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Non-salmonids in a salmonid fishway: what do 50 years of data tell us about past and future fish passage?

M. MALLEN-COOPER & D. A. BRAND

NSW Department of Primary Industries, Cronulla, New South Wales, Australia

Abstract Salmonid fishways have been used in many countries for non-salmonid fishes, including Australia, but generally with poor results. Trapping the entrance and exit of a 1:9 gradient salmonid fishway on the Murray River confirmed very poor passage of native fish, with <1% of the most abundant species ascending. Fifty years of fish passage monitoring showed the numbers of three native species declining by 95–100% and non-native fish becoming dominant. Fishways are now being designed for native fish and being quantitatively assessed, but daily flow management also needs to be addressed. The ecological model for passage of potamodromous fishes has changed from passing adults of a few species to one that incorporates the whole fish community, specifically: immature fish of large-bodied species that dominate numbers migrating upstream; a diverse range of movement strategies; and small-bodied species, crustaceans and low numbers of less-mobile species.

KEYWORDS: Australia, fish passage, fishways, migration, non-salmonids, potamodromous.

Introduction

Fishways have been used in some form for over 300 years (De Lachadenede 1931; in Nemenyi 1941). Early fishways concentrated on upstream passage of salmon in the northern hemisphere, initially with varied results (Cobb 1925), but with increasing success, particularly with the advent of the Columbia River fishways in the north-west United States (Clay 1995). That success led to salmonid fishway designs or slight variations being used for non-salmonid fishes elsewhere, including South America (Quirós 1989), South Africa (Bok 1990), Nigeria (Petts 1984), Sudan (Bernacsek 1984), Iraq (FAO/UN 1956), Pakistan (Khan 1940; Ahmad, Ali & Ahmad 1962), Thailand (Pholprasith 1995), New Zealand (Jowett 1987) and Australia (Mallen-Cooper & Harris 1990). Unfortunately, many of these fishways failed (e.g. Petts 1984); often the fishway was too steep, or water velocities within the fishway exceeded the swimming ability of the fish.

In Australia, salmonid-type fishways have been used from early last century until the mid-1980s (Hooker 1966; Mallen-Cooper & Harris 1990). Most of these are in the eastern states where there are 91, with 69 of these in the south-eastern region, which has the most intensive water resource development (Harris 1984; Mallen-Cooper 1989; Hajkowicz & Kerby 1992; Harris & Mallen-Cooper 1994).

In the extensive Murray–Darling river system of south-eastern Australia, 22 salmonid-type fishways were built from 1930 to 1975 (Mallen-Cooper 1989). These fishways were pool-type designs, built on gradients of 1:9, which is similar to typical salmonid fishways of the time, up to slopes of 1:4, which is steeper than successful salmonid fishways of the time. These fishways were mostly designed with the standard salmonid head loss (or step height) between pools of 0.3 m that produces a maximum water velocity of 2.4 m s^{-1}, which are still commonly used criteria for salmonids (Clay 1995).

Experiments in the late 1980s in experimental vertical-slot fishways concluded the optimum slope and maximum water velocity for Murray–Darling fishes were 1:18 and 1.8 m s^{-1} (Mallen-Cooper 1994). More recently, a gradient of 1:32 with a maximum water velocity of 1.4 m s^{-1} has been used on the Murray River to include the passage of small-bodied

Correspondence: Dr. Martin Mallen-Cooper, Fishway Consulting Services, 8 Tudor Place, St Ives Chase, NSW 2075, Australia (e-mail: mallencooper@optusnet.com.au)

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species (Barrett & Mallen-Cooper 2006). Passage through these new fishways has been (Mallen-Cooper 1999), or is being (Stuart, Baumgartner, Zampatti & Barrett 2004) assessed. However, there is only limited assessment of the older salmonid fishways to quantify their contribution to fish passage and evaluate the: potential for modification, ability to act as an adjunct with a new fish passage facility, or need for complete replacement. Two studies of tidal fishways on large subtropical rivers (Kowarsky & Ross 1981; Russell 1991) concluded that the fishways were inefficient, but were based largely on samples from the exit of the fishways, with no quantitative sampling downstream of the fishway or at the entrance to indicate the numbers or species of fish unable to use the fishways. These two fishways have since been modified to improve native fish passage (Stuart & Mallen-Cooper 1999; Stuart & Berghuis 2002). Beumer & Harrington (1982) investigated a fishway in a small temperate stream and noted that small fish did not use the fishway, but there was a paucity of data. Harris (1984) surveyed the design of 29 fishways in the coastal rivers of south-eastern Australia and found that there was inadequate maintenance, poor entrance locations and poor operation at low flows.

The present study investigates the Euston fishway on the Murray River, which is the least steep (1:9) of the old fishways. This particular fishway also has some fish passage records from 1937 (Cadwallader 1977) to the present, providing a rare opportunity to examine the trends in migratory fish populations over an extensive time period. The aims of the present study were to: (i) assess the effectiveness of a salmonid-type fishway for the passage of non-salmonid fishes, by comparing fish that enter the fishway with fish that successfully ascend the fishway; and (ii) examine trends in fish movements from long-term monitoring of fish passage through the fishway.

Materials and methods

Site description

The Euston fishway is one of nine on the Murray River (Fig. 1). The earliest was built at Murtho (Lock 6) in 1930, followed by Euston (Lock 15) in 1937. A vertical-slot fishway with a slope of 1:18 was built at Torrumbarry Weir (Lock 26) in 1991 (Mallen-Cooper 1999), and a fish lock was completed at Yarrawonga Weir in 1995 (Fig. 1). In the last few years, vertical-slot fishways with a slope of 1:32 have been built at Locks 7, 8, 9 and 10, and three fishways have been built at the tidal barrages of this system, with the intent of passing a wide range of species and sizes of fish (Barrett & Mallen-Cooper 2006).

The fishway at Euston is a concrete channel, rectangular in cross-section, divided into pools by a series of evenly spaced baffles and rises 4.57 m. There are 15 pools, each 2.74 m long by 1.83 m wide. They are mostly 1.22 m deep, with deeper pools at the top (2.13 m) and bottom (2.44 m) to accommodate 1.22 m variation in headwater and tailwater. The tailwater range is 27% of the 4.57 m tailwater variation in elevation, but 66% of the non-drown-out flows of the period 1937–1945. Each baffle has a square orifice on the bottom of one side that the fish swim through. The orifices are 0.30 m high by 0.50 m wide and are placed on alternate sides of each pool. The fishway is designed to have a head loss, or difference in water level between consecutive pools, of 0.305 m, which equates to a maximum water velocity in the fishway of 2.4 m s^{-1}. This is a common standard for salmonid fishways (Clay 1995). The turbulence is estimated to be between 125 and 143 W m^{-3} (assuming a Cd of 0.7–0.8). The total length of the fishway is 44.66 m and it has one major bend as the fishway turns back towards the weir and places the entrance of the fishway 7.4 m from the face of the weir.

Fishway sampling

The fishway was assessed by trapping fish at the fishway entrance and trapping fish that ascended the fishway and reached the exit. Fish were trapped at the entrance with a single cone-trap placed over the orifice of the baffle. The cone tapered from a square opening of 440 × 440 to 260 × 280 mm over a distance of 510 mm and it was covered with 20-mm square mesh. To capture fish at the exit of the fishway, a cone-trap of similar dimensions was part of an enclosed cage, again covered in 20-mm square mesh. The cage extended into the weir-pool and was lifted with a crane.

The cone-trap was placed in the lowest pool of the fishway that was above the tailwater to sample fish at the fishway entrance. A flat screen with 20-mm square mesh was placed over the upstream orifice of this pool to contain the fish. After the first three samples, another flat screen was placed in the next higher pool as an additional measure to contain fish. Both these pools were then sampled. Prior to the addition of the second screen every pool above the cone-trap was sampled.

The velocity of water passing out of the lowest pool was reduced to between 1.0 and 1.4 m s^{-1} (from 2.4 m s^{-1}) for the sampling period by partially cover-

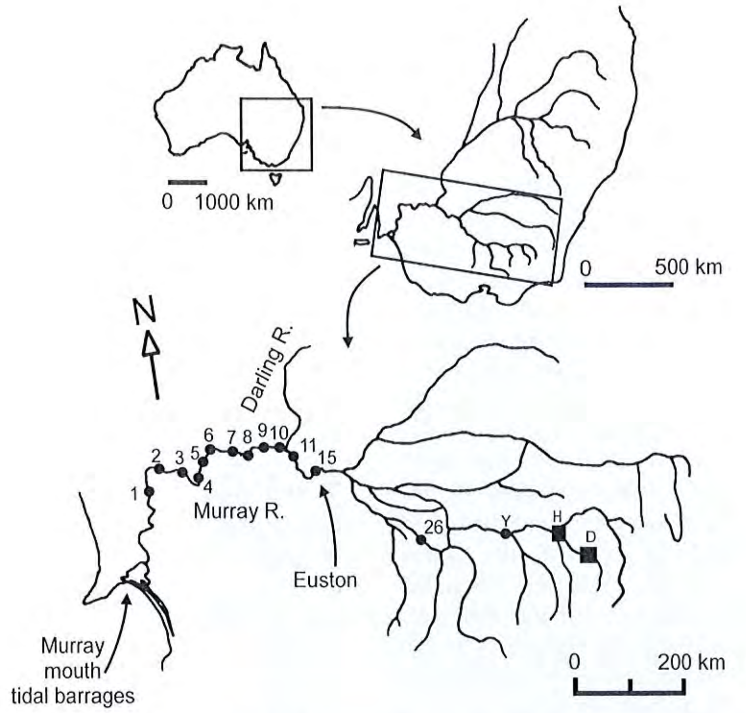

Figure 1. Location of weirs and locks on the Murray River, south-eastern Australia: locks are numbered from 1 to 26; Y, Yarrawonga Weir; H, Hume Dam; D, Dartmouth Dam.

ing one orifice at the top of the fishway with a steel plate, which restricted the flow of water. The intent was to enable small fish, with poorer swimming abilities, as well as larger fish to enter the fishway. At the end of the sampling period the cone-trap was replaced with a flat screen to facilitate the capture of fish.

The exit and entrance of the fishway were sampled for approximately 22 h each, on consecutive days from 26 November 1990 to 31 January 1991. For statistical analysis it would have been desirable to randomise the exit and entrance samples within the paired days, but it was considered more important to provide the same antecedent conditions at the fishway entrance for each sample. The important antecedent condition in this case was a high water velocity (2.4 m s^{-1}) coming from the fishway entrance for the day prior to sampling. This condition may have attracted fish to the fishway prior to the sample.

Hence, the exit of the fishway was always sampled first and this sampling regime provided eight paired-days (16 total) and replicates for the exit and entrance of the fishway. All fish collected were identified and counted and sub-samples of at least 200 fish of each species were measured for length each day. Sampling times varied up to 22 h, so catch rates were standardised to 24 h by the proportional change of time.

Fish migration

Following the end of the paired sampling, the entrance of the fishway was also sampled for an additional 6 days in the first week of February 1990. The aim was to gain information on fish migration during a rise in river level.

Long-term monitoring

Fish passage at the Euston fishway has been monitored periodically since its construction in 1937, using a trap at the exit of the fishway, which is checked daily. The monitoring was instituted by the River Murray Commission (RMC) to assess the fishway and continued to 1945. The Murray–Darling Basin Commission (formerly RMC), in cooperation with NSW Department of Primary Industries (formerly NSW Fisheries), recommenced monitoring in 1987 and continued systematically to 1992. The monitoring was done by Commission staff that operate the weir, and fish were identified, counted and released.

Results

Species composition and abundance

Of the six species found entering the fishway, four were native; golden perch *Macquaria ambigua* Richardson (Percichthyidae), bony herring *Nematalosa erebi* Günther (Clupeidae), silver perch *Bidyanus bidyanus* Mitchell (Teraponidae), Australian smelt *Retropinna semoni* Weber (Retropinnidae) and two were non-native; common carp *Cyprinus carpio* L. (Cyprinidae) and redfin perch *Perca fluviatilis* L. (Percidae). Excluding Australian smelt, 60 native fish were collected at the exit of the fishway compared with 6486 at the entrance (Fig. 2), which comprised mostly golden perch with a mean catch rate of 777 fish day^{-1}. Abundant Australian smelt were observed in the entrance pool of the fishway but their small body size enabled these fish to easily pass through the 20-mm mesh of the cone-trap. A sub-sample of Australian smelt was taken with a dip-net and the fish were measured for their length distribution (Fig. 3). Only 20 silver perch were collected, including 19 at the exit of the fishway; these are not included in Fig. 2.

The numbers of golden perch, bony herring and redfin perch at the entrance of the Euston fishway were significantly higher than the numbers at the exit of the fishway ($P < 0.02$) (Fig. 2). The numbers of common carp and silver perch were not significantly different ($P > 0.05$) at the two locations in the fishway. However, for silver perch the sample size was very small and for common carp numerous large fish were observed jumping out of the trap at the entrance, which confounded the results for that species. Large common carp were unable to jump out of the trap at the exit of the fishway because it was enclosed.

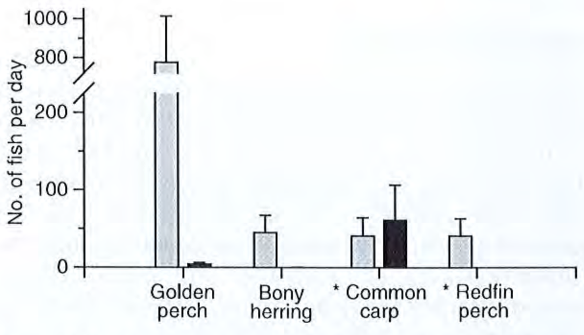

Figure 2. Catch rate (mean ± SE) for each species collected at the entry (grey bar) and exit (filled bar) of Euston fishway in eight paired days; *non-native species.

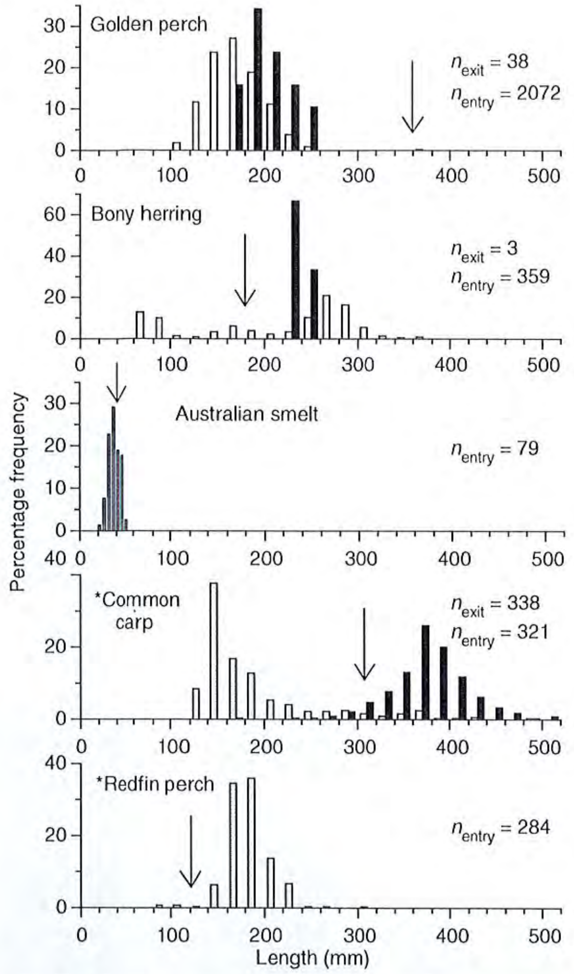

Figure 3. Length frequency of each species collected at the entry (grey bar) and exit (filled bar) of Euston fishway (arrows indicate length at maturity; *non-native species).

Significantly smaller golden perch were collected at the entrance of the fishway (Kolmogorov–Smirnov $D_{38,\ 2072} = 0.48$, $P < 0.001$) and no fish <160 mm total length (TL) was collected at the exit of the fishway (Fig. 3). The size distribution of common carp was biased as the trap at the exit caught large fish effectively. However, large common carp (300–500 mm) could ascend the fishway and small common carp 120–260 mm, which could be caught by the exit trap, were very poorly represented at the exit of the fishway. The smallest common carp at the exit of the fishway was the same as the smallest golden perch. No statistical tests on length distribution were applied to: common carp because of the biased entrance sample, bony herring because of the low sample size for the exit sample, or Australian smelt and redfin perch which were not collected at the exit of the fishway.

In the 6 days of sampling the entrance of the fishway in February 1990, the water temperature was stable between 25.0 and 25.5 °C and the river level rose and fell 13 cm (from a gauge height of 1.80–1.93 m), which reflected a maximum change in river discharge of just over 922 ML day^{-1} from 8892 to 7970 ML day^{-1}. The mean water velocity of the river changes from 0.10 to 0.13 m s^{-1} over this flow range (MDBC, unpublished gauging data). In the preceding 2 weeks, there were lower flows, with the lowest at 7220 ML d^{-1}, with a gauge height of 1.69 m.

Only low numbers of common carp, bony herring and redfin perch were collected with < 32 fish per species per day, except for one day with 193 bony herring. The major species migrating during the river rise was golden perch with 4534 entering the fishway (Fig. 4). The numbers of migrating golden perch rose and fell following the river height and flow. There was a pronounced peak in fish numbers coinciding with the peak in river height. The size distribution of these fish was similar to earlier sampling of the fishway with most fish between 100 and 250 mm.

Long-term monitoring

From the long-term monitoring from the exit of the fishway, there were irregular data entries from 1937 to 1939 but these became much more frequent from 1940 to 1945 so only these data, referred to as historical data, were used. They include 525 days of record from January 1940 to November 1945. There appear to be no data from 1945 to 1987. Monitoring from 1987 to 1992 has frequent entries but there are still missing data due to floods submerging the weir or to repairs to the trap. These later data are referred to as recent data and they include 1191 days of record from November 1987 to January 1992.

In the 47 years between the historical and recent data, the proportion of native fish decreased from 98.7% to 38%, and changed from being dominated by native silver perch to being dominated by non-native common carp (Fig. 5). In the historical data period, 25 198 native fish and 344 non-native fish were recorded passing through the fishway. The native fish comprised silver perch (65.4% of total fish numbers, including native and non-native fish), golden perch (22.2%), Macquarie perch, *Macquaria australasica* Cuvier (Percichthyidae) (8.5%), Murray cod, *Maccullochella peelii peelii* Mitchell (Percichthyidae) (2.5%) and catfish, *Tandanus tandanus* Mitchell (Plotosidae) (0.003%) (Fig. 5). The non-native fish were redfin perch and one brown trout, *Salmo trutta* L. (Salmonidae), which when combined contributed 1.3% of the total. A total of 4436 native fish used the fishway in the recent data period, comprising silver perch (9.8% of total fish numbers, including native and non-native

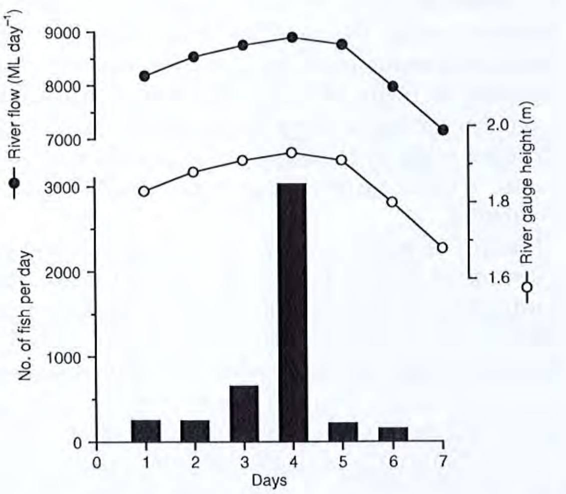

Figure 4. Catch rate of golden perch entering the Euston fishway during a rise and fall in river level.

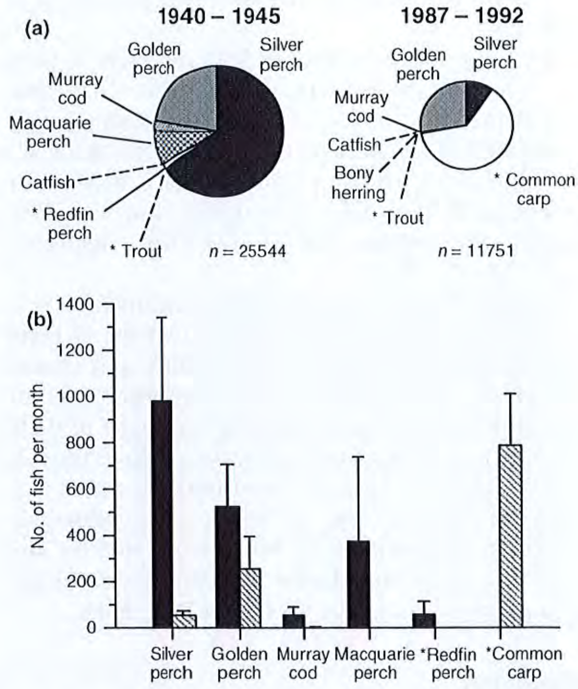

Figure 5. (a) Species composition of migratory fish ascending Euston fishway in 1940–1945 and 1987–1992 (area of pie graphs are relative to total numbers; *non-native species; dotted line to label indicates % too small to show on graph). (b) Monthly (mean ± SE) passage rates of fish, from September to February inclusive, in the Euston fishway in 1940–1945 (filled bar) and 1987–1992 (striped bar).

species), golden perch (27.3%), Murray cod (0.004%), bony herring (0.003%) and one catfish (0.00009%); and 7315 non-native common carp (62%) (Fig. 5).

Raw data from the historical and recent data sets show there is a discrete season of fish migration; primarily spring and summer, with some occasional movement in autumn. To standardise these two data sets for comparison, a migration season was considered to be from the beginning of spring (September) to the end of summer (February) of the following year. Within this period, the number of days of data for each season varied from 28 to 151 for 1940–1945 and 32 to 76 for 1987–1992. To standardise this variation, the mean number of fish per month (30 days) for each species was calculated for each season and common species (>20 fish per month) were plotted for the historical and recent data sets (Fig. 5).

After standardising the data, the variance in mean monthly passage rates was high, reflecting the variability between years within the two data sets. For example, 95% of the Macquarie perch were recorded in November 1945 and redfin perch were only recorded in January and February of 1940; both species were not recorded in 1987–1992. Common carp were present in high numbers in 1987–1992, but absent in 1940–1945.

For those species present in both data sets, a t-test was done on the log-transformed [$\ln(x + 1)$] data. This showed that the 95% decline in the mean monthly passage of silver perch over the 47 years was significant ($t = 3.83$, $P < 0.005$), as was the 96% decline for Murray cod ($t = 3.97$, $P < 0.005$), but the 51% decline for golden perch was not significant ($t = 1.39$, $P > 0.2$).

As tailwater level affects fishway performance, it is possible that changes to flow regimes over the 47 years have changed the fishway's performance and caused the drop in silver perch numbers reaching the exit of the fishway. Hence, a comparison was made of daily passage rates of silver perch at different flows (flow is directly related to tailwater level) from the recent and historical data sets (Fig. 6). There was considerable overlap of the flow rates for both periods and for any given flow consistently higher numbers of silver perch were passing through the fishway in 1940–1945.

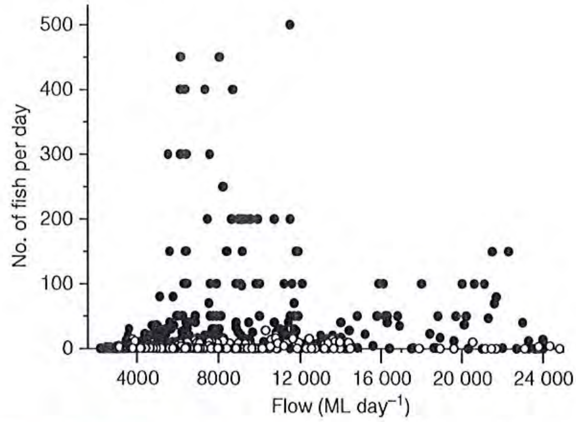

Figure 6. Numbers of silver perch ascending the Euston fishway at different flows in 1940–1945 (filled circle) and 1987–1992 (open circle).

Discussion

Changes in the migratory fish community over 50 years

The physical structure of Euston fishway remained unchanged over 50 years and fish recorded at the exit of the fishway represent a subsample of the migratory fish population in this reach of the Murray River that can locate the fishway entrance and have the swimming capacity to ascend the fishway. Water temperature and flow are important stimuli for migration (Northcote 1984) and to infer changes in fish abundance from the fishway data, it is necessary to consider changes in temperature and flow regimes caused by increasing river regulation. Water temperatures have changed little at Euston, because high dams with stratification that release cold water are over 1000 km upstream, but the frequency of some flows $< 10\ 000$ ML day^{-1} has declined by 50% during the period of regulation (Close 1990). Importantly, these flows correlate with periods of fish movement through the Euston fishway, and some decline in fish numbers using the fishway might be attributed to a reduced stimulus to migrate. However, using the most abundant species, silver perch, as an example, at any given flow more fish were migrating in 1940–1945 than 1987–1992. The declines in numbers of fish moving are far greater than can be explained solely by changes in the flow regime and they appear to reflect genuine large scale changes in the fish population.

Macquarie perch are now only found over 600 km upstream of Euston (Harris & Rowland 1996). The construction of the Euston Weir in 1937 inundated 100 km of river habitat. The early explorer, Charles Sturt, described this reach of the Murray River and the nearby lower Murrumbidgee River, as having rapids and occasional rocky substrates (Sturt 1833). These habitats may have included cobble riffles, which is a spawning habitat of Macquarie perch (Cadwallader & Rogan 1977). Hence, the upstream passage of this species in 1938 at Euston might have been the last migration, after the weir was built, of a local population to a spawning ground that had become inundated by the weir pool; this might explain why the migration of this species was never repeated at Euston.

The decline of Murray cod corroborate earlier descriptions, using explorer's diaries and commercial catch records, of a decline from 1918 to the mid-1960s (Rowland 1989). The reduced abundance of Murray cod, the disappearance of Macquarie perch, the decline of the once highly abundant silver perch, and the major increase in the proportion of non-native species because of the introduction of common carp in the 1960s (Shearer & Mulley 1978), are powerful indicators of declining river health. These changes are over an ecologically short timeframe of 50 years, especially given that golden perch, silver perch and Murray cod are long-lived (> 25 years) species (Anderson, Morison & Ray 1992; Mallen-Cooper & Stuart 2003). The reasons for the declines of these native fishes are varied (Cadwallader 1978; Pollard, Llewellyn & Tilzey 1980; Rowland 1989; Faragher & Harris 1994) but they are recognised and a strategy to rehabilitate native fish populations in the Murray–Darling river system, which includes providing fish passage, is underway (Barrett 2004; Murray–Darling Basin Commission 2004).

Fishway performance

The fishway at Euston allows the passage of some native fish that would be unlikely to migrate past this weir otherwise. Notably, over 25 000 native fish used the fishway in 1940–1945. The present study, however, suggests that number is probably < 1% of the fish that were trying to ascend the fishway. Sampling the exit and entrance of the fishway showed that it is inadequate in passing all species and size-classes of native migratory fish. Australian smelt, a small species, was excluded altogether and only minor numbers of golden perch and bony herring fully ascended the fishway. Of these latter two species, only larger fish, with correspondingly greater swimming abilities, reached the exit of the fishway. The low proportions of native fish ascending the fishway may help to maintain genetic mixing along the river but it would meet few, if any, other ecological objectives. The present study also did not assess the migratory fish population below the fishway or the efficiency of the fishway in attracting fish, which is likely to be important given that the weir is 120 m wide and the fishway passes 0.66% of the low river flow and smaller proportions of higher river flows.

The poor passage of native fish is likely related to high turbulence, high water velocities or the baffle design not suiting fish behaviour. Turbulence is created by the energy of the water entering a pool being dissipated by the volume of the pool; measured in Watts per cubic metre [W m^{-3}] and termed Power or Energy Dissipation Factor (EDF). For salmonids 200 W m^{-3} is recommended (Bell 1973), for shad 150 W m^{-3} (Larinier, Travade & Porcher 2002) and for non-salmonids in Canada 125 W m^{-3} (Katopodis 1981). The Euston fishway is within the range of the latter two but these figures are recommended for fish > 25–30 cm in length. For small (10–25 cm), potamodromous fishes in Australia, 92 W m^{-3} appears close to the upper limit these fishes can negotiate (Mallen-Cooper 1999) and lower turbulence is needed for smaller fishes (Stuart & Mallen-Cooper 1999). Calculating EDF produces a single figure for a complex, three-dimensional environment and the distribution of turbulence may also be important. The submerged-orifice design appears to dissipate turbulence unevenly as the jet of water from the orifice hits the downstream wall of the pool directly, producing high localised turbulence. This fishway design is very old and now rarely used; vertical-slot fishways are a much more common pool-type fishway and these have an even distribution of turbulence using more of the pool volume to dissipate energy. Further work on manipulating the distribution of turbulence may lead to refinements in the vertical-slot fishway design that could provide more efficient passage for migratory fish communities with a diverse range of swimming abilities.

The maximum water velocity through the orifice of the baffle very likely exceeded the swimming ability of the smaller fish. Golden perch in a vertical-slot fishway can negotiate the salmonid standard of 2.4 m s^{-1} through a single baffle but through multiple pools the velocity needs to be 1.8 m s^{-1} for fish > 120 mm in length (Mallen-Cooper 1999). Golden perch and bony herring have a very similar burst swimming ability in a fishway (Mallen-Cooper 1999), but the later species was more restricted in the size and proportion that ascended the Euston fishway. This is to be expected given the poor passage of other clupeids in submerged orifices in fishways (Monk, Weaver, Thompson & Ossiander 1989; Larinier *et al.* 2002).

The Euston fishway also restricts the passage of two non-native species. Redfin perch could not ascend the fishway, although they were recorded in the historical data, which may have been larger fish than in the present study. Very few immature common carp could ascend the fishway, but adult fish could. Unfortunately fishways that suit the behaviour and swimming ability of native fish also accommodate common carp (Mallen-Cooper 1999). Common carp is widespread in the lowlands of the Murray–Darling river system (Harris & Gehrke 2000) and fishways in this region are unlikely to aid the spread of this species, but they do offer the opportunity to reduce the abundance of this pest species, through a carp-selective trap, and influence the population dynamics (Stuart, Williams, McKenzie & Holt 2006).

There are 21 other fishways in the Murray–Darling Basin that have been built on steeper slopes than the Euston fishway (Mallen-Cooper 1989). These fishways have smaller pool volumes and greater turbulence, and would be even more difficult for native fish to use. Replacing the baffles with the vertical-slot design would not alter the fundamental hydraulics created by the steep slope, which leaves four options for improving fish passage at these sites: (1) replace with a Denil fishway by using the same channel and replacing the baffles, as these fishways function at steeper slopes than pool-type fishways; (2) create a fish lock by removing the baffles and using the fishway as a lock chamber, which depends on sufficient height of the fishway wall; (3) use the fishway for passage of large fish and add a fishway for small fish; and (4) replace with a new fishway. The Euston fishway has recently been modified to a Denil fishway (1:6.6 slope with resting pools), following research on the passage of native fish in high- and low-slope Denil fishways (Mallen-Cooper & Stuart 2007), and another old fishway has recently been converted to an experimental fish lock (Baumgartner & Harris in press). The third option would not be effective as these old fishways provide poor passage of bony herring. In many cases, replacement of the fishway will be necessary to achieve uniform ecological objectives along rivers.

Assessing fishways

Euston fishway provides an example of using the best available technology of the time and instituting an assessment programme, which is still often lacking (Čada 1998; Agostinho, Gomes, Fernandez & Suzuki 2002). The target species and sizes at the time were adult golden perch (> 36 cm), adult silver perch (> 26 cm) and adult Murray cod (> 50 cm), as knowledge of the movement of immature fish is more recent (Mallen-Cooper 1999); to some extent the fishway met these targets. Credit is due to the designer for locating the entrance well and designing the fishway to function during some rises in river levels. However, the Euston fishway also provides a graphic example of the problem of using counts from the exit of a fishway as an indicator of fishway performance. Historical data for this fishway were reported by Cadwallader (1977) and later cited as large numbers in the context of an effective fishway for non-salmonid fish. The problem lies in large numbers not having an ecological context; there is no comparison with the migratory fish community approaching the fishway or those fish that enter and are unable to ascend. The value of using exit counts from fishways lies in monitoring migratory fish populations (e.g. Jessop 1990), which can be a useful tool for management and can indicate important trends in river health, as in the long-term data set in the present study.

Inadequate assessment of fishways leads to the assumption that the fishway design is effective and the wider result is that poor fishways continue to be built. This is the case in Australia from 1914 (Hooker 1966) until the late-1980s (Mallen-Cooper & Harris 1990) and paralleled in neotropical rivers from 1911 to the present (Agostinho *et al.* 2002).

Although fishway exit counts are still a common form of assessment (Čada 1998), there are more rigorous and investigative fishway assessments that have attempted to quantify fish behaviour below fishways with radiotracking (Barry & Kynard 1986; Bunt, Katopodis & McKinley 1999; Gowans, Armstrong & Priede 1999), relative densities of netted fish above and below a fishway (Mallen-Cooper 1999), relative proportions of marked fish locating fishways (Bunt 2001), migratory biomass below a fishway with hydroacoustic surveys (Oldani & Baigún 2002), population estimates below a weir using electric fishing (Baumgartner 2006), behaviour of individual PIT-tagged (Passive Integrated Transponder) fish entering and ascending a fishway (Aarestrup, Lucas & Hansen 2003), and comparisons of fish entering and ascending fishways (Mallen-Cooper 1999; Stuart & Mallen-Cooper 1999; Stuart & Berghuis 2002). The continued refinement of these techniques will provide more quantitative assessment of fish passage, but that is only one component of developing fishways that contribute to sustaining fish populations.

As Petts (1989) noted, in discussing the failure of salmonid fishways for non-salmonid fishes, 'the design and efficiency of fishways is dependent upon a detailed knowledge of the swimming capabilities and behaviour of migrating fish'. Mallen-Cooper (1999) suggested four steps in developing an effective fishway: (1) identify the migratory fish community (fish ecology); (2) test swimming ability and behaviour of fish in experimental fishways; (3) design and build the fishway; and (4) assess the fishway with ecologically based performance standards. There is a strong argument for applying a wider ecological view to the development and application of fishways (Northcote 1998; Agostinho *et al.* 2002), which includes identifying fish distributions and habitats, as part of a broad-scale fish movement model. The successful fishways for salmonids and shad have used this process; fishways were placed where there were spawning areas upstream and an initially simple migration model was well understood and able to be constantly refined.

In the global regions where water resource development is expanding, assessment of a new fishway should not be viewed as a substitute for undertaking fish ecology studies prior to a fishway being constructed. Modifying a fishway later to accommodate new ecological data is often very difficult from an administrative, funding and construction viewpoint. Given the time frames for ecological studies, a fish movement model in a complex river system would almost always be incomplete, especially where long-lived species are present, but it would highlight the knowledge gaps to all stakeholders and indicate where the fishway design may need to be conservative or flexible.

Ecology and fish passage

The present study provides some significant ecological observations that, combined with recent research, contribute to a movement model as a framework for providing fish passage in the Murray–Darling river system. For golden perch, the abundance of immature fish confirms that yearlings and older numerically dominate the population migrating upstream (Mallen-Cooper 1999). These immature fish exhibited a strong upstream response to a relatively small increase in flow, from a low flow, in summer; which represented a subtle change in water velocity and depth in the river. Immature golden perch also migrate upstream at high river flows during spring, summer and early autumn (Mallen-Cooper 1996) having a much longer season of migration than the adults, which move mainly in spring (O'Connor, O'Mahony & O'Mahony 2005), and responding to a wider range of flows than the adults.

Eggs and larvae of golden perch have been collected drifting downstream (Koehn & Harrington 2005) confirming an earlier hypothesis (Lake 1967; Reynolds 1983). The upstream migration of immature fish would compensate for downstream drift as larvae, and doing this during increased flow would aid passage over low-level natural obstructions such as shallow riffles or rocks bars. This behaviour would also be an effective mechanism for dispersal to maximise distribution within a semi-arid river system. It is worth noting, however, that this migration pattern may not be applicable across the Murray–Darling river system as there are genetic differences between golden perch populations, especially between endorheic systems (Keenan, Watts & Serafini 1996).

These data highlight the fish passage issues of: movement of eggs and larvae in weir pools, irrigation off-takes and pumps (Koehn & Harrington 2005); designing upstream fishways to function over a wide range of flows; passing immature and mature fish in upstream fishways; identifying spawning areas and protecting migration pathways to these; and managing flows to restore small changes in river levels and daily/weekly variability. Apart from the design of upstream fishways, most of these issues are knowledge gaps, although downstream fish passage issues have recently been recognised (Lintermans & Philips 2004). Significantly, there are no data on the survival of larvae or the habitats and survival of 0+ fish (Mallen-Cooper & Stuart 2003), which are critical life stages and may be severely limiting the sustainability of populations.

Silver perch share characteristics of golden perch movements with mature and immature fish, yearlings and older, recorded migrating upstream (Mallen- Cooper 1999) and larvae drifting downstream (Koehn & Harrington 2005). Immature silver perch also respond to small increases in flow, from a low flow, in summer (Mallen-Cooper 1996). The differences compared with golden perch are that spawning fish have been recorded during small river rises in summer, as well as during larger flows in spring, and immature fish have not been recorded moving upstream in high flows (Mallen-Cooper 1996). The fish passage issues and the absence of data on the life history of 0+ fish are similar to golden perch, but the variability of low flows may be more important for silver perch. If there is more migration at low flows, low-level weirs may also have a greater impact on this species. The dramatic decline of silver perch after 1940 coincides with the expansion of irrigation and the construction of numerous small weirs in the Murray–Darling river system, which could be a contributing factor in their decline.

Bony herring, smelt, redfin perch and common carp did not exhibit any increased migration in response to a rise in river level. The bony herring were mature and immature fish, including young-of-the-year [YOY] (60–80 mm length; Puckridge & Walker 1990), while Australian smelt were mostly YOY (20–40 mm; Pusey, Kennard & Arthington 2004). Bony herring and Australian smelt are short-lived, <6 and <3 years respectively, and spawn in response to water temperature, independently of flow (Puckridge & Walker 1990; Pusey *et al.* 2004). The proportion of the population that is migratory is unknown but their abundance in fishways suggests their upstream movements are not straying of a few individuals.

Both species have larvae that drift in the river and hence the upstream movement could be compensation for downstream drift; however, both species also spawn in lentic habitats that are isolated for many years (Puckridge & Walker 1990; Koehn & Harrington 2005). These observations suggest a potamodromous strategy where the spatial scale of movement may extend over generations rather than intra-generational migrations to specific spawning, rearing or refuge habitats. Such flexibility is needed in a semi-arid environment where connections between aquatic environments vary over longer time periods than the life spans of short-lived species.

With a spawning strategy that is little affected by regulation of stream flow and a flexible migration strategy, bony herring and Australian smelt are able to maintain abundant and widespread populations in the Murray–Darling river system (Harris & Gehrke 2000), despite numerous barriers that prevent their upstream movements except during floods. The objectives for fish passage for these species would then appear to be to maintain natural gene flow to prevent fragmentation of populations and to restore their abundance and distribution where it has been reduced. As common species, they may get a lower priority in fish passage but bony herring represents an abundant detritivore/herbivore (Merrick & Schmida 1984) and Australian smelt an abundant microphagic carnivore (McDowall 1996) that are likely to be important in carbon cycling in the river ecosystem.

Catfish are considered to have limited movements with a small home range (Pollard, Davis & Llewelyn 1996; Pusey *et al.* 2004) and only very low numbers are recorded in fishways (Mallen-Cooper 1999; Stuart & Berghuis 2002; Stuart, Berghuis, Long & Mallen-Cooper 2007). The species can be locally abundant and the limited movements fit a metapopulation model with spatially separated populations connected by occasional dispersal (Hanski & Gilpin 1991). The movements recorded in fishways would be significant for recolonisation, which is particularly important for this species as it has suffered major declines in abundance. Infrequent dispersal movements of less-mobile species have not previously been fully recognised in fish passage objectives and need to be included in a fish movement model for the Murray–Darling river system, consistent with Northcote's (1998) more holistic view of fish passage. Extending the model to include crustaceans, which migrate upstream in Australian streams (Lee & Fielder 1979; Mallen-Cooper 1999; Baumgartner & Harris in press) and which is a developing issue in fish passage (Benstead, March, Pringle & Scatena 1999; Rawer-Jost, Kappus, Böhmer, Jansen & Rahmann 1999; Fièvet 2000), would also be consistent with this approach.

The minimum size of fish that a fishway needs to pass has a major influence on design and cost. Small fish species with a comparative size to Australian smelt (< 50 mm) have been recorded elsewhere in the Murray–Darling river system migrating upstream (Harris, Thorncraft & Wem 1998; Stuart *et al.* 2004; Baumgartner & Harris in press). Diverse and abundant small fish, including juvenile and immature life stages, have also been recorded making potamodromous movements upstream in other river systems (Mader, Unfer & Schmutz 1998; Bunt, van Poorten & Wong 2001; Agostinho *et al.* 2002) but, where size data are presented, these fish have generally been > 50 mm in length. The small size at migration in the Murray River appears unusual for the middle reaches of a large river, at least for non-climbing species, and is usually more characteristic of catadromous and amphidromous species in lowland, low-gradient reaches of coastal streams (e.g. Harris *et al.* 1998). However, the Murray and Darling rivers for the lower 1000 km have extremely low gradients of < 1:25 000, which provides the hydraulic environment with low water velocities that these small fish can exploit. It presents a particular challenge for fish passage and further work is needed on the ecological role of these movements to aid the development of fishway design and of performance standards to evaluate fish passage success.

The ecology and management of non-native species in fishways is an important issue. Common carp are a long-lived species (Brown, Sivakumaran, Stoessel & Giles 2005) that migrate upstream at almost all life stages, including YOY (Stuart & Jones 2006), immature sub-adults and adults, which has aided their dispersal throughout the Murray–Darling river system. Sub-adult and adult carp migrate upstream during rising temperatures in spring and early summer (Mallen-Cooper 1996); this can coincide with spring floods, which has aided their dispersal past low-level weirs. Common carp is the only species, native or non-native, known to make specific lateral movements onto floodplains to spawn; this might provide opportunities to cull the population, as well as during longitudinal movements in fishways as mentioned earlier (Stuart & Jones 2006; Stuart *et al.* 2006).

50-year perspective

What has been learnt in 50 years? Certainly that quantitative assessment of fishways is crucial. Each fishway application is unique, and assessment is essential to optimise the application at a site, evaluate the generic aspects of the design, and collect data on migration that will further refine fishway designs. Integrating flow management with fishway function is important (Baras, Lambert & Philippart 1994), particularly adding small daily variations in flow that can be an important stimulus for migration of some species.

Potamodromy is more complex, especially in semi-arid rivers, involving more species and life stages than previously thought. The ecological model for fish passage in the Murray–Darling river system has changed from adult fish of a few species migrating upstream to one that incorporates upstream and downstream movement of the whole fish community, specifically adding: immature fish of large-bodied species dominating numbers and possibly biomass migrating upstream; a diverse range of movement strategies that may extend over generations and small-bodied species, crustaceans and low numbers of less-mobile species as important components.

The long-term data from the Euston fishway showed that fishways can be powerful tools for monitoring long-lived, as well as short-lived, fishes, and provide evidence for a decline in the health of the Murray River. The question that remains is whether these declines have stabilised or the present data are a snapshot of an ongoing process.

These lessons are being put into practice with the Murray River 'Sea to Hume Dam' Fishway Programme, where the Murray–Darling Basin Commission is undertaking to build fishways at 15 sites connecting 2225 km of river (Barrett & Mallen-Cooper 2006). These fishways are the vertical-slot design with low turbulence (40 W m^{-3}) and low maximum water velocities (1.4 m s^{-1}) for small fish ($>$40 mm) (derived from Mallen-Cooper 1992; Stuart & Mallen-Cooper 1999) with pools (3×2 m) and slot widths (0.3 m) sized to pass large fish up to 1 m in length and a high biomass. The fishway channel is lined with rocks for crustaceans, which is the practice in Germany (FAO/DVWK 2002). The fishways are designed to function for the full range of headwater and tailwater until the weir is submerged by floodwater. An integral part of the programme is assessment with radio tracking, PIT tags, electric fishing, fishway sampling of the entry and exit, and establishment of long-term data sets (Stuart *et al.* 2004). From a rocky start 100 years ago, fishways in Australia have established themselves in the last 15 years as a major tool in fish management that can complement other strategies in native fish recovery. The lessons learned can be useful for other countries and river systems with either developing water resources or native fish recovery strategies.

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