**Stormwater wetlands can function as ecological traps for urban frogs**

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Running head: Frogs caught in ecological traps

## ABSTRACT

Around cities, natural wetlands are rapidly being destroyed and replaced with constructed wetlands created to capture stormwater. Although the intended purpose of these wetlands is to manage urban stormwater, they are inhabited by wildlife that might be exposed to contaminants. These effects will be exacerbated if animals are unable to differentiate between stormwater treatment wetlands of varying quality, that is, some function as 'ecological traps' (i.e. habitats that animals prefer despite fitness being lower than in other habitats). To examine if urban stormwater wetlands can be ecological traps for frogs, we tested if survival, metamorphosis-related measures and predator avoidance behaviours of frogs differed among stormwater wetlands with different contaminant levels in a mesocosm study, and paired this with a natural oviposition experiment to assess breeding-site preferences. We provide the first empirical evidence that these wetlands can function as ecological traps for frogs. Tadpoles had lower survival and were less responsive to predator olfactory cues when raised in more polluted stormwater wetlands, but also reached metamorphosis earlier and at a larger size. A greater size at metamorphosis was likely a result of the combined effects of increased food availability due to eutrophication, and reduced competition due to higher mortality rates. Breeding adults laid comparable numbers of eggs across wetlands with high and low contaminant levels, indicating no avoidance of the former. Since stormwater treatment wetlands are often the only available aquatic habitat in urban landscapes we need to better understand how they function to guide management decisions that mitigate their potential ecological costs. This may include improving wetland quality so that fitness is no longer compromised, preventing colonisation by animals, altering the cues animals use when selecting habitats, pretreating contaminated water prior to release, providing off-line wetlands nearby or simply not constructing stormwater treatment wetlands in sensitive areas. Our study confirms the potential for urban stormwater treatment wetlands to function as ecological traps and highlights the need for greater awareness of their prevalence and impact at landscape scales.

**Keywords:** *amphibians, constructed wetland, HIREC, metapopulation, predator avoidance, urbanisation*

## INTRODUCTION

Urbanisation is changing the biological, chemical and physical characteristics of natural aquatic ecosystems (Walsh et al. 2005). For example, in Oregon, USA, land-use changes from underdeveloped and agricultural to urban and residential led to almost all the palustrine emergent/open water wetlands being graded as either fair (46%) or poor (43%) quality (Kentula et al. 2004). Within urban areas, stormwater treatment wetlands are being created to provide ecosystem services such as flood control, and water filtration and purification (Hammer 1989). These habitats are becoming ubiquitous throughout urban and suburban landscapes, modifying both the distribution and nature of wetlands (Kentula et al. 2004). Often resembling natural wetlands, stormwater treatment wetlands attract animals (Tixier et al. 2011, Hassall and Anderson 2015), and thus may provide valuable aquatic habitat in urban landscapes where most natural waterbodies have been lost or heavily degraded.

One of the primary functions of stormwater treatment wetlands, however, is to capture and treat various contaminants (e.g. nutrients, sediments, heavy metals, pesticides) prior to release into receiving waters (Hammer 1989). Animals that inhabit them may thus suffer a range of deleterious effects, such as lower survival or physical abnormalities (Sparling et al. 2004, Snodgrass et al. 2008, Gallagher et al. 2014). Contaminants common in stormwater wetlands can also cause maladaptive behaviours to develop *via* olfactory disruption (Tierney et al. 2010). For example, Cuban tree frog tadpoles do not avoid predator cues and are hyperactive following exposure to the herbicide atrazine (Ehrsam et al. 2016). These behavioural changes can indirectly reduce fitness by increasing the probability of predation (Broomhall 2004).

The impact of highly polluted stormwater treatment wetlands will be exacerbated if animals do not recognise them as poor-quality habitats, and they function as ‘ecological traps’. An ecological trap occurs when there is differential fitness between available habitats, and animals show a preference (severe trap) or an equal-preference (equal-preference trap) for the habitat within which fitness is lower (Robertson and Hutto 2006). Ecological traps are defined by impacts to individuals, not by demographic effects (Robertson et al. 2013) and so the identification of ecological traps requires information about how habitats impact the fitness (i.e. survival, reproduction) and habitat preferences of individual animals.

Although the potential for stormwater wetlands to function as ecological traps has been discussed previously (Brand and Snodgrass 2010, Gallagher et al. 2014, Hale et al. 2015a, Sievers et al. *in press*), we currently lack empirical tests to confirm this. Animals may be attracted to stormwater wetlands by cues such as native vegetation but subsequently are exposed to contaminants that reduce fitness. Distinguishing when stormwater wetlands are functioning as ecological traps is important as these wetlands are often the only available aquatic habitat within urban areas. If traps are present, they will draw animals from higher- to lower-quality stormwater wetlands, potentially reducing the likelihood of both local and regional population persistence (Donovan and Thompson 2001, Hale et al. 2015b). Understanding how traps form and their consequences to animals has thus become an emerging management challenge. Although traps have been identified and studied broadly in terrestrial systems (Hale and Swearer 2016), there are few examples from aquatic environments, despite the significant environmental changes occurring in these habitats worldwide (Vörösmarty et al. 2010, Halpern et al. 2015).

For taxa that lack post-hatching parental care, the selection of oviposition sites is a critical decision that directly affects offspring fitness, and ultimately, population dynamics (Reich and Downes 2004). In particular, strong selective forces should be acting on the oviposition behaviour where larvae are unable to leave unsuitable sites (Rudolf and Rodel 2005, Resetarits and Silberbush 2016). Oviposition-site selection in amphibians can be influenced by a range of cues, such as the presence of predators (Resetarits and Wilbur 1989) and pesticides (Vonesh and Kraus 2009). Amphibians can thus respond adaptively to stimuli by preferring unpolluted or predator-free conditions. However, as stormwater wetlands are often the only available breeding habitat within urban landscapes, oviposition-site selection is likely to involve choices between wetlands that all are degraded/polluted/contaminated to varying degrees, complicating the decision-making process.

Given that approximately 40% of amphibian species face the threat of extinction (Monastersky 2014), there is an urgent need to understand the consequences to amphibians of inhabiting urban stormwater treatment wetlands. In this study, our overarching goal was to test if stormwater wetlands can function as ecological traps. While ecological trap studies often compare sites that are hypothesised to be traps with less disturbed (e.g. reference) sites, we examined responses to replicate pairs of stormwater wetlands of differing quality, which better reflect the reality of urban landscapes. We conducted a mesocosm experiment to quantify the fitness and behaviour of tadpoles, coupled with field-choice trials to examine the oviposition preferences of breeding frogs. We hypothesized that: (1.) tadpole survival and size/mass at metamorphosis would be lower within more contaminated wetlands, and (2.) their ability to respond adaptively to predator cues would be reduced. If urban stormwater treatment wetlands are ecological traps, we predicted that females would not avoid contaminated wetlands where offspring fitness was reduced. We discuss our results in the context of managing stormwater wetlands that are ecological traps to reduce the potential that they compromise the persistence of aquatic animals in urban landscapes. .

## METHODS

### *Study species*

Marsh frogs (Genus *Limnodynastes*, Fitzinger, 1843) are ground-dwelling frogs native to Australia, New Guinea and the Torres Strait Islands. We focused on the spotted marsh frog *Limnodynastes tasmaniensis* (Günther, 1858) and the striped marsh frog *Limnodynastes peronii* (Duméril and Bibron, 1841), two species common in south-eastern Australia that are known to occupy and breed in urban wetlands (Hamer and Parris 2011, Ficken and Byrne 2013). Nocturnal call surveys confirmed the presence of both species at all study sites.

### *Study sites*

As of 2015, nearly 500 wetlands had been constructed in Melbourne, with more than 80% of those built in the last two decades alone (Hale et al. 2015a). To examine frog responses to stormwater wetland quality, we selected six stormwater wetlands that differed in the level of contamination within the Greater Melbourne Region, Australia. These wetlands were grouped in three pairs, with one that we hypothesized to be of high quality (hereafter HQ) and another we hypothesized to be of low quality (hereafter LQ), based on sediment and water quality data recently collected by the Centre for Aquatic Pollution Identification and Management. In particular LQ and HQ status was based on the presence and concentrations of contaminants (heavy metals and pesticides) that we predicted could have an effect on amphibian fitness (Egea - Serrano et al. 2012).

***Mesocosm design***

We established 48 circular mesocosms (150L, 875 mm diameter x 310 mm deep) at The University of Melbourne, Burnley Campus (37°49’47.17” S, 145°01’28.64” E), with eight randomly assigned replicates of each source wetland. Each mesocosm was filled with 120 L of 100 µm-filtered water, 2 L of 64 µm-filtered sediment, and 4 L of unfiltered, autoclaved sediment from the respective source wetland in September 2016, and covered with 50% shade-cloth to keep out predators. Filtered sediment was added to allow the establishment of natural benthic communities, and unfiltered, autoclaved sediment to provide benthic structure; both methods also removed predatory invertebrates (Pettigrove and Hoffmann 2005). We collected zooplankton from an on-campus wetland using a 50 µm plankton net, with samples diluted and added to each mesocosm in 0.2 L aliquots after screening to exclude predators. Mesocosms were allowed to settle and mature for 4 weeks. We conducted fortnightly 25 L water changes using water from the respective source wetland to help maintain water quality conditions. Evaporated water was replaced with aged tap water or natural rainfall so that depth was kept consistent.

We collected ten spotted marsh frog egg masses from an off-line wetland (37°37’08.06” S, 145°00’10.67” E) created for the endangered growling grass frog *Litoria raniformis* (Keferstein, 1867). Each egg mass was hatched in an individual 2L beaker in the laboratory, and we transferred two-day-old tadpoles into the mesocosms. Each mesocosm received ten tadpoles on the 16$^{th}$ October 2016, one from each of the ten egg masses to remove any confounding genetic/parental effects.

### *Fitness responses*

Tadpoles were removed from mesocosms as they began to metamorphose and placed in individual 0.5 L plastic containers until complete tail resorption. We then recorded days to, length at, and mass at metamorphosis, and the incidence of any physical abnormalities (e.g. missing or malformed limbs, scoliosis). Survival was calculated as the number of tadpoles reaching metamorphosis relative to the number of tadpoles released into each mesocosm (after accounting for those removed for the predator avoidance trails; see below).

### *Periphyton, water and sediment analysis*

After the final tadpole metamorphosed (80 days), we collected periphyton, water and sediment samples from each mesocosm. To quantify food availability, we removed all periphyton from a 5 x 10 cm section, 5 cm below the high-water level of each mesocosm with a razor blade. Samples were oven dried at 60°C for 24 h, weighed, inserted into a blast furnace at 475°C for 18h, and re-weighed to calculate ash-free dry-weight (AFDW). We measured ammonia, nitrite, nitrate, phosphate and total alkalinity using an eXact® Eco-Check photometer (Industrial Test Systems, Inc, USA), and pH and conductivity using a WP-81 meter (TPS, Brisbane, Australia). We air-dried and filtered sediments to 1 mm, and metal concentrations were quantified by TrACEES at The University of Melbourne (Supplementary Appendix 1; www.chemicalanalysis.unimelb.edu.au).

### *Predator avoidance trials*

We harvested odours from dragonfly larvae – a ubiquitous and voracious tadpole predator commonly used in predator detection and avoidance experiments (Carlson and Langkilde 2013) – and conducted predator avoidance trails in a choice tank split into three zones: near predator, a middle zone and an away from predator zone (see Supplementary Appendix 2 for detailed experimental design). Trials were run with tadpoles from two source wetland pairs, as insufficient tadpoles were available from the third due to high mortality. For the first pair, we used 16 tadpoles per wetland, and for the second we used 12 tadpoles per wetland. For each source wetland, we conducted 12 replicates of each of four treatments: control (i.e. aged tap water) *vs* control, visual cues *vs* control, olfactory cues *vs* control, and visual and olfactory cues combined *vs* control. Therefore, we tested tadpoles from the first pair three times and tadpoles from the second pair four times, with no single tadpole experiencing the same treatment type more than once.

We placed a tadpole in the centre of a choice tank and provided cues at the beginning of the 2-min acclimation period. We recorded each 5-min trial using a GoPro Hero 3+, and calculated the proportion of time spent in the zone away from the predator cues relative to the time spent in the zone with the cues, based on recording the position of tadpoles every 30 seconds.

***Oviposition site preference***

We monitored oviposition preference concurrently with mesocosm experiments so that water and sediment quality would be largely comparable. We deployed three pairs of 100 L pre-formed, rock-style fibreglass ponds around each of the six source wetlands used in the mesocosm experiment. Each pond-pair simulated each wetland-pair used previously by reciprocally adding sediment and water from source wetlands.

This provided breeding frogs with a choice between two source wetland conditions with differing contaminant loads. We deployed pond-pairs approximately 10–40 m from the wetland edge, with the two ponds in a pair consistently deployed 50 cm apart in the same orientation. Pond-pairs were separated from each other by at least 50 m. Similar to the mesocosm experiments, we conducted fortnightly water changes.

Within each pond, we added a potted, semi-aquatic plant abundant at the source wetlands to encourage oviposition (foliage approximately 30 cm above water level; *Carex* sp.). We inspected ponds for egg masses at least weekly over the spring/summer breeding period. We removed and photographed egg masses, and emptied and refilled any pond containing eggs with sediment and water from the respective source wetland to avoid confounding future oviposition decisions.

***Statistical analysis***

*Mesocosm experiment*

We analysed survival, mass and length at metamorphosis, time to metamorphosis, total periphyton dry weight, ash-free fry weight of periphyton (AFDW), water quality, and the concentration of heavy metals in sediments using linear models. For each model, we fitted treatment (HQ or LQ) as a fixed effect and location (wetland pair) as a blocking factor. Data were log-transformed for all variables except survival in order to meet the assumptions of normality and homogeneity of variances.

*Predator-avoidance trials*

To examine responses to predator cues, we used generalised linear mixed effects negative binomial models with treatment and cue (control, visual, olfactory and visual + olfactory) fitted as fixed effects, location as a blocking factor, and tadpole ID and site (source wetland) fitted as random effects. Overdispersion was examined prior to calculation of model estimates and CIs. We constructed separate models for each treatment (i.e. HQ or LQ), and supressed the intercepts so that model estimates and 95% CIs could be extracted. Avoidance or attraction was then deemed significant when 95% CIs did not overlap 0.5 (i.e. no avoidance or attraction). In addition, since copper is a well-known olfactory disruptor in aquatic taxa (Tierney et al. 2010), we examined if the proportion of time spent away from predators was related to copper sediment concentration within mesocosms using linear regression, with treatment and copper concentration (log-transformed) fitted as fixed effects, and location as a blocking factor.

*Oviposition-site preference experiment*

We assessed oviposition-site preference using Chi-squared tests of the conditions (HQ or LQ ponds) eggs were laid into and the conditions (HQ or LQ source wetland) ovipositing frogs came from as factors. The number of eggs per egg mass was analysed using a linear model with treatment fitted as a fixed effect and location as a blocking factor.

We assessed normality and homogeneity of variances before all analyses using Q-Q and residual plots, respectively. We performed analyses on R 3.2.2 (R Development Core Team 2015), using the lmerTest package (Kuznetsova et al. 2015) to fit mixed effects models.

## RESULTS

***Fitness responses***

All measured parameters were significantly different between HQ and LQ treatments (Table 1). Based on model estimates, tadpoles raised in HQ treatments exhibited on average 42.0% higher survival, took 5.9 days longer to reach metamorphosis, and were 0.29 g lighter and 2.5 mm shorter than those raised in LQ treatments (Fig. 1). No physical abnormalities were observed in any of the tadpoles.

***Periphyton, water and sediment analysis***

Periphyton was approximately three times more abundant in LQ mesocosms (Table 1; Fig. 1). Water from LQ mesocosms also had higher pH, conductivity, total alkalinity, and phosphate levels (Table 2; Fig. 2; Supplementary Fig. 1). Although ammonia, nitrite and nitrate were often detected in LQ treatments, levels were below the detection limit for most HQ treatments (Supplementary Fig. 1). LQ mesocosms also had significantly higher concentrations of nine of the analysed metals (Fig. 2; Supplementary Table 2; Supplementary Fig. 1).

***Predator avoidance trials***

Overdispersion was not present in the full generalized linear mixed model ($\chi^2_{143} = 99.2, p = 0.99$). Control trials with no stimuli indicated no inherent biases in terms of preference for one side of the tank (Fig. 3). Tadpoles from different wetlands exhibited variable responses to olfactory cues from predators, but none responded to visual cues (Fig. 3). Tadpoles from HQ treatments significantly avoided predator odours, but those from LQ conditions showed a weak attraction to cues (Fig. 3). When visual and olfactory cues were provided simultaneously, all tadpoles responded by avoiding the stimuli (Fig. 3).

There was a significant, negative correlation between the proportion of time tadpoles spent away from predator olfactory cues and the copper concentration of mesocosm sediments ($F_{1, 29}$ = 4.22, *p* = 0.049; Fig. 4). Tadpoles from the mesocosms with the lowest copper concentration (~16 mg/kg) spent more than 85% of the observations away from predators, while those raised in the highest concentration (~33 mg/kg) spent less than 20% of the observations away from predators (Fig. 4).

***Oviposition site preference***

In total, 23 egg masses were laid into both high- and low-quality ponds, with no effect of the quality of the site they were laid at, or the conditions they were laid into ($\chi^2_1$ = 0, p = 1; Fig. 5). The number of eggs per egg mass was also similar for masses laid into different treatment conditions ($F_{1, 42}$ = 0.51, p = 0.47; Fig. 5).

**DISCUSSION**

We provide the first empirical evidence that urban stormwater treatment wetlands can be equal-preference ecological traps for frogs. Tadpoles had lower survival and were less responsive to predator cues when raised in more polluted conditions, but breeding adults laid comparable numbers of eggs at sites regardless of habitat quality.

Amphibians are often considered sensitive to habitat modification and contamination (Blaustein et al. 1994, Sievers et al. *in press*), and survival differed most for the two source wetland pairs with the greatest differences in environmental conditions (see Supplementary Fig. 1, 2). Heavy metals (Lefcort et al. 1998), pesticides (Egea-Serrano et al. 2012), hydrocarbons (Jelaso et al. 2002) and nitrogen (Marco et al. 1999) can all reduce amphibian survival, and a combination of contaminants can have even greater impacts (Relyea 2009). The concentration of metals within the mesocosms closely matched concentrations from initial wetland sampling (see Marshall et al. 2016, Sharley et al. 2017), so pesticide and hydrocarbon levels were also likely to have been higher within LQ treatments, further contributing to reduced survival. In addition, we would likely have observed even stronger fitness reductions if we also examined the survival of eggs and embryos, given these early life history stages can be very susceptible to contaminants (Brand et al. 2010).

Although survival was significantly reduced in LQ treatments, we also observed a reduction in the time taken to reach metamorphosis and an increase in the size of metamorphs. This may be a consequence of higher food availability, due to increased phosphates and nitrogen within more contaminated conditions, which enhances periphyton growth (Smith 1982). Additionally, greater food availability and enhanced development may be due to reduced competition because of density-dependent mortality effects. Alternatively, developing faster may allow animals to more quickly escape polluted conditions (Boyle et al. 2016). Although metamorphosis-related measures can be influenced by hydroperiod, predation and contamination (Edge et al. 2016), we found in a recent meta-analysis that survival and reproductive success differed more frequently between altered and reference wetlands than these measures, which were largely similar between wetland types (Sievers et al. *in press*). Indeed, many studies have reported conflicting findings on the effects of pollution on metamorphosis-related measures (Egea-Serrano et al. 2012). For example, in Ontario, Canada, tadpoles exposed to copper suffered reduced survival rates but increased growth rates (Leduc et al. 2016), while in Rhineland-Palatine, Germany, tadpoles exposed to cropland and the associated agrichemicals were smaller at, and took longer to reach, metamorphosis than tadpoles from natural wetlands (Wagner et al. 2014). Therefore, in isolation, these measures may provide less reliable information on individual fitness and habitat quality.

Behaviour also differed between treatments. Tadpoles raised in more contaminated conditions failed to avoid olfactory stimuli from predators, which will likely further reduce survival rates within low-quality wetlands. The attraction towards predator olfactory cues (particularly by tadpoles raised in the most contaminated conditions) mirrors similar responses observed in Cuban tree frog tadpoles after exposure to the pesticide atrazine (Ehrsam et al. 2016). We also found that tadpoles exposed to higher copper concentrations within sediments were less likely to avoid predator odours. Copper is a known olfactory inhibitor in aquatic animals (Tierney et al. 2010), and environmentally relevant levels of copper in the water can reduce the capacity of tadpoles to detect predators, ultimately leading to increased predation rates (Hayden et al. 2015).

Importantly, stormwater wetlands typically contain a mixture of contaminants, often at individual concentrations below those deemed biologically relevant (Allinson et al. 2017). Therefore, although frogs can discriminate and avoid unfavourable conditions (e.g. Vonesh and Buck 2007), their ability to do so may be reduced when they are exposed to complex mixtures of contaminants. As a consequence, females may lay eggs in sites that are highly polluted in multiple ways, rather than them being redirected to higher quality sites elsewhere, with implications for offspring fitness, and potentially population persistence.

It is critical that habitat preference is studied using an appropriate design to reflect the natural behaviour of the focal species. We offered breeding frogs a simultaneous choice between high- and low-quality conditions, an approach that is appropriate for urban stormwater treatment wetlands that are often made up of multiple, connected, sections, constructed in a deliberate sequence to maximise capture and treatment of contaminants (Lawrence and Breen 1998). As a consequence, within a single stormwater wetland network, breeding amphibians may be able to choose between areas of differing contaminant concentrations, with particular sections (e.g. sediment settlement ponds) more likely to act as ecological traps. Furthermore, although we tested habitat selection with two conditions provided simultaneously, it is possible that some amphibians exhibit sequential habitat selection, where multiple habitats may be assessed in turn until one meeting a certain threshold for acceptance is encountered (Stamps et al. 2005). In addition, a recent study found that more than 20% of frogs within a breeding population bred at more than one wetland (Lannoo et al. 2017), suggesting some that amphibians will be required to make multiple choices amongst breeding sites.While research into ecological traps is still in its infancy, the discovery and apparent prevalence of traps is starting to increase along with appreciation of the significant problem they pose for wildlife (Hale and Swearer 2016, Robertson and Chalfoun 2016). Regardless of how dispersing or breeding frogs are selecting sites, our results suggest that contaminated urban stormwater treatment wetlands are capable of reducing juvenile fitness, have the capacity to attract breeding frogs, and ultimately threaten population viability because of their capacity to function as ecological traps. Given the global vulnerability of amphibians and the increasing predominance of, and reliance by wildlife on, stormwater treatment wetlands in cities around the world, understanding the ecological risk these artificial systems pose is critical to successful wildlife management.

***Management implications***

When will the presence of low-quality stormwater wetlands become detrimental to amphibian populations, and how can we manage this risk? Even low-quality wetlands may be beneficial in some instances (Hale et al. 2015b). For example, if urban areas are built in landscapes previously devoid of natural wetlands, the construction of stormwater wetlands provides frogs with aquatic habitat that was not there previously. Even if the quality of these stormwater wetlands is relatively low, the local frog (meta)population may still increase in size because of this new habitat. However, if these wetlands are attracting and drawing in dispersing and breeding individuals away from high-quality habitats, the presence of these ecological traps can have serious consequences for metapopulations (Hale et al. 2015b). A logical next step would be to quantify both the net effect of urban development on the availability of stormwater and natural wetlands within the landscape, and the effect of these changes on local and regional (meta)populations. From this, the full impact of having stormwater wetlands functioning as ecological traps within urban landscapes can be evaluated at a scale more appropriate to managers.

When the construction of a stormwater wetland is likely to lead to the creation of an ecological trap, we need to consider how best to mitigate the risks it poses to animals. This could involve managers: (1) conserving and maintaining high-quality wetlands in the landscape; (2) restricting access by wildlife; (3) determining the cues used by animals when selecting wetlands and removing these cues, or adding them to high-quality sites; (4) finding alternative stormwater treatment solutions in sensitive areas; (5) strategically positioning stormwater wetlands within urban landscapes to limit their connectivity to other aquatic habitats; or (6) constructing high-quality, off-line wetlands that do not receive large volumes of stormwater near stormwater wetlands.

The active management of high-quality wetlands is an obvious management action to ensure at least some viable habitat is available for aquatic and semi-aquatic animals. For example, the dredging of accumulated sediment and organic debris to physically remove contaminants can improve habitat quality for wildlife, leading to greater amphibian richness and abundance (Stevens et al. 2002). Instigating a regular maintenance regime, however, has its limitations due to unknown pollutant accumulation rates, intermittent pollution events, and the substantial costs associated with intensive management regimes (Weiss et al. 2007).

Alternatively, stormwater wetlands can be designed or modified to exclude animals from potential traps. For example, designing wetland banks/edges with steep slopes may make sites less attractive than those with the gently sloping, vegetated banks that suit most amphibian species. However, steep banks/edges may also result in increased mortality of species whose metamorphs cannot escape the water and drown (Parris 2006), and this design likely reduces the overall capacity to vegetate wetlands. Other options to prevent animals accessing highly-polluted sites include displaying predator mimics, utilising sonic deterrents, or erecting exclusion fences (Martin 1979, Bomford and O'Brien 1990, Letnic et al. 2015).

It may also be possible to use our understanding of animal behaviour (Greggor et al. 2014) to discourage settlement. For example, removing fringing vegetation from wetlands is a broad-scale approach to eliminate a probable cue used by amphibians when selecting oviposition sites (Egan and Paton 2004). However, given the important role vegetation plays in reducing contaminants and improving the aesthetics of urban stormwater treatment wetlands, this method may only be suitable for the protection of specific species in need of conservation, with vegetation removal limited to certain host plant species. This strategy requires detailed knowledge of species' behaviour, habitat preferences and selection cues, and thus remains largely unexplored (although see Robertson 2012, Greggor et al. 2014). Indeed, relatively little is known about the specific habitat-selection cues amphibians use when choosing breeding sites and this remains an important and interesting area for future research.

Employing proactive strategies to pre-treat stormwater prior to release such as hybrid filtration methods have obvious broader benefits in that it likely benefits the entire wetland ecosystem. Methods such as deep sump catch basins, settling devices, vegetated filter strips and sand filtration would reduce overall inputs to wetlands (e.g. Johir et al. 2010), and could improve the survival of animals exposed to stormwater (Spromberg et al. 2016). For example, high mortality adult coho salmon suffer when exposed to unfiltered stormwater can be prevented when this water is pre-treated by soil infiltration (an inexpensive, green infrastructure technology; Spromberg et al. 2016).

However, despite this diversity of potential management options, combining the promotion of wildlife with pollution management (i.e. multi-objective management; Benyamine et al. 2004) may still be detrimental to animal populations. In addition, active monitoring in, and proactive management of, a large number of stormwater wetlands may be largely unrealistic in most circumstances. The strategic placement of stormwater wetlands within the landscape, or creating off-line wetlands near stormwater wetlands may enhance and promote urban wildlife. Theoretical models have shown that metapopulations will be most compromised when ecological traps represent a large proportion of available habitats, are highly attractive, and severely reduce individual fitness (Hale et al. 2015b). Therefore, creating stormwater wetlands predicted to receive considerable contamination beyond the dispersal distance of most animals will help prevent, or at least reduce, colonisation, and ultimately lower the risk that emigrants will settle into ecological traps. Alternatively, creating off-line wetlands disconnected from the urban stormwater network provides animals with habitat that is likely to be higher quality than even the least polluted stormwater wetlands. Given our current findings, research is first needed to determine whether animals are capable of making good decisions in this circumstance (i.e. choosing "unpolluted" off-line *versus* "polluted" on-line wetlands), and, thus, that the nearby on-line stormwater wetland is not functioning as an ecological trap.

Ecological traps are an emerging global ecological and conservation issue (Robertson and Chalfoun 2016). Identifying the potential for habitats to function as ecological traps – as we have done here – is important for minimising the harm and realising the ecological potential of human-altered habitats. Our results highlight the potential for some stormwater wetlands to impact individuals, with the assertion that this may have demographic implications. We need to better understand how a wider range of stormwater wetlands perform as habitat, particularly in terms of these broader landscape-level consequences if some stormwater wetlands are functioning as ecological traps. Future empirical studies should document how other animals respond when these habitats vary in quality, and what fitness costs are incurred, preferably using individual-level metrics (e.g. survival, growth) in addition to the more commonly used community- or population-level metrics which only indicate that animals are present at a site (Sievers et al. *in press*). In general, when traps exist, we recommend identifying the factors responsible for reduced fitness, determining the specific cues animals rely on when selecting habitats, and assessing the potential to manipulate these cues to eliminate the trap (Hale et al. 2015a). Ultimately, given the growth in stormwater treatment wetlands in cities around the world, minimising the ecological costs these artificial habitats can impose will be critical for sustaining aquatic animal populations in urban environments.

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## DATA ACCESSIBILITY

Data and metadata will be made publicly available on Figshare.

## LITERATURE CITED

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## Tables

Table 1. Output from ANOVA for the mesocosm experiment with treatment ('high' or 'low' quality conditions) and location (wetland pair) fitted as fixed effects. Significant p-values are in boldface. Asterisks indicate measures that were log-transformed to meet the assumptions of normality and homogeneity of variance.

| Measure | Factor | df | MS | F | *p* |
|---|---|---|---|---|---|
| Survival | Treatment | 1 | 0.180 | 5.10 | **0.030** |
| | Location | 2 | 0.087 | 2.40 | 0.104 |
| | Treat×Loc | 2 | 0.085 | 2.30 | 0.111 |
| | Residuals | 42 | 0.036 | | |
| *Mass at metamorphosis | Treatment | 1 | 0.034 | 12.61 | **0.001** |
| | Location | 2 | 0.006 | 2.24 | 0.120 |
| | Treat×Loc | 2 | 0.006 | 2.09 | 0.137 |
| | Residuals | 40 | 0.003 | | |
| *Length at metamorphosis | Treatment | 1 | 0.022 | 14.20 | **<0.001** |
| | Location | 2 | 0.006 | 3.97 | **0.027** |
| | Treat×Loc | 2 | 0.004 | 2.53 | 0.093 |
| | Residuals | 40 | 0.002 | | |
| *Time to metamorphosis | Treatment | 1 | 0.029 | 13.45 | **<0.001** |
| | Location | 2 | 0.007 | 3.18 | 0.052 |
| | Treat×Loc | 2 | 0.005 | 2.43 | 0.101 |
| | Residuals | 40 | 0.002 | | |
| *Periphyton | Treatment | 1 | 0.0034 | 9.85 | **0.003** |
| | Location | 2 | 0.0005 | 1.35 | 0.270 |
| | Treat×Loc | 2 | 0.0001 | 0.40 | 0.673 |
| | Residuals | 42 | 0.0003 | | |
| *Ash-free dry-weight | Treatment | 1 | 0.0022 | 10.30 | **0.003** |
| | Location | 2 | 0.0003 | 1.50 | 0.235 |
| | Treat×Loc | 2 | 0.0002 | 0.84 | 0.440 |
| | Residuals | 42 | 0.0002 | | |

**Figures**

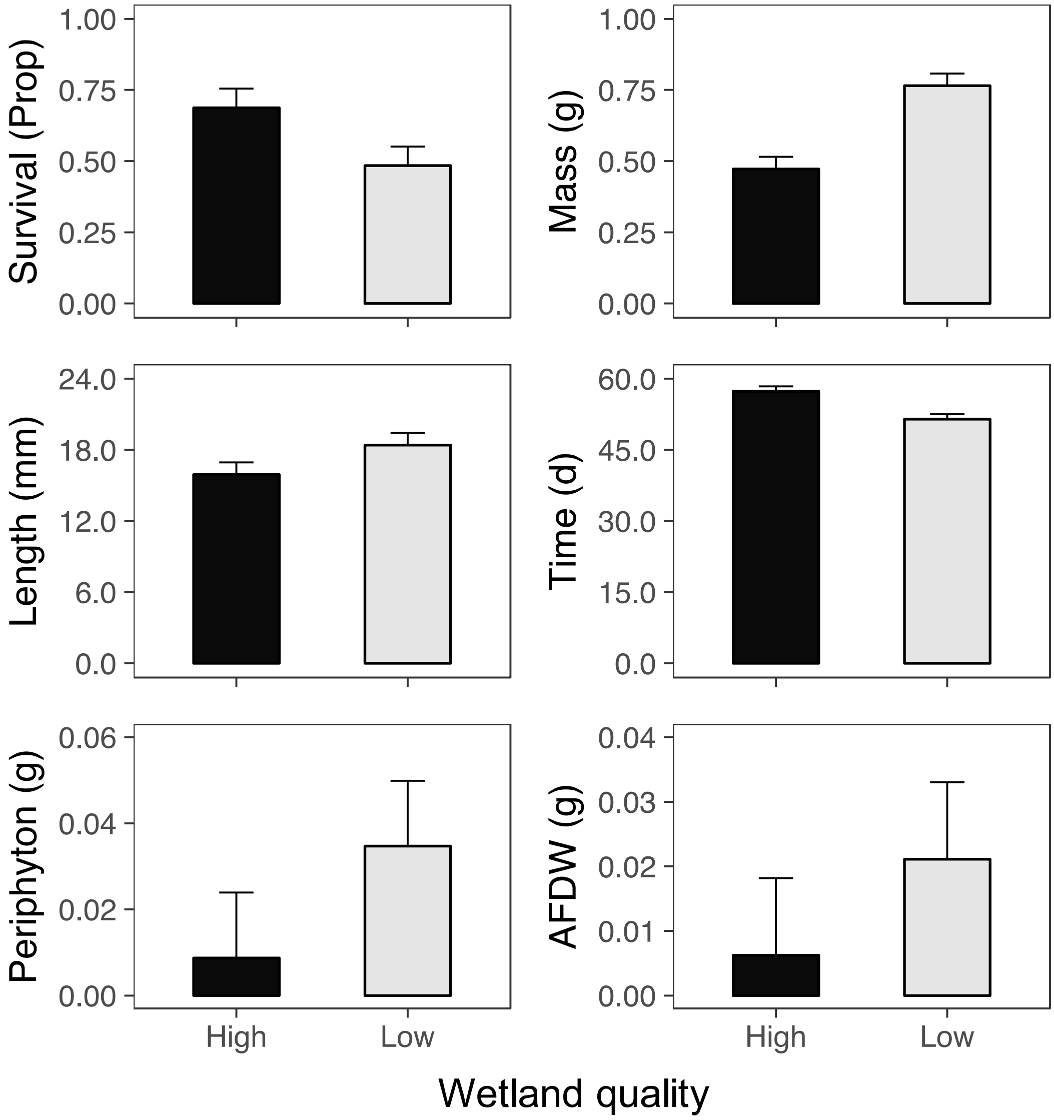



Figure 1. Model estimates (+ SE) for tadpole fitness (survival to, mass at, length at, and time to metamorphosis) and food availability (periphyton dry weight and periphyton ash-free dry weight; AFDW – g/dm$^2$) based on linear model estimates with treatment ('high' or 'low' quality conditions) and location (wetland pair) fitted as fixed effects.

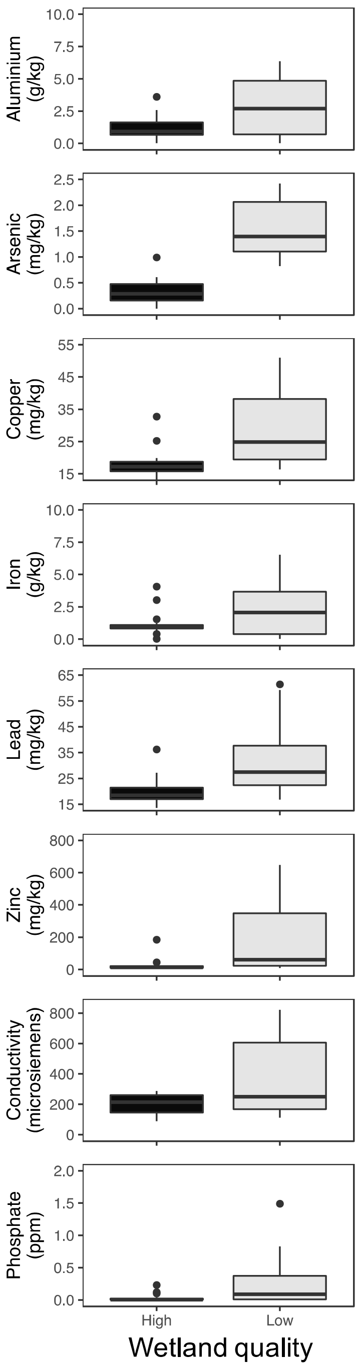



Figure 2. A subset of the sediment heavy metal concentrations (mg/kg), and the conductivity (microsiemens; µS) and phosphate ($PO_4$; ppm) concentrations within mesocosm water. N = 24 per treatment. Note different x-axis units for some metals. For vizualisations of all measured sediment and water variables see Supplementary Figure 2.

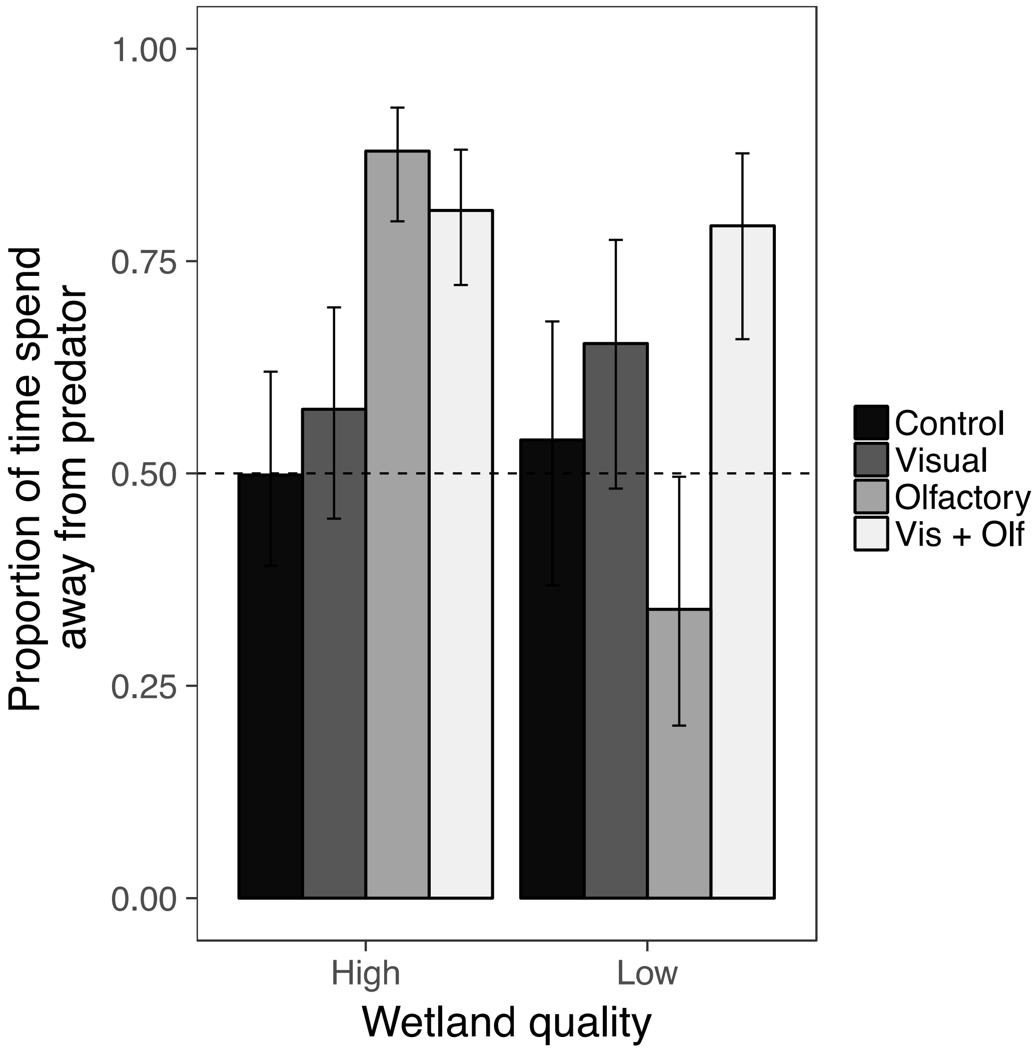



Figure 3. Median (+95% CI) model estimates from generalized linear mixed effects model for the proportion of time spent in the 'away from predator' zone relative to the 'near predator zone' in the choice tank.

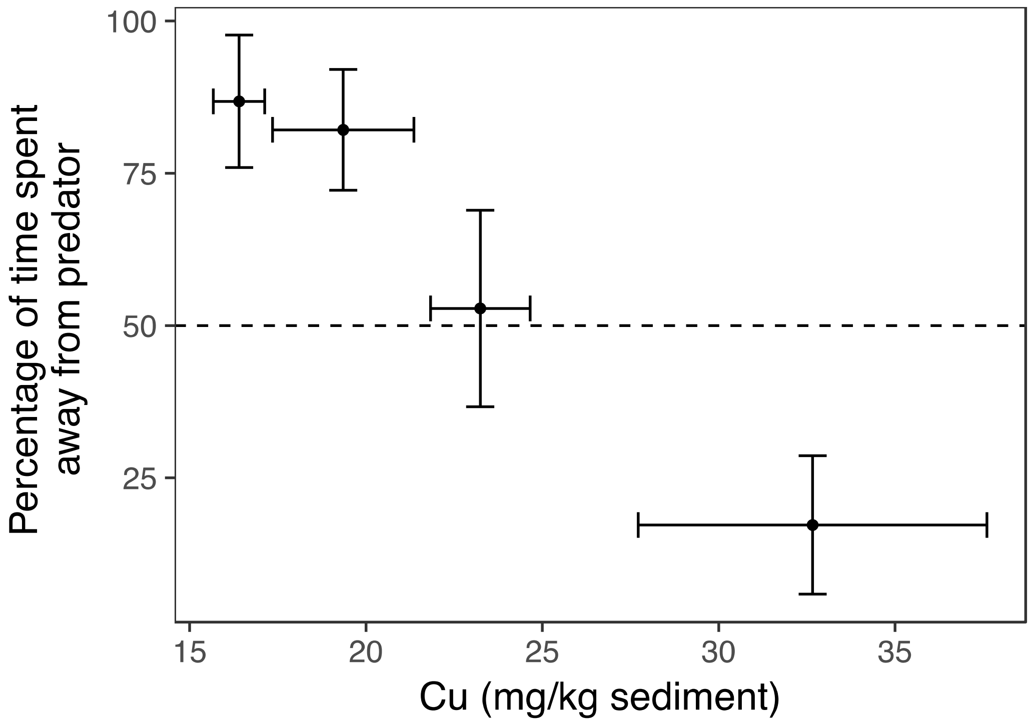



Figure 4. Relationship between copper concentration (mg/kg) within mescosom sediments and predator odour avoidance behaviour of tadpoles raised in conditions simulating Lynbrook, Chandler, Yarrabing and Ringwood wetland conditions. Error bars are standard errors.

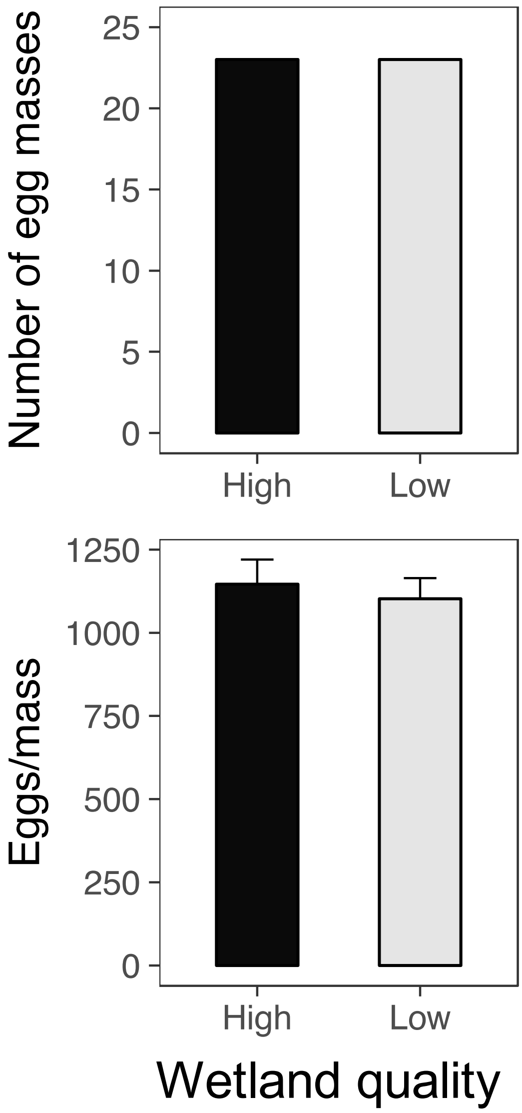



Figure 5. Oviposition site selection: the number of egg masses and the number of eggs per egg mass (+SE) laid into high- and low-quality conditions.

## SUPPLEMENTARY INFORMATION

### *Supplementary Appendix 1 - Sediment digestion and ICP-OES analyses*

All reagents were of analytical reagent grade and were used as received. Deionised water was used for the preparation of all solutions. Solutions of hydrochloric and nitric acids were prepared by dilutions of the corresponding concentrated reagents. Reverse *aqua regia* was made by mixing concentrated nitric and hydrochloric acids in a ratio of 3:1.

Oven dried (50$^{o}$C) sediment samples (0.5 g) were digested with 5 ml of reverse aqua regia solutions. Samples were weighed into 75 ml volumetric borosilicate digest tubes which were placed into a heating block digestor. Each sediment sample was mixed with the digestion solution and left for $16 \pm 1$ h (overnight) at room temperature (pre-digestion step). Then the suspension was heated at 110 °C for 5 h (digestion step). This temperature was chosen as the average digestion temperature of frequently used soil digestion methods. After the digestion, the samples were cooled down to room temperature and made up to 75 ml. Digest samples were then filtered and dispensed into sampling vials for elemental analyses using a Perkin Elmer Optima 8300 ICP-OES equipped with a charge coupled solid state detector and an auto sampler. Elemental concentrations were derived using linear calibration of the operating software was Syngistix. The analysis was performed under the following conditions: RF power at 1300 W, plasma flow at 15.0 L/min, auxiliary flow at 0.2 L/min, and nebulizer gas flow at 0.8 L/min. The spectrometer was set to normal resolution with a normal purge gas flow and read delay time of 30 s. Five replicate readings were made for each analytical sample. The sample flow rate was 1.50 ml/min with a flush time of five seconds between samples. The wash parameters were set as wash rate 1.5 ml/min and wash time 30 s.

***Supplementary Appendix 2 - Predator avoidance trials***

We obtained locally collected dragonfly larvae (Suborder: Epiprocta) – a ubiquitous and voracious tadpole predator commonly used in predator detection and avoidance experiments (e.g. Carlson and Langkilde 2013, Hanlon and Relyea 2013). To harvest predator odours, we kept 36 dragonfly larvae in 4 L of aged tap water for 18 h, which was diluted to 20 L, resulting in a concentration of 1 larva per 0.56 L of odour water (similar to Carlson and Langkilde 2013, Ehrsam et al. 2016). All cue water was made fresh and used within 12 h.

We ran predator avoidance trails with tadpoles from Pairs 1 and 3 only, as insufficient tadpoles were available from the other pair due to high mortality. We collected tadpoles from mesocosms using tadpole traps made from inverted 600 mL water bottles with lettuce inside. For the Lynbrook-Chandler trials, we collected two tadpoles from each mesocosm (16 tadpoles per source wetland; Length: 50.4±1.9 mm; mean±SE). For the Yarrabing-Ringwood trials, tadpoles were not available from two mesocosms from each source wetland, so we collected 12 tadpoles per source wetland (Length: 45.9±1.6 mm). Tadpoles were held individually in 0.5 L plastic containers with source water in the laboratory at ~25°C and under a 12L:12D light cycle before trials.

For each wetland, we conducted 12 replicates of each of four treatments: control (i.e. aged tap water) *vs* control, visual cues *vs* control, olfactory cues *vs* control, and visual and olfactory cues combined *vs* control. Given there were 32 tadpoles for the Chandler-Lynbrook trials and 24 for the Ringwood-Yarrabing, we tested the former tadpoles three times, whilst we tested the latter four times. No single tadpole experienced the same treatment type more than once.

We conducted each trial within a glass choice tank (50 x 14 x 15 cm; L x W x H), with side glass tanks (30 x 14 x 15 cm) to provide visual cues, and two containers above to provide olfactory cues (Ward and Mehner 2010). We created sections within the choice tank to create three ‘zones’ (two key zones at either end, 11 cm in length, and a middle zone, 28 cm in length). We provided visual cues by holding ten dragonfly larvae in a side tank, with the larvae pushed up to the side nearest the choice tank using a piece of plastic (reducing the functional tank length from 30cm to 10cm; Ward and Mehner 2010). We also added a similar piece of plastic to the side tank on the other side of the choice tank. We provided olfactory cues by dripping in water from containers held above each side of the choice tank at a flow rate of ~50 ml/min (Ward and Mehner 2010). We switched the side that the predator cues were applied to the choice tank every 12 trials. Each trial lasted a total of 7 min; a 2 min acclimation period and a 5 min examination period. Preliminary tests with dye revealed this was sufficient for some cue to reach the middle of the choice tank without too much mixing of cues.

We placed a single tadpole in the centre of the choice tank within a clear, perforated cylinder (9 cm diameter) and provided cues at the beginning of the acclimation period. We then lifted the cylinder using a string and pulley so that tadpoles did not see the observer. We recorded each 5-min trial using a GoPro Hero 3+, and calculated the proportion of time spent in the zone away from the predator cues, based on recording the position of tadpoles every 30 seconds and excluding the observations when tadpoles where in the middle zone. Human error led to a portion of the choice tank not being visible in some videos, so these replicates were excluded from analyses.

Supplementary Table 1. Initial sediment quality data used to select and dichotomise the wetland pairs into high and low quality (HQ and LQ, respectively). All data collected by the Centre for Aquatic Pollution and Integrated Management (www.capim.com).

| | Pair 1 | | Pair 2 | | Pair 3 | |
|---|---|---|---|---|---|---|
| | **HQ** | **LQ** | **HQ** | **LQ** | **HQ** | **L** |
| | **Lynbrook Estate Wetland** | **Chandler Rd** | **Woodlands Lake** | **Cheltenham Rd** | **Yarrabing Wetland** | **Ring L** |
| ***Heavy metals (mg kg-1 dry weight)*** | | | | | | |
| Aluminium | 21100 | 16200 | 23800 | 20000 | 7480 | |
| Antimony | 2.5* | 5 | 2.5* | 2.5* | 17 | |
| Arsenic | 16 | 39 | 17 | 37 | 2.5 | |
| Barium | 90 | 430 | 130 | 150 | 40 | |
| Beryllium | 0.5* | 0.5* | 1 | 1 | 0.5* | |
| Boron | 25* | 25* | 25* | 25* | 25* | |
| Cadmium | 0.5* | 2 | 0.5* | 2 | 0.5* | |
| Chromium | 45 | 73 | 43 | 48 | 12 | |
| Cobalt | 10 | 19 | 10 | 13 | 2 | |
| Copper | 72 | 299 | 45 | 113 | 14 | |
| Iron | 25400 | 29300 | 26300 | 28000 | 6360 | |
| Lead | 29 | 176 | 24 | 283 | 13 | |
| Manganese | 109 | 187 | 91 | 136 | 79 | |
| Mercury | 0.1 | 0.4 | 0.1 | 0.4 | 0.05* | |
| Molybdenum | 1* | 13 | 8 | 2 | 1* | |
| Nickel | 34 | 159 | 27 | 36 | 4 | |
| Selenium | 2.5* | 2.5* | 2.5* | 2.5* | 2.5* | |
| Silver | 1* | 1* | 1* | 1* | 1* | |
| Strontium | 24 | 38 | 46 | 37 | 8 | |
| Thallium | 2.5* | 2.5* | 2.5* | 2.5* | 2.5* | |
| Tin | 2.5 | 48 | 2.5 | 11 | 2.5 | |
| Titanium | 180 | 430 | 40 | 370 | 40 | |
| Vanadium | 49 | 47 | 80 | 58 | 16 | |
| Zinc | 606 | 3790 | 299 | 2390 | 138 | |
| ***Pesticides (ppm)*** | | | | | | |
| Bifenthrin | 136 | 68.4 | 2 | 37.2 | 2.5 | |
| Diethyltoluamide | 4.8 | 2.5 | 17.6 | 1* | 20 | |
| Diuron | 165 | 20 | 6 | 22 | 3 | |
| Fenamiphos | 69.6 | 10 | 20 | 10 | 5 | |
| Permethrin | 49.2 | 209 | 10 | 39.4 | 5 | |
| Prometryn | 5 | 2.5 | 5 | 5 | 2.5 | |
| Pyrimethanil | 1 | 7.6 | 4 | 1 | 1 | |

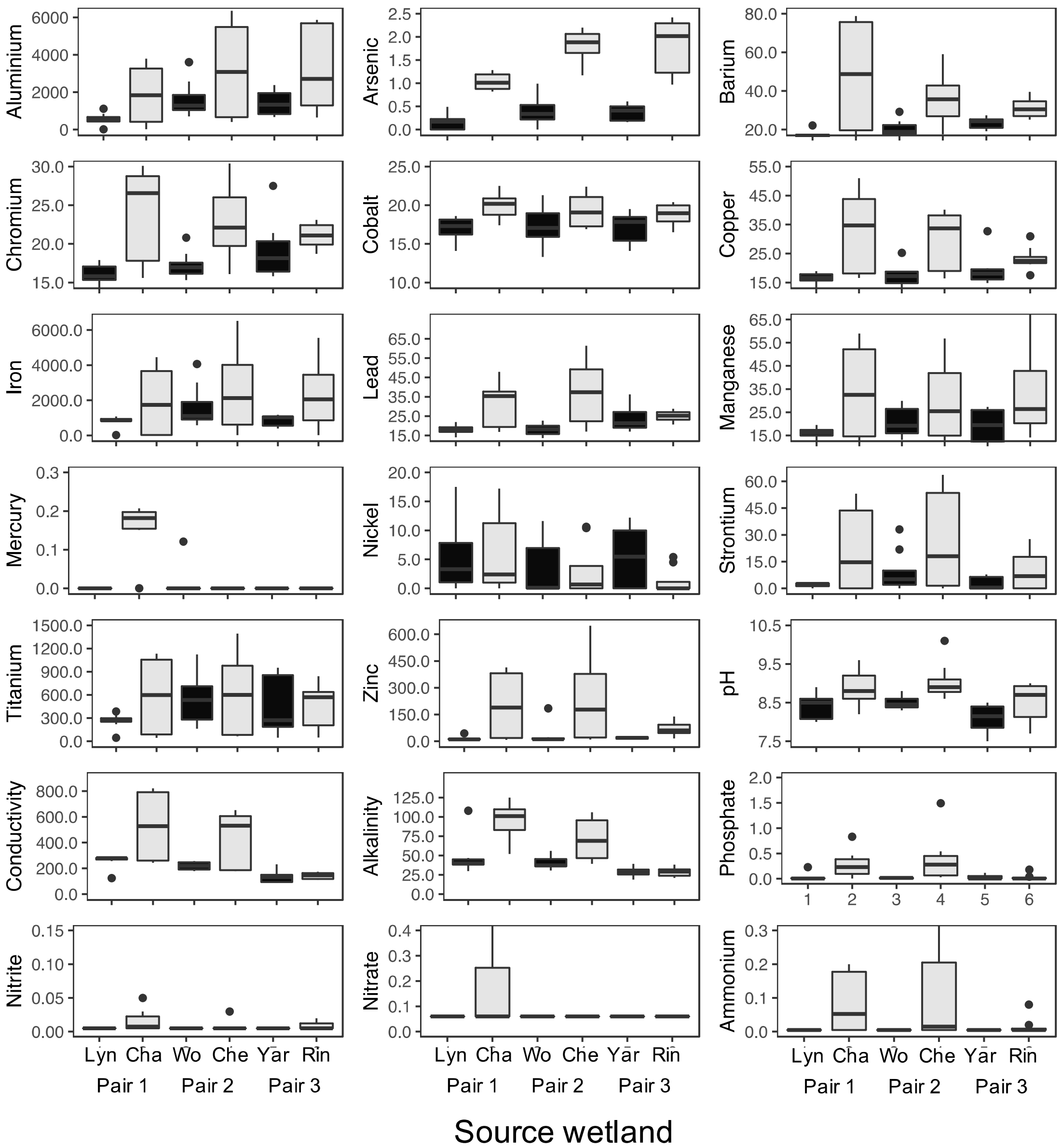
| | | | | | |
|---|---|---|---|---|---|
| Triclosan | 69.6 | 155 | 20 | 80.8 | 10 |
| Trifloxystrobin | 2.5 | 2.5 | 5 | 2.5 | 20 |
| ***Other*** | | | | | |
| Moisture Content | 4.3 | 3 | 8 | 4.6 | 2.8 |
| Total Organic Carbon | 4.22 | 4.29 | 4.19 | 4.25 | 6.14 |
| TPH | 1940 | 21810 | 500 | 5420 | 1690 |

Note: Asterisk indicates lowest detection limit

Supplementary Table 2. Output from ANOVA for the mesocosm experiment with treatment ('high' or 'low' quality conditions) fitted as a fixed effect and location (wetland pair) fitted as a blocking factor. Significant p-values are bold-face at alpha = 0.05. Asterisks indicate measures that were log-transformed to meet the assumptions of normality and homogeneity of variance. Mercury was rarely detected (8 of 48) so was not analysed.

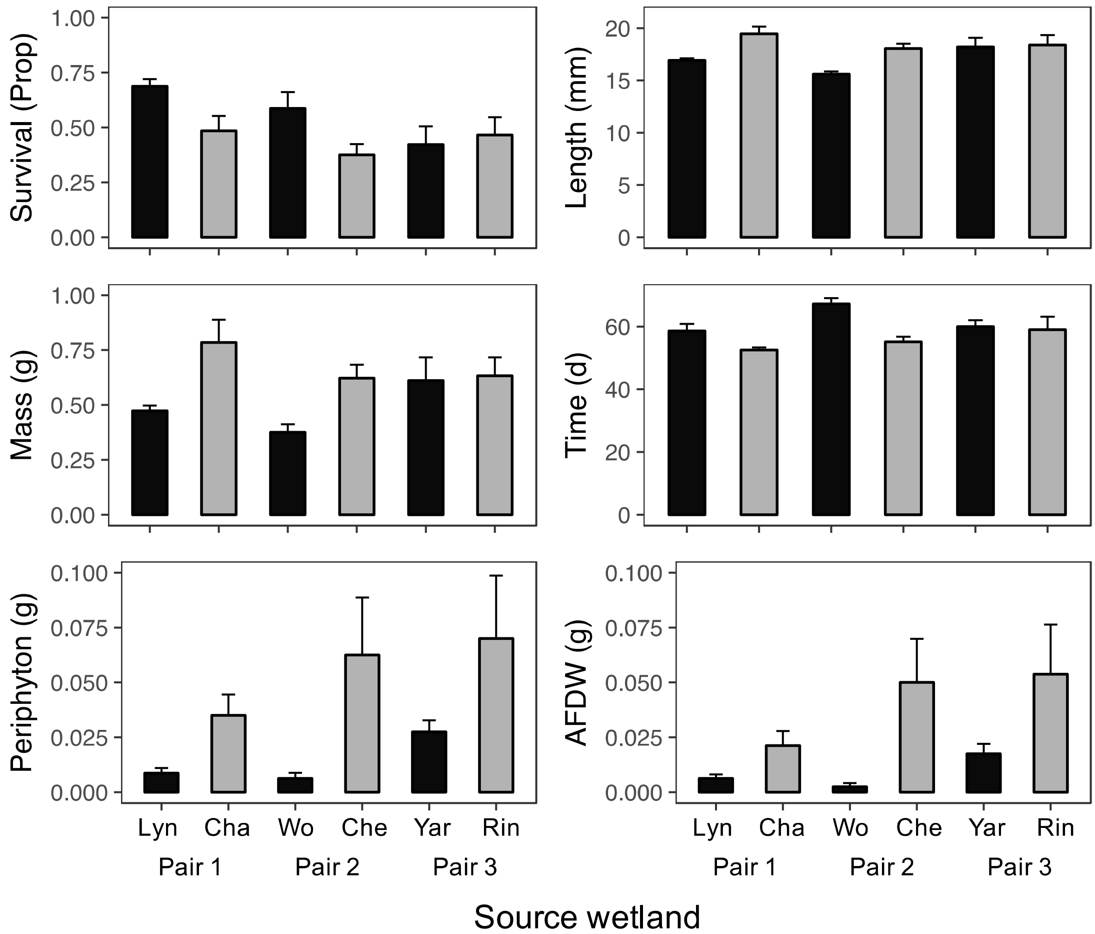
| | **Factor** | **df** | **MS** | **F** | ***p*** |
|---|---|---|---|---|---|
| *pH* | **Treatment** | **1** | **2.95** | **18.22** | **<0.001** |
| | **Pair** | **2** | **0.95** | **5.87** | **0.006** |
| | Residual | 44 | 0.16 | | |
| *Conductivity* | **Treatment** | **1** | **329528** | **13.03** | **<0.001** |
| | **Pair** | **2** | **280483** | **11.09** | **<0.001** |
| | Residual | 44 | 25296 | | |
| *Alkalinity* | **Treatment** | **1** | **7967** | **17.57** | **<0.001** |
| | **Pair** | **2** | **7480** | **16.49** | **<0.001** |
| | Residual | 44 | 454 | | |
| *PO4** | **Treatment** | **1** | **0.06** | **11.63** | **0.001** |
| | Pair | 2 | 0.01 | 2.64 | 0.082 |
| | Residual | 44 | 0.00 | | |
| *Al* | **Treatment** | **1** | **28729338** | **11.53** | **0.001** |
| | Pair | 2 | 6972271 | 2.80 | 0.072 |
| | Residual | 44 | 2491767 | | |
| *As* | **Treatment** | **1** | **18.60** | **146.52** | **<0.001** |
| | **Pair** | **2** | **1.29** | **10.18** | **<0.001** |
| | Residual | 44 | 0.13 | | |
| *Ba** | **Treatment** | **1** | **3.56** | **24.57** | **<0.001** |
| | Pair | 2 | 0.01 | 0.10 | 0.906 |
| | Residual | 44 | 0.14 | | |
| *Cr* | **Treatment** | **1** | **329.18** | **20.57** | **<0.001** |
| | Pair | 2 | 0.46 | 0.03 | 0.972 |
| | Residual | 44 | 16.00 | | |
| *Co* | **Treatment** | **1** | **62.34** | **15.64** | **<0.001** |
| | Pair | 2 | 0.88 | 0.22 | 0.802 |
| | Residual | 44 | 3.99 | | |
| *Cu* | **Treatment** | **1** | **1399.68** | **21.21** | **<0.001** |
| | Pair | 2 | 43.37 | 0.66 | 0.523 |
| | Residual | 44 | 66.00 | | |
| *Fe** | Treatment | 1 | 0.10 | 0.04 | 0.852 |
| | Pair | 2 | 4.31 | 1.58 | 0.217 |

| | | | | | |
|---|---|---|---|---|---|
| | Residual | 44 | 2.72 | | |
| *Pb* | **Treatment** | **1** | **1564.08** | **16.59** | **<0.001** |
| | Pair | 2 | 57.11 | 0.61 | 0.550 |
| | Residual | 44 | 94.28 | | |
| *Mn** | **Treatment** | **1** | **2.16** | **8.51** | **0.006** |
| | Pair | 2 | 0.02 | 0.07 | 0.930 |
| | Residual | 44 | 0.25 | | |
| *Ni* | Treatment | 1 | 26.11 | 0.99 | 0.326 |
| | Pair | 2 | 25.43 | 0.96 | 0.391 |
| | Residual | 44 | 26.49 | | |
| *Sr** | Treatment | 1 | 1.33 | 3.55 | 0.066 |
| | Pair | 2 | 0.53 | 1.41 | 0.255 |
| | Residual | 44 | 0.37 | | |
| *Ti* | Treatment | 1 | 215821 | 1.46 | 0.233 |
| | Pair | 2 | 92976 | 0.63 | 0.538 |
| | Residual | 44 | 147728 | | |
| *Zn** | **Treatment** | **1** | **27.10** | **21.16** | **<0.001** |
| | Pair | 2 | 0.19 | 0.15 | 0.862 |
| | Residual | 44 | 1.28 | | |




Supplementary Figure 1. Sediment heavy metal concentrations (mg/kg), pH, conductivity (microsiemens; μS), alkalinity (ppm), phosphate ($PO_4$), nitrite (ppm), nitrate (ppm) and ammonium (ppm). Dark bars indicate high-quality conditions and light grey bars indicate low-quality conditions. N = 8 per treatment. Lyn: Lynrook; Cha: Chandler; Wo: Woodlands; Che: Cheltenham; Yar: Yarrabing; Rin: Ringwood. Pair 1: Lynbrook (HQ; 38°03’20.84” S, 145°15’03.89” E) and Chandler (LQ; 38°00’00.56” S, 145°10’46.19” E); Pair 2: Woodlands (HQ; 38°00’11.49” S, 145°07’18.82” E) and Cheltenham (LQ; 37°59’27.68” S, 145°09’12.58” E), and; Pair

3: Yarrabing (HQ; 37°50’19.19” S, 145°13’12.39” E) and Ringwood (LQ; 37°48’48.58” S, 145°14’19.10” E).




Supplementary Figure 2. Mean (+SE) tadpole fitness (survival to, mass at, length at, and time to, metamorphosis) and food availability (periphyton dry weight and periphyton ash-free dry weight; AFDW – g/dm$^2$) for each source wetland. Dark bars indicate high-quality conditions and light grey bars indicate low-quality treatments. Lyn: Lynrook (HQ); Cha: Chandler (LQ); Wo: Woodlands (HQ); Che: Cheltenham (LQ); Yar: Yarrabing (HQ); Rin: Ringwood (LQ). Pair 1: Lynbrook (HQ; 38°03'20.84" S, 145°15'03.89" E) and Chandler (LQ; 38°00'00.56" S, 145°10'46.19" E); Pair 2: Woodlands (HQ; 38°00'11.49" S, 145°07'18.82" E) and Cheltenham (LQ; 37°59'27.68" S, 145°09'12.58" E), and; Pair 3: Yarrabing (HQ; 37°50'19.19" S, 145°13'12.39" E) and Ringwood (LQ; 37°48'48.58" S, 145°14'19.10" E).