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RESEARCH ARTICLE

Shifting baselines for species in chronic decline and assessment of conservation status. Are hazel dormice *Muscardinus avellanarius* Endangered?

Eleanor R. Scopes[1] | Cecily E. D. Goodwin[2] | Nida Al-Fulaij[3] | Ian White[3] | Steve Langton[4] | Katherine Walsh[5] | Alice Broome[6] | Robbie A. McDonald[1]

[1]Environment and Sustainability Institute, University of Exeter, Penryn, UK

[2]UK Centre for Ecology & Hydrology, Crowmarsh Gifford, Wallingford, UK

[3]People's Trust for Endangered Species, London, UK

[4]Hallgarth, Malton, UK

[5]Natural England, Crewe, UK

[6]Forest Research, Northern Research Station, Roslin, UK

Correspondence
Robbie A. McDonald
Email: r.mcdonald@exeter.ac.uk

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Abstract

1. Long-term data are beneficial for monitoring the conservation status of species. Assessments of population change over recent periods of fixed duration will, however, be subject to 'shifting baselines', where the accepted norm for the population at the start of the period already represents a reduction from historical levels. International Union for Conservation of Nature Red List criteria for categorizing conservation threat rely on assessing declines against quantitative thresholds, generally measured over 10 years, as indications of the likelihood of extinction in the near future. By contrast, legal frameworks such as the European Habitats Directive require states to achieve and sustain 'Favourable Conservation Status' for protected species, while domestic conservation legislation can have more diverse objectives and mechanisms, based on local contexts that extend beyond biological or quantitative criteria.

2. We explore the challenges associated with assessing the risk of extinction and the conservation status that arise from the availability of long-term monitoring data for hazel dormice *Muscardinus avellanarius* in the United Kingdom.

3. Numbers of adult dormice counted in the National Dormouse Monitoring Programme are in ongoing decline, amounting to an overall decline of 78% (95% confidence interval = 72%–84%) over 27 years, 1994–2020. If the observed annual rate of decline of 5.7% (95% CI = 4.7%–6.8%) were to continue unabated, dormouse counts would decline by >90% from 1994 to 2034. Despite this, the species would never be categorized as Endangered, under IUCN criteria, which specify a reduction of >50% within 10 years.

4. While such chronic decline may not indicate imminent risk of extinction, justifying a higher Red List category, it is a demonstration of unfavourable conservation status at a national scale. Prioritization based on demonstration of such chronic declines might direct more effective action towards species conservation at a

> point when their recovery is more attainable, rather than attempting later to re-verse a journey to the brink of extinction when the species is finally 'Endangered'.

1 | INTRODUCTION

Given the ongoing biodiversity crisis (IPBES, 2019), robustly assessing a species' risk of extinction has become an indispensable means of quantifying biodiversity loss and prioritizing conservation actions. Long-term monitoring allows updates to species status, and continuing or accelerating declines can indicate targets for stepping up conservation action (Miller et al., 2006), while increasing populations might confirm the validity of actions or be celebrated as successes (Roman et al., 2015). Lists of threatened species inform conservation priorities (Miller et al., 2007) and focus scientific research on knowledge gaps (Jarić et al., 2017). The International Union for Conservation of Nature (IUCN) Red List is the leading tool for categorizing conservation threats (Rodrigues et al., 2006) and has helped substantiate and highlight risks of extinction and identify conservation threats and actions that have averted extinction (Bolam et al., 2020).

Conservation is primarily enacted at the national level (Hunter & Hutchinson, 1994) and so threatened species lists are frequently compiled at this scale, to help inform national action and awareness. Most national lists are based on global IUCN Red List frameworks, adapted for regional assessments (Miller et al., 2007), where modified criteria provide for assessment of the extinction risk facing populations that form part of a species' wider range (IUCN, 2012a). IUCN criteria apply to the assessment of extinction risk, based on recent population trends (criterion A), geographic range (B), population size (C and D) and projected probability of extinction (E; IUCN, 2012b). Within each criterion, there are thresholds and qualifying statements specifying threat level. For example, trend criteria (A) assess population reduction over three generations, or 10 years, whichever is longer, and include past data as well as projections. Threshold levels of decline for each threat category are also qualified by whether the threats the species faces have ceased or are ongoing (IUCN, 2012b). These criteria are similar, whether applied globally or regionally, although regional populations may be accorded lower categories of threat if the risk of extinction is mitigated by reinforcement by other populations outside the focal region (IUCN, 2012a).

Despite the frequent use of threatened species lists in conservation planning, there is continuing debate about such application. Red List criteria and protocols are not designed for uses other than deriving measures of extinction risk, even if there is a persistent misconception that they measure conservation priority (Collen et al., 2016). Their unqualified use in planning and legislation can, therefore, often be inappropriate, as it is likely to be inefficient to allocate resources to species threatened with imminent risk of extinction, instead of preventing less severely threatened species from reaching that point (Possingham et al., 2002).

Because of these issues, several international conservation legislative measures have simpler, non-hierarchical approaches that assign priority to any species that is not thriving, shifting the focus from proximity to extinction and towards deviations from historical status. A key example is the idea of Favourable Conservation Status (FCS), which features in European Union legislative instruments, including the EC Habitats Directive (2017; Epstein et al., 2015) and member state transcriptions of this Directive. FCS encompasses standards for thriving species according to three aspects: a population that is maintaining itself on a long-term basis, the species' range is not being reduced, and there is sufficient habitat to maintain the species in the long term. A species that does not meet one of these three definitions is considered a conservation priority as it is definitively in Unfavourable Conservation Status. FCS also differs from Red List categorisation, as it defines a positive state to reach, while the IUCN emphasizes a state to avoid. The Red List defines species recovery as when they no longer qualify for any of the threat categories even if, for example, populations are still declining, albeit less steeply. It therefore seems easy for such a 'recovered' species to succumb once again to greater risk of extinction, especially if conservation priority is based on Red List status, and conservation action has ceased or reduced.

Most monitoring data used in assessing conservation status are based on direct or indirect counts that, even when repeated and standardized, are often intermittent or short term (Bonebrake et al., 2010; Mihoub et al., 2017). There are consistent calls for the collection of longer-term data because baselines can affect perceptions of population trends, with shorter study periods potentially masking overall change (Bonebrake et al., 2010; Collins et al., 2020). Setting a baseline disregards the changes that occurred beforehand and, given consistency in patterns of biodiversity loss, use of more recent baselines is more likely to underestimate overall declines. This is a form of 'shifting baseline' syndrome, where contemporary losses are underestimated, as degraded populations are accepted as norms without historical context (Soga & Gaston, 2018). This syndrome can make people and processes more tolerant of declines and produce confusion over which baselines or trends should be used (Soga & Gaston, 2018). Short monitoring periods, although able to indicate precipitous declines, are unable to capture more gradual,

chronic losses or identify extinction debt, where population decline and extinction are delayed after habitat destruction (Kuussaari et al., 2009).

Although the IUCN Red List categorisation process has been created to be as scientifically robust as possible, it has some limitations. Key among these is its stated primary application to determine the risk of extinction, rather than conservation status, population change, need for actions, priority or many other informative indicators for conservation. Furthermore, within the categorisation process, some terms in the category descriptions are deliberately vague, to enable their application in diverse contexts. This adds subjective elements to the interpretation of criteria that can introduce bias and uncertainty (Regan et al., 2000). IUCN criteria provide guidance for dealing with uncertainty, although they focus less on statistical uncertainty, and more on instances where the available evidence is limited (IUCN, 2012b). This is for the obvious reason that threatened species are often scarce, sparsely distributed and hard to monitor effectively. Hence, quantified statistical uncertainty does not often characterize the available information. Red List criteria do not encompass quantitative measures of uncertainty, only central estimates are compared with explicit, invariant thresholds to determine the category of extinction risk. Several authors have proposed methods to incorporate statistical uncertainty (Akçakaya et al., 2000; Regan et al., 2000), but it has only recently been considered in formal assessments (Sherley et al., 2020). This means that better quantification of uncertainty, resulting from higher-quality data or analyses, tends not to be accommodated in Red List assessment processes. Finally, choosing between analytical models to quantify trends, where this choice may be partly subjective, can also add uncertainty by producing slightly differing, but similarly valid, estimates of population trends. Thus, changes in the threat category might result not from actual biological changes, but from minor statistical alterations or differences in approach (Possingham et al., 2002). On the other hand, determining FCS is concerned with whether a species' population, range or available habitat is declining or stable/increasing, which requires less statistical power and is less likely to be affected by statistical uncertainty than when measuring against discrete thresholds. Overall, maximizing the value of scarce, hard-won and long-term data by using appropriate analytical approaches and assessment criteria is important to account for long-term change in species status and, thereby, to identify conservation actions and priority.

Hazel dormice *Muscardinus avellanarius* in the United Kingdom are monitored primarily through a large citizen science project, the National Dormouse Monitoring Programme (NDMP). Licensed volunteers install and check nest boxes for dormice during the active season (in the UK hazel dormice are obligate hibernators). The Programme started with six sites in 1988 but has expanded considerably, achieving a statistically useful scale in 1993 and reaching a maximum of 438 sites in 2016. Although the absolute time frame of monitoring is relatively short, compared with the ideal long-term dataset (>100 years; Bonebrake et al., 2010), it is considerably longer than the 10-year period used for Red List assessment.

This allows a robust comparison between trends over 10-year and longer periods, enabling us to examine how well assessment of extinction risk encompasses change in populations and conservation status.

Previous analyses of NDMP data have demonstrated declines in counts of dormice, suggesting population decline of 72% between 1993 and 2014 (Goodwin et al., 2017) associated with reductions in habitat quality (Goodwin et al., 2018). Based, in part, on this analysis, the 2020 Red List for British mammals classified hazel dormice as Vulnerable, on the basis of criterion A2b (Mathews & Harrower, 2020), that is, an apparent decline of >30% but <50% (central estimate = 48% decline, 95% CI = 39%–55%) over the most recently quantified 10-year period (2005–2014; Goodwin et al., 2017), where the reduction or its causes may not have ceased, or may not be understood or may not be reversible (IUCN, 2012b). The most recent assessment of the conservation status of dormice in the United Kingdom, which the Statutory Nature Conservation Bodies had been required to report under Article 17 of the EU Habitats Directive, considered their status Unfavourable, based mainly on the declining counts (JNCC, 2019). Targets set by Natural England (the statutory body for England) for dormice to reach FCS are currently: to re-occupy 49 counties where dormice were known to be present in 1885, a reversal of the population decline to return the population to the level of 1993, and an increase in lowland mixed deciduous woodland with appropriate vegetative and structural composition (Morris, 2021). The decline of dormice in Great Britain is particularly troubling, given the legal protection (Conservation of Habitats and Species Regulations 2017, Wildlife and Countryside Act 1981, as amended) and conservation attention given to this species, suggesting that current actions may not be sufficiently effective to halt the decline. We use this large-scale and long-term dataset to explore the process of categorisation of extinction risk and the assessment of conservation status and to highlight a conservation conundrum whereby the long-term trend obscures more recent losses, yet chronic declines are not reflected in the assessment of extinction risk. We also explore how analytical model choice and statistical uncertainty become particularly important as population trends approach thresholds for threat categories. More specifically, given that estimates of dormouse population trends have run close to, but not passed, the threshold (50% decline over 10 years) that would support Red List categorisation as Endangered (Goodwin et al., 2017), we update trends with new data and ask if, or when, dormice might be Endangered in the United Kingdom.

2 | MATERIALS AND METHODS

2.1 | National Dormouse Monitoring Programme

The NDMP includes sites across England and Wales, spanning the current range for dormice in the UK. Sites are predominantly located in broadleaf and mixed woodlands. Each site is equipped with a grid

of dormouse nest boxes that are checked up to once a month in the active season from May to October, with at least one check post-hibernation in May/June and another after breeding in September/October. Volunteers monitor the number of dormice, along with basic biometric data such as sex, age class and weight. From 1988 to 2021, sites have been monitored for 1–34 years, and have different numbers of boxes, from <10 up to 750. More detailed survey methods can be found in NDMP guidelines (PTES, 2019). The NDMP only surveys a sub-sample of the dormouse population, that is, those using boxes on monitoring sites, meaning the derived trends may not be representative of the entire population or all habitats. Most potential biases, however, have either previously been evaluated, such as variation in survey effort within and among sites (Goodwin et al., 2017), or are unlikely to affect population trends by systematically changing across the course of monitoring; for example, the habitats covered by the programme change as sites are added or abandoned, but this is likely to be random as it is driven by volunteer availability. For the time-being and in the absence of evidence to the contrary or from other non-woodland habitats, we take the trends in counts presented here to be representative of the wider dormouse population.

2.2 | Data analysis

We are interested in modelling variation in dormouse counts from sites where they have been recorded, taking into account recording effort and variation in time. Inclusion of data in this analysis follows the approach established by Goodwin et al. (2017). We excluded data from 1988 to 1992, as only a few sites were surveyed in this period, and excluded data from sites that had been surveyed for 2 years or less, so any site effects could be separated in part from time effects. We also excluded sites that recorded only one dormouse, or none, in the duration of their operation, to avoid zero inflating the data with sites where dormice are not present. Several sites had more than one survey section, which were grouped to reduce the non-independence of related samples. We used the counts of adult dormice in the analyses, as the numbers of younger age-class individuals are not consistently recorded, since they are harder to detect and count, especially as some volunteers dislike disturbing breeding nests. We included data from all months in which surveys took place, to reduce between-month variation impacting yearly trends. Earlier analyses did not find spatial auto-correlation among sites (Goodwin et al., 2017) and this was not included in current models.

All statistical analyses used R version 4.0.2 (R Core Team, 2020). Dormouse counts in nest boxes from 1993 to 2021 were analysed with generalized additive models (GAMs), using the package MGCV (Wood, 2010). The model we used differs slightly from that in Goodwin et al. (2017), in which we used a negative binomial model to deal with overdispersion of data. Overdispersion was measured using the Pearson estimate from the residuals of each model; the Poisson model was overdispersed (estimate: 2.54, greater than 1), and this was effectively reduced by the negative binomial model

(estimate: 1.12). We then investigated two potential model distributions, Poisson and negative binomial, by simulating datasets with different distributions, and seeing if the two models could recover the true decline values. We found that the negative binomial model exhibited some negative bias, leading to a slight overestimation (~3%–4%) of the overall decline, when the data were simulated using any distribution other than negative binomial. Meanwhile, the Poisson model showed low levels of bias irrespective of the actual data distribution, suggesting it is more applicable to this analysis, where an unbiased trend is necessary to assess conservation priority and the underlying distribution of the data is uncertain. We, therefore, used the Poisson distribution in the model. Again, differing slightly from the earlier analyses, we included month as a fixed effect, with six levels for the May to October, as we found that, after the inclusion of additional years of new data, trends varied among months. For consistency and to enable direct comparability, we also applied our earlier (Goodwin et al., 2017) model to the updated data. Thus, we present two models, the current model using Poisson distribution and a fixed effect of month, and the earlier model using a negative binomial distribution without an effect of month, akin to the earlier analysis (Goodwin et al., 2017). In all other respects, the models are similar. Comparing these two models also highlights the role of choice in the statistical approach, enabling us to compare how two similarly valid modelling approaches affect estimates of population trends and consequent assessment of threat categories.

Both GAMs included a fixed effect of site, to account for site-specific variation in counts, and the number of boxes as an offset variable, to account for varying survey efforts. We used fixed effects as our wider ecological analyses focus on site-level variation (Goodwin et al., 2017, 2018). Time was included in the model by including the calendar year in a smoothing function formulated by penalized regression splines. The level of smoothing was determined by qualitative assessment as the number of degrees of freedom (seven), which produced the long-term population signal. To account for the period when the GAM is least reliable (Hewson & Noble, 2009), we do not report trends extending to the first (1993) and last (2021) survey years. Changes for each consecutive year are, therefore, reported for the 27-year period, 1994–2020, with overall changes relative to a baseline in 1994. Bootstrapping with 1000 replicates at the site level, with replacement, was used to calculate the 95% confidence intervals for each model (Fewster et al., 2000).

Because conservation policy is devolved within the United Kingdom, in addition to a UK trend, we calculated separate trends for England and Wales by including the country in which the site was located as an interaction term within the smoothing function, as degrees of freedom are not penalized. This was an extension of the Poisson model. Bootstrapping was stratified by country and three null bootstrap samples for Wales, which were likely produced when the limited sites available at the beginning of the time series were not randomly sampled, were removed from the confidence interval calculation.

We calculated 10-year sliding windows of dormouse population change, in line with the 10-year basis for IUCN Red List criteria

(IUCN, 2012b). Dormouse population change over the entire 27-year programme was, thus, divided into sixteen 10-year windows, using the same Poisson model as above. For each window, we calculated the change in population size by year 10, relative to year 1, starting with 1994–2003, and ending with 2011–2020. Bootstrapping was used to calculate 95% confidence intervals as before. To explore how 10-year windows compared with the overall 27-year trend, we plotted all 10-year windows on the same scale as the overall trend, with results relative to an arbitrary value of one at the starting point for each time frame, thereby explicitly identifying the shifting baseline.

3 | RESULTS

3.1 | Survey summary

Between 1993 and 2021, 838 NDMP sites were monitored. 602 (72%) sites recorded more than one adult dormouse: 534 in England and 68 in Wales. The number of sites that recorded more than one dormouse in any 1 year ranged from 33 in 1995 to 383 in 2016 (Table 1). The mean number of boxes per site was 74.6 (SE = 0.4 boxes), and the mean duration of monitoring was 10.1 years (SE = 0.26). The annual mean number of adult dormice counted per site for these 602 sites varied from a high of 8.8 (SE = 0.96) in 1995 to a low of 1.6 (SE = 0.08) in 2021 (Table 1).

3.2 | Dormouse population change

A total of 552 sites had been surveyed for more than 2 years and were used in analysis. Hazel dormouse counts in nest boxes declined by 78% (95% confidence interval = 72%–84% decline; Figure 1) over the 27-year period from 1994 to 2020, according to the Poisson model. This model had an adjusted R-squared value of 0.506, and explained 49.2% of model deviance, reflecting the dominance of the trend in time explaining variation in dormouse counts. This equates to a mean annual decline of 5.7% (95% CI = 4.7%–6.8%). While confidence intervals overlap, the central estimate of decline, on which Red List criteria are based, has worsened since the previous analysis, which estimated a decline of 72% (95% CI = 62%–79%) from 1993 to 2014 (Goodwin et al., 2017). The mean annual decline, however, is similar to the earlier estimate of 5.8% (95% CI = 4.5%–7.1%; Goodwin et al., 2017). Applying the model from the earlier analysis (Goodwin et al., 2017) to the new data, with a negative binomial distribution and no inclusion of month, resulted in a slightly greater estimate of 83% decline (95% CI = 75%–86%) in the period 1994–2020 (Figure 1). This negative binomial model also had a lower adjusted R-squared value of 0.424 and explained only 41.2% of model deviance. This is a mean annual decline of 6.5% (95% CI = 5.2%–7.2%). Patterns of decline in the two models are similar, although estimates of decline in the negative binomial model are consistently slightly greater, as expected from our model testing. Furthermore, for the model comparisons, the confidence intervals overlap the central estimates.

TABLE 1 Summary of annual records of dormouse counts from the National Dormouse Monitoring Programme from 1993 to 2021. These data are for all sites where more than one dormouse was recorded and includes the number of sites, the mean (SE mean) numbers of adult dormice counted in nest boxes per site from May to October inclusive, and the mean (SE mean) number of nest boxes per site.

Year	Number of sites	Mean count of adult dormice (SE mean)	Mean number of nest boxes (SE mean)
1993	34	6.2 (0.55)	85 (4.1)
1994	36	5.7 (0.58)	90 (4.8)
1995	33	8.8 (0.96)	97 (5.4)
1996	44	6.6 (0.53)	93 (4.1)
1997	53	6.3 (0.56)	93 (4.2)
1998	70	4.4 (0.35)	75 (3.1)
1999	78	4.9 (0.34)	81 (3.3)
2000	109	4.6 (0.32)	71 (2.4)
2001	111	3.6 (0.24)	78 (2.6)
2002	103	4.2 (0.35)	86 (3.5)
2003	145	3.5 (0.18)	82 (2.8)
2004	153	4.5 (0.32)	78 (2.6)
2005	167	3.4 (0.21)	76 (2.6)
2006	191	3.6 (0.25)	75 (2.5)
2007	192	4.1 (0.23)	81 (2.6)
2008	215	3.0 (0.15)	76 (2.4)
2009	238	3.2 (0.16)	77 (2.1)
2010	268	3.2 (0.15)	74 (1.9)
2011	300	3.5 (0.16)	73 (1.8)
2012	338	2.7 (0.10)	71 (1.7)
2013	344	2.1 (0.08)	71 (1.5)
2014	365	2.4 (0.10)	73 (1.6)
2015	374	2.3 (0.08)	73 (1.7)
2016	383	2.0 (0.08)	72 (1.6)
2017	370	2.0 (0.08)	72 (1.5)
2018	383	1.9 (0.07)	71 (1.4)
2019	369	2.0 (0.08)	72 (1.5)
2020	292	2.0 (0.10)	74 (2.0)
2021	319	1.6 (0.08)	71 (1.6)

The overall decline in counts has implications for returning dormice to FCS in the United Kingdom. Favourable status would entail a return to the population size at the start of NDMP monitoring in 1993 (Morris, 2021). With an estimated current population size of 757,000 individuals (Mathews et al., 2018), to reverse the 78% decline in the Poisson model would mean increasing the population to approximately 3,441,000, while reversing the 83% decline of the negative binomial model would mean returning to 4,453,000. Practically, this means at least doubling the current population in the next 10 years, then doubling the population again in the subsequent 10 years.
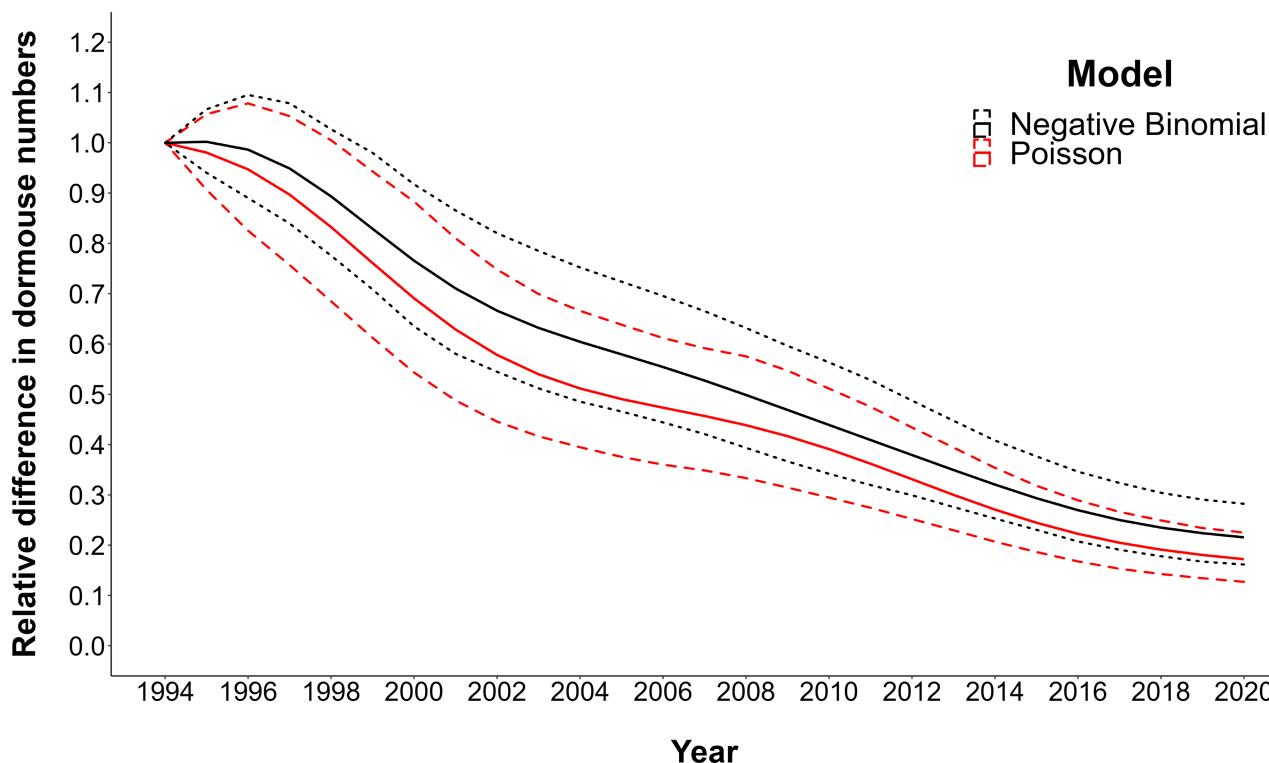
FIGURE 1 Trends in counts of adult hazel dormice from the UK National Dormouse Monitoring Programme. The current Poisson model is in black, and the earlier negative binomial model (Goodwin et al., 2017) is in red. The 27-year period 1994–2020 is shown because the first (1993) and last (2021) survey years, when the model is least accurate, have been removed. Population indices are scaled to a value of one in the first year. The area between the dashed lines shows the 95% confidence intervals for each model, calculated by bootstrapping. All NDMP sites ($n = 552$) used for this analysis recorded more than one adult dormouse and were surveyed for more than 2 years.

Estimates of overall decline in Wales are greater (89%; 95% CI = 67%–95%), than in England (78%; 95% CI = 70%–83%), although uncertainty associated with smaller sample sizes in Wales means 95% confidence intervals for estimates of decline in the two countries overlap.

The model shows declines in hazel dormouse counts have continued apace since the last analysis (Goodwin et al., 2017), with a further 27% decline (95% CI = 18%–35%) in the period 2015–2020. The additional six 10-year windows support a picture of continuing decline, as they contain the highest central estimates of decline (49.9%; 2009–2018), and the windows most closely approaching the 50% threshold, of the whole 27-year period (Figure 2). This suggests that the decline has accelerated slightly at a decadal scale since 2001. Overall, of the eighteen 10-year windows, only six differ significantly from 50%, while the most recent window (2011–2020) suggests a decline of 47% (95% CI = 38%–56%). The negative binomial model also shows the same patterns, although the magnitudes of declines in each 10-year window tend to be greater (Figure 2). Specifically, the central estimates of the five most recent 10-year windows exceed the 50% threshold and the most recent window indicates a 53% decline (95% CI = 45%–59%).

Comparison of the magnitudes of the declines in each 10-year window, relative to the full 27-year trend (Figure 3), shows how the extent of recent declines is obscured when a long-term view of the data is presented. At the same time, the shifting 10-year baseline also obscures the severe chronic decline in the overall, long-term trend. If the estimated mean annual decline (5.7%) were to continue, by 2034 counts of dormice in the United Kingdom would have declined by >90% since 1994. Yet the decline over any 10-year period would never have exceeded the >50% threshold required for dormice to be categorized in a Red List assessment as Endangered. The mean annual decline would have to accelerate to 7.4% per year to surpass a threshold of >50% in a 10-year period.

4 | DISCUSSION

The National Dormouse Monitoring Programme provides unusually extensive, consistent surveillance data, which allow for statistically robust estimates of trends in hazel dormouse counts in woodland habitats across England and Wales (Goodwin et al., 2017). Here, we have found counts of adult dormice in the NDMP have declined by 78% in the 27 years from 1994 to 2020 and declines have continued apace since the earlier analysis (Goodwin et al., 2017). A series of 10-year sliding windows indicate acceleration in the rate of decline when measured over this timeframe. The trends differ between England and Wales, with Welsh counts suggesting a more severe decline. Overall annual rates of decline are sustained, with a mean of 5.7% per year, which, if it were to continue, would mean
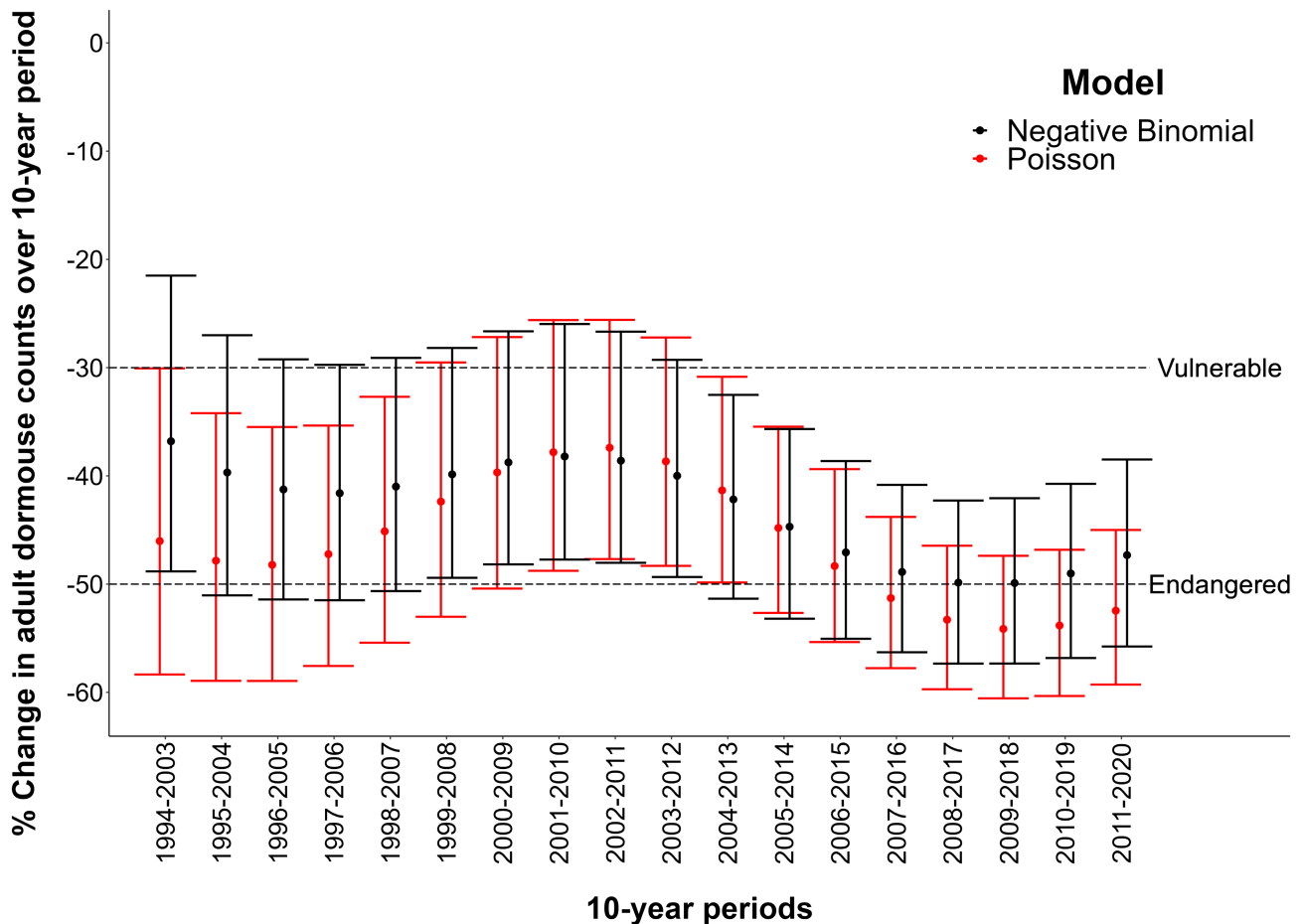
FIGURE 2 Estimates of changes in counts of adult hazel dormice from the UK National Dormouse Monitoring Programme over 10-year sliding window periods between 1994 and 2020. Points from the central estimates of the current Poisson model is in black, and the earlier negative binomial model (Goodwin et al., 2017) is in red. Bars represent 95% confidence intervals for each model calculated using bootstrapping. No population change over a 10-year period would be zero on the graph, while 50% decline is represented by −50. The thresholds for IUCN Red List criteria for changes in population size over a 10-year period for the categories Vulnerable and Endangered are indicated.

the dormouse population would decrease by >90% by 2034, some 40 years after the 1994 baseline.

When the full 27-year view is taken, the decline appears to be flattening compared with the initial reduction (Figure 1), and on the face of it, the estimate of overall decline of 78% up to 2020 differs very little from the estimate of 72% up to 2014. However, when the full period and 10-year windows are set to the same scale (Figure 3), it is clear that the long-term view obscures the magnitude of changes in decadal periods, especially the most recent. This indicates how the scale and presentation of the decline can affect its perception. Although recent declines represent the loss of a small proportion of the starting counts, they represent a significant proportional reduction in those remaining 10 years previously. Without analyses of the 10-year windows, there might be a false perception of recent abatement in the rate of decline, which supports the merits of the focus in IUCN Red List categorisation on short, recent time frames.

Conversely, however, concentrating on the shorter period means the longer term, chronic decline contributes little to threat assessment. The sustained decline, if it remains at its current level, would never surpass the threshold of a 50% decline in a 10-year period, as required for the species to be categorized on the Red List as Endangered, even if counts had declined by >90% over the entire monitoring period. This conundrum is recognized in the background for the development of IUCN criteria, which explains that a population declining by the same proportion each year will never qualify for higher threat category under criterion A, even as it goes extinct, but must instead qualify under the other criteria B–E (Mace et al., 2008). This is because a chronically declining population does not fit with the declining species paradigm (Caughley, 1994), which is the basis for criterion A (Mace et al., 2008), as the imminent risk of extinction is low. This challenge of responding to chronic declines has yet to be explicitly explored. In simulations of hip-pocket frog *Assa darlingtoni* populations, the species mostly qualifies for threatened status when its range becomes small (criterion B) or based on extinction probability (criterion E), despite consistent declines (Keith et al., 2014). The Vancouver Island marmot *Marmota vancouverensis* declined by 90% between 1973 and 2006 (Lindenmayer et al., 2013) but was initially
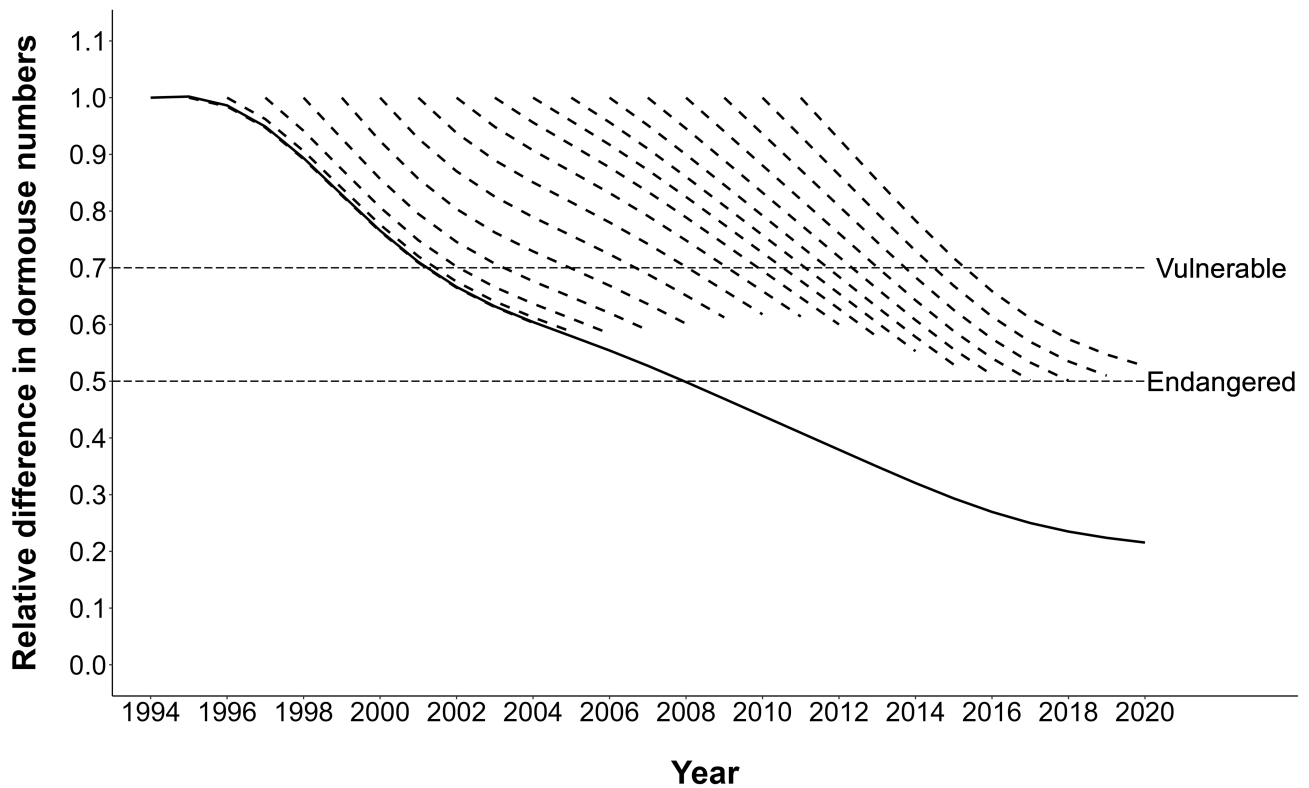
FIGURE 3 Trends in counts of adult hazel dormice from the UK National Dormouse Monitoring Programme from 1994 to 2020. The overall trend is shown as a solid line, and trends over each 10-year sliding window are shown as dashed lines. Each trend is set to the same scale to show how the change of scale effects perception of the trend. Data are from the Poisson model. Population indices are scaled to a value of one in the first year of the full 27-year period or the initial year of each 10-year window. The first 10-year window (1994–2003) is shown but as this window and the full 27-year trend share the same starting point they overlap completely. The thresholds for IUCN criteria for changes in population size over a 10-year period for the categories Vulnerable and Endangered are indicated.

only recognized as Endangered due to small and restricted populations (criteria C and D; Nagorsen, 2000).

While the Red List does as it intends, in assessing and collating imminent risks of extinction, evidence of chronic declines is material to broader assessment of conservation status, for national and international legislation. Measures of change over otherwise somewhat arbitrary (10-year) periods may particularly underplay the importance of declines in species that have short generation lengths but low productivity, which cannot recover their populations as quickly as might be expected for 'R-selected' species. For example, hibernators such as dormice, have relatively low productivity and slower life histories in general compared with similar-sized non-hibernators (Turbill et al., 2011). Such species with a slower life history may be less able to recover from chronic declines and so could, therefore, be considered to be in particular need of conservation prioritization and action, even where not categorized as Endangered by Red List criteria.

Despite IUCN guidance to the contrary, Red Lists are frequently used at least to inform conservation priority (Miller et al., 2007), stemming from the misconception that the IUCN assessment is itself a prioritization scheme (Collen et al., 2016). This misuse may be reinforced by the way the IUCN Red List also understandably dominates public discourse on conservation issues, which may also create risks

for species that are no longer categorized as Threatened, losing public interest when they still require conservation actions. This can also be seen with chronic declines, as with the hazel dormouse, which does not lead to categorisation as high risk of extinction but would be beneficial to incorporate into conservation prioritization and enactment of effective conservation measures.

On the other hand, FCS can take into account chronic declines and the associated issues for conservation prioritization. Any amount of population decline >1% per year is considered Unfavourable (JNCC, 2019), which is consistent with both chronic and acute declines. The amount of decline can be taken into account when defining FCS for the species, as this often requires a reversal of the known declines. By focusing on deviations from evolutionary viability, instead of the proximity of extinction, FCS is broader in assigning the value of conservation action and does so earlier.

Waiting for a species with known chronic decline to dwindle to the extremely small range or population size required to attain a higher category on the Red List, before they are given conservation priority, may restrict the ability to address obligations to attain FCS. Recovery may be easier to secure with earlier action as there is more time for research and to trial alternative actions, and there are likely more options for conservation before a species reaches a small population size. The conservation actions will

likely cost less, as there is less need for expensive procedures like ex situ conservation. Fundamentally, allocating resources to less threatened species, when populations are reduced but remain able to respond to effective conservation measures, is more efficient than reversing a journey to the brink of extinction when a species is finally categorized as Endangered (Possingham et al., 2002). This can be seen with dormice in the United Kingdom; although there is high awareness of their conservation need and legal protection, our analysis suggests this has not been sufficient to stabilize or increase the population. Current conservation actions, like habitat management, reintroductions and mitigation of developments are usually undertaken at very local scales, while broad-scale actions primarily focus on monitoring within woodland habitats. Practitioners have highlighted the need for more proactive landscape-scale approaches to create and improve habitat while promoting connectivity (Philips et al., 2022). Such measures are needed imminently to prevent the population from deviating further from FCS.

Systematic monitoring of threatened species has only recently been established, and follows decades, if not centuries, of anthropogenic threat, and population decline (Mihoub et al., 2017). This includes dormice in the United Kingdom, as the NDMP monitoring was established after well-described declines in range and population, relative to historical data (Bright & Morris, 1996). Thus, the ongoing declines shown here must be set in the context of prior historical losses of unquantified magnitude. Similarly, although truly long-term data (>100 years) are rare (Bonebrake et al., 2010), where available they tend to show greater losses than expected. For several hunted North American mammals, using 1970 as a baseline indicates recent population increases, but a historical baseline of 1850 reveals overall decreases (Collins et al., 2020). Such populations, depleted from their pre-anthropogenic state, may be less resilient and at greater risk of extinction (Mace et al., 2008) or even in extinction debt.

Assessments of conservation status and priority, like FCS, can incorporate this longer-term view with the aim of returning species to an earlier baseline and have the advantage of more readily integrating qualitative information, as the criteria do not include thresholds. Including long-term data could also combat shifting baseline syndrome (Soga & Gaston, 2018) but would require wider dissemination and use of FCS, and its equivalents, in the public discourse.

For hazel dormice in the United Kingdom, the available evidence from a robust monitoring scheme, albeit one confined to the distinct set of habitats where NDMP sites are established, suggests a continuation of their chronic decline. Dormice, therefore, remain in unfavourable status. However, there is further ambiguity in the threat assessment process by considering uncertainty stemming from analytical model choice. The IUCN threshold for Endangered (50%) is missed by most recent decadal declines in the Poisson model (47%) but exceeded by those in the negative binomial model (53%). Both models are valid, and the estimates do not differ statistically, but the 6% difference between these central estimates spans the threshold between Vulnerable and Endangered. Currently,

statistical uncertainty is not incorporated in the IUCN criteria, although it is starting to be discussed in formal assessments (Sherley et al., 2020). Instead, the criteria allow assessors to deal subjectively with uncertainty, providing they are precautionary, favouring the higher threat level when there is evidence for it, and the decision is well documented (IUCN, 2019, Section 3.2). For dormice, this would suggest using the negative binomial model as evidence of the higher threat level and classifying dormice as Endangered. Whether using this shift in Red List category or considering the widening gap between current populations and the 1993 reference population for FCS, our work suggests hazel dormice should be a target for further and more effective conservation action. Considerable effort will be required to return hazel dormice to Favourable Conservation Status in a meaningful time frame, requiring doubling the population in the next 10 years, and again in the following 10 years.

AUTHOR CONTRIBUTIONS

The study was conceived by Ellie Scopes, Robbie McDonald, Alice Broome and Kat Walsh. Data were analysed by Ellie Scopes, with input from Cecily Goodwin and Steve Langton. Data were collected and curated by Ian White and Nida Al-Fulaij. The manuscript was initially written by Ellie Scopes, with input from Robbie McDonald, and all authors commented on drafts and approved the manuscript for submission.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest.

PEER REVIEW

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DATA AVAILABILITY STATEMENT

The data underpinning these analyses will be made available upon request to People's Trust for Endangered Species, and the code is available at the Zenodo data repository at https://doi.org/10.5281/zenodo.7455098 (Scopes et al., 2022).

ORCID

Eleanor R. Scopes https://orcid.org/0000-0002-4410-5985
Cecily E. D. Goodwin https://orcid.org/0000-0003-0093-9838
Alice Broome https://orcid.org/0000-0003-0619-2085
Robbie A. McDonald https://orcid.org/0000-0002-6922-3195

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