# Rarity and conservation status do not predict vertebrate population trends

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**Classification:** Biological Sciences – Ecology

**Short title:** Rarity does not predict population trends

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**Keywords:** population change, rarity, vertebrates, species attributes, geographic range, habitat specificity, population size, Living Planet Database, GBIF, IUCN

**Submission type***:* Research report

**Abstract**

Species' attributes such as rarity and conservation status are often assumed to predict population change which in turn can influence conservation decision-making; however, there are very few empirical tests across taxa and biomes. We use a database of 9284 vertebrate populations from 2084 species in the UK and globally to quantify population change across rarity, Red List status, biomes and taxa. We found that rarity metrics (geographic range, population size and habitat specificity) did not predict population trends in the UK and globally, but greater fluctuations were more likely for smaller populations and species with larger ranges. There was no consistent pattern of net declines with increasing Red List status. We found higher-order taxonomic, but not geographic patterning of population change at global scales, with pronounced declines for amphibians. On a finer taxonomic scale, within amphibian, bird and reptile classes species, for which species-level phylogenies have been especially well resolved, there was no strong phylogenetic or taxanomic signal in the variation in population trends and fluctuations. Our findings suggest that rarity metrics and conservation status are poor predictors of vertebrate population trends of well monitored species, but that species with smaller populations were nevertheless more likely to fluctuate, potentially increasing their extinction risk. Thus, improved population monitoring across a diverse range of taxa is required, as species rarity and conservation status alone are insufficient to inform conservation prioritization.

## Significance statement

Understanding which factors predispose species to population change is essential for improving conservation decision-making. The rarity and conservation status of a species are often assumed to predict population change and thus could inform conservation management. However, empirical tests of this assumption are lacking. We found that rarity and conservation status did not predict population change in the UK and globally, but species with smaller populations and larger ranges were more likely to fluctuate. We found taxonomic but not geographic patterning of population change among over 9000 populations from over 2000 vertebrate species, among which amphibian populations experienced net declines. Smaller populations were more likely to fluctuate and thus could have a higher likelihood of going extinct. Our findings suggest that the rarity and conservation status of species alone do not inform conservation prioritization, and improved monitoring of diverse taxa are required to capture global population change.

## Introduction

Species' attributes such as rarity, conservation status, phylogenetic relatedness, distribution and taxon are often assumed to predict variation in extinction risk and in population change (Julliard *et al.* 2004; Shultz *et al.* 2005; Tonini *et al.* 2016; Jetz & Pyron 2018). Population trends (Hutchings & Reynolds 2004) and population fluctuations (Melbourne & Hastings 2008) have both been frequently linked with a higher likelihood of a species' extinction. For decades, conservation efforts have focused on protecting rare species (Pigott 1977; Gunn 1980; Longton & Hedderson 2000; Povilitis 2001; Gaston & Fuller 2008), those with restricted geographic extents, small population sizes or high habitat specificity, as these species are thought to be more likely to be threatened with extinction. However, there are very few empirical tests of the influence of rarity and conservation status on population change and extinction risk (e.g., Purvis *et al.* 2000; O'Grady *et al.* 2004; Ripple *et al.* 2017), and those that exist mostly focus on either specific taxa (Jones *et al.* 2003; Cardillo *et al.* 2004; Fritz *et al.* 2009) or on population declines alone (Purvis *et al.* 2000; Fritz *et al.* 2009; Gilroy *et al.* 2016; Hefley *et al.* 2016; Ceballos *et al.* 2017), and not population change across taxa and geographic regions. Population change is the fundament process leading to community reassembly (Batt *et al.* 2017). And thus, in a period of accelerating human impacts and global change (Ehrlén & Morris 2015; Hefley *et al.* 2016), global analyses of population change are required to inform conservation policy and decision-making.

Understanding what factors best explain population change and extinction risk across species has been a key focus in population ecology and conservation biology, with possible measures including species' attributes and traits, such as rarity

(Gaston & Blackburn 1995; Owens & Bennett 2000; Purvis *et al.* 2000; Cardillo *et al.* 2004; O'Grady *et al.* 2004; Olden *et al.* 2007; Collen *et al.* 2009; Ripple *et al.* 2017). Extinction risk, the likelihood that a species will go extinct within a given time frame (Mace *et al.* 2008), is widely assumed to increase when species are rare (i.e., they have small geographic ranges (Jones *et al.* 2003), small population sizes (Green 2003) and are very specialized in their habitat preferences (Gilroy *et al.* 2016). As per population dynamics theory, species with small populations are more likely to undergo stochastic fluctuations that could lead to extinction at any given point (Lande 1993; Melbourne & Hastings 2008) and local population declines and extinctions can lead to eventual global species extinction (Ceballos *et al.* 2017). Small populations are also more vulnerable to demographic stochasticity and inbreeding (Kareiva 1990; Hanski 1998). Allee effects, the relationship between individual fitness and population density, further increase the likelihood of declines due to lack of potential mates and low reproductive output once populations reach a critically low density (Dennis *et al.* 2016; Sun 2016). Thus, small populations from rare species are thought to be more likely to experience population declines (McKinney 1997; Purvis *et al.* 2000; Manne & Pimm 2001; Hutchings & Reynolds 2004).

Investigating population change over time in relation to rarity can provide insights into species' extinction risk. Rabinowitz (1981) defined 'seven forms of rarity' based on the combinations of the extremes of three characteristics: geographic range (small vs. large), habitat specificity (generalist vs. specialist), and local population size (small vs. large). Rarity can be either a state in which species naturally occur, or a state to which they are driven following external perturbations to populations

(Magurran & Henderson 2003). However, previous analyses of the influence of rarity on population trends and extinction risk have shown mixed results. Geographic range did not correlate with species' vulnerability to extinction in 45 bird and mammal taxa from all continents except Antarctica (O'Grady *et al.* 2004) and the minimum viable population size did not vary with geographic range and population size in 629 invertebrate, 529 vertebrate and 30 plant species (Brook *et al.* 2006). In contrast, in bats, geographic range was found to be the most important predictor of extinction risk (Jones *et al.* 2003). Thus, a generalizable relationship between rarity and population declines and increased extinction risk across taxa and geographic regions remains unquantified. However, simplifying the concept of rarity so that it can be modelled as a predictor of population change remains a considerable challenge (Magurran 2013; Batt *et al.* 2017).

Investigations of potential factors that predispose species to population change across geographic and taxonomic scales (e.g., from country to biome level, and from population to taxa level) are particularly valuable for guiding conservation decision-making (O'Grady *et al.* 2004; Ehrlén & Morris 2015; Hefley *et al.* 2016; Batt *et al.* 2017). Conservation actions are predominantly implemented on smaller, country-specific, regional or local scales (Mills *et al.* 2010; Guerrero *et al.* 2013), yet policy may be determined for larger regions or broader taxonomic groups (Agardy 2005; Pressey *et al.* 2007). Similarly, overall species' Red List status is determined on a global scale based on multiple criteria (Mace *et al.* 2008), but on a local scale, populations may vary in the direction and magnitude of experienced change (Ehrlén & Morris 2015; Lawson *et al.* 2015; Gilroy *et al.* 2016; van Strien *et al.* 2016; Leung *et al.* 2017). Population time-series data from the Living Planet Database (LPD),

complemented by extensive occurrence data from the Global Biodiversity Information Facility (GBIF) and BirdLife (BirdLife International 2018), and information on species habitat preferences and conservation status from the IUCN Red List, provide a unique opportunity to test which species attributes explain global and country-specific population trends across well-monitored vertebrate species.

Here, we addressed the research question: Is variation in vertebrate population change explained by differences in species attributes, such as rarity metrics and conservation status? We investigated two types of population change – trends and fluctuations. We combined open source data from the LPD, GBIF and IUCN to examine population change in the UK, a country with high-quality monitoring and abundance data, and global population change across rarity, species' Red List status, biomes, systems (freshwater, marine, terrestrial) and taxa. We considered rarity to be the state in which species exist if their geographic range is small vs. large, their local population size is low vs. high, or they are habitat generalists vs. specialists. Thus, we quantified rarity through these three individual metrics - geographic range extent, mean population size, habitat specificity, and additionally species' Red List status.

We tested the following hypotheses: (1) rare species are more likely to be experiencing negative population trends than common species on UK and global scales; (2) declining populations are more likely for species from more threatened Red List categories, (3) there are geographic, taxonomic and phylogenetic patterns in vertebrate population trends and fluctuations (e.g., hotspots of declines, increases or fluctuations in abundances in specific regions, for specific taxa and among

160 specific clades) and (4) rare species or those with a higher Red List status are more
161 likely to experience greater population fluctuations.
162
163 **Methods**
164 *Population data*
165 To quantify vertebrate population change, we extracted the abundance data for all
166 populations monitored at more than five different time points from the LPD (LPI
167 2016), resulting in 9284 populations from 2084 species for our system, biome, taxa
168 and conservation status analysis (global scale) and 508 populations from 237
169 species for our rarity analysis (UK scale, but note that habitat specificity data were
170 only available for 402 populations from 206 species, Table S1). Monitoring duration
171 differed among populations (Figures S1 and S2); however, in our analysis, we
172 retained all populations that had at least five time points of monitoring data. Before
173 analysis, we scaled the abundance of each population to a common magnitude
174 between zero and one, because the populations were monitored using different
175 metrics of abundance (e.g., population indices vs. number of individuals). This
176 allowed us to explore trends among populations relative to the variation experienced
177 across each time series.
178
179 Phylogenetic data
180 We obtained phylogenies for amphibian species from Jetz & Pyron (2018,
181 https://vertlife.org), for bird species from Jetz *et al.* (2012, https://birdtree.org), and
182 for reptile species from Tonini *et al.* (2016, https://vertlife.org). For each of the three
183 classes (*Amphibia*, *Aves* and *Reptilia*), we downloaded 100 trees and randomly
184 chose 10 for analysis (30 trees in total). We could not conduct phylogenetic analyses

185 of population change for the classes *Actinopterygii* and *Mammalia*, as species-level
186 phylogenies for them have not yet been resolved with high confidence (Foley *et al.*
187 2016; Tarver *et al.* 2016).
188
189 *Rarity metrics data*
190 We defined rarity following a simplified version of the 'seven forms of rarity' model
191 (Rabinowitz 1981), and thus consider rarity to be the state in which species exist
192 when they have a small geographic range, low population size, or narrow habitat
193 specificity. We combined publicly available data from three sources: (1) population
194 records for vertebrate species from the LPD (LPI 2016), (2) occurrence data from the
195 Global Biodiversity Information Facility (GBIF 2018) and range data from BirdLife
196 (BirdLife International 2018), and (3) habitat specificity and Red List classification
197 data for each species from the International Union for Conservation (IUCN 2017).
198
199 To estimate geographic range for species monitored in the UK, we calculated a $km^2$
200 occurrence area extent using a minimal convex hull approach based on occurrence
201 data for each species from GBIF (GBIF 2018). We filtered the GBIF data to remove
202 invalid records and outliers using the *CoordinateCleaner* package (Töpel *et al.*
203 2016). We excluded records with no decimal places in the decimal latitude or
204 longitude values, with equal latitude or longitude, within a one degree radius of the
205 GBIF headquarters in Copenhagen, within 0.0001 degrees of various biodiversity
206 institutions and within 0.1 degrees of capital cities. For each species, we excluded
207 the lower 0.02 and upper 0.98 quantile intervals of the latitude and longitude records
208 to account for outlier points that are records from zoos or other non-wild populations.
209 We drew a convex hull to most parsimoniously encompass all remaining occurrence

210 records using the *chull* function, and we calculated the area of the resulting polygon
211 using *areaPolygon* from the *geosphere* package (Hijmans 2016). We did not use
212 IUCN range maps, as they were not available for all of our study species, and
213 previous studies using GBIF occurrences to estimate range have found a positive
214 correlation between GBIF-derived and IUCN-derived geographic range (Phillips *et al.*
215 2017). Extracting and filtering GBIF data and calculating range was computationally-
216 intensive and occurrence data availability was lower for certain species; thus, we did
217 not estimate geographic range for all species globally. Instead, we focused on
218 analyzing range effects for species monitored in the UK, a country with intensive and
219 detailed biodiversity monitoring of vertebrate species, and for birds globally, as they
220 are a very well-studied taxon. To estimate geographic range for bird species
221 monitored globally, we extracted the area of occurrence in $km^2$ for all bird species in
222 the LPD that had records in the BirdLife Data Zone (BirdLife International 2018).
223
224 We calculated mean population size across the monitoring duration for each
225 population using the raw population data. To create an index of habitat specificity,
226 we extracted the number of distinct habitats a species occupies based on the IUCN
227 habitat category for each species, accessed through the package *rredlist*
228 (Chamberlain 2017). We also quantified habitat specificity by surveying the number
229 of breeding and non-breeding habitats for each species from its online IUCN profile
230 (the 'habitat and ecology' section, IUCN 2017). The two approaches yielded similar
231 results (Figure S3, Table S2, key for the profiling method is presented in Table S4).
232 We obtained global conservation status data for all study species through the IUCN
233 Red List classification (IUCN 2017).
234

*Population change modelling*

As the first stage of our analysis, we modeled population change to produce a population trend (slope and variance of population change over time). We used state-space models that model abundance over time (freely-varying intercept) to calculate the amount of overall population change experienced by each population (*mu*), following Dennis *et al.* (2006) and Humbert *et al.* (2009). State-space models account for process noise and observation error and thus deliver reliable estimates of population change when working with large datasets where records were collected using different approaches (Knape *et al.* 2011; Pedersen *et al.* 2011; Leung *et al.* 2017). Not accounting for process and measurement error could lead to over-estimation of population declines (Rueda-Cediel *et al.* 2018). Positive *mu* values indicate population increase and negative *mu* values indicate population decline. We calculated the error around *mu* for each population (half of the difference between the upper 95% confidence limit and the lower 95% confidence limit), with larger errors indicating populations that have experienced more fluctuations (variance in abundance metrics) overtime.

*Population trends across rarity metrics*

To test the influence of rarity metrics (geographic range, mean population size and habitat specificity) on variation in population trends in the UK, we conducted a second stage analysis using six Bayesian linear models (one per metric in the UK and globally) based on Markov Chain Monte Carlo Bayesian inference (Hadfield 2010). The response variable was population trend (*mu* values from state-space models), and fixed effects were geographic range (log transformed), mean population size (log transformed) and habitat specificity (number of distinct habitats

occupied). We used weakly informative priors for all coefficients (an inverse Wishart prior for the variances and a normal prior for the fixed effects), and we refer to predictors as 'significant' when the 95% credible interval for the corresponding parameter in the fitted models did not overlap zero. We did not include random effects in the Bayesian models (e.g., a species or region random intercept), because only 28% of the studied species had more than four monitored populations and spatial replication was low (Table S1).

*Population trends across species' Red List status*

To investigate the relationship between population trends in the UK and globally, and species' Red List status, we modelled population trends from the first stage of the analysis (state-space models) as a function of Red List status (categorical variable) using two Bayesian linear models (for the UK and global databases).

*Global population trends across latitude, biomes, systems and taxa*

To investigate the geographic and taxonomic patterns of population trends, we estimated population trend from the first stage of the analysis (state-space models) as a function of (1) latitude and biome (as defined by the 'biome' category in the LPD, e.g., 'temperate broadleaf forest', Olson & Dinerstein 2002), (2) system (freshwater, marine, terrestrial), and (3) taxon (*Actinopterygii*, bony fish; *Amphibia*, amphibians; *Aves*, birds; *Mammalia*, mammals; *Reptilia*, reptiles). Each categorical model from this second stage of our analyses was fitted with a zero intercept to allow us to determine if net population trends differed from zero for each of the categories under investigation.

*Global population trends within amphibian, bird and reptile classes*

To determine if there is a phylogenetic signal in the patterning of population change within amphibian, bird and reptile taxa, we tested variation in population trends among amphibian, bird and reptile species, the taxa whose phylogenies have been resolved with high confidence (Jetz *et al.* 2012; Tonini *et al.* 2016; Jetz & Pyron 2018). We modelled population change to test the phylogenetic and species-level taxonomic relatedness, which were both included as random effects in three (one per class) Bayesian linear mixed effects models using the package *MCMCglmm* (Hadfield 2010). The models did not include fixed effects. We used parameter-expanded priors, with a variance-covariance structure that allows the slopes of population trend (the *mu* values from the first stage analysis using state-space models) to covary for each random effect. To account for phylogenetic uncertainty, for each class, we ran ten models with identical structures but based on different randomly selected phylogenetic trees. We report the mean estimates and their range for each class. The phylogeny and taxonomy models were ran for 120 000 iterations with a thining factor of ten and a burn-in period of 20 000 iterations. We assessed significance of the random effects (phylogeny and species) by inspecting their posterior distributions, with a distribution pushed up against zero indicating lack of effect significance, since these distributions are always bounded by zero and have only positive values.

*Population fluctuations across systems, taxa, Red List status and rarity metrics*

To test whether rare species fluctuate more than common species and whether there is geographic and taxonomic patterning in population fluctuations, we modelled the

309 error around *mu* (half of the 95% confidence interval) as a response variable across
310 systems, taxa, Red List status and rarity metrics.
311
312 *Monitoring duration and units*
313 To assess the influence of monitoring duration on population trends, we used a
314 Bayesian linear model (Figures S1 and S2). We modelled population trend (*mu*) as a
315 function of monitoring duration (years) for each population, fitted with a zero
316 intercept, as when duration is zero, no population change has occurred. The units of
317 population monitoring across the LPD varied with 1368 different categorical levels
318 (e.g., population index, nest counts, numbers of individuals). Variation in population
319 trends were not explained by sampling method across the five most commonly used
320 units (Figure S5). We estimated population change using state-space models
321 specifically because of their suitability for analyses of large databases where records
322 were derived using different methods (Humbert *et al.* 2009; Leung *et al.* 2017).
323
324 *Comparison of modelling approaches*
325 We conducted the following supplementary analyses: in the second-stage Bayesian
326 models estimating population trends across systems, biomes, taxa and rarity
327 metrics, 1) we weighed *mu* values by the square of *tau*, the observation error
328 estimate derived from the state-space models, 2) we used slopes of linear model fits
329 of abundance (scaled at the population level, centered on zero and with a standard
330 deviation of one) instead of the *mu* estimates from state-space models, 3) we
331 modelled the standard error around the slope values of the linear models and the
332 standard deviation of the raw population data for each time-series as additional

metrics of population fluctuations. All different analytical approaches yielded very similar results (see main text and Figures S6, S7 and S10, Table S2).

All statistical analyses were conducted using R (version 3.4.2, R Core Team 2017). Code and data are publicly available on GitHub (link to be added at time of publication).

## Results

*Population trends in the UK and globally across rarity*

Across monitored vertebrate populations in the UK and globally, species with smaller ranges, smaller population sizes, or narrower habitat specificity (i.e., rare species) were not more prone to population declines than common species (Figure 1a-f, Table S2).

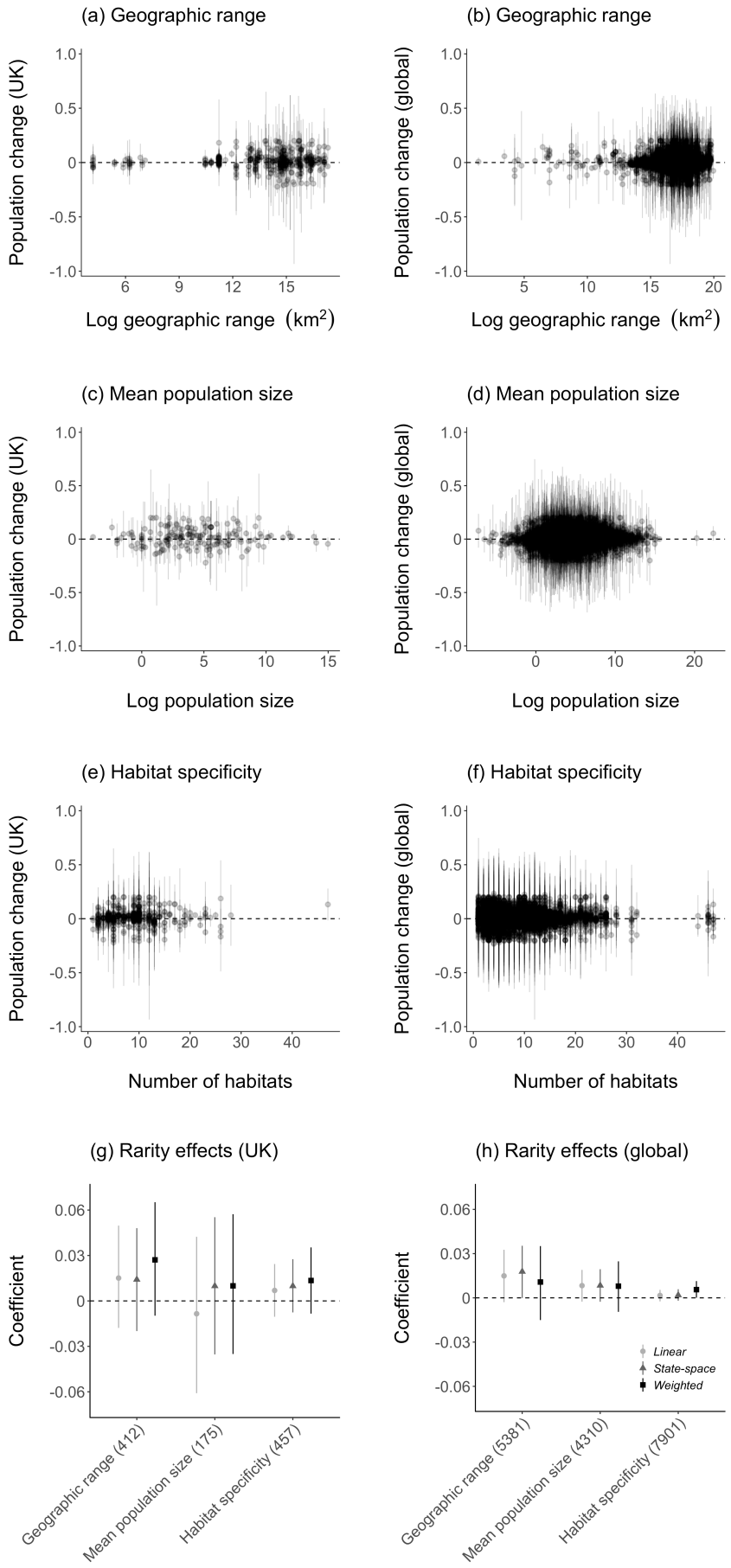
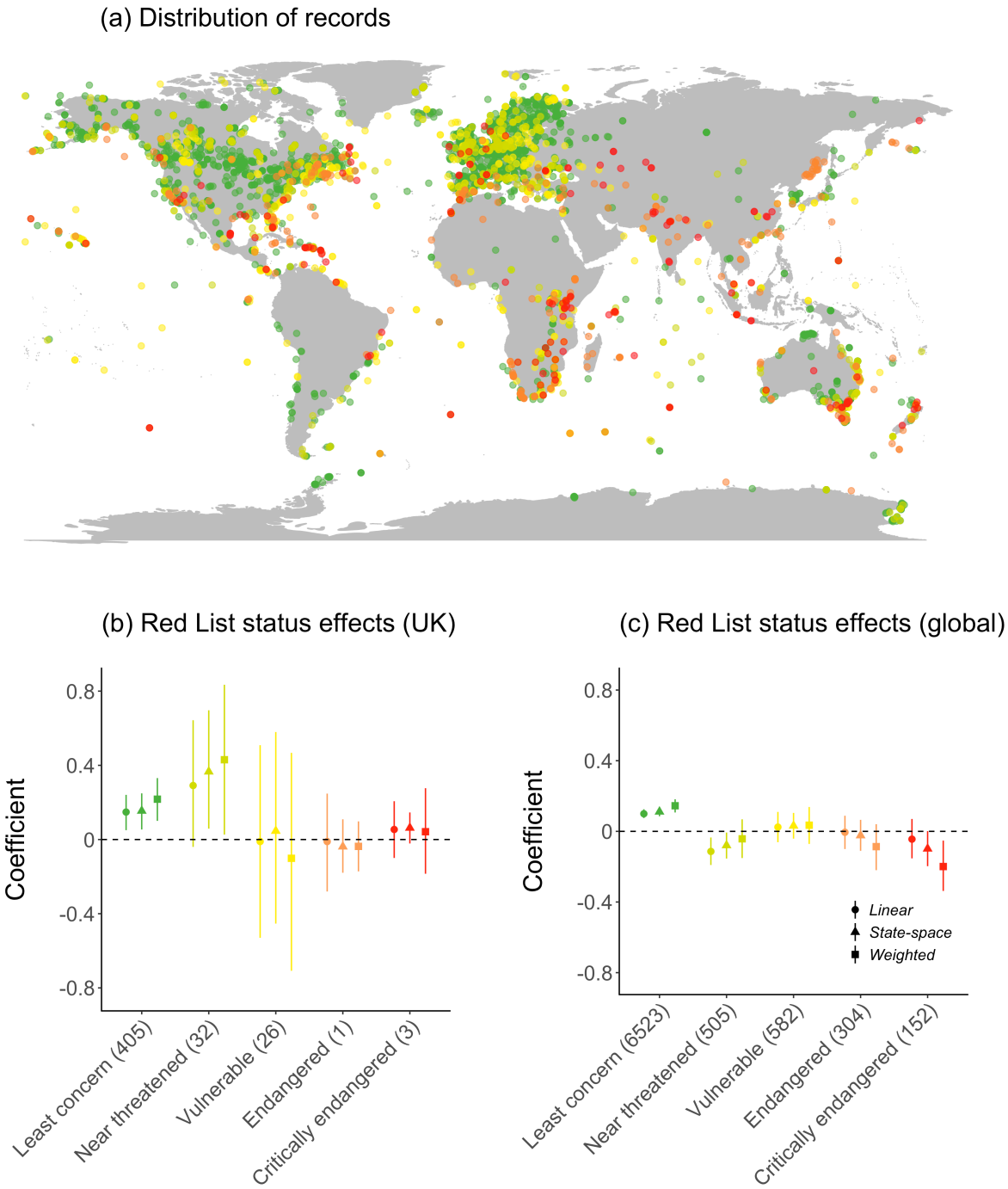

(a) Geographic range
Population change (UK)
Log geographic range (km2)
(b) Geographic range
Population change (global)
Log geographic range (km2)
(c) Mean population size
Population change (UK)
Log population size
(d) Mean population size
Population change (global)
Log population size
(e) Habitat specificity
Population change (UK)
Number of habitats
(f) Habitat specificity
Population change (global)
Number of habitats
(g) Rarity effects (UK)
Coefficient
Geographic range (412)
Mean population size (175)
Habitat specificity (457)
(h) Rarity effects (global)
Coefficient
Linear
State-space
Weighted
Geographic range (5381)
Mean population size (4310)
Habitat specificity (7901)

**Figure 1. Effects of rarity metrics (geographic range, mean population size and habitat specificity) on vertebrate population trends in the UK (plots a, c, e, g) and globally (b, d, f, h).** The numbers on x axis indicate sample size for each category. Geographic range refers to the global range of UK species (a) and the global range of the bird species in the global LPD (b). The *mu* values of population change (plots a-f) are derived from state-space model fits of changes in abundance over the monitoring duration for each population. Effect sizes (plots g and h) are population trends (*mu*) calculated using linear models (light grey circles), state-space models (grey triangles), and population trends (*mu*) weighted by *tau* (black squares), the observation error estimate from the state-space models. Error bars show confidence intervals around population trend (*mu,* plots a-f) and credible intervals for *mu* across rarity metrics (plots g and h). Effect size coefficients (g and h) were standardized by dividing the effect size by the standard deviation of the corresponding input data. See Table S2 for full model outputs.

*Population trends in the UK and globally across Red List status*

Over the monitoring duration for populations in the UK, species classified as least concern and near threatened had a positive population trend (LC: slope = 0.01, CI = 0.004 to 0.02; NT: slope = 0.03, CI = 0.01 to 0.06), whereas vulnerable, endangered and critically endangered populations did not experience directional change (Figure 2b and Table S2). Globally, we found increasing, decreasing and stable populations across all Red List categories of conservation status. For example, a population of the least concern species red deer (*Cervus elaphus*) in Canada declined by 68% over 7 years going from 606 to 194 individuals and a population of the critically endangered Hawksbill sea turtle (*Eretmochelys imbricate*) from Barbados increased

by 269% over 7 years going from 89 to 328 individuals. Populations from least concern species were more likely to increase (slope = 0.01, CI = 0.005 to 0.01), whereas populations from vulnerable, endangered and critically endangered species did not experience net directional change (Figure 2a). Populations from near threatened species were more likely to decline (slope = -0.01, CI = -0.01 to -0.001). Across all categories, there was no consistent pattern of greater population declines in the UK and globally with increasing Red List status (Table S2). Most of the populations in the LPD that were monitored at more than five time points were from species categorized as least concern (Figure 2).




382

383 **Figure 2. Distribution of population records by Red List status (plot a) and**

384 **vertebrate population trends in the UK (plot b) and globally (plot c) across**

385 **species' Red List status.** Colored points on the map indicate the Red List status

386 of each population as indicated in plots b and c. Effect sizes (plots b and c) include

387 the population trend (*mu*) calculated using linear models (circles), using state-

space and linear models (triangles), and population trend (*mu*) weighted by *tau*, the observation error estimate from the state-space models (squares). The numbers on the x axis show the sample size in each Red List category (plots b and c). Error bars show 95% credible intervals (plots b and c). Effect size coefficients (plots b and c) were standardized by dividing the effect size by the standard deviation of the corresponding input data. See Table S2 for full model outputs.

*Global population trends across latitude and biomes*

Global vertebrate population trends did not show distinct geographic patterning, as increasing, stable and declining populations were present on all continents and across all latitudes (Figure 3a and b). Populations in Mediterranean forests, montane grasslands, polar freshwaters, temperate wetlands, tropical forests and tropical coral biomes were more likely to increase, whereas populations from the remaining studied biomes were not more likely to be increasing versus decreasing (Figure 3c and Table S2). The lack of biogeographic patterning in vertebrate population trends was also apparent on a UK scale (Figure S6 and Table S2).

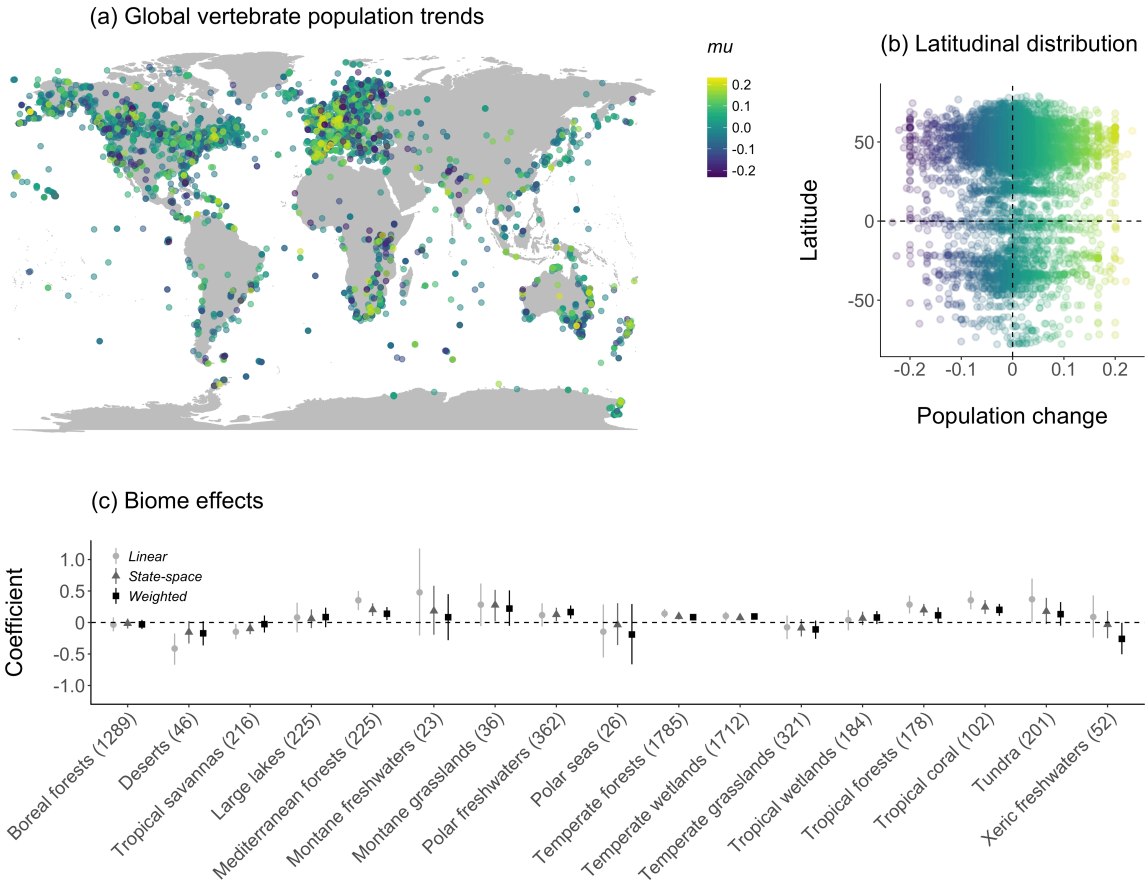
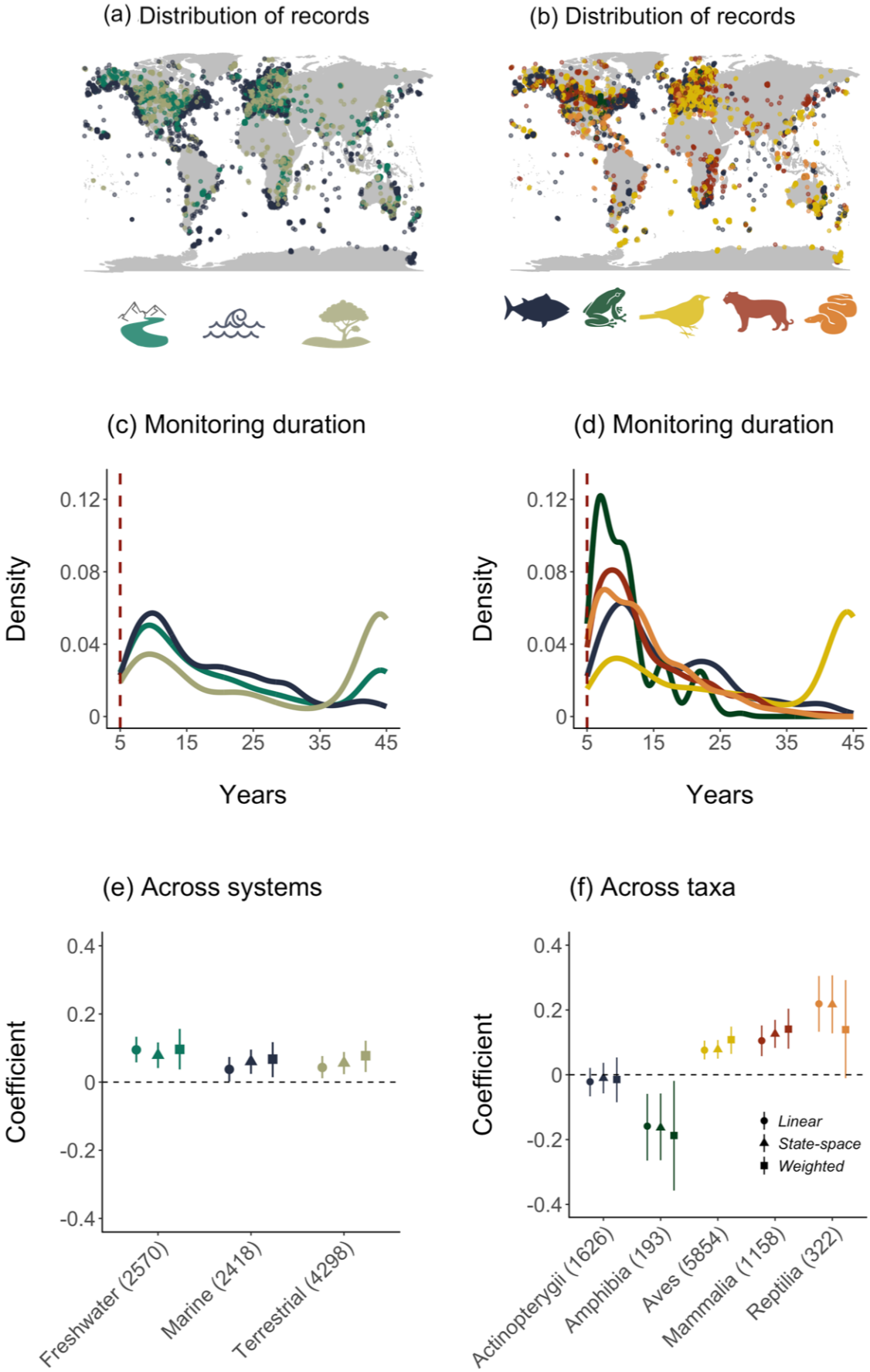



**Figure 3. Lack of biogeographic patterning in global vertebrate population trends (plot a), latitudinal distribution of population trends (plot b), and population trends across biomes (plot c).** Results include 9284 populations from 2084 species. Effect sizes (plot c) include the population trend (*mu*) calculated using linear models (light grey circles), state-space models (grey triangles), and population trend (*mu*) weighted by *tau*, the observation error estimate from the state-space models (black squares). The numbers on the x axis show the sample size for each biome category (plot c). Error bars show 95% credible intervals (plot c). Effect size coefficients (plot c) were standardized by dividing the effect size by the standard deviation of the corresponding input data. See Table S2 for full model outputs.

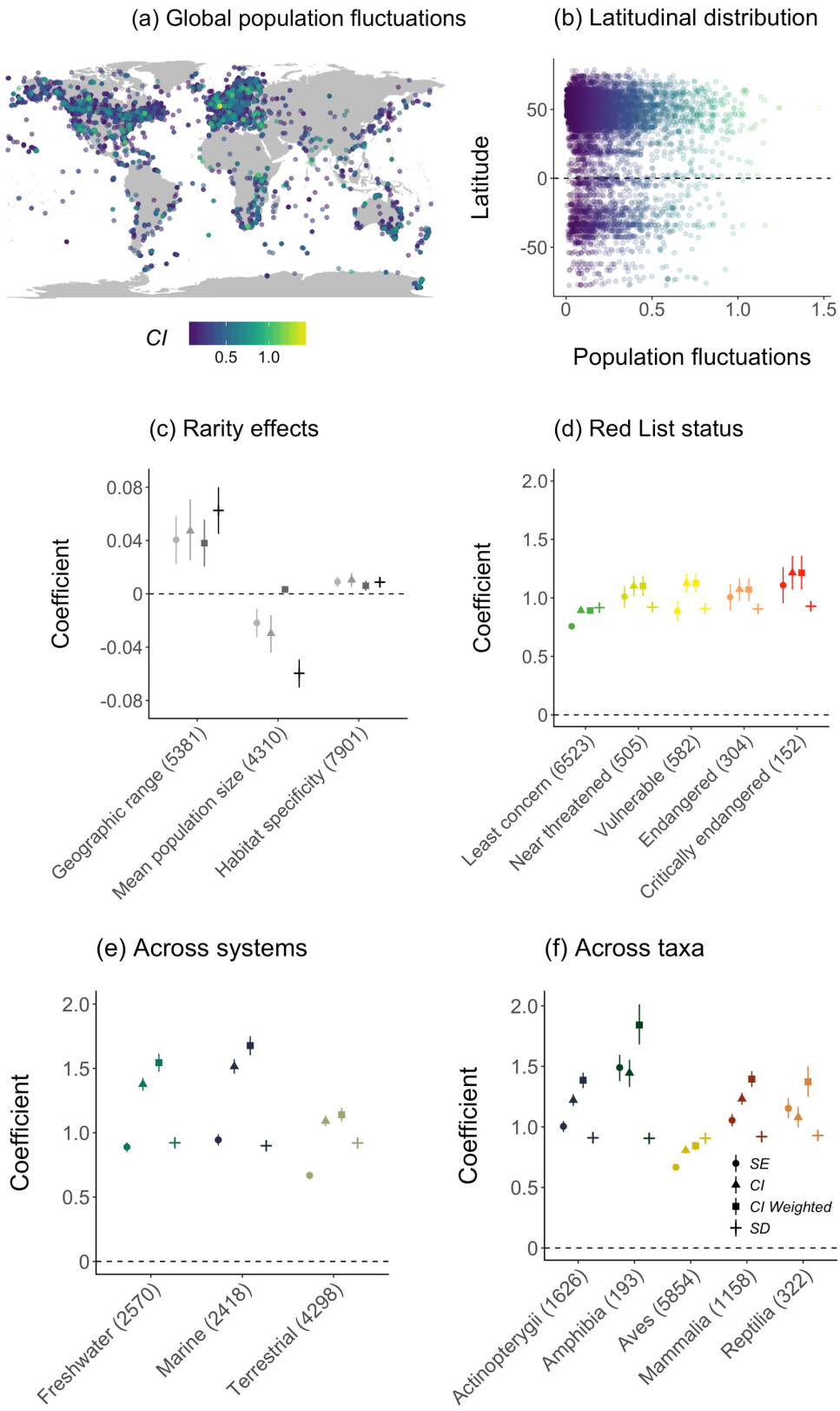
*Global population trends across systems and taxa*

Globally, monitored vertebrate populations experienced population increases in all three studied systems (freshwater slope = 0.005, CI = 0.002 to 0.01; marine slope = 0.004, CI = 0.002 to 0.01; terrestrial slope = 0.003, CI = 0.001 to 0.005, Figure 2a-b, Table S2). In freshwater and terrestrial habitats, there was a bimodal distribution of population change (Hartigans' dip test, $D = 0.04$, $p < 0.01$, Figure S7a), likely driven by the bimodal distribution for bird populations (Hartigans' dip test, $D = 0.04$, $p < 0.01$, Figure S7b). Amphibian populations had net declines (slope = -0.01, CI = -0.02 to -0.005), whereas birds, mammals and reptiles had net increases (slope = 0.004, CI = 0.003 to 0.01; slope = 0.01, CI = 0.01 to 0.01; slope = 0.02, CI = 0.01 to 0.02). Bony fish populations were centred on zero (slope = -0.001, CI = -0.004 to 0.002).


(a) Distribution of records
(b) Distribution of records
(c) Monitoring duration
(d) Monitoring duration
Density
0.12
0.08
0.04
0
5
15
25
35
45
Years
(e) Across systems
(f) Across taxa
Coefficient
0.4
0.2
0
-0.2
-0.4
Linear
State-space
Weighted
Freshwater (2570)
Marine (2418)
Terrestrial (4298)
Actinopterygii (1626)
Amphibia (193)
Aves (5854)
Mammalia (1158)
Reptilia (322)

428

429 **Figure 4. Distribution of records across (plot a) freshwater, marine and**
430 **terrestrial systems and (plot b) bony fish, amphibians, birds, mammals and**
431 **reptiles, monitoring duration (plots c and d) and their respective population**
432 **trend (*mu*) effect sizes (plots e and f).** The numbers on x axis indicate sample size
433 for each category. Effect sizes are population trends (*mu*) from linear models
434 (circles) calculated using state-space models (triangles), and population trend (*mu*)
435 weighted by *tau*, the observation error estimate from the state-space models
436 (squares, plots e and f). The numbers on the x axis show the sample size for each
437 taxa category (plots e and f). Error bars show 95% credible intervals (plots e and f).
438 Effect size coefficients (plots e and f) were standardized by dividing the effect size by
439 the standard deviation of the corresponding input data. See Table S2 for full model
440 outputs.
441
442 *Population fluctuations across systems, taxa, Red List status and rarity metrics*
443 Global population fluctuations (variation in population abundance over time) did not
444 show distinct geographic patterning (Figure 5a-b). Populations that experienced
445 more fluctuations had in general smaller mean population sizes (across all taxa,
446 slope = -0.004, CI = -0.01 to -0.002) and for bird species had larger geographic
447 ranges (slope = 0.005, CI = 0.003 to 0.01, Figure 5c). On a global scale, population
448 fluctuations were more common with increasing Red List threat level (LC: slope =
449 0.11. CI = 0.11 to 0.12; CR: slope = 0.17, CI = 0.15 to 0.19, Figure 5d, see Table S2
450 for all model outputs). Across the three studied systems, terrestrial populations
451 experienced lower population fluctuations (slope = 0.10, CI = 0.10 to 0.11, Figure
452 5e). Across taxa, fluctuations were most common for amphibian populations (slope =
453 0.22, CI = 0.20 to 0.23, Figure 5f).


(a) Global population fluctuations
CI
0.5
1.0
(b) Latitudinal distribution
Latitude
50
0
-50
0
0.5
1.0
1.5
Population fluctuations
(c) Rarity effects
Coefficient
0.08
0.04
0
-0.04
-0.08
Geographic range (5381)
Mean population size (4310)
Habitat specificity (7901)
(d) Red List status
Coefficient
2.0
1.5
1.0
0.5
0
Least concern (6523)
Near threatened (505)
Vulnerable (582)
Endangered (304)
Critically endangered (152)
(e) Across systems
Coefficient
2.0
1.5
1.0
0.5
0
Freshwater (2570)
Marine (2418)
Terrestrial (4298)
(f) Across taxa
Coefficient
2.0
1.5
1.0
0.5
0
SE
CI
CI Weighted
SD
Actinopterygii (1626)
Amphibia (193)
Aves (5854)
Mammalia (1158)
Reptilia (322)

454

**Figure 5. Global population fluctuations in vertebrate species (plot a) across latitude (plot b), rarity metrics (global scale, plot c), Red List status (global scale, plot d), systems (plot e) and taxa (plot f).** The numbers on x axis indicate sample size for each category. The response variables in the models were: 1) the standard error around the slope estimates of the linear model fits of abundance versus year (circles), 2) half of the 95% confidence interval around the *mu* value of population change (triangles), 3) half of the 95% confidence interval around *mu* weighted by *tau*, the observation error estimate from the state-space models (squares), and 4) the standard deviation of the raw data for each population time-series (plus symbols). Larger errors indicate greater population fluctuations. Effect size coefficients (c-f) were standardized by dividing the effect size by the standard deviation of the corresponding input data. Error bars show 95% credible intervals (c-f). See Table S2 for full model outputs.

*Global population trends and fluctuations within amphibian, bird and reptile classes*

When we tested for finer scale species-level taxonomic clustering of population change (both overall population trends and population fluctuations) within amphibian, bird and reptile classes, we did not detect a phylogenetic relatedness signal in the patterning of population change for either trends or fluctuations (Figures 6 and S8, Table S2). Similarly, species identity within amphibian, bird and reptile classes also did not explain variation in population trends or fluctuations (Figures 6 and S8, Table S2). There were no distinct clusters of specific clades that were more likely undergo increases, decreases or fluctuations in populations (Figure 6).

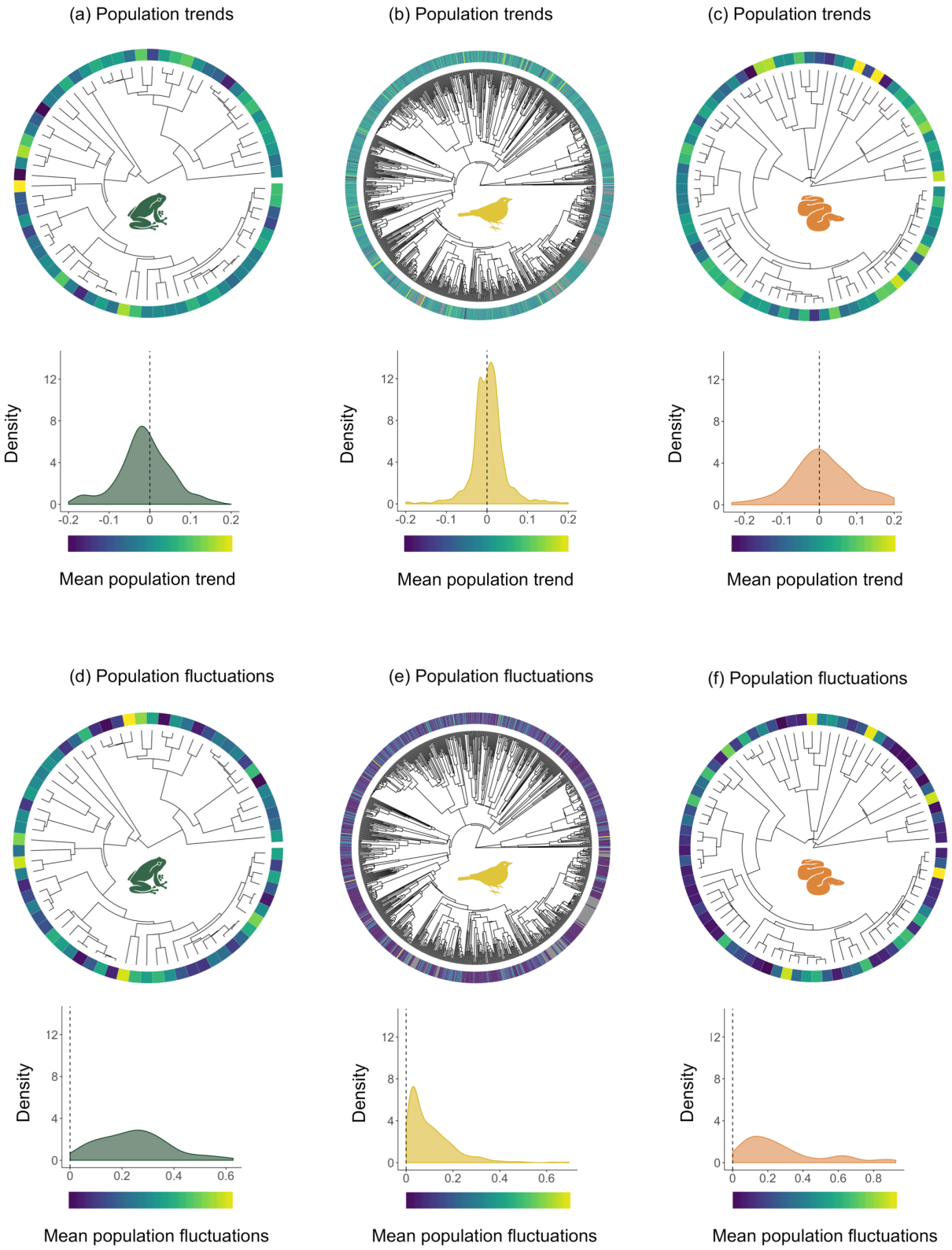



**Figure 6. No distinct patterning in population trends (average *mu* across the population trends of each species, plots a-c) and in population fluctuations**

**(plots d-f) was found across 69 amphibian species, 806 bird species and 79 reptile species.** Heat maps around phylogenetic trees indicate population change with darker blue indicating species with declining populations, green indicating species with populations with no change and yellow indicating species with increasing populations across monitored species in the Living Planet Database. Density plots show mean population change across the sampled species in each class. The phylogenies were obtained from Jetz & Pyron (2018) for ampibians (https://vertlife.org), from Jetz *et al.* (2012) for birds (https://birdtree.org), and from Tonini *et al.* (2016) for reptiles (https://vertlife.org). For model outputs, see Figure S8 and Table S2.

*Global population trends across monitoring duration*

Monitoring duration was weakly positively related to vertebrate population trends, with slightly more increasing populations found for longer duration studies (Figure S9, Table S2).

## Discussion

Taken together, our findings do not support the application of rarity metrics as generalizable predictors of population trends in well-monitored vertebrate populations. Across monitored vertebrate populations in the UK and globally, rarity metrics (geographic range, population size and habitat specificity) and species' Red List status did not explain variation in population trends (Figures 1 and 2). There was no consistent pattern of greater population declines with increasing Red List status in either the UK or globally (Figures 1e, f and 2). There was no biogeographic patterning of population trends and fluctuations, as populations increased, remained

506 stable and declined across different biomes on all continents (Figure 3). Among
507 higher-order taxonomic units (class), birds, mammals and reptiles increased, bony
508 fish did not experience net directional change and amphibians were the only taxon
509 with overall declines, as well as the taxon that experienced the most population
510 fluctuations (Figures 4-5). On a finer taxonomic scale, across amphibian, bird and
511 reptile species, there were no distinct phylogenetic clusters of closely-related species
512 experiencing similar population trends or fluctuations. Fluctuations were more likely
513 for species with smaller population sizes and broader habitat preferences (Figure 5).
514 Thus, although we did not find evidence that rarity metrics explain variation in
515 population trends, rare species were nevertheless more likely to fluctuate, potentially
516 increasing their extinction risk.
517
518 *Population change across rarity metrics*
519 The power of rarity metrics to explain population trends could be mediated by the
520 type of threats to which populations are exposed (Schemske *et al.* 1994), as well as
521 whether species are naturally rare, or have become rare due to external drivers in
522 recent time (de Lange & Norton 2004; Robbirt *et al.* 2006; Harrison *et al.* 2008).
523 Geographic range, population size and habitat specificity may be of lesser
524 importance in human-dominated landscapes where threats such as habitat change,
525 pollution and exploitation are more prevalent (Newmark 1996; Purvis *et al.* 2000;
526 O'Grady *et al.* 2004; Brook *et al.* 2006, Sullivan *et al.* 2016 but see Owens & Bennett
527 2000 and Julliard *et al.* 2004). For UK and global vertebrate populations, the lack of
528 rarity effects on population trends (Figure 1) could be related to urbanization,
529 intensive land management and/or high investment in conservation (Keith *et al.*
530 2015), which may be contributing to the observed increases in species classified as

near threatened by the IUCN (Figure 1f). Species with larger geographic ranges and small populations were most likely to fluctuate (Figure 5). A higher likelihood of fluctuations for populations with smaller mean population size (Figure 5c) is in line with population dynamics theory, suggesting that even though rarity metrics did not predict population trends, they might still be good indicators of extinction risk, as greater fluctuations could in time lead to extinction (Lande 1993; Melbourne & Hastings 2008).

*Population change across Red List status*

Across all Red List status categories, there was no consistent pattern of net declines in the UK and globally with increasing threat level (Figure 2). Populations from vulnerable and endangered species did not experience directional change, suggesting that local scale conservation actions might have stabilized populations for species which are declining on a global scale (Saterson *et al.* 2004; Vickery *et al.* 2004; Gillings *et al.* 2005; Hoffmann *et al.* 2010). The threats and mitigation actions to which populations with more threatened Red List statuses are exposed on a local scale are likely heterogeneous across different biomes (Pereira *et al.* 2012; Maxwell *et al.* 2016). The finding that populations from species with higher Red List status were more likely to fluctuate (Figure 5d) relates to one of the Red List criteria – number of breeding individuals (Mace *et al.* 2008). Extinction risk is higher when there are very few breeding individuals due to Allee effects (Dennis *et al.* 2016; Sun 2016). Thus, although we did not find evidence to support the hypothesis that species with a higher Red List status are more likely to be declining, those populations did experience higher population fluctuations, suggesting that species of concern might still have a higher risk of extinction.

*Global population change across latitude, biomes and systems*

Global vertebrate population trends varied among taxa but did not show distinct geographic patterning (Figures 3-4). Some biomes like Mediterranean forests were more likely to include increasing populations, whilst populations in other biomes did not experience directional population change (Figure 3 and Table S2). Similar heterogeneity in the patterns of population trends across habitats has been documented in the Netherlands (van Strien *et al.* 2016) and in temperate North America and Europe (Leung *et al.* 2017) also using the LPD, suggesting that local scale environmental factors or proximity to anthropogenic drivers of change may be better predictors of population trends (Brook *et al.* 2006; Fritz *et al.* 2009, but see Phillips *et al.* 2017). The lack of distinct geographic patterning of population trends and fluctuations around the world (Figures 3, 4 and 6a, b and e) might be attributable to either stochastic (random events) and deterministic (e.g., anthropogenic threats) processes influencing inter-annual population change (Kareiva 1990; Lundberg *et al.* 2000). Thus, to attribute population trends and fluctuations to anthropogenic drivers, we need to go beyond previous studies (Ceballos *et al.* 2017; Davidson *et al.* 2017) and quantify the full spectrum of population change including positive, negative and stable trends and fluctuations. Quantifying the drivers of population change is a critical first step towards disentangling the pathways from population change to biodiversity change in the Anthropocene (Leung *et al.* 2017; Yoccoz *et al.* 2018).

*Global population change across taxa*

Taxon-specific factors including reproductive strategy, trophic level, generation time, life history traits (Purvis *et al.* 2000; Cardillo *et al.* 2004), or sampling effects

581 (Magurran *et al.* 2010) could explain why some taxa were more likely to increase
582 (birds, mammals and reptiles), whereas others were more likely to decline
583 (amphibians) or remain stable (bony fish), (Figure 4 and Table S2). However, within
584 the classes *Amphibia*, *Aves* and *Reptilia*, phylogenetic relatedness and species-level
585 taxanomic classification did not predict variation in population trends and
586 fluctuations. Similar lack of phylogenetic dependencies has been detected for the
587 population growth rates of migratory birds (Moller *et al.* 2008). Among seabirds,
588 however, breeding phenology varied based on phylogeny and taxonomy (Keogan *et*
589 *al.* 2018). Among squamates and amphibians, there is evidence that phylogenetic
590 relatedness predicts extinction (Tonini *et al.* 2016; Jetz & Pyron 2018), a process
591 that occurs over much longer time-scales than the population change we study here.
592 Amphibians were the only taxa to experience net population declines as well as
593 greater population fluctuations among monitored vertebrate species in the LPD
594 (Figure 4f and 5f). The global amphibian decline (Houlahan *et al.* 2000), also
595 demonstrated with this dataset at the temperate North American and European scale
596 (Leung *et al.* 2017), has been linked to the spread of a fungal disease (chytrid
597 fungus, *Batrachochytrium dendrobatidis*), facilitated by global warming (Pounds *et al.*
598 2006), as well as habitat loss and Allee effects in small populations (Green 2003).
599

600 *Study limitations*

601 Our study is limited by three main factors: 1) the gaps in global monitoring of
602 populations, 2) challenges in estimating geographic range and habitat specificity,
603 and 3) overrepresentation of common species. The LPD under-represents
604 populations outside of Europe and North America (Pereira & Cooper 2006; Proença
605 *et al.* 2016; McRae *et al.* 2017) and includes different species abundance measures

606 across populations that have been monitored for different durations (Figures S1, S2
607 and S6). The GBIF database includes sub-optimal representation of some species
608 and there might be confusion between sub-species and occurrence outliers (Yesson
609 *et al.* 2007; Boakes *et al.* 2010), despite being the largest international open source
610 database of species locality data (Yesson *et al.* 2007). In addition, we used a static
611 measure of geographic range, which does not account for changes in species
612 distributions over time. For example, species could naturally have a small range or
613 the small range size could be due to historic habitat loss (Magurran & Henderson
614 2003; Prendergast *et al.* 1993). In addition, UK populations included in the LPD are
615 predominantly from species with wide geographic ranges (Table S4). The
616 relationship between population change and geographic range could differ for highly
617 endemic species with restricted distribution that are not monitored or included in the
618 LPD dataset.
619
620 *Conclusion*
621 In summary, rarity metrics did not explain vertebrate population trends and
622 we did not find a consistent pattern of net declines with increasing Red List status in
623 the UK and globally. We found higher-order taxonomic but not geographic patterning
624 of population change around the world, yet within-classes, phylogeny and species
625 identity did not explain variation in population trends and fluctuations among
626 amphibian, bird and reptile species. Vertebrate populations experienced different
627 magnitudes of net change and population fluctuations, which in the future may lead
628 to shifts in community composition as some species become locally extinct and
629 others expand into new habitats. Our findings caution the use of rarity metrics for the
630 prediction of future global population trends, but suggest that such metrics, in

631 particular population size, are nevertheless indicators of population fluctuations,
632 which might ultimately link to increased species extinction risk. It is only with
633 improved monitoring of diverse taxa that we can test the drivers of global population
634 change during the Anthropocene to inform wildlife conservation.
635

636 **Statement of authorship**

637 GND and IMS conceived the idea and conducted the statistical analyses. JLG
638 contributed to the calculation of geographic range estimates. All authors contributed
639 to the integration of the LPI, GBIF and IUCN databases. GND wrote the first draft of
640 the manuscript and all authors contributed to revisions.
641

642 **Conflict of interest**

643 The authors declare no conflicts of interest.
644

645 **Data accessibility statement**

646 We will make all R code for the analyses and data publicly available on GitHub and
647 will include the relevant DOIs and links.
648

649 **Acknowledgements**

650 We thank the WWF and the RZSL for compiling and maintaining the Living Planet
651 Database. We thank Dmitry Schigel and Jan Legind at GBIF for providing access to
652 a direct download link for the UK species occurrence data. We are grateful to the
653 Critical Thinking in Ecology and Environmental Sciences 2015-16 tutorial group at
654 the University of Edinburgh including Hannah Louise Stevens, Ashlie Nithsdale, Lisa
655 Kopsieker and Claudia Ardrey who helped us conceive the idea for the analysis. We

656 thank Dan Greenberg for advice on state-space models and Kat Keogan and Albert
657 Phillimore for advice on phylogenetic models. We thank Anne Magurran, Albert
658 Phillimore and Mark Vellend for providing very helpful feedback on the manuscript.
659 Funding for this research was provided by the NERC doctoral training partnership
660 grant (NE/L002558/1) at the University of Edinburgh and the Carnegie Trust for the
661 Universities of Scotland.
662

663 **References**

664 Agardy, T. (2005). Global marine conservation policy versus site-level
665 implementation. *Mar. Ecol. Prog. Ser.*, 300, 242–248.
666 Batt, R.D., Morley, J.W., Selden, R.L., Tingley, M.W. & Pinsky, M.L. (2017). Gradual
667 changes in range size accompany long-term trends in species richness. *Ecol.*
668 *Lett.*, 20, 1148–1157.
669 BirdLife International. (2018). IUCN Red List for birds. Downloaded from
670 http://www.birdlife.org in March 2018.
671 Boakes, E.H., McGowan, P.J., Fuller, R.A., Chang-qing, D., Clark, N.E., O'Connor,
672 K., *et al.* (2010). Distorted views of biodiversity: spatial and temporal bias in
673 species occurrence data. *PLoS Biol.*, 8, e1000385.
674 Brook, B.W., Traill, L.W. & Bradshaw, C.J.A. (2006). Minimum viable population
675 sizes and global extinction risk are unrelated: MVP is unrelated to global
676 extinction risk. *Ecol. Lett.*, 9, 375–382.
677 Cardillo, M., Purvis, A., Sechrest, W., Gittleman, J.L., Bielby, J. & Mace, G.M.
678 (2004). Human Population Density and Extinction Risk in the World's Carnivores.
679 *PLoS Biol.*, 2, e197.

Ceballos, G., Ehrlich, P.R. & Dirzo, R. (2017). Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. *Proc. Natl. Acad. Sci.*, 201704949.

Chamberlain, S. (2017). rredlist: “IUCN” Red List Client. R package version 0.4.0. https://CRAN.R-project.org/package=rredlist.

Collen, B., Loh, J., Whitmee, S., McRAE, L., Amin, R. & Baillie, J.E.M. (2009). Monitoring Change in Vertebrate Abundance: the Living Planet Index. *Conserv. Biol.*, 23, 317–327.

Davidson, A.D., Shoemaker, K.T., Weinstein, B., Costa, G.C., Brooks, T.M., Ceballos, G., *et al.* (2017). Geography of current and future global mammal extinction risk. *PLOS ONE*, 12, e0186934.

Dennis, B., Assas, L., Elaydi, S., Kwessi, E. & Livadiotis, G. (2016). Allee effects and resilience in stochastic populations. *Theor. Ecol.*, 9, 323–335.

Dennis, B., Ponciano, J.M., Lele, S.R., Taper, M.L. & Staples, D.F. (2006). Estimating density dependence, process noise, and observation error. *Ecol. Monogr.*, 76, 323–341.

Ehrlén, J. & Morris, W.F. (2015). Predicting changes in the distribution and abundance of species under environmental change. *Ecol. Lett.*, 18, 303–314.

Foley, N.M., Springer, M.S. & Teeling, E.C. (2016). Mammal madness: is the mammal tree of life not yet resolved? *Philos. Trans. R. Soc. B Biol. Sci.*, 371, 20150140.

Fritz, S.A., Bininda-Emonds, O.R.P. & Purvis, A. (2009). Geographical variation in predictors of mammalian extinction risk: big is bad, but only in the tropics. *Ecol. Lett.*, 12, 538–549.

Gaston, K. & Fuller, R. (2008). Commonness, population depletion and conservation biology. *Trends Ecol. Evol.*, 23, 14–19.

Gaston, K.J. & Blackburn, T.M. (1995). Birds, body size and the threat of extinction. *Philos. Trans. R. Soc. Lond. B Biol. Sci.*, 347, 205–212.

GBIF. (2017). Biodiversity occurrence data. http://www.gbif.org/ Downloaded Feb 2016 - Apr 2017.

GBIF. (2018). Species occurrence data.

Gillings, S., Newson, S.E., Noble, D.G. & Vickery, J.A. (2005). Winter availability of cereal stubbles attracts declining farmland birds and positively influences breeding population trends. *Proc. R. Soc. B Biol. Sci.*, 272, 733–739.

Gilroy, J.J., Gill, J.A., Butchart, S.H.M., Jones, V.R. & Franco, A.M.A. (2016). Migratory diversity predicts population declines in birds. *Ecol. Lett.*, 19, 308–317.

Green, D.M. (2003). The ecology of extinction: population fluctuation and decline in amphibians. *Biol. Conserv.*, 111, 331–343.

Guerrero, A.M., McALLISTER, R.R.J., Corcoran, J. & Wilson, K.A. (2013). Scale Mismatches, Conservation Planning, and the Value of Social-Network Analyses: Scale Mismatches and Conservation Planning. *Conserv. Biol.*, 27, 35–44.

Gunn, A. (1980). Why should we care about rare species? *Environ. Ethics*, 2, 17–37.

Hadfield, J.D. (2010). MCMC Methods for Multi-Response Generalized Linear Mixed Models: The MCMCglmm R Package. *J. Stat. Softw.*, 33, 1–22.

Hanski, I. (1998). Metapopulation dynamics. *Nature*, 396, 41.

Harrison, S., Viers, J.H., Thorne, J.H. & Grace, J.B. (2008). Favorable environments and the persistence of naturally rare species. *Conserv. Lett.*, 1, 65–74.

Hefley, T.J., Hooten, M.B., Drake, J.M., Russell, R.E. & Walsh, D.P. (2016). When can the cause of a population decline be determined? *Ecol. Lett.*, 19, 1353–1362.

Hijmans, R.J. (2016). geosphere: Spherical Trigonometry. R package version 1.5-5. https://CRAN.R-project.org/package=geosphere.

Hoffmann, M., Hilton-Taylor, C., Angulo, A., Bohm, M., Brooks, T.M., Butchart, S.H.M., *et al.* (2010). The Impact of Conservation on the Status of the World's Vertebrates. *Science*, 330, 1503–1509.

Houlahan, J.E., Findlay, C.S., Schmidt, B.R., Meyer, A.H. & Kuzmin, S.L. (2000). Quantitative evidence for global amphibian population declines. *Nature*, 404, 752–755.

Humbert, J.-Y., Scott Mills, L., Horne, J.S. & Dennis, B. (2009). A better way to estimate population trends. *Oikos*, 118, 1940–1946.

Hutchings, J.A. & Reynolds, J.D. (2004). Marine fish population collapses: consequences for recovery and extinction risk. *AIBS Bull.*, 54, 297–309.

IUCN. (2017). Conservation status data. http://www.iucnredlist.org/ Downloaded Jan-April 2017.

Jetz, W. & Pyron, R.A. (2018). The interplay of past diversification and evolutionary isolation with present imperilment across the amphibian tree of life. *Nat. Ecol. Evol.*

Jetz, W., Thomas, G.H., Joy, J.B., Hartmann, K. & Mooers, A.O. (2012). The global diversity of birds in space and time. *Nature*, 491, 444–448.

Jones, K.E., Purvis, A. & Gittleman, J.L. (2003). Biological Correlates of Extinction Risk in Bats. *Am. Nat.*, 161, 601–614.

Julliard, R., Jiguet, F. & Couvet, D. (2004). Common birds facing global changes: what makes a species at risk? *Glob. Change Biol.*, 10, 148–154.

Kareiva, P. (1990). Population dynamics in spatially complex environments: theory and data. *Phil Trans R Soc Lond B*, 330, 175–190.

Keith, D., Akçakaya, H.R., Butchart, S.H.M., Collen, B., Dulvy, N.K., Holmes, E.E., *et al.* (2015). Temporal correlations in population trends: Conservation implications from time-series analysis of diverse animal taxa. *Biol. Conserv.*, 192, 247–257.

Keogan, K., Daunt, F., Wanless, S., Phillips, R.A., Walling, C.A., Agnew, P., *et al.* (2018). Global phenological insensitivity to shifting ocean temperatures among seabirds. *Nat. Clim. Change*, 8, 313–318.

Knape, J., Jonzén, N. & Sköld, M. (2011). On observation distributions for state space models of population survey data: Observation models for population data. *J. Anim. Ecol.*, 80, 1269–1277.

Lande, R. (1993). Risks of population extinction from demographic and environmental stochasticity and random catastrophes. *Am. Nat.*, 142, 911–927.

de Lange, P.J. & Norton, D.A. (2004). The ecology and conservation of Kunzea sinclairii (Myrtaceae), a naturally rare plant of rhyolitic rock outcrops. *Biol. Conserv.*, 117, 49–59.

Lawson, C.R., Vindenes, Y., Bailey, L. & van de Pol, M. (2015). Environmental variation and population responses to global change. *Ecol. Lett.*, 18, 724–736.

Leung, B., Greenberg, D.A. & Green, D.M. (2017). Trends in mean growth and stability in temperate vertebrate populations. *Divers. Distrib.*, 23, 1372–1380.

Longton, R.E. & Hedderson, T.A. (2000). What are rare species and why conserve them? *Lindbergia*, 25, 53–61.

LPI. (2016). Living Planet Index database, 2016. www.livingplanetindex.org/ Downloaded February 2016.

Lundberg, P., Ranta, E., Ripa, J. & Kaitala, V. (2000). Population variability in space and time. *Trends Ecol. Evol.*, 15, 460–464.

Mace, G.M., Collar, N.J., Gaston, K.J., Hilton-Taylor, C., AkçAkaya, H.R., Leader-Williams, N., *et al.* (2008). Quantification of Extinction Risk: IUCN's System for Classifying Threatened Species. *Conserv. Biol.*, 22, 1424–1442.

Magurran, A. (2013). *Measuring biological diversity*. John Wiley & Sons.

Magurran, A.E., Baillie, S.R., Buckland, S.T., Dick, J.M., Elston, D.A., Scott, E.M., *et al.* (2010). Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends Ecol. Evol.*, 25, 574–582.

Magurran, A.E. & Henderson, P.A. (2003). Explaining the excess of rare species in natural species abundance distributions. *Nature*, 422, 714.

Manne, L.L. & Pimm, S.L. (2001). Beyond eight forms of rarity: which species are threatened and which will be next? *Anim. Conserv.*, 4, 221–229.

Maxwell, S.L., Fuller, R.A., Brooks, T.M. & Watson, J.E. (2016). Biodiversity: The ravages of guns, nets and bulldozers. *Nature*, 536, 143–145.

McKinney, M.L. (1997). Extinction vulnerability and selectivity: combining ecological and paleontological views. *Annu. Rev. Ecol. Syst.*, 28, 495–516.

McRae, L., Deinet, S. & Freeman, R. (2017). The Diversity-Weighted Living Planet Index: Controlling for Taxonomic Bias in a Global Biodiversity Indicator. *PLOS ONE*, 12, e0169156.

Melbourne, B.A. & Hastings, A. (2008). Extinction risk depends strongly on factors contributing to stochasticity. *Nature*, 454, 100–103.

Mills, M., Pressey, R.L., Weeks, R., Foale, S. & Ban, N.C. (2010). A mismatch of scales: challenges in planning for implementation of marine protected areas in the Coral Triangle: Mismatch of scales in conservation planning. *Conserv. Lett.*, 3, 291–303.

Moller, A.P., Rubolini, D. & Lehikoinen, E. (2008). Populations of migratory bird species that did not show a phenological response to climate change are declining. *Proc. Natl. Acad. Sci.*, 105, 16195–16200.

Newmark, W.D. (1996). Insularization of Tanzanian parks and the local extinction of large mammals. *Conserv. Biol.*, 10, 1549–1556.

O'Grady, J.J., Reed, D.H., Brook, B.W. & Frankham, R. (2004). What are the best correlates of predicted extinction risk? *Biol. Conserv.*, 118, 513–520.

Olden, J.D., Hogan, Z.S. & Zanden, M.J.V. (2007). Small fish, big fish, red fish, blue fish: size-biased extinction risk of the world's freshwater and marine fishes. *Glob. Ecol. Biogeogr.*, 16, 694–701.

Olson, D.M. & Dinerstein, E. (2002). The Global 200: Priority Ecoregions for Global Conservation. *Ann. Mo. Bot. Gard.*, 89, 199.

Owens, I.P. & Bennett, P.M. (2000). Ecological basis of extinction risk in birds: habitat loss versus human persecution and introduced predators. *Proc. Natl. Acad. Sci.*, 97, 12144–12148.

Pedersen, M.W., Berg, C.W., Thygesen, U.H., Nielsen, A. & Madsen, H. (2011). Estimation methods for nonlinear state-space models in ecology. *Ecol. Model.*, 222, 1394–1400.

Pereira, H.M. & Cooper, D.H. (2006). Towards the global monitoring of biodiversity change. *Trends Ecol. Evol.*, 21, 123–129.

Pereira, H.M., Navarro, L.M. & Martins, I.S. (2012). Global Biodiversity Change: The Bad, the Good, and the Unknown. *Annu. Rev. Environ. Resour.*, 37, 25–50.

Phillips, H.R.P., Newbold, T. & Purvis, A. (2017a). Land-use effects on local biodiversity in tropical forests vary between continents. *Biodivers. Conserv.*, 26, 2251–2270.

Phillips, H.R.P., Newbold, T. & Purvis, A. (2017b). Land-use effects on local biodiversity in tropical forests vary between continents. *Biodivers. Conserv.*, 26, 2251–2270.

Pigott, C.D. & Walters, S.M. (1977). The scientific basis of practical conservation: aims and methods of conservation. In: *Proc. R. Soc. Lond. B*. The Royal Society, pp. 59–68.

Pounds, A.J., Bustamante, M.R., Coloma, L.A., Consuegra, J.A., Fogden, M.P.L., Foster, P.N., *et al.* (2006). Widespread amphibian extinctions from epidemic disease driven by global warming. *Nature*, 439, 161–167.

Povilitis, T. (2001). Toward a robust natural imperative for conservation. *Conserv. Biol.*, 15, 533–535.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature*, 365, 335–337.

Pressey, R.L., Cabeza, M., Watts, M.E., Cowling, R.M. & Wilson, K.A. (2007). Conservation planning in a changing world. *Trends Ecol. Evol.*, 22, 583–592.

Proença, V., Martin, L.J., Pereira, H.M., Fernandez, M., McRae, L., Belnap, J., *et al.* (2016). Global biodiversity monitoring: From data sources to Essential Biodiversity Variables. *Biol. Conserv.*

Purvis, A., Gittleman, J.L., Cowlishaw, G. & Mace, G.M. (2000). Predicting extinction risk in declining species. *Proc. R. Soc. B Biol. Sci.*, 267, 1947–1952.

R Core Team. (2017). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rabinowitz, D. (1981). Seven forms of rarity. In: *The biological aspects of rare plants conservation*. pp. 205–217.

Ripple, W.J., Wolf, C., Newsome, T. M., Hoffmann, M., Wirsing, A. J. & McCauley, D. J. (2017). Extinction risk is most acute for the world's largest and smallest vertebrates. *114*, 10678–10683.

Robbirt, K.M., Roberts, D.L. & Hawkins, J.A. (2006). Comparing IUCN and Probabilistic Assessments of Threat: Do IUCN Red List Criteria Conflate Rarity and Threat? *Biodivers. Conserv.*, 15, 1903–1912.

Rueda-Cediel, P., Anderson, K.E., Regan, T.J. & Regan, H.M. (2018). Effects of uncertainty and variability on population declines and IUCN Red List classifications. *Conserv. Biol.*

Saterson, K.A., Christensen, N.L., Jackson, R.B., Kramer, R.A., Pimm, S.L., Smith, M.D., *et al.* (2004). Disconnects in Evaluating the Relative Effectiveness of Conservation Strategies. *Conserv. Biol.*, 18, 597–599.

Schemske, D.W., Husband, B.C., Ruckelshaus, M.H., Goodwillie, C., Parker, I.M. & Bishop, J.G. (1994). Evaluating Approaches to the Conservation of Rare and Endangered Plants. *Ecology*, 75, 584–606.

Shultz, S., B. Bradbury, R., L. Evans, K., D. Gregory, R. & M. Blackburn, T. (2005). Brain size and resource specialization predict long-term population trends in British birds. *Proc. R. Soc. B Biol. Sci.*, 272, 2305–2311.

van Strien, A.J., Meyling, A.W.G., Herder, J.E., Hollander, H., Kalkman, V.J., Poot, M.J.M., *et al.* (2016). Modest recovery of biodiversity in a western European country: The Living Planet Index for the Netherlands. *Biol. Conserv.*, 200, 44–50.

Sun, G.-Q. (2016). Mathematical modeling of population dynamics with Allee effect. *Nonlinear Dyn.*, 85, 1–12.

875 Tarver, J.E., dos Reis, M., Mirarab, S., Moran, R.J., Parker, S., O'Reilly, J.E., *et al.*
876 (2016). The Interrelationships of Placental Mammals and the Limits of
877 Phylogenetic Inference. *Genome Biol. Evol.*, 8, 330–344.
878 Tonini, J.F.R., Beard, K.H., Ferreira, R.B., Jetz, W. & Pyron, R.A. (2016). Fully-
879 sampled phylogenies of squamates reveal evolutionary patterns in threat status.
880 *Biol. Conserv.*, 204, 23–31.
881 Töpel, M., Zizka, A., Calió, M.F., Scharn, R., Silvestro, D. & Antonelli, A. (2016).
882 SpeciesGeoCoder: Fast Categorization of Species Occurrences for Analyses of
883 Biodiversity, Biogeography, Ecology, and Evolution. *Syst. Biol.*, syw064.
884 Vickery, J.A., Bradbury, R.B., Henderson, I.G., Eaton, M.A. & Grice, P.V. (2004). The
885 role of agri-environment schemes and farm management practices in reversing
886 the decline of farmland birds in England. *Biol. Conserv.*, 119, 19–39.
887 Yesson, C., Brewer, P.W., Sutton, T., Caithness, N., Pahwa, J.S., Burgess, M., *et al.*
888 (2007). How Global Is the Global Biodiversity Information Facility? *PLoS ONE*, 2,
889 e1124.
890 Yoccoz, N.G., Ellingsen, K.E. & Tveraa, T. (2018). Biodiversity may wax or wane
891 depending on metrics or taxa. *Proc. Natl. Acad. Sci.*, 115, 1681–1683.
892

893 **Supplementary Information**

894

895 **Table S1. Number of species and populations included in analyses.**

| Scale | Analysis | Number of species | Number of populations | Number of species with >3 populations |
|---|---|---|---|---|
| **UK** | Geographic range | 167 | 381 | 32 |
| | Population size | 112 | 253 | 19 |
| | Habitat specificity | 144 | 289 | 29 |
| | Red List status | 153 | 364 | 31 |
| **Global** | System | 2074 | 9284 | 834 |
| | Biome | 2074 | 9284 | 834 |
| | Taxa | | | |
| | - *Actinopterygii* | 544 | 1626 | 151 |
| | - *Amphibia* | 78 | 193 | 21 |
| | - *Aves* | 968 | 5852 | 538 |
| | - *Mammalia* | 306 | 1158 | 94 |
| | - *Reptilia* | 110 | 322 | 16 |
| | Red List status | 1702 | 8064 | 708 |
| | Latitude | 2074 | 9284 | 834 |
| | Duration | 2074 | 9284 | 834 |

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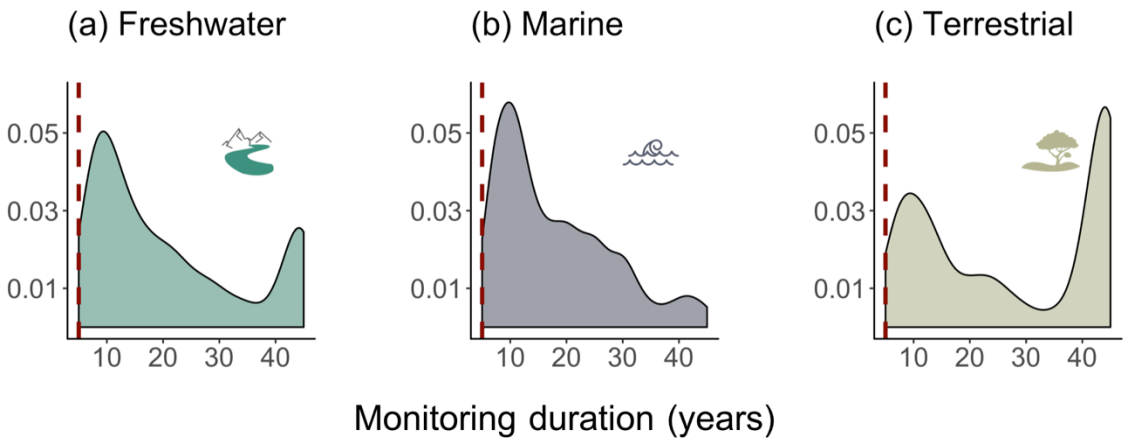


**Figure S1. Distribution of monitoring duration across (a) freshwater, (b) marine and (c) terrestrial systems. Results include 9284 populations from 2084 species.**

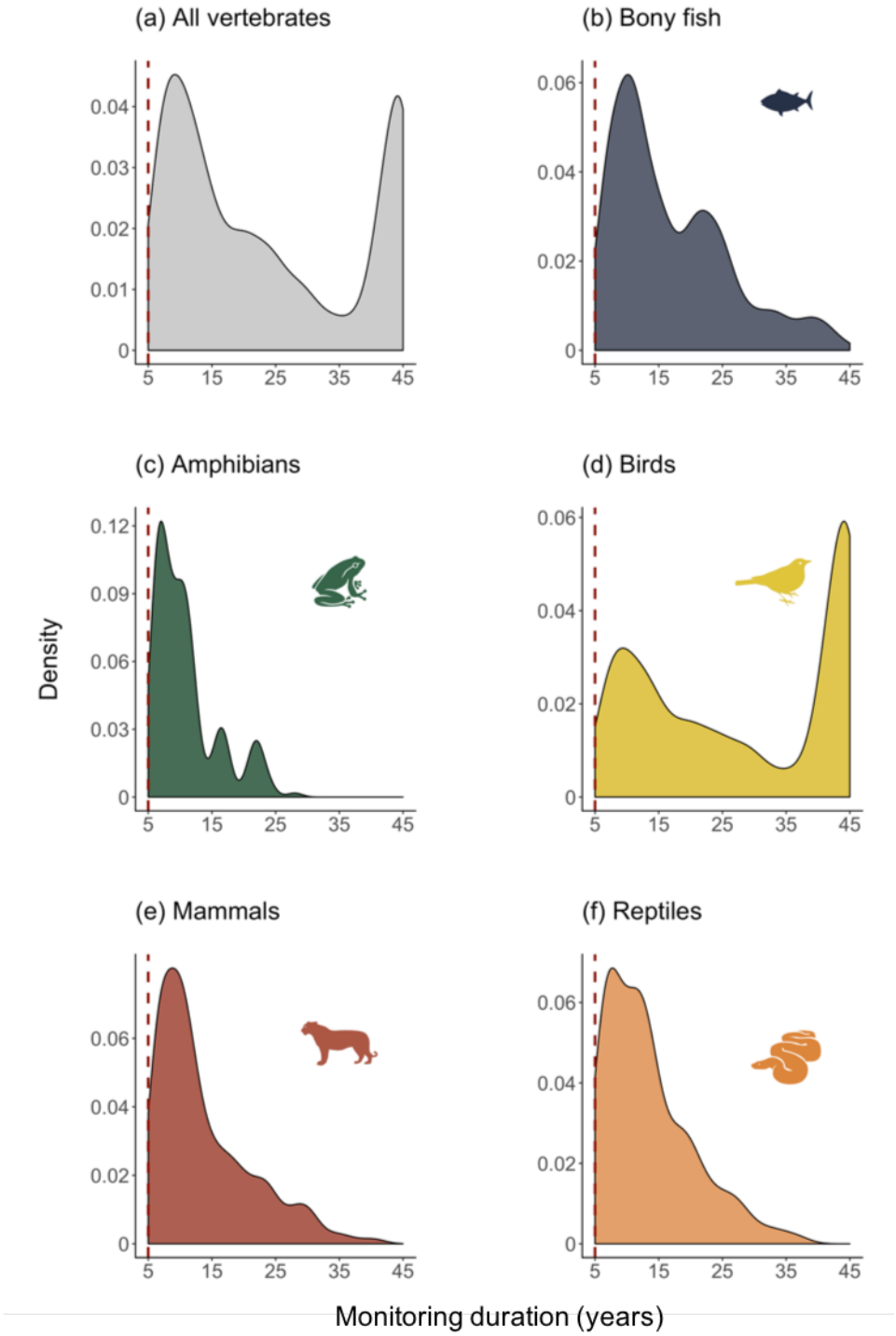


**Figure S2. Distribution of monitoring duration across all monitored vertebrate populations (plot a), bony fish (plot b), amphibians (plot c), birds (plot d), mammals (plot e) and reptiles (plot f).** See Table S1 for sample sizes of each group.

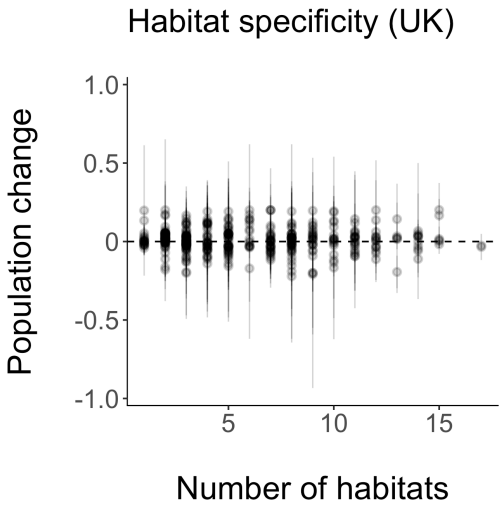


**Figure S3. Effect of habitat specificity (calculated by surveying the 'Habitat and Ecology' profile for each species on the IUCN Red List website) on vertebrate population trends in the UK.** The *mu* values of population change are derived from state-space model fits of changes in abundance over the monitoring duration for each population. Error bars show confidence intervals around population trend (mu). See Table S2 for full model outputs.

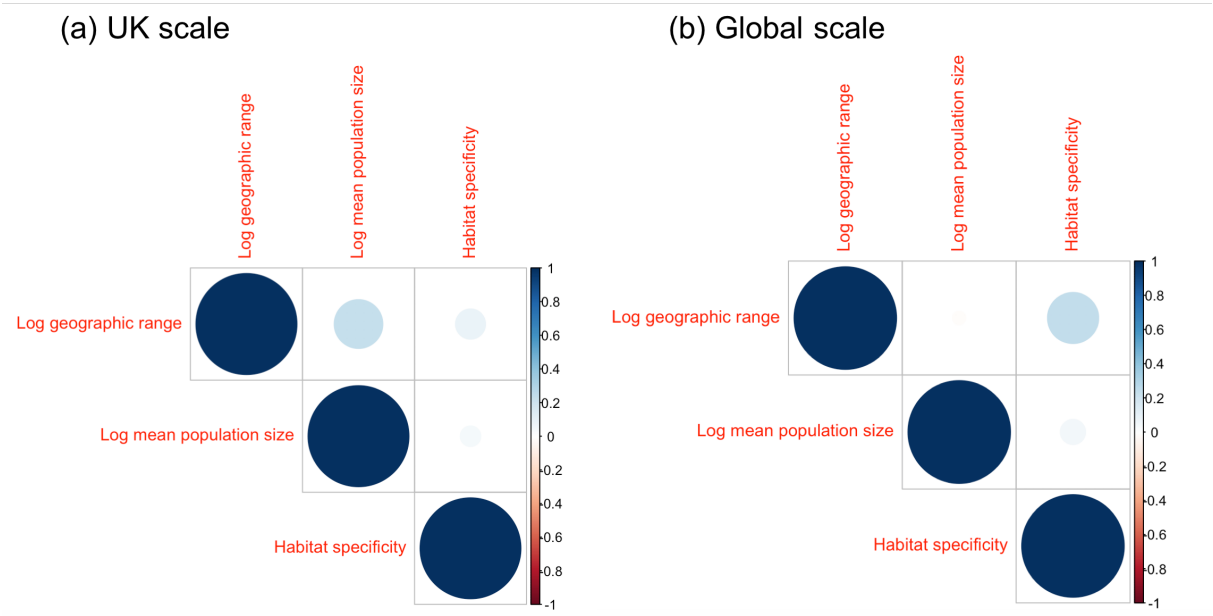


**Figure S4. Correlation matrix between rarity metrix on (a) UK and (b) global scales.** The three rarity metrics were weakly correlated on both study scales.

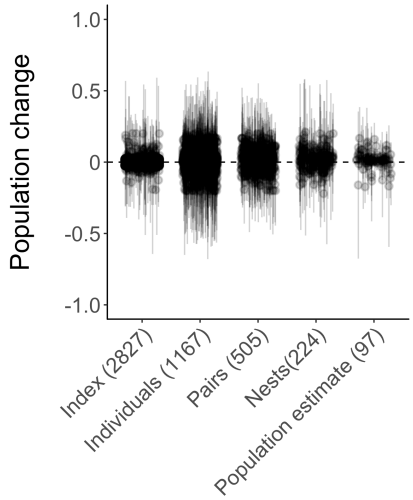


**Figure S5. Population change (*mu* values from state-space models) across the five most commonly used units in the subset of the LPD we analyzed.** Numbers on the x axis correspond to sample size in each category.

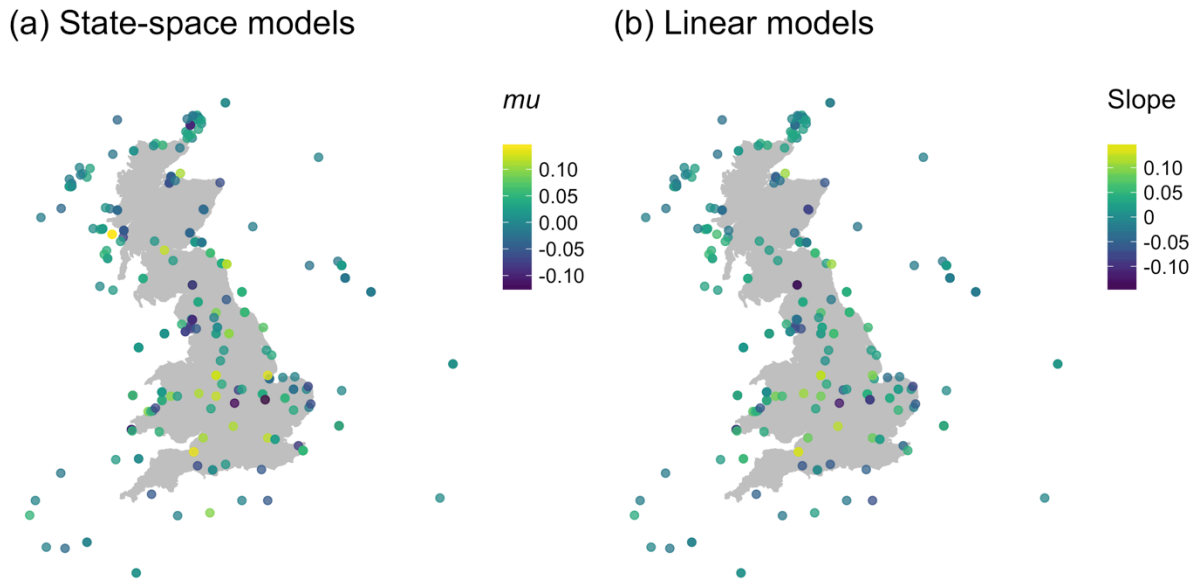


**Figure S6. Geographic distribution of vertebrate population change in the UK, from state-space models (plot a) and linear models (plot b).** Results include 508 populations from 237 species in the UK.

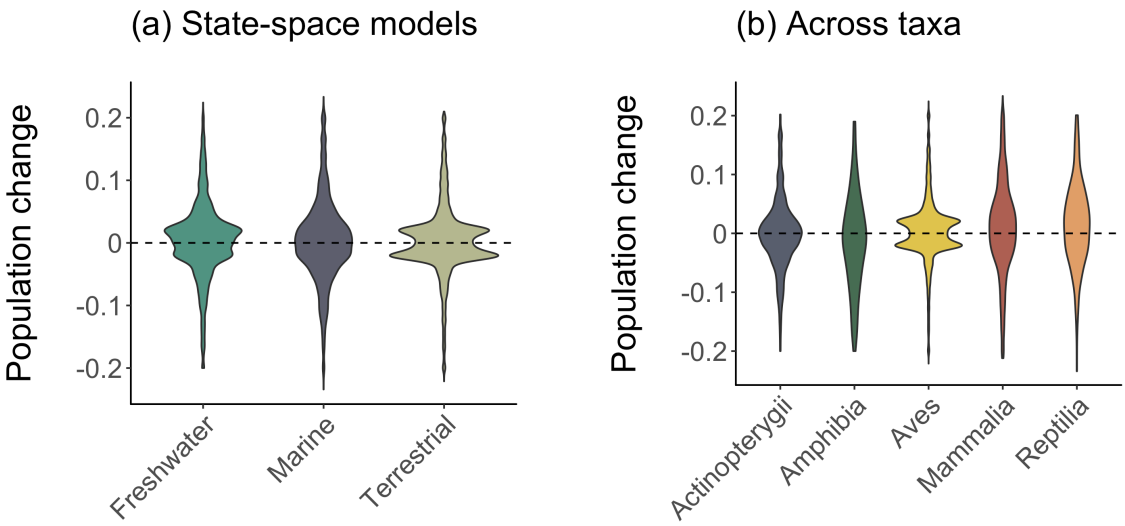


**Figure. S7. Distribution of population change (*mu*) values from state-space models of vertebrate abundance over time across systems (plot a) and across taxa (plot b).** See Table S1 for sample sizes.

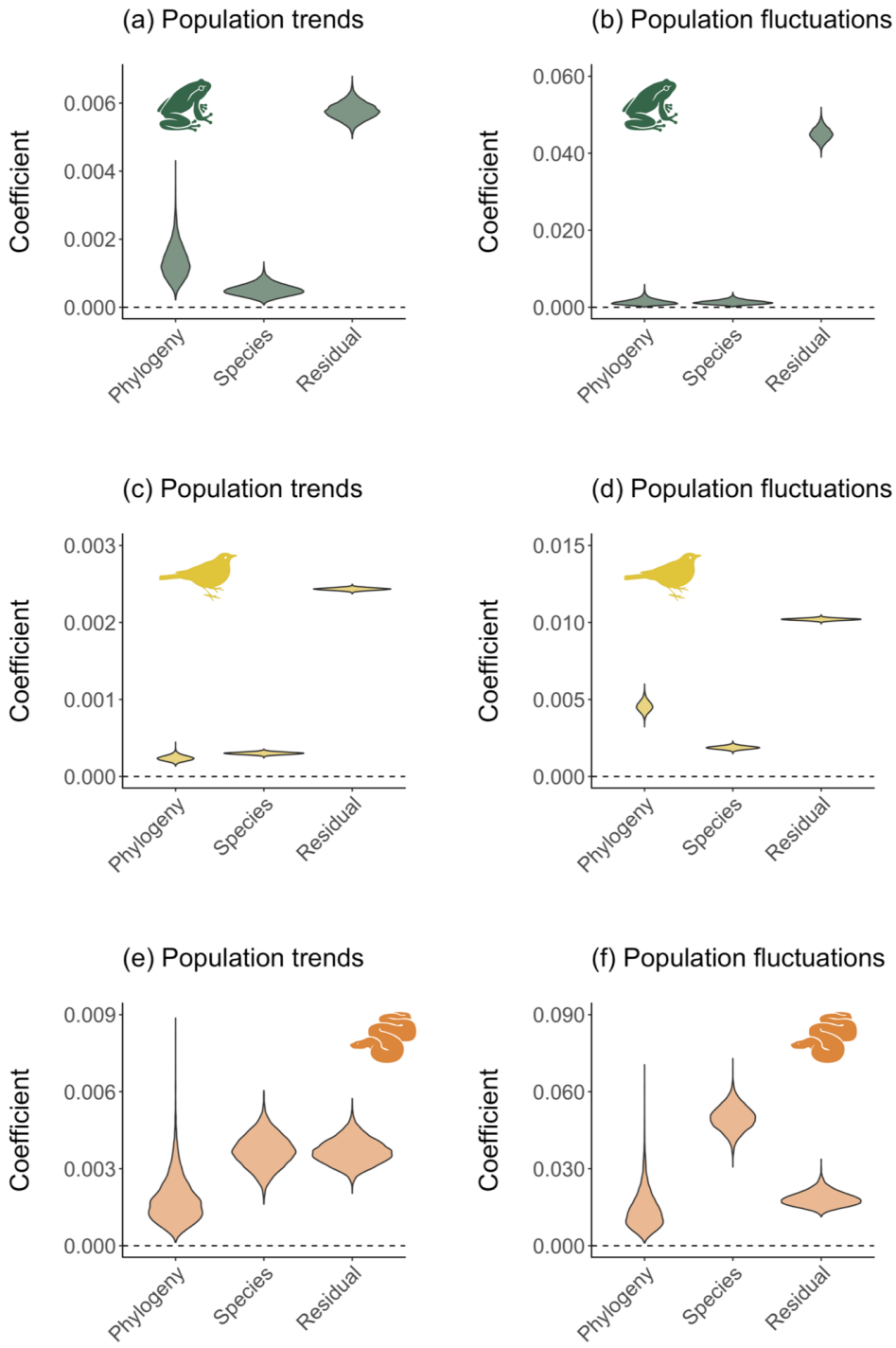


**Figure. S8. Distribution of posterior means (mean distributions across ten models for each taxa for each type of population change using ten random trees to account for phylogenetic uncertainty) for phylogenetic, species and residual variance across taxa.** Phylogeny effects were calculated based on a branch length covariance matrix. See Table S2 for full model outputs.

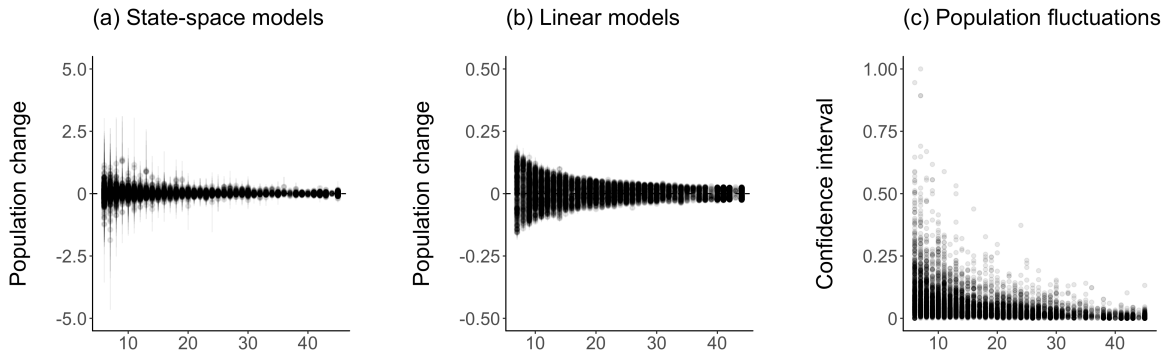


**Figure S9. Vertebrate population trends across monitoring duration.** Results are for 9284 populations from 2084 species. Population trends (*mu*) were estimated for all populations monitored for more than five time points using state-space models (plot a) and linear models (plot b). Population fluctuations (plot c) were estimated using the confidence interval (half of the 95% confidence interval) of the *mu* values (see also Figure 5). Error bars show 95% confidence intervals.

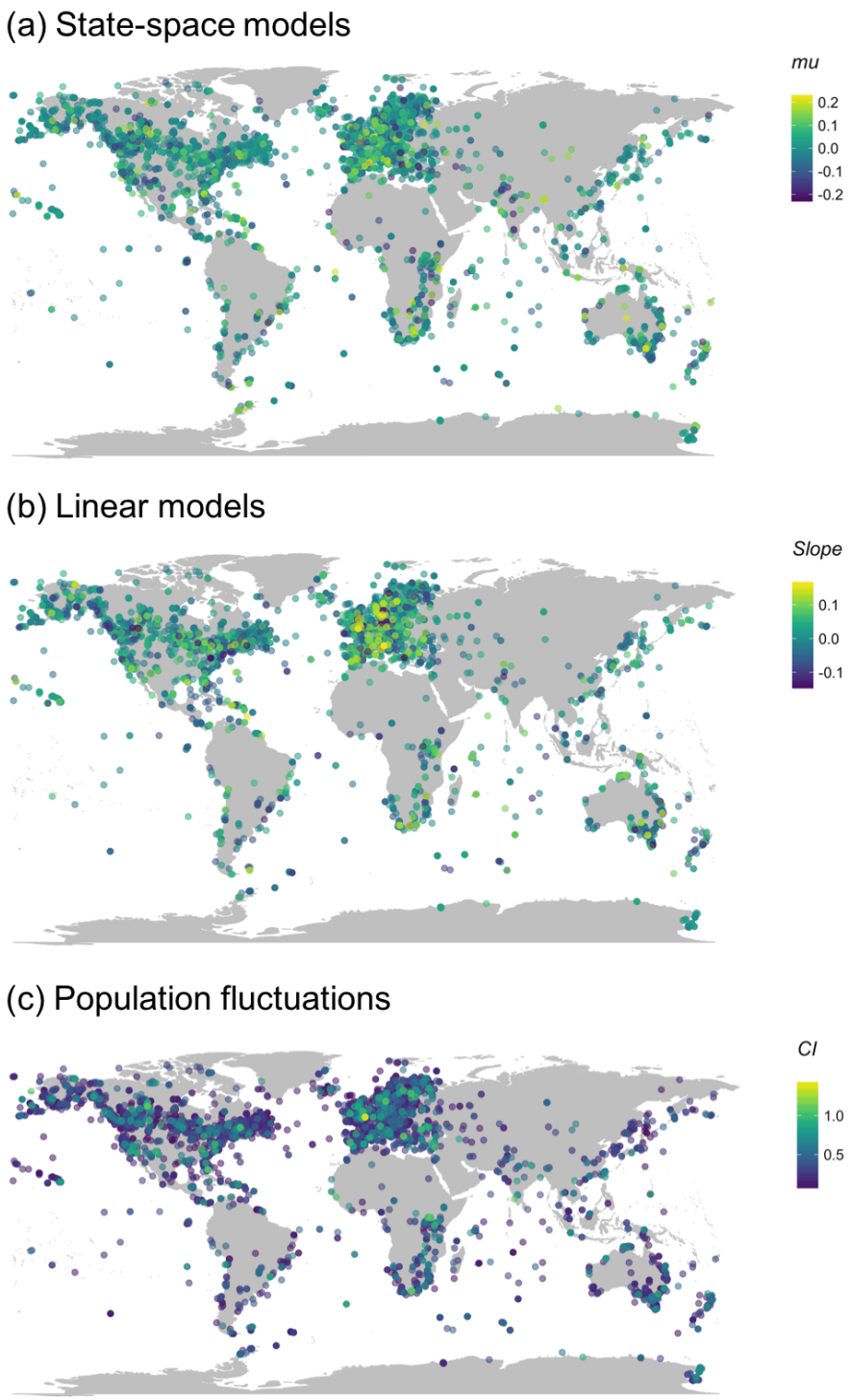


**Figure S10. Geographic distribution of population change, calculated using state-space models (plot a), linear models (plot b), and using the confidence interval (half of the 95% confidence interval) of the *mu* values (plot c).** Results include 9284 populations from 2084 species. Larger confidence intervals on (c) correspond to populations that have experienced more fluctuations.

**Table S2. Model outputs from UK and global analyses.** Sigma is the overall model residual variance. Net population change is estimated using *mu* values derived from state-space models of population abundance versus time and slopes of linear models of population abundance versus time. The weighted *mu* models included *mu* as a response variable, weighted by *tau*, the observation error estimate derived from the state-space models. The fluctuation models included half of the 95% confidence interval around the *mu* value of population change, the standard error around the slopes of linear models of population abundance versus time, and the standard deviation of the raw time-series data for each population. Larger confidence intervals correspond to populations that experienced more fluctuations. The weighted fluctuation models were weighted by *tau*, the observation error estimate derived from the state-space models. To account for phylogenetic uncertainty, we ran the phylogenetic models for amphibian, bird and reptile species using 10 different trees for each class, and here we present the mean, min and max values from the different model runs.

| Model name | Variable | Posterior mean | Lower 95% CI | Upper 95% CI | Effective sample size | pMCMC | Effect |
|---|---|---|---|---|---|---|---|
| Geographic range (UK) - mu | (Intercept) | -0.001 | -0.04 | 0.03 | 18,000 | 0.94 | fixed |
| | log(km2_range) | 0.001 | -0.001 | 0.004 | 18,000 | 0.41 | fixed |
| | sigma | 0.01 | 0.005 | 0.01 | 18,000 | | residual |
| Geographic range (UK) - weighted | (Intercept) | -0.01 | -0.05 | 0.03 | 17,509 | 0.58 | fixed |
| | log.range | 0.002 | -0.001 | 0.005 | 17,525 | 0.16 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,435 | | residual |
| Geographic range (UK) - slope | (Intercept) | -0.01 | -0.07 | 0.06 | 18,000 | 0.85 | fixed |
| | log.range | 0.002 | -0.002 | 0.01 | 17,583 | 0.38 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Geographic range (UK) - fluctuations CI | (Intercept) | 0.03 | -0.04 | 0.10 | 18,000 | 0.41 | fixed |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | log.range | 0.01 | 0.003 | 0.01 | 18,000 | 0.002 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Geographic range (UK) - fluctuations CI weighted | (Intercept) | 0.02 | -0.06 | 0.10 | 18,000 | 0.59 | fixed |
| | log.range | 0.01 | 0.003 | 0.01 | 18,000 | 0.002 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 18,000 | | residual |
| Geographic range (UK) - fluctuations SE | (Intercept) | -0.02 | -0.09 | 0.06 | 18,000 | 0.67 | fixed |
| | log.range | 0.01 | 0.005 | 0.02 | 18,000 | 0.0002 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 18,000 | | residual |
| Geographic range (UK) - fluctuations SD | (Intercept) | 0.15 | 0.08 | 0.23 | 17,410 | 0.0002 | fixed |
| | log.range | 0.02 | 0.01 | 0.02 | 18,056 | 0.0001 | fixed |
| | sigma | 0.03 | 0.02 | 0.03 | 18,000 | | residual |
| Mean population size (UK) - mu | (Intercept) | 0.003 | -0.02 | 0.03 | 18,710 | 0.79 | fixed |
| | log.meanpop | 0.001 | -0.003 | 0.005 | 21,192 | 0.67 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,000 | | residual |
| Mean population size (UK) - weighted | (Intercept) | 0.003 | -0.02 | 0.02 | 18,000 | 0.80 | fixed |
| | log.meanpop | 0.001 | -0.003 | 0.005 | 18,000 | 0.67 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,000 | | residual |
| Mean population size (UK) - slope | (Intercept) | 0.04 | -0.01 | 0.08 | 18,581 | 0.11 | fixed |
| | log(meanpop) | -0.001 | -0.01 | 0.01 | 18,717 | 0.74 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 18,000 | | residual |
| Mean population size (UK) - fluctuations CI | (Intercept) | 0.30 | 0.24 | 0.36 | 17,427 | 0.0001 | fixed |
| | log.meanpop | -0.01 | -0.02 | 0.0004 | 18,000 | 0.05 | fixed |
| | sigma | 0.03 | 0.03 | 0.04 | 18,000 | | residual |
| Mean population size (UK) - fluctuations CI weighted | (Intercept) | 0.30 | 0.24 | 0.36 | 17,570 | 0.0001 | fixed |
| | log.meanpop | -0.01 | -0.02 | 0.0001 | 18,000 | 0.05 | fixed |
| | sigma | 0.03 | 0.03 | 0.04 | 18,000 | | residual |
| Mean population size (UK) - fluctuations SE | (Intercept) | 0.21 | 0.17 | 0.25 | 18,000 | 0.0001 | fixed |
| | log.meanpop | -0.01 | -0.02 | -0.01 | 18,000 | 0.0001 | fixed |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | sigma | 0.02 | 0.01 | 0.02 | 18,000 | | residual |
| Mean population size (UK) - fluctuations SD | (Intercept) | 0.50 | 0.44 | 0.56 | 18,000 | 0.0001 | fixed |
| | log.meanpop | -0.01 | -0.03 | -0.005 | 18,000 | 0.01 | fixed |
| | sigma | 0.03 | 0.02 | 0.04 | 18,403 | | residual |
| Habitat specificity (UK) - mu | (Intercept) | 0.01 | -0.01 | 0.02 | 18,000 | 0.38 | fixed |
| | hab_spec | 0.001 | -0.001 | 0.002 | 18,000 | 0.28 | fixed |
| | sigma | 0.01 | 0.005 | 0.01 | 16,892 | | residual |
| Habitat specificity (UK) - weighted | (Intercept) | 0.01 | -0.01 | 0.02 | 18,000 | 0.40 | fixed |
| | hab_spec | 0.001 | -0.001 | 0.003 | 18,000 | 0.23 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,727 | | residual |
| Habitat specificity (UK) - slope | (Intercept) | 0.01 | -0.01 | 0.04 | 20,246 | 0.33 | fixed |
| | hab_spec | 0.001 | -0.001 | 0.003 | 18,000 | 0.44 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,436 | | residual |
| Habita specificity (UK) - profiling method | (Intercept) | 0.01 | -0.003 | 0.03 | 18,000 | 0.11 | fixed |
| | hab_spec | -0.0003 | -0.002 | 0.002 | 18,000 | 0.76 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Habitat specificity (UK) - fluctuations CI | (Intercept) | 0.11 | 0.09 | 0.14 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.003 | 0.0003 | 0.01 | 18,000 | 0.03 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Habitat specificity (UK) - fluctuations CI weighted | (Intercept) | 0.11 | 0.08 | 0.15 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.004 | 0.0005 | 0.01 | 18,000 | 0.02 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 18,000 | | residual |
| Habitat specificity (UK) - fluctuations SE | (Intercept) | 0.11 | 0.08 | 0.14 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.002 | -0.001 | 0.004 | 18,000 | 0.28 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 18,000 | | residual |
| Habitat specificity (UK) - fluctuations SD | (Intercept) | 0.37 | 0.34 | 0.40 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.001 | -0.001 | 0.004 | 18,000 | 0.32 | fixed |
| | sigma | 0.03 | 0.02 | 0.03 | 18,000 | | residual |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Red List status (UK) - mu | Least concern | 0.01 | 0.004 | 0.02 | 18,000 | 0.001 | fixed |
| | Near threatened | 0.03 | 0.005 | 0.06 | 16,876 | 0.02 | fixed |
| | Vulnerable | 0.002 | -0.02 | 0.03 | 18,000 | 0.86 | fixed |
| | Endangered | -0.04 | -0.18 | 0.11 | 18,000 | 0.60 | fixed |
| | Critically endangered | 0.06 | -0.02 | 0.15 | 18,000 | 0.15 | fixed |
| | sigma | 0.01 | 0.005 | 0.01 | 18,000 | | residual |
| Red List status (UK) - weighted | Least concern | 0.02 | 0.01 | 0.02 | 18,000 | 0.0001 | fixed |
| | Near threatened | 0.03 | 0.002 | 0.07 | 18,416 | 0.04 | fixed |
| | Vulnerable | -0.01 | -0.04 | 0.03 | 18,000 | 0.73 | fixed |
| | Endangered | -0.04 | -0.17 | 0.10 | 18,000 | 0.59 | fixed |
| | Critically endangered | 0.04 | -0.18 | 0.28 | 18,000 | 0.72 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,398 | | residual |
| Red List status (UK) - slope | Least concern | 0.02 | 0.01 | 0.03 | 18,827 | 0.002 | fixed |
| | Near threatened | 0.04 | -0.01 | 0.09 | 18,000 | 0.10 | fixed |
| | Vulnerable | -0.001 | -0.05 | 0.05 | 18,000 | 0.96 | fixed |
| | Endangered | -0.01 | -0.28 | 0.25 | 18,000 | 0.93 | fixed |
| | Critically endangered | 0.05 | -0.10 | 0.21 | 18,000 | 0.48 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Red List status (UK) - fluctuations CI | Least concern | 0.14 | 0.13 | 0.16 | 18,000 | 0.0001 | fixed |
| | Near threatened | 0.14 | 0.09 | 0.19 | 18,000 | 0.0001 | fixed |
| | Vulnerable | 0.10 | 0.05 | 0.16 | 18,000 | 0.0002 | fixed |
| | Endangered | 0.24 | -0.05 | 0.52 | 18,000 | 0.10 | fixed |
| | Critically endangered | 0.09 | -0.07 | 0.26 | 17,435 | 0.30 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Red List status (UK) - fluctuations CI weighted | Least concern | 0.15 | 0.13 | 0.17 | 17,566 | 0.0001 | fixed |
| | Near threatened | 0.16 | 0.09 | 0.22 | 18,000 | 0.0001 | fixed |
| | Vulnerable | 0.11 | 0.04 | 0.17 | 17,582 | 0.001 | fixed |
| | Endangered | 0.24 | -0.06 | 0.54 | 18,000 | 0.11 | fixed |
| | Critically endangered | 0.06 | -0.23 | 0.34 | 18,000 | 0.68 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 17,700 | | residual |
| Red List status (UK) - fluctuations SE | Least concern | 0.13 | 0.11 | 0.14 | 18,000 | 0.0001 | fixed |
| | Near threatened | 0.13 | 0.08 | 0.18 | 18,600 | 0.0001 | fixed |
| | Vulnerable | 0.07 | 0.01 | 0.13 | 18,809 | 0.02 | fixed |
| | Endangered | 0.07 | -0.23 | 0.37 | 19,204 | 0.65 | fixed |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Critically endangered | 0.07 | -0.11 | 0.24 | 18,000 | 0.45 | fixed |
| | sigma | 0.02 | 0.02 | 0.03 | 18,000 | | residual |
| Red List status (UK) - fluctuations SD | Least concern | 0.33 | 0.33 | 0.34 | 18,000 | 0.0001 | fixed |
| | Near threatened | 0.40 | 0.37 | 0.43 | 20,507 | 0.0001 | fixed |
| | Vulnerable | 0.23 | 0.21 | 0.25 | 18,000 | 0.0001 | fixed |
| | Endangered | 0.24 | -0.02 | 0.52 | 18,000 | 0.08 | fixed |
| | Critically endangered | 0.28 | 0.16 | 0.40 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| System (UK) - mu | Freshwater | 0.02 | 0.001 | 0.03 | 18,000 | 0.04 | fixed |
| | Marine | 0.01 | -0.001 | 0.02 | 18,905 | 0.07 | fixed |
| | Terrestrial | 0.02 | 0.004 | 0.03 | 18,135 | 0.01 | fixed |
| | sigma | 0.01 | 0.004 | 0.01 | 18,000 | | residual |
| System (UK) - weighted | Freshwater | 0.02 | -0.004 | 0.03 | 21,467 | 0.10 | fixed |
| | Marine | 0.01 | 0.0004 | 0.02 | 17,586 | 0.04 | fixed |
| | Terrestrial | 0.02 | 0.01 | 0.03 | 18,424 | 0.005 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,000 | | residual |
| System (UK) - slope | Freshwater | 0.03 | 0.004 | 0.06 | 18,000 | 0.02 | fixed |
| | Marine | 0.01 | -0.005 | 0.03 | 18,064 | 0.20 | fixed |
| | Terrestrial | 0.03 | 0.01 | 0.05 | 18,705 | 0.01 | fixed |
| | sigma | 0.02 | 0.01 | 0.02 | 18,000 | | residual |
| Taxa (UK) - mu | Actinopterygii | -0.004 | -0.02 | 0.01 | 18,000 | 0.51 | fixed |
| | Amphibia | -0.002 | -0.04 | 0.03 | 18,000 | 0.90 | fixed |
| | Aves | 0.01 | 0.002 | 0.02 | 18,000 | 0.02 | fixed |
| | Mammalia | 0.03 | 0.02 | 0.05 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.04 | -0.02 | 0.10 | 18,000 | 0.16 | fixed |
| | sigma | 0.005 | 0.004 | 0.01 | 18,000 | | residual |
| Taxa (UK) - weighted | Actinopterygii | -0.01 | -0.02 | 0.01 | 18,000 | 0.29 | fixed |
| | Amphibia | -0.002 | -0.05 | 0.04 | 17,451 | 0.92 | fixed |
| | Aves | 0.02 | 0.01 | 0.03 | 18,000 | 0.003 | fixed |
| | Mammalia | 0.04 | 0.02 | 0.05 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.03 | -0.04 | 0.10 | 17,161 | 0.41 | fixed |
| | sigma | 0.004 | 0.004 | 0.01 | 18,000 | | residual |
| Taxa (UK) - slope | Actinopterygii | -0.01 | -0.04 | 0.01 | 18,000 | 0.41 | fixed |
| | Amphibia | -0.003 | -0.07 | 0.06 | 18,000 | 0.92 | fixed |
| | Aves | 0.02 | 0.0001 | 0.03 | 18,468 | 0.05 | fixed |
| | Mammalia | 0.06 | 0.04 | 0.09 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.07 | -0.05 | 0.18 | 18,000 | 0.23 | fixed |
| | sigma | 0.02 | 0.01 | 0.02 | 18,000 | | residual |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| System (global) - mu | Freshwater | 0.005 | 0.002 | 0.01 | 19,286 | 0.0001 | fixed |
| | Marine | 0.004 | 0.002 | 0.01 | 18,820 | 0.0003 | fixed |
| | Terrestrial | 0.003 | 0.001 | 0.005 | 18,000 | 0.001 | fixed |
| | sigma | 0.003 | 0.003 | 0.004 | 18,000 | | residual |
| System (global) - weighted | Freshwater | 0.01 | 0.002 | 0.01 | 18,000 | 0.001 | fixed |
| | Marine | 0.004 | 0.001 | 0.01 | 18,000 | 0.01 | fixed |
| | Terrestrial | 0.004 | 0.002 | 0.01 | 17,492 | 0.001 | fixed |
| | sigma | 0.004 | 0.004 | 0.004 | 19,112 | | residual |
| System (global) - slope | Freshwater | 0.01 | 0.01 | 0.01 | 18,000 | 0.0001 | fixed |
| | Marine | 0.004 | 0.0002 | 0.01 | 18,000 | 0.04 | fixed |
| | Terrestrial | 0.004 | 0.001 | 0.01 | 18,000 | 0.01 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| System (global) - fluctuations CI | Freshwater | 0.13 | 0.13 | 0.14 | 18,000 | 0.0001 | fixed |
| | Marine | 0.13 | 0.13 | 0.14 | 18,000 | 0.0001 | fixed |
| | Terrestrial | 0.10 | 0.10 | 0.10 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| System (global) - fluctuations CI weighted | Freshwater | 0.15 | 0.14 | 0.16 | 18,000 | 0.0001 | fixed |
| | Marine | 0.15 | 0.14 | 0.16 | 18,000 | 0.0001 | fixed |
| | Terrestrial | 0.11 | 0.10 | 0.11 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| System (global) - fluctuations SE | Freshwater | 0.12 | 0.12 | 0.13 | 18,000 | 0.0001 | fixed |
| | Marine | 0.12 | 0.11 | 0.12 | 18,000 | 0.0001 | fixed |
| | Terrestrial | 0.09 | 0.09 | 0.09 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 16,149 | | residual |
| System (global) - fluctuations SD | Freshwater | 0.55 | 0.55 | 0.56 | 18,943 | 0.0001 | fixed |
| | Marine | 0.56 | 0.56 | 0.56 | 18,000 | 0.0001 | fixed |
| | Terrestrial | 0.54 | 0.54 | 0.54 | 18,000 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Biome (global) - mu | Boreal forests/taiga | -0.001 | -0.004 | 0.002 | 17,462 | 0.70 | fixed |
| | Deserts and xeric shrublands | -0.01 | -0.03 | 0.002 | 18,000 | 0.09 | fixed |
| | Trop. and subtrop. grasslands savannas and shrublands | -0.01 | -0.02 | -0.001 | 18,000 | 0.03 | fixed |
| | Large lakes | 0.003 | -0.004 | 0.01 | 18,000 | 0.42 | fixed |
| | Mediterranean forests woodlands and scrub | 0.02 | 0.01 | 0.02 | 18,000 | 0.0002 | fixed |

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|---|---|---|---|---|---|---|---|
| | Montane freshwaters | 0.01 | -0.01 | 0.04 | 18,000 | 0.34 | fixed |
| | Montane grasslands and shrublands | 0.02 | 0.001 | 0.04 | 18,000 | 0.03 | fixed |
| | Polar freshwaters | 0.01 | 0.001 | 0.01 | 18,000 | 0.02 | fixed |
| | Polar seas | -0.002 | -0.02 | 0.02 | 18,000 | 0.83 | fixed |
| | Temperate forests | 0.005 | 0.002 | 0.01 | 18,000 | 0.0001 | fixed |
| | Temperate wetlands and rivers | 0.005 | 0.002 | 0.01 | 18,000 | 0.001 | fixed |
| | Temperate grasslands savannas and shrublands | -0.004 | -0.01 | 0.002 | 18,000 | 0.22 | fixed |
| | Tropical wetlands and rivers | 0.01 | -0.003 | 0.01 | 18,000 | 0.21 | fixed |
| | Tropical and subtropical forests | 0.02 | 0.01 | 0.03 | 18,000 | 0.0001 | fixed |
| | Tropical coral | 0.02 | 0.01 | 0.04 | 18,000 | 0.0001 | fixed |
| | Tundra | 0.01 | -0.001 | 0.01 | 18,000 | 0.09 | fixed |
| | Xeric freshwaters and endorheic basins | -0.002 | -0.02 | 0.01 | 15,937 | 0.77 | fixed |
| | sigma | 0.003 | 0.003 | 0.003 | 17,444 | | residual |
| Biome (global) - slope | Boreal forests/taiga | -0.001 | -0.01 | 0.003 | 18,000 | 0.55 | fixed |
| | Deserts and xeric shrublands | -0.04 | -0.06 | -0.02 | 17,811 | 0.001 | fixed |
| | Trop. and subtrop. grasslands savannas and shrublands | -0.01 | -0.02 | -0.002 | 18,000 | 0.02 | fixed |
| | Large lakes | 0.004 | -0.01 | 0.02 | 18,000 | 0.50 | fixed |
| | Mediterranean forests woodlands and scrub | 0.03 | 0.01 | 0.04 | 18,000 | 0.0001 | fixed |
| | Montane freshwaters | 0.03 | -0.01 | 0.07 | 18,000 | 0.18 | fixed |
| | Montane grasslands and shrublands | 0.02 | -0.004 | 0.05 | 18,000 | 0.10 | fixed |
| | Polar freshwaters | 0.01 | -0.004 | 0.02 | 18,000 | 0.22 | fixed |
| | Polar seas | -0.01 | -0.04 | 0.02 | 18,000 | 0.50 | fixed |
| | Temperate forests | 0.01 | 0.004 | 0.01 | 18,000 | 0.0001 | fixed |
| | Temperate wetlands and rivers | 0.01 | 0.002 | 0.01 | 18,000 | 0.004 | fixed |
| | Temperate grasslands savannas and shrublands | -0.004 | -0.01 | 0.01 | 18,000 | 0.42 | fixed |
| | Tropical wetlands and rivers | 0.003 | -0.01 | 0.02 | 18,000 | 0.64 | fixed |

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|---|---|---|---|---|---|---|---|
| | Tropical and subtropical forests | 0.02 | 0.01 | 0.04 | 17,621 | 0.0001 | fixed |
| | Tropical coral | 0.04 | 0.02 | 0.05 | 18,960 | 0.0001 | fixed |
| | Tundra | 0.01 | 0.0001 | 0.03 | 18,000 | 0.04 | fixed |
| | Xeric freshwaters and endorheic basins | 0.01 | -0.02 | 0.03 | 18,000 | 0.60 | fixed |
| | sigma | 0.004 | 0.003 | 0.004 | 18,000 | | residual |
| Biome (global) - weighted | Boreal forests/taiga | -0.002 | -0.01 | 0.003 | 17,517 | 0.42 | fixed |
| | Deserts and xeric shrublands | -0.03 | -0.05 | 0.003 | 18,000 | 0.08 | fixed |
| | Trop. and subtrop. grasslands savannas and shrublands | -0.002 | -0.02 | 0.01 | 18,000 | 0.71 | fixed |
| | Large lakes | 0.01 | -0.01 | 0.02 | 18,000 | 0.29 | fixed |
| | Mediterranean forests woodlands and scrub | 0.02 | 0.005 | 0.03 | 19,064 | 0.01 | fixed |
| | Montane freshwaters | 0.01 | -0.03 | 0.05 | 18,469 | 0.65 | fixed |
| | Montane grasslands and shrublands | 0.02 | -0.01 | 0.06 | 18,000 | 0.12 | fixed |
| | Polar freshwaters | 0.02 | 0.01 | 0.03 | 18,630 | 0.002 | fixed |
| | Polar seas | -0.01 | -0.05 | 0.02 | 17,320 | 0.44 | fixed |
| | Temperate forests | 0.01 | 0.004 | 0.01 | 17,516 | 0.0002 | fixed |
| | Temperate wetlands and rivers | 0.01 | 0.01 | 0.01 | 17,324 | 0.0001 | fixed |
| | Temperate grasslands savannas and shrublands | -0.01 | -0.02 | 0.002 | 17,573 | 0.14 | fixed |
| | Tropical wetlands and rivers | 0.01 | -0.004 | 0.02 | 18,000 | 0.13 | fixed |
| | Tropical and subtropical forests | 0.01 | -0.001 | 0.03 | 18,000 | 0.07 | fixed |
| | Tropical coral | 0.04 | 0.02 | 0.06 | 17,511.66 | 0.0001 | fixed |
| | Tundra | 0.01 | -0.004 | 0.02 | 18,000 | 0.16 | fixed |
| | Xeric freshwaters and endorheic basins | -0.03 | -0.05 | -0.0004 | 18,000 | 0.04 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Biome (global) - fluctuations CI | Boreal forests/taiga | 0.07 | 0.07 | 0.08 | 18,000 | 0.0001 | fixed |
| | Deserts and xeric shrublands | 0.22 | 0.18 | 0.26 | 18,658 | 0.0001 | fixed |
| | Trop. and subtrop. grasslands savannas and shrublands | 0.21 | 0.19 | 0.23 | 18,000 | 0.0001 | fixed |

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|---|---|---|---|---|---|---|---|
| | Large lakes | 0.11 | 0.09 | 0.12 | 18,000 | 0.0001 | fixed |
| | Mediterranean forests woodlands and scrub | 0.15 | 0.13 | 0.16 | 17,403 | 0.0001 | fixed |
| | Montane freshwaters | 0.14 | 0.09 | 0.19 | 18,000 | 0.0001 | fixed |
| | Montane grasslands and shrublands | 0.17 | 0.13 | 0.21 | 18,000 | 0.0001 | fixed |
| | Polar freshwaters | 0.14 | 0.13 | 0.15 | 18,000 | 0.0001 | fixed |
| | Polar seas | 0.16 | 0.11 | 0.21 | 18,148 | 0.0001 | fixed |
| | Temperate forests | 0.09 | 0.09 | 0.10 | 18,000 | 0.0001 | fixed |
| | Temperate wetlands and rivers | 0.13 | 0.12 | 0.13 | 18,000 | 0.0001 | fixed |
| | Temperate grasslands savannas and shrublands | 0.08 | 0.07 | 0.09 | 18,000 | 0.0001 | fixed |
| | Tropical wetlands and rivers | 0.20 | 0.19 | 0.22 | 18,440 | 0.0001 | fixed |
| | Tropical and subtropical forests | 0.20 | 0.18 | 0.21 | 18,000 | 0.0001 | fixed |
| | Tropical coral | 0.16 | 0.14 | 0.19 | 17,964 | 0.0001 | fixed |
| | Tundra | 0.11 | 0.09 | 0.13 | 18,000 | 0.0001 | fixed |
| | Xeric freshwaters and endorheic basins | 0.18 | 0.14 | 0.21 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.01 | 0.02 | 18,000 | | residual |
| Biome (global) - fluctuations CI weighted | Boreal forests/taiga | 0.08 | 0.07 | 0.08 | 18,000 | 0.0001 | fixed |
| | Deserts and xeric shrublands | 0.26 | 0.21 | 0.30 | 18,000 | 0.0001 | fixed |
| | Trop. and subtrop. grasslands savannas and shrublands | 0.25 | 0.23 | 0.27 | 17,477 | 0.0001 | fixed |
| | Large lakes | 0.12 | 0.09 | 0.14 | 18,000 | 0.0001 | fixed |
| | Mediterranean forests woodlands and scrub | 0.16 | 0.14 | 0.19 | 18,000 | 0.0001 | fixed |
| | Montane freshwaters | 0.23 | 0.15 | 0.31 | 17,217 | 0.0001 | fixed |
| | Montane grasslands and shrublands | 0.18 | 0.12 | 0.23 | 18,681 | 0.0001 | fixed |
| | Polar freshwaters | 0.17 | 0.15 | 0.19 | 17,751 | 0.0001 | fixed |
| | Polar seas | 0.18 | 0.12 | 0.23 | 18,000 | 0.0001 | fixed |
| | Temperate forests | 0.09 | 0.09 | 0.10 | 18,000 | 0.0001 | fixed |
| | Temperate wetlands and rivers | 0.14 | 0.13 | 0.15 | 18,000 | 0.0001 | fixed |
| | Temperate grasslands | 0.08 | 0.06 | 0.10 | 18,000 | 0.0001 | fixed |

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|---|---|---|---|---|---|---|---|
| | savannas and shrublands | | | | | | |
| | Tropical wetlands and rivers | 0.22 | 0.20 | 0.25 | 18,000 | 0.0001 | fixed |
| | Tropical and subtropical forests | 0.21 | 0.19 | 0.24 | 18,000 | 0.0001 | fixed |
| | Tropical coral | 0.18 | 0.15 | 0.21 | 18,000 | 0.0001 | fixed |
| | Tundra | 0.11 | 0.09 | 0.14 | 18,000 | 0.0001 | fixed |
| | Xeric freshwaters and endorheic basins | 0.23 | 0.18 | 0.27 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Biome (global) - fluctuations SE | Boreal forests/taiga | 0.06 | 0.06 | 0.07 | 18,000 | 0.0001 | fixed |
| | Deserts and xeric shrublands | 0.21 | 0.17 | 0.25 | 18,000 | 0.0001 | fixed |
| | Trop. and subtrop. grasslands savannas and shrublands | 0.11 | 0.09 | 0.13 | 17,497 | 0.0001 | fixed |
| | Large lakes | 0.09 | 0.07 | 0.11 | 18,000 | 0.0001 | fixed |
| | Mediterranean forests woodlands and scrub | 0.16 | 0.14 | 0.18 | 18,000 | 0.0001 | fixed |
| | Montane freshwaters | 0.15 | 0.10 | 0.21 | 16,658 | 0.0001 | fixed |
| | Montane grasslands and shrublands | 0.11 | 0.07 | 0.15 | 18,000 | 0.0001 | fixed |
| | Polar freshwaters | 0.14 | 0.13 | 0.15 | 18,000 | 0.0001 | fixed |
| | Polar seas | 0.08 | 0.03 | 0.13 | 18,000 | 0.002 | fixed |
| | Temperate forests | 0.09 | 0.08 | 0.09 | 18,000 | 0.0001 | fixed |
| | Temperate wetlands and rivers | 0.11 | 0.11 | 0.12 | 18,000 | 0.0001 | fixed |
| | Temperate grasslands savannas and shrublands | 0.07 | 0.06 | 0.09 | 18,000 | 0.0001 | fixed |
| | Tropical wetlands and rivers | 0.21 | 0.19 | 0.23 | 18,000 | 0.0001 | fixed |
| | Tropical and subtropical forests | 0.15 | 0.13 | 0.17 | 18,000 | 0.0001 | fixed |
| | Tropical coral | 0.18 | 0.16 | 0.21 | 18,000 | 0.0001 | fixed |
| | Tundra | 0.13 | 0.11 | 0.14 | 17,343 | 0.0001 | fixed |
| | Xeric freshwaters and endorheic basins | 0.13 | 0.09 | 0.17 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Biome (global) - fluctuations SD | Boreal forests/taiga | 0.51 | 0.51 | 0.52 | 18,000 | 0.0001 | fixed |
| | Deserts and xeric shrublands | 0.60 | 0.57 | 0.62 | 17,053.04 | 0.0001 | fixed |

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|---|---|---|---|---|---|---|---|
| | Trop. and subtrop. grasslands savannas and shrublands | 0.61 | 0.60 | 0.62 | 18,000 | 0.0001 | fixed |
| | Large lakes | 0.54 | 0.52 | 0.55 | 18,000 | 0.0001 | fixed |
| | Mediterranean forests woodlands and scrub | 0.59 | 0.58 | 0.61 | 17,567 | 0.0001 | fixed |
| | Montane freshwaters | 0.58 | 0.54 | 0.62 | 18,000 | 0.0001 | fixed |
| | Montane grasslands and shrublands | 0.59 | 0.56 | 0.62 | 18,000 | 0.0001 | fixed |
| | Polar freshwaters | 0.55 | 0.54 | 0.56 | 18,000 | 0.0001 | fixed |
| | Polar seas | 0.55 | 0.52 | 0.59 | 18,000 | 0.0001 | fixed |
| | Temperate forests | 0.53 | 0.53 | 0.54 | 18,056 | 0.0001 | fixed |
| | Temperate wetlands and rivers | 0.55 | 0.55 | 0.56 | 18,000 | 0.0001 | fixed |
| | Temperate grasslands savannas and shrublands | 0.52 | 0.51 | 0.53 | 17,393 | 0.0001 | fixed |
| | Tropical wetlands and rivers | 0.60 | 0.59 | 0.62 | 18,000 | 0.0001 | fixed |
| | Tropical and subtropical forests | 0.62 | 0.60 | 0.63 | 18,000 | 0.0001 | fixed |
| | Tropical coral | 0.61 | 0.59 | 0.63 | 18,000 | 0.0001 | fixed |
| | Tundra | 0.54 | 0.53 | 0.56 | 18,000 | 0.0001 | fixed |
| | Xeric freshwaters and endorheic basins | 0.56 | 0.53 | 0.58 | 18,000 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Taxa (global) - mu | Actinopterygii | -0.001 | -0.003 | 0.002 | 18,000 | 0.67 | fixed |
| | Amphibia | -0.01 | -0.02 | -0.005 | 18,000 | 0.003 | fixed |
| | Aves | 0.004 | 0.003 | 0.01 | 18,000 | 0.0001 | fixed |
| | Mammalia | 0.01 | 0.01 | 0.01 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.02 | 0.01 | 0.02 | 18,000 | 0.0001 | fixed |
| | sigma | 0.003 | 0.003 | 0.004 | 18,000 | | residual |
| Taxa (global) - weighted | Actinopterygii | -0.001 | -0.01 | 0.003 | 18,000 | 0.68 | fixed |
| | Amphibia | -0.01 | -0.03 | -0.002 | 18,000 | 0.03 | fixed |
| | Aves | 0.01 | 0.003 | 0.01 | 18,000 | 0.0001 | fixed |
| | Mammalia | 0.01 | 0.01 | 0.02 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.01 | -0.001 | 0.02 | 18,000 | 0.07 | fixed |
| | sigma | 0.004 | 0.004 | 0.004 | 18,000 | | residual |
| Taxa (global) - slope | Actinopterygii | -0.002 | -0.01 | 0.002 | 18,000 | 0.34 | fixed |
| | Amphibia | -0.02 | -0.04 | -0.01 | 18,000 | 0.002 | fixed |
| | Aves | 0.01 | 0.004 | 0.01 | 17,484 | 0.0001 | fixed |
| | Mammalia | 0.01 | 0.01 | 0.02 | 16,803 | 0.0001 | fixed |

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|---|---|---|---|---|---|---|---|
| | Reptilia | 0.03 | 0.02 | 0.04 | 17,631 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Taxa (global) - fluctuations CI | Actinopterygii | 0.15 | 0.14 | 0.16 | 18,000 | 0.0001 | fixed |
| | Amphibia | 0.22 | 0.20 | 0.24 | 17,663 | 0.0001 | fixed |
| | Aves | 0.09 | 0.09 | 0.10 | 18,000 | 0.0001 | fixed |
| | Mammalia | 0.17 | 0.17 | 0.18 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.17 | 0.15 | 0.18 | 18,410 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.02 | 18,000 | | residual |
| Taxa (global) - fluctuations CI weighted | Actinopterygii | 0.17 | 0.16 | 0.18 | 18,000 | 0.0001 | fixed |
| | Amphibia | 0.28 | 0.26 | 0.31 | 18,000 | 0.0001 | fixed |
| | Aves | 0.10 | 0.09 | 0.10 | 18,000 | 0.0001 | fixed |
| | Mammalia | 0.20 | 0.19 | 0.21 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.21 | 0.19 | 0.23 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Taxa (global) - fluctuations SE | Actinopterygii | 0.14 | 0.13 | 0.14 | 18,000 | 0.0001 | fixed |
| | Amphibia | 0.24 | 0.23 | 0.26 | 18,539 | 0.0001 | fixed |
| | Aves | 0.08 | 0.07 | 0.08 | 18,000 | 0.0001 | fixed |
| | Mammalia | 0.15 | 0.14 | 0.16 | 16,776 | 0.0001 | fixed |
| | Reptilia | 0.19 | 0.18 | 0.20 | 18,576 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,836 | | residual |
| Taxa (global) - fluctuations SD | Actinopterygii | 0.56 | 0.56 | 0.57 | 18,000 | 0.0001 | fixed |
| | Amphibia | 0.62 | 0.60 | 0.63 | 17,434 | 0.0001 | fixed |
| | Aves | 0.53 | 0.53 | 0.53 | 18,000 | 0.0001 | fixed |
| | Mammalia | 0.60 | 0.59 | 0.60 | 18,000 | 0.0001 | fixed |
| | Reptilia | 0.59 | 0.58 | 0.60 | 18,000 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Amphibian population trends – mu and phylogeny | (Intercept) | -0.009 (-0.009 – -0.008) | -0.005 (-0.005 – -0.005) | 0.031 (0.027 – 0.035) | 10,047 (10,713 – 9350) | 0.59 (0.55 – 0.62) | fixed |
| | phylo | 0.001 (0.001 – 0.002) | 0.001 (0.001 – 0.001) | 0.004( 0.004 – 0.005) | 7,037 (6,171 – 7,602) | | random |
| | species | 0.001 (0.001 – 0.001) | 0.001 (0.001 – 0.001) | 0.002 (0.002 – 0.002) | 7,912 (6,541 – 8,500) | | random |
| | sigma | 0.006 (0.006 – 0.006) | 0.004 (0.004 – 0.004) | 0.007 (0.007 – 0.007) | 9,855 (9,280 – 10,041) | | residual |
| Bird population trends – mu and phylogeny | (Intercept) | 0.005 (0.004 – 0.005) | -0.009 (-0.014 – -0.005) | 0.017 (0.02 – 0.02) | 10,304 (9,515 – 12,560) | 0.45 (0.31 – 0.63) | fixed |
| | phylo | 0.001 (0.001 – 0.003) | 0.001 (0.001 – 0.001) | 0.001 (0.001 – 0.001) | 3,850 (3,305 – 4,144) | | random |

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|---|---|---|---|---|---|---|---|
| | species | 0.001 (0.001 – 0.001) | 0.001 (0.001 – 0.001) | 0.001 (0.001 – 0.001) | 6,799 (6,358 – 7,260) | | random |
| | sigma | 0.002 (0.002 – 0.002) | 0.002 (0.002 – 0.002) | 0.003 (0.003 – 0.003) | 9,991 (9,713 – 10,345) | | residual |
| Reptile population trends – mu and phylogeny | (Intercept) | 0.004 (0.003 – 0.005) | -0.004 (-0.005 – -0.003) | 0.047 (0.043 – 0.056) | 10,030 (9,679 – 10,595) | 0.83 (0.81 – 0.85) | fixed |
| | phylo | 0.002 (0.001 – 0.003) | 0.001 (0.001 – 0.001) | 0.007 (0.005 – 0.010) | 3,971 (2,850 – 4,823) | | random |
| | species | 0.004 (0.004 – 0.004) | 0.001 (0.001 – 0.001) | 0.007 (0.007 – 0.007) | 3,418 (3,058 – 3,701) | | random |
| | sigma | 0.004 (0.004 – 0.004) | 0.001 (0.001 – 0.001) | 0.007 (0.007 – 0.007) | 3,562 (3,133 – 3,765) | | residual |
| Amphibian population fluctuations – CI and phylogeny | (Intercept) | 0.271 (0.269 – 0.271) | 0.217 (0.215 – 0.219) | 0.323 (0.321 – 0.327) | 9,905 (8,988 – 10,066) | 0.001 (0.001 – 0.001) | fixed |
| | phylo | 0.001 (0.001 - 0.002) | 0.001 (0.001 - 0.001) | 0.006 (0.005 – 0.006) | 9,754 (8,283 – 10,000) | | random |
| | species | 0.001 (0.001 – 0.001) | 0.001 (0.001 - 0.001) | 0.005 (0.005 – 0.005) | 9,822 (8,557 – 10,000) | | random |
| | sigma | 0.045 (0.0349 – 0.045) | 0.035 (0.035 – 0.035) | 0.056 (0.055 – 0.056) | 10,076 (10,000 – 10,413) | | residual |
| Bird population fluctuations – CI and phylogeny | (Intercept) | 0.114 (0.109 – 0.119) | 0.059 (0.048 – 0.070) | 0.169 (0.153 – 0.188) | 10,000 (10, 000 – 10,000) | 0.001 (0.001 – 0.001) | fixed |
| | phylo | 0.005 (0.004 – 0.006) | 0.003 (0.0.002 – 0.003) | 0.007 (0.005 – 0.009) | 4,017 (3,779 – 4,336) | | random |
| | species | 0.002 (0.002 – 0.002) | 0.001 (0.001 – 0.001) | 0.003 (0.002 – 0.003) | 5,706 (4,991 – 6,092) | | random |
| | sigma | 0.010 (0.010 – 0.010) | 0.010 (0.010 – 0.010) | 0.011 (0.011 – 0.011) | 9,936 (9,223 – 11,161) | | residual |
| Reptile population fluctuations – CI and phylogeny | (Intercept) | 0.302 (0.301 – 0.306) | 0.180 (0.1625 – 0.190) | 0.424 (0.410 – 0.434) | 9,837 (9,242 – 10,009) | 0.001 (0.001 – 0.001) | fixed |
| | phylo | 0.014 (0.012 – 0.014) | 0.001 (0.001 – 0.001) | 0.056 (0.050 – 0.060) | 2,779 (2,256 – 3,213) | | random |
| | species | 0.050 (0.049 – 0.050) | 0.019 (0.016 – 0.022) | 0.082 (0.081 – 0.084) | 4,407 (3,544 – 4,730) | | random |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | sigma | 0.018 (0.018 – 0.018) | 0.006 (0.006 – 0.007) | 0.036 (0.035 – 0.036) | 3,985 (3,817 – 4,536) | | residual |
| Geographic range (birds) - mu | (Intercept) | -0.01 | -0.03 | 0.004 | 18,000 | 0.15 | fixed |
| | log(range) | 0.001 | -0.0000 | 0.002 | 18,000 | 0.05 | fixed |
| | sigma | 0.003 | 0.003 | 0.003 | 18,000 | | residual |
| Geographic range (birds) - weighted | (Intercept) | -0.003 | -0.03 | 0.02 | 18,000 | 0.77 | fixed |
| | log(range) | 0.001 | -0.001 | 0.002 | 17,620 | 0.41 | fixed |
| | sigma | 0.003 | 0.003 | 0.003 | 18,000 | | residual |
| Geographic range (birds) - slope | (Intercept) | -0.02 | -0.04 | 0.01 | 18,000 | 0.26 | fixed |
| | log.range | 0.001 | -0.0003 | 0.003 | 18,000 | 0.10 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,881 | | residual |
| Geographic range (birds) - fluctuations CI | (Intercept) | 0.02 | -0.02 | 0.05 | 18,000 | 0.37 | fixed |
| | log(range) | 0.005 | 0.003 | 0.01 | 18,000 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Geographic range (birds) - fluctuations CI weighted | (Intercept) | 0.01 | -0.04 | 0.05 | 18,000 | 0.72 | fixed |
| | log(range) | 0.01 | 0.003 | 0.01 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.01 | 0.02 | 18,657 | | residual |
| Geographic range (birds) - fluctuations SE | (Intercept) | -0.04 | -0.08 | -0.01 | 18,000 | 0.01 | fixed |
| | log(range) | 0.01 | 0.01 | 0.01 | 18,000 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Geographic range (birds) - fluctuations SD | (Intercept) | 0.47 | 0.44 | 0.50 | 18,000 | 0.0001 | fixed |
| | log(range) | 0.004 | 0.002 | 0.01 | 18,000 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Mean population size (global) - mu | (Intercept) | 0.01 | 0.001 | 0.01 | 18,000 | 0.03 | fixed |
| | log.meanpop | 0.001 | -0.0002 | 0.001 | 18,000 | 0.14 | fixed |
| | sigma | 0.005 | 0.005 | 0.01 | 17,393 | | residual |
| Mean population size (global) - weighted | (Intercept) | 0.01 | 0.0004 | 0.01 | 17,924 | 0.04 | fixed |
| | log.meanpop | 0.001 | -0.001 | 0.002 | 18,000 | 0.37 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Mean population size (global) - slope | (Intercept) | 0.01 | 0.0004 | 0.02 | 17,382 | 0.04 | fixed |
| | log.meanpop | 0.001 | -0.0003 | 0.002 | 17,360 | 0.14 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Mean population size (global) - fluctuations CI | (Intercept) | 0.21 | 0.20 | 0.23 | 18,000 | 0.0001 | fixed |
| | log.meanpop | -0.004 | -0.01 | -0.002 | 18,000 | 0.0001 | fixed |
| | sigma | 0.03 | 0.03 | 0.03 | 18,000 | | residual |
| Mean population size (global) - fluctuations CI weighted | (Intercept) | 0.25 | 0.23 | 0.26 | 18,000 | 0.0001 | fixed |
| | log.meanpop | -0.01 | -0.01 | -0.003 | 18,000 | 0.0001 | fixed |
| | sigma | 0.03 | 0.03 | 0.03 | 18,000 | | residual |
| Mean population size (global) - fluctuations SE | (Intercept) | 0.22 | 0.21 | 0.23 | 18,000 | 0.0001 | fixed |
| | log.meanpop | -0.01 | -0.01 | -0.01 | 18,000 | 0.0001 | fixed |
| | sigma | 0.03 | 0.03 | 0.03 | 17,622 | | residual |
| Mean population size (global) - fluctuations SD | (Intercept) | -0.002 | -0.002 | -0.001 | 18,000 | 0.0001 | fixed |
| | log.meanpop | 0.0003 | 0.0002 | 0.0004 | 18,000 | 0.0001 | fixed |
| | sigma | 0.0002 | 0.0001 | 0.0002 | 18,000 | | residual |
| Habitat specificity (global) - mu | (Intercept) | 0.004 | 0.001 | 0.01 | 18,000 | 0.002 | fixed |
| | hab_spec | 0.0001 | -0.0001 | 0.0004 | 18,000 | 0.36 | fixed |
| | sigma | 0.004 | 0.004 | 0.004 | 18,000 | | residual |
| Habitat specificity (global) - weighted | (Intercept) | 0.003 | -0.0002 | 0.01 | 18,000 | 0.06 | fixed |
| | hab_spec | 0.0003 | 0.0000 | 0.001 | 18,000 | 0.05 | fixed |
| | sigma | 0.004 | 0.004 | 0.004 | 18,000 | | residual |
| Habitat specificity (global) - slope | (Intercept) | 0.01 | 0.002 | 0.01 | 17,426 | 0.004 | fixed |
| | hab_spec | 0.0002 | -0.0002 | 0.001 | 17,536 | 0.41 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Habitat specificity (global) - fluctuations CI | (Intercept) | 0.11 | 0.10 | 0.11 | 17,358 | 0.0001 | fixed |
| | hab_spec | 0.001 | 0.001 | 0.002 | 18,000 | 0.0001 | fixed |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Habitat specificity (global) - fluctuations CI weighted | (Intercept) | 0.12 | 0.11 | 0.12 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.001 | 0.001 | 0.002 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Habitat specificity (global) - fluctuations SE | (Intercept) | 0.10 | 0.09 | 0.10 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.001 | 0.001 | 0.002 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Habitat specificity (global) - fluctuations SD | (Intercept) | 0.54 | 0.54 | 0.55 | 18,000 | 0.0001 | fixed |
| | hab_spec | 0.001 | 0.0002 | 0.001 | 18,563 | 0.002 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Red List status (global) - mu | Least concern | 0.01 | 0.005 | 0.01 | 18,000 | 0.0001 | fixed |
| | Near threatened | -0.01 | -0.01 | -0.0005 | 18,000 | 0.04 | fixed |
| | Vulnerable | 0.002 | -0.003 | 0.01 | 18,294 | 0.40 | fixed |
| | Endangered | -0.002 | -0.01 | 0.01 | 18,000 | 0.60 | fixed |
| | Critically endangered | -0.01 | -0.02 | 0.0001 | 18,000 | 0.05 | fixed |
| | sigma | 0.004 | 0.003 | 0.004 | 18,000 | | residual |
| Red List status (global) - weighted | Least concern | 0.01 | 0.01 | 0.01 | 19,846 | 0.0001 | fixed |
| | Near threatened | -0.003 | -0.01 | 0.005 | 18,000 | 0.45 | fixed |
| | Vulnerable | 0.002 | -0.005 | 0.01 | 18,000 | 0.52 | fixed |
| | Endangered | -0.01 | -0.02 | 0.003 | 18,000 | 0.19 | fixed |
| | Critically endangered | -0.02 | -0.03 | -0.01 | 18,000 | 0.01 | fixed |
| | sigma | 0.004 | 0.004 | 0.004 | 18,000 | | residual |
| Red List status (global) - slope | Least concern | 0.01 | 0.01 | 0.01 | 18,000 | 0.0001 | fixed |
| | Near threatened | -0.01 | -0.02 | -0.004 | 18,000 | 0.01 | fixed |
| | Vulnerable | 0.002 | -0.01 | 0.01 | 17,507 | 0.58 | fixed |
| | Endangered | -0.001 | -0.01 | 0.01 | 18,000 | 0.91 | fixed |
| | Critically endangered | -0.01 | -0.02 | 0.01 | 17,564 | 0.44 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Red List status (global) - fluctuations CI | Least concern | 0.11 | 0.11 | 0.12 | 18,000 | 0.0001 | fixed |
| | Near threatened | 0.14 | 0.13 | 0.15 | 18,000 | 0.0001 | fixed |
| | Vulnerable | 0.14 | 0.13 | 0.15 | 19,536 | 0.0001 | fixed |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Endangered | 0.16 | 0.15 | 0.17 | 18,000 | 0.0001 | fixed |
| | Critically endangered | 0.17 | 0.15 | 0.19 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Red List status (global) - fluctuations CI weighted | Least concern | 0.12 | 0.12 | 0.12 | 18,432 | 0.0001 | fixed |
| | Near threatened | 0.16 | 0.15 | 0.18 | 18,812 | 0.0001 | fixed |
| | Vulnerable | 0.15 | 0.14 | 0.17 | 18,832 | 0.0001 | fixed |
| | Endangered | 0.19 | 0.17 | 0.21 | 17,714 | 0.0001 | fixed |
| | Critically endangered | 0.19 | 0.16 | 0.22 | 18,469 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,000 | | residual |
| Red List status (global) - fluctuations SE | Least concern | 0.10 | 0.10 | 0.10 | 18,000 | 0.0001 | fixed |
| | Near threatened | 0.13 | 0.12 | 0.14 | 19,308 | 0.0001 | fixed |
| | Vulnerable | 0.11 | 0.10 | 0.12 | 18,645 | 0.0001 | fixed |
| | Endangered | 0.13 | 0.12 | 0.15 | 18,000 | 0.0001 | fixed |
| | Critically endangered | 0.15 | 0.13 | 0.17 | 18,000 | 0.0001 | fixed |
| | sigma | 0.02 | 0.02 | 0.02 | 18,555 | | residual |
| Red List status (global) - fluctuations SD | Least concern | 0.54 | 0.54 | 0.55 | 18,390 | 0.0001 | fixed |
| | Near threatened | 0.57 | 0.56 | 0.58 | 18,000 | 0.0001 | fixed |
| | Vulnerable | 0.56 | 0.55 | 0.57 | 18,214 | 0.0001 | fixed |
| | Endangered | 0.59 | 0.58 | 0.60 | 18,000 | 0.0001 | fixed |
| | Critically endangered | 0.61 | 0.59 | 0.62 | 17,295 | 0.0001 | fixed |
| | sigma | 0.01 | 0.01 | 0.01 | 18,000 | | residual |
| Latitude (global) - mu | (Intercept) | 0.004 | 0.002 | 0.01 | 18,254 | 0.0001 | fixed |
| | Decimal.Latitude | -0.0000 | -0.0000 | 0.0000 | 18,670 | 0.90 | fixed |
| | sigma | 0.003 | 0.003 | 0.004 | 18,000 | | residual |
| Duration (global) - mu | End | 0.0000 | 0.0000 | 0.0001 | 18,561 | 0.03 | fixed |
| | sigma | 0.003 | 0.003 | 0.004 | 18,000 | | residual |

967 **Table S3. List of species included in the UK scale analysis of population**

968 **change across rarity metrics.**

969

| Species name | Number of populations |
|---|---|
| *Acrocephalus schoenobaenus* | 1 |
| *Acrocephalus scirpaceus* | 1 |
| *Agonus cataphractus* | 1 |
| *Alca torda* | 4 |
| *Anarhichas lupus* | 1 |
| *Anas acuta* | 1 |
| *Anas crecca* | 1 |
| *Anas platyrhynchos* | 2 |
| *Anser albifrons* | 4 |
| *Anser fabalis* | 1 |
| *Anthus pratensis* | 2 |
| *Ardea cinerea* | 1 |
| *Arenaria interpres* | 1 |
| *Argentina silus* | 1 |
| *Argentina sphyraena* | 1 |
| *Arnoglossus laterna* | 1 |
| *Asio flammeus* | 1 |
| *Aythya ferina* | 4 |
| *Aythya fuligula* | 2 |
| *Botaurus stellaris* | 14 |
| *Branta bernicla* | 54 |
| *Branta canadensis* | 1 |
| *Branta leucopsis* | 3 |
| *Brosme brosme* | 1 |
| *Bucephala clangula* | 1 |
| *Bufo bufo* | 1 |
| *Burhinus oedicnemus* | 1 |
| *Buteo buteo* | 1 |
| *Calidris alba* | 1 |
| *Calidris alpina* | 1 |
| *Calidris canutus* | 2 |
| *Calidris maritima* | 2 |
| *Callionymus maculatus* | 1 |

| | |
|---|---|
| *Capreolus capreolus* | 3 |
| *Carduelis cannabina* | 2 |
| *Cepphus grylle* | 2 |
| *Cervus elaphus* | 2 |
| *Cetorhinus maximus* | 1 |
| *Cettia cetti* | 1 |
| *Charadrius hiaticula* | 1 |
| *Chelidonichthys lucerna* | 2 |
| *Circus aeruginosus* | 1 |
| *Circus cyaneus* | 1 |
| *Clupea harengus* | 2 |
| *Columba oenas* | 1 |
| *Coronella austriaca* | 1 |
| *Corvus corax* | 1 |
| *Corvus corone* | 1 |
| *Corvus monedula* | 1 |
| *Crex crex* | 1 |
| *Cyclopterus lumpus* | 1 |
| *Cygnus columbianus* | 3 |
| *Cygnus cygnus* | 1 |
| *Cygnus olor* | 1 |
| *Delphinus delphis* | 2 |
| *Echiichthys vipera* | 1 |
| *Egretta garzetta* | 1 |
| *Emberiza cirlus* | 1 |
| *Emberiza citrinella* | 1 |
| *Emberiza schoeniclus* | 3 |
| *Eptesicus serotinus* | 1 |
| *Esox lucius* | 2 |
| *Falco peregrinus* | 1 |
| *Falco tinnunculus* | 1 |
| *Fulica atra* | 1 |
| *Fulmarus glacialis* | 7 |
| *Gadus morhua* | 7 |
| *Glyptocephalus cynoglossus* | 1 |
| *Haematopus ostralegus* | 2 |
| *Haliaeetus albicilla* | 1 |
| *Halichoerus grypus* | 52 |

| | |
|---|---|
| *Hippoglossus hippoglossus* | 1 |
| *Lagopus lagopus* | 3 |
| *Larus argentatus* | 1 |
| *Larus canus* | 1 |
| *Larus fuscus* | 1 |
| *Larus melanocephalus* | 1 |
| *Lepidorhombus whiffiagonis* | 2 |
| *Lepus timidus* | 1 |
| *Limosa lapponica* | 2 |
| *Limosa limosa* | 2 |
| *Lissotriton vulgaris* | 5 |
| *Lophius budegassa* | 1 |
| *Lophius piscatorius* | 2 |
| *Lullula arborea* | 1 |
| *Melanogrammus aeglefinus* | 9 |
| *Meles meles* | 1 |
| *Mergus serrator* | 1 |
| *Merlangius merlangus* | 8 |
| *Merluccius merluccius* | 2 |
| *Micromesistius poutassou* | 1 |
| *Milvus milvus* | 3 |
| *Molva molva* | 1 |
| *Morus bassanus* | 4 |
| *Muscicapa striata* | 1 |
| *Myotis nattereri* | 1 |
| *Natrix natrix* | 1 |
| *Netta rufina* | 1 |
| *Numenius arquata* | 2 |
| *Nyctalus noctula* | 1 |
| *Oenanthe oenanthe* | 1 |
| *Orcinus orca* | 1 |
| *Oriolus oriolus* | 1 |
| *Oryctolagus cuniculus* | 6 |
| *Oxyura jamaicensis* | 2 |
| *Pandion haliaetus* | 1 |
| *Parus major* | 2 |
| *Passer domesticus* | 1 |
| *Passer montanus* | 1 |

| | |
|---|---|
| *Perca fluviatilis* | 3 |
| *Perdix perdix* | 1 |
| *Phalacrocorax aristotelis* | 8 |
| *Phoca vitulina* | 28 |
| *Phrynorhombus norvegicus* | 1 |
| *Pipistrellus pipistrellus* | 1 |
| *Pipistrellus pygmaeus* | 1 |
| *Platichthys flesus* | 1 |
| *Plecotus auritus* | 1 |
| *Plectrophenax nivalis* | 2 |
| *Pleuronectes platessa* | 6 |
| *Pluvialis apricaria* | 2 |
| *Pluvialis squatarola* | 1 |
| *Podiceps cristatus* | 1 |
| *Pollachius pollachius* | 1 |
| *Pollachius virens* | 3 |
| *Prunella modularis* | 1 |
| *Puffinus mauretanicus* | 2 |
| *Pyrrhula pyrrhula* | 1 |
| *Raja brachyura* | 1 |
| *Raja clavata* | 1 |
| *Raja microocellata* | 1 |
| *Raja montagui* | 1 |
| *Rhinolophus ferrumequinum* | 2 |
| *Rhinolophus hipposideros* | 4 |
| *Rissa tridactyla* | 9 |
| *Salmo salar* | 1 |
| *Salmo trutta* | 1 |
| *Scomber scombrus* | 2 |
| *Scyliorhinus canicula* | 1 |
| *Sitta europaea* | 1 |
| *Sprattus sprattus* | 1 |
| *Stenella coeruleoalba* | 1 |
| *Stercorarius parasiticus* | 3 |
| *Sterna dougallii* | 5 |
| *Sterna hirundo* | 1 |
| *Sterna paradisaea* | 1 |
| *Sternula albifrons* | 1 |

| | |
|---|---|
| *Streptopelia turtur* | 1 |
| *Strix aluco* | 1 |
| *Sturnus vulgaris* | 1 |
| *Sylvia communis* | 1 |
| *Syngnathus rostellatus* | 1 |
| *Tachybaptus ruficollis* | 1 |
| *Tadorna tadorna* | 1 |
| *Thalasseus sandvicensis* | 2 |
| *Trachurus trachurus* | 2 |
| *Tringa nebularia* | 1 |
| *Tringa totanus* | 1 |
| *Trisopterus esmarkii* | 2 |
| *Trisopterus luscus* | 1 |
| *Trisopterus minutus* | 1 |
| *Triturus cristatus* | 5 |
| *Turdus philomelos* | 1 |
| *Tursiops truncatus* | 4 |
| *Uria aalge* | 10 |
| *Vanellus vanellus* | 3 |
| *Xiphias gladius* | 1 |

971 **Table S4. Profiling method for estimating habitat specificity for 144 species**
972 **with populations in the UK in the LPD.** We extracted the habitats in which each
973 species occurs from their IUCN profiles (http://www.iucnredlist.org/) and we followed
974 this key for consistency.

| Habitat | Considered to be the same as: | Considered to be different to: |
|---|---|---|
| **Rural park** | Suburban park, urban park, rural garden, suburban garden, urban garden | |
| **Lake** | Big lake, small lake, pond, pool, dam, oxbow lake, reservoir | |
| **Bog** | Swamp, bogland | Lagoon |
| **Coastal cliff** | Island cliff | |
| **Shingle beach** | Pebble beach, rock beach | Sandy beach |
| **Stream** | River | Weir |
| **Fruit Tree Plantation** | Fruit Garden, orchard | |
| **Thicket** | Copse, grove, small stand | |
| **Forest** | | Woodland |
| **Glade** | Forest Clearing | |
| **Broadleaf** | Deciduous | |
| **Urban** | Suburban | |
| **River margin** | Various types of river margins | |
| **Tidal Creek** | | Estuary |
| **Harbour** | Dock, jetty, pier | |
| **Bush lands** | Shrublands | |
| **Irrigation channel** | Ditch | |
| **Heath** | Moorland | |
| **Sandy beach** | Spit, dune | Shingle beach, pebble beach, rock beach |
| **Crag** | Rocky outcrop, cliff, rocky slope | |

| | |
|---|---|
| **Marsh** | Wet meadow |
| **Islet** | Island |

975