ÖGH Herpetozoa
Austrian Herpetological Society

Herpetozoa 38: 231–240 (2025)
DOI 10.3897/herpetozoa.38.e153679

Early amphibian colonizers of man-made mini-ponds: first results of a citizen science experiment

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Academic editor: Peter Mikulíček ♦ **Received** 23 March 2025 ♦ **Accepted** 15 August 2025 ♦ **Published** 10 September 2025

Abstract

The loss of suitable spawning habitats is a global threat to amphibians. Thus, establishing man-made ponds may serve as a successful conservation practice to counteract this threat. In this experiment, we used a citizen science (CS) approach to create mini-ponds in private gardens and asked participants to monitor amphibian activity. We provided 302 polyethylene mini-ponds (120 L × 90 W × 40 D cm) to citizen scientists across the range of the target species, the European green toad (*Bufotes viridis*), in Austria. In the first year of the experiment (2024), *B. viridis* and 12 other amphibian taxa were recorded at 38% of the monitored sites. Signs of amphibian reproduction were detected in 10% of the mini-ponds. Bayesian occupancy models showed detection probabilities consistent with other studies. Over time, we expect more amphibians to discover the mini-ponds and more individuals to use them for reproduction. Our experiment demonstrates that the creation of relatively small ponds is a promising method for supporting and monitoring amphibian populations.

Key Words

Anura, breeding sites, citizen science, frogs, newts, succession, toads, Urodela

Introduction

Amphibians are among the most threatened taxa globally, with 41% of species threatened with extinction (Luedtke et al. 2023). Factors contributing to their decline include habitat destruction and degradation (especially for aquatic habitats), pollution, and climate change (Cushman 2006; Jaureguiberry et al. 2022). To counteract amphibian population declines, man-made aquatic habitats may alleviate such negative effects. For instance, Moor et al. (2022) showed that the number of occupied man-made ponds increased over time in the Kanton Aargau, Switzerland, for 10 of the 12 amphibian species studied, with one species remaining stable and one species continuing to decline. They suggested that creating a large number of ponds could mitigate amphibian population decline. Furthermore, due to their impermeability, artificial water bodies may maintain water levels longer than small ephemeral water bodies, thus providing important breeding habitats for amphibians, especially during dry periods (Wagner et al. 2025). However, water bodies with short hydroperiods may be beneficial for multiple amphibian species due to reduced interspecific competition and lower predation pressure from invertebrates and fish

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(reviewed in Buxton and Sperry 2017). In such environments, newts, for example, exhibit increased courtship behavior and deposit more eggs (Winandy et al. 2017). Utilizing such breeding habitats is a reproductive strategy particularly favored by species such as the European green toad, *Bufotes viridis* (Zahn and Niedermeier 2004; Landler et al. 2023), or the yellow-bellied toad, *Bombina variegata* (Moor et al. 2022).

Bufotes viridis is a pioneer species of typically ephemeral ponds (natural and man-made), with a broad geographic range from Central Europe to Western Asia. In Austria, it predominantly occurs at low elevations in the east (federal states of Vienna, Lower Austria, and Burgenland), with some regional populations throughout Styria, Upper Austria, and Tyrol, typically at elevations of 200–400 m and up to 1150 m (Cabela et al. 2001; Stöck et al. 2008). Since 2000, the range has decreased dramatically overall, and some local populations may have recently disappeared or are considered lost [e.g., in Carinthia in southern Austria (Maletzky 2025; Fig. 1)]. In its natural habitats, *B. viridis* typically spawns in steppe lakes and other open areas filled by rainfall or in puddles in wild-river floodplains. These areas lack vegetative structures such as aquatic plants, which are sometimes used by other amphibian species to lay their eggs (Stöck et al. 2008; Indermaur et al. 2010). Water bodies that result from man-made activities, such as construction sites, vehicle tracks, and surface depressions filled by rainfall or groundwater, are often colonized by *B. viridis* (Kaczmarski et al. 2019; Conan et al. 2022; Landler et al. 2023). We chose *B. viridis* as our umbrella species due to its ecological requirements, high conservation status (included in Annex IV of EU Council Directive 92/43/EEC), and visual attractiveness for science communication.

To test the effectiveness of large-scale establishment of man-made ponds for amphibian conservation, we started an Austria-wide citizen science (CS) experiment. We used a CS approach to both establish new mini-ponds in private gardens and have them monitored for colonization by *B. viridis* and other amphibian species by the citizen scientists. In CS, laypeople (citizen scientists) actively participate in basic and applied research and conservation, such as documenting the distribution or assessing the status of taxa, recording animals killed by road traffic, or predicting amphibian migration (Devictor et al. 2010; Heigl et al. 2017; McKinley et al. 2017; Peer et al. 2021; Fontaine et al. 2022). Recording amphibian species by citizen scientists has been used successfully in previous projects (Lee et al. 2021; Peer et al. 2021; Márton et al. 2025).

Among the advantages of including citizen scientists in ecological research is the possibility to obtain data from areas otherwise difficult to access by scientists, such as private gardens (Egerer et al. 2019). For example, in a recent study by Márton et al. (2025), more than 800 citizen scientists with garden ponds in Hungary participated in an online survey to determine the importance of these small water bodies for biodiversity. They identified characteristics such as pond age, area, and the extent of aquatic and shoreline vegetation as important factors influencing amphibian presence.

However, a common challenge in species monitoring is that not all target individuals present in a given area are observed (imperfect detection). Hence, reliable information about detection probabilities is needed to produce accurate inferences of species' occupancy status, distribution ranges, or abundance estimates. Detection probabilities are influenced by factors such as the target species' biology and behavior, seasons, climatic and weather conditions, and observer skill (Weir et al. 2005). While citizen scientists can produce data of comparable quality to that of professionals (Edgar et al. 2017; McKinley et al. 2017)—particularly as their experience grows (Falk et al. 2019)—studies focusing on amphibian monitoring report that citizen scientists tend to have lower average detection probabilities than professionals (Lee et al. 2021; Schmidt et al. 2023). These findings underscore the necessity of accounting for imperfect detection, including in studies involving CS. Occupancy models are particularly well suited for addressing imperfect detection, as they disentangle the probability of detecting a species from the probability of its true presence through repeated surveys (MacKenzie et al. 2002). Thus, we used Bayesian occupancy models (Doser et al. 2022) to estimate species-specific detection probabilities from our occurrence data, enabling more accurate inferences about the occupancy of our mini-ponds.

We hypothesized that the mini-ponds would be used by amphibians already during the first months after installation. Of the 21 amphibian taxa occurring in Austria, we anticipated that the mini-ponds might be used by common generalists such as *Bufo bufo*, *Pelophylax ridibundus*, and *Lissotriton vulgaris*, as well as pioneer species such as *Bombina variegata* and *Bufotes viridis*. We also expected that reproduction (amplexus, deposition of spawn, developed tadpoles, and metamorphs) might occur in the mini-ponds.

Methods

Choice of mini-ponds

We provided 300 mini-ponds (120 L × 90 W × 40 D cm, 150 L, 0.98 m² surface area, polyethylene, drainable; Ubbink model Iris SI, Outside Living Industries Deutschland GmbH, Bocholt, Germany) to 300 citizen scientists in Austria. Two additional mini-ponds were installed at the Vienna Zoo. Typically, ponds created as a conservation measure for *B. viridis* are shallow (0.3–1 m) but have a larger surface area (> 15 m²), created by using a soft liner as the foundation, with gravel added as a substrate (Rienesl 2017). However, these were impractical for our experiment, as the work required to install them, along with the space requirements, would have greatly limited the number of willing participants and introduced too much habitat variability to allow comparisons.

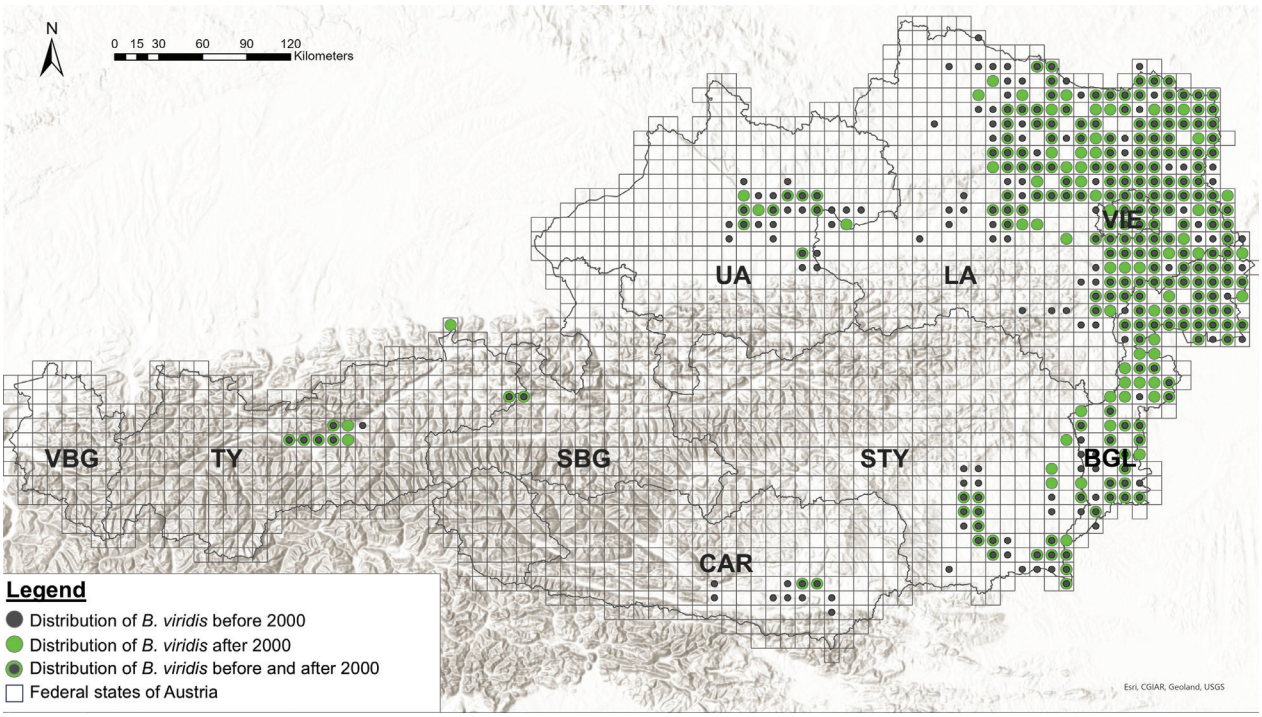

Figure 1. Observations of *B. viridis* in Austria (1970–2025), based on HFDÖ and iNaturalist. Federal states: BGL – Burgenland, CAR – Carinthia, LA – Lower Austria, SBG – Salzburg, STY – Styria, TY – Tyrol, UA – Upper Austria, VBG – Vorarlberg, VIE – Vienna.

There are many examples of green toads spawning in concrete pools without any substrate, of various sizes and depths (e.g., Conan et al. 2022; Gruszka et al. 2024). Still, the survival and development of larvae and tadpoles can be influenced by both the surface area and the volume of the ponds (Pearman 1993). We therefore chose to use the mini-pond model with the largest surface-to-volume ratio that was commercially and readily available to us. To aid animals in exiting the ponds, we specifically chose a design with ribbing on the sides and supplemented this by providing a coconut-fiber mesh to be attached on the side (Fig. 2). We asked the citizen scientists not to add any substrate (e.g., soil, gravel, sand, or stones), plants, or wood to the pond in order to create comparable early successional oligotrophic habitats and to refill the mini-ponds regularly with water to ensure sufficient water levels (Fig. 2A, B).

Selection of mini-pond locations

We prioritized locations within the known distribution of *B. viridis* in Austria after 2000, based on information from the Austrian Herpetofauna Database (HFDÖ), unpublished data from the authors, and the nonprofit social network iNaturalist (Fig. 1). We provided citizen scientists with six ponds within areas of likely extirpated populations, such as in Carinthia.

We allocated mini-ponds to citizen scientist applicants, who placed them in accessible, amphibian-friendly locations. To assess potential localities, we examined satellite images focusing on impassable barriers and the availability of nearby water bodies. We also asked the applicants whether they already had a pond in their garden and whether they had observed amphibians on their property. Thus, we selected only applicants whose properties were accessible to amphibians. We delivered the ponds between November 2023 and February 2024. The citizen scientists were asked to install the ponds in their gardens by 1 March 2024.

Monitoring

Citizen scientists monitored their mini-ponds and recorded observed amphibians using a custom-designed smartphone application (“AmphiApp”), freely available for Android and iOS. The application automatically uploaded data to a web-integrated database. Citizen scientists were instructed to monitor their ponds every second weekend from 2 March 2024 until 1 September 2024. Each monitoring record included the user ID, the relative water level in the mini-pond and whether water was added, the date/time of the observation, the geographical coordinates, a tentative species identification, a description of the observation, a photograph, and optionally an audio recording of vocalizing amphibians. Data were compiled into a database and validated by us. We considered an amphibian observation to be associated with a mini-pond if the animal was recorded within five meters of the pond. However, such cases were rare; generally, the animals were documented in the mini-pond. Furthermore, we encouraged unscheduled visits to report additional amphibian observations, so we received records until the end of October.

Figure 2. Example mini-ponds in March 2024 (**A**) and four months later, with visible algal growth (**B**). Note the ribbed sides and the coconut-fiber mesh.

We attempted to identify each recorded amphibian to the species level. However, the sometimes low quality of single photographs (e.g., lacking distinct species characteristics) and poor audio recordings precluded identification at that level for some records. We grouped all *Pelophylax* spp. (*P. lessonae*, *P. ridibundus*, and *P. esculentus*) into one category due to their morphological similarity, which precludes unambiguous distinction, and treated them as one taxon for the rest of the manuscript. Brown frogs (*Rana temporaria*, *R. dalmatina*, and *R. arvalis*) that could not be clearly identified were assigned to *Rana* spp. Some tadpoles and some frogs could only be identified as Anura.

Detection probabilities

To estimate species-specific detection probabilities, we fitted single-species occupancy models in a Bayesian framework using the *PGOcc()* function in the *spOccupancy* R package (Doser et al. 2022). We restricted the dataset in two steps. First, we retained only records from the main sampling period when citizen scientists were instructed to regularly monitor their mini-ponds, plus an additional week at the beginning and end (24 February to 7 September). This timeframe was divided into 14 two-week intervals, with the regular monitoring dates positioned at the midpoint of each interval. This approach ensured that samples collected a few days early or late were still attributed to the same sampling date. Second, we applied a species-specific restriction by defining a core activity period for each species, corresponding to its phenological window when individuals are typically found in or near water for reproduction (Suppl. material 1: fig. S1; Nöllert and Nöllert 1992a; Cabela et al. 2001). Only data from within these core activity periods were used for model fitting (MacKenzie et al. 2002). Data outside these periods were excluded to minimize heterogeneity in detection probability and to avoid potential violations of the closure assumption from emigration. Species were counted as present or absent in each interval, with multiple detections within the same interval counted only once. Furthermore, we retained only species detected in ≥ 10 intervals across all sites during their core activity period. We applied this threshold to ensure adequate data for reliable detection probability estimates (MacKenzie et al. 2005). Three species (*L. vulgaris*, *B. viridis*, and *B. variegata*) and *Pelophylax* spp. fulfilled this criterion.

We assessed model adequacy using posterior predictive checks (Gelman et al. 2025), and MCMC convergence was evaluated using the Gelman–Rubin diagnostic ($\hat{R}$; Gelman and Rubin 1992). Initial model formulations included a ran-

dom intercept for detection across sites, but convergence diagnostics ($\hat{R} > 1.1$) indicated poor mixing for some species. To avoid biased or unstable estimates, we therefore fit intercept-only models without covariates or random effects (occ. formula = ~1; det.formula = ~1). For each model, we ran three chains, from which we collected 10,000 posterior samples each. The first 2,000 samples per chain were discarded as burn-in, and we kept every draw (thinning interval = 1). We used default priors ($\mu = 0$; $\sigma^2 = 2.72$).

Per-interval detection probability estimates and 95% Bayesian credible intervals were obtained directly from posterior samples. We additionally computed cumulative detection probabilities over all intervals per site as

$$p_c = 1 - (1 - p)^K,$$

where p is the per-interval detection probability per site and K is the number of intervals (MacKenzie et al. 2005). Finally, we conducted Bayesian pairwise comparisons to quantify the posterior probability that detection probabilities (both p and p_c) differed between species. These comparisons calculated a mean and 95% credible intervals for the difference between the posterior samples of all species pairs. They also reported the probability that the difference between the mean posterior samples of a species pair was larger than zero (P (> 0)). We interpreted posterior probabilities P (> 0) > 0.95 or < 0.05 as statistically significant differences in the Bayesian sense, corresponding to strong evidence that one species had a higher detection probability than the other.

Results

Of the 302 mini-ponds, 289 were set up (confirmed by photographs), as some participants dropped out of the experiment immediately. Even though we provided instructions on how to prepare the mini-ponds, initial pond conditions varied (e.g., some people added substrate, plants, or multiple tree branches, or did not place the coconut-fiber mesh properly). We were able to contact most participants and resolved most problems by the start of sampling. Most mini-ponds were installed and filled with water in the first week of March 2024.

Overall, citizen scientists detected amphibians in 38% (n = 111) of the 289 mini-ponds (Fig. 3). Considering the geographical regions of Austria (Lehner et al. 2025), most mini-ponds (n = 157) were located in Northern Austria (purple), where amphibians (n = 23) were found in 15% of all sites. In the Southeastern Alpine Foothills (light blue; n = 6), amphibians were found in 26% of all mini-pond sites (n = 23). In the Northeastern Basins (red; n = 23), amphibians were found in 32% of all mini-pond sites (n = 71). In the Eastern Alps–West (green), where we had only nine mini-ponds, three were visited by amphibians (33%).

Overall, we collected 353 records, including observations of reproductive activity (amplexus, presence of eggs, larvae, or metamorphs), comprising 13 amphibian taxa (Table 1). Records were unevenly distributed among species. Of the 353 reports from 111 mini-ponds, 42.3% were of *Pelophylax* spp. (n = 107,

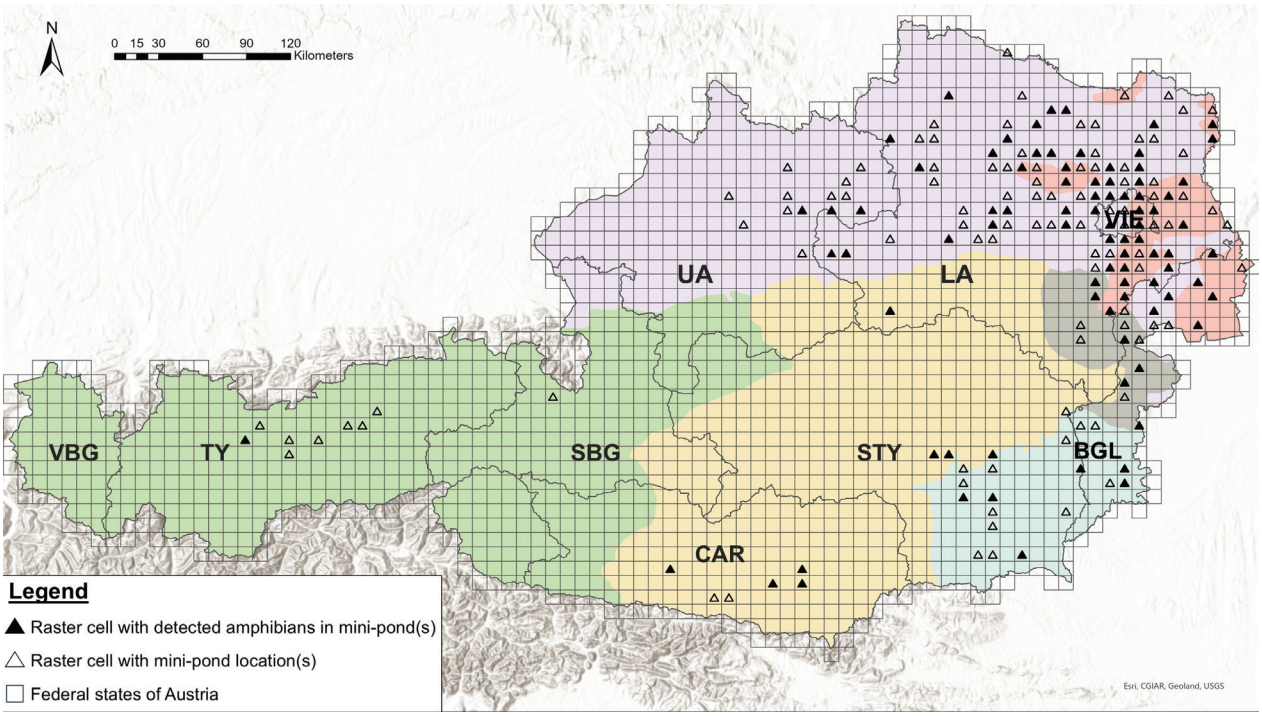

Figure 3. Mini-pond sites in Austria and the amphibians recorded there in March–October 2024. BGL – Burgenland, CAR – Carinthia, LA – Lower Austria, SBG – Salzburg, STY – Styria, TY – Tyrol, UA – Upper Austria, VBG – Vorarlberg, VIE – Vienna. The coloration refers to five subregions of Austria: Green – Eastern Alps-West, Yellow – Eastern Alps-East, Light Blue – Southeastern Alpine Foothills, Purple – Northern Austria, Red – Northeastern Basins, and Grey – no clear assignment; modified after Lehner et al. (2025).

from 47 ponds), followed by *B. viridis* (15.3%, n = 59, from 17 ponds) and *B. bufo* (12.6%, n = 17, from 14 ponds). *Lissotriton vulgaris* (n = 27) was found in 12 ponds, and *R. dalmatina* (n = 15) was found in 13 ponds. *Rana temporaria*, *B. bombina*, *B. variegata*, *H. arborea*, *Mesotriton alpestris*, and *T. carnifex* were observed in fewer than eight mini-ponds each. Additionally, in 23 of the 111 mini-ponds, we collected records of amphibians that could only be classified within the order Anura. The number of records per month generally followed the expected pattern of aquatic activity for amphibians, with most records made between May and September.

Citizen scientists recorded reproductive activity in 29 (10%) of 289 mini-ponds. Reproductive activity was detected for eight of the 13 identified taxa: *L. vulgaris* (six of 12 mini-ponds with records of the species), green frogs (5/47), *B. viridis* (3/17), *B. bufo* (2/14), *B. variegata* (2/5), *H. arborea* (1/4), *R. temporaria* (1/6), and *M. alpestris* (1/2). We could not confirm reproduction of *R. dalmatina*, *B. bombina*, or *T. carnifex*. Additionally, in 19 of the 29 mini-ponds, we observed signs of reproduction that could only be classified within the order Anura.

Detection probabilities

Estimated per-interval detection probabilities (p) were generally low, ranging from 0.107 (CI: 0.052–0.176) for *B. viridis* to 0.154 (CI: 0.062–0.274) for *B. variegata* (Fig. 4A). Estimated cumulative detection probabilities over the respective species' core activity periods (p_c) ranged from 0.658 (CI: 0.413–0.855) for *B. viridis* to 0.836 (CI: 0.659–0.947) for *L. vulgaris* (Fig. 4B). We did not find significant differences between species for both p and p_c (Suppl. material 1: tables S1, S2). All four species-specific occupancy models showed adequate fit in posterior predictive checks (PPC Bayesian p-values ranged from 0.28 to 0.38), indicating no evidence of lack of fit. Convergence diagnostics were satisfactory for all models, with all $\hat{R} < 1.1$. Of the ten species identified during the main sampling period (including *Pelophylax* spp.; *T. carnifex* was not detected), six were observed exclusively during their respective core activity periods, while four were also detected outside the core activity period (*B. bufo*, *B. viridis*, *R. dalmatina*, and *R. temporaria*) (Suppl. material 1: fig. S1).

Table 1. Records of amphibians documented in 111 of 289 mini-ponds, March–October 2024, throughout Austria. Red List conservation status: VU – Vulnerable, NT – Near Threatened (Gollmann 2007).

Taxa	Red List status	Month								Total
		III	IV	V	VI	VII	VIII	IX	X	
Anura	–	1	12	16	12	18	12	7	0	**78**
Bufo bufo	NT	2	2	3	2	3	1	4	0	**17**
Bufotes viridis	VU	2	3	15	11	6	8	9	5	**59**
***Pelophylax* spp.**	NT/VU	0	4	17	14	11	28	28	5	**107**
***Rana* spp.**	NT/VU	2	3	3	2	3	0	7	0	**20**
Rana dalmatina	NT	2	4	1	0	3	2	3	0	**15**
Rana temporaria	NT	1	0	1	2	0	0	2	0	**6**
Bombina bombina	VU	0	0	0	2	0	1	0	0	**3**
Bombina variegata	VU	0	1	5	3	3	0	0	0	**12**
Hyla arborea	VU	0	2	0	2	0	0	1	0	**5**
Mesotriton alpestris	NT	0	0	0	0	0	1	0	0	**1**
Lissotriton vulgaris	NT	2	6	2	5	4	6	2	0	**27**
Triturus carnifex	VU	0	0	0	0	0	0	1	2	**3**
Total		**12**	**37**	**63**	**55**	**51**	**59**	**64**	**12**	**353**

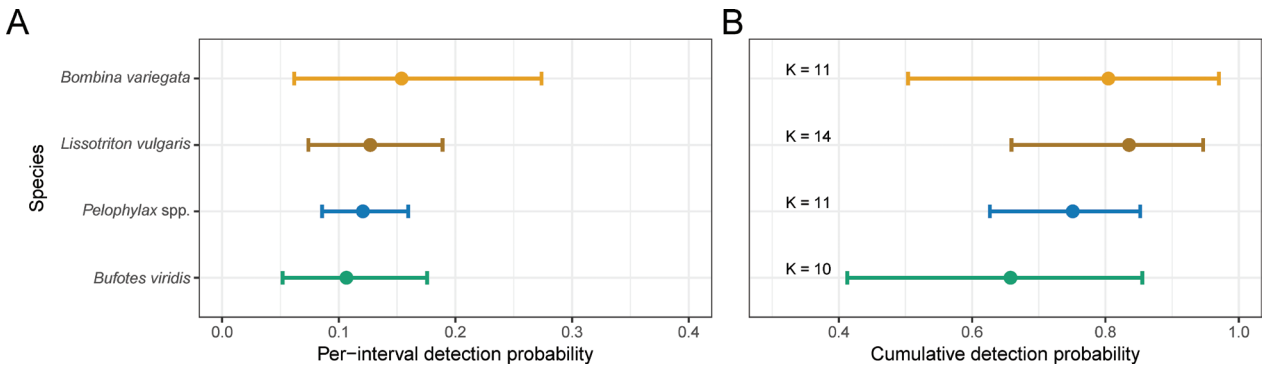

Figure 4. Estimated detection probabilities for the taxa *B. viridis*, *Pelophylax* spp., *L. vulgaris*, and *B. variegata* derived from single-species occupancy models. Points represent posterior means and error bars indicate 95% credible intervals. (**A**) shows per-interval detection probabilities (p), while (**B**) shows the cumulative detection probabilities over the respective species' core activity period (p_c). K indicates the length of each taxon's core activity period, measured as multiples of 14-day intervals.

Discussion

Our citizen science experiment showed that man-made mini-ponds were effective in attracting amphibians, with signs of reproduction in 10% of ponds in the same season they were installed. The specific amphibian occurrences may be partially explained by the geographical distribution of the mini-ponds, which were mostly placed in the lowlands and in eastern Austria. These areas generally support a higher diversity and density of amphibians than higher elevation areas and may therefore be representative for such an endeavor.

Especially highly mobile generalists such as *B. bufo* and *Pelophylax* spp. (Moor et al. 2022), but also less mobile species such as newts (Cayuela et al. 2020), may benefit from the presence of ponds by having more potential breeding habitats available within a shorter distance, compensating for other already occupied potential breeding habitats (Moor et al. 2024). This allows for habitat switching (Winandy et al. 2017), which can be crucial in areas with ongoing landscape changes or anthropogenic pressures (Bounas et al. 2020). In our experiment, *Pelophylax* spp. were the most common amphibian taxa recorded colonizing the newly installed mini-ponds during the first monitoring season, followed by *B. viridis* and *L. vulgaris*. Our results were in line with those of Moor et al. (2022), which indicated a higher probability of colonization in newly created ponds by more common species (e.g., *R. temporaria*, *Pelophylax* spp., *B. variegata*). However, in the aforementioned study, the surveyed ponds were installed over a longer period (up to several decades), while our mini-ponds were installed only a few weeks before the first amphibian records. Fog (2024) indicated highly species-specific movement patterns as one of the reasons why *R. dalmatina* may colonize ponds even faster than *R. temporaria*, which is also in line with our results.

Although our results are limited to a single active season and biweekly sampling by untrained citizen scientists, the detection of amphibians in 38% of the mini-ponds highlights the potential conservation value of even small man-made aquatic habitats. Of the species recorded, six are endangered in Austria and listed in Appendix IV of the Habitats Directive (https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A31992L0043, visited 21 March 2025), and nine are listed as Near Threatened or Vulnerable in Austria's Red List (Gollmann 2007).

While habitat loss affects all species, the decreased dynamics of our landscapes (e.g., channelization of rivers) leads to an overrepresentation of late-succession ponds, which particularly negatively affects pioneer species. In addition, many aquatic habitats are maintained in a relatively stable state by people for their recreational or ornamental functions and therefore may contain many aquatic plants and predatory fish. Given that the mini-ponds were newly established with low predator abundance (e.g., no fish and no last-season dragonfly larvae) but available prey (e.g., mosquito larvae) and algae, we expect a high density-dependent survival of the developing larvae. We advised citizen scientists to drain the ponds in winter and refill them in spring, as described in the instructions provided. Thus, these ponds could provide suitable habitats for a longer period of time. However, even if the mini-ponds were suboptimal breeding habitats, they might be essential for metapopulations by serving as stepping stones within fragmented landscapes, providing connectivity between larger, key breeding locations (e.g., O'Brien et al. 2021). Recolonization requires the establishment of a continuous network of closely placed and connected aquatic and terrestrial habitats and takes time (O'Brien et al. 2021). We therefore anticipate that more species and individual amphibians will find the mini-ponds and that more mini-ponds will be colonized (including for breeding) during the second monitoring season, once animals have had more opportunities to encounter the new potential breeding habitats.

The citizen scientists detected most amphibians in the late spring and early autumn months, which is not entirely consistent with the known peaks of breeding activity of the three main taxa (*Pelophylax* spp., *B. viridis*, and *B. variegata*) detected in our mini-ponds (Nöllert and Nöllert 1992b; Stöck et al. 2008). However, we also noted a slight decline in the number of total reports from citizen scientists in summer (July–August), likely due to people being on vacation.

When accumulating detection probabilities over each taxon's core activity period, detection probabilities increased substantially (0.658–0.836), although the estimates for per-interval detection probabilities were generally low (0.107–0.154; Fig. 4A). These estimates imply that, depending on the taxa, 16–34% more ponds might have been occupied but not detected. However, considering the large credible intervals, the proportion of undetected occupancy might have been as high as 59% for *B. viridis* or as low as 3% for *B. variegata* (Fig. 4B). Our estimates were lower than those reported by Schmidt et al. (2023), who found average per-survey detection probabilities of 0.30 ± 0.16 and cumulative detection probabilities of > 0.9 (number of surveys > 10) for various amphibian species sampled by volunteers (non-professionals). Our lower estimates might be the result of our sampling sites (mini-ponds) being very small. This situation could potentially lead to temporary absences due to roaming behavior, thereby lowering estimated detection probabilities. Schmidt et al. (2023) found little effect of closure and abundance on occupancy model estimates; however, for very small sampling sites, the effects might be larger.

Although our occupancy models included only detections within each taxon's core activity period to reduce potential heterogeneity in detection probability, four species were still detected outside their respective core activity periods during the main sampling period. Combined with the data from 2025, it may be possible to use these data to estimate detection probabilities outside core activity periods, yielding more accurate cumulative detection estimates for the main sampling period.

We did not find any significant differences between taxa regarding their detection probabilities (p and p_c; Suppl. material 1: tables S1, S2), which might be partly due to the large credible intervals. Noticeably, among the four taxa detected often enough to fit occupancy models, the three taxa with the highest p and p_c estimates are at least partly diurnal (*B. variegata*, *L. vulgaris*, and *Pelophylax* spp.). Most nocturnal species were detected during fewer than 10 intervals across all ponds, likely because citizen scientists conducted most of their sampling during the day. Only *B. viridis* was detected more frequently, likely because it was our target species and most mini-ponds were located within its distribution range. Future studies should instruct citizen scientists to sample equally during the day and night to improve detection probabilities for nocturnal species.

We were unable to fit models for six of the ten taxa detected during the main sampling period (excluding Anura and *Rana* spp.) due to insufficient detections. This may be due to the tendency of the citizen scientists to sample primarily during the day, to specific aquatic habitat preferences not met by the mini-ponds, or to too few ponds located within the respective species' distribution ranges. Additionally, explosive breeders such as *B. bufo* and the three *Rana* spp. stay in breeding waters for only a few days to a maximum of several weeks, limiting the period in which they could be detected within our mini-ponds and thus lowering cumulative detection probability. For that reason, citizen science approaches with more frequent sampling events during the respective activity periods would be better suited for these species.

Conclusion

Our experiment shows that the installation of man-made mini-ponds may provide suitable breeding habitats for amphibians. Moreover, the mini-ponds used in this experiment are easy to obtain, install, and maintain. A properly installed mini-pond could last for multiple years, does not take up much space in a garden, and is therefore an accessible option for people who are willing to get involved in conservation efforts but have limited resources. Given the habitat preferences of many amphibians, CS-driven initiatives can have a conservation impact, especially if scaled up and implemented as part of comprehensive amphibian protection measures. We therefore recommend the use of man-made mini-ponds in conservation and, to a limited degree, monitoring strategies, but we emphasize the importance of extensive communication when working with citizen scientists to ensure the success of such an endeavor. Our first results are promising, and the second season of this experiment should show whether the results can serve as a guide for future conservation efforts and research into man-made breeding ponds for amphibians.

Acknowledgments

We are indebted to all the citizen scientists who actively participated in our experiment, without whom data collection at this scale would not have been possible. We thank Alejandro López-de Sancha and Benedikt Schmidt for providing valuable suggestions that improved the manuscript. We thank Christoph Leeb for his advice and help with data analysis. This research was funded by the Biodiversity Fund of the Federal Ministry of Austria for Agriculture, Forestry, Climate and Environmental Protection, Regions, and Water Management, and Next Generation EU (project no. C321025).

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Supplementary material 1

Additional figure and tables

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Data type: pdf

Explanation note: **table S1.** Pairwise comparison of posterior per-interval detection probabilities between taxa. **table S2.** Pairwise comparison of posterior cumulative detection probabilities between species. **fig. S1.** Core activity periods for each taxon detected in the mini-ponds.

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