Amphibian communities are structured by local habitat quality in garden ponds and spatial factors in urban ponds

Márton Uhrin[1,2,3], Barbara Barta[1,2], Beáta Szabó[1,2], Thu-Hương Huỳnh[1,2,3], Irene Tornero[1,2,4,5], *Andrew J. Hamer[1,2], *Zsófia Horváth[1,2]

[1]Institute of Aquatic Ecology, HUN-REN Centre for Ecological Research
[2]National Multidisciplinary Laboratory for Climate Change, HUN-REN CER
[3]Doctoral School of Biology, ELTE Eötvös Loránd University
[4]GRECO, Institute of Aquatic Ecology, University of Girona
[5]Department of Zoology, Faculty of Biology, University of Seville
*equal contribution

Corresponding author: Márton Uhrin: uhrin.marton@ecolres.hu

ORCID iDs

Márton Uhrin: 0009-0005-5584-1063
Barbara Barta: 0000-0002-6674-3275
Beáta Szabó: 0000-0002-2972-0306
Thu-Hương Huỳnh: 0009-0005-7524-205X
Irene Tornero: 0000-0001-7337-4263
Andrew J. Hamer: 0000-0001-6031-7841
Zsófia Horváth: 0000-0003-2857-1094

Author contribution statement

Márton Uhrin: Data curation, Formal analysis, Visualization, Writing - original draft, Writing - review & editing. **Barbara Barta**: Conceptualization, Data curation, Investigation, Project administration, Writing - review & editing. **Beáta Szabó**: Investigation, Writing - review & editing. **Thu-Hương Huỳnh**: Writing - review & editing. **Irene Tornero**: Investigation, Writing - review & editing. **Andrew J Hamer**: Conceptualization, Data curation, Funding acquisition, Investigation, Project administration, Writing - original draft, Writing - review & editing. **Zsófia Horváth**: Conceptualization, Formal analysis, Funding acquisition, Investigation, Project administration, Resources, Writing - original draft, Writing - review & editing.

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Conflict of interest statement

The authors declare no conflict of interest.

Data availability statement

The data associated with the manuscript will be available via the Figshare repository at https://doi.org/10.6084/m9.figshare.31812748. Pond coordinates are available from the authors upon reasonable request. To protect the privacy of private property owners participating in the study, the coordinates of private ponds are provided only at reduced spatial precision.

Abstract [182 words]

Amphibians are among the most threatened vertebrates, and urbanisation contributes to their decline through habitat loss, fragmentation, pollution, and the spread of invasive species. At the same time, urban freshwater habitats, such as ponds, can serve as important refuges within highly modified landscapes. While the role of urban ponds in supporting freshwater biodiversity is increasingly recognised, the contribution of privately managed garden ponds remains poorly understood. Here, we compared amphibian communities in 90 ponds across

Budapest, Hungary, including privately managed garden ponds and other urban ponds, to evaluate the relative importance of local environmental conditions, landscape context, and spatial connectivity. Amphibian communities differed less in taxonomic composition than in the processes structuring them. Garden pond communities were primarily shaped by local environmental filtering, whereas spatial configuration within the pond network played a stronger role in structuring communities in larger urban ponds. These results highlight contrasting mechanisms of community assembly across urban pond types. Urban amphibian conservation can therefore benefit simultaneously from local stewardship of garden ponds and strategic planning of connected urban pond networks that reduce dispersal barriers across the urban landscape.

Keywords: biodiversity; urban ecology; conservation; freshwater ecology; metacommunity

Introduction

Urbanisation is an inevitable outcome of rapid population growth and associated economic development (Seto, Güneralp & Hutyra, 2012; United Nations, 2022). However, it remains one of the major drivers of biodiversity loss worldwide (McKinney, 2002; Oertli & Parris, 2019; McDonald *et al.*, 2020). Urban development exerts multiple stressors on ecosystems, including habitat loss and fragmentation, pollution, altered hydrology, the spread of invasive species, and increased human disturbance (Dudgeon & Strayer, 2025). These stressors modify the physical environment in urban ecosystems and can act as strong abiotic filters for the local biota (Grimm *et al.*, 2008; Francis & Chadwick, 2013).

Urban ponds have been increasingly recognised for their potential to conserve biodiversity within cities, acting as valuable refuges and contributing to urban ecological networks (Hassall, 2014; Krivtsov *et al.*, 2022). These small water bodies can support diverse taxa (Oertli *et al.*, 2002; Hill *et al.*, 2017), including amphibians, which often rely on them for breeding, dispersal, and shelter within otherwise fragmented urban landscapes (Parris, 2006). Different types of urban ponds - such as stormwater ponds (Brand & Snodgrass, 2010; Scheffers & Paszkowski, 2013), drainage ponds (O'Brien, 2015), and garden ponds (Banks & Lavericki, 1986; Butterworth, Van Helden & Close, 2025) - can all provide important habitats for amphibians, even though they differ strongly in management regimes and environmental conditions (Oertli & Parris, 2019; Hassall, 2014; Horváth *et al.*, 2025). Urban ponds may therefore function as critical components of conservation networks, supporting metapopulation dynamics and providing refugia for threatened taxa in heavily modified landscapes where many natural aquatic habitats have been lost.

Urban ponds found in parks, natural reserves or other publicly accessible areas within an urban landscape generally tend to be larger, older, and more heterogeneous (Hassall, 2014; Peeters *et al.*, 2023), whereas garden ponds, as their specific subtype, tend to be smaller, younger, and subject to more intensive management by their owners (Loram *et al.*, 2011; Hassall, 2014; Hamer *et al.*, 2024). Urban ponds differ not only in their physical and environmental characteristics but also in their origin, ownership, and management, creating distinct ecological settings within the same urban matrix. Current understanding of biodiversity in urban pond networks remains strongly biased toward larger, publicly accessible water bodies, whereas privately managed garden ponds are still understudied, despite representing a substantial proportion of potential amphibian habitat in urban areas (Horváth *et al.*, 2025). Garden ponds can also increase habitat connectivity in the urban landscape and provide aquatic refugia for multiple species during droughts, thereby stabilising metapopulation dynamics within urban landscapes.

Pond-breeding amphibians are among the most endangered vertebrate groups, with approximately 41% of species currently threatened by habitat loss and climate change (Luedtke *et al.*, 2023). Beyond their conservation status, amphibians play key ecological roles in freshwater and terrestrial ecosystems by regulating invertebrate populations and mediating energy and nutrient fluxes between aquatic and terrestrial habitats, thereby contributing to ecosystem structure and function (Hocking & Babbitt, 2014). Moreover, because of their permeable skin, biphasic life cycles, and limited dispersal ability, amphibians are widely regarded as sensitive indicator taxa for assessing habitat quality, pollution, and functional connectivity in freshwater networks, including urban pond systems (Estes-Zumpf *et al.*, 2022). Urbanisation generally exerts negative effects on amphibian assemblages, with increasing levels of urban intensity often leading to declines in both species richness and functional diversity (Pereyra *et al.*, 2021; Demartin, Ghirardi & Lopez, 2024). Their dependence on both aquatic and terrestrial habitats makes them particularly sensitive to multiple urban stressors operating across spatial scales (Hamer & McDonnell, 2008; Smallbone, Luck & Wassens, 2011).

Amphibian occupancy is typically shaped by factors across multiple spatial scales due to their lifestyle: at the local scale, pond size, vegetation cover, water quality and predation pressure play a key role (Hamer & Parris, 2011; Falaschi *et al.*, 2020; López-de Sancha *et al.*, 2025). Landscape-scale features, including green space availability or the proximity to other ponds or aquatic habitats, affect dispersal and long-term persistence within habitat networks (Collins & Fahrig, 2017; Zhang, Li & Yang, 2022). In addition, amphibians often rely on temporary ponds as stepping-stone habitats that facilitate movement between suitable breeding sites across otherwise inhospitable matrices (Fortuna, Gómez-Rodríguez & Bascompte, 2006). In urban ponds, human management can further modify local conditions by regulating vegetation, fish or macroinvertebrate populations, or applying chemical treatments (Maerz, Blossey & Nuzzo, 2005; Oertli & Parris, 2019; Márton *et al.*, 2025). These local effects interact with landscape-scale urbanisation processes that reshape ecological dynamics and constrain habitat quality for amphibians (Kurz *et al.*, 2014; Mendenhall *et al.*, 2014; Thompson, Nowakowski & Donnelly, 2016). Despite growing research on urban freshwater systems, it remains unclear whether amphibian communities in different types of urban ponds are structured by similar environmental and spatial processes. Comparative studies disentangling environmental and spatial drivers across different types of ponds remain scarce, limiting evidence-based management in urban landscapes.

Here, we present the first comparative study of amphibian communities in privately managed garden ponds and other urban ponds (in parks and nature reserves), evaluating how local habitat quality and spatial connectivity jointly structure urban amphibian metacommunities. Using a city-wide dataset from Budapest, Hungary, we tested whether amphibian species richness and community composition differ between pond types and whether the relative importance of environmental and spatial drivers varies between them. As garden ponds are typically smaller and more intensively managed than other urban ponds, we expected stronger local environmental filtering in these systems. However, given that amphibians frequently move among ponds during their life cycle, we expected spatial connectivity within the pond network to influence amphibian occurrence in both pond types. Understanding these mechanisms can inform conservation strategies that integrate local habitat management with landscape-scale connectivity in urban environments.

Methods

Site selection

Garden ponds were selected from the participant pool of the citizen science project called MyPond (https://mypond.hu/) (Hamer *et al.*, 2024; Márton *et al.*, 2025; Huỳnh *et al.*, 2025), all found within a 36 km radius of the centre of Budapest, covering a substantially larger area than the city. The level of urbanisation around each pond was quantified as the proportion of urban land cover (i.e., urban area including buildings and roads as well as other artificial surfaces but not urban green spaces) within a 1 km radius, extracted from the Ecosystem Map of Hungary (project KEHOP-430-VEKOP-15-2016-00001, Ministry of Agriculture, 2019). This index was used solely for the stratified selection of garden ponds. The level of urbanisation surrounding these ponds differed substantially; therefore, we divided the full gradient into ten equal-width urbanisation categories (0-1 in 0.1 increments). We then randomly selected 6 ponds from each of the ten categories using the 'ddply' function from the 'plyr' package (v1.8.9; Wickham, 2011a) and contacted their owners. As not all owners responded positively, we contacted additional owners, which eventually resulted in 40 garden ponds.

In parallel, we compiled an inventory of publicly accessible urban ponds across the metropolitan area of Budapest. The selection was based on field surveys and aerial imagery. In addition, several ponds located within protected areas were included following consultations with local nature conservation officers. Altogether, 54 urban ponds were identified. Although some small or temporary ponds may have been overlooked, the dataset represents the vast majority of urban ponds in Budapest (**Figure 1**).

Upon investigating the whole dataset compiled of 40 garden and 54 urban ponds, we decided to exclude four ponds from the analysis. Two garden ponds were identified as agricultural farmland ponds rather than actual ornamental garden ponds. From the urban pond subset, one was excluded due to the lack of sampling permission, and another was also removed because of its substantially smaller surface area (7 m^2) compared to the rest (**Table S6**). This resulted in a final dataset of 90 ponds in total, comprising 38 garden and 52 urban ponds.

Field surveys and laboratory measurements

Each pond was sampled once between April 4th and May 17th, 2022. At the beginning of each survey, a 5-minute pre-disturbance acoustic survey was conducted to detect any anuran species calling. Then, dipnetting was carried out with a net (mesh size 500 µm) to sample the water body for larval stage anurans and newts. The number of sweeps was scaled to pond surface area (**Table S1**) and allocated proportionally across microhabitats. We recorded species presence and counted all visually encountered individuals during the surveys, including larvae and adults. We recorded the presence of fish and the abundance of macroinvertebrates per litre obtained during aquatic sampling. Macroinvertebrates were preserved in alcohol (70%) and kept at 18-20°C until sorting and identification to the lowest taxonomic level in the laboratory.

Pond substrate (natural, concrete, plastic or rubber) was recorded, and data on approximate age were collected from the pond owners. The percentage cover of water level, emergent, submerged, fringing and floating vegetation, surface algae, open water, canopy cover and woody debris was estimated visually at each pond. Pond depth was determined based on information from their owners. The area of small ponds was measured with a measuring tape, while the area of larger ponds was delineated and calculated using GIS (QGIS Development Team, 2022). On-site measurements of physical and chemical parameters included temperature, dissolved oxygen concentration (Hanna HI 9142N probe), pH, and electrical conductivity (WTW 3620 IDS probe).

Chlorophyll-a fluorescence was measured with an AquaPen (AP 100-C, Photon System Instruments). Water samples were collected from each site within 1 m of the shoreline for chemical analysis. We used a Multi N/C 3100 TC-TN analyzer (Analytik Jena, Germany) to measure total organic carbon (TOC) and total nitrogen concentrations (TN). We used standard spectrophotometric methods following the protocols of Eaton et al. (2005) to measure the concentration of total phosphorus (TP) and additional water chemistry variables (major ions, metalloids, and heavy metals) (**Table S3**).

Landscape and spatial data

We used the Ecosystem Map of Hungary (project KEHOP-430-VEKOP-15-2016-00001, Ministry of Agriculture, 2019) through QGIS 3.16.14 (QGIS Development Team, 2022) to determine the effects of landscape cover on amphibian communities. A 1-km radius buffer was mapped around each pond, which represents the maximal dispersal distance of most species expected to occur in the study area (Smith & Green, 2005; Kovar *et al.*, 2009). Data on landscape cover was extracted using the Zonal Histogram tool by counting the number of 20x20 m raster pixels of different landscape categories within the 1-km buffers. We recorded the proportions of the urban land cover (tall buildings, short buildings, other artificial surfaces, impermeable roads, railways), dirt roads, urban green space, agricultural area, grassland, forest, and standing and flowing water (wetland, rivers) around the ponds. Tall, short buildings and other artificial surfaces were merged into a single urban area variable, while impermeable roads and railways were merged into the roads variable. Dirt roads were kept as a separate variable since they are more permeable to amphibians than concrete roads or railways. We extracted human population density from the JRC-GEOSTAT 2018 population grid (Joint Research Centre, European Commission, 2021) using the Zonal Statistics tool to characterise anthropogenic pressure around ponds, using population data from April 2021.

To measure the effects of pond connectivity within the urban landscape, we calculated closeness centrality and nearest-neighbour distance. We calculated the closeness centrality of each pond based on the spatial distance matrix among all ponds, thereby quantifying each pond's relative connectedness within the pond network. For this, we used the 'igraph' (v2.1.1; Csárdi et al., 2026), 'fields' (v16.3; Nychka et al., 2021) and 'reshape2' (v1.4.4; Wickham, 2007) packages of the R software. Nearest-neighbour distance (distance to the closest pond) was calculated using the 'fields' package (v16.3; Nychka et al., 2021). To calculate both indices, we used a pond network with a larger dataset of 384 ponds from a citizen science program (Hamer *et al.*, 2024). We then extracted the corresponding index values for the 90 sampled ponds used in the subsequent analysis.

To detect spatial patterns in the data, we used Moran's Eigenvector Maps (MEM) generated with the R package 'adespatial' (v0.3.24; Dray et al., 2024). We only retained MEM eigenvectors with positive autocorrelations as these represent broad- to intermediate-scale spatial structures typically linked to ecological processes such as dispersal and spatially structured environmental variation, whereas negative MEMs primarily capture fine-scale noise or checkerboard patterns that are difficult to interpret ecologically (Dray, Legendre & Peres-Neto, 2006). MEMs were calculated for the three datasets (for all ponds, only for garden ponds, and only for urban ponds), matching the community datasets (i.e., we only used locations of garden ponds for the garden pond dataset), resulting in three different sets of MEMs for the three datasets.

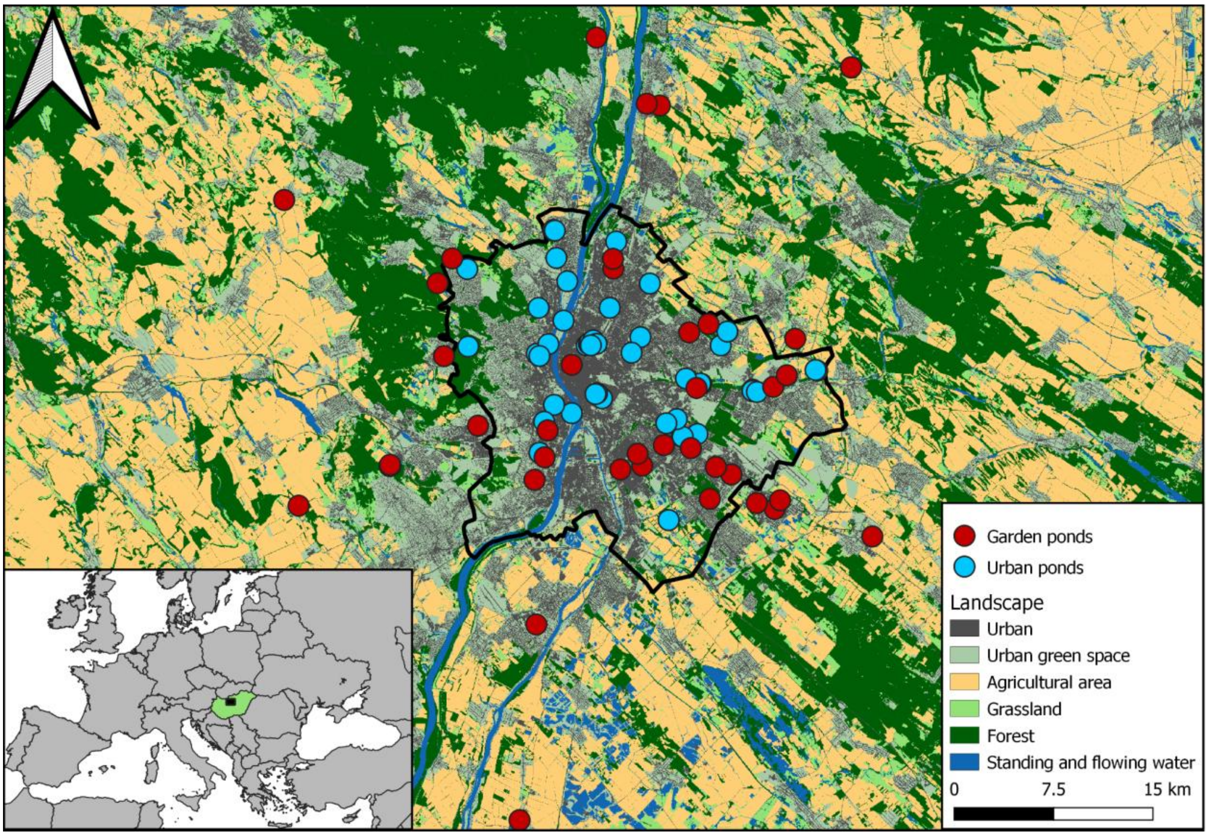

Figure 1: Location of the study sites within the urban landscape of Budapest, Hungary. The map illustrates the spatial distribution of surveyed garden and urban ponds across the metropolitan area, providing the spatial context for subsequent environmental and spatial analyses.

Response variable preparation

Species data of all detected larvae and adult individuals were converted to presence–absence (0/1) before analysis. Sites at which no amphibian species were detected were removed. Multivariate community analyses were conducted on the remaining 14 garden and 19 public ponds.

For species-level habitat-use analyses, we employed an analogous modelling framework using all ponds of each dataset. To ensure robust estimation, model convergence and adequate representation, analyses were restricted to species recorded at least four times in each pond type (*Pelophylax* spp., *Rana dalmatina*, *Bufo bufo*, *Bufotes viridis*).

Predictor data preparation

Some of the measured variables were excluded before analysis: water temperature and dissolved oxygen due to high short-term variability. Water depth was removed from the analysis due to having a high proportion of missing values. Some of the other environmental variables (submerged vegetation, surface algae, predatory threat index and fish) were also missing a smaller share of data (1 to 6 out of 61 ponds) due to field or lab measurement issues. Here, missing values were imputed using the ‘mice’ package (v3.16.0; Buuren and Groothuis-Oudshoorn, 2011), employing predictive mean matching (pmm) with five imputations. The imputation model was based on correlations among the remaining environmental variables.

The four substrate categories were represented by three dummy variables, with the rarest category (plastic) used as the reference level and therefore coded as 0 across the other three variables. Fish presence was used as a binary variable. Pond age was transformed into an ordinal variable based on the following formula: ponds aged 1 - 5 years = 1, 6 -10 years = 2, > 10 years = 3.

To account for biotic interactions with predatory macroinvertebrates, we constructed a predatory threat index (Van Buskirk, 2005; Hamer & Parris, 2013). Predator abundances were weighted by their relative risk to amphibian larvae and subsequently summed into a single composite variable (**Table S2**).

To approximate Gaussian distributions (assessed using the Shapiro–Wilk test) for the analysis, variables were subjected to various transformations (**Table S3**). Following transformations, all predictors were standardised using z-score. We ran a principal component analysis (PCA) on the water chemistry variables (ionic components only), extracted the site scores of the first principal axis (PCA1), and proceeded to use these values as a single water chemistry variable for the analysis. Variance inflation factors (VIFs) were used to assess multicollinearity among the predictors independently for each pond subset (**Table S4**). Variables displaying high collinearity (VIF > 5; Kim, 2019) were removed iteratively. All substrate types were removed from the analysis due to having high variance inflating power in all subsequent analyses.

Community analysis

We tested for overall differences in amphibian community composition between garden and urban ponds using a PERMANOVA based on Jaccard dissimilarities using the ‘vegan’ package (v2.6.8; Oksanen et al., 2024). Differences in species richness were tested using a generalized linear model with a quasi-Poisson error distribution, which relaxes the equidispersion assumption of the Poisson model. Pond type was included as a categorical predictor in all models.

To identify the environmental and spatial drivers of amphibian community composition, predictors were grouped into four categories: environment, landscape, connectivity and spatial autocorrelations (MEMs). We conducted distance-based redundancy analysis (db-RDA) using Jaccard dissimilarities to assess the proportion of community variation explained by each predictor group. Then, we applied permutation-based stepwise model selection (‘ordistep’; 5000 permutations; bidirectional) to identify significant predictors using the ‘vegan’ package (v2.6.8; Oksanen et al., 2024). Due to the lack of collinearity (VIF > 5; Kim, 2019), connectivity and MEM variables were merged into a single category, space for subsequent variance partitioning. Using these significant predictors, variance components were partitioned among environment, landscape, and space using Jaccard dissimilarity as species distances with the varpart() function of the ‘vegan’ package, and final diagrams were produced using the ‘VennDiagram’ package (v1.7.3; Chen, 2022). These analyses were performed separately for garden ponds, urban ponds, and the combined dataset.

Species analysis

To analyse species occurrence, we first used random forests with the ‘randomForest’ package (v4.7.1.2; Liaw and Wiener, 2002) to rank predictor importance and reduce dimensionality, thereby minimising overfitting. Following this pre-selection step, variance inflation factors with the ‘car’ package (v3.1.3; Fox and Weisberg, 2019) were calculated to diagnose collinearity. Due to the prescreening effect of the random forest, high VIF values were limited to a subset of landscape predictors, in which case either urban area, roads, or agricultural area was removed, depending on which exhibited the highest VIF (**Table S4**).

Then, we applied a similar framework using distribution-based generalized linear models to identify the key drivers of species occurrence. We fitted binomial GLMs using the ‘MASS’ package (v7.3.60.2; Venables and Ripley, 2002) with the selected predictors. To assess the independent contribution of each predictor, we conducted likelihood ratio Type II analyses of deviance using the ‘car’ package (v3.1.3; Fox and Weisberg, 2019). When complete or quasi-complete separation was detected, we refitted models using Firth’s bias-reduced penalized likelihood logistic regression using the ‘logistf’ package (v1.26.1; Heinzer et al., 2010; Firth, 1993) to obtain stable parameter estimates. As in the community-level analysis, significant variables were assigned to environmental, landscape, connectivity, and MEM groups; connectivity and MEMs were later merged into a space group for comparability. Variance partitioning was conducted using ‘vegan’, applying Euclidean distances on the binary data, appropriate for univariate responses. Venn diagrams were produced using the ‘VennDiagram’ package (v1.7.3; Chen, 2022).

All the statistical analyses were carried out using the software R (v4.4.1; R Core Team, 2024), plots were created using the ‘ggplot2’ package (v4.0.0; Wickham, 2011b).

Results

We detected 1465 individuals of eight species in the 90 ponds: *Bufo bufo* (1170), *Bufotes viridis* (122), *Pelophylax* spp. complex (101), *Rana dalmatina* (43), *Hyla arborea* (11), *Bombina bombina* (10), *Lissotriton vulgaris* (5) and *Pelobates fuscus* (3). In garden ponds, we recorded 207 individuals of four species, while in urban ponds, we recorded 1272 individuals of eight species. Species richness differed significantly between pond types (quasi-Poisson GLM: t = 2.71, p = 0.011), with higher richness in urban ponds (**Figure 2**).

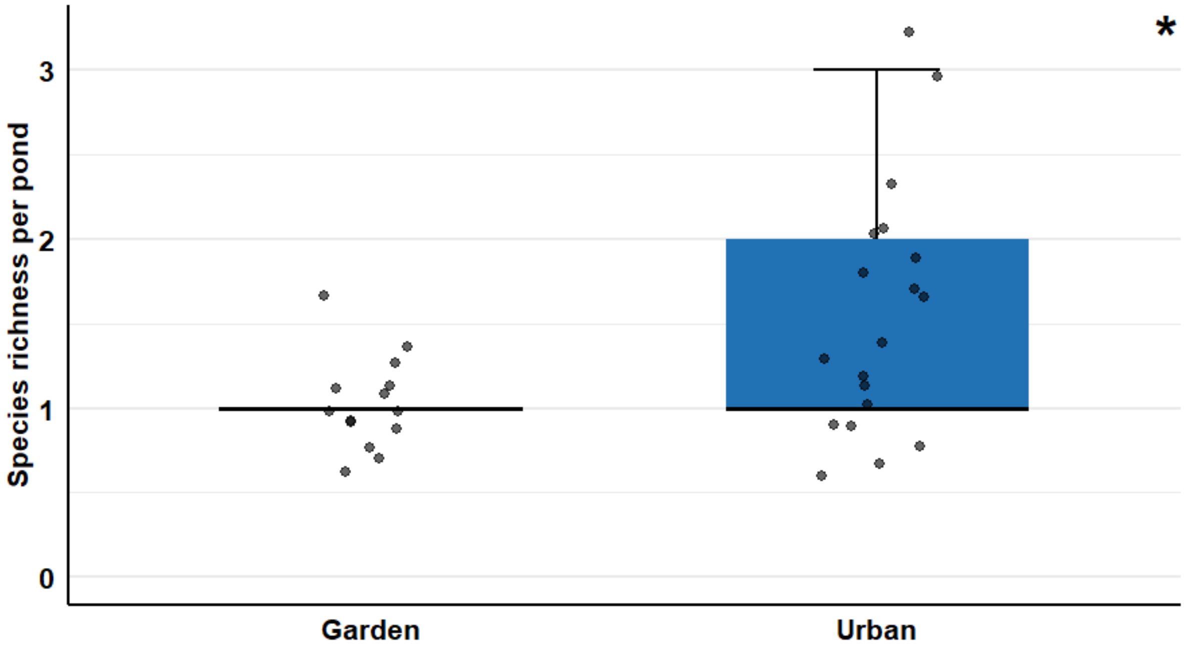

Figure 2: Boxplots displaying species richness for garden and urban ponds. An asterisk indicates the significant difference between the two habitat types based on the generalized linear model.

Pelophylax spp., *R. dalmatina*, *B. bufo*, and *B. viridis* were detected both in garden and urban ponds, while *L. vulgaris*, *B. bombina*, *P. fuscus* and *H. arborea* were only encountered in urban ponds (**Figure 3**). No significant difference was found in community composition between pond types (PERMANOVA, F=1.549, R^2=0.048, p=0.201).

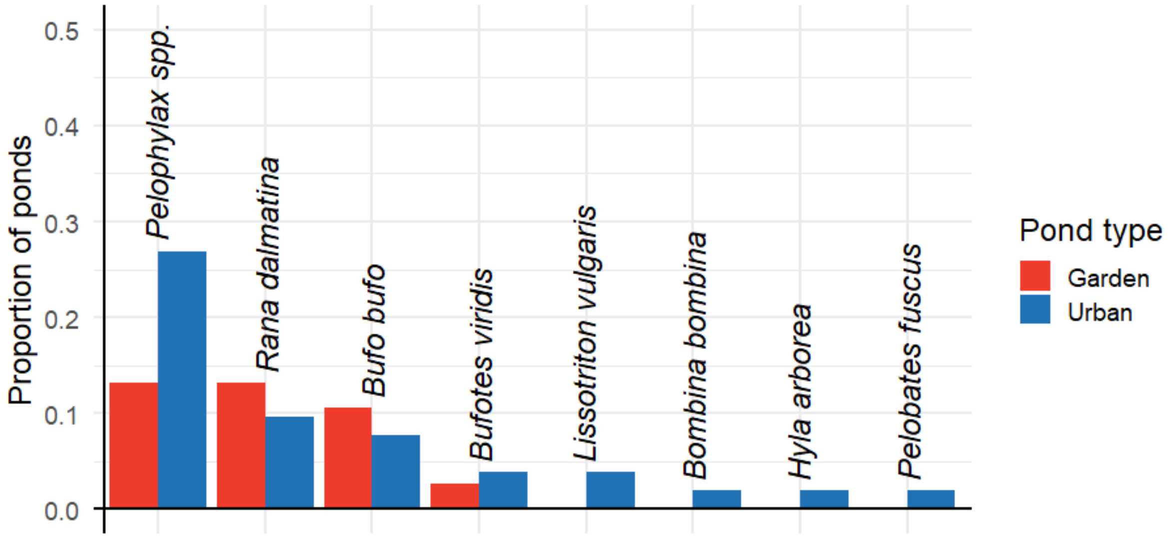

Figure 3: Proportion of ponds in which each species was recorded in garden ponds (red, n=38) and urban ponds (blue, n=52).

Models explained 26% of the total community variance when all ponds were analysed together (**Figure 4**). However, when garden and urban ponds were separated, different patterns emerged in the relative importance of environmental and spatial drivers. In garden ponds, environmental factors explained most of the variance (pure effect: 47%), which included pond area, total organic carbon, submerged vegetation, total phosphorus, and pH (**Figure 4**, **Table S5**). In contrast, spatial variables, including nearest-neighbour distance and MEM6 played a more prominent role in structuring amphibian communities in urban ponds (pure effect: 20%; **Figure 4**).

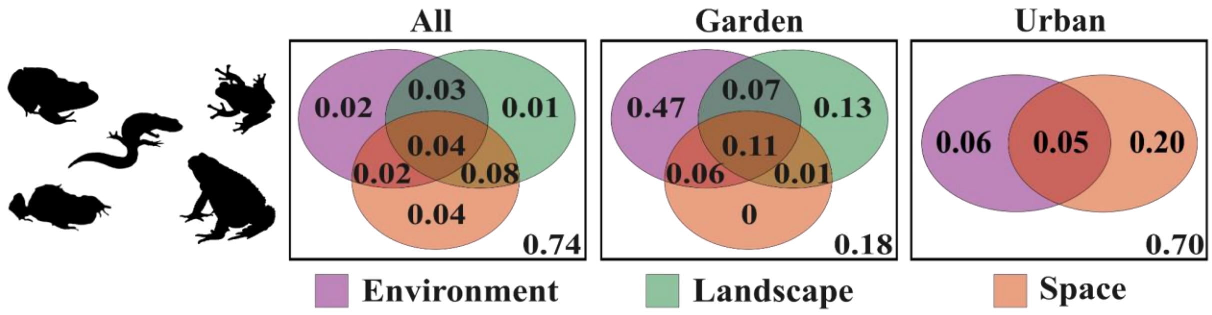

Figure 4: Variance partitioning of amphibian community composition explained by environmental, landscape, and spatial predictors for all ponds, garden ponds, and urban ponds. Residual variance is displayed outside the Venn diagrams.

Species-level analyses of occurrence revealed contrasting patterns. *Pelophylax* spp. responded mostly to environment (pure effect: 9%) and space (pure effect: 15%) (**Figure 5a**) when analysing all ponds jointly. In garden ponds, however, spatial predictors were stronger (pure effect: 17%), while in urban ponds the strong influence of environment dominated (pure effect: 17%). *Rana dalmatina* showed little response to environment and was instead affected by spatial factors and landscape cover when analysing all ponds together (**Figure 5b**). *Bufo bufo* exhibited largely unexplained variation (87%); however, when analysed separately by pond type, environment and landscape variables together explained most variance in garden ponds, while in urban ponds, residual variance remained high (**Figure 5c**).

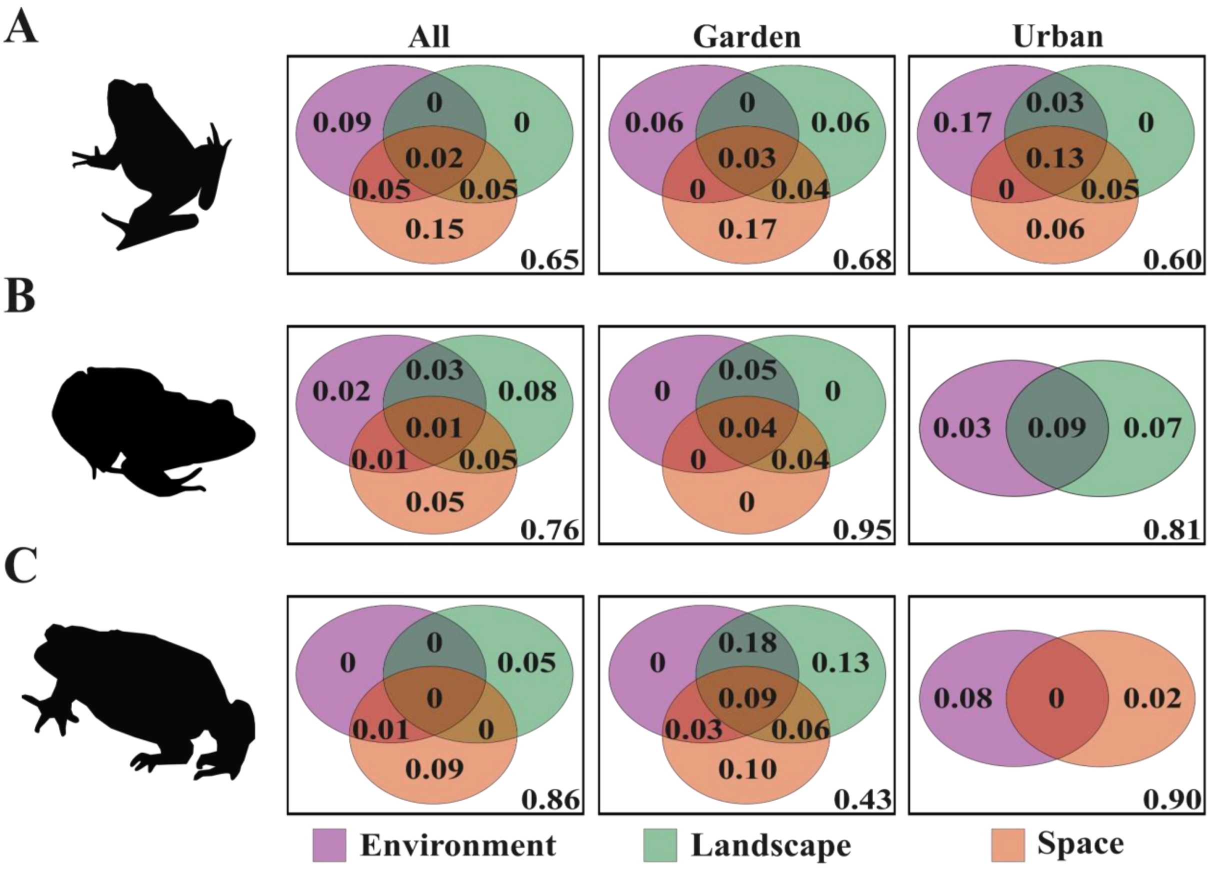

Figure 5: Variance partitioning of the occurrence of three species: *Pelophylax* spp. (A), *Rana dalmatina* (B) and *Bufo bufo* (C) explained by environmental, landscape and spatial predictors for all ponds, garden ponds, and urban ponds. Residual variance is displayed outside the Venn diagrams for each variance partitioning.

Discussion

Here, we show that amphibian communities in garden and urban ponds differ more in the processes structuring them than in their taxonomic composition. Urban ponds supported higher species richness, yet the overall community composition did not differ significantly between pond types. This indicates that garden ponds have the potential to support amphibian communities comparable to those of urban ponds, despite their structural differences. However, the drivers of community assembly differed between habitats. In garden ponds, local environmental variables explained most of the variation in amphibian species composition, indicating strong abiotic filtering at the pond scale. In contrast, spatial structure played a stronger role in urban ponds, suggesting that dispersal processes and the spatial configuration of the pond network more strongly influence amphibian communities in these systems.

Compared to garden ponds, urban ponds hosted higher amphibian species richness, yet both pond types exhibited overall similar community composition. Urban ponds supported a wider range of amphibian species, including species of higher conservation relevance, such as *Bombina bombina* and *Pelobates fuscus*, which are typically less tolerant of habitat modification and therefore less frequently recorded in highly urbanised environments (Konowalik *et al.*, 2020; Kaczmarski, Benedetti & Morelli, 2020). In particular, *P. fuscus* has experienced widespread population declines across Europe (Eggert *et al.*, 2006). Garden ponds, on the other hand, hosted more common species, e.g. *Pelophylax* spp., *Rana dalmatina* or *Bufo bufo*. This pattern is consistent with the smaller size, younger age, and higher levels of human management typical of garden ponds, which can constrain colonisation and persistence (Gaston *et al.*, 2005; Gledhill, James & Davies, 2008; Oertli, 2018; Márton *et al.*, 2025). Despite these structural differences, the limited compositional separation suggests that both pond types draw from the same regional species pool, dominated by urban-tolerant generalist species (Parris, 2006; Hamer & McDonnell, 2008; Hamer & Parris, 2011). Within this shared pool, rarer or more sensitive species appear largely restricted to the larger and less intensively managed urban ponds, compared to the smaller and more intensively managed garden ponds, which may explain their higher species richness despite the overall similarity of community composition.

Amphibian communities in garden and urban ponds had a pronounced difference in the relative importance of environmental and spatial drivers. When all ponds were analysed together, we could only explain a relatively low share of community variation, with a weak spatial trend, indicating that pooling obscured the type-specific patterns. This weak spatial structure reflected a broad central–peripheral gradient across the study area, with garden and urban ponds unevenly distributed along this gradient(**Table S5**, **S6**). When pond types were analysed separately, contrasting assembly mechanisms emerged. In garden ponds, the community structure was primarily driven by local environmental filtering. These ponds were smaller, had higher submerged vegetation cover (**Table S6**), and are typically embedded within fine-scale microhabitat settings (Hill *et al.*, 2021; Loram *et al.*, 2011; Hamer *et al.*, 2024). In line with this, landscape variables had a weak effect, and no pure spatial effect was detected, suggesting that suitable local pond quality might allow colonisation even in relatively isolated garden ponds. In contrast, the amphibian community in urban ponds was structured mostly by spatial processes. In urban landscapes, movement pathways are often constrained by infrastructure, and connectivity depends heavily on the configuration of surrounding habitats (Johnson *et al.*, 2013; Guderyahn, Smithers & Mims, 2016). Urban barriers such as roads and built infrastructure can reduce functional connectivity among water bodies (Cushman, 2006; Ribeiro *et al.*, 2011), making ponds located within connected green corridors particularly important as stepping-stone habitats (Lee *et al.*, 2022).

Our species-level analyses revealed contrasting habitat-use patterns among the three dominant amphibians. For *Pelophylax* spp., fringing vegetation emerged as an important predictor, consistent with existing evidence that emphasises the role of dense marginal vegetation in increasing habitat heterogeneity, providing refuge from predators, and improving microclimatic conditions (Buttemer *et al.*, 1996; Parris, 2006; Wassens *et al.*, 2010). In contrast, *Rana dalmatina* responded more strongly to spatial and landscape-scale drivers when all ponds were analysed together, reflecting its reliance both on surrounding aquatic habitats (Dalpasso *et al.*, 2022) and permeable green matrices for its terrestrial movement (Hartel *et al.*, 2009). Finally, *Bufo bufo* showed more variable responses, broadly mirroring community-level patterns. In garden ponds, local environmental and landscape predictors, particularly the predatory threat index and proximity to rivers, were most influential. The positive association with predatory macroinvertebrates likely reflects coexistence rather than direct facilitation, while the river effect is consistent with existing evidence that riparian habitats enhance anuran dispersal and occupancy (Birx-Raybuck, Price & Dorcas, 2010) and may also increase macroinvertebrate abundance (Hill *et al.*, 2016), contributing to correlated patterns in the variance partitioning.

While we included an extensive set of potential environmental, landscape, and spatial predictors, a substantial proportion of the variation in community composition and species occurrence remained unexplained in some models. Patterns generally became more deterministic once garden and urban ponds were analysed separately, reflecting general differences in how communities are assembled in these habitats. In garden ponds, the model explained a large proportion of community variation, whereas lower explained variation in urban ponds suggests that additional processes may influence amphibian communities in these systems. Nevertheless, unexplained variation remained high in some species-specific models, e.g. for *B. bufo* in urban ponds and *R. dalmatina* in garden ponds. Such residual variance is common in amphibian metacommunity studies (Van Buskirk, 2005; Péntek *et al.*, 2017; Knutson *et al.*, 1999; Dalmolin, Tozetti & Ramos Pereira, 2019), and may reflect unmeasured predictors, including predators, pathogens, stochastic processes, legacy effects in terms of population dynamics or local management, or fine-scale terrestrial habitat features relevant for shelter or overwintering (Hamer & Parris, 2013; Parris & Cornelius, 2004; Green, 2003). In addition, many landscape predictors covaried strongly along the urbanisation gradient (e.g. urban cover, roads, and human population density versus forest or grassland cover), limiting the attribution of effects to a single driver. Although correlated variables were removed for statistical reasons, excluded predictors should not be interpreted as ecologically unimportant, as their influence may be represented indirectly through the retained variables. Finally, amphibian occurrences are inherently dynamic due to their frequent movements among ponds, and our single-season sampling design may have resulted in imperfect detection for some species (Gomes-Mello *et al.*, 2021), particularly under the unusually cold and dry spring of the study period (Ficetola & Maiorano, 2016). Repeated surveys have been shown to substantially improve detection probability and inference on habitat associations (Petitot *et al.*, 2014; Barata, Griffiths & Ridout, 2017). Importantly, these limitations do not alter the consistent pattern that local environmental conditions dominated community assembly in garden ponds, whereas spatial processes were more influential in urban ponds.

Our results point to two complementary conservation pathways for urban amphibians: improving local habitat quality in privately managed garden ponds and strengthening connectivity within urban pond networks. Although based on a single city, the contrasting roles of local habitat management and spatial connectivity are likely to apply broadly across urban pond systems across temperate regions. For garden-pond owners, relatively simple measures - such as retaining aquatic vegetation, avoiding fish stocking, minimising pollutant inputs, and maintaining moderate hydroperiod - can substantially increase habitat suitability (Gledhill *et al.*, 2008; Hassall, 2014; Oertli & Parris, 2019). As local environmental conditions were the primary drivers of amphibian occupancy in these ponds, such local-scale management actions can promote amphibian presence largely independent of the surrounding landscape context. At the city scale, conservation planning should prioritise the retention and creation of ponds embedded within green corridors and stepping-stone networks, while reducing dispersal barriers such as concrete embankments, steep artificial shorelines, and other hard infrastructure that restricts amphibian movement across the urban matrix (Gagné & Fahrig, 2007; Cayuela *et al.*, 2019). Adaptive monitoring frameworks that track occupancy, species richness, and basic habitat indicators (vegetation cover, fish presence, connectivity) could help evaluate the effectiveness of these interventions. Engaging private pond owners through citizen science initiatives could further support amphibian-friendly management while generating valuable long-term data on how local actions influence urban amphibian communities.

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