University Library



A gateway to Melbourne's research publications

Minerva Access is the Institutional Repository of The University of Melbourne

Author/s:
Sievers, M;Hale, R;Swearer, SE;Parris, KM

Title:
Frog occupancy of polluted wetlands in urban landscapes

Date:
2019-04-01

Citation:
Sievers, M., Hale, R., Swearer, S. E. & Parris, K. M. (2019). Frog occupancy of polluted wetlands in urban landscapes. Conservation Biology, 33 (2), pp.389-402. https://doi.org/10.1111/cobi.13210.

Persistent Link:
https://hdl.handle.net/11343/284388

**Frog occupancy of polluted wetlands in urban landscapes**

Michael Sievers[1, 2*], Robin Hale[1], Stephen E. Swearer[1] and Kirsten M. Parris[2]

[1] School of BioSciences, The University of Melbourne, Parkville, Victoria, Australia, 3010

[2] School of Ecosystem and Forest Sciences, The University of Melbourne, Parkville, Victoria, Australia, 3010

*Corresponding author: sieversm@unimelb.edu.au (Michael Sievers) +61 3 8344 7986. Postal address: School of BioSciences, The University of Melbourne, Parkville, Victoria, Australia, 3010

Running head: Urban frogs

**Keywords:** *Anuran, Bayesian regression, contamination, community ecology, landscape, pollution, stormwater management, urbanization*

## ARTICLE IMPACT STATEMENT

That contaminated urban wetlands are readily occupied by frogs has implications for conservation.

## ABSTRACT

Global urban sprawl and the rising popularity of water-sensitive urban design (WSUD) has led to a surge in the number of wetlands constructed to collect and treat stormwater run-off in cities around the world. However, contaminants such as heavy metals and pesticides present

This is the author manuscript accepted for publication and has undergone full peer review but has not been through the copyediting, typesetting, pagination and proofreading process, which may lead to differences between this version and the Version of Record. Please cite this article as doi: 10.1111/cobi.13210.

This article is protected by copyright. All rights reserved.

Author Manuscript

in stormwater adversely affect the survival, growth and reproduction of animals inhabiting these wetlands. A key question is whether wildlife can identify and avoid highly-polluted wetlands. We investigated this question using pond-breeding frogs across 67 urban wetlands in Melbourne, Australia, to determine if frogs are attempting to breed in wetlands that affect the fitness of their offspring. We found little evidence for any important relationships between species richness and the concentration of any of the contaminants assessed, and frogs were present and calling at even the most polluted wetlands. The proportion of fringing vegetation at a wetland had the greatest positive influence on the number of frog species present and the probability of occurrence of individual species, indicating that frogs inhabit wetlands with abundant vegetation, regardless of their pollution status. These wetlands contained contaminant levels similar to urban wetlands around the world, and at levels which reduce larval amphibian survival. These results are, thus, likely generalizable to other areas, suggesting that urban managers could inadvertently be creating ecological traps in countless cities, with significant conservation implications. Wetlands are important tools for the management of urban stormwater run-off, but we need to ensure their construction does not facilitate declines in wetland-dependent urban wildlife.

**INTRODUCTION**

More than half the world's population lives in urban areas (WHO 2017). As cities expand, the requirement for land, and safe, reliable and sustainable sources of water are changing the nature and distribution of wetlands within urban landscapes (Kentula et al. 2004). A rise in water-sensitive urban design (WSUD), for example, has seen wetlands constructed in cities around the world to intercept stormwater and the pollutants it carries (Kentula et al. 2004).



This article is protected by copyright. All rights reserved.

Author Manuscript

With tens of thousands of these wetlands built in residential, commercial and industrial areas (Tixier et al. 2011), they have become a common feature of urban landscapes.

While stormwater wetlands are inhabited by a variety of animals (Brand & Snodgrass 2010; Le Viol et al. 2012; Hassall & Anderson 2015), their ecological benefits are unclear because they progressively accumulate toxic contaminants such as heavy metals and pesticides that can adversely affect fitness of animals (i.e. survival, reproduction, growth; Bishop et al. 2000; Tixier et al. 2011; Egea - Serrano et al. 2012). Global analyses indicate that while the species richness and population densities within urban wetlands can be comparable to those in natural wetlands, the fitness of individuals is often compromised (Sievers et al. 2018a).

Amphibians are susceptible to anthropogenic alterations to wetlands (Sievers et al. 2018a), and are suffering high rates of extinction (Monastersky 2014). Many species are considered sensitive to the contaminants that urban wetlands sequester (Hamer & McDonnell 2008; Wake & Vredenburg 2008), and exhibit reduced survival and impaired growth following exposure (Lefcort et al. 1998; Snodgrass et al. 2008; Gallagher et al. 2014). Contaminants also cause physical abnormalities (Reeves et al. 2010; Ruiz et al. 2010) and affect behaviour (Shuman-Goodier & Propper 2016; Sievers et al. 2018b), with consequences for fitness related end-points. For example, copper increased tadpole activity levels, and combined with elevated temperatures, increased the frequency of predatory attacks from larval dragonflies (Hayden et al. 2015). Despite these impacts, amphibians are frequently found occupying and breeding in urban wetlands around the world (e.g. Brand & Snodgrass 2010; Scheffers et al. 2013).

Most studies of urban amphibian communities focus on how vegetation structure (Scheffers & Paszkowski 2013), hydroperiod (Hamer & Parris 2011), water quality (Ramesh et al. 2017), predators (Hamer & Parris 2013) and wetland size (Scheffers & Paszkowski 2013) influence species richness and occupancy. A recent survey found that amphibian richness and, for some



This article is protected by copyright. All rights reserved.

Author Manuscript

species, occurrence was negatively influenced by the concentration of heavy metals in wetland sediments (Ficken & Byrne 2013).

Contaminants might influence occupancy if frogs avoid poor-quality habitats, or if population persistence is impaired at inappropriate sites.

Although frogs can modify their oviposition behaviour to avoid pesticides (Takahashi 2007; Vonesh & Buck 2007), when all available wetlands suffer moderate to high levels of pollution, such as in highly urbanized areas, frogs may be less capable of discerning wetland quality. The ecological impact of urban wetlands will be exacerbated when frogs mistakenly live and breed in wetlands containing harmful levels of contaminants, causing these to function as ecological traps (i.e. habitats that animals find equally or more attractive than other available habitat, despite animals that prefer that habitat having lower fitness than if they selected an available alternative; Robertson & Hutto 2006). We have recently shown that urban wetlands can act as ecological traps for spotted marsh frogs (*Limnodynastes tasmaniensis*), with some individuals choosing to breed in wetlands with metal and pesticide levels that reduced tadpole survival and interfered with anti-predator behaviours (Sievers et al. 2018c).

Given the increasing prevalence of stormwater wetlands around the world, and growing evidence the pollutants they accumulate impact larval amphibians, it is pertinent to assess how frequently frogs may be attempting to breed in polluted wetlands. We used 67 urban wetlands throughout the Greater Melbourne Region, Australia as a case study to explore relationships between species richness and occupancy and a range of habitat- and landscape-scale characteristics, with a focus on the concentration of metals and pesticides within sediments. As with many other cities, constructed wetlands are a common stormwater treatment option in Melbourne, and these habitats show considerable spatial variability in their contaminant loads (Marshall et al. 2016; Sharley et al. 2017). If frogs inhabit and attempt to breed at urban



This article is protected by copyright. All rights reserved.

Author Manuscript

wetlands irrespective of contaminant levels, managers could be unintentionally creating ecological traps (Hale et al. 2015a), which have the potential to threaten conservation efforts.

Author Manuscript

## METHODS

### *Amphibian surveys*

We conducted five replicate nocturnal call surveys (runs) at 67 urban wetlands within the Greater Melbourne Region, Australia (Appendix S1) over two breeding seasons for spring/summer breeders (September 2015–March 2016 and October 2016–February 2017). The order in which we surveyed wetlands was randomized within each run, but stratified based on geographic location (i.e. clusters of sites). Following Parris et al. (1999), two people listened for advertisement calls at each wetland for 5 min, followed by a physical search of the surrounding banks and vegetation using headlamps for a minimum of 10 min.

### *Pollution levels*

As part of a monitoring program conducted during the first year of surveys, the Centre for Aquatic Pollution Identification and Management (CAPIM; www.capim.com) measured the concentrations of 24 heavy metals and nine pesticides in urban wetland sediments (Table 1; For detailed descriptions of sediment collection and analysis see Marshall et al. 2016; Sharley et al. 2017). From these data, we calculated a heavy-metal index based on benchmark values set by The Australian and New Zealand Environment Conservation Council (ANZECC & ARMCANZ 2000; Ficken & Byrne 2013). We also compared levels of heavy metals within the sediments of our study wetlands with those from Snodgrass et al (2008) and Sievers et al (2018c), which



This article is protected by copyright. All rights reserved.

caused significant mortality of amphibian larvae (Table 1). We chose to use an index in our models as the concentrations of most metals were highly correlated with each other.

We also examined the effects of bifenthrin, diuron and trifloxystrobin within wetland sediments as model variables. These three pesticides are relatively common in urban wetlands around the world (Brown et al. 2010; Mimbs IV et al. 2016), and based on limits of reporting, were detected at a greater proportion of our study wetlands (bifenthrin: 82%, diuron: 88%, trifloxystrobin: 27%) than the other six pesticides. Bifenthrin is a synthetic pyrethroid used in residential developments (e.g. for termite control); diuron is a photosynthesis-inhibiting herbicide used primarily for weed control; and trifloxystrobin is a strobilurin fungicide commonly used to control mildews and brown patch of turf grass (Marshall et al. 2016; Allinson et al. 2017). These pesticides can adversely affect fish and invertebrates at low concentrations (Nebeker & Schuytema 1998; Amweg et al. 2005; Zhu et al. 2015), but little is known about their impact on frogs (although see Belden et al. 2010). Given their considerable toxicity to algae, plants, and other aquatic animals, however, it is likely that frogs could experience direct (e.g. mortality, reduced growth) and indirect (e.g. reduced food availability, immunological impacts and endocrine disruption) effects across one or more life stages.

***Habitat and landscape characteristics***

We calculated the wetland surface area using the spatial analysis tools on www.nearmap.com. We estimated the proportion of the wetland surface area covered by fringing and emergent vegetation during the first round of surveys and these two proportions were averaged to provide a vegetation index ranging from 0 to 1. During each survey, we recorded water temperature, EC and pH.



This article is protected by copyright. All rights reserved.

Author Manuscript

We also included the underlying catchment geology (sedimentary or basalt), the area of wetland within a 500m radius of each survey wetland (using GIS software), and the proportion of the sub-catchment classified as urban and as road (Table 2; see Sharley et al 2017 for the calculation and specific definitions of these variables). Area-informed metrics, such as the amount of available habitat within a given radius of a wetland as used here, can predict the likelihood of immigration (Bender et al. 2003). Although information on the movement ecology of our species is limited, we assumed 500 m to be an achievable dispersal distance (Sinsch 2014).

***Statistical analysis***

We assessed potential explanatory variables for collinearity and only included uncorrelated variables ($r < 0.4$) in the same model (Appendix S2). We examined the influence of eleven habitat- and landscape-scale variables on the number of frog species detected as adults (i.e. species richness) using Bayesian Poisson regression modeling with uninformative priors for the intercept term ($a \sim \text{dnorm}[0, 1.0 \times 10^{-6}]$) and regression coefficients ($\beta_j \sim \text{dnorm}[0, 1.0 \times 10^{-6}]$, where $j$ is an explanatory variable) in OpenBUGS (Lunn et al. 2000; Spiegelhalter et al. 2007). We used logistic regression with the same explanatory variables to identify the influence of the variables on the occurrence of the five most common species (runs were combined to account for detectability; we considered five surveys to give a high level of confidence of detecting a species if it was present at a site; Canessa et al. 2012).

We transformed the heavy metal index ($\ln[x + 1]$), wetland area, and water and pesticide variables ($\ln[x]$) prior to analysis to meet statistical assumptions. We included one null model (M1; constant only), and to aid convergence we centered all explanatory variables by subtracting their means. We used OpenBUGS to generate 100,000 samples from the posterior



This article is protected by copyright. All rights reserved.

Author Manuscript

distribution of 12 models after discarding an initial "burn-in" of 10,000 samples. We obtained 95% Bayesian credible intervals (BCI) from the 2.5th and 97.5th percentiles of the distribution. We assessed the relative fit of the models against model complexity using the deviance information criterion (DIC) and considered the best models to be those with a ΔDIC < 2 (Spiegelhalter et al. 2007). We assessed the relative importance of the explanatory variables by calculating multiplicative effect sizes and 95% BCIs (Appendix S3; Hamer & Parris 2011; Hamer et al. 2012).

For species that occurred at fewer than 20% of wetlands ('rare species'), we conducted principal component analysis to compare sites occupied by a rare species to sites that weren't (using all eleven model variables), and to make the same comparison for a subset of the variables (proportion of vegetation in the emergent and fringing zones, the heavy-metal index and species richness).

Finally, we used canonical correspondence analysis (CCA) to assess the relative importance of the independent variables used previously on community composition. We excluded rare species and six unoccupied wetlands. CCA was performed using the vegan package (Oksanen et al. 2007) in R 3.2.2 (R Development Core Team 2015).

***Ethics and collection permits***

This research was approved by the University of Melbourne Animal Ethics Committee (permit no. 1513577.1) and conducted under DELWP research permit no. 10007589.

Author Manuscript



This article is protected by copyright. All rights reserved.

## RESULTS

### *Amphibian surveys*

We detected nine species of frogs during field surveys (Table 3). *Crinia signifera* (common eastern froglet) was the most common species, detected in 58% of wetlands, while *Litoria peronii* (Peron's tree frog), *Litoria raniformis* (growling grass frog), *L. fallax* (eastern dwarf tree frog) *and L. verreauxii* (whistling tree frog) were detected in fewer than 20% of wetlands (Table 3). We detected an average of 2.5 species per wetland (SE: 0.2; range: 0–6), with six wetlands unoccupied.

### *Species richness*

The best-supported model of species richness included only the proportion of the emergent and fringing vegetation at a wetland (Model 2, Table 4). Model 3, which also included the heavy-metal index (ΔDIC = 1.5), and Model 4, which also included the habitat area in a 500m radius (ΔDIC = 1.2) both had some support. The proportion of emergent and fringing vegetation had a substantial positive effect, with the number of species at the most vegetated wetland 2.1–2.4 times higher than the number of species at the least vegetated (Fig. 1). There was less evidence that heavy metals affected species richness; although multiplicative effect sizes were consistently negative, means were close to zero with considerable uncertainty surrounding estimates (Fig. 1). Similarly, although there was some evidence for positive effects of habitat area in a 500-m radius, urbanization, geology and trifloxystrobin concentration, BCIs overlapped zero (Fig. 1). There were also no clear relationships between species richness and the concentration of any of the specific metals or pesticides (Fig. 2), and seven of the nine frog species were present at wetlands with metal concentrations above probable effect



This article is protected by copyright. All rights reserved.

Author Manuscript

concentrations identified by consensus-based sediment guidelines (MacDonald et al. 2000). These guidelines are based on biological effects on macroinvertebrates and microcrustaceans, and while these consensus-based freshwater quality guidelines provide an interesting reference, they should be interpreted with caution.

***Species occurrence***

*Crinia signifera (common eastern froglet)*

Common eastern froglets were more prevalent at wetlands with a high cover of emergent and fringing vegetation, with the probability of occurrence increasing by 66–82% (Fig. 3; percentages here and below represent the change in occurrence probability between the wetlands with the lowest and highest value for that variable, while holding all other variables constant). We also found evidence of an important positive effect the amount of wetland habitat in a 500-m radius (57–78%), and the underlying catchment geology strongly influenced occurrence, with basalt wetlands more likely to support common eastern froglets (Table 4; Fig. 3).

*Limnodynastes tasmaniensis (spotted marsh frog)*

The best-supported model of the occurrence of this species included the vegetation cover at a wetland and the area of wetland habitat in a 500-m radius as the explanatory variables (Table 4). Spotted marsh frogs were more prevalent at wetlands with high cover of emergent and fringing vegetation (55–67%), while the influence of wetland habitat within 500 m was less certain, but consistently positive (6–33%; Fig. 3).

Author Manuscript



This article is protected by copyright. All rights reserved.

*Limnodynastes dumerilii (eastern banjo frog)*

Models including only landscape-scale variables had the most support (Table 4). The eastern banjo frog was more likely to be present at basalt wetlands in highly urbanized sub-catchments (82–85%). The probability of eastern banjo frogs occurring at a wetland was also negatively associated with the area of wetland habitat in a 500m radius (Fig. 3). Although the prevalence of roads was included in the best supported model, it had no influence on occurrence.

*Limnodynastes peronii (striped marsh frog)*

DIC values were very similar across most models, indicating support for many different models (Table 4). Most notably, there was evidence that wetlands in highly urbanized sub-catchments with a high prevalence of roads were more likely to contain striped marsh frogs (46–51% and 50%, respectively; Fig. 3). There was also some evidence for a positive effect of vegetation and heavy metals on the occurrence of striped marsh frogs, albeit with considerable uncertainty (Fig. 3).

*Litoria ewingii (southern brown tree frog)*

Models including only landscape-scale variables had the most support (Table 4). Of these, the underlying catchment geology and the prevalence of roads had a strong influence on occurrence, with southern brown tree frogs more often inhabiting sedimentary wetlands with fewer roads (75–76% decrease; Fig. 3). We also found some evidence for a positive effect of the area of wetland habitat in a 500m radius (25–58%; Fig. 3).

*Rare species*

We found no evidence that the sites occupied by rare species in the study area (Peron's tree frog, growling grass frog and the whistling tree frog) had different habitat conditions based on



This article is protected by copyright. All rights reserved.

Author Manuscript

our models (Appendix S4; S5). Although rare in the study area, we did not include the eastern dwarf tree frog as it is not native to Melbourne and is currently invading the study area.

***Community responses to urban environmental gradients***

The Canonical Correspondence Analysis (CCA) revealed considerable separation of species, with only the common eastern froglet and spotted marsh frog close together in the plot (Fig. 4). The first CCA axis explained 49.6% of the variability in the species occurrence data and the second axis explained 32.7%. The striped marsh frog and the eastern banjo frog were positively associated with urbanisation and the prevalence of roads, and negatively associated with the size of wetlands, the total area of habitat around wetlands and conductivity (Fig. 4). The spotted marsh frog and the common eastern froglet were positively associated with vegetation, heavy metal concentrations, the size of wetlands, and the total area of habitat around wetlands, and negatively associated with urbanisation and pesticide levels (Fig. 4). The occurrence of the southern brown tree frog was mostly influenced by geology (Fig. 4).

**DISCUSSION**

More than 90% of surveyed wetlands were inhabited, with vegetation cover strongly influencing both species richness and the occurrence of most species. Although there was some evidence that species richness was negatively correlated with the level of heavy metals and positively with the amount of wetland within a 500-m radius, these relationships were less clear. The proportion of the catchment classified as urban, the underlying catchment geology, the concentration of pesticides within sediments, electrical conductivity, wetland size, and the prevalence of roads had little effect, with considerable uncertainty around model estimates.



This article is protected by copyright. All rights reserved.

Author Manuscript

***The role of pollution in driving habitat use***

For the species and pollutant levels recorded in this survey area, our results suggest that urban wetlands will be readily inhabited irrespective of the concentration of heavy metals and pesticides present. Although previous studies have found amphibian richness to be influenced by heavy metals (e.g. Ficken & Byrne 2013) and pesticides (e.g. Gibbs et al. 2009), these studies typically compare wetlands with highly contrasting concentrations of contaminants. In our study, where all wetlands were in human-dominated areas and had some degree of contamination, we did not observe such strong trends, suggesting that while richness may be higher when contamination is very low (i.e. at natural wetlands), it plateaus once contamination reaches a certain level. Alternatively, there may be a wider potential pool of species to begin with in less-urban areas (such as the rural areas surveyed in Ficken & Byrne 2013).

The heavy-metal index and the pesticides were not included in any of the most supported models. We did find some evidence suggesting the occurrence of the striped marsh and eastern banjo frogs were influenced by heavy metals, with the former more prevalent and the latter less prevalent at wetlands with higher metal concentrations. Intraspecific differences in tadpole behaviour may mean tadpoles of the striped marsh frog spend less time in contact with sediments and are thus less susceptible to contaminants, but few inter-specific differences are known about larvae from this genus. Alternatively, competitive exclusion may be occurring, however, plots of the probability of species occurrence based on the presence or absence of other species show that competitive exclusion is unlikely to be a significant driver (Appendix S6).

Author Manuscript



This article is protected by copyright. All rights reserved.

Primarily relying on call surveys may have even underestimated the occupation of contaminated wetlands if pollution impacts vocalisations. Indirect effects could occur through reduced food availability that increases the marginal energetic cost of vocalising (such as in birds; Golabek et al. 2012), while direct effects could occur through impacts to physiology and behaviour *via* absorbed ions and ingestion. Cadmium, for example, adversely affects advertisement call duration and rate in black-spotted frogs (*Pelophylax nigromaculata*) (Huang et al. 2015), and the fungicide vinclozolin reduces calling activity in South African clawed frogs (*Xenopus laevis*) (Hoffmann & Kloas 2010). If similar effects were prevalent, individuals present at highly contaminated wetlands could be assumed to be absent due to reduced detection probability, strengthening our conclusion that frogs are selecting wetlands irrespective of contaminant levels.

***Habitat and landscape characteristics***

Fringing and emergent vegetation provides amphibians with shelter and protection, and a place to feed, call and breed (Tarr & Babbitt 2002; Egan & Paton 2004) and is often positively associated with the occurrence of amphibians (Hamer et al. 2012; Kruger et al. 2015). Our results are largely consistent with these findings, with species richness greatest at the most vegetated wetlands. It is also possible that vegetation has other impacts on habitat quality. For example, vegetation can mitigate the toxic effects of pesticides through their removal by sorption or alkaline hydrolysis (Brogan III & Relyea 2017). Therefore, it is possible that the relationship between vegetation and amphibian assemblages is also related to vegetation lowering the concentrations of other contaminants not measured here, making the wetland more habitable, and consequently increasing the probability of occurrence. More work examining the interactions between vegetation and different pollutants, and the consequences



This article is protected by copyright. All rights reserved.

Author Manuscript

this has for frogs would allow this possibility to be further examined.

Although the size of wetlands may influence species richness (via well-established species-area relationships; Hanski 1994; Parris 2006), we found no substantial effect of wetland area on species richness or occurrence. Unlike natural wetlands where wetland size may be a proxy for permanence or ephemerality – which may actually be the driver of increased richness at larger wetlands – urban wetlands fed by stormwater are often permanent regardless of size, and other studies have similarly found no effect of wetland size on amphibians in urban areas (e.g. Kruger et al. 2015). On the other hand, the total wetland area within a 500-m radius was important, similar to results from other urban wetland systems (Scheffers & Paszkowski 2013). More habitat typically means greater connectivity and, thus, a greater likelihood of colonization (Bender et al. 2003).

In general, landscape-scale variables exerted a substantial influence on species occurrence. However, the strong effect of catchment geology on southern brown tree frogs and common eastern froglets likely reflects the matching of species ranges with regional differences in geology rather than a causal relationship, as southern brown tree frogs were detected almost exclusively in eastern sites that have a sedimentary geology, while common eastern froglets were detected almost exclusively in western sites that have a basalt geology. Wetlands with higher levels of surrounding urban cover were more likely to harbor striped marsh and eastern banjo frogs, two species that commonly inhabit urban wetlands and that may be well adapted to urban environments (Hamer & Parris 2011; Ficken & Byrne 2013).

Author Manuscript



This article is protected by copyright. All rights reserved.

***Ecological and conservation implications***

Urban sprawl and the rising popularity of water-sensitive urban design (WSUD) has led to the global proliferation of wetlands designed and constructed to intercept stormwater run-off. Frogs occupied urban wetlands that commonly had sediment heavy-metal concentrations above published thresholds for likely biological effects (Fig. 2; MacDonald et al. 2000), and above those within sediments that impacted tadpole survival in other studies (Snodgrass et al. 2008; Sievers et al. 2018c). However, we did not consider a range of other contaminants that are common to urban wetlands (e.g. PAHs; Gallagher et al. 2011) and also known to impact amphibians (e.g. salts and PAHs; Snodgrass et al. 2008). Further, Snodgrass et al (2008) and Sievers et al (2018c) focused on only a limited number of species and identified considerable species-specific vulnerabilities. More work is needed to explore the effects of a wider range of contaminants on frogs. Nonetheless, the presence of frogs at wetlands along a strong pollution gradient here still suggests that some urban wetlands may function as sinks for some species (Pulliam 1988).

There is also a strong *prima facie* case that some urban wetlands are ecological traps when the results presented here are considered in conjunction with our previous research, which has shown that breeding adults oviposit in a range of habitats, including those where larval fitness is significantly reduced (Sievers et al. 2018c). The Greater Melbourne Region likely represents an ideal model system that replicates what is occurring in many other urban areas. For example, average concentrations of heavy-metals within stormwater pond sediments were higher in our wetlands for 5 of 7 metals measured in South Carolina (Crawford et al. 2010) and for 3 of 12 metals in Florida (Liebens 2001). Further, levels of the pesticides bifenthrin and permethrin were higher at our sites than in urban waterways in China (Li et al. 2011) and California (Weston et al. 2005), and for diuron in France (Datry et al. 2003). This



This article is protected by copyright. All rights reserved.

Author Manuscript

suggests urban managers could be unintentionally creating ecological traps in hundreds or thousands of cities around the world.

How much of a lasting threat these wetlands pose to population persistence in urban landscapes depends on whether frogs have sufficient phenotypic plasticity or evolutionary potential to adapt to high pollution environments. Amphibians living near agricultural areas with high levels of pesticide exhibit greater tolerance (e.g. Cothran et al. 2013; Hua et al. 2015), and induced tolerance to a single insecticide can lead to cross-tolerance to other insecticides (Hua et al. 2014). The presence of a range of contaminants, often at sublethal levels, within stormwater wetlands may be sufficient to drive evolutionary resistance in exposed populations (Major et al. 2018). Along with phenotypic plasticity, is an important avenue for future research to consider the full impact of stormwater wetlands on amphibians.

Ultimately, urban managers should monitor contaminant loads within stormwater wetlands to identify those with continuously high levels based on published benchmark guidelines (e.g. ANZECC & ARMCANZ 2000), and make efforts to reduce either their occupation by wildlife or pollution input/levels. For example, emergent and fringing vegetation could be minimized or substituted for alternate vegetation (e.g. deep submerged), wetlands in areas known to produce high contaminant loads could be isolated, or runoff could be pre-treated before entering these wetlands. Wetlands are important tools for the management of urban stormwater run-off, but we need to ensure their construction does not facilitate declines in wetland-dependent urban wildlife.

**ACKNOWLEDGMENTS**

We thank M. Ioannides, D. Lenga, L. Myers and J. Rodriguez for assisting with fieldwork, and Rhys Coleman, Mark Burgman and two anonymous reviewers for helpful feedback on the



This article is protected by copyright. All rights reserved.

Author Manuscript

manuscript. We also thank the Centre for Aquatic Pollution Identification and Management (CAPIM) for access to their databases. Research was funded by the Australian Research Council (LP140100343), the Holsworth Wildlife Research Endowment, Melbourne Water, the Nature Conservancy, and was supported in part by the Clean Air and Urban Landscapes Hub of the Australian Government's National Environmental Science Program.

**REFERENCES**

Allinson, M., P. Zhang, A. Bui, J. H. Myers, V. Pettigrove, G. Rose, S. A. Salzman, R. Walters, and G. Allinson. 2017. Herbicides and trace metals in urban waters in Melbourne, Australia (2011–12): concentrations and potential impact. Environmental Science and Pollution Research **24**:7274-7284.

Amweg, E. L., D. P. Weston, and N. M. Ureda. 2005. Use and toxicity of pyrethroid pesticides in the Central Valley, California, USA. Environmental Toxicology and Chemistry **24**:966-972.

ANZECC, and ARMCANZ. 2000. Australian guidelines for water quality monitoring and reporting. Australian and New Zealand Environment and Conservation Council, and Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

Belden, J., S. McMurry, L. Smith, and P. Reilley. 2010. Acute toxicity of fungicide formulations to amphibians at environmentally relevant concentrations. Environmental Toxicology and Chemistry **29**:2477-2480.

Bender, D. J., L. Tischendorf, and L. Fahrig. 2003. Using patch isolation metrics to predict animal movement in binary landscapes. Landscape Ecology **18**:17-39.

Bishop, C. A., J. Struger, L. J. Shirose, L. Dunn, and G. D. Campbell. 2000. Contamination and wildlife communities in stormwater detention ponds in Guelph and the Greater Toronto Area,



This article is protected by copyright. All rights reserved.

Author Manuscript

Ontario, 1997 and 1998. Part II - Contamination and biological effects of contamination. Water Quality Research Journal of Canada **35**:437-474.

Brand, A. B., and J. W. Snodgrass. 2010. Value of Artificial Habitats for Amphibian Reproduction in Altered Landscapes. Conservation Biology **24**:295-301.

Brogan III, W. R., and R. A. Relyea. 2017. Multiple mitigation mechanisms: Effects of submerged plants on the toxicity of nine insecticides to aquatic animals. Environmental Pollution **220**:688-695.

Brown, J. S., M. Sutula, C. Stransky, J. Rudolph, and E. Byron. 2010. Sediment Contaminant Chemistry and Toxicity of Freshwater Urban Wetlands in Southern California1. Journal of the American Water Resources Association **46**:367-384.

Canessa, S., G. W. Heard, K. M. Parris, and M. A. McCarthy. 2012. Integrating variability in detection probabilities when designing wildlife surveys: a case study of amphibians from south-eastern Australia. Biodiversity and Conservation **21**:729-744.

Cothran, R. D., J. M. Brown, and R. A. Relyea. 2013. Proximity to agriculture is correlated with pesticide tolerance: evidence for the evolution of amphibian resistance to modern pesticides. Evolutionary Applications **6**:832-841.

Crawford, K. D., J. E. Weinstein, R. E. Hemingway, T. R. Garner, and G. Globensky. 2010. A survey of metal and pesticide levels in stormwater retention pond sediments in coastal South Carolina. Archives of environmental contamination and toxicology **58**:9-23.

Datry, T., F. Malard, L. Vitry, F. Hervant, and J. Gibert. 2003. Solute dynamics in the bed sediments of a stormwater infiltration basin. Journal of Hydrology **273**:217-233.

Egan, R. S., and P. W. Paton. 2004. Within-pond parameters affecting oviposition by wood frogs and spotted salamanders. Wetlands **24**:1-13.



This article is protected by copyright. All rights reserved.

Author Manuscript

Egea-Serrano, A., R. A. Relyea, M. Tejedo, and M. Torralva. 2012. Understanding of the impact of chemicals on amphibians: a meta-analytic review. Ecology and evolution **2**:1382-1397.

Ficken, K., and P. G. Byrne. 2013. Heavy metal pollution negatively correlates with anuran species richenss and distribution in south-eastern Australia. Austral Ecology **38**:523-533.

Gallagher, M. T., J. W. Snodgrass, A. B. Brand, R. E. Casey, S. M. Lev, and R. J. Van Meter. 2014. The role of pollutant accumulation in determining the use of stormwater ponds by amphibians. Wetlands Ecology and Management **22**:551-564.

Gallagher, M. T., J. W. Snodgrass, D. R. Ownby, A. B. Brand, R. E. Casey, and S. Lev. 2011. Watershed-scale analysis of pollutant distributions in stormwater management ponds. Urban Ecosystems **14**:469-484.

Gibbs, K. E., R. L. Mackey, and D. J. Currie. 2009. Human land use, agriculture, pesticides and losses of imperiled species. Diversity and Distributions **15**:242-253.

Golabek, K., A. Ridley, and A. Radford. 2012. Food availability affects strength of seasonal territorial behaviour in a cooperatively breeding bird. Animal Behaviour **83**:613-619.

Hamer, A., and K. Parris. 2013. Predation Modifies Larval Amphibian Communities in Urban Wetlands. Wetlands **33**:641-652.

Hamer, A. J., and M. J. McDonnell. 2008. Amphibian ecology and conservation in the urbanising world: A review. Biological Conservation **141**:2432-2449.

Hamer, A. J., and K. M. Parris. 2011. Local and landscape determinants of amphibian communities in urban ponds. Ecological Applications **21**:378-390.

Author Manuscript



This article is protected by copyright. All rights reserved.

Hamer, A. J., P. J. Smith, and M. J. McDonnell. 2012. The importance of habitat design and aquatic connectivity in amphibian use of urban stormwater retention ponds. Urban Ecosystems **15**:451-471.

Hanski, I. 1994. Patch-occupancy dynamics in fragmented landscapes. Trends in Ecology & Evolution **9**:131-135.

Hassall, C., and S. Anderson. 2015. Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. Hydrobiologia **745**:137-149.

Hayden, M. T., M. K. Reeves, M. Holyoak, M. Perdue, A. L. King, and S. C. Tobin. 2015. Thrice as easy to catch! Copper and temperature modulate predator-prey interactions in larval dragonflies and anurans. Ecosphere **6**:1-17.

Hoffmann, F., and W. Kloas. 2010. An environmentally relevant endocrine-disrupting antiandrogen, vinclozolin, affects calling behavior of male Xenopus laevis. Hormones and Behavior **58**:653-659.

Hua, J., D. K. Jones, B. M. Mattes, R. D. Cothran, R. A. Relyea, and J. T. Hoverman. 2015. The contribution of phenotypic plasticity to the evolution of insecticide tolerance in amphibian populations. Evolutionary applications **8**:586-596.

Hua, J., D. K. Jones, and R. A. Relyea. 2014. Induced tolerance from a sublethal insecticide leads to cross-tolerance to other insecticides. Environmental science & technology **48**:4078-4085.

Huang, M. Y., R. Y. Duan, and X. Ji. 2015. The influence of long-term cadmium exposure on phonotaxis in male Pelophylax nigromaculata. Chemosphere **119**:763-768.

Kentula, M. E., S. E. Gwin, and S. M. Pierson. 2004. Tracking changes in wetlands with urbanization: Sixteen years of experience in Portland, Oregon, USA. Wetlands **24**:734-743.



This article is protected by copyright. All rights reserved.

Author Manuscript

Kruger, D. J. D., A. J. Hamer, and L. H. Du Preez. 2015. Urbanization affects frog communities at multiple scales in a rapidly developing African city. Urban Ecosystems **18**:1333-1352.

Le Viol, I., F. Chiron, R. Julliard, and C. Kerbiriou. 2012. More amphibians than expected in highway stormwater ponds. Ecological Engineering **47**:146-154.

Lefcort, H., R. Meguire, L. Wilson, and W. Ettinger. 1998. Heavy metals alter the survival, growth, metamorphosis, and antipredatory behavior of Columbia spotted frog (*Rana luteiventris*) tadpoles. Archives of Environmental Contamination and Toxicology **35**:447-456.

Li, H., W. T. Mehler, M. J. Lydy, and J. You. 2011. Occurrence and distribution of sediment-associated insecticides in urban waterways in the Pearl River Delta, China. Chemosphere **82**:1373-1379.

Liebens, J. 2001. Heavy metal contamination of sediments in stormwater management systems: the effect of land use, particle size, and age. Environmental Geology **41**:341-351.

Lunn, D. J., A. Thomas, N. Best, and D. Spiegelhalter. 2000. WinBUGS-a Bayesian modelling framework: concepts, structure, and extensibility. Statistics and computing **10**:325-337.

MacDonald, D. D., C. G. Ingersoll, and T. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of Environmental Contamination and Toxicology **39**:20-31.

Major, K. M., D. P. Weston, M. J. Lydy, G. A. Wellborn, and H. C. Poynton. 2018. Unintentional exposure to terrestrial pesticides drives widespread and predictable evolution of resistance in freshwater crustaceans. Evolutionary Applications **11**:748-761.

Marshall, S., D. Sharley, K. Jeppe, S. Sharp, G. Rose, and V. Pettigrove. 2016. Potentially toxic concentrations of synthetic pyrethroids associated with low density residential land use. Frontiers in Environmental Science **4**:75.



This article is protected by copyright. All rights reserved.

Author Manuscript

Mimbs IV, W. H., J. P. W. Cusaac, L. M. Smith, S. T. McMurry, and J. B. Belden. 2016. Occurrence of current-use fungicides and bifenthrin in Rainwater Basin wetlands. Chemosphere **159**:275-281.

Monastersky, R. 2014. Biodiversity: Life--a status report. Nature **516**:158-161.

Nebeker, A., and G. Schuytema. 1998. Chronic effects of the herbicide diuron on freshwater cladocerans, amphipods, midges, minnows, worms, and snails. Archives of Environmental Contamination and Toxicology **35**:441-446.

Oksanen, J., R. Kindt, P. Legendre, B. O'Hara, M. H. H. Stevens, M. J. Oksanen, and M. Suggests. 2007. The vegan package. Community ecology package **10**:631-637.

Parris, K. M. 2006. Urban amphibian assemblages as metacommunities. Journal of Animal Ecology **75**:757-764.

Parris, K. M., T. W. Norton, and R. B. Cunningham. 1999. A comparison of techniques for sampling amphibians in the forests of south-east Queensland, Australia. Herpetologica:271-283.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. American Naturalist **132**:652-661.

R Development Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ramesh, R., G. Perry, M. Farmer, and K. Griffis-Kyle, L. 2017. Limited Data Used to Make Valid Inference about Targeting Sites for Conservation: A Case Study in Urban Amphibian Ecology. Herpetological Review **48**:1-5.

Reeves, M. K., P. Jensen, C. L. Dolph, M. Holyoak, and K. A. Trust. 2010. Multiple stressors and the cause of amphibian abnormalities. Ecological Monographs **80**:423-440.

Author Manuscript



This article is protected by copyright. All rights reserved.

Ruiz, A. M., J. C. Maerz, A. K. Davis, M. K. Keel, A. R. Ferreira, M. J. Conroy, L. A. Morris, and A. T. Fisk. 2010. Patterns of Development and Abnormalities among Tadpoles in a Constructed Wetland Receiving Treated Wastewater. Environmental Science & Technology **44**:4862-4868.

Scheffers, B. R., and C. A. Paszkowski. 2013. Amphibian use of urban stormwater wetlands: The role of natural habitat features. Landscape and Urban Planning **113**:139-149.

Sharley, D. J., S. M. Sharp, S. Marshall, K. Jeppe, and V. J. Pettigrove. 2017. Linking urban land use to pollutants in constructed wetlands: Implications for stormwater and urban planning. Landscape and Urban Planning **162**:80-91.

Shuman-Goodier, M. E., and C. R. Propper. 2016. A meta-analysis synthesizing the effects of pesticides on swim speed and activity of aquatic vertebrates. Science of the Total Environment **565**:758-766.

Sievers, M., R. Hale, K. M. Parris, and S. E. Swearer. 2018a. Impacts of human-induced environmental change in wetlands on aquatic animals. Biological Reviews **93**:529-554.

Sievers, M., R. Hale, S. E. Swearer, and K. M. Parris. 2018b. Contaminant mixtures interact to impair predator-avoidance behaviours and survival in a larval amphibian. Ecotoxicology and environmental safety **161**:482-488.

Sievers, M., K. M. Parris, S. E. Swearer, and R. Hale. 2018c. Stormwater wetlands can function as ecological traps for urban frogs. Ecological Applications.

Sinsch, U. 2014. Movement ecology of amphibians: from individual migratory behaviour to spatially structured populations in heterogeneous landscapes 1, 2. Canadian Journal of Zoology **92**:491-502.

Author Manuscript



This article is protected by copyright. All rights reserved.

Snodgrass, J. W., R. E. Casey, D. Joseph, and J. A. Simon. 2008. Microcosm investigations of stormwater pond sediment toxicity to embryonic and larval amphibians: Variation in sensitivity among species. Environmental Pollution **154**:291-297.

Spiegelhalter, D., A. Thomas, N. Best, and D. Lunn. 2007. OpenBUGS user manual, version 3.0. 2. MRC Biostatistics Unit, Cambridge.

Takahashi, M. 2007. Oviposition site selection: Pesticide avoidance by Gray Treefrogs. Environmental Toxicology and Chemistry **26**:1476-1480.

Tarr, T. L., and K. J. Babbitt. 2002. Effects of habitat complexity and predator identity on predation of Rana clamitans larvae. Amphibia-Reptilia **23**:13-20.

Tixier, G. T., M. Lafont, L. Grapentine, Q. Rochfort, and J. Marsalek. 2011. Ecological risk assessment of urban stormwater ponds: literature review and proposal of a new conceptual approach providing ecological quality goals and the associated bioassessment tools. Ecological Indicators **11**:1497-1506.

Vonesh, J. R., and J. C. Buck. 2007. Pesticide alters oviposition site selection in gray treefrogs. Oecologia **154**:219-226.

Wake, D. B., and V. T. Vredenburg. 2008. Are we in the midst of the sixth mass extinction? A view from the world of amphibians. Proceedings of the National Academy of Sciences **105**:11466-11473.

Weston, D., R. Holmes, J. You, and M. Lydy. 2005. Aquatic toxicity due to residential use of pyrethroid insecticides. Environmental Science & Technology **39**:9778-9784.

WHO. 2017. World Health Organisation. Available from: http://www.who.int/gho/urban_health/situation_trends/urban_population_growth/en/.



This article is protected by copyright. All rights reserved.

Author Manuscript

Zhu, L., H. Wang, H. Liu, and W. Li. 2015. Effect of trifloxystrobin on hatching, survival, and gene expression of endocrine biomarkers in early life stages of medaka (Oryzias latipes). Environmental toxicology **30**:648-655.

Author Manuscript

**TABLES**

Table 1. Minimum, maximum and mean (± SD) sediment quality variables for the 67 wetlands, and where possible, the low and high interim sediment quality guideline (ISQG) values from ANZECC and ARMCANZ (2000), and the concentrations of metals within wetland sediments from Snodgrass et al 2008 (Snodgrass I – Owings Mills; Snodgrass II – New Cut Road), Sievers et al 2018c (Sievers I – Chandler Rd; Sievers II – Cheltenham; Sievers III Ringwood Lake), and Gallagher et al (2011).

| Contaminant | Min | Max | Mean ± SE | ISQG-Low | ISQG-High | Snod I | Snod II | Sievers I | Sievers II | Sievers III | Gallagher (Mean ± SD) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *Heavy metals (mg kg-1 dry weight)* | | | | | | | | | | | |
| Aluminium | 6850 | 37100 | 17311 ± 691 | | | | | 16200 | 20000 | 9570 | |
| Antimony | 2.5* | 17 | 4.2 ± 0.4 | 2 | 25 | | | 5 | 2.5* | 9 | |
| Arsenic | 2.5* | 47 | 7.9 ± 1.0 | 5 | 70 | 0.3 | 0.6 | 39 | 37 | 10 | |
| Barium | 40 | 430 | 130 ± 9 | | | | | 430 | 150 | 60 | |
| Beryllium | 0.5* | 2 | 0.73 ± 0.04 | | | | | 0.5* | 1 | 0.5* | |
| Cadmium | 0.5* | 11 | 0.77 ± 0.16 | 1.5 | 10 | 0.3 | 0.6 | 2 | 2 | 0.5* | |
| Chromium | 12 | 121 | 40.5 ± 2.3 | 80 | 370 | 14.9 | 9.4 | 73 | 48 | 21 | 55.2 ± 8.1 |



This article is protected by copyright. All rights reserved.

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Cobalt | 2 | 22 | 10.7 ± 0.7 | | | | | 19 | 13 | 3 | |
| Copper | 11 | 1090 | 64 ± 16 | 65 | 270 | 13.7 | 7 | 299 | 113 | 40 | 46.6 ± 17.6 |
| Iron | 6360 | 36900 | 23859 ± 801 | | | | | 29300 | 28000 | 14300 | |
| Lead | 10 | 456 | 46.1 ± 8.6 | 50 | 220 | 8.4 | 10.6 | 176 | 283 | 48 | 24.7 ± 10.0 |
| Manganese | 36 | 496 | 154 ± 13 | | | | | 187 | 136 | 55 | |
| Mercury | 0.05* | 2.9 | 0.13 ± 0.04 | 0.15 | 1 | | | 0.4 | 0.4 | 0.05* | |
| Molybdenum | 1* | 13 | 1.5 ± 0.2 | | | | | 13 | 2 | 1* | |
| Nickel | 4 | 159 | 26.7 ± 2.6 | 21 | 52 | 14.2 | 4.3 | 159 | 36 | 10 | 63.0 ± 152.4 |
| Silver | 1* | 6 | 1.07 ± 0.07 | 1 | 3.7 | | | 1* | 1* | 1* | |
| Strontium | 7 | 151 | 25.6 ± 2.8 | | | | | 38 | 37 | 11 | |
| Thallium | 2.5* | 2.5* | | | | | | 2.5* | 2.5* | 2.5* | |
| Tin | 2.5* | 48 | 7.3 ± 1.0 | | | | | 48 | 11 | 2.5 | |
| Titanium | 20 | 620 | 150.4 ± 14.2 | | | | | 430 | 370 | 90 | |
| Vanadium | 14 | 130 | 41.0 ± 2.2 | | | | | 47 | 58 | 27 | |
| Zinc | 23 | 3790 | 458 ± 82 | 200 | 410 | 56.5 | 62.7 | 3790 | 2390 | 597 | 189.1 ± 128.5 |
| *Pesticides (mg kg-1 dry weight)* | | | | | | | | | | | |
| Bifenthrin | 1* | 330 | 39.5 ± 6.6 | | | | | 68.4 | 37.2 | 13 | |
| Diethyltoluamide | 1* | 33 | 8.1 ± 0.9 | | | | | 2.5 | 1* | 15 | |
| Diuron | 1.25* | 5316 | 118 ± 78 | | | | | 20 | 22 | 91 | |

Author Manuscript



This article is protected by copyright. All rights reserved.

| | | | | | | |
|---|---|---|---|---|---|---|
| Fenamiphos | 5* | 69.9 | 11.5 ± 1.3 | 10 | 10 | 15 |
| Permethrin | 1.5* | 930.5 | 26.6 ± 13.9 | 209 | 39.4 | 32.8 |
| Prometryn | 1* | 30.8 | 8.1 ± 1.2 | 2.5 | 5 | 27 |
| Pyrimethanil | 0.5* | 7.6 | 2.0 ± 0.2 | 7.6 | 1 | 5 |
| Triclosan | 5* | 155 | 21.4 ± 3.3 | 155 | 80.8 | 60 |
| Trifloxystrobin | 1* | 49 | 7.1 ± 1.1 | 2.5 | 2.5 | 5 |

Table 2. Minimum, maximum and mean (± SE) wetland and landscape-level attributes for the 67 wetlands.

| Variable | Min | Max | Mean ± SE |
|---|---|---|---|
| *Wetland attributes* | | | |
| Area ($m^2$) | 136 | 97282 | 10127 ± 2114 |
| Perimeter (m) | 55 | 4781 | 649 ± 102 |
| Vegetation (prop) | 0.025 | 0.925 | 58.9 ± 0.03 |
| Closest road (m) | 1 | 425 | 64.1 ± 8.0 |
| pH | 6.8 | 8.6 | 7.66 ± 0.04 |
| Electrical conductance (µS) | 87 | 3339 | 526 ± 71 |
| *Landscape-level characteristics* | | | |

Author Manuscript



This article is protected by copyright. All rights reserved.

| | | | |
|---|---|---|---|
| Urban (proportion of sub-catchment) | 0 | 1.00 | 0.67 ± 0.03 |
| Road (proportion of sub-catchment) | 0 | 0.18 | 0.03 ± 0.005 |
| Habitat 500m* ($m^2$) | 723 | 323207 | 38966 ± 8172 |

* Habitat 500m refers to the amount ($m^2$) of wetland habitat in a 500m radius surrounding each wetland.

Table 3. Frog species detected as adults during the study and the percentage of wetlands at which they were detected.

| Family/species | Common name | % occupied |
|---|---|---|
| Hylidae | | |
| *Litoria ewingii* | Southern brown tree frog | 43.3% |
| *Litoria peronii* | Peron's tree frog | 17.9% |
| *Litoria raniformis* | Growling grass frog | 6.0% |
| *Litoria fallax* | Eastern dwarf tree frog | 4.5% |
| *Litoria verreauxii* | Whistling tree frog | 3.0% |
| Myobatrachidae | | |
| *Crinia signifera* | Common eastern froglet | 58.2% |
| *Limnodynastes dumerilii* | Eastern banjo frog | 40.3% |
| *Limnodynastes peronii* | Striped marsh frog | 40.3% |
| *Limnodynastes tasmaniensis* | Spotted marsh frog | 34.3% |



This article is protected by copyright. All rights reserved.

Author Manuscript

Table 4. Deviance information criterion (DIC) values for the 12 regression models of species richness (Poisson) and species occurrence (Logistic) at a wetland, with boldface values indicating models for which ΔDIC<2.

| | | | | | | | Species richness | | *Crinia signifera* | | *Limnodynastes tasmaniensis* | | *Limnodynastes dumerilii* | | *Limnodynastes peronii* | | *Litoria ewingi* | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Model | Variables | | | | | | DIC | ΔDIC | DIC | ΔDIC | DIC | ΔDIC | DIC | ΔDIC | DIC | ΔDIC | DIC | ΔDIC |
| 1 | Const. | | | | | | 241.9 | 3.6 | 93.1 | 10.3 | 88.2 | 2.3 | 92.4 | 8.4 | **92.4** | **0.0** | 93.7 | 35.0 |
| 2 | Const. | Veg | | | | | **238.3** | **0.0** | 88.3 | 5.6 | **85.9** | **0.0** | 94.2 | 10.2 | **93.5** | **1.1** | 95.8 | 37.1 |
| 3 | Const. | Veg | HM | | | | **239.8** | **1.5** | 90.3 | 7.6 | 88.0 | 2.0 | 95.0 | 11.0 | **92.7** | **0.4** | 96.7 | 38.0 |
| 4 | Const. | Veg | Hab | | | | **239.5** | **1.2** | **83.9** | **1.1** | **87.2** | **1.3** | 94.1 | 10.1 | 95.5 | 3.1 | 96.9 | 38.2 |
| 5 | Const. | Veg | HM | Hab | | | 241.1 | 2.8 | 86.0 | 3.2 | 89.4 | 3.4 | 94.4 | 10.4 | 94.9 | 2.5 | 98.1 | 39.4 |
| 6 | Const. | Veg | HM | Bifen | Diu | Trif | 242.7 | 4.4 | 94.3 | 11.5 | 91.7 | 5.7 | 95.2 | 11.2 | 97.0 | 4.6 | 89.2 | 30.5 |
| 7 | Co | V | H | C | | | 2 | 2. | 9 | 9. | 9 | 4. | 9 | 9. | **9** | **1.** | 9 | 3 |



This article is protected by copyright. All rights reserved.

Author Manuscript

| | | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | nst. | eg | M | ond | | | 41.1 | 8 | 2.4 | 6 | 0.0 | 0 | 3.5 | 5 | **3.9** | **5** | 8.4 | 9.7 |
| 8 | Const. | Hab | Bifen | Diu | Trif | | 246.7 | 8.4 | 95.9 | 13.2 | 92.9 | 7.0 | 91.8 | 7.8 | 98.6 | 6.3 | 90.3 | 31.6 |
| 9 | Const. | Urb | Hab | Geol | Road | | 245.9 | 7.6 | 92.6 | 9.8 | 96.3 | 10.3 | **84.0** | **0.0** | 94.9 | 2.6 | **59.4** | **0.7** |
| 10 | Const. | Urb | Geol | Road | | | 244.3 | 6.0 | 94.2 | 11.5 | 94.4 | 8.5 | **84.9** | **0.9** | **92.8** | **0.4** | **58.7** | **0.0** |
| 11 | Const. | Veg | Area | Hab | Geol | | 241.4 | 3.1 | **82.8** | **0.0** | 91.3 | 5.3 | 95.3 | 11.3 | 99.4 | 7.0 | 63.3 | 4.6 |
| 12 | Const. | Veg | HM | Hab | Urb | Area | 242.4 | 4.1 | 90.1 | 7.3 | 93.4 | 7.4 | **85.6** | **1.6** | 96.4 | 4.0 | 102.3 | 43.6 |

Key to model variable abbreviations: Const, model constant; Veg, total proportion of aquatic vegetation; HM, sediment heavy metal concentration index (see Ficken and Byrne 2014; ln+1 transformed); Hab, area of wetland habitat within a 500m radius (ln-transformed); Bif, sediment concentration of the insecticide bifenthrin (ln-transformed); Diu, sediment concentration of the herbicide diuron (ln-transformed); Trif, sediment concentration of the fungicide trifloxystrobin (ln-transformed); Urb, proportion of the sub-catchment classified as urban; Road, proportion of the sub-catchment classified as road; Geology, sedimentary or basalt catchment geology ; Area, wetland area (ln-transformed), and; Cond, electrical conductivity (ln-transformed).



This article is protected by copyright. All rights reserved.

Author Manuscript

**FIGURES LEGENDS**

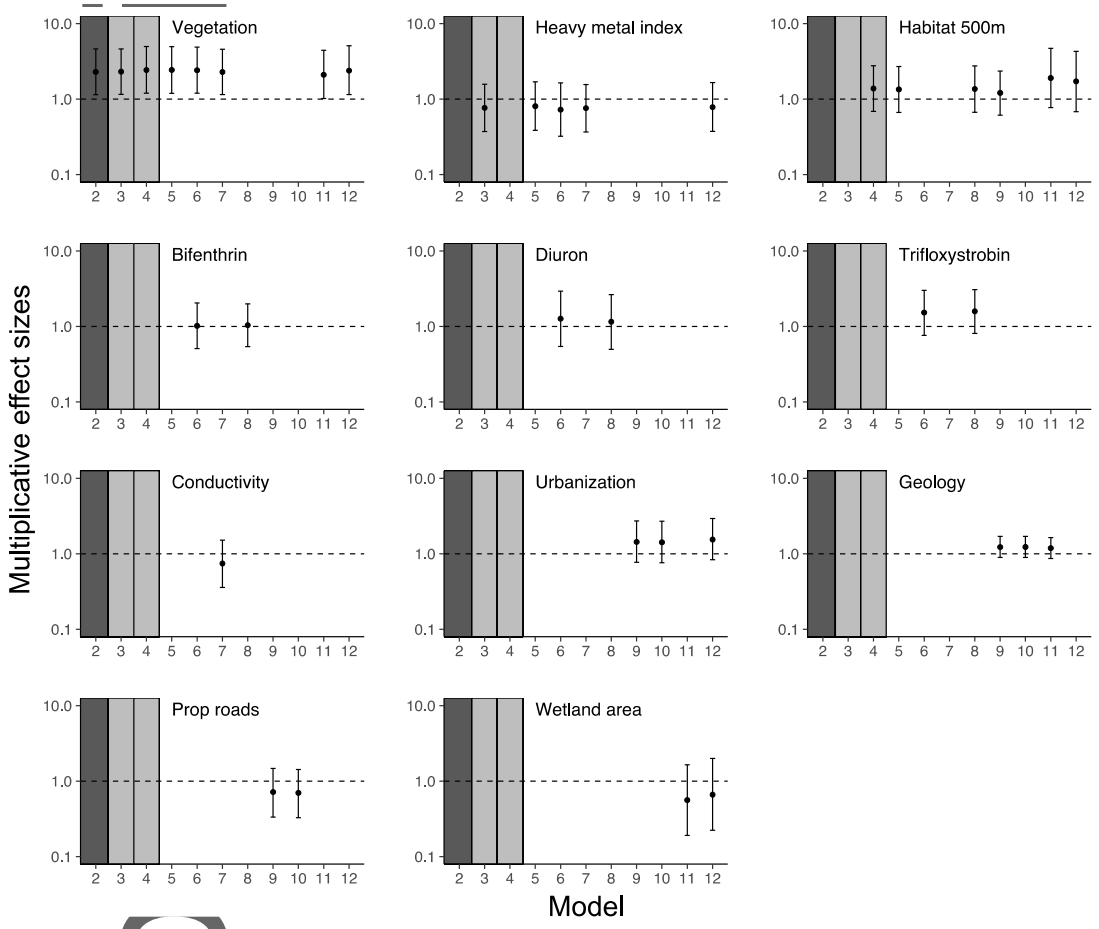



Figure 1. The multiplicative effect of the explanatory variables (mean and BCIs) on adult species richness at 67 wetlands. The dark grey band indicates the best-supported model (i.e. ΔDIC = 0), and the light grey band indicates models with some support (ΔDIC < 2). See Table 3 for model numbers and structure. Effect sizes > 1 indicate a positive effect of the explanatory variable on species richness while effect sizes < 1 indicate negative effects. Model 1 (constant) is not shown.

Author Manuscript



This article is protected by copyright. All rights reserved.

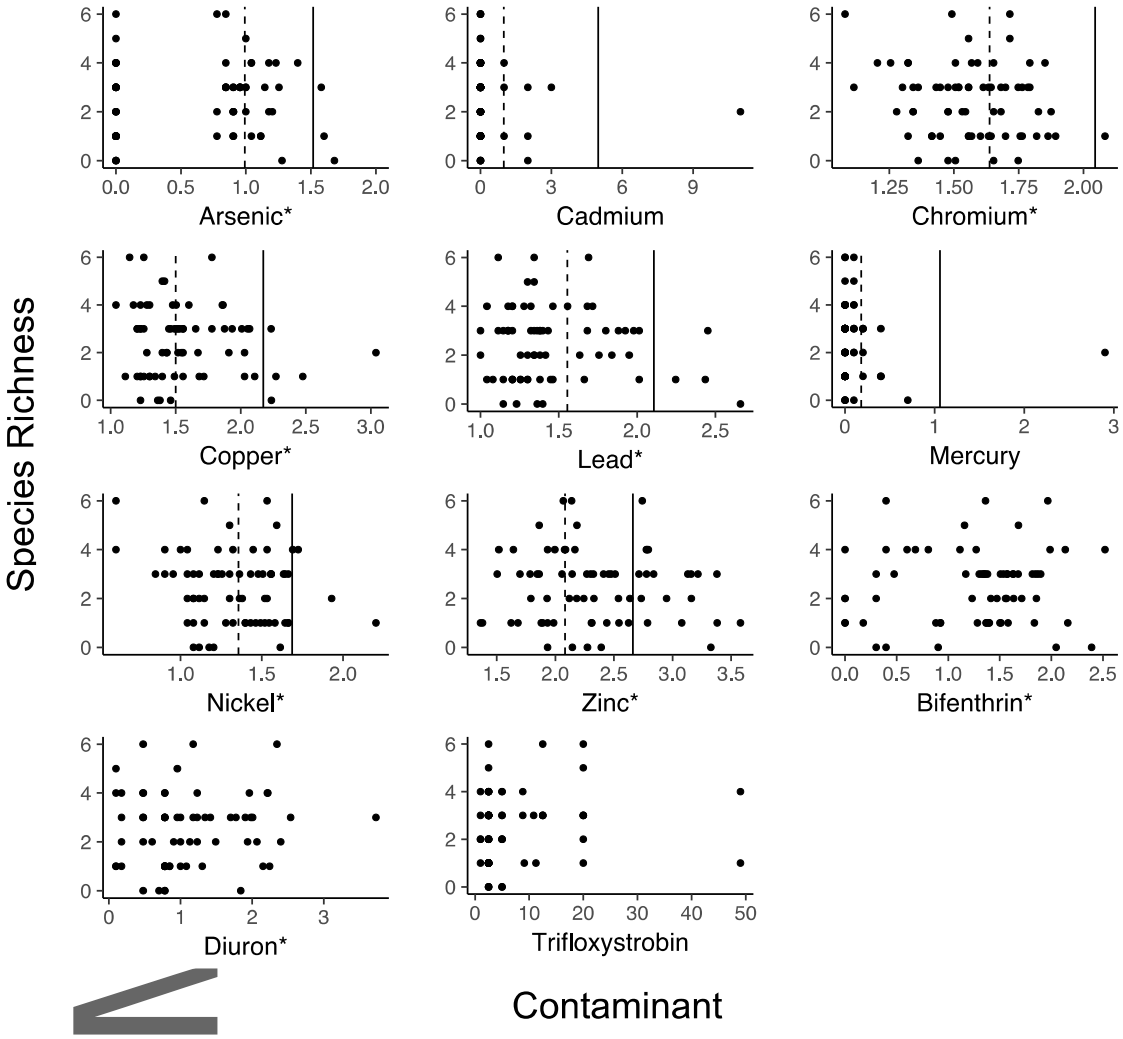
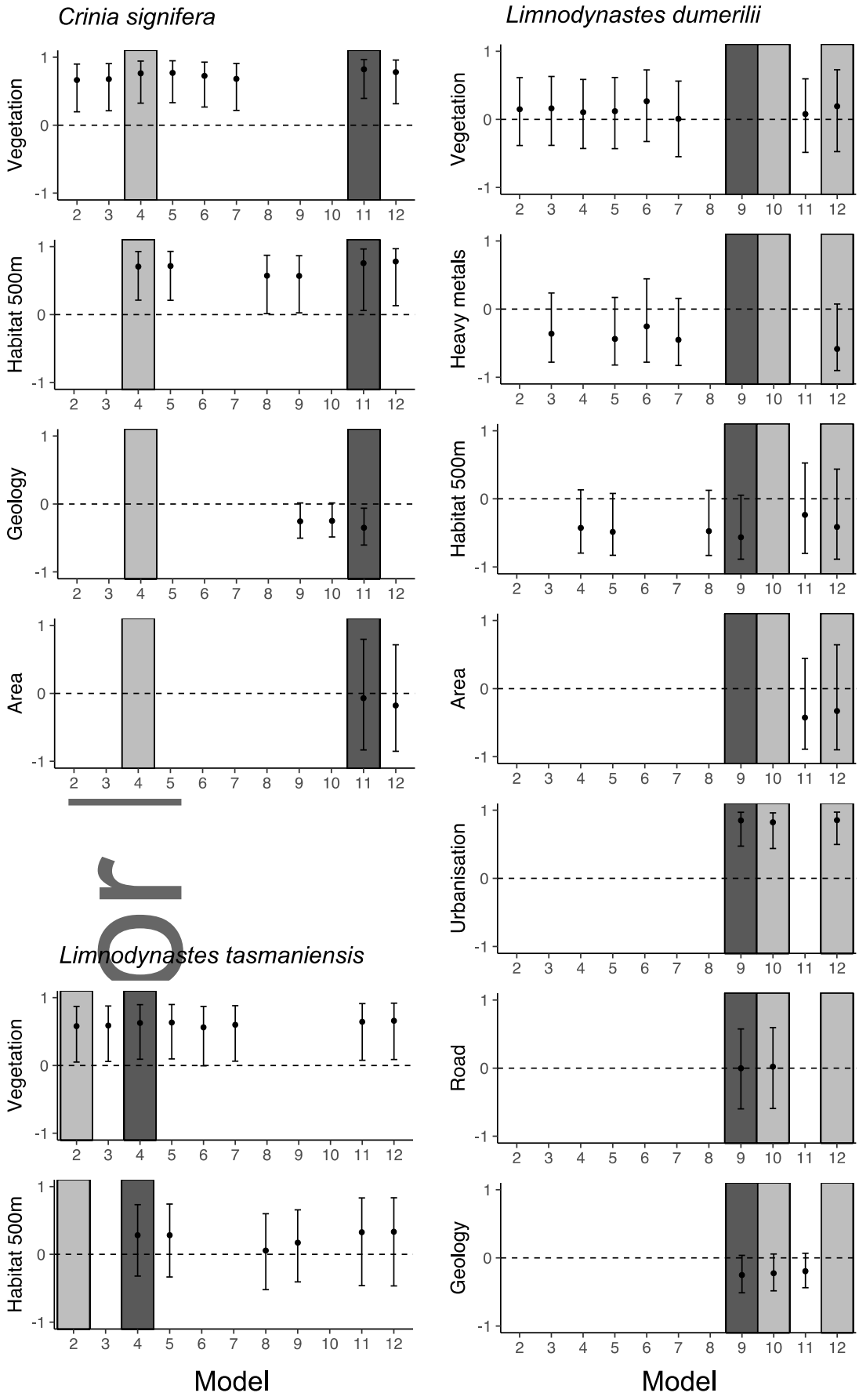



Figure 2. Frog species richness plotted against the concentration of heavy metals (mg/kg) and pesticides (mg/kg) in wetland sediments. Dashed lines represent the threshold effect concentration (below which harmful effects are unlikely to be observed) and solid lines represent the probable effect concentration (above which harmful effects are likely to be observed) based on MacDonald et al. (2000). Some contaminants are log-transformed to aid visualization (asterisks).

Author



This article is protected by copyright. All rights reserved.






This article is protected by copyright. All rights reserved.

Figure 3. The multiplicative effect of the explanatory variables (mean and BCIs) on species occurrence at 67 wetlands. Only variables present in well-supported models for each species (ΔDIC < 2) are shown. The dark grey band indicates the best-supported model (i.e. ΔDIC = 0), and the light grey bands indicate models with some support (ΔDIC < 2). See Table 3 for model numbers and structure. Effect sizes > 0 indicate a positive effect of the explanatory variable on species occurrence while effect sizes < 0 indicate negative effects. Model 1 (constant) is not shown.

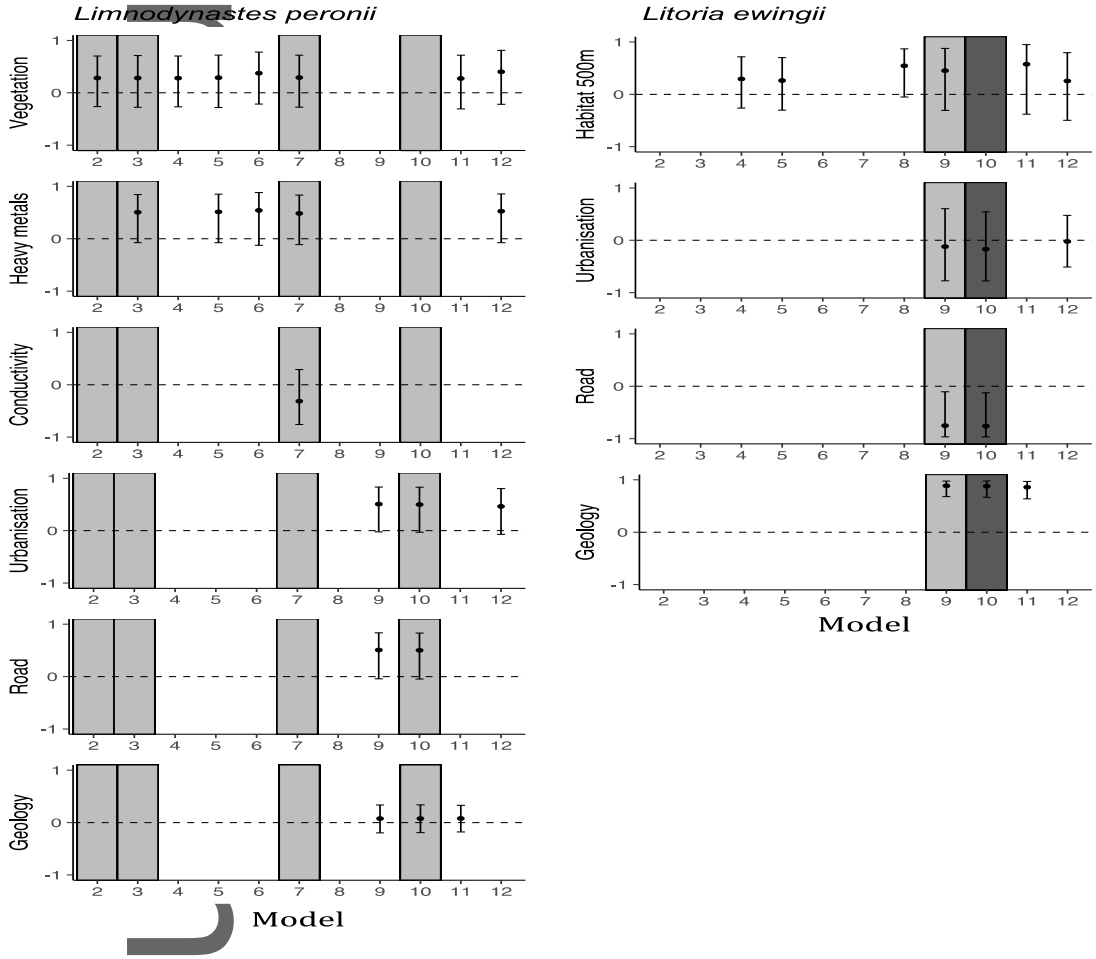



Figure 3 continued.



This article is protected by copyright. All rights reserved.

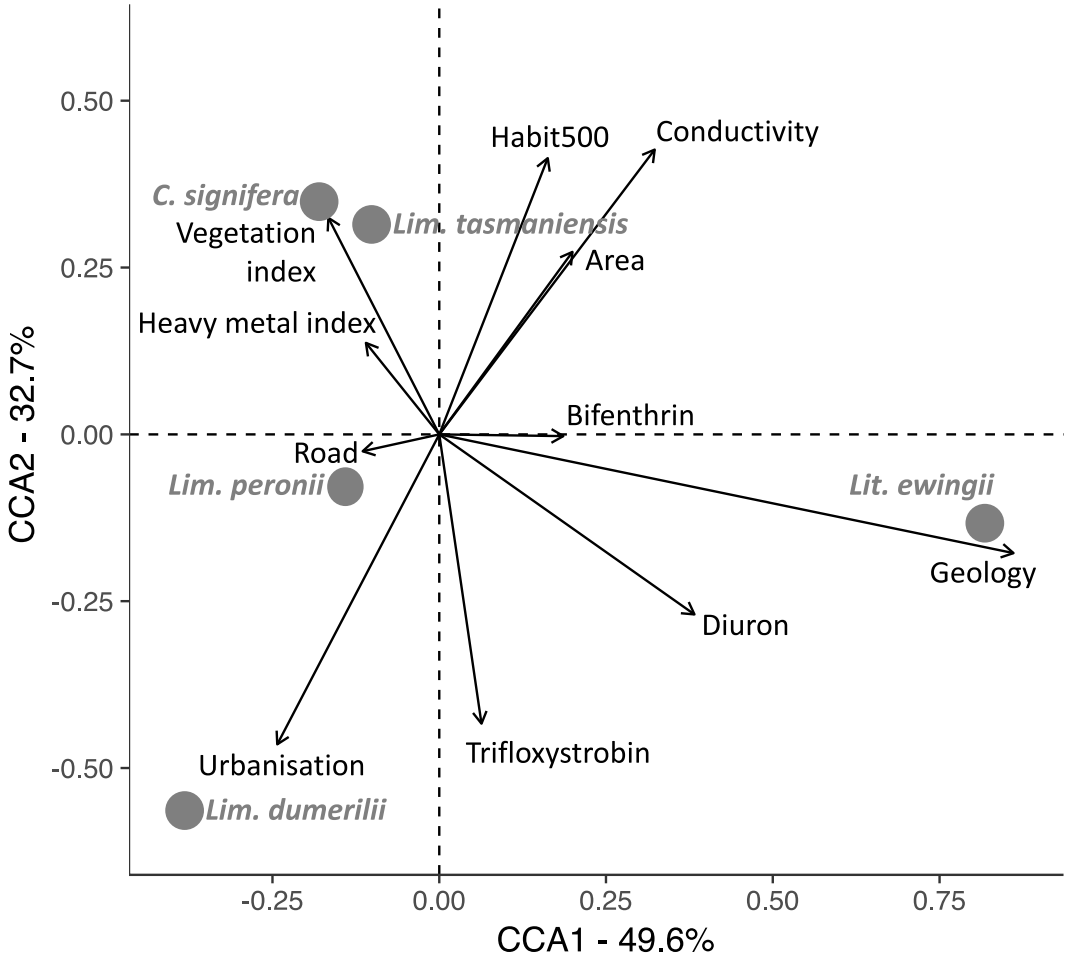



Figure 4. Ordination diagram (biplot) of the canonical correspondence analysis (CCA) for the detection/non-detection of adults according to eleven explanatory variables at 61 wetlands in the Greater Melbourne area, Australia. See Table 3 for a description of variables.

Author



This article is protected by copyright. All rights reserved.