

Gwydir River Selected Area

2021-22 Annual Summary Report

FLOW | Monitoring
Evaluation
Research



Appendix I – The effect of environmental watering on frog activity and chorusing behaviour in the Gwydir wetlands, Australia

Summary

- This project was funded as part of a master's degree through the University of New England. The study combined long-term visual survey data of frog diversity collected by NSW DPIE-EES staff from 2015-19 in the Gwydir Wetlands and acoustic recorder observations undertaken in 2019-20 aimed at monitoring the frog chorus before and after the arrival of river flows.
- A diverse frog community was documented including 12 species of frog, one of which (Knife-footed frog (*Cyclorana cultripes*)) had not been previously reported in the Gwydir Wetlands.
- Floodplain inundation from environmental watering was beneficial for frog populations occurring in riverine wetlands, especially during low rainfall and dry periods.
- Decreasing wetland inundation due to river regulation or absence of natural flooding from rainfall may further contribute to declination of frog density or diversity in floodplain wetlands.
- Passive, long duration overnight acoustic recording of frog community revealed complex chorusing behaviour.
- Frogs can be a potential surrogate species to help understand inundation requirements to optimise ecological outcomes for conservation of riverine floodplains.

The following is a draft manuscript that has been submitted to Ecological Indicators Conservation journal in early 2022.

The effect of river flows including environmental watering on frog populations and chorusing behaviour in the Gwydir wetlands, Australia

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Highlights

- Floodplain inundation from river flows, including environmental water delivery, was beneficial for frog populations occurring in floodplain wetlands, associated with higher abundance and species richness, especially during low river inflows and dry period
- Overnight continuous passive acoustic recordings of frog communities revealed complex chorusing behaviour in response to the arrival of flows
- Based on our findings, frogs are potentially a good indicator taxon for better understating inundation requirements and for guiding conservation and management of riverine floodplains

Abstract

Globally, river regulation has degraded wetlands and continues to threaten sensitive biota. The Murray-Darling Basin, Australia's most economically significant drainage basin, has declined in condition because of heavy regulation. We tested how wetland inundation affected the frog community in a catchment of the northern Murray-Darling Basin. We surveyed frogs between 2015 and 2019 to determine long-term changes in the community composition with floodplain inundation. Additionally, we recorded nightly soundscapes for four days before and after the arrival of river flows between 2019 and 2020. The abundance and richness of frog species increased during larger inundation events and altered the community composition (beta diversity). Warmer temperatures increased the detection of frog species, and frog community dominance decreased with decreasing vegetation cover. The abundance of five frog species (*Limnodynastes tasmaniensis*, *Limnodynastes fletcheri*, *Crinia parinsignifera*, *Litoria peronii*, and *Litoria latopalmata*) was higher during larger inundation events. The total species richness of chorusing frogs increased after the arrival of flows, but the responses varied among species and sites. After inundation, the choruses of *Limnodynastes tasmaniensis* increased whereas *Limnodynastes fletcheri* decreased. Our findings indicate inundation as a key driver of breeding for these species and suggest complex overnight chorusing behaviour in response to river flows inundating floodplain wetlands.

Keywords: River regulation; Environmental watering; Floodplain wetlands; Frogs; Long term survey; Passive acoustic monitoring

1 Introduction

River flows drive the community composition of floodplain biota by connecting river and floodplain wetlands (Kuiper et al., 2014; Thomaz et al., 2007), while river regulation impedes wetland connectivity and reduces biodiversity (Frazier & Page, 2006; Kingsford, 2000a; Nilsson & Berggren, 2000). Environmental watering is one useful strategy to restore connectivity and inundation in floodplain wetlands, and measuring its long-term success is important for optimizing its delivery (Arthington et al., 2018; Dube et al., 2015). Understanding responses of aquatic taxa, such as frogs, to inundation is crucial for understanding ecological outcomes for water management and wetland conservation (Holgerson et al., 2019; McGinness et al., 2014).

In floodplain wetlands, various biotic and abiotic factors influence how frogs respond to inundation, for example rainfall, temperature, and season in combination with wetland inundation, influence frog populations (Ocock et al., 2014, 2016; Wassens & Maher, 2011). Such relationships vary between species, for example, some species only call in warmer temperatures (e.g., *Litoria peronii* above 20 °C), whereas others can call in inundated habitat year-round, regardless of temperature e.g., *Crinia parinsignifera* (Amos, 2017; Spencer & Wassens, 2009). Additionally, some species only respond to watering events that coincide with their breeding season (Jakob et al., 2003; McGinness et al., 2014). For example, summer breeders such as *Litoria peronii* call only in summer, even if habitat is otherwise available (Wassens et al., 2009). Due to favourable habitat conditions in periodically inundated sites, frog richness and abundance is often higher in temporary wetlands compared to permanent sites (Hoffmann, 2018; Ocock et al., 2016; Porej & Hetherington, 2005), which can influence the beta diversity (comparisons of community composition between communities) (Davis et al., 2017; Henning & Schirato, 2006). Variability of inundation during flood pulses can also create differential habitats and influence frog richness, species composition, and beta diversity (Ramalho et al., 2018).

Little is known about factors that limit frog diversity, community composition, and chorusing activities in the northern Murray-Darling Basin, despite the regions' conservation significance and the heavy impact of water regulation and land modification (Southwell et al., 2014; Wilson et al., 2009). Most studies have been limited to examining short-term (i.e., seasonal and annual) variation in frog communities (Brodie et al., 2020a; Henning & Schirato, 2006; McGinness et al., 2014; Moreira et al., 2017; Ocock et al., 2016), and longer-term trends are still poorly understood, especially the immediate calling response of frog communities to wetting and drying cycles.

The aims of our study were to: 1) examine variation in frog abundance, richness, dominance (the inverse of species evenness), and beta diversity (comparisons of community composition between communities) over a five-year period and look for associations with wetland inundation, temperature, rainfall, and vegetation cover and; 2) use passive acoustic monitoring to compare the variation in frog chorusing activity before and after river flows inundated sites following two different flow events from environmental water release.

2 Methods

2.1 Study sites

Our study sites were located in the Gwydir River catchment of the northern Murray-Darling Basin, downstream and west of Moree, New South Wales (NSW), Australia (Fig 1). The survey area included six sites in Central Gingham: Bunnor Bird Hide (BNNB), Bunnor floodplain (BNNF), Gingham Waterhole (GINW), Little Lagoon (LITL), Lynworth Floodplain (LYNF), and Munwonga Wetland (MUNW); three sites in Lower Gingham: Ardblair Waterhole (previously known as Boyanga Waterhole, BOYW), Gingham Bridge (GINB), and Old Boyanga Wetland (OLDB); three sites in Lower Gwydir: Allambie Bridge (ALLA), Old Dromana floodplain (OLDF), and Old Dromana Ramsar (OLDR); two sites in Mallowa watercourses: Bungunya Wetland (BUNG) and Valetta Wetland (VALE); and two sites in Mehi River system: Derra Waterhole (DERW) and Whittaker's Lagoon (WHIT) (Fig 1). Habitats surveyed were permanent and temporary waterbodies including shallow open waterholes, in-stream channels, and lagoons, mixed with marshes and woodland (Walcott et al., 2020).

2.1.1 Five-year frog surveys

From 2015–2019, acoustic and visual encounter surveys (VES) were completed in September and November of each year at 16 sites across the Gwydir River catchment with additional surveys in February 2017 and March 2019 at a subset of sites (Fig 1). Timed VES were conducted by two experienced surveyors for a total of one person-hour followed by a five-minute listening survey. VES surveys commenced at least 30 minutes after sunset using spotlights and each individual was counted and identified including individuals that were heard (but not seen) in close proximity to the surveyor (within a radius of approximately one metre). For each species of frog calling during the timed-audio surveys the number heard calling was estimated using the following categories: no calling, rare (1–5 individuals), common (6–10 individuals), abundant (11–20 individuals), or very abundant (≥ 20 individuals). VES and auditory data were combined for analyses. For abundance, evenness, and beta diversity, the midpoints of each audio category were used (e.g., if 12 individuals were seen and 1–5 were heard calling, the total number was recorded as 15).

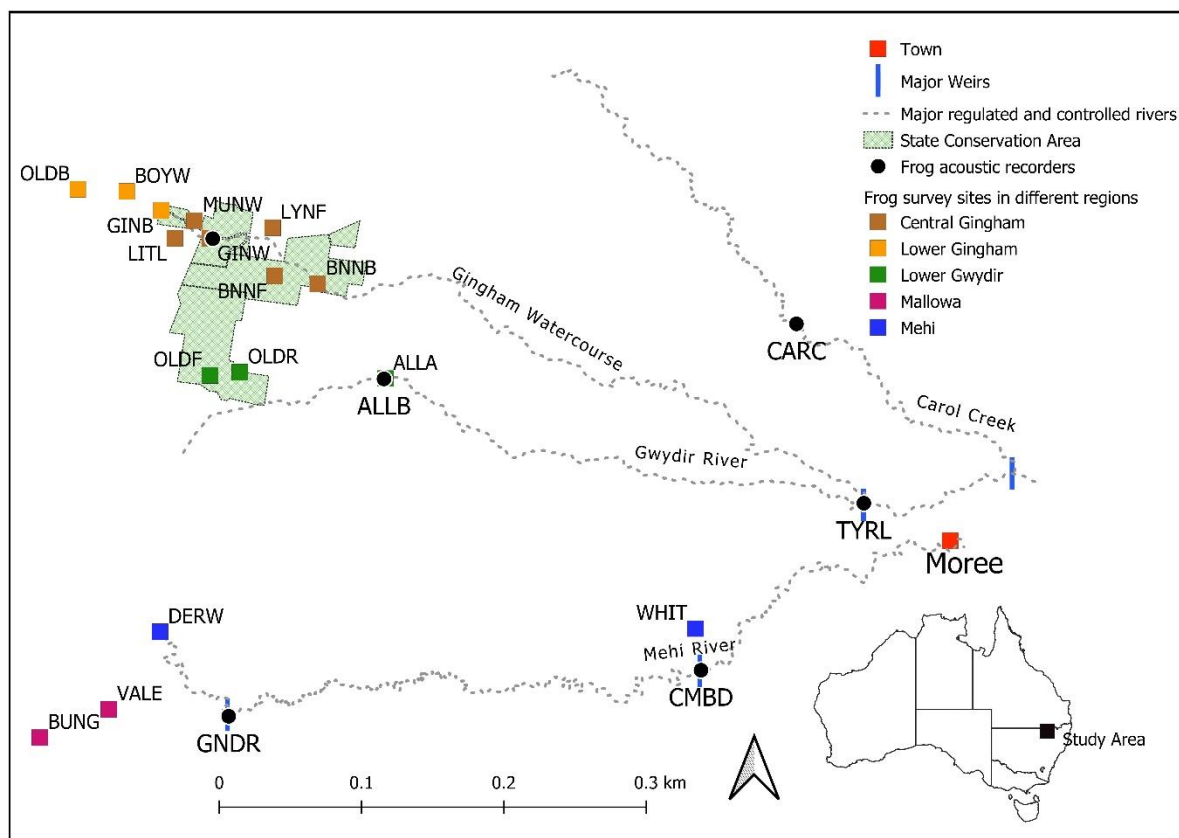


Fig. 1. Map showing locations of the 16 frog survey sites and six acoustic data collection sites. See section 2.1 (study sites) and Table 2 for abbreviation used for sites.

2.1.2 Acoustic recordings

To examine frog calling responses to river flows (including environmental water delivery) that inundated sites during severe drought in 2019–2020, we recorded soundscapes in six sites across the Gwydir River catchment for four nights immediately before and after flow events (Fig 1, Table 1; one flow event per site). The sites were either dry (e.g., Allambie Bridge) or only held shallow water (e.g., Gundare) unless filled with river flow or environmental water released from Copeton Dam, upstream of the Gwydir River. We deployed a Bioacoustic Audio Recorder (BAR) (Firmware: V2.99, FRONTIER LABS) at each site to record choruses continuously from 1700 h to 0700 h with sample rate of 44.1 kHz in stereo WAV file format, resulting in 14 hours of recordings per night. Acoustic recordings were made at five sites in October–November 2019 and one site in February–March 2020, with one night of recording missed at the Gingham site (fourth night after flow) due to equipment failure (Table 1).

Table 1. Key site characteristics, acoustic recording dates, and rainfall recorded 24 hours before flow arrival, minimum daily temperature, and general site hydrology (Temporary-T, Permanent-P).

Sites	Type	Total rainfall with 24 hours (in mm)		Mean minimum temperature (°C)		Soundscape recording dates	
		<i>Before flow</i>	<i>After flow</i>	<i>Before flow</i>	<i>After flow</i>	<i>Before flow</i>	<i>After flow</i>
<i>Allambie Bridge (ALLB)</i>	<i>River channel (T)</i>	0	0	11.4	13.0	7-10 Nov 2019	11-14 Nov 2019
<i>Carol Creek (CARC)</i>	<i>Creek (P)</i>	0	4.6	18.8	14.4	30 Oct to 2 Nov 2019	3-6 Nov 2019
<i>Combadello (CMBD)</i>	<i>In-stream weir (P)</i>	0	4.6	18.8	14.4	30 Oct to 2 Nov 2019	3-6 Nov 2019
<i>Gingham Waterhole (GINW)</i>	<i>Floodplain wetlands (T)</i>	35.5	0	19.4	17.6	24-27 Feb 2020	28 Feb - 1 Mar 2020*
<i>Gundare (GNDR)</i>	<i>In-stream weir (P)</i>	0	0	13.3	15.1	13-16 Nov 2019	17-20 Nov 2019
<i>Tyreel (TYRL)</i>	<i>In-stream weir (P)</i>	0	4.6	18.8	14.4	30-31 Oct, 1-2 Nov 2019	3-6 Nov 2019

*Sites with 7 nights of data (one night missed due to equipment failure).

2.2 Environmental variables

2.2.1 Five-year frog surveys

Ambient air temperature was recorded at the beginning of the survey using a Kestrel pocket weather meter (Model: 3500 Delta T). The percent of each wetland that was inundated at the time of each survey (hereafter “inundated area”) was estimated by visually assessing each site and comparing the observed inundated area with wetland delineation maps of the sites. Extensive inundation in the central Gingham watercourse during the September 2018 surveys resulted in water extending beyond the defined survey areas at two sites. Rainfall data were derived from the Australian Bureau of Meteorology (BoM, 2020) website using the nearest weather station within a 12–70 km radius of each survey site. Using these data, cumulative rainfall was calculated over four periods prior to each survey (rain in past 24, 48, 72, and 96 hours). Several vegetation categories were measured at each site on each survey occasion: emergent vegetation (tall and short), aquatic vegetation (low growing, floating, and submerged), total vegetation (sum of emergent and aquatic vegetation), dry bare ground, terrestrial vegetation cover, and open water (i.e., no vegetation). To determine the percent area covered by each vegetation category, three random 5 x 5 m quadrats were placed at the site. For each quadrat, vegetation category estimates could total more than 100% when combined because layers of different vegetation types could grow above and among one another.

2.2.2 Acoustic recordings

We recorded river flow as a binary category (wet= true or dry= false). Flow nights were determined by using river flow measurements (megalitres/day) from the nearest gauging stations (<https://realtimedata.watersw.com.au/>) and examining sentinel hub images for site inundation (<https://apps.sentinel-hub.com>) following the two environmental watering events. We obtained daily rainfall and minimum temperature from the Australian Bureau of Meteorology (<http://www.bom.gov.au/climate/data/> [Accessed 20 January 2021]) using the nearest weather station for each site (9–43 km).

2.3 Detecting and calculating frog chorus duration

To analyse our acoustic recordings, we used long-duration ‘false-colour’ spectrograms (FCS) to visually detect different frog choruses (Fig 2). FCS are graphical display tools calculated from acoustic indices (Indraswari et al., 2020; Pieretti et al., 2011) to visualize long duration audio recordings (up to 24 h), compressed into a single spectrogram and viewable as a single image (Towsey et al., 2014). Acoustic indices are numeric summaries of the energy distribution in a recording based on amplitude or spectral content (Gan et al., 2019; Sueur et al., 2014). We used QUT Ecoacoustics Audio Analysis Software v20.11.2.0 (Towsey et al., 2017) to generate the FCSs following methods found in Towsey et al., (2014, 2015). Each frog chorus was identified by the colours, call frequency, and signal shape (Table 2) in the FCS, and further confirmed by listening to the corresponding sections of the recordings.

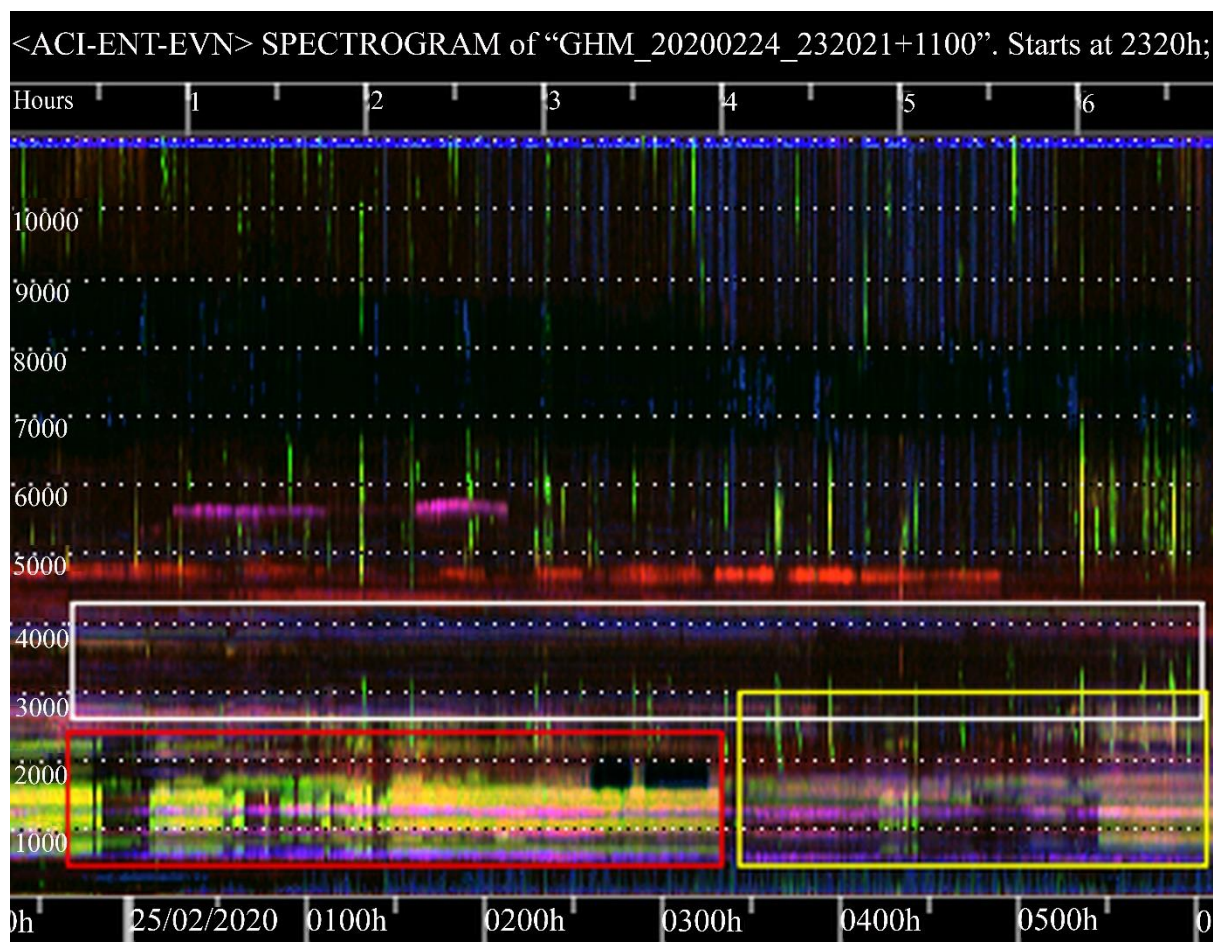


Fig. 2. An example of part of a false-colour spectrogram at Gingham waterhole (abridged for clarity). Time is shown on the x-axis (2320 h–0605 h), and sound frequency on the y-axis (0–11,000 Hz). One pixel represents 1 min and approximately 43 Hz frequency range. Horizontal dotted lines delineate 1000 Hz frequency intervals. Frog chorus identified are: *Crinia parinsignifera* by purple shade marks in the frequency range 2200–4200 Hz (white square) from 0000 h to 0647 h, *Cyclorana alboguttata* by light green marks in the frequency range 500–2400 Hz (red square) from 0000 h to 0700 h, and *Limnodynastes fletcheri* by purple marks in the frequency range 400–3000 Hz (yellow square) from 2320 h to 0700 h. Other sounds identified are insects - red 4000–5000 Hz and bright pink 5000–6000 Hz.

Table 2. Frog species detected during monitoring and the frequency range of typical calls. Mean and standard deviation (SD) of call duration of each species are shown before (BF) and after flow (AF) as well as the number of sites where each species was detected.

Species	Abbr.	Call frequency range (Hz)	Mean and SD of call duration in minutes		No. of sites frogs detected	
			BF	AF	BF	AF
<i>Crinia parinsignifera</i>	CP	2200-4200	303.3(SD±325.4)	571.4(SD±151.1)	1	2
<i>Cyclorana alboguttata</i>	CA	500-2400	381.8(SD±223.4)	0	1	0
<i>Cyclorana cultripes</i>	CC	600-800	0	202.3(SD±350.5)	0	1
<i>Cyclorana verrucosa</i>	CV	200-1900	0	96.8(SD±193.5)	0	1
<i>Limnodynastes fletcheri</i>	LF	400-3000	224.2(SD±255.4)	95.5(SD±187.8)	3	2
<i>Limnodynastes salmini</i>	LS	300-4000	0	283.0(SD±335.9)	0	1
<i>Limnodynastes tasmaniensis</i>	LT	650-3600	132.3(SD±192.5)	283.4(SD±225.4)	3	5
<i>Litoria latopalmata</i>	LL	1300-3700	121.8(SD±186.8)	44.3(SD±118.9)	4	4
<i>Litoria peronii</i>	LP	800-2600	261.6(SD±239.0)	345.4(SD±258.5)	4	4

Our aim was to detect and calculate the duration (in minutes) of chorusing for each frog species per night recorded. Choruses were sometimes interrupted by brief pauses that made it difficult to precisely delineate choruses using the FCS. Therefore, to standardise the chorus calculation methods, we considered five minutes or more of continuous calling by a single species as a chorus and we ignored pauses <20 min if continuous choruses for at least 5 min or more occurred on either side of the pause. Most pauses were substantially shorter than 20 min, and most choruses were longer than 5 min, so this approach should have had minimal impact on the results.

We used the imager package (Barthelme, 2020) and interactive functions in R (Brodie, 2020b) to explore the FCS and identify the frog calling in audio segments. We used Raven Pro 1.6 (<https://ravensoundsoftware.com/>) to measure frog call frequency and Audacity (version 2.4.2, <http://www.audacityteam.org>) to analyse recordings by looking at species-specific spectrogram to calculate chorus minutes.

2.4 Statistical analyses

Statistical analyses were carried out using R- 4.0.1 (R Core Team 2020) for the five-year frog survey data and R- 4.1.0. (R Core Team 2021) for the acoustic data.

2.4.1 Five-year frog surveys

We used mixed effect models to examine the influence of environmental variables on total frog abundance, species richness, and species dominance (the inverse of Shannon's evenness). We assessed species richness and dominance separately because combined alpha diversity measures (e.g., Shannon's diversity index) confound richness and evenness (e.g., a high Shannon's diversity can result from either high richness and low evenness or low richness and high evenness). Different types of models were used for each response variable based on the data distributions. Abundance data were analysed with zero-inflated negative binomial models via glmmTMB package (Brooks et al., 2017), and richness and dominance were analysed with general linear mixed effects models via the lmer function in the lme4 package (Bates et al., 2015). Only surveys where at least two species of frog were documented were included in the dominance models, whereas all surveys were included in the richness models, even if no frogs were documented. Dominance was calculated by subtracting Shannon's evenness from one, and a square root transformation was applied to achieve an acceptable model fit.

For all models, inundated area, temperature, rain, and vegetation were included as the fixed effects, and study site and survey date (nested in season) were included as random intercepts (survey date was defined as each month * year combination, and seasons were: autumn, spring, and summer). For the rain category, several cumulative measurements were calculated (rain in the preceding 24, 48, 72, or 96 hours). Therefore, for each analysis, we determined which measurement of rainfall was most appropriate by constructing a model for each measurement and selecting the model with the lowest AIC as the final model for analysis (REML was set to false for model selection, and true for the final analysis). Similarly, vegetation variables: emergent aquatic, aquatic, total vegetation (emergent and aquatic), dry bare ground, terrestrial vegetation cover, and open water, most of which were correlated because they were estimates of percent cover. To select the most appropriate vegetation variable for each model, we again constructed models using each variable and selected the model with the lowest AIC for the final analysis.

Finally, we tested the effects of the environmental variables on frog community composition (beta diversity measured with Bray-Curtis dissimilarities) using permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001) via the `adonis2` function in the `vegan` package (Oksanen et al., 2020). We used survey month as strata and included study site as the first term in the model. Inundated area, air temperature, rainfall, and vegetation variables were also included. The order of these terms was based on the order of significance (most to least) in the abundance models, while the rainfall and vegetation variables were the same as those selected for the abundance models. All numeric data were transformed to proportions prior to calculating Bray-Curtis dissimilarities (McKnight et al., 2019). Additionally, when sample sizes allowed, a separate abundance model was constructed for each frog species.

2.4.2 Acoustic recordings

We used mixed effects models to test the effect of flow on frog chorusing. First, we ran a model using data from all frog species. For each species, we calculated the mean chorus duration for all nights with and without flow at each site (four nights before flow averaged and four nights after flow averaged) and included chorus duration as the response variable. We included flow, frog species, rainfall in the previous 24 hours (summed across all nights per flow category), and minimum air temperature (averaged across all nights per flow category) as fixed effects (with an interaction between flow and species), and study site as a random effect. Following this model, we ran species specific models for the four species that called in at least three sites (Fig 5): *Limnodynastes fletcheri*, *Limnodynastes tasmaniensis*, *Litoria latopalmata*, and *Litoria peronii*. These models were structured in the same way as the previous model with the exclusion of the species term and interaction. Additionally, rainfall was not included for the *Limnodynastes fletcheri* model because it only rained twice.

We used a similar model to test the effect of flow on frog species richness. We calculated frog species richness across all nights before and all nights after flow for each site. We again included flow, rain, and temperature as fixed effects (calculated as before) with site as a random effect.

We constructed the models in the `lme4` package (version 1.1-27) (Bates et al., 2015) and assessed significance with the `Anova` function in the `car` package (Fox & Weisberg, 2019) using a type II sum of squares. For all models, we checked model assumptions using QQ plots and residual plots. For the chorus duration model using all species, we had to square root transform the chorus duration data to achieve acceptable model fit. All other models used the raw data. For all models, we used a combination of all four nights (flow and no flow) rather than treating each night as a replicate because treating each night as a replicate resulted in models that were too complex to be fit reliably with the small samples sizes available to us.

3 Results

3.1 Five-year frog surveys

A total of 12 species and 6,651 frog encounters (visual or auditory) were recorded. The highest number of encounters was recorded for *Limnodynastes tasmaniensis* ($n = 2,810$) and the lowest for *Cyclorana verrucosa* ($n = 1$, further taxonomic verification required). The most widespread species were *Limnodynastes fletcheri*,

Limnodynastes tasmaniensis, and *Crinia parinsignifera* which were found at all the sites, followed by *Litoria peronii* which was found in 14 (87.5 %) of the sites surveyed. Among years (all surveyed sites combined), observed species richness was highest in 2018 (n = 10) and lowest in 2016 and 2019 (n = 8 for both). The highest frog abundance was recorded in 2017 (n = 2,159), followed by 2016 (n = 1,651) and 2018 (n = 1,386), and lowest was in 2019 (n = 262).

Abundance and richness of frog species were significantly higher in sites with larger inundated areas (both $P < 0.0001$; Table 3, Fig 3A and 3B). Species richness was significantly higher with warmer air temperatures ($P = 0.045$; Table 3, Fig 3D). Dominance (the inverse of species evenness) of frogs was significantly lower in sites with a higher proportion of open water (less vegetation cover) ($P = 0.004$), but it was not affected by inundation, air temperature, or rainfall (all $P > 0.05$) (Table 3, Fig 3C). The frog community composition (beta diversity) differed significantly among sites ($P = 0.0002$) and inundation, temperature, and vegetation significantly affected frog community composition (all $P < 0.05$), but rainfall did not significantly affect community composition ($P = 0.15$) (Table 3). Abundance of *Limnodynastes tasmaniensis*, *Limnodynastes fletcheri*, *Crinia parinsignifera*, *Litoria peronii*, and *Litoria latopalmata* were significantly higher in sites with greater inundated areas (all $P < 0.05$; Table 3, Fig 4).

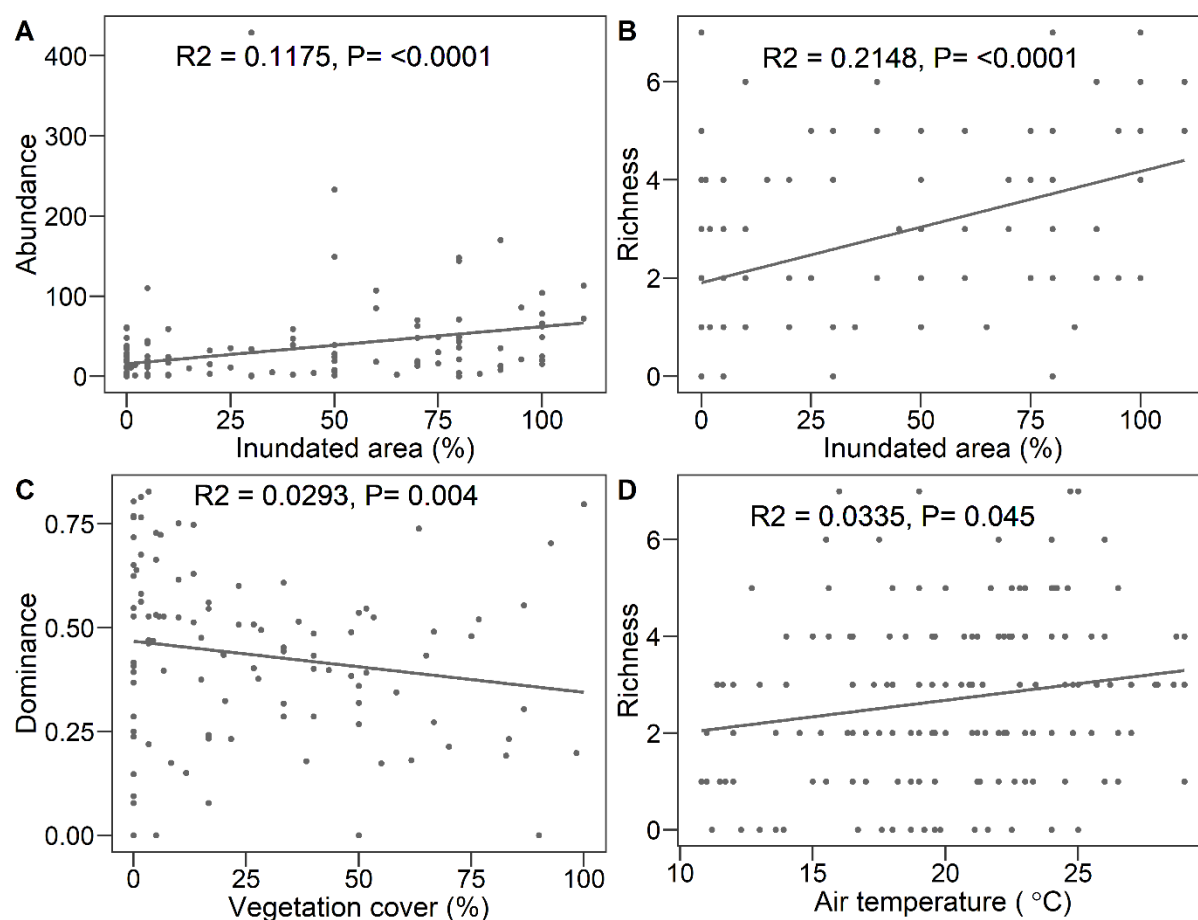


Fig. 3. Effect of inundated areas on frog (A) abundance (all species combined), (B) richness, (C) effect of vegetation cover on frog dominance (for this model, the "open water" variable was the habitat variable with the best fit; higher proportions of open water indicated a lack of vegetation within the waterbody, and (D) effect of air temperature on frog richness. Each point represents an individual survey.

Table 3. Fixed terms in the final models and associated p-values for abundance, richness, dominance, beta diversity (PERMANOVA), and single-species abundance.

	Inundated area	Air temperature	Rainfall	Vegetation cover
Abundance	<0.001	0.510	0.122	0.500
Richness	<0.001	0.045	0.432	0.872
Dominance	0.097	0.072	0.855	0.004*
Beta diversity	0.003	0.024	0.150	0.019
<i>Limnodynastes tasmaniensis</i>	<0.001	0.365	0.311	0.992
<i>Limnodynastes fletcheri</i>	0.0004	0.304	0.298	0.103
<i>Crinia parinsignifera</i>	<0.001	0.669	0.996	0.771
<i>Litoria peronii</i>	0.0003	0.099	0.175	0.103
	0.041	0.053	0.104	0.652

*For this model, "open water" (i.e., areas of water with no vegetation) had negative association and was selected by lowest AIC indicating the best fit. Thus, this negative influence suggests a positive association with vegetation.

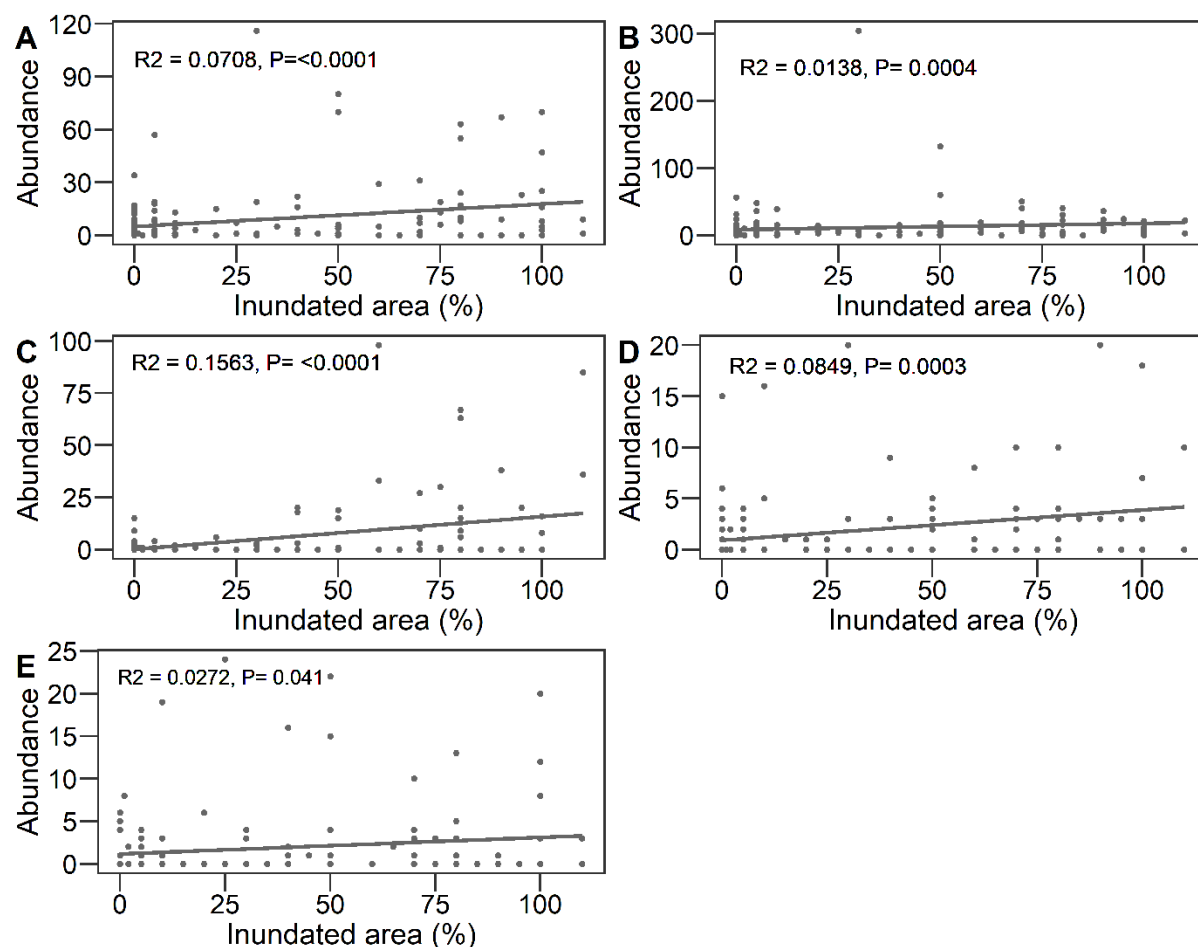


Fig. 4. Effect of inundated areas on abundance of (A) *Limnodynastes tasmaniensis*, (B) *Limnodynastes fletcheri*, (C) *Crinia parinsignifera*, (D) *Litoria peronii*, and (E) *Litoria latopalmata*.

3.2 Acoustic recordings

During the acoustic monitoring, a total of nine frog species were detected (Table 2) with highest richness ($n = 7$) at the Gingham waterhole and lowest ($n = 2$) at Allambie Bridge (Fig 5B). There was a small but statistically significant increase in richness following river flows ($\chi^2 = 14.3$, $P < 0.001$; Fig 5B). Richness also increased with increasing temperature ($\chi^2 = 7.2$, $P = 0.007$), but was not affected by rainfall ($\chi^2 = 2.7$, $P = 0.103$).

In the call duration model with all nine species, there were significant main effects of species ($\chi^2 = 16.9$, $P = 0.031$), but the main effects of flow ($\chi^2 = 2.6$, $P = 0.107$), temperature ($\chi^2 = 1.1$, $P = 0.298$), and rain ($\chi^2 = 0.005$, $P = 0.945$) were not significant. However, the interaction between flow and species was significant ($\chi^2 = 18.8$, $P = 0.016$), indicating that the arrival of water had different effects on different species (Fig 5A, Table 2). Subsequent species-specific models showed that call duration increased significantly with flow for *Limnodynastes tasmaniensis* ($\chi^2 = 41.9$, $P < 0.001$), decreased significantly for *Limnodynastes fletcheri* ($\chi^2 = 25.0$, $P < 0.001$), and was not significantly affected for *Litoria latopalmata* ($\chi^2 = 0.1$, $P = 0.770$) and *Litoria peronii* ($\chi^2 = 0.3$, $P = 0.583$). Temperature and chorus duration had a significant negative relationship for *Limnodynastes fletcheri* ($\chi^2 = 9.47$, $P = 0.002$), *Limnodynastes tasmaniensis* ($\chi^2 = 5.5$, $P = 0.021$), and *Litoria latopalmata* ($\chi^2 = 5.6$, $P = 0.018$).

Rainfall was not significant in any model (all $P > 0.143$), but only a few nights of rain occurred during the study, resulting in limited power.

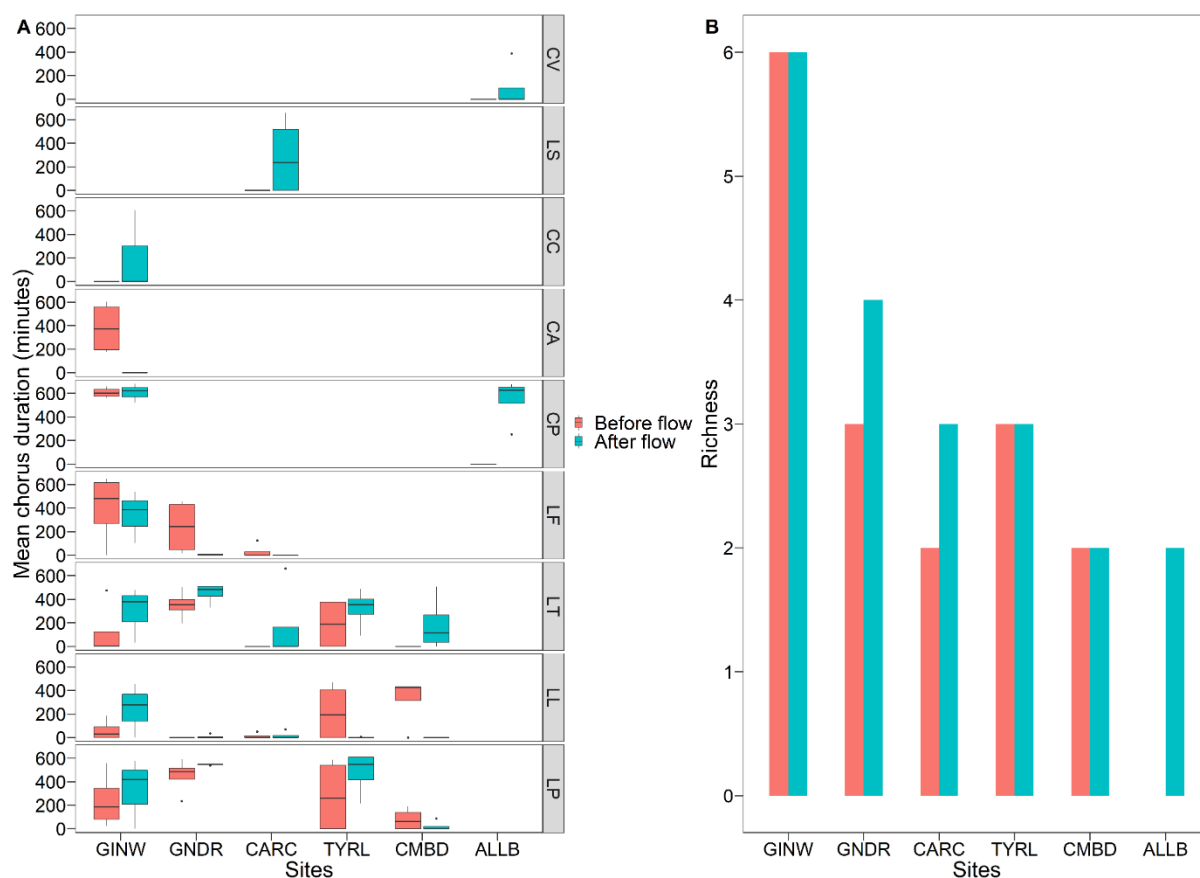


Fig. 5. (A) Mean chorus variation of the nine frog species detected at the six sites before and after river flows inundated the wetland sites. Each data point is the total chorus duration of each species for a given night at a given site before and after flow. See Table 2 for abbreviation used for species names and sites. Arrival of flow significantly affected two species (negatively for LF and positively for LT). (B) Bar plot showing frog richness before and after flow ($\chi^2 = 14.3$, $P < 0.001$). No calling was detected at the Allambie Bridge site before flow.

4 Discussion

Wetland inundation had multiple effects on the frog populations occurring in the floodplain habitats. At the broadest scale, community composition (beta diversity — comparisons of community composition between communities) was significantly affected by wetland inundation and community dominance (the inverse of species evenness) decreased significantly with an increasing proportion of open water (no vegetation) over the course of the five-year survey. Frog occupancy and community composition across inundated floodplain habitat are not consistent, because different species with their different physiological and behavioural traits showed broadly different responses to variety of environmental factors (Deoniziak et al., 2017; Eskew et al., 2012; Lytle, 2001; Ocock et al., 2016). Therefore, frogs in floodplain habitats rely to a great extent of micro habitats created by inundation for breeding and foraging which may influence frog community structure (Bateman et al., 2008; Henning and Schirato, 2006; Indermaur et al., 2010; McGinness et al., 2014; Tockner et al., 2006). Significant results of community composition (beta diversity patterns) and dominance

with open water model indicate that the broad habitat plays an interactive role in structuring frog communities in the Gwydir Wetlands.

Further, consistent with previous studies inundated areas create more favourable habitats for frogs, we found that the total frog abundance was significantly higher in inundated sites (Babbitt & Tanner, 2000; Mac Nally et al., 2014; Ocock et al., 2016). More specifically, the abundance of *Limnodynastes tasmaniensis*, *Limnodynastes fletcheri*, *Crinia parinsignifera*, *Litoria peronii*, and *Litoria latopalmata* was higher in inundated areas, which is consistent with previous studies in the middle and southern parts of the Murray-Darling Basin (Amos, 2017; McGinness et al., 2014; Ocock et al., 2016; Wassens & Maher, 2011). Though previous studies reported positive association of these five species with vegetation, rainfall, and temperature (Hoffmann, 2018; Jansen & Healey, 2003; Ocock et al., 2016; Wassens et al., 2012), we did not detect those effects at the species-specific level. However, this may have been partially due to reduced sample sizes within species, and some of our community-level models did detect significant effects of those factors (though less consistently than inundated area).

River flows inundating sites revealed the complex chorusing patterns of the frog acoustic community. Consecutive overnight chorusing varied among species and among sites upon arrival of river flows. This may reflect differential behavioural responses of different species among habitats or could be due to limited spatial sampling at our sites. The increase in chorusing duration in *Limnodynastes tasmaniensis* after inundation from river flows is consistent with previous findings (Heard et al., 2015; McGinness et al., 2014). The negative effect of flow on *Limnodynastes fletcheri* was surprising and one possible explanation is that *Limnodynastes fletcheri* has good dispersal ability and calling males can move to nearby low-laying vegetated waterbodies which allowed them to disperse beyond the recorders during inundation (Bishop-Taylor et al., 2015; Ocock et al., 2014). Future acoustic sampling would benefit from paired trail cameras with acoustic recorders to allow researchers to gauge the specific spatial layout of water at the site.

Both higher inundation over the five-year period and inundation immediately after river flows increased frog richness, which is consistent with previous studies (Hoffmann, 2018; Mac Nally et al., 2014; McGinness et al., 2014) and indicates that higher inundation in wetland creates favourable breeding habitats that supports higher frog richness.

Over the five-year period, in addition to examining the effect of environmental watering, we documented diverse frog communities, including twelve species, two of which (*Cyclorana cultripes* and *C. verrucosa*) remain taxonomically indistinct (Southwell et al., 2014; Wilson et al., 2009). However, some burrowing species had been previously recorded from this region were not detected during our study (e.g., *Neobatrachus sudelli*, *Notaden bennetti*, *Uperoleia rugosa*, and *Platyplectrum ornatum*) (Southwell et al., 2014; Wilson et al., 2009). This may be a result of low rainfall at the time of the five-year surveys (cumulative rainfall of past four days from frog survey dates: 2.2 mm in 2018, 34.1 mm in 2017, 25.4 mm in 2016, 10 mm in 2015) as burrowing species are responsive to heavy rainfall (Ocock et al., 2016; Read, 1999) and this demonstrates the importance of long-term sampling through different environmental cycles for detection of the full suite of resident species. Additionally, some of the missing species are explosive breeders that are only active for a very brief period and are, therefore, easy to miss during surveys.

5 Conclusion

Our results suggest an overall positive effect of wetland inundation on frog populations in terms of abundance and species richness. This has important implications for legislative revisions of the water management plan to include provisions specifically for amphibians and provides insight into the inundation requirements of frog populations occurring in floodplain wetlands which is crucial to mitigate the negative impact of river regulation on this taxon. Our results from acoustic monitoring were hindered by low sample size and an inability to assess the movement of frogs in relation to the recorders, but these trends will become more clearer with advancement of automatic detection of frog calls.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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