Volume 36 (April 2026), 87–97

Herpetological Journal

FULL PAPER

https://doi.org/10.33256/36.2.8797

Published by the British Herpetological Society

Updated assessment of population size and habitat of the Critically Endangered frog *Boophis williamsi* (Guibé, 1974) in Madagascar's central highlands

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Amphibians are among the most threatened vertebrates, yet many species lack the ecological and demographic data needed to implement effective conservation measures, especially in tropical regions. William's bright-eyed frog *Boophis williamsi* is one of Madagascar's most highly threatened frog species, being restricted to a small area of the Ankaratra Massif in the central highlands. We conducted fieldwork during 2018, 2019 and 2021 to identify habitat associations across life stages and estimate abundance along ten stream transects. Our results confirmed *B. williamsi* presence at seven localities within a small area of only a few square kilometres, the only location the species is known to occur. Habitat associations varied by life stage, with tadpoles more often found in shaded, rocky streams within humid montane forest, while adults were more often found along rocky streams in open savanna habitats. Overall, streams were shallow (0.18–0.83 m), cool (7.70–17.80 °C), and often slightly alkaline (pH 6.70–8.26). Using N-mixture models, we estimated fewer than 600 individuals (tadpoles and frogs combined) across four surveyed stream transects during the 2018–2019 austral summer. Detection probabilities for frogs were seasonally variable, peaking in December at the onset of the rainy season. Conversely, tadpoles had higher and relatively constant detectability across September, December and March. Despite ongoing threats from deforestation and fire, *B. williamsi* persists at Ankaratra, likely aided by recent conservation efforts. We recommend the development of a standardised, long-term monitoring protocol incorporating both tadpoles and frogs, as well as increased protection of high-elevation forests near stream habitats that support this imperilled species.

Keywords: amphibian conservation, Ankaratra, biodiversity conservation, detection probability, range-restricted species

INTRODUCTION

Amphibians are the most threatened vertebrate class globally, yet many species and major clades remain poorly studied (da Silva et al., 2020; Womack et al., 2022; Luedtke et al., 2023). The lack of accurate species-specific data, such as distribution, population size and key habitat features, can hinder effective conservation planning and recovery efforts (Cushman, 2006; Camaclang et al., 2015; Hortal et al., 2015). This challenge is especially acute in tropical regions, where amphibian diversity is highest and research efforts are disproportionately low (Ficetola et al., 2014; Christie et al., 2021). Furthermore, threatened amphibian species, and especially those in countries with historically limited scientific funding and capacity, are least likely to be the focus of research and are often overlooked (da Silva et al., 2020). Addressing such knowledge gaps is essential to guide targeted conservation actions for the most at-risk species.

The island of Madagascar supports remarkable amphibian species richness and endemism amid urgent conservation challenges (Antonelli et al., 2022; Ralimanana et al., 2022). The island supports an estimated 500–700 endemic frog species, many of which remain undescribed (Carné & Vieites, 2024). The ecological characteristics of most species are similarly poorly known. Nearly half of the 313 species that have been assessed by the IUCN Red List are at risk of extinction (IUCN, 2025), and Madagascar ranks among the countries with the highest proportion of amphibian species occurring entirely outside of protected areas (Nori et al., 2015). Given that habitat loss and degradation are the greatest threats to amphibians both locally and

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Figure 1. An adult *Boophis williamsi* (left) and its highland stream habitat at Tsimiaramianadahy (right) on Ankaratra Massif, central Madagascar.

globally (Andreone et al., 2021; Ralimanana et al., 2022), identifying critical habitat characteristics is vital to inform protection efforts. Moreover, with only a handful of exceptions (e.g. Andreone, 1998; Rabemananjara et al., 2007; Lehtinen & Carfagno, 2011; Edmonds et al., 2024; Rasoarimanana et al., 2024), demographic traits and baseline population size estimates are lacking for most frog species on the island. Such information is fundamental for assessing species' conservation status and guiding management decisions (Mills, 2013).

William's bright-eyed frog *Boophis williamsi* (Fig. 1) is among the most highly threatened frogs in Madagascar (Andreone et al., 2008). The species is restricted to a tiny area of the Ankaratra Massif, a highland volcanic range in central Madagascar that has experienced a disproportionately high rate of deforestation (Monteiro et al., 2023). *Boophis williamsi* is associated with small rocky streams that flow in and around a severely fragmented forest that covers <25 ha (Goodman et al., 2018). Although known from several localised sites, the species is most frequently encountered at Ambohimirandrana, where Vences et al. (2002) first described its live colouration after observing a wild male for the first time in nearly three decades. Since then, *B. williamsi* has been the focus of increased fieldwork, though it appears to occur at relatively low densities. The Malagasy Herpetological Association Langaha conducted a 15-day survey and found fewer than ten individuals per 600–1000 m of stream (Rabemananjara et al., 2011). Andreone et al. (2014) later provided the first population size estimates, reporting 15–46 frogs and 17–51 tadpoles in a 100 m stream section at Ambohimirandrana. Despite its precarious status, *B. williamsi* has since been promoted as a flagship species to support forest protection and restoration efforts at Ankaratra (Rabemananjara et al., 2012), yet recent data on its population status remain scarce.

Here, we present ecological data on *B. williamsi* based on fieldwork conducted on the Ankaratra Massif, as part of a broader effort to study the region's amphibian fauna. Specifically, our objectives were to: 1) verify the species' distribution and determine whether *B. williamsi* is present at ten stream sites; 2) assess habitat characteristics (forest type, canopy cover and stream substrate) and water-quality parameters (pH, flow velocity, discharge, depth and temperature); and 3) estimate frog and tadpole abundance and detection probabilities to update baseline knowledge and inform future monitoring, including the most effective timing and conditions for surveys. Collectively, these efforts aim to support long-term monitoring and conservation of one of Madagascar's most highly threatened amphibian species.

MATERIALS & METHODS

Study site

The Ankaratra Massif is located in the Vakinankaratra Region of Madagascar's central highlands, approximately 15 km west of the city of Ambatolampy. The massif has a sub-humid cool climate. Average daily temperatures range 10.7–22.9 °C but can fall as low as 3.1 °C during the austral winter and reach up to 29.4 °C in the summer (Goodman et al., 2018). The degree to which central Madagascar was forested during the last millennium is debated (see Bond et al., 2008; 2023; Joseph et al., 2021; 2024; Phelps et al., 2025), but what is clear is that the Ankaratra Massif has had almost no forest and has been dominated by an anthropogenic landscape of grasses for >100 years (Perrier de la Bâthie, 1927). Natural forest on the mountain range is restricted to a few square hectares and falls within the boundaries of the larger Réserve de Ressources Naturelles de Manjakatompo Ankaratra, a New Protected Area (NAP) established in 2015 (Goodman et al., 2018). Eight local community-based groups (Vondron'Olona Ifotony or VOI) are responsible for managing the site in collaboration with the NGO Vondrona Ivon'ny Fampandrosoa (VIF) and

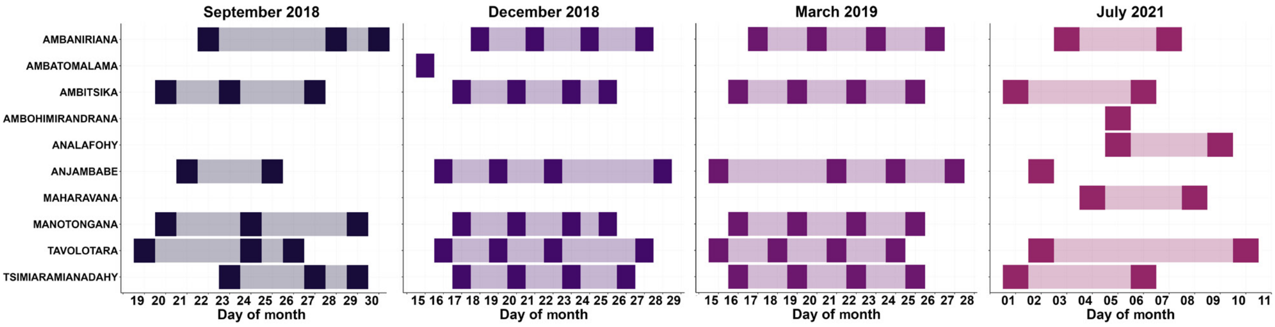

Figure 2. Survey effort across sites and months at Ankaratra Massif in the central highlands of Madagascar. Dark tiles indicate individual days when a transect was surveyed; lighter horizontal bars represent the full date range over which surveys occurred within each month. Site names correspond to those listed in Table 1.

Table 1. Site names, geographic co-ordinates, altitude in metres above sea level, number of days surveyed (days) and estimated frog abundance (N̂). "Present" indicates sites where *Boophis williamsi* was detected. Each site had a 100 m stream transect divided into ten 10 m sections, and N̂ = estimated mean number of frogs per 10 m section, derived from the top model in Table 2. CI = 95% credible interval for the abundance estimate. Asterisk (*) denotes sites where we only observed tadpoles.

Site	Latitude	Longitude	Altitude (m)	Days	N̂	95% CI
Present						
Ambaniriana	-19.3432	47.2528	2177	13	6.9	3.1 – 17.6
Ambitsika	-19.3358	47.2810	2213	13	4.4	1.9 – 11.3
Ambohimirandrana*	-19.3399	47.2736	2326	1	-	-
Analafohy	-19.3449	47.2750	2130	2	-	-
Maharavana*	-19.3409	47.2680	2118	2	-	-
Tavolotara	-19.3459	47.2791	2020	13	1.5	0.5 – 4.0
Tsimiaramianadahy	-19.3334	47.2626	2391	13	11.5	4.9 – 28.4
Undetected						
Ambatomalama	-19.3528	47.2774	2029	2		
Anjambabe	-19.3491	47.2940	1881	12		
Manotongana	-19.3347	47.2831	2177	11		

the regional forest administration (Rabemananjara et al., 2012). However, uncontrolled fires and charcoal production continue to impact forest habitat (Goodman et al., 2018), with the deforestation rate on Ankaratra in recent decades nearly four times the national average (Monteiro et al., 2023).

Survey methods and data collection

We surveyed eight sites on the Ankaratra Massif during September, December and March 2018–2019 for 1–4 days each month. Additionally, we revisited sites and surveyed an additional two sites in July 2021 (Fig. 2; Table 1). We used visual encounter surveys along 100 m stream transects, with the start and end points marked with a pink flag. We further subdivided transects into 10 m sections and marked the start and end points with green flags. We surveyed each transect for 2–4 hours at three times of day: morning (09:00–12:00 h), afternoon (14:00–16:00 h) and evening (19:00–23:00 h). At each time point, two observers visually searched for tadpoles and frogs, while two additional team members recorded data on captured individuals.

We captured animals by hand and temporarily placed them in individual plastic bags with a small amount of stream water. The animals and bags were kept in shaded areas during handling and were released at the point of capture after data collection (typically held in the bag for <10 min). For each individual encountered, we recorded sex (male, female, unknown), developmental stage (tadpole, immature frog, adult frog) and snout-vent-length (SVL) measured to the nearest millimetre with vernier calipers. Sex was determined based on the presence or absence of nuptial pads and secondary sexual characteristics following Glaw & Vences (2007). Individuals that lacked secondary sexual characteristics and measured ≤37 mm SVL were classified as immature, and their sex was recorded as unknown.

Habitat and environmental data

We qualitatively characterised habitat features at the start of each 10 m stream section within a transect. Habitat types included forest edge (the transitional zone between forest and adjacent vegetation), degraded forest (forest heavily impacted by anthropogenic

Table 2. Comparison of N-mixture models for estimating *Boophis williamsi* abundance at Ankaratra Massif, Madagascar. Models varied only by the detection probability covariates. ΔELPD = difference in expected log-predictive density, a measure of model fit. The standard error (SE) of ΔELPD shows the uncertainty in the difference between models. Given the high SE of ΔELPD for tadpole models, we used the same month-varying detection model for estimates as we did for frogs so that the two were comparable.

Detection covariate	ΔELPD	SE of ΔELPD
Frogs		
Month	0	0
Month * water temp	-4.867	2.313
Discharge volume * water temp	-20.460	9.110
Depth * water temp	-24.143	8.897
Depth	-39.536	15.596
Discharge volume	-39.758	14.965
Null	-43.123	18.743
Water temperature	-47.401	23.594
Tadpoles		
Month * water temp	0	0
Month	-4.705	4.675
Discharge volume * water temp	-5.831	6.075
Depth * water temp	-6.826	5.213
Null	-6.913	5.84
Water temperature	-7.118	5.241
Depth	-7.852	6.185
Discharge volume	-7.868	5.688

activities and dominated by *Pinus* spp.), semi-degraded forest (forest retaining some native plant species but also invaded by *Pinus* spp. and other non-native plants), intact forest (minimally disturbed forest maintaining its native plant assemblages), gallery forest (riparian forest occurring along streams between mountain slopes) and savanna (grassland with scattered shrubs). Stream substrate type was classified as predominantly mud, mixed substrate (mud, sand and plant debris), mostly rocks, or all rocks. Canopy cover categories were visually estimated and included medium shade (~25–75% canopy cover), closed canopy (>75% canopy cover), shaded understory (little to no overhead canopy, but the stream segment was shaded by shrubs, grasses or low vegetation) and open (typically corresponding to savanna habitat with no canopy). Additionally, we recorded stream physicochemical variables at the start of each 10 m stream section before surveying using a multimetre, including pH, water temperature, flow rate and discharge. We also measured stream depth and width.

Analysis of habitat associations and abundance modelling

To explore habitat characteristics, we examined the distribution of tadpole and frog captures across habitat type, stream substrate and canopy cover, using proportional bar plots. Habitat associations were examined separately for adults and tadpoles to assess potential differences in microhabitat use across life stages. We cleaned, analysed and visualised data in R version 4.3.1 (R Core Team, 2023) using packages dplyr (Wickham et al., 2025) and ggplot2 (Wickham, 2011).

We estimated frog and tadpole abundance from count data with N-mixture models implemented in the ubms R package (Kellner et al., 2022). N-mixture models allow estimation of population size from repeated counts of unmarked individuals and can produce results comparable to more intensive capture-mark-recapture methods (Ficetola et al., 2018; Madsen & Royle, 2023). We developed separate models for frogs and tadpoles, as we assumed their detection probabilities differed, and the ubms package does not currently support modelling distinct detection probabilities for multiple life stages within a single framework. Full data and code are available in the Illinois Data Bank at https://doi.org/10.13012/B2IDB-7088746_V1.

Abundance estimates were based on data from four of the ten surveyed sites where sufficient observations were obtained: Ambaniriana, Ambitska, Tavolotra and Tsimiaramianadahy. Three sites where *B. williamsi* was detected (Ambohimirandrana, Analafohy and Maharavana) were excluded from the analysis due to limited sampling (one or two site visits only) and three sites yielded no detections (Table 1). We used a Bayesian framework to implement a random site effect on abundance, accounting for the non-independence of 10 m sections within each stream transect while enabling site-specific abundance estimates. To meet the assumption of population closure, we restricted the analysis to surveys conducted between September and March, assuming no births, deaths, immigration or emigration during this period. We acknowledge that some violations of closure, such as tadpole metamorphosis or mortality, may have biased estimates by confounding detection probability and abundance estimates (see Discussion). However, given the low number of frogs detected overall, we retained all surveys from the 2018–2019 field season to maximise sample size, excluding only the July 2021 surveys from abundance estimation.

We compared eight models with varying detection covariate structures, including month, water depth, water discharge, water temperature, and interactions between water temperature and the other three covariates (Table 2). Although additional water quality variables were measured, they were excluded from the analysis due to multicollinearity concerns or because they were considered unlikely to affect detection probability (e.g. pH). For each model, we set the upper bound on latent abundance per transect section as K = 100 and applied broad priors for both state and detection coefficients, specifying a normal distribution with mean 0 and a standard deviation of 3. Each model was run for 50,000 iterations, with the first 25,000 iterations discarded as warmup samples. Model convergence was

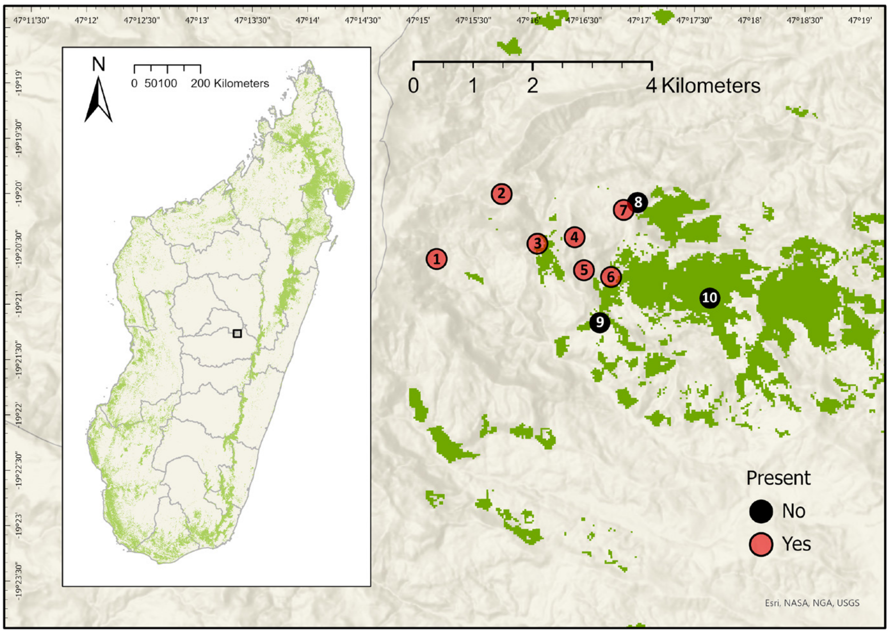

Figure 3. Study location showing the entire known distribution of William's bright-eyed frog *Boophis williamsi* in the central highlands of Madagascar. The green forest cover is from the 2017 update of Vieilledent et al. (2018). The grey lines in the inset are regional political boundaries. The sampled localities are: **(1)** Ambaniriana, **(2)** Tsimiaramianadahy, **(3)** Maharavana, **(4)** Ambohimirandrana, **(5)** Analafohy, **(6)** Tavolotara, **(7)** Ambitsika, **(8)** Manotongana, **(9)** Ambatomalama and **(10)** Anjambabe.

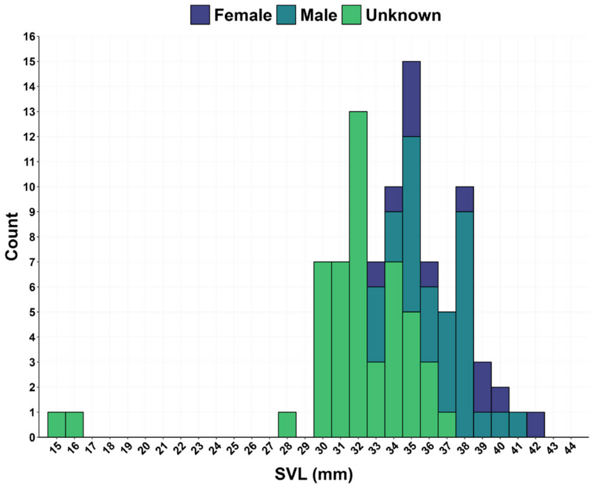

Figure 4. A histogram of all *Boophis williamsi* frogs captured by snout-vent-length (SVL) and sex.

Table 3. Water parameters of sites surveyed at Ankaratra Massif in the central highlands of Madagascar where William's bright-eyed frog *Boophis williamsi* was present.

Variable	Mean	SD	Range
Water temperature (°C)	14.77	1.99	7.70 – 17.80
pH	7.77	0.36	6.70 – 8.26
Depth (m)	0.46	0.21	0.18 – 0.83
Speed (m/s)	1.16	0.46	0.19 – 2.67
Water discharge (m^3/s)	0.23	0.31	0.00 – 0.91

supported by trace plots, R-hat values (all <1.01) and high effective sample sizes. We assessed model fit by examining residuals, conducting posterior predictive checks and compared models using differences in expected log-predictive density (Table 2).

RESULTS

We recorded *B. williamsi* at seven of ten sites on the Ankaratra Massif (Fig. 3; Table 1). Across the seven sites and all survey periods, we observed 546 tadpoles and 95 frogs. Of the frogs, 42 were classified as immature and 53 as adults (32 males, 14 females and 7 undetermined sex). Male frogs were slightly smaller than females, with a mean SVL of 36.6 mm (SD = 1.9) compared to 37.6 mm (SD = 2.8) for females (Fig. 4).

The highest number of frog captures occurred at Tsimiaramianadahy (30 individuals), a savanna-dominated site, and Ambaniriana (28 individuals), a degraded forest. At sites with limited survey effort (one or two visits), we recorded a single frog at Analafohy (semi-degraded forest), tadpoles at Ambohimirandrana (semi-degraded forest) and Maharavana (degraded forest dominated by invasives), and no detections at the lower-elevation intact forest site Ambatomalama. Most frog captures (71 of 95 frogs) occurred in open savanna habitat or gallery forest with sparse canopy cover (Fig. 5). The remaining captures were made along stream transects in degraded forest (17 frogs) or intact forest (7 frogs). In contrast, tadpoles were predominantly found in forested habitat, with a smaller proportion of captures (<20%) made

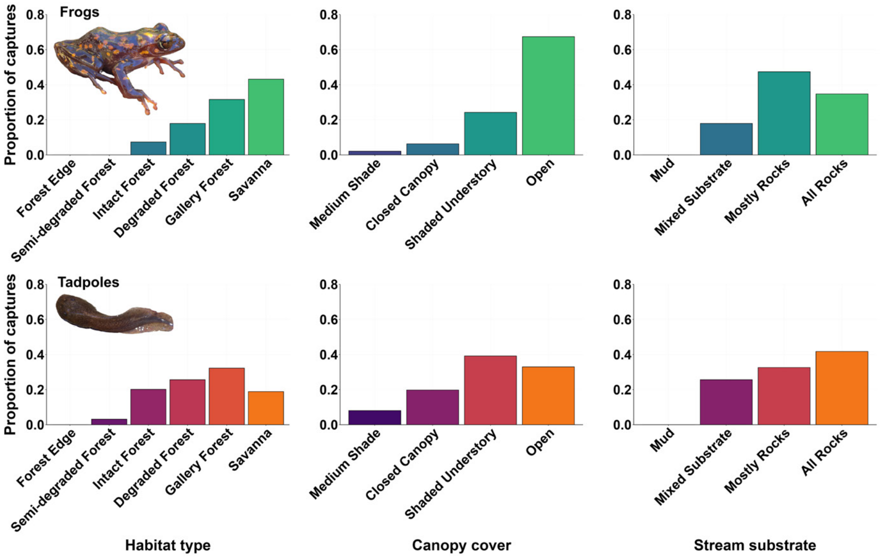

Figure 5. The proportion of total frogs (top) and tadpoles (bottom) captured by habitat type, canopy cover, and stream substrate.

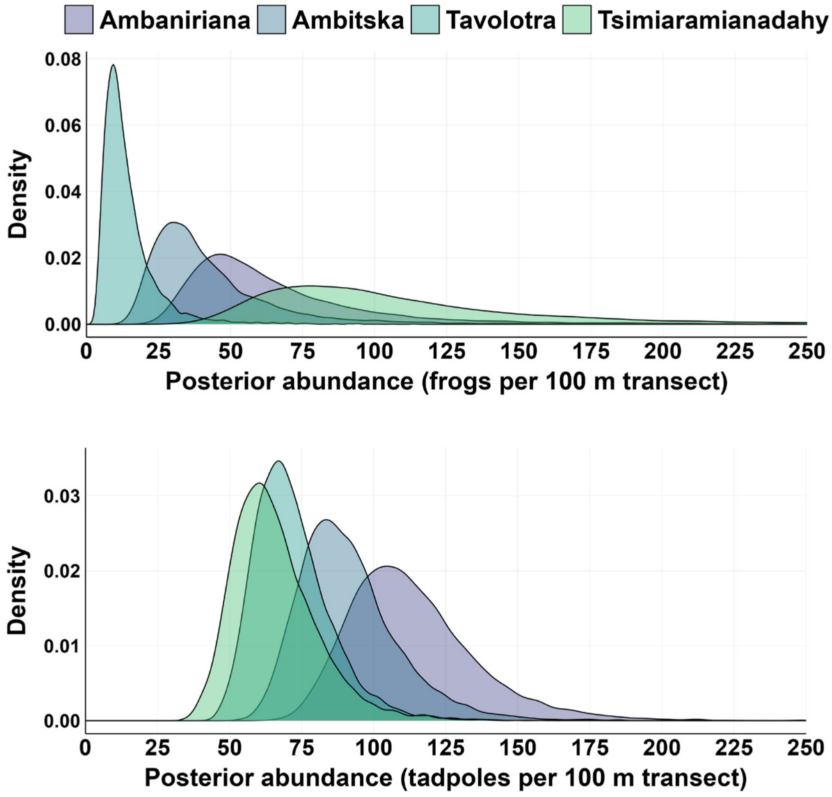

Figure 6. Estimated abundance of William's bright-eyed frogs (top) and tadpoles (bottom) within 100-metre transects across four sites on the Ankaratra Massif in the central highlands of Madagascar. The plot displays the posterior predictive distribution of abundance based on 10,000 posterior draws. The shape shows the range of likely values for each site, where wider posterior densities indicate greater uncertainty around the abundance estimate.

in savanna. No frogs or tadpoles were detected along forest edge transects (Fig. 5). Capture locations also varied by stream substrate type. Both frogs and tadpoles were most frequently captured in transect sections with a predominantly rocky or all-rock substrate, while no individuals were recorded along transect sections with a predominantly muddy substrate (Fig. 5). Water quality measurements indicated that streams were generally shallow, cool and slightly alkaline, with a broad range of water velocities and discharge rates (Table 3).

The model with detection probability varying by month performed best for frogs (Table 2), providing an estimated total abundance of 241 individuals (95% credible interval: 93–634) across all four sites and 400 m

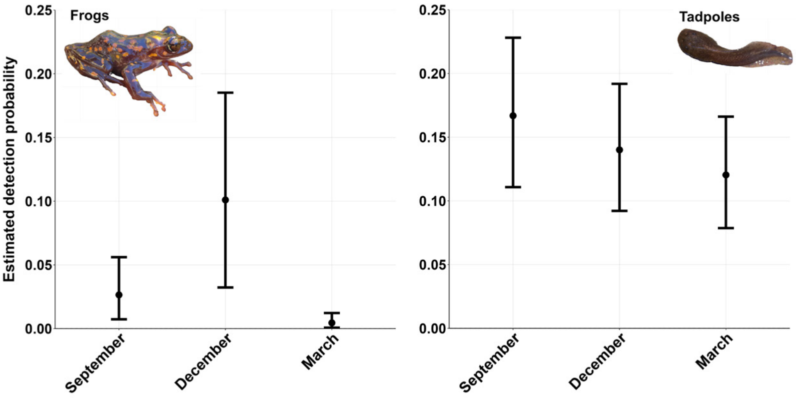

Figure 7. Estimated mean detection probability by month of William's bright-eyed frogs (left) and tadpoles (right). Error bars are 95% credible intervals.

of total stream (Table 1). There was greater uncertainty comparing models for tadpoles, with large standard errors relative to ΔELPD for several models (Table 2). As such, we decided to use the same model for tadpoles as we did for frogs (detection varying by month), which provided an estimate of 343 tadpoles (95% CI: 221–521; Table 1). Among the four sites, Tavolotara had the lowest estimated mean frog abundance per transect, whereas Tsimiaramianadahy had the highest, albeit with greater uncertainty (Fig. 6). Conversely, the estimate of tadpole abundance was lowest at Tsimiaramianadahy and highest at Ambaniriana (Fig. 6). Estimated frog detection probability was generally very low (range = 0.004–0.102) with a peak in December (Fig. 7). The estimated detection probabilities for tadpoles from the month-varying model were higher overall (range = 0.120–0.167), with a slight seasonal decline from a high in September to a low in March, although 95% credible intervals overlapped (Fig. 7).

DISCUSSION

We detected *B. williamsi* at seven sites on the Ankaratra Massif, all located within a relatively small area already known to support the species. Notably, all but one of these sites had also been reported by Monteiro et al. (2023) who examined the effects of land use change on amphibian diversity at Ankaratra. All the streams drain in the same general direction and likely belong to the same watershed. This is especially evident at Ambohimirandrana, Analafohy and Tavolotara, which are clustered within ~0.5 km of one another and flow into a shared downstream river network. Such spatial configuration may allow for passive dispersal of tadpoles or occasional adult movement between localities, especially during periods of high rainfall. We document the first confirmed occurrence of *B. williamsi* at Ambaniriana (Fig. 3), a stream over 1 km from the other occupied sites, with gallery forest that has been the focus of surveys for the endangered and locally endemic frog *Mantidacytlus pauliani* (Radonirina et al., 2023). We suggest that future surveys verify the full extent of the distribution of *B. williamsi* by sampling additional streams on the massif, especially considering the species' low and seasonally variable detection probability.

Tadpoles had a higher detection probability than frogs, emphasising the importance of larval identification as a tool for confirming site occupancy. The tadpoles of *B. williamsi* are thought to take over one year to complete metamorphosis (Blommers-Schlösser, 1979; Gehring et al., 2011), and larval counts could therefore provide a stable index of site occupancy and reproduction over time. In other amphibian species, detecting larvae can sometimes be more effective than surveying post-metamorphic life stages (e.g. Petitot et al., 2014). Additionally, data on larval vital rates are necessary for projecting population persistence and evaluating population trajectories (Skelly & Richardson, 2010; Ribeiro & Rebelo, 2011), thus collecting data on all life stages including larvae is valuable. Given the higher detectability of tadpoles in our study, we recommend continuing to include larvae in future monitoring protocols.

Regarding the best time of year to survey, we found that detection probability of frogs was highest in December, which coincides with the onset of the rainy season in Madagascar. The higher detection in December likely reflects increased *B. williamsi* activity during the warmer, wetter austral summer, which coincides with reproduction for many frog species in Madagascar (Heinermann et al., 2015; Rakotoarisoa et al., 2025). Nonetheless, the reproductive timing and phenology of *B. williamsi* remain completely unknown; we did not observe any eggs or reproductive activity, nor have other authors. In contrast, detection was extremely low for frogs at the end of the rainy season in March, so we recommend surveys for post-metamorphic stages occur earlier. On the other hand, tadpole detection probability peaked in September and, like frogs, declined by March. However, overlapping credible intervals around tadpole detection probability estimates suggest tadpoles may remain consistently detectable, unlike frogs. Interestingly, Monteiro et al. (2023) reported finding the greatest number of *B. williamsi* during the austral winter months (May–August), whereas our limited surveys in July yielded few detections, with most individuals encountered instead during surveys in September and December. Considering the high variation in interannual abundance and activity in amphibian species elsewhere in Madagascar (see Dubos et al., 2020), it is also possible that seasonal peaks of *B. williamsi* may not be consistent across years.

There were distinct habitat associations for different life stages of *B. williamsi*. Tadpoles were more frequently encountered in shaded stream segments running through gallery forest and in stream sections with a substrate composed entirely of rocks. In contrast, frogs were more often captured along sections with a mostly rocky substrate in open savanna habitat. The tadpoles' extensive labial tooth rows indicate adaptation to fast-flowing, rocky streams (Schmidt et al., 2008), and the association with shaded environments in our study may reflect a need for cooler water temperatures or higher oxygenation. On the other hand, adult frogs may benefit from the ability to thermoregulate in open habitats (Spranger et al., 2024). Other possible explanations for the differing habitat associations between life stages include variation in food availability or breeding requirements, for example if larvae feed on the biofilm growing on rocks within forested streams or if adults move downstream to gallery forest to reproduce but spend the rest of the year in the savanna. Notably, we did not detect any tadpoles or frogs in forest edge habitat, which supports earlier suggestions that the species may be particularly vulnerable to edge effects (Rosa, 2017). Still, we note that these observations are based on presence-only data and do not account for habitat availability. Further research incorporating random unused sites (e.g. Lehtinen & Carfagno, 2011; Edwards et al., 2019; Rasoarimanana et al., 2024) would help clarify the strength of habitat associations.

The low abundance of *B. williamsi* reaffirms its high extinction risk, with fewer than 600 individuals (frogs and tadpoles combined) estimated across four of the seven surveyed streams during the 2018–2019 austral summer. To estimate abundance, we assumed population closure; as a result, if individuals died or tadpoles metamorphosed during the survey period, we may have slightly underestimated tadpole abundance and overestimated frog abundance. Future work could consider more flexible open N-mixture models, which also allow estimation of additional demographic parameters (Zipkin et al., 2014). Nonetheless, we believe the abundance estimates produced here reasonably reflect the species' small population size, an important finding given that small populations are subject to high variance in population growth rates, increasing their risk of extinction (Green, 2003).

The small population size of *B. williamsi* raises particular concern when taken into consideration with the species' likely slow life history pace. Skeletochronology data indicate the species requires two years post-metamorphosis to reach sexual maturity, with frog age ranging from 3–6 years in a sample of 14 individuals (Andreone et al., 2014). Other life history traits of *B. williamsi*, such as the number of eggs in a mass, size at metamorphosis, growth patterns and associated mortality schedules, are unknown and require further study. Nevertheless, because *B. williamsi* occurs on one of the highest mountain ranges in Madagascar and considering the inverse relationship between elevation and life history pace (Morrison & Hero, 2003; Zhang & Lu, 2012; Hardy et al., 2023), we expect the species to have relatively low fecundity and slow growth compared to other *Boophis* species. Such characteristics allow for persistence under stable conditions but also leave populations vulnerable to disturbances impacting adult survival rates (Cayuela et al., 2015) and are associated with high extinction risk at high altitudes (Guirguis et al., 2023).

Despite its precarious situation having been recognised for decades (Vences et al., 2002; Andreone et al., 2008), *B. williamsi* continues to persist amid deforestation and wildfires. Encouragingly, the area has seen increased conservation attention over the past 10–15 years, including the development of community-led reforestation projects and legal protection of habitat (Rabemananjara et al., 2012; Goodman et al., 2018). These conservation efforts may have helped buffer some threats. Further, although *B. williamsi* individuals tested positive for the amphibian chytrid fungus *Bd* in 2013 (Bletz et al., 2015), no signs of disease have been reported in the species or elsewhere in Madagascar (Andreone et al., 2021).

Still, *B. williamsi* remains restricted to just a handful of mountain streams and occurs at low densities, showing the need for continued conservation and monitoring. Long-term persistence will depend on sustained financial support to manage its habitat, meaningful engagement with local communities who rely on forest resources, and active participation from government agencies, which are ongoing challenges given the limited resources for conservation in Madagascar (Eklund et al., 2022). To guide future efforts, we recommend developing a standardised, long-term monitoring protocol and emphasise the need for continued protection of remaining high-elevation gallery forest along rocky streams on the massif.

ACKNOWLEDGEMENTS

We thank the Mohamed Bin Zayed Species Conservation Fund for supporting this research. Fieldwork was made possible through the collaboration of the Vondrona Ivon'ny Fampandrosoana (VIF), and we are especially grateful to the local guides and cook from the local community groups (VOI) of the Ankaratra Massif who assisted R. Fanomezantsoa in the field, as well as to H.O. Radonirina. We also thank C. Brooks, S. Horigan and the Coding 4 Conservation mentorship program, and acknowledge the Direction Régionale de l'Environnement et du Développement Durable (DREDD) Vakinankaratra for providing research permissions. Institutional support was provided by the University of Mahajanga and the École Doctorale Écosystème Naturel (EDEN). All fieldwork and research for the study was authorised by Malagasy authorities and conducted under the research permit N° 392/21/MEDD/SG/DGEF/DGNRE/SCBE.Re

Author contributions

Rebecca Fanomezantsoa and Nirhy Rabibisoa designed the study and collected the data; Nirhy Rabibisoa supervised the research; Rebecca Fanomezantsoa and Sam Hyde Roberts acquired funding; Rebecca Fanomezantsoa and Devin Edmonds performed the analyses; Devin Edmonds drafted the manuscript; all authors reviewed and edited the final version.

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Accepted: 11 November 2025