Screens to Manage the Alien Invasive Common Carp, **Cyprinus carpio** *L. (Chapter 5)*

Chapter 5 aimed to field test and compare the new carp screen designs developed (Chapter 4) with those most widely-used carp screen design in the MDB to confirm the new designs achieve their intended goals of targeting sexually mature carp while allowing most native fish to pass. Chapter 5 also describes morphometric data for small-bodied fish, not previously reported in Chapter 4, and uses these data to determine the likelihood of small-bodied species being physically excluded by new and existing carp screen designs.

1.4.3. A note on thesis structure

The research chapters of this thesis have been written in a style intended for publication in scientific journals (Chapters 2 and 4 have been published) and can be read as stand-alone studies; thus, there is some repetition among the Introduction and Methods sections. Further, as these chapters have co-authors, they are written in plural.

Each data chapter is preceded by a preface, which includes an abstract, information on the publication status of the chapter at the time of thesis submission, and describes the contributions of all co-authors to the research therein.

All tables and figures appear embedded within the text and the numbering of figures and tables begins at one for each chapter to simplify cross-references to the results. All literature cited in the thesis chapters has been compiled at the end of the thesis and not at the end of each chapter. Appendices are referred to in the text as appropriate.

This thesis complements that of fellow PhD candidate Anthony (Rex) Conallin, whose research focuses on the adaptation of the Williams Carp Separation Cage (Stuart *et al.* 2006a; Stuart *et al.* 2006b) for use in wetland inlets. Both theses concern carp management at wetland inlets and highlight that improved knowledge of lateral fish movements is critical to effectively target carp and reduce impacts on native fish.

1.5. Study Area

1.5.1. The Murray-Darling Basin and Lower Murray

The MDB is Australia’s largest river catchment (1 073 000 km^2 in area). The Murray is its principal river, though the Murray receives significant inflows from the Darling, Murrumbidgee and Goulburn Rivers. Discharge in the MDB is characterised by low and variable flows, driven by a predominantly semi-arid climate (Walker 1992). The River Murray is highly regulated (Maheshwari *et al.* 1995) to provide stable water levels for irrigated agriculture that generates ≈AUS$5 billion annually (Quiggin 2001; Pink 2008).

The Lower Murray (Fig. 1.3), the area where lateral fish movements were studied and carp screens were tested, is the 830-km tract downstream of the Murray and Darling confluence, and is a distinct sub region of the MDB (Walker and Thoms 1993). The Lower Murray is the most regulated section of the MDB, owing to the presence of 10 (of 14) low-level weirs, five river-mouth barrages, and numerous levees and ancillary regulators (Zampatti and Leigh 2004; Walker

2006). Regulation has resulted in approximately 70% of the wetland area becoming perennially inundated (Pressey 1990).

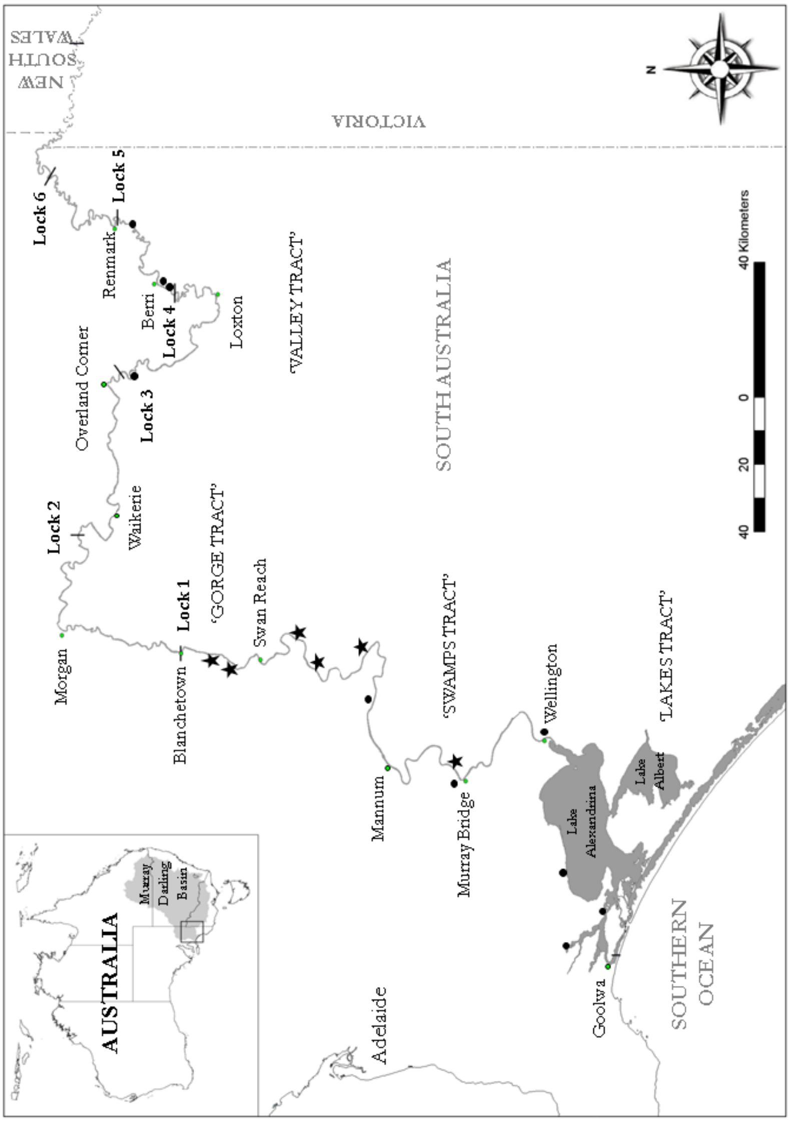

Figure 1.3. Map of South Australian MDB (Lower Murray) showing lateral movement sampling sites (stars, Chapters 2-4) and field carp screen trial locations (black circles, Chapter 5; note that three trials were conducted at the site marked adjacent to Lock 3). Inset Australia map shows the MDB in grey, the area covered by the carp screen survey (Chapter 4).

Four subregions occur within the Lower Murray (Walker and Thoms 1993) (Fig. 1.3). From its confluence with the Darling, the Murray flows through the 'Valley' tract, an area defined by a broad (5-10 km) floodplain which the river meanders across. In this region the floodplain includes many wetlands, anabranches and deflation basins. The Murray enters the 'Gorge' tract (Walker and Thoms, 1993) at Overland Corner; this area is characterised by a constrained floodplain 2-3 km wide with 30 m limestone cliffs on one side and long straight reaches of river interspersed with meanders. Wetlands in this region are characterised as ‘channel margin swales’ (Walker 2006). The ‘Swamps’ tract exists downstream of Mannum. Here the river flows through reclaimed riparian swampland, with fewer meanders and a broader floodplain (Walker and Thoms, 1993). From Wellington the Murray enters the ‘Lower Lakes’. This region is defined by two large, shallow lakes, Alexandrina and Albert (64 900 ha and 17 100 ha respectively). The Lower Lakes are separated from the Murray mouth by five barrages constructed in 1939-1940; these maintain a 450-600 mm increase in the river level under normal regulated conditions.

Flow in the Lower Murray is controlled by inflows from the mid-Murray and the Darling, although two small tributaries, the Finniss River and Currency Creek, which flow to Lake Alexandrina near the Murray mouth, provide localised inflows. Despite regulation, flow in the Lower Murray retains its natural seasonality, but average monthly flows have been greatly reduced and the frequency of low-flows (<5000 ML d^{-1}) is greatly increased (Walker and Thoms 1993). Regulation has reduced the incidence of small floods, but large floods remain unchanged (Maheshwari *et al.* 1995). The last pool of the Lower Murray (between Lock and Weir 1 at Blanchetown and the Barrages near Goolwa), is affected by wind-driven seiches that create variable river levels (±30 cm) causing bi-directional flushing flows through wetland inlets (Webster *et al.* 1997).

The monitoring of lateral fish movements (Chapters 2 & 3) was undertaken at six wetlands within the ‘Gorge’ and ‘Swamps’ tracts of the Lower Murray (Fig. 1.3). The field survey of existing carp screens (Chapter 4) encompassed the entire MDB (Fig. 1.3 inset map). The field testing of new carp screen designs (Chapter 5) was conducted at 12 sites spanning almost the entire length of the South Australian section of the River Murray (Fig. 1.3).

1.5.2. Impacts of over-extraction and drought in the study area during field sampling

From 2001 up to the completion of field sampling for this study, the MDB experienced a severe drought (Bond *et al.* 2008; Murphy and Timbal 2008) which, coupled with extraction for irrigation (Craik and Cleaver 2008), resulted in a period of low, stable, within-channel flows in the Lower Murray. At the start of this study (May 2006), mean river height was 0.81 m (± 0.08 s.d.) (Department of Water, Land and Biodiversity Conservation, unpublished data), which was a typical bankfull level. However, during the assessment of lateral fish movements (spring 2006, chapters 2-3) the flows to the study reach were only 18% and 46% of the 1949-2006 mean and median flows, respectively.

In the summer of 2006-2007 the river level in the reach between Lock 1 and the river-mouth barrages (Fig. 1.3), where lateral movements were assessed (Chapters 2-3), was lowered (*c.* -0.8 m) by reducing down-stream flows to lessen the potential for evaporative water loss, particularly in the large shallow Lower Lakes. This resulted in the 45 wetland complexes below Lock 1, including the study sites, drying for the first time since the construction of the river-mouth barrages (Smith *et al.* 2009b).

2. Chapter Two - OFFSTREAM MOVEMENTS OF FISH DURING DROUGHT IN A REGULATED LOWLAND RIVER

ANTHONY J. CONALLIN,[a b] KARL A. HILLYARD,[a] KEITH F. WALKER,[a] BRONWYN M. GILLANDERS,[a] AND BENJAMIN B. SMITH[b]

[a] *School of Earth & Environmental Sciences, The University of Adelaide,*
South Australia 5005, Australia

[b] *Inland Waters & Catchment Ecology Program, SARDI Aquatic Sciences, PO Box 120, Henley Beach, South Australia 5022, Australia*

Conallin, A. J., Hillyard, K. A., Gillanders, B. M., Walker, K. F., and Smith, B. B. (2010). Offstream movements of fish during drought in a regulated lowland river. *River Research and Applications* doi:10.1002/rra.1419

Statement of Authorship

In this chapter, Karl Hillyard and Anthony Conallin developed the study, collected and analysed the data, and drafted the manuscript. Equal contribution was made by Karl Hillyard and Anthony Conallin. Ben Smith, Bronwyn Gillanders and Keith Walker advised on development of the study, data analysis and contributed to the synthesis and finalisation of the manuscript. Ben Smith also assisted with collection of data in the field.

Certification that the statement of contribution is accurate.

Karl A. Hillyard (Candidate)

Signed *Date* 11/5/2011

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

Anthony J. Conallin

Signed *Date* 11/5/2011

Bronwyn M. Gillanders

Signed *Date* 11/5/2011

Keith F. Walker

Signed *Date* 11/5/2011

Benjamin B. Smith

Signed *Date* 11/5/2011

2.1. ABSTRACT

Access to offstream habitats is vital for many freshwater fish, but details of their lateral movements are scarce. We describe the movements of fish between the channel of the River Murray and six perennially inundated wetlands in South Australia from August to November 2006. At this time there were unprecedented low flows in the river owing to the combined effects of river regulation, drought and over-allocation to upstream users. Some 210 000 fish from 18 species (14 native, 4 alien) were recorded, including two uncommon native species listed by conservation agencies. Movements of juveniles and adults varied among wetlands despite the shared river reach and the proximity of the wetlands to each other, but showed no consistent directionality. This may reflect the prevailing low-flow conditions, the virtually permanent connections between the wetlands and channel maintained by weirs, levees and barrages, and the dominance of ‘generalist’ species. We speculate that movements facilitate efficient resource utilization and nutrient exchange between homogenized river and wetland habitats in the absence of the flood-pulse. We anticipate directional movements will become apparent when flows are increased, so that our data could provide a comparative baseline for future studies. As modifications to natural flow paths may impede access to and from wetlands by fish and other aquatic fauna, provisions for access should be incorporated into flow control structures, used locally to manipulate wetland hydrology.

2.1. INTRODUCTION

Flow regulation has undermined the ecological integrity of many of the world's rivers (Kingsford *et al.* 2006), changing the dynamics of the flood pulse (Junk *et al.* 1989; Tockner *et al.* 1999) and altering patterns of hydraulic connectivity between habitats (Ward and Stanford 1995). In southeastern Australia, the flow regime of the Lower Murray (the 830-km tract below the confluence of the Murray and Darling Rivers: Walker 1992) has been modified by upstream diversions and 10 low-level weirs, five river mouth barrages, floodplain levees and numerous offstream regulators (Walker and Thoms 1993). Although the seasonality of the flow regime is largely intact, regulation has virtually eliminated small to medium floods (<20 000 ML day^{-1}) and maintains low flows (<5000 ML day^{-1}) and bankfull conditions (Walker and Thoms 1993; Maheshwari *et al.* 1995). As a consequence, about 75 per cent of the 1100 floodplain wetlands are perennially inundated and connected to the river, while others are indefinitely isolated (Smith *et al.* 2009b). There have been corresponding ecological changes in river and floodplain environments, and in the flora and fauna. For example, there has been an invasion of wetland biota (e.g. freshwater mussel *Velesunio ambiguus*; crayfish *Cherax destructor* and many plants including *Phragmites australis*, *Typha* spp.) to the main channel (Walker 2006), a loss of some habitat specialists (e.g. olive perchlet *Ambassis agassizii*: Hammer and Walker 2004), an increase in the abundance of habitat generalists (e.g. bony herring *Nematalosa erebi*: Puckridge and Walker 1990), proliferation of alien plants (e.g. Taylor and Ganf 2005) and animals (e.g. Gehrke *et al.* 1995). Perennial inundation has reduced the 'pulse' of wetland productivity associated with less-regulated rivers (Briggs and Maher 1985; Kingsford *et al.* 2006). The biological responses to regulation in the River Murray mirror those in other large regulated rivers in Europe (Aarts *et al.* 2004) and North America (Poff *et al.* 1997).

Riverine fish are vulnerable to the effects of river regulation, as many require longitudinal and lateral connections among habitats to complete their life cycles. For example, off-channel habitats can provide refuge from adverse conditions (Copp 1997; Brown *et al.* 2001), are areas of significant spawning (Molls 1999) and nursery habitat (Bénech and Peňáz 1995) and elevated food resources resulting in higher growth and survival rates compared to the main channel (Sommer *et al.* 2001). In Australia, regulation is implicated in the decline in the range and abundance of native fish populations in the Murray–Darling Basin (MDB) over the last 50 years (e.g. Lintermans 2007), particularly in the Lower Murray where the effects of river regulation are greatest (cf. Lloyd and Walker 1986; Hammer and Walker 2004). Thirty five of 46 fish species in the MDB occur in the Lower Murray, and most occur in both channel (Davies *et al.* 2008) and floodplain habitats (Smith *et al.* 2009b). Whilst floodplain access is important for similar fish assemblages in less regulated areas of the MDB (Balcombe *et al.* 2006) and unregulated rivers such as Cooper Creek in central Australia (Arthington *et al.* 2005; Balcombe and Arthington 2009), the importance of offstream movements in the perennially inundated Lower Murray is less clear (cf. Stuart and Jones 2006a).

Longitudinal fish movements within rivers are well documented (Ward and Stanford 1995; Mallen-Cooper 1999; Ward *et al.* 2001), but less is known of lateral movements between rivers and wetlands (cf. Zeug *et al.* 2009). Lateral fish movement studies in regulated systems have primarily focused on fish moving to 'refugia' during high flows (e.g. Copp 1997) or in response to changing connectivity related to seasonal water-level fluctuations (e.g. Barko *et al.* 2006; Castello 2008; Lyon *et al.* 2010) with few studies (cf. Schultz *et al.* 2007) concerned with movements during stable flows and water levels in perennial inundated systems. Despite the lack of understanding of lateral fish movements in perennial systems, modifications to lateral connectivity are occurring through wetland rehabilitation programmes in the Mississippi (Schultz *et al.* 2007) and the MDB (Jensen 2002). Rehabilitation of perennial wetlands has encouraged construction of inlet regulators (e.g. Chapter 4; Hillyard *et al.* 2010) primarily to restore natural wetting and drying cycles. In addition, a severe, decade-long drought now prevailing in the southern MDB has substantially reduced river flows and prompted closure of many wetlands with regulators and earthen banks to conserve water for 'critical human needs' (e.g. http://tinyurl.com/cxc9ux (accessed 24 February 2010)). The impacts of disconnection and drying of perennial wetlands on the Lower Murray fish community are unknown.

In this study, we examined the movements of fish between the channel of the Lower Murray and six perennially-inundated (under regulation, since *c.* 1940) wetlands in South Australia, under low and stable flow (drought) conditions. Our aims were to (1) describe the fish assemblage utilizing wetland inlets; (2) compare these species with the known regional fish assemblage; (3) determine whether the fish assemblage differed among wetland inlets; (4) compare differences in abundance and size distribution of fish species moving into and out of wetlands and (5) determine if movements were influenced by environmental variables. Based on previous studies on the distribution of fish in the Lower Murray (e.g. Davies *et al.* 2008) and adjacent wetlands (Smith *et al.* 2009b), we anticipated that a diverse assemblage of species would be recorded moving to and from the wetlands, including most of the extant local species. We also expected that assemblages would be similar among wetlands, given their proximity to one another, but that species abundances would differ among wetlands, depending on local variables. Finally, we expected to find differences in the abundance and size of fish moving into and out of wetlands, reflecting ontogenetic habitat shifts and other behavioural responses.

2.2. METHODS

2.2.1. Study area and site descriptions

The study area was a 140-km reach of the Lower Murray between Lake Alexandrina and Lock and Weir 1 (hereafter 'Lock 1') at Blanchetown, South Australia (Figure 2.1). The wetlands are 117–260 km upstream of the river mouth (Table 2.1, Figure 2.1) and have surface areas of 3-85 ha. At the time of sampling, all were permanently connected to the Murray at 'pool level' *via* inlet channels 18–758 m long and 6–35 m wide. Three wetlands had flow-control regulators (two with box culverts, one with concrete pipes) and screens to exclude common carp *Cyprinus carpio* ('carp exclusion screens': Chapter 4; Hillyard *et al.* 2010).

Five of the wetlands (Morgans, Noonawirra, Nildottie, Kroehns, North Purnong) are situated in the 'Murray Gorge' Tract (Walker and Thoms 1993), with a 2–3 km wide floodplain flanked by 30-m limestone cliffs and long straight reaches with some meanders (Figure 2.1). The remaining wetland (Riverglades) is in the 'Lower Swamps' Tract, where the river flows through reclaimed riparian swampland with few meanders and a broader floodplain (Fig. 2.1: Walker and Thoms 1993). There are no weirs within the study reach, but barrages along the seaward margins of Lake Alexandrina maintain a 450–600 mm increase in the river level under normal, regulated-flow conditions. River flows in the study reach are controlled by releases through Lock 1 and through the barrages to the sea, and daily water-level changes of ±30 cm may occur under the influence of south-westerly winds (Webster *et al.* 1997).

A comparative seasonal study was planned to investigate fish movements in spring and autumn 2006–2007, but falling river levels due to drought and upstream diversions in the intervening summer dried the study wetlands for the first time since 1940. Accordingly, data for spring only are reported here.

2.2.2. Collection and processing

Sampling occurred over 13 weeks from mid-August to mid-November 2006. Two custom-made, double-winged, 8-mm stretched mesh fyke nets were employed per wetland. The nets were set for approximately 24 h once weekly, placed back-to-back inside the inlet channel within 50 m of its confluence with the Murray, to trap fish entering and leaving the wetland (Figure 2.2). The nets covered the entire inlet channel and were floated and weighted to prevent fish by-pass. Where present, carp exclusion screens were removed during sampling.

The catch from each net was transferred to 60-L tubs of aerated river water. Fish were identified to species, except carp gudgeons which are an unresolved ‘species complex’ (*Hypseleotris* spp.) (Bertozzi *et al.* 2000). Carp X goldfish *Carassius auratus* hybrids were pooled with carp. Up to 50 individuals of all large-bodied species (>100 mm Total Length, TL, at maturity) were randomly sub-sampled and measured (TL, mm) and the remaining fish were counted. Due to high abundances of small-bodied fish, up to 50 individuals of each species were randomly sub-sampled, measured (TL, mm) and bulk-weighed, and an average individual weight calculated. The remainder of each small-bodied species sample was then bulk-weighed, and total counts for abundance were estimated by extrapolation.

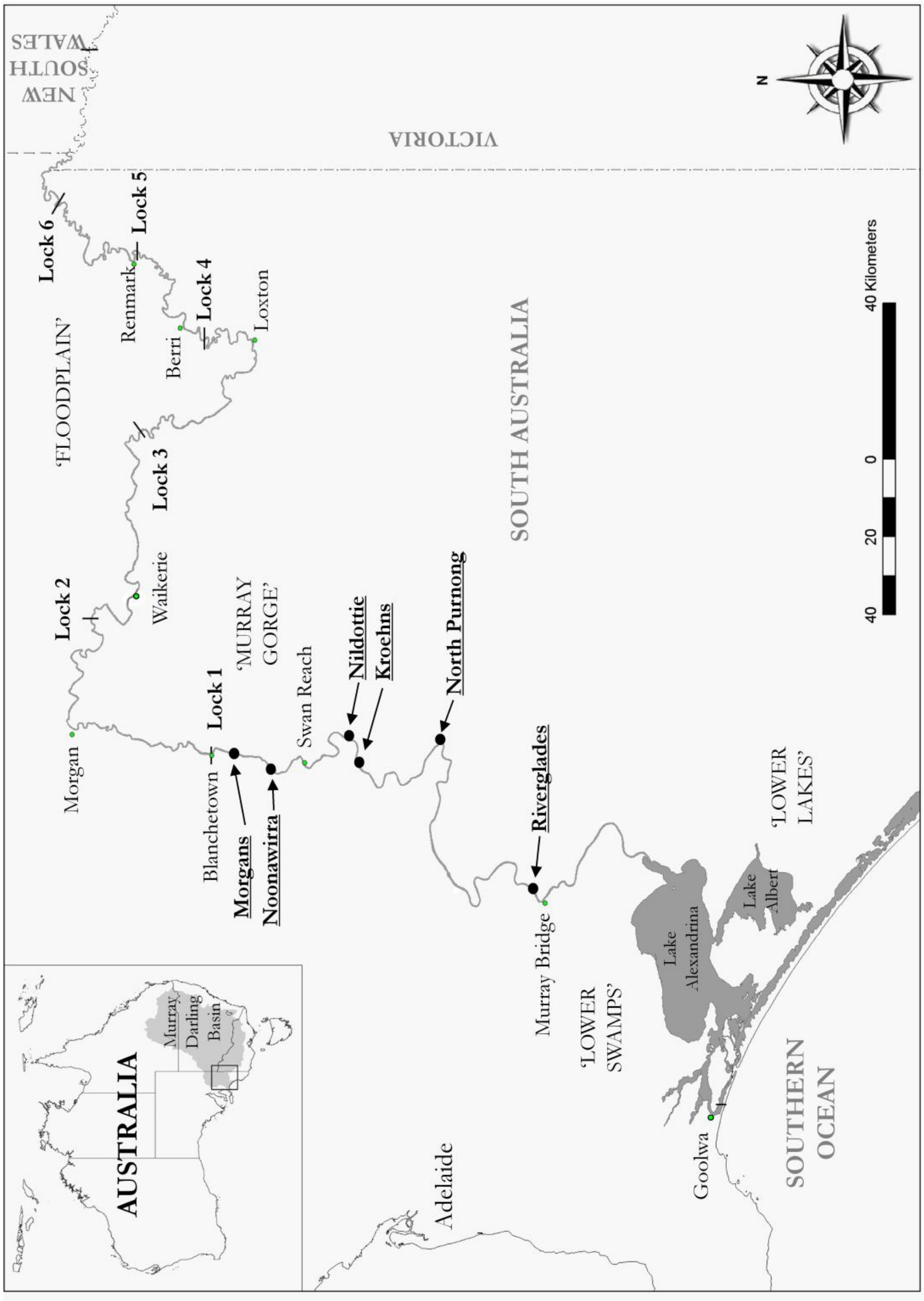

Figure 2.1. The River Murray in South Australia, showing major towns and the six wetlands surveyed in this study (bold and underlined).

Table 2.1. Characteristics of the six wetlands sampled, including subjective descriptors (relative to each other) of the wetland and inlet physical features.

Description	Morgans	Noonawirra	Nildottie	Kroehns	North Purnong	Riverglades
Wetland area (ha)	23	6	3	46	85	21
Distance to river mouth (km)	260	255	225	215	194	117
Inlet length (m)	184	60	18	758	272	165
Inlet width at net location (m)	8	35	6	8	7	15
Shoreline development index	1.94	3.17	3.32	2.40	2.90	2.91
Presence of a regulator / type	6-cell box culvert	5-cell pipe culvert	No	No	No	6-cell box culvert
Woody debris (1-5, absent-abundant)	1	2	5	3	2	2
Emergent vegetation(1-5, sparse-abundant)	1	2	3	5	3	4
Submerged vegetation(1-5, absent-abundant)	1	2	5	1	2	4
Relative ‘access’ of inlet (1-5, low-high)	1	3	5	1	4	5
Relative depth of inlet (1-5, shallow-deep)	1	1	1	3	4	5

2.2.3. Environmental data

Electrical Conductivity (EC) was measured weekly in the wetland, the inlet and the river using a TPS 90 FL–T water quality meter (TPS, Springwood, Queensland, Australia). Hydrological data were obtained from SA Water gauging stations at Blanchetown, Swan Reach and Goolwa (Figure 2.1). River height (stage) data for Swan Reach were obtained from the Department of Water, Land and Biodiversity Conservation, Adelaide, and compared to data from flow/depth loggers (Starflow loggers: cf. Webster *et al.* 1997) in three of the six wetland inlets; a close relationship between river and inlet stage enabled descriptions of flow in all six inlets. Stage data were also used to describe relative inlet depth, variability in depth (Coefficient of Variation, CV), the duration that the water level was rising or falling and the relative flow magnitudes through each inlet during each net-set period. Wind speed/direction data were obtained from the Bureau of Meteorology to account for seiche effects on river stage. Wetland areas (ha) and shoreline development indices (Dodd 2002) were calculated using aerial imagery from Google Earth®. The index was calculated as the ratio of the length of the shoreline of a wetland to the length of the circumference of a circle of area equal to that of the wetland. Other factors describing individual wetlands included the distance from the river mouth and scored subjective assessments of the relative abundance and complexity of woody debris and emergent and submerged vegetation, inlet depth and inlet ‘access’ (i.e. the density of macrophytes, woody debris and other structures that could hinder fish passage) (Table 2.1).

Figure 2.2. Arrangement of back-to-back fyke nets set in wetland inlet used in lateral fish movement sampling. Photo shows Nildottie wetland inlet, facing the river.

2.2.4. Data analysis

Records of individual fish species moving between wetlands and the river were compared to recent fish surveys (Davies *et al.* 2008; Smith *et al.* 2009b) in the Lower Murray. Data were standardized for Catch Per Unit Effort (CPUE) prior to analysis, using a standard set time of 24 h for each net set. Relative numbers of individual species were used as a descriptor of the fauna at each inlet, and were derived from pooled abundance data over the 13 weeks of sampling (refer to Chapter 3 for similar temporal analysis of these data).

A Non-metric Multidimensional Scaling (NMS) ordination using the Sorensen (Bray–Curtis) metric was performed (PC-ORD v. 4.36: MjM Software, Oregon, USA) on the CPUE data to display the relative differences among samples. The fish assemblage (primary) data were then correlated with ordination scores and presented as ‘joint plots’. The line length and angle indicate the relative strength and direction of species correlations in ordination space (McCune and Grace 2002). Prior to analysis, data were fourth-root transformed to reduce the influence of abundant species (Clarke and Warwick 2001).

To compare the composition and directionality of assemblages moving to and from the wetlands, two-way PERmutational Multivariate ANalysis Of VAriance (PERMANOVA) was applied, using PRIMER v. 6 (Plymouth Marine Laboratories, UK). The data were fourth-root transformed and converted to a Bray–Curtis distance matrix, and the test statistic (essentially an F ratio) was calculated as the ratio of the sum of squared distances among groups to the sum of squared distances within groups (Anderson 2001; McArdle and Anderson 2001).

Two-way ANalysis Of VAriance (ANOVA) were used to compare total species richness, abundances and individual abundances of the six most common (three small-bodied and three large-bodied) species among wetlands and the direction of movement. Sampling weeks were treated as replicates. Data were $\log_{10}(x + 1)$ transformed and checked for normality (graphical inspection) and heterogeneity of variances (Cochran's C test). Analyses were still performed if variances remained heterogeneous after transformation, as ANOVA is a robust method where (as in this case) data are balanced and sample sizes are relatively large (Underwood 1997). The significance level (α) was reduced from 0.05 to 0.01 to lessen the chance of Type I error. Student-Newman-Keuls (SNK) post hoc tests were conducted where appropriate and considered significant if $p < 0.05$.

The size distributions of dominant species moving to and from wetlands were compared using a two-tailed Kolmogorov–Smirnov large-sample test (Sokal and Rohlf 1994). Fish-length data were pooled over wetlands and weeks to ensure large sample sizes ($n > 30$). Mature female body size data for Australian smelt *Retropinna semoni*, un-specked hardyhead *Craterocephalus stercusmuscarum fulvus* and carp gudgeons were from Pusey *et al.* (2004), data for bony herring were from Puckridge and Walker (1990) and data for golden perch *Macquaria ambigua ambigua* and common carp were from Mallen-Cooper and Stuart (2003) and Brown *et al.* (2005), respectively.

2.2.5. Relationships with environmental variables

Spearman's rank correlation coefficients were calculated among biological variables (total species richness, mean species richness, mean abundance of all species and individual abundances of dominant species) and physical variables (wetland size, distance from the river mouth and inlet length). In addition, the fish assemblage NMS plot was overlaid with a secondary matrix of environmental variables to show correlation vectors with the ordination axes.

2.3. RESULTS

2.3.1. Assemblage data

2.3.1.1. Species presence/absence and richness

Eighteen species (14 native, 4 alien) were recorded moving between the river and the wetlands (Table 2.2), including two native species (freshwater catfish *Tandanus tandanus*, Murray cod *Maccullochella peelii*), which have current conservation listings. Eight native species and four alien species were recorded from the inlets of all six wetlands (Table 2.2). In contrast, five native species were restricted to the inlets of single wetlands: these were Murray cod and freshwater catfish at Kroehns, congolli *Pseudaphritis urvillii* and pouched lamprey *Geotria australis* at North Purnong and small-mouthed hardyhead *Atherinosoma microstoma* at Riverglades. Common galaxias *Galaxias maculatus* were recorded from three wetlands, namely Kroehns, North Purnong and Riverglades. The total numbers of species sampled ranged from 15 at Kroehns and North Purnong to 12 at Morgans, Noonawirra and Nildottie (Table 2.2). Mean species richness also differed among wetlands (Figure 2.3, Table 2.3), with consistently more species recorded at Riverglades than at other wetlands (SNK tests). Conversely, Morgans had fewer species compared to other wetlands, but was similar to Noonawirra.

Table 2.2. CPUE abundance, total abundance, and species richness of fish collected moving in and out of six wetland inlets in the Lower Murray SA between August and November 2006. Note small-bodied <≈250 mm TL, large bodied >≈250 mm TL maximum length.

Species name	Scientific name	Morgans		Noonawirra		Nildottie		Kroehns		North Purnong		Riverglades		Total	
		In	Out	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
Small-bodied															
Australian smelt	*Retropinna semoni*	4097	7890	7822	8929	5634	4275	3648	26088	18740	25176	20419	15128	60361	87487
Carp gudgeons	*Hypseleotris spp.*	284	213	1041	1241	3671	5389	654	377	1864	1150	2111	313	9625	8682
Un-specked hardyhead	*Craterocephalus stercusmuscarum fulvus*	70	47	50	345	3884	2744	259	544	1008	2497	289	2764	5560	8941
Flat-headed gudgeon	*Philypnodon grandiceps*	8	15	15	58	90	74	62	29	49	42	393	1156	617	1375
Murray-Darling rainbowfish	*Melanotaenia fluviatilis*	8	1	13	27	286	176	5	14	27	25	6	34	345	276
Dwarf flat-headed gudgeon	*Philypnodon macrostomus*	6	19	25	33	17	30	7	19	6	17	87	245	148	362
Common galaxias	*Galaxias maculatus*	0	0	0	0	0	0	0	1	0	1	32	86	32	88
Eastern gambusia*	*Gambusia holbrooki*	46	8	7	6	11	0	3	0	2	0	1	6	70	19
Small-mouthed hardyhead	*Atherinosoma microstoma*	0	0	0	0	0	0	0	0	0	0	1	0	1	0
Large-bodied															
Bony herring	*Nematalosa erebi*	932	2814	1344	1157	492	2210	2830	7110	2103	3368	490	449	8192	17107
Golden perch	*Macquaria ambigua ambigua*	6	12	25	25	13	8	26	23	27	25	93	129	189	222
Common carp*	*Cyprinus carpio*	9	30	9	25	29	9	41	98	21	57	11	3	121	223
Redfin perch*	*Perca fluviatilis*	13	41	6	10	10	6	4	67	28	12	14	8	75	144
Goldfish*	*Carassius auratus*	1	0	13	27	11	1	26	11	11	5	98	7	161	51
Pouched lamprey	*Geotria australis*	0	0	0	0	0	0	0	0	1	1	0	0	1	1
Congolli	*Pseudaphritis urvillii*	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Freshwater catfish	*Tandanus tandanus*	0	0	0	0	0	0	1	0	0	0	0	0	1	0
Murray cod	*Mucculloch ella peelii*	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Total in & out		**5482**	**11089**	**10371**	**11882**	**14150**	**14922**	**7566**	**34382**	**23886**	**32377**	**24045**	**20328**	**85499**	**124979**
Total wetland		**16570**		**22253**		**29072**		**41947**		**56263**		**44373**		**210479**	
Species richness		**12**		**12**		**12**		**15**		**15**		**14**		**18**	

* alien species

2.3.1.2. Abundance

Of 210 479 (CPUE) fish sampled, 70% were Australian smelt, 12% bony herring, 9% carp gudgeons and 7% un-specked hardyhead (total = 98% of catch, Table 2.2). Total abundance varied from 16 570 at Morgans to 56 263 at North Purnong. Mean abundance of all species also varied among wetlands, but there was an interaction between ‘wetland’ and ‘direction’ (Figure 2.3, Table 2.3). The abundance of fish moving into or out of wetlands was lower at Morgans and Noonawirra compared to other wetlands (SNK tests). Kroehns was the only site to show a difference in abundance relative to direction, having a greater abundance of fish moving out of the wetland. Due to the dominance of Australian smelt, a strongly schooling species (Lintermans 2007), an analysis of mean abundance without this species was conducted. With Australian smelt removed, mean abundance differed among all wetlands, but not for direction (Figure 2.3, Table 2.3).

2.3.1.3. Abundance of dominant species

The mean abundances of five of the six dominant species (carp gudgeons, un-specked hardyhead, bony herring, common carp, golden perch) varied significantly among wetlands, but not in the direction of their movements (Figure 2.4, Table 2.3). Kroehns had more common carp and bony herring than other wetlands, Riverglades had more golden perch and Nildottie had more carp gudgeons and un-specked hardyheads (SNK tests). Data for Australian smelt indicated an interaction between ‘wetland’ and ‘direction’ (Figure 2.4, Table 2.3). The abundance of Australian smelt moving into wetlands differed between most wetland pairs as did abundance moving out, but Kroehns was the only wetland to show a difference in abundance relative to direction, having more Australian smelt moving out of the wetland.

2.3.1.4. Ordination and Per-MANOVA

The data yielded a 3-D ordination (stress 0.12) with Axes 1 ($r = 0.32$) and 3 ($r = 0.21$) selected for display (Figure 2.5a, b), based on separation of samples into distinct wetland groups. Per-MANOVA confirmed that the individual wetlands differed from each other, but this was not so for the direction of movement (Table 2.3). Pair-wise tests confirmed that all wetland assemblages differed from each other (Student's t, $p < 0.05$).

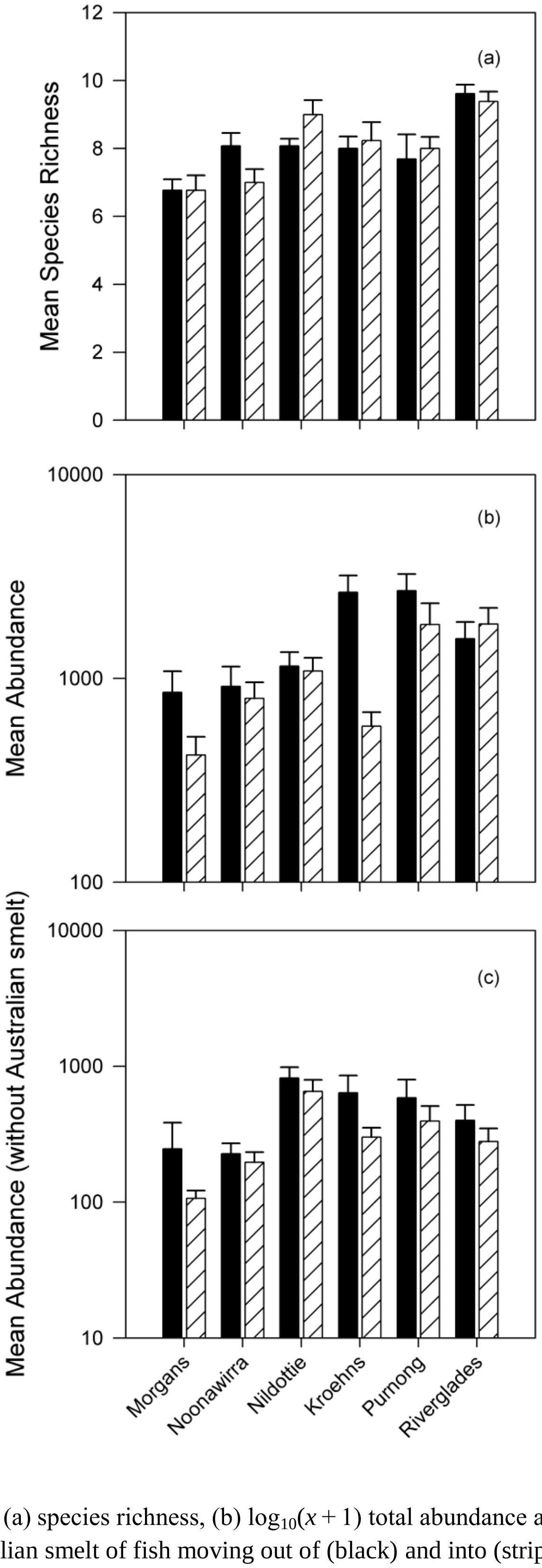

Figure 2.3. Mean (+1 SE) (a) species richness, (b) $\log_{10}(x + 1)$ total abundance and (c) $\log_{10}(x + 1)$ total abundance without Australian smelt of fish moving out of (black) and into (striped columns) the six wetlands sampled from August–November 2006. All abundance values represent fish per net per 24 hours set. Note varying *y*-axis scale.

Table 2.3. Mean squares (MS) and significance levels for two-way ANOVAs of species richness, total fish abundance (with/without Australian smelt), abundance of Australian smelt, bony herring, carp gudgeons, un-specked hardyhead, golden perch, common carp and Per-MANOVA of assemblage by ‘wetland’ and ‘direction’. Significant *p* values highlighted in bold.

Source	Wetland		Direction		Wetland × Direction		Error
	MS	*p*	MS	*p*	MS	*p*	MS
Species richness*	22.06	**<0.0001**	0.05	0.9003	2.75	0.2482	2.04
Total abundance*	1.34	**<0.0001**	1.04	0.2015	0.48	**0.0045**	0.13
Total abundance (excl. Australian smelt)*	1.56	**<0.0001**	0.06	0.3139	0.05	0.9566	0.22
Australian smelt	1.79	**<0.0001**	0.92	0.3914	1.04	**0.0004**	0.22
Bony herring*	2.63	**<0.0001**	0.03	0.7037	0.18	0.7516	0.33
Carp gudgeons	4.06	**<0.0001**	0.88	0.2619	0.55	0.2619	0.31
Un-specked hardyhead	8.91	**<0.0001**	2.26	0.1162	0.63	0.1635	0.39
Common carp	0.63	**0.0002**	0.21	0.3545	0.20	0.1491	0.12
Golden perch	1.67	**<0.0001**	0.01	0.6365	0.04	0.7943	0.08
Assemblage	5,851.4	**0.001**	1,399.7	0.059	593.9	0.052	401.54
Degrees of freedom	5		1		5		144

* heterogeneous variance after transformation

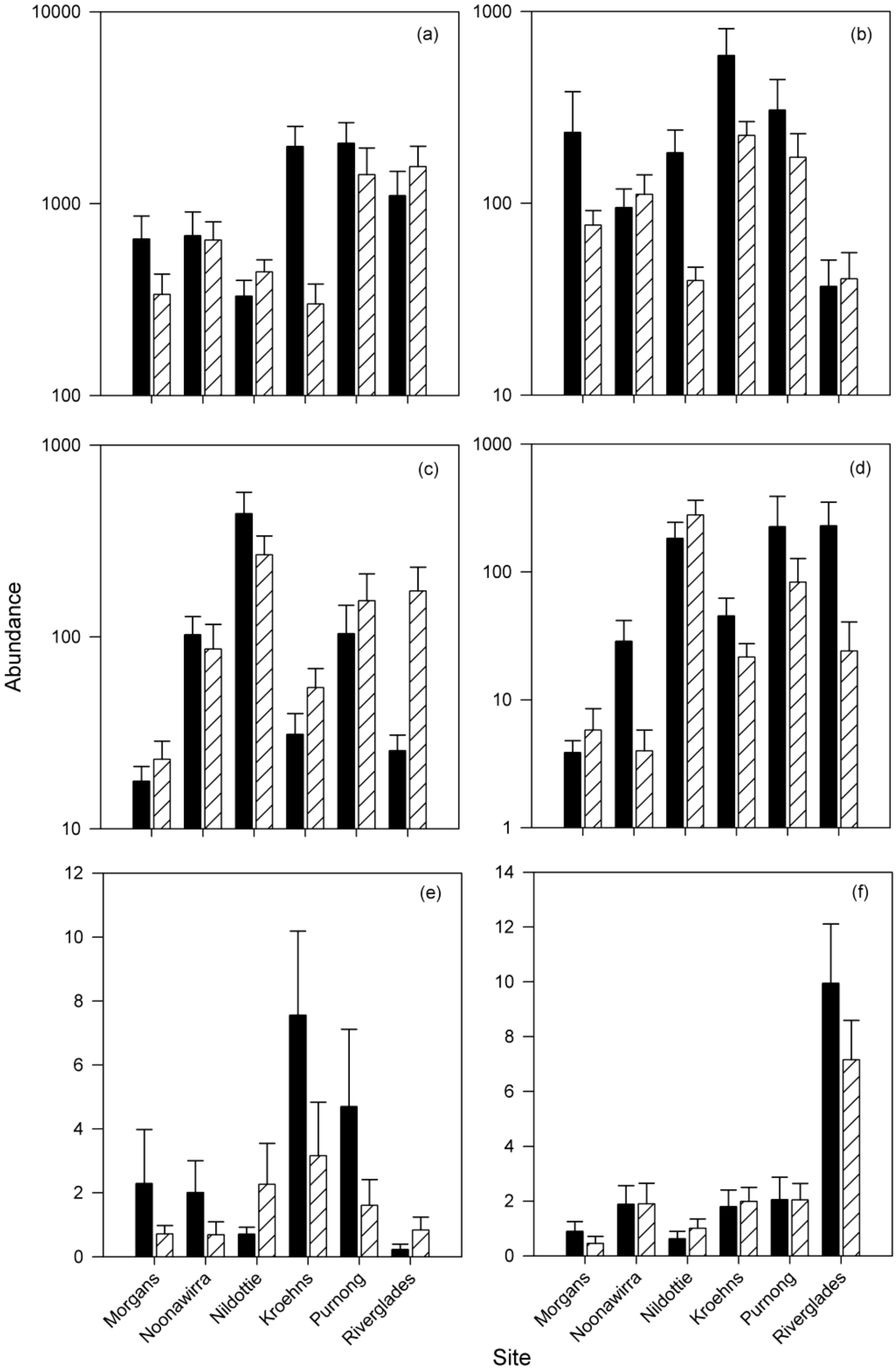

Figure 2.4. Mean (+1 SE) abundance of the most commonly sampled species: (a) $\log_{10}(x + 1)$ Australian smelt, (b) $\log_{10}(x + 1)$ bony herring, (c) $\log_{10}(x + 1)$ carp gudgeons, (d) $\log_{10}(x + 1)$ un-specked hardyhead, (e) common carp and (f) golden perch moving out of (black) and into (striped columns) wetland inlets sampled from August to November 2006. All abundance values represent fish per net per 24 hours set. Note varying *y*-axis scale.

Several species influenced the separation of wetland samples in the NMS (Figure 2.5a). For example, the separation of wetlands along Axis 3 was influenced by common galaxias (correlation between abundance and axis score: $r = -0.45$), golden perch ($r=-0.45$) and flat-headed gudgeons *Philypnodon grandiceps* ($r=-0.49$), all in greater abundances at Riverglades than other wetlands. Carp gudgeons ($r = 0.40$) and Murray–Darling rainbowfish *Melanotaenia fluviatilis* ($r=0.38$) were most abundant at Nildottie. Eastern gambusia *Gambusia holbrooki* abundance was correlated ($r = 0.39$) with Axis 1, given highest abundances at Morgans and Noonawirra compared to other wetlands. In contrast, Australian smelt abundances were correlated with Axis 1 ($r = -0.36$) and Axis 3 ($r=-0.29$), with highest abundances at Riverglades, North Purnong and Kroehns, compared to other wetlands.

2.3.2. Relationships with environmental variables

Few environmental variables were correlated (Spearman rank) with biological variables. Mean weekly total abundance ($r_s = -0.943$, $p = 0.005$) and the mean weekly abundance of common galaxias ($r_s = -0.941$, $p = 0.005$) and golden perch ($r_s = -0.943$, $p = 0.005$) were negatively related to distance from the river mouth. Golden perch were positively related to river EC ($r_s = 0.943$, $p = 0.005$) and inlet depth ($r_s = 0.941$, $p = 0.005$), which increased towards the river mouth.

Several environmental variables influenced the separation of samples in ordination space (Figure 2.5b). Inlet access ($r = -0.82$), shoreline development index ($r=-0.74$), submerged vegetation ($r=-0.58$) and wetland EC ($r=-0.46$) separated sites along Axis 1, including a positive relationship with Australian smelt and un-specked hardyhead and a negative relationship with eastern gambusia (Figure 2.5a, b). Distance from the river mouth ($r = 0.76$), woody debris ($r=0.48$), river EC ($r=-0.62$) and wetland area ($r=-0.32$) influenced the separation of wetlands on Axis 3. Distances from the river mouth and woody debris were positively correlated to the abundances of carp gudgeons, Murray–Darling rainbowfish and bony herring, and negatively with common galaxias, golden perch and flat-headed gudgeons. Conversely, river EC and inlet depth were negatively correlated with carp gudgeons, Murray–Darling rainbowfish and bony herring, and positively correlated with common galaxias, golden perch and flat-headed gudgeons.

2.3.3. Directional differences in size classes

A broad range of size classes of large- and small-bodied native and alien species were recorded (Figures 2.6 and 2.7). Common carp ranged from 20–770 mm TL with a bimodal distribution dominated by adults (Figure 2.6). In contrast, bony herring (22–457 mm) and golden perch (81–478 mm) showed uni-modal size distributions and were dominated by juveniles (Figure 2.6).

Small-bodied fish including Australian smelt (32–82 mm), un-specked hardyhead (25–68 mm) and carp gudgeons (24–71 mm TL) had uni-modal distributions dominated by adults (Figure 2.7).

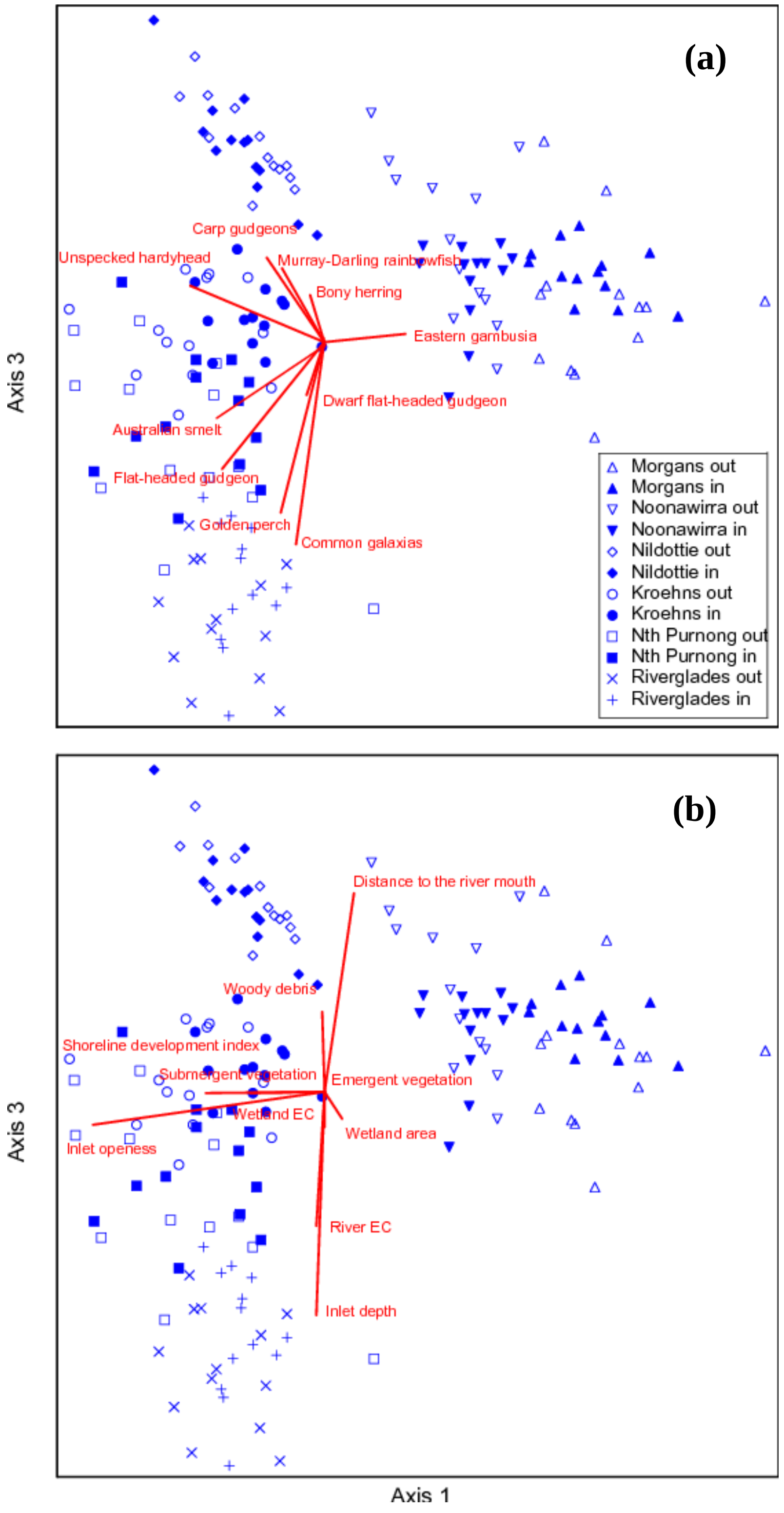

Figure 2.5. Ordination of assemblages of fish moving into and out of the six wetlands, with correlations for (a) species abundance and (b) environmental variables overlaid. Stress = 0.12.

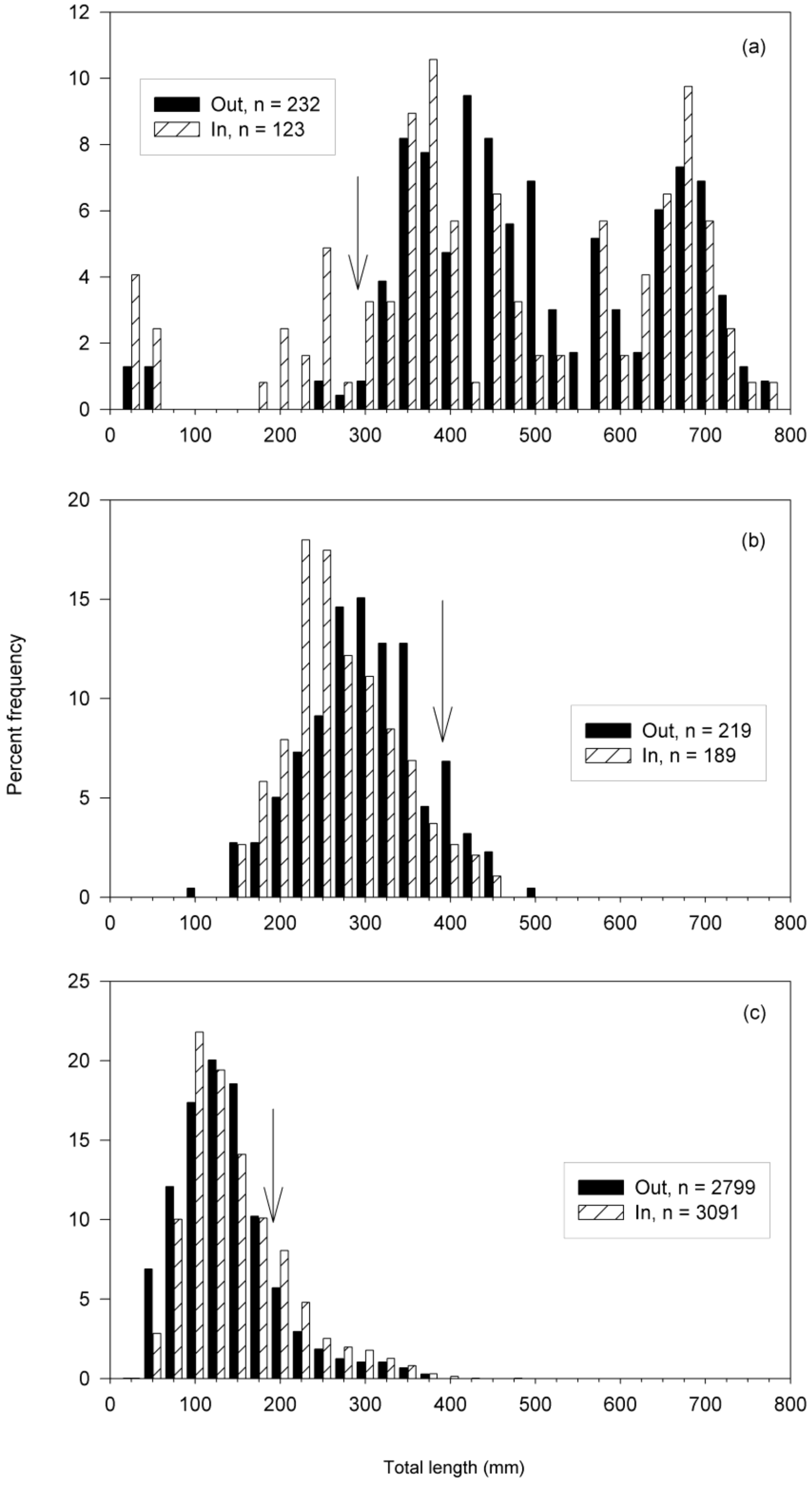

Figure 2.6. Size distributions of the three dominant large-bodied species, (a) common carp, (b) golden perch and (c) bony herring, moving into and out of wetlands from August–November 2006. Arrows indicate the approximate minimum length at maturity for females. Note varying *y*-axis scale.

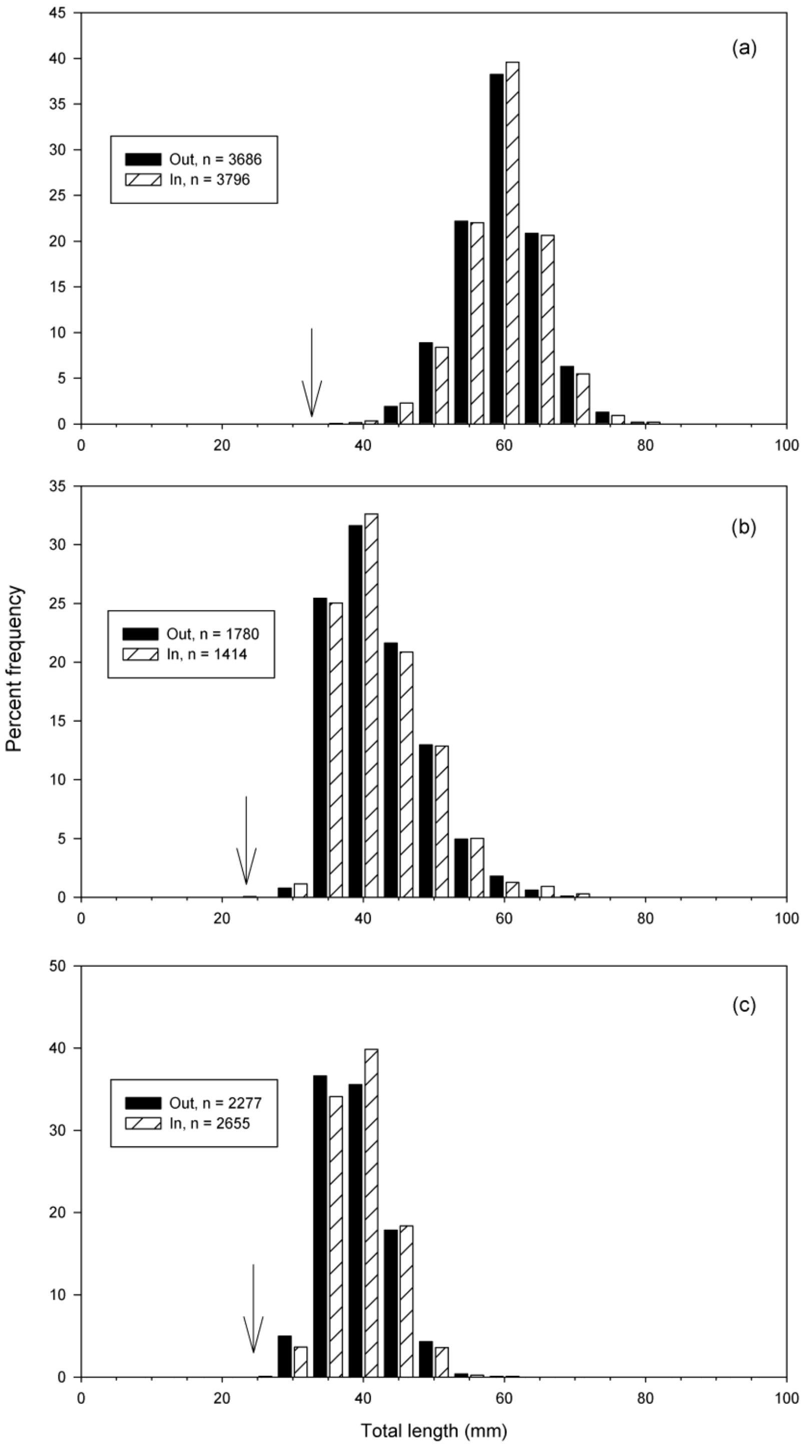

Figure 2.7. Size distributions of the three dominant small-bodied species, (a) Australian smelt, (b) un-specked hardyhead and (c) carp gudgeons, moving into and out of wetlands from August–November 2006. Arrows indicate the approximate minimum length at maturity for females. Note varying *y*-axis scale.

Three of the six dominant species (common carp, golden perch, bony herring) showed differences in size distributions relative to the direction of movement (Table 2.4). Larger common carp and golden perch were recorded moving out of wetlands, and larger bony herring were recorded moving in. No differences in size distribution related to the direction of movement were detected for Australian smelt, un-specked hardyhead or carp gudgeons.

Table 2.4. A comparison of size classes (Total Length, TL mm) of the most abundant fish moving in or out of wetlands in the Lower Murray in August-November 2006. Shown are sample sizes, mean (± SD) total length, and Kolmogorov-Smirnov statistic (KS). Directional size differences are shown where present. Significant *p* values highlighted in bold.

	Out		In				
Species	*n*	Mean ± S.D. (mm)	*n*	Mean ± S.D. (mm)	KS	*P*	Direction significantly larger fish moving
Common carp	232	486 ± 150	123	441 ± 191	0.195	**0.005**	Out
Golden perch	219	289 ± 70	189	259 ± 64	0.228	**<0.001**	Out
Bony herring	2799	125 ± 58	3091	135 ± 63	0.074	**<0.001**	In
Australian smelt	3686	58 ± 6	3796	57 ± 6	0.013	0.921	-
Un-specked hardyhead	1780	40 ± 6	1414	40 ± 6	0.020	0.786	-
Carp gudgeons	2277	37 ± 5	2655	37 ± 4	0.041	0.055	-

2.4. DISCUSSION

2.4.1. Lateral movements

The importance of lateral connectivity for riverine fish is widely recognized (Junk *et al.* 1989; Welcomme 1995; Górski *et al.* 2011), but descriptions of lateral movements are scarce. Studies that have considered lateral movements are typically in systems where access to off-channel habitats is influenced by seasonal flows that inundate temporary habitats (e.g. Kwak 1988; Castello 2008; Lyon *et al.* 2010), providing a stimulus for movement. In contrast, this study describes lateral fish movements into and out of perennially inundated wetlands where river regulation exacerbated by drought has diminished flows and seasonal water level variations.

As anticipated, our data revealed a diverse (18 species) and abundant (some 210 000 individuals) fish assemblage moving between perennially inundated wetlands and the channel of the Lower Murray. Movements involved all fish species recorded in recent surveys of fish in wetlands (Smith *et al.* 2009b) and the river (Davies *et al.* 2008) in this region, as well as an additional three species (pouched lamprey, Murray cod, congolli) not recorded in these studies. Indeed 14 of 27 native species (with all *Hypseleotris* spp. treated as one species complex; *cf.* Bertozzi *et al*., 2000) known from the Murray–Darling Basin in South Australia (Hammer and Walker, 2004) were recorded moving laterally.

Lateral movements were dominated by five native species, including small-bodied (Australian smelt, carp gudgeons, un-specked hardyhead) and large-bodied species (bony herring, golden perch). These species are considered ‘ecological generalists’ (Balcombe and Arthington 2009; Smith *et al.* 2009b), with broad or flexible habitat, food and reproductive traits that allow them to survive and often dominate in rivers modified by regulation (Copp 1990; Galat and Zweimuller 2001; Aarts and Nienhuis 2003). For example, each of these species, excluding golden perch, has a reproductive cycle that is independent of the seasonal flood pulse of the Murray (Humphries *et al.* 1999), and therefore may not be disadvantaged by regulated conditions. Indeed, bony herring in the Lower Murray are thought to have increased in abundance as a consequence of regulation (Puckridge and Walker 1990).

Regulation of floodplain rivers is often attributed to the loss of native species diversity due to modification of the flow regime and habitat connectivity (e.g. Schiemer and Spindler 1989; Aarts *et al.* 2004). Indeed river regulation is implicated in the regional extinction of five fish species (Lloyd and Walker, 1986; Walker and Thoms, 1993), and in the decline of nine other species now protected by state and federal legislation (Hammer and Walker, 2004; South Australian *Fisheries Management Act 2007*). In this study, two native species of conservation significance (freshwater catfish, Murray cod) were recorded, albeit at only one wetland inlet

(Kroehns) and in very low abundance. These species occur in both wetland (Jones and Stuart 2008a; Smith *et al.* 2009b) and riverine (Davies *et al.* 2008) habitats, but the importance of lateral connectivity for them is unclear and is hampered by a lack of historical data on their distribution prior to regulation (cf. Stuart and Jones 2006a). Hence, a precautionary approach should be followed prior to modification of wetland inlets where these species occur.

Four alien species (common carp, redfin perch *Perca fluviatilis*, goldfish, eastern gambusia) common throughout the MDB (Lintermans 2007) and regulated rivers worldwide (e.g. Copp *et al.* 2005) were recorded at all surveyed inlets. Common carp, a notorious pest in wetlands (Koehn 2004), was the most abundant alien species in this study and is the dominant large-bodied fish in the River Murray (Davies *et al.* 2008). Lateral movements by common carp have been recorded in the lower (Reynolds 1983) and middle reaches (Jones and Stuart 2008a) of the Murray, where wetland habitats provide favourable conditions for spawning and recruitment (Vilizzi and Walker 1999; Smith and Walker 2004; Stuart and Jones 2006a). However, the propensity of common carp to move to and from wetlands may make them vulnerable to trapping (Stuart *et al.* 2006a; Thwaites *et al.* 2010; Daniel *et al.* 2011) and exclusion screens (Chapter 4; Hillyard *et al.* 2010).

2.4.2. Variation among wetlands

Spatial variability is a common attribute of fish assemblages in freshwater habitats (see Jackson *et al.* 2001). In this study, there was substantial variability in species presence and abundance among wetland inlets, despite their proximity to each other and their permanent connection within the same river reach. For example, species richness and total abundance were higher towards the river mouth, owing to the presence of diadromous species such as common galaxias, congolli and pouched lamprey. This is consistent with other studies of diadromous species as they generally occur in highest abundance in the lower reaches of coastal rivers, rarely penetrating more than 50 km upstream (Leathwick *et al.* 2005; Smith *et al.* 2009b).

Spatial variability in lateral movements was also noteworthy for species common to all wetland inlets. For example, Nildottie inlet had more small-bodied species such as carp gudgeons, un-specked hardyhead and Murray–Darling rainbowfish relative to other wetlands. Differences in species abundance at Nildottie were related to several physical attributes with all three species positively correlated with large woody debris, and negatively with wetland size and inlet depth. This suggests that lateral movements of these species may be more prevalent between the river and small shallow wetlands. Conversely, these wetlands may also be less attractive for large-bodied predatory fish such as golden perch that were negatively correlated with the above characteristics. Indeed, golden perch are generally found in higher abundance in large-deep wetlands of the Lower Murray compared to small-shallow wetlands (Smith *et al.* 2009b). Although several environmental parameters influenced the fish community at the six wetland inlets surveyed, further investigation is required to determine the broader applicability of these findings to predict the use of wetland inlets by fish at a regional scale.

2.4.3. Patterns of movement

Lateral movements of most fish species were bidirectional, without clear pattern, which contrasts to other studies (Niger River, Mali: Bénech and Peñáz 1995; Morava River, Czech Republic: Hohausova *et al.* 2003; Murray River, Australia: Stuart and Jones 2006b; Amazon floodplain, Brazil: Castello 2008; Murray River, Australia: Jones and Stuart 2008a; Murray River, Australia: Lyon *et al.* 2010). Distinct one-way movements in these studies reflect seasonal changes in hydraulic connectivity where fishes benefit from moving into relatively warm, food rich and productive temporary habitats, and leave prior to disconnection and drying (cf. Flood-pulse concept: Junk *et al.* 1989). The lack of directionality evident in the present study may therefore be an artefact of stable water levels imposed by river regulation, which maintains a low level of productivity in both wetland and river habitats compared with less regulated systems (Walker 2006).

Fish of a range of sizes moved laterally, including juvenile and adult common carp, golden perch and bony herring. The general absence of juvenile small-bodied species is likely an artefact of the mesh size employed (8-mm stretched) (Smith *et al.* 2009b). Ontogenetic habitat shifts are commonplace in freshwater fish (Werner and Gilliam 1984), and are reported for some native Australian species (King 2004). Whilst three of the six dominant species in this study (common carp, golden perch, bony herring) varied in size in relation to direction of movement, the differences were relatively small and not indicative of an ontogenetic habitat shift. However, we speculate that had follow-up sampling in autumn been undertaken, it is likely that juveniles of several species (e.g. common carp, bony herring) would have been observed dispersing from wetlands (see Stuart and Jones 2006a; Balcombe *et al.* 2007; Jones and Stuart 2008a). We anticipate that, when higher flows resume in the Lower Murray, ontogenetic patterns of lateral movements will be more apparent, as in the mid reaches of the river (cf. Lyon *et al.*, 2010). In that regard, the present study provides an invaluable comparative database for future research.

2.4.4. Importance of lateral movements

Lateral movements of fish are undertaken to fulfil ecological functions related to reproduction, refuge or feeding (Lucas and Baras 2001), but whether these functions drive movements in perennially inundated systems is unclear. We were unable to discount any of the above drivers of movement due to the absence of clear directional or ontogenetic movement patterns. However, we postulate that lateral movements in the Lower Murray during low flows and stable river levels represent haphazard searching for suitable spawning (ripe individuals of several species were observed), feeding or refuge locations and that bidirectional movements facilitate efficient resource utilization (Lucas and Baras 2001) in a system characterized by homogenized river and wetland habitats with sustained low levels of productivity (Walker 2006). This notion is supported indirectly by studies of lateral movements in systems where connectivity is controlled by seasonal river-level variability (e.g. Bénech and Peňáz 1995; Hohausova *et al.* 2003; Castello 2008), and directly by Lyon *et al.* (2010), who describes similar two-way movements when river and wetland water levels stabilize in the middle reaches of the Murray.

The high abundance of fish moving (some 210 000) in this study may provide an important ecosystem function by facilitating the exchange of biological resources (nutrients) between the river and floodplain in the absence of floods. The 'flood pulse' is often cited as the main driver of biological exchange between the river and floodplain environments in large river systems and is responsible for the bulk of productivity generated in these systems (Junk *et al.* 1989; Walker *et al.* 1995; Burford *et al.* 2008). Several studies have highlighted the significance of fish movements in facilitating the exchange of organic matter between rivers and floodplains (Winemiller and Jepsen 1998; Roach *et al.* 2009), embayments and lakes (Brazner *et al.* 2001), and estuaries and oceans (Deegan 1993), but no studies have addressed the role of fish-driven exchange in perennially inundated river systems under low flows. Extrapolation of the number of fish moving suggests that in the absence of flow-driven exchange of organic matter, fish movement may play an important role in nutrient transport within this system. In light of predicted modifications to lateral connectivity in the Lower Murray, to maintain water for 'critical human needs'(Smith *et al.* 2009b), the importance of fish-mediated biological exchange to the ecology of the Lower Murray requires further investigation.

2.4.5. Conclusion

An abundant and diverse fish community was recorded moving between the river and wetlands during this study. Whilst the movements were predominantly of 'generalist' native and alien species, two species of conservation significance were represented. Movements of juveniles and adults varied among wetlands despite the shared river reach and the proximity of the wetlands to each other, but showed no consistent directionality. The lack of directionality likely reflects the effects of reduced flow and river level variability caused by river regulation, exacerbated by drought. We speculate that directional movements will become apparent when flows are increased, so that our data could provide a comparative baseline for future studies.

Variability in the presence and abundance of some species could be explained partly by the environmental attributes of wetlands, their inlets and the adjacent river, but the high variability among wetlands demonstrates the need for individual assessment of wetlands prior to management interventions. Modifications to wetland inlets, including the installation of flow control structures and ‘water saving’ initiatives (disconnection of wetlands *via* earthen levee banks) will act to compound the effects of regulation by diminishing river–floodplain connectivity, and will also act as physical barriers to the lateral migration of aquatic fauna. Thus, any future management interventions likely to affect lateral connectivity between rivers and wetlands should accommodate the passage requirements of the resident aquatic fauna.

3. Chapter Three - CONNECTIONS WITHOUT DIRECTIONS: LATERAL MOVEMENTS OF FISH IN A DROUGHT-AFFECTED REGULATED RIVER

KARL A. HILLYARD[a], ANTHONY J. CONALLIN[a,b] , KEITH F. WALKER[a], BRONWYN M. GILLANDERS[a] AND BENJAMIN B. SMITH[b]

[a] *Southern Seas Ecology Laboratories, School of Earth and Environmental Sciences, Darling Building, DX 650 418, University of Adelaide, SA 5005, Australia.*

[b] *Inland Waters and Catchment Ecology Program, SARDI Aquatic Sciences, PO Box 120, Henley Beach, SA 5022, Australia.*

Text in manuscript

Statement of Authorship

In this chapter, Karl Hillyard developed the study, collected and analysed the data, and drafted the manuscript. Anthony Conallin, Ben Smith, Bronwyn Gillanders and Keith Walker advised on development of the study, data analysis and contributed to the synthesis and finalisation of the manuscript. Anthony Conallin and Ben Smith also assisted with collection of data in the field.

Certification that the statement of contribution is accurate.

Karl A. Hillyard (Candidate)

Signed *Date* 11/5/2011

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

Anthony J. Conallin

Signed *Date* 11/5/2011

Bronwyn M. Gillanders

Signed *Date* 11/5/2011

Keith F. Walker

Signed *Date* 11/5/2011

Benjamin B. Smith

Signed *Date* 11/5/2011

3.1. ABSTRACT

Many riverine fish move freely between instream and offstream habitats, in response to seasonal variations in flow and water level. For several years these cues have been suppressed in the lower River Murray, Australia, by the effects of prolonged drought and upstream diversions, and so we anticipated that fish movements would be haphazard, asynchronous and bi-directional. We placed bi-directional fyke nets (24 h, once weekly) in the inlets to six perennially-inundated wetlands adjoining the river, and monitored movements over 13 weeks in spring (mid-August to mid-November) 2006. Samples of some 210 000 fish from 18 species (14 native, four alien) indicated that movements were haphazard, as predicted. The balance of fish movements, for most species, was from, rather than to, wetlands, possibly in response to falling wetland water levels. The abundance of several small-bodied species increased, apparently in response to increasing water temperature and day length. These results demonstrate that any flow control regulators installed to facilitate wetland management must include provision for fish passage, as bi-directional movements will occur whenever water levels allow.

3.2. INTRODUCTION

Riverine fish move within their environment to gain access to habitats for refuge, feeding and spawning (e.g. Dodson 1997). The stimuli may be intrinsic, including genetic, ontogenetic or behavioural factors, or extrinsic, like day length, temperature and water discharge (Lucas and Baras 2001). Depending on the species and conditions, movements may extend from metres to thousands of kilometres, and from diel through seasonal, annual and multi-annual cycles (McKeown 1984). They may include upstream-downstream and instream-offstream transfers, or combinations of these (Jungwirth *et al.* 2000). Although lateral movements generally occur at smaller spatial and temporal scales (e.g. Hohausova 2000), their significance is less well understood (Humphries *et al.* 1999).

Connectivity between habitats clearly is a prerequisite for movements. Longitudinal movements are often blocked or constrained by physical barriers, including dams, weirs and other regulating structures (e.g. Marmulla and Welcomme 2002), and by flow diversions (Larinier 2001). Lateral movements also are affected by instream structures, and by small structures (e.g. culverts, causeways, weirs) associated with tributaries, wetlands and distributary channels. Structures may not necessarily impede lateral movements, as riverine impoundments often connect, rather than disconnect, instream and offstream habitats (e.g. Nilsson and Berggren 2000).

Lateral movements are typical of fish in floodplain rivers (e.g. Lowe-McConnell 1987; Junk *et al.* 1989; Schlosser 1991; Welcomme 1995; Winemiller and Jepsen 1998). Cues for lateral fish movements have been investigated in the Amazon (Fernandes 1997; Castello 2008), upper River Murray (Lyon *et al.* 2010), Volga (Górski *et al.* 2010; Górski *et al.* 2011) and Niger rivers (Bénech and Peñáz 1995). In these studies, lateral fish movements, are typically directionally skewed by floods or managed flows, suggesting that lateral movements may be directionally balanced in the absence of flow-related cues. This hypothesis conforms to a movement model proposed by Lyon *et al.* (2010) which predicts approximately equal movements between instream and offstream habitats when water levels maintain a stable connection between habitats.

An opportunity to test this hypothesis is provided by the lower River Murray (hereafter ‘Lower Murray’), in the Murray-Darling Basin in south-eastern Australia, where flow regulation maintains permanent connections to many offstream wetlands, but there is little variation in river stage owing to the presence of a series of weirs (Walker 2006). In recent years the seasonal and inter-annual stability of river levels has been emphasized by the effects of prolonged drought (Bond *et al.* 2008; Murphy and Timbal 2008; Smith *et al.* 2009b).

The fish fauna of the Lower Murray includes 35 native and seven alien freshwater species spanning Total Lengths (TL) of *c.* 40-1000 mm (Hammer and Walker 2004), plus diadromous species whose local range and abundance depend on connectivity between the Lower Lakes (Alexandrina, Albert) and Coorong, near the river mouth, and the Southern Ocean. Observations of fish movements in the Lower Murray have focussed on large-bodied (adults >250 mm TL), commercially and recreationally important species such as golden perch *Macquaria ambigua*, whose longitudinal movements are cued by changes in flow (Reynolds 1983). Temporal patterns of lateral movements have not been described in the Lower Murray, but in more northerly regions of the Murray-Darling Basin they are reported to be cued by changes in flow and/or water levels (Jones and Stuart 2008a; Lyon *et al.* 2010). Spatial differences in fish lateral movements in the Lower Murray, explained by the proximity of each wetland to the river mouth and variation in wetland and inlet morphology, have been reported by Conallin *et al.* (2010; Chapter 2).

This study explores the temporal nature and extent of lateral fish movements between the Murray main channel and six perennial offstream wetlands during a period of prolonged drought (Bond *et al.* 2008; Murphy and Timbal 2008; Smith *et al.* 2009b) which, coupled with extraction for irrigation (Craik and Cleaver 2008), resulted in a period of low river flow and water level variation. It is an extension of analyses of spatial patterns in offstream fish movements in the same study reach where fish movements were found to differ among wetlands despite their proximity to each other (Chapter 2; Conallin *et al.* 2010). The study occurred over 13 weeks in spring 2006, and was designed to test the hypothesis that hydraulic connections, without hydraulic cues, lead to temporally asynchronous and haphazard fish movements. This paper (i) documents the nature, abundance and size structure (using length as a surrogate for age) of populations moving through the wetland inlets; (ii) records temporal changes in patterns of movement to and from the wetlands, and (iii) seeks relationships between fish movements and environmental variables.

3.3. METHODS

3.3.1. Study area

The Lower River Murray is part of Australia's largest river system, the Murray-Darling Basin (MDB, 1 073 000 km^2). Rivers in the MDB are characterised by low and variable flows, driven by a predominantly semi-arid climate (Walker and Thoms 1993). The Lower Murray is highly regulated by 10 low-level weirs (< 3 m high), five river-mouth barrages, floodplain levees and numerous smaller regulators (Walker and Thoms 1993; Maheshwari *et al.* 1995; Walker 2006). Although the 830-km Lower Murray (below the Murray-Darling junction) receives no major tributaries and flows for 350 km in a limestone gorge, there are extensive wetlands along most reaches. River regulation has deprived some wetlands of water, but the majority (70%) have been permanently connected as a consequence of raised, stabilized river levels (Pressey 1986), altering the habitat structure and function of the floodplain (Walker and Thoms 1993). Since 2001 the MDB has experienced a severe drought (Bond *et al.* 2008; Murphy and Timbal 2008; Smith *et al.* 2009b) which, coupled with extraction for irrigation (Craik and Cleaver 2008), resulted in a period of low, stable, within-channel flows in the Lower Murray (see 'Hydrology' below).

The six study wetlands were in a 140-km section of the Lower Murray between Lake Alexandrina and Lock and Weir 1 (hereafter 'Lock 1') at Blanchetown. They were chosen to represent a range of different wetland and inlet characteristics (Table 3.1). The inlets were 117-260 km upstream of the Murray mouth and the wetlands were *c.* 2.5-85 ha in area. At the time of sampling, all were permanently connected to the River Murray *via* single inlets of *c.* 20-750 m length and *c.* 5-35 m width where nets were set. Three wetlands had flow-control regulators (two with box culverts, one with concrete pipes) and removable screens ('carp exclusion screens') to exclude common carp *Cyprinus carpio* (Chapter 4; Hillyard *et al.* 2010).

3.3.2. Fish sampling

Sampling was over 13 weeks from mid-August to mid-November 2006. Two custom-made, double-winged, 8-mm stretched mesh fyke nets were used in each wetland. The nets were set once weekly for approximately 24 h, and placed back-to-back inside the inlet channel within 50 m of its confluence with the Murray, to trap fish entering and leaving the wetland. All nets covered

Table 3.1. Location and characteristics of the six wetlands sampled.

Wetland Name	Latitude	Longitude	Surface area (ha)	Flow regulating Structure	Distance to river mouth (km)	Inlet length (m)	Inlet width at net location (m)
Morgans	34 28.526’S	139 36.095’E	23	5 box culverts	260	184	8
Noonawirra	34 29.926’S	139 34.509’E	6	5 pipe culverts	255	60	35
Nildottie	34 40.356’S	139 39.017’E	3	None	225	18	6
Kroehns	34 42.747’S	139 34.467’E	46	None	215	758	7
Purnong	34 50.501’S	139 35.784’E	85	None	194	272	8
Riverglades	35 05.417’S	139 18.319’E	21	6 box culverts	117	165	15

the entire inlet channel and were floated and weighted to prevent fish from by-passing them. Carp exclusion screens (Chapter 4; Hillyard *et al.* 2010) (if present) were removed prior to sampling.

The catch from each net was transferred to 60-L tubs of aerated river water. Fish were identified to species, except for carp gudgeons (*Hypseleotris* spp.: see Bertozzi *et al.* 2000). Carp X goldfish *Carassius auratus* hybrids were pooled with carp. Up to 50 individuals of all large-bodied species (>100 mm TL, at maturity) were randomly sub-sampled and measured (TL, mm) and the remainder counted. Due to high abundances, up to 50 individuals of each small-bodied species were randomly sub-sampled, measured (TL, mm) and bulk-weighed, and an average individual weight was calculated. The remainder of each small-bodied species was then bulk-weighed, and total counts for abundance were estimated by dividing the bulk species weight by the average individual weight for each species.

3.3.3. Hydrology

Pool level in the study reach at the start of the study in mid-August was near bankfull (0.81 m ± 0.08 SD AHD (Australian Height Datum (AHD), standardised national height datum); Department of Water, Land and Biodiversity Conservation, unpublished data), but the river level fell over the 13-week study period to 0.64 m AHD (Figs 3.1-3.2). Flows to the study reach were 18% and 46% of the 1949-2006 mean and median flows, respectively (Figs 3.1-3.2). Wind-driven seiches affected river levels and inlet depths (±30 cm: see also Webster *et al.* 1997) and caused bi-directional flows within the wetland inlets. Whilst this study initially was planned to include a seasonal comparison investigating fish movements in spring and autumn 2006-2007, river levels were lowered in the intervening summer and the study wetlands were disconnected and dried. Accordingly, data for spring only are reported here.

3.3.4. Environmental and flow data

Weather data (wind speed and direction, Mean Sea Level air Pressure (MSLP), at the nearby Murray Bridge airport: 35.065°S, 139.226°E) were supplied by the Bureau of Meteorology. Lunar phase data for the region were sourced from Geoscience Australia. Water temperature was recorded using temperature loggers (Onset HOBO UA-001-08, Bourne, Massachusetts) placed in the wetlands, wetland inlets and adjacent river, and an average value for each of three sub-locations was used for analyses. Electrical conductivity was measured weekly in the wetlands, inlets and the adjacent river using a TPS 90 FL-T Water Quality Meter (TPS, Springwood, Queensland, Australia). Hydrological data were obtained from SA Water gauging stations at Blanchetown (34.351°S, 139.616°E), Swan Reach (34.590°S, 139.599°E) and Murray Bridge (35.123°S, 139.289°E). Data for river flow (at Blanchetown) and height (at Murray Bridge and Swan Reach) were sourced from the South Australian Department of Water, Land and

Biodiversity Conservation and compared to data from flow and depth loggers (Starflow loggers, cf. Webster *et al.* 1997) in three of the six wetland inlets; a close relationship between river and inlet stage enabled descriptions of flow in all six inlets. Stage data were also used to describe relative inlet depth, variability in depth (coefficient of variation, CV), the time that the water level was rising or falling and the relative flow magnitudes through each inlet during each net-set period.

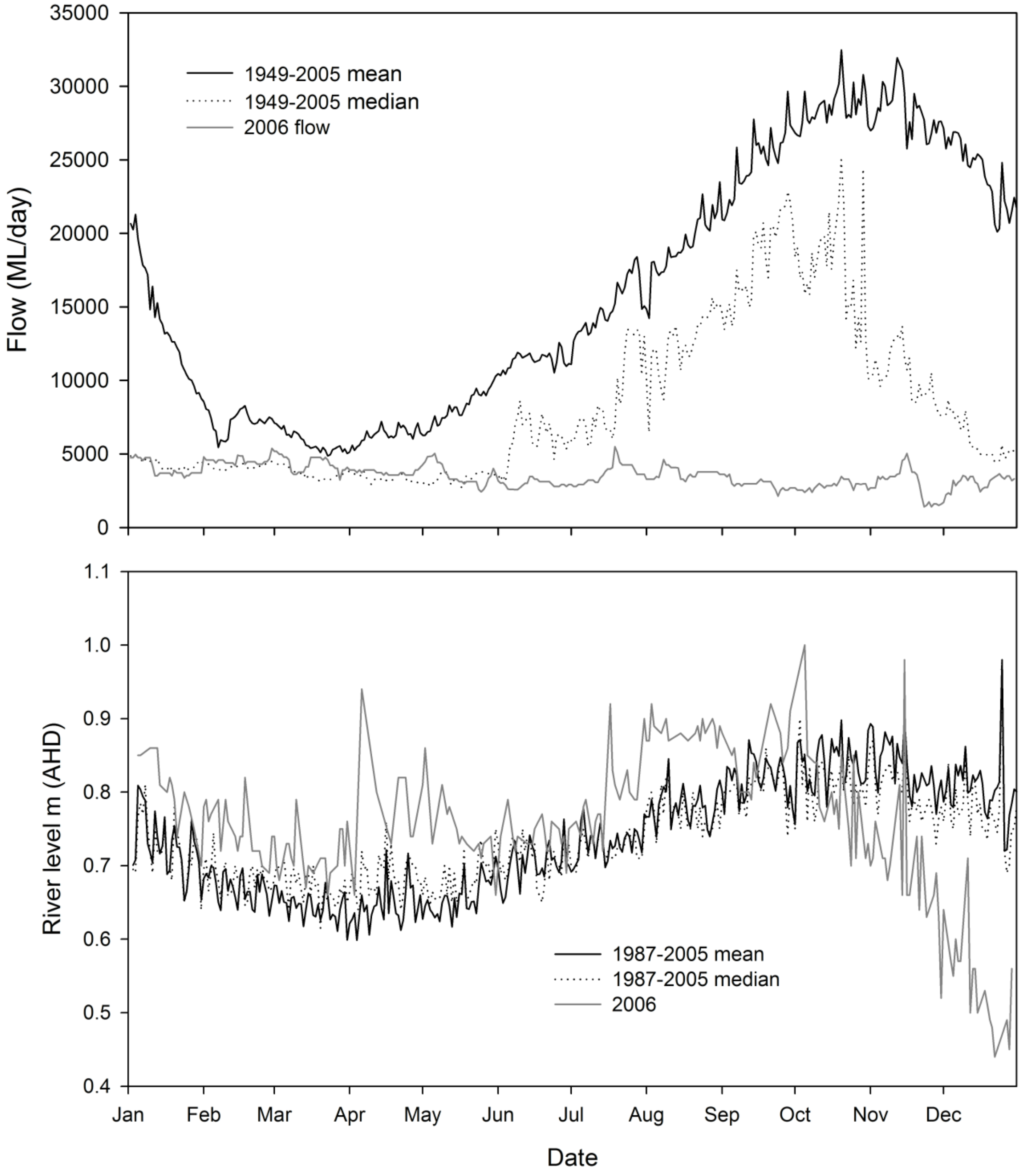

Figure 3.1. Comparison between long-term and 2006 flow in the River Murray at Blanchetown and river level (stage) at Murray Bridge, South Australia.

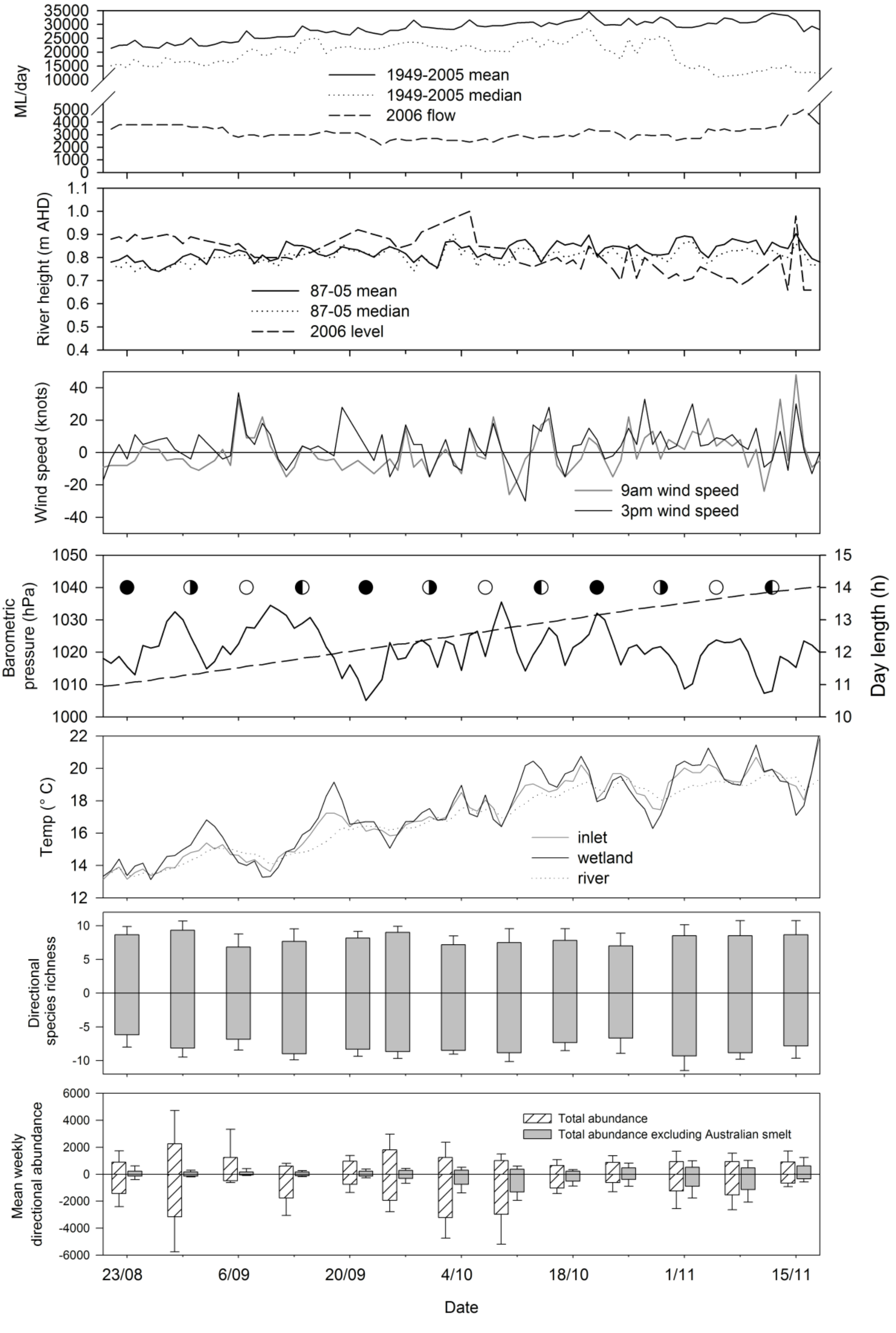

Figure 3. 2. Sampling period, environmental data, directional species richness and mean weekly directional abundance (with and without Australian smelt) data. Negative wind speed indicates northerly winds, positive wind speeds indicate southerly winds. Positive values for directional species richness and mean weekly directional abundance show fish moving from river to wetland, negative values show fish moving from wetland to river.

3.3.5. Data analysis

All catch data were standardised to a 24 h Catch Per Unit Effort (CPUE). Wetland inlets were treated as replicate sites to assess temporal changes in the fish assemblage over the 13 weeks. Two-way ANOVAs were used to compare total species richness, total abundance, total abundance with Australian smelt *Retropinna semoni*, a schooling species, removed (due to its numerical dominance), and individual abundances and total lengths of the nine most common species among weeks and between the direction of travel (into or out of the wetlands). Abundance data were $\log_{10}(x+1)$ transformed and checked for normality (graphical inspection) and heterogeneity of variances (Levene's test), although analyses were still performed if variances remained heterogeneous, as ANOVA is a robust method where (as in this case) data are balanced and samples are relatively large (Underwood 1997). The significance level (α) was reduced from 0.05 to 0.01 to lessen the chance of Type I error.

To compare the composition and directionality of assemblages moving to and from the wetlands, two-way Permutational Multivariate ANalysis Of VAriance (Per-MANOVA) was applied, using PRIMER (version 6: Plymouth Marine Laboratories, UK). The data were fourth-root transformed to reduce the influence of abundant species (Clarke and Warwick 2001) and converted to a Bray-Curtis distance matrix. The test statistic (essentially an *F* ratio) was calculated as the ratio of the sum of squared distances among groups to the sum of squared distances within groups. A Non-metric Multidimensional Scaling (NMS) ordination using the Sorensen (Bray-Curtis) metric was performed in PC-ORD (version 4.36: MjM Software, Oregon, USA). The fish assemblage (primary) data were then correlated with ordination scores and presented as 'joint plots'. The relative strength and direction of species correlations in ordination space are indicated by length and angle of the overlaid line on the joint plot (McCune and Grace 2002). In addition, the primary NMS matrix was overlaid with a secondary matrix of environmental variables to reveal correlations with ordination scores.

Spearman's rank correlation coefficients were calculated between biological variables (i.e. total species richness, mean species richness, mean abundance and abundance of dominant species) and temporally variable environmental factors (climatic, hydraulic and astronomical variables). Due to the large number of comparisons, the significance level (α) was reduced from 0.05 to 0.01 to lessen the chance of Type I error.

3.4. RESULTS

3.4.1. Assemblage

3.4.1.1 Species diversity and richness

Eighteen species, including 14 native and four alien species, were recorded in the inlets of the six off-channel wetlands over the 13 weeks of sampling (Table 3.2). Species richness, pooled among wetlands, ranged from 12 (week 9) to 15 (week 11). Mean species richness differed significantly among weeks, but there was no difference in direction or an interaction between 'Week' and 'Direction' (Table 3.3, Fig. 3.2).

Twelve of the 18 species were recorded every week over the study period (Table 3.2). Common galaxias *Galaxias maculatus* was recorded in 12 of the 13 weeks (not in week 9). The remaining five species (pouched lamprey *Geotria australis*, small-mouthed hardyhead *Atherinosoma microstoma*, congolli *Pseudaphritis urvillii*, Murray cod *Maccullochella peelii*, freshwater catfish *Tandanus tandanus*) were recorded in only one or two of the sampling weeks (Table 3.2).

3.4.1.2. Abundance

Some 210 000 fish were caught moving between the river and off-channel wetlands (Table 3.2). Most (87%) were small-bodied species, dominated by Australian smelt (70% of all fish caught), carp gudgeons (8.5%) and un-specked hardyheads *Craterocephalus stercusmuscarum fulvus* (7%). The most abundant large-bodied species was bony herring *Nematalosa erebi* (12.5%) (Table 3.2). Weekly abundance, pooled among wetlands, ranged from some 32 000 fish in week two to 8900 fish in week 10. With Australian smelt excluded, abundance ranged from 1300 fish in week three to 10 000 fish in week eight (Table 3.2). Mean abundance varied among weeks and direction (Table 3.3, Fig. 3.2), with slightly more (*c.* 60% total CPUE abundance) fish moving from wetland to the river over the 13 weeks. Mean total abundance peaked during week two and weeks five to seven. With Australian smelt removed from the analysis, similar variation among weeks, but not direction, was apparent (Table 3.3, Fig. 3.2). Mean weekly abundance increased over the sampling period when Australian smelt were removed (Fig. 3.2).

Table 3.2. Fish species captured moving between off-channel wetlands and the main river channel showing weekly % Catch Per Unit Effort (CPUE) and total catch. * indicates alien species.

Common name	Taxon	Family	actual catch	Week 1	Week 2	Week 3	Week 4	Week 5	Week 6	Week 7	Week 8	Week 9	Week 10	Week 11	Week 12	Week 13
Australian smelt	*Retropinna semoni* (Weber, 1895)	Retropinnidae	150 220	84.23	94.32	86.86	88.16	78.56	84.09	76.48	57.85	55.77	41.42	35.72	34.85	39.60
Bony herring	*Nematalosa erebi* (Günther, 1868)	Clupeidae	26 531	1.82	3.43	4.37	8.47	10.92	6.46	9.71	28.17	20.29	16.08	12.55	31.04	7.23
Carp gudgeons	*Hypseleotris* spp.	Eleotridae	18 172	4.53	0.93	2.71	1.04	4.72	5.60	8.24	9.39	14.70	19.13	20.99	14.55	28.32
Un-specked hardyhead	*Craterocephalus stercusmuscarum fulvus* Ivanstoff, Crowley & Allen,1987	Atherinidae	14 746	8.08	0.72	4.65	1.06	3.13	2.07	4.05	1.80	6.08	19.24	27.57	16.13	19.89
Flat-headed gudgeon	*Philypnodon grandiceps* (Krefft, 1864)	Eleotridae	2016	0.24	0.21	0.37	0.41	0.79	1.06	0.64	1.61	1.10	1.76	1.21	1.89	2.29
Murray-Darling rainbowfish	*Melanotaenia fluviatilis* (Castelnau, 1878)	Melanotaeniidae	605	0.09	0.05	0.12	0.10	0.05	0.05	0.46	0.31	0.92	0.67	0.78	0.29	0.60
Dwarf flat-headed gudgeon	*Philypnodon macrostomus* Hoese & Reader, 2006	Eleotridae	511	0.09	0.03	0.13	0.12	0.18	0.20	0.14	0.51	0.37	0.11	0.45	0.56	0.49
Golden perch	*Macquaria ambigua ambigua* (Richardson, 1845)	Percichthyidae	410	0.42	0.10	0.26	0.15	0.51	0.13	0.07	0.15	0.27	0.15	0.18	0.24	0.41
Common carp *	*Cyprinus carpio* Linnaeus, 1758	Cyprinidae	359	0.02	0.05	0.05	0.32	0.76	0.14	0.05	0.14	0.36	0.20	0.29	0.11	0.11
Redfin perch *	*Perca fluviatilis* Linnaeus, 1758	Percidae	223	0.12	0.02	0.14	0.06	0.06	0.02	<0.01	0.01	0.02	0.15	0.16	0.27	0.86
Goldfish *	*Carassius auratus* Linnaeus, 1758	Cyprinidae	222	0.18	0.04	0.06	0.03	0.14	0.10	0.03	0.03	0.07	1.03	<0.01	0.02	0.07
Common galaxias	*Galaxias maculatus* (Jenyns, 1842)	Galaciidae	119	0.11	0.02	0.25	0.02	0.14	0.01	0.09	0.02	0.00	0.04	0.05	0.02	0.08
Eastern gambusia *	*Gambusia holbrooki* (Girard, 1859)	Peocilidae	91	0.06	0.07	0.02	0.05	0.04	0.06	0.05	0.01	0.05	0.02	0.02	0.01	0.04
Pouched lamprey	*Geotria australis* Grey, 1851	Petromyzontidae	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01	<0.01	0.00
Small-mouthed hardyhead	*Atherinosoma microstoma* (Günther, 1861)	Atherinidae	1	<0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Congolli	*Pseudaphritis urvillii* (Valenciennes, 1832)	Pseudaphritidae	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01	0.00	0.00
Murray cod	*Maccullochella peelii* (Mitchell, 1838)	Percichthyidae	1	0.00	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Freshwater catfish	*Tandanus tandanus* Mitchell, 1838	Plotosidae	1	0.00	<0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total CPUE abundance				13 884	32 444	10 381	14 241	10 258	22 403	26 700	23 812	10 040	8866	13 136	14 846	9468
Total CPUE abundance (excluding *R. semoni*)				2189	1843	1364	1686	2199	3564	6280	10 037	4441	5194	8444	9672	5719
Species richness				14	14	13	14	13	13	13	13	12	13	15	14	13

Table 3.3. Mean square and significance level for two-way ANOVAs of species richness, total fish abundance (with and without Australian smelt), abundance of Australian smelt, carp gudgeons, un-specked hardyhead, flat-headed gudgeon, Murray-Darling rainbowfish, dwarf flat-headed gudgeon, bony herring, common carp, golden perch and Per-MANOVA of assemblage by week and direction. Significant *P* values are shown in bold type.

Source	Week		Direction		Week x Direction		Error
	MS	*P*	MS	*P*	MS	*P*	MS
Species richness *	6.035	**0.008**	0.026	0.920	3.734	0.137	2.510
Total abundance	0.507	**0.002**	0.706	**0.048**	0.198	0.356	0.178
Total abundance excluding Australian smelt *	1.250	**<0.001**	0.014	0.777	0.309	0.068	0.179
Australian smelt	1.000	**<0.001**	0.555	0.149	0.234	0.561	0.264
Carp gudgeons *	2.242	**<0.001**	1.092	0.059	0.175	0.852	0.300
Un-specked hardyhead *	1.500	**0.009**	2.076	0.073	0.282	0.943	0.636
Flat-headed gudgeon	0.276	0.663	0.033	0.760	0.142	0.959	0.350
Murray-Darling rainbowfish *	0.282	0.185	0.003	0.901	0.063	0.989	0.214
Dwarf flat-headed gudgeon *	0.372	**0.002**	1.315	**0.002**	0.136	0.424	0.132
Bony herring *	1.305	**<0.001**	0.001	0.962	0.409	0.284	0.338
Common carp	0.232	0.057	0.166	0.261	0.137	0.406	0.130
Golden perch *	0.120	0.587	0.004	0.862	0.126	0.529	0.137
Assemblage	1450.3	**0.001**	1353.4	0.017	494.08	0.634	530.96
Degrees of freedom	12		1		12		130

* Heterogeneous variance after transformation

3.4.2. Dominant species abundance and length

Temporal and directional differences in mean abundance were recorded for the nine dominant species (Fig. 3.3). Dwarf flat-headed gudgeons *Philypnodon macrostomus* showed significant differences among weeks and between direction, with more fish moving out (*c.* 70% total CPUE abundance) of the wetlands than in (Table 3.3). Conversely, the mean abundance of Australian smelt, carp gudgeons, un-specked hardyhead and bony herring varied significantly among weeks only, whereas no statistically significant temporal or directional patterns of movement were apparent for flat-headed gudgeon *Philypnodon grandiceps*, Murray-Darling rainbowfish *Melanotaenia fluviatilis*, common carp or golden perch.

Four trends in mean weekly abundance (decreasing, increasing, stable, variable) were evident among the nine dominant species over the study period (Fig. 3.3): Australian smelt decreased in abundance; carp gudgeons, un-specked hardyhead, flat-headed gudgeon and Murray-Darling rainbowfish generally increased in abundance; the catch of golden perch was stable, and the mean abundance of bony herring, common carp and dwarf flat-headed gudgeon was variable. Despite their variable abundance, peak catches of bony herring and dwarf flat-headed gudgeon occurred in weeks six to eight and weeks 11-12. The catch of common carp was low until it peaked in weeks four and five, after which it decreased for the remainder of the study (Fig. 3.3).

A broad size range of fish was collected moving between the river and off-channel sites. Directional differences in length were not observed between fish moving into or out of the wetlands (Fig. 3.4). The majority of the nine dominant species showed little difference in mean length over the 13 weeks. Bony herring showed a gradual (non-significant) trend of increased length over the 13 weeks. Of the nine dominant species, golden perch and common carp showed most variability in length over the 13 weeks.

3.4.3. Per-MANOVA and ordination

The fish species assemblage varied among weeks but showed no difference in direction (Table 3.3). The data yielded a 3-D ordination (stress 0.12) with Axes 2 (r = 0.36) and 3 (r = 0.19) selected for display based on the separation of samples across the 13 sampling weeks (Fig. 3.5). Axis 2 best showed the changes in assemblage over the 13 sampling weeks and was influenced by decreasing Australian smelt abundance (correlation between abundance and axis 2 score: r = 0.368) and increasing abundance of carp gudgeons (-0.568), un-specked hardyheads (-0.358), dwarf flat-headed gudgeons (-0.299) and Murray-Darling rainbowfish (-0.285). These trends in the ordination mirrored the individual mean abundance graphs (Fig. 3.3).

3.4.4. Relationship with environmental and flow variables

River flow at Blanchetown averaged 3136 ML d^{-1} (±530 SD) over the study and ranged from *c.* 2100 to 5000 ML d^{-1}. During the study the river level at Swan Reach averaged 0.81 m AHD (±0.08 SD) and ranged between 0.66 and 1.0 m AHD. Day length increased from approximately 11 to 14 h over the 13 weeks. Water temperatures (inlet, wetland, river) increased over the 13 weeks. River water temperature was most stable and averaged 16.87 °C (±1.98 SD), but ranged from 13.33 °C to 19.58 °C. Wetland water temperature was most variable (range 13.13 °C-21.45 °C), averaging 17.32 °C (±2.22 SD). The mean inlet water temperature (17.15 °C (±2.15 SD)) was similar to the wetland water temperature and ranged from 13.16 °C to 20.69 °C. The study period encompassed three full lunar cycles. Daily average air pressure (Mean Sea-Level Pressure (MSLP)) was highly variable ranging from 1005.10 to 1035.50 hPa, averaging 1021.22 hPa (±6.52hPa SD). Wind speed and direction were highly variable over the 13 weeks (Fig. 3.2).

The dominant environmental variables correlated with the fish assemblage data in ordination space were day length (r = -0.968) and water temperatures measured in the river (-0.956), inlet (-0.929) and wetland (-0.850) which separated weeks along Axis 2 (Fig. 3.5). These variables showed a positive relationship with carp gudgeons, un-specked hardyheads, dwarf flat-headed gudgeons and Murray-Darling rainbowfish. A negative relationship was apparent with Australian smelt.

Spearman rank correlations between fish abundance and environmental variables showed similar patterns to those seen in ordination space (Table 3.4). Climatic variables (principally river, wetland and inlet temperatures) were commonly correlated with fish abundance, followed by hydraulic (mean inlet depth, river level and river flow) and astronomical (day length and % lunar illumination, a proxy for lunar phase) variables. Water temperature (river, wetland and inlet) was correlated with several species. For example, increasing water temperatures (river and inlet) were positively correlated with carp gudgeon abundance and total abundance (excluding Australian smelt), and negatively correlated with Australian smelt abundance. Day length was strongly and positively correlated with the abundance of carp gudgeons, flat-headed gudgeons, un-specked hardyheads, dwarf flat-headed gudgeons and total abundance (excluding Australian smelt), and negatively correlated with Australian smelt (Fig. 3.5). Increasing inlet depth and river level were negatively correlated with the abundance of bony herring. Goldfish *Carassius auratus* abundance decreased around the time of the each full moon and peaked close to each new moon. Northerly winds at 9 am were negatively correlated with goldfish abundance but were positively correlated with redfin perch *Perca fluviatilis* abundance at 3 pm.

Table 3.4. Significant Spearman rank correlations (r_s) between fish species abundance, and environmental variables through time.

	Environmental factor	Biological factor	r_s	*P*
Climatic	River water temperature	Carp gudgeons	0.775	0.002
		Australian smelt	-0.747	0.003
		Total abundance (Australian smelt removed)	0.742	0.004
	Wetland water temperature	Total abundance (Australian smelt removed)	0.758	0.003
		Murray-Darling rainbowfish	0.758	0.003
		Carp gudgeons	0.725	0.005
		Bony herring	0.725	0.005
	Inlet water temperature	Carp gudgeons	0.742	0.004
		Total abundance (Australian smelt removed)	0.725	0.005
		Murray-Darling rainbowfish	0.698	0.008
		Australian smelt	-0.687	0.010
	9 am Mean sea level air pressure	Total abundance	0.786	0.001
		Australian smelt	0.736	0.004
	9 am Wind direction	Goldfish	-0.900	<0.001
	3 pm Wind direction	Redfin perch	0.720	0.005
Hydraulic	Mean inlet depth	Bony herring	-0.896	<0.001
	River level	Bony herring	-0.687	0.009
	Flow at Blanchetown	Golden perch	0.684	0.010
Astronomical	Day length	Carp gudgeons	0.813	0.001
		Total abundance (Australian smelt removed)	0.791	0.001
		Flat-headed gudgeon	0.736	0.004
		Un-specked hardyhead	0.681	0.010
		Australian smelt	-0.681	0.010
		Dwarf flat-headed gudgeon	0.670	0.010
	% Lunar illumination	Goldfish	-0.680	0.010

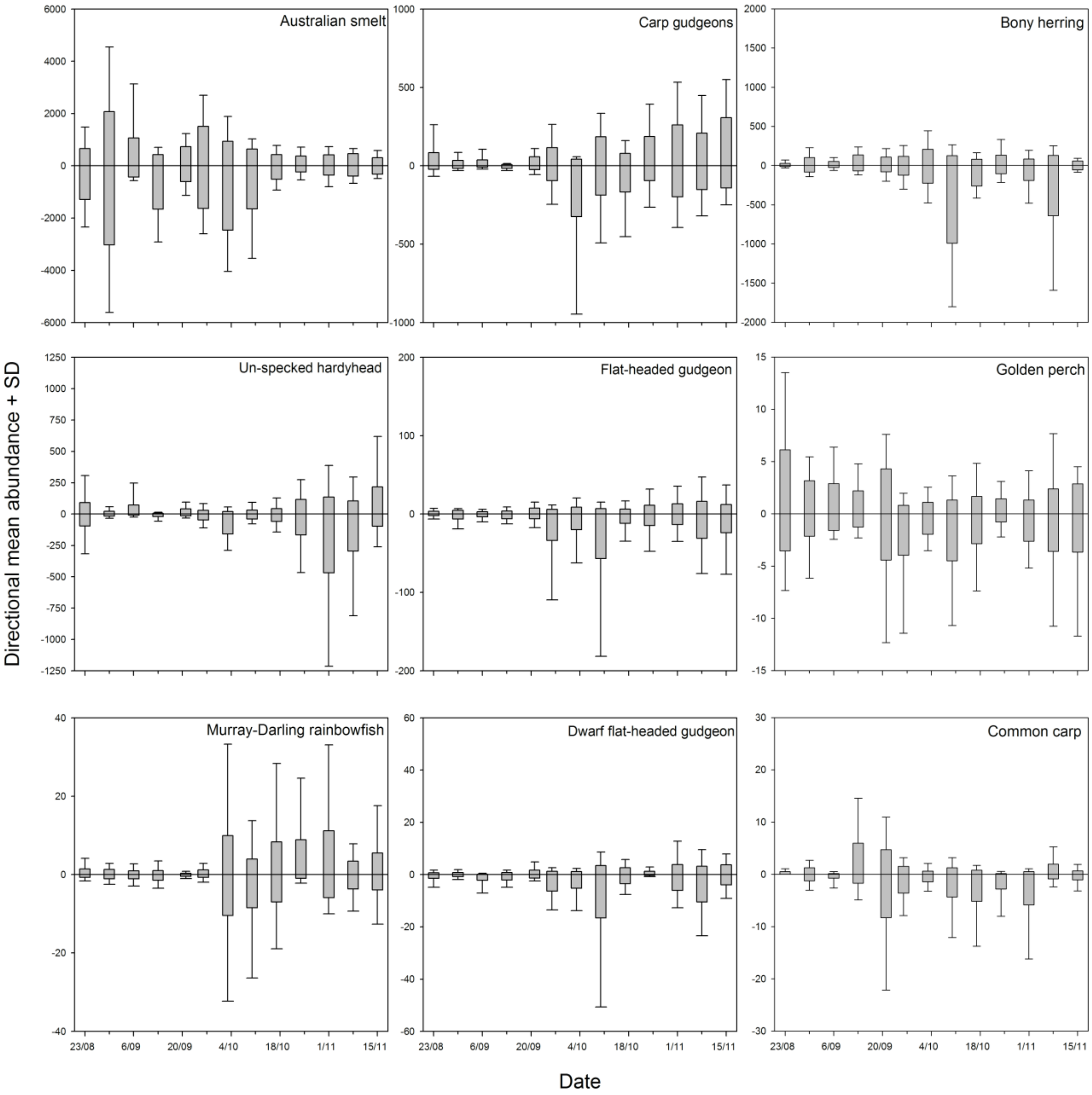

Figure 3.3. Weekly mean (± 1 SD) abundance ($\log_{10}(x+1)$) of the nine most common fish species moving between off-channel wetlands and the main river channel. Positive values indicate fish moving from river to wetland and negative values indicate fish moving from wetland to river. Note: scale of y-axis varies.

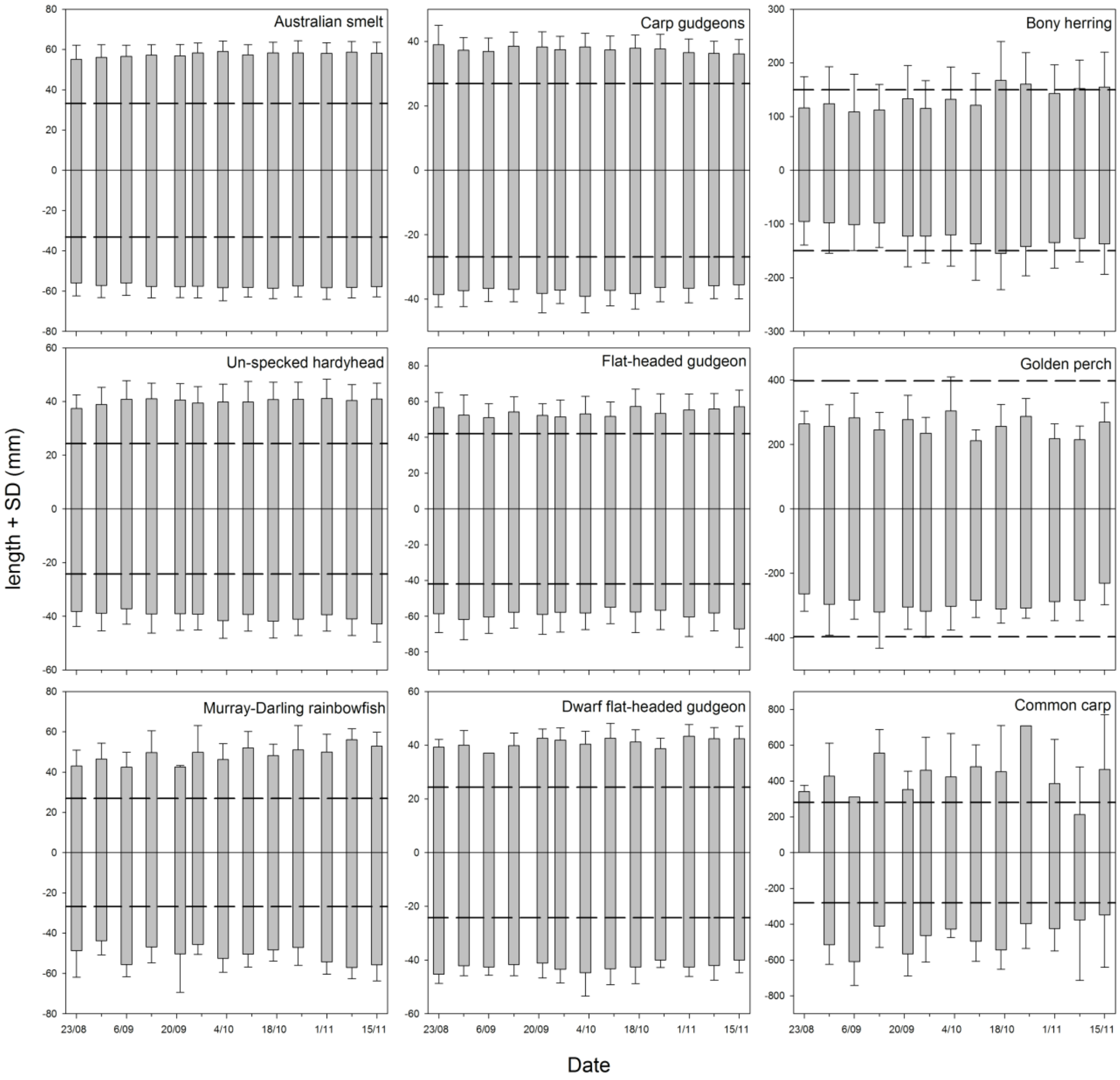

Figure 3.4. Weekly mean (± 1 SD) total length (mm) of the nine most common fish species moving between off-channel wetlands and the main river channel. Positive values indicate fish moving from river to wetland and negative values indicate fish moving from wetland to river. Minimum length at maturity of females is shown by dashed lines; lengths from Pusey *et al.* (2004) (Australian smelt, carp gudgeons, un-specked hardyheads, flat-headed gudgeon, Murray-Darling rainbowfish and dwarf flat-headed gudgeon), Puckridge and Walker (1990) (bony herring), Mallen-Cooper and Stuart (2003) (golden perch) and Brown *et al.* (2005) (common carp). Note: scale of y-axis varies.

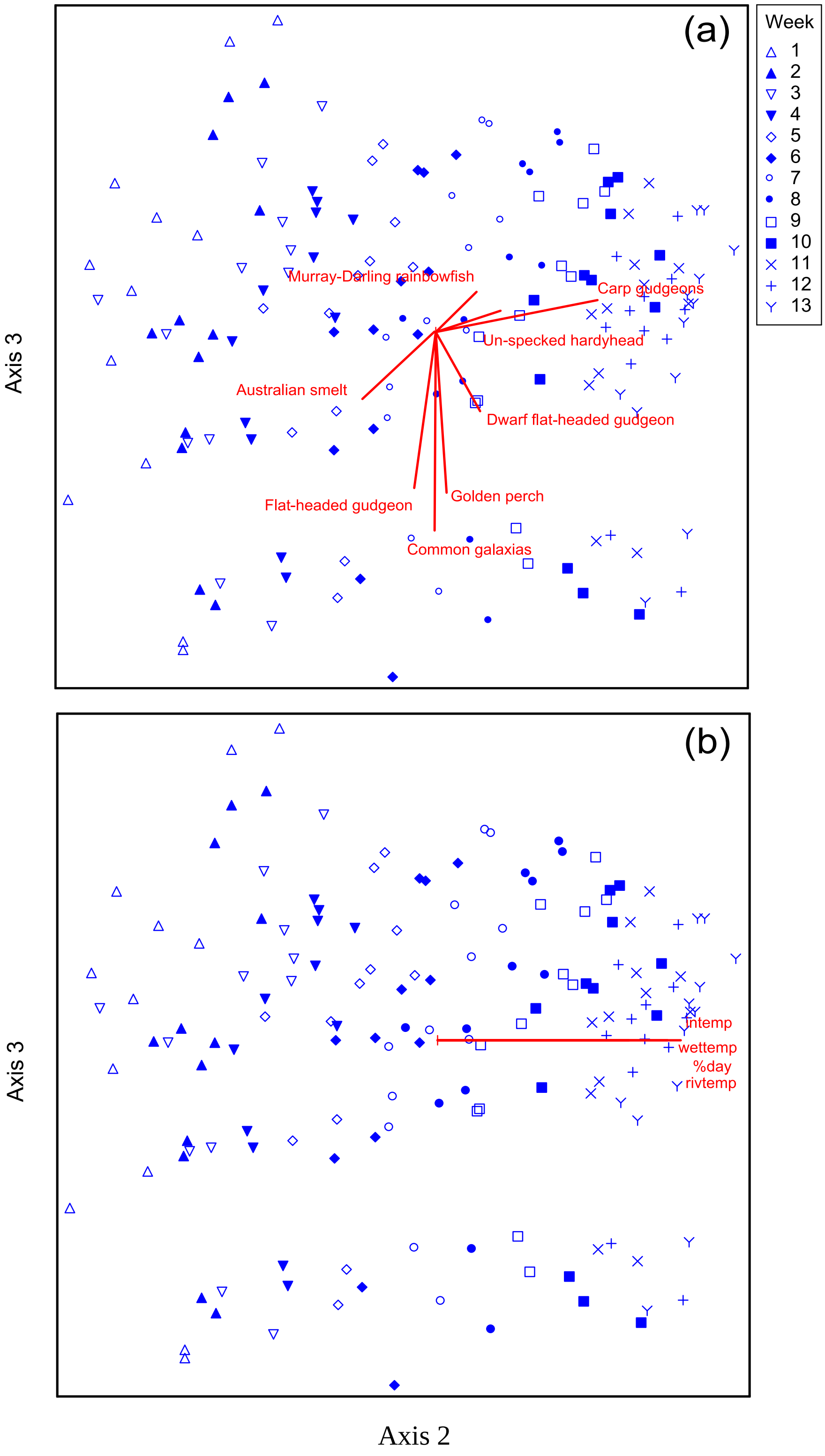

Figure 3.5. NMS ordination plot showing (a) species superimposed on NMS and (b) environmental variables superimposed on assemblage data (intemp = inlet temperature, wettemp = wetland temperature, %day = day length, rivtemp = river temperature).

3.5. DISCUSSION

3.5.1. Temporal and directional patterns

Large numbers of fish were recorded moving through the six wetland inlets over the 13 weeks of this study. Fish movements were typically bi-directional, asynchronous and haphazard, consistent with our original hypothesis and the movement model proposed by Lyon *et al.* (2010), where movements between instream and offstream habitats are approximately equal when water levels maintain a stable connection between habitats. The net movement of all fishes from the wetlands may relate to the gradual draw-down of the river level over the 13 weeks (mean water level reduction of 17 cm, *c.* 10-30% of wetland depth), with emigration being a typical fish response to falling water levels in temporary habitats (Poizat and Crivelli 1997; Cucherousset *et al.* 2007; Lyon *et al.* 2010).

Analysis of the lengths of the dominant species did not indicate differences in the size structure moving in and out of wetlands, though the absence of larval small-bodied fish may be an artefact of the net mesh size (8-mm stretched) used. Spring-summer is a typical period for fish spawning and recruitment in the MDB (e.g. King *et al.* 2003) and freshwater fish often make ontogenetic habitat shifts (Werner and Gilliam 1984; King 2004). Anecdotal observations (e.g. river fishers, shack owners, lock operators) suggest that typical carp spawning movements did not occur in spring 2006, possibly due to the absence of flow related cues. Directional length differences may have been apparent (more small fish leaving wetlands, reflecting recruitment in wetlands over summer) had autumn sampling been undertaken as initially planned.

Mean total abundance and the mean abundance of dwarf flat-headed gudgeons moving out of wetlands were the only variables to increase over the 13 weeks. Dwarf flat-headed gudgeons were generally caught in low numbers (<10 individuals) moving in either direction, but higher numbers (~50) were recorded on three occasions emigrating from the Riverglades wetland. The high abundance of dwarf flat-headed gudgeon at Riverglades is likely explained by the rocky, ‘rip-rap’ lined banks of the inlet channel, an ideal habitat for this species (Lintermans 2007).

The haphazard movement patterns that we report correspond to the spatially variable, generally non-directional movement patterns described at the six wetland inlets by Conallin *et al.* (2010), when the same fish movement data were pooled among the 13 weeks, rather than being pooled among the six wetlands as in this study. This finding corresponds to our hypothesis that fish movements are temporally asynchronous and haphazard when hydraulic connections are present, but hydraulic cues are absent. Differences in fish assemblages were apparent when fish movements were analysed among wetlands. The spatial differences were best explained by variable wetland and inlet morphology, and the distance of each wetland from the river mouth with diadromous fishes recorded at wetlands closer to the river mouth. As in this study, fish lengths did not differ with direction of movement among wetlands (Chapter 2; Conallin *et al.* 2010). When considered both temporally and spatially, fish movements in the Lower Murray varied over time and among wetlands, though directional movement patterns were absent regardless of whether data were analysed temporally or spatially.

3.5.2. Temporal cues

Only five of the most abundant species, all small bodied, displayed temporal trends in catch over the study period, ranging from decreasing (Australian smelt) to increasing (carp gudgeons, un-specked hardyheads, flat-headed gudgeons and Murray-Darling rainbowfish) abundance. Ordination and correlation analysis revealed that increasing day length and water temperature (wetland, inlet and adjacent river) were correlated with these patterns.

Fish are typically more active at higher water temperatures (Lucas and Baras 2001). In our study, warming water temperatures had the greatest effect on the movement activity of small-bodied native fish and had no effect on large-bodied natives or common carp. In contrast, movements of common carp elsewhere in the MDB are triggered by increasing water temperature (as well as increasing river discharge (lateral movements only)) (Mallen-Cooper 1996; Jones and Stuart 2008a). The influence of increasing water temperature alone, without influence from rising water level or increasing flow, on large-bodied native species movements is variable. For example, longitudinal golden perch movements are influenced by the combination of rising water temperatures and increasing flow and river level (Mallen-Cooper 1996; Mallen-Cooper 2001; O'Connor *et al.* 2005), whereas longitudinal bony herring movement is stimulated by increasing temperature alone (Mallen-Cooper 1996; Mallen-Cooper 2001; Baumgartner 2006).

Day length showed similar patterns to water temperature (and the two variables were correlated), whereby movements of small-bodied species (e.g. Australian smelt, carp gudgeons, flat-headed gudgeon) were most influenced by increasing day length. While not apparent for all species, day length similarly influences movements of some fish in North American streams (Albanese *et al.* 2004).

Hydraulic variables such as flow or river height had little influence on fish movements. An exception was bony herring, whose abundance was negatively correlated with daily fluctuations in river level, and hence inlet depth. River level and inlet depth, hence direction of flow to or from wetlands, are typically driven by wind-induced seiches which can cause fluctuations in river level of up to 30 cm (between *c.* 20-40% of inlet depth) in the study reach (Webster *et al.* 1997). While river level gradually dropped over the 13 weeks, seiches rapidly altered river and wetland level on a far shorter temporal scale (*c.* 24 h).

The influence of lunar cycles on the movements of some riverine fish is well recognised. For example, some species migrate in higher abundances during the various phases of the moon (e.g. Cossette and Rodriguez 2004; Miyai *et al.* 2004; Bizzotto *et al.* 2009), although this is not reported from Australia (Humphries 2005; Butler and Rowland 2009). Goldfish were the only species whose movements were apparently influenced by lunar cycles, in that the abundance of goldfish peaked near the new moon and was lowest around the full moon. It is possible that this may be an artefact of the sampling duration which spanned only three full lunar cycles. Few studies of Australian riverine fishes have specifically considered lunar influence on fish movements. Carp movement through a fishway was poorly correlated to lunar phase (Champion *et al.* 2001). Similarly, studies of the spawning behaviour of Australian cod species (*Maccullochella* spp.) have failed to detect any lunar influence (Humphries 2005; Butler and Rowland 2009).

Other environmental factors were correlated with fish activity, but it is unclear whether these factors are directly correlated with movements or represented the synergistic effects of other interrelated variables. For example, increasing air pressure was correlated with greater Australian smelt abundance and morning northerly winds were negatively correlated with the abundance of goldfish but afternoon northerly winds were positively correlated with redfin perch. Seiching in the study reach is influenced by both air pressure and wind direction; falling air pressure is linked to frontal weather systems which are preceded by strong northerly winds and followed by strong southerly winds (Webster *et al.* 1997).

3.5.3. Why are fish moving laterally?

Fish make lateral movements to exploit inundated habitats for food or reproduction (e.g. Castello 2008; Zeug and Winemiller 2008) or to seek refuge (e.g. Copp 1997; Borcherding *et al.* 2002). The dominant fish recorded moving laterally in the Lower Murray are considered to be ‘ecological generalists’ (Mallen-Cooper 2001; Smith *et al.* 2009b) with flexible spawning behaviours that enable them to exploit river and wetland habitats.

The haphazard lateral movements observed may relate to feeding movements during a period of nutrient limitation associated with the stable water levels affecting primary productivity (cf. Baker *et al.* 2000). While not quantified in the study reach, terrestrial nutrient inputs can drive food webs in freshwater systems elsewhere in the MDB (Reid *et al.* 2008) and in rivers worldwide (e.g. Wipfli 1997; Herwig *et al.* 2004; Mehner *et al.* 2005). As the most abundant fish species sampled are microphagic carnivores (Australian smelt, carp gudgeons, un-specked hardyheads and flat-headed gudgeons) or detritivores (bony herring) (Harris 1995), it is possible that terrestrial inputs (e.g. insect and leaf litter) may be important dietary components. It may be that the extension of the riparian zone (important for contribution of organic matter and habitat structure: Pusey and Arthington 2003), through ongoing maintenance of connected wetlands with extended shorelines has compensated for the reduction in flood-driven productivity pulses, sustaining native fish in the Lower Murray despite the recent drought and prolonged period of perennial wetland inundation.

3.5.4. Management implications

Numerous projects to restore the ecological integrity of wetlands artificially inundated by river regulation are underway in the Lower Murray (e.g. Little Duck Lagoon: Jensen 2002). Inundated wetlands are periodically dried to extirpate carp, reduce turbidity by drying the substrate and promote nutrient cycling (Jensen 2002). Periodic drying is achieved by installing wetland regulators which modify lateral connectivity and can affect lateral fish movements (Mallen-Cooper 2001). However, as wetlands are ideally dried over relatively short time spans (drying for 2-4 months over summer-autumn) and only every 2-3 years (Pressey 1987), it is likely that the benefits associated with wetland drying (e.g. improved nutrient cycling, sediment consolidation and aeration) offset any localised loss of habitat for the Lower Murray fish caused by drying the few (*c.* 10% of permanently connected wetlands) regulated wetlands. As many native fish species will move laterally when wetlands are inundated, any flow control regulators involved with wetland rehabilitation projects should be designed and operated to maximise fish passage (Mallen-Cooper 2001). How fish in the Lower Murray will be affected by current management trends, which dry some wetlands over longer periods (*c.* 12+ months) to reduce evaporative water losses, remains unclear.

Common carp are the dominant alien species in the Lower Murray (Smith *et al.* 2009b) where, as in many regions, they damage wetlands (Jensen 2002) and have potential to breed in large numbers (Smith and Walker 2004). To reduce the localised impacts caused by common carp, their access to wetlands in the Lower Murray is currently (unsuccessfully) managed by carp exclusion screens (Chapter 4; Hillyard *et al.* 2010). Carp separation cages (Stuart *et al.* 2006a) adapted for use in wetland inlets show promise (Smith *et al.* 2009a; Thwaites *et al.* 2010). Our data showed that common carp moved continuously during spring, a pattern which has also been observed in other regulated rivers with permanent off-channel habitats (e.g. Mississippi: Schultz *et al.* 2007). Therefore carp screens and traps on perennially inundated wetlands in the Lower Murray need to be applied over long time frames, at least from spring through autumn, as carp move to comparatively deep, thermally stable river habitats for overwintering (Penne and Pierce 2008). Hence traps should be designed and operated to minimise impacts on native fish which also move laterally over the warmer months (see Smith *et al.* 2009a; Hillyard *et al.* 2010). Future monitoring of lateral fish movements in the Lower Murray over longer time scales and during periods of higher river flow may demonstrate clearer patterns of common carp movement, including confirmation of anecdotal observations of mass movements of carp to wetlands each spring.

3.5.5. Conclusion

Movements of the abundant and diverse fish assemblage were temporally asynchronous and haphazard in the presence of permanent hydraulic connections between the river and the wetlands, corresponding to previous 'spatial' analysis of data (Conallin *et al.* 2010). While typically bidirectional, the net movement of fish from the wetlands to the river over the 13 weeks of the study is likely a result of the river level being lowered to reduce evaporative water loss by disconnecting wetlands, to conserve water supplies in the drought-impacted Lower Murray for 'critical human needs'. The broad-scale loss of wetland habitat (some 45 wetland complexes: Smith *et al.* 2009b) below Lock 1 caused by the lowering of the river level may have little effect on the ecological generalists which dominate the Lower Murray fish assemblage, but is likely to further exacerbate the decline of the region's threatened fish which are dependent on wetlands (e.g. Agassiz's glassfish *Ambassis agassizii*, southern purple-spotted gudgeon *Mogurnda adspersa*: Hammer *et al.* 2009). The low river levels below Lock 1 have provided an opportunity to enact change in how the region's wetlands, artificially inundated since the 1940s, are managed. Further installation of fish-friendly regulators, which incorporate appropriate carp management tools (carp traps and screens), would allow a reintroduction of regular wetland drying and facilitate local carp management in a region recognised as a carp spawning hotspot. As fish will move between instream and offstream habitats when water levels allow, any regulators installed to facilitate wetland management must include provision for native fish passage and should allow for the management of alien species, especially carp, through the use of carp screens or similar tools.

4. Chapter Four - OPTIMISING EXCLUSION SCREENS TO CONTROL EXOTIC CARP IN AN AUSTRALIAN LOWLAND RIVER

KARL A. HILLYARD[a], BENJAMIN B. SMITH[b], ANTHONY J. CONALLIN[a,b] AND BRONWYN M. GILLANDERS[a]

[A] *Southern Seas Ecology Laboratories, School of Earth and Environmental Sciences, Darling Building, DX 650 418, University of Adelaide, SA 5005, Australia.*

[B] *Inland Waters and Catchment Ecology Program, SARDI Aquatic Sciences, PO Box 120, Henley Beach, SA 5022, Australia.*

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Statement of Authorship

In this chapter, Karl Hillyard developed the study, collected and analysed data, and drafted the manuscript. Ben Smith and Bronwyn Gillanders advised on development of the study, data analysis and contributed to the synthesis and finalisation of the manuscript. Ben Smith and Anthony Conallin also assisted with collection of data in the field.

Certification that the statement of contribution is accurate.

Karl Hillyard (Candidate)

Signed *Date* 11/5/2011

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

Benjamin B. Smith

Signed *Date* 11/5/2011

Anthony J. Conallin

Signed *Date* 11/5/2011

Bronwyn M. Gillanders

Signed *Date* 11/5/2011

4.1. ABSTRACT

Carp exclusion screens (CES) are used to restrict adult common carp from entering wetlands thereby minimising their ecological impacts, and spawning and recruitment potential, but there is marked variation in current CES design and management. We quantified current CES designs, dimensions and locations within the Murray-Darling Basin, Australia. Directional fyke nets at inlets of six permanently inundated wetlands were used to identify fish using wetlands and therefore potentially vulnerable to CES. Morphometric data from captured fish were then used to design CES that excluded sexually mature carp. The ability of optimised and existing CES designs to exclude large-bodied fishes that used wetlands was then assessed. Fifty-four CES with eight mesh designs and varied dimensions were identified. We recorded 18 species comprising some 210 000 fish in the wetland inlets. Two optimised meshes to exclude sexually mature carp were developed: a 44-mm square grid mesh and a ‘jail bar’ mesh with 31.4-mm gaps. Modelling revealed that up to 92% of carp could be excluded by either optimised mesh design, although few young-of-year carp were caught. Optimised and existing CES designs would also exclude 2-65% of large-bodied native fishes. Optimised CES may allow localised carp control without restricting passage of some key native fishes.

4.2. INTRODUCTION

Common carp *Cyprinus carpio* (hereafter carp) have established self-sustaining populations on all habitable continents (Lever 1994). In the Murray-Darling Basin (MDB) carp may compete with native species for resources (Cadwallader 1978) and their benthic feeding behaviour is linked to increases in turbidity (King *et al.* 1997), total nitrogen (Driver *et al.* 2005a), and damage of shallow-rooted, soft-leaved submerged macrophytes (Fletcher *et al.* 1985; Roberts *et al.* 1995). These adverse impacts are pronounced in lentic, well-vegetated, warm, shallow habitats such as wetlands (Gehrke *et al.* 1995; Koehn *et al.* 2000) where carp congregate to breed (Stuart and Jones 2006a; Jones and Stuart 2008a).

Controlling alien invasive fish is a worldwide issue (Moyle and Light 1996). For carp, genetic (Thresher 2008), biological (McColl *et al.* 2007) and chemical (Sorensen and Stacey 2004) control technologies are being developed, but are still many years from deployment. Current control options have application at the individual wetland or river reach scale. They include “Judas” carp (Diggle *et al.* 2004), jumping (Stuart *et al.* 2006a) and pushing (Thwaites *et al.* 2010) traps, water level manipulation to expose spawned eggs to desiccation (Sheilds 1958), electrical barriers (Verrill and Berry 1995), non-specific chemical piscicides (e.g. Marking 1992), blocking nets to reduce carp access to spawning substrate (Parkos *et al.* 2006) and carp exclusion screens (French *et al.* 1999). However, carp control in Australia has been described as “sporadic, short lived and rarely successful” (Stuart *et al.* 2006a). New control options are needed, and the design, use and management of some existing technologies require further investigation.

One area that warrants further research and field application is to control carp at their ‘source habitats’ to minimise their spawning and recruitment potential. Source habitats include seasonally inundated wetlands (as opposed to main-channel habitats) in the mid reaches of the River Murray (Stuart and Jones 2006a) where up to 98% of carp recruits are produced (Crook and Gillanders 2006). Similarly, 70% of wetlands in the Lower Murray are perennially inundated and permanently connected, and are recognised as important carp spawning habitats (Vilizzi 1998a; Smith and Walker 2004). These permanent connections have altered the lateral connectivity, the connection between the river and its floodplain, within the Lower River Murray. Maintenance of lateral connectivity is important to allow fauna access to key spawning and feeding grounds, and to sustain the exchange of resources in the riverine environment (Ward 1989; Lucas and Baras 2001). Regulation of the Lower River Murray has artificially maintained lateral connectivity between the river and many wetlands which would have seasonally filled and dried under a natural flow regime (Walker and Thoms 1993), providing ideal carp habitat (Smith *et al.* 2009b).

Drying artificially inundated wetlands *via* evaporation can eradicate stranded carp. Drying in this way is facilitated by installing flow control structures (e.g. box or pipe culverts) which have stop logs (cf. Schultz *et al.* 2007) or sluice gates to prevent water from entering the wetland. During filling and under normal bankfull conditions, carp exclusion screens (CES) are used in place of the logs or gates. Carp exclusion screens are physical mesh barriers that are placed across wetland inlets to exclude large fish from entering a wetland (Meredith *et al.* 2006). The perceived benefits of carp exclusion at wetland inlets using CES are: (1) limiting carp access to potential spawning habitat (Stuart and Jones 2006a; Jones and Stuart 2008a), (2) improved water quality and increased diversity and abundance of macrophytes (Lougheed *et al.* 1998; Lougheed and Chow-Fraser 2001; Lougheed *et al.* 2004), (3) the ability to extirpate populations of adult carp *via* stranding during drying (Jensen 2002), and (4) allowing the passage of small-bodied native fishes, which comprise the majority of fish in wetlands (Smith *et al.* 2009b). However, CES have potential disadvantages as they are not species-selective, and they exclude or inhibit any large-bodied aquatic species (e.g. native fish and turtles) that are too large, unable or unwilling to bypass the screen. Furthermore, flow through a screen may cause small fishes to contact the mesh, potentially resulting in injury (Swanson *et al.* 2005), and large-bodied native species that enter as juveniles can grow to a size incapable of passing back through a CES and are therefore also stranded during drying (assuming the screens are not removed or other actions are not undertaken to rescue the fish). Clearly, screens are also not effective during over bank floods, they will not exclude carp that are smaller than the mesh aperture (Navarro and Johnson 1992), and they may also become fouled with debris, which could affect their exclusion efficiency and may result in increased current velocities (velocities >0.15 m.s^{-1} are a physical barrier to small native fishes such as carp gudgeons *Hypseleotris* spp.: Mallen-Cooper 2001). Like all structures, they require ongoing maintenance.

This study aimed to (1) document the current locations and designs of the flow control structures fitted with CES in the MDB, and the designs and dimensions of the CES mesh, (2) compare the composition and size of aquatic fauna using inlets of several permanently inundated wetlands with relatively stable water levels, and (3) use morphometric data from common large-bodied species (adult length >100 mm total length, TL) to design optimised CES to restrict sexually mature carp (≥250 mm TL). We then modelled the exclusionary efficiency of optimised and existing designs on other large-bodied fishes recorded using the wetland inlets.

4.3. METHODS

4.3.1. Site description

Discharge of the MDB (1 073 000 km^2) is characterised by low and variable flows, driven by a predominantly semi-arid climate (Walker and Thoms 1993). The Murray River is highly regulated by 14 low-level weirs, floodplain levees and numerous smaller regulators (Maheshwari *et al.* 1995) to provide stable water levels for irrigated agriculture. Regulation of the Lower Murray has resulted in approximately 70% of the wetlands becoming perennially inundated (Pressey 1990). At the time of this study, the MDB was experiencing severe drought which, coupled with extraction for human consumption, resulted in a period of low, stable, within-channel flows, especially in the Lower Murray where fish surveys for this study were undertaken (see below). Mean river height in the study reach during the surveys was 0.78 m (± 0.01 s.d.) (Department of Water, Land and Biodiversity Conservation, unpublished data), a typical bankfull level, and flows to the study reach were only 18% and 46% of the 1949-2006 mean and median flows respectively.

4.3.2. Field survey of existing carp exclusion screens in the Murray-Darling Basin

Wetlands with CES fitted to flow control structures within the MDB were initially identified from the literature (e.g. Nichols and Gilligan 2003), and consultation with catchment management authorities and regional researchers. Wetland CES were surveyed over three weeks during June and July 2007. For each CES, details of the shape of the mesh, construction materials and the dimensions of the mesh were recorded to identify designs which could be easily manipulated to design ‘optimised’ CES to exclude all carp larger 250 mm TL. This exclusion threshold was chosen based on the known size at sexual maturity of carp in Australia (approximately 300 mm caudal fork length, 50% of population: Brown *et al.* 2005). The flow control structure associated with the CES was also described in terms of its type (box culvert, pipe culvert, sheet piling system, stand-alone carp exclusion screen) and ability to manipulate wetland water levels (structural integrity, presence or absence of stop-logs or sluice gates).

4.3.3. Use of wetland inlets by native and alien fish

To evaluate the use of wetland inlets by native and alien fish and other fauna, wetlands in the Lower River Murray, South Australia, were sampled weekly from mid-August to mid-November 2006 (austral spring), coinciding with the peak season for the movement of MDB fishes for feeding and spawning (Humphries *et al.* 1999; King *et al.* 2003). Six perennially inundated wetlands were chosen for sampling to represent a diversity of wetland types (Table 4.1). During sampling, we set two double-winged, small-mesh (8-mm stretched) fyke nets within the main inlet of each wetland, once per week, overnight for approximately 24 h. The nets used at each wetland were custom-made to cover the entire width of the inlet channel, and they were heavily weighted and floated to ensure that the whole water column was sampled. One net was set facing the river to capture river fish moving toward the wetland and the other faced the wetland to capture wetland fish moving toward the river. All captured fish were sorted and identified to species. Random sub-samples of up to 50 fish per species were then measured to the nearest mm for TL, body width (W, across the widest span, generally anterior to the pectoral fin) and body depth (D, across the deepest span, generally anterior to the dorsal fin) for morphometric analysis.

4.3.4. Conceptual design of optimised carp exclusion screens

Based upon CES examined during the field survey, two designs (‘grid mesh’ and vertical ‘jail bars’ (after French *et al.* 1999)) were identified which could be optimised to exclude all carp larger than 250 mm TL (Appendix 2). The required dimensions for each optimised mesh design were calculated from morphometric data from 113 carp (see below). Regressions were developed for TL-W, TL-D and W-D. A 10% compression factor was applied to W and D data to compensate for carp’s ability to compress their bodies and squeeze through gaps (French *et al.* 1999). For the square grid mesh, it was assumed that fish could pass through the mesh on a 45° angle (from vertical: as per Webb *et al.* 1996) and would therefore be constrained by their body depth. The mean compressed body *depth* of a 250-mm TL carp was then used as the hypotenuse and the length of the square sides were calculated by applying a trigonometric formula. In contrast, the optimal dimension for the vertical jail bars was derived from the mean compressed body *width* of a 250-mm TL carp.

To assess likely impacts of the optimised screens on other large-bodied (>100 mm TL at maturity) fishes, morphometric data were collected from a sub-sample of the four most common large-bodied fishes captured during the surveys of wetland inlets: native bony herring *Nematalosa erebi* (*n* = 184) and golden perch *Macquaria ambigua* (*n* = 227), and the alien species goldfish *Carassius auratus* (*n* = 45) and redfin perch *Perca fluviatilis* (*n* = 30). For each species, regressions were again developed for TL-W, TL-D and W-D. The 10 % compression factor was applied to all species, except bony herring, which have laterally compressed bodies that likely cannot be further compressed (as per the “squeezability hypothesis”: Robichaud *et al.* 1999). The slopes of TL-W, TL-D and W-D regressions were compared among species using three one-way analyses of covariance (ANCOVA), using TL and W as covariates to remove their effect on W and D respectively. All statistical analysis was conducted using SPSS 15 software (SPSS Inc. Chicago, Illinois).

4.3.5. Modelling the likely efficiency of optimised carp exclusion screens

The morphometric regressions developed for all five large-bodied species were used to calculate the theoretical exclusion efficiency of both optimised designs (31.4-mm jail bars and 44-mm grid mesh) and Alu-Tread Series 13 walkway mesh (Locker Group, Melbourne Australia), which is the most common CES design currently in use in the MDB (see Results) (Appendix 2). Modelling was undertaken using mesh dimensions calculated to exclude carp ≥250 mm TL for the optimised designs and the fixed dimensions of Alu-Tread, a diamond-shaped mesh, with a maximum aperture of 97 mm by 34 mm (Table 4.2). The morphological regressions were used to determine the maximum size of the five fishes that would pass the three screen designs. The maximum sizes were then extrapolated to the length frequency distributions for each species captured during the spring 2006 wetland inlets surveys to determine the percentage of the total population of each large-bodied fish species that would have been excluded by each screen design.

Table 4.1. Details of the six wetland inlets where fish were sampled in spring 2006.

Description	Latitude (°S)	Longitude (°E)	Wetland area (ha)	Distance to river mouth (km)	Inlet length (m)
Morgans	34.4709	139.6010	22.99	260	184.42
Noonawirra	34.4977	139.5745	6.25	255	60.28
Nildottie	34.6726	139.6495	2.59	225	18.93
Kroehns	34.7126	139.5743	46.25	215	758.79
North Purnong	34.8414	139.5947	84.94	194	272.03
Riverglades	35.0893	139.3066	21.30	117	165.46

Table 4.2. Summary information for CES meshes, including the number (*n*) and shape of each mesh, and the maximum vertical (Y_{max}) and horizontal (X_{max}) aperture range (s), in use in the Murray-Darling Basin in June-July 2007.

Name	*n*	Shape	Y_{max} (mm)	X_{max} (mm)
Alu-Tread Series 13 (walkway)	24	Octagonal	34	97
Welded grid mesh	10	square or rectangular	21-50	22-75
Security mesh	5	Octagonal	52-58	55-63
Woven grid mesh	5	Square	22.5	22.5
Reinforced Alu-Tread Series 13	4	Hexagonal	34	43
Reinforced security mesh	3	Hexagonal	26	34
Horizontal bars	4	Rectangular	15	290-450
Jail bars	1	Rectangular	330	7

4.4. RESULTS

4.4.1. Field survey

Fifty-four flow control structures fitted with CES were visited (Appendix 1); most were within the inlets to wetlands located adjacent to the River Murray. The majority of screens were in South Australia (*n* = 45) with eight in Victoria and only one in New South Wales. None were found in Queensland or the Australian Capital Territory. Eight mesh designs were identified (Table 4.2); the most common being 'walk-way' mesh (*n* = 24, 44%; identified as Alu-Tread Series 13 mesh), followed by welded grid mesh (*n* = 10, 19%).

Carp exclusion screens were most commonly fitted to box (*n* = 32, 59%) and pipe culverts (*n* = 13, 24%). However, some screens were free-standing (*n* = 6, 11%), or fitted to sheet-piling systems (modular plastic panels inserted between piles sunk in the inlet substrate, *n* = 2, 4%), or in one case, a well (2%). Most flow control structures had stop-logs or sluice gates to control water entry (*n* = 43, 80%). We also observed a number of structures that appeared to leak, despite the presence of stop-logs or sluice gates. Other sites had unsecured screens that were liable to tampering.

4.4.2. Use of wetland inlets by native and alien fish

Some 210,000 fish from 18 species, including 13 native and 5 alien species were captured (Table 4.3). Small-bodied (<100 mm TL at maturity) native fishes, primarily Australian smelt comprised the bulk of the catch (*n* = 185 086, 86.9%). Of the large-bodied (>100 mm TL at maturity) fishes, native bony herring and golden perch along with the invasive alien carp, redfin perch and goldfish were the most common species (Table 4.3).

Of the large-bodied species likely to be impacted by CES, carp had a tri-modal length-frequency distribution, dominated by adults peaking at 450 mm and 700 mm TL and a smaller number of juveniles at 50 mm TL (Fig. 4.1). In contrast, the catch of golden perch, bony herring, goldfish and redfin perch was dominated by individuals <400 mm TL (Fig. 4.1). For golden perch, sub-adult fish (<300 mm TL), dominated the catch, whereas bony herring were dominated by large numbers of age 1+ fish (150 mm TL: Puckridge and Walker 1990). Goldfish catch was dominated by large numbers at 200 mm TL. Redfin perch were mostly young-of-year (YOY) (<100 mm TL), with a secondary peak in the length-frequency distribution at 300 mm TL (Fig. 4.1).

Three turtle species were also captured, including the eastern snake-necked turtle *Chelodina longicolis* (*n* = 104), Murray short-necked turtle *Emydura macquarii* (*n* = 16) and the broad-shelled turtle *Chelodina expansa* (*n* = 7). Freshwater shrimp *Paratya australiensis*,

Table 4.3. Catch summary and morphometric data (mean and range (mm) for total length (TL), body width (W) and body depth (D)) for large-bodied (>100 mm TL at maturity) and small-bodied (<100 mm TL at maturity) fishes captured at six wetland inlets in the Lower River Murray, South Australia, during mid-August to mid-November 2006.

Species	*n*	TL	W	D
Large-bodied native species				
Bony herring *Nematalosa erebi*	26,531	130.2 (22-457)	14.8 (3-45)	38.2 (7-163)
Golden perch *Macquaria ambigua ambigua*	410	275.2 (81-478)	40.4 (16-83)	80.2 (30-305)
Pouched lamprey *Geotria australis*	2	425 (350-500)	-	-
Freshwater catfish *Tandanus tandanus*	1	280.0	-	-
Murray cod *Maccullochella peelii*	1	97.0	-	-
Congolli *Pseudaphritis urvillii*	1	142.0	-	-
Small-bodied native species				
Australian smelt *Retropinna semoni*	148,917	57.6 (32-82)	-	-
Carp gudgeon complex *Hypseleotris* spp.	18,172	37.2 (24-71)	-	-
Un-specked hardyhead *Craterocephalus stercusmuscarum fulvus*	14,746	40.3 (25-68)	-	-
Flat-headed gudgeon *Philypnodon grandiceps*	2,016	56.8 (20-87)	-	-
Murray-Darling rainbowfish *Melanotaenia fluviatilis*	605	50.5 (30-90)	-	-
Dwarf flat-headed gudgeon *Philypnodon macrostomus*	511	42.5 (14-82)	-	-
Common galaxias *Galaxias maculatus*	119	87.3 (39-114)	-	-
Large-bodied alien invasive species				
Common carp *Cyprinus carpio*	351	473.5 (20-770)	62.9 (27-151)	116.0 (44-195)
Redfin *Perca fluviatilis*	223	124.2 (32-358)	40 (8-62)	70.5 (38-104)
Goldfish *Carassius auratus*	222	156.7 (78-300)	27.4 (11-70)	52.1 (22-165)
Cyprinus X Carassius Hybrid	8	315.0 (215-407)	47.8 (30-73)	90.5 (58-120)
Small-bodied alien invasive species				
Eastern gambusia *Gambusia holbrooki*	91	33.2 (24-63)	-	-

freshwater prawns *Macrobrachium australiense* and yabbies *Cherax destructor* were also caught, but were not enumerated.

4.4.3. Conceptual design of optimised carp exclusion screens

After adjusting for length or width, significant differences in the length-width ($F_{4,590} = 57.49$, $P < 0.001$), length-depth ($F_{4,589} = 78.03$, $P < 0.001$) and width-depth ($F_{4,589} = 124.72$, $P < 0.001$) ratios were found between the five large-bodied fishes examined. Similar body morphologies were observed between golden perch, carp, redfin perch and goldfish, but all were different to bony herring (Fig. 4.2*a*-*c*). Bony herring are laterally compressed (slope of width vs. depth = 2.55; Fig. 4.2*c*), compared with the remaining species (slope of width vs. depth = 1.34-1.76; Fig. 4.2*c*).

The mesh dimensions required to exclude carp ≥250 mm TL using grid mesh were 44 mm (each axis of the mesh, Tables 4.4 and 4.5, Figs 4.1 and 4.3*a*). The aperture between bars required to exclude carp ≥250 mm using 'jail bar' style mesh was 31.4 mm (Tables 4.4 and 4.5, Figs 4.1 and 4.3*b*).

4.4.4. Modelling the likely efficiency of optimised carp exclusion screens

Modelling was used to determine the percentage of each large-bodied fish species (in terms of abundance) captured during the wetland surveys which would have been excluded by both the optimised grid mesh and jail bar designs (with an exclusion threshold for carp of ≥250 mm TL) had they been in use at the time of sampling. Approximately 92% of carp and 65% of native golden perch would have been excluded by either optimised design (Table 4.6). Jail bars (with aperture between the bars of 31.4 mm) would have excluded only 2% (vs. ~7% by grid mesh) of native bony herring, but excluded more goldfish (10% jail bars vs. 6% grid mesh) and redfin perch (24% jail bars vs. 20% grid mesh) than a 44 x 44 mm grid mesh.

The smaller axis (34 mm, Table 4.2) of Alu-Tread determines its exclusion threshold, for all large-bodied species. As Alu-Tread excludes fishes in a similar manner to jail bars (body width determining exclusion threshold), the exclusion threshold, in terms of carp length (TL), is slightly longer than our two optimised designs (Table 4.7). The exclusion threshold for the other large-bodied species is within ~10% of our jail bar and grid mesh designs (Fig. 4.1 and Table 4.7).

Table 4.4. Summary of the models used to design conceptual 'grid mesh' and 'jail bar' carp screens based of fish morphology detailing regressions and data boundaries.

Species	Grid mesh	Jail bars	Bounds (mm)	
			minimum	maximum
Common carp	D = 13.29+0.223TL	W = -5.39+0.161TL	20	770
Bony herring	D = -4.11-0.283TL	W = -0.67+0.105TL	22	457
Golden perch	D = -11.73+0.330TL	W = -8.30+0.178TL	81	478
Goldfish	D = 1.16+0.301TL	W = -2.09+1.73TL	78	300
Redfin perch	D = -8.20+0.297TL	W = -8.86+0.185TL	32	358

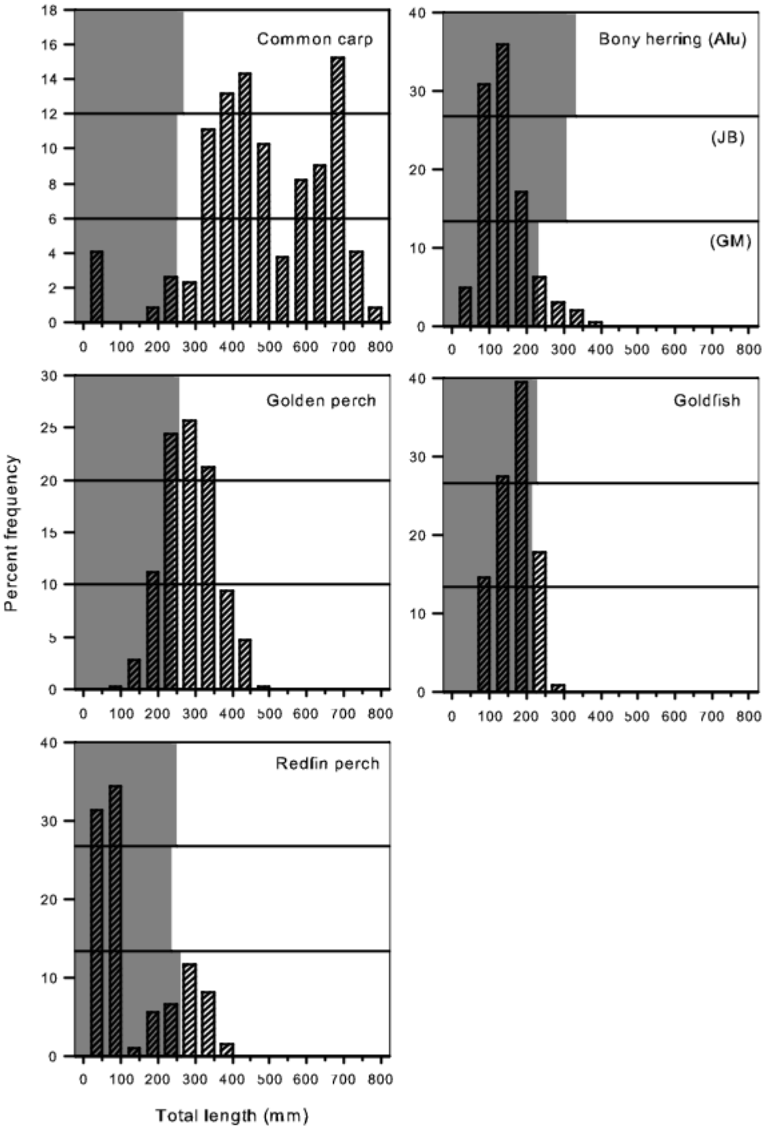

Figure 4.1. Size distribution of common carp (*n* = 342), bony herring (*n* = 5890), golden perch (*n* = 385), goldfish (*n* = 124) and redfin perch (*n* = 198) recorded in wetland inlets in the Lower River Murray, South Australia. Shaded area (labelled on bony herring only) shows sizes which would pass (Alu) Alu-Tread Series 13, (JB) 31.4-mm jail bars and (GM) 44-mm grid mesh, the latter two designed to exclude ≥250 mm TL carp. Note variable y-axis.

Table 4.5. Summary of the conceptual ‘grid mesh’ and ‘jail bars’ dimensions required to exclude carp of 100 – 500 mm (TL) with the 250 mm target size class highlighted. Also shown are the corresponding lengths of other species expected to be excluded by meshes of different dimensions. Dimensions above the 250 mm (TL) threshold are provided to illustrate the lengths at maturity of the other large-bodied species. Approximate lengths at maturity for all species are shown in bold (information from the following sources: bony herring (Puckridge and Walker 1990), golden perch (Mallen-Cooper and Stuart 2003), redfin perch (Morgan *et al.* 2002), goldfish (McDowall 1996; Lintermans 2007) and carp (Brown *et al.* 2005).

	Grid mesh						Jail bars					
To exclude carp of: total length (mm)	Depth (mm)	Aperture (x & y) mm required	Lengths of other species likely to be prevented passage: Golden perch	Bony herring	Goldfish	Redfin perch	Width (mm)	Aperture mm required	Lengths of other species likely to be prevented passage: Golden perch	Bony herring	Goldfish	Redfin perch
100	35.6	22.7	143	128	**114 (♂ & ♀)**	**148 (♂ & ♀)**	10.7	9.6	107	99	74	**106 (♂ & ♀)**
150	46.8	29.8	177	**163 (♂)**	152	185	18.8	16.9	152	**168 (♂)**	**120 (♂ & ♀)**	149
200	58.0	36.9	211	**199 (♀)**	189	223	26.8	24.1	198	**237 (♀)**	167	193
250	69.1	44.0	245	234	226	260	34.9	31.4	243	307	213	236
300 (♂)	80.3	51.1	279	270	263	298	42.9	38.6	288	376	260	280
350 (♀)	91.5	58.2	312	305	300	336	51.0	45.9	**334 (♂)**	445	306	323
400	102.6	65.3	**346 (♂)**	341	337	373	59.0	53.1	379	514	353	367
450	113.8	72.4	380	377	374	411	67.0	60.3	**424 (♀)**	584	399	410
500	125.0	79.5	**414 (♀)**	412	411	448	75.1	67.6	470	653	446	454

Table 4.6 Summary of the percentage of the total population of each large-bodied fish species captured at wetland inlets, which would have been excluded by the optimised ‘grid mesh’ and ‘jail bar’ designs, based on the size of fishes recorded in wetland inlets in the Lower River Murray, South Australia, during mid-August to mid-November 2006. Changes in exclusion efficiencies are modelled for each 50 mm size class of carp. The 250-mm carp size class is highlighted.

		Grid mesh				Jail bars			
Excluded carp threshold (TL mm)	% Carp population	% Bony herring population	% Golden perch population	% Goldfish population	% Redfin perch population	% Bony herring population	% Golden perch population	% Goldfish population	% Redfin perch population
50.0	95.9	70.8	99.7	100.0	34.3	99.9	100.0	100.0	35.9
100.0	95.9	43.0	97.9	76.6	33.8	65.8	99.7	100.0	34.3
150.0	95.9	23.2	91.7	58.1	30.3	21.5	96.6	70.2	33.3
200.0	95.0	11.8	82.9	22.6	25.8	6.4	86.8	46.8	30.3
250.0	92.4	6.7	64.9	6.5	20.2	2.1	65.2	10.5	23.7
300.0	90.1	4.2	46.0	0.8	10.1	0.1	39.0	0.8	15.7

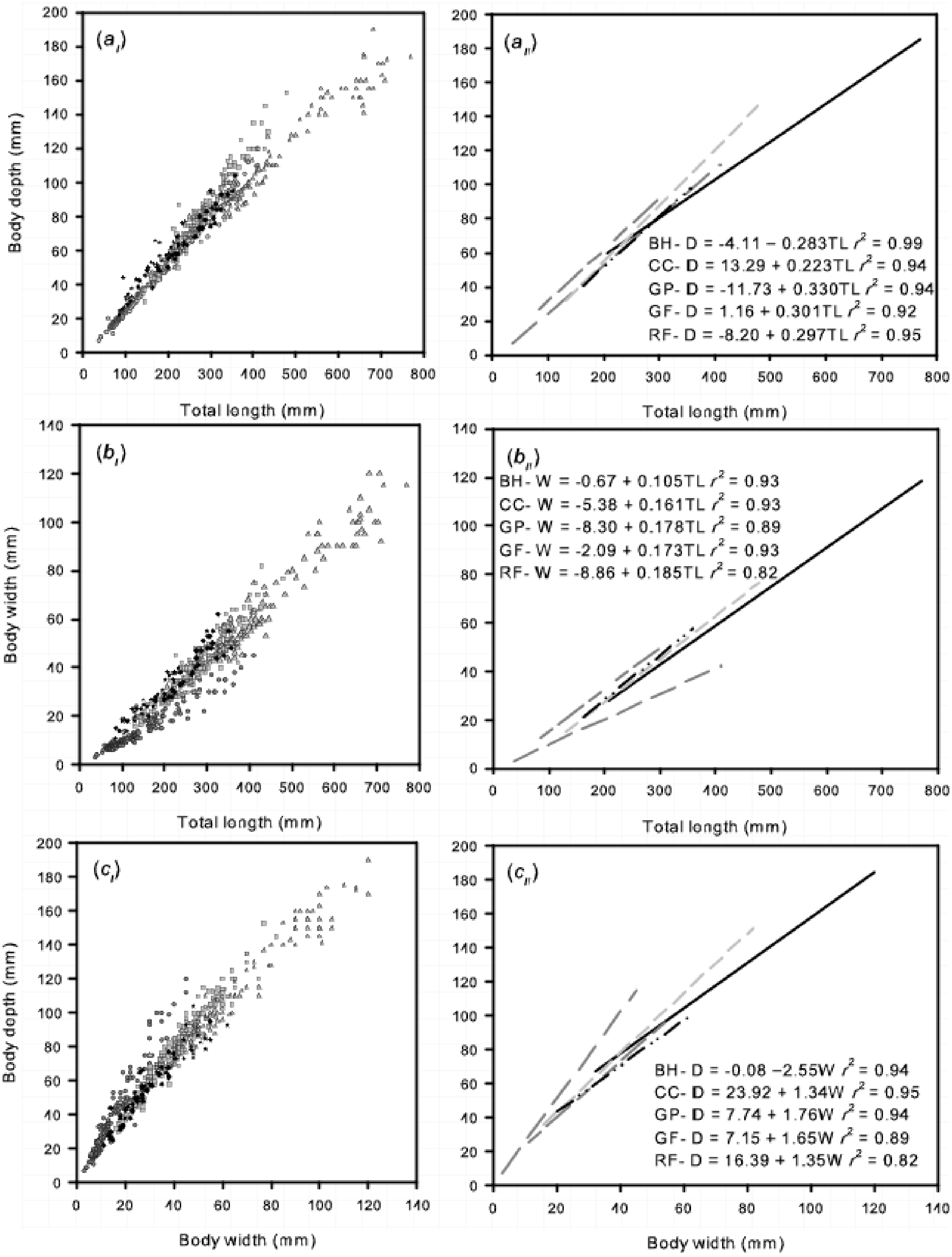

Figure 4.2. Relationship between (*a*) body depth and total length, (*b*) body width and total length and (*c*) body depth and body width, showing (*i*) scatter plot of raw data and (*ii*) fitted regressions for common carp (CC, *n* = 113, ▲, black solid line), golden perch (GP, *n* = 227, ■, light grey short dashed line), bony herring (BH, *n* = 184, ●, dark grey long dashed line), goldfish (GF, *n* = 45, ★, dark grey short-long-short dashed line) and redfin perch (RF, *n* = 30, ◆, black dash-dot-dot line) collected from six wetland inlets in the Lower River Murray, South Australia.

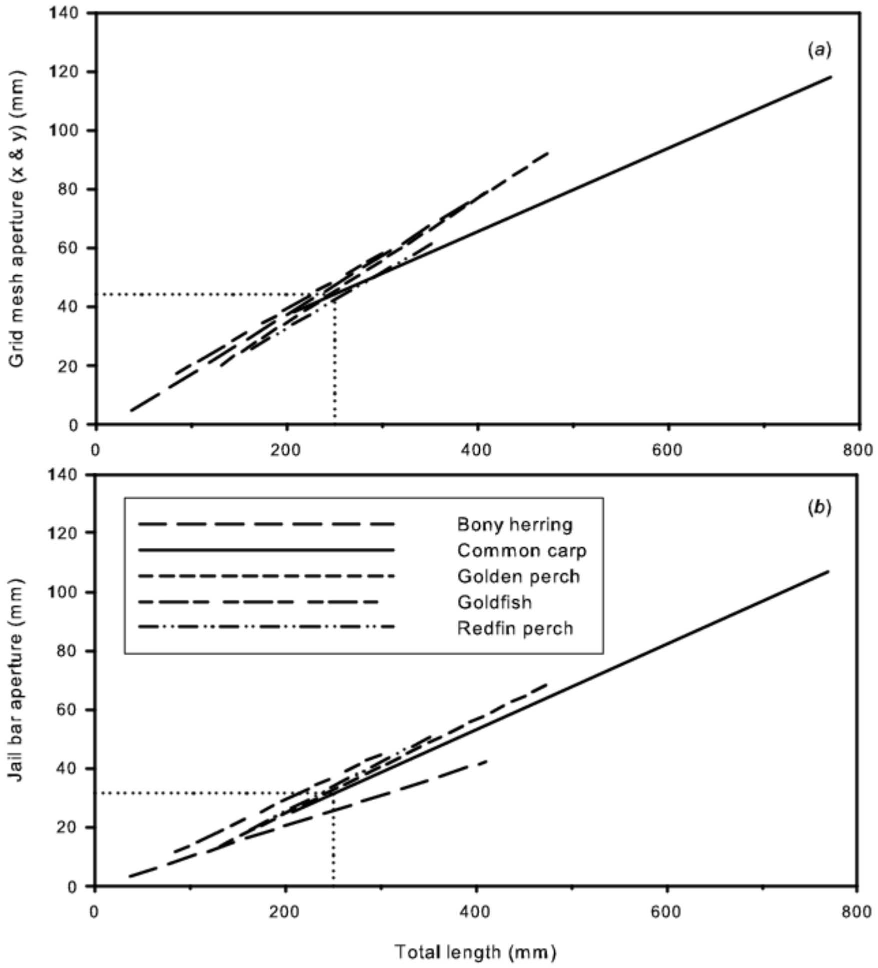

Figure 4.3. Modelled relationships between the aperture of optimised carp exclusion screen mesh, (*a*) vertical and horizontal grid mesh bars and (*b*) vertical jail bars and the total length of large-bodied species. Dotted lines indicate the 250 mm TL carp exclusion threshold.

Table 4.7. A summary of the size (total length, mm) and percentage of the total population of each large-bodied fish species captured during the wetland inlets survey, which would have been excluded by our proposed optimised grid mesh and jail bars designs (250 mm TL carp exclusion threshold) versus Alu-tread Series 13. The relative percentage of open aperture through which a fish could pass compared with the corresponding percentage of solid mesh of each screen design is also shown.

Species	Grid mesh		Jail bars		Alu-Tread Series 13	
	% excluded	TL	% excluded	TL	% excluded	TL
Common carp	92.40	250	92.40	250	91.81	268
Bony herring	6.71	234	2.07	307	0.83	332
Golden perch	64.94	245	65.19	243	57.40	259
Goldfish	6.45	226	10.48	213	6.45	230
Redfin perch	20.20	260	23.74	236	21.21	252
% open area	88		75		71	

4.5. DISCUSSION

4.5.1. Field survey of existing carp exclusion screens in the Murray-Darling Basin

Almost all CES were located in the inlets to wetlands adjacent to the River Murray main channel. While broadly distributed from the lower reaches in South Australia to the upper reaches on the New South Wales–Victoria border, the majority of screens were concentrated downstream of the confluence of the Murray and the Darling Rivers at Wentworth, NSW. This is potentially related to the high number of wetlands in the Lower River Murray that are permanently inundated by river regulation (Walker 2006), and are being actively managed to restore a ‘natural’ hydrological regime (Jensen 2002).

The current screen designs have sufficiently small apertures to restrict the movement of large carp, as well as the four other large-bodied species that we considered in this study. Based on a small scale survey of CES, Nichols and Gilligan (2003) found current mesh designs (Alu-Tread; jail bars, 10 mm aperture) successfully excluded large (≥300 mm TL) breeding carp. However, they also reported that carp occurred in similar abundance and biomass in adjacent wetlands with and without CES. Also, large carp (defined by Nichols and Gilligan (2003) as those >250-300 mm) were found in similar frequencies in wetlands fitted with CES and the adjacent river. Nichols and Gilligan (2003) suggest that either large carp were entering wetlands despite the use of CES or juvenile carp were passing the screen and growing to adult size before they could be killed by desiccation. Their findings indicate that despite having effective screens, wetland water level management was ineffective in controlling the carp population at their study sites through wetland drying (see also Smith *et al.* 2009b). Without regular manipulation of the wetland water level (e.g. drying every 1-2 years) to kill any small carp which enter, carp may become concentrated in the wetland by the CES, diminishing the potential benefits of carp exclusion.

Several flow control structures fitted with CES were of poor design or leaking, meaning that complete drying of the wetland was not possible; a partial dry with CES installed would concentrate large carp and intensify their ecological effects. Furthermore, a kangaroo was observed trapped in a dry regulator, highlighting the need to consider potential impacts on terrestrial species, particularly where a flow regulating structure may stand dry for a period of time. Thus, careful design and management of CES and the flow control structures to which they are fitted is needed to maximise the intended ecological outcomes.

4.5.2. Use of wetland inlets by native and alien fish

Thirteen native and five introduced fish species were captured in the inlets of six wetlands in the Lower River Murray. This diverse range of species includes most of the species (60%) known to occur within the region, and two (freshwater catfish and Murray cod) of conservation concern at either the state or national level. However, we acknowledge that our sampling was both spatially (in terms of the entire MDB) and temporally limited, and occurred during a period of low, stable within bank flows. This may have affected the composition and size of the fish assemblage sampled, especially in regard to juveniles. For example, if we had sampled in autumn we would have expected to detect YOY recruits of large-bodied species such as carp (Vilizzi 1998a). Each of the large-bodied species recorded could potentially be affected by the use of CES, however, our data suggest that only the largest golden perch (~250 mm TL) and turtles would be excluded by the optimised screens (see below). Importantly, the majority of fish that were sampled using wetland inlets in this study, and wetlands previously (Smith *et al.* 2009b) are small-bodied (<100 mm TL at maturity). Their body dimensions are well within the optimised mesh designs and should not be *physically* restricted by CES (Karl Hillyard, unpublished data). However, CES may affect small-bodied fish behaviour, but this requires further investigation.

Apart from carp, large-bodied species that are likely to be physically restricted by CES, were dominated primarily by bony herring and to a lesser extent by golden perch. Bony herring are an integral part of aquatic food webs (Sternberg *et al.* 2008) and are similarly abundant in both off-channel habitats (Balcombe *et al.* 2006; Smith *et al.* 2009b) and the main river channel (Baumgartner *et al.* 2008; Davies *et al.* 2008). The numerical dominance of bony herring in the MDB riverine environment, coupled with records of spawning in inundated backwaters (Pusey *et al.* 2004), suggests that care must be taken with the design and application of CES to minimise exclusion of this species from inundated wetlands which may be used as a spawning habitat. Fortunately, our modelling suggests that only ~2-6% of bony herring would be excluded by our optimised designs. Slightly more golden perch were caught than carp within the six wetland inlets. The similar abundance of these two species contrasts with the greater catch of carp in recent fish surveys in the same reach of the main Murray channel (Davies *et al.* 2008). Carp are also known to be far more abundant (almost 40x) than golden perch in wetlands of the Lower Murray (Smith *et al.* 2009b). The low catch of carp is speculated to relate to the comparatively low flow conditions at the time of sampling. However, the presence of golden perch in wetland inlets highlights the potential impacts of CES on the lateral movements of this species. We recommend that site-specific monitoring occur wherever CES are being considered for installation to ensure that the use of CES does not conflict with other wetland management objectives.

Other fauna sampled using wetland inlets, which may also be impacted by CES, included three species of turtle and three species of decapod crustaceans. Of particular note is that broad-shelled and Murray turtles have conservation status and are not well adapted for movement over land, which would be needed to bypass the screens, potentially exposing them to predation by foxes (Cann 1998; Spencer and Thompson 2005).

4.5.3. Conceptual design of optimised carp exclusion screens

The five common large-bodied fishes using wetland inlets had similar body dimensions, except bony herring, which is laterally compressed. Hence, whilst we recommend a jail bar CES design to allow the passage of most bony herring, we acknowledge that the larger individuals of other fishes with similar body dimensions to carp (such as golden perch) will also be excluded by this design. Besides carp, golden perch are the only other commonly recorded large-bodied species with a breeding size greater than what would be excluded by our optimised meshes; 325 and 397 mm TL for males and females respectively (Mallen-Cooper and Stuart 2003). However, little evidence exists suggesting that golden perch spawn on the floodplain and would be impeded in their reproductive movements by CES. Drifting eggs have been sampled in the main river channel (Stuart and Jones 2006a) but larvae are rarely collected (Humphries and King 2004). Juvenile golden perch use inundated wetlands (Pusey *et al.* 2004) but are unlikely to be *physically* restricted in their movement by our CES designs.

Several other large-bodied native species are found in the Lower Murray with a maximum adult size close to that of carp (aside from bony herring and golden perch) but were not collected in our study. These include Murray cod, silver perch *Bidyanus bidyanus* and freshwater catfish. However, radio tracking has shown that Murray cod are principally found in the main river channel, rather than off-channel habitats such as wetlands (Jones and Stuart 2007) and are unlikely to be adversely affected by CES. Despite this, we sampled a juvenile Murray cod in a wetland inlet and Murray cod have also been recorded stranded in off-channel habitat behind a flow regulating structure elsewhere in the MDB (Jones and Stuart 2008b). Silver perch, not recorded in wetland inlets in this study, are known to utilise inundated floodplain habitats (Jones and Stuart 2008b; Smith *et al.* 2009b). Spawning of silver perch is believed to occur in the main channel (King *et al.* 2009) and they are not thought to recruit on the floodplain (King *et al.* 2003), therefore it is unlikely that CES would interfere with their reproduction. A single freshwater catfish was recorded using a wetland inlet in this study, and they are reported in inundated floodplain habitats elsewhere in the Lower River Murray (Smith *et al.* 2009b). Freshwater catfish have similar movement patterns (Reynolds 1983) and occupy a similar niche (Clunie and Koehn 2001) to carp but their use of wetland habitat for breeding is unclear. Regardless of their need for wetland access, with lengths at maturity of ~250-300 mm TL (silver perch; Mallen-Cooper and

Stuart 2003) and ~300-400 mm TL (freshwater catfish; Pusey *et al.* 2004), CES could restrict the movements of mature individuals, though analysis of their morphology is required to confirm this. Similar modelling of body dimensions versus CES dimensions of other large-bodied species found elsewhere in the MDB, or not recorded in this study (e.g. trout cod, *Maccullochella macquariensis*) will identify the extent our optimised CES designs may affect their movements.

Appropriate CES design and application at a wetland site needs to reflect a range of factors potentially unique to each wetland such as the presence of threatened species, site access, inlet morphology and intended management objectives. If carp management is a desired objective, our optimised designs targeting ≥250 mm TL carp will prevent entry of sexually mature carp.

4.5.4. Modelling the likely efficiency of optimised carp exclusion screens

Modelling suggested that our optimised meshes, designed to exclude carp ≥250 mm TL would have been successful in excluding a large proportion (~92%) of carp, based on the sizes recorded in wetland inlets in spring 2006. Furthermore, ~93% (grid mesh) or ~97% (jail bars) of native bony herring, which make up the majority (96%) of large-bodied fish using wetland inlets, could have passed the optimised designs. However, most (~65%) native golden perch and a few redfin perch (~20%) and goldfish (~10%) would have been excluded. These exclusionary efficiency models are limited by the fish sizes used to generate them. Young-of-year recruits were virtually absent in our sampling. Had we sampled more YOY carp, the modelled exclusionary efficiencies would likely be reduced. Increasing the carp exclusion threshold from 250 mm to 300 mm (TL) would reduce the proportion of golden perch excluded from ~65% to ~45% using grid mesh or ~40% using jail bars, but only allow an extra 2 - 3% of (potentially sexually mature) carp passage. This may be preferential if golden perch are common in a wetland designated to receive a CES.

Modelling suggests that our optimised designs appear to have similar exclusion efficiency to Alu-Tread, which is currently used on 43% of CES in the MDB. However, Alu-Tread (carp exclusion threshold 268 mm TL) is likely to allow the passage of sexually mature carp ≥250 mm TL into wetlands. Similarly, a higher percentage of the other four large-bodied species could pass this mesh than our optimised designs. Alu-Tread has fewer apertures for fish to pass through than either optimised design (Table 4.7) and has a rougher surface, factors which may cause injury to fish given the greater chance of contacting the mesh. Similarly, less open area may increase the chance of screens clogging with debris. Preliminary flume and field testing of both our optimised designs and those currently in use (e.g. Alu-Tread) has shown some small-bodied fishes, such as Australian smelt, are less likely to pass Alu-Tread than the optimised designs. Hence, we believe that while Alu-Tread has similar exclusion efficiency to the optimised designs, it may be more likely to adversely impact the movement of small-bodied native fishes which use wetland inlets in large numbers.

In conclusion, a diverse range of species use wetland inlets and these species span a wide size range. Our two optimised designs (jail bars and grid mesh) may prevent the passage of carp ≥250 mm TL, while allowing the passage of small-bodied native fishes (which comprise the majority of fish in wetlands), the juveniles of large-bodied native fishes (e.g. golden perch) and >95% of the most abundant large-bodied native fish in wetlands, bony herring. The key drawback is that large-bodied native fauna with body dimensions similar to carp (e.g. sexually mature golden perch and turtles) will be excluded. Hence, careful evaluation of a wetland's management objectives, including carp management *via* the use of CES, must be made in the context of potential effects on the local fish assemblage and other site-specific factors. If large-bodied native fish passage is critical, then wetland carp separation cages currently in development, which incorporate novel pushing (Thwaites *et al.* 2010) and jumping (Stuart *et al.* 2006a) trap components, could be considered.

This study provides a basis for future testing and validation of CES, with the intention of coordinating and improving wetland management and rehabilitation throughout the MDB. CES may aid wetland rehabilitation along with careful management at an appropriate site, and unlike other carp control techniques, this technology is relatively inexpensive and requires little further development. The removal of existing CES at wetland sites that cannot be dried regularly to kill carp by desiccation should be considered.

5. Chapter Five - TESTING EXCLUSION SCREENS TO MANAGE THE ALIEN INVASIVE COMMON CARP, *CYPRINUS CARPIO* L.

KARL A. HILLYARD[a], BENJAMIN B. SMITH[b] AND BRONWYN M. GILLANDERS[a]

[a] *Southern Seas Ecology Laboratories, School of Earth and Environmental Sciences, Darling Building, DX 650 418, University of Adelaide, SA 5005, Australia.*

[b] *Inland Waters and Catchment Ecology Program, SARDI Aquatic Sciences, PO Box 120, Henley Beach, SA 5022, Australia.*

Text in manuscript

Statement of Authorship

In this chapter, Karl Hillyard developed the study, collected and analysed data, and drafted the manuscript. Bronwyn Gillanders and Ben Smith advised on development of the study, data analysis and contributed to the synthesis and finalisation of the manuscript.

Certification that the statement of contribution is accurate.

Karl A. Hillyard (Candidate)

Signed *Date* 11/5/2011

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

Bronwyn M. Gillanders

Signed *Date* 11/5/2011

Benjamin B. Smith

Signed *Date* 11/5/2011

5.1. ABSTRACT

The rehabilitation of regulated off-channel wetlands has relied heavily on flow control structures to reinstate natural patterns of connectivity and carp exclusion screens (‘carp screens’) to manage abundant alien common carp, *Cyprinus carpio*. The ability of current carp screen designs to manage adult carp, without detrimental effects to native fauna, has been questioned. We evaluated the passage of carp and other aquatic fauna through three screen designs, at 12 wetlands spanning three bioregions of the River Murray in South Australia. Two new untested designs (grid mesh with 45 mm x 45 mm internal dimensions and ‘jail bars’ with 31.4 mm gaps (actual mean gap = 31.3 mm ± 0.34 s.d.)) allowed the passage of more fish (including all small and medium-sized native species <200 mm total length), and larger fish, than the most common current design (Alu-Tread walkway mesh). Adult common carp and turtles passed none of the screens. Further testing of the favoured ‘31 mm jail bars’ carp screen design is recommended at wetland flow control structures and during higher flow conditions.

5.2. INTRODUCTION

The common carp, *Cyprinus carpio* (hereafter ‘carp’), originally from Asia, now occurs in fresh waters on every continent except Antarctica. It is established in 91 of the 121 countries where it has been introduced (Casal 2006), and it is a recognised invasive species in Europe (Copp *et al.* 2005), North and South America (Zambrano *et al.* 2006), Africa (Moreau and Costa-Pierce 1997) and Australasia (Koehn 2004; Rowe 2007). In many regions, the species is considered naturalised (Lever 1994).

Carp have been termed ‘ecosystem engineers’ (Matsuzaki *et al.* 2009), reflecting their capacity to modify their habitat. They have been implicated in causing increased turbidity (Fletcher *et al.* 1985; Breukelaar *et al.* 1994; Lougheed *et al.* 1998), the destruction of some delicate, shallow-rooted macrophytes (Fletcher *et al.* 1985; Roberts *et al.* 1995; Sidorkewicj *et al.* 1998), declines of benthic macroinvertebrates (Hinojosa-Garro and Zambrano 2004; Bartsch *et al.* 2005) and increases in phytoplankton (Gehrke and Harris 1994). Impacts are greatest in shallow offstream wetlands where carp congregate to feed and spawn (Stuart and Jones 2006a). In lowland rivers, carp move between the main channel and adjacent wetlands (Schultz *et al.* 2007; Jones and Stuart 2008a; Conallin *et al.* 2010), providing opportunities for exclusion or capture in the connecting channels using carp exclusion screens (hereafter ‘carp screens’) (French *et al.* 1999; Hillyard *et al.* 2010), jumping traps (Stuart *et al.* 2006a), pushing traps (Thwaites *et al.* 2010), or combination jumping and pushing traps (Smith *et al.* 2009a). Carp screens have only been deployed in North America and Australia (Jensen 2002; Lougheed *et al.* 2004), and jumping and pushing traps in Australia.

Carp screens are metal screens that are typically attached to flow control structures (e.g. box or pipe culverts) in the inlets to off-channel wetlands. The aim of carp screens is to prevent the passage of adult carp (Chapter 4; Hillyard *et al.* 2010), but small carp may pass (Navarro and Johnson 1992) and grow to adults within the wetland where they become trapped, highlighting a need for regular wetland drying (Smith *et al*. 2009a). Where a wetland is perennially inundated due to river regulation, carp screens may still compliment wetland remediation projects by managing carp, if the wetland can be disconnected from the river and dried *via* the use of stop-logs or sluice-gates (Jensen 2002; Hillyard *et al.* 2010).

Contact with screens has been reported to kill or injure small fish (Swanson *et al.* 2005), and carp screens are a barrier to all fauna of a comparable size to the carp they exclude. Rationale supporting the use, design and management of carp screens has been limited (cf. French *et al.* 1999; Hillyard *et al.* 2010). Empirical evidence of habitat improvements (cf. Recknagel *et al.* 1998; Nichols and Gilligan 2003; Lougheed *et al.* 2004) or a reduction in carp abundance (Smith *et al.* 2009b) resulting from the use of carp screens is also scant. However, localised improvements in water quality and wetland condition have been associated with reduced carp biomass in wetland trials not involving carp screen use (King *et al.* 1997; Robertson *et al.* 1997).

In the Murray-Darling Basin (MDB), south-eastern Australia, an exhaustive survey of river stakeholders followed by a field reconnaissance identified 54 existing carp screens in current use of varying material, shapes and dimensions (Chapter 4; Hillyard *et al.* 2010). The first installations occurred at least 17 years ago (Nichols and Gilligan 2003), but all carp screens have been constructed and operated with little scientific input. This paper describes the results of field trials designed to evaluate the passage of carp, native fishes and turtles through the most common existing design in the Murray-Darling Basin (Alu-Tread walkway mesh (Locker Group, Melbourne, Australia), 44 % of deployed carp screens, Table 5.1) and two untested ‘optimised’ designs proposed by Hillyard *et al.* (2010), including a square grid mesh (GM) with 45 mm x 45 mm internal dimensions and vertical ‘jail bars’ with 31.4 mm gaps (actual mean gap = 31.3 mm ±0.34 s.d.) between bars (Appendix 2). The latter screen meshes were designed to prevent the passage of carp ≥250 mm total length (TL, mm) (≈ 225 mm fork length: Karl Hillyard, unpublished data), whilst maximising the passage of small (<100 mm TL at maturity) and medium sized (mature TL 100-200 mm) native fishes (which comprise some 80 % of the abundance of fish in wetlands along the South Australian Murray: Smith *et al.* 2009b), as well as bony herring, *Nematalosa erebi* (a large-bodied, laterally compressed native fish that comprises 12% of the individuals and 42 % of the biomass of fish recorded moving between the main river channel and off-channel wetlands during stable river levels and low flows: Chapter 2; Conallin *et al.* 2010).

5.3. METHODS

5.3.1. Study locations

This study occurred along the River Murray in South Australia (Fig. 5.1) where carp screens are most abundant within the Murray-Darling Basin (Chapter 4; Hillyard *et al.* 2010). As fish communities differ across River Murray wetlands (Chapter 2; Conallin *et al.* 2010) and among hydro-geographical regions (Smith *et al.* 2009b), carp screens were trialled in 12 wetlands spanning the 'Lower Lakes' (barrages to Wellington), 'Lower Swamps' (Wellington to Mannum) and 'Riverland' (Overland Corner to border) hydro-geographical regions (Fig. 5.1). Locations were selected according to their known fish communities (e.g. high abundance of carp, or presence of a diversity of native and/or threatened species) and ease of access.

The River Murray in South Australia experiences a semi-arid climate, where average annual evaporation far exceeds rainfall (Walker 1992). River flow predominantly originates from diversions from upstream storages, and is further regulated in South Australia by six serial weirs, riverbank levees, offstream regulators and five barrages at the Murray mouth (Fig. 5.1). Regulation of the Murray has reduced the frequency of low flows (100-300 GL per month) and annual and medium interval floods (1-20 year), but the frequency of medium flows (500-1500 GL per month) has increased and large floods (>20 year interval) remain unchanged (Walker and Thoms 1993; Maheshwari *et al.* 1995). Regulation has also perennially inundated around 70% of the offstream wetland area in the Lower Murray (Pressey 1986).

5.3.2. Field trials

Trials of the three carp screen meshes, fitted in the mouths of fyke nets, were conducted at twelve wetlands over four consecutive weeks from the 13 October until the 7 November 2008. Originally, the carp screen meshes were to be fitted to flow control structures (e.g. culverts: Schultz *et al.* 2007), rather than fyke nets, but ongoing drought conditions (Bond *et al.* 2008; Murphy and Timbal 2008) forced the 'closure' of all managed wetlands with flow control structures in South Australia, to minimise evaporative losses from the wetlands (Smith *et al.* 2009b).

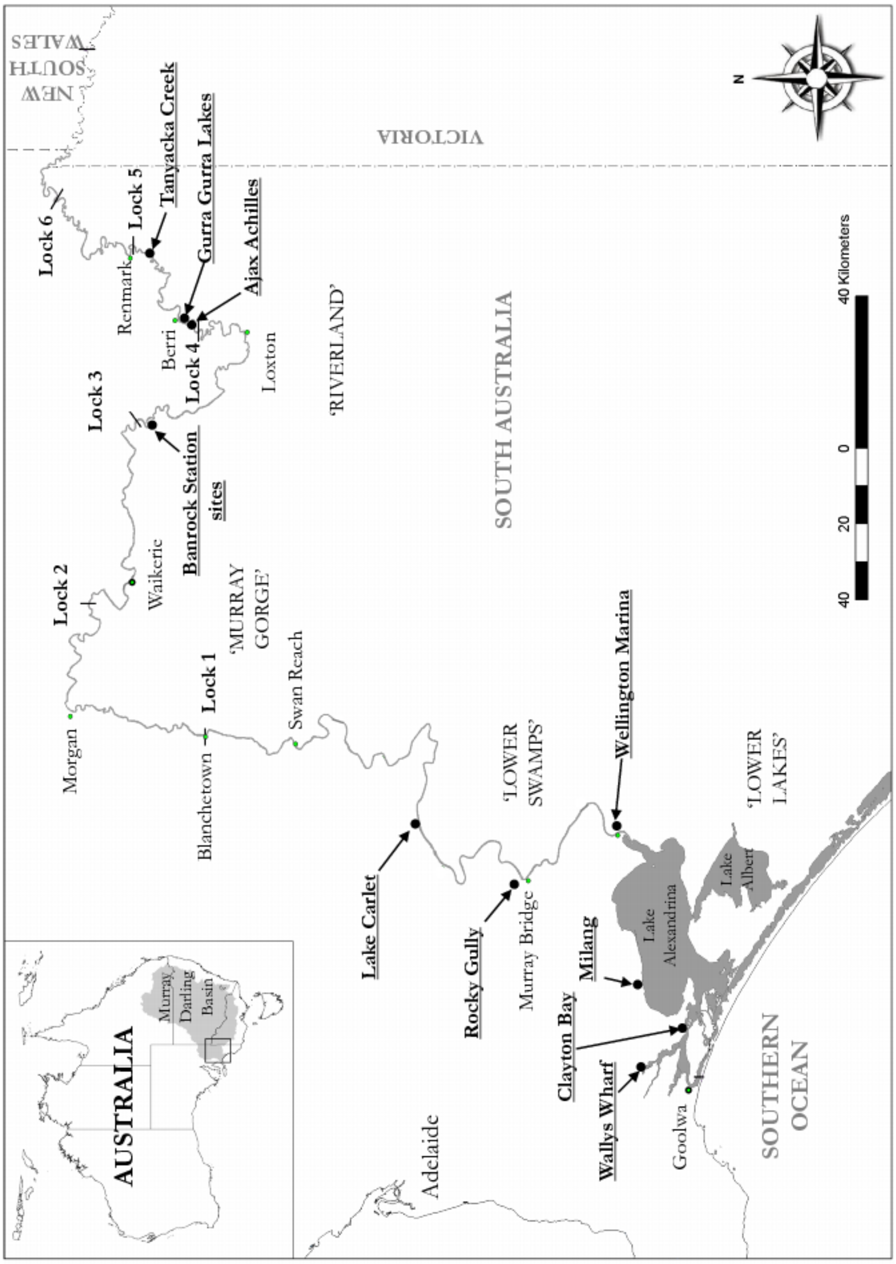

Figure 5.1. Map showing the location of wetlands (names underlined) where carp screens were field tested (note that trials were conducted at three sites at Banrock Station).

Three locations were sampled for four nights each week. At each location, a set of four fyke nets were used. Within each set, three nets were fitted with one of each of the three trial (treatment) screens (Appendix 2) and the fourth (control) net had no mesh fitted. Nets were randomly positioned within locations and positions were rotated across the four net nights to reduce any localised net effect, e.g. greater catch in nets closer to wetland inlets.

Fyke nets were double-winged (directional) with an 80 cm square mouth to which the trial screens were fitted. Each wing was 3 m long and dropped 80 cm. The body of the net was 4 m long, with a funnel immediately behind the mouth and two more funnels along the body of the net to prevent escape. Wings and the mouth of each net were fitted with a buoyant float line and a heavily weighted lead line to ensure the entire water column was sampled. Nets were set for approximately 24 hours.

For each net, the length (TL, mm) and weight (g) of up to 30 individuals of each species were measured and weighed, and an average individual weight calculated. The remainder of each species was then bulk-weighed to determine its biomass. Total abundance was estimated by dividing the species biomass by the average individual species weight. All data were subsequently standardised to a 24 hour Catch Per Unit Effort (CPUE).

5.3.3. Morphologic regressions and theoretical exclusion thresholds of small and medium-bodied species

Between five and 143 individuals of each fish species caught in the field trials were measured (TL, width and depth; to the nearest mm) to construct length-width, length-depth and width-depth regressions to calculate species-specific exclusion thresholds for each of the three mesh designs. Reported maximum lengths from the literature (Lintermans 2007; Gomon *et al.* 2008) were used to derive theoretical maximum widths and depths of small and medium-bodied species, which were subsequently compared with the maximum dimensions of each mesh design (Table 5.1).

5.3.4. Data analysis

Two-way ANOVAs, using screen type and location as factors, were used to analyse the catch of fish (abundance and biomass) and decapod crustaceans (biomass only). All data were $\log_{10}(x+1)$ transformed and checked for normality (graphical inspection) and heterogeneity of variances (Levene’s test). Analyses were still performed if variances remained heterogeneous after transformation, as ANOVA is a robust method where data, as in this case, are balanced and sample sizes are relatively large (Underwood 1997). Student-Newman-Keuls (SNK) *post hoc* tests were conducted where appropriate. Kruskal-Wallis tests followed by multiple Bonferroni corrected Mann-Whitney tests were used to investigate differences in fish lengths between the field tested carp screen designs and control nets.

Table 5.1. Dimensions, shape and theoretical carp exclusion thresholds (derived from Chapter 4; Hillyard *et al.* 2010) of the three carp screens mesh designs tested.

Name	Reason for testing	Effective aperture shape	Material	Maximum horizontal aperture (mm)	Maximum vertical aperture (mm)	Aperture area (mm^2)	Theoretical carp exclusion threshold (TL mm)	Morphologic axis carp excluded by
Jail bars	Optimised design	Vertically elongated rectangles	Aluminium	31.4	800	25120	250.00	Width
Grid mesh	Optimised design	Square	Stainless Steel	45	45	2309	250.00	Depth
Alu-Tread Series 13	Field survey	Horizontally compressed hexagon	Aluminium	97	34	2309	268.26	Width

Differences in the fish assemblage caught in the fyke nets were further compared using two-way PERmutational Multivariate ANalysis Of VAriance (PERMANOVA) (PRIMER, version 6: Plymouth Marine Laboratories, UK), using screen type and location as factors. The data were fourth-root transformed and converted to a Bray-Curtis distance matrix, and the test statistic (essentially an F ratio) was calculated as the ratio of the sum of squared distances among groups to the sum of squared distances within groups.

5.4. RESULTS

5.4.1 Catch summary

Some 153 000 fish representing 15 native and four alien species, as well as 88 turtles from three species were caught (Table 5.2). Turtles (109-365 mm carapace length) were exclusively caught in control nets suggesting that all screens prevented their passage. Numerous decapod crustaceans (freshwater shrimps, *Paratya australiensis;* freshwater prawns, *Macrobrachium australiense*) were also caught but not enumerated, although biomass was recorded on 12 of the 16 net-nights.

Fish abundance and biomass (ANOVA) and fish assemblage (PERMANOVA) differed significantly across screen type and location, though there was no interaction (Table 5.3). *Post hoc* SNK tests showed jail bars (mean = 1099 ± 245 S.E.) and 45 mm grid mesh (mean = 1017 ± 212 S.E.) nets caught more fish than the Alu-Tread (mean = 633 ± 126 S.E.) or the control nets (mean = 803 ± 169 S.E.). In slight contrast, the biomass of fishes caught was greater in jail bars (mean = 585 g ± 98 S.E.), 45 mm grid mesh (mean = 569 g ± 109 S.E.) and control (mean = 701 g ± 140 S.E.) nets than Alu-Tread (mean = 289 g ± 47 S.E.).

Decapod crustaceans were caught at 10 of the 12 locations (mean = 28.37 g ± SE 3.39). The biomass of decapod crustaceans varied between locations, but there was no significant difference among screen types (Table 5.3).

Table 5.2. Abundance and dimensions of fish and turtles (carapace length, mm) caught in trial and control fyke nets across the 12 study wetlands in South Australia.

Family	Common name	Species	*n*	Mean TL (mm)	SE	TL range (mm)
Clupeidae	Bony herring	*Nematalosa erebi* (Günther, 1868)	689	96.51	1.52	30-465
Galaxiidae	Common galaxias	*Galaxias maculatus* (Jenyns, 1842)	1113	57.78	0.63	26-141
Retropinnidae	Australian smelt	*Retropinna semoni* (Weber, 1895)	22023	50.56	0.05	25-73
Atherinidae	Small-mouthed hardyhead	*Atherinosoma microstoma* (Günther, 1861)	34835	50.61	0.07	25-88
	Murray hardyhead	*Craterocephalus fluviatilis* McCulloch, 1913	3392	48.23	0.10	30-71
	Un-specked hardyhead	*Craterocephalus stercusmuscarum fulvus* Ivantsoff, Crowley & Allen, 1987	10766	32.88	0.08	20-72
Melanotaeniidae	Murray-Darling rainbowfish	*Melanotaenia fluviatilis* (Castelnau, 1878)	139	52.79	0.83	24-79
Hemiramphidae	River gar	*Hyporhamphus regularis* (Günther, 1866)	10	134.70	4.75	106-162
Percichthyidae	Murray-Darling golden perch	*Macquaria ambigua ambigua* (Richardson, 1845)	5	237.20	49.06	90-344
Eleotridae	Carp gudgeon complex	*Hypseleotris* spp.	72295	31.46	0.02	19-56
	Flat-headed gudgeon	*Philypnodon grandiceps* (Krefft, 1864)	3513	49.35	0.18	21-99
	Dwarf flat-headed gudgeon	*Philypnodon macrostomus* Hoese & Reader 2006	233	36.69	0.31	25-52
Gobiidae	Western blue-spot goby	*Pseudogobius olorum* (Sauvage, 1880)	1447	47.14	0.16	28-62
	Lagoon goby	*Tasmanogobius lasti* Hoese, 1991	806	44.41	0.24	20-70
	Tamar goby	*Afurcagobius tamarensis* Johnston, 1883	120	58.01	1.51	31-93
Cyprinidae	Common carp	*Cyprinus carpio* Linnaeus, 1758	420	46.67	2.86	14-645
	Goldfish	*Carassius auratus* Linnaeus, 1758	48	97.83	4.97	47-200
Poecilidae	Eastern gambusia	*Gambusia holbrooki* (Girard, 1859)	1552	32.36	0.14	21-53
Percidae	Redfin perch	*Perca fluviatilis* Linnaeus, 1758	173	65.10	2.35	21-156
Chelidae	Broad-shelled turtle	*Chelodina expansa* Gray, 1885	1	365.00	-	-
	Long-necked turtle	*Chelodina longicollis* (Shaw, 1794)	80	205.51	2.72	149-294
	Murray turtle	*Emydura macquarii* (Gray, 1830)	7	244.86	27.90	109-339

Table 5.3. Summaries of (a) two-way ANOVA testing differences in fish abundance (all $\log_{10}X+1$ CPUE) and biomass, and decapod crustacean biomass, and (b) two-way PERMANOVA testing difference in fish assemblages among trial carp exclusion screens and locations.

Variable	Effect	d.f.	MS	(a) *F*, (b) Pseudo-*F*	*p*
(a)					
Fish abundance	Screen type	3	0.792	11.380	**<0.001**
	Location	11	3.495	50.190	**<0.001**
	Screen type x Location	33	0.070	0.542	0.979
	Error	144	0.128		
Fish biomass	Screen type	3	1.102	9.682	**<0.001**
	Location	11	2.037	17.896	**<0.001**
	Screen type x Location	33	0.114	0.826	0.735
	Error	144	0.138		
Decapod biomass	Screen type	3	341.544	1.572	0.222
	Location	8	19993.108	92.021	**<0.001**
	Screen type x Location	24	217.267	0.331	0.999
	Error	108	657.159		
(b)					
Fish abundance	Screen type	3	1392	5.1435	**0.001**
	Location	11	27597	70.831	**0.001**
	Screen type x Location	33	270.64	0.69462	0.99
	Residual	144	389.62		

5.4.2. Lengths of abundant fish species

The lengths of abundant fish species (small-bodied species: un-specked hardyhead, *Craterocephalus stercusmuscarum fulvus*; carp gudgeons *Hypseleotris* spp. (an unresolved species complex: Bertozzi *et al.* 2000) and Australian smelt *Retropinna semoni*; medium-bodied species: flat-headed gudgeon, *Philypnodon grandiceps*; and large-bodied species: bony herring and carp), either caught in high numbers during this trial (>70 % of total catch) or during a recent survey of lateral movements of fishes in wetland inlets (Chapter 2; Conallin *et al.* 2010), were compared between the three carp screen designs and the unscreened control nets. The four small and medium-bodied species showed little difference in length among screen types (Fig. 5.2), whereas a Kruskal-Wallis test showed significant differences ($H_3 = 24.296$, $p < 0.001$) in the lengths of large-bodied bony herring between the control and treatment nets. Multiple Bonferroni corrected Mann-Whitney (level of significance = 0.0083) tests showed bony herring caught in the control (mean TL = 104 mm ± 50 s.d.) and jail bars (mean TL = 102 mm ± 39 s.d.) nets were larger than those caught in the Alu-Tread (mean TL = 81 mm ± 21 s.d.) and the 45 mm grid mesh (mean TL = 88 mm ± 24 s.d.). Large carp (>100 mm TL) were only caught in the control net, but no statistically significant difference between screen types was found ($H_3 = 1.333$, $p = 0.721$, Fig. 5.2).

5.4.3. Morphologic regressions and theoretical exclusion thresholds of small and medium-bodied species

Using regressions of length-width and length-depth (Table 5.4), and published maximum lengths (Lintermans 2007; Gomon *et al.* 2008), maximum body width and depth of the 19 fish species collected in the field ($n = 5$ to 143 depending on species) were calculated. For the 13 species of small and medium-bodied fish (i.e. maximum length < 200 mm TL) the maximum width and depth were 18 mm and 21 mm, which is much smaller than the smallest carp screen mesh tested (Table 5.5). Of the six large-bodied fish species (max length > 200 mm TL), the elongate river gar (maximum fork length 350 mm: Gomon *et al.* 2008) was the only species that would not be physically encumbered by the carp screens tested. The remaining five large-bodied fish species all had maximum widths and depths greater than all three carp screen meshes (Table 5.5).

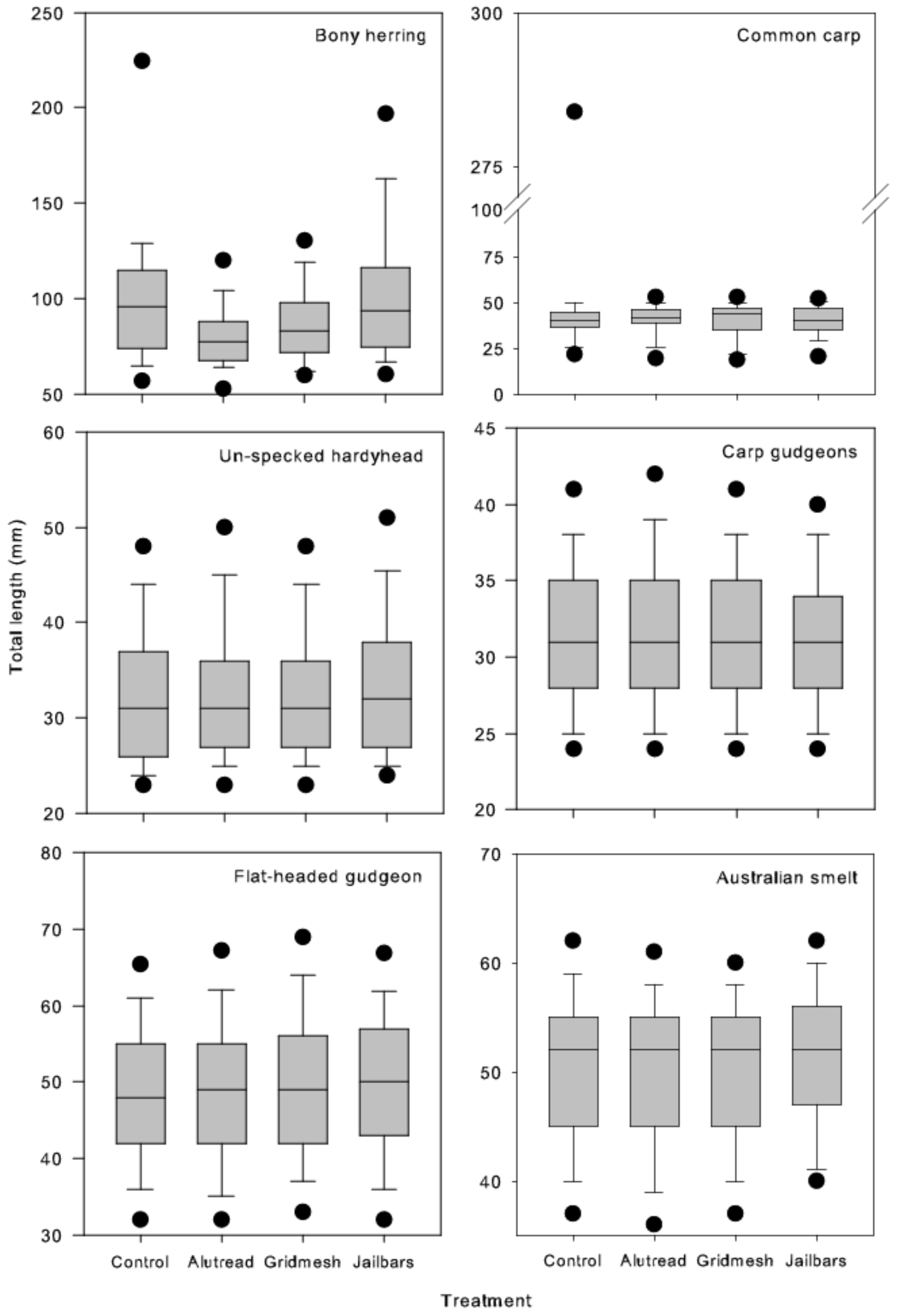

Figure 5.2. Box-plots showing the difference between control and treatment nets of the median (solid line) total length (mm) of six key fish species sampled in Carp Exclusion Screen trials at wetlands in the Lower River Murray, South Australia. Dots indicate the 5th and 95th percentiles, whiskers the 10th and 90th percentile and the box shows the 25th to 75th percentile range. Note varying y-axis scale.

Table 5.4. Morphometric relationships of fish caught during carp exclusion screen trials in 12 wetlands in South Australia in spring 2008, shown are the slope, y-intercept (Y int) and R^2. L-W: Length-Width, L-D: Length-Depth and W-D: Width-Depth.

Species		L-W			L-D			W-D		
	n	Slope	Y int	R^2	Slope	Y int	R^2	Slope	Y int	R^2
Native species										
Bony herring	72	0.095	-0.139	0.908	0.245	-1.003	0.954	2.431	1.074	0.926
Common galaxias	143	0.077	0.125	0.867	0.102	-1.024	0.844	1.237	-0.722	0.850
Australian smelt	117	0.066	0.223	0.405	0.094	1.756	0.376	0.737	3.899	0.254
Small-mouthed hardyhead	70	0.104	-0.686	0.767	0.138	-0.673	0.846	1.102	1.335	0.765
Murray hardyhead	51	0.138	-2.022	0.728	0.239	-4.591	0.797	1.441	0.402	0.751
Un-specked hardyhead	85	0.100	-0.153	0.733	0.154	-0.599	0.896	1.221	0.702	0.776
Murray-Darling rainbowfish	48	0.098	-0.211	0.800	0.266	-2.612	0.815	2.206	0.444	0.676
River gar	10	0.029	4.108	0.184	0.088	-0.034	0.720	0.952	2.952	0.381
Murray-Darling golden perch	5	0.097	9.792	0.832	0.259	8.041	0.988	2.297	-5.941	0.881
Carp gudgeon complex	68	0.098	-0.002	0.577	0.208	-1.551	0.758	1.418	0.720	0.585
Flat-headed gudgeon	77	0.173	-3.010	0.911	0.138	-0.605	0.820	0.744	2.152	0.782
Dwarf flat-headed gudgeon	94	0.175	-1.385	0.510	0.112	-0.367	0.523	0.473	1.352	0.558
Western blue-spot goby	48	0.118	-0.653	0.414	0.122	-0.512	0.466	0.549	2.512	0.317
Lagoon goby	100	0.102	0.107	0.438	0.105	0.131	0.409	0.786	1.185	0.545
Tamar goby	62	0.206	-4.022	0.911	0.139	-1.543	0.673	0.656	1.391	0.691
Alien species										
Common carp	81	0.149	-0.623	0.998	0.254	-0.679	0.992	1.699	0.381	0.994
Goldfish	45	0.119	1.621	0.613	0.286	0.922	0.694	2.188	-0.024	0.933
Eastern gambusia	61	0.133	-0.322	0.583	0.272	-3.522	0.620	1.316	0.032	0.442
Redfin perch	44	0.116	-0.410	0.815	0.225	-2.024	0.886	1.627	1.375	0.762

Table 5.5. Reported maximum sizes and derived maximum width and depths (using regressions reported in Table 5.4) and potential for exclusion of maximum length individuals by carp screens mesh designs reported in Table 1. + denotes species likely to be able to pass carp screens at maximum length. The theoretical maximum length of large-bodied species that would be excluded by the three carp screens designs are also shown (lengths adapted from Chapter 4; Hillyard *et al.* 2010). Maximum sizes sourced from Lintermans (2007), with the exception of river gar, which was sourced from Gomon *et al.* (2008).

Species	Reported maximum length (mm)	Derived max depth (mm)	Derived max width (mm)	Alu	JB	45 mm GM
Small-bodied						
Lagoon goby	55	5.93	5.74	+	+	+
Western blue-spot goby	60	6.80	6.41	+	+	+
Eastern gambusia	60	12.80	7.68	+	+	+
Dwarf flat-headed gudgeon	65	6.93	10.00	+	+	+
Carp gudgeon complex	70	13.03	6.86	+	+	+
Murray hardyhead	76	13.58	8.43	+	+	+
Un-specked hardyhead	78	11.40	7.68	+	+	+
Murray-Darling rainbowfish	90	21.28	8.61	+	+	+
Australian smelt	100	11.13	6.87	+	+	+
Medium-bodied						
Small-mouthed hardyhead	107	14.05	10.44	+	+	+
Tamar goby	110	13.80	18.60	+	+	+
Flat-headed gudgeon	115	15.26	16.86	+	+	+
Common galaxias	190	18.39	14.79	+	+	+
Large-bodied						
River gar	350	30.69	14.19	+	+	+
Goldfish	400	115.29	49.04	230.38	211.12	228.64
Bony herring	470	114.02	44.28	331.73	303.03	237.18
Redfin perch	600	132.89	68.97	252.08	234.06	263.33
Murray-Darling golden perch	760	204.64	83.51	259.48	240.71	247.47
Common carp	1200	303.79	178.56	268.26	250.00	250.00

5.5. DISCUSSION

Few large-bodied fishes were caught throughout the study, with large carp only caught in the control nets. As predicted *via* modelling by Hillyard *et al.* (2010), jail bars mesh allowed passage of the largest bony herring to pass any screen (max TL = 241 mm), a key large-bodied riverine and wetland species known for offstream movements, and a species that may be affected by alterations to lateral connectivity (Davies *et al.* 2008; Smith *et al.* 2009b; Conallin *et al.* 2010).

The low catch of carp and other fishes >250 mm TL makes it problematic to extrapolate potential impacts of carp screens to large-bodied native fauna. Whilst there was potential for net 'shyness' (Stuart *et al.* 2008b), it is likely the low catch of large-bodied native species (e.g. golden perch, *Macquaria ambigua ambigua*; Murray cod, *Maccullochella peelii*; silver perch, *Bidyanus bidyanus*; freshwater catfish, *Tandanus tandanus*) reflects their abundance in riverine habitats, or deeper wetlands that are dissimilar to the shallow, regulated wetlands considered in this study (where carp screens are typically used) (Davies *et al.* 2008; Smith *et al*. 2009b; Conallin *et al*. 2010; Hillyard *et al*. 2010). Indeed, few large-bodied native fish have been recorded from wetlands in more northerly regions (cf. Lyon *et al.* 2010), so it is unlikely that carp screens would affect the movement of these species in any region of the MDB. However, knowledge of large-bodied native fish movements which underpin assumptions regarding carp screen designs and possible impacts has been largely gained during a prolonged drought with below average flows (Craik and Cleaver 2008; Murphy and Timbal 2008). Large-bodied fish behaviour might change outside of drought conditions, which may require refinement of carp screen designs and operational guidelines. Future collection of morphological data on large-bodied species will allow comparison with the exclusion regressions described by Hillyard *et al.* (2010).

While all three screens performed similarly in terms of excluding large-bodied fishes, more small and medium-bodied species (e.g. Australian smelt; small-mouthed hardyhead, *Atherinosoma microstoma*) passed the two optimised carp screen designs. Such species are numerically dominant in riverine and wetland habitats (Davies *et al.* 2008; Smith *et al.* 2009b) and are known to move laterally between these habitats in large numbers (Conallin *et al.* 2010; Hillyard *et al.* 2010). Hence, their passage may be susceptible to the impacts of poorly designed carp screens.

The presence of turtles in control nets is likely to explain the reduced abundance and biomass of fish in un-screened control nets relative to jail bars and grid mesh screened nets. Partially consumed fish were found in control nets which caught turtles, suggesting turtles were preying upon fish. Similar observations have been made in trials testing turtle by-catch reduction devices in fyke nets in North America (Fratto *et al.* 2008).

Turtles were exclusively caught in control nets, highlighting the likely impact of carp screens on these species. Turtles may be exposed to predation by foxes (Spencer and Thompson 2005) when navigating on land around a flow control structure fitted with a carp screen, particularly species that are less adapted to movement over land and are unable to fully retract their limbs into their shell such as the broad-shelled turtle, *Chelodina expansa* (Cann 1998). Similarly, turtles have been impinged on carp screens in high-velocity water during wetland filling events (E. Byrne, Brenda Park/Scotts Creek Wetland Community Group; G. Warwick, Commercial Fisher, Lake Bonney; pers. comms.), highlighting a need for frequent inspection and thoughtful consideration of key design requirements in lotic habitats.

No difference in the catch of freshwater decapods (e.g. freshwater shrimps, freshwater prawns) was found between treatment and control nets. Decapods are widely distributed across the MDB (Sheldon and Walker 1998) and are key prey of 'iconic' species such as golden perch (Ebner 2006). Decapods make both longitudinal (Lee and Fielder 1979; Stuart *et al.* 2008a) and lateral movements (Chapter 4; Hillyard *et al.* 2010) in river systems. Hence it is advantageous that all screens tested had little impact on these species.

Like turtles, fish may also be impinged on carp screens. Impingement on screens has been investigated in regard to irrigation and power station cooling off-takes and hydro-power intakes (e.g. Ehrler and Raifsnider 2000; Swanson *et al.* 2005; Baumgartner *et al.* 2009). These situations typically employ angled or travelling screens to reduce the chance of fish impingement or injury (McMichael *et al.* 2004). Although carp screens currently installed in the MDB are normally constructed perpendicular to the flow path, they are typically used at lentic- rather than lotic wetlands, with stable water levels (Chapter 4; Hillyard *et al.* 2010). Further investigation of the impingement of native fauna on carp screens is required in lotic conditions. Finally, risk of fish impingement occurs when dry wetlands are being filled; hence care should be taken to fill wetlands at a flow rate compatible with the swimming ability of the regional fish fauna (cf. Mallen-Cooper 2001).

Utilising morphology of fish to adapt exclusion screens to target pest species has previously been used in the Great Lakes region of North America to exclude alien carp while promoting the passage of native northern pike, *Esox lucius* (French *et al.* 1999). Carp screens could be locally adapted to target carp or other pest species while passing native species in any river system where flow control structures already exist, e.g. the Mississippi River (Schultz *et al.* 2007), or where lakes or wetlands have defined channels where carp screen or traps could be installed, e.g. the Niger River (Bénech and Peñáz 1995) or the Lower Rhine (Borcherding *et al.* 2002).

Conclusions and future considerations

We recommend the jail bars mesh, with 31 mm gaps between bars, as the most promising carp screen design for future trialling and application of carp screens at wetland inlets in the MDB, for incorporation into other carp management technologies, and for testing under high flow conditions. Whilst the grid mesh design compared favourably with the jail bars design in terms of passage of small- and medium-sized native species, and both performed better than the Alutread design, the jail bars allowed the passage of more Australian smelt and larger bony herring. Further, recent field application of both carp screen designs in high-velocity water (≈1.5 $m.s^{-1}$) revealed that the grid-mesh clogs with debris much faster than the jail bars design, is harder to clean and requires more frequent inspection (Thwaites and Smith 2010).

Given that carp have been observed attempting to pass a carp screen, despite no prospect of success, even to the point of injury (B. Smith, SARDI Aquatic Sciences, pers. obs. 2006), it is likely that the jail bars design will require some (horizontal) reinforcement to prevent carp flexing the bars and passing the screen when suitable conditions return with strong movement drivers. Any horizontal reinforcement of jail bars should be spaced greater than the potential maximum body depth of key large-bodied species (~400 mm) to ensure the benefits of the jail bar design in passing key native species are preserved. It is likely that any horizontal reinforcement may increase the incidence of clogging with debris, though this will require validation in future field trials.

Modifying the jail bars design to a less specific dimension may offer more practical fabrication options. Re-analysis of the data and exclusion thresholds proposed by Hillyard *et al.* (2010) reveals that standardising the aperture between jail bars from 31 mm to 30 or 35 mm would alter the carp exclusion threshold to 240 mm TL or 275 mm TL, respectively. Thus, even a 35 mm gap would still ensure sexually mature common carp are excluded (Brown *et al.* 2005). These standardised dimensions would have only minor influence on the percentage of the carp excluded in the population analysed by Hillyard *et al.* (2010) (30 mm = 93 %, 31 mm = 92 %, 35 mm = 91 %). Similar outcomes would result for native large-bodied golden perch (30 mm = 70 %, 31.4 mm = 65 %, 35 mm = 53 %) and virtually no effects would be realised for laterally compressed bony herring (30 mm = 97 %, 31 mm = 98 %, 35 mm = 99 %). These examples highlight the flexibility of the jail bars design and the potential to adapt the apertures between bars to suit local fish assemblages. However, given the principal purpose of carp exclusion, we advocate against making the apertures any larger than 31 mm.

The 31 mm jail bars carp screen proposed herein is applicable to other carp management technologies such as carp separation cages (CSC: Stuart *et al.* 2006a; Smith *et al.* 2009a; Thwaites *et al.* 2010). Indeed, the 31 mm jail bars design corresponds to the improvements to current CSC designs suggested by Stuart *et al.* (2006a). With plans to install CSC on fishways on the River Murray (Barrett and Mallen-Cooper 2006) and at wetlands considered to be carp recruitment ‘hotspots’, construction from a jail bars style mesh would maximise the passage of small and medium-sized native fish, while still trapping adult carp.

While the notion of using carp screens to exclude large carp from wetlands initially appears simple, designing carp screens which exclude carp yet promote volitional passage by native species requires detailed information about the local aquatic fauna and careful consideration, planning and integration with wetland and/or carp management objectives (cf. Smith *et al.* 2009a).

6. Chapter Six- GENERAL DISCUSSION

Carp screens, metal mesh screens typically placed in flow-control structures on wetland inlets, are designed to reduce the damage caused by carp in wetlands and to deny them access to spawning habitat. From a field survey of carp screens (Chapter 4; Hillyard *et al.* 2010), and discussions with wetland managers, it is apparent that carp screens have been designed and used without reference to ecological information about carp, and without regard for the potential impacts on native fish and other fauna. In this thesis, these shortcomings are addressed by designing and testing two new carp screens based on carp morphology, and by comparing the impacts of new and existing designs on other fauna. A complementary study was also undertaken of fish movements to and from wetlands in the Lower Murray.

This investigation had four main objectives:

1. To describe the spatial, temporal and directional patterns of fish movements in spring and autumn, when carp move to and from wetlands;
2. To gather morphological data from fish making lateral movements, to 'optimise' the design of carp screens to prevent the passage of mature carp and limit the effects on other fauna;
3. To catalogue the locations and designs of existing carp screens in the Murray-Darling Basin (MDB), to determine where they are used and to assess their effectiveness; and
4. To design and test two new carp screen designs.

6.1. Key results, implications and future directions

6.1.1. Lateral movements

Chapters 2 and 3 describe the faunal movement through six wetland inlets in the Lower Murray over 13 weeks in spring 2006 (autumn monitoring was planned, but abandoned because the wetlands dried out). Morphological data from carp and large-bodied (>250 mm TL) native fish were collected; variations in spatial, temporal and directional movements were described, and some insights were gained into cues for movements.

Samples included some 210,000 fish, from 14 native and four alien species, including all species commonly recorded in local wetlands (Smith *et al.* 2009b) and the Murray channel (Davies *et al.* 2008). Four species have current or proposed state or federal conservation listings (viz. pouched lamprey, freshwater catfish, Murray cod, congolli: Hammer *et al.* 2009). Diadromous and estuarine species were more prevalent at wetland inlets closest to the river mouth.

Numbers of fish moving laterally, pooled among wetlands were dominated by small-bodied (<250 mm TL, and in most cases <100 mm TL) Australian smelt, carp gudgeons and un-specked hardyhead and large-bodied bony herring (including both adults and juveniles). Biomass was dominated by Australian smelt, bony herring, carp and golden perch.

Two native species (bony herring, golden perch) and two alien species (redfin perch, goldfish) emerged as large-bodied species whose morphology and behaviour should be considered when designing carp screens due to their abundance in lateral movement sampling. Screens could restrict the movements of other large-bodied species (e.g. freshwater catfish, Murray cod), but these are not currently common locally (Davies *et al.* 2008; Smith *et al.* 2009b) and were caught in insufficient numbers for analysis of potential impacts. The detection of these uncommon species highlights that a precautionary approach should be followed in the carp screen planning process to ensure impacts are minimised on species such as these.

Lateral movements varied among wetlands, but clear, directional patterns were not apparent (Chapter 2; Conallin *et al.* 2010). The variation was attributed to differences in wetland and inlet morphology. Thus, shallow inlets (<1 m) with woody debris had more small-bodied species and fewer large-bodied species (except bony herring), and deeper (>1 m) inlets without debris contained larger species, particularly carp and golden perch (but not bony herring). As the habitats of carp and golden perch may overlap in the Murray-Darling Basin (Koehn and Nicol 1998), it appears that wetlands attractive to carp may also be attractive to golden perch. Monitoring is needed during routine screen operations, or prior to installation, to assess the potential risk of screens hindering local movements of golden perch, bony herring and other large-bodied native species.

This field study was undertaken in spring 2006 during a period of below-average flows, when the Lower Murray and its wetlands remained connected, and the absence of flow cues may have been responsible for the generally haphazard, bi-directional movements recorded. In years with higher spring flows, the anticipated mass river-to-wetland carp movements probably would become apparent again (cf. Jones and Stuart 2008a). Similarly, the composition and size structure of the assemblage and the responses of the species during higher flows could differ from those reported here. The lateral movement study described in Chapters 2-3 therefore should be repeated during high flows, in spring and autumn. The size distributions of fish moving in autumn may shift, relative to those in spring, by the inclusion of young-of-year fish spawned from spring through autumn. Carp are likely to move from wetlands to deeper, cooler riverine habitats in autumn, as in North American lakes (Penne and Pierce 2008). Monitoring emigrations from wetlands could present opportunities to harvest carp, if they moved *en masse*, although depending on what is found, a range of logistics would need to be considered (e.g. technology to separate carp from native large-bodied species). Further carp harvesting opportunities could follow from identification of over-wintering aggregations (Penne and Pierce 2008). Carp captured in spring or summer could be implanted with radio or acoustic tags and their movements monitored over autumn and winter to locate aggregations accessible to conventional fishing methods. This strategy has been employed successfully to manage carp in Lakes Crescent and Sorrell, Tasmania, since early 2000 (Diggle *et al.* 2004; Inland Fisheries Service 2004). Net avoidance (cf. Hunter and Wisby 1964) may be one additional factor explaining the low number of carp recorded moving laterally. Future radio or acoustic tagging studies may resolve whether net avoidance was the cause of the low carp catch in this study.

Bony herring and golden perch were the two most common large-bodied native species recorded moving laterally (Chapter 2; Conallin *et al.* 2010), and are considered likely to be affected by carp screens. As detailed in Chapter 4 (Hillyard *et al.* 2010), there is little evidence that golden perch recruitment would be adversely affected by the use of carp screens. Golden perch do not spawn on floodplains (Humphries *et al.* 1999) and their larvae avoid floodplain habitats (Gehrke 1991), implying that these may represent suboptimal spawning habitats. However, juvenile golden perch may use floodplain habitats (Pusey *et al.* 2004), demonstrating the potential for interaction with carp screens, although the new carp screen designs are unlikely to physically restrict their movements (Chapter 4; Hillyard *et al.* 2010). In contrast, bony herring do spawn in wetland habitats (Pusey *et al.* 2004). The new carp screens described herein (Chapter 4; Hillyard *et al.* 2010) were designed to allow movements of mature bony herring, maintaining wetland access and reducing impacts on spawning and potential recruitment. Carp screens probably will not adversely affect most large-bodied species (e.g. Murray cod, freshwater catfish, silver perch) (Chapter 4; Hillyard *et al.* 2010), but site-by-site assessments are required prior to screen installation.

The study showed some temporal movement patterns for some species. For example, four small-bodied species (carp gudgeons, un-specked hardyhead, flat-headed gudgeon, Murray-Darling rainbowfish) became more abundant over the 13-week sampling period, whereas Australian smelt, a schooling species, became less abundant. These trends were associated with increasing water temperature and photoperiod. It is unclear why numbers of Australian smelt decreased, when fish generally are more active at higher temperatures (e.g. Lucas and Baras 2001). A similar trend of declining Australian smelt abundance has also been observed at the inlet and outlet of Banrock Station wetland, South Australia (Fredberg *et al.* 2009). This decline may be related to mortalities caused by capture (netting) and handling, as Australian smelt are a particularly fragile species (McDowall 1996). Alternately, the decline in Australian smelt abundance may reflect variation in the species schooling behaviour, whereby large schools, possibly related to spawning which begins in late-winter (Humphries *et al.* 2002), were caught early in the study, distorting the temporal abundance. Further monitoring with a less-invasive sampling method (e.g. split-beam sonar: Matveev 2007) could determine whether the decline reflects a biological pattern or is an artefact of sampling. No such trend was observed, however, for the large-bodied species.

In general, more fish were recorded moving from, rather than to, the wetlands. The net emigration was attributed to low river levels associated with drought, and is a typical response of fish to gradually-falling water level (Poizat and Crivelli 1997; Cucherousset *et al.* 2007; Lyon *et al.* 2010). The mass immigration of carp to wetlands that had been anticipated did not occur, due perhaps to a combination of the relatively stable, perennial river-wetland connection (cf. Schultz *et al.* 2007; Lyon *et al.* 2010) and the absence of seasonal flow-related cues (cf. Bénech and Peñáz 1995; Stuart and Jones 2006b). The sampling interval (*c.* 24 h weekly) could have missed a migration, but anecdotal observations (e.g. river fishermen, shack owners, lock operators) suggest that it did not occur in spring 2006. Nevertheless, the mean abundance of carp did peak, in mid to late September (sampling weeks 4-5, mean *c.* 5-7 fish per inlet per week), representing a 3-fold increase over the preceding three weeks (mean *c.* 2 fish per inlet per week). This coincided with a water temperature rise from 13 °C to 16 °C, potentially a trigger for spawning movements (Rodriguez-Ruiz and Granado-Lorencio 1992; Jones and Stuart 2008a) and spawning (Crivelli 1981; Hume *et al.* 1983b). Some young-of-year carp (20-49 mm TL; $n = 14$) were captured (moving to and from wetlands) in weeks 12-13 (early to mid-November), showing (from application of Vilizzi's (1998a) length-at-age models) that spawning had taken place no earlier than week 5 (late September, when mean water temperatures reached the 16 °C spawning threshold). This was expected as the initial spawning peak is typically between September and December, although carp may spawn multiple times in one season (approximately September to March, but recorded extremes are from mid-August to early-May: Vilizzi 1998a; Sivakumaran *et al.* 2003; Smith 2004; Brown *et al.* 2005) when the mean water temperature exceeds 16 °C and day length exceeds 10 hours (Crivelli 1981; Davies *et al.* 1986).

Understanding carp movement and spawning triggers is needed for localised management of carp spawning. If screens are removed in winter, to allow large carp to escape, they should ideally be re-instated before river-to-wetland movements begin, and certainly before temperature and photoperiod spawning triggers are reached. In the Lower Murray, where carp generally have year-round access to perennially-inundated wetlands, carp screens should be installed by mid-July, when photoperiod exceeds the 10-hour spawning threshold. Installing carp screens by mid-July would buffer any inter-annual climatic variation in the onset of the 16 °C spawning threshold. Longer duration sampling of lateral movements may reveal when carp actually begin moving, allowing refinement of carp screen applications based on specific environmental triggers. Where river levels are less stable, knowledge of commence-to-fill thresholds should guide when carp screens are installed, as carp typically enter wetlands as seasonal flows and levels increase e.g. Barmah-Millewa Forest (Stuart and Jones 2002; Jones and Stuart 2008a), Upper Mississippi (Barko *et al.* 2006), Camargue marshes (Poizat and Crivelli 1997).

6.1.2. Carp screens

Chapter 4 (Hillyard *et al.* 2010) describes a survey of existing carp screen designs and associated structures, and the ways that morphological data could be used to exclude mature carp and estimate the exclusion of other species. Chapter 5 describes field trials of the two new screen designs, compared to the most common existing design, and also considered the potential for incidental effects on small-bodied fish.

The survey showed that carp screens are used throughout the southern Murray-Darling Basin, but most are in South Australia (SA). This probably reflects the efforts of Wetland Care Australia, SA Murray-Darling Basin Natural Resources Management (SAMDBNRM) Board and the SA Department for Environment and Natural Resources to rehabilitate wetlands in the Lower Murray in SA, which have been degraded by perennial inundation and are now ideal carp habitat (e.g. Jensen 2002). It is likely that carp screens will be installed more widely in future as water demands increase and concern intensifies over ecosystem ‘health’, leading to increased management of off-channel habitats ('The Living Murray Environmental Works and Measures Program’ Program: Murray-Darling Basin Authority 2009), hence the new designs described here could be implemented as further regulators are installed. In South Australia, wetland flow-control structures (including carp screens) are currently funded by the SAMDBNRM Board, subject to guidelines (Allder 2008) designed to maintain fish passage (cf. Mallen-Cooper 2001). The guidelines say little of carp screens and their design, but suggest only that the choice of mesh should be based on the size of fish (no specific species) that managers wish to enter a wetland.

There is a diverse range of screen designs and dimensions in use across the MDB, with some coordination evident in South Australia and in the Gunbower Forest, Victoria. This is due to structures being designed by the same organisation, and to the ease of accessing some materials, rather than a biologically-driven rationale. Regulators in Gunbower Forest have 45-mm square (internal dimension) grid mesh screens, coincidentally matching one of the screen designs proposed here (on the basis of fish morphology). The design documents for these regulators stated that 50-mm (45-mm square internal dimension) mesh would restrict adult carp, but without substantiation (cf. Sinclair Knight Merz 2005). The survey of carp screens highlights the wide variation in flow-control structure design, and has helped to inform the design of morphologically-optimised screens (Chapter 4; Hillyard *et al.* 2010).

New carp screens were designed (31 mm jail bars and 45 mm square grid mesh), using a method adapted from French *et al.* (1999), to exclude >250 mm TL carp based on morphological data (length, width, depth) for the five dominant large-bodied species (bony herring, carp, golden perch, redfin perch, goldfish) in the lateral movement study. The >250 mm TL (*c.* 225 mm FL: K. Hillyard, unpublished data) exclusion threshold is a conservative value relative to known lengths at maturity (median: males 307 mm FL; females 328 mm FL: Brown *et al.* 2005), although initial lengths at maturity may fall close to this threshold (initial: males 230 mm FL; females 280 mm FL: Brown *et al.* 2005). An alternate exclusion threshold could have been the length at which carp adopt an environmentally damaging benthivorous habit (Sibbing 1988; Matsuzaki *et al.* 2009; Badiou and Goldsborough 2010). However, as benthivory in carp occurs at the start of the juvenile period (~35 mm Standard Length (SL): Vilizzi 1998b), excluding carp this small would affect many native species. A screen sufficiently fine to prevent passage of *c.* 35 mm SL carp would be prone to clogging with debris, and would require regular maintenance (comparable to issues with fine mesh screens used to control carp in Lakes Crescent and Sorell, Tasmania: Inland Fisheries Service 2004).

Further investigation is needed to assess the ability of fish to compress their bodies to pass through screens. From published research (e.g. Robichaud *et al.* 1999), a 10% compression factor was applied here to all large-bodied fish, aside from the laterally-compressed bony herring, when calculating the screen exclusion thresholds (Chapter 4; Hillyard *et al.* 2010). This acts as a ‘factor of safety’ in design, effectively making the screen design more conservative in terms of the theoretical length of fish able to pass. Validation of the actual ability of various species to compress their body would allow further refinements. Rough estimates of actual compression could be determined by applying a mass to the dorsal or lateral surface of a fish equivalent to its size-relative swimming force (cf. Thwaites *et al.* 2007). Analysis of spatial (e.g. latitudinal) and temporal variation in carp morphology (cf. Phillips and Shine 2006) would also allow further localised refinement of carp screen apertures.

Length-width and length-depth morphological ratios were calculated for all common small- (Chapter 5) and large-bodied fish (Chapter 4; Hillyard *et al.* 2010) and compared with the aperture dimensions of the new and existing screen designs to assess the theoretical lengths at which screens would exclude each species. Published maximum lengths (e.g. Lintermans 2007; Gomon *et al.* 2008) were used to extrapolate theoretical maximum widths and depths for small-bodied species, suggesting that none of the small-bodied species considered should be physically excluded by either of the new, or most frequently used, screen designs. Modelling large-bodied fish morphology against the two new and the most common existing screen designs showed that Alu-Tread (the most common existing design) would allow larger individuals of all species (native and alien) to pass than the two new designs (Chapter 4; Hillyard *et al.* 2010). Alu-Tread would still restrict carp below median reproductive length, and allow only slightly more and larger carp to pass (Alu-Tread = 91.81%/268 mm TL vs. jail bar or grid mesh = 92.40%/250 mm TL). From morphology alone, Alu-Tread appeared to be an adequate option for continued use in carp screens, relative to the two new designs.

Field testing showed the two new carp screen designs passed more and larger native fish than Alu-Tread. Large carp (>100 mm TL) were caught only in the control nets. Significantly larger bony herring passed the jail bar design than the other screens (Chapter 5), despite larger bony herring being theoretically capable of passing Alu-Tread, based on their laterally-compressed morphology corresponding with the shape of Alu-Tread apertures. Preliminary testing of the two new screen designs was also undertaken in a culvert at the inlet of Lake Bonney, SA, to ascertain the performance (e.g. clogging, durability) of the new screen designs in flowing water. Though the conditions in the Lake Bonney inlet were not typical of where carp screens are normally used (very high flow, >1.5 m.s^{-1}, into a large terminal lake) the trial provided insight into how the screens may perform during filling of a dry wetland. The Lake Bonney trial suggested that the jail-bar design was less prone to fouling by debris than the grid-mesh design (Thwaites and Smith 2010). Reducing the need to clean carp screens is important given most are in remote locations, typically without access to electricity (Chapter 4; Hillyard *et al.* 2010) limiting automatic options for cleaning such as those used on power station water off takes (e.g. Taft 2000). As many screens are designed to rotate, cleaning is typically undertaken by *in situ* back-flushing, normally on a quarterly or biannual basis (K. Mason SA MDB NRM Board, pers. comm. 2011). Future studies, field testing the jail-bar design, should be designed to compare the rate of fouling with the grid mesh and existing designs.

Aside from reduced clogging observed at Lake Bonney, the field results showed that more, and larger native species were able to pass the jail-bar screen; hence, it is the preferred option for general use in wetland inlets. Encouragingly, the new jail bar design is starting to replace existing screens as flow-control structures are refurbished (K. Hillyard, pers. obs.). Given the favourable results of the field trials (the need for further *in situ* trials notwithstanding), the guidelines for wetland flow control structures (Allder 2008) should be updated to specify use of the *c.* 31 mm jail-bar screen design (Chapter 5).

Initial field testing in wetlands throughout the SA Murray-Darling Basin successfully demonstrated the benefits of the new screen designs, particularly the jail-bar design, relative to Alu-Tread. However, low water levels associated with drought, and associated logistical constraints, prevented *in situ* testing of the new screen designs in wetland flow-control structures. Testing screens in directional fyke nets was effective for initial trials, but the motivation for fish to move laterally into off-channel habitat through a flow-control structure is likely to differ from encountering a net in a wetland. For example, carp repeatedly attempt to swim through carp screens to enter wetlands, despite no prospect of physically passing the screen, even to the point of injury (B. Smith, SARDI Aquatic Sciences, pers. comm. 2006). Therefore, despite the promising performance of the new screen designs, further field testing in flow-control structures where carp screens are traditionally used, under a wide range of conditions, is recommended to verify the results in Chapter 5. As well as assessing the size and abundance of fish which pass the screens, the potential for fouling, clogging with debris and the influence of high velocity flows should be investigated in wetland inlets where carp screens are typically used (cf. Lake Bonney trial: Thwaites and Smith 2010).

During the lateral movement study (Chapter 4; Hillyard *et al.* 2010) and field screen trials (Chapter 5), three species of turtles (Chelidae) were caught regularly in un-screened nets, but never in screened nets (Chapter 5). The turtles may be vulnerable to predators if forced to move overland to bypass a screen, particularly species not well adapted for movement over land (e.g. broad-shelled and Murray turtles: Cann 1998), and may be trapped on screens in high-velocity flows (Chapter 5; Thwaites and Smith 2010). These effects should be considered where strong currents occur, such as when wetlands are filling, and when carp screens are installed on a flow-through wetland. Some success in improving turtle movement past carp screens in high velocity flows (during a wetland re-filling event) has been achieved using newly designed 'carp pivot screens' angled in the direction of the flow with the top of the screen set just below the water surface (Thwaites and Smith 2010). This allowed turtles to move directly over the screens, and when combined with ‘carp deflector screens’, prevented carp from jumping over the screens and escaping the wetland. While carp pivot screens appeared to successfully reduce turtle entrapment, further investigation (e.g. movement behaviour through regulated v. unregulated inlets, determination of swimming ability) is needed to address the issue of turtle movement past carp screens, and indeed wetland regulators in general.

6.2. Contributions to future carp management

This investigation has produced a scientifically-robust carp control option for use at wetland inlets (cf. French *et al.* 1999; Jensen 2002; Lougheed *et al.* 2004), and one that could be combined with other carp control technologies such as carp separation cages (e.g. Stuart *et al.* 2006a; Thwaites and Smith 2010). Improved understanding of where and when carp, native and alien fish species move through wetland inlets may improve habitat restoration projects, even where carp screens are not installed. For example, particular wetland inlet habitat characteristics could be used to attract native species (e.g. carp gudgeons, which have an affinity for woody debris: Chapter 2; Conallin *et al.* 2010) to wetland inlets.

This study has applications beyond the Murray-Darling Basin. In particular, the carp screen design process described herein could be applied in North and South America (Zambrano *et al.* 2006), Africa (Moreau and Costa-Pierce 1997), New Zealand (Tempero *et al.* 2006; Rowe 2007) and elsewhere in Australia (e.g. Pinto *et al.* 2005). Further, the process and methodology for designing screens to restrict a specific species at a certain size (thus, analysis of the fish community, collection of morphological data, design of new screen followed by validation and testing) could be applied to other pest fish species.

While the screens described herein are a useful tool for managing carp at the wetland scale, they remain one option among many, some of them currently available [e.g. carp separation cages (Stuart *et al.* 2006a; Thwaites and Smith 2010), Judas carp and traditional fishing (Diggle *et al.* 2004; Inland Fisheries Service 2004)] and others still in development [e.g. pheromones (Sorensen and Stacey 2004), environmental attractants (Elkins *et al.* 2009), pathogens (McColl *et al.* 2007; Saunders *et al.* 2010) or daughterless carp (Thresher and Bax 2003; Thresher *et al.* 2009)]. Ultimate success in managing carp populations on a river-basin scale probably will see carp screens used to mitigate recruitment at identified carp recruitment ‘hot-spots’ (Gilligan *et al.* 2009), as part of a broader integrated strategy involving a variety of methods (e.g. Harris 1997).

6.3. Impact of carp screens on wetland ecology

Improvements to wetland ecology and condition can be effected using carp screens. The exclusion of carp from wetlands will modify wetland ecology, particularly where carp have become naturalised, in a number of ways. Preventing carp from accessing wetlands will likely improve water quality and increase the diversity and abundance of macrophytes (Lougheed *et al.* 1998; Lougheed and Chow-Fraser 2001; Lougheed *et al.* 2004; Chow-Fraser 2005) by removing disturbances associated with benthic feeding (‘mumbling’: Sibbing 1988), where the sediment is resuspended, increasing turbidity and liberating nutrients (also coupled with reduced nutrient excretion associated with reduced prey consumption) (e.g. Williams *et al.* 2002; Parkos *et al.* 2003; Matsuzaki *et al.* 2007; Matsuzaki *et al.* 2009). Impacts on macrophytes may involve direct herbivory and undermining of plants (Crivelli 1983; Sidorkewicj *et al.* 1996; Miller and Crowl 2006), as well as indirectly *via* reduced water quality (e.g. high turbidity, nutrients) affecting growth (Parkos *et al.* 2003; Matsuzaki *et al.* 2007; Matsuzaki *et al.* 2009). Predation on benthic macroinvertebrates is likely to be reduced (Khan 2003; Miller and Crowl 2006), and changes to nutrient dynamics are likely to have cascading effects on phytoplankton, zooplankton and species which prey upon them (Tatrai *et al.* 1997; Matsuzaki *et al.* 2009; Weber and Brown 2009). Although the level of impact requires quantification, localised removal of carp may reduce predation, competition or suppression (e.g. *via* habitat degradation) of frog populations (Gillespie and Hero 1999; Germaine and Hays 2009; Kloskowski 2011). Competitive pressure may ease for species with a similar diet to carp, found both within the MDB, e.g. freshwater catfish *Tandanus tandanus*, Australian smelt *Retropinna semoni*, common galaxias *Galaxias maculatus*; and elsewhere, e.g. axolotl *Ambystoma mexicanum*, largemouth bass *Micropterus salmoides* and bluegills *Lepomis macrochirus* among others (Clunie and Koehn 2001; Khan *et al.* 2002; Wolfe *et al.* 2009; Zambrano *et al.* 2010; Weber and Brown 2011). Small carp (<300 mm TL) are prey for numerous species of birds such as cormorants, herons, grebes and eagles (Miller 1979; Crivelli 1981; Moser 1986; Brown *et al.* 1991; Kloskowski 2004), as well as predatory fish in the MDB and elsewhere (e.g. Murray cod, northern pike *Esox lucius*: Navarro and Johnson 1992; Ebner 2006). Thus, when carp screens are used and carp recruitment is reduced or prevented, the removal of small carp might alter food-chain dynamics.

The foregoing changes to wetland ecology assume the exclusion of all carp, but carp <250 mm TL will still be able to pass the new screen designs described here. Hence, the use of jail bars or grid mesh may not yield all the benefits (or modifications) to wetland ecology of overall carp exclusion or removal. Exclusion of all carp may appear desirable, but it would also require exclusion of all native fish and limits on movement of other fauna (e.g. macroinvertebrates, turtles, tadpoles). Although carp <250 mm TL are not sexually mature (Brown *et al.* 2005), they are benthic feeders from ~30 mm TL (Vilizzi 1998b). Thus, carp >30 mm TL can contribute to increases in turbidity, modifying nutrient levels and phytoplankton and adversely affecting aquatic plants (by herbivory, increased turbidity or nutrient release) (e.g. Roberts *et al.* 1995; Sidorkewicj *et al.* 1999; Matsuzaki *et al.* 2007; Weber and Brown 2009). Carp in this size range (30-250 mm TL) consume algae, detritus, zooplankton and microcrustaceans (García-Berthou 2001; Khan 2003). In Victorian lakes <250 mm TL carp share similar diets with native Australian smelt and common galaxias (Khan *et al.* 2002). Many of the perturbations and potential competitive impacts associated with carp in general occur even with immature carp. Despite this, the impacts of allowing <250 mm TL carp passage will still result in overall positive outcomes for wetland condition as carp impacts are both density- and size-dependent (Roberts *et al.* 1995; Chumchal *et al.* 2005; Driver *et al.* 2005a; Badiou and Goldsborough 2010).

The outcomes of preventing adult carp from accessing wetland habitats are generally positive, but determining the actual benefits to wetland condition from carp screens alone can be difficult owing to confounding impacts of climatic variation and water level manipulation and the need for comparable ‘control’ wetlands (Recknagel *et al.* 1998; Lougheed *et al.* 2004; Chow-Fraser 2005; Weber and Brown 2009). Though likely to be both expensive and logistically difficult to achieve, well-designed experimentation and monitoring, beyond mesocosm or exclusion type experiments scale (i.e. entire wetland scale: cf. Wallace *et al.* 2009), are required for confirmation.

6.4. Future priorities

The following recommendations would increase the understanding of lateral fish movements and improve the use of the optimised carp screen designs proposed in this thesis.

1. The lateral movement component of this study was conducted during a period of low flow with little river-level variation. These conditions are likely to recur as climate change advances (CSIRO 2008), but lateral movements during ‘normal’ conditions also require investigation. Prolonged lateral movement sampling (at least spring to autumn) under normal flow conditions may better indicate the cues and timing of carp movements, allowing refinement in the application of carp screens and possibly identifying further opportunities for carp control.
2. The low river levels associated with drought conditions precluded the testing of the optimised carp screen designs in wetland flow control structures where they would typically be used. Testing the jail bar and grid mesh designs using fyke nets in wetlands (Chapter 5) and the inlet of Lake Bonney (Thwaites and Smith 2010) demonstrated positive initial results, and further testing in wetland flow control structures should be undertaken to confirm the benefits of jail bars in places where they would be typically used.
3. Many of the assumptions regarding improvements in wetland condition brought about by carp screens are derived from mesocosm or within-wetland ‘barrier’ type experiments which focus on a limited suite of parameters (e.g. King *et al.* 1997; Robertson *et al.* 1997; Williams *et al.* 2002; Parkos *et al.* 2003; Wolfe *et al.* 2009; Kloskowski 2011). Wetland-scale assessment of changes in wetland condition or ecology from carp screen use are lacking, particularly under Australian conditions (cf. Wallace *et al.* 2009). Although likely to be challenging in design, cost and implementation, a robust, well-planned multi wetland assessment of how carp screens alter or improve wetland condition should be undertaken. Care should be taken to avoid the pitfalls of previous wetland-scale experiments which have included a lack

(e.g. Lougheed *et al.* 2004), or poor selection (e.g. Recknagel *et al.* 1998), of ‘control’ wetlands.

4. The results of the existing carp screen survey (Chapter 4; Hillyard *et al.* 2010) and conversations with wetland managers have suggested that some managers may not understand how and where carp screens should be used, and the actual benefits (and disadvantages) of their use, despite published guidelines to inform the selection and implementation of carp management options (including carp screens and carp separation cages: Smith *et al.* 2009a). Effective communication of the results of this study to wetland managers, and integration of the recommendations into wetland structure guidelines (e.g. Allder 2008), could have considerable benefits.

The jail bar screen design proposed in this thesis offers a simple, effective and immediately available technology for denying large carp access to their desired spawning grounds, where their impacts are pronounced, while minimising impacts on offstream movements of native fish. However, screens should only be utilised when carp management is a recognised priority, and considered achievable, during the wetland management planning process. Carp management at the wetland-scale can locally improve wetland condition, but application of the carp screens at wetlands identified as spawning ‘hotspots’ may help achieve basin-scale carp control when integrated with other management techniques.

7. References

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8. Appendices

8.1. Appendix 1- Details of carp screen locations and designs

Details of the location and design of wetland regulating structures in the Murray-Darling Basin with CES *in situ* recorded in June-July 2007. The design and dimension of the CES mesh at each site are also provided (Chapter4; Hillyard *et al.* 2010).

Wetland complex	Structure name	Latitude (° S)	Longitude (° E)	Structure design	CES mesh design	CES aperture dimension (mm)
Akuna Station	Akuna Station 01	34.1800	140.2378	box culvert	reinforced Alu-Tread	34x43
	Akuna Station 02	34.1750	140.2460	well	Alu-Tread	34x97
	Akuna Station 03	34.1748	140.2476	box culvert	reinforced Alu-Tread	34x43
Banrock Station	Banrock Station 01	34.2100	140.3484	box culvert	reinforced horizontal bars	15x290
	Banrock Station 02	34.1996	140.3582	independent screen	reinforced horizontal bars	15x450
Beldora Lagoon	Beldora Lagoon 01	34.3392	140.3995	box culvert	weldmesh	46x22
Brenda Park	Brenda Park 01	34.0908	139.6692	box culvert	Alu-Tread	34x97
	Brenda Park 02	34.0683	139.6826	pipe culvert	Alu-Tread	34x97
Chowilla	Lake Litra 01	33.9411	141.0010	box culvert	reinforced Alu-Tread	34x43
	Lake Merreti 01	34.0410	140.7512	pipe culvert with cage	Alu-Tread	34x97
	Lake Merreti 02	34.0447	140.7618	independent screen	Alu-Tread	34x97
	Lake Merreti 03	34.0472	140.7712	independent screen	Alu-Tread	34x97
	Pilby Creek 01	33.9852	140.8927	box culvert	reinforced vertical bars	330x7
Cobdogla	Cobdogla 01	34.2399	140.3981	box culvert	Alu-Tread	34x97
	Cobdogla 02	34.2315	140.3957	box culvert	Alu-Tread	34x97
	Cobdogla 03	34.2244	140.4025	box culvert	Alu-Tread	34x97
Gunbower	Barham Cut 01	35.6926	144.2124	independent screen	weldmesh	45x45
	Little Gunbower Creek 01	35.6951	144.1742	box culvert	weldmesh	45x45
	Shillinglaws 01	35.7395	144.2536	box culvert	weldmesh	45x45

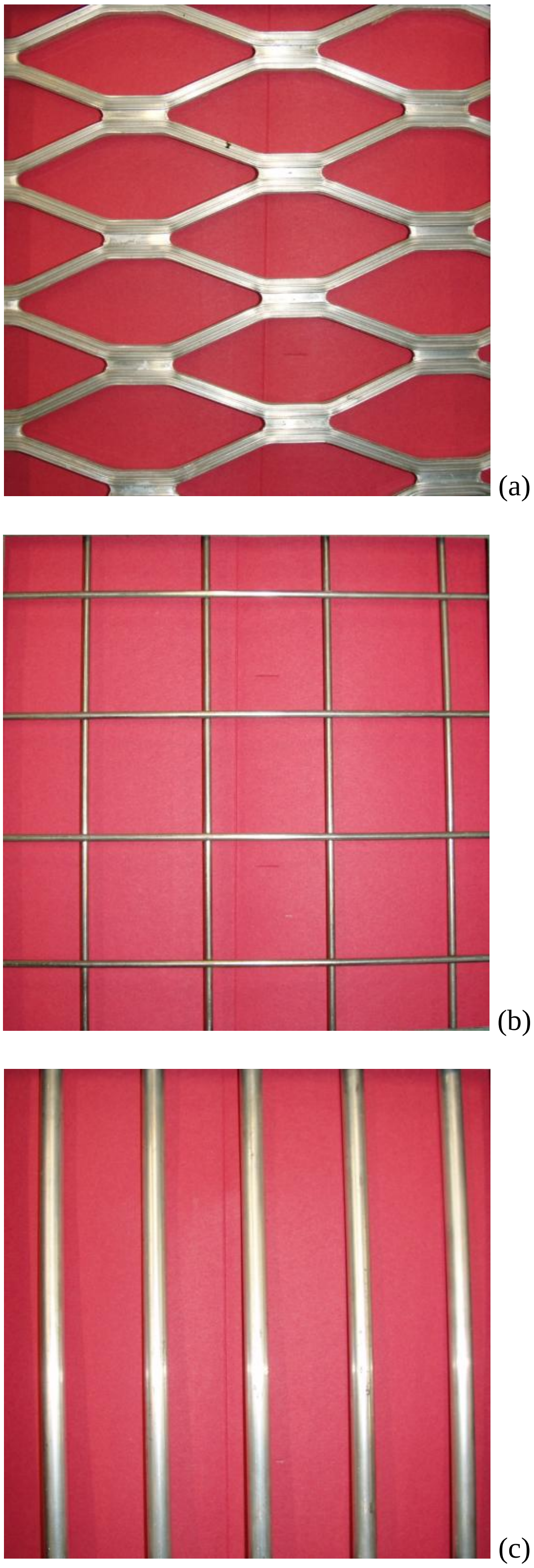
Appendix 1- Details of carp screen locations and designs- continued.

Wetland complex	Structure name	Latitude (° S)	Longitude (° E)	Structure design	CES mesh design	CES aperture dimension (mm)
Gurra Gurra Lakes	Causeway Lagoon 01	34.3104	140.5961	pipe culvert	Alu-Tread	34x97
	Little Duck Lagoon 01	34.3109	140.5956	pipe culvert	Alu-Tread	34x97
	Old Loxton Road 01	34.3053	140.5927	box culvert	Alu-Tread	34x97
	Winding Creek 01	34.3115	140.6095	sheet piling system	Alu-Tread	97x34
Hart Lagoon	Hart Lagoon 01	34.1653	139.9530	box culvert	Alu-Tread	34x97
Lindsay - Wallpolla	Horseshoe Lagoon 01	34.1364	141.8401	box culvert	woven mesh	22x22
	Lake Wallawalla 01	34.1676	141.1843	box culvert	woven mesh	22x22
	Lake Wallawalla 02	34.1657	141.1902	box culvert	woven mesh	22x22
	Websters Lagoon 01	34.0899	141.1378	box culvert	woven mesh	22x22
Little Toolunka Flat	Little Toolunka Flat 01	34.1429	139.9215	box culvert	Alu-Tread	34x97
	Little Toolunka Flat 02	34.1144	139.9607	pipe culvert	Alu-Tread	34x97
Loveday Swamp	Loveday Swamp 01	34.2672	140.3880	box culvert	reinforced Alu-Tread	34x43
Martins Bend	Martins Bend 01	34.2905	140.6171	box culvert	Alu-Tread	34x97
Morgan's Lagoon	Morgan's Lagoon 01	34.4708	139.6012	box culvert	security mesh	58x63
	Morgan's Lagoon 02	34.4652	139.6002	box culvert	security mesh	58x63
Ngak Indau	Ngak Indau 01	34.3282	140.5712	pipe culvert with cage	woven mesh	23.5x25.3
Overland Corner	Overland Corner 01	34.1594	140.3396	box culvert	weldmesh	46x22
Paisley Creek	Paisley Creek 01	34.3431	139.6187	pipe culvert	Alu-Tread	34x97
	Paisley Creek 02	34.3403	139.6198	pipe culvert	weldmesh	45x45
	Paisley Creek 03	34.3439	139.6215	pipe culvert	Alu-Tread	34x97
Paiwalla	Paiwalla 01	35.0317	139.3684	pipe culvert	double security mesh	26.15x34.14
	Paiwalla 02	35.0342	139.3683	pipe culvert	double security mesh	26.15x34.14

Appendix 1- Details of carp screen locations and designs- continued.

Wetland complex	Structure name	Latitude (° S)	Longitude (° E)	Structure design	CES mesh design	CES aperture dimension (mm)
Ramco Lagoon	Ramco Lagoon 01	34.1565	139.9139	box culvert	Alu-Tread	34x97
Reedy Creek	Reedy Creek 01	34.9478	139.2515	sheet piling system	Alu-Tread	34x97
Richardson's Lagoon	Richardson's Lagoon 01	36.0396	144.5679	box culvert	weldmesh	22x22
Riverglades	Riverglades 01	35.0892	139.3065	box culvert	Alu-Tread	34x97
	Riverglades 02	35.0998	139.2971	pipe culvert	Alu-Tread	34x97
	Riverglades 03	35.0996	139.2968	pipe culvert	Alu-Tread	34x97
Rocky Gully	Rocky Gully 01	35.1133	139.2733	independent screen	reinforced security mesh	26.15x34.14
Spectacle Lakes	Spectacle Lakes 01	34.3337	140.4008	box culvert	weldmesh	46x22
Sweeney's Lagoon	Sweeney's Lagoon 01	34.3718	139.6256	box culvert	weldmesh	21x47
Templeton	Templeton 01	34.0545	140.8195	box culvert	security mesh	52x55
	Templeton 01	34.0540	140.8174	box culvert	security mesh	52x55
	Templeton 03	34.0623	140.8142	box culvert	security mesh	52x55
Wonga Wetlands	Wonga Wetlands 01	36.0666	146.8486	independent screen	weldmesh	50x75

8.2. Appendix 2- Photos of common carp screen designs



Figures showing (a) Alu-Tread Series 13 ®, (b) 45 mm grid mesh, and (c) 31 mm jail bar carp screen designs.

8.3. Appendix 3- Permission to reproduce published chapters

Chapter 2

From: Karl Hillyard [mailto:karl.hillyard@staff.adelaide.edu.au]
Sent: 06 December 2010 01:09
To: G.Petts@westminster.ac.uk
Subject: Conallin et al 2010

Dear Professor Petts

I am enquiring as to whether it is possible to obtain a statement from you in your capacity as editor-in-chief of River Research and Applications, and on behalf of Wiley Publishing, giving permission to include the following manuscript, which has been accepted for publication, as a chapter in my doctoral dissertation.

Conallin, A. J., Hillyard, K. A., Gillanders, B. M., Walker, K. F., and Smith, B. B. (2010). Offstream Movements of Fish during Drought in a Regulated Lowland River. *River Research and Applications* doi:10.1002/rra.1419

Yours sincerely,

Karl Hillyard

From: Geoffrey Petts [mailto:G.Petts@westminster.ac.uk]
Sent: Wednesday, 15 December 2010 4:56 AM
To: karl.hillyard@staff.adelaide.edu.au
Subject: RE: Conallin et al 2010

Dear Karl,

I am happy for you to include you manuscript as a chapter in your thesis subject as always to full acknowledgement.
With all best wishes,

Geoff

Professor Geoffrey E. Petts
Vice Chancellor and Rector
The University of Westminster

From**:** Karl Hillyard [mailto:karl.hillyard@staff.adelaide.edu.au]
Sent: Monday, 6 December 2010 11:39 AM
To: 'aboulton@une.edu.au'
Subject: Hillyard et al 2010

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Hillyard, K. A., Smith, B. B., Conallin, A. J., and Gillanders, B. M. (2010). Optimising exclusion screens to control exotic carp in an Australian lowland river. *Marine and Freshwater Research* **61**, 418–429. doi:10.1071/MF09017

Yours sincerely,

Karl Hillyard

From: Andrew Boulton [mailto:aboulton@une.edu.au]

Sent: Tuesday, 7 December 2010 8:06 PM

To: karl.hillyard@staff.adelaide.edu.au

Cc: Hadley, Caroline (Publishing, Collingwood)

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