

Pump my wetland: potential benefits of using water pumps fitted with large-mesh screens to conserve anurans in regulated floodplain environments

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ABSTRACT

Context. Water pumps fitted with filtering screens are effective for delivering water to floodplain environments and excluding large-bodied exotic fish; yet, the benefits to wetland fish and anurans are unknown. **Aims.** To quantify fish and tadpole responses to refilling wetlands with water pumps fitted with large-mesh screens following drawdown v. overland reconnections in the mid-Murrumbidgee region of New South Wales. **Methods.** Frog and fish communities were compared between wetlands under managed inundation and overland reconnections by using PERMANOVA, and non-metric multidimensional scaling was used to evaluate community divergence between watering strategies. Classification and regression trees were used to identify thresholds in explanatory variables and predicted threshold responses in fish and tadpole abundance. **Key results.** Fish and tadpole communities differed in relation to watering strategies. Managed inundation resulted in a higher abundance of tadpole species and one native fish, whereas overland reconnections resulted in a high abundance of exotic fish species and fewer tadpoles. Water depth was a driver of tadpole abundance in two species, including the threatened southern bell frog (*Litoria raniformis*). **Conclusions and implications.** Using filtered pumps with large-mesh screens to deliver water to wetlands of high conservation value may be an effective strategy for reducing large-bodied exotic fish and enhancing frog and fish populations in regulated floodplain systems.

Keywords: alien species, environmental water, fish screens, frog conservation, invasive fish management, larval anurans, management intervention, pumping, wetland restoration.

Introduction

River regulation is a significant threat to freshwater ecosystems worldwide (Nilsson and Dynesius 1994; Hawke *et al.* 2021). Changes to the frequency and magnitude of flow pulses have contributed to decreased frequency of natural connections between rivers and floodplains (Frazier *et al.* 2005; Page *et al.* 2005), permanent drying or inundation of ephemeral wetlands (Walker and Thoms 1993; Kingsford 2000; Rayner *et al.* 2015) and significant changes to the composition of wetland biodiversity (Kingsford 2000; Beesley *et al.* 2014; Littlefair *et al.* 2021). Small-bodied fish and frog communities can be particularly affected by long-term changes in wetland hydrology (Gilligan 2005; Wassens and Maher 2011), whereas the creation of permanent weir pools may favour exotic fish populations (Jellyman and Harding 2012). In Australia, fish biomass is often dominated by invasive common carp (*Cyprinus carpio*) and gambusia (*Gambusia holbrooki*), two species responsible for causing broad-scale wetland degradation and influencing the structure of aquatic communities (Koehn 2004; Gilligan 2005; Kloskowski 2011). Both species pose a significant threat to amphibians and their larvae (Morgan and Buttemer 1996; Webb and Joss 1997; Gillespie and Hero 1999; Pyke and White 2000; Hunter *et al.* 2011; Kloskowski 2011; Atobe *et al.* 2014) and have been implicated in the decline of endangered frog and fish species (Hunter *et al.* 2011; Mac Nally *et al.* 2014; Marshall *et al.* 2019).

Rivers that have a high percentage of their annual flows extracted from the system often need water to be reinstated to maintain ecological processes (Linke and Deretic 2020). This can be achieved through environmental water allocations to restore overbank flows or re-water disconnected wetlands (Arthington *et al.* 2006). In the Murray–Darling Basin, environmental water allocations (often termed water for the environment) are used by State and Commonwealth organisations to restore natural hydrological regimes in rivers and floodplain wetlands (Arthington *et al.* 2006; Gawne *et al.* 2020). Although highly desirable in replicating natural connectivity between the river and floodplain wetlands, provision of larger flow pulses poses a significant challenge in multi-use river systems (Banks and Docker 2014) and may inadvertently benefit invasive fish species by delivering them to suitable spawning sites (Conallin *et al.* 2012), impeding the recovery of native wetland biodiversity (Rayner *et al.* 2015).

In an era of water scarcity (Arthington *et al.* 2018; Berbel and Esteban 2019), managed inundation of freshwater habitats using pumps may be an important strategy to improve native species diversity (Vilizzi *et al.* 2013; Hoffmann 2018). Large irrigation pumps can transfer significant volumes of water directly from the main river channel or through irrigation distributary networks to floodplain wetlands (King and O'Connor 2007; Boys *et al.* 2021a), and, when fitted with filtering screens, can reduce entrainment of native fish (Baumgartner and Boys 2012; Boys 2021; Boys *et al.* 2021a, 2021b). Pumping water from river channels can maintain water levels in refuge wetlands during non-flood years, and thus sustain diversity of wetland fauna such as small-bodied native fish (Vilizzi *et al.* 2013), freshwater turtles (Ocock *et al.* 2018), amphibians (Hoffmann 2018; Littlefair *et al.* 2021) and floodplain snakes (Michael *et al.* 2023). Delivering water by pumping infrastructure to wetlands can be further enhanced by using large-mesh filtering screens to exclude large-bodied exotic fish (Hillyard *et al.* 2010; Vilizzi *et al.* 2013), while delivering small-bodied native species to targeted wetlands of high conservation value. Significant efforts are underway to implement modern fish-protection screens on water infrastructure (Baumgartner and Boys 2012; Boys *et al.* 2021a, 2021b). Prioritisation of sites for modernisation requires an assessment of the ecological goals for restoration. Central to this assessment are the desired outcomes of waterways receiving allocations. Modern fish-protection screens feature a fine mesh (2–3-mm mesh size), with the goal of protecting native biota by avoiding entrainment into water delivered through water pumps. This technology is appropriate for sites where water is extracted for irrigated agriculture. However, at sites where water is delivered to habitats with ecological significance (such as floodplain wetlands), managers may seek to exclude native fish and amphibians from pumped water. At these sites, use of large-mesh screens (>30-mm mesh size) may be appropriate. However, the benefits of using large-mesh screens in floodplain anuran conservation is largely unknown. To address this knowledge

gap, we compared responses of fish and larval frog (tadpole) over a 10-year period with two water-delivery methods, managed inundation using large-mesh screens v. overland reconnections, in four ecologically significant wetlands in south-eastern Australia. We hypothesised that managed inundation using large-mesh screens would reduce adult carp abundance (Vilizzi *et al.* 2013), thereby increasing tadpole numbers owing to predator release (Hunter *et al.* 2011; Atobe *et al.* 2014). We also postulated that small-bodied native fish abundance would be similar between methods as large-mesh screens do not protect native fish from entrainment (Boys *et al.* 2021a, 2021b). Threshold responses to water level and temperature were also examined because changes in these metrics are important for structuring fish and frog communities (Bino *et al.* 2018).

Methods

Study sites

This study was conducted in the mid-Murrumbidgee region of southern New South Wales (NSW) (Fig. 1). Wetlands in this region are part of a semi-arid floodplain ecosystem and support large stands of river red gum (*Eucalyptus camaldulensis*) forest as well as significant populations of native fish and the threatened southern bell frog (*Litoria raniformis*) (Wassens *et al.* 2021), listed as Endangered in NSW (Biodiversity Conservation Act 2016), nationally Vulnerable (Environment Protection and Biodiversity Conservation Act 1999) and globally Endangered (IUCN Red List). Four wetlands, classified as ox-bow lagoons (Yarradda, Sunshower, Gooragool and McKennas), form part of a broader long-term ecological monitoring program in the Murrumbidgee catchment (Wassens *et al.* 2021) and were chosen because they have experienced significant shifts in hydrology following river regulation and recent changes in water management. Yarradda and Gooragool lagoons are in the Murrumbidgee Valley National Park. Grazing is absent from Gooragool and ceased at Yarradda in 2018, whereas McKennas and Sunshower lagoons are located on private property and are intermittently grazed by cattle. Prior to the Millennium Drought (2001–2009), this wetland cluster experienced regular floods and was inundated, on average, in 9 or 10 years by natural river to wetland overland reconnection events (Hall *et al.* 2019). Between 2000 and 2009, all four wetlands remained continuously dry and filled in 2010–11 during a natural flood (Hall *et al.* 2019).

Water delivery and timeline

In response to changes in hydrology and declining wetland condition, the NSW Department of Planning, Industry and Environment (DPIE) and Commonwealth Environmental Water Office (CEWO) returned water to Yarradda and Sunshower lagoons by delivering environmental water from the Murrumbidgee River by pumping infrastructure

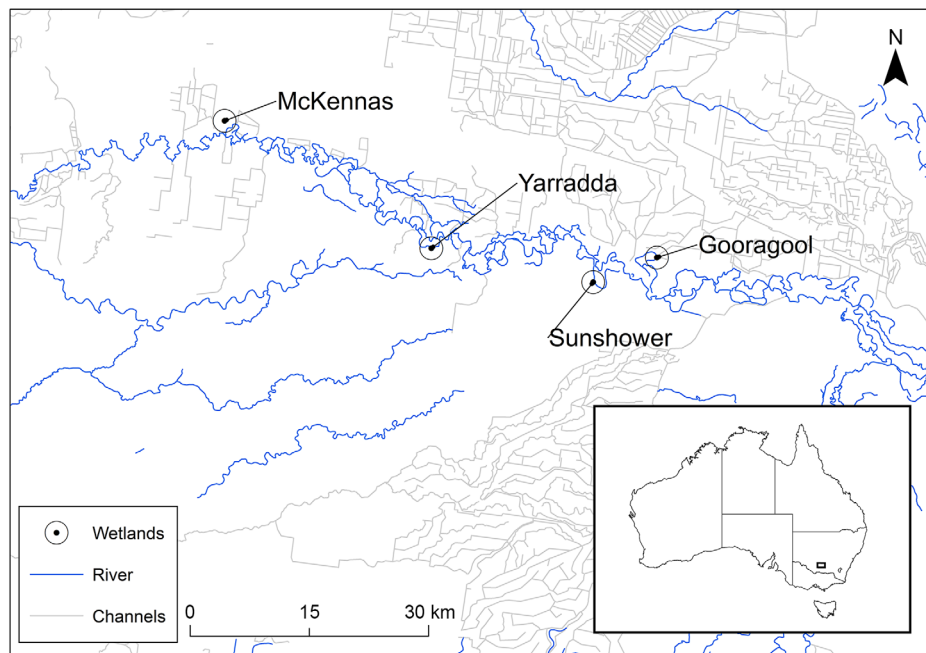


Fig. 1. Study area, showing location of four monitored wetlands in the mid-Murrumbidgee catchment, NSW.

(hereafter termed managed inundation) (Fig. 2). In the austral winter of 2014, Yarradda Lagoon dried up, after which, CEWO and DPIE refilled the wetland in summer 2015 by using filtering screens (~ 80 -mm mesh) fitted to the inlet of an impeller-type pump to control excessive carp numbers. The irrigation pump delivered 40 mL day^{-1} and water level was maintained to at least 50% capacity by additional pumping actions between January 2014 and mid-2016 (Fig. 2). A large and natural overland reconnection (flood event) occurred between August and November 2016, complemented with a managed environmental water allocation used to facilitate a river to wetland reconnection in August 2017. The wetland was allowed to briefly dry (less than 5 days) in spring 2018, to remove carp that had entered the wetland and was subsequently refilled through a filtering pump in November 2018. At Sunshower Lagoon, three small pumps delivered a combined total of $\sim 20 \text{ mL day}^{-1}$ in 2020 after a 2-year dry phase. Each pump inlet was fitted with ~ 30 -mm holes and, post pump, a 400-m poly pipeline was installed to deliver the environmental water to Sunshower Lagoon via a small tributary. Gooragool and McKennas lagoons connected to the main channel during unregulated flood events (hereafter termed overland reconnections) during 2011–12 and 2015–18 (Fig. 2). Natural flood events and managed reconnections using environmental water allocations were both classified as overland reconnections in this study.

Sampling protocol

At each of the four wetlands, annual fish and tadpole surveys were conducted between 2011 and 2020 water years

(July–June) by using a combination of two large (5-m double wing and 12-mm mesh) and two small (3-m double wing and 4-mm mesh) fyke nets set overnight for an average of 14 h. When water levels were low, two D-fyke nets (3-m double wing and 12-mm mesh) were deployed. Surveys were conducted during September, November, January and March each year. Fish and tadpole catch-per-unit-effort (CPUE, tadpole $\text{fish}^{-1} \text{ h}^{-1}$) was calculated for each wetland by dividing net soak time by the number of individuals collected. Small-bodied fish were defined as being $< 100 \text{ mm}$ and adult carp were defined as being sexually mature at $> 150 \text{ mm}$. Water depth was measured using a water-level logger (Hobo U20L) at each wetland. Developed models of water depths as a function of days since last commence-to-fill were used to predict water depth between 2010 and 2012 and in Sunshower Lagoon following logger failure during the 2019–2020 surveys (Fig. S1 of the Supplementary material). Water temperature was measured using a hand-held water-quality meter (Horiba U50).

Data analysis

Survey-level data were pooled for each water year (June–July). Fish and tadpole CPUE was averaged for each water year and underwent fourth-root transformation. Comparisons of frog and fish communities between wetlands under managed inundation *v.* overland reconnections were undertaken using PERMANOVA on the basis of a Euclidian distance resemblance matrix, sums of squares Type III (partial) and unrestricted permutation of the transformed

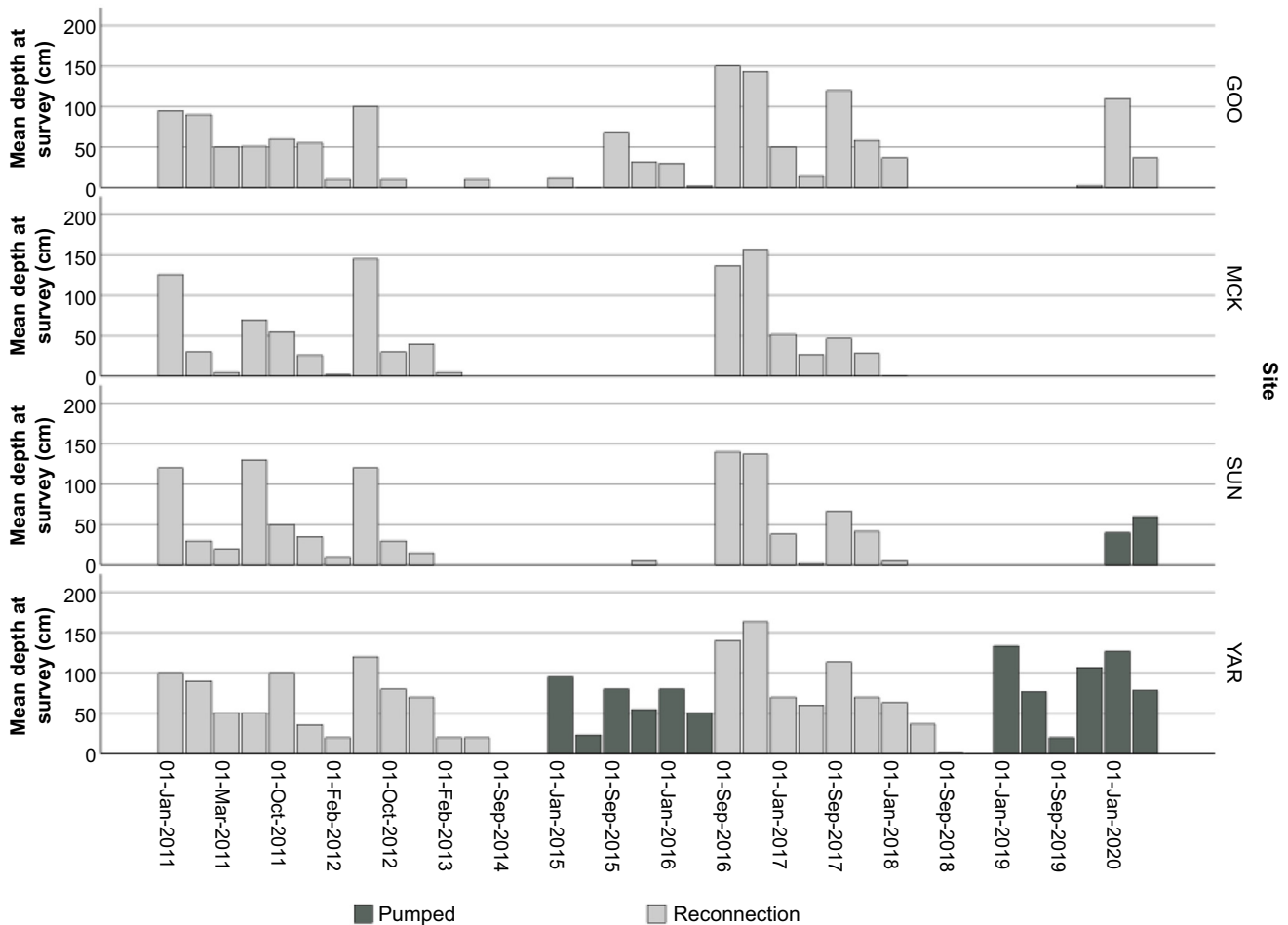


Fig. 2. Timeline of watering actions and water depth across four wetlands in the mid-Murrumbidgee catchment between January 2011 and March 2020.

data, with site being included as a random factor. Non-metric multidimensional scaling (nMDS) was used to examine community divergence between watering strategies. Key species contribution to differences between managed inundation and overland reconnections were assessed using SIMPER (Primer, ver. 7, Primer-E Ltd, Lutton, UK, see <https://www.primer-e.com/our-software/>). A Wilcoxon test was used to compare mean CPUE between reconnection events and managed inundation actions, and a one-way Student's *t*-test was used for species with low numbers under managed inundation events by using R (ver. 4.3.2, R Foundation for Statistical Computing, Vienna, Austria, see <https://www.R-project.org/>).

To identify thresholds in explanatory variables that best predicted threshold in response variables, a regression tree approach (Classification and Regression Trees, CART) was used to explore data for Yarradda Lagoon because this wetland experienced repeated managed inundation and overland reconnection events (Fig. 2). Responses in the CPUE of each of the five detected tadpole species, the total combined tadpole CPUE, common carp CPUE and gambusia CPUE were examined. Tadpoles of *Limnodynastes tasmaniensis* and *Limnodynastes*

fletcheri are indistinguishable in the field and were combined for analysis. Ecological response thresholds against water depth, water temperature and water delivery method (managed inundation v. overland reconnections) were then evaluated. The Gini index in the 'rpart' package (ver. 4.1.21, T Therneau, B Atkinson and B Ripley, see <https://cran.r-project.org/package=rpart>) was used to progressively identify thresholds in explanatory variables. No assumptions were made regarding the distribution of response variables (Breiman *et al.* 1984). Analysis was implemented using the 'rpart' function in the 'rpart' package in R.

Results

Summary statistics

Between 2011 and 2020, we recorded 109 603 individuals from 14 fish species (five introduced, $n = 54\,747$; and nine native species, $n = 54\,856$) (Table 1). Carp gudgeon

Table 1. Mean ($\pm$ s.d.) total tadpole and carp catch-per-unit-effort (CPUE), water depth, and water temperature in four wetlands in the mid-Murrumbidgee catchment between 2011 and 2020 water years.

Site	Metric	2011	2012	2013	2015	2016	2017	2018	2019	2020
Sunshower	Tadpole (n)	0 $\pm$ 0	0.33 $\pm$ 0.47	0.12 $\pm$ 0.13	0 $\pm$ 0	0 $\pm$ 0	1.56 $\pm$ 2.71	4.72 $\pm$ 6.67	0 $\pm$ 0	2.94 $\pm$ 0.89
	Carp (n)	0 $\pm$ 0	58.84 $\pm$ 114.19	0.92 $\pm$ 1.09	0 $\pm$ 0	0 $\pm$ 0	62.36 $\pm$ 92.71	22.69 $\pm$ 32.08	0 $\pm$ 0	0 $\pm$ 0
	Water depth (m)	0.86 $\pm$ 0.21	0.53 $\pm$ 0.32	0.35 $\pm$ 0.31	0 $\pm$ 0	0.14 $\pm$ 0.16	1.25 $\pm$ 1.1	0.59 $\pm$ 0.54	0 $\pm$ 0	0.34 $\pm$ 0.44
	Water temperature ($^{\circ}$ C)	27.34 $\pm$ 1.65	25.37 $\pm$ 4.98	28.35 $\pm$ 7.05	0 $\pm$ 0	0 $\pm$ 0	25.2 $\pm$ 10.08	21.16 $\pm$ 4.73	0 $\pm$ 0	22.52 $\pm$ 1.66
Gooragool	Tadpole (n)	0 $\pm$ 0	0.41 $\pm$ 0.81	0 $\pm$ 0	0 $\pm$ 0	0.29 $\pm$ 0.25	0.08 $\pm$ 0.17	0.05 $\pm$ 0.04	0 $\pm$ 0	0.5 $\pm$ 0.79
	Carp (n)	0 $\pm$ 0	11.19 $\pm$ 7.45	0 $\pm$ 0	0 $\pm$ 0	17.42 $\pm$ 25.05	4.83 $\pm$ 5.3	2.37 $\pm$ 2.37	0 $\pm$ 0	123.78 $\pm$ 146.37
	Water depth (m)	0.83 $\pm$ 0.26	0.76 $\pm$ 0.38	0.15 $\pm$ 0.09	0.08 $\pm$ 0.1	0.68 $\pm$ 0.36	1.1 $\pm$ 0.75	0.49 $\pm$ 0.41	0 $\pm$ 0	0.8 $\pm$ 0.57
	Water temperature ($^{\circ}$ C)	30.2 $\pm$ 1.7	27.13 $\pm$ 6.25	28.1 $\pm$ 4.05	27.2 $\pm$ 1.94	23.94 $\pm$ 6.52	23.99 $\pm$ 7.85	29.02 $\pm$ 8	0 $\pm$ 0	27.14 $\pm$ 2.77
McKennas	Tadpole (n)	0 $\pm$ 0	0.79 $\pm$ 1.05	0.1 $\pm$ 0.11	0 $\pm$ 0	0 $\pm$ 0	2.95 $\pm$ 5.91	6.47 $\pm$ 9.15	0 $\pm$ 0	0 $\pm$ 0
	Carp (n)	0 $\pm$ 0	197.83 $\pm$ 340.01	8.25 $\pm$ 8.05	0 $\pm$ 0	0 $\pm$ 0	62.73 $\pm$ 119.06	0.53 $\pm$ 0.66	0 $\pm$ 0	0 $\pm$ 0
	Water depth (m)	1.7 $\pm$ 0.27	1.17 $\pm$ 0.7	0.82 $\pm$ 0.72	0 $\pm$ 0	0 $\pm$ 0	1.71 $\pm$ 0.85	0.59 $\pm$ 0.46	0 $\pm$ 0	0 $\pm$ 0
	Water temperature ($^{\circ}$ C)	28.71 $\pm$ 3.11	26.74 $\pm$ 7.61	27.68 $\pm$ 5.77	0 $\pm$ 0	0 $\pm$ 0	26.66 $\pm$ 8.06	27.79 $\pm$ 12.02	0 $\pm$ 0	0 $\pm$ 0
Yarradda	Tadpole (n)	0 $\pm$ 0	0.08 $\pm$ 0.12	0.07 $\pm$ 0.13	3.53 $\pm$ 3.58	4.45 $\pm$ 5.16	0.05 $\pm$ 0.1	0.28 $\pm$ 0.57	15.23 $\pm$ 26.21	1.71 $\pm$ 1.94
	Carp (n)	0 $\pm$ 0	7.96 $\pm$ 13.49	0.46 $\pm$ 0.51	0 $\pm$ 0	7.58 $\pm$ 9.52	52.47 $\pm$ 48.56	5.77 $\pm$ 8.93	26.83 $\pm$ 31.72	2.97 $\pm$ 3.77
	Water depth (m)	2.57 $\pm$ 0.18	2.14 $\pm$ 0.4	1.86 $\pm$ 0.47	0.96 $\pm$ 1.11	2.07 $\pm$ 0.29	2.78 $\pm$ 0.92	2.14 $\pm$ 0.4	2.02 $\pm$ 0.69	2.03 $\pm$ 0.44
	Water temperature ($^{\circ}$ C)	26.76 $\pm$ 5.52	25.13 $\pm$ 5.67	25.68 $\pm$ 2.72	27.61 $\pm$ 5.06	22.51 $\pm$ 5.77	25.21 $\pm$ 7.17	26.07 $\pm$ 4.21	24.84 $\pm$ 5.2	24 $\pm$ 4.46

SUN, Sunshower lagoon; GOO, Gooragool lagoon; MCK, McKennas lagoon; YAR, Yarradda lagoon.

(*Hypseleotris* sp.) was the most abundant species ($n = 49\,063$) and contributed 89% of the total native fish catch. Carp were the most abundant exotic species ($n = 37\,348$, 62% of total exotic fish catch), followed by gambusia ($n = 18\,360$, 31% of total exotic fish catch). During the same period, 1950 tadpoles of five frog species were recorded (*Crinia parinsignifera*, $n = 4$; *Limnodynastes* sp., $n = 367$; *Litoria peronii*, $n = 1173$; *L. raniformis*, $n = 42$; and *L. interioris*, $n = 364$). *Limnodynastes* sp. (*L. tasmaniensis* and *L. fletcheri*), *Litoria interioris* and *L. peronii* tadpoles were recorded across all wetlands. *C. parinsignifera* and *Litoria raniformis* tadpoles were recorded only at Yarradda Lagoon.

Managed inundation versus overland reconnections

Overall, fish and tadpole community composition differed significantly between managed inundation and overland reconnections (PERMANOVA pseudo- F 2.088, $P = 0.009$) (Fig. 3). Differences were driven by higher average abundance of *C. parinsignifera* ($P < 0.001$), *L. peronii* ($P < 0.01$), *L. interioris* ($P < 0.01$), *L. raniformis* ($P < 0.01$) tadpoles and bony herring (*Nematalosa erebi*) ($P < 0.05$) following pumping (Fig. 4). Fish communities associated with overland reconnections had higher abundance of carp gudgeon, Australian smelt (*Retropinna semoni*), carp, gambusia, unspotted hardyhead (*Craterocephalus fulvus*) and goldfish (*Carassius auratus*) (Fig. 4), although differences were not

statistically significant. We found no statistical differences in average CPUE between managed inundation and overland reconnections for small-bodied native fish such as Murray–Darling rainbowfish (*Melanotaenia fluviatilis*), flat-headed gudgeon (*Phlypnodon grandiceps*) or oriental weatherloach (*Misgurnus anguillicaudatus*). Large-bodied native fish such as golden perch (*Macquaria ambigua*) were absent following managed inundation events (Fig. 4).

Tadpole and fish threshold responses in Yarradda Lagoon

Regression tree analysis identified managed inundation as a key driver of total tadpole CPUE, as well as for CPUE of *C. parinsignifera*, *Limnodynastes* sp. and *L. peronii* (Fig. 5a, b, d), with a higher CPUE being associated with managed inundation compared with reconnections. *L. raniformis* tadpoles were associated with managed inundation, although water depth was also an important predictor. *L. raniformis* tadpoles were not detected when water depth was below 2.3 m, but CPUE increased when water levels exceeded 2.3 m (Fig. 5c). *Limnodynastes interioris* CPUE was driven by a combination of water depth and managed inundation, having high CPUE when water levels exceeded 2.6 m. When water levels dropped below 2.6 m then managed inundation was associated with high *L. interioris* CPUE (Fig. 5f). Water delivery method was not a significant predictor of carp CPUE, instead, a water depth threshold was identified as a key predictor, with a higher carp

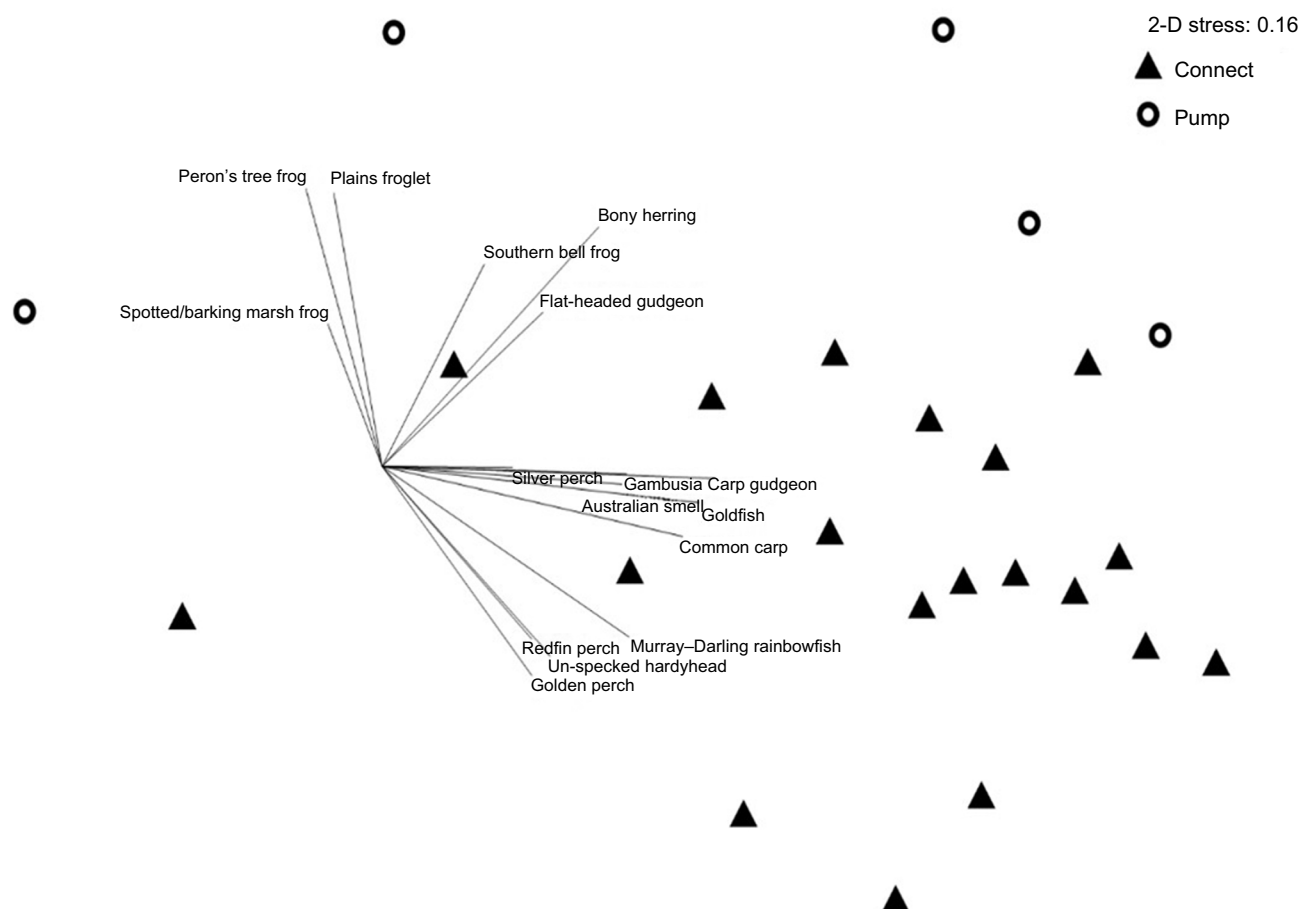


Fig. 3. Two-dimensional non-metric multi-dimensional scaling of the wetland community (CPUE) from Sunshower Lagoon and Yarradda Lagoon based on a Euclidean distance matrix and stratified by water delivery method (connect, overland reconnection; and pump, managed inundation events).

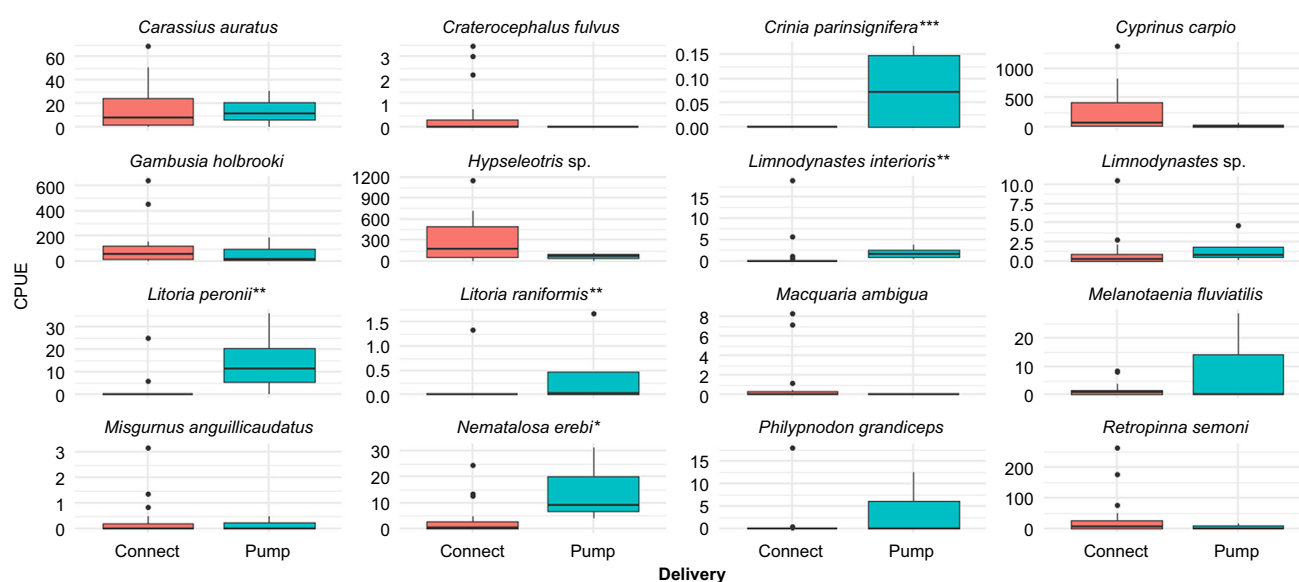


Fig. 4. Box plots of mean catch-per-unit-effort (CPUE) for the frog and fish community following managed inundation (pumping) and overland reconnection events. Probabilities are significant at: *, $P < 0.05$; **, $P < 0.01$; and ***, $P < 0.001$.

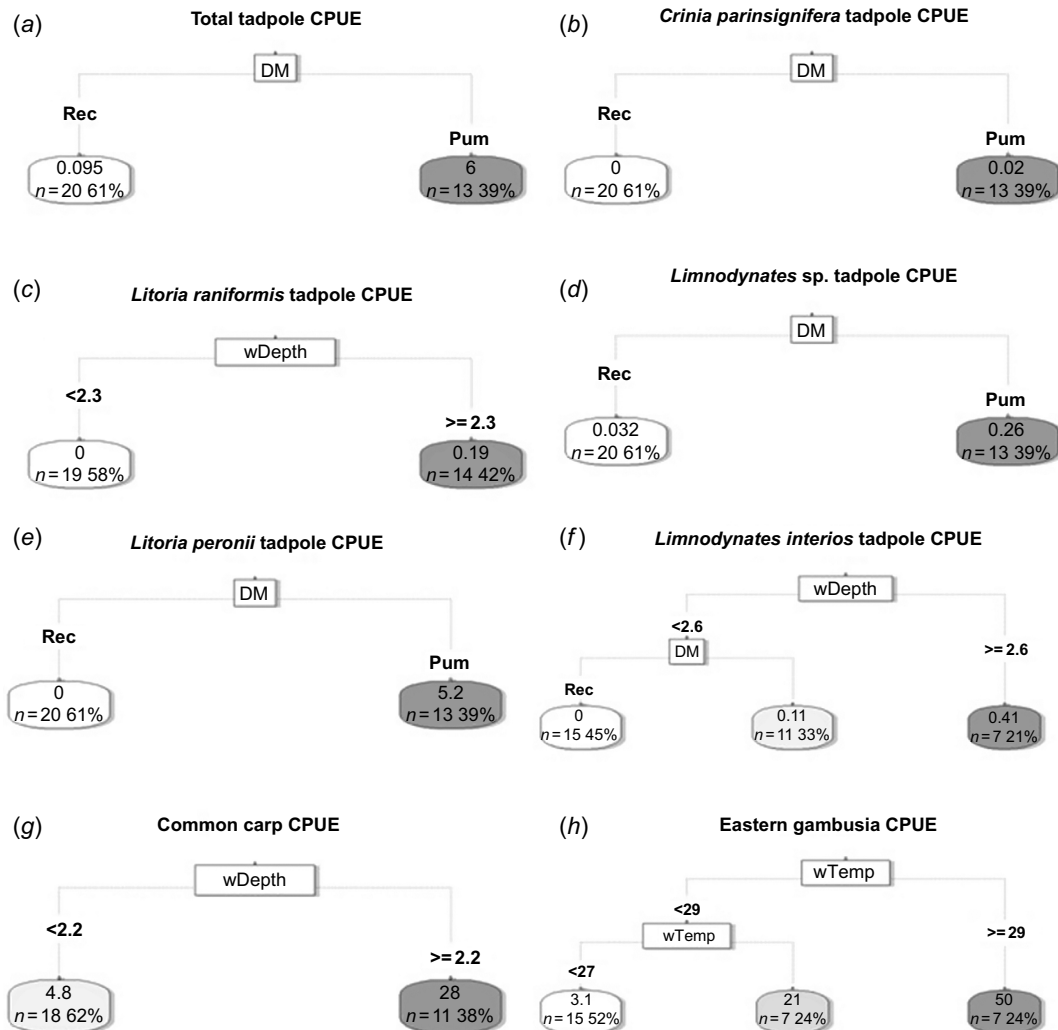


Fig. 5. Regression tree thresholds for (a) total tadpole; (b) *Crinia parinsignifera*, (c) *Litoria raniformis*, (d) *Limnodynates* sp., (e) *Litoria peronii*, (f) *Limnodynates interioris*, (g) common carp and (h) eastern gambusia catch-per-unit-effort (CPUE) in Yarradda Lagoon. DM, delivery mode; rec, river to wetland reconnection event; pum, pumping with environmental water allocations, dark shading represents higher values.

CPUE when depths exceeded 2.2 m (Fig. 5g). Water temperature was the key predictor of gambusia CPUE, with an initial threshold when temperatures were greater than 27°C and a second threshold reached when temperatures exceeded 29°C (Fig. 5h).

Discussion

In managed wetland ecosystems, the timing, frequency and volume of water delivered can have a profound effect on the structure of aquatic communities (Wassens and Maher 2011; Vilizzi *et al.* 2013; Beesley *et al.* 2014; Wedderburn *et al.* 2014). This study took advantage of a shift in wetland management (water delivery) and builds on the mesh screening literature by demonstrating the relationship between water

delivery method (managed inundation through pumping v. overland reconnections), water depth, and water temperature and the structure of frog and fish communities in a semi-arid floodplain ecosystem. Managed inundation following dry phases was found to be associated with significant increases in tadpole abundance, especially of common and widespread species such as *C. parinsignifera*, *L. interioris* and *L. peronii*, but also of the threatened *L. raniformis*. Managed inundations were also found to reduce large-bodied exotic fish species from entering the system, a finding that is consistent with that of a similar study which investigated fish communities following managed inundation events (Vilizzi *et al.* 2013). Our findings are also congruent with studies that have demonstrated amphibian population recovery following eradication of non-native fish (Knapp *et al.* 2007; Pope 2008; Tiberti *et al.* 2019; Miró *et al.* 2020). Loss of common species can have

profound ecological and economic consequences (Winfree et al. 2015; Gaston et al. 2018). Therefore, interventions that maintain or increase the abundance of widespread species are important for maintaining ecosystem stability. The use of pumps fitted with large-mesh screens, therefore, has broad implications for recovering wetland frog populations in regulated floodplain environments affected by reduced flows and water availability. However, we acknowledge that due to our small sample size, our capacity to draw general conclusions may be limited.

Environmental watering of wetlands by pumping may provide critical breeding opportunities and refugia for maintaining threatened floodplain frog species. The results of this study indicated that tadpoles of *L. raniformis* were higher in abundance following managed inundation than after overland reconnections, although regression tree analyses identified water depth as a strong variable explaining CPUE, highlighting the importance of restoring natural flood conditions to create suitable breeding conditions for threatened floodplain frog species. Ensuring that wetlands are inundated to capacity and for extended durations is likely to benefit *L. raniformis* because the species requires rising water levels to trigger breeding (Hamer et al. 2016) and sustained water levels to reach maturity. *L. raniformis* successfully bred in Yarradda Lagoon following managed inundation in 2014–15 when high tadpole counts were recorded. However, no *L. raniformis* tadpoles were recorded in the following years when water levels dropped to 50% capacity and exotic fish numbers increased. A second breeding event occurred following a natural reconnection event in 2017–2018, which raised water levels to 80% capacity. By contrast, we found no evidence that *L. raniformis* successfully bred in 2018–19 or 2019–20 during managed inundation, likely owing to sustained predation by exotic fish or our limited sample size or survey method. Larvae of this species are difficult to detect because of their cryptic foraging behaviour in shallow waters as well as their susceptibility to predation by gambusia (Pyke and White 2000), a species capable of passing through fish screens used in this study, possibly preventing *L. raniformis* from undergoing larval development and recruitment. Ongoing monitoring is required to determine whether managed inundation will benefit this species as well as other threatened floodplain frog species in the medium to long term.

Managed inundation using filtering screens fitted to pumping infrastructure is an effective tool for excluding large-bodied fish such as carp (Vilizzi et al. 2013). However, larval and juvenile carp can still enter the system and will require ongoing control; so, any potential benefits will be short-lived. In temperate environments, carp grow rapidly, reaching sexual maturity within 2 years and are highly fertile (Vilizzi and Walker 1999; Tessema et al. 2020). Thus, the amount of time that elapses following managed inundation after a dry phase before carp impose significant predation pressure on resident frog species is likely to be 2–3 years (Kloskowski 2011). River to wetland reconnections

can also rapidly introduce high densities of carp to the floodplain (Conallin et al. 2012), undoing past management actions. The findings from Yarradda Lagoon highlighted this pattern, as over 200 adult carp (CPUE) and 300 gambusia (CPUE) were recorded in 2016–2017 following an overland reconnection. Allowing ephemeral wetlands to enter a dry phase is crucial for maintaining wetland health and will be important for reducing carp numbers. One promising pattern that emerged from this study was higher abundances of native bony herring following managed inundation, indicating that smaller individuals were able to safely pass through the large-mesh screens. Managed inundation also did not appear to affect small-bodied native fish with similar numbers of Murray–Darling rainbowfish and flat-headed gudgeon being recorded between water delivery methods, suggesting that timing managed inundation with native fish supply from the main channel is important. Therefore, using managed inundation through pumping during native fish breeding periods, coupled with strategically allowing wetlands to draw down annually or biannually during winter months, may be an effective management tool for restocking small-bodied native fish communities and reducing the incursion of large exotic carp between unregulated overland reconnections.

Associative relationships among watering methods, fish communities and tadpole numbers were identified in this study. However, the mechanism behind the patterns remains unclear because other factors influencing frog breeding success may be at play. Two potential mechanisms include released predation pressure on adult frogs and their eggs (Gillespie and Hero 1999; Komak and Crossland 2000; Hunter et al. 2011) and improved condition of aquatic vegetation or water quality, leading to greater tadpole survival (Bajer and Sorensen 2015). Thus, the interactive effects of predator release on other stages of amphibian life cycle and concurrent improvements in water quality or wetland vegetation could not be disentangled in this study. Other factors not measured in this study may have contributed to frog breeding success, including annual rainfall, inundation duration, temperature (Mac Nally et al. 2014) and grazing intensity (Jansen and Healey 2003). Future studies should include a broader set of wetlands stratified by different management interventions to decouple the interactive effects of different recovery pathways.

Conclusions and management implication

Our findings confirmed that the structure of fish and tadpole communities in floodplain wetlands differ depending on the method of water delivery. Following a dry phase, managed inundation using pumps fitted with large-mesh filtering screens can reduce incursions of large-bodied fish (including carp) and promote breeding in floodplain frog species, and some small-bodied native fish. By contrast, river to wetland

reconnections contribute to a higher abundance of exotic fish species and reduced tadpole numbers, which must be considered in the long-term management of wetlands that support threatened frogs. Attempts to improve frog populations worldwide are often hampered by non-native fish. Thus, improved water delivery methods using screening technology provide novel opportunities for manipulating fish communities and improving breeding outcomes for anurans in regulated floodplain environments.

Supplementary material

Supplementary material is available [online](#).

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Data availability. The data that support the findings of this study are available from the corresponding author upon reasonable request.

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