

Spangled perch (*Leiopotherapon unicolor*) in the southern Murray-Darling Basin: Flood dispersal and short-term persistence outside its core range

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Abstract The spangled perch *Leiopotherapon unicolor* is considered a rare vagrant in the southern Murray-Darling Basin, Australia, due to its intolerance of the relatively cool water temperatures that prevail during winter months. This study details 1342 records of the species from 68 locations between 2010 and 2014 outside its accepted ‘core adult range’ following widespread flooding during 2010 and 2011. Although records of the species declined over 2013, *L. unicolor* remained resident in the southern Murray-Darling Basin as of April 2014. The species persisted in several locations for three consecutive winters with recruitment documented at two sites. This study represents the first identification of the dispersal of large numbers of *L. unicolor* into the southern Murray-Darling Basin, persistence beyond a single winter, and recruitment by the species in habitats south of its recognized ‘core adult range’. Targeted research would determine the potential for predicted environmental changes (artificially warmer drainage wetlands, climate change and greater floodplain connectivity) to facilitate longer term persistence and range expansion by the species in the southern Murray-Darling Basin.

Key words: flood dispersal, range expansion, spangled perch, core range, recruitment.

INTRODUCTION

Fishes are considered to be spatially constrained to a core range, but with the potential to disperse to explore new habitats and resources (Radinger & Wolter 2014). Dispersal forms a critical part of the life strategies of many freshwater fishes, influencing population dynamics, gene flow and access to habitat, food and breeding opportunities (Koehn & Crook 2013; Radinger & Wolter 2014). Patterns of dispersal vary spatially and temporally between species or life stages of fishes, incorporating active swimming migrations and/or passive dispersal of eggs or early life stages drifting with a current (Koehn & Crook 2013). Many Australian fishes disperse to exploit the boom–bust nature of ephemeral aquatic environments, with periodic contractions to isolated refuges (Puckridge *et al.* 1998; Bunn *et al.* 2006; Balcombe & Arthington 2009). Inter-annual variation in rainfall therefore strongly influences patterns of habitat persistence and thus persistence by fish (Unmack 2001; Arthington *et al.* 2005; Bunn *et al.* 2006; Booth *et al.* 2011). Although decreasing inflows, increased water abstraction and river regulation in the Murray-Darling Basin

(MDB) are likely to alter hydrological connectivity and refuge habitat persistence (Huey *et al.* 2011), one of the expected impacts of warming climate (Chessman 2013) is a southward expansion in the range of organisms.

One of Australia’s most widespread native freshwater fishes, the spangled perch *Leiopotherapon unicolor* (Günther, 1859) occurs in freshwater systems across much of northern Australia, and the western and northern sections of the MDB. Using the layered model of species distribution proposed by Booth *et al.* (2011), the accepted ‘core adult range’ of *L. unicolor* extends south to the Menindee Lakes on the Darling River (Fig. 1) (Schiller *et al.* 1997; Pusey *et al.* 2004; Lintermans 2007). Prior to 2010, only four records of the species had been documented in the southern MDB downstream of the Menindee Lakes, coinciding with extensive flooding events in the Darling River that would have actively dispersed individuals downstream (Lintermans 2007). To date, there is no documented evidence that *L. unicolor* has persisted for extended periods (i.e. beyond the subsequent winter) in the southern MDB following such dispersal events. This most likely reflects an inability to tolerate the relatively cool water temperatures that prevail over winter months, with reduced survivorship below 7.2°C (Llewellyn 1973). Therefore, the species is considered

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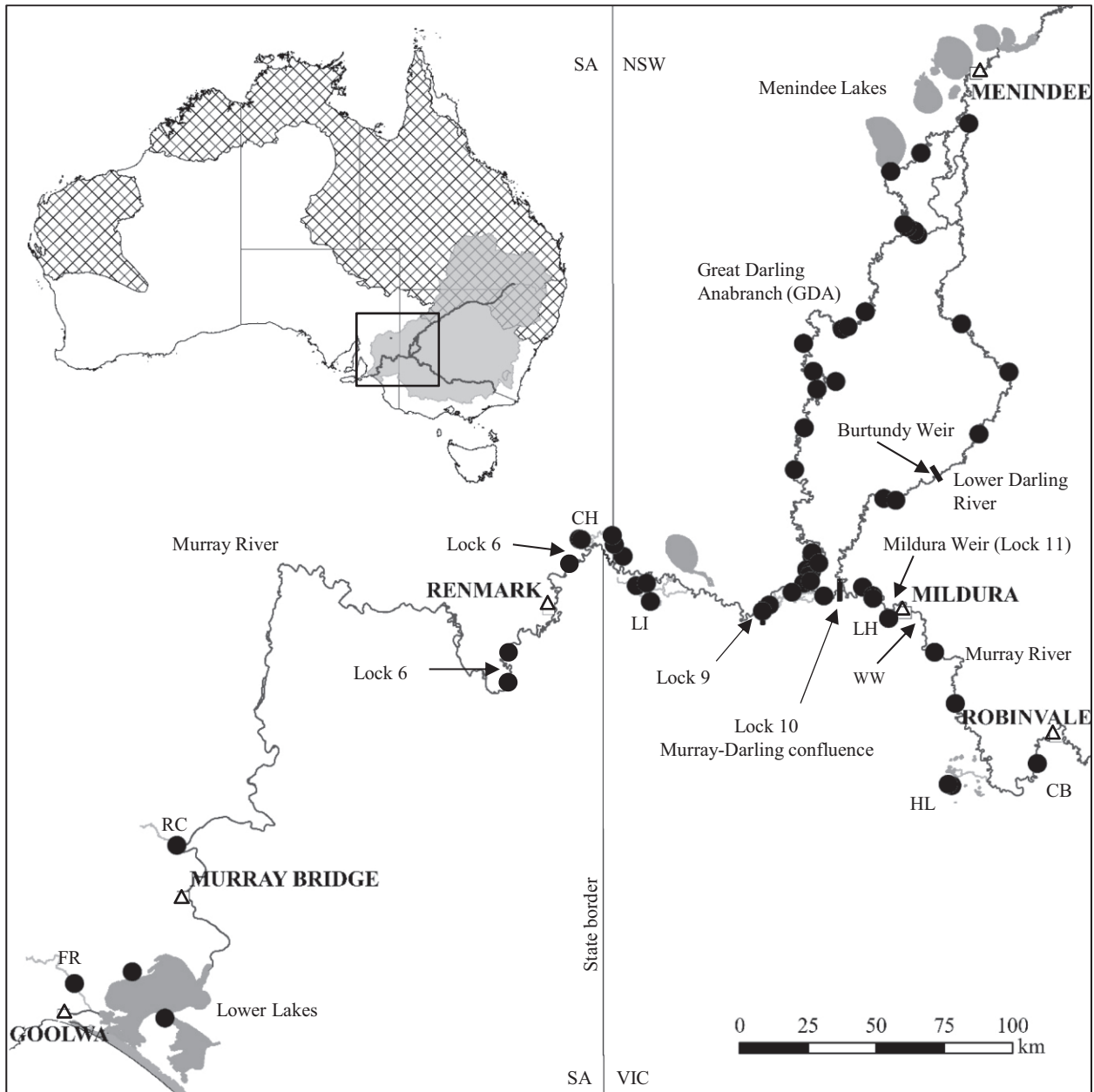


Fig. 1. The location of recent records of the species across the southern Murray-Darling Basin (MDB) (black dots). Specific locations referred to in the text are Finnis River (FR), Reedy Creek (RC), Chowilla Floodplain (CH) and Lindsay Island (LI) floodplains, Lake Hawthorn (LH), Woorlong Wetland (WW), Hattah Lakes (HL), and Carina Bend (CB). Inset figure shows the currently recognized 'core adult range' of *Leipothorapon unicolor* across Australia (hatched area) (based on Pusey *et al.* 2004) with the Murray-Darling Basin shown in light grey.

a rare migrant in the southern MDB, which represents 'non-breeding adult range' (see Booth *et al.* 2011).

During 2010 and 2011, high rainfall ended a decadal drought and generated widespread flooding across the MDB (Leblanc *et al.* 2012; van Dijk *et al.* 2013). Flood events of similar or greater magnitude occurred in the Darling River every 10–15 years in recent history (see Fig. 2). In this study, we aimed to assess the extent and significance of dispersal of the

species into and throughout the southern MDB with these floodwaters (i.e. outside of the species 'core adult range') and to investigate instances of extended persistence (>1 year) and recruitment by these migrants.

Relevant ecology of the species

Leipothorapon unicolor is a hardy species, surviving in a diverse range of habitats including flowing rivers,

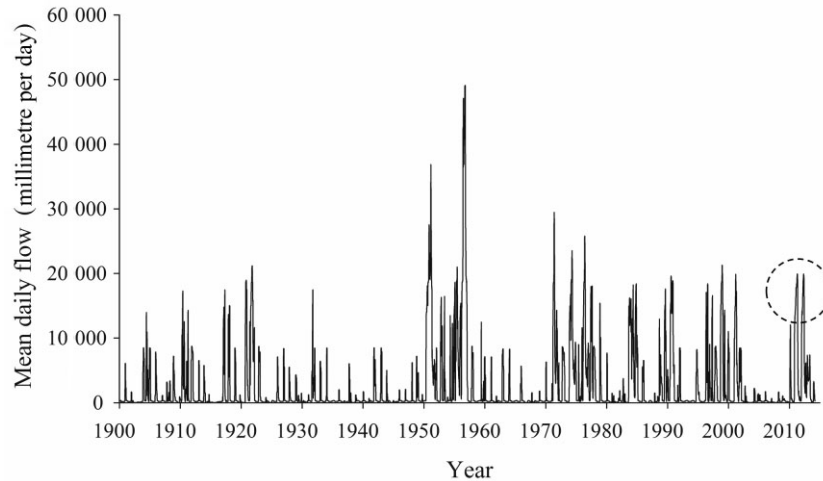


Fig. 2. Simulated daily flow (megalitre per day) in the lower Darling River (Burtundy, 425007) between 1900 and 2014, with the recent flood events in which *Leiopotherapon unicolor* dispersed southward indicated by a dashed circle (data courtesy of Murray-Darling Basin Authority).

lakes, wetlands, temporary streams and constructed drains (Merrick 1996; Lintermans 2007). It has broad tolerances, occurring in salinities range of 0.2–35.5 g L⁻¹, pH between 4.0 and 10.2 and hypoxic conditions (dissolved oxygen <2 mg L⁻¹) (Llewellyn 1973; Merrick 1996; Pusey *et al.* 2004). The species tolerates temperatures between 5.3°C and 39°C, with a reduction in survival below 7.2°C (Llewellyn 1973).

Given the wide geographic range of the species and its ability to spawn in impoundments and river-floodplain systems, *L. unicolor* exhibits some plasticity in its life cycle (Llewellyn 1973). A study of captive fishes suggests that spawning generally occurs in lentic water bodies during spring and summer wet seasons at temperatures > 20°C (Llewellyn 1973). Although flooding is not a requirement for spawning, recruitment is greater in years with significant flooding due to increased habitat and feeding opportunities (Pusey *et al.* 2004; Balcombe & Arthington 2009; Kerezszy *et al.* 2013; Balcombe *et al.* 2014). *Leiopotherapon unicolor* typically live for 2–4 years and reach 330 mm total length (TL) (Pusey *et al.* 2004). It grows rapidly with juveniles attaining a length of 25 mm TL in 40–60 days, and sexual maturity is achieved in the first year (at 58 mm TL males, 78 mm TL females) (Llewellyn 1973). The species exhibits pronounced ontogenetic dietary shifts; juveniles are meiophagous omnivores consuming aquatic invertebrates and plant materials, whereas the diet of adults consists predominately of fishes and macrocrustaceans (Gehrke 1988; Davis *et al.* 2011).

Leiopotherapon unicolor is capable of rapid and extensive movement upstream, downstream and laterally at rates of at least 9.4 km day⁻¹ during flooding (Bishop *et al.* 1995; Pusey *et al.* 2004). Kerezszy *et al.* (2013) characterize *L. unicolor* as an ‘extreme dispersing

species’ capable of moving over 300 km following opening of dispersal pathways such as flood connection between isolated waterhole refuges. Individuals are reported to move overland in very shallow temporary surface water, and the appearance of *L. unicolor* in isolated water holes after rain events offers testament to its dispersal capabilities (Merrick 1996; Lintermans 2007, I. Ellis, MDFRC, *unpublished data* 2011). *Leiopotherapon unicolor* exhibits remarkably little genetic variation across its distribution, further highlighting the vagility and adaptability of the species, (Bostock *et al.* 2006).

METHODS

Between October 2010 and April 2014, incidental records of *L. unicolor* in the southern MDB (downstream of the Menindee Lakes) were collected during monitoring programmes conducted by a number of research organizations including: Fish Communities of the Lower Murray Darling Catchment: Status and Trends (New South Wales Department of Primary Industries) (Gilligan 2012); The Darling Anabranch Adaptive Management Monitoring Program (NSW Office of Environment and Heritage) (Bogenhuber *et al.* 2013); The Living Murray Condition Monitoring at Lindsay, Mulcra and Wallpolla Islands and Hattah Lakes (Mallee Catchment Management Authority, the Victorian Department of Environment and Primary Industries and the Murray-Darling Basin Authority) (Sharpe & Vilizzi 2011; Wood *et al.* 2011; Henderson *et al.* 2014); Chowilla Icon Site – fish assemblage condition monitoring (the Murray-Darling Basin Authority through the Department of Water Land and Biodiversity Conservation and South Australian Murray-Darling Basin NRM Board) (Leigh *et al.* 2012); Condition Monitoring of Threatened Fish Species at Lake Alexandrina and Lake Albert (Adelaide University, Murray-Darling Basin Authority and the South Australian Department for Water)

Table 1. Summary of location and capture records of *Leiopotherapon unicolor* in the southern MDB between 2010 and 2014

Region	No. of locations	No. of records	Date range	Collectors
Darling River	12	254	May 2011 to August 2012	NSW DPI
Great Darling Anabranch (GDA)	27	963	October 2010 to January 2014	MDFRC, NSW DPI
Murray River (upstream junction)	11	97	March 2011 to November 2012	MDFRC, NSW DPI
Murray River (downstream junction)	10	18	March 2011 to April 2014	DEWNR, MDFRC, NSW DPI
Lower Murray (South Australia)	8	10	February 2011 to March 2014	Adelaide University, DEWNR, SARDI, private fisherman

DEWNR, Department of Environment, Water and Natural Resources; MDFRC, Murray-Darling Freshwater Research Centre; NSW DPI, NSW Department of Primary Industries; SARDI, South Australian Research and Development Institute.

(Wedderburn & Barnes 2012; Wedderburn & Sutor 2012); and Monitoring programme for Murray hardyhead in the Victorian Mallee (Mallee Catchment Management Authority and the Victorian Department of Environment and Primary Industries) (Ellis 2013).

The monitoring data utilized were collected through surveys designed to sample the fish community using (predominantly) fyke netting and/or boat electrofishing techniques along with verified records reported by recreational fishers and were compiled in the present study (see Supplementary information). Notably, the species was detected regularly and routinely post-flooding throughout the southern MDB in both electrofishing and fyke-netting assessments. For all collected *L. unicolor*, the geographical region, date and location of capture, habitat type (creek, riverine or wetland), sampling method, number captured and length (standard length (SL) and/or TL) were recorded (see Supplementary information). Additionally, we refer to the proportion of total catch in the lower Darling River showing symptoms of disease or parasites such as anchor worm, or ulcers, lesions, fin rot or emaciation as reported by Gilligan (2012, 2013). In total, the compiled *L. unicolor* data set consisted of 1342 records (ranging 10–230 mm TL) collected from 68 locations, up to 1000 km outside of the species recognized range (Fig. 1 and Table 1).

Simulated daily flow (megalitre per day, 1900 to 2014) and surface water temperature (°C, 1996–2014) is presented for the lower Darling River (at Burtundy) with surface water temperature also provided for the Murray River upstream of the confluence with the Darling River (at Mildura Weir) and a representative irrigation drainage outlet (Woorlong Wetland near Mildura) over this time (courtesy: Murray-Darling Basin Authority and Victorian Department of Environment and Primary Industries).

RESULTS

Historical records in the southern MDB

Major floods are historically common in lower Darling River (e.g. 1950s, 1970s, mid-1980s; 1990s, 2000s and most recently 2010–2011) (Thoms *et al.* 2000) (Fig. 2). However, only four verified records of individual *L. unicolor* have been documented in the MDB south of Menindee prior to the present study (see

Gehrke & Harris 2001; Lintermans 2007). One of these was from the Edward River near Deniliquin in 1887, whereas two were from the lower lakes near the terminus of the Murray River in 1970 and 1976 (Atlas of Living Australia 2014; M. Lintermans, personal communication, 2014). The fourth was collected in the Murrumbidgee River below Balranald during intensive and repeated surveys in the NSW Rivers Survey (Schiller *et al.* 1997). The 1970, 1976 and 1997 records were all collected post-flooding in the Darling River.

Leiopotherapon unicolor was absent in other documented historical surveys in the southern MDB. In 1949–1950 Langtry undertook extensive surveys throughout the Murray River and its floodplain system from the South Australian Border to Yarrawonga Weir, capturing no *L. unicolor* (Cadwallader 1977). Whereas Darling River flows had been low in the years preceding Langtry's surveys (reducing the potential for downstream dispersal), survey and museum collections data compiled by Llewellyn (1983) from 1960 to 1976, a period during which the Darling River had experienced major flooding, also reported an absence of *L. unicolor* records south of the Menindee Lakes.

The species was also absent in surveys conducted in more recent decades, which incorporated greater sampling frequency and effort. Between 1998 and 2010, the species was not detected during targeted fish monitoring in the lower Darling River and Great Darling Anabranch (GDA) (Gehrke & McLean 1988; Scholz *et al.* 2003; Gilligan 2005; Wallace *et al.* 2008; Gilligan 2013) or Murray River regions (Douglas *et al.* 1998; Leigh *et al.* 2010; Sharpe & Vilizzi 2011; Gilligan 2013). Annual fish community monitoring each winter from 2001 to 2007, and each autumn from 2008 to 2010 across a range of aquatic habitats on the Lindsay Island floodplain (immediately downstream of the Darling River confluence with the Murray River) did not detect any *L. unicolor* (Vilizzi *et al.* 2007; Sharpe & Vilizzi 2011). Basin and statewide assessments such as the NSW Rivers Survey and the Sustainable Rivers Audit also failed to detect the species south of the

Menindee Lakes (Harris & Gehrke 1997; Davies *et al.* 2008).

Flood dispersal (2010–2012)

Leiopotherapon unicolor rapidly dispersed southward with floodwaters that originated in the northern rivers of the Darling River system in late 2010. These floodwaters combined with floodwaters from the upper Murray River to create widespread flooding over much of the southern MDB. By October 2010, individuals were recorded in the lower Darling River and GDA downstream of Menindee Lakes (Fig. 1 and Table 1) (Gilligan 2012; Bogenhuber *et al.* 2013). *Leiopotherapon unicolor* was subsequently detected in the Murray River and its associated floodplain downstream of the confluence of the Murray and Darling Rivers. Most records were in the vicinity of the Murray River's confluence with the GDA and Darling River (i.e. between Lock 9 and 10), but records were also documented near the Victoria–South Australian state border (i.e. the Lindsay Island–Chowilla floodplain) (Sharpe & Vilizzi 2011; Wood *et al.* 2011; Henderson *et al.* 2014).

The earliest of these recent records from the Murray River system was in February 2011 at Lake Littra on the Chowilla floodplain (Lenon & Sutor 2012), highlighting that downstream dispersal of *L. unicolor* was rapid. As the floodwaters progressed, *L. unicolor* was dispersed further downstream throughout the lower Murray River with four individuals captured between Locks 3 and 6 later in 2011 (Leigh *et al.* 2012). The species reached Reedy Creek wetland (688 river km downstream of the confluence of the Murray and Darling Rivers) by November 2011 (Wedderburn & Sutor 2012), and the Lower Lakes (approximately 880 river km downstream of the confluence of the Murray and Darling Rivers) by March 2012 (Wedderburn & Barnes 2012). The detection of the species in the Finnis River near Goolwa in September 2013, a tributary stream of the Lower Lakes, is also the most downstream record (see Fig. 1).

Although the passive downstream transport of fishes during flooding is unsurprising, active upstream dispersal of *L. unicolor* in the Murray River upstream of the confluence of the Murray and Darling Rivers was also detected after the 2010–2011 flooding. The species was recorded in several floodplain wetlands near Mildura throughout 2011 (approximately 40 river kilometre upstream of the Murray–Darling confluence). *Leiopotherapon unicolor* was also recorded at the Hattah Lakes in March 2011 (Little Lake Hattah) and again in March 2012 (Lake Bulla) (Wood *et al.* 2011). Both of these lakes filled from the Murray River approximately 216 river kilometre upstream of its confluence with the Darling

River. Finally, the most upstream record after the 2010–2011 flood was of an individual fish from the Murray River at Carina Bend near Robinvale, some 270 river kilometre upstream of the Murray–Darling confluence (Gilligan 2012).

Persistence and reproduction

Whereas individual observations of *L. unicolor* from the southern MDB may simply indicate vagrancy after flood dispersal, the persistence of 'groups' of individuals (i.e. multiple fishes) with demonstrated spawning and recruitment in locations outside of its recognized range has not been documented prior to the present study. A broad size range (10–230 mm TL) of *L. unicolor* was recorded in the GDA and lower Darling River between 2010 and 2014 (see Supplementary information) (Bogenhuber *et al.* 2013; D. Bogenhuber, MDFRC, unpublished data, 2014). Furthermore, in December 2013 meta-larval *L. unicolor* ($n = 13$, 10–16 mm TL) was collected from the southern reaches of the GDA close to its terminus at the Murray River. These individuals are estimated to be <40 days old based on growth curves of Llewellyn (1973). As water released from the Menindee Lakes took approximately 75 days to reach the sampled location (D. Bogenhuber, MDFRC, unpublished data, 2014), we conclude that these fishes were spawned in the southern reaches of the GDA (potentially in the Anabranch Lakes) or in the nearby Murray River, and thus south of the recognized species range. Although a high incidence of disease was recorded for *L. unicolor* in the lower Darling River (e.g. 22.8% and 14% of individuals in 2011 and 2012, respectively) (Gilligan 2013), the species continues to be recorded in the mid and southern reaches of the GDA, with 76 individuals (50–88 mm TL) recorded most recently in January 2014.

Short-term persistence (beyond a winter period) and presence of young-of-year (YOY) fishes were also detected in Lake Hawthorn (Fig. 3) (Ellis *et al.* 2012). The species colonized the wetland in January 2011 during flood connection with the Murray River, before the lake was again disconnected from the main river channel in March 2011. Following disconnection, routine fish monitoring continued to record *L. unicolor*, and importantly, cohorts of YOY fishes (<20 mm SL) were detected in November 2011 and February 2012. Maturation of these cohorts (increase in size) was evident in subsequent surveys in autumn 2012 (Fig. 3). The recruitment of these YOY fishes in an isolated system clearly reflects localized spawning in Lake Hawthorn. As Lake Hawthorn contracted in 2012–2013 and salinity increased to over 20 g L⁻¹, *L. unicolor* decreased in abundance and was extirpated from the wetland in 2013 (Ellis 2013).

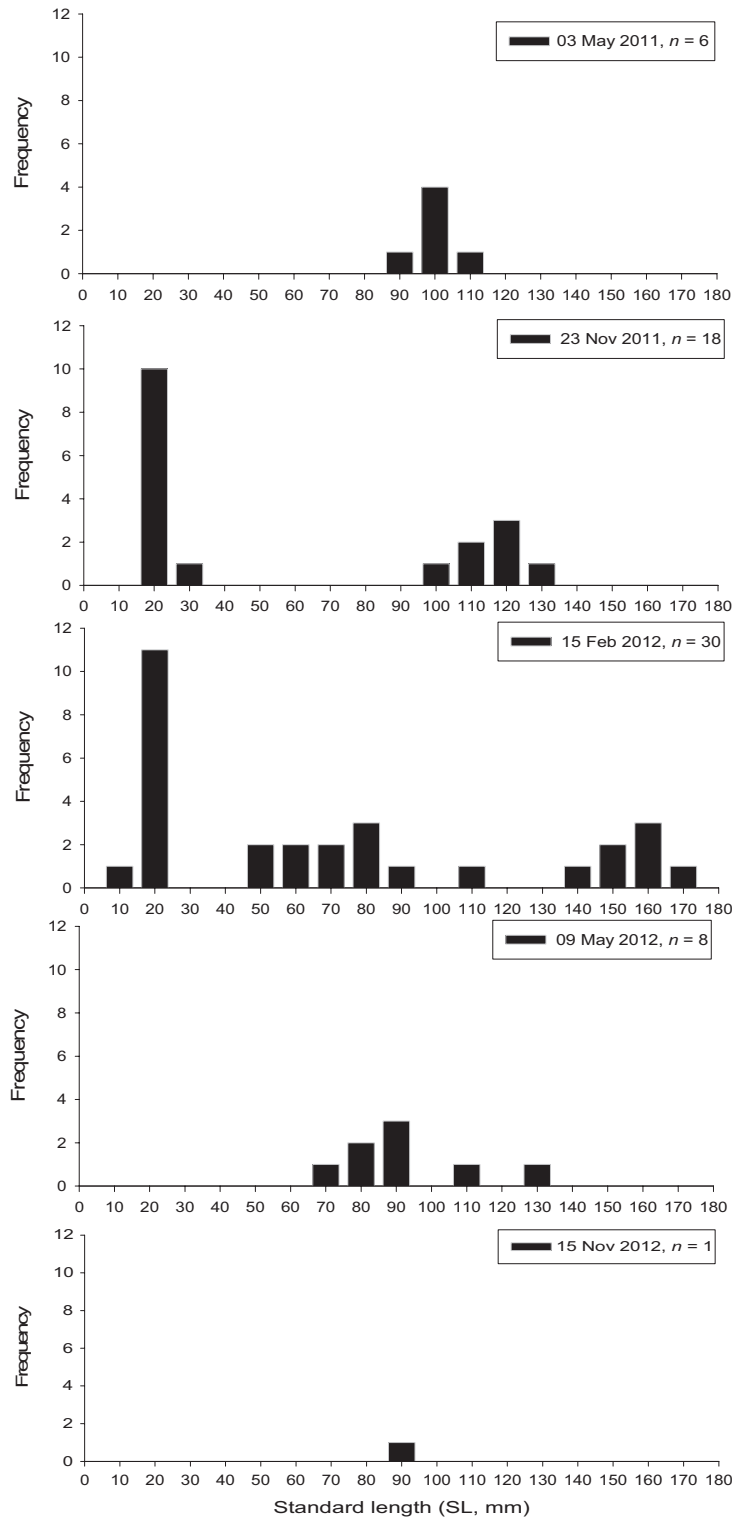


Fig. 3. Temporal changes in the population structure of *Leiotrotherapon unicolor* in Lake Hawthorn between May 2011 and November 2012 showing appearance and growth of a juvenile cohort (<20 mm standard length (SL)).

Examination of several long-term data sets by the authors suggests that water temperatures in many southern MDB habitats have generally remained above the suggested lower tolerance of *L. unicolor* since

2009 (i.e. did not fall below 7.2°C, see Fig. 4). Similarly, Hutchison *et al.* (2011) collated winter temperature minima across the MDB, in an investigation of the potential for pest Mozambique tilapia *Oreochromis*

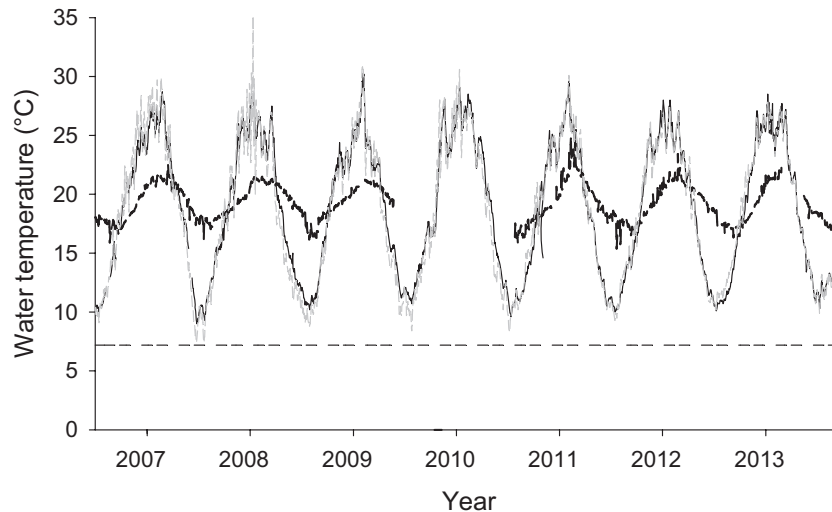


Fig. 4. Surface water temperature ($^{\circ}\text{C}$) in the Murray River (black; ds Mildura Weir, 414216A), the lower Darling River (grey; Burtundy, 425007) and an irrigation drainage outlet (bold black; at Woorlong Wetland near Mildura, 414703) from 2006 to 2013 (data courtesy of Murray-Darling Basin Authority and Department of Environment and Primary Industries). Suggested temperature below, which survival is significantly reduced (7.2°C) (Llewellyn 1973), is indicated by a black dotted line.

mossambicus to invade the MDB. Many southern MDB locations included in this assessment demonstrated winter minima $>7.2^{\circ}\text{C}$, hence above the lower threshold for *L. unicolor*, particularly at sites with large volumes of water (like weir pools on the lower Murray River), which were considered more resistant to chilling (Hutchison *et al.* 2011).

DISCUSSION

The observations presented here demonstrate rapid and extensive dispersal of *Leiopotherapon unicolor* into and throughout the southern MDB with floodwaters originating in the northern rivers of the Darling River system in late 2010 and 2011. Dispersal upon accessing the Murray River progressed both downstream (to the Lower Lakes at the terminus of the MDB) and upstream as far as Robinvale (Fig. 1). Importantly, this study is the first to identify short-term persistence (1–3 years) and recruitment of the species in the southern MDB. The observations presented here may simply represent common but previously undocumented southern occurrences of species. We suggest that this is however unlikely given the extent of historical assessments and monitoring programmes, many of which have corresponded with flooding events in the Darling River. The significance of the observations of the present study remains unclear, and given the decline in records since 2012, it is likely that the thermal tolerances of the species (Llewellyn 1973; Pusey *et al.* 2004) continue to prevent longer term persistence in the southern MDB.

The temperature limitations of the species are likely to manifest through winter stress syndrome (WSS),

which is a condition of severe lipid depletion in fishes brought on by external stressors (e.g. pollution, parasites or disease) in combination with cold weather leading to increased disease infection and mortality (Langdon *et al.* 1985; Lemly 1996). For example, during winter, cultured carp reduce feeding rates and their tissue reserves are depleted with greatest mortality due to infectious diseases (Snieszko 1974). In the MDB, high incidences of diseases in silver perch *Bidyanus bidyanus* are due partly to suppression of the immune system following rapid decreases in water temperatures in winter and to increased susceptibility to infection by pathogens including protozoans and fungi (Rowland *et al.* 2007). Infection of bony bream *Nematalosa erebi* in the lower Murray River, by both bacteria and a fungus-like oomycete *Saprolegnia*, mainly affects fishes whose susceptibility is increased by stress due to winter cold, which lowered immunity (Puckridge *et al.* 1989). Winter temperatures in the southern MDB are likely therefore to indirectly impact on *L. unicolor* through suppression of immune capacity to withstand infection, particularly during low winter flows (see Pusey *et al.* 2004). Receding flow volumes post-flood concentrate fishes in refuge waterholes or stream reaches, with declining food resources and fish condition (i.e. energy reserves) and increased cross-infection rates likely to enhance disease susceptibility (Langdon *et al.* 1985; Burford *et al.* 2008; Balcombe *et al.* 2014). Indeed, there was a high incidence of disease infection for *L. unicolor* recorded in the lower Darling River post-flooding (e.g. 22.8% and 14% of individuals in 2011 and 2012, respectively), suggesting that conditions became sub-optimal after floodwaters receded (Gilligan 2013).

Many managed wetland systems in the southern MDB receive irrigation drainage water via sub-surface infrastructure (often year round), which may result in water temperatures that are several degrees warmer near outfall locations than adjacent riverine and floodplain habitats during winter months (for example see Fig. 4). The warmer habitats may afford *L. unicolor* greater resistance to WSS and facilitate longer persistence and opportunities for recruitment. Such elevated water temperatures are likely to have contributed to the observations in Lake Hawthorn. Based on Hutchison *et al.* (2011) assessment of minimal water temperatures, there will be other wetlands that could potentially afford *L. unicolor* suitable winter refuge. Unless such locations are included in existing monitoring programmes or are subject to environmental management (e.g. receive environmental water), they are unlikely to have been subject to fish community assessment, and consequently, we suggest that the evidence for persistence of *L. unicolor* in the southern MDB may be understated. Although misidentification of larval or juvenile *L. unicolor* as another morphologically similar species (e.g. silver perch *B. bidyanus* or redfin perch *Perca fluviatilis*) could also potentially understate the scale of persistence in the southern MDB, we consider misidentifications to be unlikely given the diagnostic marking on the caudal fin of juvenile *L. unicolor* (Leggett & Merrick 1987). Given the propensity of *L. unicolor* for 'boom-bust' population dynamics (Pusey *et al.* 2004; Bunn *et al.* 2006), further evidence of *L. unicolor* persistence in the southern MDB might be expected following future Murray River flooding, should it occur in the absence of Darling River flood dispersal.

Although present anthropogenic stressors act to reduce river flow, which can impair dispersal (Balcombe *et al.* 2011), the proposed increased floodplain connectivity (i.e. connection every 1–2 years) under MDB environmental watering initiatives (e.g. under the Basin Plan: MDBA 2012) may favour *L. unicolor* by increasing dispersal opportunities. Under such circumstances, temporary winter refuge locations (such as Lake Hawthorn) could act as nodes for dispersal allowing more frequent and persistent establishment of *L. unicolor* populations across the southern MDB.

Furthermore, despite projections for climate change to reduce rainfall and exacerbate anthropogenic stressors on Australian freshwater fishes (Morrongiello *et al.* 2011a,b), *L. unicolor* is considered one of the less vulnerable species and may expand its range southwards (Balcombe *et al.* 2011; Booth *et al.* 2011; Morrongiello *et al.* 2011a; Chessman 2013). Indeed, a warming climate may contribute to a reduction in the occurrence of WSS increasingly, creating conditions in which self-sustaining populations of *L. unicolor* could persist in the southern MDB. The species possesses an

opportunistic life strategy with traits suited to range shifting (i.e. early maturation, rapid growth and short generation time (Perry *et al.* 2005)) and therefore may represent an early indicator of climate change-dependent range shifting in Australian freshwater fish (Pusey *et al.* 2004; Chessman 2013).

Conclusions

The observations presented here may represent early signs of the range expansion of *L. unicolor* into the southern MDB. Warming climates (associated with climate change) and artificially warmer wetlands combined with greater floodplain connectivity (anticipated under proposed environmental watering initiatives in Australia) may further aid the establishment of self-sustaining populations in the southern MDB. The detection and confirmation of range shifts in fish species is difficult (Booth *et al.* 2011) and targeted research will be required to (1) identify self-sustaining populations and (2) determine how climate change is influencing the distribution of the species before a southern expansion can be confirmed. Such a programme would incorporate surveys for adult and juvenile *L. unicolor*-targeted habitats identified by expert fish ecologists from both the southern and northern MDB as likely to act as refuges for the species (e.g. farm dams, semi-permanent waterholes, artificial drainage systems). Where detected, real-time logging of water physico-chemical parameters (particularly temperature) would be recorded for comparison with historical data sets (where they exist) to highlight evidence for warming of southern MDB aquatic habitats. A southern range expansion by *L. unicolor* will add novel biotic pressures to existing aquatic communities (Ficke *et al.* 2007) and would highlight the potential for other northern species, including pest species (e.g. *O. mossambicus*), to colonize the southern MDB.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article at the publisher's web-site:

Table S1. Summary of *L. unicolor* catch records collated for this study.