Impact of invasive exotic fishes on wetland ecosystems in the Murray-Darling Basin

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Abstract

Wetlands can harbour significant populations of exotic fishes which may be a key source of colonists for the broader catchment. There are at least eleven species of exotic fish in the Murray-Darling Basin, and potential for several other species to colonise its warmer regions. Five species are routinely encountered in wetlands, and five others may occur in localised or low abundances. While these attract a variety of public opinions, based on perceptions of ecological impact and human utility, most are likely to have some impact on wetland ecosystems. Such impacts may be categorised as (1) direct predation, (2) habitat modification, (3) disease or parasite introduction, and (4) resource and interference competition.

From current knowledge, Carp, Redfin and Mosquitofish represent the principal threats to Murray-Darling Basin wetlands, while lesser impacts may result from Goldfish and Trout. Neither Tench or Roach are thought to have a significant effect on Murray-Darling Basin wetlands. There is insufficient field knowledge of Oriental weatherloach to gauge its impact on Australian habitats, although recent range expansions and occasional high densities point to the need for urgent assessment. Overseas and experimental studies of weatherloach suggest possible reductions in water quality and macroinvertebrate assemblages and its introduction of at least one parasite. Tilapia could significantly reduce macrophyte, macroinvertebrate and fish assemblages and affect water quality if introduced to the Basin. Translocated native fishes such as Banded grunter may also pose a threat in the future.

The main knowledge gaps reducing our capacity to manage the threats to wetland from exotic fishes are (1) understanding the mechanisms and scales of dispersal and colonisation, (2) understanding the impact of smaller species such as Mosquitofish and Oriental weatherloach, and (3) establishing decision support mechanisms for selecting appropriate control strategies. Little effort to date has been directed at assessing impacts, or the efficacy, of specific control strategies other than for Carp. Sound public education will be critical in managing impacts of these species, as will integration with other interventions such as flow-regime improvements.

Introduction

Off-channel, wetland ecosystems are a key habitat for freshwater fishes (e.g. Copp, 1989; Schlosser, 1991; Wilson and Wright, 2005; Closs *et al*., this volume). For stage-structured populations, wetlands can provide spawning habitat, nursery refuge during early life history (Morton *et al*., 1995) and act as a key source of dispersive offspring to the broader catchment (Stuart and Jones, 2002; cf beaver ponds in the US: Schlosser, 1995, 1998). They can also represent a key habitat throughout the lifecycle of small species such as Glass perch (Ambassidae) or rainbowfish (Melanotaeniidae) (Allen *et al*., 2002), and contribute energy (carbon) or food items to nearby riverine populations during periods of floodplain inundation (Boulton and Jenkins, 1998; Hillman, 1998; McGinness *et al*., 2002). However, these properties may apply equally to the population maintenance of exotic species (e.g. Stuart and Jones, 2002). Determining the links between wetlands and introduced species is a key step in any effort to manage their abundances and environmental impacts.

A range of exotic fishes has become established throughout the Murray-Darling Basin, including wetland ecosystems (Clunie *et al*., 2002). They reflect a variety of life history modes (Growns, 2004) and foraging behaviours and trophic levels (Kennard *et al*., 2001). Numerous reviews have examined the status of these species in Australia, focusing either on Carp (Roberts and Ebner, 1997; Braysher and Barrett, 2000; CCCG, 2000; Koehn *et al*., 2000) or a broad range of species (Braysher, 1993; Arthington and Blüdhorn, 1995; Arthington and McKenzie, 1997; Arthington *et al*., 1999; Clunie *et al*., 2002).

Nevertheless, with few exceptions, past studies have tended to emphasise the population ecology, environmental impact or control potential within channel environments. The issue of wetland impacts has received little consideration, despite the acknowledged role of wetlands in the life history of riverine fishes.

This review assesses the current knowledge of impacts from exotic fishes on wetlands, and also examines the management risks for controlling pest fish impact and the associated priority knowledge gaps. It suggests that environmental variability, difficulties with isolating the impacts of exotic fishes from those of other stressors, a paucity of wetland-specific fish data, phenotypic plasticity in exotic species, and the potential for future problems from 'sleeper species' represent key barriers to establishing a greater understanding of this issue. This paper uses as a working definition of a pest species, *"Any species that has invaded habitats within the MDB where it does not naturally occur and that is known or suspected to be detrimental in some way, which may include environmental, economic and social impacts"* (Clunie *et al*., 2002). It focuses on the impact of such species on either ephemeral or permanent 'off main-channel' or floodplain habitats, including billabongs, swamps, backwaters, flood-runners and small/shallow natural lakes.

Species with a wetland distribution

Five exotic fishes are routinely encountered within Murray-Darling Basin wetlands, namely Carp (*Cyprinus carpio*), Goldfish (*Carassius auratus*), Mosquitofish (*Gambusia holbrooki*), Redfin or English perch (*Perca fluviatilis*) and Oriental weatherloach (*Misgurnus anguillicaudatus*) (Roberts and Ebner, 1997; Allen *et al*., 2002; Clunie *et al*., 2002;). Of these, the latter two species are largely restricted to southern catchments (Allen *et al*., 2002) although Redfin also occur in upland tributaries of northern catchments such as the Border Rivers or Gwydir (Faragher and Lintermans, 1997; NSW DPI, unpubl. data). Clunie *et al*. (2002) ranked invasive fishes (both current Murray-Darling Basin species and those with the potential for future establishment) on the basis of stakeholder perceptions of threat to aquatic habitats (Table 1.). These five 'wetland species' were ranked the highest in their perceived threat to basin aquatic habitats.

A further three species (Rainbow trout, *Onchorhynchus mykiss*; Tench, *Tinca tinca*; Roach, *Rutilis rutilis*) may occur in localised or minor wetland abundances (Clunie *et al*., 2002).

Table 1. Stakeholder ranking of threat from exotic fishes on Murray-Darling Basin aquatic ecosystems. Taken from Clunie et al. (2002).

Species	Ranking
Carp, *Cyprinus carpio*	1
Mosquitofish, *Gambusia holbrooki*	2
Goldfish, *Carassius auratus*	3
Redfin, *Perca fluviatilis*	4
Weatherloach, *Misgurnus anguillicaudatus*	5
Brown trout, *Salmo trutta*	6
Tilapia, *Oreochromis mossambicus*	7
Rainbow trout, *Onchorhynchus mykiss*	8
Tench, *Tinca tinca*	9
Roach, *Rutilis rutilis*	10
Guppy, *Poecilia reticulata*	11
Atlantic salmon, *Salmo salar*	12
Brook trout, *Salvelinus fontinalis*	13

Tilapia (*Oreochromis mossambicus*) and Guppy (*Poecilia reticulata*) are thought to have the potential to colonise Murray-Darling Basin wetlands should they become established in the system. Both are currently known from coastal Queensland catchments (Arthington and McKenzie, 1997; G. Wilson, pers. obs.). A number of other exotic aquarium species have also established populations in the same catchments (Arthington *et al*., 1999), with the potential to invade northern Murray-Darling Basin habitats should further introductions occur.

Beyond these is the issue of translocation of native species beyond their natural range. For instance, the tropical Banded grunter (*Amniataba percoides*) has established strong populations in dams on the Brisbane River, and has become well known for its high densities in shallow macrophyte areas and a voracious appetite (Brian Dare, pers. comm.). Other species such as the Sooty grunter (*Hephaestus fuliginosus*) and Sleepy cod (*Oxeleotris lineolatus*) have also been translocated outside their natural range within Queensland (Clunie *et al*., 2002), largely for angling purposes. These species would all be expected to have a significant impact on wetland ecosystems of the northern Murray-Darling Basin should they be translocated into the region.

Barriers to assessing impact risk

A number of factors make the task of determining the risk to Basin wetlands from exotic fishes a difficult one. They represent a combination of the intrinsic properties of wetlands themselves, the modified landscape in which they now exist, the availability of appropriate monitoring data, and the biological characteristics of pest species generally. Collectively, they illustrate the complex nature of quantifying (or even qualifying) wetland impacts from exotic fishes and, therefore, determining the most appropriate control strategy to adopt.

Most river systems throughout the Murray-Darling Basin reflect highly variable and often unpredictable patterns of rainfall and hydrology (e.g. McMahon *et al*., 1992; Walker *et al*., 1995; Puckridge *et al*., 1998; Thoms and Parsons, 2003). This will have a subsequent effect on the condition of wetlands through their connectivity with nearby river channels or the extent to which they receive local rainfall runoff. Wetlands can cycle through a diverse range of ecological states in between inundation and drying (e.g. Scholz *et al*., 2000, 2002). Their condition at any point in time will in part reflect antecedent patterns of local wetting and drying and the biotic responses to this (e.g. faunal recruitment) at a range of spatial scales. Along with 'background' patchiness in parameters such as riparian vegetation or macrophyte cover and soil type, this may mask the localised impacts from exotic fishes.

In agricultural landscapes such as those of much of the Murray-Darling Basin, reductions in wetland condition could result from a number of stressors. In addition to exotic fishes, these may include flow regulation, weed infestations, livestock grazing, draining or levee construction, pesticide contamination, natural disturbances such as droughts, and other exotic fauna such as feral pigs (various references in Williams, 1998). Under such a complex disturbance regime, it will be difficult to isolate causal links between exotic fishes and wetland condition from the influence of other factors. This underscores the value of carefully planned field experiments for determining impact agents and restoration success in wetlands (Chapman and Underwood, 2000).

Relatively few field data exist on the aquatic impacts of exotic fishes in the Murray-Darling Basin, especially those describing wetland-specific patterns. Indeed, few widespread surveys of wetland condition have been undertaken across the Basin, particularly for fish assemblages (Closs *et al*., this volume; but see SKM and MDFRC 2004 for South Australia's River Murray valley). These are an urgent prerequisite to a deeper understanding of impacts by exotic fishes, and should be considered a high priority. Attempts to assess both past and potential wetland impacts from exotic fishes are hampered by a lack of field data, and much of our current understanding has arisen from overseas, laboratory or pond studies. Adequate field information would allow more refined estimates of the occurrence and spread of exotic species, highlight sites of particular conservation concern, and provide a reference point against which to gauge future impacts ad spread. Again, the need for rigorous field experiments in describing impacts cannot be overstated.

A further issue is the inherent adaptability or phenotypic plasticity of pest species (Courtenay and Hensley, 1980; Moyle, 1985; Ehrlich, 1989; Rejmánek and Richardson, 1996; Rosecchi *et al*., 2001). Although species such as Carp and Redfin have relatively well defined geographical distributions within the Basin (Driver *et al*., 1997; Faragher and Lintermans, 1997), they display a degree of flexibility in traits such as diet and growth rate across their range. Under some circumstances, this may make it difficult to establish a consistent pattern of impact. For example, redfin growth can become stunted at high population densities (Cadwallader and Backhouse, 1983; McDowell, 1996b), which may lead to varying levels of impact given the likely scaling of body size with foraging intensity or dietary composition.

Species with an unrealised potential for impacting Basin wetland ecosystems are a further barrier to determining impact risks from exotic fishes. Such 'sleeper species' may be exotic species at significant risk of invading Basin habitats from elsewhere in Australia or else species that have already escaped into the Basin and either remained undetected or are thought to exist in only minor abundances. Species established in south-east Queensland catchments such as Red devil (*Amphilophus citrinellus*), Tinfoil barb (*Barbodes schwanenfeldii*) or Tilapia would all fall into this category (Arthington *et al*., 1999), along with translocated native fishes such as the Banded grunter.

Impact categories

Four categories of wetland impact from exotic fishes are considered in this review. First, Direct *predation on macrofauna* is taken to include predation on native fish or larger invertebrates such as decapod crustacea. Second, *Habitat modification* includes disturbance to structural elements such as the benthic environment or macrophyte beds. Third, *Disease or parasite introductions* includes the harbouring and transmission of bacterial, viral or fungal pathogens and a variety of parasite taxa. Lastly, *Resource and interference competition* covers more cryptic impacts such as behavioural interference towards other fishes, consumption of zooplankton or benthic algal assemblages, and the associated potential for creating energetic blocks within aquatic food webs. While there are also potential economic and social impacts from these species, these are beyond the scope of this review and will not be examined further.

Impacts on Murray-Darling Basin wetlands

Direct predation on macrofauna

A number of exotic fishes have been implicated in reductions of wetland macrofauna, either through direct predation or other antagonistic interactions. Redfin are probably the greatest threat, and appear capable of reducing abundances of small native fishes such as Pygmy perch, Rainbow fishes, Gudgeons or galaxiads in wetland areas. Redfin are highly piscivorous in their diet and also occur in large shoals (Cadwallader and Backhouse, 1983). Lintermans (unpubl. data, cited in Faragher and Lintermans, 1997) observed a strong coincidence between sudden declines in redfin due to disease and increased abundances of Western carp gudgeon in Lake Burley Griffin, Australian Capital Territory. Studies on the Ovens River floodplain have also provided evidence of redfin impacts. Stoffels (1998) and Stoffels and Humphries (2003) observed interactions between Redfin perch, Hypseleotrid gudgeon and Mosquitofish under field and laboratory conditions. Redfin perch were noted to prey on both gudgeon and Mosquitofish, although Mosquitofish appeared more vulnerable due to their mid-water schooling behaviour. Similarly, Flat-headed galaxias (Galaxius rostratus) on the same floodplain were also thought to be highly susceptible to Redfin perch predation as a result of their mid-water schooling (McNeil, 2004). This possibly explained the absence of galaxiads from otherwise suitable billabongs inhabited by Redfin perch. Clunie and Koehn (2001) noted observations of small Redfin perch appearing to forage in and around nests of Eel tailed catfish (*Tandanus tandanus*), although it was unclear whether they were targeting the catfish eggs. Negative effects on the distribution of Western pygmy perch have also been implicated in

Western Australia's Murray River catchment (Hutchison, 1991), and other (riverine) work in Western Australia found small fish comprising up to 17% of Redfin perch stomach contents (Pen and Potter, 1992).

Redfin perch are also thought to be a potentially major threat to reintroductions of threatened native fishes in the Basin (Faragher and Lintermans, 1997; Clunie and Koehn, 2001). This is corroborated by observations from Victoria's Lake Burrumbeet (Baxter *et al.*, 1985) of Redfin perch preying heavily on newly-stocked Rainbow trout fingerlings.

Mosquitofish may also have a predatory impact on wetland communities, although probably act more as agents of sub-lethal, interference competition through fin-nipping or other aggressive behaviours (Schoenherr, 1981; Lloyd, 1990; McDowell, 1996a; Warburton and Madden, 2003). Ivantsoff and Aarn (1999) found remains of small fish in the gut of 21 of 631 mosquitofish collected from a northern New South Wales river, including rainbowfish larvae. Similarly, Mosquitofish have been found to have significant predatory impacts on several small native fishes in the USA (Schoenherr, 1981; Schaefer *et al.*, 1994).

In some circumstances, Carp may also have a predatory impact given their often concentrated abundances and high biomass (Driver *et al.*, 1997; Stuart and Jones, 2002). Carp are known to be capable of consuming large crustacea and small fishes (e.g. juvenile Bony bream: R. Ping Kee, pers. com.), although these are generally minor dietary components (Jester, 1974; Crivelli, 1981; Hume *et al.*, 1983; Khan, 2003). The greater proportion of smaller invertebrates and zooplankton in the Carp diet suggests that their trophic impact may be reflected more as an energy block (see 'Resource and interference competition' below).

Rainbow and Brown trout are known to have major impact on upland stream galaxiids (e.g. Tilzey, 1976; Crook and Sanger, 1997; Raadik, 1993; Lintermans, 2000), but as they are seldom abundant in wetland habitats, their impact on wetland macrofauna is difficult to gauge and is probably minor. Nevertheless, Cadwallader and Eden (1982) noted predation of the Common galaxias (*Galaxias maculates*) by Rainbow trout in Victoria's Lake Purrumbete.

Habitat modification

Both Carp and (to a lesser extent) Goldfish have been credited with significant structural damage to wetlands in Australia and overseas. The general scientific view seems to be that wetland impacts are likely to be greater than those to lotic habitats (Table 3.5 in Roberts and Ebner, 1997). Unfortunately, much of the Australian work on these has been undertaken in pond or other artificial environments, and extrapolation of findings to natural systems needs to be undertaken cautiously. The physical impacts of carp on the benthic environment have received the most attention, and result primarily from their post-juvenile benthic mode of feeding. Changes to the water chemistry, suspended sediment, macrophytes and bank geomorphology have all been examined, and many of the findings indicate complex effects whose expression may also relate to other factors such as sediment type. Tilapia are also thought capable of significant damage to wetland macrophytes and possibly macroinvertebrates (Arthington and McKenzie, 1997), though have not as yet established Basin populations.

Evidence for Carp benthic feeding behaviour leading to bank erosion is equivocal, and difficult to relate to natural environments. Roberts and McCorkelle (1995) tested the role of Carp in bank erosion in irrigation supply channels with 'mature banks'. After a nine month period of Carp exclusion through caging, no significant effects or changes to the bank profile were detected. Bank slumping and under-cutting was evident along banks both with and without carp. Nevertheless, this finding may have been influenced by the comparatively short study duration and the already disturbed nature of the site (constructed irrigation channels). In a further example, numerous foraging marks were evident in the banks of new ponds on the lower Darling Anabranch after adult Carp had been in them for only 2–3 months (G. Wilson, pers. obs.; Figure 1).

Varying influences of Carp on water chemistry have also been detected in Australian research. In the early 1980s, the Victorian Carp Program found no relationship between Carp biomass density and turbidity over nine wetland sites and two years along the Goulburn River (Hume *et al.*, 1983; Fletcher *et al.*, 1985).

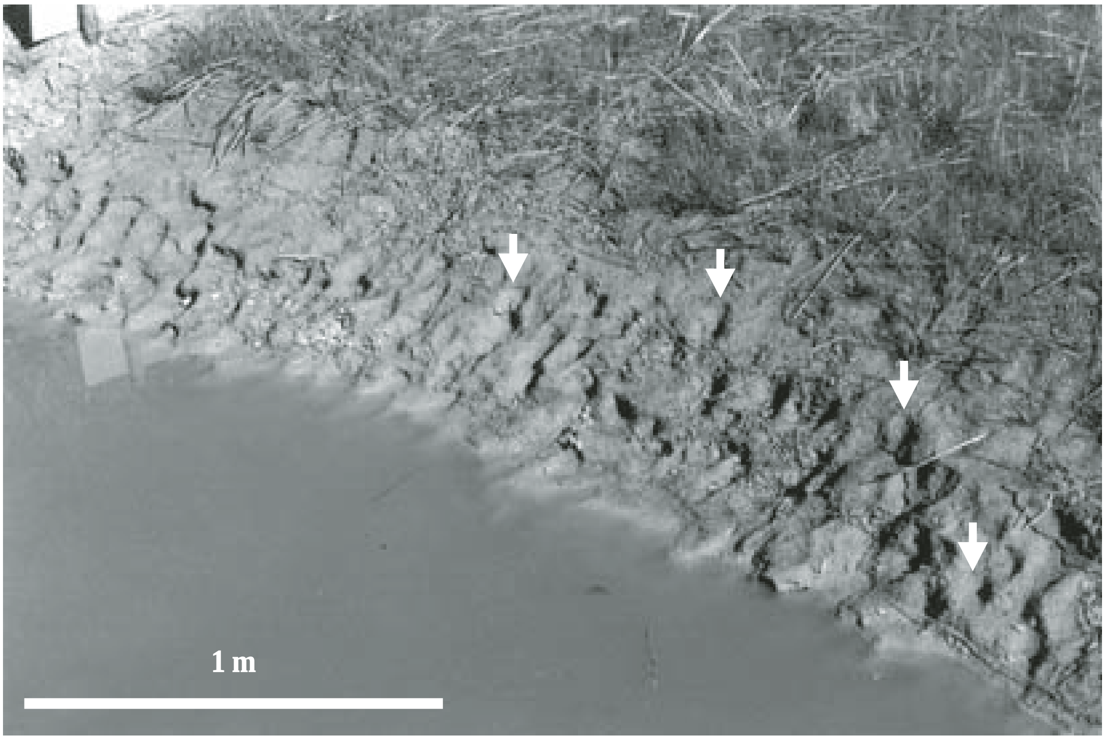

Figure 1. Bank erosion from benthic foraging by adult carp in an experimental pond, lower Darling Anabranch. Examples of individual foraging marks are indicated.

Instead, flow variability was suggested as the key driver of water quality in these floodplain billabongs. A later study of Carp impacts in two Murrumbidgee billabongs split into 'high' and 'low' Carp biomass treatments (King *et al*., 1997; Robertson *et al*., 1997) found a significant impact of Carp on turbidity, total phosphorus levels (six out of seven sampling dates) and algal bloom intensity, but that these effects varied with Carp biomass and sediment type. In the 'high Carp' ends, higher rates of particle settlement, reduced algal production on wood blocks, and increased sediment oxygen demand was also detected. Nevertheless, it was suggested that factors other than Carp had contributed more to water quality variation, sounding a note of caution to oversimplified views of Carp impact as well as highlighting the challenges of understanding impacts of this nature under natural conditions.

Three other studies in artificial habitats near Griffith each indicated significant links between carp biomass and altered water quality. Bowmer *et al*. (1994) used a 3.2 km irrigation drain divided into two sections of 1.2 km (bare) and 2 km (vegetated) through a concrete weir to examine the effect of Carp removal on water quality and macrophyte recovery. Carp were removed from each treatment using poisoning and were found to be present at 432 kg ha^{1} and 474 kg ha^{1} in the bare and vegetated treatments, respectively. Turbidity in the bare treatment shifted from between 20–50 NTU down to around 10 NTU after Carp removal. However, turbidity in the vegetated end remained unchanged though did decrease along the length of the vegetated section to nearly 0 NTU. This suggested the importance of macrophytes to limit disturbance of the sediment and reduce suspended material loads in the water column. Roberts *et al*. (1995) examined the short-term effects of Carp on turbidity in two ponds stocked with high (510 kg ha^{1}) and low (226 kg ha^{1}) densities, respectively. After only four days, turbidity in the high density pond had increased from 7 to 73 NTU. Macrophytes also appeared to modify the effects of Carp on turbidity, similar to the findings of Bowmer *et al*. (1994). Meredith (1996) also examined the capacity of Carp to reduce pond water quality, particularly the issue of sediment bioturbation leading to increased phosphorus levels. In two ponds with high (1000 µg g^{1})

and low (180 μg g^{1}) phosphorus sediment added, total phosphorus levels were similar but turbidity was nearly twice as high in the low-phosphorus pond. However, levels of bio-available phosphorus were two-thirds lower in the low-phosphorus pond.

Carp may also interfere with the recruitment, growth and diversity of wetland macrophyte beds. While research findings generally indicate the capacity of Carp to impact macrophyte assemblages, they also demonstrate the importance of plant growth form and sediment structure to the expression of any impacts. The Victorian Carp Program concluded that Carp were not a general agent of macrophyte damage, but suggested that the abundances of soft-bodied *Potamogeton* had decreased in their floodplain billabongs as a response to Carp (Fletcher *et al*., 1985). They also concluded that a density of 450 kg ha^{1} was the critical point beyond which plant impacts would be more widespread. However, the pond experiment of Roberts *et al*. (1995) found no impacts to *Juncus*, *Myriophyllum papillosum* or *Schoenoplectus validus* despite the 'high density' treatment of 510 kg ha^{1}. By contrast, a complete loss of *Chara fibrosa* and *Vasllisneria* sp. occurred at both carp densities by day six, accompanied by raised surface water temperatures. Similarly, the presence of both bare and densely-vegetated sections of the irrigation channel in Bowmer *et al*. (1994) confirms that some macrophytes can persist under moderate to high carp densities. Instead, the rapid recruitment of young plants to the bare end of the channel following Carp removal suggested that recruitment limitation was a major impact by this fish. Interestingly, Swirepik (1999) also concluded that Carp significantly reduced vegetative regrowth of *Potamogeton tricarinatus* in her large ponds and that this led to increased reliance on seed-bank regeneration. These findings have important implications for plant regeneration in ephemeral wetlands following flood inundation, which would also be expected to attract numbers of Carp for feeding (potential substrate disturbance) and spawning.

A further potential impact of Carp on wetland plants is their reduction in seed banks through the ingestion of seeds. Seeds as a significant portion of the carp diet have been recorded in France (Crivelli, 1981), the USA (Jester, 1974) and Victoria (Hume *et al*., 1983; but *cf*. Khan, 2003). Nevertheless, no study has yet demonstrated that this limits subsequent plant recruitment.

Disease or parasite introductions

Exotic fish species have been credited with the introduction and dispersal of a number of diseases and parasites into Basin aquatic habitats (Arthington and McKenzie, 1997). Cyprinids, Redfin perch, weatherloach and trout are all implicated, and have the potential to impact both riverine and wetland fish assemblages. Nevertheless, wetland-specific data on infection of native fish are scarce, and direct links between disease outbreak and decline of any native fish have not been firmly established.

Table 2. Parasites and pathogens known from carp and goldfish. 'No/Yes' refers to groups where only some of the known parasites/pathogens have been identified in Australian populations.

Group	No. taxa	Recorded in Australia?	Notable examples	Known native fish susceptibility?
Viruses	1	No		No
Bacteria	1	Yes	Goldfish Ulcer Disease	silver perch
Fungi	2	No/Yes		bony bream
Protozoa	14	No/Yes		Various
Monogenea	2	No		Potential
Digenea	4	No		No
Cestoda	2	No/Yes		Various
Crustacea	1	Yes	*Lernea cypracea*	Various

Carp and Goldfish are known to harbour at least 27 parasite and pathogen taxa (Table 2). The two most relevant ones in Australia are the bacterium *Aeromonas salmonicida*, causing Goldfish ulcer disease (GUD), and the parasitic copepod *Lernea cypracea*. GUD is known to infect Silver perch (Bidyanis bidyanis) (Humphrey and Ashburner, 1993), but otherwise appears restricted to cyprinids and salmonids (all exotic species). By contrast, *Lernea* is a widespread problem in native fishes (Rowland and Ingram, 1991). Harris and Gehrke (1997) found a positive relationship between the proportion of Carp in New South Wales (river) fish populations and the incidence of visible abnormalities such as *Lernea* infection in native species. The extent of infection of juvenile Carp and small native species may also vary markedly among wetlands (G. Wilson, pers. obs.).

A further serious pathogen is the Epizootic Haematopoietic Necrosis Virus (EHNV), first isolated from Australian redfin (Langdon *et al*., 1986). EHNV is known to be pathogenic to a range of native fishes (at least *Galaxias*, Macquarie perch, Silver perch, Murray cod), and so could feasibly impact wetland fish assemblages. It is thought to possibly have contributed to declines in Silver perch abundances (Langdon, 1989). Importantly, it is also carried by Rainbow trout and so stocking of this fish could potentially aid in dispersal of the virus. EHNV is impossible to eradicate once present in a waterbody, although it is not known how prevalent the disease is throughout the Basin (Clunie and Koehn, 2001).

Weatherloach have been found to harbour at least 19 protozoans, cestodes and trematodes in Asian populations (Koster *et al*., 2002). To date, at least one monogenean parasite, *Gyrodactylus macracanthus*, is thought to have been introduced into Australia by weatherloach and has been found in the Australian Capital Territory population (Dove and Ernst, 1998). Insufficient work has been undertaken to confirm the spread of this fluke throughout the remainder of the Weatherloach's Australian distribution, or the presence of further exotic parasites.

The Asian Fish Tapeworm *Bothriocephalus acheilognathi* is a more recent arrival in Murray-Darling Basin fishes (1997: Dove *et al*., 1997). It appears to have spread throughout populations of wetland species such as Hypseleotrid gudgeons, Flat-headed gudgeon and Australian smelt (Retropinna semoni). Previously known hosts overseas include various cyprinids occurring in the Basin as well as Mosquitofish. In the Australian Capital Territory and surrounds, Dove *et al*. (1997) found this parasite infecting Carp, Mosquitofish and Western carp gudgeon. Mass mortality events in Australian Capital Territory populations of Western carp gudgeon are thought to have possibly been due to outbreaks of this parasite (Clunie and Koehn, 2001).

Resource and interference competition

Wetland impacts from resource or interference competition have been linked to Carp, Goldfish, Mosquitofish and possibly weatherloach. Nevertheless, many of these are of a more cryptic nature, and more difficult to establish evidence for than issues such as habitat modification. They could arise through such mechanisms as foraging behaviour, dietary preferences or antagonistic behaviour.

A combination of the frequent high densities of young Carp and their zooplanktivorous diet (e.g. Khan, 2003) could potentially produce an 'energy block' in the flow of resources between producers and higher consumers. Carp larvae and juveniles potentially lead to a major reduction in food resources for small native fishes or early life history stages. Khan (2003) and Khan *et al*. (2003) also identified the potential for Carp to induce a trophic cascade in shallow western Victorian lakes through their heavy predation on micro-crustacea. Shirley (pers. comm.) also detected significant reductions in zooplankton in the presence of larger carp. Localised reductions in zooplankton could feasibly limit the survivorship of small fishes or (particularly) their larval stages at the same spatial scale.

Furthermore, the concentration of large numbers of Carp in wetlands, for example for spawning or during wetland drying, may displace other fish or macroinvertebrates. In situations where weatherloach have established high localised abundances (Koster *et al*., 2002), similar interference with wetland fishes could feasibly result.

A further possibility is the effect on food web structure from reductions in benthic algal production. Such algal mats are thought to be a key driver of aquatic food webs on floodplains of the northern Murray-Darling Basin and nearby catchments (Bunn *et al*. 2003), and their loss could result in chronic reductions in faunal assemblages at higher trophic levels. Although algae are not a major dietary component for carp in wetlands (e.g. Khan, 2003) their foraging mode in shallow areas could significantly disturb algal mats along the littoral margins. The possibility of this having influenced the abundance and diversity of macroinvertebrates such as the herbivorous gastropod *Notopala* still requires testing.

There is also the potential for carp benthic foraging to have contributed to the decline of eel-tailed catfish in Basin wetlands, either directly through disturbance to nest sites or indirectly through raised levels of suspended sediment smothering eggs or larvae (Clunie and Koehn, 2001). Nevertheless, Carp and catfish can coexist in high numbers too (Clunie and Koehn, 2001; G. Wilson, pers. obs) and Carp are likely to be only one of many factors to have influenced the decline of this and other threatened native fishes.

Possibly the greatest impact of Mosquitofish on wetland fish assemblages arises from their highly antagonistic approaches towards other small fishes (Schoenherr, 1981; McDowell, 1996a; Warburton and Madden, 2003). Lloyd and Walker (1986) speculated that the spread of Mosquitofish had played a significant role in the decline of the purple-spotted gudgeon (*Mogurnda adspersa*) and pygmy perch from the lower Murray River. Apart from chasing and predation on eggs or larvae (See 'Direct predation on macrofauna' above), nipping at fins and scales can lead to fungal infection and subsequent mortality (Meffe *et al*., 1983).

Management risks

Despite the above potential for impacts to Murray-Darling Basin wetlands from exotic fish, a number of risks exist in managing these habitats for this purpose. Some of the main ones include parallel management decisions inadvertently benefiting exotic species, varying community attitudes and ignorance of the impact potential of such species, and selecting the most appropriate control strategy for particular sites or species. Each of these issues is discussed briefly below.

Improved wetland connectivity

Managing aquatic ecosystems will inevitably require decisions to be made about conflicting objectives (Braysher and Barrett, 2000). Decisions aimed at improving one aspect of ecosystem function may inadvertently promote the maintenance of exotic fish populations. Environmental flow regimes that reinstate or improve rates of floodplain connectivity are a strong example of this, whereby survivorship rates, wetland access, and/or the dispersal of eggs or larvae of exotic species may be facilitated under increased inundation regimes. For instance, McNeil (2004) found fish assemblages in highly ephemeral wetlands on the Ovens River floodplain to be dominated by Goldfish, although these perished as the wetlands dried out if no further flooding occurred. Redfin perch were similarly affected by hypoxia as wetland water levels receded. McNeil (2004) also found wetlands to be a key winter refuge for redfin which would presumably disperse to other habitats should a flood be experienced. Numerous studies have also observed adult Carp entering wetlands after flooding (e.g. Scholz *et al*., 1999; Stuart and Jones, 2002), potentially leading to decreased plant recruitment and increased turbidity (e.g. Crivelli, 1983; Roberts *et al*., 1995). As wetlands appear to be a key spawning or refuge habitat for Carp, Goldfish, Redfin and Mosquitofish (Stoffels and Humphries, 2003; Stuart and Jones, 2002; McNeil, 2004), increased flooding frequency and duration would also be expected to promote the survivorship of these species and dispersal of offspring into channels or other wetlands.

Community awareness, perceptions and attitudes

Exotic fishes attract a range of opinion in the Australian community, based partly on perceptions of ecological impact but also human utility (Table 3). Indeed, these species have become established in Murray-Darling Basin wetlands through a variety of mechanisms, including migration from adjacent river channels and deliberate releases (unwanted aquarium fish, unused bait, stocked fingerlings for angling). While Carp have nearly uniformly ascribed pest status, Goldfish are often viewed as a 'native Carp' species and being useful for bait (pers. obs.). Salmonids (trout, salmon) are revered as game fish and subject to substantive

Table 3. Variation in public opinion and awareness of the impact of exotic fishes on Murray-Darling Basin aquatic ecosystems. ✓, slight view or awareness; ✓✓, moderate view or awareness; ✓✓✓, widespread view or awareness; ?, extent of community awareness of species' impact potential unknown.

Species	Negative	Positive	Benign	Impact awareness
Carp	✓✓✓	✓		✓✓✓
Goldfish	✓	✓	✓	✓
Redfin Perch	✓	✓✓		✓
Oriental weatherloach	✓		✓	?
Mosquitofish	✓		✓	?
Trout	✓	✓✓✓	✓	?

agency and private stocking efforts. Similarly, redfin are often a popular angling target. Other species, thought to have originated from aquariums (e.g. Oriental weatherloach) or other deliberate releases (e.g. Mosquitofish), are generally viewed as a benign or unknown threat. Yet most of these species are likely to have some impact on wetland ecosystems. This diversity of attitudes parallels that seen in the USA (Courtenay, 1990), and represents a management risk in terms of levels of community support for control efforts against particular species and the potential to continued releases into waterways through ignorance. Clearly, past calls for increased community education on pest species (e.g. Clunie *et al*., 2002) remain highly relevant.

Choice of inappropriate control techniques

Formal analysis of the projected benefit of alternative control strategies is an important step in managing the impact of invasive species. No single technique should be viewed as a panacea ('silver bullet') and a range of methods will invariably be necessary at the landscape scale to maximise the chances of control success (Braysher, 1993). The application and success of individual control methods will depend on factors such as intrinsic site characteristics (e.g. geomorphology, hydrology, presence of species of high conservation concern), logistical constraints (e.g. site accessibility, proximity to major population centres), and the abundance and biology of the invasive species. Failing to fully account for these factors may risk having a minimal net reduction in the population in question and/or parallel impacts on desirable habitat attributes such as the abundance of native fish.

Past control efforts for pest fishes in the Murray-Darling Basin have mostly focused on carp. Roberts *et al*. (1997) provided an example of a cost-benefit analysis of the most effective carp control techniques across varying habitat types, including a specific assessment of exclusion screens and large-scale netting for wetlands. Various papers in Roberts and Tilzey (1997) also provided a general treatment of alternative control options, although without a specific wetland focus. Brown and Walker (2004) used computer modeling to predict the efficacy of a range of techniques for carp under varying scenarios of abundance, growth and reproduction. Their assessment included analysis of carp from Barmah Forest, a significant wetland complex adjacent to the Murray River. Unfortunately, there have been no comparable studies for other Murray-Darling Basin pest species, which in itself represents a management risk for Basin wetlands.

Geomorphology, hydrology and logistical factors are interrelated and, for example, will influence the ease of deploying particular trapping methods or exclusion devices. Determining the biological attributes of invasive fishes, particularly those relating to reproductive mode, will also be critical to the selection of control measures. Factors such as egg and larval duration, seasonal spawning phenology, and whether the use of wetlands for spawning is obligate or facultative will be particularly relevant. A recent multivariate classification of the reproductive characteristics of native and exotic freshwater fishes from south eastern Australia (Growns, 2004) detected five major reproductive guilds. Exotic fishes, including all of the above 'wetland species', were spread across all five guilds (Mosquitofish was the sole member of the fifth guild) although were predictably clustered within multi-species families (*Cyprinidae, Salmonidae*). This reproductive diversity will invariably necessitate a range of control strategies for the suite of exotic species occurring in Murray-Darling Basin wetlands.

The use of screens across wetland inlets to exclude Carp is a strong example of the need for careful selection of control method. Since the early 1990s, these have become a popular mechanism for limiting wetland access by Carp along the lower Murray River. Sites such as Pilby Creek, an anabranch wetland on South Australia's Chowilla floodplain, became well known for their screens and the purported improvements in water quality and macrophyte abundance resulting from the exclusion of large Carp (e.g. Recknagel *et al.*, 1998). Yet, screens suffer from becoming clogged with algae or other debris, do not exclude the ingress of dispersive larvae during flow periods (cf. Gilligan and Schiller, 2004; Humphries and King, 2004), may easily be breached by adult Carp during floods (P. Teakle, pers. comm.), and are likely to exclude significant numbers of larger native vertebrates such as Eel-tailed catfish, Golden perch, Bony bream or turtles. Any ecological benefits from inlet screens are likely to be highly localised (limited to single wetlands) and their use needs to be considered in light of the initial high installation cost and requirements for ongoing and time-consuming maintenance such as removal of algae and other debris. The efficacy of such screens is likely to be restricted to areas of particularly high conservation need and wetlands with a single, discrete flow path into them.

Key knowledge gaps in managing wetland impacts from exotic fishes

Dispersal

The mechanisms and scales of dispersal, particularly during early life history are poorly understood for all exotic fishes in the Murray-Darling Basin. This includes stage or size-structured phenomena such as larval drift (Humphries and King, 2004) or shifts in juveniles away from any nursery habitat (McDowell, 1996b; Vilizzi and Walker, 1999). The importance of such 'source–sink' dynamics to the control of exotic fishes is underscored by the need to understand the scales over which control efforts need to be managed. For example, managing a suite of wetland sites with continual re-introduction from uncontrolled re-colonist sources upstream will be fruitless until the scale of management is broadened to include both source and sink populations. It will also be necessary to determine the role of anthropogenic factors such as flow regulation and deliberate translocations to further dispersal and range expansion.

Impacts from small species

Ecological impacts of small exotic fishes such as Mosquitofish and weatherloach have received little attention in comparison to those of larger species such as Carp. This undoubtedly relates to the cryptic habits of small fishes and associated difficulties with tracking interspecific or habitat interactions in the field. Nevertheless, preliminary observations and studies from overseas and elsewhere in Australia suggest that smaller taxa may still have a substantial impact on the ecology of wetland systems (references above). This need is particularly pressing for weatherloach which, despite its comparatively short history in the Basin, appears to be steadily expanding its range across southern catchments and achieving localised high densities (Koster *et al.*, 2002). Similarly for Mosquitofish, while some laboratory evidence exists of negative interactions with small native fishes, corroborative observations are still required under field conditions.

Guidance in the choice of control strategy

With the exception of carp (Roberts and Tilzey, 1997; Roberts *et al.*, 1997; Brown and Walker, 2004), virtually no information is available to guide managers in choosing the most appropriate control strategy for exotic fish in Murray-Darling Basin wetlands. For larger species, the range of potential strategies includes poisoning, exclusion, egg destruction, direct removal and/or commercial harvesting, habitat restoration and genetic modification. Yet for species such as Mosquitofish or Weatherloach, a smaller body size and their cryptic habits suggest that techniques such as exclusion or direct removal/harvesting would have minimal chance of control success. By contrast, methods such as allowing ephemeral habitats to dry out or driving populations towards male dominance through genetic manipulation may show more promise with these species, particularly given their comparatively short generation times. Formal analyses of the efficacy of alternative strategies for a range of species are critically needed, although a greater understanding of the basic population dynamics of most species will first be necessary across a range of habitats.

Linked to this is the need for rigorous comparisons of control success under field conditions. Again, some information is available for carp (e.g. Stuart and Jones, 2002) although further work is needed for all species. Such studies are needed to confirm modeling projections from studies such as Roberts *et al.* (1997) and Brown and Walker (2004). Ideally, any outcomes would be delivered as a decision support tool that both management agencies and community groups could utilise.

Conclusions

The topic of exotic fishes and wetlands combines two complex management issues. Wetlands in the Murray-Darling Basin are increasingly gaining recognition for their critical role in the functioning of river systems and maintaining biodiversity at the catchment scale (Phillips, this volume). Yet, while there is a growing appreciation of how dynamic wetland ecosystems can be under the Basins' typically variable flow regimes, the establishment of self-sustaining wetland populations of exotic fishes has also created a further layer of ecological complexity. There is an urgent need for a more rigorous understanding of wetland function, the population dynamics of pest species, and strategies for their effective control or elimination throughout the Basin. While we have assembled some understanding of the wetland impacts of species such as Carp or Redfin perch, much further work is needed on the smaller, cryptic species whose impacts may prove equally significant. Field tests of control measures for any species are still needed. Far more community education is required to address the risk of continued introductions through such mechanisms as the disposal of unwanted live bait or aquarium fish. Managers will also need to account for the risks in managing pest fishes such as the cost–benefit balance from initiatives such as environmental flow increases. Nevertheless, awareness in many community sectors of the parallel importance of wetlands and reducing pest fish abundances continues to grow, and fortunately the need for knowledge-guided management interventions is well accepted.

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