# The impacts of agricultural practices on the occurrence of frogs in Southwest Victoria

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Photo: A crop of broad beans sown through an ephemeral wetland.

## STATEMENT OF RESPONSIBILITY

This thesis is submitted in accordance with the regulations of Deakin University in partial fulfilment of the requirements of the degree of Environmental Science Honours. I, Kim Lewandowski, hereby certify that the information presented in this thesis is the result of my own research, except where otherwise acknowledged or referenced, and that none of the material has been presented for any degree at another university or institution.

Signature of candidate: Signature Redacted by Library Date: 17$^{th}$ May 2023

## ETHICS AND RESEARCH PERMITS

This project was conducted with the approval of the Animal Ethics Committee Wildlife chapter of the Deakin University Animal Ethics Committees (B29-2022). A permit to carry out research on private land was not required by the Department of Environment, Land, Water and Planning under the Wildlife Act 1975.

Principle Investigator: Don Driscoll

Co-Investigator: Kim Lewandowski

**ABSTRACT**

As a result of an ever-growing and rapidly expanding human population, the demand for food production has reached new levels. Predictions to meet this demand in the coming decades will see agricultural expansion on a scale like never before with more natural landscapes set to succumb to modification and destruction. The impacts of agricultural practices on global wildlife through habitat loss and destruction have been widely evident for all taxa and cause for alarm, however, the rapid decline of amphibians over the last few decades has been extraordinary. In this study, we investigated the impacts of cropping, grazing, wetland type, refuge and tree availability, and vegetation type on the occurrence of frog species in the lake district of south-west Victoria. We carried out two nocturnal audio surveys on 23 cropped and 23 uncropped swamps on either privately owned land or public access roadsides. Using occupancy modelling and principal components analysis, we found one species (*Limnodynastes tasmaniensis*) responded to the types of swamps available whereas another species (*Litoria ewingii*) negatively reacted to grazing levels and showed favourable vegetation types. No species were impacted by cropping, refuge availability or tree availability. With agricultural intensification on the rise, the importance of understanding wildlife responses to these practices is critical. This knowledge will help guide and implement land management practices and conservation efforts to assist in slowing down the rate of biodiversity loss.

**ACKNOWLEDGEMENTS**

I would like to thank my supervisor, Don Driscoll, for his patience and guidance throughout a rather long honours time frame. His advice and knowledge, as well as encouragement kept me going through what was a difficult course for me.

I would also like to thank my one and only volunteer, Renata Lewandowski, who without her encouragement, I would have given up way before completion. Thank you for braving the mosquito and snake infested swamps with me during the dark of night as well as that one landowner from whom we feared for our lives. Thank you for your patience during the gruelling vegetation surveys and for keeping me sane while counting each and every refuge at a site. Thank you for encouraging me every step of the way and not allowing me to give up during the analysis and write-up phase of the project.

Thank you to the many landowners who allowed me to come onto their properties at ridiculous hours of the night to listen to frogs and for happily talking to me about their properties and what they use their land for. I would also like to thank Ayesha Burdett for putting me in contact with landowners in the area. Without these people, this project would not have been possible.

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This thesis is based upon the format presented within the Austral Ecology Journal.

## INTRODUCTION

Global agricultural practices are a key factor in land modification, unsustainable water use, green-house gas emissions and loss of biodiversity (Cayuela, Lambrey, Vacher, & Miaud, 2015; Jones, Monjeau, Perez-Guzman, & Harrison, 2023; Matson, Parton, Power, & Swift, 1997). Presently, the global human population is more than 7.9 billion (Bureau, 2023) and the requirement for agricultural products is higher than it has ever been (Pradhan, Fischer, van Velthuizen, Reusser, & Kropp, 2015). The human population is estimated to increase further in the next few decades and with it, the demand for food (Hromada et al., 2021; Matson et al., 1997). From 1700 to 1980, the amount of land used for agriculture rose by 466% (Matson et al., 1997) and currently 48% of Australia is being used for agriculture (Statistics, 2023). It has been estimated that, to meet demand, global food production needs to expand by 60 to 110% by the year 2050 (Pradhan et al., 2015). While there is a strong focus on the need to feed a rapidly growing population, less importance is being placed on the significance of ecosystem services that aid agricultural practices (Henderson & Loreau, 2018). These services include water supply and purification, soil fertility, biodiversity, pollination, atmospheric balancing, and pest control (Henderson & Loreau, 2018; Matson et al., 1997). Since the 1950's, agricultural practices have rapidly increased and, coupled with urbanisation and revolutionary transport networks, have caused a major decline in suitable wildlife habitats and therefore contributed to extensive declines in farmland biodiversity (Arntzen, Abrahams, Meilink, Iosif, & Zuiderwijk, 2017; Cushman, 2006; Pulsford, Barton, Driscoll, & Lindenmayer, 2019). Globally, these effects are reaching a critical turning point and measures are urgently needed to move food production to a sustainable course (Jones et al., 2023; Matson et al., 1997).

Since the 1980's, there has been a rapid decline in amphibian numbers (Beebee & Griffiths, 2005; Burton, Gray, Schmutzer, & Miller, 2009; Cushman, 2006; Pabijan et al., 2023). This has been attributed to commercial use, introduced species that prey on/compete with native species, habitat loss and degradation, contaminants, disease and climate change (Collins, 2010; Cushman, 2006; Pabijan et al., 2023). Habitat loss and degradation, disease and exotic pathogens are three issues that play a part in modern amphibian decline and extinction (Burton et al., 2009; Collins, 2010). The IUCN (2023), currently lists 41% of assessed amphibians worldwide as being threatened with extinction, which is more than either mammals or bird (Beebee & Griffiths, 2005; Cayuela et al., 2015; Cushman, 2006). This number is vastly underestimated as many species are too inadequately studied to determine their status (Stuart et al., 2004).

Habitat loss and degradation, due to agriculture, has been suggested as a key factor in amphibian decline and a greater comprehension of how they can co-exist with this practice is required to conserve biodiversity (Arntzen et al., 2017; Cushman, 2006; Hromada et al., 2021; Pabijan et al., 2023). Deforestation and wetland drainage has destroyed amphibian populations globally (Pabijan et al., 2023) . Conserving forest cover in an agricultural landscape can be crucial for forest-dwelling amphibians (Beebee & Griffiths, 2005; Hromada et al., 2021). Amphibians have low vagilities therefore wildlife barriers, such as those imposed by habitat loss and infrastructure development, inhibit connectivity, and reduce dispersal rates (Arntzen et al., 2017; Cushman, 2006). Water quality is crucial to the survival of amphibians and contaminants found in pesticides and fertilisers are known to impact the development of larvae and increase the risk of diseases such as chytridiomycosis and ranavirus (Hromada et al., 2021). For example, Atrazine, a common herbicide, has been suggested to cause feminisation in frogs (Beebee & Griffiths, 2005). Livestock grazing can wear away shorelines, leave behind nitrogenous waste and destroy foraging and shelter sites through vegetation consumption and trampling (Burton et al., 2009; Hromada et al., 2021; Jansen & Healey, 2003). Changes to land use has a more prolific effect on taxa that are dependent on specific ecosystems, such as amphibians, which require both terrestrial and aquatic habitats making them particularly at risk to changes in either setting (Arntzen et al., 2017; Cayuela et al., 2015; Hazell, Cunnningham, Lindenmayer, Mackey, & Osborne, 2001). Loss and destruction of wetlands decreases the amount of suitable breeding sites for amphibians and loss of terrestrial landscaping reduces the amount of habitat that they could use for dispersal, foraging and hibernation (Cayuela et al., 2015; Hamer, Smith, & McDonnell, 2012; Hazell et al., 2001). Furthermore, fragmentation of ecosystems creates isolated populations in which genetic drift and inbreeding can cause further declines in species (Cayuela et al., 2015).

Naturally occurring wetlands in the lakes district of southwestern Victoria are being advanced upon by large-scale cropping to produce grain and seed such as wheat, barley, oats, pulses and canola (M. C. Casanova, A., 2016). Eighty-one percent of wetlands in this area are found on private land (P. Papas, and Moloney, P., 2012) and their conservation is therefore determined by individual land owners (M. T. Casanova & Powling, 2014). To produce maximum yield crops, land must be made suitable by removing trees, rocks and vegetation and draining water systems to avoid waterlog. Once a crop is planted, pesticides and herbicides are required to eliminate any threats and fertilisers are used to increase mineral uptake (M. C. Casanova, A., 2016). Through land-clearing to sow crops and chemical use to produce maximum yield, frogs in these areas may be susceptible to habitat loss and destruction.

An increase in concerns about the environmental effects of agricultural practices has opened up an interest in biodiversity conservation in farming areas (Hromada et al., 2021). Understanding how local species are being affected by agriculture is crucial to allow stronger conservation and management strategies to be implemented in these areas (Cayuela et al., 2015; Pabijan et al., 2023)

This study examines the relationships between swamp cropping and frog species occurrence in the western lakes district of southwestern Victoria, a highly agricultural area. We asked: 1) Does cropping affect the occurrence of frogs at swamps? 2) Is frog occurrence affected by grazing, refuge availability, tree cover, vegetation structure and vegetation type?

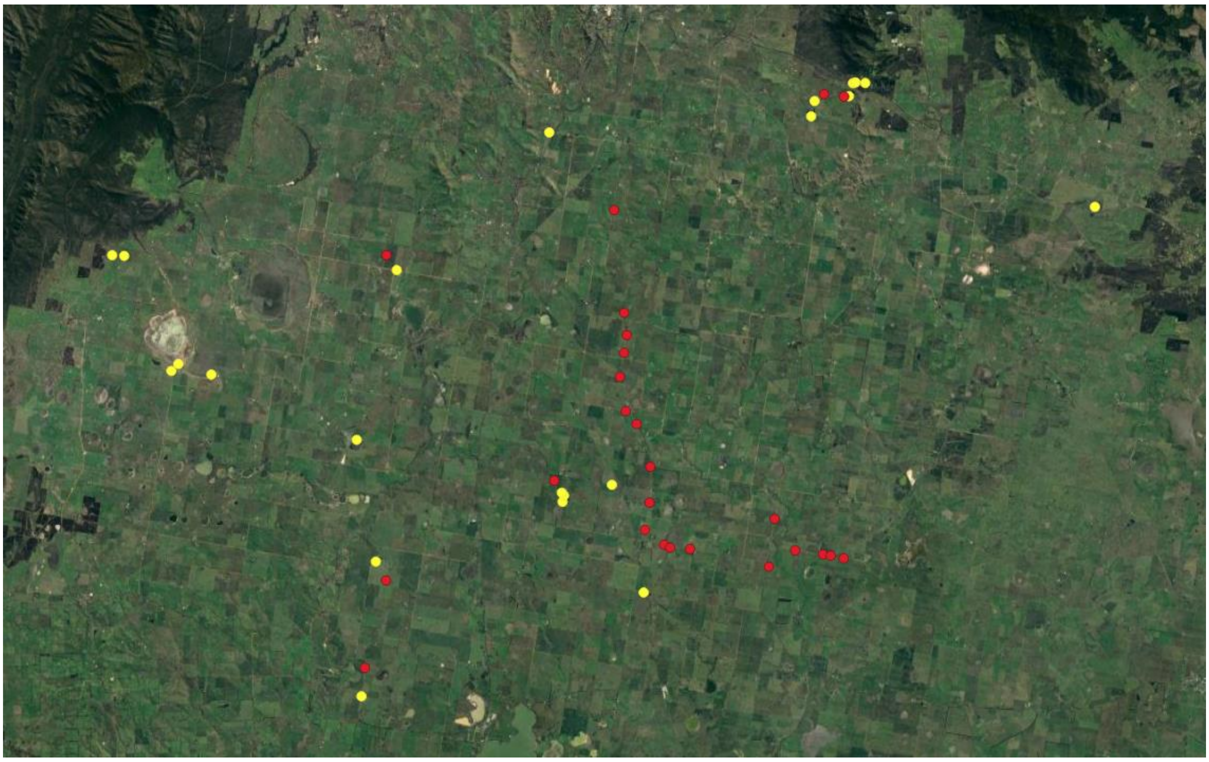
We predicted that frog occurrence would be greater in swamps that had not been cropped and that swamps with more refuges, less grazing, higher tree density and vegetation cover would support more occurrences.

## MATERIALS AND METHODS

### *Study Area*

Our study area was within the lake district of south-western Victoria which has a high density of ephemeral swamps. In this study, we defined the term 'swamp' to include lakes, wetlands, marshes and soaks, the distinctive characteristic being the nature of their ephemeral flooding (M. T. Casanova & Powling, 2014). The climate in Victoria's Western District lakes region is classed as temperate. Winters are cool and wet with average temperatures ranging from 11 to 13 $^{o}$C and summers are mild to hot with temperatures ranging from 22 to 25 $^{o}$C. Rainfall in the region can range from 600 to 1000mm per annum (Leahy, Robinson, Patten, & Kramer, 2010).

A)



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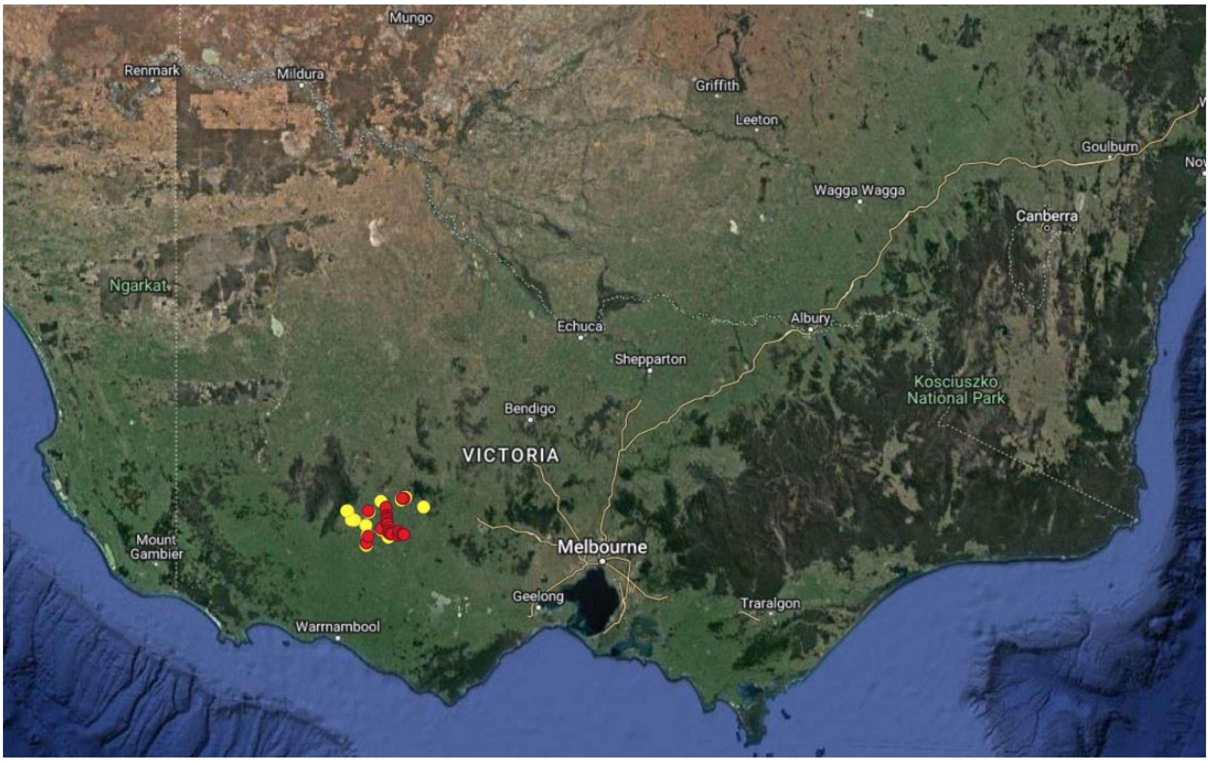


**Figure 1.** A) Spatial arrangements of study sites denoted by red circles for cropped sites and yellow circles for uncropped sites, and B) Location of the study area in relation to Victoria (Google Earth, 2023)

***Study Design***

We conducted two separate site surveys across 50 swamps during December and January. Sites were selected to include varying levels of frog habitat, with the assumption that suitable habitat would decline where cropping was present. Sites were located on either privately owned land, both agricultural and non-agricultural, or public roadsides north of Lake Bolac but south of the town of Ararat. Permission to access private land was gained from landowners, facilitated by Ayesha Burdett, Freshwater Ecologist for Riverbend Ecology.

Swamps were classified by either presence or absence of cropping within the last five years and were randomly selected from maps (privately owned land) and roadside access (if accessible and suitable). All swamps were freshwater and were categorised as either a swamp or a dam. Of the 46 swamps, 23 were classed as cropped and 23 uncropped. Four of the uncropped swamps were identified as dams and 19 as natural swamps.

Two separate nocturnal surveys were carried out at each site, starting 30 minutes after sunset. Surveys were carried out at night as the targeted species are more likely to be calling during this time period (Dodd, 2011). Ten minute surveys were conducted in this study as per suitable detection probabilities (Pierce & Gutzwiller, 2004). We allocated two minutes before each survey started to allow any frogs disturbed by our arrival to begin calling again (Dodd, 2011). Frogs were recorded regardless of distance from the swamp.

A third survey was conducted at all sites during the day. Site covariates (Table 1), likely to impact occurrence and detectability, were recorded using a Kestrel Weather Meter and the Bureau of Meteorology App, taken from the Ararat Prison Weather Station.

The number of refuges within a 150m radius of the swamp edge were also counted as a covariate. Refuges, for this study, are defined as a spatial protection available all year round that can provide shelter from predation and climatic changes, such as, logs, rocks and farm debris (Keppel et al., 2012). Trees were also counted within a 150m radius of the swamp edge. Grazing level was categorised at each survey site as either grazed or ungrazed.

Vegetation surveys were carried out according to (P. H. Papas, R. Amtstaetter, F. Clunie, P. Rogers, D., 2021). Percentages of vegetation cover were acquired by placing a 1m x 1m quadrat, every meter, along two five meter transects along the swamp edge. Vegetation was categorised into plant groups: litter, bryophyte, herb, grass, sedge, rush, shrub, crop and bare ground.

**Table 1.** Summary of site and survey covariates

| Covariate | Type | Description |
|---|---|---|
| Percentage vegetation cover | Site | Percent cover of vegetation types along the swamp edge, categorised into plant groups |
| Percent bare ground | Site | Percent cover of bare ground along the swamp edge |
| Vegetation height | Site | Average vegetation height along the swamp edge |
| Grazing level | Site | Categorized into none, low or high |
| Wetland type | Site | Categorised as either a swamp or a dam |
| No. of trees within 150m | Site | Number of trees, counted manually, within 150m of the swamp edge |
| No. of refuges within 150m | Site | Number of refuges, counted manually, within 150m of the swamp edge |

***Data Analysis***

To assess the role of vegetation type and structure in species occupancy, we used principal component analysis (PCA) to create uncorrelated variables (Lauck, Swain, & Barmuta, 2005; Milner, Hayes, Evans, & Starrs, 2015). I highlight vegetation variables as contributing important variation to a PCA axis if it had an R-squared value higher than 0.5.

We used binomial generalised linear models (GLM), with the car and lme packages within R (Bates, 2023; Fox, 2023). We fitted predictor variables individually in separate models, including cropping, refuge count, damming, tree count, grazing and the first three PCA axis. Presence/absence of frog species with enough records was the response variable. Only species detected at 6 or more sites were included in the analysis. We calculated a binomial GLM for each species to assess whether individual predictor variables could explain frog occurrence.

**RESULTS**

Of the 50 sites surveyed, four of them had dried up before surveys were completed and therefore, only data from 46 sites was used in analyses. We detected nine species of frogs throughout the survey however, we failed to detect any species at eight of these sites. Table 2 shows the number of sites each species was present at. Six species were used for analysis (top six).

**Table 2.** Target species and the number of sites they were present at.

| Scientific name | Common name | Sites present at |
|---|---|---|
| *Crinia signifera* | Common eastern froglet | 26 |
| *Limnodynastes dumerilii* | Southern banjo frog | 24 |
| *Litoria ewingii* | Southern brown tree frog | 22 |
| *Limnodynastes peronii* | Striped marsh frog | 19 |
| *Limnodynastes tasmaniensis* | Spotted marsh frog | 7 |
| *Crinia parinsignifera* | eastern sign-bearing froglet | 6 |
| *Litoria raniformis* | Growling grass frog | 1 |
| *Litoria verreauxii* | Whistling tree frog | 1 |
| *Neobatrachus sudellae* | Sudell's frog | 1 |


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***Effects of wetland type***

*Limnodynastes tasmaniensis* was significantly affected by wetland type (Appendix, Table 2.) with the likelihood of occurrence decreasing from 0.17 down to 0.04 if the wetland is a dam (Figure 2).

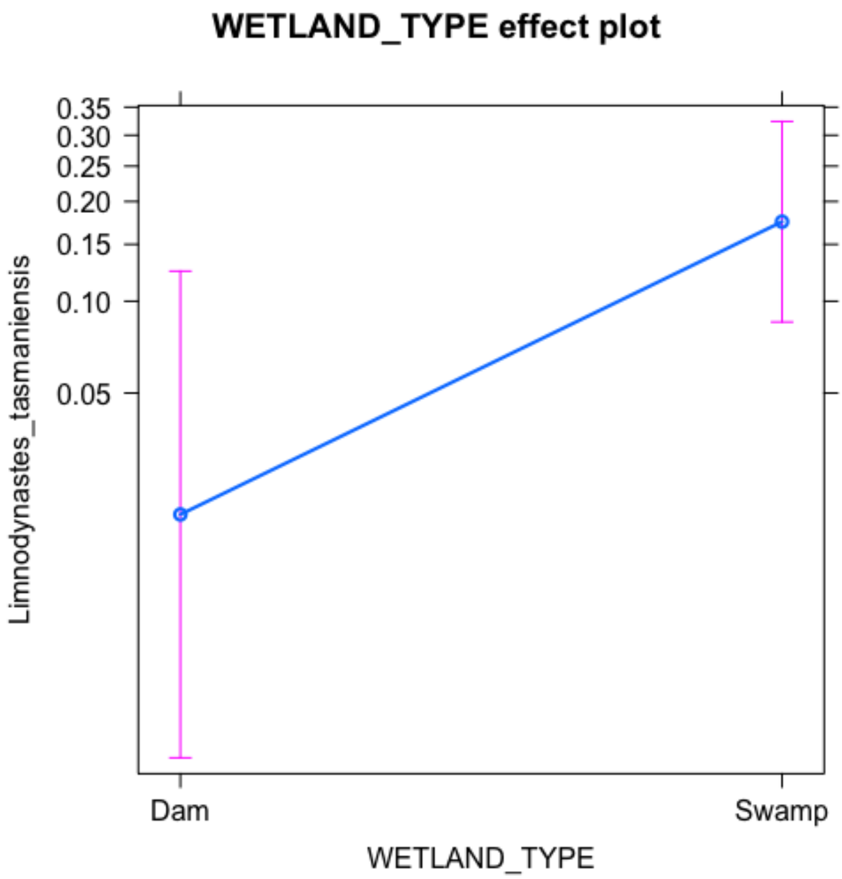


**Figure 2.** The impact of wetland type on the probability of occurrence of *Limnodynastes tasmaniensis.*

### *Effects of grazing*

*Litoria ewingii* was significantly affected by grazing (Appendix, Table 3.). Grazing had a negative influence on the occurrence of *Litoria ewingii*, likelihood of occurrence decreasing from 0.37 down to 0.09 if the swamp surrounds had been grazed (Figure 3).

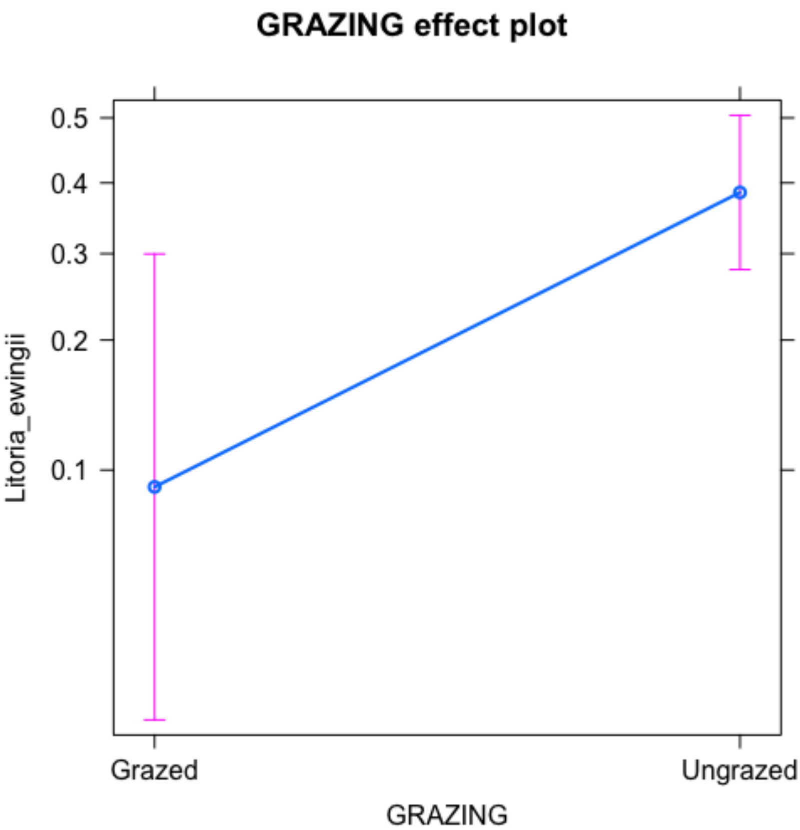


**Figure 3.** The impact of grazing on the probability of occurrence of *Litoria ewingii.*

### *Effects of vegetation type*

Principal components analysis reduced the seven habitat variables to three components that explained 76% of the total variance of the original variables (Table 3).

Vegetation type surrounding swamp sites impacted *Litoria ewingii*. Principal component two displayed a negative association with *Litoria ewingii* occurrence (Figure 4). PC2 was representative of places with more herbs and grasses (Table 3).

**Table 3.** Proportion of variance from PCA loadings

| | Comp. 1 | Comp. 2 | Comp. 3 | Comp. 4 | Comp. 5 | Comp. 6 | Comp. 7 |
|---|---|---|---|---|---|---|---|
| Litter | 0.467 | | 0.559 | | | 0.34 | 0.58 |
| Bryophyte | 0.547 | | 0.412 | | | -0.356 | -0.634 |
| Herb | 0.113 | 0.591 | -0.24 | 0.391 | 0.61 | -0.223 | |
| Grass | | 0.667 | | 0.122 | -0.734 | | |
| Rush | | 0.393 | | -0.892 | 0.208 | | |
| Shrub | 0.479 | -0.209 | -0.446 | -0.184 | -0.191 | -0.529 | 0.421 |
| Bare ground | 0.486 | | -0.509 | | | 0.652 | -0.28 |
| Proportion of variance | 0.354546 | 0.221329 | 0.191591 | 0.130847 | 0.077771 | 0.02133 | 0.002586 |

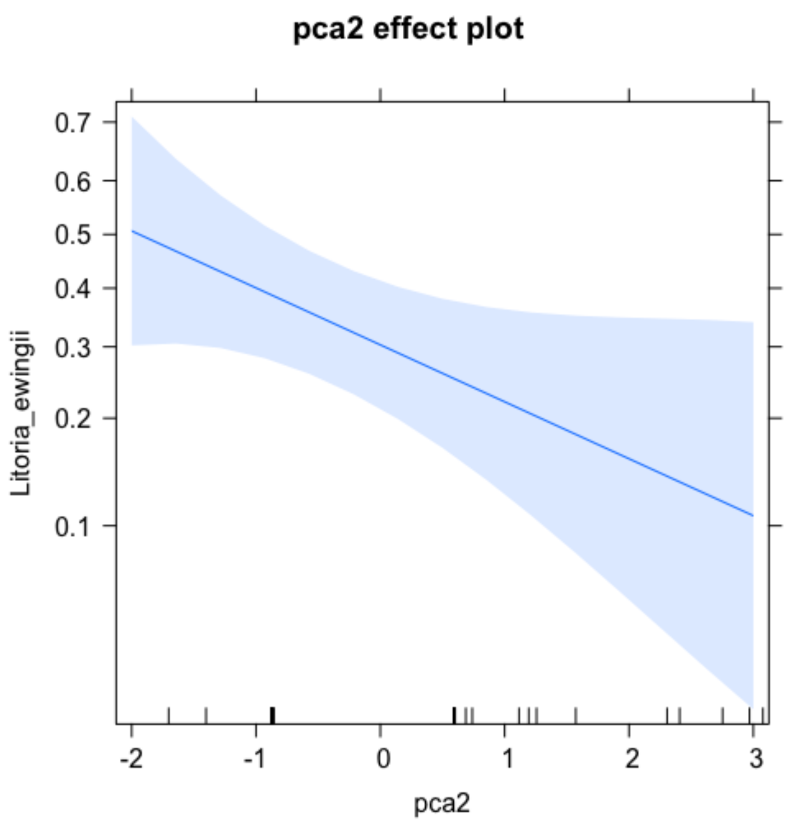


**Figure 4.** The impact of PC2 on the probability of occurrence of *Litoria ewingii.*

### *Effects of swamp cropping, refuge availability and tree availability*

No effects of swamp cropping, refuge availability or tree availability were significant in any species of this study (Appendix, Table 1.).

## DISCUSSION

### *Agricultural intensification*

While the rate of agricultural expansion has decreased in the past few decades, crop yields (output per area) have increased drastically through changes to crop varieties, more effective fertilisers and pesticides, and better technologies for irrigation and machine use (Egli, Meyer, Scherber, Kreft, & Tscharntke, 2018; Matson et al., 1997). These are extraordinary scientific and technological successes however, the impacts of such advances on environmental components have come under scrutiny (Lotze-Campen et al., 2008; Matson et al., 1997). Agricultural intensification can impact species through modifying their physical environment and thereby changing the availability of suitable resources such as breeding and foraging sites, shelter and locomotive paths for dispersal (Matson et al., 1997; Pulsford et al., 2019). These modifications to our natural world are having consequential impacts at local, regional and global scales (Henderson & Loreau, 2018; Matson et al., 1997)

### *The importance of habitat type and structure*

The availability and suitability of breeding habitat is key to a species distribution and therefore its long-term population survival (Hazell et al., 2001; Lauck et al., 2005). This is particularly important for amphibians who require binate environments, aquatic and terrestrial, for different life stages (Cayuela et al., 2015; Lauck et al., 2005). The components of a habitat required by frogs are connected and often several, species specific, coinciding factors are needed around a body of water to make it a suitable breeding site (Lauck et al., 2005; Pulsford, Lindenmayer, & Driscoll, 2017). The terrestrial habitat, as well as wetland features such as aquatic vegetation, heavily influence how frogs use modified landscapes and it is during the metamorphic stage that the connection between these two environments becomes crucial (Burton et al., 2009; Hazell et al., 2001; Pulsford et al., 2017). Vegetation components surrounding the edges of wetlands provides vital protection from predators throughout all life stages: egg-laying, metamorphs with limited mobility and adults vocalising/feeding (Burton et al., 2009; Hazell et al., 2001; Jansen & Healey, 2003; Lauck et al., 2005).

### *Habitat type and structure impacts on Limnodynastes tasmaniensis*

Part of the modification of land for agricultural purposes includes the construction of farm dams, which are deemed as important and often the only wetland habitat available for frogs in agricultural landscapes (Burton et al., 2009; Hazell et al., 2001). Dams, especially those regularly maintained by landowners, generally have a steeper slope, a smaller area of shallow water and less emergent vegetation (Lauck et al., 2005). In this study, *Limnodynastes tasmaniensis* was significantly affected by wetland type, favouring naturally occurring swamplands over constructed dams. This is consistent with other studies that have found *L. tasmaniensis* occurrence to be positively associated with shallow shores (Hamer et al., 2012) and percentage of emergent vegetation (Hazell et al., 2001). Generally considered the productive zone of a swamp, shorelines can provide microhabitats for vocalisation, foraging and egg-laying (Anstis, 2002; Hamer et al., 2012). Shallower water allows solar radiation to reach the swamps substrate that positively affects vegetation growth and produces warmer water temperatures which influences the development of larvae (Lauck et al., 2005; Semlitsch, 2002).

***Litoria ewingii***

Despite being a habitat generalist (Lauck et al., 2005), the occurrence of *Litoria ewingii* was negatively affected by grazing. Grazing alters and removes vital vegetation that provides shelter and habitat for foraging and breeding frogs (Pulsford et al., 2019). Jansen and Healey (2003), found species richness in frogs to be significantly higher in wetlands that had been grazed at low intensity and attributed this to aquatic vegetation and water being of much higher quality in wetlands with less grazing activity. The capture and abundance rates of green frogs (*Rana clamitans*) in Tennessee, USA have been negatively influenced by cattle grazing in wetlands due to a loss of emergent shoreline vegetation (Burton et al., 2009; Woodford & Meyer, 2003). Wallace (2018), found that *Litoria ewingii* had a negative association with cropped swamps. Whilst this study did not find any significant results between frogs and cropping, it does show this species is affected by vegetation changes and prefer taller vegetation (Pulsford et al., 2019).

Vegetation type also had an impact on the occurrence of this species as they preferred wetlands with less grass and herb and higher levels of bare ground, shrubs and trees. These favoured wetlands were representative of a more natural wetland opposed to wetlands surrounded by homogenous grazing vegetation. Hamer et al. (2012), found the *L. ewingii* was positively associated with older, more established sites and negatively impacted by dredging as it removed vital aquatic vegetation. This implies they are slower to recolonise wetlands that have experienced some form of modification such as sites that are grazed regularly (Hamer et al., 2012)

***Why did other species not respond?***

We only used species that had been detected at six or more sites for analysis and *C. parinsignifera* was at the lowest range with only six detections. We attribute there being no significant affects found with this species due to a small dataset. *L. ewingii* has been identified as being fairly tolerant of disturbance and is widespread in both urban and agricultural settings (Lauck et al., 2005) however, Wallace (2018) did find that this species was more likely to occur in uncropped swamps. In similar study about amphibian occurrence and vegetation attributes, Littlefair, Nimmo, Ocock, Michael, and Wassens (2021) also found no significant impacts on *Lim. Peronii*. However, Hamer et al. (2012) found that *Lim. peronii* had similar positive influences from shore depth as did *L. tasmaniensis.* This leads us to believe that limitations in this study (small sample size and poor survey effort) have led to there being no significant impact of habitat type and structure on this species. *C signifera* was the most common species in this study with occurrence at over 56% (26 sites). Hamer et al. (2012) found this species occurred at 90% of their study sites. We hypothesize that this species' widespread abundance is the cause for no variable impacts found in this study. Wallace (2018) found that *Lim. Dumerilii* had a negative detectability association with wind. As per limitations in this study, we can attribute some of the reasons for this species having no significant results to insufficient survey effort as the second surveys for each site were done in extremely high winds and no third surveys were completed.

***Refuge availability and cropping***

Refuges provide sheltering points for amphibians to assist in reduced desiccation and predator avoidance however, agricultural practices and their associated activities of clearing land, remove these vital elements from habitats (Semlitsch & Bodie, 2003; Todd, 2007). Amphibians can use anything available to them as refugia, such as, leaf litter, burrows, farm debris, logs and rocks (Keppel et al., 2012; Valdez et al., 2017). A study by Scheffers, Phillips, and Shoo (2014), in the rainforests of the Philippines, found that bird nest ferns positively influenced frog occurrence serving as important refuges for adults and breeding sites. Similarly, Evans et al. (2020), found occurrence of *L. tasmaniensis* in south-east Australia to be higher in sites where course woody debris had been added to act as refuges against desiccation and predation. Given the high level of evaporative water loss shown by amphibians and their reliance on suitable refuges to combat the risk of desiccation, it is surprising that no species in this study were impacted by refuge availability (Evans et al., 2020; Todd, 2007). Refuge values in this study were quite large, ranging from zero to 322, however 63% of sites had no refuges. Therefore, we suggest that the small sample size and insufficient survey efforts in this study led to insignificant results.

Modifications of natural landscapes to harvest crops in a monocultural fashion has led to the loss of amphibian biodiversity across the globe (Cunningham, Young, & Lindenmayer, 2012; Johansson, Sahlsten, Primmer, & Merilä, 2005; Zedler, 2003). Jordan et al. (2016) found a strong connection between amphibian composition and water quality from a highly agricultural setting in Indiana and Ohio, USA. Intensive farming of rice fields in Japan, as well as the abandonment of rice paddy's have negatively affected the Japanese brown frog (*Rana japonica*) by reducing wet areas (Kidera et al., 2018). Rice frog (*Fejervarya* *multistriata*) abundance was positively correlated to the amount of forest found in the agricultural landscapes of the Yangtze River Delta, China (Ben et al., 2020).

This study found no significant effects of cropping on frog occurrence. Surveys were carried out in December of 2022 and January of 2023, following record annual rainfall in Australia due to La Niña, a natural cycle that is part of the El Niño-Southern Oscillation (Royal Meteorological Society, 2022). Changes in rainfall impact most amphibian species due to their reliance on humid environments and water availability for breeding, however, the influence on biodiversity is not easily foreseen (Ficetola & Maiorano, 2016). In a meta-data analysis study by Ficetola and Maiorano (2016), increased periods of rain showed an increase in survival of some species but a decrease in survival for other species. Some frog occurrences have been shown to decrease when temperatures increase and rainfall decreases (Evans et al., 2020). We suggest that higher rainfall may limit otherwise negative impacts of agricultural land modification such as cropping and recommend further research be conducted in this area.

### ***Limitations***

All sites were either on private land or accessed from the roadside. This impacted our site selection as we either needed consent from private landowners or we had to find swamps that could be accessed and properly assessed from the side of the road. Due to these limitations, study sites were often clumped within a small area and may not be an accurate representation of swamps within south-west Victoria. There is the potential of non-independence due to the closeness and non-random selection of sites and occasionally, where there were multiple swamps in close proximity, we had difficulty determining which swamps the frogs were calling from. Whilst we only surveyed one swamp in each clump, and ensured all selected sites were a minimum of 1km apart, mistaking frog calls from an adjacent swamp could present issues. This was a risk for many sites, and we would recommend creating a distance buffer for future surveys. This would ensure that species recorded are in fact in the same swamp where all other variables have been measured.

Due to La Nina, the rainfall this year has been unusually high compared to past years and therefore some swamps may have even merged into one big wetland area which changes not only the depth and width of the wetland but also the vegetation types and structure.

The configuration of cropped sites versus uncropped sites could introduce spatial confounding as uncropped sites were mostly scattered around the outskirts of the study region while cropped sites were all closely located down the centre (Figure 1. A.). However, there were no noticeable differences in soil type and landforms between cropped and uncropped sites and they all received the same amount of rainfall, so we don't believe this was a big issue.

Due to time constraints, we were unable to complete a third site visit which precluded detectability models. Furthermore, the weather during the second site visit to most sites was extremely windy and could have impacted on frogs calling and our ability to hear them. In a similar study, Wallace (2018) found that detection of *C. signifera and Lim. Dumerilii* decreased with higher wind speeds. In a study conducted in Maryland, USA, Royle and Jung (2005) also found that detection decreased in 5 out of ten species as wind speeds increased.

### *Conservation and management implications*

This study showed that grazing, wetland type and vegetation type play an important role in the occurrence of frogs in agricultural landscapes. As the detection of some species was too small to analyse, it is important to identify the true extent of these species to determine their conservation status within this region. Further research is needed to establish stronger relationships between habitat modification and amphibian responses to aid landowners and governing bodies in land management decisions.

### *Conclusion*

In conclusion, this study has shown that some aspects of agricultural intensification can have negative impacts on local frog species already in decline. The structural changes that livestock grazing has on vegetation can impact the occurrence of frogs in farmland areas. Whilst dams are an important component of agriculture and do provide habitat for some frogs, not all species can thrive in these ecosystems. When assessing the connectivity of wetlands across a region, one should be careful about including dams in their assessment as some species may need natural wetlands for breeding, dispersal and therefore, survival. Agricultural practices will only increase in future years, leading to less suitable habitat for frog species in these settings. The human need to feed is fast growing but with sustainable agricultural practices, research, and monitoring, we can achieve a balance between humankind and nature.

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**APPENDIX**

**Table 1.** Summary of GLM for each species vs. cropping

| Species | | Estimated | Standard error | Z-value | P-value |
|---|---|---|---|---|---|
| Limnodynastes dumerilii | intercept | -0.2624 | 0.2974 | -0.882 | 0.378000 |
| | CROPPED | -0.3662 | 0.4293 | -0.853 | 0.394000 |
| Crinia signifera | intercept | -0.1744 | 0.2960 | -0.589 | 0.556000 |
| | CROPPED | -0.0880 | 0.4196 | -0.210 | 0.834000 |
| Limnodynastes peronii | intercept | -0.8267 | 0.3204 | -2.580 | 0.009880 |
| | CROPPED | -0.2148 | 0.4641 | -0.463 | 0.643550 |
| Litoria ewingii | intercept | -1.0415 | 0.3358 | -3.102 | 0.001920 |
| | CROPPED | 0.5074 | 0.4539 | 1.118 | 0.263680 |
| Limnodynastes tasmaniensis | intercept | -2.3510 | 0.5233 | -4.494 | 0.000007 |
| | CROPPED | 0.0000 | 0.7400 | 0.000 | 1.000000 |
| Crinia parinsignifera | intercept | -1.8971 | 0.4378 | -4.333 | 0.000015 |
| | CROPPED | 1.9095 | 1.1018 | -1.733 | 0.083100 |

**Table 2.** Summary of GLM for each species vs. damming

| Species | | Estimated | Standard error | Z-value | P-value |
|---|---|---|---|---|---|
| Limnodynastes dumerilii | intercept | -0.2007 | 0.3178 | -0.631 | 0.528000 |
| | DAM | -0.4353 | 0.4313 | -1.009 | 0.313000 |
| Crinia signifera | intercept | 0.1001 | 0.3166 | 0.316 | 0.752000 |
| | DAM | -0.5701 | 0.426 | -1.338 | 0.181000 |
| Limnodynastes peronii | intercept | -0.619 | 0.3315 | -1.867 | 0.061800 |
| | DAM | -0.5849 | 0.4671 | -1.252 | 0.210500 |
| Litoria ewingii | intercept | -0.5108 | 0.3266 | -1.564 | 0.118000 |
| | DAM | -0.4877 | 0.4521 | -1.079 | 0.281000 |
| Limnodynastes tasmaniensis | intercept | -1.5506 | 0.4161 | -3.726 | 0.000194 |
| | DAM | -2.3812 | 1.0921 | -2.180 | 0.029231 |
| Crinia parinsignifera | intercept | -2.1972 | 0.527 | -4.169 | 0.000031 |
| | DAM | -0.596 | 0.7947 | -0.750 | 0.453000 |

**Table 3.** Summary of GLM for each species vs. grazing

| Species | | Estimated | Standard error | Z-value | P-value |
|---|---|---|---|---|---|
| Limnodynastes dumerilii | intercept | -0.5661 | 0.2441 | -2.319 | 0.020400 |
| | GRAZING | 0.2907 | 0.2649 | 1.097 | 0.272500 |
| Crinia signifera | intercept | -0.3066 | 0.2376 | -1.290 | 0.197000 |
| | GRAZING | 0.2118 | 0.2636 | 0.803 | 0.422000 |
| Limnodynastes peronii | intercept | -0.94068 | 0.26131 | -3.600 | 0.000318 |
| | GRAZING | 0.02193 | 0.28902 | 0.076 | 0.939522 |
| Litoria ewingii | intercept | -0.5107 | 0.2447 | -2.087 | 0.036900 |
| | GRAZING | -0.8944 | 0.4353 | -2.055 | 0.039900 |
| Limnodynastes tasmaniensis | intercept | -2.1486 | 0.3883 | -5.533 | 0.000000 |
| | GRAZING | -0.8083 | 0.8192 | -0.987 | 0.324000 |
| Crinia parinsignifera | intercept | -2.1972 | 0.3984 | -5.515 | 0.000000 |
| | GRAZING | -16.8246 | 2896.2805 | -0.006 | 0.995000 |

**Table 4.** Summary of GLM for each species vs. refuges

| Species | | Estimated | Standard error | Z-value | P-value |
|---|---|---|---|---|---|
| Limnodynastes dumerilii | intercept | -0.497678 | 0.234863 | -2.119 | 0.034100 |
| | REFUGES | 0.001701 | 0.002899 | 0.587 | 0.557400 |
| Crinia signifera | intercept | -0.177243 | 0.229178 | -0.773 | 0.439000 |
| | REFUGES | -0.001298 | 0.002965 | -0.438 | 0.661000 |
| Limnodynastes peronii | intercept | -0.954721 | 0.2542243 | -3.755 | 0.000173 |
| | REFUGES | 0.000705 | 0.0031167 | 0.226 | 0.821123 |
| Litoria ewingii | intercept | -0.835307 | 0.247597 | -3.374 | 0.000742 |
| | REFUGES | 0.001769 | 0.002966 | 0.596 | 0.550992 |
| Limnodynastes tasmaniensis | intercept | -2.263139 | 0.395223 | -5.726 | 0.000000 |
| | REFUGES | -0.00344 | 0.006815 | -0.505 | 0.614000 |
| Crinia parinsignifera | intercept | -2.843887 | 0.485281 | -5.860 | 0.000000 |
| | REFUGES | 0.006818 | 0.003769 | 1.809 | 0.070500 |

**Table 5.** Summary of GLM for each species vs. tree count

| Species | | Estimated | Standard error | Z-value | P-value |
|---|---|---|---|---|---|
| Limnodynastes dumerilii | intercept | -0.494203 | 0.241618 | -2.045 | 0.040800 |
| | TREES | 0.002995 | 0.006333 | 0.473 | 0.636300 |
| Crinia signifera | intercept | -0.090659 | 0.23534 | -0.385 | 0.700000 |
| | TREES | -0.007787 | 0.006764 | -1.151 | 0.250000 |
| Limnodynastes peronii | intercept | -0.913616 | 0.259842 | -3.516 | 0.000438 |
| | TREES | -0.001061 | 0.007076 | -0.150 | 0.880853 |
| Litoria ewingii | intercept | -0.859024 | 0.255882 | -3.357 | 0.000788 |
| | TREES | 0.004614 | 0.006482 | 0.712 | 0.476613 |
| Limnodynastes tasmaniensis | intercept | -2.37183 | 0.419284 | -5.657 | 0.000000 |
| | TREES | 0.001155 | 0.010802 | 0.107 | 0.915000 |
| Crinia parinsignifera | intercept | -2.763095 | 0.487484 | -5.668 | 0.000000 |
| | TREES | 0.011411 | 0.009373 | 1.217 | 0.223000 |

**Tables 6.** Summary of principal components extracted from the PCA.

| Species | | Estimated | Standard error | Z-value | P-value |
|---|---|---|---|---|---|
| *Limnodynastes dumerilii* | intercept | -0.4436 | 0.2144 | -2.069 | 0.038600 |
| | PCA 1 | 0.1063 | 0.1335 | 0.796 | 0.426100 |
| *Limnodynastes dumerilii* | intercept | -0.4449 | 0.2146 | -2.073 | 0.038100 |
| | PCA 2 | 0.1434 | 0.1703 | 0.842 | 0.399700 |
| *Limnodynastes dumerilii* | intercept | -0.4522 | 0.2177 | -2.077 | 0.037800 |
| | PCA 3 | 0.4076 | 0.2557 | 1.594 | 0.110900 |
| *Crinia signifera* | intercept | -0.2189 | 0.2100 | -1.042 | 0.297000 |
| | PCA 1 | -0.0568 | 0.1373 | -0.414 | 0.679000 |
| *Crinia signifera* | intercept | -0.2212 | 0.2125 | -1.041 | 0.298000 |
| | PCA 2 | 0.2604 | 0.1720 | 1.514 | 0.130000 |
| *Crinia signifera* | intercept | -0.2201 | 0.2148 | -1.025 | 0.305400 |
| | PCA 3 | 0.5095 | 0.2966 | 1.718 | 0.085800 |
| *Limnodynastes peronii* | intercept | -0.9795 | 0.2463 | -3.978 | 0.000070 |
| | PCA 1 | -0.2974 | 0.2627 | -1.132 | 0.258000 |
| *Limnodynastes peronii* | intercept | -0.9316 | 0.2316 | -4.023 | 0.000057 |
| | PCA 2 | -0.0073 | 0.1864 | -0.039 | 0.969000 |
| *Limnodynastes peronii* | intercept | -0.9331 | 0.2319 | -4.024 | 0.000057 |
| | PCA 3 | -0.0718 | 0.2061 | -0.348 | 0.728000 |
| *Litoria ewingii* | intercept | -0.8027 | 0.2313 | -3.471 | 0.000519 |
| | PCA 1 | -0.2276 | 0.2103 | -1.082 | 0.279094 |
| *Litoria ewingii* | intercept | -0.8329 | 0.2370 | -3.515 | 0.000440 |
| | PCA 2 | -0.4296 | 0.2169 | -1.981 | 0.047640 |
| *Litoria ewingii* | intercept | -0.8070 | 0.2313 | -3.489 | 0.000485 |
| | PCA 3 | 0.5269 | 0.3045 | 1.731 | 0.083498 |
| *Limnodynastes tasmaniensis* | intercept | -2.3675 | 0.3747 | -6.319 | 0.000000 |
| | PCA 1 | 0.1180 | 0.1855 | 0.636 | 0.525000 |
| *Limnodynastes tasmaniensis* | intercept | -2.3522 | 0.3703 | -6.353 | 0.000000 |
| | PCA 2 | 0.0351 | 0.2921 | 0.120 | 0.904000 |
| *Limnodynastes tasmaniensis* | intercept | -2.4715 | 0.3977 | -6.214 | 0.000000 |
| | PCA 3 | 0.4658 | 0.2492 | 1.869 | 0.061600 |
| *Crinia parinsignifera* | intercept | -2.5750 | 0.4504 | -5.717 | 0.000000 |
| | PCA 1 | -0.3390 | 0.5940 | -0.571 | 0.568000 |
| *Crinia parinsignifera* | intercept | -2.5334 | 0.4057 | -6.245 | 0.000000 |
| | PCA 2 | 0.2298 | 0.2873 | 0.800 | 0.424000 |
| *Crinia parinsignifera* | intercept | -2.4968 | 0.3932 | -6.350 | 0.000000 |
| | PCA 3 | 0.0133 | 0.3365 | 0.039 | 0.969000 |