



## Integrating historical and recent data to measure long-term trends of endangered subterranean species

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### ABSTRACT

Documenting population trends is pivotal to identify the underlying drivers of biodiversity changes and setting conservation priorities. Ascertaining trends often requires long-term, standardized, monitoring data that are not always available. Historical data provide important information on past species distribution, but their integration with recent data to obtain trend estimates is challenging. Here we show how site occupancy-detection models (SODMs) can allow combining data from recent monitoring with historical ones from the gray literature. Using data on the endangered cave salamander, *Speleomantes strinatii*, we tested whether SODMs can provide reliable trend estimates if *i*) historical data include repeated within-season surveys enabling the estimation of past detectability, or if *ii*) information on detection/non-detection is not available. We conducted repeated surveys across 40 caves covering the species range, for which historical (1940–1982) biospeleological data were available. We then developed Bayesian SODMs *i*) estimating species detectability from both recent and past surveys, and then assessing trends; *ii*) in absence of estimates of past detectability, assessing trends by comparing scenarios on the potential misdetection rate during historical surveys. Salamanders were widespread in the study sites. SODM estimated high detectability for both recent and historical surveys and suggested a growing occupancy. Changes in occupancy were unrelated to landscape modifications. Even without historical detection/non-detection data, realistic scenarios of past misdetection consistently suggested an increasing or stable trend. The application of tailored analytical approaches is fundamental to exploit the vault of information available in historical data, and can be linked to adaptive management to promote efficient conservation actions.

### 1. Introduction

Documenting declines is pivotal to identify species that are threatened by extinction, is a prerequisite to pinpoint the underlying drivers of biodiversity changes and to propose conservation strategies. Standardized monitoring protocols are the ideal framework for measuring distribution changes and triggering conservation interventions when a certain level of decline is detected (Lindenmayer et al., 2013; McDonald, 2003; Wintle et al., 2010; Yoccoz et al., 2001). However, the majority of

monitoring schemes have been launched rather recently, and mostly cover the last 10–40 years (Mandeville et al., 2023; Vihervaaram et al., 2021). Furthermore, these data often suffer geographical and taxonomic biases, with long-term programs only covering a few taxonomic groups and geographical areas (Bowler et al., 2022; Chapman et al., 2024; Freeman et al., 2018; Mandeville et al., 2023; Venne and Currie, 2021). These biases can influence decision-making, potentially undermining the effectiveness of data-driven solutions to biodiversity loss (Chapman et al., 2024). As such, there is a great interest in obtaining robust

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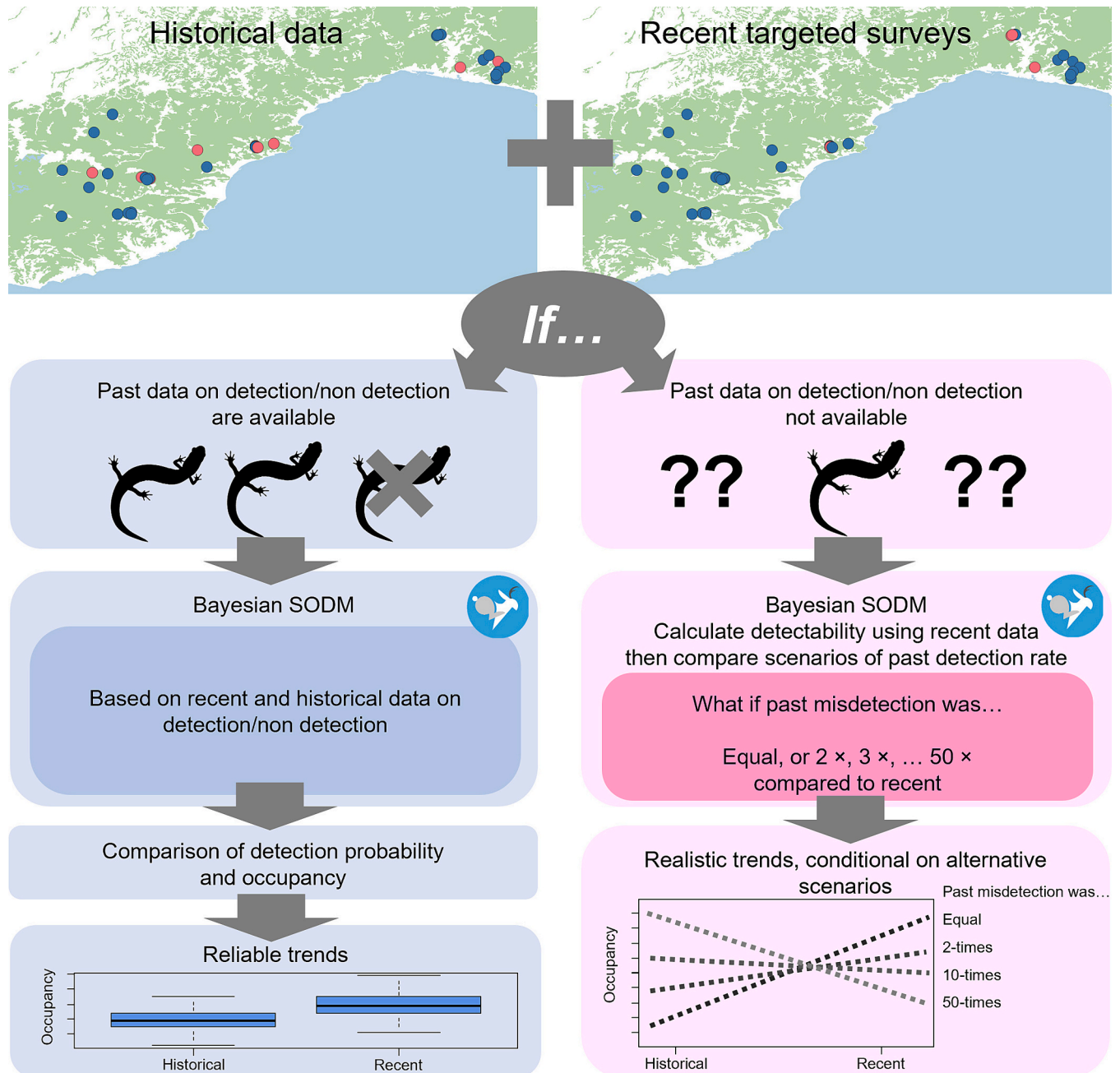
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estimates of how populations have changed in the long term, which is key to identify the actual conservation priorities (Hughes et al., 2023).

Given the scarcity of long-term data obtained through standardized monitoring, researchers are extensively working on approaches to extract information from available sources, while correcting for their inherent biases (Chapman et al., 2024; Hughes et al., 2023; Maes et al., 2015; Monsarrat et al., 2019; Rousset and Ferdy, 2014; Valdez et al., 2023; Warton et al., 2013). The gray literature is potentially a great source of historical information on biodiversity (Haddaway and Bayliss, 2015). The gray literature comprises data published on a broad range of sources such as government papers, theses, organizational reports, and monitoring reports, and can include the outcome of surveys performed in the past, even if they were not part of standardized monitoring

programs (Haddaway and Bayliss, 2015). Nonetheless, integrating the gray literature with recent data to obtain trend estimates is challenging, for instance because differences in protocols and effort can affect the detection probability of target species, thus influencing estimates (Altwegg and Nichols, 2019; Guillera-Arroita, 2017).

Imperfect detection of target species is pervasive in biodiversity monitoring (Guillera-Arroita, 2017; MacKenzie et al., 2017). In the last few decades, approaches have been developed to explicitly integrate imperfect detection into modelling; such approaches have greatly improved the analysis of biodiversity patterns and provide more robust estimates of species trends and of the underlying drivers (Devarajan et al., 2020; Ficetola et al., 2018b; Guillera-Arroita, 2017; MacKenzie et al., 2017; Schmidt, 2005). However, these frameworks generally



**Fig. 1.** Study outline. Data from recent and historical surveys can be combined using different approaches. If past data include information on detection/non-detection, we can use species site occupancy-detection models (SODM) to obtain robust estimates of detection probability, occupancy, and trends. Past data can be useful even in absence of reliable information on detection/non-detection, as SODMs can incorporate alternative scenarios to evaluate the verisimilitude of proposed trends.

require that data are collected using approaches that are informative about the observation process (Guillera-Arroita, 2017). Major efforts are thus ongoing to directly or indirectly extract this information from a variety of sources, such as citizen-science databases (Altwegg and Nichols, 2019; Chapman et al., 2024; Ellwood et al., 2017; Gorta et al., 2019; Robinson et al., 2018). Still, very limited attempts have been made to use a detection probability framework (e.g. site occupancy-detection models; SODMs) to integrate recent monitoring with historical data available from the gray literature.

Underground life is a major – yet understudied – component of Earth's biodiversity that provides key ecosystem services (Canedoli et al., 2022; Culver and Pipan, 2009; Ficetola et al., 2019; Mammola et al., 2020; Mammola et al., 2022). In underground environments, the difficulties of monitoring are exacerbated, and only in the last few years proposals have emerged for standardized assessments of subterranean biodiversity (Culver et al., 2012; Lunghi et al., 2022b; Mammola et al., 2016; Mammola et al., 2021; Popovic et al., 2020; Saccò et al., 2019; van der Heyde et al., 2023; Wynne et al., 2021). Nevertheless, caves have been the target of biodiversity explorations for more than two centuries, and these activities produced a rich gray literature that often remains unexploited by biodiversity analyses (Ficetola et al., 2019; Romero, 2009). Can we exploit this historical information to estimate long-term trends of cave-dwelling species?

In this study, we show that SODMs can allow to integrate recent surveys with historical data to obtain realistic estimates of long-term (>50 years) trends of occupancy. As model species, we considered the Strinati's cave salamander (*Speleomantes strinati*), a ground-dwelling species endemic to a small region between NW Italy and SE France. This salamander is considered “endangered” by the IUCN and with a decreasing population trend (IUCN SSC Amphibian Specialist Group, 2022). However, actual trends are only available for very few populations (Salvidio et al., 2020; Salvidio et al., 2016) and quantitative broad-scale estimates are lacking. As a consequence, there is disagreement on the actual conservation status of this species between organizations (IUCN SSC Amphibian Specialist Group, 2022; Rondinini et al., 2022). Historical contributions report detailed information on the occurrence of this salamander in caves during the period 1940–1983 (Bologna and Vigna Taglianti, 1985; Sanfilippo, 1950). Here we propose a framework for the explicit integration of recent and historical data in SODM (Fig. 1). *i*) We identified multiple caves for which historical (1940–1983) biospeleological data were available (Bologna and Vigna Taglianti, 1985; Sanfilippo, 1950), covering the majority of the species range and reporting both detections and non-detections of salamanders. *ii*) We performed repeated surveys in the same caves, using monitoring approaches that are appropriate to run SODMs. This allowed us to test whether the detection probability of the target species differed between historical and present-day approaches. *iii*) We integrated historical and recent data using Bayesian SODMs, to obtain quantitative estimates of population changes, and to evaluate whether long-term environmental modifications (land-cover change) can explain the observed trends in occupancy. *iv*) We show that our framework can be expanded in order to apply it even if historical sources do not report the detection/non-detection of species for repeated surveys (i.e., sources do not report the information usually required to run SODMs). In absence of the information required to estimate historical detection probability, we compared different scenarios, assuming that past misdetection probability was a) equal; b) 2, 3, 4, 5, 10, 20 or 50 -times higher than in recent, targeted surveys. These scenarios can be used to evaluate the verisimilitude of trends suspected on the basis of expert opinion or sparse data (as is the case of many species assessments; IUCN, 2001). Our framework allows the production of realistic estimates of population trends even in the absence of complete historical data, and can enable integrating data from standardized monitoring with other heterogeneous sources (Gómez-Rodríguez et al., 2012; Simmonds et al., 2020).

## 2. Methods

### 2.1. Study species and historical data sources

The region between NW Italy and SE France (Western Apennines, Ligurian and Maritime Alps; Fig. 1) is one of the areas of highest species richness and endemism within the Mediterranean Basin (Casazza et al., 2016; Médail and Myers, 2004). The Strinati's cave salamander, *Speleomantes strinati*, is endemic to this region; this species generally lives in caves, in near-surface underground environments, but, under suitable weather conditions (cold, rain), it can also be active outdoors, particularly for feeding (Lanza et al., 2006; Rosa et al., 2023). Nevertheless, caves are the site where this species is most easily detected and monitored, particularly from late spring to early autumn (Ficetola et al., 2012; Salvidio et al., 2016).

Historical data were obtained from two main sources (Bologna and Vigna Taglianti, 1985; Sanfilippo, 1950). We were able to identify the location of 40 caves, 12 monitored by Sanfilippo (1950) and 28 described by Bologna and Vigna Taglianti (1985). Sanfilippo (1950) performed biospeleological surveys in the caves of the Genova Province (East of the study area; Fig. 1) from 1940 to 1950 and reported all the detected animals. Despite several cavities being monitored multiple times, in most cases Sanfilippo (1950) did not report the detection/non-detection of species during each survey. Only in 2 cases he reported that a specific cave was monitored multiple times during suitable months (2 and 8 times, respectively) but he did not detect salamanders.

Bologna and Vigna Taglianti (1985) combined direct surveys (performed in 1958–1982) with a thoughtful research of the literature to provide data on the caves of Western Liguria and Southern Piedmont. We only considered caves that were surveyed by a biospeleologist who usually reported the occurrence of salamanders (i.e., those biospeleologists that detected salamanders in multiple cavities). In several caves, only detections are reported, while surveys without detection are not reported. We considered the cave as occupied if there were records of salamanders, while we assumed that a cave was visited without detection of salamanders if 1) it was surveyed by a biospeleologist that usually reported the occurrence of salamanders (i.e. if this biospeleologist reported salamanders in several other cavities) and 2) this biospeleologist recorded cave-dwelling organisms in that cave but not cave salamanders. We found six caves where a biospeleologist performed more than one survey per year and reported the entire community of cave-dwelling organisms detected during each survey. In these cases, we used these data to obtain information on the detection/non-detection of cave salamanders.

### 2.2. Recent surveys

In 2011–2023, we used visual encounter surveys to assess the occurrence of cave salamanders in the 40 cavities identified from the literature. Surveys were performed in late spring / early summer (June–July), when underground activity of cave salamanders is high (Lunghi et al., 2015). The surveys were performed during the central hours of sunny and dry days. Caves were explored entirely or until the point where the progression required the use of speleological equipment. In each survey, 3–7 people actively searched for active salamanders. In order to assess the detection probability of the species, most caves (70 %) were surveyed twice during the same season; 3–10 days elapsed from the first to the second survey. Only a few caves were not surveyed multiple times during the same season due to logistic constraints and accessibility issues. Previous research showed that this sampling method allows robust assessment of cave salamander distribution and that, given the high detection probability of the species, two surveys provide reliable estimates of species occupancy (Ficetola et al., 2018a; Ficetola et al., 2012).

### 2.3. Habitat changes

We considered the cover of natural vegetation in the landscape surrounding each cave (hereafter: natural vegetation cover) as a potential environmental driver of population change. Natural vegetation cover was calculated on the basis of land-cover maps; each survey was associated with maps representing the land cover that occurred roughly at the moment of the survey. For historical data, we used the land cover map produced by the National Research Council using cadastral datasets from 1956 to 1968, at a geographic scale of 1:200,000 (MCL1960) and digitized by Falcucci et al. (2007). For recent data, we used the Corine Land Cover 2018 map (resolution: 100 m), which was mostly based on satellite images acquired in 2017–2018 (Buttner et al., 2021; European Environment Agency, 2020). For both land-cover maps, natural vegetation cover was calculated as the percentage cover of natural vegetation (all types of forests; shrublands, natural grasslands, and heathlands) within a radius of 1000 m around each cave.

### 2.4. Statistical analyses

We implemented three types of SODMs in a Bayesian framework to assess long-term changes in site occupancy. In the first model, we estimated detection probabilities of historical and recent surveys separately on the basis of available repeated detection/non-detection data (Table 1), to assess possible differences in detection probabilities in the two periods, and to obtain quantitative estimates of population changes exploiting the information from both past and present surveys. As in classical SODMs, the model included an ecological component and an observational component. In the ecological part of the model, the true state (presence/absence) in a site  $i$  in the period  $k$  followed a Bernoulli distribution, with probability  $\psi$ . In the observational part of the model, the detection probability was defined separately for historical ( $k = 1$ ) and recent ( $k = 2$ ) records, so that the general structure of the models is:

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

$$y_{i,j,1} | z_{i,1} \sim \text{Bernoulli}(z_{i,1}p_1)$$

$$y_{i,j,2} | z_{i,2} \sim \text{Bernoulli}(z_{i,2}p_2)$$

where  $y_{i,j,1}$  and  $y_{i,j,2}$  are detection/non-detection at site  $i$ , survey  $j$ , and period 1 or 2. Then, to assess whether detection probability of historical and recent records differ, we calculated a derived parameter  $p_{diff}$ , defined as the difference between the detection probabilities of recent and historical records:

$$p_{diff} = p_2 - p_1$$

Additionally, we estimated the “mis-detection multiplier” indicating how much mis-detection probability (i.e. the probability of not seeing the species during a survey, when it was actually present:  $1 - \text{detection probability}$ ) of historical surveys was compared to recent surveys. Mis-detection multiplier was calculated as the ratio between mis-detection probability of historical and recent data:

$$m = \frac{(1 - p_1)}{(1 - p_2)}$$

**Table 1**

Summary of detections/non-detections of cave salamanders in the study caves, on the basis of historical and recent data.

	Historical data (1940–1982)	Recent data (2011–2023)
Salamanders detected	27	33
Salamanders not detected	13	7
Caves with multiple surveys	8	24

Finally, to estimate population changes, we calculated a derived parameter describing the proportional change in occupied sites between historical and recent periods, as:

$$\psi_{prop} = \frac{\sum_{i=1}^{n_{site}} z_{i,2} - \sum_{i=1}^{n_{site}} z_{i,1}}{\sum_{i=1}^{n_{site}} z_{i,1}}$$

with  $n_{site}$  being the number of sampled caves. Subsequently, percentage change was transformed into annual population trend  $D_a$  with the following formula:

$$D_a = 1 - (1 - \psi_{prop})^{\frac{1}{L}}$$

where  $L$  is the average timespan (in years) between recent and historical surveys (i.e. 58.3 years) (Bird et al., 2012; Tracewski et al., 2016).

In the second model, we assessed the potential drivers of temporal changes in cave occupancy using a dynamic occupancy model. Occupancy for recent surveys was defined as follows:

$$z_{i,2} | z_{i,1} \sim \text{Bernoulli}(z_{i,1}\Phi_i + (1 - z_{i,1})\gamma_i)$$

with  $\Phi$  and  $\gamma$  being persistence and colonization parameters respectively. We then assessed the relationships between these parameters and the change in natural vegetation cover.

In the third model, we expanded our framework to allow the estimation of temporal trends when repeated surveys are not available for historical records. In this model, the mis-detection multiplier parameter was used as a constant to calculate historical detection probability ( $p_s$ ), as follows:

$$p_s = 1 - m \times (1 - p)$$

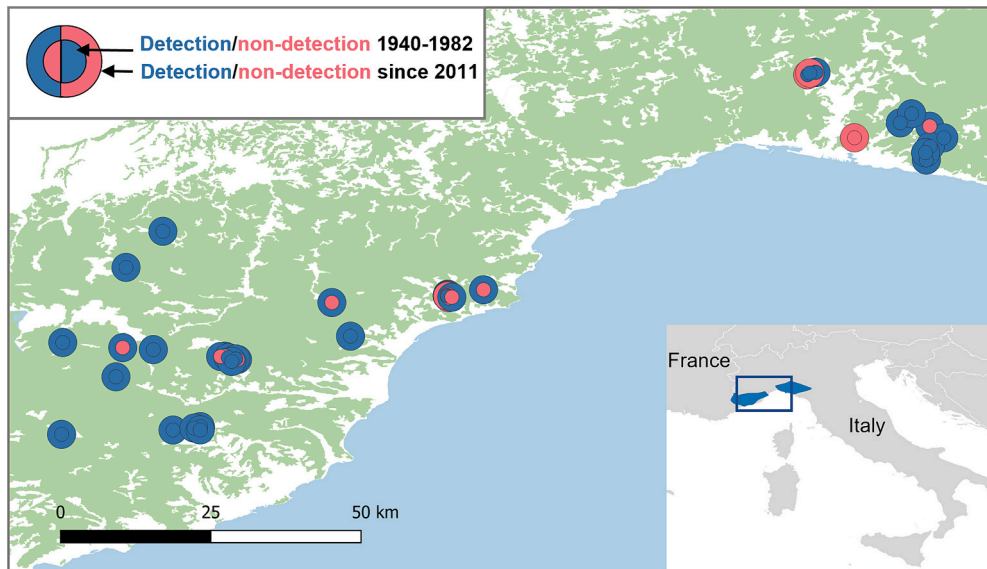
here,  $m$  is the mis-detection multiplier, and  $1 - p$  is the mis-detection probability of recent surveys. To avoid unrealistic estimates of detection probabilities, the lower bound of  $p_s$  was set to 0.1. We run this model with different mis-detection multipliers, hence assuming that mis-detection probability was equal, 2, 3, 4, 5, 10, 20 or 50 times higher than in recent surveys (the code is available at figshare: <https://doi.org/10.6084/m9.figshare.25429027.v1>).

## 3. Results

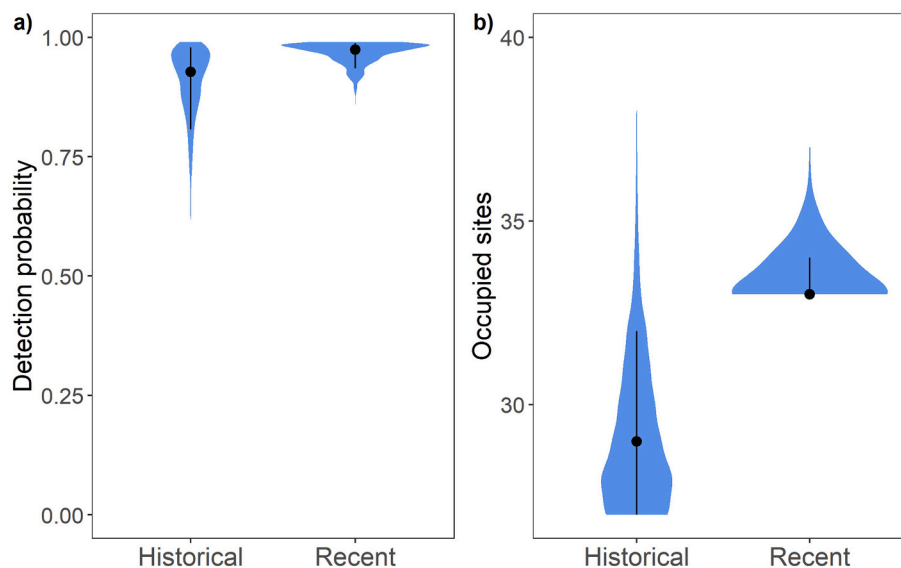
The historical sources reported cave salamanders in 27 out of 40 caves, while recent surveys detected them in 33 caves (Table 1). Both recent and historical sources detected salamanders throughout the whole range (Fig. 2).

### 3.1. Comparison of detection probability and occupancy between historical and recent surveys

Site occupancy-detection models (SODMs) showed a very high detection probability of salamanders both in historical and recent surveys (historical surveys: median estimated detection probability = 0.92, 95 % credible intervals [CI] = 0.73–0.99; recent survey: detection probability = 0.97, 95 % CI = 0.91–0.99; Fig. 3a). The difference in detection probability between recent and historical surveys was close to zero (median: 0.04; 95 % CI = –0.04–0.23); still, the mis-detection multiplier was estimated to be positive, indicating that mis-detection was probably higher during historical surveys (median: 2.67; 95 % CI = 0.33–14.68). SODM estimated that, in historical times, 29 caves were occupied by cave salamanders (95 % CI = 27–34), while the present occupancy was 33 (95 % CI = 33–35, Fig. 3b). Considering mis-detection, occupancy showed an average annual increase of 0.3 % (95 % CI = 0.00 %–0.514 % Fig. 4, blue bar).



**Fig. 2.** Detection/non-detection of cave salamanders in 40 caves where historical (inner circles) and recent (outer circles) data are available. The green background indicates the present cover of natural vegetation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article)



**Fig. 3.** a) Detection probability and b) number of caves occupied by salamanders, as estimated by SODMs for historical and recent surveys. Violin plots show the range of the posterior distribution, while bars represent the 80 % credible intervals of the posterior distribution. Smoothing bandwidth was set to 0.5 in b).

### 3.2. Potential drivers of occupancy changes

Natural vegetation cover increased over time, with historical records showing an average natural vegetation cover of 71.3 % and recent records showing an average cover of 90.1 % (Fig. S1).

Habitat change (i.e. difference in the cover of natural vegetation) was not strongly related to population persistence ( $\beta = 0.65$ ; 95 % CI =  $-0.65$ – $2.47$ ) or colonization ( $\beta = -0.14$ ; 95 % CI =  $-2.39$ – $1.86$ ).

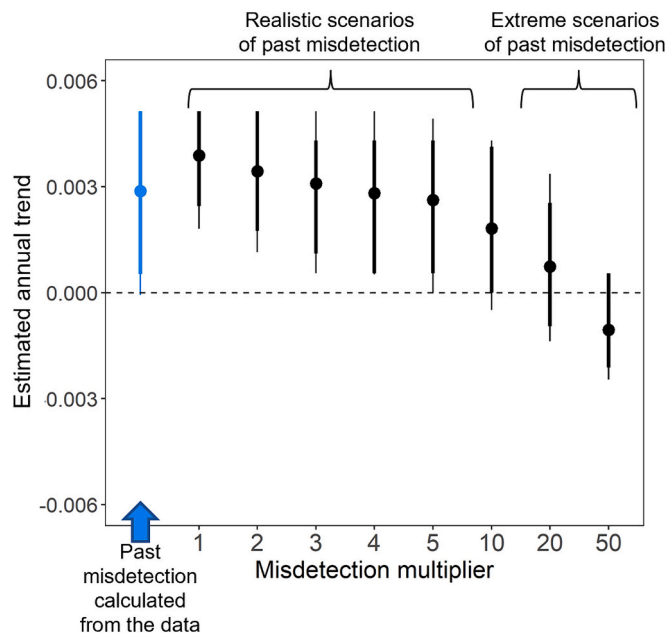
### 3.3. Estimating long term changes in absence of estimates of detection probability in the past

In the absence of historical repeated surveys, SODM suggested an increase in species occupancy if, in historical surveys, the misdetection probability was up to 2–5 times higher than in recent surveys (Fig. 4).

Even extreme scenarios (historical rate of misdetection 10 to 20 times higher than recent surveys) suggested an increase in the occupancy of the species, even though in this case confidence intervals of population trends overlapped zero. Unrealistic misdetections rates (e.g. misdetection in the past 50-times higher than recent surveys) would be required to obtain negative trends, and even in this case the confidence intervals of trends clearly overlapped zero (Fig. 4).

## 4. Discussion

Historical data are a great resource for conservation biologists and are fundamental to document long-term changes of species distribution. However, many features can differ between historical and recent data sources, and this may make comparisons and trend estimations challenging (Skelly et al., 2003). Our analysis showed that flexible analytical



**Fig. 4.** Annual trend of cave salamanders in the study period, estimated using the misdetection rate obtained from historical data (left blue bar), and estimated using alternative scenarios of past misdetection (past misdetection 1 ×, 2 ×, 3 ×, 4 ×, 5 ×, 10 ×, 20 ×, or 50 × compared to the present one). Circles indicate the median estimate; thick and thin lines indicate 90 % and 95 % Bayesian credible intervals of the estimates. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

frameworks can allow the integration of historical and recent surveys, providing data-driven estimates of species trends. On the one hand, a few historical surveys with complete community assessments can be used to extract information on detection/non-detection. This allows testing if past detection probability was comparable to the present one, producing robust trend estimates. On the other hand, even if historical data on detection/non-detection are absent, we can compare scenarios of the reliability of past absences to tease apart realistic trends from less realistic hypotheses.

#### 4.1. Long-term trends of the Strinati's cave salamander

Cave salamanders generally are easy to detect due to their confident behavior and the high abundances they can reach in suitable cavities (Ficetola et al., 2018a; Lunghi et al., 2022a). Our analysis supports the high detectability of these animals and suggests a similar detection rate in the past. Accessing, exploring, and completely sampling a cave is time- and resource-consuming (Lunghi et al., 2022b; Mammola et al., 2021); as a consequence, there were very few cases in which the same cavity was surveyed multiple times during the same season (Bologna and Vigna Taglianti, 1985; Sanfilippo, 1950). Nevertheless, a few bio-speleologists performed repeated assessments of the whole community, generally in the most accessible caves. Such surveys confirmed that, when present, cave salamanders are generally detected at each survey. In fact, in all sites that received multiple historical surveys, cave salamanders were either detected at all the surveys, either never detected. Therefore, the slightly lower historical detection probability compared to the recent one (Fig. 2a) mostly reflects the larger incertitude, related to the small number of caves with multiple historical surveys.

SODM integrating this incertitude can be used to provide robust estimates of the historical and recent occupancy of the species, that showed a ~ 14 % increase in about 60 years. The IUCN Red List infers that Strinati's cave salamanders are decreasing because of the "decline in the extent and quality of its habitat" (IUCN SSC Amphibian Specialist

Group, 2022). However, such habitat decline has mostly been observed in small areas at the boundary of the specie's range (e.g. the Principality of Monaco) (Renet et al., 2012), while through most of the range the habitat does not seem to have declined. In fact, the majority of landscapes surrounding the study caves underwent an increase in natural vegetation during the last 60 years, in agreement with observations from other Mediterranean regions, where the abandonment of traditional agricultural areas triggered an expansion of forests and other natural habitats (Falcucci et al., 2007; Marta et al., 2021; Queiroz et al., 2014). We highlight that the Strinati's cave salamander is not a strictly cave-dwelling species, as it lives in a large number of sub-surface environments (e.g. soil, small crevices...) and, during cold and humid seasons, is often active in surface natural environments such as the forest floor (Rosa et al., 2023; Salvidio et al., 2017). Our sites were restricted to the Italian portion of the species range. Most (~75 %) of the range of the Strinati's cave salamander lies in Italy, thus our data probably represent the situation of the majority of populations, still, additional data covering range portions not considered here (Renet et al., 2012) would be important for a more complete view of the species status.

Changes in natural vegetation cover did not explain changes of occupancy. This is probably related to the limited number of sites undergoing extinction/colonization, which reduces statistical power. Our dataset could be integrated with additional data covering the whole range, with information on populations that do not exploit caves (e.g. populations exploiting the soil of forests) and, ideally, with assessments of local abundances (Ficetola et al., 2020; Renet et al., 2012; Rosa et al., 2023; Salvidio et al., 2016) to provide a robust baseline for prompt detection of local declines in the future, and can be used as reference to understand the fate of cave salamanders under emerging threats such as infectious diseases or climate change (Dondero et al., 2023; Falaschi et al., 2019).

The integration of different sources imposed some limitations to our models. First, past data on detection/non-detection were only available for very few sites (Table 1), and this increased the uncertainty of the estimates of past detection probability. Such uncertainty has repercussions on downstream analyses and resulted in broad confidence intervals of several parameter estimates. For instance, the estimated trend obtained using past detection/non-detection data showed broader confidence intervals than most hypothetical scenarios (Fig. 4). Second, our estimate of population trends was calculated assuming an average interval of 58 years, but past surveys were conducted along a broad timeframe, and the actual intervals ranged between 40 and 83 years. Unfortunately, most caves were sampled in just one year during both historical and recent surveys. This hampered a precise estimation of the number of occupied sites on a yearly basis, which would be required to calculate occupancy trends taking into account different time intervals across caves. Under these circumstances, the use of the average time interval is expected to provide an overall estimate for the whole study period. Finally, due to data limitations, we used the same past land-use for the whole period 1940–1982, despite land-use changes probably occurred in this interval (Marta et al., 2021). This might have reduced the power of analyses testing drivers of occupancy changes. Awareness of these limitations is important, and conservation decisions should consider both worst-case and best-case scenarios.

#### 4.2. Can we apply this approach also to other species?

Our analysis focused on cave salamanders, but the same approach can be also applied to many cave-dwelling species. For instance, a large number of invertebrates are endemic or sub-endemic of the caves of the study area (Bologna and Vigna Taglianti, 1985; Lana et al., 2021). Monitoring efforts toward cave invertebrates are even scarcer than the ones toward vertebrates; still, they are particularly urgent given the many threats affecting them (Hughes et al., 2023; Mammola et al., 2020; Mammola et al., 2018; Mammola et al., 2022). Historical bio-speleological surveys are a precious yet underappreciated source of data

that often remains confined to the gray literature. These data can have limitations such as lack of standardization, lack of absence data, and taxonomic uncertainty. Nevertheless, strict filtering criteria can allow identifying the most reliable information that can be integrated with recent surveys, allowing the exploitation of these neglected data. For instance, >90 invertebrate species occur exclusively or almost exclusively in the caves of the study area, including taxa playing key functional roles in subterranean environments or that are of conservation interest. For most of these taxa, it is generally assumed that information is insufficient to draw any conservation recommendations. However, biospeleologists and entomologists have reported the occurrence of several of them since the beginning of the last century across a large number of cavities. These include several species of carabid beetles (genus *Duvalius*) and of millipedes (e.g. >10 endemic species belonging to genera *Crossosoma*, *Litogona*, and *Plectogona*), for which good-quality information is available (Bologna and Vigna Taglianti, 1985). These data, once appropriately integrated with the outcome of recent surveys, could be used to infer trends of endemic species in the study area, which is one of the regions within the Mediterranean hotspot showing the highest concentration of endemic taxa (Casazza et al., 2016; Médail and Myers, 2004). This may allow unprecedented analyses of temporal trends across multiple taxa, potentially revealing the long-term trajectories of significant portions of underground communities.

Analogous strategies can be used for many species that live in discrete patches that can be re-surveyed after long periods such as ponds, glacier forelands, or mountain summits (Klopsch et al., 2022; Kulonen et al., 2018). Many of these environments are heavily affected by ongoing environmental changes, and the use of an appropriate analytical approach is pivotal to distinguish genuine biodiversity changes from the effects of methodological changes.

#### 4.3. Can we assess long-term trends without estimates of detection probability in the past?

Careful inspections sometimes allow retrieving detection/non-detection data from the gray literature; but, in many cases, non-detections have not been reported and we can only rely on presence records. The lack of detection/non-detection data makes impossible estimating species detectability in the past and can determine an underestimation of past species occupancy, thus reducing the possibility of detecting declines. Such risk is exacerbated if recent surveys are conducted with resource-intensive approaches that maximize detectability. How can we estimate long-term trends under these conditions? Even when direct estimates of past detectability are impossible, we can make assumptions on which conditions should be met to achieve a substantial (or non-substantial) decline. For instance, we can test what would be the effect of a past misdetection probability that was much higher (or lower) than the present one. The flexible Bayesian framework is particularly appropriate for such exercises, given that alternative scenarios can be incorporated directly into the models, thus allowing taking into account uncertainties. In our case, the species trend was robust to a range of past misdetection rates, with realistic scenarios suggesting a stable or weakly increasing trend (Fig. 4). Such comparison of alternative scenarios can be used to assess whether a stable or a declining trend is robust. For instance, in our case, a range-wide species decline is unlikely even with extreme assumptions such as the very improbable hypothesis that, in the past, biospeleologists misdetected salamanders 50 times more than in recent surveys. Similar approaches can be applied to a broad range of data, particularly species checklists from the past.

#### 4.4. Conclusion

We increasingly rely on new monitoring approaches for a better, faster, and more accurate assessment of biodiversity. This has resulted in an exponential growth of available data that provide prompt indications on recent trends and underlying drivers. Nevertheless, older data

sources remain fundamental to identify long-term trends. While recent standardized monitoring schemes accumulate new information, it is important to compare them with the vault of information collected by naturalists through the decades. The application of tailored analytical approaches remains fundamental to fully exploit these data, to retrieve the most accurate information and maximize the efficiency of conservation actions. Still information on species trends, alone, is not enough for effective conservation, and must be explicitly linked to management actions, should a decline be detected. The detection of concerning declines should trigger rapid conservation decisions, such as reducing the exploitation of caves, the protection of disturbed sites, and the restoration of natural vegetation surrounding caves (Hughes et al., 2023; Lindenmayer et al., 2013). Such adoption of trigger points remains rare, particularly for underground species, but can be especially effective for small species linked to specific microhabitat, for which local management actions are extremely beneficial (Hughes et al., 2023; Lindenmayer et al., 2013; Wynne et al., 2021).

#### CRedit authorship contribution statement

**Gentile Francesco Ficetola:** Writing – original draft, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Raoul Manenti:** Writing – review & editing, Methodology, Investigation, Data curation, Conceptualization. **Benedetta Barzaghi:** Investigation. **Samuele Romagnoli:** Investigation. **Elia Lo Parrino:** Investigation. **Andrea Melotto:** Data curation, Investigation. **Silvio Marta:** Investigation. **Simone Giachello:** Investigation. **Valentina Balestra:** Writing – review & editing, Investigation. **Enrico Lana:** Investigation. **Luigi Maiorano:** Validation, Methodology. **Roberta Pennati:** Investigation. **Enrico Lunghi:** Investigation. **Mattia Falaschi:** Writing – review & editing, Methodology, Investigation, Formal analysis, Conceptualization.

#### 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.

#### Data availability

Data and scripts used to run the analyses are available at figshare: <https://doi.org/10.6084/m9.figshare.25429027.v1>.

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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.2024.110695>.

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