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Long-term detection data reveals widespread range contractions in Spanish herpetofauna

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ABSTRACT

Populations of amphibians and reptiles have undergone substantial declines in recent decades worldwide. The Iberian Peninsula, a European diversity hotspot for this group, is suffering severe droughts, rising temperatures, and abandonment of traditional agricultural practices. Despite the importance and sensitivity of Iberian herpetofauna, a data-driven, national, long-term, and multi-species analysis of the distribution and demographic situation of amphibians and reptiles has been lacking. We used dynamic occupancy models to study how the distributions of 25 amphibian and 37 reptile species have changed over the last three decades in Spain. We analysed a detection dataset compiled from multiple sources including citizen science, monitoring programmes and opportunistic observations. Our results showed widespread declines in amphibian and reptile occupancy, which could also suggest generalised population declines. Moreover, we observed a mismatch between the IUCN Red List categories and the estimated occupancy trend, with some species appearing stable at the IUCN global level but presenting substantial range contractions at the regional level. The causes for these declines are unclear. According to our models, amphibian persistence was negatively correlated with rainfall and colonization negatively correlated with altitude. Reptile persistence was negatively correlated with temperature. Our analysis highlights the value of national-scale assessments to identify occupancy trends at fine resolution and investigate regional threats driving these changes. We encourage further research and early conservation action before regional trends and threats become global issues. We further highlight the need for long-term, standardized monitoring programmes to contribute high-quality data to regional and global assessments.

1. Introduction

Amphibians and reptiles are experiencing among the most severe population declines documented in vertebrates (Wake and Vredenburg, 2008; Böhm et al., 2013). A recent global amphibian assessment estimated that around 41% of all known species are globally threatened (Luedtke et al., 2023). Additionally, global analyses using fossil-calibrated baselines confirmed that current amphibian extinction rates exceed background rates by several orders of magnitude (Alroy, 2015). Reptiles, on the other hand, have been historically less studied. However, they also display vulnerability, with roughly 21% of evaluated species now considered threatened, and a similar proportion showing evidence of declining populations (Böhm et al., 2013; Cox et al., 2022).

The causes of these declines are multifaceted and often synergistic. Habitat loss and degradation remain the most pervasive threats worldwide (Brooks et al., 2002; Newbold et al., 2015). Land-use change has led to the fragmentation and simplification of landscapes, reducing their potential to sustain herpetofaunal diversity (Böhm et al., 2013). Climate change exerts additional stress by altering temperature and precipitation regimes, affecting phenology and the availability of suitable microhabitats (Carey and Alexander, 2003; Sinervo et al., 2010). Pollution, invasive predators, and road mortality are factors that further

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exacerbate these pressures locally. Infectious diseases—especially chytridiomycosis caused by *Batrachochytrium dendrobatidis*—have been responsible for catastrophic collapses in amphibian populations across multiple continents (Skerratt et al., 2007; Scheele et al., 2019). However, only recently has chytridiomycosis become a global issue; whereas disease seemed to dominate amphibian declines in tropical montane systems during the 1980s and 1990s, land-use and climatic factors appeared to play stronger roles in temperate regions such as Europe (Buckley et al., 2014; Bosch et al., 2018).

The Mediterranean Basin hosts exceptional amphibian and reptile endemism (Myers et al., 2000). Iberia harbours ~37 amphibian and ~ 63 reptile species, including several narrow-range endemics of conservation concern. The Mediterranean lowlands and mountain systems concentrates a disproportionately high number of threatened European species (Crnobrnja-Isailović et al., 2025; Bowles et al., 2025). In the Iberian context, herpetofauna faces agricultural intensification in lowlands and the abandonment of traditional farming in mountain areas, altering habitats essential for breeding and thermoregulation. The progressive increase in drought frequency and temperature extremes has led to reduced hydroperiods in temporary ponds, affecting reproductive success and larval survival in amphibian species (Gómez-Rodríguez et al., 2010; Puig-Gironès et al., 2024; Crnobrnja-Isailović et al., 2025). Simultaneously, expanding forest cover in depopulated rural areas has modified open habitats critical for reptiles (Santos et al., 2016; Bowles et al., 2025). Invasive alien species pose additional localised but severe threats to native reptile assemblages, mainly through predation (Bowles et al., 2025). Chytridiomycosis has caused dramatic mass mortalities in Iberian-mountain amphibians (Thumsová et al., 2025), and its role driving widespread amphibian declines in temperate regions could be underestimated (Goodyear et al., 2026). The interplay between these pressures suggests that Iberian herpetofauna is undergoing complex, species-specific responses whose net outcome cannot be inferred from single-factor explanations (Puig-Gironès et al., 2024).

Despite the ecological and conservation relevance of Iberia, existing studies on population or range trends are often taxonomically restricted, spatially limited, or based on analyses of opportunistic data that do not explicitly account for imperfect detection and survey effort (Pleguezuelos et al., 2002; Loureiro et al., 2008). Modern analytical frameworks that explicitly separate detection artefacts from presence or abundance offer more reliable bases for inference (MacKenzie et al., 2003; Royle and Kéry, 2007; Guillera-Arroita, 2017). Here, we use dynamic occupancy models to estimate range dynamics by characterising local colonization and persistence processes, while correcting for imperfect detection. This framework is increasingly applied in large-scale biodiversity monitoring and to estimate temporal trends in species' distributions (Kéry et al., 2013; Briscoe et al., 2021; Kelleher et al., 2025). Although less sensitive than abundance estimates, occupancy analyses can also serve as useful indicators of population trends (Briscoe et al., 2021; Gaston et al., 2000; Passy, 2012). It is well established that sustained reductions in abundance are often associated with range contractions, allowing occupancy-based approaches to detect broad patterns of population declines (Fordham et al., 2012; Steenweg et al., 2018).

Our study provides the first comprehensive, quantitative synthesis of distributional trends in Iberian amphibians and reptiles over the last three decades, contributing to filling an important knowledge gap. We compiled an extensive database combining detection records from scientists and citizen-science programs collected between 1990 and 2024, covering the Spanish territory. We analysed a total of 62 species (25 amphibians and 37 reptiles) using hierarchical dynamic occupancy models fitted in a Bayesian framework. By focusing on species for which sufficient long-term data exist, we (1) identify broad-scale temporal range trends across the Iberian herpetofauna, (2) discuss plausible explanations for the differences in trends among taxa, and (3) highlight knowledge gaps where further mechanistic research is needed.

2. Methods

2.1. Data

The study was conducted across Peninsular Spain and the Balearic Islands, encompassing an area of approximately 500,000 km^2 defined by 5337 UTM grid cells of 10 km × 10 km based on the European Environment Agency reference grid. We chose this scale taking into account the characteristics of the detection data, computation demands and the level of detail in distribution estimates. While a 1 km × 1 km grid would be more homogenous in terms of local habitat, some of the data already came on a 10 km × 10 km resolution, and this scale is computationally more amenable for this multispecies nation-wide analysis. A 50 km × 50 km grid would have been too coarse to effectively detect changes in distributions.

Species data were obtained through the SIARE (Servidor de Información de Anfibios y Reptiles de España, https://siare.herpetologica.es) monitoring platform, coordinated by the Spanish Herpetological Association. This platform is supported by a long-term volunteer network aimed at monitoring the status and distribution of herpetofauna in Spain. The records used in this study, aggregated at the UTM grid cell level and limited to presence data, were based on observations collected during semi-structured field visits between 1990 and 2024. Visits are considered semi-structured because they correspond to citizen-science monitoring programmes as well as other opportunistic field surveys with various purposes. We excluded records derived from regional or national herpetofaunal atlases, which are typically based on intensive sampling strategies designed to maximize species detection rather than reflect standard observation conditions. With effort potentially spanning multiple years, these records could introduce a mismatch in temporal resolution compared to the rest of the records.

Detection/non-detection data are considered more informative than presence-only data, and allow using more powerful modelling techniques (Guillera-Arroita et al., 2015). To deal with the lack of recorded absences in the original dataset, for each species we imputed zeros at those sites and visits with no records, but with records from other species in the same group (considering reptiles and amphibians different groups). This imputation assumes that if a species belonging to one group was recorded, then any other species of the same group would have been recorded if it was detected. Separate survey records within sites and years were kept disaggregated and treated as replicate visits to account for detectability. For computational efficiency, and given our focus on quantifying distributional trends, we restricted our analyses of range dynamics to the known realised distribution range of each species, considering only those sites where the species was ever recorded at in the dataset. Therefore, we might not have captured the full extent of species' ranges, but a representative portion of it.

Recently described species with uncertain geographic boundaries were not considered. Thus, five species of the *Podarcis* genus (*P. hispanicus*, *P. virescens*, *P. vaucheri*, *P. liolepis*, *P. guadarramae*) were grouped into *Podarcis hispanica sensu lato*; the two *Timon* species (*Timon lepidus* and *Timon nevadensis*) were included into *Timon lepidus sensu lato*; *Psammodromus edwardsianus* and *Psammodromus occidentalis* were grouped into *Psammodromus hispanicus sensu lato; Alytes almogavarii* was included in *Alytes obstetricans sensu lato* and *Rana parvimalpata* was included into *Rana temporaria sensu lato*. After all the filtering and grouping we were left with 25 amphibian species and 37 reptile species.

2.2. Modelling framework

We used dynamic occupancy models to estimate occupancy probabilities for each species at each site and season from the detection/non-detection data (MacKenzie et al., 2003; Royle and Kéry, 2007; Kelleher et al., 2025). Dynamic occupancy models describe changes in site occupancy by explicitly modelling colonization and persistence (or alternatively extinction) probabilities as functions of environmental

covariates and other factors (such as temporal effects). For our analyses, we considered each UTM grid cell a sampling site, and each year a sampling season (reflecting the annual breeding cycles of many species). We assumed that occupancy could change between seasons but not within seasons, so we treated separate visits to the same site within a season as sampling replicates, which allowed us to estimate species detectability and account for imperfect detection (Guillera-Arroita, 2017).

The dynamic occupancy model assumes that there is a true occupancy state of each site i in season t, $z_{i,t} \in \{0,1\}$, where 1 represents that a site is occupied and 0 that it is not. This latent state is imperfectly observed, and governed by colonization and persistence probabilities as follows:

$$z_{i,1} \sim \text{Bernoulli}\left(\psi_{i,1}\right),$$

$$z_{i,t}|z_{i,t-1} \sim \text{Bernoulli}\left(\phi_{i,t-1}z_{i,t-1} + \gamma_{i,t-1}\left(1 - z_{i,t-1}\right)\right), \quad (1)$$

where $\psi_{i,1}$ is an initial occupancy probability, $\phi_{i,t-1}$ is a probability of persistence, which is the probability that a site remains occupied in season t given that it was occupied in season $t-1$, and $\gamma_{i,t-1}$ is a probability of colonization, which is the probability that a site becomes occupied in season t if it was not occupied in season $t-1$. In our model implementation, we made $\phi_{i,t}$ and $\gamma_{i,t}$ depend on covariates and temporal effects such that:

$$\phi_{i,t} = \text{logit}^{-1}\left(\text{X}_{i,t}\beta_\phi + \eta_{\phi,t}\right), \quad (2)$$

$$\eta_{\phi,t} \sim \text{normal}\left(0, \sigma_\phi\right)$$

$$\gamma_{i,t} = \text{logit}^{-1}\left(\text{X}_{i,t}\beta_\gamma + \eta_{\gamma,t}\right), \quad (3)$$

$$\eta_{\gamma,t} \sim \text{normal}\left(0, \sigma_\gamma\right)$$

where β_ϕ, β_γ are regression coefficients, η_ϕ, η_γ are temporal random effects for ϕ_t and γ_t, respectively, and $X_{i,t}$ is a matrix of covariate values. Given that we restricted our analyses to the sites with detections of the species, that we mostly expected range contractions and also that many of the species had relatively small distribution ranges, we did not expect much variation to explain with covariates in the first year of data, so we modelled initial occupancy probability as a constant across sites. Although this approach may limit detailed spatial inference, it is appropriate for estimating mean occupancy and therefore, it is unlikely to affect the assessment of temporal occupancy trends.

Then, recognising that the latent occupancy state $z_{i,t}$ might be unobserved, the dynamic occupancy model has an additional observation submodel which relates the observed binary detection/non-detection data $y_{i,t,j} \in \{0,1\}$, for site i, season t and visit j, to the latent occupancy states as follows:

$$y_{i,t,j}|z_{i,t} \sim \begin{cases} \text{Bernoulli}\left(p_{i,t,j}\right), & z_{i,t} = 1, \\ 0, & z_{i,t} = 0. \end{cases} \quad (4)$$

That is, if the species is absent, it cannot be detected, and if the species is present, it can be detected with a detection probability $p_{i,t,j}$. Due to lack of visit-level information (*e.g.*, effort or environmental conditions at the time of survey), in our implementation, we had no visit-level covariates for $p_{i,t,j}$, but we specified a temporal random effect, reflecting possible general changes in detectability across seasons, such that:

$$p_{i,t,j} = \text{logit}^{-1}\left(\beta_p + \eta_{p,t}\right), \quad (5)$$

$$\eta_{p,t} \sim \text{normal}\left(0, \sigma_p\right)$$

where β_p is an intercept and $\eta_{p,t}$ a temporal random effect for detectability in season t. Residual spatial heterogeneity in observer effort was unaccounted for, but it should not influence temporal trends.

All analyses were conducted in R (R Core Team, 2024) using some of the functionality from the tidyverse packages (Wickham et al., 2019) and the sf package (Pebesma and Bivand, 2023) to handle spatial objects. Models were fitted within the Bayesian framework, implemented using the probabilistic programming language Stan (Carpenter et al., 2017), through the R-package rstan (Stan Development Team, 2025). We ran four chains of the No-U-Turn Hamiltonian Monte Carlo (HMC) algorithm, with 4000 samples each. We used the first 2000 samples as adaptation and discarded them. We used the Gelman-Rubin statistic (Rhat) to assess model convergence, which compares within chain with between chain variance, considering that values below 1.01 indicate no evidence of lack of convergence (Vehtari et al., 2021).

We used relatively uninformative priors for the model parameters, as follows:

$$\beta_\phi, \beta_\gamma \sim \text{normal}(0, 1.5)$$

$$\sigma_\phi, \sigma_\gamma \sim \text{Half-Cauchy}\left(0, 1\right)$$

$$\sigma_p \sim \text{Half} - \text{Cauchy}(0, 2)$$

Thus, considering that data came from multiple observers and surveys occurred at different times of the year, we allowed greater potential annual variation in detection probability than in colonization or persistence probabilities. More uninformative priors produced similar results but on occasions prevented convergence.

2.3. Model covariates and selection

We considered four covariates that we thought could capture drivers of colonization and persistence processes in amphibians and reptiles: 1) average annual minimum temperature per UTM (Abatzoglou et al., 2018), 2) average annual rainfall per UTM (Abatzoglou et al., 2018), 3) average altitude per UTM (CNIG, 2021), and 4) percentage surface covered by protected areas per UTM (MITECO, 2024). Note that we used the most recent version of the Spanish Natural Protected Areas layer. Unlike the two climatic variables, this is a static variable aimed at explaining differences in persistence and colonization rates among sites, rather than changes over time. We assumed designated areas were, on average, in better conservation status even before formal protection.

Given the large number of species we had to run models for, and to facilitate interpretation of results across taxa, we analysed all species within a common modelling framework. More complex models could capture finer ecological differences, but they would require different sets of covariates and model structures for each species. Given that detailed causal inference on the drivers of change for every species analysed was beyond the scope of this work, we fitted relatively simple candidate models. The following three model structures were sufficiently general to be applied consistently and to converge for all species:

1) mod1 was spatially constant, with only (time-specific) intercepts in colonization and extinction.
2) mod2 included altitude, average annual rainfall and percentage of protected areas in the UTM as potential explanatory variables of temporal and spatial variation in colonization and extinction,
3) mod3 was similar to mod2, but included annual minimum temperature instead of altitude. We made this choice because there was collinearity between minimum temperature and altitude, and including both variables together led to identifiability issues during model fitting.

We always made colonization and persistence depend on the same variables and we standardized covariates before model fitting.

Although the objective of the analysis was not future prediction, we conducted model selection based on the ability of the models to predict

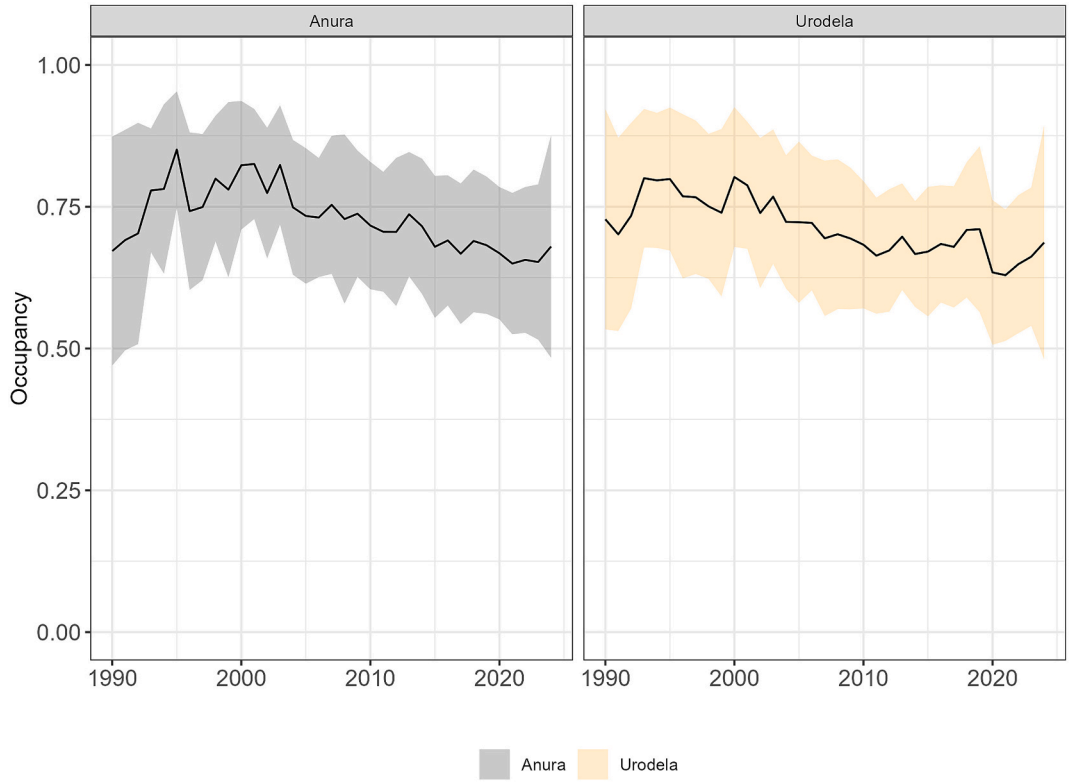

Fig. 1. Occupancy dynamics of the different amphibian groups between 1990 and 2024. The solid line corresponds to the estimated mean occupancy, while the shaded band corresponds to a 95% credible interval.

unobserved data, assuming that a model that predicts better is also more likely to describe better occupancy and detection processes, and their trends. We therefore fitted models using the first 23 years of data (1990–2012) and used the other 12 years (2013–2024) for testing. For each species, we chose the best of the three candidate models based on their mean log posterior predictive density across sites (LPPD),

$$\log P(\tilde{y}|y) = \log \frac{1}{M} \sum_{m=1}^{M} P(\tilde{y}|\theta^{(m)}),$$

where $\tilde{y}$ are left-out data, y are training data and $\theta^{(m)}$ are model parameters estimated in HMC iteration m with $min1,\ldots,M$. In our case, for each site i

$$P(\tilde{y}_i|\theta^{(m)}) = \sum_{\forall z_i} \left\{ \prod_{t=13}^{24} P\left(y_{i,t}|z_{i,t}\right) P(z_{i,t}|z_{i,t-1}) \right\}. \quad (6)$$

In Eq. 6, we compute the probability of the observation history at site i, given the sequence of latent species occupancy states at that site z_i, integrating over all possible state sequences. To compute the overall LPPD, we averaged over the LPPD values across all sites.

Once the best model was selected, we fitted the chosen model to the full data series for each species. We also calculated a season-level AUC (area under the receiver operating characteristic curve), as a complementary assessment of model performance to evaluate the ability of the model to discriminate between sites with any detections in a given year, from those with no detections. As an estimator for whether a site had detections in a given year ($\sum_j^J y_{i,t,j} > 0$ for cases with detection), we used the estimated probability of having at least one detection in site i and season t, assuming the same number of visits J in the dataset for that site and year (note that the number of visits J may change between sites and seasons):

$$P(\geq 1 \text{ detection})_{i,t} = \psi_{i,t}\left(1 - \prod_{j=1}^{J}\left(1 - p_{i,t,j}\right)\right). \quad (7)$$

As a means to synthesize results, we estimated a posterior distribution of linear trends in occupancy, fitting a linear model to the posterior samples of site occupancy over time (ψ_t) for each species. That is, for each species we estimated a linear fit to each of the posterior HMC samples of ψ_t for $t = 1,\ldots,35$, which resulted in a distribution of slopes of the temporal occupancy trend for each species. We repeated this process to estimate slopes for the last 5, 10, 15, 20, 25, 30 and 35 years.

We used Bayes factors to determine statistical support for our inference (Kass and Raftery, 1995). Thus, for any hypothesis H_1 on a parameter θ (*e.g.*, temporal occupancy trend < 0), against a null hypothesis H_0 (*e.g.*, temporal occupancy trend ≥ 0), we computed

$$BF = \frac{P(data|H_1)}{P(data|H_0)}.$$

We considered $3 < BF \leq 10$ to provide moderate statistical support for H_1 and $BF > 10$ to provide strong support.

3. Results

On average, amphibian species had 15,481 visits (SD = 12,567) with data for 34.8 years (SD = 0.5), and reptiles had 16,816 visits (SD = 14,693) with data also for 34.8 years (SD = 0.9). On average 29% (SD = 5.4%) of site-seasons had visit replicates for amphibians, resulting in 1.9 visits per site and season on average (SD = 0.3). For reptiles, an average of 31% (SD = 5.4%) had replicates, resulting in 2.1 visits per site and season on average (SD = 0.4). From the amphibian data, we calculated an observed average detection rate (number of visits where a species was observed divided by the total number of visits) of 0.27 (SD = 0.1), while for reptiles it was 0.19 (SD = 0.06). Both amphibian and reptile observed ranges were quite variable, with an average number of sites where

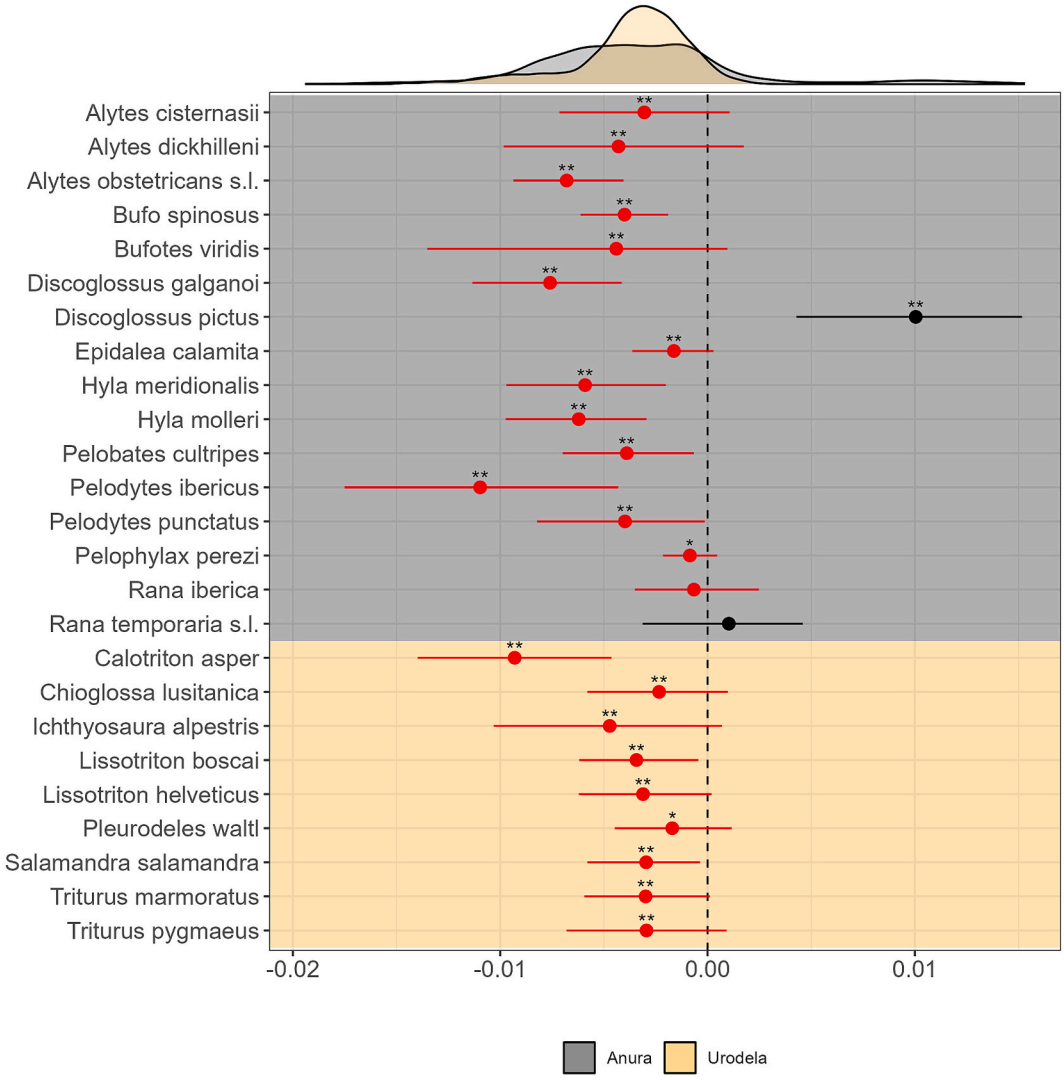

Fig. 2. Occupancy linear trend per amphibian species between 1990 and 2024. Negative trends are presented in red and positive in black. Statistical support for the trends is based on Bayes factors (BF): * moderate support (BF > 3), ** strong support (BF > 10). The densities shown on top correspond to the overall trend estimated for each taxonomic group. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

species were detected at least once of 4,233 (SD = 4,689) for amphibians and 3,077 (SD = 3,285) for reptiles (out of a total 5,337 sites).

Although for some reptile models AUC scores were above 0.8, most AUC scores were between 0.6 and 0.8 (mean AUC = 0.69, SD = 0.04 for amphibians, and mean AUC = 0.72, SD = 0.04 for reptiles) indicating that our models had relatively low discrimination capacity between site-seasons with species detection *vs* site-seasons with no detection (SI-fig. A1).

Overall, our model results indicated amphibian range contractions (amphibian mean trend = −0.003, SD = 0.004, BF = 7.7, see Figure 1and Fig. 2). In particular, 22 amphibian species showed occupancy declines (20 of them with BF > 10, Fig. 2) and only one species showed an increase (with BF > 10, Fig. 2). Urodele species consistently presented declines in their distributions (mean trend = 0.004, SD = 0.003, BF = 21.5). Most anurans also suffered range contractions (mean trend = −0.003, SD = 0.005, BF = 5.48), although some species seemed to have stable or even increasing ranges (Fig. 2). The largest range expansion relative to its current range was observed for *Discoglossus pictus* (mean trend = 0.010, SD = 0.003, BF = 499), although it is still restricted to the northeast corner of Spain and across the border with France. *Pelodytes ibericus* suffered the most severe range contraction of all amphibian species (mean trend = −0.011, SD = 0.003, BF = 1,332), being particularly pronounced in the early 2000s (SI-fig. A2). In terms of covariate effects across species, we found moderate support for the hypotheses that amphibian persistence was negatively correlated with rainfall (mean = −0.538, SD = 0.538, BF = 5.140, Fig. 3) and colonization was negatively correlated with elevation, although effects differed between species (mean = −0.316, SD = 0.616, BF = 3.13, Fig. 4).

Reptiles also showed a general negative trend in their range dynamics (reptile mean trend = −0.005, SD = 0.006, Fig. 5, Fig. 6, SI-fig. A3). Only 4 species increased their range (only 2 with BF > 10, Fig. 6), while 27 contracted their range (25 of them with BF > 10, Fig. 6). The most consistent declines in distribution ranges appeared in the testudines group (mean trend = −0.008, SD = 0.004, BF = 23, Fig. 5) - all four species suffered range contractions with strong statistical support (Fig. 6). Although less consistently across species, lacertids (mean trend = −0.004, SD = 0.007, BF = 4.73, Fig. 6) and ophidians (mean trend = −0.003, SD = 0.005, BF = 2.32, Fig. 6) also experienced generalised range contractions. We found statistical evidence for one lacertid species increasing its distribution range and 10 decreasing their range (Fig. 7). We found no evidence of change in the other three species of the group (Fig. 7). The largest expansion in the lacertids group was observed for *Podarcis hispanica* s.l. (mean trend = 0.010, SD = 0.001, BF > 1000,

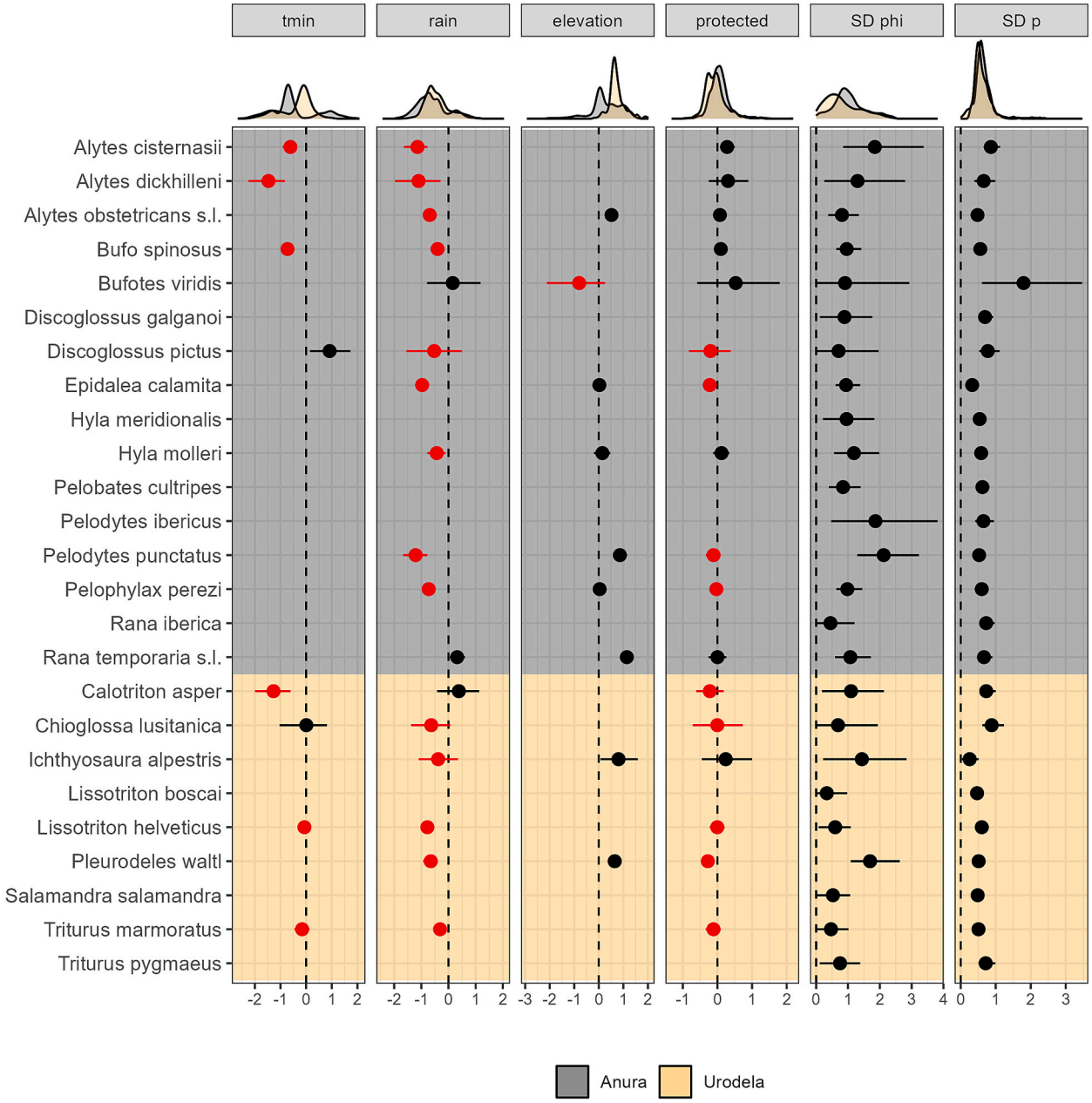

Fig. 3. Covariate effects on persistence of the different amphibian species. The densities shown on top correspond to the overall parameter value estimated for each taxonomic group.

Fig. 6) and the strongest decline in occupancy was observed for *Psammodromus hispanicus* s.l. (mean trend = −0.018, SD = 0.002, BF > 1,000, Fig. 6), a species that occupies a large portion of the Iberian peninsula, and showed a generalised decline in occupancy that was particularly strong in the early 2000s (SI-fig. A3). We found statistical support for range declines in 8 species of ophidians, for range increases in 3 species, and we found no evidence of change in the other 2 species (Fig. 6). We found the strongest decline in the ophidians group for *Hemorrhois hippocrepis* (mean trend = −0.012, SD = 0.002, BF > 1000, Fig. 6) and the strongest increase for *Macroprotodon cucullatus* (mean trend = 0.003, SD = 0.003, BF 4.34, Fig. 6). Other species not belonging to any of the previously mentioned groups also suffered overall contractions (mean trend = −0.007, SD = 0.004, BF = 53, Fig. 5). In relation to environmental conditions, effect differed among species and we only found moderate statistical support for reptile mean persistence being negatively correlated with minimum annual temperature (mean = −0.575, SD = 0.622, BF = 3.64, Fig. 7, Fig. 8).

Both for amphibians and reptiles, temporal effects were quite large compared to the effects of covariates (Fig. 3, Fig. 4, Fig. 7, Fig. 8).

4. Discussion

Our analyses provide a comprehensive, empirically-grounded assessment of long-term changes in the realised distributions of Spanish amphibians and reptiles. We address a major gap in large-scale evaluations capable of disentangling distribution trends from short-term fluctuations and observer bias. By applying dynamic occupancy models to several decades of monitoring data and explicitly accounting for imperfect detection, we demonstrate that range contractions are generalised across Iberian herpetofauna. Across the 62 taxa analysed, a large majority of species showed declining trajectories in realised distributions, although of different magnitudes and with variable certainty.

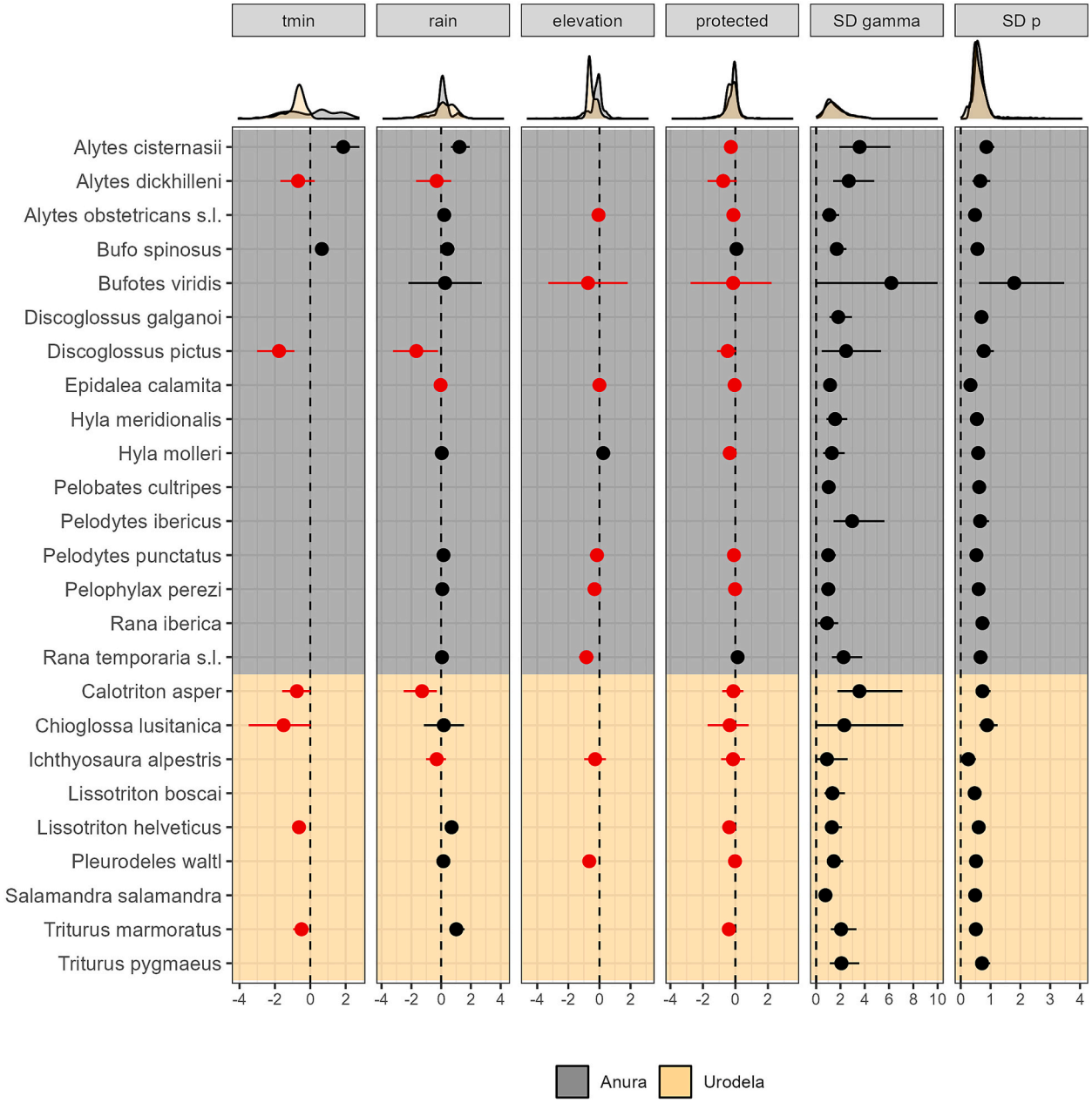

Fig. 4. Covariate effects on colonization of the different amphibian species. The densities shown on top correspond to the overall parameter value estimated for each taxonomic group.

Assuming that distribution dynamics are driven by population changes (Gaston et al., 2000; Steenweg et al., 2018), these findings would also suggest generalised population declines. However, it is important to note that occupancy-abundance relationships might be affected by survey design and by species mobility (Steenweg et al., 2018). Although occupancy-based approaches can serve as useful indicators of broad patterns of population decline, they are less sensitive than explicit abundance studies. For example, we observed that *Podarcis pityusensis* appeared stable, but the species is undergoing population declines due to introduced snakes (Montes et al., 2022). Therefore, it seems that the species remains consistently present in most of its range, but probably at lower densities. Despite this limitation, our study shows declining trends representing a widespread phenomenon affecting multiple taxonomic groups, ecological strategies and distribution types. We provide one of the most extensive empirical assessments of amphibian and reptile population change conducted to date in southwestern Europe.

When trends are examined by major taxonomic groups, we identify differential responses among amphibians and reptiles. Amphibians show an overall negative trend, with declines affecting nearly all species analysed. Urodeles are particularly consistent in this respect, with all species exhibiting negative trends, indicating range contraction even in taxa that remain globally classified as Least Concern, such as *Calotriton asper* or *Ichthyosaura alpestris*. Consistency across species suggests that declines in salamanders and newts are not driven by isolated events or localised collapses, but rather by broad-scale processes acting across much of their Iberian range. Anurans show greater heterogeneity, with most species declining, but with *Discoglossus pictus* exhibiting a positive trend and *Rana temporaria* s.l. with a stable range. This variability likely reflects differences in life-history strategies, habitat breadth and sensitivity to short-term environmental variability, yet the predominance of negative trajectories indicates that anurans are also subject to widespread pressures.

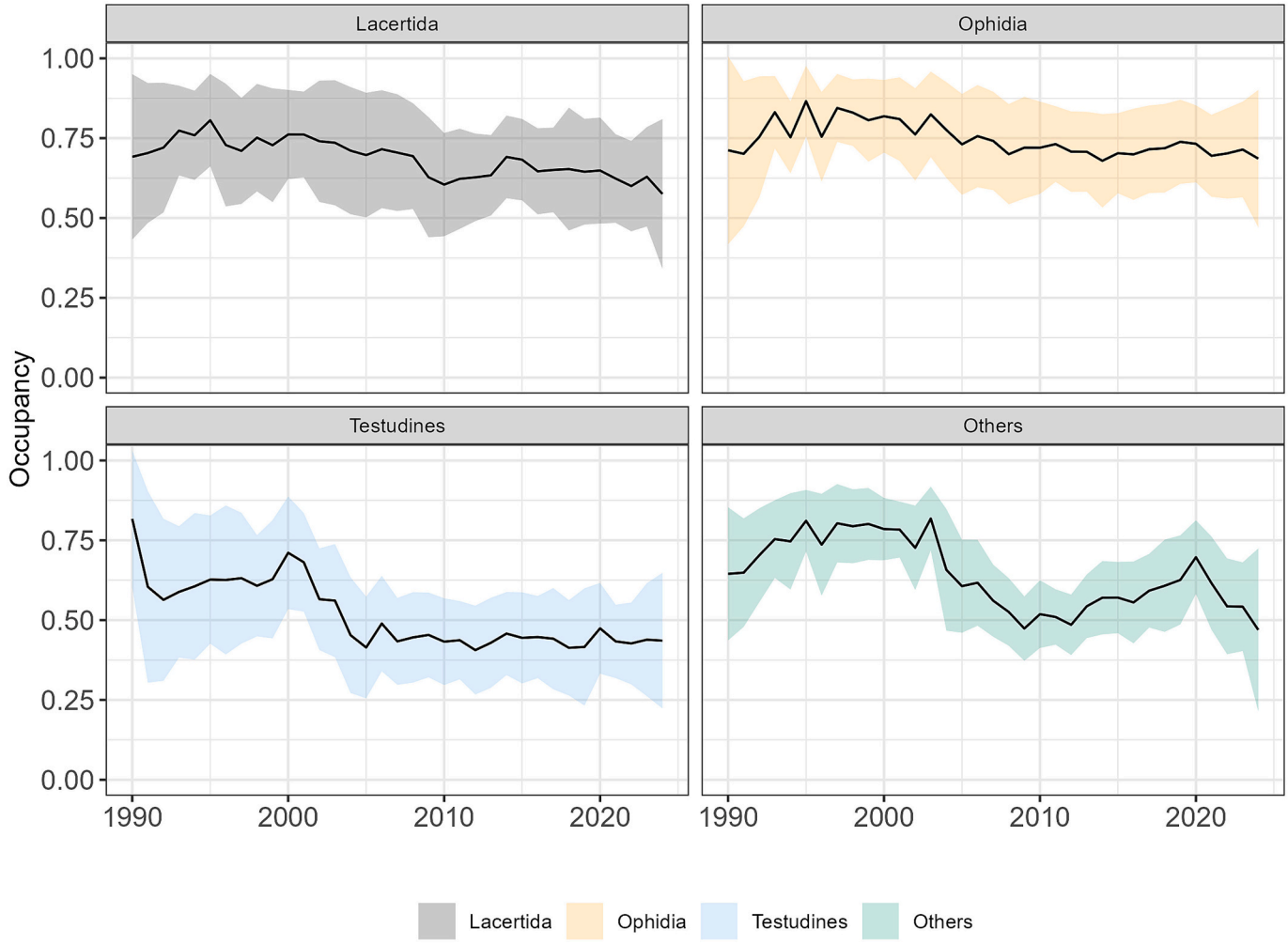

Fig. 5. Occupancy dynamics of the different reptile groups between 1990 and 2024. The solid line corresponds to the estimated mean occupancy, while the shaded band corresponds to a 95% credible interval.

Reptiles display stronger average declines than amphibians (although also more uncertain), with particularly pronounced negative trends in chelonians. This group includes long-lived species with delayed maturity and low reproductive rates, traits that can buffer populations against short-term environmental variability, but also make them especially vulnerable to chronic pressures (Morris et al., 2008). For this reason, we hypothesise that the steep declines detected in chelonians may reflect long-standing demographic erosion. Squamate reptiles also exhibit widespread declines, though with greater interspecific variability. Importantly, some of the strongest negative trends occur in widespread species that are often perceived as resilient or common, such as lizards in the *Timon* genus, snakes like *Hemorrhois hippocrepis* or geckos like *Tarentola mauritanica*. This finding challenges a persistent assumption in herpetological conservation and underscores that broad distribution does not necessarily confer demographic stability in landscapes undergoing rapid environmental change (Nowakowski et al., 2017; Xu et al., 2023). Furthermore, for snakes, our results obtained through dynamic occupancy modelling differ from previous assessments. Particularly notable, are discrepancies in the trends of *Hemorrhois hippocrepis* and *Hierophis viridiflavus*, for which our results show clear range contractions, while other analyses using the software TRIM, considered their populations stable (Santos et al., 2022). It is unclear though whether these differences are produced by the different methodologies or differences in the data used.

The temporal structure of the estimated occupancy dynamics provides insights into how these declines may have unfolded, consistent with progressive and temporally lagged population changes rather than abrupt collapses. For amphibians, mean species occupancy increase from approximately 0.7 in the earliest seasons to values exceeding 0.8 during intermediate periods, before stabilising or showing incipient declines in later years (Fig. 1). This pattern suggests negative overall trends may stem from cumulative processes acting after an initial phase of relative stability or even expansion. Emerging evidence indicates that temperate amphibians may experience slow, disease-mediated declines lacking conspicuous mass mortality events, yet capable of driving long-term population erosion (Goodyear et al., 2026). Therefore, we hypothesise that disease-driven impacts may be contributing more broadly to recent declines than currently acknowledged. Subgroup-specific temporal dynamics further refine this interpretation. Urodeles display relatively muted temporal variability, with gradual declines emerging only in later periods (Fig. 1). Although just a hypothesis, this pattern aligns with long-term attrition rather than episodic crashes and suggests that population erosion in salamanders and newts may proceed slowly and evade detection without extended time series. Conversely, Anurans exhibit stronger temporal fluctuations, including early increases followed by stabilisation or decline (Fig. 1) - potentially reflecting greater sensitivity to short-term environmental variables, like hydroperiod (Crnobrnja-Isailović et al., 2025). In general, we observed that amphibian persistence decreased in those areas within their range with more rainfall and colonization decreased at higher altitudes. Reptiles showed a clearer sustained occupancy decline over time, with mean occupancy decreasing from around 0.74 in early seasons to

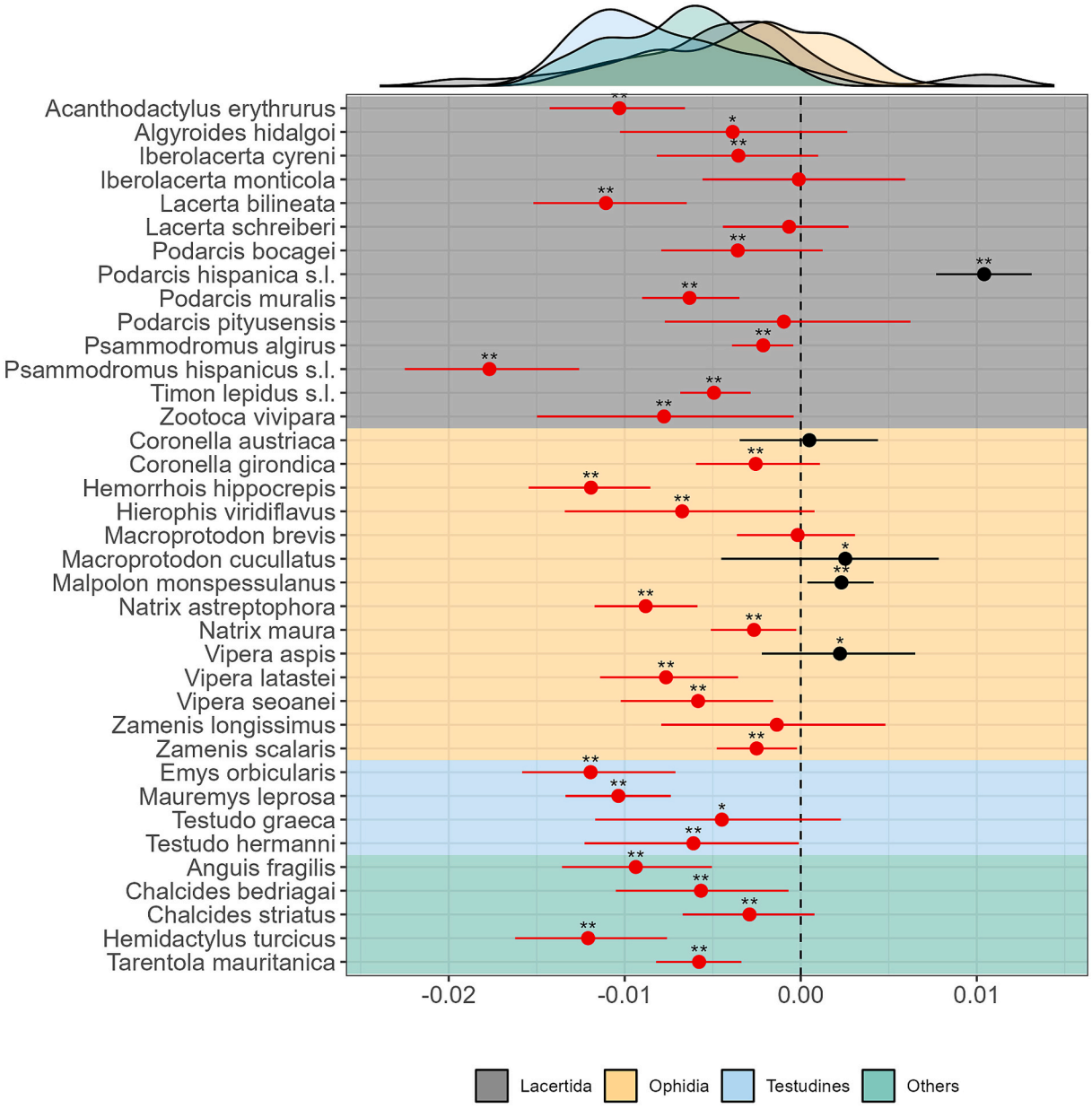

Fig. 6. Occupancy linear trend per reptile species between 1990 and 2024. Negative trends are presented in red and positive in black. Statistical support for the trends is based on Bayes factors (BF): * moderate support (BF > 3), ** strong support (BF > 10). The densities shown on top correspond to the overall trend estimated for each taxonomic group. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

approximately 0.64 in the most recent periods (Fig. 5). Looking at subgroups, chelonians show high occupancy early in the time series followed by declines in later periods (Fig. 5). This pattern could be interpreted as being indicative of delayed demographic responses to chronic threats such as habitat loss, fragmentation and road mortality (Bowles et al., 2025). Squamate reptiles show more heterogeneous trajectories, but declines generally become more pronounced in recent periods (Fig. 5), suggesting that population losses may be relatively recent rather than long-standing. We found that reptiles' persistence mostly decreased in warmer areas within their range.

Although the effects of environmental covariates are perhaps counterintuitive at a first glance, these might be explained by the scope of our study. On a large scale, reptiles might be selecting areas with high temperatures, but within their distribution range (scope of our analysis), areas with the highest temperatures might not be optimal. Same could be said about amphibians and high rainfall areas: areas of highest rainfall within their range might not be optimal. However, we note that these interpretations arise from the scale and design of the analysis, rather than from direct tests of ecological mechanisms. In addition, we might not see clearer effects of environmental variables in persistence and colonization, because we are analysing the conditions within each species range and not across the whole of Spain. Variation in environmental conditions within species ranges is smaller and so patterns more subtle. Then, the large estimated temporal random effects in relation to the effect of covariates could suggest that there are relevant environmental drivers or other sources of heterogeneity not included in the models. This would have greater implications if our main aim was spatial prediction, but less so for quantifying range trends.

In a broader context, the magnitude and prevalence of declines detected in Spain are consistent with global assessments of amphibian and reptile conservation status (Crnobrnja-Isailović et al., 2025; Bowles et al., 2025), yet they also reveal important discrepancies that are highly informative from a conservation perspective. While global IUCN assessments integrate information across entire species ranges, our results

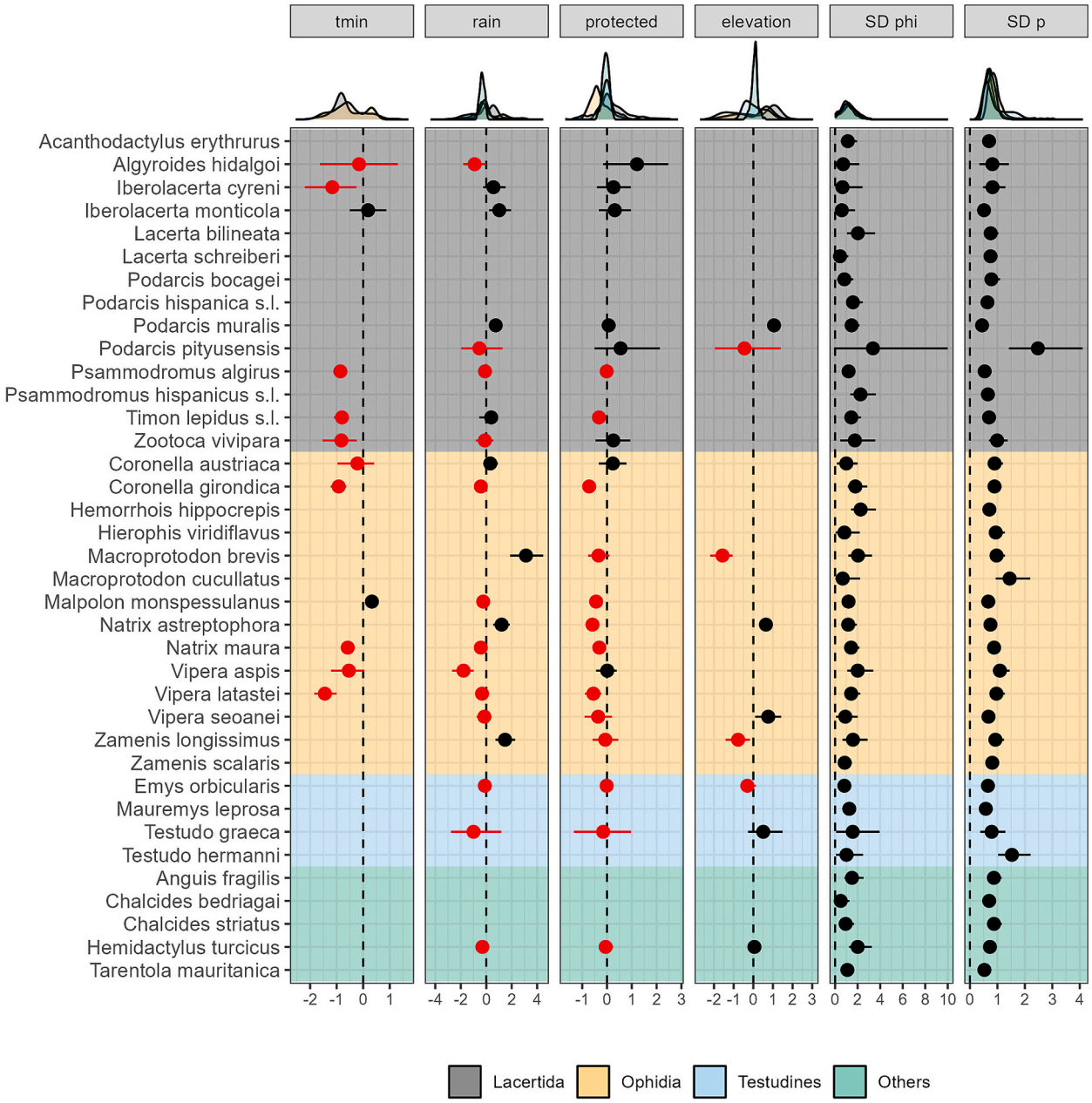

Fig. 7. Covariate effects on persistence of the different reptile species. The densities shown on top correspond to the overall parameter value estimated for each taxonomic group.

show that many species classified as Least Concern or Near Threatened at the global level, exhibit sustained and sometimes strong declines within Spain. This decoupling highlights a key limitation of range-wide assessments for taxa with broad distributions: regional declines may be masked when trends are averaged across global scales (Rodríguez et al., 2000; Gärdenfors, 2001; Moser et al., 2016). Our findings therefore reinforce the argument that national- or regional-scale analyses are essential complements to global assessments, particularly in biodiversity hotspots such as the Iberian Peninsula, where environmental gradients, land-use pressures and historical trajectories differ markedly from those of other parts of a species' range. Our research also considers a longer time span than those of IUCN assessments (typically 10 years or 3 generations, IUCN (2012)) and, in some instances, long-term trends are still negative, even though in recent years they might have reverted (SI-Table A1 and SI-Table A2), as it is the case of *Alytes cisternasii* or *Discoglossus galganoi*.

Comparing the empirical population trends with the global conservation status of the species, as well as their status under the Spanish legal protection framework, reveals a systematic lag between demographic change and conservation recognition. Many species experiencing strong and sustained declines, such as *Psammodromus hispanicus* s.l., *Hemorrhois hippocrepis* or *Hemidactylus turcicus*, are classified globally as Least Concern or Near Threatened and are protected nationally only through inclusion in the Spanish List of Wild Species under Special Protection (LESPRE), without formal recognition as Vulnerable or Endangered in the Spanish Catalogue. While inclusion in LESPRE provides baseline legal protection, it does not necessarily imply prioritisation, targeted management or enhanced monitoring. Consequently, conservation frameworks may be slow to respond to declines in broadly distributed species whose regional dynamics are difficult to infer from local observations alone. These findings highlight a mismatch between the scale of observed declines and the scope of effective conservation action.

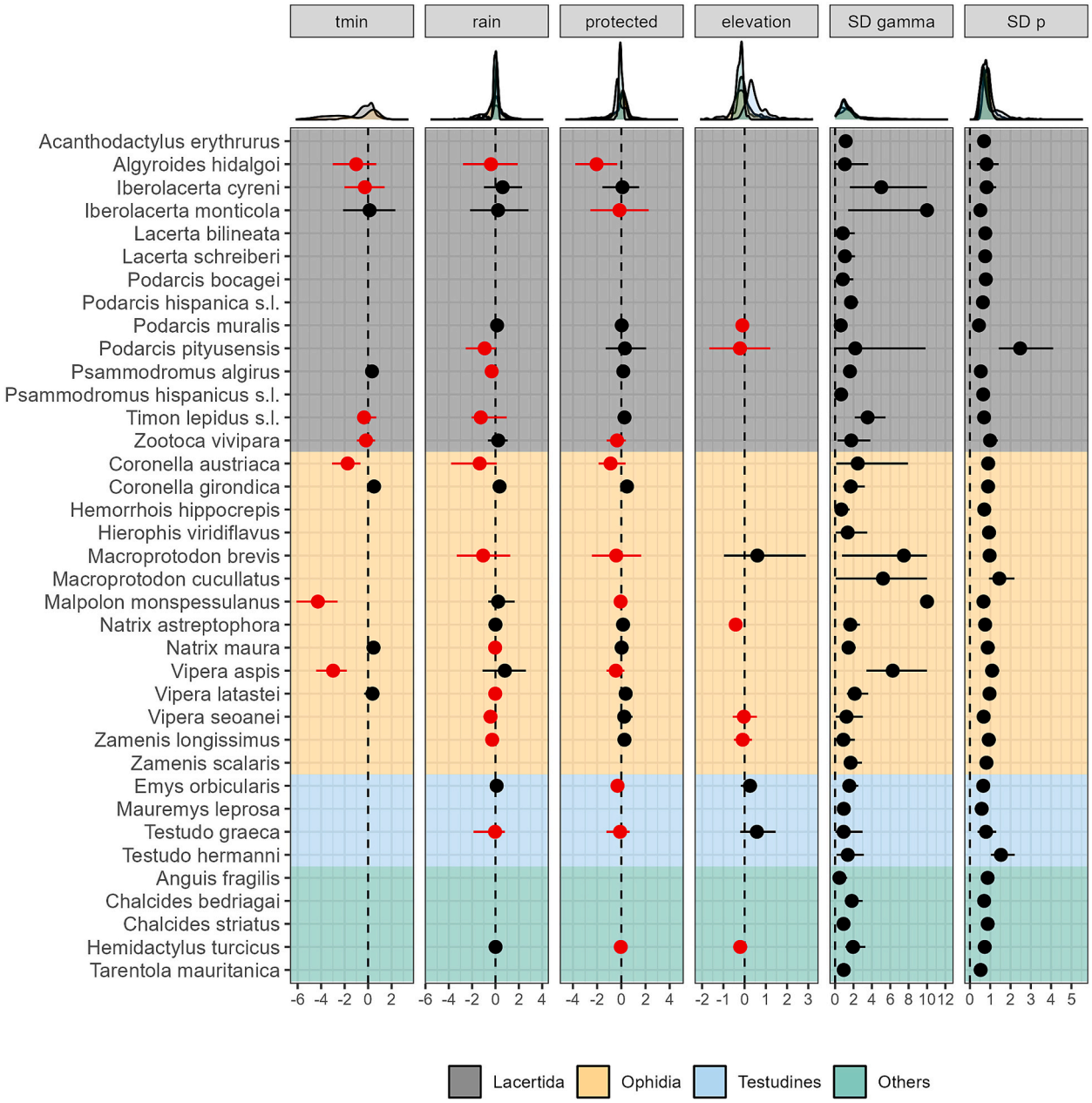

Fig. 8. Covariate effects on colonization of the different reptile species. The densities shown on top correspond to the overall parameter value estimated for each taxonomic group.

Conservation efforts at both European and national levels remain concentrated on a relatively small subset of emblematic or narrowly distributed species, such as *Alytes dickhilleni* or *Algyroides hidalgoi*, while many declining but widespread taxa are assumed to be secure and receive little targeted intervention. Although expert knowledge is indispensable—particularly for endemics and highly localised taxa—our results demonstrate that reliance on expert judgement and coarse global indicators alone may delay recognition of declines in widespread species, adding to similar observations made for other groups (Aizpurua et al., 2015). Population dynamics in widespread taxa can vary markedly across regions, and declines may be spatially heterogeneous or recent, making them difficult to detect without large-scale empirical analyses.

Overall, this study demonstrates the critical value of long-term, observation-based analyses for understanding range and population change in amphibians and reptiles, particularly in taxa with complex life histories or demographic buffering. However, it is important to acknowledge the limitations of the available data. Many of the records used in this study are opportunistic in the sense that they come from miscellaneous sources, and therefore visits to sites are not framed within a structured monitoring programme. Observers decide when and if they want to report on the species detected, which may result in under-reporting of some species that might be perceived as common or otherwise uninteresting. Therefore, there is no guarantee that those species not reported were actually not observed, or that all visits followed the same protocol. Despite the specialized background of AHE participants, observer reporting bias may have artificially inflated the detectability of notable species while underestimating that of common ones. Modelling and data wrangling can help with certain issues, like inferring non-detections from surveys detecting other species, but these fixes are limited in what they can do, and they may contribute to uncertainty in occupancy estimates. Other modelling choices, although

thought to minimise potential bias, might have also influenced the results. For example, considering only the obser ed realised range of the species to study their range dynamics might have biased our results towards what happens in more accessible areas, where data are easier to obtain. However, given the time length of the dataset it is unlikely that the proportion of undetected populations is large. The lack of information about survey conditions prevented us from modelling the observation process in more detail, which could have helped better separate the ecological and observation processes. Thus, we advocate for more efforts and funding for structured, long-term monitoring programmes and observation networks that provide consistently structured and accessible data.

By integrating three decades of monitoring data within a robust modelling framework, we reveal widespread, progressive range contractions that are only partially reflected in current global assessments and conservation frameworks. While expert knowledge remains essential, particularly for species with restricted distributions, our results show that empirical, large-scale trend analyses are indispensable for detecting regional declines in widespread taxa. Incorporating such quantitative evidence into conservation assessment and prioritisation processes would improve the sensitivity and responsiveness of conservation frameworks, helping to identify emerging declines before they become entrenched and substantially more difficult to reverse.

CRediT authorship contribution statement

Francisco Cervantes Peralta: Writing – original draft, Software, Methodology, Formal analysis. **Jaime Bosch:** Writing – original draft, Methodology, Data curation, Conceptualization. **Barbora Thumsová:** Writing – review & editing, Data curation, Conceptualization. **José J. Lahoz-Monfort:** Writing – review & editing, Methodology, Formal analysis, Conceptualization. **Gurutzeta Guillera-Arroita:** Writing – original draft, Methodology, Formal analysis, Conceptualization.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work the authors used ChatGPT (chatgpt.com) in order to improve the English language. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

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. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2026.112056.

Data availability

All data used in this study are property of the Spanish Herpetological Association (https://herpetologica.es/) and contains sensitive information affecting the protection of threatened species. Access to these data can be provided for review and upon justified request. All the code used is available on GitHub (repository to be confirmed).

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