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Artificial water sites increase amphibian resilience in a changing Mediterranean landscape

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ABSTRACT

Amphibians are highly vulnerable to climate change, habitat loss, and fragmentation, particularly in agricultural landscapes, where land use practices are a substantial driver of biodiversity loss. However, in the Mediterranean region, the continuation of traditional land use practices over centuries has shaped the landscape, determining the coexistence of natural and artificial habitats, such as Mediterranean artificial water reserves, that are supposed to exert a positive effect on biodiversity. Our goal was to assess how landscape configuration, land-use heterogeneity, and the presence of artificial water bodies in the Mediterranean biodiversity hotspot shape amphibian community composition. Using spatial multi-species occupancy models, we analysed detection/non-detection data from 11 species across 1729 freshwater sites along the entire Apennine Mountain chain in Italy. Our findings indicate that among environmental variables, elevation is the only driver affecting the whole amphibian occupancy patterns. At the species level, the presence of artificial water sites is the factor which contributes the most to amphibian occupancy. Although the majority of species generally showed lower occupancy in artificial sites compared to natural habitats, these structures are capable of increasing amphibian resilience amid increasing droughts and changing climate conditions by modulating the ecological effect of environmental predictors. Amphibian resilience in Mediterranean agricultural landscapes can be strengthened by fostering the coexistence of natural and artificial habitats. The latter sites offer a cost-effective management solution that facilitates positive interactions between biodiversity conservation policies and agricultural stakeholders, supporting broader conservation goals in light of future environmental changes.

1. Introduction

In the face of a changing climate and a rapid increase in human population, balancing food security with biodiversity conservation and climate change mitigation is an overriding global challenge (Leclère et al., 2020). As the single largest form of land use on Earth, agriculture covers approximately 38 % of the global land surface (Foley et al., 2005; IPBES, 2019). Agricultural practices, ranging from land conversion to the release of pollutants, are recognised as significant global drivers of biodiversity loss (e.g., Dudley and Alexander, 2017; Jaureguiberry et al., 2022; Wan et al., 2025). For example, the recent IUCN global amphibian assessment – the most imperilled group of vertebrates worldwide - identifies agriculture and related practices as the single most important threat to amphibians, threatening 77 % of the species evaluated (Luedtke et al., 2023). Consequently, reconciling agricultural production with nature conservation has become a compelling issue in the debate surrounding the European Green Deal's associated plans, such as the Common Agricultural Policy (CAP 2023–2027), the European Union's Biodiversity strategy for 2030 and the recently adopted EU Nature Restoration Law (EU Regulation 2024/1991). These international recovery policies are ambitious and share a common foundation in recognising the importance of farmed landscapes for biodiversity conservation across Europe, for instance, by mandating that 4 % of agricultural land incorporates high-diversity landscape features or non-productive areas.

The Mediterranean biodiversity hotspot, a geographical region around the Mediterranean Basin, is recognised for its high levels of endemism and species richness (Myers et al., 2000). Within this area,

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millennia of human land-use practices have profoundly reshaped landscapes and habitats. These traditional land-use systems have persisted over time, significantly influencing species distribution, community composition, and ecological interactions (Blondel, 2006). In contrast to intensive agricultural areas, the long-lasting interplay of natural ecosystems with historical rural activities favoured the establishment of new habitats and species (Blondel, 2006), becoming pivotal sources of Mediterranean biodiversity (Plieninger et al., 2006) and ultimately resulting in a mosaic of natural and artificial habitat patches.

Examples of human-made landscape features, or configurations, important for biodiversity include hedgerows, dry stone walls and similar linear elements of the landscape, which provide shelter and connectivity pathways across inhospitable agricultural areas (Guiller et al., 2022). The alternation of productive areas with abandoned or re-naturalised ones creates a mosaic of small habitat patches that are crucial for biodiversity (Riva and Fahrig, 2022; Wintle et al., 2019). Even completely artificial landscape features, such as earthworks, terraces, or dry-stone walls offer shelters (Collier, 2013), reproductive sites (Assandri et al., 2018), and microclimatic refugia (Manenti, 2014) for biodiversity.

Based on the impact of land-use practices and landscape composition on biodiversity, our study specifically focuses on the drivers of amphibian occupancy patterns at freshwater sites within the Mediterranean region. In the context of the global biodiversity crisis, the loss and degradation of wetlands and the associated freshwater biodiversity outpaces declines observed in marine or terrestrial environments and has been identified as "*the ultimate conservation challenge*" (Dudgeon et al., 2006; Reid et al., 2019). Freshwater habitats and their associated biodiversity are pivotal natural resources for economic, social, cultural and aesthetic values, which are also critical for sustainable development (Dudgeon et al., 2006). The majority of amphibians display a biphasic life cycle (Wells, 2007), being dependent on both aquatic ecosystems and the surrounding terrestrial habitats. Habitat loss, alteration, and fragmentation are among the most severe threats to these species, driving current and future population declines and extinctions (Cushman, 2006; Luedtke et al., 2023). Their life history traits further increase their susceptibility to these impacts, as human-induced habitat changes can disrupt the connection between terrestrial and aquatic habitats, making them even more vulnerable (Becker et al., 2007). In relation to their habitats, the vulnerability of amphibian populations may be attributed to a combination of factors, including: (i) limited vagility, which increases the impact of habitat fragmentation (Bowne and Bowers, 2004), (ii) high desiccation and mortality risk when moving across unsuitable habitat patches (Haggerty et al., 2019), and (iii) narrow ecological requirements (e.g., microclimatic conditions), which exacerbate the impact of anthropogenic disturbances, such as habitat loss, degradation, and edge effects (Liu et al., 2021).

Among the various artificial landscape features found in Mediterranean agricultural settings, the most significant for this study are the traditional Mediterranean water management systems. In general, water management systems in human-modified landscapes are known to provide beneficial effects on biodiversity. With proper management, artificial livestock watering ponds can host as much biodiversity as natural ponds (Malerba et al., 2023; Sayer et al., 2012; Zamora-Marín et al., 2024). However, the general significance of human-made water bodies is usually assessed for those that mimic natural aquatic habitats, such as shallow artificial ponds with aquatic vegetation (e.g., Knutson et al., 2004; Contreras et al., 2009; Moor et al., 2022). By contrast, traditional Mediterranean artificial water sites, primarily designed as water reserves for irrigation or livestock watering, differ significantly in structure and size to natural water bodies (Romano, 2025). Nevertheless, they are supposed to play a crucial role in sustaining amphibian communities (Caballero-Díaz et al., 2020). For instance, entirely artificial water sites, such as drinking troughs, agricultural stone wells or irrigation tanks, can sustain functional food webs (Salvidio et al., 2019; Bissattini et al., 2025) and also contribute to enhancing landscape connectivity and mitigating climate change effects for endangered amphibians (Romano et al., 2023a).

For these reasons, it is crucial to determine how landscape configuration, land-use practices and the occurrence of artificial landscape features influence amphibian communities. This understanding is essential not only for effective management planning but also because it offers a unique opportunity to investigate the influence of Mediterranean agricultural landscapes on amphibian occupancy patterns. Here, by leveraging the Mediterranean's mosaic of productive agricultural, re-naturalised, and natural environments, we model the drivers of amphibian community occupancy patterns, including effects of landscape configuration, the share of agricultural land and the presence of human-made traditional water management systems. To do this, we used a comprehensive dataset of repeated detection/non-detection surveys, encompassing 11 amphibian taxa across 1729 water sites on the southern, central, and northern Apennine mountain chain of Italy, spanning over 1300 km. We analysed these data using recently developed spatial latent factor multi-species occupancy models (Doser et al., 2023).

Specifically, we aim to: 1) identify the drivers of amphibian occupancy at both the community and species levels, examining climatic, topographic, and anthropogenic factors, such as landscape configuration and agricultural intensity; 2) investigate the role of artificial water sites in shaping amphibian occupancy patterns; 3) and determine whether amphibian occurrences in artificial water sites modulate how a species, or the entire community, responds to specific environmental predictors. This latter objective arises from the nature of Mediterranean artificial water sites. Unlike natural water bodies, these sites were constructed by humans to mitigate water shortages and ensure a consistent water supply in all seasons. As a result, these sites may experience a more stable hydroperiod than natural sites, potentially serving as reliable alternative habitats and modulating or buffering the effects of environmental features (Romano et al., 2023a). Ultimately, our goal is to provide valuable insights into managing and planning agricultural landscapes, effectively integrating amphibian biodiversity conservation with sustainable agricultural production.

2. Materials and methods

2.1. Dataset, study area and study framework

The 1729 sampling sites composing our dataset are clustered in three main study areas corresponding to the northern, central and southern parts of the Apennine mountain chain (Fig. 1), spanning 1300 km along the Tyrrhenian side of the Apennines, from central Liguria (province of Genoa, NW Italy) to the southern tip of the Italian peninsula (Calabria region). The aquatic sampling sites included in the dataset were sourced from three main origins: i) original fieldwork data; ii) previously published datasets by the authors (Bernabò et al., 2022; Corsetti and Romano, 2007; Romano et al., 2014, 2012, 2010, 2007); and iii) citizen science projects coordinated by two of the authors (AR and IB). All aquatic sites were visited at least once between 2002 and 2023, by trained herpetologists (AC, IB, SS and AR). Additionally, a subset of 194 sampling sites was visited multiple times (by AC, IB, SS and AR) during the same sampling season, according to a double occupancy sampling design (MacKenzie et al., 2017), allowing for the consideration of species-specific imperfect detection. Specifically, 52 sites were visited 5 times/season, 41 sites were visited 4 times/season, and 101 sites were visited 3 times/season, resulting in a total of 727 repeated surveys. Amphibian occurrence data were collected using standard field methodologies (Dodd, 2010). In particular, aquatic sites were identified following the methodology reported by Romano et al. (2010), (2012) for the identification of potential aquatic habitat types. During each visit, the presence/pseudo-absence of amphibians was recorded.

We excluded from the dataset species with a highly fragmented range (i.e. *Ichthyosaura alpestris*) and not relying on aquatic sites for

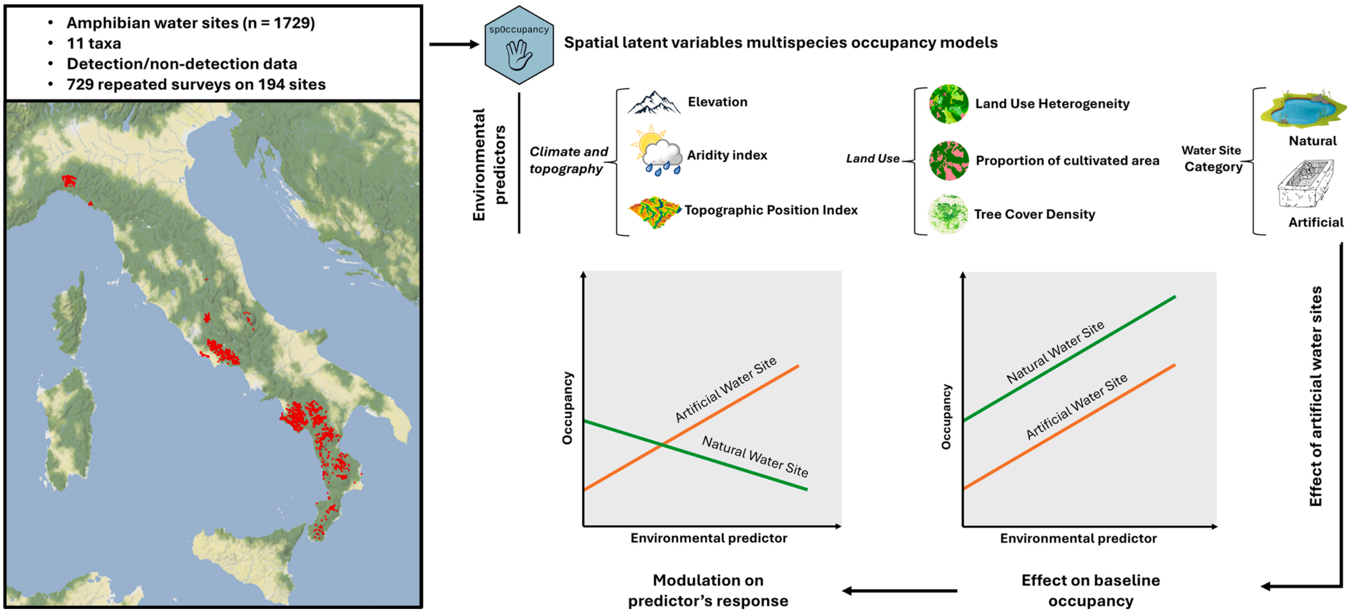


Fig. 1. Graphic representation of the study area (red dots are water sites included in the analysis) and the study framework.

reproduction (i.e. Plethodontid salamanders belonging to the genus *Speleomantes*), while genetically close or parapatric species (e.g., *Salamandrina perspicillata* and *S. terdigitata*; *Lissotriton italicus* and *L. vulgaris*) have been considered as a single entity at the genus taxonomic level. Thereby, we focused on 11 native biphasic amphibians distributed within the study area: *Rana dalmatina*, *Rana italica*, *Pelophylax* sp. (including *P. ridibundus* and *P. bergeri*), *Bombina variegata*, *Hyla* sp. (including *H. meridionalis* and *H. intermedia*), *Bufo bufo*, *Bufotes balearicus*, *Salamandra salamandra*, *Salamandrina* sp. (including *S. perspicillata* and *S. terdigitata*), *Triturus carnifex* and *Lissotriton* sp. (including *L. vulgaris* and *L. italicus*).

Study sites encompass a wide variety of both natural and anthropic land cover categories. The former spans from temperate and supra-Mediterranean broadleaved forests to meso-Mediterranean evergreen oak forests and thermo-Mediterranean shrublands. The latter include more structured productive agricultural areas, such as terraced vineyards in the Cinque Terre National Park, and extensive pastures or meadows interspersed with more natural or abandoned patches. The distance from the coastline varies from less than 50 m to 71 km, and the elevations range from a few metres to about 1850 m asl. Regarding climate, according to the Köppen-Geiger classification system (Peel et al., 2007), study sites range from a Hot-summer Mediterranean climate (Csa: temperate, dry summer, hot summer) and Warm-summer Mediterranean climate (Csb: temperate, dry summer, warm summer) in the southern part of the study area, to a Temperate oceanic climate (Cfb: temperate, fully humid, warm summer) in the inner sections of the northern part.

2.2. Environmental predictors

We selected and computed seven variables for each water site included in the dataset to evaluate potential drivers of amphibians occurrence at both the community and species levels. Regarding climate-related drivers, we focused on three variables: (i) elevation above sea level: site-specific elevation a.s.l., extracted from a digital elevation model (20 m spatial resolution, provided by the Italian Military Geographic Institute); (ii) the De Martonne aridity index (*DMA*): a climatic index used to quantify the degree of aridity of a region (De Martonne, 1926) with lower values corresponding to more arid climate (Pellicone et al., 2019); (iii) the topographic position index (*TPI*), which was calculated from the digital elevation model. *DMA* values for each site were spatially downscaled using the ClimateDT downscaling service (Marchi et al., 2024; Romano et al., 2023a), a geo-web service that allows scale-free downscaling of climatic variables and indices, averaging the values of the index for the period covered by data collection (2002–2023). As regards *TPI*, this index is significant for representing potential microclimatic conditions relevant to amphibian occurrence (Basile et al., 2017; Romano et al., 2021, 2017) as it quantifies the relative position of a cell within the surrounding landscape - positive *TPI* values indicate that the point is higher than its surroundings (suggesting ridge or hilltop positions), while negative *TPI* values indicate lower positions, such as valleys or depressions (Guisan et al., 1999). Concerning the drivers related to agricultural landscape configuration and the presence of landscape features, we selected and computed four additional variables: (i) we extracted the percentage of tree cover from the High-Resolution Layer Tree Cover Density (*TCD* – 100 m spatial resolution) provided by the Copernicus Land Monitoring Service; (ii) we built a buffer of 250 m radius around each water site and, using the Corine Land Cover (100 m spatial resolution; provided by the Copernicus Land Monitoring Service), we calculated Land Use Heterogeneity (*LUH*) based on Shannon's Diversity Index (Farina, 2007); (iii) we recategorised the Corine Land Cover layer into a binary format (0 = natural landscape class; 1 = cultivated landscape class) and calculated the proportion of cultivated area (*Cultivated*) within a 250 m buffer around each water site; and finally, (iv) we added a categorical variable to consider the occurrence of specific human-made landscape features, which have been shown to be relevant for amphibian landscape connectivity within the Mediterranean region, and their presence has been formerly supposed to modulate the effect of other environmental predictors on amphibian occupancy (Romano et al., 2023a). This variable (*Type*) categorises each water site as either natural (e.g., pond, stream) or artificial (e.g., water tank, drinking trough, stone well), using a binary format (0 for natural sites and 1 for artificial sites). The extraction of values from raster layers, the calculation of land use heterogeneity, and the proportion of cultivated area were performed in the R environment using packages *raster* (Hijmans, 2024) and *landscapemetrics* (Hesselbarth et al., 2019), while TPI computation was performed in SAGA GIS v9.5.1.

2.3. Data analysis

We analysed detection/non-detection data of 11 amphibian taxa through hierarchical multi-species occupancy models proposed by

Tobler et al. (2019) and Doser et al. (2023). We employed the package *spOccupancy* (Doser et al., 2022) within the R programming environment to fit multi-species occupancy models considering the effect of environmental covariates on occurrence, both on the whole amphibian community and for each species separately. We also accounted for species-specific imperfect detection, residual species correlations and spatial autocorrelation (Doser et al., 2023).

Specifically, imperfect detection was accounted for using an observation model, exploiting the subset of sites with repeated surveys. Detection probabilities are assumed to be species-specific, i.e. we fit an intercept-only model, not accounting for the effect of survey-specific covariates. For all our models, the observation sub-model can be defined as follows:

$$y_{i,j,t} \sim Bernoulli(p_{i,j,t}|z_{i,j})$$

$$logit(p_{i,j,t}) \sim \alpha_{0i}$$

Where: $y_{i,j}$ is the observed detection (1) or non-detection (0) of species *i* at site *j* in survey *t; $p_{i,j,t}$*is the probability of detecting species *i* at site *j* in survey *t*, conditional on its true occurrence state $z_{i,j}$; and α_{0i} is the detection probability intercept for species *i.* Occurrence probabilities were modelled using a logit-linear combination of site-specific environmental predictors and latent variables, each associated with species-specific factor loadings. These latent variables potentially represent unmeasured predictors, with factor loadings reflecting species-specific responses to them, and the correlation between species-specific factor loadings helping to identify patterns of species co-occurrence (Doser et al., 2023). In our models, we assumed occupancy as a function of environmental predictors, and modelled it via logit link, as follows:

$$z_{i,j} \sim Bernoulli(\psi_{i,j})$$

$$\begin{aligned} logit(\psi_{i,j}) \sim\ & \beta_{0i} + \beta_{1i} * Elevation_j + \beta_{2i} * Elevation_j^2 + \beta_{3i} \\ & * TPI_j + \beta_{4i} * DMA_j + \beta_{5i} * TCD_j + \beta_{6i} * LUH_j + \beta_{7i} \\ & * Cultivated_j + \beta_{8i} * Type_j \end{aligned}$$

Where: $z_{i,j}$ is the true occurrence state of species *i* at site *j; $\psi_{i,j}$* is the occupancy probability for species *i* at site *j; β_{0i}* is the intercept for species *i; $\beta_{1i-}\beta_{8i}$* are species specific regression coefficients to account for environmental predictors effect. Specifically, we both included a linear and a quadratic effect for elevation. This model structure was used to fit both non-spatial and spatial models (i.e. models 1 and 2). We built non-spatial models, in which the latent factors are assumed as independent normal random variables using function *lfMsPGOcc*, and models accounting for spatial autocorrelation, which is resolved in the *spOccupancy* package using a nearest neighbour Gaussian process, using function *sfMsPGOcc* (Datta et al., 2016; Doser et al., 2022). Additionally, we built two more models — one spatial and one non-spatial (i.e. models 3 and 4) — that also account for the interaction between site type (i.e., natural or artificial water sites) and all other predictors included in the model. These models account for both varying intercepts and slopes, depending on whether a site is natural or artificial (Fig. 1). The goal was to evaluate whether occurrences in artificial water sites could modulate the response of a given species, or the entire community, to specific environmental predictors (Romano et al., 2023a). To account for these additional interactions, we modified the model structure as follows:

$$\begin{aligned} logit(\psi_{i,j}) \sim\ & \beta_{0i} + \beta_{1i} * Elevation_j + \beta_{2i} * Elevation_j^2 + \beta_{3i} \\ & * TPI_j + \beta_{4i} * DMA_j + \beta_{5i} * TCD_j + \beta_{6i} * LUH_j + \beta_{7i} \\ & * Cultivated_j + \beta_{8i} * Type_j + \beta_{9i} * Elevation_j \\ & * Type_j + \beta_{10i} * Elevation_j^2 * Type_j + \beta_{11i} * TPI_j \\ & * Type_j + \beta_{12i} * DMA_j * Type_j + \beta_{13i} * TCD_j \\ & * Type_j + \beta_{14i} * LUH_j * Type_j + \beta_{15i} * Cultivated_j \\ & * Type_j \end{aligned}$$

Prior to model building, we scaled and centred environmental predictors and checked them for collinearity, considering a cut-off for inclusion of Pearson's $r < 0.75$ (Dormann et al., 2013; Figure S1). We assumed species-specific coefficients follow a common normal distribution with community-level mean and variance parameters. Priors for community-level mean parameters were assumed as normal, with mean equal to 0 and variance equal to 2.72, while for the community-level variance parameters we used an inverse gamma distribution with shape and scale parameters equal to 0.1, following Doser et al. (2023). We used 2 latent factors, both in non-spatial and spatial models, which are the number of factors that best balance computational efficiency and results' robustness for an 11-species community (Doser et al., 2023). Spatial models were fitted using a Nearest Neighbour Gaussian Process with an exponential correlation function, utilizing 5 nearest neighbours for each point. This number was deemed sufficient to capture the spatial autocorrelation in the dataset while reducing the computation time required to run the model (Datta et al., 2016). All models were run exploiting the *spOccupancy* package's model fitting functions which utilize a Pólya-Gamma data augmentation approach, combined with a Markov Chain Monte Carlo algorithm, to enhance computational efficiency (Doser et al., 2022). For non-spatial models, we ran 4 chains with 25,000 iterations each, discarding the first 10,000 iterations as a burn-in for each chain, and thinning by 10, yielding 6000 posterior samples for each parameter. Spatial models usually require longer chains to achieve convergence (Doser et al., 2022; Doser et al., 2023), therefore, we ran 3 chains with 600,000 iterations each, discarding the first 200,000 iterations from each chain as a burn-in, and thinning by 200, thus yielding 6000 posterior samples for each parameter. The resulting models were compared using the widely applicable information criterion (WAIC; Watanabe and Opper, 2010). We evaluated convergence by visually inspecting trace plots and by means of the Gelman-Rubin statistic (Gelman and Rubin, 1992), considering that chains reached convergence when the R-hat statistic was < 1.1. To assess model fit, a task of primary importance in hierarchical models (Costa et al., 2021, 2020), we relied on posterior predictive checks based on MacKenzie and Bailey's (2004) goodness-of-fit test, by grouping detection/non-detection data for each site, comparing the χ^2 statistic computed for the observed and generated data and computing a Bayesian p-value. To assess the effects of covariates and the significance of parameters, we calculated the 89 % Highest Density Interval (HDI), which is recognised as a particularly stable credible interval for posterior distributions (Kruschke, 2014; Vélez et al., 2024). Additionally, we computed the probability of direction (*Pd*) for each parameter from its posterior distribution. A covariate effect was considered to have strong support if the 89 % HDI did not include zero, and even when the HDI overlapped zero, we considered the effect to have moderate biological relevance if the probability of direction exceeded 75 % (Perea et al., 2023; Tilker et al., 2020). Model building, model selection and posterior predictive checking were conducted through package *spOccupancy* (Doser et al., 2022), while HDIs and probabilities of direction were calculated in package *bayestestR* (Makowski et al., 2019).

3. Results

From our dataset, based on repeated surveys conducted across 1729 water sites, we recorded a total of 3492 detections belonging to the 11 amphibian taxa (see Materials and Methods). At least one taxon was encountered in 1687 of the 1729 (98 %) water sites surveyed. The 11 taxa showed high variability in both the number of detections and occurrences, from *Rana italica* being the most frequently encountered species (number of detections = 722; number of occurrences = 630), to *Bufotes balearicus,* which was the least encountered (number of detections = 26; number of occurrences = 23). Detailed numbers of detections and occurrences for each taxon are provided in Table 1. From the model selection procedure, one model was supported (i.e. model 4), accounting for spatial autocorrelation in model structure and for the interaction of site type with the other predictors (Table 2), also resulting in a good fit (Bayesian p-value = 0.45). According to this model, no single environmental predictor had a strong, significant, and consistent effect on the whole amphibian community considered (Table S1; Figure S2). Nevertheless, *Elevation* and *Elevation*$^{\wedge 2}$ displayed a moderate biological effect on the amphibian community (β *Elevation* = 0.28, β *Elevation*$^{\wedge 2}$ = −0.17; *Pd Elevation* = 82 %, *Pd Elevation*$^{\wedge 2}$ = 85 %), indicating that community occupancy probability was higher at both lower and higher elevations but decreased at intermediate altitudes (Fig. 2).

At the species level, the supported model revealed varying detection probabilities and occupancy values among the 11 taxa included in the community (Table 1). Consistent with the raw data, *Rana italica* had the highest occupancy probability, while *Bufotes balearicus* had the lowest (Table 1). The set of environmental predictors had varying effects on the occupancy of the 11 taxa considered, with some predictors being significant for certain taxa and not for others, highlighting opposing responses to a given predictor within the community (Fig. 3; Fig. 4; Figure S2). Apart from *Elevation,* site *Type* and *Cultivated* had a significant, or plausible, biological effect on 10 out of the 11 taxa considered, accounting for both positive and negative effects on occupancy probability (Fig. 3; Fig. 4). *TCD* and *TPI* also had a substantial effect on occupancy probability for 8 taxa, each with diverse directions and intensities (Fig. 3; Fig. 4). Finally, *DMA* and *LUH* affected the occupancy probability of 6 out of 11 species (Fig. 3; Fig. 4).

Mediterranean artificial waters resulted to be important drivers of amphibian occupancy patterns, exerting a significant, or plausible, biological effect on 10 taxa. Moreover, a significant modulating effect was also observed. The most supported model, accounting for the interactions between environmental predictors and site *Type*, indicated a plausible modulation effect of *Type* on the effect of *TCD* for 7 taxa, on *Elevation* and *Elevation*$^{\wedge 2}$ for 7 and 5 taxa, respectively, and on *Cultivated* and *TPI* for 6 taxa. For *LUH* and *DMA* a plausible modulation effect was detected for 3 taxa (Fig. 5; Figure S2). Finally, significant residual correlations were found among several taxon co-occurrences (Fig. 6).

4. Discussion

In examining the effects of landscape-scale predictors on amphibians at the community level, we found that only *Elevation* had a biologically significant impact (Fig. 2). For this variable, occupancy probability exhibited a bimodal pattern, with higher values at both low and high altitudes, and notably lower at mid-elevations. This pattern is likely to reflect the availability of water sites, which is greater at intermediate elevations (Figure S3). Consequently, at lower and higher elevations – i.e. where water site availability may be a limiting factor for reproduction – amphibians tend to colonize the majority of available sites, resulting in a higher occupancy rate. In contrast, at mid-elevations, the greater availability of water sites results in a lower proportion of sites being colonized. This suggests that water site availability is not a limiting factor for the occupancy of the amphibian community at this elevation. Therefore, management actions aimed at increasing water site availability should focus on areas at lower or higher elevations. It is important to note that this occupancy pattern has moderate statistical support, with a probability of direction slightly over 80 %.

As a second consideration, at the species level, our analyses revealed that site-specific environmental variables can differently affect amphibian occupancy, reflecting the species' ecological requirements within the community (Fig. 4). This result is largely expected, as the 11 species in our community are known to have distinct ecological needs. Most species responded differently to the environmental predictors, suggesting that their ecological requirements have been largely identified. Therefore, the remaining variability captured by the two latent variables in the model can be interpreted as an indicator of species co-occurrences rather than representing an additional environmental feature not accounted for in the analysis (Doser et al., 2023; Romano et al., 2022). Indeed, significant species co-occurrences are widely supported by their ecological requirements known from the literature (Fig. 6a). This finding is also supported by the fact that latent variables allowed us to order species in ecologically sound sub-communities (Fig. 6b): identifying a forest/stream related community, a community more related to standing waters or open habitats, and a set of generalist or pioneer species usually associated with early successional habitats.

A main focus of our study was to explore the response of the amphibians to land use practices in Mediterranean agricultural landscapes. In this context, the significant effects of *Cultivated* land, *TCD* and *LUH* on occupancy for most taxa highlighted the existence of complex interactions between extant biodiversity and human influences. In particular, the contrasting responses of certain species to land-use

Table 2
List of candidate models, ranked by WAIC.

Model	WAIC	ΔWAIC
M4. Spatial model + interactions with site type	13,425.32	-
M3. Spatial model	13,524.81	99.49
M2. Non-spatial model + interactions with site type	13,868.98	443.66
M1. Non-spatial model	14,081.58	656.26

Table 1
Summary of the taxa dataset. For each taxon, the following metrics are reported: number of detections, number of occurrences, estimated number of occurrences, naïve occupancy, estimated occupancy, and estimated detection probability.

Taxon	Number of detections	Number of occurrences	Estimated occurrences (95 %HDI)	Naive occupancy	Estimated occupancy (95 %HDI)	Detection probability (95 %HDI)
Rana dalmatina	153	131	262 (201−335)	0.08	0.15 (0.12–0.19)	0.47 (0.36–0.59)
Rana italica	722	630	822 (780−868)	0.36	0.48 (0.45–0.50)	0.75 (0.70–0.79)
Pelophylax sp.	413	330	491 (442−548)	0.19	0.28 (0.26–0.32)	0.65 (0.58–0.71)
Bombina variegata	253	171	196 (182−215)	0.10	0.11 (0.11–0.12)	0.85 (0.78–0.90)
Hyla sp	190	172	404 (315−520)	0.10	0.23 (0.18–0.30)	0.38 (0.28–0.52)
Bufo bufo	386	323	408 (376−447)	0.19	0.24 (0.22–0.26)	0.76 (0.69–0.82)
Bufotes balearicus	26	23	44 (26−101)	0.01	0.03 (0.02–0.06)	0.56 (0.23–0.80)
Salamandra salamandra	260	156	188 (174−205)	0.09	0.11 (0.10–0.12)	0.71 (0.65–0.76)
Salamandrina sp.	357	294	358 (333−388)	0.17	0.21 (0.19–0.22)	0.77 (0.70–0.82)
Triturus carnifex	240	204	254 (231−284)	0.12	0.15 (0.13–0.16)	0.78 (0.69–0.85)
Lissotriton sp.	492	451	529 (499−563)	0.26	0.31 (0.29–0.33)	0.82 (0.76–0.87)

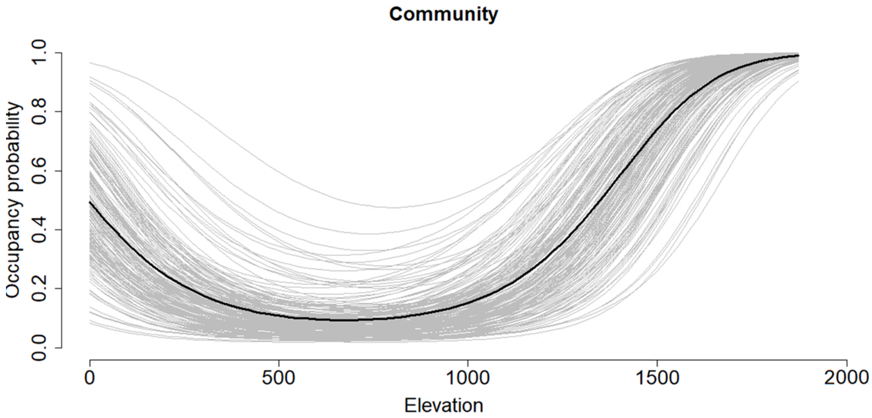
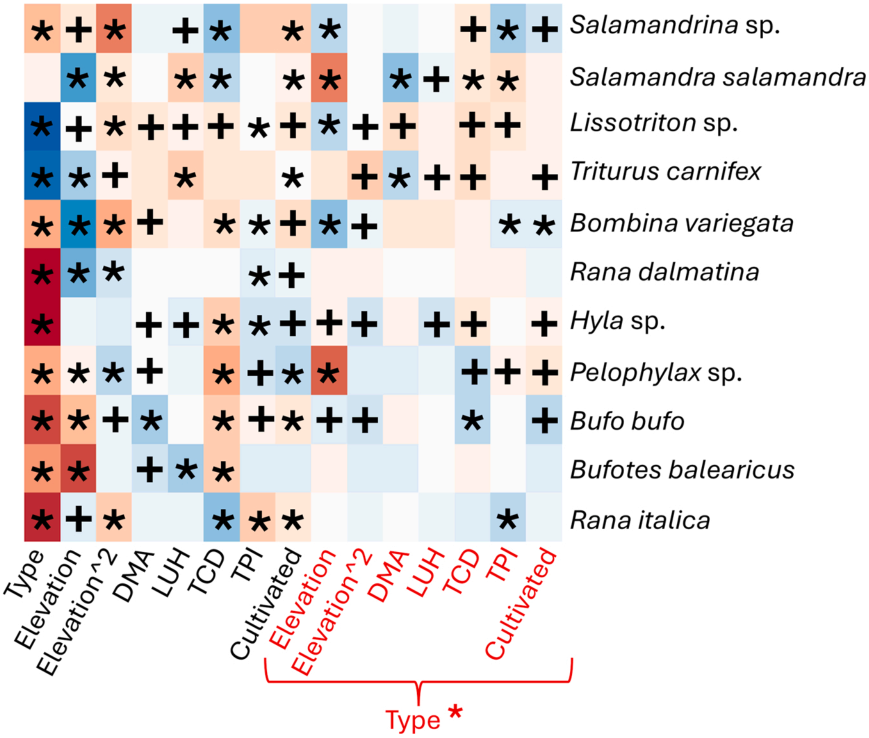


Fig. 2. Response plot showing the effect of Elevation on amphibian community under the best model. The black solid line represents mean community response, while grey lines represent dispersion around the mean and are constituted by 200 random draws from the posterior distribution.



	Type	Elevation	Elevation^2	DMA	LUH	TCD	TPI	Cultivated	Elevation	Elevation^2	DMA	LUH	TCD	TPI	Cultivated
Salamandrina sp.	*	+	*		+	*		*	*				+	*	+
Salamandra salamandra		*	*		*	*		*	*		*	+	*	*	
Lissotriton sp.	*	+	*	+	+	+	*	+	*	+	+		+	+	
Triturus carnifex	*	*	+		*			*		+	*	+	+		+
Bombina variegata	*	*	*	+		*	*	+	*	+				*	*
Rana dalmatina	*	*	*				*	+							
Hyla sp.	*			+	+	*	*	+	+	+		+	+		+
Pelophylax sp.	*	*	*	+		*	+	*	*				+	+	+
Bufo bufo	*	*	+	*		*	+	*	+	+			*		+
Bufotes balearicus	*	*		+	*	*									
Rana italica	*	+	*			*	*	*						*	

Fig. 3. Heatmap showing the effect of environmental predictors for the 11 amphibian taxa studied, under the best model. Red labels indicate the interaction between environmental predictors and water site category (natural or artificial). Blue indicates a positive effect for a given predictor, while red denotes a negative effect. In each case, the intensity of the colour represents the strength of the effect. The asterisk (*) indicates that the 89 % HDI of the considered parameter does not include 0, while plus symbol (+) indicates a probability of direction > 75 %.

changes may reflect differing ecological strategies, with some taxa benefiting from historical anthropogenic activities in cultivated areas. For instance, *S. perspicillata* and *S. salamandra*, which are forest-associated species and positively affected by *TCD*, are negatively affected by the proportion of cultivated land surrounding the water sites. By contrast, the occupancy of those taxa related to open habitats, such as *Pelophylax* sp., *Hyla* sp. and, to a lesser extent, *R. dalmatina*, resulted to be positively affected by *Cultivated*. For example, *Bombina variegata* and *Hyla sp.* are known to be well adapted to small artificial water bodies which offer ideal conditions (e.g., limited canopy cover, high insolation, low predatory pressure) for rapid and massive larval development (Barandun, 1990; Cayuela et al., 2022). From a landscape management planning perspective, these results suggest that a single management measure (sensu Margules and Pressey, 2000) is not viable for the community of amphibian species inhabiting the Apennine mountain chain. We stress the importance of conducting a more comprehensive analysis to better understand how the identified sub-communities respond to different sets of environmental predictors. This will help determine whether a shared management approach is feasible.

In amphibian conservation, the creation of artificial water sites capable of mimicking natural water bodies, such as ponds or lakes, is a common technique to increase the availability of breeding habitats in altered landscapes, producing a concrete conservation result (Pechmann et al., 2001; Petranka et al., 2007; Lambert et al., 2021; Moor et al., 2022). Furthermore, amphibian resilience to climate change can be improved by creating and managing a dense pond network that ensures the presence of reproductive sites (Blaustein et al., 2001). However, also human-made water structures not explicitly created for conservation

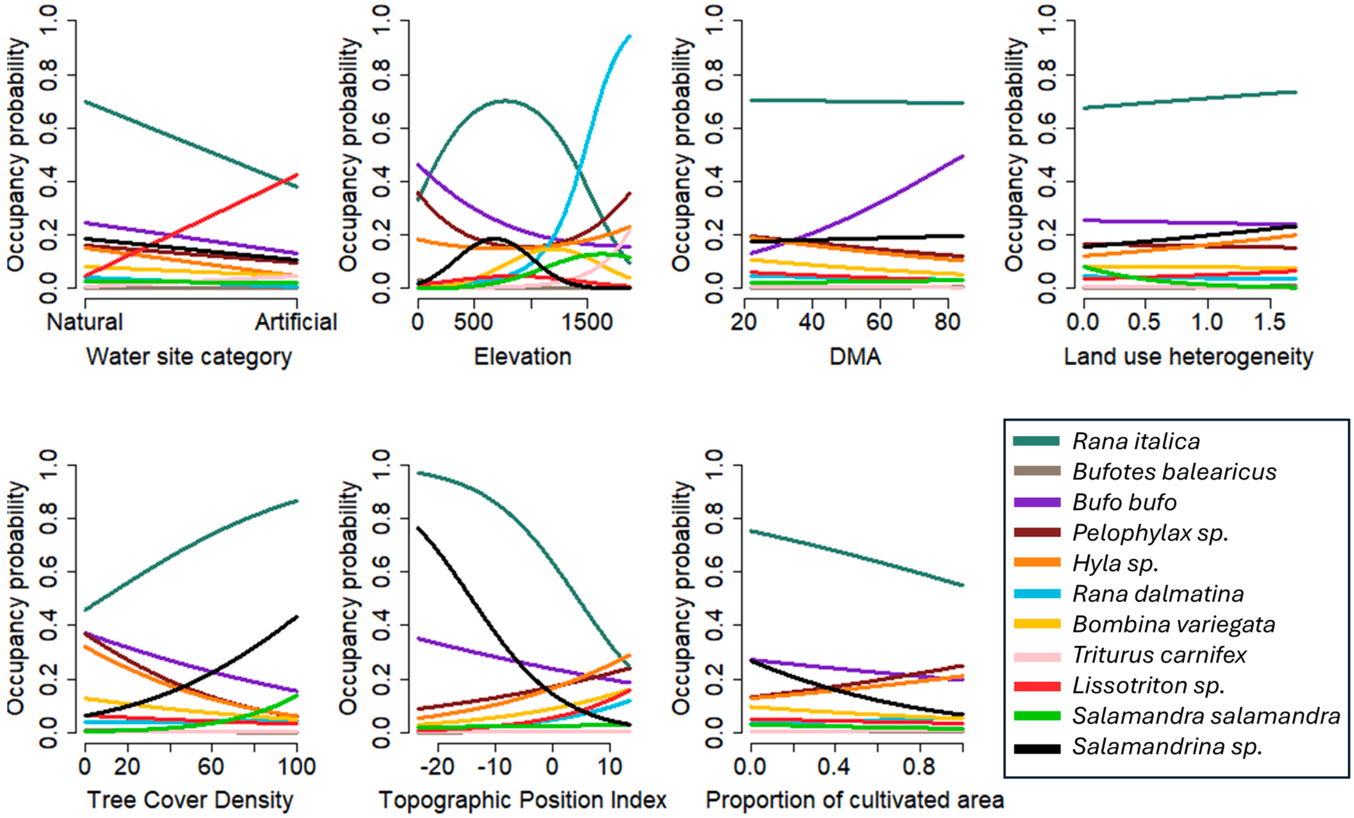


Fig. 4. Response plots showing the effect of environmental predictors for the 11 amphibian taxa considered, under the best model. Each taxon has been assigned a different colour.

purposes, such as stormwater ponds, livestock ponds and dams, or fortuitous anthropogenic habitats (e.g. flooded furrows in logged forests, puddles in quarries) can sustain viable amphibian populations (e.g., Brand and Snodgrass, 2010; Cayuela et al., 2022; Malerba et al., 2023) and act as important connectivity nodes within the landscape (Scheffers and Paszkowski, 2013). In this context, another key objective of our study is to evaluate the role of a specific artificial water site category: the Mediterranean artificial water sites (see also Romano, 2025). Indeed, *Type* of the water site is a significant predictor for almost all species examined, although displaying different intensity and effect (Fig. 3). In fact, occupancy for certain species is higher in artificial water sites, while for other species it is lower. Specifically, newts (*Triturus carnifex* and *Lissotriton* sp.) are the species that benefit the most from the presence of Mediterranean water sites (Romano et al., 2023b). This finding is supported by the literature, which indicates that newts take advantage of artificial water bodies as key sites for foraging and reproduction (Salvidio et al., 2019; Bissattini et al., 2025).

Furthermore, significant interaction terms between *Type* and other predictors illustrate how artificial water bodies can influence amphibians' responses to environmental predictors. Specifically, these results indicate that amphibian occupancy in natural and artificial sites often responds differently to environmental variables, and artificial water sites may modulate the effects of environmental features. Future climate change projection for the Mediterranean region reveal rising temperatures and shifts in precipitation regimes. (Giorgi and Lionello, 2008). Consequently, natural freshwater habitats may dry out or experience altered hydroperiods. In this framework, artificial water reservoirs could provide critical refugia, helping sustain amphibian populations, especially in areas with intensive agriculture. Notably, for taxa of conservation concern, these structures appeared to buffer the effect of environmental features linked to climate (e.g., *DMA* and *Elevation* and *TPI*), potentially enabling species to persist in less favourable areas (Fig. 5). This outcome suggests that artificial water sites not only serve as supplementary habitats but could also contribute to enhance resilience of amphibian populations to landscape and climatic changes, highlighting their potential role in species conservation planning.

However, while artificial sites can increase the occupancy of some species, they generally support lower amphibian occupancy in comparison to natural freshwater habitats. This indicates that although artificial water bodies are important, they may not be as effective as natural habitats in supporting amphibian diversity over the long-term.

Finally, our study can also have methodological entailments. We found a significant variation in detection probability across the 11 amphibian taxa considered (Table 1). The mean detection probability for these species was $p = 0.68$, with some exceeding $p = 0.75$. These relatively high values, in the context of imperfect detection, raise questions about the necessity of the additional modelling burden and extensive field effort required for repeated surveys – both of which are needed to directly estimate detection probability in similar situations. However, we emphasize that with a detection probability of $p = 0.68$, at least three surveys without detection are required to confirm species absence with 95 % confidence if detection is not properly accounted for in a site-occupancy mode (calculated as: N surveys = log(alpha level) / log(1-p), following Kery, 2002). In cases with significant species-specific variation in detectability, as in our study (estimates ranging from $p = 0.38$ to $p = 0.85$), the lowest detection probability for the entire community should be considered. Under such conditions, at least six surveys would be required to confirm the true absence of the species with the lower detection probability values, unless a method that explicitly accounts for detection probability (e.g., site occupancy models) is used. Failing to account for this variability could lead to unreliable and biased estimates of species occurrence, potentially misrepresenting the influence of environmental predictors (e.g., Hamer and Horányi, 2024). This concern is particularly relevant when applying species distribution or habitat suitability models that rely on presence/absence data or generate pseudo-absences. Such biases have

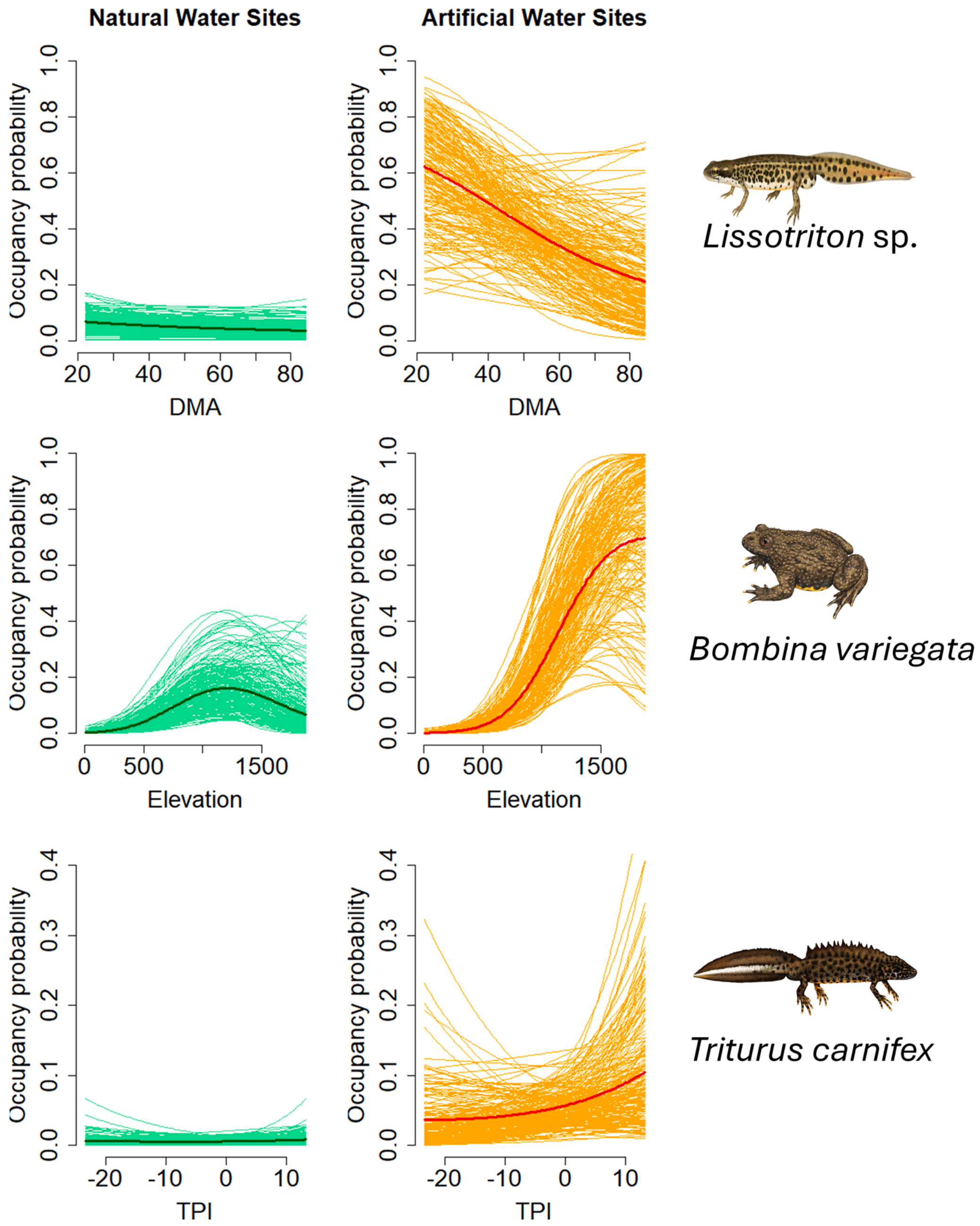


Fig. 5. Plots illustrating the modulation effect of site *Type* on different predictors for three species of conservation concern. For each predictor and taxon, the plots on the left side (green) represent occupancy response on natural water sites, while the plots on the right side (orange) represent occupancy response on artificial water sites. The thick dark lines represent the mean response, while the thin light lines depict the dispersion around the mean, consisting of 200 random draws from the posterior distribution.

direct conservation implications, as neglecting detection probabilities may lead to misguided landscape scale management recommendations (Hamer and Horányi, 2024). Therefore, accurate ecological assessments should incorporate detection probabilities into the modelling process, to guide informed conservation and management actions. The issue of imperfect detection in ecological studies and conservation planning has long been debated, particularly for taxa of conservation concern such as amphibians (Schmidt, 2003). Despite this attention, the problem remains unresolved, given that a recent study (Bennett et al., 2024) found that detection probabilities have been estimated for fewer than 40 % of North American threatened amphibians and less than 20 % of threatened North American reptiles. Nevertheless, it is plausible that conservation actions for these species have already been planned and implemented without accounting for detection probability.

5. Conclusions

Our study supports previous research emphasizing the importance of traditionally managed anthropogenic habitats for multi-taxa biodiversity conservation in Mediterranean ecosystems (e.g., Caballero-Díaz et al., 2020; Cerini et al., 2019; Costa et al., 2025; Romano et al., 2023b; Zamora-Marín et al., 2021). The integration of natural and semi-natural habitat patches with cultivated areas creates a mosaic landscape that can mitigate some negative effects of agricultural intensification (Tscharntke et al., 2005; Fischer and Lindenmayer, 2007; Guerra and Aráoz, 2015). Our findings confirm that both natural and artificial aquatic habitats in Mediterranean landscapes contribute to maintaining amphibian diversity (Guerra and Aráoz, 2015; Martínez-Abraín and Galán, 2018). Although artificial sites generally host lower amphibian

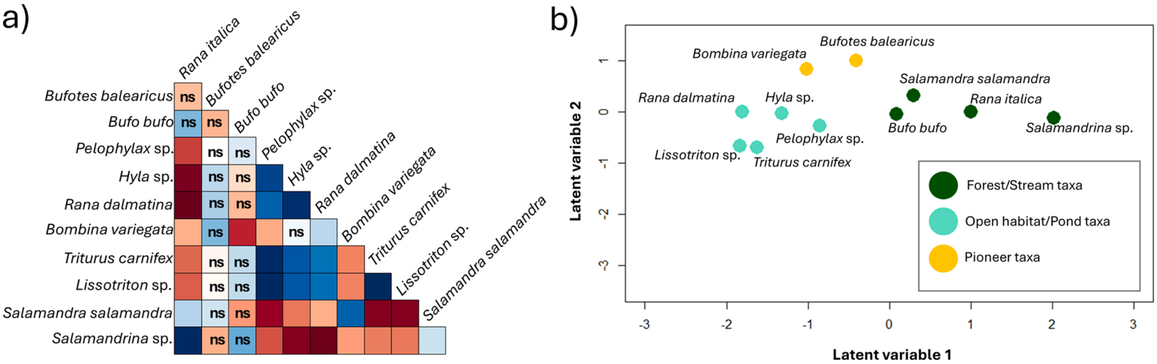


Fig. 6. a) Residual correlation matrix, based on latent variables' values, representing species co-occurrences; ns = not significant residual correlation. b) Ordination plot, based on latent variables' values, depicting taxa associations; colours represent three possible communities/associations.

occupancy than natural habitats, they serve as critical refugia, enhance landscape connectivity and provide essential habitats in areas where natural wetlands are diminishing (Romano et al., 2023a). Notably, their role in supporting amphibian diversity and resilience within agricultural landscapes may not be as immediately apparent as that of natural aquatic habitats. The conservation or ex-novo creation of small artificial aquatic sites, while often practical and cost-effective, should be regarded as a supplementary measure rather than a replacement for the protection and restoration of natural wetlands. Indeed, natural wetlands are complex ecosystems that sustain considerably higher levels of biodiversity and provide unique ecological functions. As agricultural intensification and land abandonment reshape Mediterranean ecosystems, with possible adverse outcomes for biodiversity (Falcucci et al., 2007; Perpina-Castillo et al., 2021), the goal of maintaining a mosaic of diverse habitats, including artificial water bodies, will be key to preserving amphibian populations and broader biodiversity.

From a management perspective, recent EU environmental and agricultural policies in the Mediterranean region, as well as the Nature Restoration Law, already recognise the importance of incorporating high-diversity landscape features into farmlands. It is, therefore, evident that some of these features should include artificial water sites, which should be made accessible to wildlife, particularly amphibians, as recently highlighted by the proposal of an index to evaluate the accessibility of artificial water siters to amphibians (Romano, 2025). Our study supports these policies by demonstrating that, when properly managed, artificial landscape elements can contribute significantly to biodiversity conservation in the midst of the global climate crisis.

CRediT authorship contribution statement

Antonio Romano: Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Andrea Costa:** Writing – original draft, Supervision, Methodology, Formal analysis, Data curation, Conceptualization. **Sebastiano Salvidio:** Writing – original draft, Data curation, Conceptualization. **Giacomo Rosa:** Writing – original draft, Conceptualization. **Ilaria Bernabò:** Writing – original draft, Data curation, Conceptualization.

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Declaration of Competing Interest

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

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2025.109912.

Data availability

Data will be made available on request.

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