Susceptibility of Australian fish to entrainment through irrigation systems with a review of research and potential mitigation strategies.

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Summary

Cotton is one of Australia's important agricultural industries. There are over 1500 cotton farms across the country and much of the production of cotton relies on an irrigated water supply. The cotton industry is environmentally conscious and continues to strive for excellence in achieving sustainable cotton production. One area where the industry will be able to improve its environmental footprint is through fish friendly use of water.

Fish are known to be entrained through irrigation systems and once in irrigation storages or channels, are permanently lost from the river system. This report reviews the current literature on the susceptibility of Australian native fish to entrainment and impingement and potential mitigation options and their relative merits and costs.

The likelihood of a fish being entrained depends in part on their swimming ability and behaviour. This varies between species and life history stages. Juvenile and larval fish have poorer swimming abilities than adults. Life history phases that swim with the current rather than against it are more likely to be entrained. Some knowledge of the swimming abilities of Australian native fishes exists, but many species have no available data. The burst speed and sprint speeds of fish are probably the most important indicator of a fish's ability to avoid entrainment. To cater for the weakest swimming species and life history stages, an approach velocity of $0.1ms^{-1}$ is conservatively recommended for Australian fish species.

There are various mitigation options available to prevent fish from becoming entrained or impinged in irrigation systems. These can be designed to maintain desired pumping volumes without harming fish. Fish mitigation options can consist of both behavioural and physical barriers. Many of the modern physical screens have self-cleaning mechanisms to help maintain optimum flow volumes. Screens manufactured from wedge-wire have good water flow characteristics and lend themselves to efficient cleaning. For pumped offtakes wedge-wire cylinder screens with self-cleaning brushes offer an effective and proven screening technology. Cone screens are also suitable for pumped offtakes that are in shallow water sites with low flow velocities. These can also be used to screen some gravity fed diversions. Gravity fed diversions can be effectively screened within channel with a range of other options, including rotating drum screens, horizontal fixed plate screens, vertical fixed plate screens and travelling screens. Within channel screens normally divert fish via a bypass channel back to the river. For screens to function properly they need to be orientated correctly to the flow to ensure there is adequate sweeping velocity to help prevent impingement of fish. Approach velocities need to be kept around $0.1ms^{-1}$ and are minimised through correct orientation of the screen to the current, adequate screen surface area and the use of internal baffles.

Behavioural screens use electric fields, bubbles, lighting, turbulence, or sound to deter fish from approaching irrigation diversions. These screens have no impact on pumping efficiency but are only partially effective at preventing entrainment. Limited effectiveness on small juvenile fish and some species groups have been reported. Of these technologies, acoustic deterrents, or acoustic deterrents in combination with bubbles and strobe lights may offer some potential mitigation at sites where it would be logistically difficult or cost prohibitive to install a physical screen.

Although it is likely that physical screens will have benefits to farmers through reduced debris and cleaner water leading to less time unblocking centre pivots, and sprinkler mechanisms, screens are

still a major capital expense. Subsidies or grants to assist with purchase and installation costs would be a useful mechanism to encourage voluntary uptake of fish screening in Australia.

Introduction

Cotton is an important agricultural industry in Australia, and there are over 1,500 cotton farms across the country. The cotton industry employs over 12,000 people and operates primarily in New South Wales and Queensland, with small areas of production in Victoria, the Northern Territory and Western Australia. Much of the cotton crop relies on an irrigated water supply and the industry has become increasingly water efficient (Cotton Australia, 2020). The Cotton industry is environmentally conscious and continues to strive for excellence in achieving sustainable cotton production. The Cotton Research and Development Corporation (CRDC) has developed a strategic RD&E plan for the period 2018-2023 (CRDC 2018). Through this plan the Cotton industry aims to increase economic, social, and environmental benefits for the Australian cotton industry and wider community, by investing in knowledge, innovation and its adoption. One part of the plan focuses investments in RD&E that ensure Australian cotton continues to be produced to the highest environmental and social standards, with an improved environmental footprint.

An area of investment where the cotton industry can reduce its environmental footprint is through improvements to fish friendly cotton production. Australian freshwater fishes are of social, economic and cultural value (MDBC 2013), and indeed freshwater fish and fishing are highly valued by many who work in the cotton industry. The potential for native freshwater fish to be entrained through irrigation systems, including pumped diversions and gravity fed diversions, has been recognised since early this century. Blackley (2003), the MDBC (2004), Baumgartner (2005) and King *and O'Connor* (2007) were among the first to draw attention to this potential impact on fish in Australia. Subsequent research on pumped diversions (*e.g.* Baumgartner *et al.* 2009; Boys *et al.* 2012; Norris 2015) has demonstrated that native freshwater fish are indeed susceptible to entrainment through Australian irrigation systems and can suffer significant injuries and mortalities. Fish that are entrained in irrigation diversions and survive are effectively permanently lost from the river system. Some species and size classes appear to be more susceptible to entrainment than others, and this may be in part due to swimming ability, behaviour, and the location of the offtake (Ehrler and Raifsnider 2000; Baumgartner *et al.* 2009; Norris *et al.* 2020).

In recognition that cotton irrigation has the potential to entrain native fish, the CRDC has commissioned on-ground research to investigate the relative impact of different irrigation offtake systems on native fish, in order to develop best management practices and to prioritise where mitigation efforts should be directed.

This review document is comprised of two sections. The first part reviews current available knowledge of the swimming performance of native fish in two key Australian irrigation zones (The Murray-Darling Basin and the Fitzroy Basin). This information will enable determination of which species or size classes may be most susceptible to entrainment and provide a basis for the approach velocities that may be required to prevent impingement of the weaker swimming species of fish around irrigation screening systems. The second part reviews current knowledge of different mitigation systems used at water diversions around the world, including various types of physical barriers (screens) and behavioural barriers or deterrents. The different options are examined, with consideration to cost, operating efficiency and their effectiveness for preventing entrainment and impingement of fish.

The objective of this research is to present options that allow cotton irrigation to continue, with minimal or no impacts on pumping volumes or pumping efficiency, whilst minimising loss of fish from rivers and weir pools. Some options may even have beneficial outcomes for on farm irrigation system efficiency, especially for those systems that use centre pivot type water distribution systems, that can become clogged with small debris and even small fish.

Chapter 1: A review of swimming performances of fishes in the Murray-Darling Basin and the Fitzroy River: A summary for approach velocities required for irrigation pump screening

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Introduction

In 2017-18, 10.5 million mega litres were used to irrigate farmed lands across Australia (ABS 2019), and early this century 72% of all irrigated lands occured within the Murray-Darling Basin (MDB) (Meyer 2005). Fish extracted from rivers in water for irrigation use are permanently removed from the ecosystem if no return mechanisms are in place (Baumgartner *et al.* 2009). There are multiple methods for extracting water for irrigation, including gravity–fed diversion channels, direct pumping and syphoning from the river. Of these methods, mechanical pumps are frequently used, particularly in northern MDB inland rivers. This method of water extraction generally removes large quantities of fish of different size classes from the river (Kingsford 2000a; Baumgartner *et al.* 2007). Entrained fish can be either severely injured or killed instantly by the impeller and the sudden increase in pressure and turbulence within the irrigation pipes (Neitzel *et al.* 2000; Baumgartner *et al.* 2007). Fish that survive passing through the pump system will become permanently isolated to the irrigation channel or off-river storages. Unscreened water off-take pumps can account for entrainment of up to 1130 fish per ML in the northern Murray-Darling Basin (Norris 2015). In the southern MDB, irrigation channels can entrain multiple fish species, ranging from small to large-sized species. Common small-bodied species, such as carp gudgeons and smelts, are frequently extracted, as well as important recreational species, including juvenile Murray cod (King and O’Connor 2007) and larval and adult golden perch (O’Connor *et al.* 2008). Solutions to prevent or reduce fish removal via water extraction need to be developed and employed to reduce the number of direct mortalities and loss of fish from Australian rivers and help decrease the environmental footprint of the Cotton industry.

In the UK, legislation has been in effect since 1975 for protecting salmon and freshwater fisheries to prevent entrainment of fish (Turnpenny *et al.* 1998). The associated regulations detail the minimum requirements needed for physical screens and the use of behavioural screens and fish by-passes. Similarly, in the US, extensive guidelines and criteria are in place to facilitate salmonid passage, including regulations for the use of screened intakes for hydropower and irrigation use (NMFS 2008). In Australia there are currently no legal obligations or guidelines for pump operators to install screens for irrigation use.

Screening pump intakes can reduce the number of fish entrained and reduce the intake of debris that causes pipe blockages and impeller damage (Boys *et al.* 2013a). The effectiveness of pump screening largely depends on the pore size of the mesh screen, intake current velocity adjacent to the screen and the surface area of the screen. Velocity and mesh size are important to exclude small size classes of fish, and screen surface area is important to prevent loss of pumping efficiency. Although there are commercially produced pump screens available on the market and being used in New Zealand, the USA, Canada and parts of Europe, the availability and the use of screens are limited in Australia (Boys *et al.* 2013b). Currently, there is only one manufacturer (AWMA) in Australia that designs and installs fish exclusion screens, although suppliers of imported fish screens are increasing. However, the effectiveness of these screens on native Australian fish species is still not well understood as most studies have been conducted outside of Australia. Preliminary research has been undertaken on the effectiveness of pump screens for reducing entrainment for some Australian species (Baumgartner *et al.* 2009; Boys *et al.* 2013a; Boys *et al.* 2013b, Norris *et al.* 2020) and some field trials are currently underway, particularly in New South Wales.

However, improperly designed or installed screens can result in fish being impinged on the screen due to high approach water velocities. Although some studies in the US suggest that screen impingements have only minor effects (Danley *et al.* 2002; Rose *et al.* 2008), the full extent of fish impingement on screens is still largely unpublished, especially regarding Australian species. Despite limited knowledge of its impacts, efforts can be made to greatly reduce impingement by controlling the water velocity near the surface of the pump screen.

Of the many factors that influence impingement, the approach velocity and the sweep velocity are the most influential. The approach velocity (water velocity vector perpendicular to the screen) increases exponentially the smaller the distance is to the screen's surface (McMichael *et al.* 2004; Peake 2004; Boys *et al.* 2013a). The sweeping velocity is the water velocity vector running parallel to the screen surface (McMichael *et al.* 2004). Factors that can influence the water velocity at a screen's surface can include head loss and screen submergence (McMichael *et al.* 2004). The approach velocity, relative to the sweeping velocity, is generally the primary factor used when focusing on fish impingement. In an ideal scenario, the approach velocity for pump screens should be governed by the weakest swimming species in the local rivers to maximise the probability of all species being able to escape impingement. This can be determined based on swimming performance data of various species. Traditionally, screen design criteria on approach velocity in the U.S. are based on laboratory studies from fish swim tunnel experiments using prolonged swimming performance (NMFS 2008). These are generally unrealistic scenarios to real-life conditions. However, acknowledgement of the different swimming modes fish use to escape impingement can provide valuable knowledge in determining approach velocity criteria.

This review focuses on swimming abilities of fish species from the northern MDB and Fitzroy River in Queensland. These two river systems encompass the two largest catchments in Queensland and form the two major sources of water extraction in the state for irrigation use. Many of the species in these catchments are more widely distributed in Australia, so the information collated has wider application. Within the Fitzroy River Basin, 90% of water extraction is supplied to irrigated agricultural crops, with many irrigators having off-river ring tank storages (Loch and Rolfe 2000). Similarly, the northern MDB has had significant water extractions into private off-river storages since the 1980s (Kingsford 2000b). The two river systems have many species and genera in common. The MDB contains more species representative of temperate regions, whereas the Fitzroy system includes species more representative of tropical and sub-tropical regions. An understanding of the swimming speeds of fish from these catchments will provide insight into those species more likely to be susceptible to entrainment by irrigation infrastructure and identify estimates for approach velocities that should permit the majority of fish to escape entrainment or impingement.

Categories of swimming performance and its uses

Prior to the 1960s, the swimming speeds of fish were originally categorised into 'burst' and 'cruising speeds' (Brainbridge 1960). Categorisations of swimming speeds were redefined in the subsequent literature. Beamish (1978) divided swimming performance of fish into three main categories: burst, prolonged and sustained speeds. Burst speed can be defined as the maximum swimming speed obtained via anaerobic metabolism in less than 20 seconds (Beamish 1978; Hammer 1995). Prolonged speed is the speed obtained involving both anaerobic and aerobic metabolism between 20 seconds to 200 minutes (Beamish 1978). Sustained speed is the speed maintained for over 200 minutes using only aerobic metabolism (Beamish 1978; Hammer 1995). Subsequent testing of a fish's swimming ability within these categories was examined by the introduction of incremental velocity tests involving swim tunnel respirometers and this is known as the critical swimming speed, *Ucrit* (Brett 1964; Beamish 1978; Hammer 1995). Arguably, the critical swimming speed can be considered as an extension of the prolonged swimming speed, accounting for a lesser timeframe. The testing protocol for critical swimming speed generally involves acclimating individual fish for a certain amount of time. The fish is then subjected to an increment of set water velocities for the duration of set time increments until the fish reaches exhaustion and swimming failure is observed (Brett 1964; Hammer 1995; Farrell 2008; Peake 2008). The critical swimming speed is a way of measuring the prolonged swimming speed and can be calculated by the formula: $Ucrit = Ui + [U(ti/t)]$ where Ui is the penultimate velocity at which an individual fish reaches swimming fatigue; U is the set velocity increment; ti is the time spent swimming in the final velocity increment; t is the time increment (Brett 1964; Hammer 1995; Peake 2008; Starrs *et al.* 2011).

Among the various methodological uses of *Ucrit* in current literature, the use of shorter time increments has been widely adopted to suit the needs of independent studies. The original incremental time of 60 minutes recommended by Brett (1967) was later recognized as too time consuming and has its limitations (Beamish 1978; Hammer 1995). Subsequent studies in the 1990s and 2000s have identified the use of shorter incremental times to as short as less than 300 seconds (although commonly in 20 minute increments) as a more appropriate measure for the physiological locomotive ability of fish (Reidy *et al.* 1995; Jain *et al.* 1997; Nelson *et al.* 2002; Peake 2004; Castro-Santos 2005; Farrell 2008). Particularly, the adaptation of burst and sprint swimming durations have been categorized as constant acceleration tests and are termed as *Uburst* and *Usprint*, respectively (Reidy *et al*. 1995; Nelson *et al.* 2002; Starrs *et al.* 2011). Sprint performance can be described as the swimming speed obtained between 20 to 300 seconds involving multiple bursts with a time increment of 10 seconds (Starrs *et al.* 2011). For clarity, the aforementioned definitions for burst and sprint performance will be used in this paper. Thus, the swimming performance of individual fish can be divided into categories based on endurance time of burst (<20s), sprint (20-300s), critical (5-60 min), prolonged (60-200 min) and sustained (>200 min) performances. Here, only burst, sprint and critical swimming speeds are reported for simplicity, with special mentions of prolonged and sustained swimming speeds.

The swimming speed measures, and their specific testing protocols provide empirical data necessary to answer specific physio-ecological questions. The most prevalent use of *Ucrit* is to determine a species ability to negotiate fishways or to investigate fish passage through instream barriers Applications of swimming performance data can also be used to design solutions to mitigate infrastructure impacts, such as dams, weirs, culverts and irrigation offtakes (e.g. Shiau *et al.* 2020). Peake (2004) used swim tunnel experiments to determine the approach velocity at which juvenile northern pike become impinged on irrigation pump screens and concluded fish need to only swim for a short period to avoid impingement. Peake also demonstrated that in a more realistic intake screen scenario, the northern pike was able to escape impingement at approach velocities higher than those employed in constant swim tunnel protocols used for prolonged swimming speeds. This was primarily due to the reduction in water velocity as the perpendicular distance from the screen increased, and thus only burst swimming is required to escape impingement (Peake 2004).

Boys *et al.* (2013a) have subsequently experimented with variable approach velocities with increasing distance. Both Peake (2004) and Boys *et al.* (2013a) concluded that the use of prolonged swimming speeds is an inappropriate laboratory measure to determine screen approach velocity criteria. The use of constant acceleration tests of burst and sprint performance would be more applicable to evaluate intake screen approach velocities. Burst and sprint are the primary swimming modes that fish will be using to escape impingement. Essentially, sprint performance can provide a conservative measure where burst, critical or prolonged performance may not be the most effective measure (Starrs *et al.* 2011).

One major constraint in the current literature regarding fish swimming performance data is the lack of consistent swimming trial protocols. Methodologies have been tailored to specific research needs, with no two test protocols identical, and all have different incremental times, velocities, definitions and different flume designs (Wolter and Arlinghaus 2003). Swimming speeds are further confounded greatly by the influence of temperature, population genetic differences, oxygen availability, turbulence, size and life stage of the fish, diet, trained or untrained, reared or wild-caught and behavioural conditions (Beamish 1978; Domenici 2001; Danley *et al.* 2002; Wolter and Arlinghaus 2003; Kopf *et al.* 2014). These are all factors that vary between experiments and are further exacerbated in the field by temporal limitations.

Murray-Darling Basin Species

The recreational species in the northern MDB (*i.e*. Murray cod, golden perch and silver perch) being larger bodied species, have large differences in swimming performances across life stages (Kopf *et al.* 2014). No burst or sprint speeds of pre-juvenile (larval) life stages were found for these species in the literature. However, Kopf *et al.* (2014) examined the critical swimming speeds of the larvae stages of six native species, including silver perch, golden perch and Murray cod. (Table 1.1). Although the critical swimming speed (*Ucrit*) is not as explicit as the sprint speed in determining a fish's ability to escape impingement, it still provides intrinsic insight into a fish's swimming capability to overcome certain instream barriers under normal swimming behaviour. Critical speeds would provide a much more conservative velocity to screen designs, as burst or sprint speeds are generally higher than critical speeds.

It is also important to understand the variable risk applied to the various life stages of susceptible fish. Kopf *et al.* (2014) demonstrated, critical swimming speeds significantly differed with the ontogeny and size of the fish. Larval stages are significantly weaker swimmers than their juvenile or adult conspecifics. This was also reflected in a field study by Boys *et al.* (2013b) in that smaller sized fish (<150mm) show greater impingement rates, particularly at higher velocity. This was further exacerbated by the rheotactic behaviour of fish (Danley *et al.* 2002, Boys *et al.* 2013b). Fish subjected to higher velocities tended to swim with the current, rather than against it (Danley *et al.* 2002).

Table 1.1: The critical and prolonged swimming speeds of the different larval stages of silver perch, Murray cod and golden perch. This table was adapted from Figure 3 and Table VII from Kopf *et al.* (2014).

**Ucrit* at 5min and 0.048 m/s increments; prolonged speed based on when >75% swum over 60min.

Species	Life stages	Swimming Capability (m/s)	
		Critical (<60min)	Prolonged (60-200min)
Bidyanus bidyanus (Silver Perch)	Metalarva	0.25-0.46	0.011
	Post-flexion	0-0.1	0
	Flexion	0-0.1	0
	Pre-flexion	0-0.1	0
Maccullochella peelii (Murray Cod)	Metalarva	0.25-0.4	0.107
	Post-flexion	0.15-0.35	0.107
	Flexion	0.1-0.25	0.059
	Pre-flexion	0.05-0.2	0.011
Macquaria ambigua (Golden Perch)	Metalarva	0.05-0.25	0.059
	Post-flexion	0-0.2	0.011
	Flexion	0-0.15	0.011
	Pre-flexion	0-0.15	0.011

For silver perch, the critical swimming speed ranges from 0.001 ms-1 for pre-flexion stages up to 0.46 m.s-1 for silver perch meta-larvae (Kopf *et al.* 2014). It seems silver perch are the weakest swimmer for life stages earlier than meta-larvae when compared to Murray cod and golden perch (Table 1.1; Kopf *et al.* 2014). This is a massive difference to the swimming performance of juvenile or adult silver perch (Table 1.2; Watson *et al.* 2019a, b). Juvenile Murray cod and silver perch have an exceptional sprint speed upwards to 1.2 m.s-1 and 1 m.s-1, respectively (Watson *et al.* 2019a). Juvenile golden perch and eel-tailed catfish are the weaker swimmers of the recreational species with sprint speeds of 0.4-0.75 m.s-1 (Watson *et al.* 2019a, b). This data can be equated to the lifestyle and preferred natural habitats these species are normally found in (Lintermans 2009).

Examination of the current literature found no empirical swimming performance data available for pre-juvenile stages of smaller-bodied native fish (i.e. hardyheads, gudgeons, rainbowfish, glassfish, etc). Limited data on adult stages for some species is available (Table 1.2). Australian smelt had the lowest burst speed of 0.5 m.s-1 (Kilsby 2008) and olive perchlets had the highest burst speed of 0.5-0.9 m.s-1 (Watson *et al.* 2019a). Swimming performance data for some other small-bodied species are also in Table 1.2.

Out of the invasive species found in the northern MDB, mosquito fish had the lowest sprint speed of 0.35 m.s-1 (Starrs *et al.* 2017), but Wilson (2005) reported that they are able to burst swim between 0.73 to 1.2 m.s-1. Other larger-bodied adult invasive species (i.e. redfin perch, rainbow trout, carp and goldfish) have sprint speeds between 0.62 m.s-1 for goldfish to 0.92 m.s-1 for rainbow trout (Starrs *et al.* 2017). Carp are capable of burst speeds of up to 2 m.s-1 (Tudorache *et al.* 2007). Two of these species are unlikely to be encountered around irrigation systems in Queensland. Of these species rainbow trout has a highly restricted distribution in the upper Condamine catchment in Queensland and redfin perch is a possible rare vagrant to some sections of the Queensland Border Rivers, but redfin perch are more widespread in south-eastern Australia. The data provides some evidence of invasive species outperforming our native species in their ability to negotiate through high velocity waters and perhaps to avoiding entrainment through irrigation offtakes.

Fitzroy River System Species

Besides the species found in both the Fitzroy and Murray-Darling river systems, many of the Fitzroy system species have no available empirical data on swimming performance (Table 1.2). There was only some data in the literature for barramundi, tarpon, crimson-spotted rainbowfish, mullet, pacific blue-eyes, and speckled goby (Table 1.2). Of the available data on native species, the speckled goby had the slowest sprint speed of 0.39ms-1 (Donaldson *et al.* 2013). Like pest species in the MDB, the invasive species found in the Fitzroy had remarkable swimming speeds. Guppies (*Poecilia reticulata*) have burst speeds of up to 0.854ms-1 (Ghalambor *et al.* 2004) and juvenile Mozambique tilapia (*Oreochromis mossambicus*) have a critical swimming speed of 1.76-2.28m.s^{-1} (Table 1.2; Botha *et al.* 2018).

Table 1.2: Compilation of burst, sprint and critical swimming speeds of juvenile and adult species that occur within the Fitzroy River Basin and the Murray-Darling Basin in Queensland. Introduced species are in shaded rows. This species list is compiled from Pusey *et al.* (2004) and Lintermans (2009) with the various swimming velocities divided into adult and juvenile life stages, according to the source reference and to the species natural size range. Note Murray cod are an established introduced population in the Fitzroy River Basin near Emerald and silver perch have been stocked in Fairbairn Dam near Emerald.

*Mean velocities are presented here unless if the mean velocity is not clearly reported in the source reference then the range is provided based on the best available information acquirable.

*Swimming speeds reported in the source reference as body length per second ($bl.s^{-1}$) or total length per second ($tl.s^{-1}$) were converted to $m.s^{-1}$ based on mean length of fish stated in the source reference.

Species	River System	Swimming Capability ($m.s^{-1}$, unless otherwise stated)						Notes on methodology
		Juvenile			Adult			
		Burst	Sprint	Critical	Burst	Sprint	Critical	
Ambassis agassizii (Olive perchlet)	Fitzroy, MDB	-	-	-	-	0.5-0.9[1] 0.57[2]	0.3-0.7[1] 0.53[2]	**Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[1] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Amniataba percoides (Barred grunter)	Fitzroy	-	-	-	-	-	-	Nil
Anguilla obscura (Pacific short-finned eel)	Fitzroy	-	-	-	-	-	-	Nil
Anguilla reinhardtii (Long-finned eel)	Fitzroy, MDB	0.63-0.73[3]	0.42-0.63[3]	Þ0.32-0.42[3] §0.32[3]	-	-	-	*Varied swim category definition; used group swimming; different testing methods[3] Þ Denotes prolonged speed § Denotes sustained swimming
Arius graeffei (Fork-tailed catfish0	Fitzroy	-	-	-	-	-	-	Nil
Arrhamphus sclerolepsis (Snub-nosed garfish)	Fitzroy	-	-	-	-	-	-	Nil

Bidyanus bidyanus (Silver perch)	MDB, Fitzroy	-	0.75-1[1] 0.74[2]	0.5-0.8[1] 0.68[2]	-	-	2.63[4]	**Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[1] *Velocity based on passing a fishway[4] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Craterocephalus amniculus (Darling River hardyhead)	MDB	-	-	-	-	-	-	Nil
Craterocephalus fulvus (Unspecked hardyhead)	MDB	-	-	-	-	-	0.3[5]	*Swim at 0.3 m/s for 10 min[5]
Craterocephalus marjoriae (Marjorie's hardyhead)	Fitzroy	-	-	-	-	-	-	Nil
Craterocephalus stercusmuscarum (Flyspecked hardyhead)	Fitzroy, MDB	-	0.55[2]	0.49[2]	-	-	-	**Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Gadopsis marmoratus (Northern river blackfish)	MDB	-	-	-	-	-	-	Nil
Galaxias olidus (Mountain galaxias)	MDB	-	-	-	-	-	-	Nil
Glossamia aprion (Mouth almighty)	Fitzroy	-	-	-	-	-	-	Nil
Glossogobius giurus (Flathead goby)	Fitzroy	-	-	-	-	-	-	Nil
Gobiomorphus australis (Striped gudgeon)	Fitzroy	-	-	-	-	-	-	Nil
Hephaestus fuliginosus (Sooty grunter)	Fitzroy	-	-	-	-	-	-	Nil

Hypseleotris compressa (Empire gudgeon)	Fitzroy	-	-	-	-	0.3-0.7[1] 0.51[2]	0.09-0.15[6] 0.15-0.7[1] 0.44[2]	**Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[1] **Ucrit* at 5min and 0.02 m.s^{-1} increment[6] **Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[2]
Hypseleotris galii (Firetail gudgeon)	Fitzroy, MDB	-	-	-	-	0.5[2]	0.38[2]	*0-0.4 m.s^{-1} on metalarva; *Ucrit* at 5 min and 0.048 m.s^{-1} increment (Kopf *et al* 2014) **Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[2]
Hypseleotris klunzingeri (Western carp gudgeon)	Fitzroy, MDB	-	-	-	-	-	-	*0-0.4 m.s^{-1} on metalarva; *Ucrit* at 5 min and 0.048 m.s^{-1} increment (Kopf *et al* 2014)
Kuhlia rupestris (Jungle perch)	Fitzroy	-	-	-	-	-	-	Nil
Lates calcarifer (Barramundi)	Fitzroy	-	-	0.3-0.55[7]	-	-	0.66[8]	**Ucrit* at 20min and 0.33 tl.s^{-1} increments; varied velocities for different population and temperature[7] *Velocity based on passing a fishway[8]
Leiopotherapon unicolor (Spangled perch)	Fitzroy, MDB	-	0.66[2]	0.41[2]	-	-	-	**Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[2]
Maccullochella peelii (Murray cod)	Fitzroy, MDB	-	0.8-1.2[1] 0.61[2]	0.3-0.8[1] 0.46[2]	-	-	-	**Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[1] **Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[2]
Macquaria ambigua (Golden perch)	Fitzroy, MDB	-	0.4-0.75[1] 0.47[2]	0.1-0.5[1] 0.32[2]	-	-	-	**Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[1] **Usprint* at 10s and 0.05 m.s^{-1} increment; *Ucrit* at 5min and 0.1 m.s^{-1} increment[2]

Megalops cyprinoides (Tarpon)	Fitzroy	0.19-1.38[9]	-	-	-	-	0.10-0.49[10]	*Maximum speed tested[10] *Different burst speed definition[9]
Melanotaenia duboulayi (Crimson spotted rainbowfish)	Fitzroy	-	-	-	0.7-0.82[11]	0.6-1.1[1] 0.70[2]	0.47-0.53[11] 0.55-0.75[1] 0.63[2]	**Ucrit* at 20min and 0.063 $m.s^{-1}$increments; burst speed based on max velocity[11] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[1] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Melanotaenia fluviatilis (Murray-darling rainbowfish)	MDB	-	-	-	-	0.68[2]	0.45[2]	*0-0.4 $m.s^{-1}$ on metalarva; *Ucrit* at 5 min and 0.048 $m.s^{-1}$ increment (Kopf *et al* 2014) **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Melanotaenia splendida splendida (Eastern rainbowfish)	Fitzroy	-	-	-	-	-	-	Nil
Melanotaenia taitei (Desert rainbowfish)	MDB	-	-	-	-	-	-	Nil
Mogurnda adspersa (Purple-spotted gudgeon)	Fitzroy, MDB	-	-	-	-	0.57[2]	0.21[2]	**Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Mugil cephalus (Sea mullet)	Fitzroy	0.25[12]	0.19[12]	0.15[12]	-	-	-	*Varied swim category definitions and testing methods used[12]
Nematalosa erebi (Bony bream)	Fitzroy, MDB	-	-	-	-	-	0.98-1.66[13]	*Velocity based on passing a fishway[13]
Neosilurus ater (Black catfish)	Fitzroy	-	-	-	-	-	-	Nil
Neosilurus hyrtlii (Hyrtl's catfish)	Fitzroy, MDB	-	-	-	-	-	-	Nil

Notesthes robusta (Bullrout)	Fitzroy	-	-	-	-	-	-	Nil
Oxyeleotris lineolatus (Sleepy cod)	Fitzroy	-	-	-	-	-	-	Nil
Philypnodon grandiceps (Flathead gudgeon)	Fitzroy, MDB	-	-	-	-	0.47[2]	0.23[14] 0.33[2]	**Ucrit* at 5min and 0.04 $m.s^{-1}$ increment[14] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Philypnodon macrostomus (Dwarf flathead gudgeon)	MDB	-	-	-	-	-	-	Nil
Porochilus rendahli (Rendahl's catfish)	Fitzroy, MDB	-	-	-	-	-	-	Nil
Pseudomugil signifier (Pacific blue-eyes)	Fitzroy	-	-	-	1.17[15]	0.52[2]	0.46[2]	*Different burst speed definition[15] **Usprint* at 10s and 0.05 $m.s^{-1}$increment; *Ucrit* at 5min and 0.1 m/s increment[2]
Redigobius bikolanus (Speckled goby)	Fitzroy	-	-	-	0.39[16]	0.43[2]	0.34[2]	**Usprint* at 10sec and 0.5 $tl.s^{-1}$ increment[16] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Retropinna semoni (Australian smelt)	Fitzroy, MDB	-	-	-	0.5[12] 0.46[14]	0.27[12] 0.70[2]	0.19[12] 0.66[2]	**Utest* as maximum swimming speed in[14] *Different swim speed definition and testing methods; data for *Retropinna retropinna*[12] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
Scleropages leichardti (Saratoga)	Fitzroy	-	-	-	-	-	-	Nil
Scortum hillii (Leathery grunter)	Fitzroy	-	-	-	-	-	-	Nil

Strongylura krefftii (Longtom)	Fitzroy	-	-	-	-	-	-	Nil
Synbranchidae Spp. (Swamp eel)	Fitzroy	-	-	-	-	-	-	Nil
Tandanus tandanus (Eel-tailed catfish)	Fitzroy, MDB	-	0.5-0.75[1] 0.5[2]	0.45-0.65[1] 0.41[2]	-	-	-	*Usprint at 10s and 0.05 $m.s^{-1}$ increment; Ucrit at 5min and 0.1 $m.s^{-1}$ increment[1] **Usprint* at 10s and 0.05 $m.s^{-1}$ increment; *Ucrit* at 5min and 0.1 $m.s^{-1}$ increment[2]
***Carassius auratus* (introduced)** (Goldfish)	Fitzroy, MDB	-	-	0.36[17] 0.38-0.44[18] 0.2-0.40[19]	-	0.62[20]	0.43[20]	**Ucrit* at 20min and 5 $cm.s^{-1}$ increments[17] **Ucrit* at 20min and 6 $cm.s^{-1}$ increments[18] **Ucrit* at 20min and 5 cm/s increments[19] **Usprint* at 10s and 0.05 $cm.s^{-1}$ increment; *Ucrit* at 15min increment with 30min acclimation period[20]
***Cyprinus carpio* (introduced)** (Common Carp)	MDB	1.2[21]	-	0.38-0.45[21] 0.46-0.53[18] 0.44-0.56[19]	1.6-2[21]	-	0.75-1.1[21]	**Ucrit* at 20min and 6 $cm.s^{-1}$ increments[18] **Ucrit* at 20min and 5 $cm.s^{-1}$ increments[19] **Ucrit* at 20min and 5 $cm.s^{-1}$ increments; different burst speed testing definition[21]
***Gambusia holbrooki* (introduced)** (Mosquito fish)	Fitzroy, MDB	-	-	-	0.73-1.2[22]	0.35[20]	0.24[20]	*Different burst speed testing procedure[22] **Usprint* at 10sec increments; *Ucrit* at 15min increments and 30min acclimation[20]

***Oncorhynchus mykiss* (introduced)** (Rainbow trout)	MDB	0.86[23]	-	0.61[24] 0.23-0.34[25]	-	0.92[20]	0.59-0.78[26] 0.70[20]	**Uburst* at 1min and 0.05 m.s^{-1} increments with 20min acclimation; used group swimming[23] **Ucrit* at 20min and 0.1 m.s^{-1} increments with 40min acclimation[26] **Ucrit* at 60min and 0.1 m.s^{-1} increments[25] **Ucrit* at 2min and 0.05 m.s^{-1} increments[24]
***Oreochromis mossambicus* (introduced)** (Mozambique tilapia)	Fitzroy	-	-	1.76-2.28[27]	-	-	-	**Ucrit* at 5min and 0.5 bl.s^{-1} increments[27]
***Perca fluviatilis* (introduced)** (Redfin perch)	MDB	1.6[28] 1.45[29]	-	0.8[30] 0.17[31]	0.71[20]	-	0.45[20] 1.13[30]	**Usprint* at 10sec and 0.05 m.s^{-1} increment; *Ucrit* at 15min and 0.05 m.s^{-1} increment with 30min acclimation period[20] **Ucrit* at 20 min and 0.05 m.s^{-1} increment with 120min acclimation[30]
***Poecilia reticulata* (introduced)** (Guppy)	Fitzroy	-	-	-	0.854[32]	-	0.21-0.23[33]	**Ucrit* at 3min and 0.029 m.s^{-1} increment[33]

References in order or appearance: Watson *et al.* 2019a[1], Watson *et al.* 2019b[2], Langdons and Collins 2000[3], Mallen-Cooper 1994[4], Bice and Zampatti 2005[5], Rodgers *et al.* 2014[6], Edmunds *et al.* 2010[7], Mallen-Cooper 1992[8], Tran *et al.* 2010[9], Lefevre *et al.* 2014[10], McGuigan *et al.* 2003[11], Mitchell 1989[12], Stuart *et al.* 2008[13], Kilsby 2008[14], Wilson *et al.* 2010[15], Donaldson *et al.* 2013[16], Sinha *et al.* 2012[17], Pang *et al.* 2011[18], Liew *et al.* 2012[19], Starrs *et al.* 2017[20], Tudorache *et al.* 2007[21], Wilson 2005[22], Osachoff *et al.* 2014[23], Webb 1993[24], Kieffer *et al.* 1998[25], Jain and Farrell 2003[26], Botha *et al.* 2018[27], Blaxter 1969[28], Komorov 1971[29], Tudorache *et al.* 2008[30], Davies 2000[31], Ghalambor *et al.* 2004[32], Nicoletto 1991[33].

Movement and behavioural risks

The risk of impingement or entrainment varies depending upon a species behavioural and migratory patterns. Migrating fish, particularly those that exhibit catadromous or anadromous migrations, can travel over hundreds of kilometres through the river system (Reynolds 1983), potentially exposing them to many irrigation offtakes. Of the species in the Murray-Darling Basin, the migratory pattern of golden perch is the most well studied and most extensive. Adult golden perch are known to migrate more than 1000km upstream (Llewellyn 1968; Reynolds 1983). O'Connor *et al.* (2005) further examined golden perch migratory patterns and found that some adults also travel downstream and concluded that golden perch migrate not solely upstream, but to specific spawning sites with favourable spawning conditions. Stuart and Sharpe (2019) studied the displacement of golden perch larvae downstream and reported that golden perch spawning follows flow events, enabling larvae to reach refugia and disperse over 1600km. Stuart and Sharpe (2019) also noted that golden perch's requirement of flow events and larvae long-distance dispersal are at high risk due to reducing base-flows and water abstractions. Small golden perch at 44 days old and 17.4 mm standard length had near 100% unscreened entrainment rate with impingement occurring at as low as 0.05 $m.s^{-1}$ at a 2 mm exclusion screen (Stocks *et al.* 2019). With species having large distance movement across the basin, the risk of fish becoming entrained in unscreened pumps or diversion channels increases with the number of encounters with these infrastructure types.

Other species may also be at risk of entrainment at unscreened irrigation pumps or channels and this can be exacerbated by the coincidence of peak flow pumping activity and spawning migrations. In the northern Murray-Darling Basin and the Fitzroy catchment, wet seasons with high natural flows begin in late spring and end at the start of the dry season in mid-autumn. Species such as *Hypseleotris spp*., spangled perch, olive perchlet, and Hyrtl's tandan generally move upstream on rising wet season flow events, with peak movement in spring. There is a tendency for juvenile fish to move downstream (Hutchison *et al.* 2008), and on falling flows increased numbers of adults move downstream also. With several species moving downstream, particularly juveniles and sub-adults (Hutchison *et al.* 2008), entrainment risks are highly likely. For screened pumps or diversions, impingement risks are speculated to be higher for the weaker swimming larvae and juveniles. This would be inclusive of species that don't exhibit extensive migratory movements such as *T. tandanus* (Reynolds 1983).

Regardless of reproductive strategies, upstream and downstream migrants are subjected to variable behavioural risks of being entrained or impinged. Upstream migrants can be selective of velocity gradients to minimise head currents and may prefer to swim near riverbanks or at the bottom for lower velocities (Williams *et al.* 2012). Conversely, downstream migrants may move with bulk flow (Haro *et al.* 1998; Enders *et al.* 2009; Williams *et al.* 2012), especially eels (Jansen *et al.* 2007; Økland *et al.* 2018). Fish tend to display positive rheotaxis when exposed to accelerating flows to maintain position and have more control (Danley *et al.* 2002; Enders *et al.* 2009; Vowles *et al.* 2014). However, Enders *et al.* (2009) observed that velocity gradients are higher in high flows and that juvenile salmon only exhibited avoidance behaviour at 0.2m/s. Further, Vowles *et al.* (2014) observed that more smolts travelling downstream displayed non-avoidance behaviour and went with the flow and this was exacerbated by darkness. Therefore, it is likely that for downstream movement, weaker swimming fish are more likely to be rheo-negative and have a higher probability of being entrained or impinged, particularly in turbid waters. This hypothesis has been partially investigated by Boys *et al.* (2013b) and they have concluded it to be significant.

Variations in impingement risk by species and life stage

Survival post-impingement among species can also be variable. In a field study in the US, Rose *et al.* (2008) reported survival rates of juvenile *O. mykiss* passage over an inverted-weir screen was 99% after 24 hours. Contrarily, Young *et al.* (2010) reported a higher stress response and a number of deaths of small-bodied adult delta smelt after passing a diversion screen with approach velocity of 0.1-0.15 $m.s^{-1}$. Stocks *et al.* (2019) also recorded mortality of juvenile golden perch at an approach velocity of 0.1 $m.s^{-1}$. However, the experimental design of Stocks *et al.* (2019) involved a screen perpendicular to the flow, and therefore did not include any effect of sweeping velocities, that may have reduced entrainment. These studies suggest possible species-specific impacts of impingement. Extensive impingement survival studies in the U.S. based on marine desalination intake screens have demonstrated that survival is dependent on species life stages and hardiness, as well as screen operating characteristics (Hogan 2015). The real impact is still unknown for freshwater Australian species due to a lack of data on the swimming performance of most species, particularly for early life stages.

Key knowledge gaps regarding the impacts of irrigation screening on fish irrigation screening include the survival and mortality rate of fish after impingement and the entrainment rates of larval fish. Although the natural mortality of fish larvae is generally considered to be naturally high, the additional mortality or loss to the system via irrigation pumps or diversion channels could be driving a decline in recruitment of fish into juvenile stages. Screened pumps can prevent entrainment of certain juvenile fish species, but continuing entrainment of larval life stages could substantially reduce recruitment (McMichael *et al.* 2004; Boys *et al.* 2013a). Entrainment of fish larvae would be most prevalent during the downstream drift phase of species that spawn upstream (Humphries *et al.* 1999; King and O'Connor 2007; Stocks *et al.* 2019). A study examining the impingement and entrainment of larval fish at a marine water desalination plant intake found that larval fish of less than 5mm had an entrainment rate of 100% through a 2mm mesh screen, after which the probability depended on individual species morphology and growth characteristics (Hogan 2015). However, the application of designing screens to exclude larval fish based on mesh size is not feasible. Managing screen approach velocities would seem to be the better option to reduce larval entrainment. The location of the pump inlet (*e.g.* near-bank or mid-channel) could also potentially make a difference in the number of larvae that pass close enough to be at risk of impingement or entrainment (see below). Research into inlet location could provide further options to reduce larval entrainment and impingement.

Stocks *et al.* (2019) recommended in their study that screens that limit approach velocities or impingement duration should be fitted to pump offtakes. They also suggested cycling pumps on and off at 10-minute intervals to allow for impinged fish to escape. Cycling of pumps may not be very practicable for irrigators. Based on gathered empirical evidence from the literature, the most practical application of screens at the moment would be to design screens with approach velocities appropriate to the weakest performing species and to fit them to take advantage of sweeping velocities to reduce impingement. Future studies to investigate fish-friendly screen designs should also facilitate for the ease of behavioural avoidance of fish. Screen designs should also take into consideration the local flow conditions and species spawning and migratory patterns.

The role of pump offtake positioning within a waterway, on the risk of impingement or entrainment of adult and larval fish remains unclear and requires further investigation. For species like the golden perch with long-distance movements spanning their life history, variable risk could exist for adults and juveniles such that adults may migrate upstream nearer to the river banks for a lower velocity gradient and juveniles and larvae tend to move downstream along with bulk flow (Williams *et al.* 2012). Therefore, it is likely that offtakes can entrain or impinge more larval fish if positioned in the middle of the river channel than nearer to the bank. Planned research within the CRDC is expected to provide answers to some of these issues.

Pump operators may choose to install offtakes at variable positions as well as at different depths to minimise debris intake. Therefore, future studies into screening designs should also investigate the difference in the positioning of irrigation offtakes for maximal pumping efficiency and fish exclusion. By applying optimal fish exclusion screens, we could drastically reduce the number of fish removed from river systems whilst maintaining the required water use by irrigators. It is through investigating appropriate approach velocities and optimal screen designs that we will be able to inform river managers and the irrigation industry and construct suitable guidelines for Australia. This will help irrigators, including those in the cotton industry, to reduce their environmental footprint.

Conclusion

Larval fish would be at the most risk of entrainment as they are the weakest swimming stage. Risk of entrainment would be increased in catchments where there are greater numbers of offtakes, as the probability of a larva encountering an offtake as it drifted downstream would be increased. Similarly, fish species that migrate long distances would be at greater risk of encountering offtakes as the probability of passing near an offtake would be greater than for a more sedentary species.

Currently, there is a lack of empirical data on the swimming performances of many Australian native fish species. When concerning the swimming speeds of different life stages of native fish, there is very little published research at all. Based on the available data, an approach velocity near the screen of 0.25 $m.s^{-1}$ would suffice for adults and juveniles of most species for which there is data to prevent impingement on the screen. However, this would not be sufficient for fish in any earlier life stages. Data is still lacking for many small species and juveniles of larger species, so a conservative approach is warranted. Similarly, as recommended by Boys *et al.* (2013a), an approach velocity of less than 0.4 $m.s^{-1}$ would significantly reduce entrainment, however, Boys *et al.* (2013b) stated that the true effects of impingement are still unknown, and suggested that the velocity be kept under 0.1 $m.s^{-1}$. This recommendation is relevant to the scarcity of knowledge and data on the effects of impingement for the majority of the native species in our river systems. Likewise, similar velocity restrictions of 0.1 to 0.15 $m.s^{-1}$ have been applied to the through-slot velocities on intake screens for marine desalination plants in Australia (Craig 2015). This would be adequate for the burst or sprint speeds of many species, including the burst and sprint speeds of some larval stages (see Tables 1.1 and 1.2) for which there is existing data.

Chapter 2: A review of fish entrainment mitigation options and research.

Michael Hutchison, Andrew Norris, Jenny Shiau and David Nixon

Introduction

Research into entrainment of fish through irrigation systems is still in its infancy in Australia. Existing studies in Australia have shown quite a wide variation in the numbers of fish entrained and the mortality rates of entrained fish. For example in a study of entrainment through a 36 ML per day and a 150 ML per day pump on the Namoi River NSW, Baumgartner *et al.* (2009) recorded a maximum entrainment rate of 232 fish in a day, with an overall mortality and injury rate of 7.5%. In contrast Norris (2015) monitored a 29 ML per day pump and a 36 ML per day pump in Oakey Creek near Dalby, and found a mean of 1130 fish per ML were entrained, with mortality rates ranging from 70% to 80%. Norris (2015) also found considerable variation in the numbers of fish entrained between the two pumps, with more fish entrained through the smaller pump. Localised factors such as pump inlet locations, the abundance of fish in the pumped reach and the type of flow event pumped could all have influenced the results of these two studies. Variation in mortality and injury rates is probably a moot point, because once entrained into an irrigation system, fish are essentially lost to the river population (Baumgartner *et al.* 2009). Despite the variation between these studies, it is still clear that entrainment of fish through irrigation systems is an issue in Australia and measures to mitigate entrainment will be beneficial to fish populations.

Technologies to prevent entrainment of fish through hydro-electric turbines and irrigation diversions have been in place in the USA, Britain, New Zealand and parts of Europe long before such technologies were first considered in Australia. Irrigation diversions and offtakes began to be screened for fish in New Zealand in the 1980s and by 2005 regulatory authorities required fish screening on irrigation and stock-water offtakes in New Zealand rivers (Jamieson *et al.* 2007). Guidelines for New Zealand irrigators were produced in 2007 (Jamieson *et al.* 2007). In Britain, abstraction of water for hydroelectric generation and certain other purposes is subject to regulations requiring screens or other effective barriers to be put in place to prevent entrainment of fish. A screen may be interpreted as either a physical mesh or a behavioural screen with a deterrent stimulus (*e.g*. electrical, acoustic, light). Hydropower screening normally diverts fish to a bypass channel (Turnpenny *et al.* 1998).

The impact of diversions on economically important migrating salmonids (salmon and trout) was a driver for screening of irrigation offtakes in the USA. Perhaps the earliest attempt to prevent loss of fish through irrigation diversions was in New York in 1865 (Leitritz 1952). Further recognition of impacts of irrigation diversions on fish and early attempts at screening were made in Montana in the 1890s (Clothier 1953a, 1953b, Spindler 1955). Since that time many different types of physical screens have been tried and tested and backed by an evidence-based approach to the development of physical design criteria (reviewed in Boys *et al.* 2012), with several highly successful designs currently in use. This provides Australia with a great opportunity to implement mitigation strategies that work, without repeating the mistakes that have been made elsewhere. Australia has its own unique suite of fish species and some refinements may be required to adjust for local conditions. However, the general principals learned from the international studies can be applied to Australia. We can also learn from some US designed screening technologies that have recently been tested at several Australian locations.

As well as physical screens, there have also been various behavioural screens developed and tested in Europe and North America. In this chapter we review a range of physical and behavioural screens in terms of their effectiveness, relative cost and potential for application in Australia.

General considerations

Unfortunately, there is no universal solution for fish entrainment at all sites. To select a suitable screen for a site it is necessary to consider the site characteristics. Each site needs to be evaluated based on several localised factors. Some of the key factors to consider when selecting screens for a site, as described by Mefford (2013) include:

1. Is there power at the site?
2. What is the allowable head loss through the screen?
3. What is the typical depth of flow at the site?
4. What are typical flow velocities at the site? (Sites with higher velocities allow for wider use of passively cleaned screens that are cleaned by sweeping flows that exceed approach velocities by at least 10 times)
5. Do site constraints strongly limit the allowable footprint for screen installation?
6. What are typical debris loads at the site?
7. What is the average ratio of screened flow to channelled (river) flow? (This mainly applies to screened gravity fed diversion channels and not to pumped irrigation sites).

Broadly, screens can be described as either physical or behavioural barriers which are used to prevent or limit entrainment of fish. Behavioural screens do not totally exclude all fish from entrainment but may reduce the numbers of fish entrained. However, well designed and operated physical screens can eliminate entrainment of non-larval stages and greatly reduce larval entrainment. In the following sections we consider the various types of physical and behavioural screens, evidence for their efficacy and where they may be most appropriately applied. Where available we draw on some Australian studies and experiences, but most of the information is from locations other than Australia.

Physical screens

Physical screens can be manufactured from a range of fabric types. According to Mefford (2013) common fabrics include woven wire mesh, perforated metal plate and wedge wire screens. Woven wire is generally a low-cost fabric, mounted on closely spaced structural supports to prevent tearing under load. Woven wire fabric is more difficult to clean and is suitable only for small screens where the debris loads are light to medium. Norris *et al.* (2020) noted difficulties with cleaning algae off woven wire fabric on a cylindrical screen (Figure 2.1) and this led to a loss in pumping efficiency. Perforated plate is a widely used screen fabric of relatively low cost. Most commonly the plates are stainless steel. They provide a smooth surface with round holes that are less likely to trap debris. Perforated plates require spaced frame supports to prevent bending under load. Wedge wire is another commonly used screening material. Wedge wires are run parallel to each other leaving narrow slit openings between the wires. The wedge wires are supported by cross wires running at regular intervals. Wedge wire is ideal for screening due to its smooth upstream face. The wedge shape leads to the smallest opening being at the upstream face, which reduces debris wedging below the screen surface. Wedge wire is more expensive than other screen fabrics, but is more durable than the other fabrics, requiring less support. It is therefore ideal for situations with higher debris loads. The parallel wedge wires make them ideal for efficient automated cleaning by brushes. There are also some screens available in synthetic moulded materials, but these need to be UV protected and must have low expansion and contraction ratios from changing temperatures.

Figure 2.1: Algal accumulation on a woven wire mesh screen

Although screens may be self-cleaning of debris under flow conditions, most screen types will eventually need some form of mechanical cleaning or brushing as they can become bio-fouled by growths of algae, and in some regions, aggregations of molluscs (Hanna, 2010; Mefford, 2013).

If physical screens are used for screening diversion channels several mechanisms that cause injury, migrational delay or mortality must be considered when designing a physical barrier screen. According to Nordlund (2008) these include the following:

1. physical contact with the screen.
2. impingement onto the screen.
3. entrainment through the screen mesh.
4. predation in the screen forebay.
5. predation at the bypass return pipe and at the outfall in the river.
6. water quality in the ditch.
7. water quantity in the ditch, bypass return pipe and river.
8. debris accumulations in bypass pipes, head gates or trash-racks.
9. excessive delay of fish due to poor hydraulic guidance conditions

According to Nordlund (2008) approach velocity must be less than the sustained swimming speed of the fish, although based on the data in Chapter 1, less than the sprint or burst speed of the weakest swimming fishes should be sufficient. Sweeping velocity should be at least double the approach velocity and preferably more. Issues relating to the need for bypass channels or bypass pipes that return fish to the river can be avoided if the screen is positioned at the entrance to the diversion channel, rather than in the irrigation diversion channel. Screens for pumped diversions will almost always be in the river channel.

Types of physical screens include rotary drums, Coanda screens, horizontal fixed plate screens, cylindrical screens, cone screens, travelling screens and vertical fixed plate screens.

Rotary drum screens

According to Nordlund (2008) in a report focusing on screened channel diversions, over 200 physical barrier screens had been installed in Oregon, Idaho and Washington state since 1992. Most of these screens were rotary drum screens (Figure 2.2). Rotary drum screens are best suited to gravity fed diversion channels or ditches. Baumgartner and Boys (2012) suggest they are suitable for small (<1,000 ML per day) and large (1,000-10,000 ML per day) gradient fed canals. Most rotary drum screens need access to electrical power to drive the rotating mechanism, but there are some versions that can be connected by a geared mechanism to a paddle wheel that sits in the canal and is driven by the flowing water. Paddle wheel drive versions are usually restricted to flow rates less than 15 cubic feet per second (cf.s^{-1}) which equates to 425 litres per second (L.s^{-1}), whereas electrically powered rotating drum screens can cope with greater flows (Mefford 2013). Solar power is usually sufficient to drive smaller drum screen units (Mefford 2013)

Drum filters rotate to flush debris off on the downstream side of the screen. The water level must sit between 0.65 and 0.85 the diameter of the screen for effective cleaning. Therefore, they operate best at sites where the water level is relatively stable. High silt loads can result in deposits either side of the screen (Mefford 2013; U.S. Department of Interior 2006.).

The advantages of drum screens are they are self-cleaning and have excellent debris handling qualities. The disadvantages are that they are more costly than passive flat plate screens and that they are suited only to sites with well-regulated stable water levels such as in canals. Seals at the bottom and sides of the drum require maintenance. Drum screens have moving parts, so bearings and drive chains also require maintenance.

Neitzel *et al.* (1996) compared a drum screen set perpendicular to an approach channel with one set at 45 degrees to the approach channel. The screens were tested using chinook salmon (*Oncorhynchus tshawytscha*) fry 47-68 mm in length. They found no significant difference between the two screen orientations. No fish passed over, through or around the drum screen. Approach velocities at the face of the drum screen did not exceed 0.12 m.s^{-1}. in both cases. Boys *et al.* 2012 and Boys *et al.* 2013b recommend that in Australia approach velocities should not exceed 0.1 m.s^{-1}.

The importance of maintaining the correct approach velocities to a drum screen and installing the screen correctly is demonstrated by the following example. A poorly installed drum screen in an irrigation channel in New Zealand had approach velocities of 0.4 m.s^{-1}, more than three times the recommended 0.12 m.s^{-1} for New Zealand species. This screen was field evaluated with 500 juvenile rainbow trout *O. mykiss* 25-35 mm in length and 500 juvenile Chinook salmon 60-80 mm in length. Fish were not effectively diverted to a bypass channel, as there was very little sweep across the screen and the bypass channel was poorly positioned. Fish were either impinged on the screen or passed over or through the screen. The screen meshes were 5 mm wide, whereas the recommended drum screen mesh size in New Zealand is 3 mm. This permitted some fish to pass through the screen. The seals on the sides and bottom of the drum screen were also poorly maintained and not operative, enabling some fish to bypass the screen into the irrigation canal downstream (Bonnett *et al.* 2014).

Figure 2.2: A rotary drum screen. Image reproduced with permission from Hydroscreen Co.LLC http://www.hydroscreen.com/products/rotary_fish_barriers/index.html#sthash.cUCqm8ua.dpbs

Coanda screens

Coanda screens are screens made of tilted wedge wire to increase flow through efficiency. Coanda screens have no moving parts, and can be flat, but frequently have a slight curvature to form a concave arc (U.S. Department of Interior 2006, Nordlund 2008, Mefford 2013). These screens form an inclined ramp. Water passes over the screen face, with most water passing through the screen to an irrigation diversion (Figure 2.3). The sweep velocity carries fish and debris over the screen face to a bypass channel (Mefford 2013). Coanda screens have a high flow capacity and are typically set on the downstream face of an overflow weir and are usually mounted in a channel (U.S. Department of Interior 2006). Flow passes over the weir crest, down an acceleration plate and across the screen panel (see Figure 2.3). These screens normally require at least 90 cm of drop (U.S. Department of Interior 2006). Coanda screens only work well as fish bypasses if sufficient flow depth exists at the downstream end of the screen. This allows debris to be removed, where it won't pose a hazard to fish passing over the screen. Without sufficient flow, Coanda screens can become dewatered at the toe of the screen (Mefford 2013). There is potential for injury of fish (including descaling) when flow conditions aren't sufficient for safe bypass of fish. Coanda screens are not widely used in the Pacific Northwest of the U.S.A. due to a lack of sites with suitable conditions (Nordlund 2008). These screens are relatively compact and are self-cleaning of debris and can help remove sediment from diversions but may require occasional brushing to remove debris jammed between the wires or to remove algae growth (Wahl 2003).

One of the more frequent uses of Coanda screens is to exclude invasive species of fish (Wahl 2003). For example, a 0.5 mm mesh Coanda style screen has been used in the Mareeba-Dimbulah water distribution network in north Queensland to stop inter-catchment transfer of the pest fish Tilapia (Pest Smart 2014), including eggs and larvae. Coanda screens have also been applied in situations where fish survival is the objective (Wahl 2003). One example is the passage of salmonids past an irrigation diversion in the East Fork Irrigation district in Oregon U.S.A. at a sand-trap and fish screen facility (Buell 2000). Tests involving chinook salmon and steelhead fry (30-50 mm FL) and steelhead smolts (130-260 mm FL) at this facility indicated that no injuries, behavioural anomalies or latent mortalities resulted from passage over the Coanda screen for any of the three species or life stages of fish tested (Buell 2000).

However, not all Coanda screens function well for fish survival. Bestgen *et al.* (2001) examined survival and entrainment of early life stages of fathead minnows *Pimephales promelas* released over inclined wedge wire-screens (Coanda screens). They looked at the effect of screen angle, screen slot width and overflow rate on entrainment rates. Five size groups of minnows were tested consisting of nominal sizes of 5.0, 7.5, 12.5, 22.5 and 45 mm in total length.

Exclusion of 45 mm and 22.5 mm minnows was 100 per cent on a 45-degree screen with a 1 mm slot width and low (10%) overflow rate. Mortalities ranged from 0% to 12 %. High overflow rates (25%) were found to lead to high impingement and mortality rates. Exclusion rates of fish declined with declining size. Larger fish survived better than smaller fish regardless of screen configuration. From 96% to 100 % of 12.5 mm fish were excluded from entrainment but survival ranged from 62% to 86%.

A 0.5 mm slot width led to only 15% survival of 12.5 mm length fish. This was possibly due to more frequent contact with closely spaced screen wedge wires Changes of screen angle did not affect survival rates of fish. Almost all 5 mm fish died in this study and survival of 12.5 mm fish was worse in low flow treatments than in high flow treatments. Overall, the Coanda screens trialled in this experiment do not seem very conducive to fish survival. We would not recommend use of Coanda screens where the objective is to achieve high levels of fish survival when screening fish from irrigation intakes.

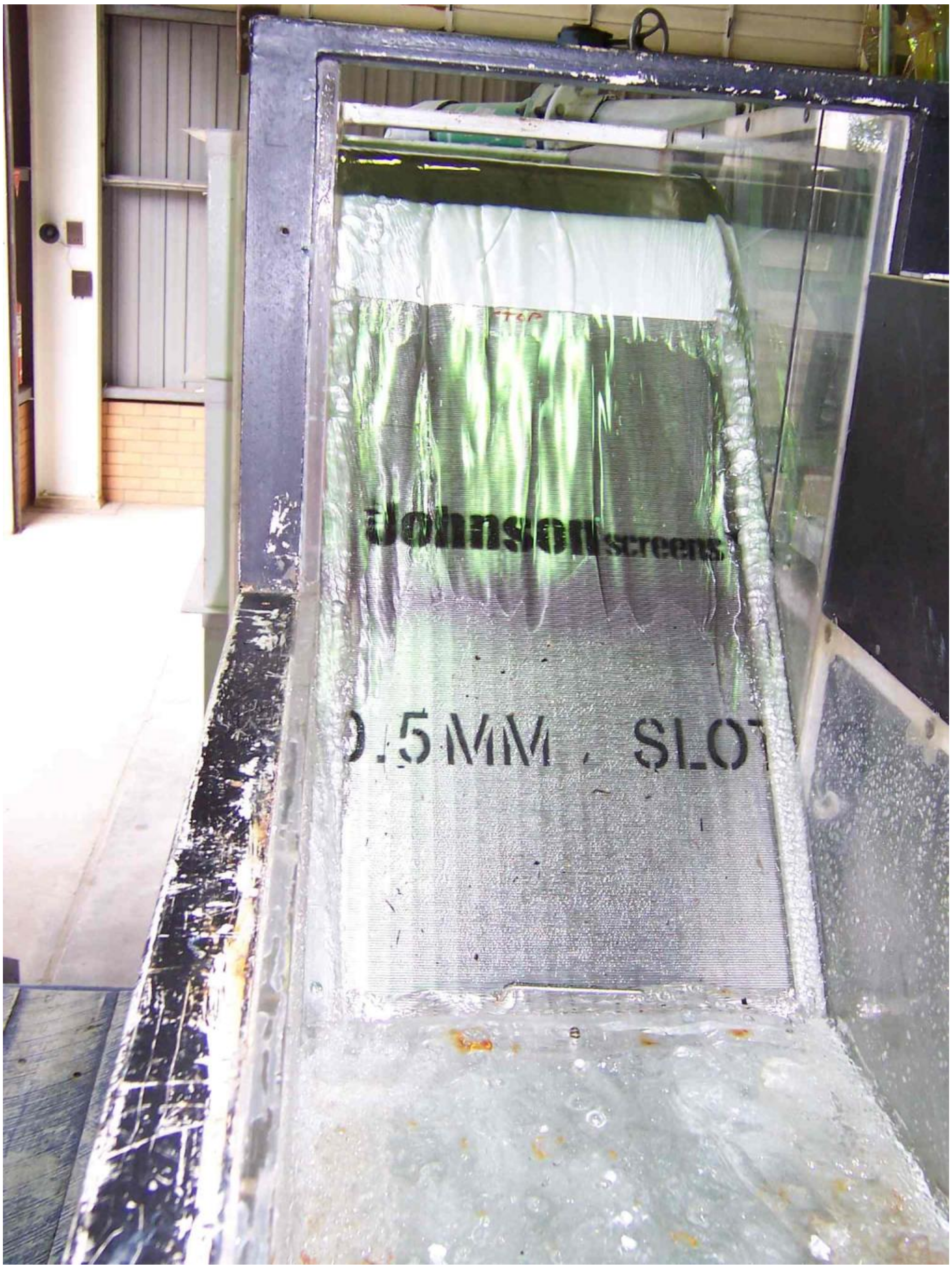

Figure 2.3: A Coanda screen being tested in a flume tank. Water that passes through the screen in a field situation would pass into an irrigation diversion. These screens are very efficient at passing water, but if flow over the screen is inadequate fish can become stranded or injured.

Horizontal fixed plate screens (*e.g.* Farmer's Screens)

Horizontal fixed plate screens should only be used in streams and canals where flow fluctuations are small (Nordlund 2008). These screens are not suitable at weir sites with highly fluctuating water levels (farmerscreen.org). There can be major issues if the cleaning mechanism fails as the weight of the water can overcome the structural ability of the frame (Nordlund 2008). These screens are designed for within diversion canal use. Horizontal plate screens may be made of perforated plate or wedge-wire and are run submerged and horizontal above the channel bottom. Water flows above the screen and is passed through the screen into an irrigation diversion channel. A baffle (weir) maintains water depth above the plate to prevent dewatering of the screen, and a side wall keeps fish within a channelised section above the plate. Sweeping flows help prevent fish from becoming impinged on the plate. The wall wedges fish towards a bypass channel (see Figure 2.4). These screens are used to pass water into the irrigation scheme whilst diverting fish to a bypass channel or pipe back to the river or reservoir from where they came.

The Farmer's Screen is one example of a horizontal fixed plate screen (Figure 2.4). This screen design has been quite successful at diverting salmonids to bypass channels with no entrainment through the screen and with little or no injury to the fish. These screens can provide safe efficient downstream passage of salmonids when operated within their design criteria. (Mesa *et al.* 2012). The Farmers screens are made from stainless steel and come in various sizes and can accommodate flows ranging from 0.01 to 4.3 m^3s^{-1}. The Farmers Screens tested by Mesa *et al.* (2012) had approach velocities ranging from 0 to 5 cms^{-1}. Sweeping velocities ranged from 36 to 178 cms^{-1}.

Rose *et al.* (2008) tested two different type of horizontal screens. Approach velocities ranged from 3 to 8 $cm.s^{-1}$ and sweeping velocities from 69 to 143 $cm.s^{-1}$. Survival rates of fish held for 24 hours after passage over these screens exceeded 98%.

Farmers Screens require sufficient flow to ensure 5-10% of the flow is diverted to a fish bypass channel. Two square metres of screen area is required for every 100 $L.s^{-1}$ of flow (farmerscreen.org). The screen should be in the irrigation channel, off-river, with a functioning head gate upstream to control flow. However, the screen should be located close enough to the river to enable the bypass flow to be easily diverted back into the river (farmerscreen.org). Fish can become stranded on horizontal flat plate screens if inflow rates are not sufficient (Rose *et al.* 2008; Mesa *et al.* 2012). Head requirement for horizontal plate screens is 3-9 cm (Mefford 2013). Horizontal plate screens have no moving parts and are generally cost-effective (U.S. Department of Interior 2006). Debris and sediment can sometimes be a problem. Diversion flow rates will vary as a function of surface water elevation and fouling. The biggest issue with fouling has generally been algae growth but this can be cleaned. Overall, tests suggest that Farmers Screens provide safe downstream passage for fish at irrigation diversions (Salalila *et. al*. 2019), and Baumgartner and Boys (2012) consider them suitable for small gradient fed canals in Australia with flows of less than 1000 ML per day.

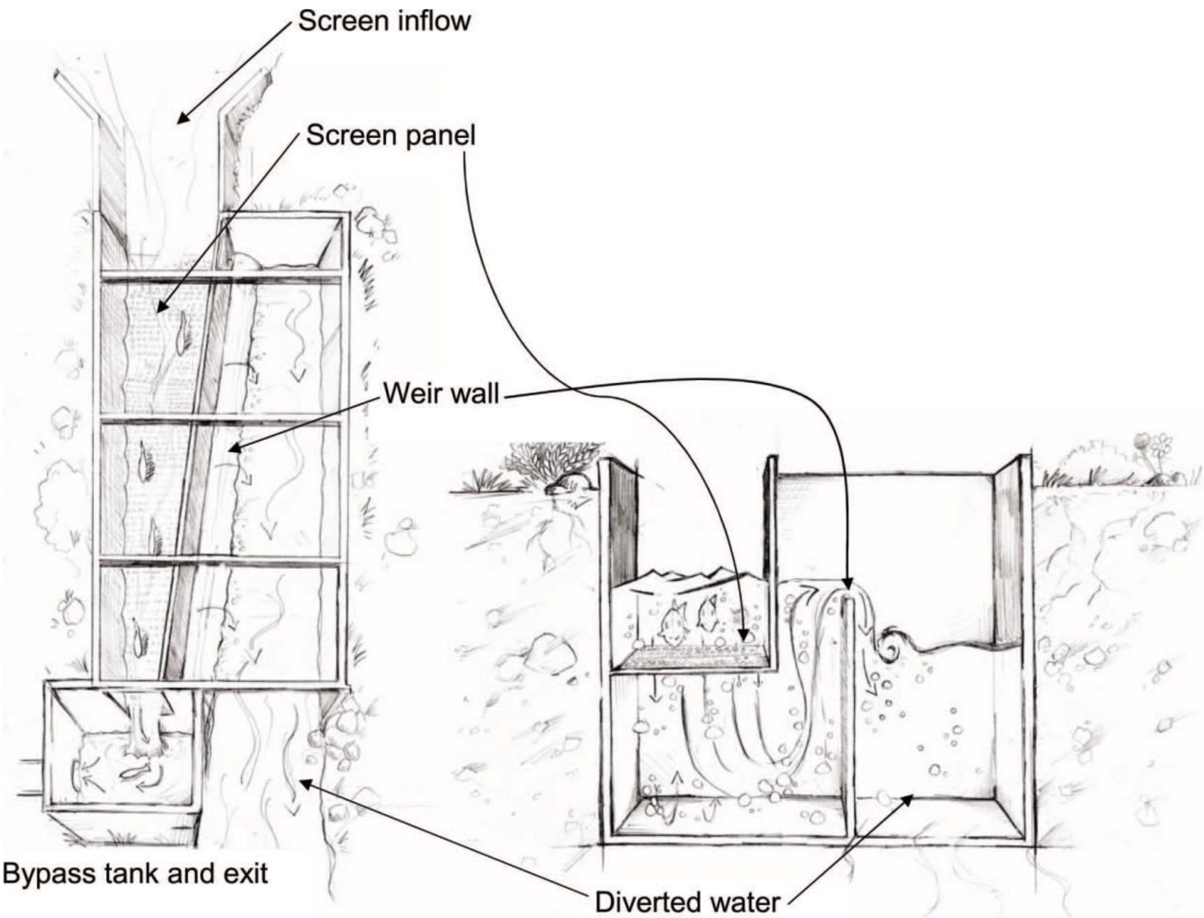

Figure 2.4: A conceptual diagram of a Farmer's screen (Horizontal Flat-Plate Screen). Reproduced with permission from the American Fisheries Society. Source: Mesa *et. al*. 2012 (North American Journal of Fisheries Management).

Pump Intake screens 1: Cylindrical screens

Cylindrical screens are operated fully submerged and are typically used on pumped diversions within the river or weir pool where the pump inlet is located. These screens can also be used on gravity fed conduits (U.S. Department of Interior 2006). Water should submerge the screen by a minimum of half the screen diameter for effective operation (Mefford 2013). As the name implies these screens are cylindrical in shape (Figure 2.5), and this shape provides a large surface area per unit length (Mefford 2013), which is important for helping to reduce approach flow velocities. Cylindrical screens should be placed parallel to the flow to achieve best through screen velocity uniformity and to provide a sweeping flow from the river current. Cylindrical screens can be mounted as a single unit or end to end as a T shaped unit with an exit pipe in between (Mefford 2013).

Baumgartner and Boys (2012) recommended rotating cylinder screens for pump intakes. They stated water jet cleaned mesh fabric screens may be suitable for pumps with capacities less than 30ML per day, but recommended brush cleaned, wedge wire cylindrical screens for larger pumps. The brushed screens appear to be very efficient at keeping free of blockages by debris.

The prices of these screens vary considerably. Good quality smaller wedge-wire screens (e.g. 12 $ML.D^{-1}$) start from around $20,000 AU. Woven mesh screens may be cheaper. A wedge-wire screen for a 30 $ML.D^{-1}$ system costs around $25,000. Costs for screens for larger systems will depend on factors such as site access and retrieval systems. Installation costs will vary according to site characteristics and existing infrastructure. Costs per unit of flow can vary quite widely. Typically, costs per unit volume go down for larger structures (U.S. Department of Interior 2006).

Rotating cylinder screens generally require access to power, but some smaller models use water current driven propellor systems to rotate the screen (AWMA undated (a)). Some cylindrical screens are static and use sweeping flows of the river current to clean them (U.S. Department of Interior 2006). However, static cylindrical screens will eventually need manual cleaning and are not suited to areas with high debris loads such as backwater areas where debris tends to accumulate.

Cylindrical screens have the option of being fitted to retrieval systems that can raise screens between use for maintenance (Figures 2.6 and 2.7) and lower screens back in place when required (U.S. Department of Interior 2006). Having screens raised between usages should reduce the risk of biofouling and reduce maintenance costs. Screens can also be raised to avoid flood debris, then lowered for pumping when risk of heavy debris is reduced.

Many cylindrical screens have internal baffling to generate uniform through screen flow velocities, but this adds to head loss. Head-loss through a medium sized electrically powered rotating AWMA cylinder screen is around 110 mm (4.4 inches) at 20 $ML.D^{-1}$ and 250 mm (10 inches) at 30$ML.D^{-1}$. The water powered propeller driven version of the screen has slightly higher head-loss.

Norris *et al.* (2020) demonstrated that a woven wire mesh cylinder screen with water jet cleaning, fitted to a 36 $ML.D^{-1}$ pump eliminated entrainment of fish at a pumped irrigation site near Dalby in Queensland. An unscreened pump in the same reach entrained small native fish, including carp gudgeon and juvenile eel-tailed catfish. However, the screen had some problems with algae biofouling parts of the screen, which the water jets did not remove effectively. This may have been avoided if the screen was left out of the water prior to pumping. A brushed, wedge-wire cylinder screen is likely to have been better for pumping efficiency and cleaning properties on this sized pump.

Passive wedge wire screens fitted to a water offtake in the Thames River UK, accumulated algae growth on the wedge wire, but the openings between the wires were largely kept clear by compressed air backwash. No noticeable drops in summer abstraction rates were observed (Bromley *et al.* 2014). The weakest swimming fish are fish larvae. They are also small enough to pass through most mesh sizes. Entrainment of fish larvae through the same cylindrical wedge-wire screens set in the Thames River was significantly reduced compared to unscreened controls. Interestingly 3 mm and 2 mm slot widths performed better at excluding larvae than 1 mm slot widths, and it is thought biofouling may

Figure 2.5: An example of a rotating cylindrical screen unit just prior to installation. This one has a water jet cleaning system and woven wire mesh fabric. Other models available have brush cleaning systems and a wedge wire construction and may be more suitable for Australian conditions. Wedge wire is recommended for pumps larger than 30 ML per day. Note the elbow to ensure the screen is set parallel to the river flow.

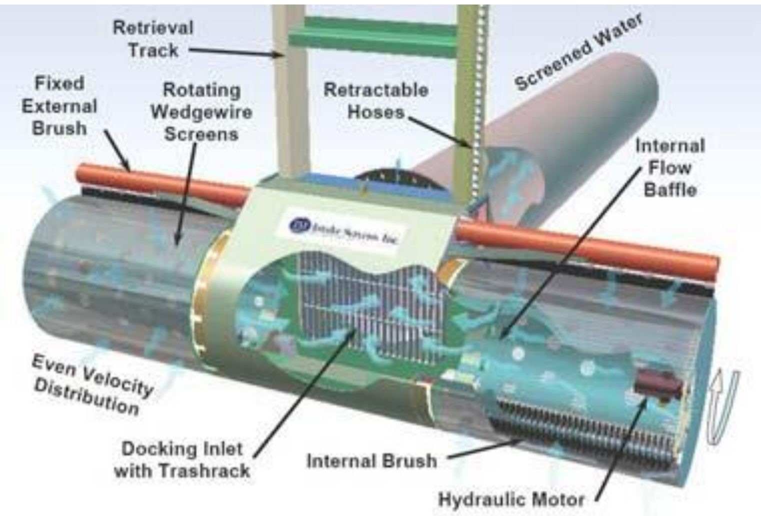

Figure 2.6: A diagram of a brush cleaned T style wedge-wire cylinder screen that can be raised up a retrieval track for maintenance. Internal workings are shown. Image reproduced with permission from AWMA. https://www.awmawatercontrol.com.au/

have increased slot flow velocities through the smaller slot width configuration. Unscreened control pumps removed a mean of 205 fry or larvae per ML. Cylindrical screens with 3 mm, 2 mm and 1 mm slot widths removed a mean of 69, 61 and 161 fry or larvae per ML, respectively. The majority of entrained fry were <6 mm TL (Bromley *et al.* 2014). A wedge-wire cylinder screen in the Delaware River in New Jersey was evaluated and found to reduce entrainment of stiped bass larvae to one tenth of non-screened conditions (Ehrler and Raifsnider 2000). Wedgewire cylindrical screens with 0.5 mm slot widths significantly reduced entrainment of the larvae of several estuarine species in Narragansett Bay Rhode Island. Use of 1 mm slot widths reduced entrainment of some larval species, but not all. In freshwater (Portage River Mouth, Lake Erie), entrainment of shad larvae was reduced by 50% with a 0.5 mm slot width, but not with a 1 mm slot width (EPRI 2005).

Figure 2.7: A T style retrievable wedge-wire self-cleaning cylinder screen in the raised position. Image reproduced with permission from AWMA https://www.awmawatercontrol.com.au/products/cylinder-screens-powered/

Experiments with a perforated cylinder screen (with 5 cm holes) in a flume tank showed the screen effectively reduced entrainment of juvenile sturgeon to 2% compared to 40% for unscreened controls (Poletto *et al.* 2015). The same screen reduced entrainment of juvenile Chinook salmon by 93%. The device also worked at night and in turbid conditions, but fish were at greater risk of being entrained at night (Mussen *et al.* 2015). Given the large hole sizes in this cylindrical screen, it is reasonable to assume that a different configuration with smaller gaps such as a wedge-wire screen may have produced an even more impressive result, provided approach velocities were kept low.

Pump intake screens 2: Cone screens

Cone screens (Figure 2.8 and 2.9), as their name suggests are cone shaped and come in wedge wire, perforated plate and woven wire versions (Mefford 2013). The standard application for these screens is fitted to a pump intake or gravity flow through a head wall. The cone shape offers a large surface area for a small stream depth and a small footprint (Mefford 2013, AWMA undated (b)). These screens can operate fully or partially submerged. Most cone screens come with cleaning brushes. These screens generally need power to operate the cleaning system, but there are some versions available with propeller drives located in the discharge pipe to operate the brush cleaning system. The head requirement for operation is low, ranging from 3-9cm. These screens are best suited to low flow (<0.15 m.s^{-1}) velocity areas such as backwaters and impoundments (Mefford 2013). If exposed to currents these screens can have approach velocity hotspots (Gard *et al.* 2010). A combination of internal and external baffles can be used to address this to some extent (Hanna 2013). We believe cone screens may be suited for installation in perpendicular (to the river channel) side channels constructed for pump intakes, where they will be protected from the current, or for fitting to pumps in weir pools and large impoundments where the current is low. These screens have also been used to screen water for diversion channels *e.g*. at Cohuna in Australia, where brush cleaned wedge wire constructed cone screens have been used (AWMA undated (b)). Evaluation of the Cohuna screens was difficult because the irrigation channel had some resident small fish present that had entered pre-screening. However, the screens appear to have been effective at preventing entrainment of silver perch and Murray cod larvae (North Central Catchment Management Authority 2020). From an irrigation perspective, there have been no issues with water delivery or changes to head loss after three full irrigation seasons (Peter Rose, North Central Catchment Management Authority, pers. comm.) No fish bypass is required for these types of screens (U.S. Department of Interior 2006).

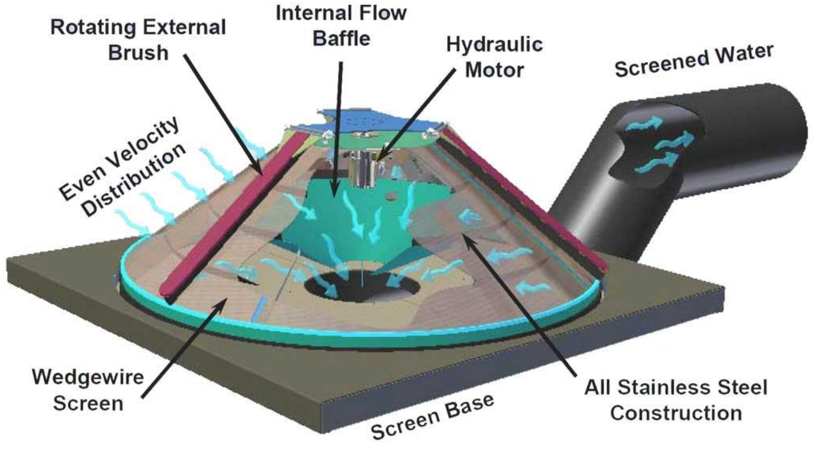

Figure 2.8: A diagram of a cone screen showing internal workings. Image reproduced with permission from AWMA. https://www.awmawatercontrol.com.au/products/cone-screens/

Figure 2.9: Installation of cone screens. Reproduced with permission from AWMA.
https://www.awmawatercontrol.com.au/project/trangie-nevertire-irrigation-scheme-fish-screens/

Flow capacity of these screens is generally larger than 142 Ls^{-1} (Mefford 2013). Cone Wedge wire screens can have a capacity of up to 150 ML/D per screen. Multiple screens in tandem can be used to pass larger volumes. Approach velocities are designed to be 0.1 $m.s^{-1}$ (AWMA undated (c)) There is little in the peer reviewed literature on fish entrainment or impingement on these screens, but it can be assumed to be low when approach velocities are kept to 0.1 m.s-^{1}, and this appears achievable in low flow areas or with appropriate external baffling in riverine areas.

Travelling (vertical or inclined) screens

Travelling screens do not require a controlled water depth and have been widely applied at water diversions in the USA for many years. They are mostly used at small diversions or at secondary dewatering structures in fish bypasses. These screens require a powered site to operate. Travelling screens have seals around the margins and moving parts including a spray pump (or other cleaning mechanism such as a brush or backwash) and a conveyor which require maintenance. Occasional adjustment of belts and drives is required (U.S. Department of Interior 2006; Mefford 2013). Travelling screens are generally not economically viable at large diversions (U.S. Department of Interior 2006). Baumgartner and Boys (2012) considered these screens to have limited application in the Murray-Darling Basin. Potentially they could be used at large gravity fed canals or in large dams in conjunction with fish bypasses. Baumgartner and Boys (2012) suggested these screens may be more expensive than other types. Figure 2.10 shows an example of a travelling screen.

These screens are most useful at sites needing a screen that can operate under a wide range of flow (depth) conditions. Typically, the screens are made of wire fabric or articulated slotted panels. These screens can be set in a river at a diversion channel entrance or can be set in a diversion channel with a bypass (Mefford 2013). The head requirement for these screens is 6-18cm, depending on screen porosity and internal bracing. These screens may be set vertically or inclined at up to 30 degrees (Mefford 2013). For the screens to be effective they need low approach velocities at the screen to prevent impingement (Baumgartner and Boys 2012). In some cases, they are set parallel to the channel flow or skewed to the flow, to ensure adequate sweeping velocity. The downstream end of the channel generally leads to a fish bypass and the screened water flows to the irrigation diversion. Some travelling screens are set as an end screen across a channel. If the channel velocity is low and the screen approach velocity is low, then these can function effectively as a fish barrier (U.S.

Department of Interior 2006). If these screens are positioned facing a strong current, they are likely to impinge fish.

Some travelling screens are modified by fitting with bucket (trough) like structures across the screen to aid survival of impinged fish. These are also known as Ristroph screens and they improved survival of impinged marine life from 15% on an unmodified screen to 90% with the Ristroph modification (Pankratz 2015). Impinged fish are washed into the buckets by a water spray, then carried up with the mobile screen and tipped into a fish bypass (Black and Perry 2014). Survival rates of ten species of impinged freshwater fish were tested on this this type of travelling screen by Black and Perry (2014). Fish were impinged at approach velocities of 0.3, 0.6 and 0.9 m.s-1. There was a tendency for higher mortalities with increasing velocities, but this was only significant for bluegills *Lepomis macrochirus*. Overall, there was a tendency for decreasing mortality, injury, and scale loss with increasing size of fish, with fish length being a significant factor. Mortality rates did not exceed 5% for all species and velocities tested. It was concluded the trough/bucket modification helped reduce impingement losses.

Figure 2.10: Travelling (inclined) polymer screen being installed in New Zealand. Reproduced with permission from AWMA. https://www.awmawatercontrol.com.au/products/travelling-polymer-screens/

Vertical panel or fixed plate screens

Vertical panel or fixed plate screens are generally constructed from wedge-wire or perforated plate. As the name implies, these screens are installed in a vertical or near vertical position. These screens generally do not need to be powered, relying on sweeping flows to passively clean them, however some screens have the option of installing electric or paddlewheel driven brush cleaning systems. Passive cleaning works best if the sweeping velocity is at least 15 times the approach velocity (Mefford 2013). Mechanical cleaning is recommended if the diversion is greater than half the upstream channel flow (Mefford 2013). Vertical panel screens can be either set as bank or wall side screens, parallel to the stream flow for gravity diversions (with no bypass channel) or set at the entry to pump sumps. Alternatively, these screens can be set at an angle (15-45 degrees) within a diversion channel to divert fish to a bypass channel (Mefford 2013). In longer screens, baffling can be used to help create uniformity of approach flow. To cope with large flows in channels, fixed plate screens can be set in a V formation to increase surface area. Fish are diverted to a centre bypass channel through the V (Mefford 2013).

Boys and Baumgartner (2012) stated these screens can be used for pumps, but they did not recommend them as a first choice. However, they suggest these screens are suitable for both small and large diversion canals. The screens are usually effective barriers to fish entrainment, and they do not require a controlled operating depth (U.S. Department of Interior 2006). Vertical panel screens normally have a head requirement of 6 cm-18 cm, depending on baffling and channel velocity (Mefford 2013). Facilities designed correctly, have resulted in guidance rates of juvenile salmonids to bypass channels at greater than 98% (NMFS 2008). Because of their excellent fish protection performance and generally low operating cost, flat plate screens are widely applied at small to large irrigation diversions in Washington, Oregon, and California where total fish exclusion is required (U.S. Department of Interior 2006).

Wedge wire in fixed plate screens can be oriented either horizontally or vertically. de Bie *et al.* (2018) compared the efficacy of horizontal and vertically aligned wedge-wire to guide juvenile chub *Squalius cephalus* to a bypass. The horizontally oriented wedge wire performed better than vertically oriented wedge-wire under low discharge conditions, but there was no significant difference in performance between the two wedge-wire configurations at high discharge conditions. de Bie *et al.* (2018) reported avoidance behaviour around the screens trialled but did not report any impingement. They suggested the horizontal wedge-wire passed water more efficiently, which may be beneficial for water abstraction. As with all screens, installation needs to ensure that approach velocities are kept within a range that prevents or minimises impingement of fish.

Behavioural screens

Behavioural screens do not use any form of physical barrier to prevent entrainment of fish. Instead they rely on some form of stimulus to deter fish from approaching irrigation infrastructure. Deterrents may include bubbles, turbulence, sound, flashing lights, an electric current, or combinations of these. The main advantage of behavioural screens is that there is no impact at all on head loss or pumping efficiency, but the key disadvantage is that these methods tend to be less effective than physical barriers at preventing entrainment of fish (NMFS 2008; Nordlund 2008). They also do not prevent debris from entering irrigation systems (U.S. Department of Interior 2006). Baumgartner and Boys (2012) did not include any behavioural screens in their list of diversion screens with high potential for direct application in the Murray-Darling Basin. Behavioural barriers may have lower capital costs than physical screens, but they only partially reduce entrainment. They may offer an option at sites that are otherwise difficult to physically screen (U.S. Department of Interior 2006) or are cost prohibitive to physically screen (e.g. a bank of flood-lifters).

Louvers

Louvers are intermediate between a physical barrier and a behavioural barrier. Louver barriers comprise of an array of vertical slats placed on a diagonal across a structure. The Louvers generate flow turbulence that fish tend to avoid (US Department of Interior 2006). Louvers are of variable effectiveness depending on the size of the fish. They are more effective for larger fish with better swimming capabilities (Skinner 1974; Nordlund 2008).

Louvers have most frequently been used at hydropower diversions but may also be used at the entrance of diversion channels. They have also been used experimentally, fitted to a box at the end of an intake pipe (Poletto 2015). The louvered box at the end of an intake pipe was tested on juvenile sturgeons. Only 5% of fish tested were entrained. These were of smaller average size (22.7 cm ± 0.4) compared to the non-entrained fish (28.9 cm ± 0.4). Juvenile sturgeon of this size would have reasonable swimming abilities and the boxed louver arrangement was not tested on smaller fish.

Louver screens operate with higher approach velocities than physical screens. They pass small debris and are less likely to have flow blockages than physical screens, but they are not absolute barriers. Their effectiveness varies with fish species, fish size life history stage and site conditions. some debris, including aquatic plants will intertwine or embed in a louver. Louver screens are not well accepted by resource agencies in the USA. (U.S. Department of Interior 2006).

Bubbles, sound and lights

Bubbles, sound and lights and combinations of these have all been trialled as fish deterrents. Sager and Hocutt (1987) tested strobe lights and bubble curtains as a deterrent to various species of North American estuarine fish species. All species tested showed little avoidance of bubble curtains. Avoidance of strobe lights varied from 8% to 100%. whilst avoidance of strobe light, bubble curtain combinations ranged from 3% to 81%. The system worked best at low flow rates. Although the strobe lights showed promise as a deterrent, they would not be suitable for highly turbid Australian river conditions.

Patrick and Christie (1985) also examined fish responses to a strobe light, air bubble barrier combination. They tested the system on various North American freshwater and estuarine fish species. They found that the three freshwater species tested, gizzard shad (*Dorosoma cepedianum*) alewife (*Alosa pseudoharengus*) and smelt (*Osmerus mordax*), all avoided an air bubble barrier (38%-73%). Bubble barriers worked best in clearwater conditions. All species tested (freshwater and estuarine) showed avoidance behaviour to strobe lights. Increased avoidance was evident for most species when strobe lights were combined with bubbles to illuminate the bubble barrier, with effectiveness ranging from 90% to 98%. Strobe lighting was found to be more effective than continuous lighting. Unfortunately, the lack of clearwater conditions at most Australian irrigation sites, would render bubble and bubble strobe-light combinations ineffective as barriers.

Sound and pressure waves can be detected by both the inner ear and lateral line of fish (Noatch and Suski 2012). Swim bladders enhance the hearing ability of fish, and in species with a coupling between the swim bladder and the ear, hearing is even further enhanced (Blaxter 1981).Hearing specialists that have an anatomical structure that connects the inner ear and a gas bubble (swim bladder) to the inner ear are able to detect frequencies up to several KHz. Generalists (with a swim bladder but no inner ear connection), detect sounds < 1KHz (Schilt 2007).

There are variable results reported for acoustic barrier systems (Noatch and Suski 2012). Ultrasound has generally been found to be ineffective (Sonny *et al.* 2006). However, Teague and Clough (2014) found ultrasound of 45 kHz caused a startle response in juvenile twaite shad (*Allosa falax*) and had potential as a deterrent. This species has particularly good hearing so this should be considered a species-specific deterrent. Successful treatments have been reported for sounds between 20 Hz and 20 KHz and for infrasound 0.1-2.0 Hz (Noatch and Suski 2012). Maes *et al.* (2004) tested an acoustic deterrent system in the Scheldt Estuary Belgium, using sounds that ranged in frequency from 20 to 600Hz. The system was installed to repel estuarine fishes from a power station cooling water intake. The sound projector array was designed by Fish Guidance Systems Ltd (FGS, UK). FGS Mk II 30-60 sound projectors were used, each with a power of 600 W. Sound output was at 174 dB. Impingement was reduced on average by 60% across the fish community. Up to 94.7% and 87.9 % reduction was noted for herring (*Clupea harengus*) and sprats (*Spattus sprattus*). The system was less effective for lampreys and Pleuronectiforms (Flatfishes *e.g.* flounders and soles) which lack swim bladders. This difference in response was probably related to hearing ability.

Generally, species with swim bladders responded well to the FGS Acoustic system, while those species without swim-bladders showed no response or only a moderate response. As noted above, fish with swim bladders have better hearing. Avoidance response success was also in part related to swimming ability. The overall reduction in impingement rates will probably depend on local conditions including water entrance speeds, background noise and fish species composition. Most Australian freshwater species have swim-bladders, so most Australian freshwater species are likely to show some response to sound. Larval flatfishes have swim-bladders, but they are lost after they metamorphose into bottom dwelling flat fish, and lampreys do not have swim bladders at any stage. A light, sound and bubble screen did not deter sea lampreys (*Petromyzon marinus*) from moving upstream in another experiment (Miehls *et al.* 2017) and this provides further evidence of sound barriers only being effective for fish with swim bladders. Some temperate and subtropical Australian freshwaters have lampreys, and some tropical Australian rivers contain freshwater soles, but these are only a small component of the total fish fauna and these species are generally absent from the major irrigation regions. Freshwater soles do occur in Gulf catchments where future irrigation developments are planned, and lampreys occur in the Murray River irrigation area but are absent from the northern Murray-Darling Basin.

Knudsen *et al.* (1994) examined the effect of sound as a deterrent on Atlantic salmon smolts. They found intense sound at 150 Hz had no observable effect on the smolts, but sound at 10Hz was an effective deterrent. Six smolts were captured moving through a slit into a fish trap when the 10Hz sound was active, and 338 smolts moved into the trap when the sound was turned off. Fish reacted up to 3 m away from the sound source.

Pegg and Chick (2004) were able to reduce passage of bighead carp *Hypophthalmichthys nobilis* and silver carp *H. molitrix*, by 95% using acoustic barriers with a signal sound of 20-2000Hz. A signal of 20-500Hz only repelled 57% of these Asian carp species.

Sound as a deterrence has also been used in conjunction with bubble curtains. Zielinski *et al.* (2014) evaluated the performance of fine, graded, and coarse bubble curtains to reduce passage of common carp (*Cyprinus carpio*). They also evaluated acoustically enhanced bubble curtains. Coarse and graded bubble systems reduced passage of carp by 75-85%. These bubble systems were found to produce sound near 200 Hz at approximately 130 dB. Combining fine bubbles with a speaker array resulted in reduced passage equivalent to the coarse bubble system. Further testing with speaker arrays and lighting indicated carp avoidance of the bubble curtains involved responses to sound and fluid motion, rather than visual cues.

Welton *et al.* (2002) examined the efficacy of acoustic bubble screens to deflect Atlantic salmon smolts. The system known as a bioacoustics fish fence (BAFF) which is manufactured by FGS, used a sound generator in conjunction with an air bubble sheet to create a wall of sound. The BAFF unit works on the principal of sound propagation in air-water mixtures. A significant number of smolts were deflected. Efficiencies were better at night than in the day, due to smolt behaviour, with 72.9-73.8% being deflected at night and 20.35-43.85% being deflected in the day. All were significantly better than control treatments when the acoustic bubble barriers were switched off. Welton *et al.* (2002) did caution that the style of acoustic bubble barrier they used could potentially be damage by bedload movement.

Deleau *et al.* (2020) examined the use of acoustics to enhance the efficiency of a physical screen to divert migrating European eels *Anguilla anguilla*. They found that the acoustic treatment enhanced the guidance efficiency of the physical screen.

Putland and Mesinger (2019) reviewed 286 experimental acoustic deterrent studies on 111 species. They noted advantages of acoustic systems were their potential long range, their independence of light conditions, and no clogging by debris. This was offset by variability in reported effectiveness, ranging from 0% to 97%. They found a promising avenue was to place acoustic deterrents at strategic bottlenecks. (Perhaps this could apply to side channels for pumped offtakes in Australian irrigation systems). They also observed that many of the publications on acoustic deterrents occurred in the grey literature (not included in this current review of acoustic systems), often including authors who developed or sell the systems, which could be construed as a conflict of interest. Most studies did not examine habituation effects, including those reported in this review. *i.e.* fish may become used to the sounds and stop avoiding them. Habituation could be an issue at sites where water diversion is nearly continuous, where acoustic deterrents would operate for long periods, such as at hydropower diversions or some irrigation diversion channels. However, where irrigation offtakes pump for only short durations of several days at a time (*e.g.* flood flow harvest, or allocated flow harvest), then habituation is less likely to be of concern.

Acoustic deterrents might provide an option for some Australian irrigation offtakes, where more conventional physical screening options are logistically difficult or cost prohibitive. Although acoustic deterrence is never likely to achieve 100% exclusion, it could still potentially substantially reduce entrainment of fish. However field testing of acoustic deterrents has not yet been done in Australia, so research into this type of technology under Australian field conditions, with Australian fish species needs to be completed by independent researchers before any firm recommendations can be made on the use of this technology.

Electric barriers

Electric barriers pass a current through the water from an anode to a cathode. This is meant to induce behavioural avoidance of the electric field by fish. Electric currents tend to be size selective, with smaller fish being less affected by electric currents than larger fish (Noatch and Suski 2012). Electric barriers have mostly been used to prevent invasive species expanding their range up shipping canals and constricted waterways, but they have also been used to prevent entrainment (Noatch and Suski 2012). Electrical fields have not been shown to be very successful in guiding fish and have had limited success as fish barriers (U.S. Department of interior 2006). Water conductivity, voltage, pulse frequency and duration and electrode configuration can all influence the performance of an electric barrier (Utz *et al.* 2017)

Uz *et al.* (2017) investigated using a high frequency pulsed electric current to exclude fish (fathead minnows) and crayfish in a lab situation. In low conductivity waters of (<50 µS.cm^{-1}) fish were deterred at >200V. In waters with conductivities exceeding 250 µS cm^{-1} the electric current caused immobilisation of the fish. Immobilisation would not prevent entrainment. Pulse durations of 150 µs were more effective than 50 µs.

Egg *et al.* (2019) found that an electric fish fence (an electrified fence with 50 mm gaps consisting of a frame with horizontally tightened steel ropes) developed by Aufleeger *et al.* (2014) was able to deter up to 72% of fish at an intake pumping station by inducing the fish to turn away. This behaviour was most significant in large, streamlined fish. No turning behaviour was observed when the fence was not powered up. The mean current velocity around the fence was 0.05 m.s^{-1} and the mean temperature was 4.3ºC. Bullen and Carlson (2003) reported that an electric barrier was 98.7 % effective in containing grass carp *Ctenopharyngodon idella* within a single bay of a lake.

The U.S. Department of Interior (2006) considered electric barriers to be not very effective as a stand-alone technique. Nordlund (2008) considered electric barriers, along with other behavioural barriers to be less successful than physical barriers and to have lesser degrees of success when unusual hydraulic conditions occur. The advantages of electric deterrent systems include flexible deployment and their general effectiveness for larger sized fish. Disadvantages include variable performance at different conductivities, reduced effect on smaller fish and potential safety concerns for humans and livestock that may come into contact with electrified water. Smith Root require their electrical barriers to be fenced. Smith Root state that the electric barriers produced by them use pulse frequencies that are set lower than those used in traditional electrofishing with the aim of achieving changed fish behaviour and not galvanotaxis (forced swimming) or tetany (stunning). The systems are designed to be non-lethal and use low frequency pulsed DC (Smith Root 2020). However, these settings may have less of a behavioural impact on small fish.

Summary of screen types

Table 2.1 contains a summary of the different screen types and lists the types of situations they are most suited to, and the pros and cons of the different screening options. It would appear that wedgewire self-cleaning cylinder screens are a potential option for many pumped irrigation offtakes, with self-cleaning cone screens being another option for pumped offtakes, especially at more sheltered or shallow sites. Irrigation diversion channels can use a range of options, with vertical fixed plate screens, horizontal plate screens, rotating drums and travelling screens all being options. Cone screens have also been used at the entrance of irrigation diversion channels. If screens are situated within a diversion channel, then a fish bypass is normally required, but if screens can successfully be installed at the diversion channel entrance, then a bypass is not normally required.

Behavioural screens have the advantage of unimpeded flow, but none are 100% effective at excluding fish. Bubble curtains and strobe lights would appear to have little merit in highly turbid Australian waters. The merits of electrical barriers to keep fish away from diversions is still questionable, but acoustic barriers, including acoustic barriers in conjunction with bubble screens may have merit in some situations in Australia where other physical barriers may be cost prohibitive or logistically difficult to install. However, these should first be evaluated under Australian conditions before proceeding with implementing them anywhere.

Table 2.1: Summary of screen types, their suitability for different applications, advantages and disadvantages. Physical screens are shaded pale yellow and behavioural screens light blue. This table is based on references as listed in the preceding pages of this chapter, including U.S. Department of Interior (2006), Jamieson *et al.* (2007), Nordlund (2008), Baumgartner and Boys (2012), Noatch and Suski (2012) and Mefford (2013). Small diversion channels are defined as <1000 ML/D, large diversion channels as >1000 ML/D. large pumps are defined as >100 ML/D and small pumps as <100ML/D as per Baumgartner and Boys (2012).

Screen type	Diversion type where used	Advantages	Disadvantages	Relative cost	Comments
Rotary drums	Within small and large diversion channels Entrance to diversion channels. Mostly used for smaller flows.	Active cleaning, proven technology. Better than 98% survival of juvenile fish when installed correctly.	Water level must be between 0.65 and 0.85 drum diameter for effective cleaning. Increased impingement of fish when water depth exceeds 0.85 drum diameter. Seals need regular monitoring for wear.	Capital costs tend to be higher than flat plate screens	Requires a fish bypass if installed within channel. Need to ensure approach velocities are below 0.12 $m.s^{-1}$. Screens mounted within a canal recommended to be at 15-30 degrees to the flow.
Coanda screens	Within diversion channels Entrance to diversion channels	Passive cleaning. High flow capacity. No moving parts. Exclude sediment from diversion.	Tends to strand or injure fish if flow across the screen is insufficient. Difficult to control bypass flow. Possible dewatering of screen toe. Requires elevation drop >90 cm. Substantial bypass flow may be required.	No information	Better for preventing invasive fish movements. Not so useful where aim is for diverted fish to survive. Fish need to be diverted over the screen to a bypass or the River. Not widely used in Pacific Northwest of USA.
Horizontal fixed plate	Within small diversion channels	Passive cleaning. Good for shallow flow. No moving parts.	Some internal baffling below screen may be required. Screen may be blocked with bottom sediments. Will require occasional physical cleaning to remove sediment or biofouling. Bottom oriented fish exposed to full screen length. Requires flow fluctuations to be small. Limited to small diversions.	Cost effective	Fish bypass required. Sweep velocity needs to be high for effective passive cleaning
Cylindrical Screens	Pump intakes, small and large. Gravity fed conduits.	Many have cleaning mechanisms installed for efficient water flow. Retrievable versions available. Large surface area per unit length. Small footprint. No need for a fish bypass. Exclude debris from diversion.	May be subject to impacts from debris. Minimum depth of water and clearance requirements	Start from around $20,000 Au for wedgewire screens on small pump systems. Woven mesh screens may be cheaper.	Wedgewire screens more likely to be cleaned effectively. Recommend brush cleaned screens. Should be set parallel to stream flow. Wedgewire screens more resilient to impacts than woven mesh.

Screen type	Diversion type where used	Advantages	Disadvantages	Relative cost	Comments
Cone screens	Pump intakes, small and large. Entrance to small and large diversion channels through head wall	Many have cleaning mechanisms installed for efficient water flow. Suitable for shallow water. Large surface area achievable in shallow water depths. No need for a fish bypass. Good performance record. Exclude debris from diversion.	Uniform distribution of flow through the screen decreases with increasing channel velocity. Without appropriate external baffling restricted to low flow velocity areas.	No information	Wedgewire screens more likely to be cleaned effectively. Recommend brush cleaned screens
Travelling screens (Vertical or inclined)	Mostly within small and large diversion channels to direct fish to a bypass placed at the entrance to a diversion channel	Good cleaning characteristics. Can work well in sediment laden flows. Can operate over a wide range of depths.	Numerous moving parts with periodic adjustments of belts required. If not installed correctly impinged fish can be carried on travelling screen mesh seal problems difficult to identify.	Because of the relatively high cost, usually only used for smaller flows	Addition of travelling bucket system can improve survival of impinged fish. Belt screens not a first choice according to Baumgartner & Boys 2012
Vertical panel or fixed plate screens	Mostly within small and large diversion channels or entrance to diversion channel. May screen pump intake sump.	Passive cleaning. Good cleaning characteristics with appropriate site. No power required. When installed correctly have excellent fish protection performance.	Occasional manual cleaning required. Cleaning effectiveness can be impacted by changes in stream conditions. Mechanical cleaner is recommended if diversion flow is >0.5 times the stream channel flow.	Low operating cost	Sweep velocity needs to be high (>15 x approach velocity) for effective passive cleaning. Within diversion channel use will require a fish bypass back to the river. Not a first choice for pumped diversions according to Baumgartner and Boys (2012)
Louvers	Within small and large diversion channels or entry to diversion channel	Reduced risk of flow blockages. Can pass small debris. Higher approach velocities than physical screens. Effective for larger stronger swimming fish.	Requires high velocity sweeping flow. Less effective for small fish. Can become entangled with weed.	Relatively low cost	Fish bypass required if installed within channel Response of Australian species unknown. Not widely accepted in USA. Opposed by US west coast fisheries agencies.
Bubble screens	Entrance to small and large diversion channels or around small and large pump intakes	No loss of pumping efficiency	Low effectiveness as stand-alone screens for most species	Potentially low cost	Bubbles alone not an effective barrier for most species. Untested on Australian species. **Not recommended**
Strobe lights (including bubble combinations)	Entrance to diversion channel, or around pump intake	No loss of pumping efficiency. Less infrastructure	Will not be effective in turbid waters, which prevail in many Australian irrigation areas	Potentially low cost	Strobe lights will not be effective in many flow events due to high turbidity levels. **Not recommended**

Screen type	Diversion type where used	Advantages	Disadvantages	Relative cost	Comments
Acoustic screens (including light and bubble combinations)	Entrance to diversion channel or around pump intake	No loss of pumping efficiency. Trials overseas have excluded more than 90% of fish with this technology. Effective across a wide range of environmental conditions.	Variable in effectiveness. The right frequency range needs to be selected. Potential for damage of systems by debris. Not tested on Australian species. May require research to find most effective acoustic frequencies for Australian species. Probably not effective for fish larvae.	Potentially lower cost than physical barrier systems at sites with difficult configurations	Acoustic screens may have potential to partially exclude fish, but untested on Australian species. A possible option for sites where physical screens are logistically difficult or cost-prohibitive to install. Site needs adequate acoustic characteristics
Electric barriers	Entrance to small and large diversion channels. Possibly adjacent to pump sites.	No loss of pumping efficiency. Efficient against recruited fish	Effectiveness will vary with temperature and conductivity. If settings are wrong could stun fish. Not so effective on small fish. Fish fatigue around the electric field. Possible safety concerns. Need for electrical control station on site	Installation potentially costly	May possibly be used at entry to side channel from which intake water is pumped, but if fish are stunned rather than deterred, could increase entrainment. More widely used to prevent upstream movement of pest fish. May be less effective at preventing downstream movement into an irrigation diversion, especially if fish are inadvertently stunned.

All physical screens need to be installed such that the approach velocities are below 0.12 $m.s^{-1}$ and in the case of passive screens they need to be positioned so that sweeping velocities exceed approach velocities.

Unfortunately, not much has been published on the cost of the different screening options other than some references to relative costs. According to the U.S. Department of Interior (2006), capital costs depend on the type of facility required, the site characteristics, and the flow rate. Unit costs for a facility (cost per delivered volume) can vary widely because of site characteristics. However, unit costs tend to reduce with increasing volume.

What is in it for the farmer?

There is very little published in peer reviewed literature on the costs and benefits of fish screening options for irrigators. Well designed, self-cleaning fish screens have the benefit of reduced fouling, and hence maintenance requirements (Baumgartner and Boys 2012). Unscreened diversions not only entrain fish, but also sticks and debris that can clog irrigation systems. The experience of Pacific Coast Irrigators in the USA suggests reduced running and maintenance costs from fish screening can be a major motivator for irrigators to install and maintain screens (Boys *et al.* 2012). The Fish Screens Australia website suggests that fish screens will lead to less fouling and more reliable water supply than is obtained through current trash racks, whilst maintaining pumping volume. https://fishscreens.org.au/screens/ The website features images of severely clogged trash racks compared with unclogged self-cleaning screens. The finer screens will mean less debris entering the irrigation system. Cleaner water provides more options and leads to less time unblocking centre pivots, and sprinkler mechanisms.

A study on the costs and benefits of screening is needed to better inform the wider irrigation community. NSW Local Land Services has embarked on such a study in the Macquarie and Lachlan River systems. Irrigators who have installed screens are being asked to monitor water quality improvements and savings. www.centralwest.lls.nsw.gov.au Hopefully the findings will be published in a report. The Fish Screens Australia website also features screen showcase sites. The experience at these sites should be able to provide feedback to the irrigation community at some time in the future. For example, the Trangie Nevertire Irrigation Scheme was recently fitted with cone screens. Scheme members are hopeful that the expected reduction in debris will benefit lateral move or pivot irrigation systems with nozzles prone to blocking with debris. https://www.abc.net.au/news/2020-06-19/state-first-as-fish-exclusion-screen-installed-near-trangie-nsw/12343218 (see also Spotlight on Cotton R&D summer 2020/21 edition).

Screen installation is a capital expense and screening larger systems can be costly. Despite the possible benefits to farmers from better water quality and reduced maintenance, the expense of installing a screen is a real issue for farmers to consider. Uptake of screens has generally been best where financial incentives are provided. For example, the Oregon Department of Fish and Wildlife has a cost share program, covering up to 60% of design, engineering and installation costs, and there are also additional tax incentives (Baumgartner and Boys 2012, Boys et al, 2012, Oregon Department of Fish and Wildlife 2013). The Oregon state law also requires government agencies to be responsible for screen repairs at diversions less than 73.4 ML/D (Baumgartner and Boys 2012). Recently in NSW Australia, Local Land Services has been offering subsidies of up to $25,000 for installation of fish screens along the Lachlan and Macquarie Rivers. https://www.lls.nsw.gov.au/__data/assets/pdf_file/0005/1248584/Fish-screens-Info-sheet-.pdf

A similar scheme also exists for the Darling River with subsidies of $5,000. The funding has been provided by Catchment Action NSW (Local Land Services 2020). Further schemes like these may be required in Australia to assist and motivate irrigators to uptake fish screening.

Those properties that do install fish screens will be able to use the low impact on fish as an opportunity to portray Australian cotton as an ecofriendly product. This could provide a marketing advantage in some sectors.

Conclusions

Several screening options are available that have evidence for being effective at preventing entrainment and impingement of fish so long as they are installed correctly for the prevailing conditions. These screens can maintain desired diversion volumes, and many have self-cleaning mechanisms to prevent accumulation of debris, increased head loss and loss of flow volume. For pump offtakes, brush-cleaned wedge-wire cylinder screens and cone screens appear to be two promising options. Suitable fish screens for gravity fed irrigation diversions include fixed vertical plate screens, wedge-wire cone screens and horizontal fixed plate screens. Rotary drum screens and travelling screens are also effective for gravity diversions but may have a higher unit cost. Ongoing evaluations of screens installed Australian waters should provide further information on the efficacy of these screens for excluding fish and the costs and benefits to irrigators. This will assist irrigators to make informed choices. It is imperative that the economic costs and benefits of these screens for farming operations be evaluated and published.

Most of the behavioural screening options, although potentially of lower cost do not appear to have merit for Australian irrigation conditions, except perhaps acoustic screens. Experimental evaluation of acoustic screens in Australian waters is recommended, as this technology may provide an alternative option to reduce losses of fish in sites where installation of a physical screen is logistically difficult or cost prohibitive. When data is available on the behaviour of Australian fish species around acoustic screens an evaluation of their suitability for Australian conditions can be made. These screens would not physically alter existing pumping operations, but they would also not result in reduced debris entering the irrigation system, which is a potential benefit of physical screens.

There will be capital costs involved with any fish entrainment mitigation system for irrigation offtakes. Subsidies or grants to assist with purchase and installation costs would be a useful mechanism to encourage voluntary uptake of fish screening in Australia. Reducing impacts of irrigation on entrainment of fish will provide a positive marketing opportunity for the cotton industry and other irrigated agriculture to portray a clean and green image for Australian produce. Through use of appropriate screening this can be achieved without reducing the amount of water harvested for irrigation.

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