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Ecology, Monitoring, Conservation

Conservation translocation handbook for New South Wales threatened freshwater fishes

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A report to the NSW Department of Primary Industries Fisheries

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Acknowledgement of the Traditional Owners of the Murray–Darling Basin

We respectfully acknowledge the Traditional Owners, their Elders past, present and emerging, their Nations of the Murray–Darling Basin, and their cultural, social, environmental, spiritual, and economic connection to their lands and waters.

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1. BACKGROUND

Amid a global biodiversity crisis that is currently occurring (Harrison et al. 2018), freshwater fishes appear disproportionately at risk (Darwall and Freyhof 2016). For example, 28% of freshwater fish species assessed for the International Union for Conservation of Nature (IUCN) Red List of Threatened Species are deemed threatened with extinction (Tickner et al. 2020). This number is probably much higher as our knowledge of true levels of diversity is incomplete (Adams et al. 2014). Many threats have been imposed on freshwater fishes including habitat loss and degradation, invasive species, over-exploitation, pesticides, pollution, water abstraction and flow alteration and climate change (Arthington et al. 2016; Darwall and Freyhof 2016; Dudgeon et al. 2006). In Australia, these threats place freshwater fish at extreme risk, with almost one-third of Australia's freshwater fish species threatened (under international listings). At least 20 species are at risk (with > 50% probability) of becoming extinct in the next ~20 years (Lintermans et al. 2020), some of which were only recently discovered or described (Raadik 2014). Several of these species are among the 49 species of native freshwater fishes that occur in the expansive Murray-Darling Basin (MDB) (Lintermans 2007; Lintermans, unpublished data). Some of these species endemic to the MDB, such as Flathead Galaxias *Galaxias rostratus* and Murray Hardyhead *Craterocephalus fluviatilis*, are also considered at risk of extinction. Indeed it is forecast that the natural habitat of five freshwater fishes will be completely lost by the end of the century (Galego de Oliveira et al. 2019).

Additionally, freshwater fishes of the MDB have also been heavily impacted by recent prolonged drought (Millennium drought from 2001-2009, as well as severe drought and lowest-on-record inflows through 2016-19 for some basin regions) and widespread bushfires over 2019−20. In New South Wales (NSW) a Native Fish Drought Response (NFDR) was initiated in 2019 to provide a strategic, proactive response to protect native fish through the prevailing drought. This included assessing all potential response options (including flow delivery, aeration, fish rescue and relocation, compliance, and communication), to guide drought response actions and native fish recovery. As part of the NSW drought response, contingency populations of many species from at risk locations were salvaged. From a group of threatened small-bodied threatened wetland specialists, Olive Perchlet, (~630 individuals), Southern Pygmy Perch (~740), and Oxleyan Pygmy Perch (~292), among several other species, were rescued with the view to implement ex situ maintenance and production. By definition, ex-situ maintenance and production involves the management of individuals in a controlled or modified setting away from known wild populations of the species (IUCN/SSC 2014). Similar emergency actions were implemented for fish species, including Stocky Galaxias *Galaxias tantangara* and Short-tail Galaxias *Galaxias brevissimus*, in response to the 2019−20 bushfires.

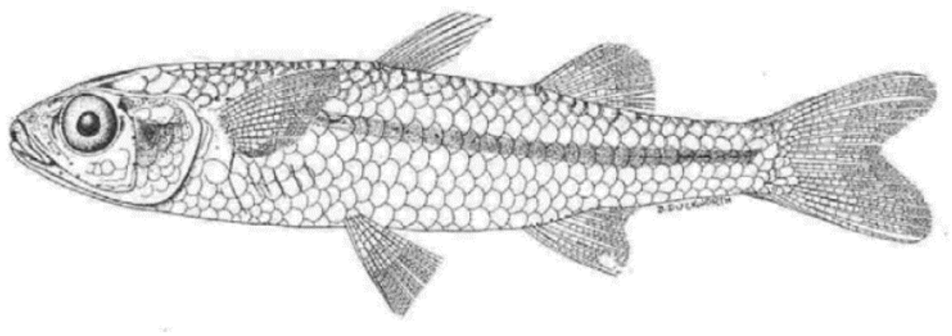
This conservation translocation handbook provides a synthesis of knowledge of nine target threatened freshwater fishes in NSW to guide conservation actions in relation to the NFDR and other initiatives such as the Native Fish Recovery Strategy (MDBA 2020). The included freshwater fishes have all experienced historical declines in distribution and abundance and have been further impacted by the prolonged drought and/or the 2019−20 bushfires. These include:

- Murray Hardyhead *Craterocephalus fluviatilis*
- Olive Perchlet (MDB population) *Ambassis agassizii*
- Oxleyan Pygmy Perch *Nannoperca oxleyana*
- River Blackfish (Snowy River population) *Gadopsis marmoratus*
- Round-snout Galaxias *Galaxias terenasus*
- Short-tail Galaxias *Galaxias brevissimus*
- Southern Purple-spotted Gudgeon *Mogurnda adspersa*
- Southern Pygmy Perch *Nannoperca australis*
- Stocky Galaxias *Galaxias tantangara*

Following this introduction (Section 1), Section 2 provides a summary of the conservation and population status, biological information, genetic management and known threats and knowledge gaps of each of the nine target species (up until December 2020). In Section 3, information about ex-situ maintenance and production of each species is provided. Section 4 provides guidance on conservation translocations including a background, planning, implementation and monitoring and evaluation. Section 5 summarises priority actions and recommendations. This comprehensive and informed handbook should not be viewed as standalone, but as supporting to other planning documents to achieve effective recovery of the threatened freshwater fishes.

2. SPECIES SUMMARIES

2.1 Murray Hardyhead *Craterocephalus fluviatilis*



(B Duckworth and Crowley and Ivantsoff (1990))

2.1.1 Conservation status

International: *Critically Endangered*
National: *Endangered*
NSW: *Critically Endangered*
Rest of range: *Critically Endangered* (SA); *Threatened* (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales: *Fisheries Management Act 1994*; South Australia: Action Plan for South Australian Freshwater Fishes 2009 *and Fisheries Management Act 2007*; Victoria: *Flora and Fauna Guarantee Act 1988*.

2.1.2 Population status

Murray Hardyhead is endemic to lowland floodplains of the Murray and Murrumbidgee rivers where it is was historically common (Crowley and Ivantsoff 1990; Ellis et al. 2013; Lintermans 2007; Stoessel 2010). With river regulation (1920s-1980s) and a general reduction in the availability of shallow saline, and vegetated wetland habitats, the species experienced significant declines through the latter half of the 20th century. Murray Hardyhead were considered absent from the mid-Murray and Murrumbidgee rivers (NSW) in the early 2000s, but persisted in fragmented, often isolated, populations in the Murray River below Lock 1 and the Lower Lakes, lowland Murray River floodplains in the South Australian Riverland Victorian Mallee region, and wetlands in the Kerrang Lakes region of north central Victoria (DELWP

2017; Ellis et al. 2013; Hammer et al. 2013). The impacts of river regulation were exacerbated by critical water shortages during the Millennium Drought during which (or shortly after), some remnant populations across these regions were lost while others experienced dramatic declines in abundance (DELWP 2017). Yet, a number of key sites were maintained with environmental watering, including Berri Evaporation Basin and Disher Creek and Boggy Creek (South Australia); and the Cardross Lakes and Round Lake (Victoria) (Bice et al. 2014; Ellis et al. 2013; Wedderburn et al. 2014). Captive breeding populations were also salvaged from at risk sites across Victoria and South Australia and maintained as a contingency back up measure from 2008-2013 (Ellis and Carr 2011; Ellis and Kavanagh 2014; Ellis et al. 2009).

Since the Millennium drought there has been some localised recovery in several locations in part due to delivery of environmental watering and reintroduction efforts (Bice et al. 2014; Ellis et al. 2013; Wedderburn et al. 2014). In the Lower Lakes, the species has seen limited recovery attributed to the persistence of the Boggy Creek site with environmental water and reintroduction of 7520 fish (Bice et al. 2014). Reintroductions have also helped the species persist in the Rocky Gully wetland (Whiterod and Gannon 2020). Unfortunately, reintroductions to three sites across Lake Albert on four occasions between 2016 and 2019 do not appear to have facilitated the successful reestablishment of the species.

Subpopulations in the Riverland region (including Berri Evaporation Basin and Disher Creek) have been maintained through targeted water delivery, and the species was recently rediscovered in the Gurra Gurra Wetland Complex (Whiterod and Gannon 2019; authors, unpublished data). In the Victoria Mallee region, the only historically known remnant population in the state exists in Round Lake (near Kerang), while translocated populations are established in Koorlong Lake, Brickworks Billabong and Lake Elizabeth. An attempted reintroduction of the species to Lake Hawthorn in 2018 was unsuccessful (Dan Stoessel, ARI, unpublished data; Whiterod and Wood 2019). A population discovered in 2012 in Lake Kelly (near Kerang) following widespread flooding in the region, and which persisted in a channel system (Tutchewop Main Drain) in direct association with the lake for several years, is now likely to be extirpated (Stoessel and Dedini 2013). In NSW, Murray Hardyhead sourced from the Disher Creek population in South Australia were released into Little Frenchman's Creek and an associated surrogate refuge dam in late 2018. Monitoring has demonstrated survival and subsequent breeding and recruitment, suggesting reestablishment of Murray Hardyhead in Far West NSW (where it had not been detected since 2005), and the first successful attempt in NSW to reestablish a regionally extinct native fish (Ellis et al. 2020; Ellis et al. 2018).

The estimated area of occupancy (AOO) is 96km^2 and extent of occurrence (EOO) is 46,038km^2, and the overall population trend is deemed to be declining (Stoessel et al. 2019). Furthermore, Stoessel et al. (2019) infer that based on current trends, where a subpopulation has been lost every one to two years for the past 40 years, the loss of the remaining subpopulations may occur within the next 10 years without further intervention.

2.1.3 Biological Information

Murray Hardyhead grow up to 100 mm TL, have a small protruding mouth, large silvery eyes, moderately rounded snout, two small and short-based dorsal fins, a forked tail, with pectoral fins positioned high on the body (Lintermans 2007). Murray Hardyhead can be distinguished from other hardyhead species by several attributes. It is distinguished from the Lake Eyre Hardyhead *Craterocephalus eyresii* by its non-overlapping geographical range. Although it is unlikely to co-exist with Darling River Hardyhead *Craterocephalus amniculus* given known ranges, it has fewer mid-lateral scales than the Darling River Hardyhead (Murray Hardyhead: 31-35 scales; Darling River Hardyhead: >38 scales). Murray hardyhead frequently co-occur with Unspecked Hardyhead *Craterocephalus stercusmuscarum fulvus* but is distinguished primarily by possessing a differing number of transverse scales compared (Murray Hardyhead: 10 or 11 scale rows with 3 rows above the lateral line; Unspecked Hardyhead: 7 or 8 scales). Additionally, Murray Hardyhead scales are generally roundish with pigment around the margin, while Unspecked Hardyhead appear diamond shaped and are arranged in uniform rows, with pigment through the scale as well as around the margin (Ellis and Kavanagh 2014). In the Lower lakes region where it co-occurs with Smallmouthed Hardyhead *Atherinosoma microstoma*, Murray Hardyhead have a deeper body and shorter gill rakers (Hammer and Wedderburn 2008).

Murray Hardyhead is a short-lived (<2 years), salt-tolerant species (Ellis 2006). Salinity tolerance is dependent on life-stage, with juveniles (and therefore likely adults) capable of surviving salinities in excess of 90 ppt or >100,000 µScm^{-1} electrical conductivity (hereby referred to as 'EC' throughout), while egg hatch rates and larvae survival cease at salinities approaching 30 ppt, or 50,000 EC at water temperatures of 24°C (Stoessel et al. 2020a). Apart from the freshwater Lower Lakes in SA, the species appears to be restricted to isolated moderately to highly saline wetlands. The species is frequently observed schooling in open-water and amongst aquatic vegetation such as fringing emergent rushes (i.e. *Cumbungi* and *Juncus*), and submerged macrophytes, including *Ruppia* and *Myriophyllum* (Ellis 2005) and recently submerged terrestrial plants. Juvenile and adult Murray Hardyhead feed predominantly on micro-crustaceans, although larger fish tend to have a more diverse diet, consuming larger prey items such as dipteran larvae and pupae (Ellis 2006). Increased water level and surface area or flooding during the spawning period (spring to summer) enhances rotifer and zooplankton abundance which likely benefits recruitment success of the species (Ellis 2005; Wedderburn et al. 2010).

Murray Hardyhead have a prolonged spawning season from September to March (spring and summer), with peak larval abundance usually occurring in late October to early November (Ellis 2005). The species is a batch spawner, with females depositing clutches of up to 80 (possibly more) adhesive eggs on submerged vegetation. Hatching takes an average of 13 days after fertilisation at temperatures between 24 and 25°C. The species reaches sexual maturity at around 25−30 mm standard length (SL), which may be reached within 120 days for fish spawned early in the breeding season, meaning fish produced early in a breeding may themselves spawn later in the same breeding season (Ellis 2005; Ellis 2006). The abundance of adults declines at the end of the breeding season (January/February), with replacement by the maturing juveniles (Ellis 2005). Failed spawning and recruitment may therefore result in the rapid local extinction of populations (Ellis 2005; Stoessel 2010).

2.1.4 Genetic management

The remnant subpopulations of the species have been managed as five conservation units on the basis of genetic distinction, these being the (1) lower Murray River and Lower Lakes; (2) Riverland and Victorian Mallee regions; but with separate units for the (3) Kerang Lakes:

Round Lake and Lake Kelly; and (4) Woorinen North Lake (believed to be extinct); and (5) Lake Elizabeth (Adams et al. 2011). Recent population genetic analyses effectively consolidates these units into two meta-populations for which there are nine partially isolated subpopulations (Thiele et al. 2020). The meta-populations being the (1) lower Murray River and Lower Lakes, (2) the Riverland and Victorian Mallee regions and, although they did not form part of the analyses of Thiele et al. (2020), NSW subpopulations (e.g. the reintroduced subpopulation as well as any future subpopulations). It is increasingly recognised that separate management of subpopulations may be reinforcing genetic isolation, thus managing more broadly (e.g. meta-population level) is now recommended (Thiele et al. 2020).

2.1.5 Known threats and knowledge gaps

The species has experienced rapid and ongoing decline, attributed to multiple, compounding threats (DELWP 2017; Ellis et al. 2013; Hammer et al. 2013). These include the impact of river regulation and water abstraction that have contributed to the deterioration and loss of the floodplain wetlands and changing of salinity regime, as well as the impact of alien species (Ellis et al. 2013; Hammer et al. 2013; Wedderburn et al. 2017). Many of these threats relate to the deterioration and loss of shallow vegetated saline wetland habitats preferred by Murray Hardyhead (Wedderburn et al. 2007). These habitats have been impacted by river regulation for decades, but these impacts were compounded during the Millennium Drought. Several known sites became extinct or were relegated to small captive breeding populations during the drought, while others experienced substantial declines in abundance (DELWP 2017). Fortunately, captive breeding attempts were largely successful, with knowledge regarding the biology and behavior of the species gained via captive management later informing the in-situ management of wild and re-introduced populations (Ellis et al. 2013). Furthermore, research determining the salinity tolerance of various life stages has informed environmental watering of known (and reintroduction) sites that support the species (Stoessel et al. 2020a).

Building on the summary of Koehn et al. (2017), Koehn et al. (2020b) summarises the amount of available knowledge that exists for the species (Table 2-1), indicating that, typically, less than 59% of knowledge that is needed is presently available.

Table 2-1. Status of available knowledge for life stages of Murray Hardyhead (available knowledge was scored as follows: 1: 0–19% of knowledge needed is available; 2: 20–39% of knowledge needed is available; 3: 40–59% of knowledge needed is available; 4: 60–79% of knowledge needed is available; 5: 80% of knowledge needed is available): adapted from Koehn et al. (2020b).

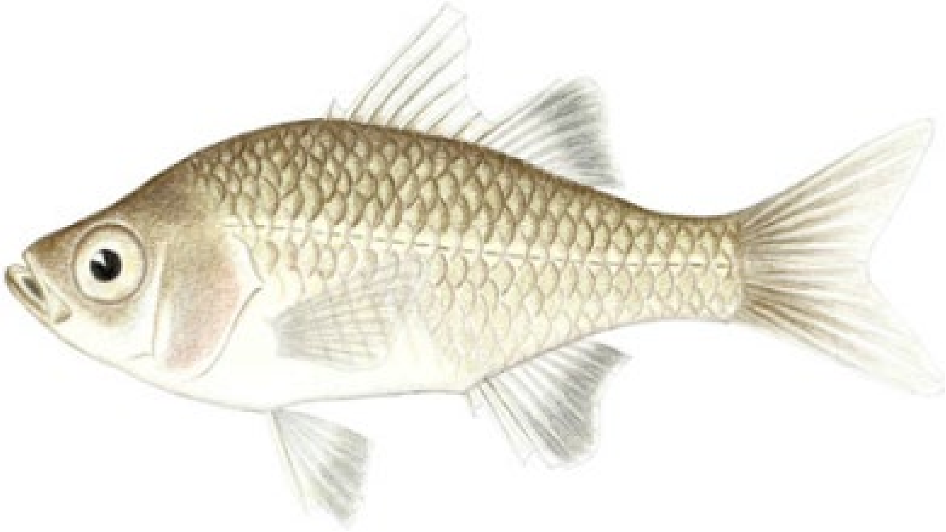
Spawning	Eggs	Larvae	Juveniles	Adults
Spawning	2.5			3.0
Spawning conditions				3.0
Survival (recruitment)	2.0	2.0	2.0	2.0
Growth and condition		2.0	2.0	3.0
Movements		2.0	2.0	2.0
Physical habitat requirements	3.0	3.0	3.5	4.0
Water quality tolerances	2.0	2.0	3.0	3.5
Flows requirements	2.5	2.5	2.5	2.5

2.1.6 Overall summary

Murray Hardyhead persists in the wild often benefitting from strategic environmental water delivery. The discovery of formerly unidentified subpopulations and reintroductions have also improved the outlook for the species. In Lake Alexandrina, the species persists at multiple locations, but has not been detected in Lake Albert since 2008 despite reintroduction attempts. Across the Riverland region, the Berri Evaporation Basin and Disher Creek subpopulations appear secure so long as a managed environmental watering regime is adhered to (noting these subpopulations do exhibit variability in abundance). Identification of a population at the Gurra Gurra Wetland Complex provides further security for the species in the Riverland. In NSW, the reintroduced population at Little Frenchman’s Creek has demonstrated successful recruitment over three consecutive breeding seasons (2018-19, 2029-20 and 2020-21) and appears secure as long as a managed environmental watering regime is adhered to. In the Victoria Mallee region, an isolated translocated subpopulation persists in Koorlong Lake, Brickworks Billabong. In north central Victoria (Kerrang Lakes region) a remnant subpopulation persists in Round Lake, and a re-introduced population survives in Lake Elizabeth. Subpopulations may also remain in the Reedy Lakes and Tutchewop Drain system. Managed “surrogate refuge” populations in South Australia provide both a backup and contingency measure, as well as an opportunity to contribute to future reintroductions should appropriate sites be determined.

We have sufficient understanding of the breeding ecology and salinity tolerances of the species different life-history stages to inform appropriate conservation management of existing populations. The challenges ahead include ongoing commitment to this conservation management, and a shift in focus towards 'recovery' whereby additional translocated populations are established in locations that serve as nodes for dispersal during future flood events (i.e. flow mitigated dispersal and genetic mixing between populations). Continued active management and reintroductions are required as well as mitigation of pest species and other prevailing threats to ensure persistence and meaningful recovery of this (and other) species.

2.2 Olive Perchlet (MDB population) *Ambassis agassizii*



(NSW DPI Fisheries)

This section focuses on the MDB population of Olive Perchlet.

2.2.1 Conservation status

International: *Least Concern*
NSW: *Endangered population* (MDB population)
Rest of range: *Critically Endangered* & *Protected* (SA); not listed (QLD), *Threatened* (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales: *Fisheries Management Act 1994*; South Australia: Action Plan for South Australian Freshwater Fishes 2009 *and Fisheries Management Act 2007*; Victoria: *Flora and Fauna Guarantee Act 1988*; QLD: *Nature Conservation Act 1992*.

2.2.2 Population status

Olive Perchlet was historically widespread throughout the MDB in NSW, Queensland, South Australia and Victoria, and in coastal streams of north eastern NSW and south eastern Queensland (Allen and Burgess 1990). In the MDB, it was found broadly from both the northern (Darling River, Border Rivers, Bogan River, Clarence River and Condamine-Balonne, Nebine and Warrego River) and southern catchments (Lachlan River, Murrumbidgee River and Murray River downstream to the Lower Lakes) (Lintermans 2007). Although the species remains common within the rivers of coastal Queensland and NSW, the MDB population has declined and is now patchily distributed or absent. In the NSW section of the northern Basin, there have been recent captures from Esk River drainage in the NE of the Clarence Catchment (M. Birch, personal communication, 2020) and a heavily restricted (but locally abundant), population exists in the Bogan River Catchment. In the Border Rivers Catchment, the species is widespread in the Dumarseq River and adjacent Macintyre River below the Severn River-Macintyre River junction, and is occasionally found downstream as far as Goondiwindi. This population expands and disperses downstream along the Barwon-Darling Rivers during floods (as far as below Wilcannia), but does not appear to persist. In the Gwydir River, the species was first detected in the mid-2010s from two wetlands, which subsequently dried and it has not been sampled since. It was considered extirpated from the southern MDB (e.g. NSW (last recorded in 1970); SA (last recorded in 1983); and Victoria (last recorded in 1922)) until it was rediscovered in large numbers (almost 5000 fish) from the Lachlan River Catchment in 2007 (McNeil et al. 2008). The rediscovered population in the Lachlan Catchment is restricted to the weirpool upstream of Brewster Weir and a short distance downstream of Brewster Weir, as well as reaches within the lower end of Mountain Creek (the outlet channel for Lake Brewster). It has been reintroduced to the Cargelligo weirpool (upstream of Brewster in the Lachlan) as well as the Thegoa Lagoon near Wentworth (700 captive bred fish in May 2011) but neither population has established (D. Gilligan, personal communication, 2020).

The overall estimated EOO for the species is 1,705,786 km^2, with relatively stable population trend, yet the MDB populations have been deemed to be continuing to decline (Raadik and Unmack 2019).

2.2.3 Biological information

Olive Perchlet are small bodied (up to 76 mm TL but more often to 50 mm TL), oval, laterally compressed, olive to semitransparent, with brown margins on the scales (Lintermans 2007). Individuals have a proportionately large mouth and eyes, a single prominent dorsal fin and a forked tail. Olive Perchlet often reside in large schools inhabiting shallow, low flow areas of vegetated creeks, wetlands, swamps and rivers (Allen and Burgess 1990). During habitat preference trials, Olive Perchlet preferred structure over open sandy habitat, showed a significant preference for submerged macrophyte over sand, preferred submerged macrophytes over emergent plants and used sand and emergent plant habitats in similar proportions (Hutchison et al. 2020).

In the wild, both sexes reach sexual maturity at approximately one year of age and live between two to four years, with females tending to live longer than males. Olive perchlet commence spawning when water temperatures reach between 22–23°C in spring/early summer (Lintermans 2007; McNeil et al. 2008; Milton and Arthington 1985), with the maturation of gonads of both male and females beginning in September and ripe fish found in wild populations in October and November (Milton and Arthington 1985). Fecundity is usually 200–700 eggs, but can be as high as 9966 eggs under captive breeding conditions (Llewellyn 2008). Eggs are small (0.7 mm diameter), and adhesive which allows them to attach to aquatic plants and rocks on the stream bed. Hatching times range from one day at temperatures between 20-29°C (Llewellyn 2008) and five to seven days at 22°C, and larvae are approximately 3 mm long at hatching (Pusey et al. 2004). The species can have multiple spawnings over two to possibly three years (Llewellyn 2008; Milton and Arthington 1985).

2.2.4 Genetic management

Significant genetic differentiation is evident across the present range of the species, indicating four main groupings: southern coastal QLD, northern coastal NSW, southern MDB (Lachlan catchment) and northern MDB (Burnett, Warrego, Condamine, Macintyre, Gwydir and Bogan; P. Unmack, unpublished data). At this stage, managing the species separately across the distinct groupings is recommended but, consistent with other target species, evaluation of the validity of assisted gene flow is required.

2.2.5 Known threats and knowledge gaps

Although no individual threat has been attributed to the ongoing decline of the species, potential threats include impacts from alien fish (including predation by Redfin Perch *Perca fluviatilis*, egg predation and resource competition with Eastern Gambusia *Gambusia holbrooki* and competition and habitat alteration by Common Carp *Cyprinus carpio)*, spawning and recruitment restrictions and habitat loss and degradation caused by cold water pollution and river regulation (Lintermans 2007). There is limited ecological knowledge of the species (Table 3-2). There are records of migration through tidal barrage fishways in coastal streams but there is limited knowledge of movement patterns of the MDB populations of the species (Lintermans 2007). Building on the summary of Koehn et al. (2017), Koehn et al. (2020b) summarises the amount of available knowledge that exists for the species (Table 2-2), indicating that typically less than 39% of knowledge that is needed is presently available (and in many cases <19%).

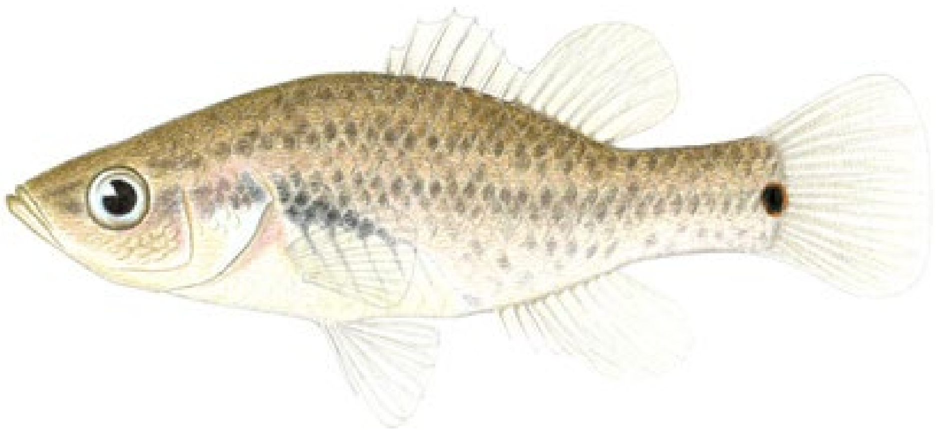
Table 2-2. Status of available knowledge for life stages of Olive Perchlet (available knowledge was scored as follows: 1: 0–19% of knowledge needed is available; 2: 20–39% of knowledge needed is available; 3: 40–59% of knowledge needed is available; 4: 60–79% of knowledge needed is available; 5: 80% of knowledge needed is available): adapted from Koehn et al. (2020b).

Spawning	Eggs	Larvae	Juveniles	Adults
Spawning	2.0			2.0
Spawning conditions				2.0
Survival (recruitment)	1.0	1.0	1.0	1.5
Growth and condition		1.0	1.0	1.0
Movements		1.0	1.0	1.0
Physical habitat requirements	3.0	1.0	2.0	3.0
Water quality tolerances	1.0	1.0	1.0	1.0
Flows requirements	2.0	1.0	2.0	2.0

2.2.6 Overall summary

The MDB population of Olive Perchlet persists as several distinct lineages with populations within the lineages. There is an urgent need to address knowledge gaps and known threats whilst securing known populations and reestablishing populations within the distinct lineages throughout their known historical range.

2.3 Oxleyan Pygmy Perch *Nannoperca oxleyana*



(NSW DPI Fisheries)

2.3.1 Conservation status

International: *Endangered*
National: *Endangered*
NSW: *Endangered*
Rest of range: *Vulnerable (QLD)*

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales*: Fisheries Management Act 1994*; QLD: *Nature Conservation Act 1992*.

2.3.2 Population status

Oxleyan Pygmy Perch is the most northerly distributed species of *Nannoperca* and is endemic to low-lying coastal plains of southern Queensland and northern NSW. On the mainland remnant populations persist from Tin Can Bay, north of Noosa, to the Richmond River and there have been recent captures from the Esk River (NE Clarence Catchment) within northern NSW. The species also occurs on Fraser, Moreton and Stradbroke islands (Knight 2016; Knight and Arthington 2008; Knight et al. 2012). Surveys since 2000 have resulted in the capture of the species from a total of 86 waterbodies, including lakes, swamps, creeks and smaller tributary streams within 67 permanently connected, unfragmented (i.e. contiguous) drainage systems in NSW (Knight 2016; Knight et al. 2012). Ranges in the Tabbimoble Swamp area and Broadwater National Park area were both reduced after drought and fire in 2019/2020. Their preferred habitats were particularly susceptible to drought conditions as they are mostly found in shallow depressions over sandy soils and much of the available habitat is dry in years of drought (M. Birch, personal communication, 2020). The present AOO of the species was recently predicted to be 292 km^2, within which populations are considered to be severely fragmented and in continued decline (Butler et al. (2019).

2.3.3 Biological information

Oxleyan Pygmy Perch is the most northerly distributed species of *Nannoperca* and is endemic to low-lying coastal plains of southern Queensland and northern NSW. On the mainland remnant populations persist from Tin Can Bay, north of Noosa, to the Richmond River and there have been recent captures from the Esk River (NE Clarence Catchment) within northern NSW. The species also occurs on Fraser, Moreton and Stradbroke islands (Knight 2016; Knight and Arthington 2008; Knight et al. 2012). Surveys since 2000 have resulted in the capture of the species from a total of 86 waterbodies, including lakes, swamps, creeks and smaller tributary streams within 67 permanently connected, unfragmented (i.e. contiguous) drainage systems in NSW (Knight 2016; Knight et al. 2012). Ranges in the Tabbimoble Swamp area and Broadwater National Park area were both reduced after drought and fire in 2019/2020. Their preferred habitats were particularly susceptible to drought conditions as they are mostly found in shallow depressions over sandy soils and much of the available habitat is dry in years of drought (M. Birch, personal communication, 2020). The present AOO of the species was recently predicted to be 292 km^2, within which populations are considered to be severely fragmented and in continued decline (Butler et al. (2019).

Oxleyan Pygmy Perch is a small freshwater fish, growing to a maximum of 60 mm TL (Knight et al. 2012). Individuals are characterised by a moderately laterally compressed body, one deeply notched dorsal fin and a truncate caudal fin. Further distinguishing features include the absence of a lateral line, a small mouth reaching to just below the eye and enlarged teeth in its lower jaw. The body is covered in ctenoid scales and is light brown to olive in colour, darker on the back, with a conspicuous round black spot with an orange margin at the base of the caudal fin. The fins of both sexes are mainly clear, except during the breeding season when the fins and body of males become intense red and brown and their pelvic fins become jet black (Knight et al. 2007).

Oxleyan Pygmy Perch inhabit slightly acidic and tannin-stained water in slow flowing pools and backwaters in coastal streams, river channels, lakes and swampy drainages (DPI 2015). Coastal waterways inhabited by Oxleyan Pygmy Perch are characterized by slightly acidic and tannin-stained water with gentle flow with water velocities generally below 0.4 m/sec, often over sandy soil types (Pusey et al. 2004). Preferred microhabitat include areas where there is a high abundance of structure, such as emergent plants including Jointed Rush, Grey Rush, Zig Zag Rush, Maundia and Water Ribbon and submerged plants including Bladderwort and Sphagnum moss plants (M. Birch, personal communication, 2020), or steep undercut banks, fringed by semi-submerged branches and fine rootlets from adjacent land-based trees and scrubs (Knight and Arthington 2008).

Its diet largely consisting of insects and their larvae. Oxleyan Pygmy Perch prefer fresh habitats with conductivity below 830 EC and an acidic pH range of 3.3−6.9, they have also been shown to prefer well oxygenated water with a mean dissolved oxygen concentration of 6.42 mgL^{-1} (Knight et al. 2012), however have frequently been captured at sites with very low dissolved oxygen concentrations around 2 mgL^{-1} (M. Birch, personal communication, 2020). They are mostly found at water depths of approximately 50 cm but have been collected from depths of up to 130 cm (Pusey et al. 2004).

The protracted breeding season of Oxleyan Pygmy Perch occurs September to May when water temperatures increase beyond 16.6°C (Knight and Arthington 2008) with females producing an average of 587 eggs, releasing approximately eight eggs per day throughout the season (Knight and Arthington 2008). Male Oxleyan Pygmy Perch exhibit territorial behavior during the breeding season to defend their nesting site (Knight et al. 2007).

2.3.4 Genetic management

Oxleyan Pygmy Perch is the most northerly distributed species of *Nannoperca* and is endemic to low-lying coastal plains of southern Queensland and northern NSW. On the mainland remnant populations persist from Tin Can Bay, north of Noosa, to the Richmond River and there have been recent captures from the Esk River (NE Clarence Catchment) within northern NSW. The species also occurs on Fraser, Moreton and Stradbroke islands (Knight 2016; Knight and Arthington 2008; Knight et al. 2012). Surveys since 2000 have resulted in the capture of the species from a total of 86 waterbodies, including lakes, swamps, creeks and smaller tributary streams within 67 permanently connected, unfragmented (i.e. contiguous) drainage systems in NSW (Knight 2016; Knight et al. 2012). Ranges in the Tabbimoble Swamp area and Broadwater National Park area were both reduced after drought and fire in 2019/2020. Their preferred habitats were particularly susceptible to drought conditions as they are mostly found in shallow depressions over sandy soils and much of the available habitat is dry in years of drought (M. Birch, personal communication, 2020). The present AOO of the species was recently predicted to be 292 km^2, within which populations are considered to be severely fragmented and in continued decline (Butler et al. (2019).

Genetic analyses has identified distinct genetic structuring consistent with the isolation of populations over an extended period (Knight et al. 2009). Genetic diversity is moderate across populations, within populations in the South Evans Head subcatchment and Marcus Creek being more genetically diverse than other populations. Knight et al. (2009) conclude that a genetic rescue to strategically mix populations is warranted but must be carefully managed. This is consistent with the genetic management strategy of this present report.

2.3.5 Known threats and knowledge gaps

Due to the preference of shallow waters, Oxleyan Pygmy Perch are under threat from drought conditions where their shallow sandy habitats dry out and can lead to extreme population fluctuations. The species is not thought to be able to adopt specific strategies for surviving drought conditions (such as aestivation) (Knight 2000). Oxleyan Pygmy Perch are exposed to many other threats across its present range (DPI 2015), primarily, anthropogenic activities such as land clearing for urbanisation, agriculture, forestry and mining that has fragmented key habitats throughout their range. Other threats include collection of fish for aquaria and competition from alien fish species, particularly Eastern Gambusia.

2.3.6 Overall summary

Oxleyan Pygmy Perch are a small bodied species of percichthyid found throughout coastal Wallum Heath habitats of south eastern Queensland and northern NSW. The species has specific habitat requirements within its range, much of which has been severely impacted by drought conditions and fragmented due to human actions. Generally, there is limited knowledge of its present status.

2.4 River Blackfish (Snowy River population) *Gadopsis marmoratus*

(NSW DPI Fisheries)

River Blackfish represent a complex of five undescribed but recognised candidate species (Hammer et al. 2014; Unmack et al. 2017). This section (and section 3) focuses on the Snowy River population of River Blackfish, which is among the eastern sub-lineage of the South East Victoria (SEV) candidate species but draws on knowledge across the species complex.

2.4.1 Conservation status

International: *Least Concern*
NSW: *Endangered population* (Snowy River population)
Rest of range: *Endangered* & *Protected* (SA); not listed (QLD), not listed (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales: *Fisheries Management Act 1994*; South Australia: Action Plan for South Australian Freshwater Fishes 2009 *and Fisheries Management Act 2007*; Victoria: *Flora and Fauna Guarantee Act 1988*; QLD: *Nature Conservation Act 1992*.

2.4.2 Population status

THE SEV candidate species of River Blackfish is known from the East Gippsland, Snowy, Tambo River, Mitchell, Thomson and La Trobe River basins in Victoria. In NSW it is only known from the Snowy River Catchment. In NSW, the Snowy River population of River Blackfish (hereby referred to as River Blackfish) was historically abundant and widely distributed across the mid and upper reaches of the catchment. This population has declined substantially in abundance and distribution, now confined to the Delegate River to the upper reaches of its catchment, being recorded at only 11 sites across a 50 km area.

2.4.3 Biological information

The SEV candidate species of River Blackfish grows to 330 mm TL. As with all River Blackfish, the Snowy River population is characterised by a slightly mottled appearance and long slender body. Colouration varies among and within candidate species, from golden, green or brown to almost black. River Blackfish have a large mouth, the upper jaw longer than the lower, and a dorsal fin that extents almost the length of the body; they are also characterised by the pelvic fins being reduced to a single branched ray emerging from underneath the throat. Although capable of larger movements, River Blackfish are generally a non-migratory fish with a small home range. Much of their movements and foraging are carried out within nocturnal hours (Koster and Crook 2008).

River Blackfish within the Snowy River favour clear flowing streams where it feeds on insects, crustaceans, molluscs and small fish. In the wild, River Blackfish are reported to breed from spring to early summer when temperatures exceed 16°C. The species exhibits low fecundity and eggs are deposited on submerged structures such as logs or rock before being cared for by the male (Jackson 1978). Egg hatching time is temperature dependent; varying from 14−16 days at water temperatures between 12 to 20°C (Jackson 1978; McDowall 1996) and eggs are approximately 7 mm when they hatch. At around 26 days and at a size of 15 mm, the yolk-sac is almost completely absorbed, and juveniles commence active foraging (Jackson 1978).

2.4.4 Genetic management

No population genetic analyses of the Snowy River population of River Blackfish have been conducted and it is managed as a single conservation unit.

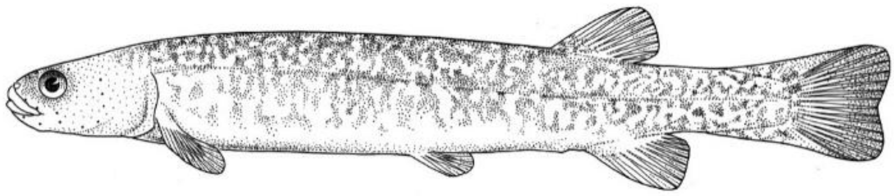
2.4.5 Known threats and knowledge gaps

There are many threats to the River Blackfish within the Snowy River Catchment, including increased habitat degradation from land clearing, erosion, unseasonal shifts in water temperature due to water releases from large impoundments and increased siltation. Alien species such as trout and Redfin Perch may also provide competition for resources and predate on young River Blackfish. Climate change may also present a threat in the future, influencing changes within the species' range including increases in bushfire, drought and flooding events which in turn can lead to rising water temperatures, destruction of important habitats (such as large woody debris) and drying of waterways.

2.4.6 Overall summary

Historically abundant, River Blackfish within the Snowy River catchment have drastically reduced in abundance and spatial distribution and are now largely confined to the Delegate River in the upper reaches of the Snowy River catchment. Within the Snowy River, the River Blackfish is exposed to an array of threats including habitat degradation and competition from alien species.

2.5 Roundsnout Galaxias *Galaxias terenasus*



(Rhyll Plant and Raadik (2014))

2.5.1 Conservation status

International: *Endangered*
National: *not listed*
NSW: *not listed*
Rest of range: *Threatened* (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999;* New South Wales: *Fisheries Management Act 1994*; Victoria: *Flora and Fauna Guarantee Act 1988*.

2.5.2 Population status

Roundsnout Galaxias occupies a restricted range in southern NSW and East Gippsland in Victoria (Raadik 2014; Raadik 2019). In NSW, it is restricted to the Bombala and Delegate sub-catchments of the Snowy River Catchment and the Genoa River. In Victoria, it is restricted to the Genoa River just downstream of the Victoria/NSW border, and in the mid to upper Cann River Catchment. Its present estimated EOO (2112km^2) and AOO (92km^2) are considered to be continuing to decline (Raadik 2019). Very recently, species level genetic analysis has demonstrated deeper divergence between the Cann River population and the other two, indicating the Cann River population can be considered a separate candidate species (T.A. Raadik, unpublished data).

2.5.3 Biological information

Roundsnout Galaxias can grow to 70 mm fork length (FL), but is typically 45–55 mm FL (Raadik 2014). It is characterised by having a largely olive-brown body with brownish, irregularly shaped blotches that extend onto the head and snout. The blotches and bands are sometimes obscured by fine dark stippling. The fins and gill covers are translucent and have a golden patch.

As described by Raadik (2014), Roundsnout Galaxias differs from the other *Galaxias* species by a combination of the following characters: diminutive size; long anterior nostrils, usually visible anterio-laterally from ventral view; distinctive body colour pattern and thin fins; low mean total pectoral fin segmented ray count of 13; low mean vertebral count of 51; dorsal and ventral trunk profiles straight or nearly so; lateral snout profile usually rounded; body depth shallow through pectoral fin base (11.2–14.3% SL); dorsal midline usually distinctly flattened anteriorly from dorsal fin base; mouth small, usually reaching back to anterior margin of eye with posterior extent of mouth about 0.4 ED below ventral margin of eye; head and inter-orbital narrow (49.8–64.0% and 31.4–40.5% HL respectively) but head length greater than PelAn distance; eye large (17.5–27.7% HL and 45.7–73.3% HD); gape narrow (26.4–34.4% HL and 48.7–64.2% HW); snout, upper and lower jaws short (17.6–29.8%, 24.4–29.9% and 21.2–29.2% HL respectively); lower jaw about 95 (82.9–100%) length of upper; caudal peduncle moderately long and longer than length of caudal fin; caudal peduncle flanges moderately developed but short, usually not reaching to adpressed anal fin; dorsal fin base short (7.1–11.6% SL); distance between pelvic and anal fins short (17.8–24.6% SL); pelvic fin very short (6.3–11.2% SL), only about 74.4% of length of pectoral fin; raised lamellae absent from ventral surface of rays of paired fins; accessory lateral line absent; anal fin origin usually under 0.42 distance posteriorly along dorsal fin base; 2 thin to moderately thick and long (5.1% SL) pyloric caecae; gill rakers short and stout; and, lack of distinct black bars along lateral line (Raadik 2014).

Roundsnout Galaxias reaches sexual maturity at about 30–35 mm and although the spawning period for the species is not confirmed, it is suggested to be sometime during spring to early summer months (Raadik 2014). Gravid females usually have stippling along the body between the pectoral fin base and vent, sometimes extending almost to the mid-lateral region. The fecundity of the species is considered low as gravid females have been shown to produce 220–240 eggs (Raadik 2014). Roundsnout Galaxias has been recorded in variable sized rivers and streams with clear water, slow to medium rates of flow with a substrate primarily consisting of bedrock, boulder, cobble and coarse sand. Instream structure at sites where the species has been recorded is rock, timber snags with little aquatic vegetation. Vegetation surrounding sites where this species occurs has varied between heavily forested and almost completely cleared for grazing (Raadik 2014).

2.5.4 Genetic management

Until recently, the species was thought to be part of the Mountain Galaxias *Galaxias olidus* species complex before its description by Raadik (2014). No population genetic analysis has been undertaken, but Raadik (2014) found one fixed allozymic difference between the Genoa and Snowy populations when undertaking taxonomic analysis. Consequently, the species should be managed as two conservation units.

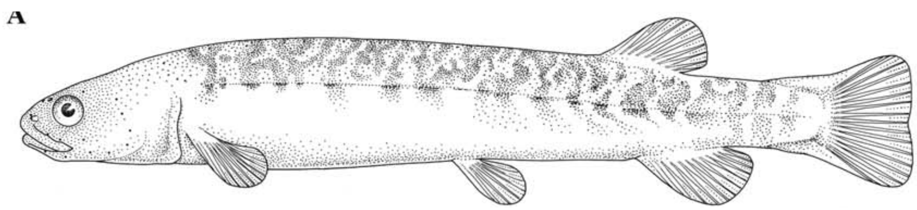
2.5.5 Known threats and knowledge gaps

Increased sedimentation from agriculture and forestry may lead to habitat degradation within this species' range. Climate change presents a long-term threat influencing changes within the species' range including increases in bushfire, drought and flooding events which in turn can lead to rising water temperatures, drying of waterways, destruction of important habitats (e.g. instream structures), transportation of predatory alien species and a host of other challenging changes to the species' surrounding environment. Although predation from alien trout species may be a threat to Roundsnout Galaxias, the species has been observed to be able to survive amongst populations of alien trout species, possibly by occupying habitats not used by trout (Raadik 2014).

2.5.6 Overall summary

This newly described species has been assessed as Endangered globally (but not nationally at present) and many knowledge gaps exist. These must be addressed as the Roundsnout Galaxias is susceptible to sedimentation and extreme weather events such as drought and bushfires.

2.6 Short-tail Galaxias *Galaxias brevissimus*



(Rhyll Plant and Raadik (2014))

2.6.1 Conservation status

International: *Critically Endangered*
National: *not listed*
NSW: *not listed*
Rest of range: *not listed* (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999;* New South Wales: *Fisheries Management Act 1994*; Victoria: *Flora and Fauna Guarantee Act 1988*.

2.6.2 Population status

Short-tail Galaxias occur across a very restricted range, being known from only three creeks (Guinea, Jibolaro and Bumberry creeks) in the upper reaches of the Tuross River Catchment in southern NSW. There are two subpopulations (Guinea and Jibolaro creeks, and Bumberry Creek, separated by ~10 km of trout-infested Tuross River. The estimated AOO is 16 km^2 and the EOO is 22 km^2 with it predicted to have experienced a 53% population decline over the last 10 years (Lintermans and Raadik 2019). Lintermans et al. (2020) assessed Short-tail Galaxias as having a >50% probability of extinction in the next ~20 years.

2.6.3 Biological information

Short-tail Galaxias has a maximum recorded length of 95 mm FL, but commonly 70–75 mm FL (Raadik 2014). It has a moderately elongate body, which is brown on the upper sides and back and extends to the top and sides of snout and head, with lighter brown on the lower lateral trunk sides and is overlain by dark brown to black spots and blotches that are small to moderate and irregularly shaped (Raadik 2014).

As per the summary by Raadik (2014), Short-tail Galaxias can be distinguished from the other *Galaxias* species by a combination of the following characters: short caudal peduncle (10.3–12.0% SL) and caudal fin length (10.1–12.2% SL); anal fin and pelvic fins set far back at about 76 and 53% SL respectively; anal and dorsal fin lengths short and dorsal fin base short (8.0–9.6% SL); small pectoral fin (9.6–12.0% SL); dorsal midline of trunk usually flattened anteriorly from above midpoint between pectoral and pelvic fin bases; head quite narrow (55.9–59.6% HL) and eye relatively large (18.3–21.0% HL); nostrils moderately long, not visible from ventral view; gape about as wide as length of lower jaw; often a single, sometimes two, unbranched, segmented rays in the dorsal fin (versus usually 2); low mean number of vertebrae (52); raised lamellae on the ventral surface of paired fins appear to be absent; caudal peduncle flanges relatively short, occasionally just reaching adpressed anal fin; single, moderately short (1.7% SL) and thin pyloric caecum; anal fin origin usually under 0.8 distance posteriorly along dorsal fin base; gill rakers sharply pointed; and, lack of black bars along lateral line (Raadik 2014).

Known populations occur within narrow, shallow and clear water flowing through pool and riffle habitats. The Jibolaro and Guinea creeks sub population occurs over a substrate of clay and sand with some areas of silt (Raadik 2014) and the Bumberry Creek sub population is in a rocky-bottomed stream (M. Lintermans and T.A. Raadik, unpublished data). The Jibolaro and Guinea creeks are largely cleared of riparian vegetation and replaced with agricultural pasture, leaving much of their habitat unshaded. The Bumberry Creek sub population occurs on forested catchment as part of Wadbilliga National Park and is heavily shaded. Although not certain, it is likely Short-tail Galaxias breed during the late winter-early spring months (Raadik 2014) with ripe males and gravid females observed in late July (M. Lintermans, unpublished data).

2.6.4 Genetic management

Until recently, the species was thought to be part of the Mountain Galaxias species complex before its description by Raadik (2014). No population genetic analysis has been undertaken, so the species is managed as a single conservation unit.

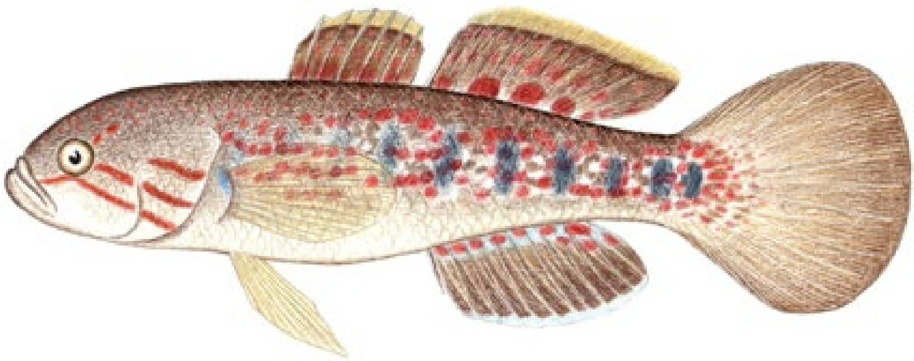
2.6.5 Known threats and knowledge gaps

The geographic range of the species has been impacted over time by alien Brown Trout (*Salmo trutta*) and Rainbow Trout (*Oncorhynchus mykiss*), which have likely reduced and fragmented the range of Short-tail Galaxias (Raadik 2014). Habitat degradation through removal of riparian vegetation through agricultural grazing is a current threat within the species' range. The increased frequency and ferocity of bushfires due to climate change may further erode the quality of riparian vegetation as well as sedimentation and reduce water quality due to post-bushfire run off. The Jibolaro and Guinea creek sub population was significantly affected by drought with the stream having ceased to flow and being reduced to isolated pools at the end of 2019 (M. Lintermans, unpublished data).

2.6.6 Overall summary

This newly described species has been assessed as Critically Endangered globally (but not nationally at present) and many knowledge gaps exist. It occurs across a heavily restricted range making it susceptible to disturbance. There is a need to list the species as threatened in NSW to ensure the immediate protection of the species. Further research is also required to increase understanding of the species to inform conservation and management.

2.7 Southern Purple-spotted Gudgeon *Mogurnda adspersa*



(NSW DPI Fisheries)

2.7.1 Conservation status

International: *Least Concern*
National: *not listed*
NSW: *Endangered*
Rest of range: *not listed (QLD): Critically Endangered & Protected* (SA); *Threatened* (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales: *Fisheries Management Act 1994*; QLD: *Nature Conservation Act 1992*; South Australia: Action Plan for South Australian Freshwater Fishes 2009 and *Fisheries Management Act 2007*; Victoria: *Flora and Fauna Guarantee Act 1988*.

2.7.2 Population status

Historically, Southern Purple-spotted Gudgeon was broadly distributed across coastal areas of Queensland and northern NSW as well as patchily occurring in the MDB. In the southern MDB, it was once widespread and common in wetland and fringing river habitats. Specifically, it was known from the Murrumbidgee and Murray (and possibly Lachlan) catchments, including the Lower Murray (Cardross Lakes and SA section). It persists in coastal QLD and coastal NSW (although only one remnant population is known), but has experienced declines across the MDB and was considered extirpated from the southern MDB: SA (last recorded in

1973: Hammer et al. 2009b); NSW (last recorded in 1968: Llewellyn 2006); and Victoria (last recorded in 1990s), before chance rediscoveries in SA and Victoria. In SA, it was detected from a single Lower Murray wetland in 2002, which has been maintained through regular reintroductions as the site dried soon after the rediscovery (Hammer et al. 2015; Whiterod 2019). In Victoria, there was short-lived redetection during the 1990s in Cardross Lakes (Raadik et al. 1999) and in late 2019 it was rediscovered from Third Reedy Lake (Kerang Lakes) (Iervasi 2019) with subsequent surveys confirming a small population (Stoessel 2020).

The northern MDB includes remnant populations in Wuuluman Creek and various tributaries of the Little River in the Macquarie catchment, populations (perhaps connected) in Halls Creek and Keera Creek sub-catchments (with individuals rarely collected from the adjacent Gwydir River channel) near Bingara, and in the upper reaches of Tycannah Creek in the Gwydir Valley, and populations in the Tenterfield Creek and Mole River sub-catchments (and in the interconnected Dumaresq River) in the Border Rivers catchment. Populations were also present near Dundas in the upper Severn River and in the upper Macintyre River around Inverell two decades ago, however none have been detected there recently. There is a possibility that some populations still exist in some of these (as well as in the Beardy River) (D. Gilligan, personal communication, 2020).

Southern Purple-spotted Gudgeon were presumed to be extinct on the NSW coast with the most-recent record from the 1970s. However, in 2012, a Charles Sturt University undergrad student (Matt Miles) recorded the species in Tucki Tucki Creek at Goonellabah (a suburb of Lismore). They were also discovered in Goonrangoola Creek (a tributary of the Hunter River) in 2009 which was outside their previous known range. Genetic work undertaken by Peter Unmack to determine if it was likely they were endemic or a new translocated population showed that the Goonrangoola Creek population mtDNA matched that of the central Queensland origin exactly, but their nuclear DNA had some unique alleles. It has been suggested that this population (Goonrangoola Creek) may have been introduced, whilst the

Tucki Tucki Creek population is endemic to coastal NSW (D. Gilligan, personal communication, 2020).

In NSW, approximately 1,250 captively bred fish were released at two sites in Adjungbilly Creek (Murrumbidgee catchment) over a period of 2 years (2004 and 2005) However, only a single fish was recaptured several weeks following release and no fish were found during later post-release surveys, suggesting a population at these sites was not established (D. Gilligan, personal communication, 2020). In 2006, captive bred fish (n=153) were released into a managed rehabilitated wetland off Goobang Creek near Condobolin (Lachlan catchment). However, no fish were recaptured during post-release surveys and a population was not established (D. Gilligan, personal communication, 2020).

Captive-bred fish have also been released into several locations in the northern MDB in NSW. In August 2007, 101 captive bred individuals were released into waterways within the grounds of Western Plains Zoo (Dubbo, NSW). The zoo reported recapturing large adult fish in May 2008. In October 2012, 16 remaining broodfish were released at the same site. However, no further reports from the zoo were received and it was presumed that the fish died out. In February 2008, approximately 2000 captive bred fish were released at two sites in the headwaters of the Castlereagh River. The Castlereagh headwaters population did establish, as evidenced by the capture of several sub-adults five and six years after the initial release. However, the current existence of this population is unknown. In October 2009, 116 captive bred fish were released into a rehabilitated managed wetland (Gulligal Lagoon) off the mid-Namoi River near Boggabri. It is unknown whether this population established (D. Gilligan, personal communication, 2020).

2.7.3 Biological information

Southern Purple-spotted Gudgeon reach a maximum of 150 mm TL, but more typically attain between 60 and 120 mm TL. It has a rounded head, small mouth, rounded tail and two dorsal fins (Lintermans 2007). The species has several distinguishing markings; a row of darkish blotches present on the sides from the start of the second dorsal fin to the start of the caudal fin, surrounded by numerous red and white spots and, at times, a series of iridescent blue blotches toward the tail and brown to purple facial strips (3–4 in males; two in females).

Throughout NSW, Southern Purple-spotted Gudgeon have a strong preference for very small tributary streams and are generally found in perennial spring fed streams. This habitat preference also applies to Queensland (MDB and coastal populations). In SA and Victoria, Southern Purple-spotted Gudgeon is a benthic and sedentary wetland specialist with a strong preference for dense physical (woody structure and rocks) and aquatic vegetation cover (Hammer et al. 2015; Lintermans 2007). It is found in small streams, rainforest streams, large rivers and dune lake systems, as well as slow-flowing weedy pools (Pusey et al. 2004). Maxent species-distribution modelling indicates that Southern Purple-spotted Gudgeon in the northern MDB have a strong preference for small spring-fed streams, however these habitats have to date not been highly targeted (D. Gilligan, personal communication, 2020).

The species spawns in summer with the timing of spawning likely dependent on water temperature (i.e. above 20°C), day length, abundance of food and availability of spawning sites (Hansen 1988). Females may produce 7–10 spawnings in one season, where clusters of eggs are deposited on firm substrates and guarded by the male. Fecundity is size dependent and varies across the range of the species (Pusey et al. 2004).

2.7.4 Genetic management

The MDB population is one of three genetic lineages identified in the species (Sasaki et al. 2016). Within this lineage, the southern MDB populations are genetically distinct from those of the northern MDB (Hammer et al. 2015; Sasaki et al. 2016). As such, the southern MDB subpopulation of the species is considered a separate conservation unit.

2.7.5 Known threats and knowledge gaps

The species has declined due to intensive flow regulation and diversions resulting in habitat alteration and loss, as well as predation and competition with alien species. Recently, the species has been significantly threatened by the 2018–19 drought and 2019–20 bushfires. Knowledge gaps exist regarding both biology and ecology (see Table 2-3 which provides a summary of knowledge status of this species in the MDB (Koehn et al. 2017).

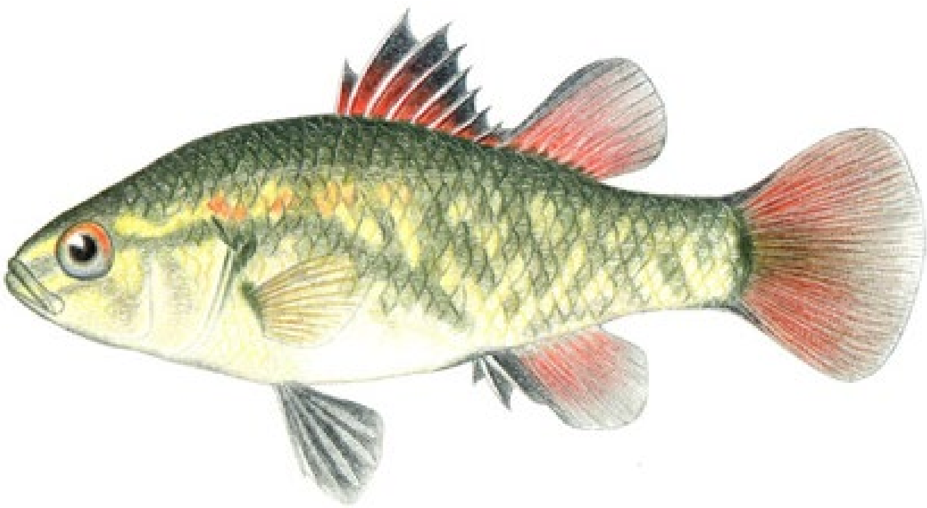
Table 2-3. Status of knowledge of the biology and ecology for life stages of Southern Purple-spotted Gudgeon (low (1–3); moderate (4–7), and high (7–10) knowledge: adapted from Koehn et al. (2017)).

Spawning	Eggs	Larvae	Juveniles	Adults
Moderate	Moderate	Low	Low	Moderate

2.7.6 Overall summary

The future of the species remains precarious throughout the MDB. Further information is required about current distribution and population status throughout NSW. In the Lower Murray, it persists but has yet to re-establish a self-sustaining population. In NSW, reintroductions have not been successful within the southern MDB, however there has been one successful establishment of an additional population within the Castlereagh River in the northern MDB. On a more positive note, captive/backup populations are maintained for the species. A captive population has been established at Taronga Western Plains Zoo (Dubbo, NSW) using drought-rescued broodstock, and recent monitoring has demonstrated breeding and recruitment within this population. Identification of priority sites for re-introducing the species that consider localised threats and the availability of environmental water, are required to re-establish a network of populations across the MDB.

2.8 Southern Pygmy Perch *Nannoperca australis*



(NSW DPI Fisheries)

2.8.1 Conservation status

International: *not listed*
National: *Vulnerable* (*MDB lineage*)
NSW: *Endangered*
Rest of range: *Critically Endangered & Protected* (SA); *Threatened* (Vic)

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales: *Fisheries Management Act 1994*; QLD: *Nature Conservation Act 1992*; South Australia: Action Plan for South Australian Freshwater Fishes 2009 and *Fisheries Management Act 2007*; Victoria: *Flora and Fauna Guarantee Act 1988*.

2.8.2 Population status

Southern Pygmy Perch historically occur in the coastal catchments of south-east SA and southern Victoria, the South Australian Gulf, the north of Tasmania, King and Flinders Islands and the MDB. In the MDB, it was historically widespread across the southern Basin, including the headwaters of Lachlan Catchment as well as the Murrumbidgee and Murray catchments. In the Murray, it occurred in headwater streams, the mid-Murray and tributaries (Broken, Ovens, Goulburn and Kiewa rivers) and the lower Murray River (including the Lower Lakes and Mt Lofty tributaries). The species has experienced significant range reductions since European settlement, associated with the degradation and loss of wetland habitat and the impact of alien species (Lintermans 2007). The species remained widely distributed but persisted as fragmented subpopulations. These subpopulations were further impacted by the Millennium Drought, with local extirpation occurring from mid- and headwater Murray River sites (including Barmah-Millewa, Normans Lagoon, Happy Valley Creek, Tallangatta Creek, Khancoban Lagoon, Oolong Creek and likely the lower Ovens River floodplain) as well as sites in Mount Lofty Ranges and Lake Alexandrina (and Hindmarsh Island). At this time, fish from Lake Alexandrina and surrounding areas (Turvey's Drain and Mundoo Island) were rescued to establish backup populations (initially captive maintenance and breeding facility) (Cole et al. 2016; Hammer 2008).

Over the past 10 years, the species has continued to decline across the MDB. In NSW, current populations include: a remnant population in Blakney Creek (Lachlan catchment: discovered 2002) (Lintermans and Osborne 2002) that has experienced recent range retraction, Billabong Creek headwaters (e.g. Mountain Creek sub-catchment - experienced large recent range retraction), Coppabella Creek (stable population) and a reintroduced population in Pudman Creek (established, but only just persisting) (Pearce 2015; Pearce et al. 2018).

Reintroductions were attempted in Thegoa Lagoon (near Wentworth) and Washpen Creek (near Euston) in May 2011 (4500 captive-bred fish to each) but despite short-term success (e.g. 2 individuals recaptured 20 days after release in Washpen Creek), the species did not establish. In the Victorian MDB, it persists in Middle Creek and Mountain Creek as well as the Avoca River and Campapse River catchments (Rose 2018). In SA, despite declines, there are locally strong subpopulations in the Lake Alexandrina and tributary streams of the Mount Lofty Ranges (Whiterod et al. 2019).

2.8.3 Biological information

Southern Pygmy Perch are a small freshwater perch attaining a maximum size of ~85 mm TL. The species has a slightly rounded head, a small mouth that extends to just in front of eye and a rounded tail (Lintermans 2007). The body colour is cream to gold to greenish-brown. These features, along with a round pupil, distinguish the species from the Yarra Pygmy Perch, with which it is often confused. Additionally, male Southern Pygmy Perch develop bright red fins during spawning, whereas the fins of a breeding male Yarra Pygmy Perch are black. Southern Pygmy Perch generally occurs in still and slow-flowing water, with abundant aquatic vegetation cover; it is rarely found in fast-flowing sections of streams.

2.8.4 Genetic management

Although once considered to historically form one contiguous meta-population across the southern MDB (particularly Murray Catchment), the species has now contracted to 14 genetically distinct subpopulations (Cole et al. 2016; Hammer 2008). These are (1) Angas River; (2) Finniss River; (3) Lake Alexandrina and surrounds: lower reaches of Tookayerta Creek, Turvey's Drain and Mundoo and Hindmarsh islands; (4) mid- to upper-reaches of Tookayerta Creek; (5) Avoca River; (6) Goulburn and Broken rivers; (7) upper Broken River; (8) Campapse River; (9) Upper Murray (Norman Lagoon); (10) Coppabella Creek; (11) Kiewa River; (12) Ovens River; (13) Mitta Mitta River; and (14) Lachlan River. These genetic distinctions have been used to define conservation units, but in recognition of the negative impacts of fragmentation, a strategy of translocation (including genetic rescue) is recommended to enhance genetic diversity (Brauer and Beheregaray 2020; Brauer et al. 2016).

2.8.5 Known threats and knowledge gaps

River regulation, cold water pollution and associated habitat deterioration including loss of aquatic vegetation, floodplain alienation and flow changes as well as predation and competition with alien species (including Redfin Perch, Trout species, and possibly Eastern Gambusia, and competition with/habitat alteration by Common Carp) have contributed to population declines in Southern Pygmy Perch. The Urumwalla Creek population was severely reduced by drought in 2019 (M. Lintermans, personal communication, 2020). Knowledge gaps exist regarding the biology and ecology of this species (Lintermans 2007). Table 2-4 provides a summary of knowledge status of this species in the southern MDB.

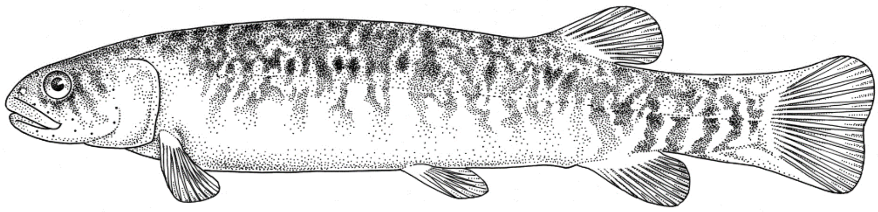
Table 2-4. Status of available knowledge for life stages of Southern Pygmy Perch (available knowledge was scored as follows: 1: 0–19% of knowledge needed is available; 2: 20–39% of knowledge needed is available; 3: 40–59% of knowledge needed is available; 4: 60–79% of knowledge needed is available; 5: 80% of knowledge needed is available): adapted from Koehn et al. (2020b).

Spawning	Eggs	Larvae	Juveniles	Adults
Spawning	2.5			3.0
Spawning conditions				3.0
Survival (recruitment)	2.0	2.0	2.0	2.0
Growth and condition		2.0	2.0	2.5
Movements		2.0	2.0	2.0
Physical habitat requirements	2.0	2.0	3.0	3.0
Water quality tolerances	2.0	2.0	2.5	3.0
Flows requirements	2.0	2.0	2.0	2.0

2.8.6 Overall summary

Although occurring broadly and despite some post-drought recovery, the species continues to decline across the MDB. Equally, backup populations are limited, with previous efforts hampered by numerous conservation units (requiring separate consideration) identified for the species.

2.9 Stocky Galaxias *Galaxias tantangara*



(Rhyll Plant and Raadik (2014))

2.9.1 Conservation status

International: *Critically Endangered*
National: *not listed*
NSW: *Critically Endangered*
Rest of range: -

The conservation status assessed under the following legislation: International: Union for the Conservation (IUCN) Nature Red List of Threatened Species; National: *Environment Protection and Biodiversity Conservation Act 1999*; New South Wales: *Fisheries Management Act 1994*.

2.9.2 Population status

Stocky Galaxias are only known from a single 3 km reach of a shallow alpine creek in the upper Murrumbidgee Catchment. The AOO and EOO are both estimated at 4 km^2, with a 93.1% decline in AOO inferred from its anticipated historical range(Lintermans and Allan 2019). It has been assessed as having a >65% probability of extinction in the next ~20 years (Lintermans et al. 2020).

2.9.3 Biological information

Stocky Galaxias has a maximum recorded length of 113 mm FL (M. Lintermans, personal communication, 2020), but commonly 75–85 mm FL (Raadik 2014). The body is mostly dark olive-brown in colour on the back and upper sides becoming lighter brown to centrally on the belly. It has dense, dark brown to almost black spots with moderately large, irregularly shaped, blotches mostly on the head and snout.

Detailed diagnostic characteristics which distinguish Stocky Galaxias from other *Galaxias* species as provided by Raadik (2014) include a mean total gill raker count of 10; body distinctly stocky and deep through vent and pectoral fin base (12.6–15.6 and 14.9–17.9% SL); caudal peduncle deep (8.5–10.2% SL); head obtuse to slightly bulbous in lateral profile, moderately deep (41.4–48.2% HL) but wide (63.4–72.8% HL); gape wide (40.2–51.0% HL and 59.6–72.2% HW); eye profiles usually not visible laterally from ventral view; nostrils short, not visible from ventral view; caudal fin weakly emarginate to truncate, about as long or slightly longer than caudal peduncle, vertical width of expanded rays usually equal to body depth through pectoral fin base; caudal peduncle flanges long, reaching more than half distance to anal fin base; anal fin long (16.3% SL); most posterior extent of mouth 0.8 ED below ventral margin of eye; dorsal midline usually flattened anteriorly from above or slightly posterior to pectoral fin bases; raised lamellae absent from ventral surface of rays of paired fins; anal fin origin usually under 0.73 distance posteriorly along dorsal fin base; usually 2, occasionally 1, relatively thin and long (4.7% SL) pyloric caecae; gill rakers short to very short; and, lack of distinct black bars along lateral line (Raadik 2014).

The reproductive development of Stocky Galaxias has recently been documented with males maturing in their second year (at approximately 52 mm LCF), and females in their third or fourth year (at approximately 70 mm LCF) (Allan et al. 2020). The spawning period for Stocky Galaxias is in late spring (Allan et al. 2020). The habitat for Stocky Galaxias is characterized by cold, clear water with high flow over a substrate of bedrock, boulder, cobble, pebble and gravel and areas of silt deposit. Pools average 0.3 m in depth with structural habitat comprised of rock and overhanging riparian vegetation. Water temperatures at the sole remaining site where this species occurs (Tantangara Creek) regularly drop to 2-3 degrees between early May and early September, and regularly fall below 1 degree in July (Allan et al. 2020).

2.9.4 Genetic management

Until recently, the species was thought to be part of the Mountain Galaxias species complex before its description by Raadik (2014). No population genetics has not taken place, so the species is managed as a single conservation unit.

2.9.5 Known threats and knowledge gaps

Its exceedingly small geographic range place the species in a precarious position, where it is vulnerable to multiple threats and limited spatial area to escape any perils within its range. The range of Stocky Galaxias is thought to have been severely reduced due to direct predation and competition from alien salmonids Brown Trout and Rainbow Trout (Lintermans and Allan 2019). Other than trout, other species highlighted to potentially impact Stocky Galaxias through competition for resources are the alien Oriental Weatherloach *Misgurnus anguillicaudatus* that are currently increasing throughout the upper Murrumbidgee River Catchment and the translocated Climbing Galaxias *Galaxias brevipinnis* (Lintermans and Allan 2019).

Habitat degradation through loss of riparian vegetation, bank degradation, mesohabitat alteration and increased sedimentation through pest animal grazing and bushfires is another threat to Stocky Galaxias (Driscoll et al. 2019; M Lintermans, unpublished data), furthermore, climate change presents a long term threat influencing changes in many environmental parameters within the species range (Lintermans and Allan 2019).

2.9.6 Overall summary

Due to its heavily restricted range, Stocky Galaxias is Critically Endangered globally and at ongoing risk of extinction. Hampering its conservation, is a lack of knowledge on its biology and ecology.

3. EX SITU MAINTENANCE AND PRODUCTION

Central to the implementation of conservation translocations is the ability to release enough evolutionarily viable individuals to allow population establishment and persistence. This section focuses on ex-situ maintenance and production given the critical role that it can play in producing fish for release. In the context of this report, we consider ex-situ maintenance and production relating to aquarium (typically 40 to 200 L), tubs and tanks (200 to 10,000 L) and ponds (typically >10,000 L), each of which have been utilised previously for some of the target species. The objective of ex situ maintenance and production should be to contribute to conservation in the wild (reinforce known populations and establish new populations) or safeguard the species against extinction (NSW OEH 2019).

(NSW DPI Fisheries)

Whilst ex-situ hatchery production is well established for several large-bodied freshwater fish species, there is a deficiency of published information on the target species. For this report, published scientific papers and technical reports were reviewed but information was also sought from those involved with maintaining and breeding the fish, such as hatchery managers, private practitioners and researchers, that have considerable knowledge that has not been documented or published in the literature. This knowledge often develops from trial and error and in many cases has helped to successfully maintain and breed some of the target species. Extensive consultation formed an important component of the preparation of this section.

There are fundamental requirements for the ex-situ maintenance and production of any freshwater fish, including the provision of water of suitable quality, knowledge of basic husbandry principles (e.g., broodstock management, biosecurity, disease and infection control) and the capacity for active management (relevant to species and the setting). Further, it is assumed that all necessary permits and approvals have been secured. For this report, it is assumed that there will be a commitment to maintain these fundamental requirements for the ex-situ maintenance and production of any of the target species.

The following sections provide detailed descriptions of available knowledge on ex-situ maintenance and production of the target species (with the three Galaxias species grouped due to a lack of information).

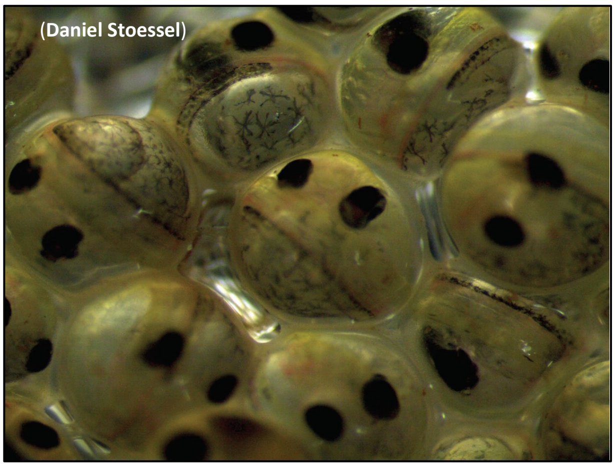
3.1 Galaxias species (Round-snout Galaxias, Short-tail Galaxias and Stocky Galaxias)

There is limited knowledge on the maintenance and breeding of any of the target Galaxias species. In early 2020, Stocky Galaxias and Short-tail Galaxias, were rescued from bushfire impacted areas, but the attempt to maintain them at the Gaden Trout Hatchery (Jindabyne) has been largely unsuccessful, and the remaining fish were transferred to aquarium facilities at Charles Sturt University, Albury. In Victoria, several closely related species, in the *Galaxias olidus* complex, such as Barred Galaxias *Galaxias fuscus* and West Gippsland Galaxias *Galaxias longifundus*, have been successfully maintained and bred in aquaria (Stoessel et al. 2020b). More recently, McDowall's Galaxias *Galaxias mcdowalli*, Yalmy Galaxias *Galaxias* sp. Yalmy, Dargo Galaxias *Galaxias mungadhan*, Shaw Galaxias *Galaxias gunaikurnai*, East Gippsland Galaxias *Galaxias aequipinni* and Cann Galaxias *Galaxias* sp. 'Cann', have also been successfully maintained in captivity (T.A. Raadik, personal communication, 2020). Knowledge gained from these efforts are universally applicable to the three target galaxiid species, although species-specific knowledge is also required.

The ex-situ maintenance of these galaxiids, and production of Barred Galaxias and West Gippsland Galaxias, was guided by available knowledge from wild populations and is detailed in Stoessel et al. (2020b). Most notably, photoperiod and water temperature were controlled to mimic prevailing conditions during the spawning period for wild population (Stoessel et al. 2015; Stoessel et al. 2020b). Initially, only reproductively mature and/or developing fish were collected from wild populations. Fish were housed in glass aquaria (297 L; 900 x 550 x 600 mm) (6-8 fish per aquarium) filled with aged (chlorine-free), carbon-filtered tap water (EC 550) to a depth of 300 mm which was trickle-fed, recirculated, filtered (Eheim 2217 canister filter) and chilled (Teco TC20 water chiller: Stoessel et al. 2020b). All fish were monitored daily and fed live tubifex worms (*Tubifex* sp.).

To promote spawning, conditions within the aquaria were adjusted to mimic known spawning conditions for wild populations (cf. Stoessel et al. 2015). For Barred Galaxias, water temperature was increased from 9.5–11.5°C and natural light was utilised (with 70% shade cloth and 66% roof shade). For West Gippsland Galaxias, water temperature was slowly decreased (15–7°C), and indoor photoperiod reduced from 9 h 59 min to 9 h 32 min over 12 days (to mimic the winter solstice), before water temperature and photoperiod were gradually increased over 48 days to 12°C and 11 h 31 minutes.

When females of each species achieved reproductive maturity (e.g. ovaries visually determined to fill approximately >90% of the body cavity, which was clearly distended), fish were hand-stripped with West Gippsland Galaxias producing more oocytes per female (~509 oocytes) compared to Barred Galaxias (~106 oocytes). For both species, the adhesive, transparent and spherical eggs were placed as a single layer on the bottom of plastic petri dishes lined with 1.5 mm plastic mesh, and within 30 sec, males were stripped and milt was spread over the oocytes using a soft fine brush. Immediately after stripping, adult fish were treated for 20 minutes in a saline solution (~16000 EC), then transferred to a 40 L recovery aquaria containing aerated, chilled water (~11°C) treated with fungicide (Aquatopia® fungus eliminator), before fish (only individuals showing no adverse reactions) were returned to the wild.


(Daniel Stoessel)

After fertilisation, eggs were placed in separate incubators in 20 L glass aquaria with aerated and recirculated water chilled to 9.0°C and 11.0°C for Barred Galaxias and West Gippsland Galaxias, respectively. Eggs underwent daily fungus checks and salt solution sterilisation (~16000 EC for 20 mins). Hatching period varied, with Barred Galaxias eggs hatching between 38 to 50 days (mean 44 days) with West Gippsland Galaxias eggs hatching sooner (between 26 and 30 days; mean 27 days) after fertilisation. The majority of eggs successfully hatched (Barred Galaxias: 80.5%; West Gippsland Galaxias: 63.4%), with the inferior hatching rate in West Gippsland Galaxias attributed to oocytes not developing or developing with significant deformities. When hatched, larvae were transferred into static aerated 4 L aquaria and fed twice daily (starting with a liquid feed (Aquasonic Pty Ltd Complete Fry Starter), encapsulated food (JBL NovoBaby 01), frozen brine shrimp and lastly Artemia nauplii). Larvae were transferred into 10 L aquaria when free swimming and into 100

L aquaria once yolk sack was absorbed (3 fish L^{-1}). Newly hatched Barred Galaxias and West Gippsland Galaxias larvae were 8.4–9.7 mm (mean 9.0 mm) and 7.1–8.9 mm (mean 8.3 mm) in length, respectively. Temperature was slowly increased in larval tanks over 60 (9.0–12.0°C) and 48 (11.0–18.0°C) days for Barred Galaxias and West Gippsland Galaxias, respectively (Stoessel et al. 2020b). Individuals were released as approximately one to three month old larvae.

These methods and procedures from galaxiid maintenance and the Barred Galaxias and West Gippsland Galaxias breeding will help to guide the ex-situ maintenance and production for the three target species. Generalities include mimicking prevailing conditions during the spawning season of wild populations and undertaking intensive management of the spawning and early development, as well as keeping the fish cool. There are clear similarities between the galaxiid species relative to reproduction and there are clearly differences between the requirements of galaxiid species, which may be linked to the habitats that each species occupy.

It should also be noted that for Stocky Galaxias and Short-tail Galaxias at Jindabyne and Stocky galaxias at Charles Sturt University, fish were found to be extremely aggressive after several months in captivity and needed to be housed individually (D. Gilligan, unpublished data). However, in the case of other species of galaxias maintained in captivity (i.e. *Galaxias mungadhan*, *Galaxias aequipinnis, Galaxias sp.* 14, *Galaxias mcdowalli, Galaxias sp.* 17, *Galaxias fuscus*, *Galaxias longifundus* fish have not displayed such behavior and some of these species have been maintained at densities of 100 individuals without any issues, with the majority reaching reproductive maturity prior to release back to the wild (D. Stoessel, personal communication, 2020).

Again, it is important to inform the ex-situ maintenance and production with knowledge of wild populations. There have been no attempts to continue maintenance to produce juveniles or adults and therefore the requirements for tub and pond maintenance are unknown. Further, it remains unresolved whether the three target species can be maintained as self-sustaining populations in ponds.

Table 3-1 is a summary of known parameters for breeding Galaxias species (Round-snout Galaxias, Stocky Galaxias and Short-tail Galaxias) in captivity.

Table 3-1. Breeding parameters summary for Galaxias species (Round-snout Galaxias, Stocky Galaxias and Short-tail Galaxias.

Adult holding	• Glass aquaria (297 L; 900 x 550 x 600 mm) (6-8 fish per aquarium) filled with aged (chlorine-free), carbon-filtered tap water (EC 550) to a depth of 300 which was trickle-fed, recirculated, filtered (Eheim 2217 canister filter) and chilled
Water quality	• Temperature for spawning 11.5°C, eggs 9.0°C, larvae 12°C for Barred Galaxias • Temperature for spawning 12°C, eggs 11.0°C, larvae 18°C for West Gippsland Galaxias
Photoperiod	• Natural light with shade cloth for Barred Galaxias • Photoperiod at 11 h, 31 mins for West Gippsland Galaxias
Fry food	• Aquasonic Pty Ltd Complete Fry Starter, encapsulated food (JBL NovoBaby 01), frozen brine shrimp and lastly Artemia nauplii
Adult diet	• Live tubifex worms (*Tubifex* sp.)
Hatching time	• Barred Galaxias eggs between 38 to 50 days; mean 44 days • West Gippsland Galaxias eggs between 26 and 30 days; mean 27 days
Brood stock treatment	• After stripping, adult fish treated for 20 minutes in a saline solution (~16000 EC), then transferred to a 40 L recovery aquaria containing aerated, chilled water (~11°C) treated with fungicide (Aquatopia® fungus eliminator)

3.2 Murray Hardyhead

Murray Hardyhead have been successfully maintained and bred ex situ in aquaria and tubs whilst populations have been maintained in ponds. In response to deteriorating conditions during the Millennium Drought, ex-situ maintenance and production was commenced for several wild subpopulations at the Murray-Darling Freshwater Research Centre (Mildura) (Ellis et al. 2013). The methodology was broadly consistent (Ellis and Carr 2011; Ellis and Pyke 2009) with adults collected from target subpopulation housed in separate glass aquaria fitted with mechanical and bio-filtration as well as 200 L recirculating tubs, and with overflow solids removal systems, and a 400 L sump fitted with two rotating sprinkler bars to trickle water over bio-filtration media. Water was supplied back to the tubs via a UV steriliser and coarse filtration.

The aquaria and tubs were maintained at a range of salinities (7000–25,000 EC) by adding locally sourced salt, to reflect salinities at which successful recruitment had been observed at source wetlands for each subpopulation (e.g. Ellis 2005). Water hardness were manipulated to maintain higher pH (~8.5) similar that observed successful recruitment events. Each aquaria and tub were given a 10–20% water change each week, with tap water treated with commercial chlorine neutraliser and conditioned for at least two days prior to use. Filter media was cleaned monthly. Adult fish were fed a mix of frozen bloodworms, dried flake and live zooplankton.

Ellis and Pyke (2009) indicated fish condition and timing of collection (given the species is largely annual with a spring and summer breeding period) were key determinates of spawning success.

Spawning was induced through artificial manipulation of room temperature to mimic spring water temperatures at which wild spawning been observed (>24°C) (Ellis and Carr 2011). Pots of live *Ruppia* and artificial spawning substrate (plastic and wool floating materials) were added to breeding aquaria, on which adults laid adhesive eggs. After several days of maturation, the artificial media was moved to separate rearing aquaria for hatching and larval development (Ellis and Carr 2011).

Hatching took place five to ten days after fertilization in waters between 20 and 26°C (I. Ellis, unpublished data). Most hatching success was achieved when egg batches were allowed to mature for several days in the adult aquaria they were spawned in, before later relocation of spawning material to hatch in smaller aquaria incorporating sponge filters and water taken from the adult aquaria they were spawned in (Ellis and Pyke 2009).

Larvae were fed a variety of food sources twice daily (live planktonic artemia, rotifers, baby brine, flakes and liquid larval food). The smaller aquaria allowed larvae to find the supplied food sources with limited effort and flow related disturbance. Larval rearing tanks included sponge filtration to minimise flow and entrainment within filters and pumps. Once juvenile size was attained (15–25 mm SL) they were moved to larger aquaria, where a slow transition to adult food sources was completed (Ellis and Pyke 2009).

More recently, Stoessel et al. (2020a) successfully maintained and bred the species on mass in conjunction with salinity tolerance experimentation. Stoessel et al. (2020a) achieved production (e.g. spawning, egg, larvae and juvenile survival and development) out of the normal breeding season. Initially, wild adult fish were transferred to two isolated quarantine aquaria (360 L, 1200 × 500 × 600 mm), which were trickle-fed, recirculated, filtered (Eheim

2217 canister filter) and maintained at ~6250 EC to mimic the salinity of the wetland where fish were captured. Stoessel (personal communication, 2020) suggested that although a high pH (8.5) is found in the wild, in captivity maintaining aquaria for long periods at a high pH can prove deadly for fish due to it affecting other water quality compounds and parameters. Adult fish were fed twice daily with a diet of rotifers *Brachionus rotundiformis*, *Artemia* nauplii (Ocean Nutrition™), tubifex worms *Tubifex* sp., and pelletised food (Nutrigard Dust, Primo Aquaculture). Fish were treated with a dewormer (BluePlanet®, Masterpet Corp Ltd) three days after arrival. Reproductive development in adults was encouraged over two weeks by slowly raising water temperature within both quarantine aquaria (from 17 to 25°C), increasing photoperiod (11 to 16 h) and reducing salinity (from 6250 to ~1500 EC). Stoessel et al. (2020a) indicate that courtship behaviour (e.g. chasing and nudging), reproductive maturation and spawning was not observed until water temperature increased above 24°C.

When reproductively mature, fish were divided equally into 10 trickle-fed, recirculated and filtered broodstock aquaria (~62 L, 590 × 300 × 350 mm), maintained at 25°C and a salinity of ~1500 EC. In each aquarium, two spawning mops (~50 strands of green wool, ~30 cm long, tied to a float) were added overnight. The clear, small (1.79–2.16 mm, mean 1.95 mm), water hardened eggs that had been laid overnight were handpicked from the mops each morning and placed into floating hatching baskets and assigned to one of 27 independent treatment or three control aquaria (12 L, 410 × 150 × 200 mm) with aerated water maintained at ~1500 EC. After 1 h of immersion in either treatment or control aquaria (i.e. to allow eggs to water harden), hatching baskets (containing eggs) were placed in a sterilisation bath containing 3% hydrogen peroxide at 250 ppm for 15 minutes (Small 2004) to assist in the control of fungus (repeated daily until no eggs remained in aquaria). After spawning, there was no adult mortality and fish regained reproductive condition in as little as one week. In the weeks following the first spawning, five sequential spawning's, separated by one week, were achieved prior to budgetary constraints halting breeding (D. Stoessel, personal communication, 2020). Adult fish, nevertheless, remained in reproductive condition up until their release to the wild in spring (D. Stoessel, personal communication, 2020). Excluding the eggs, larvae and juveniles used in experiments, well over a thousand larvae were hatched and released to the wild, thereby indicating a captive breeding protocol that was sound, and one that was also easily repeatable (D. Stoessel, personal communication, 2020). Newly hatched Murray Hardyhead larvae were 4.2–6.24 mm (mean 5.19 mm).

Whiterod (2019) details the maintenance and production of the species in three (0.1–0.3 ha) ponds (e.g. in this case, constructed wetlands and farm dams), which are characterised by aquatic habitat (including *Vallisneria*), secure water supply and moderate salinities (~1000 to 5000 EC) and no predatory fish species.

The species has been relatively easy to maintain and produce in ex situ situations. Adhering to the requirements of the species (e.g. water temperatures above 24°C during spawning season, provision of spawning structure, maintaining elevated salinities) will help to ensure the production of large numbers of individuals. Given the species is short-lived, it will be clearly important (as with wild sites) to avoid spawning and recruitment failure in any season or have capacity to replenish the breeding population.

Table 3-2 is a summary of known parameters for breeding Murray Hardyhead in captivity.

Table 3-2. Breeding parameters summary for Murray Hardyhead

Water quality	• Salinities (1500–25,000 EC) • Temperature for spawning >24°C
Photoperiod	• Photoperiod at 16 h
Fry food	• Live planktonic artemia, rotifers, baby brine, flakes and liquid larval food
Adult diet	• Frozen bloodworms, dried flake, live zooplankton, rotifers (Brachionus rotundiformis), Artemia nauplii (Ocean Nutrition™), tubifex worms Tubifex sp., and pelletised food (Nutrigard Dust, Primo Aquaculture)
Spawning substrate	• Live Ruppia and artificial spawning substrate (plastic and wool floating materials), spawning mops
Brood stock treatment	• Dewormer (BluePlanet®, Masterpet Corp Ltd)

3.3 Olive Perchlet (MDB population)

Olive Perchlet have been successfully maintained and bred at Narrandera Fisheries Centre on multiple occasions. Llewellyn (2008) details the historical (1968 to 1971) breeding program of the species, which followed the methodology that was established for Flathead Galaxias (Llewellyn 2005). Two earthen ponds (0.01 ha each) were utilised, which were purposefully designed to allow continual water flow through the pond if needed, whilst maintaining water at a constant level. Ponds were emptied and filled via a screened penstock and were fitted with a raceway in the base to allow for fish collection upon emptying the ponds. Water flow was continued through the ponds and breeding was noted after water level was raised in the ponds on at least one occasion.

Fish were bred in ponds on ten occasions between November and early January, and during February (water temperatures 19–27°C). For example, in Pond One which was stocked with 133 fish, fish spawned for the first two spawnings between 23–24°C and at 25°C for the third spawning. Pond Two was stocked with 146 fish and fish spawned at temperatures of 26°C in early to mid-January. In Pond Three (259 fish), spawning occurred at approximately 26°C after food had been added and water levels were increased. There was no evidence found of spawning above water temperatures of 27°C. The survival of adults in ponds varied with mortality attributed to low water temperatures and avian predation (Llewellyn 2008). Fish were fed on zooplankton, phytoplankton and small shrimp.

Breeding was also attempted in glass aquaria (90L) which were fitted with aquatic plants and floating weeds. Water temperature was increased (10 to 27°C), as were feeding regimes, however in all attempts, fish died over several months. pH of aquarium water was slightly alkaline (7.7–8.5). Fertilisation was unsuccessful when efforts were made to strip and fertilise ova from ripe females (Llewellyn 2008).

Following the rediscovery of the MDB population in the Lachlan River Catchment, ex situ maintenance and production of Olive Perchlet was initiated in a pond at the Narrandera Fisheries Centre. Fish (n=264) were captured from the wild in late 2009 and released into Pond Four at Narrandera Fisheries Centre. The pond had a stable water level, minimal (or no) flow-through flow, no aeration and no supplementary food was supplied. The pond had sparse submerged macrophyte cover (primarily floating pondweed). Bi-monthly pond monitoring determined recruitment within the first six months. The pond was harvested in May 2011 and 3200 fish were recovered. Fish (n=700) were released into Thegoa Lagoon, 2000 fish were released into Cargelligo weir pool and 500 fish were retained (returned to the same pond). The pond was harvested in May 2012 and 5000 fish were recovered. From this, 4500 fish were released into Cargelligo weir pool and 500 were retained (returned to the same pond). The pond was harvested in May 2013 and only 600 fish were recovered. All were retained (returned to the same pond). The pond was harvested in February 2014 and 1200 individuals were recovered. All were retained (returned to the same pond). Subsequent routine monitoring of the pond failed to detect Olive Perchlet over several months and upon draining the pond in November 2014, no fish were recovered. While the cause of the dissappearance of the fish is unknown, it was suspected to be the result of an unseasonal 'cold-snap' event in Spring 2014 and/or increased predation by birds due to increased water clarity in the pond. For future breeding, it is recommended to cycle fish through ‘fresh’ ponds, rather than repeatedly re-seeding the same pond over several years. Post-release surveys over several years failed to detect any individuals at either release location (L. Jess, personal communication, 2020).

Olive Perchlet were held in tanks and captive breeding was attempted but was not successful (L. Jess, personal communication, 2020). Past efforts with the species indicate that maintenance and production in ponds is a viable strategy, but some management is required.

Table 3-3 is a summary of known parameters for breeding Olive Perchlet in captivity.

Table 3-3. Breeding parameters summary for Olive Perchlet

Water quality	• Spawning at 23-26°C
Photoperiod	• Natural outdoor conditions
Adult diet	• Zooplankton, phytoplankton and small shrimp
Spawning substrate	• Macrophytes should be present in pond

3.4 Oxleyan Pygmy Perch

Aquaria have been central to previous ex situ maintenance and production of Oxleyan Pygmy Perch. Wager (1992) observed daily serial spawning behaviour at water temperatures above 20°C; pairs casually approached each other, shuddered while releasing a few eggs and milt, and then moved past one another. Subsequently, Knight et al. (2007) undertook spawning trials using two glass aquaria (240 L) divided into three equal portions, with gravel substratum, a recirculating system employing an external canister filter (Eheim) and chiller (Resun CL65) with water consistent with water quality of the Evans Head subpopulation. Fish were maintained with a daily diet of frozen bloodworms intermittently supplemented with live *Artemia salina* and mosquito (*Aedes vigilax*) larvae. Artificial spawning habitat (~75 strands of 250 mm black acrylic wool as a 'mop') were introduced into each portion of the aquaria (hung both horizontally and vertically). The spawning 'mops' were searched routinely for eggs, with non-developing eggs discarded. Developing eggs were transferred to floating Petri dishes to facilitate hatching. Spawning behaviour was apparent during the prolonged spawning period (e.g. September to May) with spawning commencing as water temperatures increased to 16.6°C and daylight to 11.9 h (Knight et al. 2007). Spawning frequency increased across the spawning season with a peak in November at water temperatures of ~20°C.

Over 2019–20, Oxleyan Pygmy Perch have been successfully bred at the Grafton Primary Industries Institute (M. Turner, personal communication, 2020). To achieve this, six adult fish were placed in individual glass aquaria (51 L) that were kept indoors with water supplied by aged town water trickling through them to ensure water exchange and aeration. Water quality was maintained within suitable tolerance ranges (e.g. pH: ~6.4; DO: >4 mgL^{-1}; EC initially increased to 5000 EC when adult fish were first introduced, but diluted down to ~1600 EC by the time breeding had commenced through the constant filtration system). Substrate including river rocks was added to tanks and it was noted that without the addition of the rocks the fish would not spawn. Consistent with Knight et al. (2007) spawning 'mops' were hung in each aquarium to take up the entire length of the water column and were either black or dark green shade (it was observed the fish preferred to spawn on the dark green shade mops). Fish were first induced to spawn in early August by increasing temperatures to 27°C and keeping them at this constant temperature. Photoperiod was adjusted to 12 h and 10 minutes of light each day (M. Turner, personal communication, 2020).

Fish were observed spawning directly onto the mops and eggs and larvae were left in the tanks until larvae were swimming freely, at which point free swimming larvae was removed. Approximately 50 larvae were placed into individual glass aquaria (51 L) with no filtration or flow through water provided whilst larvae were small (only slight aeration). When larvae were larger, a 300 micron mesh was placed on filters and water flow-through was recommenced as well as the aeration. Only slight cannibalism within larvae was noted throughout the project. Larvae were fed three times a day with Artemia to help decrease cannibalism rates.

Larvae were grown out to juveniles. All adult fish and larvae were fed Artemia throughout the entire project (M. Turner, personal communication, 2020).

On multiple occasions, successful maintenance and breeding of the species has been documented. As with other target species, juvenile grow out has not been undertaken and it is unclear whether tub or pond production is feasible. Table 3-4 is a summary of known parameters for breeding Oxleyan Pygmy Perch in captivity.

Table 3-4. Breeding parameters summary for Oxleyan Pygmy Perch

Water quality	• Spawning temperature >20°C, pH: ~6.4, DO: >4 mgL^{-1}, EC initially increased to 5000 EC when adult fish were first introduced, but diluted down to ~1600 EC prior to spawning
Photoperiod	• 12 h and 10 minutes
Fry food	• Artemia
Adult diet	• Frozen bloodworms, live Artemia salina and mosquito (*Aedes vigilax*) larvae
Spawning substrate	• Artificial spawning habitat (~75 strands of 250 mm black acrylic wool as a 'mop')

3.5 River Blackfish (Snowy River Population)

This section draws on knowledge of all species within the River Blackfish complex, which are assumed to be broadly relevant to the Snowy River population. However, this must be taken with caution, as the Snowy population occurs are high elevation in cool stream. River Blackfish have not been widely maintained or bred in captivity. However, captive rearing was attempted at the Narrandera Fisheries Centre in 2005, whilst Westergaard and Ye (2010) provide the first account of ex situ maintenance and breeding in SA, and the species has been successfully bred privately, on a small scale in ponds at a fish farm in Cooriemungle, Victoria.

To develop captive rearing methods for the Snowy River population of the SEV candidate species, fifty-nine adult fish were collected and transported to the Narrandera Fisheries Centre in 2005. Fifteen fish were placed in an outdoor pond and 44 individuals were housed in two 3000L indoor tanks. No fish were recovered from outdoor ponds. Fish in indoor tanks were observed to fight and had a high mortality rate. Pathology tests of sick fish found high loads of microsporidian parasites not found on other species at Narrandera Fisheries Centre and some Chilodonella. The few surviving tank fish were transferred to a pond but were not recovered (D. Gilligan, personal communication, 2020).

The efforts of Westergaard and Ye (2010) focused on a subpopulation of the 'northern' candidate species (NMD, which is only found in the MDB) from Mt Lofty tributary streams in the lower MDB. A total of nine River Blackfish were captured during autumn and transported in aerated buckets, allowed to acclimate to new conditions over 2 h and placed in a large (2000 L) tank with recirculating water (7500L h^{-1}). The water temperature was maintained at 10–14°C and 17–21°C during winter and summer, respectively. Fish were fed live *Daphnia* sp., live earthworms and chopped prawn (Westergaard and Ye 2010). Fish were maintained for 18 months with two mortalities occurring.

In preparation for spawning, pairs of fish were moved to a controlled environment room in October and held in 300 L aquaria (photoperiod 12:12 increased to 14:10, mean water temperature 17°C; DO: 9.46 mgL^{-1}; EC: ~720 EC; pH: 7.49). To condition fish, they were fed live *Daphnia* sp., live earthworms and chopped prawn (Westergaard and Ye 2010). Fish spawned naturally and no hormone manipulation was utilised. As soon as spawning was complete, eggs were removed and held in hatching trays held horizontally in glass aquaria (50L) with gentle airlifts and daily water changes. Methylene blue was added (<100 ppm) to the first spawn of eggs and formalin was used for the second spawn of eggs to prevent and treat fungal infections. From day 20, juveniles were given *Artemia* sp. nauplii and live *Daphnia* sp. and frozen bloodworms provided after day 30 (Westergaard and Ye 2010).

The candidate species from Victoria and Tasmanian basins draining to Bass Strait (SBA) has been the focus of captive breeding efforts of Stephen Mueller. Specifically, fish have been collected from the wild in the Otway Ranges region. During this process it was vital that the fish were handled with extreme care and collecting was undertaken only in the warmer months and at night. Fish were kept in a keeper net in the water they came from until they were transported. The container used for transport was of a generous size and aerated, and the fish were released quickly into their new habitat at the breeding farm after ensuring the water temperatures were similar (S. Mueller, personal communication, 2020).

Their new habitat consisted of large cement farm (4500 L) troughs with a flat bottom ensuring a maximum of four fish were placed in one tank. Fish were observed often through the first few days and nights to make sure there was no fighting or cotton wool fungal disease (saprolegniosis). If this fungal infection was suspected to be occurring, a salt bath was provided. Tanks with sloping floors were not used as they force the fish together and this can lead to fighting. Suitable habitat was placed into the tanks including pieces of poly pipe, long enough so a fish can hide (S. Mueller, personal communication, 2020).

Ponds in which fish were kept were maintained as natural as possible, vegetation was added and encouraged both in and around the ponds. Pond size was approximately 9 m long x 6 m wide x 2 m deep. This size was used to ensure the pond surface size was not too large to prevent retrieval of the young fish. Noise levels were kept to a minimum around the tanks or ponds during the breeding period. All ponds were aerated however the aeration was kept minimal and off the bottom of the pond so as not to stir up sediment that is on the bottom (S. Mueller, personal communication, 2020).

Where possible rainwater was used with no added chemicals. Water was changed more often during warmer temperatures. Water temperatures at the farm varied between 9–11°C in winter to approximately 18°C in high summer. If water temperatures spiked, water was cooled by the addition of dam water noting that a flush of clean water at the start of spring which can help the spawning process. Where dam water was used and if it was cloudy, it was run through constantly. Tannin was added to the water by placing limbs of blackwood trees into the tanks, to provide similar conditions to that in the wild and act as an anti-bacterial (S. Mueller, personal communication, 2020).

The fish were fed on natural diets that were either grown or collected from the fish farm, which was assessed as an important step in the success of the breeding program. Food included yabbies, crickets, grasshoppers, grubs and earthworms, and other insects collected when they were in season. Yabbies' were a good food source as they could be placed into the tanks or ponds and the fish could decide when to eat. If natural food was not available, then mealworms were used. Further investigation is required to determine if River Blackfish would take a manufactured food pellet (S. Mueller, personal communication, 2020).

Juvenile fish were removed from the breeding ponds as eggs could not be removed. The juveniles were collected when they started to swim away from the nest. Shrimp nets were placed around the pond approximately 1.5 m apart and checked regularly (hourly) (to prevent yabbies injuring the fish). The nets were placed on the bottom of the pond and removed at dusk. In the tanks the young were either taken out (with shrimp nets) or the adults were removed. It is recommended to remove the young as they start to wander about as they are difficult to catch at night when they are sheltering under leaves and litter (S. Mueller, personal communication, 2020).

Two 2000 L poly tanks were used, with modified outlets so very small fish could not escape. During approximately early October, commencement of raising copepods and other plankton in these tanks was undertaken. This was done by filtering dam water and collecting the bugs and then seeding the poly tanks with them. No running water was allowed into the tanks at this point to keep the bugs in the tanks and to turn the water green. A handful of blood and bone was added to the water to feed them (S. Mueller, personal communication, 2020).

Important points detected whilst breeding include (S. Mueller, personal communication, 2020):

- The fish are not strictly nocturnal they will feed during the day especially once they are settled in, this will be important in a hatchery setting where night feeding is a problem.
- These fish stress easily when handling them.
- Their eating declines rapidly during cold periods.
- Fish should not be forced to eat.
- Fish can hide in a four-gallon bucket and can be difficult to detect.

- Fish will escape and jump out from tanks if they are not secure.
- They will assume the color of the water they are in and can go from very dark to light grey.

Lessons can also be garnered from Two-spined Blackfish which when held in 1.2 m aquaria are aggressive toward each other, often resulting in death and need to be housed separately. This occurs even when there are multiple hiding places in a tank. Fungal infections can be cured with a saline solution (900 g coarse salt in a 180 L tank) in which they can live for months (M. Lintermans, unpublished data).

Table 3-5 is a summary of known parameters for breeding River Blackfish in captivity.

Table 3-5. Breeding parameters summary for River Blackfish

Water quality	• Indoor 300 L aquaria: mean water temperature 17°C, DO: 9.46 mgL^{-1}, EC: ~720 EC, pH: 7.49
Photoperiod	• Indoor 300 L aquaria photoperiod 12:12, increased to 14:10
Fry food	• Artemia sp. Nauplii, live Daphnia sp., live Copepods
Adult diet	• Live Daphnia sp., live earthworms, chopped prawn yabbies, crickets, grasshoppers, grubs and earthworms
Spawning substrate	• PVC pipes
Brood stock treatment	• If cotton wool fungal disease (saprolegniosis) suspected a salt bath is required

3.6 Southern Purple-spotted Gudgeon

The breeding of Southern Purple-spotted Gudgeon is well established, and it is relatively straightforward to produce moderate numbers of large larvae (Blewett 1929; Llewellyn 2006). Ex situ maintenance and production has occurred on northern MDB populations at the Narrandera Fisheries Centre and by Aquasave–NGT on the southern MDB (lower Murray River) population in various aquaria, tub and pond settings (Whiterod 2019) as well as in various South Australian schools under guidance of Aquasave-NGT.

Llewellyn (2006) details maintenance and breeding in aquaria and ponds (0.01 ha) between 1965 and 1969 at the Narrandera Fisheries Centre. It is suggested that increasing water temperature alone was not sufficient to induce spawning but that increasing food (at least twice daily) is the principle cue for stimulating spawning. Colour intensifies and darkens during spectacular courting displays before pairs choose a nesting site. Fish bred at water temperatures between 20.0 and 29.9°C between October and February. After fertilisation, eggs hatched after 3 to 10 days dependent on water temperature. Between 200–1300 adhesive eggs attached to solid surfaces which the male guards and fans until semi-pelagic larvae hatch after c. 10 days (Llewellyn 2006).

In NSW, ex situ maintenance and production occurred at Narrandera Fisheries Centre between 2004 to 2009 during which time - in the best production year - 6,435 larvae were produced from 73 broodfish. One key point learnt during these production trials was that over-winter pond survival from larvae to juveniles was only 3%. But pond survival from larvae to juveniles was 60-70% in spring-summer. The captive bred fish were stocked into five locations between 2004 and 2012: Adjungbilly Creek near Gundagai, Burrawang West Lagoon (Lachlan), Western Plains Zoo waterways (Education Creek), Gulligal Lagoon (Namoi) and Castlereagh headwaters. Only small numbers were released at three of the five sites. Successful populations were not established at these sites however a population was successfully established in the Castlereagh (present in 2016) (D. Gilligan, personal communication, 2020).


(John Marriott)

The methodology for aquarium maintenance and production by Aquasave–NGT (Hammer et al. 2009a; Whiterod 2019) involves the use of large aquaria (200 L) maintained with aerated filtered water in temperature-controlled rooms. Broodstock are maintained separately and fed frozen bloodworms, dried flake and live zooplankton daily before pairing up in preparation for spawning. Similar sized fish can be paired but this is not essential. Fish can be spawned on demand during spring and summer by increasing water temperature to above 24°C and feeding stimuli (increase to twice a day). Spawning substrate is provided in each tank (firm objects such as slate, tile, or pipe) and males and females are kept separately outside of spawning. Immediately after successful spawning, females are preferably (and consistent with efforts in NSW) removed with males left to tend the nest until hatching is completed (regular observation is required to ensure male is adequately fanning eggs). Alternatively, both broodstock can be removed after spawning and very light aeration provided to mimic the male fanning the eggs (with filtration turned off). Upon hatching, larvae can be reared in the aquaria or siphoned off (ie: most parents do not appear to predate heavily on fry, but fry should be siphoned off within 24 hours of hatching). Filtration should be kept off whilst larvae are young and only slight aeration added. Larvae are fed live planktonic artemia, rotifers, baby brine shrimp, flakes and liquid larval food two to three times daily. Fish should be graded weekly and separated by size class. Maintenance involves 20–30% water changes at weekly to fortnightly intervals depending on feeding regime, temperature and tank condition (e.g. more frequent during summer breeding events) (Hammer et al. 2009a; Whiterod 2019).

Whiterod (2019) details the maintenance and production of the species in medium-sized (~0.1 ha) ponds (e.g. in this case, constructed wetlands and farm dams), which are characterised by aquatic habitat (undercut grassy edge, rock and *Vallisneria*), secure water supply and no predatory fish species. The species has been relatively easy to maintain and produce in ex situ situations. Adhering to the requirements of the species will help to ensure the production of large numbers of individuals.

Table 3-6 is a summary of known parameters for breeding Southern Purple-spotted Gudgeon in captivity.

Table 3-6. Breeding parameters summary for Southern Purple-spotted Gudgeon

Water quality	• Temperature for spawning 24°C
Photoperiod	• Natural
Fry food	• Live planktonic artemia, rotifers, baby brine, flakes and liquid larval food
Adult diet	• Frozen bloodworms, dried flake, live zooplankton
Spawning substrate	• Firm objects such as slate, tile, or PVC pipe
Hatching time	• 3-10 days (temperature dependent)

3.7 Southern Pygmy Perch

Having been in decline for several years, Southern Pygmy Perch has been subject to ex situ maintenance and production across the three states in which it occurs. The early work of Llewellyn (1974) utilised both aquaria and ponds at the Narrandera Fisheries Centre. Breeding was attempted in aerated triplicate large (90 L) and small (10 L) aquaria set up with dense aquatic plants, with daily fish feeding, temperatures manually increased (12 to 22°C), and slightly alkaline pH (7.2 to 7.8). However, there was no successful breeding in any of the aquaria. Llewellyn (1974) also transferred adult fish to two ponds (both 0.01 ha, depth: 137 and 183 cm). Plankton samples and benthic samples were used to determine presence of pelagic eggs or larvae between August and December. Ponds were only emptied and refilled to check on the condition and presence of adult fish and water levels were kept relatively stable at other times (Llewellyn 1974). Successful breeding occurred on three occasions in the ponds during September and October when water temperatures exceeded 21.0°C at the surface and 19.3°C at the bottom. When males were brightly coloured and females were markedly gravid, the pond was refilled, and fish food source was increased with large amounts of small insect larvae and crustacea added to the pond. Eggs were first recorded in the pond when water temperatures increased from 16.0°C at the surface and 15.5°C on the bottom in the morning to 21.0°C at the surface and 19.3°C on the bottom in late afternoon. Eggs hatched after two and three days at water temperatures between 15.8 to 25.3°C (Llewellyn 1974). Llewellyn (1974) concluded that water temperature (surface water temperatures 21.0°C to 22.1°C) was the predominate spawning cue for Southern Pygmy Perch and there was no requirement for inundation to promote spawning.

Southern Pygmy Perch were held, and captive breeding was undertaken in ponds at Narrandera Fisheries Centre between 2007 and 2013-2014 with mixed success. In one year, approximately 14,000 fish were produced, but during most years, there was negative production or mass mortality. Unfortunately, there was not any clear indication as to why fish bred so well one year, and not during others. Pond conditions were kept as for Olive Perchlet with stable water levels, minimal flow-through, no aeration and no supplementary feeding. Submerged macrophyte density varied across ponds and across years (D. Gilligan, personal communication, 2020).

In 2006, fish were sourced from two populations of rescued fish, 50 from Blakney Creek (redfin invasion) and 122 from Coppabella Creek (drought) and transported to the Narrandera Fisheries Centre. The 50 Blakney fish increased to 400 when the pond (Pond 3) was harvested in May 2007. Fifty of these fish were retained and 50 new broodfish were collected, and these 100 fish were returned to the same pond after leaving it dry for several weeks. The 350 captive bred fish were released in Pudman Creek. The pond was re-harvested in April 2008 and only

6 fish were recovered. The Coppabella Creek pygmy perch pond (No. 4) was harvested in June 2007. Ninety-three (of 122 initially put into the pond) were recovered and returned to the pond. The pond was re-harvested in February 2008 and only 69 were recovered. These were returned to Coppabella Creek. Both ponds were found to be heavily ‘contaminated’ with several other small fish species (D. Gilligan, personal communication, 2020).

In early 2009, approximately 2000 Southern Pygmy Perch were rescued from drying refugia in Coppabella Creek. They were housed in two outdoor earthen ponds at Narrandera Fisheries Centre (Pond 5 (556 individuals) and Pond 41 (1000 individuals)), and at a facility at Tumut (444 individuals) managed by Luke Pearce. A *Costia* outbreak occurred in pond 41 in May 2009 killing 209 fish (diagnosed by EMAI). The pond was drained, and the survivors were treated with formalin then transferred to pond 5 (making a total ~1,347 fish in the pond). Pond 5 was monitored bi-monthly over a 12-month period and the fish were known to have recruited. Pond 5 was harvested in May 2011 and approximately 13,000 fish were recovered. Approximately 4,500 fish were released into each of Thegoa Lagoon and Washpen Creek in the lower Murray catchment and 3,500 into wetlands on the Charles Sturt University campus. The remaining 500 were retained and replaced in Pond 5. The pond was re-harvested in March 2012 and only 400 fish were recovered. Two hundred were transferred to Dr John Conallin from the Murray CMA and were released into two wetlands in Warring Gardens, and 200 were retained and re-seeded in Pond 5. Subsequent routine monitoring of the pond captured very few adult fish or recruits and the pond was not harvested in 2013. When harvested in 2014, no Southern Pygmy Perch were recovered (D. Gilligan, personal communication, 2020).

From the 444 fish initially held at Tumut, 22 died shortly after capture and a further 360 died several days after being transferred from Tumut to Charles Sturt University in Albury. Luke Pearce used the remaining approximately 60 fish for studies undertaken as part of his Masters degree. Fifty-three Southern Pygmy perch where collected from the Blakney Creek population in 2008 for preliminary captive breeding trials at Tumut as part of Luke Pearce’s Masters research. These fish where keep in plastic ponds with habitat added. The fish spawned in each of the ponds, however there was high larval mortality due to predation by the adults. Thirty-seven juveniles were released into Pudman Creek in April 2009. Forty-seven adults were retained for further breeding trials in 2009. Eventually, a total of 106 fish collected or bred from the Blakney Creek population and held at the Tumut facility were released into Pudman Creek (D. Gilligan, personal communication, 2020).

A second pond was stocked with 92 Southern Pygmy Perch collected from the Lachlan catchment population in Blakney Creek in November 2010 in order to produce offspring to bolster the genetic diversity present in the now established Pudman Creek population. This pond was harvested in May 2012 and no Southern Pygmy Perch were recovered (D. Gilligan, personal communication, 2020).

More recently, ex situ maintenance and production has successfully been achieved by Middle Creek Aquaculture, Flinders University, Aquasave–NGT and Arthur Rylah Institute. At Middle Creek Aquaculture, Southern Pygmy Perch are bred in both aquaria and tanks. Aquaria consist of 10 x 350 L glass aquariums plumbed into a 2000 L sump. Water exchange occurs at a flow rate of 15 L h^{-1}, which is varied according to fish stocked and fry age and development. External poly tanks (10 x 3800 L) are static and above ground with aeration and internal bio-tube blocks. Bore water is used to fill aquaria and tanks. Due to some iron present and low pH levels in bore water, 5 x 200 L barrels are filled with bore water, aeration added (to oxidise iron, then pumped from barrel). pH is adjusted to 6.6–6.8 with bicarbonate of soda. After 48 hours the water can be used for top ups and water changes in the aquariums. Large static tanks are filled with bore water, aeration added, and pH adjustments made as required, are then left to age for approximately 6 weeks before use. When external water temperatures reach 15°C and stabilise, in late winter to early spring, the tanks are fertilised to ensure good green water development. Tanks are also seeded with Daphnia and copepods.


(Chris Lamin)

During brood stock management, males and females are separated when the colour and body shape difference can still be easily observed during late summer. Fish are placed in 350 L aquariums indoors and treated for internal and external parasites. In late April, feeding of live foods begins to ensure excellent body condition leading into spawning. Water temperature and parameters are checked twice weekly and pH adjusted as required. During breeding management, spawning media is added to the external tanks. Once external tank temperatures are consistently above 15°C, green water is created and on day 8–10 depending on algae development, pH should be between 6.6–7.2. Brood stock are counted and even numbers of males and females are added to 3800 L external tanks. Females should be very heavily gravid. Tanks are monitored daily after the first 72 h and checked for fry using a small micron net. Once fry are detected the adults are then trapped, counted, and removed from external tanks. Any females that have not spawned are separated, equal numbers of males selected and are placed into another green water tank until spawning occurs.


(Chris Lamin)

Fry feeding occurs with the addition of liquid fertiliser which may be required at weekly intervals to ensure good green water bloom is maintained for 21–28 days for optimum fry numbers. Newly hatched brine shrimp are added at day 10 and continued four times daily until fry are caught, counted, and moved into 350 L aquaria at approximately 6 weeks of age. They are then weaned onto a commercial fry 'dust' until 30 days pre-release when they are fed a combination of commercial feed, live artemia, copepods and daphnia. Small numbers of rotifers are added when available.

At Flinders University, maintenance and production of Southern Pygmy Perch has occurred in outside tanks (10,000 L and 2000 L) (Attard et al. 2016a; Attard et al. 2016b). The species has been maintained at salinities of approximately ~4600 EC for most of the year, and salinity is increased to approximately ~7800 EC during the coldest months to help with saprolegnia (winter fungus infection). Other water nutrient targets are ammonia <0.1 ppm, nitrite <0.01 ppm, nitrate <25 ppm and phosphate <3 ppm. Reasonable numbers (e.g. 1000s) of the species have been produced, which have contributed to reintroduction to former habitats (Hammer et al. 2013). Table 3-7 is a summary of parameters used by Middle Creek Aquaculture in breeding Southern Pygmy Perch in captivity.

Table 3-7. Breeding parameters used at Middle Creek Aquaculture for Southern Pygmy Perch.

Water quality	• Temperature for spawning 15°C • pH 6.6–7.2
Green water recipe per 3800 L tank	• 3 kg Lucerne chaff • 2 cups Dynamic Lifter • 100 ml Nutromol Liquid fertiliser
Fry food	• Live – copepods, daphnia, artemia, rotifers • Dry – Otohime A fry dust
Adult conditioning diet	• Dry - Otohime B2, freeze dried blackworms • Live – Copepods, daphnia, mosquito larvae • Frozen – blood worms
Spawning media	• These can be anything artificial such as nylon rope that has been untwined and held together with cable ties, nylon knitting yarn etc. These should be positioned on tank bottom and weighted if necessary.
Brood stock external and internal parasite program	• Elevated salinity (~6250 EC) for 28 days • Kusuri Wormer Plus (Flubendazole) as directed • Levamisole 14 gL^{-1} (dosage 1 mL per 7 L)

Both Whiterod (2019) and Arthur Rylah Institute (unpublished data) have achieved successful maintenance and production of the species in ponds (e.g. constructed wetlands and farm dams) in South Australia and Victoria. Whiterod (2019) indicates that ponds require aquatic habitat (including *Vallisneria*), secure water supply and no predatory fish species.

The species has been relatively easy to maintain and produce in ex situ situations. Adhering to the requirements of the species (e.g., water temperatures above 24°C during spawning season, provision of spawning structure, maintaining elevated salinities will help to ensure the captive production of the species for translocation.

4. CONSERVATION TRANSLOCATION

4.1 Background

Translocations are becoming increasingly proposed as tools to aid threatened species persistence and recovery in the face of the combined pressures of habitat degradation, changes in water availability and climate change (Armstrong et al. 2015; Corlett 2016; IUCN/SSC 2013). Specifically, conservation translocations are defined as the '*the intentional movement and release of living organisms where the primary objective is a conservation benefit*' with differentiation as either population restoration or conservation introduction (IUCN/SSC 2013). Population restoration involves the intentional release of individuals within the natural range to either enhance existing populations (*reinforcement*) or reestablish populations from where they have disappeared (*reintroduction*). Conservation introduction focuses on releasing a species outside its natural range to avoid extinction of populations (*assisted colonisation*) or to perform a specific ecological function (*ecological replacement*). For this conservation translocation handbook only translocations relating to population restoration will be considered as it is viewed that conservation introduction is not yet warranted for each of the species.

The translocation of threatened freshwater fishes must consider several factors that may influence the likelihood of success. Firstly, in contrast to terrestrial ecosystems, freshwater habitats are linear and highly dynamic in terms of habitat availability and connectivity (Lintermans et al. 2015). Secondly, these ecosystems are often impacted by threats, such as river regulation and alien species that cannot be effectively controlled. Thirdly, traits linked to vulnerability and extinction, such as small body size, small home range, limited dispersal and high degree of ecological specialisation, will influence the ability of small fishes to persist and reestablish (Kopf et al. 2017; Olden et al. 2007). Lastly, persistence as fragmented populations and reduced capacity for natural recolonisation emphasise the importance of translocations to reestablish locally-extinct populations. These considerations must be taken into account during planning and implementation of translocations that focus on threatened small freshwater fishes.

4.2 Planning

Conservation translocations require a series of deliberative decisions to move forward and maximise the likelihood of success. The first decision is whether translocations are necessary to manage the target species. This acknowledges that the target species may not benefit from translocations, or that translocations are not feasible. The feasibility of translocations relates to the ability to achieve a series of other steps (Figure 4-1). These steps include not only those specifically related to translocation, but also consider the protection and maintenance of presently known subpopulations; identification of additional existing subpopulations; mitigation of threats, site and regional habitat and flow management. In terms of translocations, the step of reestablishing new subpopulations is paramount, which can be achieved through reintroductions and then reinforcement. Reinforcement may also be necessary to maintain known subpopulations. Consideration of site habitat and flow management, broader flow connectivity and the capacity to rapidly respond to emerging threats will assist with persistence of both known and new subpopulations.

The investigation of threats to populations and habitats will provide information for the management steps above. The identification and assessment of potential translocations will be necessary to enable the reestablishment of new subpopulations. Conservation translocations need to be underpinned by the capacity to produce sufficient numbers of fish for reintroduction and reinforcement (see Section 3).

Successful implementation of the translocation of the target species will require appropriate permitting and approvals, governance and formalisation of a working group (including exploration of multi-jurisdiction collaboration), improving knowledge of production capacity and reintroduction ecology through specific research and monitoring, ensuring appropriate communication amongst stakeholders and a willingness to raise awareness and garner broader support. It will also be necessary for the maintenance of basic habitat and water flow, monitoring and evaluation of success of each subpopulation, and appropriate communication with stakeholders and wider community to achieve appropriate outcomes. There also needs to be a commitment of appropriate effort and investment into each step.

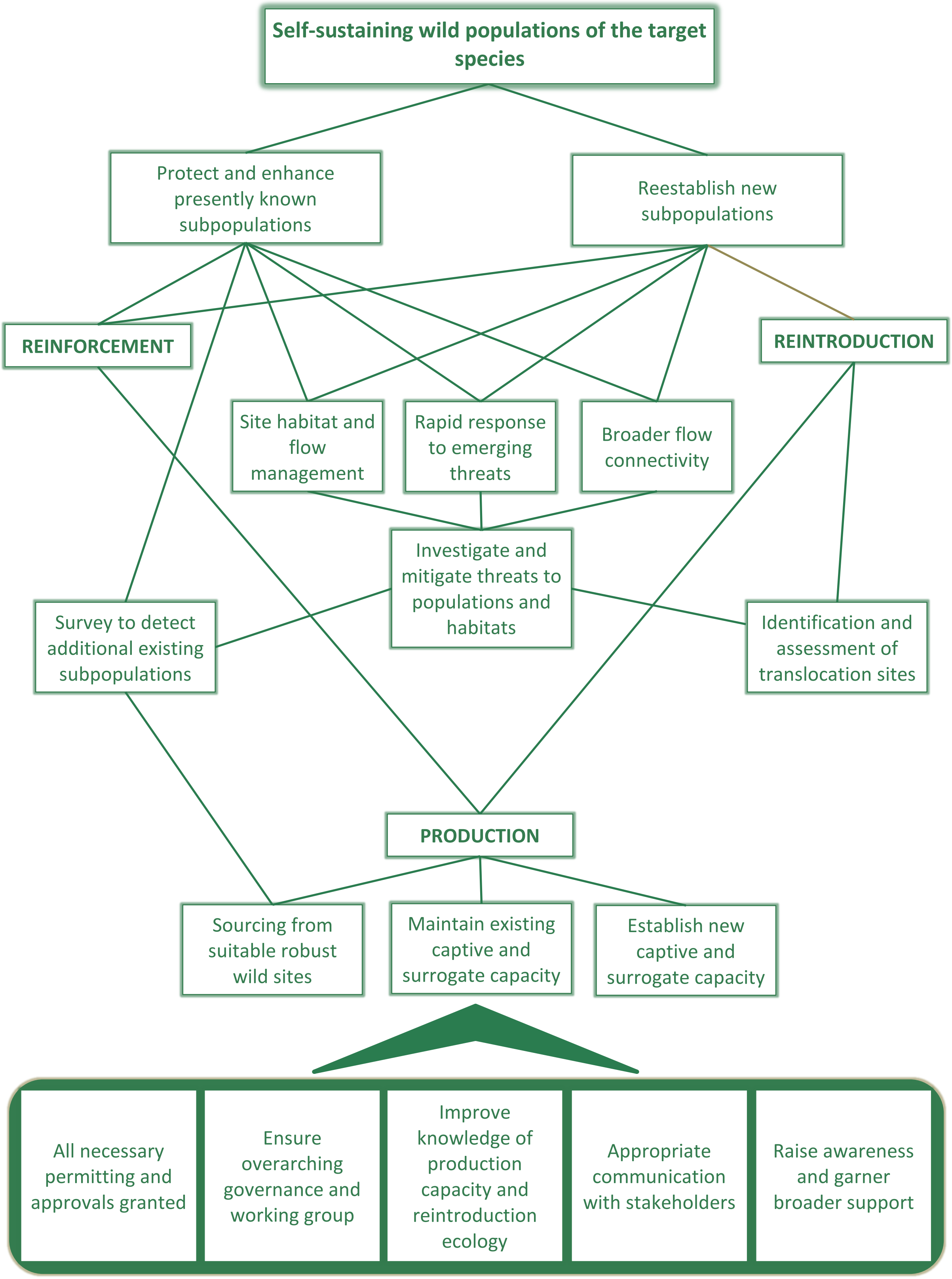


Figure 4-1. Conceptual summary of steps required to implement a translocation strategy for threatened freshwater fishes.

For a target species, if the steps outlined in Figure 4-1 (and this section) are deemed to be achievable, then translocations can be recommended. At this stage, translocation project specific planning is required. This planning includes setting translocation objectives, fulfilling permitting and approval requirements, understanding expectations of the scope required (i.e. numbers, duration), sourcing individuals and articulating robust strategies for implementation, and evaluation (Batson et al. 2015; IUCN/SSC 2013; Pérez et al. 2012). As part of implementation, it is necessary to consider the status of source populations, receiving habitats (habitat quality, resource availability and competitors/predators), genetic status (Attard et al. 2016a; Weeks et al. 2011) as well as logistics (timing, holding and transfer) and biosecurity. The success of translocation must be assessed through adequate monitoring against defined objectives.

4.2.1 Translocation objectives

There are multiple scale-dependent objectives relating to conservation translocation fish for population restoration. The overarching objective is to increase the number and status of individual subpopulations that ensure the long-term persistence of each target species (IUCN/SSC 2013). In turn, this objective relates to increasing the geographic range (i.e. extent of occurrence and area of occupancy) and improving the trend in condition of the species to link with improving the conservation status of the species. At the individual subpopulation scale, the objective is to reestablish self-sustaining individual subpopulations of the target species. There are also translocation-event objectives relating to the success of the release strategy that is implemented. During the initiation of each conservation translocation project, these objectives must be clearly defined to allow the success to be assessed and refinements to be implemented.

For each species, it remains unclear how many subpopulations are required to achieve long-term persistence: however, it is assumed to be considerably more than present. It is recommended to define species-specific targets for the number of subpopulations based on insight into present status of each target species. Additionally, the species-specific optimal release scenario, in terms of numbers and duration and type (e.g. wild-to-wild v captively produced) are largely unknown so need to be explored and then determined. Regardless, it is acknowledged that initiation of a translocation project must be combined with, at least, a minimum five-year commitment in order to maximise the likelihood of successful population establishments. Experience with large fish restocking programs demonstrates that higher numbers of fish and more frequent restocking provides a greater chance of long-term success. Thus, it is also acknowledged that substantial numbers of fish will be required, either sourced from secure wild sites or produced to allow the multi-year release of individuals across more sites.

4.2.2 Permitting and approvals

It is essential that all translocation projects align with relevant legislation and guidelines and all necessary permits and approvals must be obtained. Broadly, translocations will adhere to the *National policy guidelines for the translocation of live aquatic animals* (DAWE 2020), which is implemented in each state under separate policy. For freshwater fish in NSW the national policy is articulated in the NSW Freshwater Fish Stocking Fishery Management Strategy (FMS: DPI 2005) in accordance with the *Fisheries Management Act 1994* (FM Act). The FMS provides guidance for conservation translocation (stocking) projects in terms of the management goals and objectives and provides the management framework for future conservation stocking. It also outlines a program for monitoring the environmental, social and economic performance of the fishery, establishes trigger points for the review of the strategy, and requires regular public reporting on performance.

In NSW, assessment of the merits and risk of conservation translocation (stocking) projects is achieved through the use of a Review of Environmental Factors (REF), which is a requirement of the FMS. The REF provides an evaluation and assessment of the potential environmental risks and impacts associated with a conservation translocation project. If the REF is approved, permits are issued under the NSW Fisheries Management Act 1994 to allow the catch, possession and transport (section 37) and release/stocking (section 216) of the fish species involved. The permits cover both the collection, possession and transport of broodstock and/or progeny from captive breeding of the target species and their release (stocking) into the wild. Previously a separate REF was required for each conservation translocation project; but recent refinement ensures that a single overarching REF now covers all freshwater threatened fish recovery projects in NSW. This is a positive step forward that will provide a consistent decision–making process to streamline project approvals. Under this revised approach, each project is subject to a Threatened Species Conservation Stocking Approval. This approval ensures specific compliance via a decision-making framework and justification through a 7-part test of significance to determine whether the proposed project is likely to significantly affect threatened species, populations or ecological communities, or their habitats. Finally, the details of each translocation release must be notified through a Conservation Stocking Verification Form. This process is managed through the NSW DPI Fisheries Threatened Species Unit, and collaboration both internally and externally is necessary.

Several broadly occurring target species require strong multi-jurisdiction collaboration to achieve effective whole-of-range conservation. However, the cross-border movement of threatened fish creates administrative and implementation complexity. Thus, the development of a strategy to streamline planning (including legislation requirements), and implementation to facilitate multi-jurisdictional conservation translocations is required. This strategy will benefit from broad stakeholder engagement and communication and must be framed with whole-of-range conservation in mind. Recent translocation of Murray Hardyhead from SA to NSW has demonstrated the potential, but also highlighted aspects that need to be streamlined.

4.2.3 Genetic management

The preservation of gene flow amongst populations and genetic diversity is critical to adaptive potential and species viability (Frankham et al. 2010). Typically, species that maintain large populations across a broad range exhibit sufficient levels of gene flow and genetic diversity. Yet, for species that have declined to small and fragmented subpopulations, genetic differentiation amongst subpopulations and the loss of genetic variation and inbreeding is considered inevitable (Frankham et al. 2010; Smith et al. 2014). These subpopulations will have less ability to persist and adapt to environmental change and, are at greater risk of extinction, which in turn influences the viability of the species (Frankham 2005; Hoffmann and Parsons 1997). Translocations seek to redress genetic deterioration by mimicking gene flow through maintaining or enhancing genetic diversity, build adaptive potential and lessen extinction risk (Weeks et al. 2011).

Adaptive genetic management will be critical to the long-term survival of the target species. Whilst there is knowledge of contemporary patterns of gene flow and the genetic diversity amongst known subpopulations for some of the target species (e.g. Murray Hardyhead, Southern Pygmy Perch), for others no genetic studies have been undertaken. With knowledge of the genetic structure and diversity, an adaptive genetic management framework can accompany translocation projects for each target species. (cf. Attard et al. 2016a; Flanagan et al. 2018). The framework can help to define the scope of release strategies (i.e., numbers, frequency), scope (i.e., genetic rescue), identify source populations and assist broodstock management to ensure the translocations achieve objectives relating to genetic status.

Effective genetic management is not possible without an understanding of how genetic status changes over time (Attard et al. 2016a; Flanagan et al. 2018). Thus, genetic monitoring is vital to adaptively implement and assess the present translocation strategy in combination with information provided by population monitoring. Specifically, genetic monitoring can be used not only for assessing genetic status (e.g. genetic diversity, relatedness, population connectivity) but can also provide an indication of survival, recruitment, and abundance within the population (Attard et al. 2016a). Genetic monitoring can be equally insightful for wild, captive or surrogate populations. Thus, we believe genetic monitoring must become routine as part of the implementation of the present threatened fish conservation strategy. The transition to genomic-based monitoring will be particularly useful to provide more powerful insight into genetic status and local adaptation (Allendorf et al. 2010; Flanagan et al. 2018).

4.2.4 Accounting for climate change

This handbook was instigated as a consequence of the prolonged drought and widespread bushfires. The future climates are forecast to be warmer and drier with increased frequency and periods of extreme drought (Timbal et al. 2015). With reduced river flow volumes and less frequent flooding (Colloff et al. 2016; CSIRO 2008; Neave et al. 2015), rivers and wetlands will experience longer dry periods or be lost completely (Colloff et al. 2016). Likely increased water temperatures and decreased dissolved oxygen levels, as well as increased pollutant toxicity will further exacerbate native fish impacts as will possible increases in groundwater temperatures which could affect fish habitat quality (Ficke et al. 2007). Further, more frequent and intense bushfires are anticipated into the future (Di Virgilio et al. 2019). Considerations should also be given to human responses to climate change such as increased water diversions which will increase the effects of climate change on fish species (Ficke et al. 2007). Broadly, there is a need to adequately acknowledge the implications for future climates on freshwater fishes (Balcombe et al. 2011; Morrongiello et al. 2011) during any translocation projects.

4.3 Implementation

4.3.1 Identifying potential sites

The single most important consideration for translocations to any site is an understanding of the underlying drivers of local extirpation. Without mitigation of these drivers, translocations are unlikely to be successful in reestablishing self-sustaining populations. Broadly, many of the target species has been impacted by the consequences of river regulation through reduced overall flow volumes, altered flow regimes and frequency of flooding, as well as floodplain reclamation and levee construction. As such, many inland wetlands are nowadays permanently inundated, with others infrequently flooded, while others are permanently dry (Mallen-Cooper and Zampatti 2018; Walker 2006). This has led to a substantial reduction in habitat diversity, which along with habitat degradation (which includes loss of aquatic vegetation, poor water quality, contamination and eutrophication) and the predation and competition influence of alien species has had dire impacts on native fish populations (DELWP 2017; Hammer et al. 2009b; Saddlier and Hammer 2010).

Presently, potential surrogate and wild translocation sites for the target species are identified predominately through knowledge of former habitats and their characteristics that supported the target species immediately prior to their local disappearance (e.g. before the Millennium Drought). This approach has been logical as post-drought reestablishment was deemed most likely at these former habitats. Expert opinion has also been useful to identify additional sites that may be suitable for each of the target species and as such a workshop is proposed to identify key potential sites for each target species. Ellis and Kavanagh (2014), for instance, utilised an expert workshop to identify potential translocation sites for Murray Hardyhead across the range of the species.

Whilst this approach is suggested to be utilised here, it is acknowledged that a more quantitative approach is required to identify the number of sites required for the scale of translocations proposed under the present strategy. By way of an example, species distribution models (SDMs) can help to derive spatially explicit predictions of environmental suitability as to guide translocation strategies (Guisan et al. 2013; Malone et al. 2018). SDMs are developed using knowledge of fish distribution and environmental predictors such as landscape and river character and water quality. They allow comparison of the availability of suitable habitat under current and future climates that can inform assessment of wild populations as well as the source populations and potential release sites for translocations. Relevantly, SDMs developed for threatened fish species in NSW (de Oliveira et al. 2019; Riches et al. 2016) could provide broadscale evaluation of suitable habitats, from which more regional – or site-specific assessment could be made and are currently used for TS-MaxEnt based mapping together with expert advice.

4.3.2 Site suitability criteria

Whiterod (2019) detailed a semi-quantitative (i.e. expert opinion and on-ground data collection) two-stage site suitability criteria to assess potential translocation sites (Table 4-1). Initially, pre-assessment (stage 1) of general site suitability is made in the context of the long-term suitability of the site. This is predominately achieved as a desktop pre-assessment, which draws on the expertise of relevant stakeholders, although some of the considerations can be assessed through preliminary site inspection. Secondly, for each site differential habitat and water quality requirements of each target species is assessed (Table 4-1). Equally, the criteria are relevant to both surrogate and wild sites.

Prevailing hydrology is the predominant criterion, with assessment relating to water permanency at the site. In acknowledging the benefit of variable water levels, the criteria are linked to the persistence of refuge pools (that could support the target species) at the site as opposed to the maintenance of stable water levels. Whilst preference is given to sites that exhibit a long history of some water permanency (5–10 years) it is acknowledged that shorter periods may be suitable, particularly if suitable habitats occur nearby.

Appropriate site management is equally important, in terms of landowners and stakeholder commitment to conservation of the target species, and a willingness to manage the site in a manner that predominately benefits the target species; this commitment would ideally be articulated in a wetland management plan for the site. In terms of location, wild sites must be in the natural range of the species, with additional preference given to sites where the species had previously been abundant rather than present in low numbers. For surrogate sites, the location can be outside the natural range of the species, but it must be deemed as isolated from the catchment and relevant genetic and biosecurity issues considered. General assessment of likely habitat suitability, water quality and prevailing fish species is made at this stage.

If the pre-assessment of general site suitability is favourable, assessment moves to specific site suitability achieved through field assessment (stage 2). The assessment of overall site suitability is achieved through on-site evaluation of habitat cover, water quality parameters, food resources and prevailing fish species as well as confirmation of criteria relating to hydrology, site management and location. Habitat cover is described (by visual estimation) as the percentage of aquatic habitat cover (i.e. below the water surface) comprised of submerged vegetation, emergent vegetation, other physical structure (e.g. woody debris, rock) and open water. Water quality parameters, including water temperature, pH, dissolved oxygen concentration, electrical conductivity and water transparency, are assessed. Assessment of macroinvertebrate diversity and abundance is undertaken to investigate the presence of adequate food resources. Importantly, prevailing fish species are evaluated through targeted fish sampling, using appropriate sampling gear (e.g. fyke and seine netting) with specific focus on the presence of large-bodied predators (such as Redfin Perch, trout), small-bodied competitors (e.g. Eastern Gambusia) and other pest fish (e.g. carp) that would act to lessen the likelihood of establishment of the specific target species.

Table 4-1. Two-stage criteria for assessing suitability of surrogate and wild translocation sites.

Stage	Considerations	Requirements
Stage 1: General site suitability	Hydrology	• Water level variability • History of water permanency (preference for long history, e.g. 5–10 years)
	Site management	• Landowner/stakeholder commitment to target species conservation? • Under appropriate management regime?
	Location	• Within natural range of the species (wild) or isolated (surrogate refuge)? • Nearby potentially suitable habitats? • Suitable access to site?
	Habitat suitability	• Good levels of habitat cover (e.g. submerged and emergent vegetation, woody structure)?
	Water Quality	

	Fishes	• Suitable water quality for target species (see Table 4-3)? • Prior knowledge of prevailing fish species?
Stage 2: Specific site suitability	Hydrology Site management Location Habitat cover Water quality Food resources Fish survey	Confirm stage 1 assessment via ground truthing Confirm stage 1 assessment via ground truthing Under appropriate management regime High stable cover and submerged plants linked to species-specific requirements (see Table 4-3)? More detailed assessment at number of locations, linked to criteria in Table 4-3. Adequate availability of macroinvertebrates Prevailing fish species, with large-bodied predators as well as small-bodied competitors (see Table 4-3).

In combination, these criteria are evaluated against the species-specific tolerances and habitat preferences, which should be continually updated as new information becomes available, (Table 4-2) to provide the final assessment of overall site suitability. At this stage, a site can be recommended or rejected as a translocation site, but also identified as requiring potential management actions (e.g. habitat improvement) to improve site suitability.

Table 4-2. Summary of species-specific tolerances and preferences including percentage preferred habitat cover, key habitat preferred, water quality requirements, food resources and prevailing fish species (competitors and predators) preferences for assessing translocation sites. EC= Electrical conductivity, DO= Dissolved oxygen. Species are MHH: Murray Hardyhead; OP: Olive Perchlet; OPP: Oxleyan Pygmy Perch; RB: River Blackfish; RSG: Round-snout Galaxias; STG: Short-tail Galaxias; SPSG: Southern Purple-spotted Gudgeon; SPP: Southern Pygmy Perch; SG: Stocky Galaxias.

Target species	**Habitat cover**		**Water quality**			Food resources	**Prevailing fish species**	
	Percentage (%) physical habitat	Key habitat	EC (µScm^{-1})	DO (mgL^{-1})	pH		Competitors	Predators
MHH	>30%	Submerged (*Ruppia, Myriophyllum, otamogeton, Vallisneria*) and emergent (*Paspalum*) vegetation	400–~50,000	>2.0	4–10	Abundant foood resources appropriate for different life stages (e.g. larvae/juveniles: microcrustaceans, macroinvertebrates; advanced juveniles & adults: macroinvertebrates and small fish)	Absent or low numbers of freshwater generalists, and introduced species (such as trout species, Redfin Perch and Eastern Gambusia)	Absence or low numbers of predatory species (such as trout species, Redfin Perch, and Eastern Gambusia)
OP	>50%	Structure, submerged (*Vallisneria*, *Potamogeton*) and emergent (*Eleocharis*, *Typha*) vegetation. Preference for submerged macrophytes.	400–800	>2.0	7–9			
OPP	>50%	Shallow depressions over sandy soils, slightly acidic and tannin-stained water with gentle flow with water velocities generally below 0.4 m/sec. High abundance of structure such as emergent or submerged plants or steep undercut banks fringed by semi-submerged branches and fine rootlets from adjacent land-based trees and scrubs	<830	>6.0	4–7			
RB	>40%	Undercut banks and boulders and cover of rocks, fallen timber, snaggy areas. Submerged and emergent vegetation	200–4000	>4.0	5–9			
RSG	*Unknown*	Substrate primarily consisting of bedrock, boulder, cobble and coarse sand. Instream structure are rock, timber snags and some aquatic vegetation.	*Unknown*	*Unknown*	*Unknown*			
STG	*Unknown*	Substrate varies between the two known populations. For one population major substrate is clay and sand with some areas of silt; for the other population it is cobble, pebble gravel (steeper gradient stream). Riparian shade is present for one population but not the other. One pop is in a relatively steep gradient stream, whereas the other is much flatter. Absence of trout is a key feature for both populations.	*Unknown*	>4.0	*Unknown*			

SPSG	>30%	Submerged *(Myriophyllum, Ceratophyllum* and *Vallisneria)* and emergent *(Schoenoplectus)* vegetation	800–5000	>3.0	7–10			
SPP	>50%	Submerged *(Myriophyllum, Ceratophyllum and Vallisneria)* and emergent *(Schoenoplectus, Triglogchin, Typha)* vegetation as well as physical (rock, woody structure)	<3000	>2.0	4–10			
SG	*Unknown*	Substrate of bedrock, boulder, cobble, pebble and gravel and areas of silt deposit and structural habitat of rock and overhanging riparian vegetation. Major mesohabitats are runs and riffles (virtually no pools in this steep gradient montane stream) with steep cascades and waterfalls likely fragmenting upstream connectivity within the single population. Riparian cover varies from open snowgum/black sallee woodland to dense, stream-covering shrubs (ti-tree) to shorter heathy veg with some sphagnum. Absence of trout is a key feature.	*Unknown*	*Unknown*	*Unknown*			

4.3.3 Site enhancement

Where potential surrogate and wild sites are deemed as requiring management actions to bring them to a suitable state for translocation, certain objectives should be addressed. This can include enhancing water quality, emergent and submerged vegetation, presence of macroinvertebrates and absence of introduced or other predatory and competitor fish species. Water quality in translocation sites need to meet criteria outlined in Table 4-2.

The target species need to have suitable habitat with physical opportunities to lay eggs and have refuge from predators including predatory/competing fish species and birds as well as providing a food source from both the plants and microinvertebrates that reside in them. Habitat at a site can be improved by encouraging the growth (potentially by targeted establishment) of native submerged vegetation (such as *Myriophyllum, Ceratophyllum* and *Vallisneria*) and emergent vegetation (for example *Schoenoplectus, Triglogchin and Typha*) and through the addition of substrate such as rock and woody structure (Figure 4-1).

4.3.4 Release considerations

The practical release considerations are a critical aspect of the translocation process (Moehrenschlager and Lloyd 2016). Undoubtedly, the ability to collect, transport and then release healthy fish will influence post-release survival. Thus, the successful establishment of translocated populations. As such, efforts should be made to minimise the stress experienced by fish during the translocation process (DPI 2005; Sampaio and Freire 2016). In the following section, key considerations are discussed in the context of the translocations and release of the target species.

4.3.5 Minimising transport-related stress

Transport-related stress during live fish transport adversely impacts fish health and post-release survival (Brown and Day 2002; Sampaio and Freire 2016). Paramount to stress reduction during the transportation of fish is the maintenance of water quality parameters, as well as accounting for the accumulation of metabolic wastes (Sampaio and Freire 2016). Table 4-3 provides guiding principles to minimize transport-related stress. To minimise transport stress: 1) pure oxygen should be released into transport tanks, 2) water temperature should be maintained below species tolerances, and 3) fish should be transported in near-isosmotic water to minimise the metabolic cost of osmoregulation, thus lessening oxygen demand and waste production.

Table 4-3. Concern and suggested solutions for managing fish stress during transportation.

Aspect	Concern for fish stress	Solution
Dissolved oxygen	• Low dissolved oxygen (hypoxia) conditions increase stress	• Ensure adequate oxygen supply (preferably pure O_2) to meet oxygen demand of fish
Temperature	• Higher temperatures lead to greater oxygen demand and water production	• Transport in well-insulated tanks • Transport fish during cooler periods
Electrical conductivity	• Departure from the isosmotic point results in greater metabolic demand of osmoregulation (thus greater oxygen demand and waste production)	• Maintain transport water near isosmotic point for the target species
Metabolic waste (carbon dioxide and ammonia)	• Accumulation of metabolic wastes • Waste build-up can pose increased stress	• Utilise ammonia-reducing agents, such as Stress Coat, to mitigate the build-up of ammonia • Utilise pH buffer to achieve optimal pH
Suspended solids	• The build-up of suspended solids can influence fish stress	• Avoid feeding for 24–48 h prior to transportation • Source clean water to fill transport tanks
General	• All aspects of the transportation process can promote stress in fish	• Minimise transport time • Fish are handled as little as possible as it increases stress and oxygen demand • Avoid turbulent mixing of the transport water (from air stone or water movement) through the use of baffles and filling transport tank up completely • Use appropriately-sized transport tanks • Avoid high fish densities to avoid overcrowding • Regularly check fish and oxygen supply during transport • Monitoring stress responses of fish

Besides these water quality parameters, the complex interaction between pH and the build-up of metabolic wastes (carbon dioxide and ammonia) needs to be considered (Sampaio and

Freire 2016). The accumulation of ammonia is considered a major concern, which can be ameliorated through the addition of commercial-available ammonia-reducing agents or fasting prior to transportation. Fish (and bacterial) metabolism produces carbon dioxide, which can directly impact fish by reducing the oxygen-carrying capacity of fish blood and making them more prone to low dissolved oxygen concentrations. Carbon dioxide can also indirectly impact fish by acidifying transport water so that pH levels become lethal. During fish transportation, the build-up of carbon dioxide is typically gradual, but pH decline can be rapid. To combat carbon dioxide build-up, a combination of adequate oxygen supply and ventilation (to allow carbon dioxide to dissipate) is needed. Buffers can be used to control pH levels in the transport water. Lastly, in acknowledgement that fish transport is an inherently stressful process, a range of general solutions, such as minimizing overall transport time and handling, is recommended. Various options exist to transport fish, from small tanks (60 and 120L) to transportation tanks to avoid overcrowding (Figure 4-2).

Figure 4-2. Various scales of fish transport options that be explored for transportation of fish as part of the present strategy.

The mitigation of transport-related stress will require ongoing review and evaluation of the translocation process. This will be achieved through trial-and-error, discussion with colleagues and periodic review of the scientific literature. It will also require a greater understanding of changes in water quality and metabolic wastes as well as physiological stress in transported fish. As such, it is recommended that comprehensive monitoring of water quality and stress responses becomes routine during the transportation of fish. This should incorporate real-time monitoring of key water quality (dissolved oxygen, temperature, pH) and metabolic waste parameters. Equally, thresholds for physiological markers of stress, such as cortisol and blood glucose, should be established for each target species, which can allow for assessment of release considerations that act to lessen transport-related stress. Considerations such as the size of fish and size grouping should be considered to decrease possible aggression and predation during transport.

4.3.6 Release considerations

Upon arrival at translocation site, transport water should be gradually mixed with water from the translocation site to equilibrate water quality (namely water temperature and electrical conductivity). Once satisfied with water quality equilibration, the condition of fish should be assessed by visual inspection (with release not to proceed if fish considered unhealthy), and then fish released in a manner appropriate for each targeted species. While release of fish in larger groups is appropriate for a schooling species such as Murray Hardyhead, release in small groups is more effective for other species, such as Southern Purple-spotted Gudgeon (Figure 4-3).

Figure 4- 3. Fish release approaches for Southern Purple-spotted Gudgeon (left) and Murray Hardyhead (right).

A combination of direct release and soft release methods can be utilised. Direct release simply involves the direct liberation of fish at the release site following a period of onsite acclimation (to prevailing water). In contrast, soft release allows for a period of acclimatisation to the prevailing conditions, so that fish become accustomed to the prevailing conditions and develop accompanying natural behaviour that are likely to elicit a greater survival rate. Soft-release enclosures have been utilised successfully in previous reintroductions in the Lower Lakes; they should be sufficiently large (>1 m x 1 m), clad with small mesh (4 mm) (Figure 4-4) (Bice et al. 2014). Prior to releases, all soft release enclosures should be sampled by dip net to eliminate other fish species allowing a subsequent recovery period from netting disturbance (i.e. disturbed sediment/silt). A period of 24 hours has been chosen to allow for adequate recovery from transportation and acclimation, whilst limiting density-dependent negative impacts from holding fish for longer periods (e.g. aggression and limited dispersal) (Brown and Day 2002).

Figure 4-4. Utilisation of soft-release enclosures for the translocation of the target species.

4.3.7 Biosecurity and disease

Disease is an important consideration when reintroducing endangered species back into the wild. Not only is disease capable of nullifying the potential benefits of captive breeding programs, it can also have deleterious effects on wild populations (Viggers et al. 1993). Ongoing inspection of fish to be released is required and fish presenting poor health should be quarantined and treated, and any suspected disease reported to the Aquatic Biosecurity section of NSW DPI. Previously, fish were taken from captive and surrogate populations, held for up to three weeks to monitor health before being transported and released at the translocation site. Over time, this approach was streamlined to reduce holding and transport time (and stress) whilst permitting a greater number of translocations to take place. This approach should continue in the future as it is deemed most appropriate for threatened species where prior approval assessment has concluded there is a low or negligible biosecurity risk.

4.3.8 Timing

Generally, translocations should be undertaken in (1) spring/early summer and (2) late summer/autumn to maximise the number of fish released and account for the greatest range of conditions that will be experienced. During spring to early summer, increased food abundance and habitat availability (e.g. growth of aquatic plants) will allow fish to establish (and grow if released as juveniles) before summer, whereas individuals released in early autumn will have sufficient time to establish at the site prior to winter. Typically, releasing fish in winter or mid-summer is not recommended due to the likelihood of extreme conditions (e.g. flooding and high flows, low water levels) impacting on potential success although it may be appropriate in certain situations for certain species.

4.4 Monitoring and evaluation

Ongoing monitoring of the translocated populations of the target species is critical to document presence, distribution and abundance, and to examine population demographics to allow for regular status assessments (Bice et al. 2014; Saddlier et al. 2013). It is important to conduct monitoring both at the release sites and at several of the originally selected reintroduction sites to detect any recolonisation occurring as a result of the dispersal of released individuals (Bice et al. 2014). In this approach, three monitoring levels are proposed. Seasonal monitoring at reintroduction sites is necessary over the duration of the translocation strategy (i.e. repeat translocations over several years) to confirm short-term survival (Level 1; see below). Once fish are established, monitoring can subsequently become annual to assess ongoing survival and recruitment as part of broader condition monitoring across the region (Level 2). Statistically robust pre-translocation baselines and repeat monitoring every 5 years can determine the long-term success of the strategy for each species as related to the objective (Level 3).

Monitoring outcomes will provide an improved understanding of the factors driving the presence, abundance and recolonisation of the threatened fish. Consequently, monitoring will improve the opportunities to successfully establish populations through translocations by better understanding the needs of each fish species. For example, monitoring may identify the factors that were responsible for failure of a released fish species to establish at a site. Therefore, findings from the monitoring may also trigger targeted actions at reintroduction sites to assist populations that have been translocated (e.g. environmental watering, predator removal, habitat enhancement). The long term monitoring (Level 3), using replicate surveys at multiple sites, will be ideal for inferring patterns and dynamics of threatened fish occurrences related to environmental variables, including water levels, water quality, and predator abundances (Mackenzie et al. 2018). There may be an opportunity to combine the strategy's monitoring with other long-term monitoring, should they continue in the future, so methods must be consistent.

4.4.1 Level 1: site-based seasonal monitoring

Aim: Determine the immediate success or failure of reintroductions and understand the factors that cause discrepancies.

Seasonal monitoring conducted at the reintroduction site will confirm the short-term survival of reintroduced fish by measuring abundance (total number in catch), breeding condition, general health (e.g. parasites visible) and conditions that may affect fish numbers (e.g. water quality parameters, water depth, predator abundances). The findings will determine if there are any continuing or new threats to the fish that may be addressed. The seasonal monitoring will also determine if follow up reintroductions are required in the same season (i.e. if initial reintroduction appears unsuccessful and cause has abated).

4.4.2 Level 2: site-based annual monitoring

Aim: Determine if the fish species has established a self-sustaining population at the reintroduction sites.

Apart from measuring the same factors in level 1 monitoring, the annual monitoring approach will determine if released individuals have bred at the reintroduction sites and, if so, assess recruitment based on population size structure by measuring the total length of all threatened fish. Therefore, the annual monitoring should be conducted between February and April, at the end of the breeding–recruitment period, for each of the target species. Annual monitoring conducted at the reintroduction sites will also assess the ongoing survival of reintroduced fish (possibly excluding Murray Hardyhead which lives for only 12–18 months). Other current, ongoing monitoring programs may cover some of the future reintroduction sites in this manner (Wedderburn and Barnes 2018), so data sharing may be applicable in some cases.

4.4.3 Level 3: regional occupancy estimation (long-term)

Aim: Determine changes in occupancy and range of the fish species to examine the overall success of the translocation strategy over a decade (resilient, connected populations).

Broader spatial scale surveys are required to determine the long-term success of the translocation strategy at 5-year intervals for at least a decade. These surveys will provide an estimate of occupancy, which is the proportion of habitat (sites) occupied within the species potential range. The broad surveys, covering reintroduction sites and other sites that the species could potentially colonise naturally, must be replicated within a short period of time during the monitoring to account for false absences (probability of detection: Mackenzie et al. 2018). Based on a previous study of three of the target species, imperfect detection may be accounted for by conducting three replicate surveys (fyke nets: see below) for Southern Pygmy Perch, and four replicate surveys for Murray Hardyhead (seine, or fyke nets and seine) (Wedderburn 2018). Data for Southern Purple-spotted Gudgeon is lacking. Initially, three replicate surveys using fyke nets would provide adequate information, as well as the potential addition of eDNA, and these methods could be modified if necessary.

Ideally, this level of monitoring would include a comprehensive baseline survey prior to the commencement of the translocation strategy so that the objective can be assessed by tracking the extent of occupancy for each species from the beginning of the program. This approach, using a baseline survey, provides a statistically robust method of determining any long-term changes in occupancy of the fish species and, just importantly, the reasons for any changes. For example, an increase in occupancy (i.e. establishment at reintroduction sites and additional sites) may be significantly related to rates of river or stream flows, or water levels or quality. The assessment may also be used to determine the success of habitat enhancement efforts.

4.4.4 Evaluation criteria

The expected outcomes for the nine target species are framed in terms of restoring distribution and abundance to levels recorded prior to 2007, before major population declines and extirpations were caused by extreme drought and other threats. This includes the expansion of existing populations (e.g. range extension) and/or the establishment of new populations (e.g. additional populations), which may be facilitated through translocations. Over a decade, this is articulated as expanding the range of each species and establishing 3–4 additional locations (sites) for each of the target species, which can be evaluated from the level 3 monitoring data (i.e. occupancy estimation).

For example, in broader terms, the national recovery plan for Murray Hardyhead details recovery objectives relating to the protection and maintenance of key presently known populations (i.e. primary populations) as well as identifying and undertaking translocations to establish secondary populations to increase area of occupancy (DELWP 2017; MDBA 2014; Saddlier and Hammer 2010). For Murray Hardyhead, it is recommended to establish three secondary populations (one for each genetic management unit). Recovery plans emphasise the importance of establishing surrogate and captive populations. The findings of population monitoring may also be used to evaluate state and federal government objectives within NSW and the Murray–Darling Basin. For example, assuring that key species show improved length structure and movement, and expanded distribution – an objective of the Basin Plan and associated Basin-wide environmental watering strategy (MDBA 2014).

5. THE WAY FORWARD

5.1 Summary

Across Australia, many threatened freshwater fish face the risk of extinction in the medium-term future without substantial intervention. For instance, Lintermans et al*.* (2020) identified 20 fish species with a greater than 50% probability of extinction in the next ~20 years. In the MDB, almost half of the fish species are listed as threatened under national and state legislation (MDBA 2020), and the first freshwater fish extirpation has already been documented (Wedderburn et al. 2019). In NSW, nine target freshwater fishes (that are the focus on this handbook) are either absent or persisting as small, fragmented populations across contracted parts of their historical range. It is clear that wide-ranging recovery actions are required to redress these declines and affect recovery of threatened freshwater fishes in NSW and the MDB (Koehn et al. 2020a). In some cases, well planned and implemented conservation translocations are essential to compliment other recovery actions.

The present conservation translocation handbook provides a platform to guide the conservation translocation projects of the nine threatened freshwater fish in NSW. For each species, the present status (wild populations, biological information, genetic management, known threats and knowledge gaps) is provided along with principles to guide ex situ maintenance and production. It is a comprehensive and informed handbook that has benefited from extensive consultation with key stakeholders, including hatchery mangers, private operations, fisheries and conservation managers and scientists. The handbook should not be viewed as standalone, rather as a supporting document that aligns with the objectives of species-specific (e.g., national recovery plans) and fish-specific planning documents (e.g., The NSW Freshwater Fish Stocking Fishery Management Strategy, The Native Fish Recovery Strategy) as well as other strategies (e.g., Basin-wide environmental watering strategy). It should be viewed as a 'live' handbook that is routinely updated as new knowledge is gained.

5.2 Recommendations and priority actions

Recent broad guidance has been provided on the management actions needed to restore native fish in the MDB (Koehn et al. 2020a; MDBA 2020). The Native Fish Recovery Strategy, for instance, outlines five actions (across four outcomes) necessary to achieve four outcomes of (1) recovery and persistence of native fish; (2) identify and mitigate threats to native fish; (3) communities are actively involved in native fish recovery; and (4) recovery actions are informed by the best available knowledge (MDBA 2020). Similarly, Lintermans et al. (2020) provide nine recommendations to avert the extinction of threatened fish species. Koehn et al. (2020a) recommend 30 priority actions, relating to flow management, infrastructure and other restoration, and support and engagement, which are deemed as critical to 'providing a legacy of native fish recovery in the MDB, rather than extinctions'.

A range of recommendations and priority actions have resulted from the present handbook, which strongly align with those provided by Lintermans et al. (2020), Koehn et al. (2020a) and MDBA (2020). The recommendations and priority actions relate to undertaking long-term conservation actions, namely conservation translocations, in a strategic, effective and appropriately resourced manner that considers the whole-of-range of each target threatened species. There is a need for species-specific planning in terms of threatened fish production and translocations, as well as regular updating of the status of wild (remnant and translocated) subpopulations and captive fish. The recommendations and priority actions are summarised below:

- Acknowledge long-term commitment necessary to achieve threatened fish recovery
  - ACTION: commit sufficient multi-year resources
  - ACTION: ensure each translocation event has resources for a five-year period
- Develop whole-of-range strategies to achieve appropriate scale of actions
  - ACTION: engage relevant stakeholders to develop whole-of-range plans
  - ACTION: seek out opportunities for better inclusion of target species in natural resource management plans
  - ACTION: establish multi-jurisdictional working groups for appropriate target species
- Ensure legislative requirements do not impede ability to implement recovery actions
  - ACTION: continue with revised Statewide NSW approval process (e.g., overarching REF and site specific approvals)
  - ACTION: streamline multi-jurisdictional (where necessary) permit and approval processes
  - ACTION: undertake a review of the NSW Freshwater Fish Stocking Fishery Management Strategy (NSW DPI 2005) to ensure that it is appropriately supporting conservation outcomes for threatened fish in NSW

- Evaluation of the species-specific feasibility and scope of translocations
  - ACTION: development of translocation plan for each target species
  - ACTION: determine necessary scope (number of reintroduction sites, numbers of fish to release) for each target species
  - ACTION: identify, assess, and prioritise translocation sites for each target species
- Ensure captive maintenance and breeding is appropriate for each target species
  - ACTION: produce production manual for each target species
  - ACTION: establish network of those producing fish of each target species
- Ensure conservation translocations are guided by appropriate genetic management
  - ACTION: establish and maintain genetic status of known and captive subpopulations
  - ACTION: Development and implementation of adaptive genetic framework for each target species
- Routinely obtain information on status of each target species
  - ACTION: targeted monitoring field surveys (and consolidate with information from non-target surveys) to determine status of known wild subpopulations
  - ACTION: targeted field surveys of translocated subpopulations to determine status and identify necessary follow-up actions
  - ACTION: conduct research into aspects of the biology and ecology of each target species to address knowledge gaps (including through Fisheries Scientific Committee Student Research Grants)
- Timely compilation of new knowledge as it becomes available
  - ACTION: biennial revision of this handbook (next revision in 2023)

5.3 Conclusions

Each of the nine target threatened species face an uncertain future in NSW and urgent conservation actions are required. Conservation translocations will, in many cases, represent a necessary action for each target species. As such, the present conservation translocation handbook provides a platform to guide the conservation translocations projects of the nine threatened freshwater fish in NSW. Comprehensive and sustained actions are required in relation to conservation translocation (and conservation more generally) of each of the target species. Without these actions, some of the target species will undoubtedly be lost to NSW in the immediate future. Echoing the sentiments of Koehn et al. (2020a), it is hoped that this handbook assists with the process of recovering of each of the target species before it's too late.

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