

Keynote presentations

The importance of floodplain wetlands to Murray-Darling fish: What's there? What do we know? What do we need to know?

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Abstract

Despite the iconic status of floodplain wetlands and billabongs across the Murray-Darling Basin river system, remarkably little is known of their fish communities. No records exist of community composition prior to European colonisation. Limited palaeo-limnological studies suggest conditions in floodplain wetlands changed significantly in the 1880's or thereabouts, however the first direct records of fish communities date from as late as the 1940-50s.

Knowledge of current communities is geographically fragmented with accessible records being largely restricted to the upper and lower Murray wetlands. Most of the existing information is in the form of unpublished reports and theses. Whilst these records suggest that fish communities within wetlands are greatly altered from their natural state, with widespread dominance of exotic fish species, wetlands of the Murray-Darling system continue to support a variety of native fish species. The presence of several species of high conservation significance, sometimes in high numbers, highlights the importance of these systems for conserving native fish biodiversity.

Understanding of the ecological processes driving fish community structure is limited, and is largely based on a number of relatively recent studies conducted in wetlands along the upper Murray and tributaries. The general absence

of species that are wholly restricted to off-channel wetland systems suggests wetland and riverine fish communities are continuous across the floodplain. Studies of larval fish feeding, habitat use and dispersal indicate low velocity habitats along river channels may be crucial for the recruitment of a number of fish species, suggesting recruitment of native fish may be generally improved where river-floodplain linkages are maintained.

Studies in the upper Murray region indicate that fish communities are structured by a variety of factors including wetland bathymetry, physical and chemical conditions (particularly temperature, oxygen and salinity), microhabitat and food availability, and the presence of exotic fish species. Whilst these variables are likely to be important for the structuring of wetland fish communities across the Murray-Darling Basin, the relative importance of each may vary greatly between and within regions.

The limited research to date indicates that management strategies targeting: (i) maintenance of floodplain wetland habitat diversity; (ii) river and floodplain water regimes and linkage; (iii) riparian and littoral management; and (iv) introduced species are critical for achieving effective management of wetland fish communities.

Expanding our knowledge base of ecological processes beyond the upper Murray region is also crucial for whole-basin management.

Currently, we are poorly placed to confidently predict community and ecosystem response following any significant change to management regimes. There is an urgent need to conduct cost-benefit analyses of various management options to allow for cost-effective targeting of limited resources. Not doing such analyses risks expensive failures of otherwise well-intentioned management interventions. Such analyses require whole-system manipulations and careful monitoring of responses over time frames that are likely to generate longer-term ecological responses (i.e. >1 year).

Introduction

Despite the iconic status of floodplain billabongs and wetlands (defined as off main-channel or floodplain habitats, including ephemeral or permanent habitats such as billabongs, swamps, backwaters, flood-runners, small/shallow natural lakes, etc.) along the Murray-Darling river system, remarkably little is known of the fish communities they support.

To our knowledge, no records exist of the pre-European fish communities; hence we can only speculate as to their exact composition based on present day patterns of distribution and current knowledge of habitat use and life histories.

The little that is known of present-day wetland fish communities has occurred through the specific interests of a few individual researchers, generally with limited funding, rather than as a result of a coordinated broad-scale research program. Consequently, the information that is available is limited in spatial and temporal scope and geographically fragmented. Despite this, some generalisations regarding the past and present state of billabong fish communities and the processes structuring them can be made, and directions for future research and management options identified.

Past and present fish communities

Billabong fish communities are undoubtedly greatly altered from their natural state given the now widespread occurrence of various introduced fish species, at least within the Murray and Murrumbidgee floodplain wetlands

(e.g. King *et al.* 1997; Shirley 2002, McNeil 2004). However, it is difficult to assess the degree of change given that no records of the pre-European fish communities appear to exist. Indirect evidence based on records obtained from sediment cores extracted from billabongs along the Murray, Kiewa, Ovens and Goulburn Rivers indicate that widespread changes in environmental and biotic conditions within billabongs occurred soon after European settlement within the region (Thoms *et al.* 1999; Ogden 2000; Gell *et al.* 2005). Identifying sections of core corresponding with pre- and post-European settlement periods can be conveniently determined by the first appearance of *Pinus* pollen which corresponds approximately with 1875 (Thoms *et al.* 1999; Ogden 2002; Gell *et al.* 2005).

Following European settlement, analyses indicate markedly increased rates of sediment deposition, changes in the composition of sediments deposited, declines in macrophyte abundance as indicated by declines in *Chydorid cladocerans*, and changes in diatom community composition across a widespread area (Thoms *et al.* 1999; Ogden 2000). Such changes are presumed to have been associated with changes to land management and water regimes (Thoms *et al.* 1999; Gell *et al.* 2005).

The introduction of exotic fish species may have also played an early role given that Redfin perch and Tench were released into these systems from the late 1800's onwards (McDowall 1996). The impact of changes to the environmental conditions prevailing in wetlands following European colonisation on native fish communities are unknown. However, given that they indicate significant alterations to the wider physical, chemical and biotic environments within billabongs, changes to wetland fish communities were probably significant and widespread.

The first documented evidence of the composition of fish communities present in Murray-Darling Basin (MDB) wetland systems was collected in the period 1949–50 by J.O. Langtry (Cadwallader 1977). Langtry's collections indicate the extent of the change that had previously occurred with introduced Redfin perch and Tench dominating the large fish community in the limited number of billabongs surveyed. No detail is available



on the composition of small fish species. Further changes have since occurred with the widespread replacement of Tench by Carp through the 1970's and 80's, culminating in the frequent dominance of large fish communities by Redfin perch, Carp and Goldfish in large permanent billabongs, at least along the Murray River and major tributaries (Shirley 2002; McNeil 2004; Conallin *et al.* 2003; McCarthy *et al.* 2003, South Australian Wetland Baseline Monitoring Survey 2005).

Records of commercial fish catch from southern MDB lakes in the 1950's and 1990's suggest similar patterns, although Redfin perch have declined in these larger systems over this period (Table 1). However, despite the considerable changes that have occurred in MDB wetland fish communities, recent studies of these indicate that they remain key habitats for native fish, particularly smaller species (e.g. Shirley 2002; King *et al.* 2003; McNeil 2004).

Despite the lack of direct evidence regarding the pre-European composition of MDB wetland communities, reasonable inferences regarding their composition can be drawn using current regional species lists and known habitat preferences summarised in various texts, papers, theses, reports and personal communications

(Koehn and O'Connor 1990; McDowall 1996; Shirley 2002; King *et al.* 2003; McNeil 2004; Conallin *et al.* 2003; McCarthy *et al.* 2003, South Australian Wetland Baseline Monitoring Survey 2005; Arthington *et al.* 2005; Stephen Balcombe, pers. comm.; Glen Wilson, pers. comm.). Based on a range of records, a total of 23 native species can be considered to be regular inhabitants of wetland habitats across the MDB (Table 2). Five introduced species have established breeding populations in MDB wetlands (Table 2). Whilst our lack of knowledge regarding the natural communities originally present in wetlands limits our ability to restore them to their original condition (even if that was a realistic option), the continued presence of abundant native fish populations across a range of species highlights their importance for the continued maintenance of native fish biodiversity and floodplain community resilience.

The composition of wetland fish communities across the MDB reflects catchment-scale variation related to biogeography and climate. Wetland fish communities across the MDB generally reflect similar patterns in fish diversity observed in riverine fish communities (Gehrke *et al.* 1995, 1999). The highest diversity of native species occurs in the Lower Murray wetlands

Table 1. Southern Murray-Darling Basin lakes commercial finfish catch (%; with 5 and 95 percentile on % catch) for 1957–58 versus 1994–95. Medians and percentiles based on four southern MDB lake 'samples' (data from Reid *et al.* 1997): Lake Cowal (Lachlan River catchment), Lake Victoria (Murray River catchment), Lake Yanga (Murrumbidgee River catchment) and pooled data from Lakes Cawndilla, Menindee and Pamamaroo (Darling River catchment).

Fish species	Median of % catch (with 5th and 95th percentiles; N = 4) by catch year		Proportional change in % abundance based on medians	Increase or decrease
	1957–58	1994–95		
<i>Nematalosa erebi</i>	0 (0–0)	6 (1–7)	from 0 to 6% of catch	increase
<i>Cyprinus carpio</i>	0.2 (0–1)	59 (48–78)	29500%, or 300-fold	increase
<i>Tandanus tandanus</i>	8 (1–15)	0 (0–0)	100%	decrease
<i>Maccullochella peelii peelii</i>	2 (0.2–6)	0.3 (0–0.8)	85%	decrease
Unspecified freshwater fish	0 (0–5)	5 (0.1–39)	from 0 to 5% of catch	increase
<i>Macquaria ambigua</i>	67 (47–83)	15 (2–36)	78%	decrease
<i>Perca fluviatilis</i>	6 (1–14)	1 (0–4)	83%	decrease
<i>Bidyanus bidyanus</i>	10 (1–19)	0 (0–0)	100%	decrease
<i>Tinca tinca</i>	1 (0–18)	0 (0–0)	100%	decrease

Table 2. Status of fish in Murray-Darling Basin wetlands by region, based on reviews of publications cited in text.

A – absent, P – present, Ex – believed extinct, Uncert – status uncertain due to a lack of information.

Family	Species	Common name	Regions			
			Upper Murray	Murray South Australian Border	Lower Murray Wetlands	Northern rivers
Clupeidae	<i>Nematolosa erebi</i>	Bony Bream	A	P	P	P
Galaxiidae	<i>Galaxias maculatus</i>	Common Jollytail	A	A	A	A
	<i>Galaxias rostratus</i>	Flathead galaxias	P	A/Ex	A/Ex	A
Retropinnidae	<i>Retropinna semoni</i>	Australian smelt	P	P	P	P
Plotosidae	<i>Tandanus tandanus</i>	Eel-tailed catfish	Ex	P	Ex	Uncert
	<i>Neosilurus hyrtlui</i>	Hyrtl's tandan	A	A	A	P
Atherinidae	<i>Craterocephalus fluviatilis</i>	Murray Hardyhead	A	A	P	A
	<i>Craterocephalus stercusmuscarum</i>	Flyspecked hardyhead	A	P	A	Uncert
	<i>Atherinosoma microstoma</i>	Smallmouthed hardyhead	A	A	P	A
Melanotaeniidae	<i>Melanotaenia fluviatilis</i>	Crimsonspotted rainbowfish	A	P	Ex?	P
Chandidae	<i>Ambassis agassizii</i>	Olive perchlet	A	P	Ex	P
Percichthyidae	<i>Macquaria ambigua</i>	Golden perch	P	P	P	P
Terapontidae	<i>Bidyanus bidyanus</i>	Silver perch	Ex	P	Ex	P
	<i>Leiopotherapon unicolor</i>	Spangled perch	A	A	A	P
Nannopercidae	<i>Nannoperca australis</i>	Southern pygmy perch	P	P	P	A
Gadopsidae	<i>Gadopsis marmoratus</i>	River Blackfish	P	Ex	Ex	A
	<i>Nannoperca obscura</i>	Yarra pygmy perch	A	A	P	A
Bovichtidae	<i>Pseudaphritis urvilli</i>	Congolli	A	A	P	A
Gobiidae	<i>Philypnodon grandiceps</i>	Flathead gudgeon	P	P	P	P
	<i>Philypnodon sp.</i>	Dwarf flathead gudgeon	P	P	P	A
	<i>Mogurnda adspersa</i>	Southern purple-spotted gudgeon	Ex	P	Ex	Ex?
	<i>Hypseleotris spp.</i>	Carp gudgeon	P	P	P	P
	<i>Pseudogobius olorum</i>	Swan river goby	A	A	P	A
Total extinct			3	12		
Total present			7			
Introduced Species						
Cyrpinidae	<i>Carassius auratus</i>	Goldfish	P	P	P	P
	<i>Cyprinus carpio</i>	Carp	P	P	P	P
Cobitidae	<i>Misgurnus anguillicaudatus</i>	Weatherloach	P	A	A	A
Poeciliidae	<i>Gambusia holbrooki</i>	Gambusia	P	P	P	P
Percidae	<i>Perca fluviatilis</i>	Redfin perch	P	P	P	A?



(originally approx. 18 spp.) where diadromous species, such as Common jollytail and Congolli, contribute to species counts (Table 2).

Warmer regions along the Murray River around the South Australian Border and further north across the Northern Rivers region originally supported about 14 native fish species (Table 2), with warm-water species such as Southern purple-spotted gudgeon and Olive perchlet contributing to diversity. Across the Northern Rivers region, inland species, such as Spangled perch and Hyrtl's tandan, are also important components of the fauna. Wetlands in the cooler upper Murray River region upstream of the confluence with the Ovens Region are naturally the least diverse within the system, supporting about 10 native fish species (Table 2). The lack of historical records means knowledge of the original geographical distribution of some species remains uncertain, hence regional estimates of diversity are somewhat imprecise.

Patterns of regional extinction, native species decline and exotic species distribution reflects anthropogenic impacts related to changes in land-use, water regimes and the original sites of introduction of various exotic fish species. To our knowledge, no fish species have become extinct within the MDB since European colonisation, however regional extinction, range contraction and significant declines in abundance have occurred for several species (Table 2). Range contraction and declines in abundance have been particularly significant in wetlands associated with the Murray River. Native species that are probably extinct in wetlands along the Murray River include Olive perchlet and Southern purple-spotted gudgeon. Eel-tailed catfish are very rare and are also possibly extinct from Murray River wetlands. Flathead galaxias and Southern pygmy perch have also suffered major declines and are now absent from widespread areas of the Murray River floodplain, although they remain locally abundant in some wetlands (e.g. lower Ovens River). Determining the status of rare native species such as Southern purple-spotted gudgeon and Eel-tailed catfish, across the Northern Rivers region is hampered by a lack of data (Stephen Balcombe, pers. comm.; Glen Wilson, pers. comm.). At least three exotic species, Carp, Goldfish and Gambusia, appear to be virtually ubiquitous in lowland MDB wetlands (Table 2), and at

times can dominate the biomass (e.g. King *et al.* 1997; Shirley 2002; McNeil 2004). Redfin perch are largely restricted to the cooler southern regions of the MDB, being particularly abundant in wetlands across the upper Murray River floodplain (Shirley 2002; McNeil 2004). The range of Weatherloach appears to be expanding with records from various locations around the MDB, particularly in the upper Murray region and parts of the Murrumbidgee River catchment (Koster *et al.* 2002; McNeil 2004)

Ecological patterns in Murray-Darling Basin wetland fish communities

Knowledge of the ecological patterns in MDB wetland fish communities is largely restricted to fish communities along the Murray River, and in particular the catchment upstream of Echuca. Elsewhere, data is restricted to baseline surveys of fish community composition across selected wetlands. Studies of larval fish recruitment, diet and habitat use across riverine and wetland habitats in the Murray River and various tributaries (e.g. Humphries *et al.* 2002, King 2004, 2005; King *et al.* 2003), and studies of the composition and habitat use of fish communities in wetlands in Murray River and Ovens River floodplain billabongs (e.g. Lieschke and Closs 1999; Shirley 2002; Stoffels and Humphries 2003; Balcombe and Closs 2004; McNeil 2004) have been completed.

Larval fish recruitment and ontogenetic patterns of habitat use and diet across floodplains

Most species of fish, both native and introduced, occur across a broad range of habitats in both wetland and riverine habitats in the MDB. The lack of a specialist riverine or wetland fauna probably reflects the naturally unpredictable nature of floodplain systems, Australia's dry climate and recent marine origins of the native fish fauna (Darlington 1957; Gehrke and Harris 2000). Additionally, in unregulated and unmodified river systems, riverine and off-channel wetland/billabong systems naturally inter-grade rather than forming discrete divisions, and low velocity environments may be abundant even along river channels

(King 2004). However, some differences in the distribution of fish species across riverine and wetland habitats appear to exist between the northern and southern parts of the MDB.

In the northern MDB, there tends to be little difference between species richness in main channel habitats compared to off-channel waterbodies (S. Balcombe unpubl. data). In the Warrego River catchment, a largely unregulated system with limited floodplain development, off-channel habitats may support the same species as main channel (but often in higher abundance), including Silver perch, Golden perch and Eel-tailed catfish. In contrast, in the southern MDB, whilst most species can be found in both riverine or wetland habitats at various times, most are typically more abundant in one or the other of these types of environment. Only Murray cod appears to be exclusively restricted to riverine habitats across all stages of its life history (Humphries *et al.* 2002; King 2004; Koehn and Harrington 2005).

Differences in the distribution of fish species across floodplains of the northern and southern MDB suggest river regulation, floodplain development and introduced fish species may be having significant impacts on wetland fish communities (see also Humphries *et al.* 2002). Reasons for the general absence of large-bodied native fish species (particularly Silver perch, Golden perch and Eel-tailed catfish) from southern MDB wetlands are unclear. The regular occurrence of (usually) low numbers of Golden perch and the occasional presence of other fish species including Silver perch, River blackfish and Eel-tailed catfish in wetlands suggests they could become more abundant in wetland habitats if their abundance across floodplain systems was generally increased. King (2004) found that low velocity environments, such as backwaters and still littoral habitats, are important larval rearing environments for a wide variety of species present in the Broken and Campaspe Rivers. Their use by larval fish may be related to the availability of small prey items that can be abundant in such habitats (King 2005), and the loss of such environments along modified river channels has been suggested as a possible contributing factor to recruitment failure (Humphries *et al.* 2002; King *et al.* 2003; King 2005). Similarly, the absence of small-bodied species, such as Southern pygmy perch and

Flathead galaxias, that are typically associated with wetlands may also reflect the loss of low velocity, weedy, shallow habitats that are preferred by these species (see McNeil 2002), but are generally absent or severely degraded along regulated river channels and on intensively developed floodplains.

Comparing fish community structure across floodplain wetlands

Knowledge of the ecological patterns and processes determining wetland/billabong fish assemblages is limited and largely restricted to studies along the Murray River in northeast Victoria and southern New South Wales. Completed studies provide some information on responses of fish assemblages to key floodplain environmental gradients, patterns of fish habitat selection and use across a range of billabong types and microhabitats, and the impact of some introduced species. However, any inferences drawn must be tempered by the limited spatial and temporal scope of these studies. Strictly, they are specific only to the billabongs and wetlands of the upper Murray, although similar patterns and processes (albeit on a slightly different species complement) are likely to operate to varying extents across the MDB. Other factors, such as salinity, may also drive community patterns in certain parts of the MDB (see Gell *et al.* 2005, Meredith *et al.* 2005).

Fish communities in billabongs on the floodplain of the Upper Murray and Ovens River systems exhibit considerable variation in composition and patterns of dominance despite a relatively limited species pool (Shirley 2002; McNeil 2004). Variation in the communities present in these systems appears to be strongly related to billabong size and bathymetry which in turn influences a wide range of key abiotic and biotic variables (see Jackson *et al.* 2001 for review of roles of biotic and abiotic factors structuring lake fish communities).

In large permanent billabongs, physical and chemical variation is buffered by the volume and depth of water, hence temperature and oxygen levels rarely become lethal for large bodied fish species (Jackson *et al.* 2001; Shirley 2002). Consequently, such communities often support high abundances of large-bodied, relatively long-lived fish species. In the upper Murray region, Redfin perch, Goldfish and Carp



overwhelmingly dominate the large fish community. Small native species, particularly Australian smelt and carp gudgeons, are also very abundant within certain habitats (see Lieschke and Closs 1999; Balcombe and Closs 2004). The environmental conditions and species composition of large billabongs is highly consistent, both spatially and temporally, over large areas of the mid to upper Murray floodplain. This assemblage structure has probably been relatively stable since Carp replaced Tench as the dominant cyprinid fish in the 1970's (Shirley 2002). Similar assemblages dominated by Carp, Goldfish and Redfin perch are also found in the large billabongs on the Ovens River floodplain (McNeil 2004).

In billabongs along the lower Murray floodplain, a similar large fish assemblage is observed in large billabongs along with abundant Australian smelt and carp gudgeons, although Redfin perch tend to be relatively less abundant and Golden perch relatively more abundant (Unpublished Data; Murray-Darling Freshwater Research Centre – Lower Basin Laboratory; Connalin *et al.* 2003). Abundances of smaller fish species may exhibit greater variation between and within years due to their shorter life cycles (e.g. annual life cycle for smelt) and variable recruitment success from year to year (Lieschke and Closs 1999; Shirley 2002; Balcombe and Closs 2004).

In contrast to the relative stability of large billabongs, physical and chemical conditions in smaller wetland and lentic systems can be highly variable both spatially and temporally (McNeil 2004). Relative to the generally larger billabongs systems still present on the Murray River floodplain, fish communities exhibit considerably greater variation both in terms of species composition and relative abundances in the comparatively small floodplain wetlands present on the lesser-developed lower Ovens River floodplain (McNeil 2004).

In the Ovens River floodplain system, high summer water temperatures and hypoxic conditions can eliminate intolerant species, most significantly large piscivorous such as Redfin perch. In the absence of Redfin perch, smaller native and introduced fish species can be abundant although the actual composition will be determined by various factors including species complement, microhabitat availability and physical and chemical conditions. Under

conditions of severe environmental harshness such as near-desiccation, high temperature or hypoxia, communities are likely to be dominated by introduced *Gambusia* with a small component of native carp gudgeons and occasionally other tolerant, small bodied species such as goldfish or Southern pygmy perch. Under intermediate conditions, environmental pressures such as hypoxia may be temporarily severe, but overall environmental harshness is intermediate (i.e. pools have high permanence, depth and cooler temperatures). Under these conditions, environmental impacts may be sufficiently severe to eliminate Redfin perch but not so harsh so as to eliminate the smaller bodied less hypoxia tolerant native species. Under such conditions, significant populations of native Australian smelt, Southern pygmy perch and Flathead galaxias may occur (McNeil 2004). Low oxygen refugia from piscivory have previously been described for cichlid-Nile perch assemblages in Lake Victoria (Chapman *et al.* 1995; Chapman *et al.* 2002). Currently, little is known of the fish community structure or dynamics in smaller wetland systems elsewhere in the MDB.

Microhabitat partitioning within billabongs

Microhabitat partitioning of fish species and ontogenetic shifts in habitat use are widely evident across various wetland systems in the upper Murray region (Jones 1996; Stoffels 1998; Lieschke and Closs 1999; Balcombe 2002; Shirley 2002; Stoffels and Humphries 2003). In larger billabongs, several fish species including Carp, Goldfish and juvenile Redfin perch associate strongly with woody debris and littoral macrophytes (Jones 1996; Syarifuddin 2000; Balcombe 2002; Shirley 2002; Balcombe and Closs 2004). Abundances of carp gudgeons in particular can be extremely high in littoral macrophyte beds (Balcombe and Closs 2002, 2004). In smaller billabongs on the Ovens River floodplain, carp gudgeons, Southern pygmy perch and *Gambusia* are strongly associated with macrophytes, particularly if piscivorous Redfin perch are present within the system (Stoffels 1998; Stoffels and Humphries 2003; McNeil 2004). Such structurally complex habitats are likely to provide refuges from predators and food resources. Indeed, the availability of food within these habitats may be the critical factor determining their use as fine scale

variation in carp gudgeon abundance within macrophyte stands is closely related to biofilm and chironomid density on macrophyte stems (Balcombe 2002). Furthermore, architectural complexity and density of plant stems does not appear to be an important driver of carp gudgeon abundance within macrophyte stands (Balcombe and Closs 2004).

Considerable diel and ontogenetic variation in habitat use has also been observed amongst various fish species in billabongs. In Owens River billabongs, gudgeons become increasingly benthic as they grow, a pattern that may be influenced by competition for space and food with *Gambusia* (Stoffels and Humphries 2003). Over shorter temporal scales, Australian smelt exhibit strong diel vertical migration in areas of open water, remaining close to the bottom during the day but spreading vertically throughout the water column by night (Lieschke and Closs 1999). This diel variation in habitat use may be driven by a combination of avoidance of visually feeding Redfin perch and variation in the abundance of zooplankton prey which also exhibit significant diel vertical migration (Lieschke and Closs 1999).

Experimental studies examining the role of fish in Murray-Darling Basin wetlands

Experimental studies examining the role of fish in the ecology of MDB wetlands are limited, and have focused on native carp gudgeons and introduced Carp and Redfin perch. Studies of carp gudgeon have examined the role that food and refuge availability play in determining their distribution within macrophyte stands (Balcombe 2002) and the impact of their predation on zooplankton (Nielsen *et al.* 2000) and benthic invertebrate communities (Jones 1996; Syarifuddin 2000). Experimental studies examining the role of Redfin perch in wetlands have examined their impacts on zooplankton and benthic invertebrates (Shirley 2002). Separate experimental studies of carp have examined their impact on physical and chemical conditions, macrophytes, macroinvertebrates, zooplankton and Redfin perch in artificial and constructed floodplain ponds (Roberts *et al.* 1995; Shirley 2002; Driver 2002; Driver *et al.* 2005b) and natural billabongs (King *et al.* 1997; Robertson *et al.* 1997). Details of those studies are reviewed elsewhere in this volume (see Driver *et al.* 2005c).

Carp gudgeons are virtually ubiquitous throughout the MDB (McDowall 1996), and frequently occur in exceptional abundances in littoral habitats (Balcombe 2002; Balcombe and Closs 2000, 2004; McNeil 2004). Descriptive and experimental studies of carp gudgeon distribution and abundance clearly indicate that 'bottom-up' food-web processes play a major role in their ecology (Balcombe 2002; Balcombe and Closs 2004). The impact of carp gudgeon predation on invertebrate communities can be significant, albeit variable. Carp gudgeon predation alters the composition of benthic invertebrate communities on woody substrates, although their degree of impact is strongly mediated by the surface complexity of decaying wood (Jones 1996; Syarifuddin 2000). In contrast, Nielsen *et al.* (2000) found that the impact of carp gudgeon predation on zooplankton in small experimental billabongs to be relatively small. The impact of Redfin perch predation on the zooplankton community in a floodplain pond enclosure experiment was variable through time, with a significant negative impact on *Daphnia* sp. (Shirley 2002), a pattern consistent with studies of percid predation elsewhere (Post and McQueen 1987).

Implications and research needs for management

Realistically, MDB wetland and billabong fish communities cannot be restored to their original conditions. Even if we knew what those original conditions were, introduced fish and macrophyte species, altered water regimes and eutrophication will continue to be intractable problems. However, accepting such constraints on ultimate management aims, we can still realistically explore management approaches that have the potential to maintain and even enhance conditions. Of crucial management significance is recognising that even in their present highly modified form, MDB wetlands support highly significant native fish communities, both in terms of abundance and diversity. The limited research to date indicates that critical areas of understanding required for effective management of native wetland fish species include; (i) maintenance of floodplain wetland habitat diversity; (ii) impacts of river and floodplain water regimes; (iii) maintenance



of floodplain, riparian and littoral conditions and; (iv) impacts and cost-effective control of introduced species (particularly perch, Gambusia and Carp).

Maintenance of floodplain wetland habitat diversity

The diversity of wetland habitats across a natural floodplain is considerable, ranging from ephemeral shallow pans to large permanent ox-bow lakes. Numerous wetland habitats may also remain connected to main river channels to varying degrees. At least for the Ovens floodplain, distinct fish communities associated with these various floodplain wetland types occur (McNeil 2004). Given that many fish species are typically most strongly associated with only one or two types of wetland, maintaining the full range of habitats across the floodplain is critical for the full complement of wetland fish species (Shirley 2002; McNeil 2004). In particular, the loss of shallow wetland pans is a conspicuous feature of floodplains where intensive vegetation clearance and agricultural development has occurred or where river regulation ensures flooding is infrequent (G. Closs, S. Balcombe, P. Driver, D. McNeil, M. Shirley, pers. obs.). Such habitats are typically macrophyte-rich and warm temperatures or hypoxia restrict the abundance of large piscivores such as Redfin perch, creating conditions where small native species can persist (McNeil 2004).

Loss of wetland diversity across the floodplains is an issue throughout the MDB, irrespective of the processes driving such losses. In the northern part of the Basin (e.g. the upper Darling River), perhaps the greatest impacts on wetland fish assemblages is the loss of connectivity across in-channel and off-channel waterholes, floodplains and wetlands brought about by floodplain alteration and hydrological manipulation for agricultural needs (S. Balcombe, pers. obs.). However, published data on the impacts of these management practices are currently lacking. In the Lachlan River the combination of flow regulation and land management, which has depleted soils and vegetation that retain moisture, has resulted in the almost-complete desiccation of even the largest, once permanent wetlands. The impact of these changes on the Lachlan's fish assemblage is potentially immense, but again is largely unmeasured. It is also highly

likely that the interaction of regulation and siltation has resulted in the loss of macrophyte-rich refuge pools in rivers during low flow periods (see Sainty and Roberts 1996).

To effectively understand and manage wetland diversity appropriate for specific regions, we need quantitative assessments of existing fish communities and the wetland features with which they are typically associated. Without such data, regional management of fish communities will be largely based on speculation and anecdote.

Managing water regimes for wetland fish communities

The water regime clearly determines a number of key factors structuring floodplain fish communities. Across floodplains, flood return frequency and magnitude determines the frequency of filling, drying and finally desiccation, if flood return times are sufficiently infrequent. Observations from the Ovens floodplain would suggest that rainfall alone is insufficient to maintain water levels in shallower pans, hence maintenance of such habitats is critically dependent on filling as a consequence of large floods that spill out from channels and across floodplains (McNeil 2004).

Flood timing and magnitude also plays a crucial role in dispersing both native and introduced fish across the floodplain (McNeil 2004). At present there is no information that would allow modelling of flood timing, return times and magnitudes required to maintain the full diversity of floodplain fish communities and wetlands (particularly relationships between filling and drying) specific for the requirements of fish communities across floodplains (akin to models developed for bird nesting and water allocations for the Lachlan River (Driver *et al.* 2005a). The lack of such knowledge has serious implications for continued effective management of fish diversity where increased river regulation is proposed (e.g. Ovens River), or where improvements to current water regimes are sought.

Water levels within billabongs may also be controlled more directly by river regulation through direct connection or ground water exchange (Balcombe 2002; Shirley 2002). Sudden changes in river level due to the management of flows for irrigation can produce



time-lagged variation in adjacent billabongs with severe consequences for littoral fauna (Balcombe 2002; Balcombe and Closs 2004). Impacts of flow manipulation on littoral fauna in turn have flow-on effects to billabong food-webs with associated impacts on littoral fish assemblages (Balcombe and Humphries in press). During low flow periods, extensive dewatering of littoral vegetation may occur for extended periods, thus depriving littoral macrophyte-dependent fish of critical habitat (Balcombe and Closs 2004). These low flow periods may differ from the natural cycle of wetting and drying as irrigation management has reversed the seasonality of flows to rivers and wetlands. High flows can be also detrimental if they also occur out of season. For example, an extended period of elevated flow coincided with a near total failure of carp gudgeon recruitment in Dugay's Billabong, possibly due to either low temperatures or an absence of key food resources (or both) (Balcombe and Closs 2004).

Rapid changes in river flow and water levels in adjacent billabongs may also occur throughout the irrigation season and may totally inundate littoral habitats and then leave them exposed with little time for flora or fauna on woody substrates or in macrophyte stands to cope with the disturbance (Syarifuddin 2000; Balcombe 2002). Currently, apart from Balcombe and Humphries (in press), information on the impact of river level variation on microhabitat quality in billabongs is lacking, and hence the short to medium-term impact of different water regimes on habitat availability and quality for wetland fish communities is largely unknown.

Long-term trends in wetland environmental conditions

Paleolimnological evidence indicates European settlement has had long-term impacts on the environmental conditions present within wetlands across the MDB, resulting in increased turbidity and sedimentation and declines in macrophyte abundance (Thoms *et al.* 1999; Ogden 2000; Gell *et al.* 2005). Significantly, increased sediment deposition and turbidity and declines in macrophyte abundance are all environmental impacts commonly attributed to the invasion of Carp (Roberts *et al.* 1995; Koehn 2004), yet significant declines in these

environmental indicators appears to have commenced long before the spread of Carp occurred in the late 1960's and 70's (see Koehn 2004).

Changes in wetland conditions associated with European settlement suggest the agricultural development of floodplains and altered water regimes had a significant impact on wetland systems from the late 1800's (Thoms *et al.* 1999; Ogden 2000; Gell *et al.* 2005). Negative relationships between the ecological condition of riparian habitats and grazing intensity have been quantitatively demonstrated along the Murrumbidgee River (Jansen and Robertson 2001). Regular incursion and damage by cattle into riparian and littoral habitats occurs in wetlands in the upper Murray region (G. Closs, S. Balcombe, P. Driver, D. McNeil and M. Shirley, pers. obs). Given the importance of littoral habitat condition for many small-bodied MDB wetland fish species (Balcombe 2002; Shirley 2002; Stoffels and Humphries 2003; Balcombe and Closs 2004; King 2004; McNeil 2004), the potential for severe impacts is high. However, at present we have limited detailed and direct knowledge of agricultural impacts on wetland fish communities. Obvious research needs are exploring impacts of grazing exclusion/modification and other agricultural impacts on fish and littoral community dynamics, at both the microhabitat and whole billabong levels of scale.

Establishing the role and impacts of introduced fish species in wetlands

The limited existing research clearly indicates that Carp, Redfin perch and *Gambusia* play significant and different roles in structuring present-day environmental conditions and fish communities in MDB wetlands (King *et al.* 1997; Robertson *et al.* 1997; Shirley 2002; Stoffels and Humphries 2003; Driver 2002; McNeil 2004; Driver *et al.* 2005b). The impact of any control attempts are complicated by interactions between these species, meaning that control or eradication of one introduced species could conceivably benefit one of the other introduced species or negatively impact on a native species (Shirley 2002; McNeil 2004). Eradication of Redfin perch could conceivably benefit *Gambusia* through reduced predation (Stoffels and Humphries 2003; McNeil 2004). Reductions in Carp may reduce turbidity, thereby benefiting



Redfin perch (Shirley 2002; Driver *et al.* 2005b) and having negative impacts on open water planktivorous fish such as Australian smelt (Lieschke and Closs 1999). Further, although introduced fish are a conspicuous management problem in many wetlands, successful control may not result in the expected benefits if other factors (adverse water regimes, riparian grazing, etc.) are creating and maintaining poor habitat quality. At present, due to the limited research completed, we have little understanding of the actual outcomes that may result following pest fish control.

Establishing cost-effective approaches to wetlands management

Clearly, multiple interactive factors control the MDB wetland fish communities. As a result, management approaches that focus on a single variable run the risk of producing unexpected and unpredictable outcomes. The lack of understanding as to relationships between controlling variables means we are in a poor position to accurately predict the outcome of changes to management responses.

Environmental responses to the alteration of a single environmental variable may range from highly positive to neutral to highly negative, the latter two having the potential to severely erode public confidence in the ability of agencies to manage and spend conservation dollars effectively.

Some wetland rehabilitation approaches have acceptably predictable benefits and should be maintained, but their effectiveness and the predictability of management effects would be greatly improved by adopting a more integrated approach to land and water management.

The restoration of natural flow regimes, which includes increasing the frequency and duration of inundation have demonstrable benefits. Flow regulation and environmental flows respectively decrease and increase the frequency of swamp and billabong inundation at or above the level of aquatic plant ('fish habitat') establishment, and increases in inundation frequency can increase aquatic plant species richness within billabongs (Chessman *et al.* 2003; Driver *et al.* 2005a). However, integration of large-scale flow manipulation with localised land and water management is required. Fish species that are dependent on complex habitat structure

(e.g. carp gudgeons) may not benefit from increased environmental flows if disadvantaged by local agricultural practices. For example, the abundance of even hardy aquatic macrophytes such as spike rush and lignum are reduced under grazing (Craig *et al.* 1991; Blanch and Brock 1994).

Modelling of wetland habitat quality and native fish recruitment is being developed, with both expert knowledge and empirical models (e.g., ARIDFLO and DRY/WET, see Puckridge *et al.* 2000 and Costelloe *et al.* 2002; IMEF, see Chessman *et al.* 2003 and Driver *et al.* 2005a and MFAT, see Young *et al.* 2003). These programs include models that qualitatively or quantitatively describe how wetlands will respond to large-scale water management changes. For example, the Murray Flow Assessment Tool (MFAT) works on the broad premise that Australian smelt, Bony herring, carp gudgeons, Southern pygmy perch, hardyheads and Flathead galaxias spawn and recruit in floodplain wetlands and lakes, anabranches and billabongs during in-channel flows, and the extent of this recruitment is dependent on other measures of habitat quality, such as timing and water temperature (Young *et al.* 2003). These models have not yet examined the effect of long-term flow regimes or the recruitment of alien wetland species. The number of individuals that achieve recruitment to a reproductive age over several years or decades is a more meaningful measure of long-term population viability (Driver *et al.* 2005d).

Cost-benefit analyses (Choquenot *et al.* 2004), and related management tools such as risk assessment (e.g., Hart *et al.* 2002) are fundamental tools for assessing the best use of resources in ecosystem manipulation (e.g., the carp control model for a wetland in Choquenot *et al.* 2004). It is critically important such models are also applied to long-term and large-scale human economy-ecosystem responses.

Wetland health responses to changed management regimes are typically modelled in isolation to the long-term effects on the human economy (e.g., flow regime change effects on agriculture and wetland ecosystems over a 100 year period; LRMC 2002; Driver *et al.* 2005a). Integration of such models would better illustrate ecosystem-economy trade-offs for laypersons on natural resource

management committees, authorities and the like. Importantly, an economy-ecosystem model allows the integration of improving ecosystem services (Constanza *et al.* 1997) into the economic component when wetland health improves (and *vice versa*). Moreover, it is likely that if the full ecosystem-economy dynamic is properly understood then it would be difficult to argue that (for the benefit of the entire human community) complex vegetation communities that improve carbon and nutrient cycling, salt intrusion etc., but also benefit native fish, should not be maintained.

A strategy is also urgently required to assess and fund ecological research that will better predict where management can be effective. Whole- and part-billabong manipulations over the temporal scales appropriate to measure likely ecological responses are crucial for effective cost-benefit evaluation of management strategies. Such experimental approaches would be most informative if they were integrated with larger scale wetland management approaches. For example, the delivery of environmental flows to wetlands along inland NSW rivers is effectively a large-scale experiment which has demonstrated that individual wetland responses are highly variable (Chessman *et al.* 2003; Driver *et al.* 2005a).

The integration of numerous scales of research, and research with (unavoidably) diminishing experimental rigour with increasing scale, is usually the best approach towards understanding ecosystem processes at realistic ecological scales (James and McCulloch 1990). The common responses to such calls for extra research is that:

- (i) it takes too long to complete such studies given the urgency of the problems we face; and
- (ii) why study responses to manipulations if we already know the factor of interest is having an adverse impact.

The response to such claims is that we have limited dollars to spend on management and it is crucial that those dollars be effectively targeted. Secondly, we have little idea of the dynamics of responses to manipulations given the complexity of wetland interactions and unpredictable and potentially adverse outcomes are a real risk in the current knowledge vacuum.

Within wetlands there are very complex interactions between water quality, physical habitat, invertebrates and fish (e.g., King *et al.* 1997; Shirley 2002; Balcombe 2002, Driver *et al.* 2005b; Lieschke and Closs 1999; McNeil 2004). The responses of individual wetlands to large-scale alterations to management regimes over longer time periods would be far better understood if we had maintained integrated wetland study programs over the past ten years. Even with relatively modest research funding, we would be in a more informed position to understand the impact of altered management regimes on long-term wetland ecosystem dynamics and therefore effectively tackle wetland management and restoration than we currently are.

Acknowledgements

Thanks to Janet Pritchard for providing PDD information on AridFlo.

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