

Effects of traffic noise on vocalisations of the rhacophorid tree frog *Kurixalus chaseni* (Anura: Rhacophoridae) in Borneo

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Abstract. Transportation networks are currently growing at rapid rates, and the increase in roads can have detrimental effects on biodiversity. Increases in anthropogenic noise have been demonstrated to negatively impact several types of behaviour across taxa, as well as to mask key elements of vocalisations required for communication. While much information has been collected for species in temperate regions, fewer data are available for Southeast (SE) Asian species. Given that SE Asia has the highest rate of deforestation, which is the largest driver of road expansion, more data are needed on the impacts of traffic noise on SE Asian species. To that end, we exposed calling tree frogs (*Kurixalus chaseni*) to traffic noise to determine the impacts of two levels (low and high) of anthropogenic noise on dominant frequency, mean amplitude, and signal rates. While there was no observed impact of exposure to low traffic noise, we observed an increase in mean call amplitude when frogs were exposed to high traffic noise. Increased amplitude is energetically expensive compared to changes in frequency or signal rate, which indicates that increased traffic noise may have negative impacts on long-term fitness in this species. We encourage further studies on the relationship between traffic noise and reproduction in this and other species across the region.

Key words. behaviour, acoustics, anurans, frogs, Malaysia, *Kurixalus appendiculatus*

INTRODUCTION

Transportation networks such as roads are currently expanding faster than human population growth rates, and are the most spatially extensive source of anthropogenic noise (Barber et al., 2010). Road traffic and its associated noise are expected to continue to intrude into pristine habitats in the coming decades, increasing both direct and indirect anthropogenic disturbance of wildlife in previously undisturbed areas (Laurance et al., 2009). While many of the long-term impacts of chronic exposure to anthropogenic noise remain unknown, demonstrated short-term impacts across taxa include changes in temporal patterns of behaviour (Dorado-Correa et al., 2016), movement away from areas with high traffic noise (Bayne et al., 2008; Blickley et al., 2012), decreased foraging efficiency (Senzaki et al., 2016), increased anti-predator behaviour (Quinn et al., 2006; Voellmy et al., 2014), and altered mate attraction and territorial defence (Francis & Barber, 2013). In addition to affecting behaviour, anthropogenic noise can negatively impact cue detectability, through acoustic masking (Brumm & Slabbekoorn, 2005).

Acoustic masking occurs when there is signal overlap between the frequency spectra of anthropogenic noise and

vocalisations of individuals (Brumm & Slabbekoorn, 2005), which can lead to changes in call characteristics such as amplitude (the Lombard effect; Junqua et al., 1999) or frequency (the acoustic adaptation hypothesis; Morton, 1975). As the use of sound for communication is widespread among vertebrates (Simmons, 2003), increases in anthropogenic noise and the resulting acoustic masking have the potential to impact both intra- and inter-specific communication in a wide array of taxa. The impact of anthropogenic noise on animal communication has already been documented in mammals (Terhune et al., 1979; Rabin et al., 2003), birds (Rheindt, 2003; Slabbekoorn & Peet, 2003; Brumm, 2004), frogs (Sun & Narins, 2005; Bee & Swanson, 2007; Lengagne, 2008; Parris et al., 2009; Cunningham & Fahrig, 2010), and fish (Popper & Hastings, 2009; Voellmy et al., 2014), as well as terrestrial (Lampe et al., 2012; Shieh et al., 2012) and marine invertebrates (de Soto, 2016). These studies have shown that anthropogenic noise can impact call rate (Terhune et al., 1979; Sun & Narins, 2005; Cunningham & Fahrig, 2010), frequency (Rabin et al., 2003; Rheindt, 2003; Slabbekoorn & Peet, 2003; Parris et al., 2009; Cunningham & Fahrig, 2010), and amplitude (Brumm, 2004; Cunningham & Fahrig, 2010), as well as behaviour (Bee & Swanson, 2007; Lengagne, 2008; Kaiser et al., 2011). Additionally, these studies demonstrate that while variation across species within a taxonomic group is always likely to exist (Hu & Cardoso, 2010; Dowling et al., 2012), some groups (e.g., birds) have shown more consistent responses (Slabbekoorn & Peet, 2003; Nemeth & Brumm, 2009; Potvin et al., 2011) than others.

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Anurans have demonstrated rather variable responses to anthropogenic noise exposure, highlighting the need for more data to predict the impacts of anthropogenic noise on this taxonomic group. In frogs, for example, at least two studies (Sun & Narins, 2005; Cunnington & Fahrig, 2010) have shown that the same stimulus presented to different species results in unique changes in call rate, frequency, and amplitude. These call elements help signal species identity, sexual receptivity, position, and size (Wells & Schwartz, 2007), sending information both to the opposite sex for mate selection, as well as to individuals of the same sex for competition (Wells & Schwartz, 2007). Thus, acoustic masking from traffic noise has the potential to impede reproductive success by altering signals that convey information critical to successful breeding (Francis & Barber, 2013). To date, most studies on anthropogenic impacts on anurans have been conducted in temperate regions (Lengagne, 2008; Cunnington & Fahrig, 2010; Love & Bee, 2010; Hanna et al., 2014), or in the neotropics (Kaiser & Hammers, 2009; Caorsi et al., 2017), but few studies have examined traffic noise impacts on Southeast Asian anurans. Southeast Asian anurans may be especially vulnerable to acoustic masking as industrial timber operations, the greatest driver of road expansion, continue to grow (Laurance et al., 2009; Asia-Pacific Forestry Commission, 2010). Given the crucial role that amphibians play in maintaining ecosystems (Whiles et al., 2006), understanding the impact of traffic noise on Southeast Asian amphibians can help improve biodiversity and ecosystem management (Barber et al., 2010; Rowley et al., 2010). To that end, we examined the impact of traffic noise on the spectral and temporal properties of calls of *Kurixalus chaseni*, a tree frog found in both primary and old secondary forests in Borneo and Peninsular Malaysia (Inger et al., 2017; Matsui et al., 2018).

MATERIAL & METHODS

Study area and species. The study was conducted at Danum Valley Field Centre (DVFC) in the Lahad Datu district of Sabah, Malaysia. Danum Valley Field Centre (5°01'43"N 117°45'5"E) is one of the few remaining lowland forested areas on Borneo and a designated conservation area of 4,380 km² within the Sabah Foundation Forest Concession (Marsh & Greer, 1992) and one of Southeast Asia's largest protected forests (Reynolds et al., 2011).

Kurixalus chaseni is known to inhabit primary or old secondary forests at low elevations (Inger et al., 2017), including DVFC. Currently, the global population of *K. chaseni* is decreasing, with a major threat being deforestation of lowland areas (Diesmos et al., 2004). Because this species is often found along roads in newly disturbed areas (JAS, pers. obs.), and because its conservation status is increasingly threatened by deforestation, it is an excellent candidate for examining impacts of traffic noise on calls.

Fieldwork methodology. Playback experiments were conducted to examine whether changes to temporal and spectral properties of vocalisations could be observed upon

exposure to traffic noise. The traffic noise used in these experiments was first recorded in Singapore in September 2017. The first author stood 3 m away from Clementi Road (1°18'36"N 103°46'18"E) at 0800 hours and recorded traffic noise for 3 min using a Sennheiser ME66 microphone with a Sennheiser K6 power module and an Olympus Linear PCM LS-P2 recorder with default recording settings (44.1 kHz/16 bit sampling rate, and wav file format).

At DVFC, we searched for calling *K. chaseni* nightly from 24–27 September 2017. All individuals were encountered in a single small pond at the junction of the road leading up to the hostel (4°58'2.9"N 117°48'14.7"E) between 1913–2147 hours. Air temperatures during recordings were 27–31° C (Table 1). Individual frogs were located before vocalisations were recorded, and recording started only after individuals were calling consistently. The A-B-A protocol (McGregor et al., 1992) was used to determine the impact of traffic noise on vocalisations, with vocalisations recorded for 3 min before playing traffic noise (A), 3 min during playback of traffic noise (B), and 3 min after playback of traffic noise ceased (A). One researcher stood 1 m away from sample frogs and recorded frog calls with a uni-directional microphone (Sennheiser ME66, K6) plugged into a recorder (Olympus Linear PCM Recorder LS-P2) with default settings (44.1 kHz/16 bit sampling rate, wav file format), while another researcher stood 1 m away from sample frogs and played traffic noise from a Creative Muvo Mini portable speaker. Sample frogs were not handled prior to the experiment. To avoid resampling, all frogs were marked by toe-clipping after the experiment, and marked frogs were not resampled. Time between recordings of individuals varied from 8–27 min on a given night, with a mean time of 17.4 min between individuals. While we cannot guarantee that this time between trials was sufficient to avoid influence of previous trials on a given individual, the first individuals recorded each night were not consistently different from subsequent individuals on the same night. A total of 10 individuals were sampled. To test the impact of different playback levels of traffic noise on vocalisations of *K. chaseni*, five individuals were subjected to low noise ($\mu = 64.1$ dB SPL, range = 48.2–70.0 dB SPL; LOW) and five individuals were subjected to high traffic noise ($\mu = 78.1$ dB SPL, range = 56.3–83.7 dB SPL; HIGH). The same traffic noise recording was used for both LOW and HIGH treatments, and noise level was measured during playback with an Extech Digital Sound Level Meter approximately 1 m from the focal frog, adjacent to the unidirectional microphone. Traffic noise mean dominant frequency was 785.1 Hz.

Acoustic analysis. Vocalisations were identified as “mating” or “territorial” (Sheridan et al., 2012) using aural and visual inspection of both call spectrograms and waveforms in Audacity® v2.1.3 (Audacity Team, 2016) with the Hanning window type and a FFT window size of 014. We calculated signal rates (vocalisations/min) by counting the number of vocalisations during each 3-min period for both “mating” and “territorial” calls in Audacity® v2.1.3, but focused further analyses on “territorial” calls as not all individuals made more than 10 “mating” calls in each 3-min period. Using the sample.

Table 1. Summary data of vocalisations for individual (Ind.) *K. chaseni* recorded at Danum Valley, Malaysia 24–27 Oct 2017. Mean dominant frequency and mean amplitude for each individual are mean values from 10 random vocalisations. PS = Pre-stimulus; S = Stimulus; PoS = Post-stimulus. Low noise level had $\mu = 64.1$ dB SPL, High noise level had $\mu = 78.1$ dB SPL.

Noise level	Ind.	Recording temp (°C)	SVL (mm)	Mean Dominant Frequency (kHz)			Mean Relative Amplitude (dB)			“Mating” signal rate (signals/min)			“Territorial” signal rate (signals/min)		
				PS	S	PoS	PS	S	PoS	PS	S	PoS	PS	S	PoS
Low	1	31.1	38	2.78	2.64	2.61	84.7	84.1	85.3	7	58	74	201	68	27
	2	29.4	47	2.87	2.85	2.56	84.1	82.9	83.2	64	18	75	77	58	48
	3	30.4	47	2.65	2.67	2.69	82.6	86.3	85.3	1	37	4	197	105	205
	4	30.2	35	2.81	2.80	2.91	82.6	84.9	86.4	0	8	56	184	15	18
	5	30.0	35	2.96	3.03	2.96	84.9	85.5	82.9	21	16	35	17	48	41
High	6	28.4	39	2.67	2.66	2.55	81.6	84.6	83.3	0	0	24	36	12	20
	7	28.1	34	2.80	2.89	2.81	82.8	87.9	84.3	13	30	43	117	25	16
	8	27.2	36	2.63	2.85	2.74	83.2	87.4	81.2	37	0	0	59	113	71
	9	27.3	28	2.85	3.04	3.00	78.6	83.0	82.5	1	0	6	14	19	67
	10	28.1	36	2.49	2.04	2.61	81.3	79.7	79.0	17	0	21	22	62	13

int function in R (R Core Team, 2017), ten random numbers smaller than the total number of vocalisations in each 3-min period were selected, and the vocalisations corresponding to these numbers were then selected for further analyses of mean dominant frequency and mean relative amplitude. Mean dominant frequency of vocalisations was measured using the `specan` function from the `warbleR` package in R (Araya-Salas & Smith-Vidaurre, 2016). We used Raven Pro v1.5 (Bioacoustics Research Program, 2014) with Hanning window type, FFT window size of 256, and overlap settings of 50% to calculate the mean relative amplitude of vocalisations. This was done by summing the values of the power spectrum between the lower and upper frequency bounds of the vocalisation and dividing that by the number of frequency bins (Charif et al., 2010). Differences in mean dominant frequency and mean amplitude before, during, and after traffic noise exposure were analysed to ascertain whether *K. chaseni* were increasing the likelihood of being heard by increasing amplitude (the Lombard effect; Junqua et al., 1999) or shifting call frequencies out of the overlap range (the acoustic adaptation hypothesis; Morton, 1975).

Data analysis. Differences in mean dominant frequency, mean relative amplitude, and signal rate were analysed with one-way analysis of variances (ANOVA) tests with blocking. Time periods were defined as the independent variable, call parameters as the dependent variables and individuals as blocks. Follow-up pairwise Tukey Honest Significant Differences (HSD) tests were used to determine the time periods in which call parameters differed when the ANOVA test returned statistically significant results. All statistical analyses were conducted in R (R Core Team, 2017).

RESULTS

We regressed pre-treatment signal rate on temperature for both “mating” and “territorial” calls to determine the impact of temperature on these variables. There was no significant change in “mating” calls over the range of temperatures in this study ($F [1,8] = 0.16, p = 0.70; R^2 = 0.02$), but “territorial” call rate increased significantly with temperature ($F [1,8] = 7.92, p = 0.02; R^2 = 0.50$). To test the impact of noise exposure on territorial call rate, we normalised all signal rates to the mean temperature (29 °C) using a factorial correction. After correction, we found no difference in signal rate of “territorial” calls across the three treatment periods (before, during, and after exposure to traffic noise) for individuals exposed to either “low” (ANOVA $F [2,12] = 2.10; p = 0.16$) or “high” (ANOVA $F [2,12] = 0.08; p = 0.93$) traffic noise.

Low treatment. We found no significant effect of traffic noise on “mating” ($F [2, 8] = 2.1, p = 0.19$) or “territorial” ($F [2, 8] = 2.8, p = 0.12$) signal rates (Table 1). We also did not detect any impact of traffic noise on mean dominant frequency ($F [2, 143] = 2.7, p = 0.07$; Table 1) or mean relative amplitude ($F [2, 143] = 1.6, p = 0.21$; Table 1) of territorial calls.

High treatment. We found no significant effect of traffic noise on either “mating” ($F [2, 8] = 1.0, p = 0.40$), or “territorial” ($F [2, 8] = 0.15, p = 0.87$) signal rates (Table 1), or on mean dominant frequency ($F [2, 143] = 0.75, p = 0.48$; Table 1). However, we found that high traffic noise had a significant effect on mean relative amplitude ($F [2, 143] = 10.0, p < 0.001$; Table 1) of territorial calls, with the Tukey HSD test indicating a significant increase of 2.45–3.01 dB in mean relative amplitude during the stimulus period compared to both pre- ($p < 0.001$) and post- ($p = 0.002$) stimulus periods.

DISCUSSION

We did not find a significant impact of traffic noise on call rate or mean dominant frequency of *K. chaseni* during exposure to either low or high traffic noise. However, we did observe an increase in relative amplitude of calls during exposure to high traffic noise. While this has been observed in birds (Brumm & Todt, 2002; Nemeth & Brumm, 2010), our study is one of the first to demonstrate an increase in relative amplitude in response to traffic noise in anurans. Contrary to our findings, Cunningham & Fahrig (2010) observed a decrease in call amplitude (and an increase in dominant frequency) in green frogs and leopard frogs in response to traffic noise, while other species have shown no change in amplitude despite changes in frequency or call rate (Cunningham & Fahrig, 2010; Love & Bee, 2010; Hanna et al., 2014). It is possible that our results represent a “shock” response to a noise above normal background sound levels being suddenly broadcast, so future studies could increase the length of time for which the sound is played before the response to the noise is recorded, or repeat the experiment multiple times with a given frog, to determine whether observed responses are simply due to the sudden change, or if they are indeed due to individuals attempting to adapt to a newly noisy environment.

Given the lack of overlap in frequency between traffic noise (< 1 kHz) and calls of *K. chaseni* (2–3 kHz), it is not entirely surprising that this species did not alter call frequency during traffic noise playback. However, this is in contrast to other species whose dominant frequencies do not overlap with low frequency traffic noise and which have been shown to increase frequency but decrease amplitude of calls (Cunningham & Fahrig, 2010). Some have suggested that these species may already be calling at maximum amplitude, and are thus unable to adjust amplitude regardless of changes in ambient noise levels (Love & Bee, 2010). Additionally, energetic costs of calling increase exponentially with call amplitude, with an approximate doubling in energetic cost for each 3 dB increase in amplitude (Parris, 2002), making it energetically “easier” for anurans to increase call frequency rather than amplitude. Thus, contrary to other anurans studied to date, *K. chaseni* is likely incurring a sizeable energetic cost by increasing call amplitude when exposed to high levels of traffic noise, while other anurans may either be unable to increase amplitude, or choosing the less energetically costly alternative of increasing call frequency.

Since amphibians have indeterminate growth (Halliday & Verrell, 1988), increased energetic costs of calling may lead to limits in growth in order to allocate more energy towards calling. This can in turn lead to lower reproductive output (Gibbons & McCarthy, 1986) and increased risk of desiccation (Heatwole et al., 1969), both of which can lead to decreases in population size. Additionally, decline in body size is likely to impact trophic levels both above and below anurans, upsetting ecological balances and community organisation (Sheridan & Bickford, 2011).

The change in amplitude during high but not low traffic noise exposure may be due to the intermediately noisy natural environment of these frogs. Natural background noise levels (57.0–63.8 dB SPL) recorded throughout the study were similar to the amplitude of the low (48.2–70.0, $\mu = 64.1$ dB SPL) but not high ($\mu = 78.1$ dB) traffic noise treatment, likely accounting for the observed differences in response between the two treatments. Lengagne (2008) played traffic noise at two different intensities to the treefrog *Hyla arborea*, whose background chorus levels (74 dB) were similar to the low traffic noise (72 dB), and similarly found that calling effort (bout duration/min), number of bouts/min, and bout duration all decreased during high (88 dB) traffic noise playback, but were unchanged during low traffic noise playback. Lengagne (2008) also found no significant difference in call duration or dominant frequency during low or high traffic noise playback, similar to results from the present study, but amplitude was not measured. Additionally, it should be mentioned that in the present study, the same traffic recording was played at different playback levels to represent different amounts of traffic. However, high and low traffic volumes differ in ways other than loudness or volume. The number of cars per minute, for example, may be an important factor in eliciting a response, as might vehicle type. Standard cars differ from large trucks (logging trucks, or palm oil tanker trucks, for example) in sound parameters, so future studies could test the impact of cars per minute, loudness of traffic, as well as vehicle type on calls of *K. chaseni*.

Our results indicate that *K. chaseni*, a species common in both primary and secondary forests of Peninsular Malaysia and Borneo, may be negatively impacted by increased noise from growing transportation networks, as evidenced by the energetically costly response of increased call amplitude during exposure to high traffic noise. As transportation networks continue to grow, it is important to understand both direct and indirect impacts on biodiversity. Road networks will fragment and reduce habitat size (direct impacts), as well as decrease habitat quality and increase noise pollution (indirect impacts). Future studies should examine the relationship between increased energetic costs of calling for anurans such as *K. chaseni* exposed to traffic noise, and factors such as breeding success, body size, and population size, in order to determine indirect effects of traffic noise on population persistence. Furthermore, studies of additional species commonly found in both disturbed and relatively undisturbed habitats will illustrate how widespread the impacts of traffic noise are likely to be on anuran communities across the region, in order to more accurately assess the impact of roads on biodiversity in tropical Southeast Asia.

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