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Challenges and opportunities for assessing trends of amphibians with heterogeneous data - a call for better metadata reporting

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Abstract

Over the last decades, the worldwide decline of amphibian populations has become a major concern of researchers and conservationists. Studies have reported a diversity of trends, with some species declining seriously, others remaining stable and still others increasing. Only a few species have been monitored annually for a long period of time by specific monitoring programs. However, there are many heterogeneous datasets that contain observations of amphibians from professional surveys as well as diverse citizen science and other voluntary surveys. The use of these data brings a number of challenges, raising concerns about their validity and use in ecological research and applied conservation. We assessed to what extent such heterogeneous occurrence data can inform on the status and trends of amphibians by contrasting different approaches to overcoming challenges with the data, using the German state of Saxony as an example. We assessed the effects of data processing decisions to infer absences, the use of survey method information, and the statistical model (generalized linear mixed-effect occurrence model [GLMM] versus occupancy-detection model) and compared the trends with expert opinions (Red lists). The different data processing decisions mainly led to similar annual occupancy estimates, newts being an exception. Annual occupancy estimates were typically less certain when attempting to account for the effects of survey methods, which could be explained by many missing values on methods. Separate models for drift fence data reduced the uncertainty in the annual occurrence probability estimates of the GLMM models, but uncertainty remained high for occupancy-detection models. For both methods, strong peaks and troughs in the annual occupancy estimates occurred for several species, which were not biologically plausible. Some peaks align with periods of lower sampling effort and were probably caused by shifts in the sampling locations or target species among years. Only for three species (*Bufotes viridis, Hyla arborea,* and *Pelophylax esculentus*) were the trend results consistent among approaches and with expert opinions. For most other species, some inconsistencies appeared among models or approaches, indicating that trend assessments are sensitive to analytical choices. While heterogeneous data have proved useful for other taxa, our results highlight the complexity of using them for amphibians. We strongly recommend better harmonization of data collection and documentation, including explicit absence data, to enable more robust trend assessments in the future.

Keywords

Amphibian conservation, Anura, Urodela, citizen science data, drift fence data, data filtering, Generalized Linear Mixed model, occupancy-detection model, survey methods, Germany, Saxony

Introduction

Over the last three decades, the worldwide decline of amphibian populations has become a major concern of researchers and conservationists (Henle and Streit 1990; Pechmann et al. 1991; Stuart et al. 2004; Luedtke et al. 2023). Currently, amphibians are among the taxa with the highest extinction rate worldwide (Henle et al. 2008; Catenazzi 2015; Falaschi et al. 2019; Luedtke et al. 2023). A range of drivers contribute to the decline of amphibians, such as habitat loss and fragmentation, alien species, climate change, pollution, trade in species and pathogens, including chydridiomycosis (Henle and Streit 1990; Stuart et al. 2004; Falaschi et al. 2019; Rote-Liste-Gremium Amphibien und Reptilien 2021; Luedtke et al. 2023). Often, several of these factors interact in driving trends of amphibian species (Hayes et al. 2010). However, within Europe, a diversity of trends has been reported, with some species increasing, others remaining stable and still others declining seriously (Henle et al. 2008; Falaschi et al. 2019).

Many species of amphibians show substantial natural fluctuations in population size over years, which make it challenging to assess trends and isolate human impacts (Pechmann et al. 1991). However, only a few species have been monitored annually for a long period of time by specific monitoring programs (Vershinin et al. 2015; Mihoub et al. 2017; Falaschi et al. 2019). On the other hand, long-term and large-scale data exist from drift fences erected to save migrating amphibians from being killed by cars when crossing roads (Petrovan and Schmidt 2016; Seyring et al. 2024a). There is also a large amount of large scale data on amphibians from citizen science (Loman and Andersson 2007; Bonardi et al. 2011; Seyring et al. 2024b) and various databases that reach back more than half a century in Central European countries and elsewhere (e.g. Cabela et al. 2001; Kuzmin 2013; Sillero et al. 2014; Vershinin et al. 2015; DGHT 2018; Bowler et al. 2022; Seyring et al. 2024a, 2004b).

Large aggregated databases, as available for amphibians and many other taxonomic groups, are a compilation from a range of activities (citizen scientists, conservation organizations, research institutions, conservation agencies, voluntary surveys and others), usually without a common standard for data collection and documentation (Cabela et al. 2001; Sindaco et al. 2006; Osborne and Hoefer 2018; Seyring et al. 2024b). Such databases are increasingly used in studies of large-scale, long-term species trends, because they are often the only data that are available over a long

time period (Vershinin et al. 2015; Powney et al. 2019; Outhwaite et al. 2020; Sheard et al. 2021; Bowler et al. 2022). However, the use of these data brings a number of challenges, raising concerns about their validity and use in ecological research and applied conservation (Burgess et al. 2017; Bayraktarov et al. 2019). Insufficient or heterogeneous metadata associated with each species observation usually prevents separating records collected based on standardized sampling protocols from those collected more opportunistically (Turner 2023). Also, usually only species detections are recorded in the database, which means that they do not provide consistent data on where species have not been recorded (absence data). Finally, these data are also often strongly spatially biased (Geldmann et al. 2016; Bowler et al. 2022), which can lead to biased trend estimates (Bowler et al. 2022).

Despite these challenges, these aggregated databases often hold the most comprehensive information on spatio-temporal patterns of species occurrences. Given the importance of knowledge on species trends to conservation decision-making, testing the use and limits of these heterogeneous data is a key research area (e.g. Bowler et al. 2022, 2004; Turner et al. 2023; Della Rocca et al. 2024). Here, we test whether these data can reveal insight into changes within amphibian populations. Amphibians provide particular challenges because of their biphase life-history (aquatic - terrestrial) and the large number of different methods that are used to survey them. At the same time, amphibians are particularly suited to address this question because they show a much lower dispersal ability (Smith and Green 2005; Trochet et al. 2014) than species groups often used for such analyses, such as dragonflies (Bowler et al. 2021), birds (Kamp et al. 2021; Rigal et al. 2023) or butterflies (Arfan et al. 2018; Warren et al. 2021). Thus, they are unlikely to show substantial distributional expansion and retraction within a few years; nonetheless, they are susceptible to many environmental changes, especially because of their semi-permeable skin. Moreover, their biology is rather well known and they are a compact, easily recognizable and distinguishable species group (Günther 1996; Grossenbacher and Thiesmeier 2004; Grossenbacher 2009, 2012, 2014).

In Germany, the last Red List assessment (Rote-Liste-Gremium Amphibien und Reptilien 2020) showed a substantial decline of most species, with both rare and common species being affected, as in other parts of Central Europe (Denoël et al. 2013; Petrovan and Schmidt 2016; Chiacchio et

al. 2022). Large-scale biodiversity data, including for amphibians, are mainly collected at the federal state level. We selected the federal state of Saxony as a case study (Fig. 1) because it has the largest number of data per grid cell of all German states (DGHT 2018; Seyring et al. 2024b). Because of continuous environmental changes after the break-up of the German Democratic Republic in 1990, we expected to reveal changes in species' distributions. In other parts of Central Europe with similar environmental changes after the associated break-up of the Soviet Union, these changes are known to have affected various taxa, such as birds (Reif et al. 2011).

We examined the value of the Amphibian species database of Saxony (LfULG undated) to assess long-term trends in species occupancy. This database is run by the Landesamt für Umwelt, Landwirtschaft und Geologie (LfULG; Federal Administrative Office for Environment, Agriculture and Geology), dedicated to store occurrence data and already used by the conservation authorities to assess the distributions of endangered species. Like other such databases, it was not specifically designed for trend analysis. It contains a mixture of truly opportunistic data from conservation workers; more systematic short-term project-based data (e.g. institutional or appointed monitoring projects) and data from long-term programs, such as from protective drift fences along roads (Seyring et al. 2024a). We assessed to what extent this heterogeneous occurrence data can inform on the status and trends of amphibians by contrasting different approaches to overcoming challenges with the data. We contrast approaches concerning the derivation of absence data and the statistical model to estimate trends. Moreover, we take advantage of available metadata on survey methods and projects to compare the trend estimates derived from all data and those derived only from drift fences, which we consider the most standardized long-term data. Finally, we assessed the plausibility of all trend estimates using expert opinions (Red List) on species trends in the study region.

Methods

Study region

The study region consists of the federal state of Saxony in the central east of Germany (Fig. 1). It is composed primarily of lowland but with hilly to montane areas in the south. It contains numerous water bodies and wetlands. Land use changed considerably since 1990 following the re-unification with Western Germany. Extensive mining areas in the south-east of Leipzig and

the lowlands of Upper Lusatia were partly abandoned and flooded. Large amounts of farmland were turned into built-up land. Streets and especially highways were reconstructed, extended or built (Haase et al. 2007, Schmidt et al. 2015). There were also increases in the intensity of land use, which has led to the loss of many small, ephemeral ponds (Walz and Stein 2014, Grunewald and Naumann 2015). Since 2018, several years of droughts have led to frequent drying up of many remaining small, ephemeral pond habitats (Boergens et al. 2020) and thus leading to amphibian losses in the broad landscape as well as in protected areas (Chiacchio et al. 2022). However, there were also positive effects of changes: the transfer of environmental standards was followed by many activities to improve the quality of air, soil and water. Unfortunately, many of these effects were reversed by the increased use of nutrients and pesticides (Brühl et al. 2013; Kleeberg et al. 2016; Berger et al. 2018; Bub et al. 2023).

Species Dataset

We used a dataset of occurrence records collected in Saxony up to the year 2020 (Fig. 1), compiled in the Central Species Database of Saxony (Zentrale Artdatenbank, ZenA) by the regional conservation authorities (LfULG undated). The data are available across most of the state but with different densities in different regions (Fig. 1). The raw database contained 174,902 records for amphibians in 2020 (after removing duplicates), with a median observation year of 2007 (interquartile range = 1997-2014, range = 1907-2019). Here, we focused on data collected between 1997 and 2019, when the majority of the data were collected (1997 was the first year with over 2000 surveys), with fluctuating effort through time (Fig. 2A).

The data cover 16 species: the newts *Lissotriton vulgaris* and *Triturus cristatus,* and the anurans *Bombina bombina, Bufo bufo, Bufotes viridis, Epidalea calamita, Hyla arborea, Pelobates fuscus, Pelophylax* kl. *esculentus, P. lessonae, P. ridibundus, Rana arvalis, R. dalmatina* and *R. temporaria*. Some records were additionally available for *Bombina variegata* and *L. helveticus* but they were excluded from the analysis due to identification uncertainty and low number of records, respectively. We also excluded *Salamandra salamandra* because this species never concentrates at breeding sites, which makes population assessment methodologically not comparable to the other species.

Organising detection methods

The database includes some metadata on detection/survey methods, but this was not standardized. Various methods were used to collect the data, including observation by acoustics or sight and of dead individuals, as well as various types of traps and drift fences and capture by hand. We grouped similar detection methods together to harmonize the method names and reduce the number of categories (Fig. 2B). A column termed “Project origin” contains information to identify fence data and data that were collected for the EU Habitats Directive reporting, among others. We used these to create additional method columns for fence data and official monitoring data. Still, even for the drift fences, the metadata is insufficient to know whether the data collection was standardized across years and sites (Seyring et al. 2024b). Moreover, in many cases, even the sampling method remained unknown (Fig. 2).

Filtering data

Some of the data rows needed to be removed for diverse reasons, often relying on specific expert knowledge of the database (LfULG, pers. comm.). We removed data with the source "LfULG: Amphibienkartierung, Zusammengefasste Nachweise" from 1997 as these indicate a duplicated summary of the previous data, which was used for compiling the publication of the latest amphibian distribution atlas for Saxony (Zöphel and Steffens 2002). We further removed data without coordinates or with coordinates with fewer than 4 decimal places (i.e., imprecise) or which did not overlap the boundaries of Saxony. The data contains an ‘Anzahl’ (count in German) column, but this had been used to record true absences as well as unknown abundances. To account for this, we removed rows if ‘Anzahl’ was zero, unless an ‘Einheit’ (unit for life stage of individuals) was specified, indicating that the zero likely reflected unknown abundance rather than zero abundance. We also removed records not identified to species-level. Finally, for our analysis, we limited the data to the months March to August since this is the period during which most sampling takes place, as the species covered by us migrate to aquatic habitats for breeding and undergo their ontogenetic development during that time period.

Identifying sites

Because the data were mostly not collected as part of structured monitoring, there was no site identifier code in the data set to identify which data were collected in the same place. However, repeated sampling is the backbone of long-term monitoring to assess change. We grouped

together occurrence point data into sites based on a cluster analysis applied to the geographic coordinates of the observations. The aim of this step was to identify neighbouring sites that may reflect combined surveys around the same natural area; for instance, wetlands or ponds. To do this, we first calculated a distance matrix between all points in m. We then applied hierarchical clustering of the distances and cut the distances at 1000 m (diameter) reflecting the limited dispersal potential of most German amphibian species and upper limit of the likely distance of any single survey (Blab 1986; Günther 1996; Trochet et al. 2014). This procedure resulted in neighboring points being grouped together into unique sampling sites (Fig. S1). In other studies, a common approach to assign unique sites to separate occurrence data points is to use a regular grid over the study area. Grids cells, however, could cut through ponds, treating a single population as two or even more. Our clustering aims to produce a more realistic identification of separate sites as populations. After assigning observations to sites, we restricted our analysis to sites surveyed in at least two years over a time span of 10 years (n=1400), since the study of temporal change has been previously shown to be less biased by focusing on sites visited in more than one year (Isaac et al. 2014).

Species data

The remaining challenges with the species data were that: (1) typically only detections of species were recorded, i.e., absences were rarely recorded and only in recent years; (2) the target species group on any given survey was unknown; hence, the lack of reporting of a species may either reflect a true absence (a species was not present during a survey), a lack of detection (false absence) or reporting (a species was detected but the observation was not recorded); (3) sampling effort (e.g., survey duration) was not recorded; and (4), as noted above, sampling method (e.g., trap or visual) was not always known (known for 77% of the observations) but was rarely reported for the earlier years (23% in 1997 versus 84% in 2019) (Fig. 2B).

To attempt to account for these issues, we employed the following approaches and compared possible decisions about how to account for the heterogeneity of the data:

First, following others using heterogeneous databases (e.g., Outhwaite et al. 2020), we attempted to reverse-identify observations that were probably collected during the same survey visit. To do

this, we organized species occurrence data into groups of observations collected on the same date, in the same site (defined by the clusters above) as part of the same project (using metadata on 'project origin'). This combination – date/site/project – was used to define a survey visit.

Second, we inferred absences using the available presence-only data of species observations. We needed to infer absences for our models, since changes in the total number of presences may either reflect changes in sampling or changes in true species occurrence. Inferring absences is also standard in species distribution models that define pseudo-absences as a reference against which to compare presences. A commonly used method in species distribution models is the target-group background method – using observations of non-focal species assumed to be surveyed with similar methods or by similar projects and people to identify absences with the same pattern of spatial bias as the presence data (Phillips et al. 2009; Botella et al. 2020). A similar approach has also been used for temporal models. For instance, Hertzog et al. (2021) used the total number of bird species detected as an indicator of sampling effort towards any bird species in a German dataset of presence-only data. We used a similar target-group background method to infer absences based on reported presences of other species that were assumed to be within the same target group (Outhwaite et al. 2020). In other studies, the target group was defined by which species tend to be reported together by the same specialist recording societies (Outhwaite et al. 2020; Bowler et al. 2021). For amphibians, the extent of the target group is less clear than for other taxa (e.g., birds) because of their biological diversity and the diversity of sampling methods tailored for different amphibian species and life stages. We compared two approaches to define the target group: (1) all taxa: we inferred an absence of any species on a visit if any other amphibian species was recorded; (2) broad taxon group: we inferred absences for anuran species when there was a record of at least one other frog or toad, and absences for newt species, when there was at least one other newt recorded.

Third, we derived a proxy of sampling effort, since absences may still reflect lack of reporting or limited survey effort rather than true absence of a species at a site. Following others (Isaac et al. 2014), we calculated a simple proxy of sampling effort – list length – that was possible with the available data. List length represents the total number of species recorded on a visit; the assumption is that a list with one species usually indicates a lower sampling effort than a list with

two or more species. We included this covariate in all models (see below) and note the limitations of this assumption in our discussion.

Fourth, in a separate analysis, we tested the value of using the metadata information on survey methods when deriving absences. This step aimed to account for the fact that species differ in how well they are sampled by the range of survey methods used for amphibians. We first calculated the main detection method of any survey visit, based on the mode detection method of all species observations on a given survey visit list. We then identified the most important detection methods for each species, based on those used in at least 5 % of each species' observations. Finally, we only inferred absences for a species when the main detection method of a survey visit was one of the important detection methods for that species. We additionally included 'survey method' as a covariate in the models (see below). For these analyses, we excluded observations without a reported survey method, which led to a considerable loss of data.

Finally, in another separate analysis, we subset the dataset to only observations collected by drift fences based on information within the 'project origin' column. These analyses had to be restricted to data for the years since 2010, since drift fence surveys were not common (at least according to the available metadata) before then (Fig. 2A).

Collectively, all these decisions led to four main data subsets that are hereafter referred to as: (1) absences (based on all taxa); (2) absences (based on broad taxon group); (3) survey method; (4) drift fence data. In each case, we ran the statistical analyses discussed below.

Statistical analysis

We used two alternative approaches to analyse the occurrence data derived from the above processing steps. Each species was analysed in a separate model. Together, this meant 4 data subsets x two statistical models = 8 analyses per species.

First, we used generalized linear mixed models (GLMM), similar to a 'reporting rate model' that has been used by others (Isaac et al. 2014), in which 'presence-absence' of species (or rather, detection-nondetection) on a visit was the response variable. The main explanatory term was

306 'year' (as a factor) to estimate the mean annual occurrence of each species in each year across all
307 visits. To account for variable sampling effort, as discussed above, we included 'list length'
308 (number of species recorded on a visit, split into a factor of 1, 2/3 or 4 or more species). To
309 account for differences in the timings of survey visits within and across years, we included
310 'month' of the survey visit as an additional variable. We included site and 'ecoregion' (derived
311 from a German spatial dataset representing distinct biogeographical subregions, n= 32 [BfN
312 2008, 2024a]) as random intercepts to account for spatial variation in the intensity of sampling
313 across the study region. For the survey method data subset, we additionally included 'survey
314 method' as a covariate, but not for the other data subsets due to the large number of records
315 collected by unknown methods.
316
317 Second, we used occupancy-detection models, which have been successfully used in the analysis
318 of similar heterogeneous data (Isaac et al. 2014; Outhwaite et al. 2020; Bowler et al. 2021). These
319 models consist of two connected submodels: one submodel aiming to describe detection
320 probability (i.e., whether a species is seen on a survey visit given it was present at the site) and
321 another submodel aiming to describe species occupancy (i.e. whether a species is present in a site
322 in a given year). We used the same covariates as for the GLMMs except they were distributed
323 between these two submodels according to which process they were assumed to affect. For the
324 detection submodel, we included list length, month, and detection method (the latter only for the
325 survey method data subset). The occupancy submodel included year (as factor) and ecoregion (as
326 a random intercept). These models were fit using the spOccupancy package (Doser et al. 2022).
327 We additionally ran models allowing for spatial and temporal autocorrelation in occupancy, but
328 these models showed poorer convergence and weak evidence that these additional terms
329 mattered. The Bayesian p-values were used as goodness of fit tests and all p-values were between
330 0.1 and 0.9, and hence there was no evidence of major fit issues.
331
332 Note, while we used the same covariates in the GLMM and occupancy-detection models, they
333 differ in fundamental ways. The occupancy-detection model can be used to predict the annual
334 occupancy probability *of a site*, accounting for imperfect detection (i.e., that sometimes species
335 are not detected at a site, even if they are present). By contrast, the GLMM predicts species
336 occurrence probability *on any given survey visit* (i.e., site occupancy on a given date, under a

given sampling method/list length). This means that we expect the two models to strongly differ in their absolute values of their predictions – the occupancy-detection site occupancy probability will typically be higher than the survey visit occurrence probability, since the former adjusts for imperfect detection on a visit. Typically, the probabilities predicted by the occupancy-detection model are closer to conservation questions of interest about the probability that a species is present at a site. However, in either case, both probabilities can be used to reveal changes in species occupancy/occurrence across years.

Based on the annual occupancies from both models, we calculated long-term (1997-2019) and short-term trends (2010-2019) using simple linear regression models with year (as a continuous variable) as the only predictor and the occupancy/occurrence probabilities as the response, including the uncertainty of the probabilities (standard deviation of the estimate) as a measurement error term, using the brms library of R (Bürkner 2017). The long-term trend covers a period similar to that of the last Red List short-term assessment in Saxony (Zöphel et al. 2015) (see next section), whereas the short-term trend allowed a comparison with the results based only on drift fence data.

Plausibility assessments of our results

To check the plausibility of our model predictions, we compared the consistencies and discrepancies of the trend results among the different models and subsets of data used. We further compared qualitative results with the most recent Red List assessment of amphibians in Saxony (Zöphel et al. 2015). The most recent Red List assessment in Saxony was done by a group of experts in 2013 (Zöphel et al. 2015; LfULG, pers. comm.). The short-term trend assessments covered the period back to the year 2000, in which the data for the Amphibian Distribution Atlas of Saxony were compiled using the ZenA database (LfULG undated; Zöphel and Steffens 2002), and for some species back to 1990 (Zöphel, pers. comm.). Thus, these periods are approximately similar to our long-term trends. The distribution atlas is based on topographical maps (TK 1:25.000) dissected into grid cells of 10 x 10 km or 5 x 5 km and the occurrence (presence) data of the amphibian species within these grid cells. For their assessments of trends, the experts used the Amphibian Distribution Atlas and more recent data of the Central Species Database (ZenA) compiled by the LfULG (undated) and their personal experience (Zöphel and Steffens 2002,

Zöphel et al. 2015). However, trend assessment was not an actual calculation of trends, but rather a mere estimation of the degree of differences in the number of occupied grid cells between the two time periods before and after 1990/2000, complemented with the experts' experience (Zöphel, pers. comm.).

Results

Annual occupancy estimates

The different data processing decisions – whether using all taxa or taxa on order level (newts or anurans) to infer absences, or including survey method as a covariate – mostly led to similar annual occupancy/occurrence estimates (Fig. 3). However, annual estimates typically were less certain when using the survey method as a detection covariate. The difference between using all taxa or broad taxa to infer absence was unimportant for anurans. Surveys only recording newts were rarer, and the differences were more pronounced, which meant that occupancy estimates were lower when all taxa were used to infer absences than when only newt surveys were used. For *Triturus cristatus*, this also led to different predictions of species dynamics (Fig. 4).

Compared to the GLMM model, the occupancy-detection led to less certain and more similar predictions across species in predicted occupancy proportions. But it is important to note that these two models predict different responses: the occupancy-detection model predicted annual site occupancy, while the GLMM model predicted occurrence on any given survey visit. However, in our results, the GLMM models were more able to separate common (e.g., *Bufo bufo*) and rare species (e.g. *Epidalea calamita*) than the occupancy-detection models, probably because of the extra uncertainty added by estimating detection probabilities within the occupancy-detection models.

For both methods, strong peaks and troughs in the annual occupancy estimates occurred for some species. Some peaks align with periods of lower sampling effort (e.g., compare 2006/2007 in Fig. 2A and Fig. 3).

Long-term trends (1997-2019)

The predictions of long-term trends in occupancy, accounting for uncertainty in a Bayesian framework (Bürkner 2017), are more consistent among assessment methods than annual estimates (compare Fig. 4 with Fig. 3). Uncertainty tends to be greater when using the survey method as a detection covariate. Predictions are generally similar between using all taxa or broad taxa as the target group to infer absences, except for *T. cristatus* where they reveal opposite trends. When including survey method as a covariate, more positive trend estimates were found with the occupancy-detection model. In some cases, this led to contrasting trend predictions, e.g., a positive trend estimate (but not significant) for *P. ridibundu*s, contrasting with negative trend estimates for all other methods. *Rana temporia* and *R. dalmatina* trend predictions also have opposite directions between using the survey method and other approaches.

The most recent expert assessment of trends of amphibians in Saxony applies to the period 1990/2000–2013 (LfULG, pers. comm.) and is called short-term trends but covers a similar period as our long-term trends (1997–2019). This assessment did not identify any species that was increasing, *P*. kl. *esculentus* remained stable and the other 12 species (of the ones also assessed by us) were judged as declining (Zöphel et al. 2015). Of the latter, the degree of the decline was stated as unknown for eight species, indicating considerable uncertainty in the assessment results (Tab. 1).

Table 1. Comparison of the model-based long-term trend in Saxony and the expert-based judgment of decline in Saxony for the period 1990/2000–2013 (Zöphel et al. 2015). If predictions among models differed, two entries are provided. Grey cells represent differences to the expert judgements. Uncertain: exceptionally high 95%-confidence intervals.

Species	Model-based trend	Expert judgment
Bombina bombina	None or increasing	Strong decline
Bufo bufo	Increasing	Decline but strength unknown
Bufotes viridis	Decreasing	Strong decline
Epidalea calamita	None or uncertain	Strong decline
Hyla arborea	Decreasing	Strong decline
Lissotriton vulgaris	None or increasing	Strong decline
Pelobates fuscus	None or increasing	Decline but strength unknown
Pelophylax kl. esculentus	None or increasing	No trend
Pelophylax lessonae	None	Decline but strength unknown
Pelophylax ridibundus	None or decreasing	Decline but strength unknown
Rana arvalis	None	Decline but strength unknown
Rana dalmatina	Decreasing or increasing	Decline but strength unknown
Rana temporaria	Decreasing or increasing	Decline but strength unknown
Triturus cristatus	Decreasing or increasing	Decline but strength unknown

Short-term trends (2010-2019)

We compared predictions from models built on fence data, the most standardised continuous source of data (Seyring et al. 2024a), with models using all other types of data. This comparison was only possible from 2010 onwards, since fence data were rare before that year. For the comparison, we also removed species that were rarely caught at the fences (*E. calamita, P. lessonae, P. ridibundus,* less than 20 records from fence surveys).

The separate models for the fence data somewhat increased the uncertainty in the annual occupancy estimates of the GLMM models compared to models that combined all the other data (Fig. 5). For the occupancy-detection models, uncertainty was even higher when using only the drift fence data, preventing reliable assessment of trends for the majority of species.

The occupancy-detection models generally led to more uncertain trend predictions, but the direction of trends was mostly the same between the GLMM and occupancy detection models both for the fence and non-fence data. Declines of *B. viridis* were clearer with the non-fence data than with the fence data. Similarly, increases of *B. bufo* were clearer with non-fence data than with fence data. By contrast, increases of *P. fuscus* were clearer with the fence data than with the non-fence data. Predicted trends of *T. cristatus* contrasted among the methods/models. Most other species were predicted to be stable over this time period, or at least the uncertainty masked any potential trends.

Discussion

Given the serious decline of amphibians (Henle et al. 2008; Falaschi et al. 2019), there is a need to assess whether we can use available databases on species occurrences reported by naturalists (e.g. Loman and Andersson 2007; Bonardi et al. 2011), despite their heterogeneity (Seyring et al. 2024a, 2024b), and apply typical statistical methods for assessing trends of species. We tested the use of occupancy-detection models already well-used for these kinds of data, but typically for other organisms, such as dragonflies (Bowler et al. 2021), birds (Kamp et al. 2021; Rigal et al. 2023) or butterflies (Arfan et al. 2018; Warren et al. 2021). For comparison, we also used standard GLMMs. We found a mix of patterns, including consistencies and contradictions among model results, subsets of data, and many disagreements with expert opinion. Overall, our findings highlight the complexity of using these data for amphibians and indicate the importance of improving data collection and documentation in the future.

Assessing trends was partly limited by the uncertainty in the model predictions. Results for annual occupancies showed considerable uncertainty for the majority of species and subsets of data, both for GLMM and occupancy detection models. An exception was for the GLMM models for the short-term trends for the non-fence data, and partially also for the fence data. Presumably, the large uncertainty is due to heterogeneity of the sampling methods that could not be completely accounted for in the models and the considerably reduced dataset when survey methods were accounted for because many data sets lacked this information and thus had to be excluded.

Another challenge affecting trend assessment was the strong interannual fluctuations. We found strong peaks and troughs in the occupancy predictions (Fig. 3), which smoothed out to some extent depending on data set and modeling method, but at the expense of a broader 95% confidence interval, as for example in the occupancy detection models applied to the short-term data set (Fig. 5). These 'outliers' mean that the estimate of the long-term trend critically depends on the time-frame over which the trend is assessed. Additionally, the magnitude of some of the year-to-year changes in occupancy are biologically not plausible. Possible reasons may have been sharp shifts in sampling effort and concomitant changes in use of the main sampling methods over the years, as visible in Figure 2, but possibly also changes in sampling locations. While the robustness of occupancy-detection models against heterogenous data or spatial and temporal biases, such as increasing survey visits through time has been well explored (Isaac et al. 2014; Isaac and Pocock 2015; MacKenzie et al. 2018), this probably does not hold for non-random changes in sampling locations, but this is difficult to account for without detailed knowledge of the causes of the changes in sampling locations. For example, shifts towards urban areas occurred in Saxony (Bowler et al. 2022), which may cause biases in trend estimates.

Methods for monitoring amphibians have changed over time and are affected by incentives and effort (Heyer et al. 1994; Weir and Mossman 2005; Schlüpmann and Kupfer 2009; Seyring et al. 2024b). For example, the wide usage of water traps to capture newts emerged only after around 2005 (Schlüpmann and Kupfer 2009). Since newts are difficult to find with standard sampling, the trap-targeted sampling may have resulted in our long-term upward trend estimates for *L. vulgaris* and *T. cristatus*, unless controlling for taxa or survey method (Fig. 4). This heterogeneity of methods could not be fully accounted for since sampling methods are unknown for many records, especially in the early periods. This also explains why the estimated trends were more precise for the short-term compared to the long-term trend assessments.

Another source of sampling bias are monitoring peaks in years of intense targeted surveys as e.g. for the regular (every six years) reporting for the European Fauna-Flora-Habitat (FFH) Directive (Council Directive 92/43/EEC). The peaks between 2004-2006 and 2010-2012 in Figure 2A can thus be attributed to the reports of 2007 and 2013 (Hettwer et al. 2015; LfULG, pers. comm.). The high peaks (Fig. 2A) in 2012 and 2015 can also be attributed to the start of the Saxony pilot

monitoring project for *R. temporaria* (Seyring et al. 2024b) and the compiling of the Saxony Red List (Zöphel et al. 2015; LfULG, pers. comm.). The ability to account for all these and other biases in sampling effort are limited due to limited and inconsistent metadata to include in the models.

Consistency and discrepancy among approaches in species trend assessments

More positive and fewer negative trend estimates were found with the occupancy-detection model including survey method as a covariate than in the GLMM models in the long-term assessments. The same tendency was apparent for the different approaches to infer absences and in the short-term trend assessments. For three species (*Triturus cristatus, Rana dalmatina, R. temporaria*), this led to contrasting trend predictions. They were positive in the occupancy-detection models that included survey method as a covariate in all three species and no trend or negative for the other models with exception of the GLMM model with survey method included for *R. temporaria*. All three species were regarded as strongly declining in the expert assessment (Zöphel et al. 2015).

The trend direction was consistent among models for five out of 14 assessed species: increasing for *Bufo bufo,* no trend for *Pelophylax lessonae* and *Rana arvalis* and declining for *Bufotes viridis* and *Hyla arborea*. For the latter two species the experts regarded the decline as strong and for the other three species as declining of unknown strength. For *B. viridis* the German wide Red List (Rote-Liste-Gremium Amphibien und Reptilien 2020) also indicates a strong decline. Likely reasons for the decline of this particular pioneering species is the loss of small, shallow, ephemeral ponds of temporarily flooded plains due to a change in land use towards river regulation, intensive agriculture and more desiccation (Sedlmeier 2008; Vences et al. 2019; Zahn et al. 2020; Neßmann and Jacob 2024). The species nowadays occurs mainly in secondary habitats, such as in mining areas (Sy and Meyer 2020; Barth et al. 2024). The loss of large active mining areas and other tremendous change in land use after 1990 (Haase et al. 2007, Schmidt et al. 2015) particularly affected this species.

The same applies to the natterjack toad (*E. calamita*), which thus also should have shown a strong decline in our assessments. However, our model-based assessments indicated an absence

of a trend. This likely is due to the fact that the loss of breeding habitats (ephemeral water bodies) usually results in an end of surveys of the lost sites and an absence of relevant data in the Species Record Database of Saxony, ZenA. Experts may adjust their judgment based on such experience. However, due to the lack of reporting absences, such information is not quantified, which means it is not possible to reliably account for the contribution of such losses to the decline of a species at larger scale, regardless of method.

For one of the remaining species, *Pelophylax esculentus,* our models and expert opinion indicated no trend, except for our occupancy detection model when survey method was included as a covariate. In that case, there was a particularly strong apparent trough with a rapid drop in occupancy from almost 1 to almost 0. Presumably, this pattern was caused by shifts in survey methods that could not be accounted for despite using survey method as covariate.

For three other species (*Lissotriton vulgaris, Bombina bombina, Pelobates fuscus*), our models also indicated either no trend or increasing trend, while expert opinion was declines. The positive trend for the newt *L. vulgaris* in our models may contain upward bias as surveys of amphibians shifted towards urban environments in several database in recent years, which tend to lead to too optimistic estimates in species common in human settlements (Bowler et al. 2021) like *L. vulgaris* (Günther 1996). In addition, the increased use of aquatic traps since around 2005 (Schlüpmann and Kupfer 2009), the most efficient method to detect newts, may have contributed to an upward bias.

Differences between abundance and occupancy

A major limitation of our trend estimates, as well as the data informing the expert based Red Lists, is that the metric of change is based on the occupancy of sites or grid cells, respectively, which change slower than abundance and is thus less sensitive in revealing changes, especially declines. Some species might be declining in local density, which is, however, not detectable yet by changes in occupancy. For example, in insects strong declines can be shown in terms of biomass (Hallmann et al. 2017) or abundance (Van Klink et al. 2024), while occupancy may show contradicting trends (Dennis et al. 2019). For example, overall moth abundance in Scotland decreased by 46% in 24 years, while occupancy increased in the same time (Dennis et al. 2019).

Particularly, species with negative abundance trends varied in occupancy responses, which might be driven by summer temperatures (Dennis et al. 2019; Bowler et al. 2021). A similar effect may apply to amphibians and further research into the drivers is required to objectively assess the plausibility of our models.

Declines in abundance have been found also for ubiquitous amphibian species, such as *B. bufo* in various parts of Europe (Petrovan and Schmidt 2016, Chiacchio et al. 2022). In Saxony, this is likely the case in an ongoing pilot monitoring project for *R. temporaria* (Seyring et al. 2024b). Declines of these generalists are potentially incompletely represented in our trend analyses, since most sites (grid cells) might be still occupied, while abundance decreases. In the expert opinions, these issues are to some extent compensated by the local experience of species abundance and habitat, which was taken into account in the trend assessments in the Red Lists. However, we did not have abundance data to include in our models.

Furthermore, in the Saxonian and German wide Red List assessments, occupancy was not modeled (Zöphel et al. 2015; Rote-Liste-Gremium Amphibien und Reptilien 2020). Rather, the relative proportion of occupied grid cells of two time intervals (new vs. old) were compared, which is sensitive to shifts in sampling location, frequency of sampling a grid and survey methods. For this reason, the contradictions in our modeled trends may reflect changes in sampling intensity, sampling location or local range expansions of a species, which was not represented in the two-time interval comparisons of the expert opinions.

How can we do better in terms of both data and models?

A model can only be as good as the data. Clearly, our results show that the lack of standardized monitoring data represents a major challenge for estimating species trends and supports the development of more standardized data collection and more precise documentation of methodology, as recommended in Seyring et al. (2024a, 2024b) and Barth et al. (2024). The potential impact of changes in spatial locations between years highlights the particular need for improved coordination of sampling effort across sites over time for more representative sampling. While fence data offer some promise in the future for standardized data collection (Seyring et al. 2024a), for most species, our analysis of only fence data did not lead to greatly different

592 conclusions than analysis of all the other data collected by heterogeneous methods, partly
593 because of the short-time frame. Moreover, only some species were frequently caught by fences.
594
595 Greater standardization and method transparency would also benefit Red List assessments
596 (Cazalis et al. 2024). In addition, more rigorous approaches for trend assessments than a mere
597 comparison between different reports (DG Environment 2017; BfN 2024b) are required to reduce
598 potential biases and increase the reliability of trend assessments.
599
600 Platforms such as eBird have proved successful for bird recording, allowing individual surveyors,
601 often citizen scientists and other volunteers, to collect data in different ways, but importantly,
602 encouraging key features of the survey to be reported. Metadata within the ZenA database has
603 already improved since recent records often have known survey methods (e.g., type of trap used),
604 but this is still a problem when using older data for assessment of change. While we argue for
605 more standardized data collection, a single survey method will not be suitable to survey the
606 whole community. Hence, multiple survey methods are needed – see Seyring et al. (2024b) for
607 recommendations for German species. In this case, on-going improvements of metadata is
608 important so that the survey method is fully documented (ideally in a short-list of possible types)
609 so that the heterogeneity can be modeled.
610
611 Metadata documentation could radically help use these data for trend assessment (Menger et al.
612 2024). We propose two main metadata attributes of surveys that could be collected, along with a
613 specific survey ID assigned to each observation. First, survey effort: we are still missing metadata
614 on sampling effort, e.g., number of hours looking, number of trap days, length of drift fence
615 (Seyring et al. 2024a 2004b). We used list length (number of species reported) as a measure of
616 effort, which has been used for other taxa, but may not work as well for amphibians when the
617 number of detected species is strongly affected by survey method and location. Second, survey
618 completeness: we are missing information on the target community of each survey method and
619 whether the list of species was complete with respect to that target community. This is
620 particularly important because the potential community composition will differ among breeding
621 sites for amphibians. Our model used the target-background method to infer absences, i.e.,
622 absences of species were inferred by presence records of other species. But this may not work

with amphibians because they are monitored in very different ways, so presences of one species may not be reliably used as evidence of absences of others. Hence, we recommend developing a culture of recording the completeness of a survey and its target species group. Alternatively, absences can be directly recorded, which has been done in recent years, but it may be more efficient – in terms of database size – to record target community and completeness of survey as additional columns to the current observations, rather than adding new rows for absences.

Conclusions

Large-scale, long-term presence databases collated by a range of different contributors have become increasingly available and used for assessing trends in species. Our analyses of such a database for amphibians from Saxony, Germany, showed high sensitivity of trend predictions to analytical choice: type of statistical model, methods to infer absences, subsetting of the data and covariables. Substantial changes in survey intensity and methods used and a lack of sufficient metadata for many survey data create major challenges for reliable trend predictions. In this regard, amphibians may be more challenging than other taxonomic groups, such as birds or butterflies, because they are a diverse group monitored with many different methods by diverse kinds of people. Given the serious decline of amphibians (Henle et al. 2008, Falaschi et al. 2019), to overcome these challenges, we argue a need to push for more comprehensive and better metadata reporting and a need to encourage a switch from presence-only to presence-absence data, including whether a survey was not continued because of the loss of the surveyed habitat. To the extent possible, we should even push for coordination and harmonisation of methods across regions. Databases of metadata of monitoring schemes and monitoring organisations, such as those created by the EuMon project and revised by the ADVANCE project (Mihoub et al. 2017, Menger et al. 2024), and national monitoring centres could help to achieve better coordination (Lengyel et al. 2018, Geschke et al. 2019). Likewise, calibration among methods and their publication (e.g. Bayliss 1987, Brown et al. 1996, Gottschalk 1997) could help to expand standardisation to abundance data. Despite these still existing deficits, amphibians are a small group with relatively easily distinguishable species in most parts of Europe. In Europe, and beyond, most species accumulate regularly at their breeding ponds, which makes them a well-suited group for large-scale monitoring.

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List of Figures

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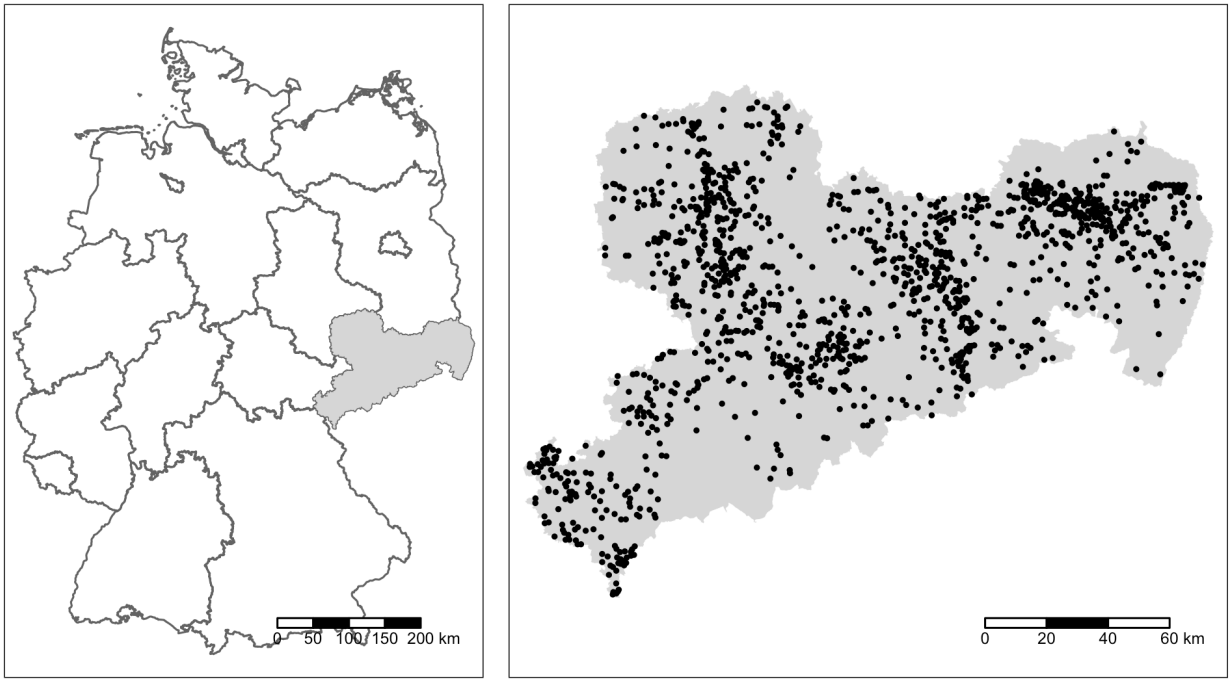
Figure 4. Predicted long-term trend (i.e., mean annual growth rates in occurrence/occupancy) for each species between 1997 and 2019. Each point shows the mean and 95% CI. The vertical dashed line represents the line of no change: points to the left indicate declines, while those to the right indicate increases. Species are ordered by the mean of their trend estimates across models/data subsets.

Figure 5. Predicted occupancy/occurrence proportions for each species in each year between 2010 and 2019 based on the GLMM model and the occupancy-detection model. The occupancy/occurrence proportion is the predicted proportion of occupied sites. Ribbons show the 95% confidence intervals. Different colours refer to different data processing decisions (see Methods section for details).

Figure 6. Estimates of short-term trends in occurrence/occupancy between 2010-2019 when based on drift fence surveys versus other sampling methods. Each point shows the mean and 95% CI of the mean annual change estimate. The vertical dashed line represents the line of no change. Species with less than 20 records from drift fences were excluded. Species are ordered by the mean of their trend estimates across models/data subsets.

1039 Fig. 1



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1042 Fig. 2
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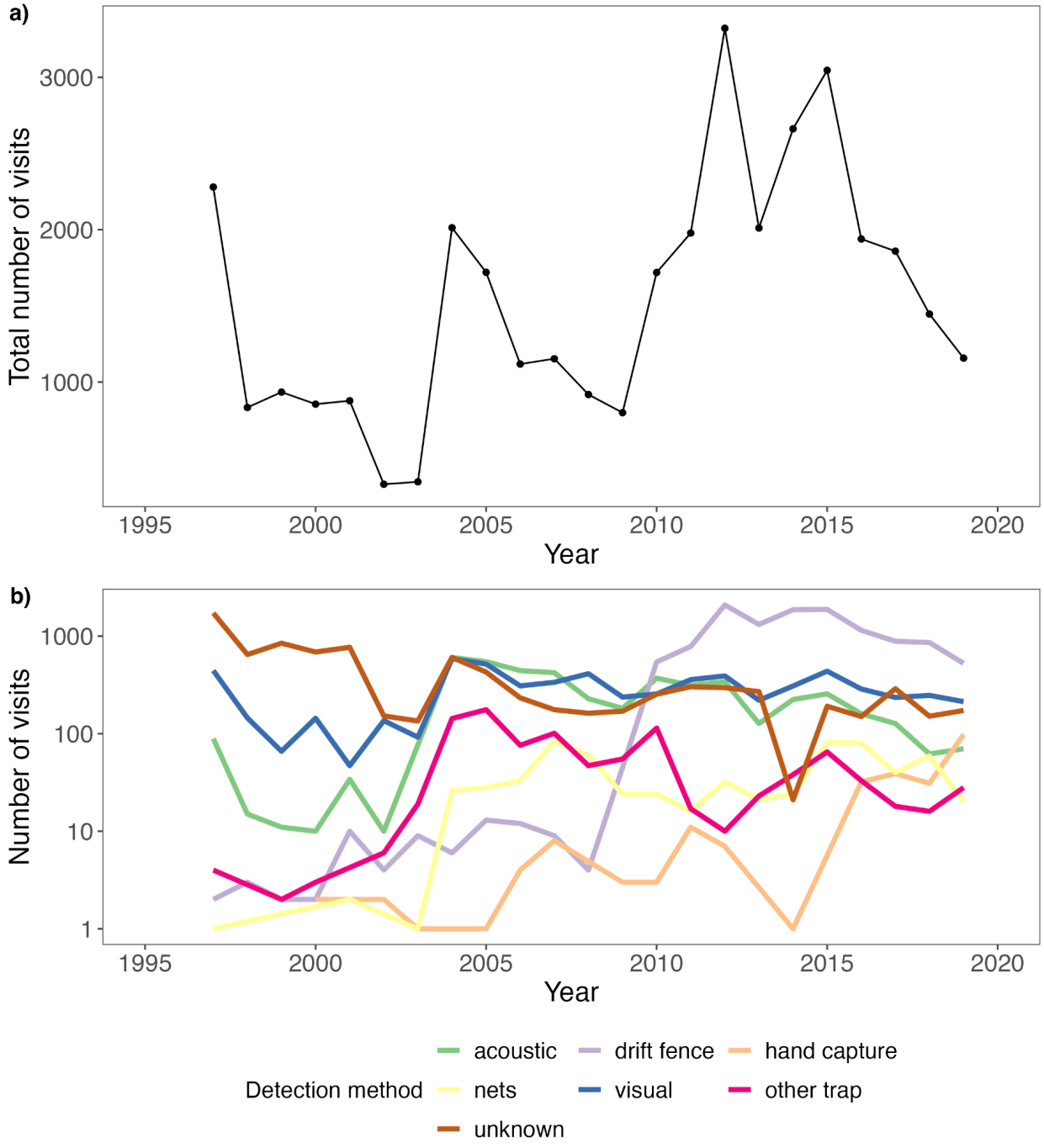

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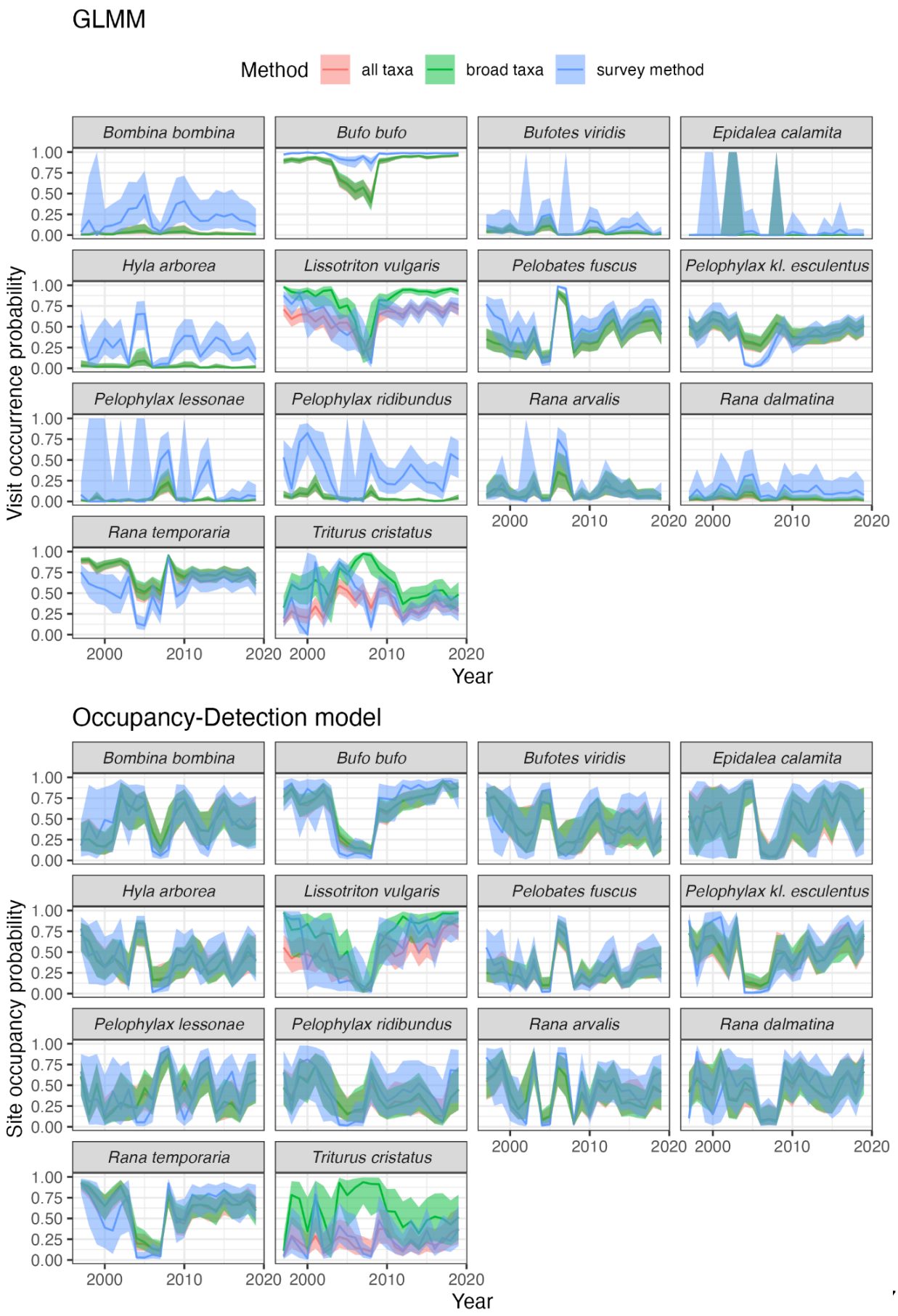
GLMM
Method
all taxa
broad taxa
survey method
Bombina bombina
Bufo bufo
Bufotes viridis
Epidalea calamita
Hyla arborea
Lissotriton vulgaris
Pelobates fuscus
Pelophylax kl. esculentus
Pelophylax lessonae
Pelophylax ridibundus
Rana arvalis
Rana dalmatina
Rana temporaria
Triturus cristatus
Visit occcurrence probability
Site occupancy probability
1.00
0.75
0.50
0.25
0.00
2000
2010
2020
Year
Occupancy-Detection model

1048 Fig. 4
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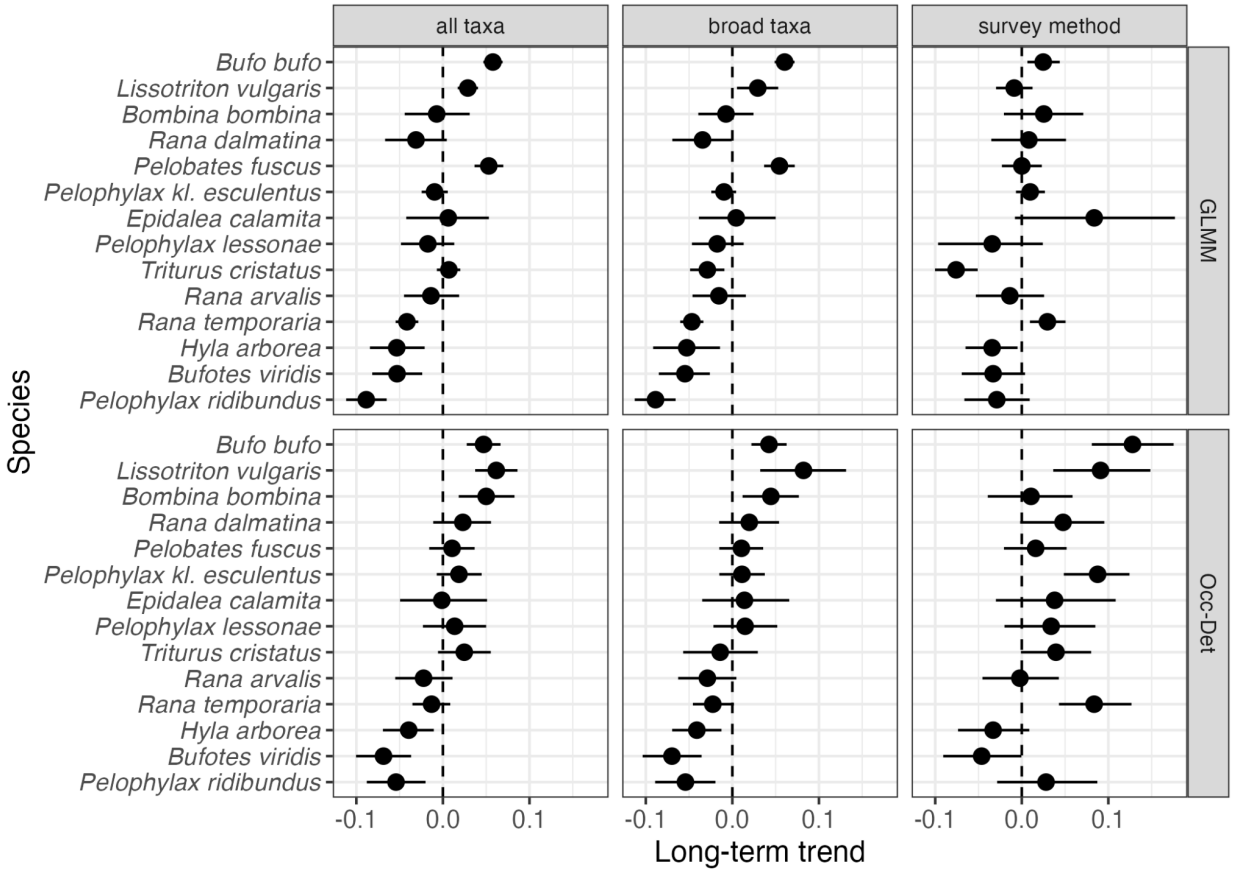

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1053 Fig. 5

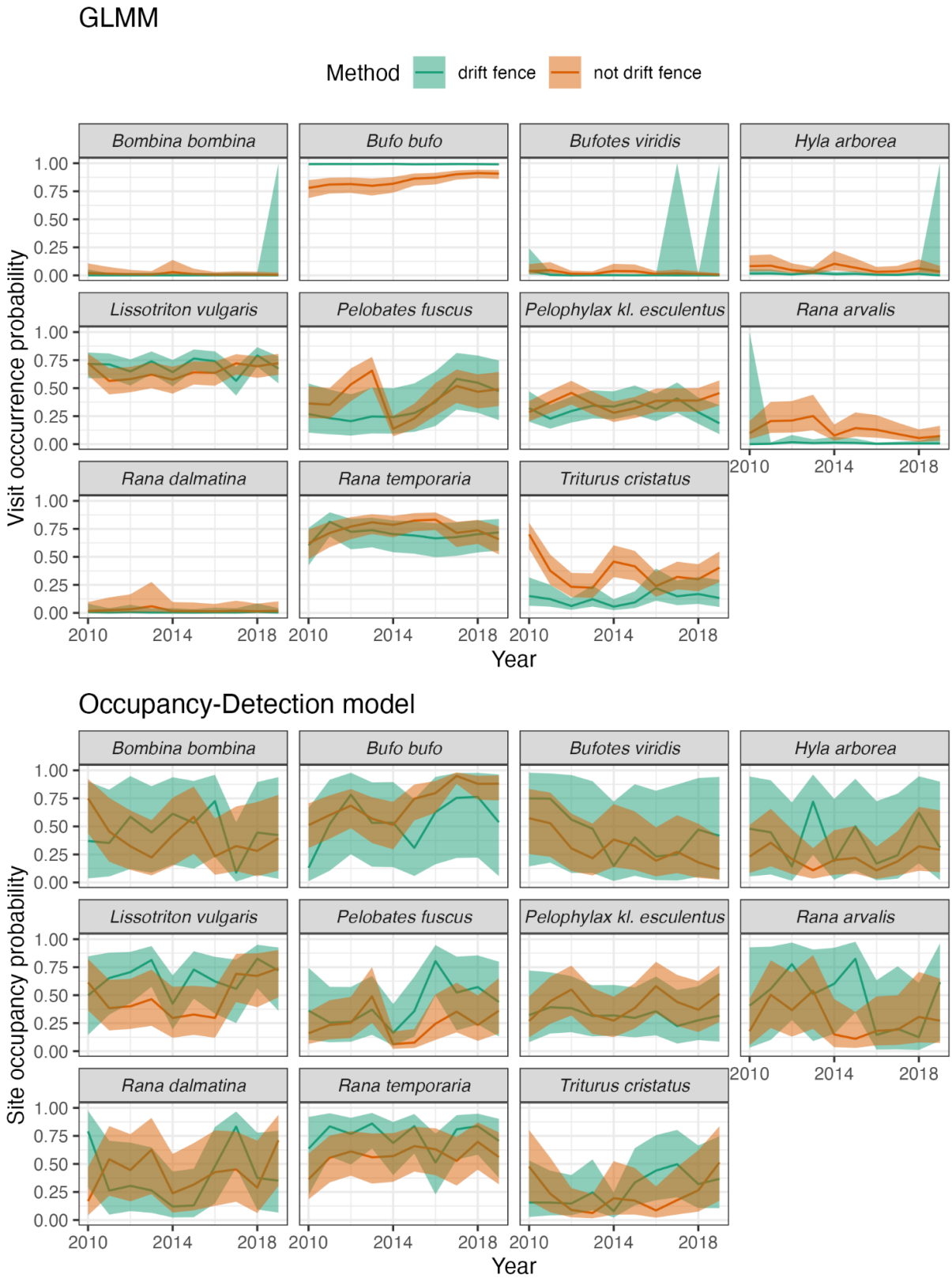

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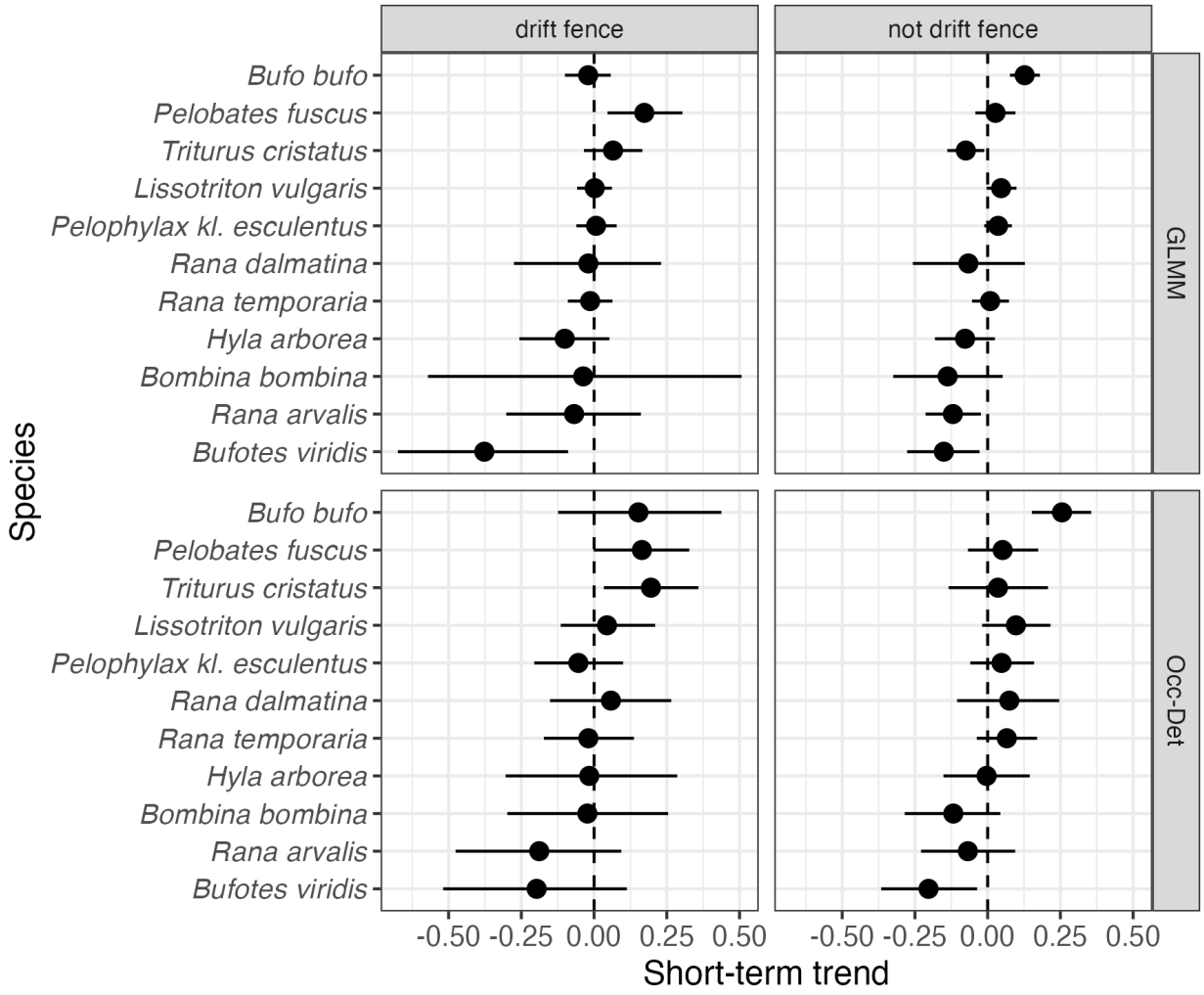
Fig. 6


drift fence
not drift fence
GLMM
Occ-Det
Species
Bufo bufo
Pelobates fuscus
Triturus cristatus
Lissotriton vulgaris
Pelophylax kl. esculentus
Rana dalmatina
Rana temporaria
Hyla arborea
Bombina bombina
Rana arvalis
Bufotes viridis
-0.50 -0.25 0.00 0.25 0.50
Short-term trend