One pond fits all? Frogs as an indicator of urban wetland health

Final report to Upper Murrumbidgee Waterwatch

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Contents

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Executive Summary

Biodiversity is threatened by increased urbanization. However, well designed urban environments can still provide quality habitat for a range of plants and animal communities, provided key functional attributes are retained and preserved. This study aimed to examine frog-habitat relationships in 33 urban and peri-urban wetlands situated in Canberra. The specific habitat requirements of frogs in Canberra, particularly around urban wetlands, were unknown prior to this study. We contended that if specific wetland attributes are correlated with frog species persistence at wetlands, it would be plausible to use frogs as an indicator of wetland health.

Using citizen science data collected by the ACT Frogwatch program, we modelled frog presence/absence data collected at these 33 wetlands, against additional data regarding connectivity to remnant forest, riparian habitat condition, aquatic habitat condition, presence of invasive fish, *Gambusia holbrooki*, and water quality, to determine if frog presence/absence is driven by attributes of wetland condition. We established a series of hypotheses to explain species richness (the number of species at a wetland) and the presence of each individual species at a wetland.

Pond connectivity to a reserve

We found that species richness (the number of frog species at a wetland) was primarily influenced by the presence of fallen timber in the surrounding riparian zone of the wetland. Conversely, mowing around urban wetlands was found to have a negative impact on frog species richness. Four species of frog were found to provide

limited information regarding wetland condition. Three of these species are habitat generalists and highly abundant at most sites, while one species is rarely detected at wetlands in the ACT region.

The remaining five species, Plains froglet *Crinia parinsignifera*, Eastern banjo frog, *Limnodynastes dumerilii*, Striped marsh frog *Limnodynastes peroni*, Peron's tree frog *Litoria peronii*, and the Smooth toadlet *Uperoleia laevigata* are highly informative, indicator species for assessing wetland health. No single trait was informative for all species. Instead, a range of connectivity, riparian condition, aquatic habitat condition, feral fish, and water quality traits explained the presence of individual species. In particular, connectivity to one hectare or larger patches of remnant forest were important to *Crinia parinsignifera*, *Limnodynastes dumerilii*, *Limnodynastes peroni*, and *Uperoleia laevigata*. Likewise, fallen logs in the riparian zone were important for the presence of *Crina parinsignifera*, *Litoria peronii*, and *Uperoleia laevigata*. The presence of *Gambusia holbrooki* had contrasting impacts on the presence of *Limnodynastes peroni* (positive effect) and *Uperoleia laevigata* (negative effect). Additional research is required to explore the interactions between *Gambusia holbrooki* and these two species of frog to further explain this result.

The Smooth toadlet, *Uperoleia laevigata* stands out as being a ‘bellwether’ species, being particularly responsive to habitat connectivity, the presence of *Gambusia holbrooki*, riparian condition and water quality. We highlight the need for additional research to understand how frog early life history phases respond to wetland condition. This study further exemplifies the benefits citizen science programs can offer to conservation and management of biodiversity and ecosystems. We make recommendations regarding future wetland design and management to maximise the benefits urban wetland environments may contribute to supporting healthy and diverse populations of frogs. We propose a monitoring program using citizen scientists to provide long-term information regarding the health of urban wetlands, and the incorporation of frog presence data as bio-indicators into ecological assessments of ecosystem health, throughout a range of urban and peri-urban environments.

Emergent pond vegetation

Introduction

Anurans (frogs and toads) are both ecologically important and iconic in natural ecosystems (Hocking and Babbitt 2014). In Australia, the frog fauna is dominated by three major taxonomic groups, the Hylidae, Microhylidae and the Myobatrachidae (Cogger 2014). Most species are endemic to Australia, with the exception of some that are shared with Papua New Guinea (Barker et al. 1995), and regional endemism is high (Slatyer et al. 2007). Biodiversity hotspots exist, primarily in the Wet Tropics, central Queensland, and south-west Western Australia (Morgan et al. 2007, Slatyer et al. 2007, Tyler & Knight 2011). In total, there are approximately 240 described species in Australia (Cogger 2014).

Anurans are often considered to be ecosystem health indicator species, which is due to their sensitivity to environmental change and the globally recognized extinction crisis they are currently experiencing (Hero & Morrison 2004, Whittaker et al. 2013). This can be attributed to their complex, bipartite life history. Typically (but many counter-examples exist), eggs and larvae exist in aquatic habitats, before undergoing a major metamorphosis into a riparian, or terrestrial adult phase (Anstis 2013). Due to this complex life cycle, they are recognized as indicator species of environmental change (Pechmann & Wilbur 1994) and can signify the habitat quality in water (Dunson et al. 1992, Fairweather & Napier 1998) and on land (Cranston et al. 1996). Their sensitivity to environmental changes is based on their highly permeable skin throughout their life span (Stebbins & Cohen 1995) and their wide range of specific habitats for breeding, development, foraging, hibernating and predator avoidance (Joly & Morand 1997, Welsh & Ollivier 1998).

Spotted marsh frog metamorphling and froglet

Over the last three decades scientists have become increasingly aware of a rapid decline in frog numbers and the extinction of many frog species globally (Campbell 1999, Whittaker et al. 2013). The key threatening processes are complex and subject of much ongoing research. Climate change effects, habitat loss, low water quality, introduced diseases and pest species are amongst the main threatening processes (Stuart et al. 2004, Whittaker et al. 2013). Within Australia, there have been 4 known extinctions, with 12.5% of the Australian frog fauna listed as critically endangered, endangered or threatened (Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act)). Within the ACT, frogs are exposed to all of the mentioned threats and three species out of 18 found in the region (Bennett 1997) are listed under the EPBC Act.

The ACT is a moderately urbanised territory, with land being urban (13%), agricultural/forestry (25%) and conservation managed (60%) (ABARES 2012). Furthermore, population growth projections for the ACT suggest an increasing population from just under 400,000 people as of 2016 to 750,000 by 2062 (ACT Treasury Report 2013). Overall estimates of rural and conservation holdings are expected to change over this time, due to infill and suburban expansion (Canberra Spatial Plan 2004). Within these new urban developments, the creation of recreational and natural amenity infrastructure is occurring, with much of the relevant frog habitats likely to be encompassed in water-sensitive urban design principles (Kazemi et al. 2011, Hamer et al. 2012). As such, the design, implementation and management of water related infrastructure could include specific design and management policies to enhance the value these infrastructure as key frog habitats within urban environments (e.g. Hazell 2003, Lemckert et al. 2006, Hamer et al. 2012).

Within the urban area of the ACT 9 frog species are regularly detected and a recent study explored the landscape-scale drivers of their distribution and contemporary changes in distribution (Westgate et al. 2015). This study and others throughout south eastern Australia have shown that a range of biotic and abiotic characteristics can influence frog occurrence, including canopy cover and pond type (Hazell et al. 2001), water body depth, bank and emergent vegetation (Lemckert et al. 2006, Hazell et al. 2004), pond size and aquatic vegetation (Hamer et al. 2012), riparian buffer zones and grazing pressure (Jansen and Healey 2003), and predatory fish (Gillespie and Hero 1999, Holbrook & Dorn 2016). However, each of these studies shows a high variance between species in terms of species-habitat relationships and what seems beneficial for one species might be less desired by another. In addition, most studies to date have focused on rural and conservation-managed lands (but see Hamer et al. 2012). Little is known about how urban areas can help sustaining diverse frog assemblages, despite the recognition of the importance of modified landscapes per se for supporting frog diversity three decades ago (Ehmann & Cogger 1985). In Australia, the lack of research in this area has resulted in

management practises for frogs in modified landscapes that are based on inappropriate assumptions, rather than on evidence based guidelines (Hazell 2003). A first step towards changing this are the guiding principles developed by Lindenmayer et al. (2003) for wildlife conservation on rural farms.

Dam at Mulligans Flat Nature Reserve

Given that urbanisation has been found to influence species-habitat relationships (Guzy et al. 2012), exploration of these patterns within an urban context is vital to producing meaningful relationships. With this information, informed decisions can be made regarding the design, implementation and management of frog-sensitive urban infrastructure, while changes in frog distributions within urban environments can be used as an indicator of environmental change, with the likely specific drivers of change identified.

Ongoing monitoring of frog populations in urban environments can fulfil many environmental and social objectives. Frogs are an excellent vehicle for community engagement in local environmental issues, and are commonly utilised educational programs both within Australia and overseas. Secondly, the sensitivity of frogs to their environment and their relative ease of detection using call surveys makes them an ideal candidate for the monitoring of wetland health in urban and rural environments. However, to facilitate direct inferences from frog species presence to specific environmental attributes requires detailed knowledge of species-habitat relationships. The overall objective of this study is to determine the species-habitat relationships of frogs at wetlands in the ACT urban and peri-urban fringe.

Aims:

1. Determine the species-habitat relationships of frogs in the ACT urban/peri-urban wetlands
2. Examine the role that frogs can form as an indicator of urban wetland health.
3. Develop specific recommendations for wetland design and management based species-habitat relationships

Spotted marsh frog male calling

Material and Methods

Study area and species

This study was conducted in the Australian Capital Territory (ACT), south-eastern Australia, in September and October 2015.

Out of the 18 frog species occurring in the ACT and nearby regions, nine species from three different families are commonly encountered in and around urban areas: Hylidae: *Litoria peronii* and *Litoria verreauxii;* Limnodynastidae: *Limnodynastes* dumerilii, *Limnodynastes peroni* and *Limnodynastes tasmaniensis*; Myobatrachidae: *Crinia signifera*, *Crinia parinsignifera*, *Uperoleia laevigata* and *Neobatrachus sudelli* (Bennett 1997, Westgate et al. 2015).

Field sites

All field sites in this study are part of the large scale annual Citizen science program *ACT and Region Frogwatch Census.* The FrogCensus runs in October when most local species are breeding, and has a strong focus on the National Water Week (third week in October). The Frogwatch program uses auditory sampling to record occurrence of calling males, following a standardized procedure (Frogwatch Census Manual 2008).

Out of the 480 established Frogwatch sites we selected 33 sites around small wetlands or isolated ponds (Figure 1) with a minimum of 3 years monitoring history with approximately ten sites each for frog densities of 6-8 species (high), 4-5 species (medium) and 1-3 species (low). For consistent terminology we will use the descriptors "site" and "wetland" for all field sites in this publication. A complete table of site codes, location descriptions and coordinates can be found in Appendix 3.

Field site FGD010

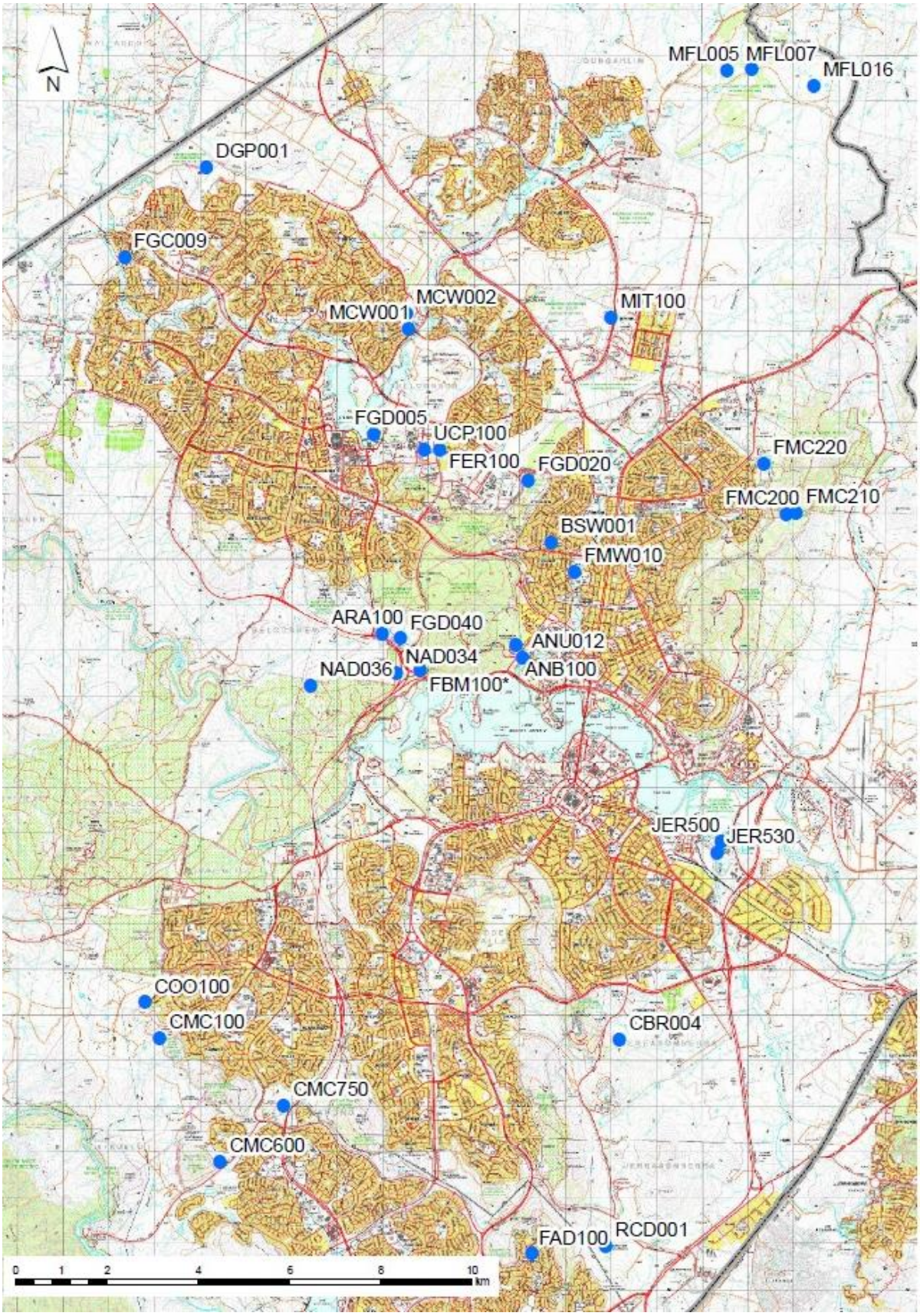


Figure 1. Distribution of 33 urban wetlands monitored in this survey. Map prepared by David Wong, Ginninderra Catchment Group.

Frog surveys

We conducted up to 5 nocturnal call surveys at each site during September and October 2015. All surveyed frog species were breeding during this time (Lintermans and Osborne 2002, Westgate et al. 2015).

Surveys were done during the first three hours after dark and included actively listening to the frog chorus at a designated spot and taking a three minute recording (Frogwatch Census Manual 2008). All recordings were validated by the Frogwatch coordinator (A.M. Hoefer) prior to inclusion in the dataset.

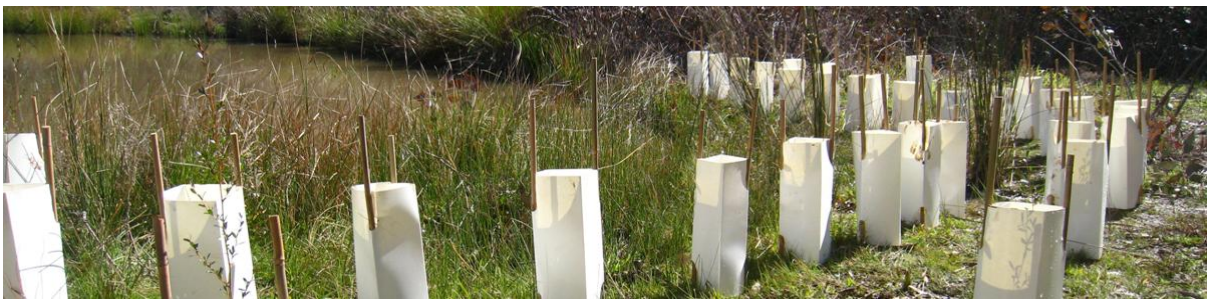
Habitat assessments - RARC

We assessed riparian condition using the "Rapid Appraisal of Riparian Condition" (RARC) tool (Jansen et al. 2005) (Appendix 1). This tool was developed to assess the biodiversity and functioning of riparian zones along creeks and rivers in the Murrumbidgee and Gippsland Regions in south-eastern Australia. It scores various indicators for five sub-indices of ecosystem functions- *Habitat, Cover, Natives, Debris* and *Features*. All RARCs were conducted in accordance with the original directions, the only adjustment being the re-positioning of transects to reflect its application for small to medium size static water bodies instead of its primary use for riverine environments.

We used transects in North, East, West, South direction from the wetland's centre instead of transects parallel along the bank (see Appendix 1). The Cardinal points were established by using a compass and detailed aerial mapping of the region. The length of transect was determined by the typology around the wetland and the extent of the riparian zone. However, transects were no more than 200m long. *Wetland area* (surface), *wetland width* and *connectivity* were determined using google maps.

Habitat assessments - FRARC

To capture additional fine-scale data related to frog-specific habitat variables and important features of wetlands that differ from riverine habitats we designed a frog-habitat assessment sheet (FRARC, Appendix 2). The FRARC scores supplementary variables along the water body, within the *full-level-zone* (water's edge to full level) and *in-water* area.



Ephemeral zone revegetation at Uriarra Village Pond

Fish surveys

We conducted visual surveys for Plague Minnow (*Gambusia holbrooki*) at all 33 wetlands. This small fresh water species, originally introduced into Australia in the 1920s, is an aggressive and voracious predator (Lintermans 2007). Its detrimental effect on fish, invertebrates and frogs has been well documented (e.g. Grubb 1972, Morgan & Buttermer 1996, Webb & Joss 1997, Gillespie and Hero 1999). This species has a very high reproductive rate, with often low population densities through the cooler months, followed by explosive population in spring and early summer (Kahn et al. 2013, Livingston et al. 2014). Females can store sperm and self-fertilize in spring and have up to nine broods with up to 300 young each per season (online resource 1). Therefore, population size increase exponential in spring and individuals can be easily spotted in the shallows of a waterway, where they "bask" in the sun in high density.

Water quality testing

We tested the water quality at 27 of the 33 sites, measuring *Electrical Conductivity, Turbidity, Total Phosphorus, pH, Nitrates, Dissolved Oxygen* and *Temperature*.

These parameters have been widely established as the best indicators of water quality while being relatively easy to measure (Waterwatch Victoria 1999). All water quality testing was conducted between the 13.12.2015 and 16.12.2015. Testing was undertaken according to the Upper Murrumbidgee Waterwatch user manual (online at www.act.waterwatch.org.au).

Frogspawn at Mulligans Flat Nature Reserve

Statistical analysis

The statistical analysis approach was extensive, but consisted of three primary stages. Firstly, data was entered into spreadsheets and categories coded to enable statistical analyses. Using all available Frogwatch data as of 31.10.2015, presence/absence and species counts at each wetland were tallied. If a species had been detected at a wetland during any reported survey (from 2002 to 2015), it was considered to be present at the wetland, and included in the tally for species richness. This approach was adopted as opposed to just using October FrogCensus data for a single year (e.g. 2014 or 2015). The main reason for this is that detecting a species in a single given year is likely to be strongly related to sampling effort, and Westgate et al. (2015) found no evidence of a decline in frog species in urban areas over the 2002-2014 period. As such, failure to detect a species in recent years is unlikely to be due to extirpation, rather driven by low detectability.

A total of 10 datasets were produced for analysis: species richness (count of number of species at each wetland) and presence/absence of each of the 9 species detected in urban wetlands in the ACT region.

Additionally, 5 separate explanatory variable datasets were compiled. Firstly, connectivity was derived from the field survey data and examination of satellite imagery (Google maps). For each wetland, a series of indicators was produced. Firstly, along each transect line (North, East, South and West from the wetland) we recorded if the wetland was adjoining a 1 hectare or larger patch of remnant forest or natural grassland. As such, this became a five-level factor (none, 1, 2, 3, or 4) depending on how imbedded the wetland was in the remnant forest. This was repeated, but this time increasing the size of the remnant patch of forest or grassland to 10 hectares or larger. This 5 level variable was also reduced to a 2 level factor to reflect if a wetland was not adjoining a 1 hectare or 10 hectare forest (0) or was adjoining a 1 or 10 hectare forest on 1 or more sides (1). Finally, covariates were produced by a) calculating the mean distance from each of the 4 compass points to the nearest patch of 10 hectare or larger forest, and b) the minimum distance from either of the 4 compass points to a patch of 10 hectare or larger forest. Each of these 8 possible explanatory variables were analysed in turn to identify the variable with the most explanatory power.

Field site at ARA 100

The second explanatory variable set was *Gambusia holbrooki* presence/absence data. Each wetland was coded as either not having *Gambusia* (0) or having *Gambusia* present (1).

Eastern gambusia gravid female. Photo credit: Tarmo Raadic. *http://www.finterest.com.au/natives-and-introduced/assessing-the-recovery-of-fish-communities-following-removal-of-the-introduced-eastern-gambusia-gambusia-holbrooki/*

The third dataset consisted of the habitat variables drawn from the RARC and FRARC data collection. These were all coded as either binary or multi-level factors, as per the categories outlined on the datasheets. Continuous variables were centred and scaled (e.g., transformed to a mean of zero and standardised variance). Initial exploration of these data revealed that in most instances many levels were empty, and ultimately provided a poor explanatory power. As such, all multi-level factors were reduced to binary outcomes, reflecting either presence or absence of the trait in question.

Examination of correlation between all these factors was undertaken in order to remove correlated variables. Given such a large and closely related set of variables, it was expected that a large number of variables would be highly correlated. We applied a rule, that if 2 variables were correlated with a correlation coefficient ($r>0.6$), one of the variables would be removed from the dataset. The decision of which variable to remove was based on its perceived relevance to frog biology and ecology, and the degree to correlation among other variables. As such, variables that were considered to only be indirectly related to frog biology and ecology, and were highly correlated with multiple other variables were preferentially removed. Following this process, 6 riparian habitat attributes remained: presence of understorey (trees and shrubs 1-5m tall), presence of fallen logs > 10cm diameter, presence of reeds and sedges, presence of rocks >20cm diameter, presence of trees > 5m tall, presence of large native tussock grasses.

The fourth dataset comprised of all variables from the FRARC dataset. Following the procedure above, multi-level variables were reduced to binary following initial examination that revealed several empty levels, and poor explanatory power. Following reduction, examination of correlation between variables was undertaken, and highly correlated variables removed, using the same conditions as outlined above. This left 3 variables explaining in-water habitat traits: presence of emergent macrophytes, water depth greater than 25cm deep at 1m from the water's edge, and wetland surface area.

Finally, water quality traits were centred and scaled, where appropriate and examination of correlation performed. Those traits that offered minimal or no variation were dropped. Remaining variables were retained for analysis: pH, total phosphorus, electrical conductivity and turbidity. This dataset was restricted to just 27 wetlands, as some did not have water quality data associated with them. As such, due to the different sample sizes, water quality could not be included in the same analyses as the other explanatory variable datasets. Instead, water quality was explored in isolation, with the restricted dataset of 27 wetlands.

The approach to statistical analysis was defined similarly to Westgate et al. (2015). Performing model selection when a large number of variables are available and a limited number of observations (n=33) has considerable problems. Instead, we developed a series of hypotheses which could be tested in turn. Identification of the best supported hypothesis was conducted using Akaike Information Criterion, corrected for small sample sizes (AICc). The decision to use AICc over AIC is often trivial. It is noted that AIC tends to favour models with more explanatory variables. A rule of thumb often applied is if n/k is >40, use AIC, where n is the number of observations, and k is the number of explanatory variables. The hypotheses to be tested are outlined in Table 1.

Dam at Callum Brae Nature Reserve

Table 1. Summary of hypotheses tested using generalised linear models, and the variables included in testing each hypothesis.

#	**Hypothesis**	**Variables**
1	Null model (no effect)	Intercept only
2	Connectivity	Best fitted (B.F.) connectivity variable
3	Gambusia	Gambusia presence
4	Riparian habitat	B.F. riparian habitat variable
5	In-water habitat	B.F. In-water habitat variable
6	Water quality*	B.F. water quality variable
7	Connectivity + Gambusia	B.F. connectivity variable + Gambusia presence
8	Connectivity + Riparian habitat	B.F. connectivity variable + B.F. Riparian habitat variable
9	Connectivity + In-water habitat	B.F. Connectivity variable + B.F. In-water habitat variable
10	Gambusia + Riparian habitat	Gambusia presence + B.F. Riparian habitat variable
11	Gambusia + In-water habitat	Gambusia presence + B.F. In-water habitat variable
12	Riparian habitat + In-water habitat	B.F. Riparian habitat variable + B.F. In-water habitat variable
13	Connectivity + Gambusia + Riparian habitat	B.F. Connectivity variable + B.F. Gambusia presence + B.F. Riparian habitat variable
14	Connectivity + Gambusia + In-water habitat	B.F. Connectivity variable + Gambusia presence + B.F. In-water habitat
15	Gambusia + Riparian habitat + In-water habitat	Gambusia presence + B.F. Riparian habitat variable + B.F. In-water habitat variable
16	Connectivity + Gambusia + Riparian habitat + In-water habitat	B.F. Connectivity variable + Gambusia presence + B.F. Riparian habitat variable + B.F. In-water habitat variable

All statistical analyses were conducted in R (R development core team 2011). Species richness was explored using generalised linear models with a poisson distribution, while individual species were explored with generalised linear models with a binomial distribution. A null model (intercept only) was fitted first for each response variable. A hypothesis was defined as having significant explanatory power if its AICc smaller than the AICc of the null model by at least 2. When multiple hypotheses were supported (defined as being within 2 AICc of the best supported model, but at least 2 smaller than the null model AICc), we used a model averaging approach, as provided by the R package “MuMIn”. This combines the variables in all the supported models to produce a composite model, and weighting of the explanatory variables in question.

Finally, a pseudo-R^2 value was produced for each final model to indicate goodness of fit. This was calculated as 1-(Residual deviance/Null deviance). This could not be calculated for composite models, so the pseudo R^2 of one of the original models was produced as a conservative R^2 for that model. Finally, graphical outputs of the models were produced with the R package “visreg” and edited in Adobe Illustrator.

Floating duck feather

Results

A total of nine frog species were detected at the 33 wetlands examined (Figure 2). Wetland sites had between 0 and 8 species present (Figure 3 and 4).

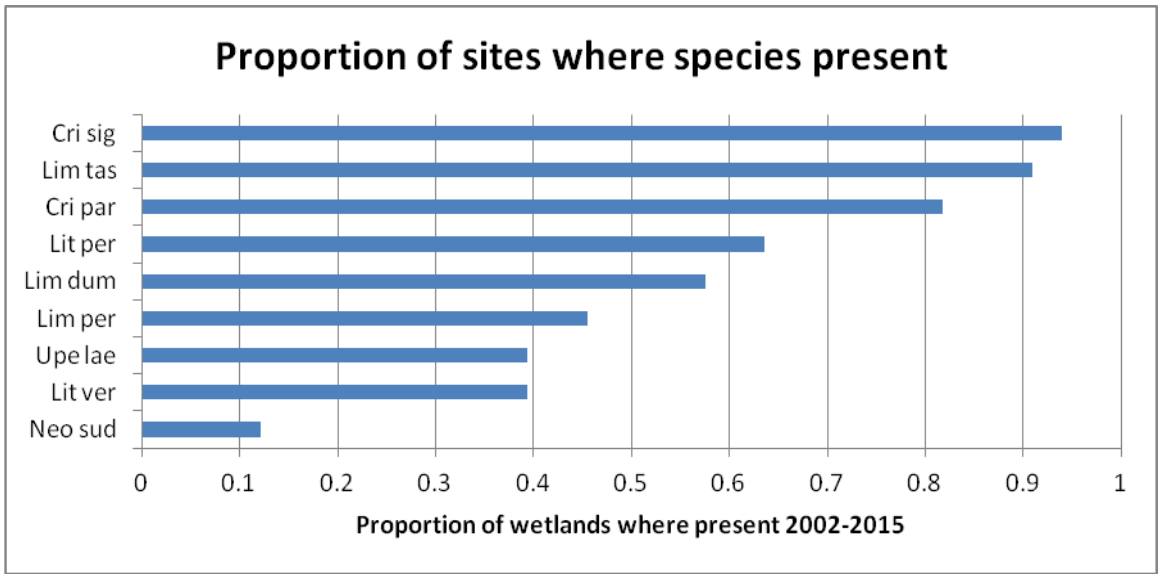


Figure 2. Proportion of sites at which each species of frog was detected, across the 33 wetlands examined in the present study.

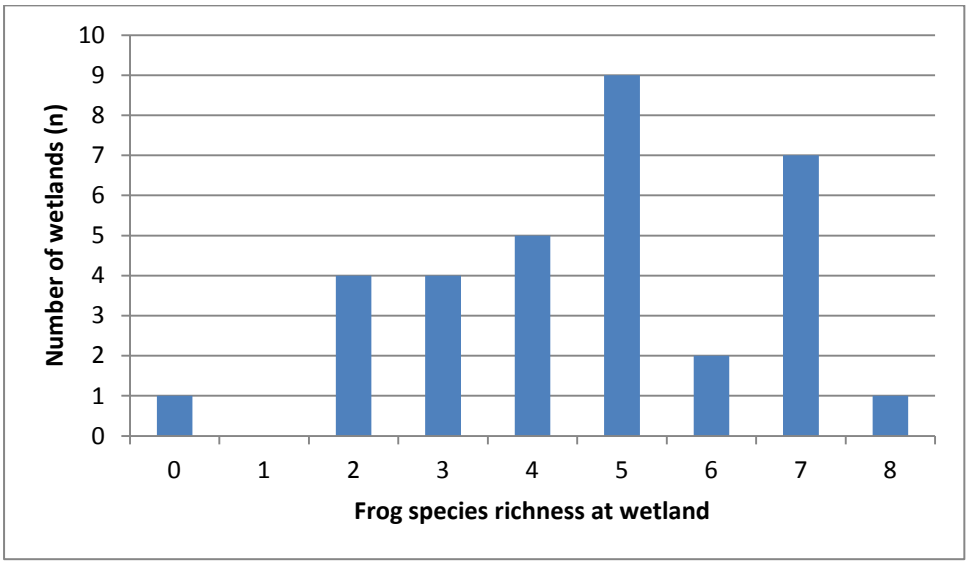


Figure 3. Frog species richness distribution at 33 urban/peri-urban wetlands between 2002 and 2015.

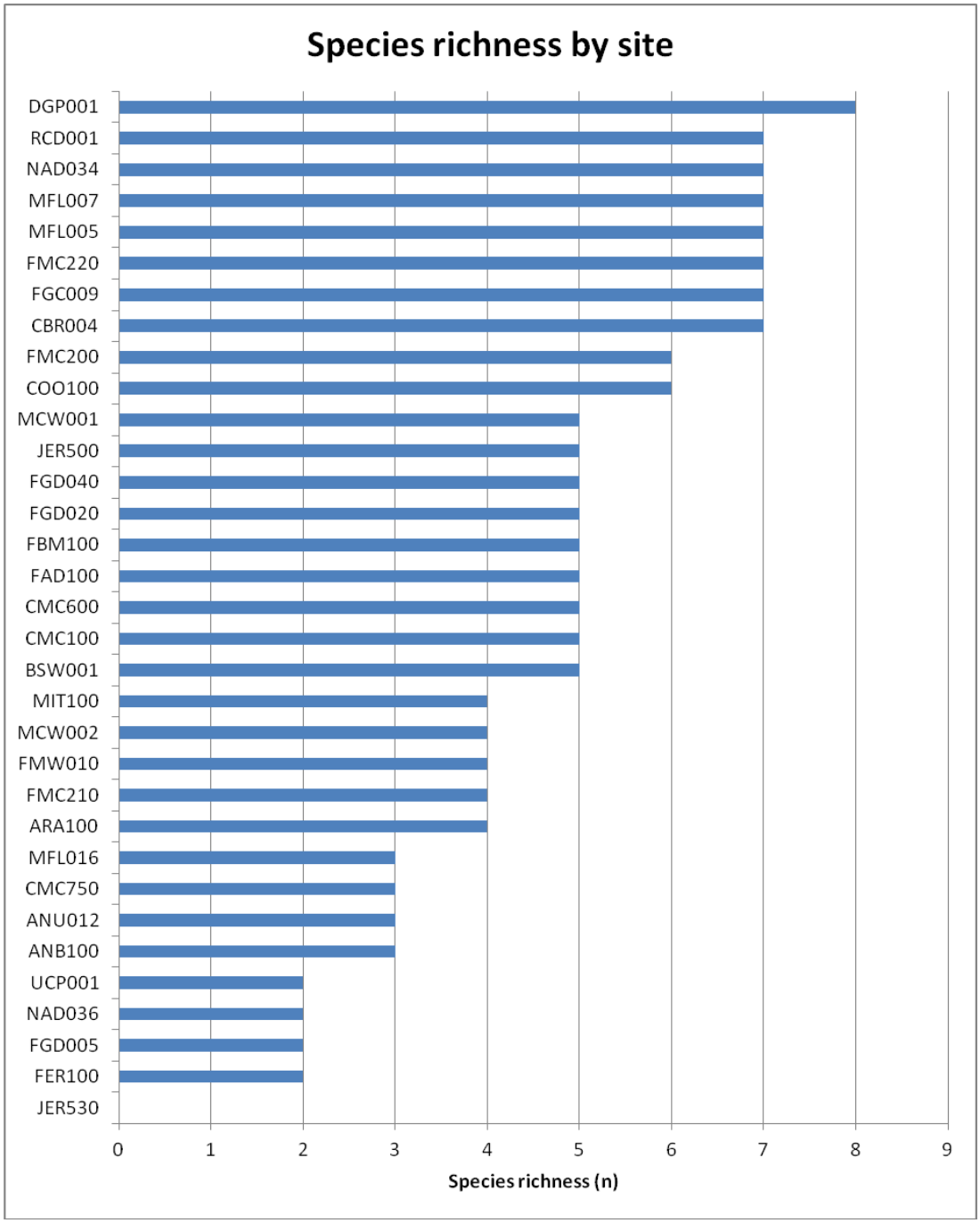


Figure 4. Number of frog species detected in 33 urban/peri-urban wetlands between 2002 and 2015. Site codes and related site information can be found in Appendix 3.

Species richness was only found to be significantly influenced by the presence of fallen logs in the riparian zone (Figure 5, Table 2). The overall goodness of fit, however, was relatively low (R^2 = 0.15). The presence of fallen logs in the riparian zone is expected to increase the number of species present by almost 2 species (Figure 5).

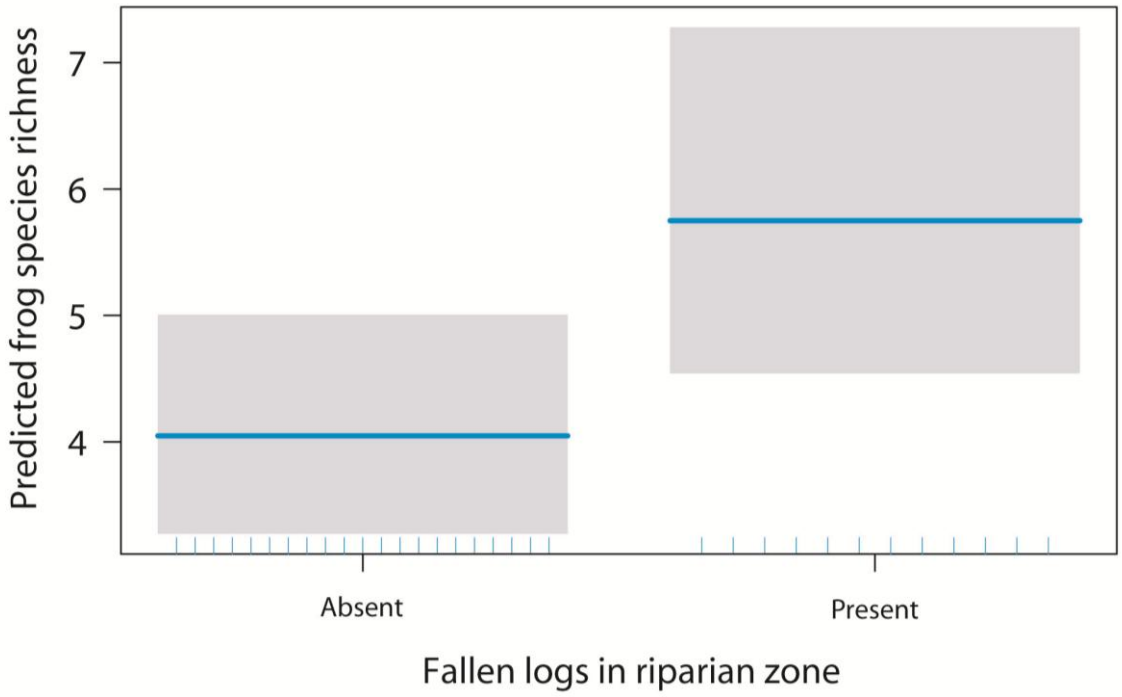


Figure 5. Predicted probability ± 95% confidence interval of the number of frog species detected at an urban/peri urban wetland in the absence or presence of fallen logs in the immediate riparian zone of the wetland.

Table 2. Summary of model outputs for species richness and presence/absence of each individual species, reporting coefficient estimates, standard errors and P values. The relative weighting each explanatory factor is shown, where model averaging was applied. A pseudo R^2 is presented for each model to give an indication of the quality of model fit.

Response	Coefficients	Estimate	Standard Error	P Value	Weighting	Pseudo-R^2
Richness	Intercept	1.4	0.11	**<0.001**		0.15
	Logs	0.35	0.16	**0.03**	NA	
Cri sig	Intercept	1.75	0.93	0.07		0.48
	1ha reserve	20.2	4624	0.99	1	
	Wetland area	-1.17	0.87	0.2	0.52	
Cri par	Intercept	0.96	0.53	0.07		0.2
	Logs	18.54	3222	0.99	1	
	Avg 10 ha	-0.416	0.5	0.4	0.3	
Lim tas	Null					
Lim dum	Intercept	-0.08	0.43	0.84		0.16
	Rocks	2.07	1.23	0.09	1	
	Min 10ha	-0.65	0.45	0.14	0.53	
Lim per	Intercept	-3.98	1.84	**0.03**		0.31
	Gambusia	2.01	0.98	**0.04**	1	
	Reeds	2.81	1.45	0.052	1	
	10ha reserve	1.81	1.04	0.08	0.52	
Lit per	Intercept	-0.46	0.64	0.48		0.24
	Logs	2.67	1.22	**0.03**	1	
	Macrophytes	1.48	0.94	0.11	0.56	
Lit ver	Null					
Neo sud	Intercept	-1.25	0.57	**0.03**		0.22
	Gambusia	-18	2776	0.99	NA	
Upe lae	Intercept	-18.5	3002	0.99		0.47
	1ha reserve	19.66	3002	0.99	1	
	Gambusia	-2.4	1.09	**0.03**	1	
	Logs	0.75	1.08	0.49	0.26	

Crinia parinsignifera was present at 27 out of 33 wetlands (Figure 2), and was found to be influenced by both logs in the riparian zone, and average distance to 10 hectare or larger patches of forest (Figure 6, Table 2). While logs had a positive, non-significant effect, and distance to forest had a non-significant, negative effect, this was the best supported model with the available data. Logs in the riparian zone was considered to be the stronger effect, compared to distance to forest patch. Overall goodness of fit was low (R^2 = 0.2, Table 2).

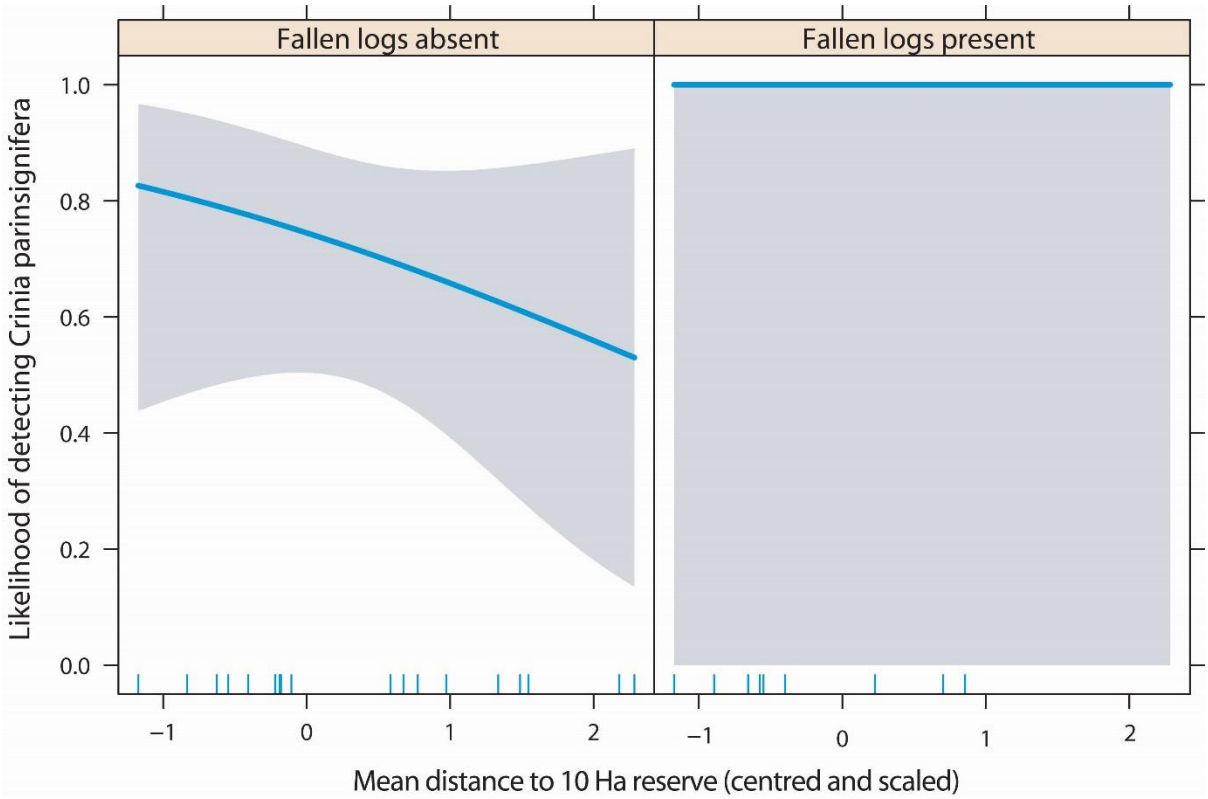


Figure 6. Probability ± 95% confidence interval of *Crinia parinsignifera* being detected at an urban/peri-urban wetland with respect to the mean distance to the 4 nearest 10 hectare or larger patches of forest, and the presence or absence of fallen logs in the immediate riparian zone of the wetland.

Crinia signifera was present at 31 out of 33 wetlands studied (Figure 2), hence the veracity of these results need to be interpreted with caution. Detection of *Crinia signifera* was found to be positively related to connectivity to a 1 hectare or larger patch of forest, and a negative effect of wetland area (Figure 7, Table 2). Wetland area was not as well supported as connectivity to 1 hectare or larger patch of forest (Table 2). Overall goodness of fit was moderately good, but should still be treated with caution (R^2 = 0.48).

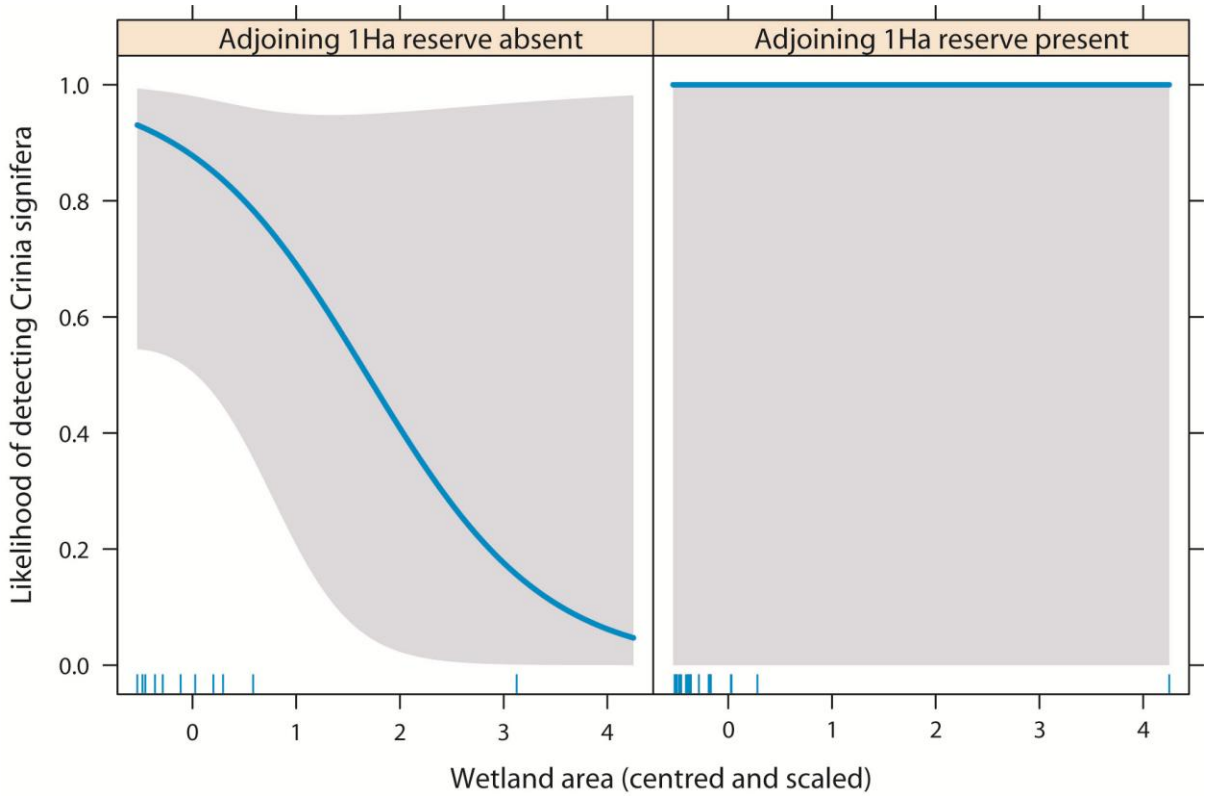


Figure 7. Probability ± 95% confidence interval of *Crinia signifera* being detected at an urban/peri-urban wetland in relation to the surface area of the wetland, and whether the wetland is adjoining a 1 hectare or larger patch of forest.

Limnodynastes dumerilii was detected at 19 out of 33 wetlands in the present study (Figure 2). The best supported model indicated that *Limnodynastes dumerilii* was positively related to rocks in the riparian zone, and negatively related to minimum distance to a 10 hectare or larger reserve (Figure 8, Table 2). Rocks in the riparian zone was considered to be the more important variable, compared to connectivity (Table 2). Overall goodness of fit was relatively low (R^2 = 0.16).

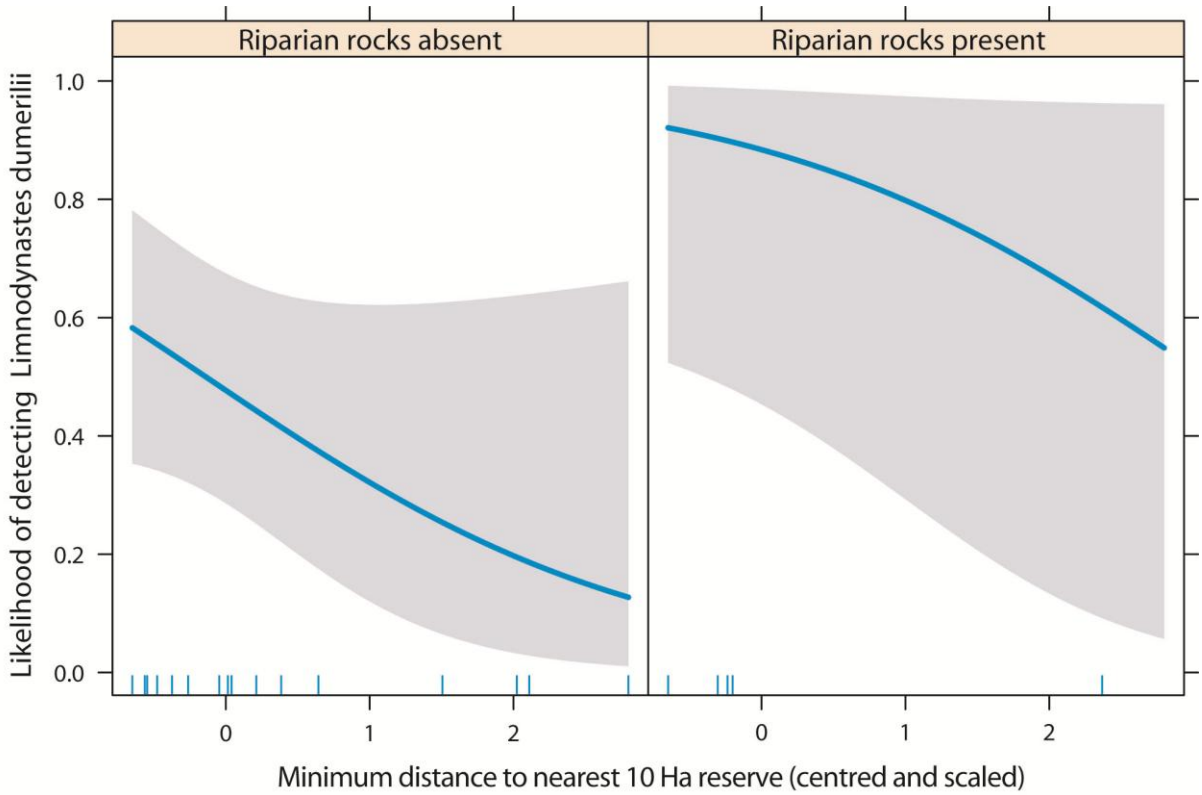


Figure 8. Probability ± 95% confidence interval of *Limnodynastes dumerilii* being detected with respect to distance to the nearest 10 hectare or larger patch of forest and the presence of rocks in the immediate riparian zone of the wetland.

Limnodynastes peroni was detected at 15 out of 33 wetlands in the present study (Figure 2). *Limnodynastes peroni* was positively related to *Gambusia holbrooki* presence in the wetland, the presence of reeds in the riparian zone, and connectivity to a 10 hectare or larger patch of forest (Figure 9). Overall, only *Gambusia holbrooki* presence was significant (P= 0.04, Table 2). *Gambusia holbrooki* and the presence of reeds were equally weighted, while connectivity was found to be less important (Table 2). Overall goodness of fit was moderate (R^2 = 0.31).

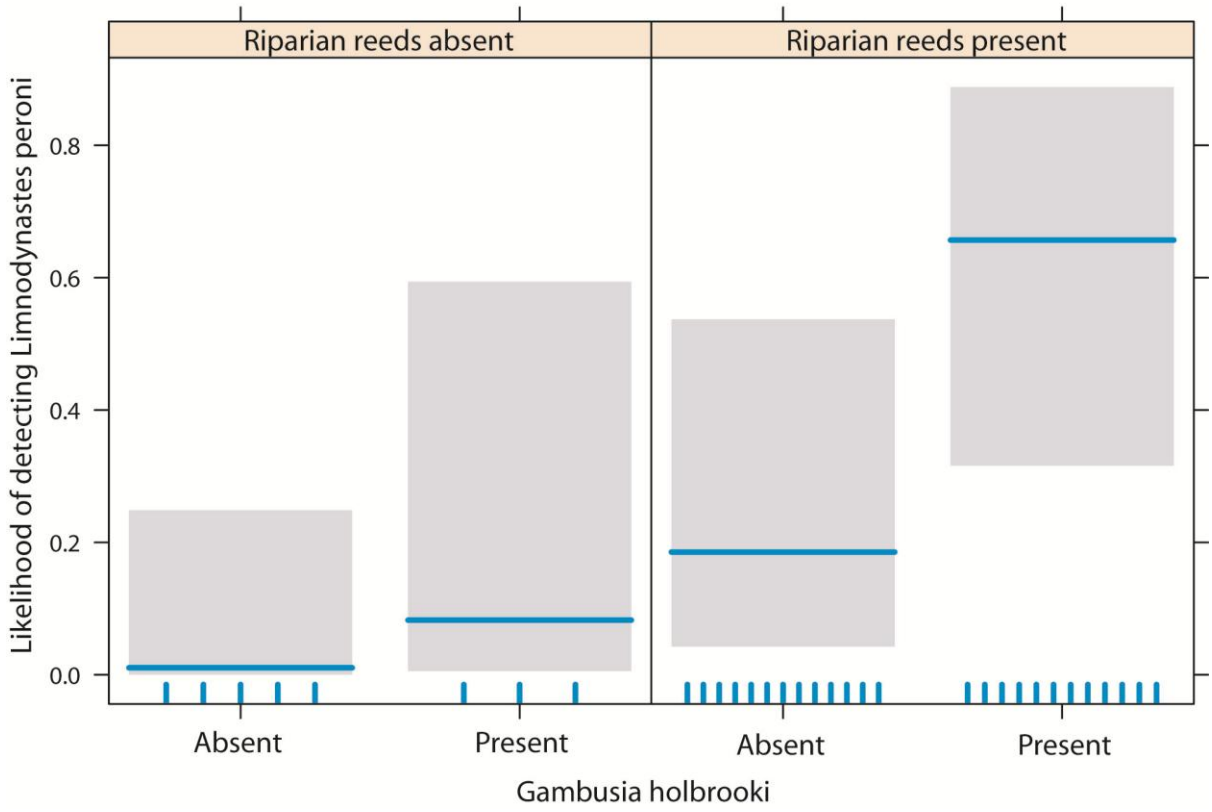


Figure 9. Probability ± 95% confidence interval of *Limnodynastes peroni* being detected at an urban/peri-urban wetland with respect to the presence of *Gambusia holbrooki* and the presence of reeds and sedges in the immediate riparian zone of the wetland.

Litoria peronii was detected at 21 out of 33 wetlands in the present study (Figure 2). *Litoria peronii* was found to be positively associated with logs in the riparian zone (P = 0.03) and positively associated with emergent macrophytes within the water (Figure 10). Logs in the riparian zone were found to be the more important variable (Table 2). Overall goodness of fit was moderate (R^2 = 0.24).

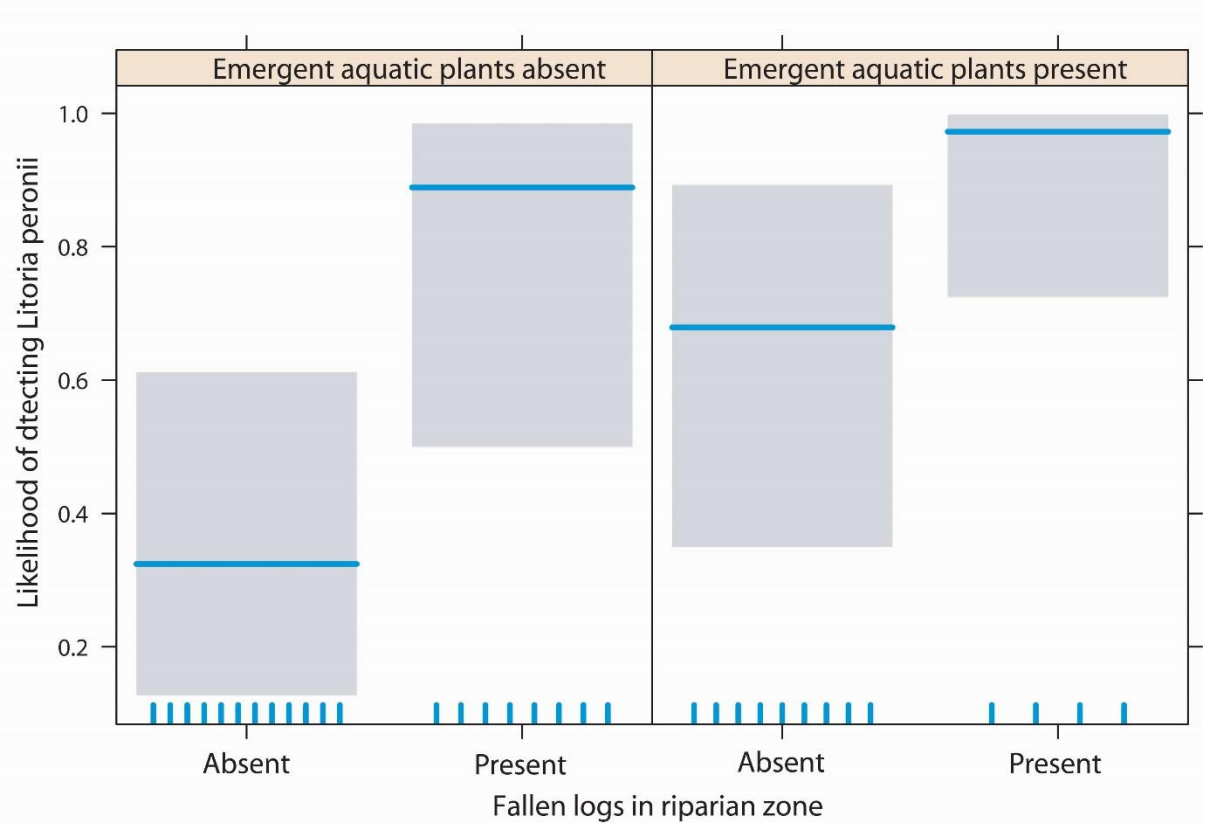


Figure 10. Probability ± 95% confidence interval of *Litoria peronii* being detected at an urban/peri-urban wetland with respect to the presence of fallen logs in the immediate riparian zone of the wetland, and the presence of emergent aquatic macrophytes in the wetland.

Neobatrachus sudelli was found at 4 out of 33 wetlands in the present study (Figure 2). As such, the following findings should be considered cautiously. *Neobatrachus sudelli* was found to be negatively associated with wetlands containing *Gambusia holbrooki* (Figure 11, Table 2). However, the variance estimate suggests that further work is needed to fully interpret this result (Table 2). Overall goodness of fit was moderate (R^2 = 0.22).

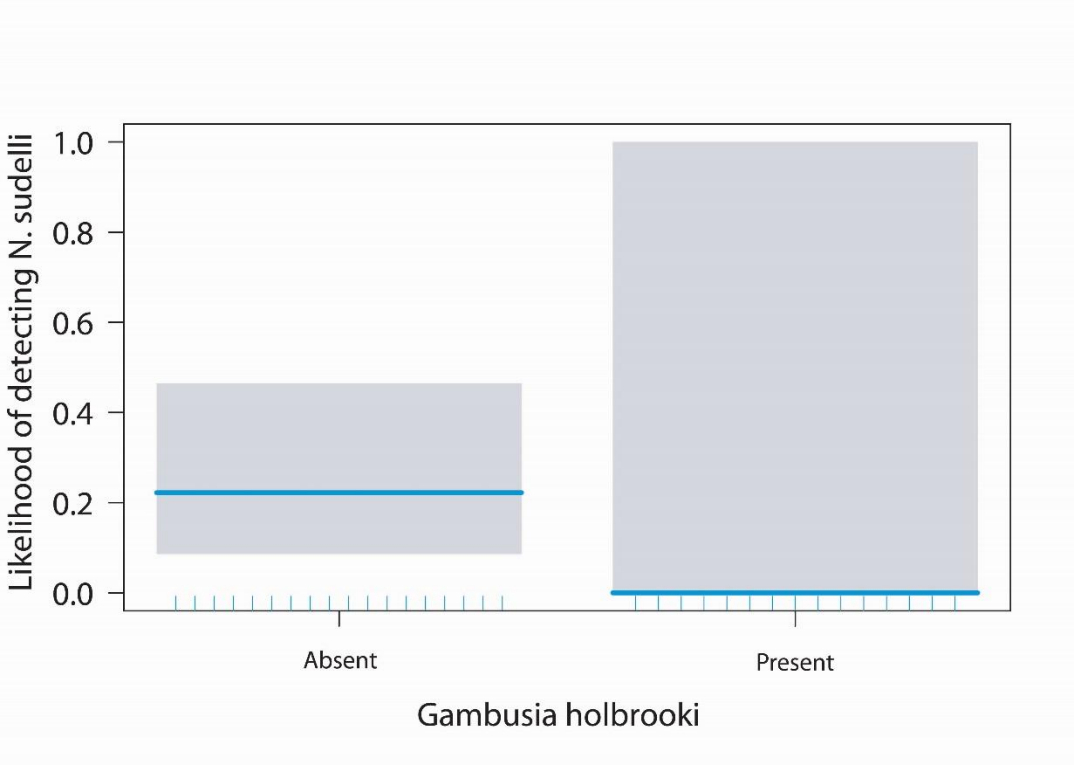


Figure 11. Probability ± 95% confidence interval of *Neobatrachus sudelli* being detected at an urban/peri-urban wetland in the presence or absence of *Gambusia holbrooki.*

Uperoleia laevigata was detected at 13 out of 33 wetlands in the present study (Figure 2). *Uperoleia laevigata* was positively associated with connectivity to a 1 hectare reserve and positively associated with logs in the riparian zone (Figure 12, Table 2). Conversely, it was negatively associated with wetlands containing *Gambusia holbrooki* (Figure 12, Table 2). *Gambusia holbrooki* presence and connectivity to a 1 hectare or larger patch of forest were equally supported, with logs in the riparian zone being less important (Table 2). Overall goodness of fit was moderately good (R^2 = 0.47).

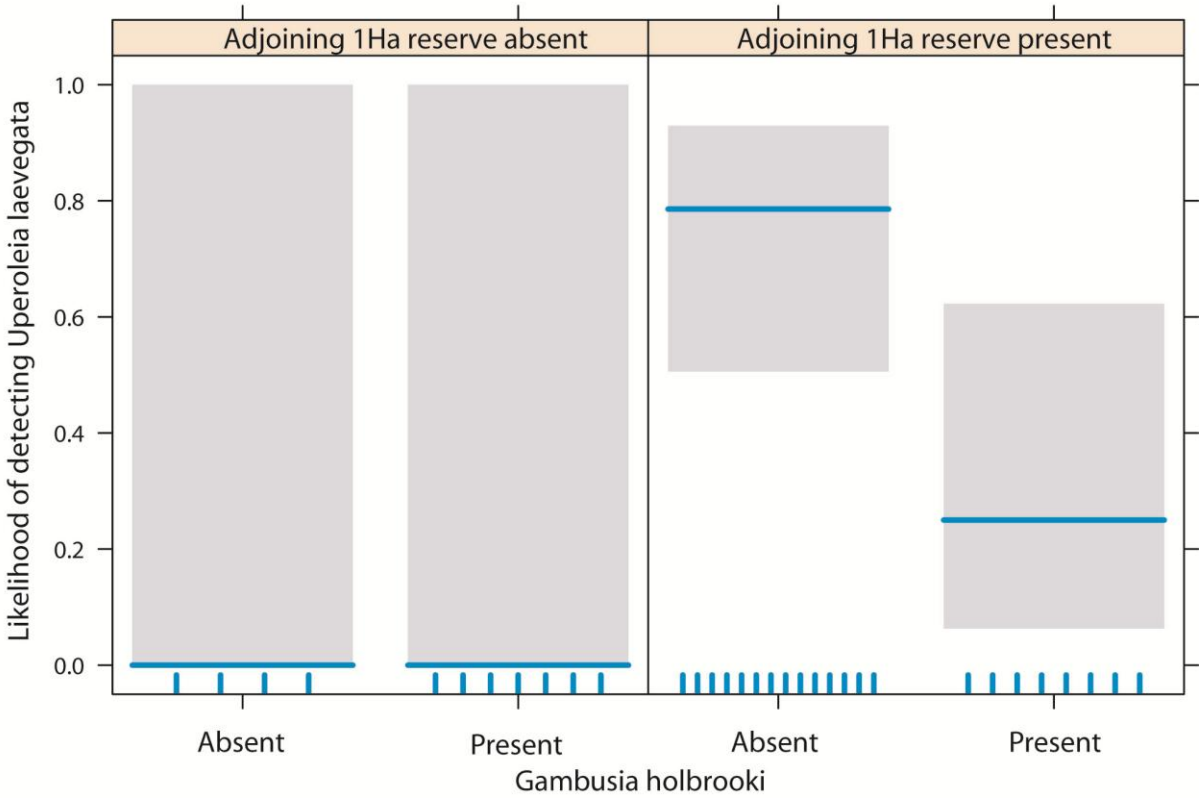


Figure 12. Probability ± 95% confidence interval of *Uperoleia laevigata* being detected at an urban/peri-urban wetland with respect to the presence of *Gambusia holbrooki* and whether the wetland is adjoining a 1 hectare or larger patch of forest.

Given that presence of logs in the riparian zone, and evidence of lawn mowing around the wetland were strongly negatively correlated, we fitted models with mowing in place of logs in the riparian zone. For frog species richness, mowing had a strongly negative effect, reducing the expected number of frog species from just over 5 species to less than 4 species, if mowing was evident (Figure 13).

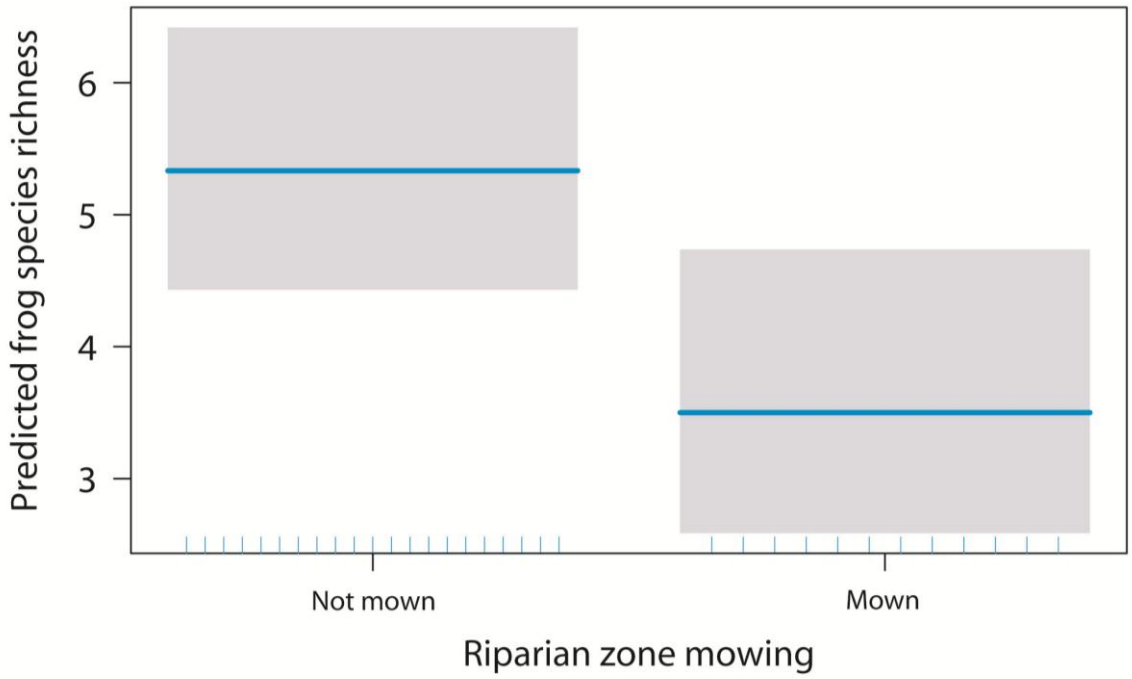


Figure 13. Predicted species richness ± 95% confidence interval at urban/peri-urban wetlands with respect to riparian zone mowing.

Litoria peronii was also found to be negatively associated with mowing, but positively associated with emergent macrophytes within the water (Figure 14).

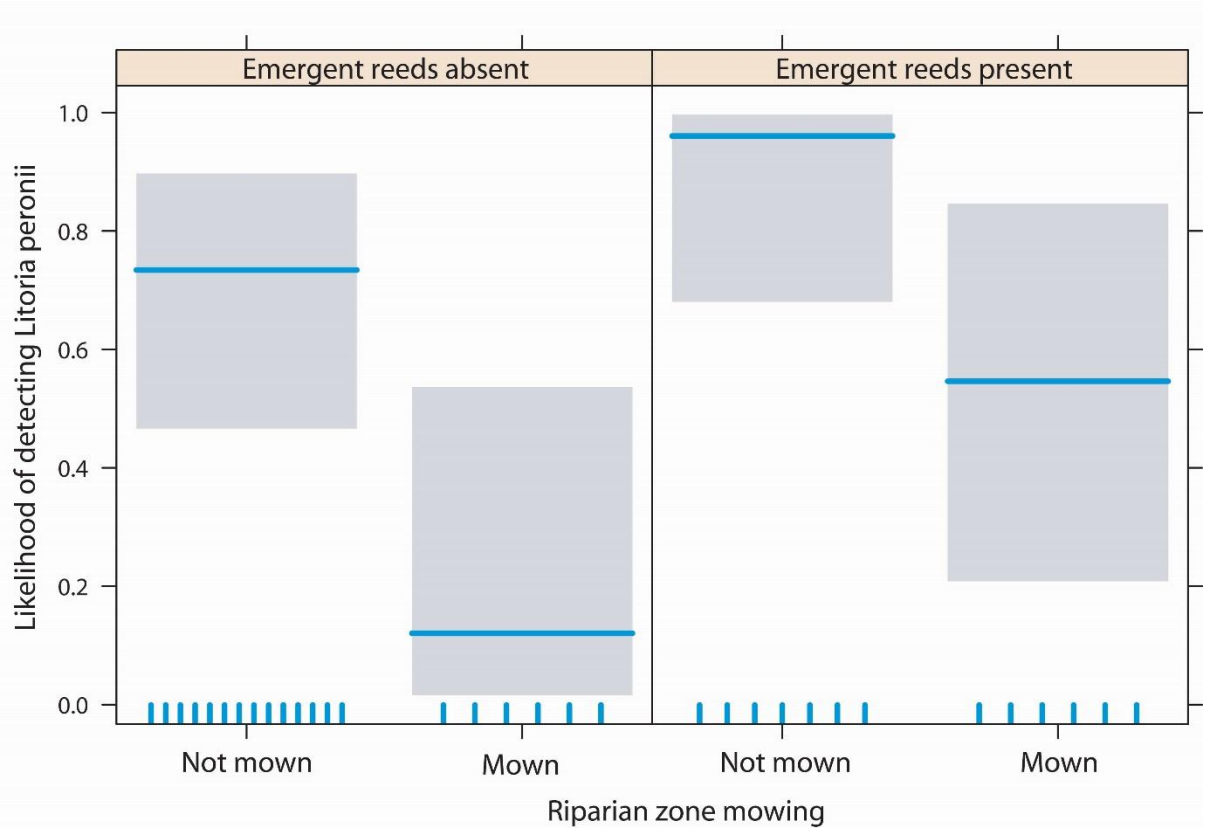


Figure 14. Probability± 95% confidence interval of detecting *Litoria peronii* at urban/peri-urban wetlands with respect to riparian zone mowing and the presence of emergent aquatic plants in the wetland.

Finally, *Uperoleia laevigata* was negatively associated with both mowing in the riparian zone, and the presence of *Gambusia holbrooki* (Figure 15). At sites where mowing occurred the likelihood of detecting *Uperoleia laevigata* was close to zero, however if mowing did not occur, and *Gambusia holbrooki* were absent, the likelihood of detecting *Uperoleia laevigata* was approximately 80% (Figure 15).

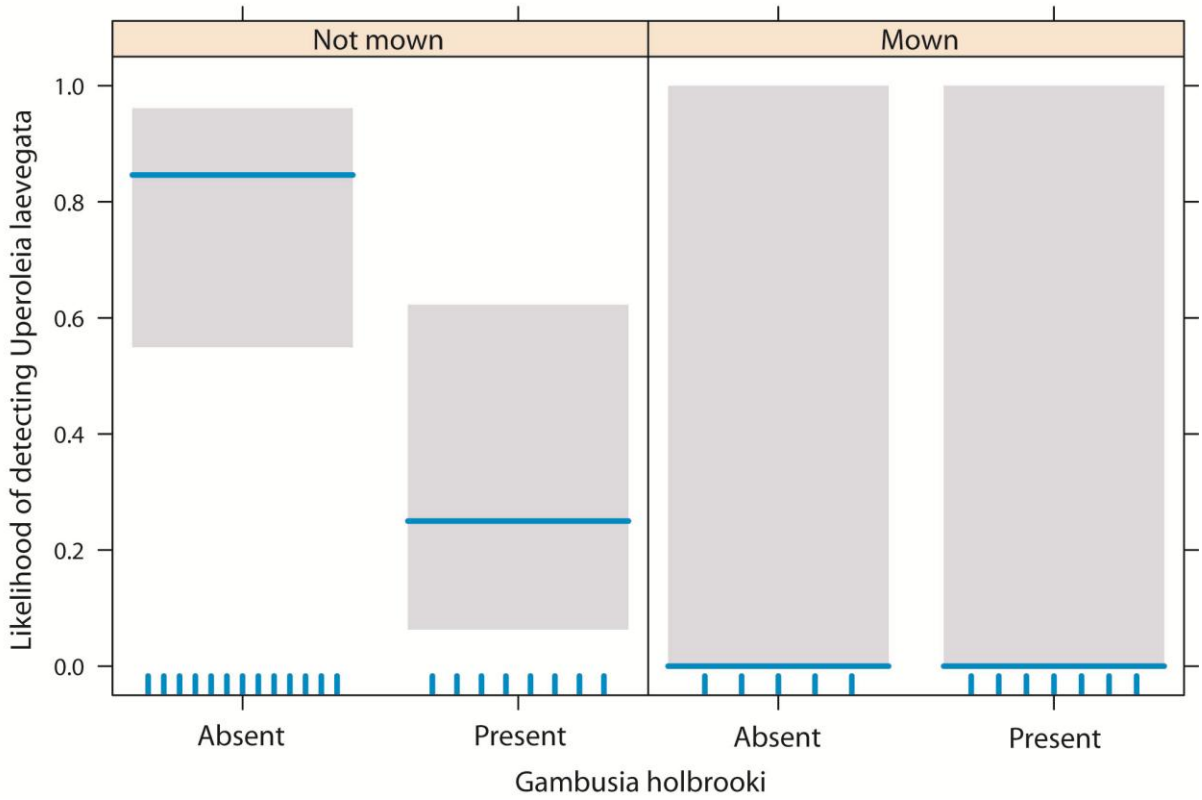


Figure 15. Probability ± 95% confidence interval of detecting *Uperoleia laevigata* at urban/peri-urban wetlands with respect to the presence of *Gambusia holbrooki* and riparian zone mowing.

Water quality was examined of a subset of 27 wetlands, where water quality data was present. *Uperoleia laevigata* was found to be strongly negatively associated with wetlands with high electrical conductivity (Figure 16). The likelihood of detecting *Uperoleia laevigata* dropped from approximately 80% to ~10% once the average electrical conductivity of 180 µS cm^{-2} had been reached (Figure 16). A full model incorporating connectivity, *Gambusia holbrooki*, logs and electrical conductivity explains 60% of the variance in *Uperoleia laevigata* presence/absence in the restricted dataset (model not shown).

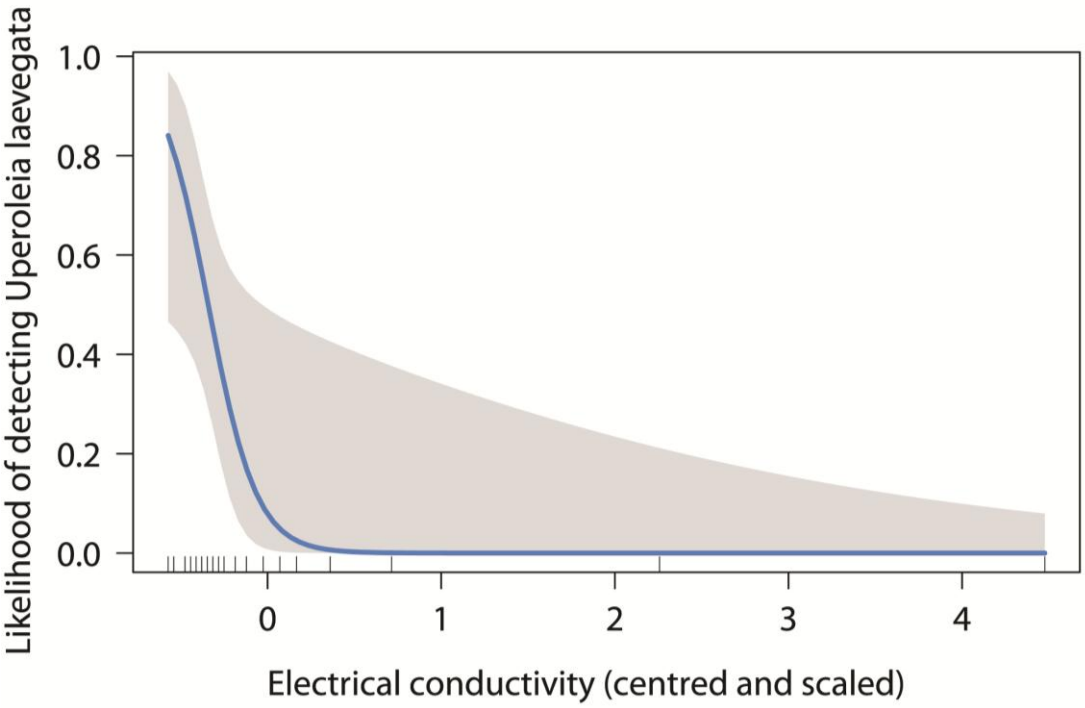


Figure 16. Probability ± 95% confidence interval of detecting *Uperoleia laevigata* at urban/peri-urban wetlands with respect to Electrical conductivity in the wetland.

Discussion

An examination of 33 wetlands in the urban/peri-urban region of the ACT revealed a varied relationship between frog species and specific habitat attributes with which they positively associate. Many of these variations are likely due to differences in each species biology and ecology. Below, we will explore some of these differences and compare our findings to previous studies. We describe how frog surveys at urban wetlands could be used to monitor the health of urban wetlands through time. We also make recommendations to aid the management of frog species at urban wetlands, and identify areas for future research.

Overall Species Richness

While overall species richness in the ACT region is low (9 species), we found a significant effect of fallen logs in the riparian zone immediately surrounding wetlands as having a positive influence on the number of species present. In the presence of logs the average number of species detected raised from 4 (no logs) to 6 species. The structural and functional components of fallen logs and other structural elements at ground level are critical for many forest and woodland ecosystems, as it increases biomass storage, decreases erosion (Lester & Boulton 2008) and provides habitat for many species including birds, beetles, fungii and vascular plants (Juutinen et al. 2006, Barton et al. 2011), reptiles (Shoo et al. 2014), and snakes (Hecnar & Hecnar 2011). The majority of our local frog species has been observed using naturally occurring woody debris as well as manually re-introduced fallen logs as a refuge and for hibernation (online resource 2, Cogger 2014, A.M. Hoefer, pers. obs.). Where logs are missing, rocks could potentially fulfil some of its ecological functions. Bush rock removal has been identified as key threatening process for some species, including frogs (Cogger et al. 1993, White & Burgin 2004, Newell & Goldingay 2005). Our study did not detect any relationships between the presence/absence of rocks and the overall species richness, but rocks were a very strong predictor for the species-specific site choice of *Limnodynastes dumerilii*.

To date, few studies have investigated the influence of natural structure at ground level, including logs and rocks, on the occurrence of frogs in Australia (Davis et al. 2010). A noteworthy exemption is the experimental study by Burgin and Wotherspoon (2009), which showed that returning logs to the forest floor increased the number of frog species. In two studies overseas adding logs to forest floor did not change the number of species, but affected the abundance of individual species (Owens et al. 2008, Wanger et al. 2009). More detailed investigations and experimental approaches to test for long-term effects of riparian timber (and rocks)

on frog assemblages and species distribution, especially in urban and peri-urban environments, is a logical next step.

The second factor which had a strong positive effect on overall species richness was the absence of mowing, however mowing and the presence of fallen logs were highly negatively correlated with each other, meaning that disentangling the ecological relevance of both is problematic. 12 of the 33 study sites were located in urban open spaces, which regularly get mowed for the ease of public access and social and recreational needs (Online resource 3). These sites had lower species richness than unmown sites, possibly for a variety of direct and indirect reasons. Firstly, regularly mowed wetland areas presumably are depleted of fallen logs and other structural elements. Fallen logs do not generally fit in the concept of a beautiful urban open space, do limit mower access and might be perceived as a fire risk in areas close to housing. Secondly, regular mowing adds a level of disturbance, which can limit frog movement, and can have detrimental effects on dispersing froglets (Campion 2010). Thirdly, the aim of mowing is to keep vegetative biomass to a minimum. However, short vegetation dries out much faster and provides less refuge for adult frogs (Smith & Sutherland 2014). An experimental study on a golf course in Sydney, located in a degraded woodland and grassland, found an increase of frog species from 7 to 10 within two years after habitat restoration in combination with a changed mowing regime to leave the riparian zone intact (Burgin & Wotherspoon 2009).

Our results demonstrate that logs in the riparian zone significantly influence species richness. However it remains unclear if mowing decreases frog species richness indirectly - through limiting log availability or disturbance, or directly - through minimising riparian vegetation. A larger scale approach will be needed to specifically test for the relationship of logs and mowing on species distribution within the ACT.

It is perhaps surprising that additional habitat attributes were not found to positively influence the species richness at the 33 wetlands examined. However, it is important to note, that differences between species, and their preferences for specific habitat attributes may cause a cancelling-out effect of individual traits. For example, a species positively associated with one environmental attribute can be obscured by another species having a disassociation with that particular variable. For this reason, failing to find strong relationships between species richness and specific attributes is commonly observed, and individual species-specific relationships tend to be more valuable. Despite this fact, previous studies have found positive relationships between species richness and specific habitat attributes.

Most notably, Westgate et al. (2015) used a nearly identical dataset as that used in the present study to investigate the effect of urbanization on patterns of frog species richness and occurrence over 13 years in the ACT and found that sites surrounded by a high proportion of bare ground had consistently lower frog occurrence, irrespectively if in rural or urban areas. For at least four of our study species a strong

preference for riparian vegetation has also been shown in other surveys across Australia. In a study of habitat correlates for 5 species on an island in the Hunter River (NSW), Lane et al. (2007) found pond area and the percentage of emergent vegetation predicted species richness but not species-specific preferences. In a study on wetland frog communities on the Murrumbidgee River floodplain, Jansen & Healey (2003) found a clear relationship between species richness and fringe vegetation as well as with overall condition scores. Increases in fringe vegetation significantly explained species richness, and abundance of a number of individual species, including some shared in this study.

Smooth toadlet male calling

Plains froglet (Crinia parinsignifera)

Previous studies on the preferences of *Crinia parinsignifera* in rural areas found positive effects of the percentage of water margins with emergent vegetation and the size of the water body (Hazell et al. 2001). Jansen & Healey (2003) found a significant relationship with bank attributes in a flood plain environment. We could not repeat these findings in our surveys of urban wetlands, however, the likelihood of detecting *Crinia parinsignifera* in the absence of fallen logs was strongly affected by the mean distance to a 10 Ha (or larger) patch of forest. This species is often associated with logs and rocks and can be observed sheltering in large numbers under one log or rock (online resource 4, Barker et al. 1995). *Crinia parinsignifera* is a generalist, highly adaptable to disturbed and altered landscapes and has a broad range of breeding habitats (Hazell et al. 2004). In a comparison of natural versus constructed ponds *Crinia parinsignifera* was equally attracted to both habitat types (Hazell et al. 2004). A lack of strong preferences for certain environments would probably be advantages for the colonisation of urban wetlands. This is also reflected by the high occupancy rate of well over 80% of our sites.

Plains froglet

Common eastern froglet (Crinia signifera)

Our data best supported the model that the likelihood of detecting *Crinia signifera* was negatively related to wetland area in relation to a small adjoining reserve, however, this result is unlikely to be overly robust, due to the high (95%) incidence of *Crinia signifera* at wetlands in our study. In contrast, Hamer et al. (2012) found a strong positive effect of increasing pond size, and shallow edges on the mean abundance of this species in urban stormwater retention ponds in Melbourne. The author argues that larger ponds generally have shallow margins, which consequently can support greater numbers of this species than a small pond. To what extent hydrological variability is involved, and the relative relationship between wetland size in our study and the previous study make placing our findings in context problematic.

Similar to our results, Lemckert (1999) found that distance to reserved land negatively affected the occurrence of *Crinia signifera* in forestry areas. Males call

from among vegetation at the water's edge or floating in open water supported by vegetation and shelter during the day under rocks, logs and other small debris, often close to the water (Online resource 5, Lintermans and Osborne 2002). Increased distances from suitable terrestrial habitats may lead to increased mortality and overall lower the likelihood of species persistence due to reduced connectivity.

Similar to *Crinia parinsignifera*, *Crinia signifera* is a generalist species preferring open and disturbed areas (Online resource 5, Barker et al. 1995) and inhabited natural and constructed water bodies equally (Hazell et al. 2004). This high adaptability to a large range of environments greatly supports the colonization of newly established water bodies and altered landscapes as suggested by our detection rate of this species at just under 95% of wetlands in the present study.

Eastern banjo frog (Limnodynastes dumerilii)

The occurrence of *Limnodynastes dumerilii* was strongly associated with rocks in the riparian zone. The importance of rocks has been described for many frog species as rocks provide shelter, congregation spaces and microhabitats supporting thermo and moisture regulation (Cogger 2014). Illegal bush rock removal from bushland reserves in the Hawkesbury area (NSW) has lead to habitat loss for numerous reptile and frog species, including *Uperoleia leavigata* and *Limnodynastes dumerilii.* Less rocks leads to a higher likelihood of predation through greater exposure, which in turn negatively affects seasonal dispersal routes (Cogger et al. 1993, White & Burgin 2004). These findings are strongly supported by our results that the occurrence of *Limnodynastes dumerilii* with respect to the nearest patch of forest ≥ 10 Ha was influenced by the presence of riparian rocks. The greater the distance from a reserve the lower the likelihood of detecting *Limnodynastes dumerilii*. This effect was more pronounced when rocks were absent. Adjoining woodlands and forests may compensate for the lack of rocks by providing important amphibian habitat for foraging and hibernation (Hecnar & M'Closkey 1996), improving landscape connectivity (Laan & Verboon 1990) and support dispersal between wetlands (Kupfer & Kneitz 1999). Incidental observations on *Limnodynastes dumerilii* suggest a gradual decline of this species over the past two decades (M. Evans, pers. comm.).

Striped marsh frog (Limnodynastes peroni)

The presence of riparian reeds had a positive influence the occurrence of *Limnodynastes peroni,* especially in the presence of *Gambusia holbrooki*. The introduced plague minnow is an extremely robust generalist, highly invasive and most abundant in urban to peri-urban environments (Gillespie & Hero 1999). It is readily considered to have a negative influence on frogs, and has been linked to the decline of frogs throughout south-eastern Australia (Gillespie & Hero 1999, Holbrook

& Dorn 2016). As such, it is surprising that we found a positive relationship between *Lymnodynastes peroni* and *Gambusia holbrooki* presence. We feel there are two, inter-related explanations for this result.

Firstly, we contend that the positive relationship between *Limnodynastes peroni* and *Gambusia holbrooki* is correlative, rather than causative. Most fish species require permanent water bodies in order to persist, as regular drying will constantly lead to localised extinctions. Furthermore, *Limnodynastes peroni* has previously been found to be positively associated with large, thick patches of emergent macrophytes around wetland habitats, where it calls, and tadpoles live (Jansen & Healey 2003). Secondly, these patches of large emergent macrophytes tend to only form in stable permanent waterbodies with non-fluctuating water supplies, where they also provide habitat for threatened native fish (Ebner and Lintermans 2007). As such, it seems likely that persistence of dense emergent macrophytes provides suitable habitat cover for *Limnodynastes peroni*, while providing micro-scale segregation from *Gambusia holbrooki*, allowing for this unexpected relationship to form (D. Hazell, pers. comm.).

For persistence to occur requires that tadpoles successfully avoid predation. Furthermore, successful hiding is even more vital in species that have relatively long tadpole phases, such as in *Limnodynastes peroni* (Anstis 2002). This species produces tadpoles from August until March (online resource 6) with most of the early tadpole stages co-existing with the *Gambusia holbrooki* population explosion during the warmer months (Kahn et al. 2013, Livingston et al. 2014) and therefore relies on successful predator avoidance. These observations are consistent with *Limnodynastes peroni* habitat preferences across urban stormwater ponds containing *Gambusia holbrooki*. Hamer et al. (2012) found a strong positive effect of aquatic vegetation with an over 0.8 likelihood of occurrence of *Limnodynastes peroni* at ponds with over 80% aquatic vegetation cover and/or a shore depths of <10cm. Very shallow wetland edges, especially in combination with vegetation, provide imperative warm-water microhabitat not only for predator avoidance but also for thermoregulation, calling, oviposition and foraging (Semlitsch 2002).However we note that *Gambusia holbrooki* also associate with these microhabitats, for probably the same reasons. When comparing *Limnodynastes peroni* occupation at natural versus constructed wetlands, the species was more likely to occur in natural ponds, however preferred in both habitat types water bodies without introduced fish predators (Hazell et al. 2004). Tadpole survival strategies are generally classified in spatial and temporal avoidance (see Gillespie and Hero 1999). Aquatic vegetation, including reeds, might play an important role in providing refuges from fish predation (Baber and Babbitt 2004) and would explain the much higher occurrence of *Limnodynastes peroni* in wetlands that have fish and reeds.

Spotted grass frog (Limnodynastes tasmaniensis)

We did not find any predictor model for habitat preferences for *Limnodynastes tasmaniensis* in urban areas. The only other study looking at habitat choice of this species in an urban setting in this species found a strong preference for shallow edges, which were less than 20cm deep (Hamer et al. 2012). However, in rural ponds *Limnodynastes tasmaniensis* occurrence was positively influenced by emergent vegetation at the edge of the pond (Hazell et al. 2001, Lane et al. 2007) with no preference for either natural or constructed water bodies but a strong preference for fish-free ponds (Hazell et al. 2004). Increasing the number of wetlands analysed may yet reveal what habitat traits are associated with *Limnodynastes tasmaniensis*, which is otherwise a widespread and common species with apparent generalist tendencies (Lintermans & Osborne 2002).

Limnodynastes tasmaniensis froglet

Peron's tree frog (Litoria peronii)

The occurrence of *Litoria peronii* was predicted by riparian zone conditions, namely the presence of fallen logs and/or the absence of mowing as well as positive association with reeds and sedges. At sites with no reeds and mowing, the detection rate of *Litoria peronii* was low, whereas sites with reeds and no mowing had a very high likelihood of *Litoria peronii* being present.

Previously, *Litoria peronii* was found to be negatively influenced by bare ground and lack of emerging riparian vegetation (Hazell et al. 2001, 2004, Lemckert et al. 2006). This might be directly linked to its habit to call from vegetated pond edges (Anstis 2002)**.** Lemckert et al. (2006) could not show any relation between bare ground and the occurrence of *Litoria peronii*, however the vast majority of the wetlands in his study were well vegetated. *Litoria verreauxii* and *Uperoleia laevigata* were both negatively affected by the percentage of bare ground in the riparian zone of rural farm dams in south-eastern Australia (Hazell et al. 2001). We expected to see a positive effect of trees within the riparian zone, as per Hazell et al. (2001), however in the present study, this was not evident. Further study is required to determine if

this is an issue of statistical power, or whether this expected relationship deteriorates in urban contexts.

Whistling tree frog (Litoria verreauxii)

In the present study, no model of occurrence explained any species-specific habitat preference for *Litoria verreauxii*. However, previous studies had found negative effects of bare ground in the riparian zone of rural ponds on the prevalence of this species (Hazell et al. 2001), but no preferential occupation of natural versus constructed ponds (Hazell et al. 2004). This species has experienced significantly decline over the past decades, most likely caused by the emergence of chytridiomycosis in the ACT during the 1980s (Osborne 1990, Scheele et al. 2014). New studies indicate that *Litoria verreauxii* are now re-expanding (Scheele et al. 2014), by around 500m per annum (B. Scheele, pers. comm.). Between 2002 and 2015 this species was detected at less than 40% of our sites. Our analysis might not have captured dynamics of a spreading population over that time, with true occupancy of wetlands yet to stabilise for this species. In addition, *Litoria verreauxii* starts breeding in winter and may decrease breeding activities when spring breeders are very active and therefore might be under-represented in our monitoring, which was tailored to capture the vocalisation of breeding frog species during the spring mating period. As this species is now re-expanding into the ACT, species-specific surveys are much needed to determine habitat preferences and population dynamics of this species.

Whistling tree frog at Mulligans Flat. Photo credit: Matthew Frawley

Sudell's frog (Neobatrachus sudelli)

Sudell's frog is a burrowing species that is rarely encountered in the ACT region. Its distribution is poorly known, and this is most likely related to the difficulties in detecting this species. It spends much of its time buried underground, and emerges during the spring and summer months after large rainfall events to breed (Cogger 2014). Across our 33 wetlands examined in this study, only 4 of these wetlands have

ever reported *Neobatrachus sudelli.* This limited distribution makes drawing inferences regarding their habitat preferences problematic. In the present study, we found a negative association with *Gambusia holbrooki*. While it is quite plausible that *Neobatrachus sudelli* may be negatively impacted by *Gambusia holbrooki*, particularly if the eggs and tadpoles of *Neobatrachus sudelli* are vulnerable to *Gambusia holbrooki* predation. In the present study we do not believe this to be a strong relationship, rather potentially another example of correlation rather than causation. Further research is required to understand the broader distribution of this species within the ACT region, and its habitat preferences, and interactions with feral fish species. Recent pit-fall surveys in grasslands around the ACT have revealed this frog many 100s of metres from the nearest water (B. Howland, pers. comm.). This suggests that habitat quality beyond the immediate margins of wetlands where they breed will be highly important to this species.

Smooth toadlet (Uperoleia laevigata)

Detection of *Uperoleia laevigata* was found to be responsive to multiple factors. Firstly, connectivity to a nearby patch of forest had a strong significant positive effect, greatly increasing the likely detection of this species. This is perhaps not surprising, as *Uperoleia laevigata* are well known to spend much of their adult lives away from water, living under cover such as rocks, logs and deep leaf litter in forests (W. Osborne, pers. comm.). Furthermore, males often call from many metres away from water, and as such, require sufficient riparian cover from which to call (D. Hazell, pers.comm). It seems likely that females exhibit wide-ranging movements away from wetlands throughout their adult lives.

Secondly, *Uperoleia laevigata* was found to be negatively associated with wetlands containing *Gambusia holbrooki*. This pattern of negative association has occurred multiple times, and warrants further study. To what extent *Gambusia holbrooki* and hydrological regime of wetlands are correlated is unknown in the present dataset and requires further study. Furthermore, the specific responses of the tadpoles of each species in relation to *Gambusia holbrooki* urgently needs to be studied. It seems likely that species-level differences in behaviour, timing and duration of the larval phase and microhabitat use are interacting to modulate the susceptibility of tadpoles to *Gambusia holbrooki* predation (Baber and Babbitt 2004). This requires urgent study in order to understand these potential interactions, and how they may best be managed through modifications to urban wetland design.

Furthermore, *Uperoleia laevigata* showed a positive response to logs in the riparian zone, and a negative response to mowing. Again, actually untangling which response variable is the causative agent requires a follow-up study, but irrespective, this result demonstrates this species is responsive to the quality of riparian condition around wetlands.

Lastly, *Uperoleia laevigata* showed a strong negative response to electrical conductivity. This was the only species to show an effect of water quality, and this requires further research. In a study of natural and constructed rural dams *Uperoleia laevigata* showed a strong preference for farm dams (Hazell et al. 2004), which had on average much lower electrical conductivity than constructed dams. To our knowledge only two other studies have found species-specific reactions in frogs to changes in electrical conductivity. Increasing levels of electrical conductivity have been shown to negatively affect frog larvae and their development through species-specific effects on stress hormone levels (Chambers 2011). A study of wetlands along the Merri Creek (Victoria) showed that species richness was negatively correlated with electrical conductivity (Ficken & Byrne 2013).

Review of factors influencing frogs at urban/peri-urban wetlands

In the present study, we examined the role of 5 competing hypotheses, regarding the drivers of frog species richness, and presence/absence of species. In the present study, we found that at various levels, all 5 hypotheses were supported. Connectivity to intact patches of forest was supported for 5 of the 9 frog species examined, suggesting that even within the urban peri-urban wetlands, connectivity is a major factor in frog detection. This result is congruent with Westgate et al. (2015) who examined a larger set of sites that exhibited a greater range of variation in connectivity attributes. As such, it was not entirely expected to observe this result in the present study. It does add further evidence to the value of maintaining forest patches, even within an urban landscape. The growing trend of in-filling within suburban environments will undoubtedly have a negative impact upon frogs, despite the small size and fragmented nature of these forest patches within the urban mosaic. Maintaining and increasing the area of environmental offset properties within urban areas may be fundamental to long-term persistence of frogs in urban environments.

The presence of the invasive fish, *Gambusia holbrooki*, had a negative impact on two species of frog, and was positively correlated with the presence of another species of frog. *Gambusia holbrooki* have been implicated in the decline and localised extinction of multiple species in the Sydney region (e.g. Gillespie and Hero 1999, NSW National Parks and Wildlife Service 2003, Holbrook and Dorn 2016). *Gambusia holbrooki* are widespread throughout the ACT region, and occur in many of the wetland habitats in the ACT. Generally considered to be poor dispersers, it seems likely that human-aided dispersal is a factor. Observations at community events have revealed that the vast majority of the public identify *Gambusia holbrooki* as being tadpoles (D. Starrs, pers. obs.). This is a major concern and further education is needed to inform the public on this important issue.

We predicted that riparian condition would be a strong predictor of frog presence absence around wetlands. However, despite the wide range of variables examined, both using the RARC and FRARC, fallen logs in the riparian zone was the main predictor, being significant for frog species richness, and having a positive impact on three individual species. Additionally, rocks and reeds in the riparian zone were also had a positive effect *on Limnodynastes dumerilii* and *Limnodynastes peroni*, respectively. The strong correlation between logs and mowing around wetlands requires further study to disentangle these confounding factors, but either way, it appears the condition of the riparian zone around wetlands is of fundamental importance to frogs within the urban/peri-urban wetland environments. Ensuring the integrity of these habitats is vital to healthy frog populations.

In-water habitat variables were less significant in the present study. Only *Litoria peronii* showed a positive effect of emergent macrophytes to their detection rate. A drawback in the present study is the strong focus on calling male frogs being the mode of determining frog presence/absence. This method does not give strong indications of either female frog presence/absence (males could be calling, but no females are present), and nor does it take into account the early life history phases of frogs.

To what extent calling frogs at wetlands are indicators of successful recruitment is unknown. It is plausible that strong source-sink dynamics occur in many wetlands, where calling frogs appear at wetlands following dispersal from other nearby sites. A wetland may not support successful recruitment, but may still support a population of calling male frogs, if they are able to disperse from nearby ‘source’ wetlands. Future research needs to focus on examining recruitment from these wetlands to determine if all are equally contributing to frog species persistence. The role of in-water cover, feral fish and water quality are likely to be far more important when considering recruitment, rather than solely the presence of calling male frogs.

Finally, water quality, specifically, electrical conductivity had a significant negative effect on *Uperoleia laevigata*. As suggested above, further examination of frog early life history phases is likely to demonstrate a greater influence of water quality on frog persistence in wetland environments. Currently, considerable efforts are underway to improve water quality in the ACT through multiple initiatives, including the Basin Priority Project. Incorporating information on indicator species such as *Uperoleia laevigata* into long-term monitoring programs would be a powerful way of examining meaningful change in water quality due to the construction of water quality control infrastructure aimed at improving the quality of water leaving the ACT.

Recommendations for Management: frogs as an indicator of wetland health

Frogs have been long identified as being good indicators of ecosystem health, due to their sensitivity to changes in ecosystems. However, actually establishing an indicator is somewhat problematic.

The overarching goal of this research project was to determine if frogs could be used as an indicator of wetland health in the urban/peri-urban wetlands of the ACT.

Of the 9 species detected in the ACT, 3 did not appear to be related to aspects of wetland condition. *Limnodynastes tasmaniensis, Litoria verreauxii* and *Neobatrachus sudelli* are currently not considered good indicators of wetland health. We anticipate, with further research, this situation may change in regards to *Litoria verreauxii*, and *Neobatrachus sudelli*, if its range and distribution are better understood. *Limnodynastes tasmaniensis* is a widespread and abundant species (similar to *Crinia signifera*) and as such, it is not surprising that it was not found to be related to wetland condition.

The remaining 6 species of frog in the ACT wetlands may be considered as indicators of wetland condition, although the value of *Crinia signifera* is debatable. A range of factors, including connectivity, riparian condition, in-water condition, feral fish and water quality all influence the presence/absence of these species. Here, we outline an approach to using these six species of frog as an indicator of wetland health:

Poor = 0-2 indicator species

Fair= 3-4 indicator species

Good= 5+ indicator species

If a wetland does not contain more than two of these six indicator species (ie, it may still contain *Limnodynastes tasmaniensis, Litoria verreauxii* and for example, *Crinia signifera*, which we consider to be poor indicators of wetland health, and having broad tolerances to a range of poor conditions), we would consider the wetland to be in poor health. It would be likely that a range of issues are present, not least poor connectivity to other suitable habitat, poor riparian condition and potentially low water quality. There may also be a considerable influence of invasive fish at this wetland.

A wetland containing three or four indicator species would be considered to be in fair condition. Depending on the species present, it would be likely that some key attributes are missing. For example, it may be suffering from limited connectivity to additional suitable forest habitat, and may have feral fish present. As we observed, *Limnodynastes peroni* is a species that may persist where feral fish may have a negative impact on other species.

A wetland containing five or more of these indicator species may be considered to be in good condition. It would be likely that connectivity to a suitable patch of forest is good, and a strong riparian zone may be present. In natural grassland environments, without a natural tree/canopy or understorey, this may be considered to be in excellent condition.

Clearly, an opportunity exists here to set goals regarding the diversity of frogs at urban wetlands. A reasonable goal would be to have at least three indicator frog species present at each wetland, and detected regularly (each species detected at least once every 3 years). Stocking of frogs and tadpoles is undesirable, as natural dispersal is a more natural process for creating and maintaining diversity in a disjointed landscape. Improving habitat connectivity to allow natural replenishment would be preferable to stocking frogs in order to achieve the desired diversity.

The adoption of current Frogwatch sampling practices (surveys for frogs calling during the winter/spring breeding seasons should be considered as a viable technique for implementing an urban/peri-urban wetland health monitoring program. Paying distinct attention to the 6 identified indicator species would provide a good indication of overall wetland health. Long-term monitoring could be used to evaluate trends in changes to wetland health, using the changes in presence of indicator frog species. Further advances could be made by extending surveys to tadpoles and metamorphlings at wetlands in order to start building a picture of recruitment dynamics within urban wetlands.

Frogspawn

Recommendations for Management: Frog sensitive wetland design

Future design and management of urban wetlands can aim to balance the demands of water management and biodiversity in urban environments. With increasing encroachment by urban development, there is a strong imperative to maximise the ecological value of green spaces within the urban extent. The present study has identified that maintaining even small patches of remnant forest are vital to many frog species, which utilise these habitats when not breeding. Education of policy makers and urban planners is required to ensure they appreciate the ecological values of these small remnant patches that are often treated as having minimal or no ecological value. The partial removal of remnant patches of forest should also be considered as detrimental, as there is clear evidence that scale of remnant patches is important for some species. All new urban developments must ensure that connectivity between wetlands and remnant forest patches not be compromised. Maintaining ‘green corridors’ between wetlands and waterways and other green spaces are required to ensure long-term persistence of frogs.

Urban wetlands themselves can be designed to enhance their value in terms of maintaining frog biodiversity. Designs and implementation that provide for fringing emergent macrophytes is vital to providing suitable breeding habitat and habitat for eggs and tadpoles, and other species (Ebner and Lintermans 2007). A diversity of native trees and understorey species around the wetlands (as opposed to manicured lawns) will provide shelter for adult phases of frogs, as well as key breeding habitats for some species. Additional provision of large woody debris and rocks in vicinity (up to 500m) around a wetland will augment the available habitat for frogs. A direct means of draining and otherwise controlling the water regime of wetlands may be beneficial, to both frogs and aquatic and riparian plant species (Roberts and Marston 2011, Hamer and Parris 2013). Being able to drain a wetland to remove feral fish and improve nutrient cycling to ensure long-term water quality will aid frog populations. However, careful consideration and appreciation of frog life histories is

needed to ensure negative effects are not incurred by well-meaning but incorrect operation of these structures. A clear opportunity exists to monitor new and old designs to explore the benefits of frog-sensitive wetland designs to gauge the benefits to frogs, and fine-tune management actions.

The role that pollution plays in the decline of frogs is poorly known. We found that water quality (electrical conductivity) may have strong negative effects on *Uperoleia laevigata*. Frog sensitive urban design should consider how potential pollutants may have an adverse impact on frogs, and consider ways to minimise these effects. A chain-of-ponds style wetland may allow for the capture and bio-processing of pollutants in discrete units, allowing for downstream-most structures to be relatively pollutant free, and suitable habitat for sensitive frogs.

Future Research Directions

This research has identified numerous knowledge gaps and opportunities for future studies.

A range of studies exist that have examined frog-habitat relationships throughout south-eastern Australia (eg. Hazell et al. 2001, 2003, 2004, Jansen and Healey 2003, Parris and Lindenmayer 2004, Lemckert et al. 2006, Lane et al. 2007, Hamer et al. 2012). These studies have examined a wide range of habitats, including urban wetlands, rural farm dams and natural chain-of-pond environments and forestry plantations. A meta-analysis that combines all these findings may help to reveal the commonality between frog-habitat relationships across a wide range of environments. It may reveal that irrespective of the environment, frogs display the same preferences for specific habitat traits. Conversely, it may reveal that land use may impact upon species-habitat relationships. This is an area for future research.

Wetland

A key knowledge gap in the present study is the hydrological regime of the wetlands examined. Previous studies have identified that ephemeral wetlands suit frogs greatly, in that they can break the life cycles of truly aquatic species, such as fish (Hamer and Parris 2013, Lowe et al. 2015). Indeed in the present study, *Gambusia*

holbrooki may be restricted to those wetlands that are semi-permanent, but not intermittent, episodic or ephemeral (Roberts and Marston 2011).

Jerrabomberra Wetlands Kelly's reserve swamp

Utilising surveys of frog calls is widespread in herpetofaunal studies. However, the relationships between calling male frogs and habitat quality in terms of species persistence and successful recruitment is unknown. A clear direction for future research is to explore the relationship between the abundance of calling frogs, and recruitment success (in terms of metamorphlings leaving a wetland, and returning as adults to breed). This research may reveal source-sink metapopulation dynamics, and further highlight the importance of connectivity between wetlands to facilitate frog persistence.

Electrical conductivity in urban environments will reflect a wide range of natural elemental factors, in addition to multiple pollutant factors. Further research examining the actual composition of highly conductive urban wetland waters is required to unravel these factors, in order to identify the likely mechanistic relationship between electrical conductivity and its impact upon *Uperoleia laevigata*. Furthermore, studies in catchments with naturally high electrical conductivity due to underlying geological processes (e.g. Yass catchment) will prove to be insightful in determining the mechanistic relationship.

Upper Murrumbidgee Waterwatch currently examines electrical conductivity at a wide range of sites throughout the upper Murrumbidgee catchment (O'Reilly et al. 2015). Further analysis of the presence-absence of *Uperoleia laevigata* at these

sites where high level electrical conductivity exists would further help to explain this relationship.

Community engagement

Involving the community in caring for the local environment has many benefits, such as increased awareness and skill development, a strong sense of community through a common goal, cultural and social connectivity, as well as strong economic returns, which can be 2 - 5 times the amount of the initial investment (online resource 7). Strong community engagement in creating and improving frog habitat, and monitoring frog populations and water quality is desirable to ensure ongoing progress and continuous and consistent data collection. Awareness raising events and projects, tailored to suit schools, the general public and special interest groups, could be used to educate about our urban wetlands and their value for biodiversity, to attract volunteer, run working bees, and facilitate frog habitat workshops. Appropriate organisations with solid existing community relationships to undertake these activities would be the ACT's Catchment Groups (CG) - Ginninderra CG, Molonglo CG and Southern ACT CG, as well as the ACT and Region Frogwatch Program.

Frogwatch volunteers in action

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Online resources

resource 1: Australian Museum http://australianmuseum.net.au/eastern-gambusia-gambusia-holbrooki-girard-1859

resource 2: www. https://frogs.org.au/frogs/

resource 3: http://www.tams.act.gov.au/city-services/public_areas/grass-mowing

resource 4: https://frogs.org.au/frogs/species/Crinia/parinsignifera/

resource 5: https://frogs.org.au/frogs/species/Crinia/signifera/

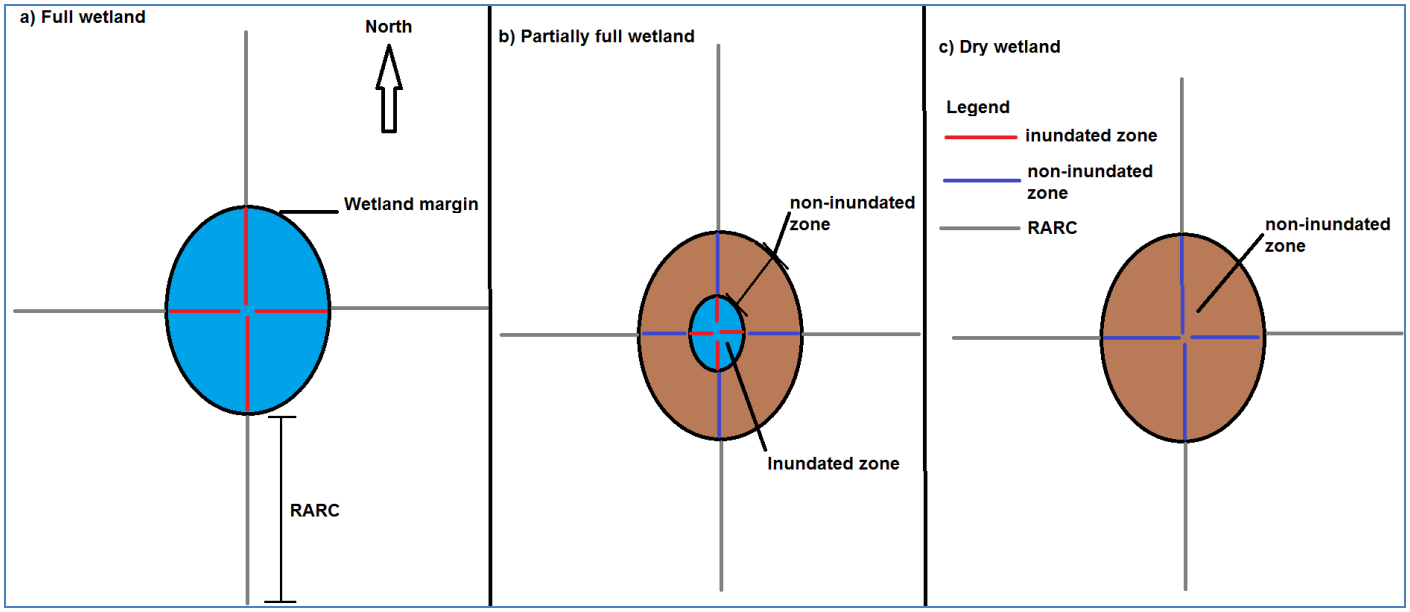
resource 6: https://frogs.org.au/frogs/species/Limnodynastes/peroni/

resource 7: http://www.landcarensw.org.au/wp-content/uploads/2015/11/Multiple-Benefits-of-Landcare.pdf

RARC for Wetlands Cheat sheet (version 2.0)

1. *Determine site size*

The majority of sites will be around small wetlands, or isolated dams. The RARC was developed for application along rivers, so we have to make an adjustment to how they are applied to a static water body that may be of small size. The following diagram depicts how a RARC is to be conducted at a wetland/farm dam, and definitions of the terms used on the frog habitat assessment sheet. On large wetlands with scope for multiple frogwatch sites, RARCs will be conducted in accordance with the original directions as outlined for rivers (eg, transects parallel along the bank). Transects are to be undertaken in a North, East, West, South direction from the wetland. RARC transects are a 'belt transect', 10m wide. Length of RARC transect should be determined based on typology around the wetland. Ideally, transects should not be less than 50m long, but not more than 200m long. Often roads, dwellings etc may limit transects.



2. *Score indicators*

Habitat

(A) Edge connectivity – % of wetland perimeter with vegetation cover (> 5m wide) (-0.5 pt for each gap > 50m in length).
(B) Width of riparian vegetation (m). Break = gap > 50m. Calculate as proportion of wetland width if wetland > 10m wide.
(C) Proximity to > 10ha patch –distance (km) to nearest patch of relatively intact native vegetation > 10Ha (+1 pt if > 50Ha).

Cover

(D) Groundcover – % cover of mosses, lichens, grasses, herbs, reeds and sedges < 1m tall.
(E) Understorey cover – % cover of herbs, reeds, shrubs and saplings 1–5m tall.
(F) Canopy cover – % cover of trees > 5m tall.
(G) Number of layers – sum of groundcover, understorey and canopy layers present.

Natives

(H) Groundcover - % cover of mosses, lichens, grasses, herbs, reeds and sedges < 1m tall that are native.
(I) Understorey cover - % cover of herbs, reeds, shrubs and saplings 1–5m tall that are native.
(J) Canopy cover - % cover of trees > 5m tall that are native.

Debris

(K) Leaf litter – % cover of leaf litter.
(L) Native leaf litter – % cover of native leaf litter.
(M) Standing dead trees – Presence/absence of dead trees > 20cm diameter at breast height.
(N) Hollow –bearing trees – Presence/absence of hollow-bearing trees.
(O) Fallen logs – abundance of fallen logs (> 10cm diameter).

Features

(P) Native canopy species regeneration – abundance of native canopy species < 1m tall minus browsing damage.
(Q) Native understorey regeneration – abundance of native understory species regenerating.
(R) Large native tussock grasses – abundance of large native tussock grasses.
(S) Reeds, sedges etc – abundance of reeds, sedges, rushes etc

Observer: ..... Date: ..... Time: .....

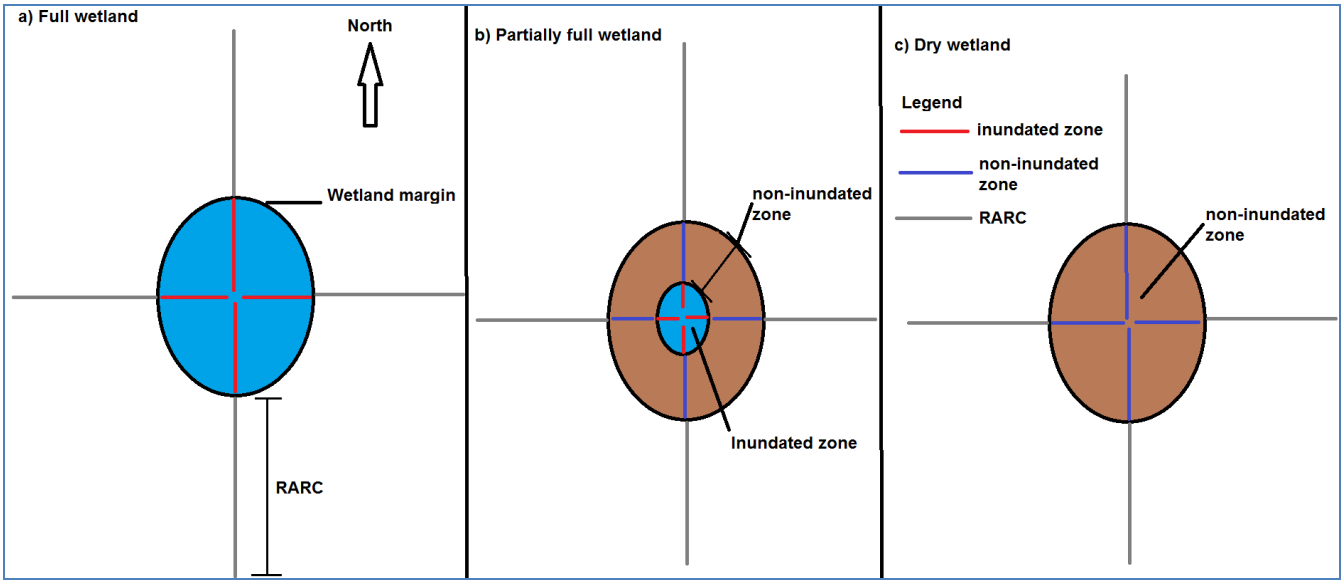
Site code/name: ..... GPS start: ..... GPS end: .....

Map longitudinal riparian veg. >5m deep | <50% | 50-64% | 65-79% | 80-94% | >95% vegetated margin	Number of breaks (>50m long)

Nearest patch of native veg. >10ha	>1km		200m-1km		Contiguous		Contig. with patch >50ha	
Transect direction (N, S, E, W)								
WIDTH	**Transect 1**		**Transect 2**		**Transect 3**		**Transect 4**	
Wetland width (centre to edge)								
Vegetation width								
COVER	**Transect 1**		**Transect 2**		**Transect 3**		**Transect 4**	
Total canopy %	None	1-30%	None	1-30%	None	1-30%	None	1-30%
	31-60%	>60%	31-60%	>60%	31-60%	>60%	31-60%	>60%
Native canopy %	None	1-30%	None	1-30%	None	1-30%	None	1-30%
	31- 60%	>60%	31- 60%	>60%	31- 60%	>60%	31- 60%	>60%
Total understorey %	None	1-5%	None	1-5%	None	1-5%	None	1-5%
	6-30%	>30%	6-30%	>30%	6-30%	>30%	6-30%	>30%
Native understorey %	None	1-5%	None	1-5%	None	1-5%	None	1-5%
	6-30%	>30%	6-30%	>30%	6-30%	>30%	6-30%	>30%
Total ground cover %	None	1-30%	None	1-30%	None	1-30%	None	1-30%
	31-60%	>60%	31-60%	>60%	31-60%	>60%	31-60%	>60%
Native ground cover %	None	1-30%	None	1-30%	None	1-30%	None	1-30%
	31-60%	>60%	31-60%	>60%	31-60%	>60%	31-60%	>60%
DEBRIS	**Transect 1**		**Transect 2**		**Transect 3**		**Transect 4**	
Total leaf litter	None	1-30%	None	1-30%	None	1-30%	None	1-30%
	31-60%	>60%	31-60%	>60%	31-60%	>60%	31-60%	>60%
Native leaf litter	None	1-30%	None	1-30%	None	1-30%	None	1-30%
	31-60%	>60%	31-60%	>60%	31-60%	>60%	31-60%	>60%
Standing dead trees	Present	Absent	Present	Absent	Present	Absent	Present	Absent
Hollow bearing trees (>20 DBH)	Present	Absent	Present	Absent	Present	Absent	Present	Absent
Fallen logs (> 10cm diam)	None	Few	None	Few	None	Few	None	Few
	Abundant		Abundant		Abundant		Abundant	
FEATURES	**Transect 1**		**Transect 2**		**Transect 3**		**Transect 4**	
Native canopy species regeneration	None	Scattered	None	Scattered	None	Scattered	None	Scattered
	Abundant		Abundant		Abundant		Abundant	
Native understorey regeneration	None	Scattered	None	Scattered	None	Scattered	None	Scattered
	Abundant		Abundant		Abundant		Abundant	
Large native tussock grasses	None	Scattered	None	Scattered	None	Scattered	None	Scattered
	Abundant		Abundant		Abundant		Abundant	
Reeds, rushes, sedges etc.	None	Scattered	None	Scattered	None	Scattered	None	Scattered
	Abundant		Abundant		Abundant		Abundant	
Evidence grazing damage	Present	Absent	Present	Absent	Present	Absent	Present	Absent

Frog Habitat Assessment cheat sheet

The frog habitat assessment sheet is designed to complement the RARC for wetland datasheet, in order to capture additional fine-scale data related to frog-specific habitat variables, and important features of wetlands that differ to riverine habitats.



Sections 1–3 should be examined adjacent to the RARC transects. Section 4 corresponds to the entire wetland.

1. *Additional RARC variables* – These are to be undertake along each transect concurrently with the RARC assessment.

- Evidence of mowing – report whether regular mowing takes place such that ground cover has been extensively modified, potentially limiting the amount of refuge habitat available.
- Number of rocks >20cm diameter – record number of large rocks that occur along the RARC transect.
- Number of trees within 10m of the wetland margin. Trees close to the water's edge are more likely to provide important habitat for breeding frogs.
- Number of tussocks within 10m of the wetland margin – record the number of tussocks (>50cm high) that are within 10m of the wetland margin. Tussocks close to the water's edge are more likely to provide important habitat for breeding frogs.

2. *Non-inundated zone variables* – These attributes relate to the zone between the water's edge (see figure). Note that when a wetland is full, this zone does not exist. When a wetland is completely dry, this zone encompasses the entire wetland.

- Width of bare ground – report the length of bare ground (ground with insufficient cover for frogs) between the water's edge and the wetland margin.
- Width of reed/rush/sedges – report the length of ground covered by reeds, rushes, sedges and other riparian plants between the water's edge and the wetland margin.
- Abundance of large woody debris, rocks, and boulders – report the number of large wood (eg. >1m long), rocks and boulders present in the non-inundated area.

3. *Inundated zone variables* – these attributes relate to the area inundated by water within the wetland.

- Width of emergent vegetation – report the width of vegetation that is growing within the inundated area that emerges from the water. This will generally consist of reeds and rushes.
- Max height of emergent vegetation – report the maximum height of the emergent vegetation within the current inundated area.
- Width of submerged/floating vegetation – report the width of the submerged and floating vegetation. Often identifying the submerged vegetation will depend on water clarity, and distance from the water's edge.
- Abundance of large woody debris, rocks, and boulders – report the number of large wood (eg. >1m long), rocks and boulders present in the inundated area.
- Water depth 1m from water's edge – report the water depth 1m (eg., arm's length) from the water's edge.

4. *Whole-of-wetland* – These variables relate to the entire wetland, rather than points associated with the RARC transects.

- % vegetation cover for whole wetland – estimate the % cover of BOTH the vegetation covering the non-inundated zone, and the in-water zone.
- % cover submerged/floating/emergent vegetation – estimate the % cover of vegetation in the inundated area of the wetland.
- Wetland water level % - estimate how full the wetland is with water.

Observer: Site Code/Name: Date/Time:

GPS Co-ordinates: RARC form Number:

Additional RARC variables (along transects)

	Transect 1		Transect 2		Transect 3		Transect 4	
Evidence of lawn mowing?	No	Yes	No	Yes	No	Yes	No	Yes
Number of rocks (> 20cm) from full level zone	None	< 5	None	< 5	None	< 5	None	< 5
	< 10	> 10	< 10	> 10	< 10	> 10	< 10	> 10
Number of trees within 10m of full level zone	None	< 5	None	< 5	None	< 5	None	< 5
	< 10	> 10	< 10	> 10	< 10	> 10	< 10	> 10
Number of tussocks within 10m of full level zone	None	< 5	None	< 5	None	< 5	None	< 5
	< 10	> 10	< 10	> 10	< 10	> 10	< 10	> 10

Full level zone variables (water's edge to full level)

	Transect 1		Transect 2		Transect 3		Transect 4	
Width of full level zone (water's edge to full level)	0m	< 2m	0m	< 2m	0m	< 2m	0m	< 2m
	< 10m	> 10m	< 10m	> 10m	< 10m	> 10m	< 10m	> 10m
Width of bare ground (water's edge to full level)	0m	< 2m	0m	< 2m	0m	< 2m	0m	< 2m
	< 10m	> 10m	< 10m	> 10m	< 10m	> 10m	< 10m	> 10m
Width of reed/rush/sedges (water's edge to full level)	0m	< 2m	0m	< 2m	0m	< 2m	0m	< 2m
	< 10m	> 10m	< 10m	> 10m	< 10m	> 10m	< 10m	> 10m
Abundance of large woody debris, rocks and boulders (n)	None	< 5	None	< 5	None	< 5	None	< 5
	< 10	> 10	< 10	> 10	< 10	> 10	< 10	> 10

In-water variables

	Transect 1		Transect 2		Transect 3		Transect 4	
Width of emergent vegetation	0m	< 1m	0m	< 1m	0m	< 1m	0m	< 1m
	< 2m	> 2m	< 2m	> 2m	< 2m	> 2m	< 2m	> 2m
Max height of emergent vegetation	0m	< 0.5m	0m	< 0.5m	0m	< 0.5m	0m	< 0.5m
	< 1m	> 1m	< 1m	> 1m	< 1m	> 1m	< 1m	> 1m
Width of submerged/floating vegetation	0m	< 1m	0m	< 1m	0m	< 1m	0m	< 1m
	< 2m	> 2m	< 2m	> 2m	< 2m	> 2m	< 2m	> 2m
Abundance of large woody debris, rocks and boulders (n)	None	< 5	None	< 5	None	< 5	None	< 5
	< 10	> 10	< 10	> 10	< 10	> 10	< 10	> 10
Water depth 1m from water's edge	<0.25m	< 0.5m	<0.25m	< 0.5m	<0.25m	< 0.5m	<0.25m	< 0.5m
	< 1m	> 1m	< 1m	> 1m	< 1m	> 1m	< 1m	> 1m

Whole-of-wetland

% vegetation cover for whole wetland (full level zone)			0%	< 10%	< 50%	> 50%
% cover of submerged/floating/emergent vegetation (in-water zone)			0%	< 10%	< 50%	> 50%
Wetland water level (%)	0%	< 25%	< 50%	< 75%	<100%	Full

Comments:

Appendix 3. List of wetland sites examined in the present study

site code	Lat	Long	Location
ANB100	-35.27813	149.11009	ANBG Pond, near Cafe
ANU012	-35.28054665	149.1116922	ANU Dickson Road Carpark
ARA100	-35.2764	149.0779	Aranda Paddock Dam
BSW001	-35.2576	149.118015	Banksia Street Wetland
CBR004	-35.3555	149.137	Callum Brae NR Site 4
CMC100	-35.357	149.0263	Cooleman Ridge Old Dam
CMC600	-35.3813	149.0414	Mt Neighbor Horse Paddock Dam
CMC750	-35.3699	149.0565	Fisher Dam
COO100	-35.350089	149.02255	Cooleman Ridge North-West Dam
DGP001	-35.185	149.0332	Dunlop Grassland Pond
FAD100	-35.398	149.117	Fadden Silt Pond
FBM100*	-35.2833	149.0871	Glenloch interchange dam
FER100	-35.2397	149.0908	Fernhill Technology Park, Bruce
FGC009	-35.2031	149.014	Jarramlee Pond
FGD005	-35.23701057	149.0748516	John Knight Park dam
FGD020	-35.2456	149.1123	O'Connor Ridge Dam
FGD040	-35.2772	149.0823	Aranda Bushland Dam
FMC200	-35.251	149.1745	Majura bottom dam, McKenzie St
FMC210	-35.2506	149.1769	Majura top dam,McKenzie St
FMC220	-35.2412	149.1688	Majura lower, via Jukes Street
FMW010	-35.2633	149.1239	David Street Wetland
JER500	-35.316	149.1605	Kelly's swamp bird hide, JWNR
JER530	-35.318029	149.15965	Siltbrap JWNR
MCW001	-35.216032	149.082582	McKellar Wetland South
MCW002	-35.212989	149.082147	McKellar Wetland North
MFL005	-35.163795	149.157845	Mulligans Flat NR
MFL007	-35.163592	149.163822	Mulligans Flat NR
MFL016	-35.1665779	149.1788925	Mulligans Flat NR
MIT100	-35.21303999	149.1313621	Gungahlin Cemetary, Mitchell
NAD034	-35.284	149.0814	National Arboretum, Cork oaks
NAD036	-35.28694444	149.0608333	National Arboretum,
RCD001	-35.396274	149.134834	Rose Cottage Dam, Paddock 8
UCP100	-35.240001	149.087031	UC pond next to childcare