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Date: 29 / 10 / 20
Resolved to publish Yes / No

CSIRO PUBLISHING

Marine and Freshwater Research
http://dx.doi.org/10.1071/MF15284

Review

Mitigating the effects of barriers to freshwater fish migrations: the Australian experience

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Abstract. Declining fish communities characterise global freshwater environments, including those in Australia. Lost river connectivity through water resource development is a key cause of decline, disrupting fish migrations and threatening species productivity, viability and fisheries. Millions of dams, weirs and lesser barriers arising from water resources projects, road and rail transport and hydro-electricity schemes obstruct fish passage in rivers worldwide. Fishways are in place at few sites in Australia and globally relative to the numbers of barriers, and few mitigate the effects of barriers adequately. Most constrain the passage of fish communities and few have performed effectively when assessed against appropriate biological standards. Herein we focus on Australian experience within the global context of obstructed fish migrations, declining fish biodiversity and inadequate fishway performance. We review the migratory characteristics of Australian freshwater fish, identify the effects of different in-stream barriers and other habitat changes on the four classes of migratory behaviour and note how Australia's highly variable hydrology presents particular challenges in mitigating fish passage barriers. Mitigation options include: basin-scale approaches; improved management of barriers, environmental flows and water quality; barrier removal; and development of improved fishway designs. Mitigation of fish-passage problems can aid in adapting to climate change effects, reversing fisheries declines and rehabilitating fish communities.

Received 5 February 2015, accepted 25 April 2016, published online 28 June 2016

Introduction

Freshwater fish are declining globally and in Australia, reflecting the generally poor condition of rivers (Butchart *et al.* 2010; Davies *et al.* 2010; Humphries and Walker 2013). Fish are the most threatened taxa among freshwater vertebrates worldwide, suffering a decline of 76% over the past 40 years (WWF 2014), primarily through damage to freshwater habitats (Fausch *et al.* 2002; http://www.iucnredlist.org, accessed 22 May 2016). Population reductions and species losses occur worldwide. For example, the Mekong River, the world's largest inland fishery, supplies >70% of regional people's animal protein, but fish catches have fallen markedly following river regulation and land use changes (Dudgeon 2000; Osborne 2004; Ferguson *et al.* 2011). Approximately 200 additional dams are planned, under construction or completed in the Mekong catchment, further threatening fish migrations and food security (Osborne 2010; Winemiller *et al.* 2016). Similarly, following dam building, habitat changes and overfishing in three rivers in the US, diadromous species declined by 95–99% (Brown *et al.* 2013). Further, on all continents, catadromous eels (Anguillidae) have declined dramatically through migration barriers and overfishing (Boubee *et al.* 2003), with glass eel stages of European eels recently at <5% of pre-1980 levels (Astrom and Dekker 2007). In Australia, historical abundance or biomass of native fishes in Murray–Darling Basin rivers are estimated to have declined by up to 90% through habitat fragmentation compounded by altered flows, overfishing, cold water pollution and invasive species (MDBC 2003). There have been widespread local extinctions and losses of biodiversity (Gehrke *et al.* 1995; MDBC 2003; Koehn and Crook 2013). Under Australian legislation, 16% of Australia's freshwater fish are listed as threatened (*Environment Protection and Biodiversity Act* 1999; see https://www.environment.gov.au/epbc, accessed 31 January 2016).

Most (perhaps all) freshwater fish migrate at some scale, being dependent on 'movements involving regular cyclic alternation between different habitats used for spawning, feeding or survival' (Northcote 1998). They require connectivity at diverse spatial–temporal scales to complete their life cycles and to maintain their viability and fisheries values (Cowx and Welcomme 1998; Lucas and Baras 2001; Winemiller *et al.* 2016).

Among the drivers of changing river conditions (altered flow regimens or water quality, land use, habitat modification, species invasions, climate change; Vörösmarty *et al.* 2010), river regulation with lost connectivity, habitat fragmentation and obstructed migrations causes degradation and reduces ecosystem services worldwide (Jungwirth *et al.* 1998; Cowx and Portocarrero Aya 2011; Gough *et al.* 2012). Water resource development fragments river ecosystems in Australia (Kingsford 2011) and globally (Nilsson *et al.* 2005; Brown *et al.* 2013), with millions of dams and smaller barriers affecting environmental, social and economic values. Dams obstructed nearly 50% of 397 assessed global freshwater ecosystems (Reidy Liermann *et al.* 2012) and demands for water supply, irrigation and hydropower in 20 countries had established 47 655 large dams by 2001, with hundreds more planned (Gough *et al.* 2012). Furthermore, the degree of regulation in developing countries, especially in south-east Asia and South America (Osborne 2010; Winemiller *et al.* 2016), is now rapidly approaching those of developed countries. Relieving the effects of drought and extreme flow variability in Australia (Kingsford 2000; Kennard *et al.* 2010) has led to river regulation with extensive impoundment. Australia has over 500 large dams (>10 m; Australian National Committee on Large Dams (see http://www.ancold.org.au/?page_id=24, accessed 22 May 2016), and 26% of large rivers are moderately or strongly affected by dams, but even this proportion is only half the average in global systems (Nilsson *et al.* 2005; Williams *et al.* 2012). Many thousands of road and rail crossings also interrupt stream connectivity in Australia (Rodgers *et al.* 2014), mirroring the situation worldwide (Gibson *et al.* 2005). In the Wet Tropics region of Far North Queensland, for example, 3748–5536 potential barriers to fish passage, predominantly road crossings, have been identified (Lawson *et al.* 2010; Kroon and Phillips 2016). Although these barriers mostly cause local impediments to fish passage, the ubiquity of such small barriers has cumulative river basin-scale effects (Rodgers *et al.* 2014). Together with the many tidal barrages and floodgates that control flows and alter fish abundance, biomass and community structure (Williams and Watford 1997; Kroon and Ansell 2006; Boys *et al.* 2012), road and rail crossings play major roles in obstructing fish migrations.

Fishways are in place at very few sites in Australia and globally relative to the numbers of barriers, and few of them mitigate the effects of barriers adequately (Bunt *et al.* 2012; Adams 2013; Winemiller *et al.* 2016). Most constrain the passage of fish communities and few have performed effectively when assessed against appropriate biological standards (Mallen-Cooper and Harris 1990; Jackson 1997; Peterken 2001).

Herein we review the effects of riverine barriers on fish with various migration strategies and consider opportunities for mitigation, including basin-scale approaches; better management of barriers, flows, water quality and temperature; removal of outmoded or ill-justified barriers; biological criteria for fishway performance assessment; and improved fishway designs. We note how improved design and operation should address the poor performance and high cost of many fishways and help combat habitat fragmentation, adapt to climate change and rehabilitate fish communities. We focus on the Australian experience within the global context of fish migration barriers and declining fish biodiversity.

Fish migrations

Migrations determine fish viability and productivity (Gross *et al.* 1988; Northcote 1998; Lucas and Baras 2001). Fish migrate in rivers to maintain ecological processes, including population recruitment, growth and production; for dispersal to redistribute population density and enhance survivorship; and for refuge during adverse conditions (Hancock *et al.* 2000; Humphries and Walker 2013). Migrations entirely within rivers are classed as potamodromy. Diadromy indicates movements between rivers and estuarine or marine areas. Diadromous species may breed in freshwater (anadromous) or marine or brackish water (catadromous), or may move between both zones for non-reproductive activity (amphidromous; Northcote 1998; Table 1). Catadromy predominates in more-tropical areas globally, with anadromy mainly in temperate zones (Gross *et al.* 1988). Approximately 42 Australian species are diadromous; four of these are listed as threatened (Harris 2001; Miles *et al.* 2014). In comparison with other continental fauna, an exceptional number (>18 species) of eastern Australia's freshwater fish species are catadromous, and frequent catadromy drives the distinctiveness of Australian north and central coastal fish communities (Pusey *et al.* 2004). Five of the seven anadromous fish species live in southern waters (McDowall 1996).

Migrations of Australia's freshwater fish species have evolved in large river streamflows that are among the world's most variable (Peel *et al.* 2004; Kennard *et al.* 2010;

Table 1. Recognised migration patterns among Australian freshwater fish
Asterisks indicate alien families established in Australia (from Harris 2001, Koehn and Crook 2013; Miles *et al.* 2014)

Migration	Number of species	Families represented
Potamodromous	31	Ambassidae, Atherinidae, Clupeidae, Cyprinidae*, Eleotridae, Gadopsidae, Galaxiidae, Percichthyidae, Percidae*, Plotosidae, Retropinnidae, Salmonidae*, Terapontidae
Catadromous	18	Anguillidae, Ambassidae, Clupeidae, Bovichtidae, Engraulidae, Galaxiidae, Kuhliidae, Latidae, Mugilidae, Percichthyidae, Scorpaenidae
Anadromous	9	Ariidae, Galaxiidae, Geotriidae, Mordaciidae, Retropinnidae, Pristidae
Amphidromous	15	Ambassidae, Eleotridae, Galaxiidae, Gobiidae, Prototroctidae

Humphries and Walker 2013), with general aridity (mean rainfall ~410 mm year^{-1}) and low relief topography (McMahon *et al.* 1992; Puckridge *et al.* 1998). Snowmelt-driven flows are uncommon in Australian rivers, contrasting with those in the Americas, Europe and Asia. Fringing mountain ranges along the south-west and east coasts capture much of the precipitation of prevailing temperate weather systems. The northern climate is tropical, with high summer rainfall and dry winter months. Seasonality in Australian streamflow is fairly weak, except in the north, and flows are strongly influenced by drought- or flood-dominated periods that may last for decades (Puckridge *et al.* 2000; Peel *et al.* 2004).

In Australia, recognition of many migrations, especially among small fish species, was delayed until detailed fishway assessments began (Barrett and Mallen-Cooper 2006; Stuart *et al.* 2008; Baumgartner *et al.* 2010). Population declines following water resource developments highlighted the effects of dams and weirs in disrupting connectivity (Mallen-Cooper and Harris 1990; Harris and Mallen-Cooper 1994; Mallen-Cooper and Brand 2007). Fish commonly traverse rivers at intermediate scales of 10–100 km. Species movements in Australian rivers probably occur at small scales of pools or reaches and days or weeks for many small-bodied, potamodromous fish like gudgeons (Eleotridae), rainbowfish (Melanotaeniidae), hardyheads (Atherinidae) and Australian smelt (Retropinnidae; McDowall 1996; Allen *et al.* 2002; Pusey *et al.* 2004). Movements of herrings (Clupeidae), galaxiids (Galaxiidae), cods and Australian bass (Percichthyidae) are at scales of 10–100 km over weeks (Harris 1988; McDowall 1996; Reinfelds *et al.* 2013). Golden perch (Percichthyidae), eels (Anguillidae), barramundi (Latidae) and silver perch (Terapontidae) may move 1000 km over months (McDowall 1996; Harris 2001).

Types of barriers to migration

The 10 types of constructed barriers (Table 2) in Australian and international rivers have different physical or habitat-related effects on fish movements, behaviour, genetics and physiology. Dams (constructed barriers with >5 m water level discontinuity) obstruct far more expansive river habitats than weirs (<5 m) and can extirpate upstream populations (Gehrke *et al.* 2002; Gough *et al.* 2012), but this is rarer for smaller barriers (weirs, floodgates, road and rail crossings), except those at tidal limit (Mallen-Cooper and Harris 1990; Harris 2001). In New South Wales (NSW), catchment areas upstream of large dams (>10 m) total ~100 000 km^2, ~12.5% of the state's total area (Table 3). Smaller dams and weirs obstruct passage to additional extensive habitats. Dams lacking fishways are absolute barriers to upstream passage and, usually, to downstream passage. But many smaller barriers lacking fishways are sometimes passable by fish (Lintermans and Phillips 2003), at least when they 'drown out' (i.e. in flood flows, with tailwater approaching upstream levels, so the structure is no longer a barrier; Harris *et al.* 1992; Gehrke and Harris 2004; Bourne *et al.* 2011).

The Murray–Darling Basin (>1×10^6 km^2) alone holds ~4000 dams and weirs (Leblanc *et al.* 2012), with more than 10 000 barriers in total if road crossings, control banks, levees, barrages and other structures are included (MDBC 2003; Steinfeld and Kingsford 2013; Baumgartner *et al.* 2014). Flow is unobstructed in only 46% of the length of Murray–Darling Basin Ecoregion watercourses, where there are 34 fish species, 10 of which are endemic and five diadromous (Reidy Liermann *et al.* 2012). Multiple barriers have changed many of the inland's extremely low gradient rivers into chains of impoundments, with lentic systems largely replacing natural lotic patterns (Crabb 1997; Gehrke and Harris 2004; Humphries and Walker 2013). Water resource projects are concentrated in the nation's southern and eastern regions, although extensive irrigation developments planned in the tropical zone will raise fish passage issues among that region's many migratory species. Victoria has nearly 2500 barriers, over 250 of which are dams (Lay and Bennett 2001), whereas NSW has 108 large dams and many thousands of smaller dams and other barriers (Kingsford 1995; Williams and Watford 1997; Gordos *et al.* 2007). In south-east coastal drainages, one or more barriers obstruct 49% of rivers, with 32% of total habitat lying upstream (Harris 2001).

Changes in water quality and flow can impose additional behavioural and physiological barriers. For example, cold water pollution downstream of thermally stratified dams in Australia can reduce summer temperatures by 10–12°C, with slow recovery >300 km down river (Phillips 2001). Sixteen such reaches exist in the Murray–Darling Basin, affecting thousands of river kilometres (Lugg and Copeland 2014). Murray cod, silver perch and presumably other species avoid cold water or reduce their activity, affecting migrations and survival below dams (Astles *et al.* 2003; Todd *et al.* 2005; Sherman *et al.* 2007). Other water quality changes also create behavioural barriers to migration. Chronic or episodic acid discharges occur from drainages with disturbed sulfidic soils, and many species avoid small changes in pH (Kroon 2005). Plumes of acid discharge from lowland tributaries may block diadromous migrations important for maintaining large upstream populations. Problems related to releases of hypolimnetic water from dams, especially hypoxia and hydrogen sulfide contamination, are usually at smaller spatial scales.

Effects of barriers to fish migration

Fish movements occur at local patch, intermediate and basin scales (Harris 2001; Lucas and Baras 2001; Jones and Stuart 2008), so the severity of barrier effects depends both on barrier distribution and the spatial scales of migrations. Multiple barriers have compounding effects on migrant communities; for example, obstruction of one-third of migrants at each of a series of four weirs results in a calculated reduction of 81% of total migrants. This is a conservative scenario; many rivers have multiple barriers with few effective fishways. Barrier types and locations (Table 2) have different effects on migration strategies of particular species (Kroon and Ansell 2006; Reidy Liermann *et al.* 2012). Diadromous species are most vulnerable to barriers in near-coastal areas, where obstruction alienates high proportions of their upstream habitat (McDowall 1992; Pollard and Hannan 1994). Barriers at or near the tidal limit may extirpate catadromous populations because recruitment depends on upstream passage of weakly swimming, immature fish (Harris 1988, 2001). Even single minor barriers (<1 m), like causeways, culverts and floodgates, may profoundly affect population

Table 2. Characteristics and effects of structures affecting fish migrations

In the absence of an effective fishway, all listed barriers obstruct. 'Structure' refers to the type of structure and vertical separation (indicative) of upstream and downstream water levels. 'Function' refers to the usual design functions of the structure. Most barriers cause downstream channel constriction and other changes

Structure	Function	Migration impact	
		Upstream passage	Downstream passage
Dam (>5 m), high dam (>10 m)	Water storage, diversion, hydroelectricity generation, flood mitigation	Prevents passage. Without remediation, significantly alters downstream flow regimen and reduces water quality and temperature. Delay and crowding of fish favour predation and disease.[A] Decoupling of stream flow and water temperature cues to fish migration and breeding.	Prevents passage except through turbines unless spilling. Impedes movement if no passage over crest is provided. High risk of injury or mortality during spills or during passage through turbines or release valves.
Flood mitigation dam or basin (2–15 m)	Local flood mitigation	Limits disruption to passage by maintaining low to moderate flows in the stream channel through the base of the dam. Water level discontinuities only occur during flooding.	Limits disruption to passage by maintaining low to moderate flows in the stream channel through the base of the dam. Water level discontinuities only occur during flooding.
Undershot weir (2–5 m)	Diversion, re-regulation, hydroelectricity generation	Obstructs passage through physical barrier, water level discontinuity and high-velocity releases. Passage may be available when weir drowns out or gates are lifted.	Obstructs passage near surface. Causes injury and mortality of eggs, larvae and larger fish when passing under gates. Passage may be available when weir drowns out or gates are lifted.
Overshot weir (0.5–5 m)	Diversion, re-regulation, hydroelectricity generation, gauging, recreation, navigation	Obstructs passage through physical barrier and water level discontinuity. Passage may be available when weir drowns out.	Obstructs passage unless spilling. Potential injury or mortality during spills. Passage available when weir drowns out, but sufficient tailwater depth needed to prevent strike injury.
Tidal barrage, floodgate (tidally variable)	Diversion, saltwater exclusion, flood control	Obstructs passage according to degree of closure, may cause water level discontinuity and high velocity. May reduce upstream and downstream water quality. Eliminates progressive salinity gradient, limiting fish physiological adaptation.	Obstructs passage according to degree of closure, may reduce upstream and downstream water quality.
Culvert (0–2 m)	Road crossing	Varies with degree of discontinuity in depth, width and form of stream bed. May cause water level discontinuity with downstream pool, high velocity and turbulence, loss of defined channel, lack of fish resting areas. Abrupt change in lighting may inhibit passage by some species.	Varies with degree of discontinuity of depth, width and form of stream bed. May cause loss of defined channel. Abrupt change in lighting may inhibit passage by some species.
Causeway (elevated above stream bed; 0.1–2 m)	Road crossing	Varies with degree of discontinuity of stream bed depth and water level, loss of defined channel, high velocity and turbulence through pipes.	Varies with degree of discontinuity of stream bed depth and water level, loss of defined channel.
Bridge (>0 m)	Road crossing	Varies with degree of discontinuity of depth, width and form of stream bed. May cause water level discontinuity with downstream pool, excessive velocity or loss of defined channel according to degree of channel modification.	Varies with degree of discontinuity of depth, width and form of stream bed. May cause loss of defined channel according to degree of channel modification.
Floodplain earthworks (0.5–3 m)	Control floodplain flows	Obstructs lateral connectivity and fish passage into floodplain channels.	May obstruct recruitment of young fish from floodplain nurseries to main channel.

[A]Delays and crowding due to all forms of fish passage barriers favour predation, disease, disrupted gonadal cycles and energy depletion among migrating fish.

Table 3. River catchment areas in New South Wales that are obstructed by large dams (>10 m)
Data are from Australian National Committee on Large Dams (see http://www.ancold.org.au/?page_id=24, accessed 22 May 2016) and NSW DPI (2016). Data from the Murray, Snowy and Macintyre rivers exclude areas outside New South Wales. Offstream pumped-storage dams and others not located on significant watercourses are excluded

River	Total catchment area ($\times 10^3$ km^2)	Number of dams	Catchment area obstructed ($\times 10^3$ km^2)	Proportion obstructed (%)
Barwon–Darling	155.00	3	1.18	0.8
Bega	2.85	2	0.44	15.4
Castlereagh	17.40	1	0.02	0.1
Gwydir	26.60	2	5.39	20.3
Hawkesbury	21.40	23	10.92	51.0
Hunter	21.50	11	2.26	10.5
Lachlan	90.00	7	10.18	11.3
Macintyre	24.50	4	7.14	29.1
Macleay	11.45	2	0.40	3.5
Macquarie–Bogan	74.80	12	16.53	22.1
Murray	14.95	5	6.19	41.4
Murrumbidgee	84.00	11	20.12	24.0
Namoi	42.00	7	8.11	19.3
Richmond	7.00	3	0.12	1.7
Shoalhaven	7.30	6	5.96	81.6
Snowy	9.07	4	2.88	31.8
Sydney[A]	1.42	4	0.10	7.0
Tweed	1.06	1	0.60	5.7
Total	612.30	108	98.54	16.1

[A]The four Sydney metropolitan dams each have total river catchments <100 km^2.

viability (Williams and Watford 1997; Gibbs *et al.* 1999; Boys *et al.* 2012). For example, tidal barriers near the mouth of the Murray River were associated with loss of almost 90% of estuarine fish production (Collis 2012). However, larger-bodied anadromous and amphidromous species may maintain populations upstream of tidal barriers with fishways or those that have sluice gates mostly remaining open or that drown out frequently (Table 2). Diadromous populations also persist when barriers lie further upstream from the tidal limit, with success depending on habitat availability; those living down river from upland dams may maintain viable populations where sufficient habitat exists.

Most potamodromous species in Australia recruit predominantly in lowland river zones, sometimes extending seawards or into upland areas (McDowall 1996). They are severely affected by barriers in lowland and slopes regions of rivers, particularly where there are multiple barriers (Gehrke and Harris 2004; Barrett and Mallen-Cooper 2006). However, potamodromous populations often persist, to varying degrees, where there are fishways or drown outs or, alternatively, where the spacing of barriers exceeds the range of their essential movements. Numerous small-bodied and immature (<60 mm) potamodromous fish and shrimp travel upstream in Australian rivers (Baumgartner and Harris 2007; Roscoe and Hinch 2010). These may simply represent dispersal movements in times of high population pressure or, alternatively, may constitute migration as an essential recruitment process (Stuart *et al.* 2008). The potamodromous Murray cod and Australian smelt often thrive in primarily lentic reaches isolated between barriers, indicating low dependence on free passage at basin scales relative to long-range species such as silver perch or golden perch. So barrier passability and distribution, as well as species' adaptations, determine levels of local recruitment and population growth. Barriers and their lentic storages also affect populations when they halt the downstream flow of semibuoyant eggs or larvae suspended by turbulence (Lintermans and Phillips 2003; Baumgartner *et al.* 2014; Table 2). They may also kill fish during downstream passage (Thorncraft and Harris 2000; Baumgartner *et al.* 2006; Williams 2012) or disorient emigrating anadromous juveniles passing downstream through Northern Hemisphere reservoirs (Odeh 1999).

A few potamodromous species recruit in upland zones in Australia (e.g. Macquarie perch (Percichthyidae), river blackfishes (Gadopsidae), some galaxiids (Galaxiidae)), as do some salmonids, cyprinids and other non-native taxa. Although these species may persist above large barriers, their evolutionary processes can be affected by reduced gene flow (Yamamoto *et al.* 2004; Huss *et al.* 2014), causing genetic fragmentation (Green 2008) and affecting physiological traits (Volpato *et al.* 2009).

Flow regimens and associated patterns of hydrodynamic diversity are critical to Australian fish ecology (Gehrke and Harris 2004; Kennard *et al.* 2010), as in Northern Hemisphere rivers (Northcote 1998; Nilsson *et al.* 2005). However, climate change is projected to continue altering run-off and water availability, increasing river flows at high latitudes and decreasing them in dry regions (Chiew *et al.* 2010; Ukkola *et al.* 2016), although projections vary regionally in Australia. Extremes of dry and wet periods will increase in coming decades (IPCC 2014) and climate change's alterations of streamflows (CSIRO 2007; Kingsford 2011; Lucasiewicz *et al.* 2013) may increase barrier effects. Further reductions in many fish migrations are likely in response to reduced flows at both natural and artificial barriers, together with suppression of natural hydrodynamic patterns, and will be exacerbated by increased water extraction. For example, potamodromous fish in south-western Australia could be severely affected by projected streamflow reductions of ~50% (Beatty *et al.* 2014). Similarly, the abundance of migrating congolli (Bovichtyidae) and common galaxias decreased markedly as freshwater inflows into the Murray River estuary diminished in the Millennium drought (Zampatti *et al.* 2010). Climate change adaptation may require new dams and weirs, but addressing non-climate stresses on freshwater ecosystems is a necessary corollary (Crabb 1997; Kingsford 2011; Lucasiewicz *et al.* 2013).

Mitigation of obstructed fish passage

Reflecting the global situation (Jungwirth *et al.* 1998; Gough *et al.* 2012), many Australian rivers have multiple barriers lacking fishways. For example, in NSW only an estimated 170 fishways serve several thousand weirs; among ~180 barriers in the Hawkesbury River catchment, there are only ~10 fishways; and 15 of the 16 weirs on the Barwon–Darling River lack modern fishways (Harris 1988; Gehrke and Harris 2004; M. Gordos, New South Wales Department of Primary Industries, pers. comm.). These deficiencies, together with recent knowledge of the migratory life cycles of Australian freshwater fish and their requirements for free passage, highlight an urgent need for much broader mitigation of riverine barriers.

Australian governments have protected fish passage under various water, fisheries and conservation legislation and policies, and fishway construction is progressing alongside other strategies, including environmental flows, pollution abatement and dam removal (Koehn and Crook 2013; Lintermans 2013). This is encouraged by international recognition that ecological integrity and fisheries are highly dependent on fish migrations (Cowx and Welcomme 1998; Jungwirth 1998; Dudgeon 2000). More recently, the European Union's Water Framework Directive (WFD; Brevé *et al.* 2014) provided a valuable model of ways in which government policies can drive broad-scale mitigation. Under the WFD, barriers that significantly hamper migration must be mitigated or removed before the end of 2027. In The Netherlands, for example, 2924 'selected barriers' have been identified and remediation is advancing (Brevé *et al.* 2014). Regulatory management of fish passage in Australia is the responsibility of state jurisdictions plus the cross-border role of the Murray–Darling Basin Authority and varies considerably, with limited national coordination. Many fish passage barriers have been documented in Victoria (Lay and Bennett 2001), NSW (Williams and Watford 1997; Gordos *et al.* 2007) and Queensland (Lawson *et al.* 2010; Kroon and Phillips 2016), with smaller numbers in other states. From the 1980s, declining fish populations and communities in Australia prompted development of fishways suited to native species (Mallen-Cooper and Harris 1990; Baumgartner and Harris 2007; Mallen-Cooper and Brand 2007), with significant investments, primarily in low-level sites (Koehn and Crook 2013; Baumgartner *et al.* 2014). Nearly 60 fishways were built in Victoria between 1997 and 2001 (Lay and Bennett 2001).

Although large dams cause disproportionately severe impacts, mitigation of their effects is lagging worldwide compared with progress in low-level sites. Only 10% of large dams in the US have facilities to pass fish upstream and downstream (Fausch *et al.* 2002). In Australia, <3% of dams have fishways. NSW has only three fishways on its 144 dams, 108 of which are >10 m (Table 3), and 10 fishways were built on the 120 large dams in Queensland (ANCOLD, see http://www.ancold.org.au/?page_id=24.html). Although Australian agencies have started investing in fishways at low barriers, high fishways are costly and their performance so far has been very problematic (Peterken 2001; Harris 2001; DEEDI 2011), reflecting international experience (Oldani and Baigun 2002; Ferguson *et al.* 2011; Brown *et al.* 2013; Winemiller *et al.* 2016). The near total obstruction of fish passage at 500 Australian large dams, each affecting extensive catchments (Table 3), combines to cause overall effects on fish communities that may be comparable with the aggregate effects of thousands of smaller barriers with generally less-than-complete obstruction of passage.

Although fishway construction at particular sites has generally taken precedence in mitigating fish passage problems, basin-scale approaches, improved management of barriers and environmental flows, pollution abatement and barrier removals may be equally important. In the face of multiple barriers and different migratory adaptations of fish, basin-scale management is often necessary to conserve system connectivity and aquatic ecology (Winemiller *et al.* 2016). This requires barrier identification, spatial analysis, fish community evaluation, environmental flow assessments, strategic prioritisation of mitigation options or avoiding the construction of some barriers (Poff and Hart 2002). Localised management lacking these insights may risk unproductive expenditure; for example, remediating sites upstream from tidal barriers will have limited benefits in rivers supporting diadromous species. Similarly, building dams at favourable engineering sites like rapids and knickpoints may selectively extirpate specialist fast-water species.

The cumulative effect of multiple barriers fragmenting rivers has prompted planning for basin-scale approaches in Australia (Barrett and Mallen-Cooper 2006; Duncan and Robinson 2014) and internationally (Cote *et al.* 2009; Kemp and O'Hanley 2010). Multiple barriers compound effects on passage, affecting potamodromous and diadromous fish differently. The few current Australian examples of basin-scale approaches include the construction or restoration of 15 fishways over 2225 km in the Murray River (Barrett and Mallen-Cooper 2006) and mitigation of the effects of 10 weirs in >90 km of the Nepean River with fishways plus environmental flows (Duncan and Robinson 2014). Many other Australian rivers require similar basin-scale mitigation, as is happening in Europe (Brevé *et al.* 2014). Programs usually need to be staged, and so relative site priority must be decided. Multifactor prioritisation schemes are available to rate priority according to fish community data, location in the system, drown out frequency, value of adjacent habitat, presence of other barriers and practical issues (Harris 2001; Nunn and Cowx 2012; Brevé *et al.* 2014).

Mitigation also implies improved methods of barrier and flow management. Structural and operational ways to reduce barrier effects listed in Table 2 have been described by Blanch (2001) and Kingsford (2000). Effects of water storage can be reduced with off-stream reservoirs that store high flows pumped from the river, thus avoiding or minimising the size of in-stream barriers. Environmental flow management below impoundments can reinstate more natural diversity of hydrodynamics and habitats while assisting fish movements at both natural and constructed barriers (Swales and Harris 1994; Arthington et al. 2006; Overton et al. 2009). Cold water pollution strongly inhibits fish movement (Astles et al. 2003; Sherman et al. 2007), so barrier mitigation needs to be accompanied by alleviation of this problem. As an example, a 'submerged curtain' designed to reverse severe cold water pollution at Burrendong Dam (Macquarie River) releases warm surface water through the hypolimnetic valves (State Water 2014). Fish-friendly designs for road and rail crossings deal with culvert profiles, stream bed conditions and baffles to manage water profile, depth and velocity, as well as biological aspects (Boubée et al. 1999; Gibson et al. 2005; Rodgers et al. 2014). Modification of floodgate operations and design can restore passage and reduce water quality problems (Williams and Watford 1997; Gibbs et al. 1999; Walsh and Copeland 2004).

Dam removal and river restoration are increasing in the US, driven by safety, economics and environmental reasons (often fisheries; Babbitt 2002; Katopodis and Aadland 2006; Rummel and Knight 2014). Fifty-one US dams were removed in 2013 alone (American Rivers 2014), with a total of 548 removed between 2006 and 2014 (Lovett 2014; O'Connor et al. 2015). Positive biotic and geomorphic responses often followed rapidly (Tullos et al. 2014). Barrier removal programs have started more recently in Australia, reflecting the briefer history of development and the importance of dams for industry, primarily irrigation. At least 14 redundant weirs have been removed under the NSW Weir Removal Program, with others under assessment (Lintermans 2013), but progress has been slow in contrast with the US. Barrier removals require stakeholder consultation, risk assessments, sediment control and riparian rehabilitation (Lay and Bennett 2001; American Rivers 2014; O'Connor et al. 2015). Although this may be expensive, there are large potential benefits from re-establishing economic and ecosystem services lost through reduced fish populations. For example, removing the most downstream barriers in multi-impoundment systems may provide sufficient access to lowland tributaries to sustain diadromous populations. Alternative 'half-way technologies' (Brown et al. 2013), such as expensive fishways and hatchery stocking, compare poorly with the ecosystem-wide restoration benefits of dam removals. Stocking is often suggested as a compromise, but very few species are artificially bred, their genetic quality, viability and ecological effects are often uncertain and providing sufficient numbers to compensate for reductions in wild parental stocks in open systems is frequently impossible (Phillips and Lintermans 2003).

Fishway design and performance

A great variety of fishways has been built. Fishways may be classified on structural or behavioural grounds: 'technical' fishways (e.g. pool-and-weir, vertical slot, Denil) are constructed in formal, engineered channels (Clay 1995; Katopodis 2005; Williams et al. 2012), whereas 'nature-like' fishways (rock ramps, bypasses) mimic natural stream channels (Harris 1997; Jungwirth et al. 1998). Fishways may also be 'volitional', in which fish choose whether to enter and then pass through the structure (e.g. bypasses, vertical slot), or 'non-volitional', where fish are transported past the barrier after entry into the fishway (e.g. fishlifts, trap-and-haul; Larinier and Marmulla 2003; O'Brien et al. 2008; Katopodis and Williams 2012). Volitional fishways usually serve small barriers, whereas non-volitional fishways are generally built at dams. Consistent with much international practice, recent Australian low-level fishways are predominantly volitional, vertical-slot and rock ramp designs.

Fishway performance

Most fishway structures do not effectively mitigate the effects of barriers (Bunt et al. 2012; Adams 2013; Winemiller et al. 2016), frequently preventing or delaying passage (Roscoe and Hinch 2010). Despite recent advances, poor fishway performance remains problematic in Australia (Mallen-Cooper and Harris 1990; Jackson 1997; Harris 2001), as it does in Europe (Larinier 2002; Larinier and Travade 2002; FAO 2002), Asia (Osborne 2010), South America (Oldani and Baigun 2002; Agostinho et al. 2012) and the US (Roscoe and Hinch 2010; Williams et al. 2012). For example, fishways in three large US rivers passed an average of <3% of diadromous fish, with prominent species failing to use conventional fishways (Brown et al. 2013).

Fishway performance should be assessed against predetermined, comprehensive biological criteria. Ideal fishways should enable passage without delay of all fish species, sizes and individuals seeking to pass barriers at all positive river discharge levels below major flooding. They should prevent genetic selectivity and avoid injuries, mortalities and predation. Four critical factors are essential for optimal performance: attraction, entry, passage and refuge (detailed in Jungwirth et al. 1998; Stuart and Mallen-Cooper 1999; Bunt 2001; FAO 2002; Stuart and Berghuis 2002; Larinier and Travade 2002; Katopodis 2005; Mallen-Cooper and Brand 2007; USBR 2007; O'Brien et al. 2008; Baumgartner et al. 2010; Roscoe and Hinch 2010; White et al. 2011; Bunt et al. 2012; Franklin et al. 2012; Noonan et al. 2012; Williams et al. 2012; Cooke and Hinch 2013). First, fish must quickly locate the fishway, a process dependent on its position relative to the barrier and riverbank, the hydraulic and structural limits to upstream travel and the flow pattern. Second, hydraulic conditions at the entrance must encourage and enable fish to enter. Third, conditions within the fishway must match the behavioural and physiological requirements of fish for passage. Finally, refuge habitat must be available at the fishway exit(s) to avoid mortalities through predation (O'Brien et al. 2008; Agostinho et al. 2012) or entrainment in adverse flows (Odeh 1999; Thorncraft and Harris 2000).

The effectiveness and efficiency of fishways may be estimated from the proportions of individuals, sizes and species in the migrating community that are observed passing (FAO 2002; Larinier and Marmulla 2003; Roscoe and Hinch 2010). Quantitative efficiency data may be used to test for delayed passage (Castro-Santos and Haro 2003). Fishways should ideally provide constant passage for whole communities. However, the impracticality of achieving passage of all individuals at all times given extreme river flows or the presence of very small or large species or life stages requires setting pragmatic targets. For example, fishways in the Murray–Darling Basin aim to allow passage of fish of 40–1000 mm (Barrett and Mallen-Cooper 2006). Although this criterion represents great improvement over earlier fish passage performance, there is evidence of large-scale movements of fish as small as 12 mm seeking upstream passage (Baumgartner and Harris 2007).

Fishway designs must integrate biology with hydraulics and hydrology, addressing hydraulic heterogeneity, operational continuity, carrying capacity, entrance arrangements and design of upstream exits (Cowx and Welcomme 1998; FAO 2002; Larinier and Marmulla 2003). Hydraulic heterogeneity is essential in catering for diverse swimming abilities and preferences, allowing all species and most sizes to pass (Stuart and Berghuis 2002; Baumgartner *et al.* 2010; Williams *et al.* 2012). Each fishway's setting is unique and there are four 'critical lessons' for design (Baumgartner *et al.* 2014): (1) fishway design should be based on local fish biology; (2) collaboration between hydraulic engineers and biologists is critical (Katopodis and Williams 2012); (3) innovation and experimental research are needed to improve effectiveness; and (4) biological, structural and hydraulic conditions need to be monitored in recording performance for ongoing improvement (Bunt *et al.* 2012). In central Victoria, only 5% of fishways built from 1999 to 2001 were monitored (Brooks and Lake 2007), although assessments and monitoring are now increasing in Australia (Barrett and Mallen-Cooper 2006; O'Brien *et al.* 2008; Walsh *et al.* 2014).

Good fishways should also have low construction, maintenance and operating costs, adaptability to different barriers, resistance to debris and flood damage, continuous operation, ease of monitoring and flexible operations to maximise performance. These objectives are particularly difficult in Australia, especially through hydrological variability requiring fishway designs that cope with marked changes in water levels above and below barriers. Most fishway types may be susceptible to flooding, debris fouling and sedimentation, and the remote location of many Australian barriers further complicates monitoring and maintenance.

Low-level fishways

Recent Australian vertical-slot fishways (Mallen-Cooper and Brand 2007; Stuart *et al.* 2008; Duncan and Robinson 2014) and a Deelder open-lock fishway (Baumgartner and Harris 2007) have been effective on weirs <4 m, enabling upstream passage for many species and sizes of fish over wide flow ranges. However, estimates of percentage passage are lacking for most species and sites. This is due, in part, to the limitations of tagging small fish for assessments (Koehn and Crook 2013). There is also difficulty in interpreting the motivation of fish holding near fishway entrances: they may be attempting passage, feeding or, at tidal barriers, adapting to freshwater. At assessed low-level fishways in Australia, comparisons between data from fishway entrances and exits have shown discrepancies indicating suboptimal passage through selection for species, sizes or behaviour, or else inadequate biomass capacity (Stuart and Berghuis 2002; Mallen-Cooper and Brand 2007; Stuart *et al.* 2008). Poor passage may also be due to slow rates of ascent, with diurnal migrants returning downstream after failing to reach fishway exits by nightfall (White *et al.* 2011). With ongoing suboptimal passage, populations are likely to decline and community diversity diminish (Cote *et al.* 2009 Kemp and O'Hanley 2010; Bourne *et al.* 2011). Interpreting fish passage outcomes by comparing abundance in upstream and downstream habitats is difficult because of habitat differences, so that tagging methods are required, albeit with their biases for species and sizes.

High-level fishways

The performance of high-level fishways in Australia and internationally has been considerably worse than at low-level sites (Jungwirth *et al.* 1998; Stuart *et al.* 2007; DEEDI 2011), discouraging further investments (Peterken 2001; Harris 2001). Dams on larger rivers (>100-m width) present particular challenges for fish passage, and high costs inhibit the desirable installation of fishways on both riverbanks. High-level fishways have considerably greater maintenance and operational requirements in relation to staffing, numbers of moving parts and energy usage. They may increase the energetic demands on fish and cause behavioural issues that limit fishway performance (Williams *et al.* 2012). Mechanical breakdown and faulty automated control systems are very common problems (Berghuis *et al.* 2000; White *et al.* 2011; Walsh *et al.* 2014) and Australian fish locks and fish lifts have generally functioned for only approximately half the time or less (Stuart *et al.* 2007; DEEDI 2011). Performance in terms of reliability and the fish species and size ranges passed has not yet approached acceptable criteria.

There are at least nine fish locks on Australian dams. Those featuring well-located entrances and designs appropriate for the variable headwater and tailwater levels have initially performed well, but most are currently non-functional through design, mechanical and maintenance failures (T. Marsden, Australian Fish Passage Services, pers. comm.). For example, the Yarrawonga Weir fish lock (Murray River) has inappropriate cycling frequencies and velocities and the adjacent hydropower intake entrains and kills fish exiting the fishway (Thorncraft and Harris 1997). This fish lock has been replaced by a manual trap-and-haul system. Different issues affected Eden Bann fish lock (Fitzroy River; Stuart *et al.* 2007) and other sites, with poorly located entrances, narrow operational ranges and mechanical and software failures.

Three fish lifts exist in Australia. Paradise Dam fish lift (Burnett River) passes individuals of some upstream-migrating species during low and medium flows but fails during high flows, when migration is greatest, primarily because of poor entrance design (DEEDI 2011). After 4 years' operation of the fish lift at Tallowa Dam (Shoalhaven River), individuals of 11 species had passed upstream. However, only 40% of individuals tagged at the entrance were passed and several of the diadromous species have not yet re-established (Walsh *et al.* 2014). Most high fishways depend on trap systems for their operation, but trap avoidance and escapement can severely limit effectiveness. Trap escapement was initially problematic at the trap-and-haul fishway on Hinze Dam (Nerang River; O'Brien *et al.* 2008), necessitating re-design (J. Harris, unpubl. data).

Mechanisms of passage failure

Early fishways (pre-1980) in Australia were adaptations of salmonid fishways that failed to suit local species, which generally have lower swimming ability and hydrodynamic tolerances (Mallen-Cooper and Harris 1990; Mallen-Cooper and Brand 2007); in addition, they rarely leap at barriers and have other differences in performance and behaviour. Various factors apart from fish behaviour and physiology determine the success of fish passage through fishways. Exogenous environmental or structural causes of failure include poorly located entrances, inadequate attraction flows, 'false attraction' in tailwaters, secondary energetic costs (escaping predators, stressful conditions) or fishways with insufficient biomass capacity (Clay 1995; Stuart and Berghuis 2002). A survey of Australian and international experts (White *et al.* 2011) emphasised the universal importance of fishway entrance conditions and the relationships between fishway structure, species behaviour and diel cycles. Katopodis (2005) stressed the need for scientific ichthyohydraulic investigations of these issues.

The effects of barriers and fishways on physiological mechanisms selectively altering fish behaviour, genetic composition, reproductive performance or rates of passage are also important but seldom examined (Roscoe and Hinch 2010). Delayed passage can alter migratory behaviour, with displacement of spawners from preferred spawning areas, reductions in production or egg viability and genetic effects (Katopodis 2005; Volpato *et al.* 2009). Statistical analyses of telemetry data tracking individuals through time commonly fail to detect the effects of delayed passage, but event–time analyses can quantify passage rates for such studies (Castro-Santos and Haro 2003).

Understanding the factors affecting fish behaviour at fishways requires detailed information on the movement patterns of individually identifiable fish, which is best accomplished using telemetry (Bunt *et al.* 2012). Sonic, radio or passive integrated transponder (PIT) tags that enable behaviour and attraction, entry and passage to be quantified are widely used. Small PIT tags are now available for use in small-bodied species. Important new knowledge is resulting from these electronic techniques (Barrett and Mallen-Cooper 2006; Koehn and Crook 2013; Reinfelds *et al.* 2013). Tagging-based studies may entail questionable assumptions about the effects of tagging on subsequent animal behaviour and physiology through stress and other effects of capture, anaesthesia and tagging (Cooke and Hinch 2013). But extrapolations from research with higher vertebrates are inappropriate, because the neuroanatomy of fish differs significantly and fish may show normal feeding and activity immediately or soon after surgery (Rose *et al.* 2014). Well-designed studies are needed to analyse this issue. Linking tagging results to data on community structure at fishways may provide helpful corroboration (FAO 2002; Cooke and Hinch 2013).

Downstream passage

Downstream fish passage has received less attention in Australia than overseas (Lintermans and Phillips 2003; Kemp and O'Hanley 2010; Williams *et al.* 2012). Technology for bypassing downstream migrants at dams has often been ineffective (Odeh 1999; Nestler *et al.* 2008; Brown *et al.* 2013) and migrants are often killed when entrained in hydropower station intakes (Odeh 1999; Thorncraft and Harris 2000; Richkus and Dixon 2003). Large-scale anadromous migrations are unusual in Australia; downstream passage at dams has rarely been implemented and results have sometimes been poor. For instance, many fish have been killed by impact with the stepped spillway at Paradise Dam (Williams 2012). Following previous fish kills in spilling flows at the original Hinze Dam, lowered spillway slots were installed to assist downstream passage when the dam was raised (O'Brien *et al.* 2008). Preliminary results from small spilling events have been generally positive, with some abrasions noted on fish but no mortalities observed (D. Roberts, Seqwater, pers. comm.), and, at Tallowa Dam, a smooth-coated slot in the ogee spillway has passed fish without injury (Walsh *et al.* 2014).

Weirs and dams with undershot gates, where water is released from the base of the structure, also kill downstream-passing fish and their eggs and larvae (Lintermans and Phillips 2003; Baumgartner *et al.* 2014). In contrast, nature-like and pool-type fishways can pass fish safely downstream (Jungwirth *et al.* 1998; Lintermans and Phillips 2003). Alternatively, lay-flat gates may be fitted at low-head weirs and can be lowered in high flows to protect downstream migrants (L. Baumgartner, unpubl. data).

Improving fishway performance

Research and development in Australia over the past two decades have produced significant advances in performance of low-elevation fishways. In particular, refinements of vertical-slot (Barrett and Mallen-Cooper 2006; Baumgartner *et al.* 2006; Stuart *et al.* 2008) and Denil open-lock (Baumgartner and Harris 2007) designs have succeeded in increasing the size range, abundance and species representation of native fish communities passing barriers <5 m. There has been substantial progress towards achieving biologically based criteria for fish passage at these sites, but important issues remain, including the high costs of fishways, which inhibit their widespread application; the critical performance failures of barriers >5 m; and ensuring passage of all species, individuals and life stages seeking to migrate. Improved measurement procedures are also needed for assessing performance, especially capacity, delayed passage (Castro-Santos and Haro 2003) and aspects of the four stages of fishway function (Roscoe and Hinch 2010; Noonan *et al.* 2012; Koehn and Crook 2013). Key areas for further development include nature-like and trap-and-haul designs and innovative fishways.

Nature-like fishways

Fishways mimicking natural stream channels have a long history in Europe and are popular in Australia and the US (Parasiewicz *et al.* 1998; FAO 2002; USBR 2007) and include rock ramps and bypass channels serving smaller barriers (Jungwirth *et al.* 1998; Katopodis 2005; Wildman *et al.* 2014), but they vary greatly in design and performance (Harris *et al.* 1998; Franklin *et al.* 2012). Poor passage results when attraction flows are too low (Bunt *et al.* 2012), when entrances are badly located or when design, construction or maintenance issues cause points of excessive head loss. Assessment and monitoring of nature-like fishways by direct sampling is difficult because of their informal structure, so that tagging methods may be required. Advantages of nature-like designs include hydraulic heterogeneity that provides multiple migration paths, adaptability to variable flows, continuous operation and compensatory aquatic habitat (Bretón *et al.* 2013). The proposed Traveston Crossing Dam (Mary River) was planned to include an innovative design for a high-level (21 m) bypass channel that adapted to storage draw downs and created compensatory aquatic habitats, but was not built (Queensland Water Infrastructure, unpubl. data).

Trap-and-haul fishways

Trap-and-haul fishways provide passage over dams where site difficulties preclude other systems. For upstream passage, fish follow a channel into a trap area, where a hopper lifts them into a tanker for relocation. This system is successfully implemented on ~16% of hydroelectric dams in the US (Cada 1998; USBR 2015), but the Hinze Dam fishway is the first of this type in Australia and is nearing completion of commissioning (O'Brien *et al.* 2008). It cost substantially less than alternative designs, operating effectively with modest manpower needs and serving a broad size range of fish. Five separate release sites are used to avoid learned behaviour among fish and bird predators. To transport downstream-migrating juvenile salmonids past large reservoirs in the US, screened collection systems, barges and bypass channels are used (Odeh 1999), but have not been required in Australia, which has few large-scale anadromous species.

Innovative fishways

There is a need for innovations in fishway design to address the high costs and poor performance currently impeding fish passage mitigation. In Australia, as in many other countries, low rates of fishway installation are due to both the high costs of fish passage programs plus dissatisfaction with the results of many fishways, especially those at high sites. Experimentation and novel approaches are needed. Applicability to diverse new and existing barriers, especially dams, modular construction systems to enable prefabrication of components, energy independence, flood protection and fewer constraints due to water levels are all important. In Australia particularly, difficulties associated with installing and operating fishways at remote, often unpowered, sites will require designs with few moving parts and low maintenance requirements, as well as independently powered operation.

Conclusions

Damaged connectivity disrupts riverine fish migrations in Australia and worldwide, severely reducing biodiversity and productivity and threatening fisheries. The Australian native fauna has low species diversity, high endemism and frequent catadromy, while lacking the Northern Hemisphere's anadromous salmonids, cyprinids and other elements. Several features characterise Australia's experience in fish passage: extremely variable hydrology, a high incidence of waterway regulation and its distinctive fish fauna. Physical barriers, flow modification and diversions, water quality and habitat changes, overfishing, invasive species and extensive cold water pollution unite to cause fish community degradation.

Notwithstanding the differences, broad patterns of migratory adaptations occur worldwide, and Australian fish passage problems and solutions generally mirror those elsewhere. To greater or lesser degrees, diadromy and potamodromy drive the viability and productivity of freshwater fish globally, and demands for water resources, energy and transportation create comparable losses of riverine connectivity. Similarly, fish passage technologies have universal general application, albeit with variations required for differences in fish physiology and behaviour, system hydrology or other factors. Successful fishway innovations are likely to be of value worldwide.

Advanced fishway designs, improved barrier management and accelerated mitigation are needed to reverse the declining condition of Australian freshwater fish. Whole communities of migratory fish, rather than 'species of concern', need to be catered for and fishway performance needs improvement to meet biologically based standards. At low-level sites, the best recent fishways have progressively reduced, but not eliminated, most adverse effects of barriers. Performance deficiencies remain severe at high-level sites. Compared with the scale of the problem, present rates of fishway construction are far too low. Enhanced research, innovation and development are needed, as well as programs for catchment-scale management, improved flow regimens and barrier removal. The comprehensive approach to mitigating fish passage barriers by Europe's WFD (Brevé *et al.* 2014) provides a valuable model. A progressive culture is necessary among water resources and fisheries agencies, with willingness to experiment and accept appropriate risk. Interdisciplinary approaches to design, assessment and monitoring are imperative. Barrier mitigation in Australia has focused on the abundant small barriers, rather than dams, but this approach requires revision because the overall values of catchments for biodiversity and fisheries are much more severely compromised by dams, where fishway performance lags far behind. Analyses of the performance and costs of high fishways are required to avoid perpetuating existing problems and to facilitate fish passage projects at dams, with their disproportionately greater ecological effects. Reducing the costs of both high and low fishways is crucial.

Research and management of fish passage problems in Australia have been communicated through 4-yearly technical workshops (Berghuis *et al.* 1997) supported by universities, water agencies and governments. This process should be formalised and occur more frequently in order to accelerate progress and to guide research and policy. A management group, possibly supported through fisheries funds and professional, water resources, conservation and angling bodies, should be established to enhance knowledge sharing among stakeholders, managers, researchers, engineers and the community.

Acknowledgements

The authors gratefully acknowledge the support of Martin Mallen-Cooper, Matthew Gordos, Tim Marsden, David Roberts and Chris Dann, who contributed data and advice on aspects of the work.

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