

## Review

# A global assessment of amphibian and reptile responses to land-use changes<sup>☆</sup>

Javier M. Cordier<sup>a,b</sup>, Ramiro Aguilar<sup>c,d,\*</sup>, Julián N. Lescano<sup>a,b</sup>, Gerardo C. Leynaud<sup>a,b</sup>, Andrea Bonino<sup>a,b</sup>, Daniela Miloch<sup>a,b</sup>, Rafael Loyola<sup>e,f</sup>, Javier Nori<sup>a,b</sup>

<sup>a</sup> Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Instituto de Diversidad y Ecología Animal (IDEA), Córdoba, Argentina

<sup>b</sup> Universidad Nacional de Córdoba, Facultad de Ciencias Exactas, Físicas y Naturales, Centro de Zoología Aplicada, Córdoba, Argentina

<sup>c</sup> Instituto Multidisciplinario de Biología Vegetal, Universidad Nacional de Córdoba – CONICET, C.C. 495, (X5000JJC), Córdoba, Argentina

<sup>d</sup> Laboratorio Nacional de Análisis y Síntesis Ecológica, Escuela Nacional de Estudios Superiores Unidad Morelia, Universidad Nacional Autónoma de México, Mexico

<sup>e</sup> Laboratório de Biogeografia da Conservação, Departamento de Ecologia, Universidade Federal de Goiás, Goiânia, Goiás, Brazil

<sup>f</sup> Fundação Brasileira para o Desenvolvimento Sustentável, Rio de Janeiro, Rio de Janeiro, Brazil



## ARTICLE INFO

## Keywords:

Reptiles  
Herpetofauna  
Land-use change  
Systematic review  
Hierarchical meta-analysis  
Amphibians

## ABSTRACT

Human land-use changes represent the most important drivers of biodiversity loss, and amphibians and reptiles represent the most threatened groups of vertebrates globally. However, today there is a general lack of knowledge and little consensus on how land-use changes affect amphibians and reptiles. In order to fill this gap, here we conduct the most comprehensive systematic quantitative review of primary research to date. By means of hierarchical meta-analyses we assessed the effects of the most common land-use changes (agriculture, cattle-raising, urbanization, deforestation, silviculture and selective logging) on the richness of amphibian and reptile communities. Our results show that almost all of the analyzed types of land-use changes have negative effects on these groups, but with different degree of magnitude. We also show that the time elapsed in disturbed conditions does not ameliorate the effects on species richness, indicating a low recovery capacity of herp communities. Another important finding is that the richest communities are the most negatively affected ones, varying the response according to the type of biome. Our synthesis provides updated empirical evidence indicating that current prevalent human land-use changes strongly reduce the richness of amphibian and reptile species as well as revealing important knowledge gaps in certain biomes of the world. These results should help providing a basis for the development of future research and contextualizing the need for effective conservation measures for these two vertebrate groups.

## 1. Introduction

Human activities represent the main causal factor of current unprecedented species extinction rates and population declines, and they are expected to continue to be so throughout the next century (Laurance et al., 2012; Pimm et al., 2014; Pimm and Raven, 2000; Sala et al., 2000). Currently, human-induced habitat loss is identified as a primary threat to 85% of all species classified as threatened by the IUCN Red List (IUCN, 2015). Even more, the current rate of extinction is between 100 and 1000 times higher than the background extinction rate ever known, thus representing the most massive extinction rate over the last 65

million years (Ceballos et al., 2015). These anthropogenic processes not only reduce biodiversity, but also imply huge impoverishment of functional ecosystem diversity and functioning (Flynn et al., 2009), thereby imperiling many ecosystem services (Jantz et al., 2014; Newbold et al., 2015).

The increased human population and the consequent demand for resources have made profound changes in land cover, with approximately 53% of the Earth's land surface currently covered by human-modified environments (Hooke et al., 2012). Some ecoregions are being transformed at alarming rates. For example, forest regions have lost over 2.3 million square kilometers in the last decade (Hansen et al.,

<sup>☆</sup> Article Impact Statement: We present the first hierarchical meta-analysis that summarizes the negative impact of human land-use changes on amphibian and reptile communities worldwide.

\* Corresponding author at: Instituto Multidisciplinario de Biología Vegetal, Centro Científico Tecnológico CONICET, Córdoba, Argentina.

E-mail address: [raguilar@imbiv.unc.edu.ar](mailto:raguilar@imbiv.unc.edu.ar) (R. Aguilar).

<https://doi.org/10.1016/j.biocon.2020.108863>

Received 7 April 2020; Received in revised form 27 August 2020; Accepted 30 October 2020

Available online 25 November 2020

0006-3207/© 2020 Elsevier Ltd. All rights reserved.

2013). Similarly, grasslands, savannas and shrublands, which are natural biomes with the least surface of protected areas, have lost approximately 45% of its original extensions during the last decade (Hoekstra et al., 2004). Likewise, accelerated loss and conversion of coastal and continental wetlands have almost halved globally their original area (Davidson, 2014).

Throughout the world, different human activities have differential impacts on biodiversity. For example, urbanization and agriculture represent strong biotic homogenization factors that oversimplify habitats and often show negative effects on population abundance and community richness of vertebrate species at local scales (Gagné and Fahrig, 2007; Glor et al., 2001; Hodgkinson et al., 2007; McKinney, 2006; Ordeñana et al., 2010; Suazo-Ortuño et al., 2008; Vallan, 2002). Similarly, the structural changes imposed by deforestation implies increased loss, fragmentation, and isolation of remaining natural habitats, all of which usually trigger negative effects on vertebrate population dynamics and community composition (Karraker and Welsh, 2006; Kutt et al., 2012; Scott et al., 2006). Other anthropogenic activities, however, show mixed response patterns on vertebrate species. For example, productive activities such as cattle raising and selective logging may negatively affect certain vertebrate species (Badillo-Saldaña et al., 2016; Beever and Brussard, 2004; Fredericksen and Fredericksen, 2002; Popescu et al., 2012), but in some cases these activities offer new vacant habitats that may be exploited by generalist species (Howell et al., 2019; Verschuyf et al., 2011).

Soon after a natural habitat is removed or degraded by human activities, habitat-specialist species will be the most prone to be locally extinct. Given their specific biological characteristics and requirements, specialist species are rarely able to adapt to drastic changes in their habitats, whereas generalist species could even benefit from these changes (Futuyma and Moreno, 1988; Östergård and Ehrlén, 2005). In fact, local increases in species richness a certain time after the occurrence of a drastic disturbance may result from the colonization of generalist species, as they are prone to take over disturbed ecosystems, at the expense of specialist ones (Blackburn et al., 2005). Thus, we may expect that altered areas are dynamically changing over time, as the colonization process gradually occurs. As a result, the time elapsed since the initial disturbance may modulate the magnitude of land-use change effects on species richness. Another relevant aspect that may affect the response of assemblage richness to land-use changes is the total pool of generalist and specialist species in the landscape, reflected by the gamma diversity. We may expect that higher-diverse communities would tend to lose more species than lower-diverse communities after a disturbance. Finally, given that the habitat characteristics determine the composition of the assemblages and the kind of prevalent land-use changes, we may find variations in the response of assemblages to land-use changes depending on the biome where the study was conducted.

Among vertebrates, amphibians and reptiles are considered the most susceptible groups to the changes imposed by human activities (Barrett and Guyer, 2008; Cushman, 2006; Wake and Vredenburg, 2008). In fact, amphibians and reptiles have a significantly higher proportion of threatened species than birds or mammals (IUCN, 2020; Nori et al., 2015; Roll et al., 2017; Stuart, 2004). Nevertheless, within these two groups there is high variability among species in their response patterns to human disturbances, and despite this, it has been classically considered that the response of both groups is similar in the face of these changes. Thus, depending on the type of disturbance, we may observe positive or negative effects on herpetofauna species richness and abundance (Adum et al., 2013; Cano and Leynaud, 2010; Guerra and Aráoz, 2015; Herrera-Montes and Brokaw, 2010; Karraker and Welsh, 2006; Leynaud and Bucher, 2005). All these factors combined make a challenging task to draw a general conclusion on how anthropogenic land-use changes are affecting the herpetofaunal communities. In part, this can be seen in previous reviews that have attempted to determine the overall effects of land-use changes on the abundance or richness of

amphibians and reptiles globally (e.g. Cushman, 2006; Gardner et al., 2007; Thompson et al., 2016). However, even these previous reviews have reported mixed results, making it difficult to attain reliable conclusions or generalizations about the effect of land-use changes on these vertebrate groups (Gardner et al., 2007; Scheffers and Paszkowski, 2013).

Thompson et al. (2016) synthesized the effects of four prevalent human-made land-use changes (urbanization, agriculture, cattle raising and silviculture) on the diversity of amphibian and reptile species. They gathered 132 studies assessing land-use change effects on richness or abundance of amphibian or reptile species, and used a vote-counting synthesis approach by comparing the relative frequency of studies showing negative, positive or null effects. However, the vote-counting technique has several flaws as a statistical procedure to synthesizing research, as it not only has low statistical power but also it fails to provide information on the overall results across the studies, which could lead to biased generalizations (Gurevitch and Hedges, 1999; Koricheva et al., 2017). Within the same study, Thompson et al. (2016) also run a meta-analytical approach by calculating response ratios as standardized measures (i.e., effect sizes) of species richness change from control versus disturbed conditions in a subsample of 39 studies. As a result of mixing up two different ways of approaching a systematic generalization, no clear response patterns raised from their review, as they explicitly express in their contribution. In the end, up to date, there is still no systematic and statistically powerful attempt to synthesize the impact of land-use changes, on amphibian and reptile species richness across different ecosystems around the world.

Quantitative statistical approaches such as meta-analysis, integrate and synthesize the available information across time and space from cumulated research, representing key tools to arrive at generalizations (Arnqvist and Wooster, 1995). Meta-analysis allows us to reach general conclusions about a domain of research, despite the apparent contradictory response patterns of individual studies (Gurevitch and Hedges, 1993). In this paper, we conduct an updated meta-analysis of the published literature, to assess the impact of different types of human land-use changes on amphibians and reptile's richness globally. More specifically, we determine: (I) the overall effects of the most prevalent human land-use changes (urbanization, agriculture, cattle raising, deforestation, selective logging and silviculture) on the richness of amphibian and reptile assemblages; (II) whether the time elapsed since the onset of land-use change affects the magnitude of amphibian and reptiles richness response; (III) how amphibian and reptile assemblages response to land-use changes is modified according to the total number of species (i.e. gamma diversity) present in a given area; and (IV) how the response of amphibians and reptiles to land-use changes varies according to different biomes.

## 2. Methods

### 2.1. Literature search and compilation of dataset

We conducted a systematic literature search aimed at recruiting all the studies that assessed the most prevalent anthropogenic land-use changes effects (namely: agriculture, cattle raising, urbanization, deforestation, silviculture and selective logging; based on Klein Goldewijk et al., 2011; Kissinger et al., 2012; Smith et al., 2014) on the richness of amphibian and reptile assemblages across the globe. To accomplish this, we used the following keyword combination: ("human disturbance" OR "anthropogenic disturbance" OR disturbance OR fragment\* OR "habitat loss" OR "land-use change" OR urban\* OR agricult\* OR farm\* OR deforest\* OR logging\* OR cattle\* OR grazing OR plantation\*) AND (amphibian\* OR reptil\* OR herpeto\*) AND (richness OR diversity OR biodiversity OR sensitivity OR community OR Simpson\* OR Shannon\* OR evenness OR Bayesian OR intactness OR "community composition" OR similarity OR correlation OR regression OR "linear model").

The search was done in two online scientific databases: ISI Web of Knowledge® and Scopus®, including all the articles published during the period between 1985 and 2018. Additionally, publications used in previous systematic reviews and meta-analyses covering similar topics were included (Cushman, 2006; Echeverría-Londoño et al., 2016; Gardner et al., 2007; Newbold et al., 2015; Thompson et al., 2016). We obtained 2047 studies that were examined to determine whether they met the conditions to be included in our meta-analysis. We excluded articles analyzing habitat degradation, without mentioning the specific type of land-use change.

Only those studies that reported the effects of land-use change on species richness as response variable were included in our analysis. In most cases, species richness was reported as the number of amphibians or reptile species present at each contrasting habitat condition sites (e.g., deforested vs. forested sites, fragmented vs continuous forests, etc.). In some cases, richness data had to be obtained from figures, tables or supplementary material. When two or more independent response records were obtained from the same study, we handled them as separate observations (Aguilar et al., 2006).

To assess whether the response of species richness varied depending on the time elapsed from the onset of the anthropogenic land-use change, we grouped the studies in two broad categories: (i) recent land-use change ranging from 0 to 20 years, and (ii) longer land-use change with more than 21 years since they occurred. As an additional measure, we consider the gamma diversity for each group as the total richness of species for both sites: disturbed and undisturbed -as defined by Hunter (2002)- as an indicator of the potential complexity that a landscape can support. Finally, given that the climatic and soil variables are actively conditioning the structure of species assemblages, we also considered the biome of each study site, using the World Earth Ecoregions model proposed by Olson et al. (2001).

## 2.2. Meta-analysis

We used *Hedges' d* as an estimate of the standardized mean difference (i.e. the effect size) that has the advantage of being unbiased by small sample size (Gurevitch et al., 2001). To calculate *Hedges' d*, we used the mean richness value, sample sizes and standard deviations from each of the two contrasting landscape conditions: the control (natural or preserved habitats) and the treatment (any habitat subjected to one of the above-mentioned human land-use changes). Negative *Hedges' d* effect sizes imply that species richness of amphibians or reptiles decreases in human-modified habitats.

We conducted hierarchical meta-analyses, which allowed us to incorporate in the overall effect size calculations, the hierarchical dependence in our data when multiple observations (i.e. effect sizes) were obtained from the same study. Thus, we included a publication-level random effect as a nesting factor to incorporate the hierarchical dependence of multiple outcomes within a study (Stevens and Taylor, 2009). The effects of land-use change were considered significant if the 95% biased-corrected bootstrap confidence intervals (CIs) of the effect size (*d*) did not overlap zero (Borenstein et al., 2009; Koricheva et al., 2017). The data were analyzed with a random effects model, assuming that differences between studies are due to sampling errors and also to true random variation among studies (Raudenbush, 2009). Heterogeneity among effect sizes was assessed with *Q* statistics, which are weighted sums of squares tested against a chi-square distribution (Borenstein et al., 2009). Specifically, we examined the *P* values of *QM* statistics that describe the variation in effect sizes that can be attributed to differences among categories of each predictor variable (i.e., fixed effects such as land-use change type, time elapsed since the onset of land-use change or biome of each study) in the model.

An intrinsic problem in any systematic quantitative review is the possibility of publication bias; i.e., studies showing significant results have a higher probability of being published. We explored the potential presence of publication bias in our dataset by assessing the relationship

between the effect sizes and sample sizes across the studies with Kendall's rank correlation test (Jennions et al., 2013). Significant *P*-values indicate the presence of publication bias whereby studies with small sample size are only published if they show large effect sizes. We also computed Rosenthal's fail-safe number, which calculates the number of non-significant, unpublished studies that need to be added to a meta-analysis to change its overall results from significant to non-significant. A fail-safe number is considered robust if it is larger than  $5n + 10$ , where *n* is the original number of studies included in the review (Jennions et al., 2013). Finally, we used the 'trim and fill' method as a sensitivity analysis that provides an estimate of how the overall effect size would change if we were able to incorporate all potentially missing studies (Møller and Jennions, 2002). All the analyses were conducted in R environment using the 'metafor' package (Viechtbauer, 2010; R Core Team, 2016).

## 3. Results

From the original database of 2047 studies, 94 of them met the requirements to be included in our analyses. Several studies provided two or more observations; therefore, we obtained a total of 133 observations (i.e., effect sizes), 83 for amphibians, and 50 for reptiles. The final list of the studies included in the meta-analysis is found in Supplementary Material: Dataset 1.

The geographic coverage of this meta-analysis comprised 37 countries across all the continents. However, most research up to now has been concentrated in the USA, Australia and Brazil, encompassing more than one half of the total observations (Fig. 1). In addition, there is a strong research bias regarding the analyzed biomes, with more than 75% of the studies conducted in temperate, tropical or subtropical forests (Fig. 2). Deforestation, silviculture and urbanization were the most frequently studied human land-use changes for both amphibian and reptile groups, encompassing 66% of the total observations (Fig. 2).

The overall effect of all land-use changes combined was significantly negative for the species richness of both groups, with a comparatively stronger negative overall effect observed in reptiles (Fig. 3). We found no significant heterogeneity of effects among land-use change types ( $QM = 6.545$ ,  $df = 5$ ,  $P = 0.257$  for amphibians; and  $QM = 1.244$ ,  $df = 5$ ,  $P = 0.941$  for reptiles), implying that all land-use changes analyzed have similar negative effects on amphibian and reptile richness. However, despite such homogeneity among them, some of these land-use changes had significant negative effects while others had negative but non-significant effect trends (i.e., 95% CIs overlapping zero effect size values). For amphibians, deforestation, silviculture, and urbanization showed significantly negative impacts on species richness. For reptiles, only cattle-raising and urbanization showed negative effects on species richness (Fig. 3).

The time elapsed since the onset of land-use change did not influence the magnitude of the negative effects on species richness, as observed from the non-significant heterogeneity between studies of more and less than 20 years for the two groups (amphibians:  $QM = 0.160$ ,  $df = 1$ ,  $P = 0.682$ ; reptiles:  $QM = 0.054$ ,  $df = 1$ ,  $P = 0.815$ ). Studies assessing impacts of land-use changes of less than 20 years showed slightly stronger negative effects for both reptiles and amphibians as compared to longer-term land-use changes (Fig. 4). Interestingly, after 21 or more years after land-use change took place, species richness continues to be negatively affected in both groups (Fig. 4).

When incorporating the total number of species within each studied regional community (i.e. gamma diversity) as a continuous predictor variable for the magnitude of anthropogenic effects on herps communities, we found a negative significant relationship for amphibians ( $\beta = -0.0124$ ;  $P$ -value = 0.028,  $N = 80$ ; Fig. 5). Such result implies that assemblages with larger number of amphibian species were proportionally more negatively affected by human disturbances than less-diverse assemblages. In contrast, gamma diversity of reptile communities did not influence the magnitude of land-use change effects on reptile richness ( $\beta$

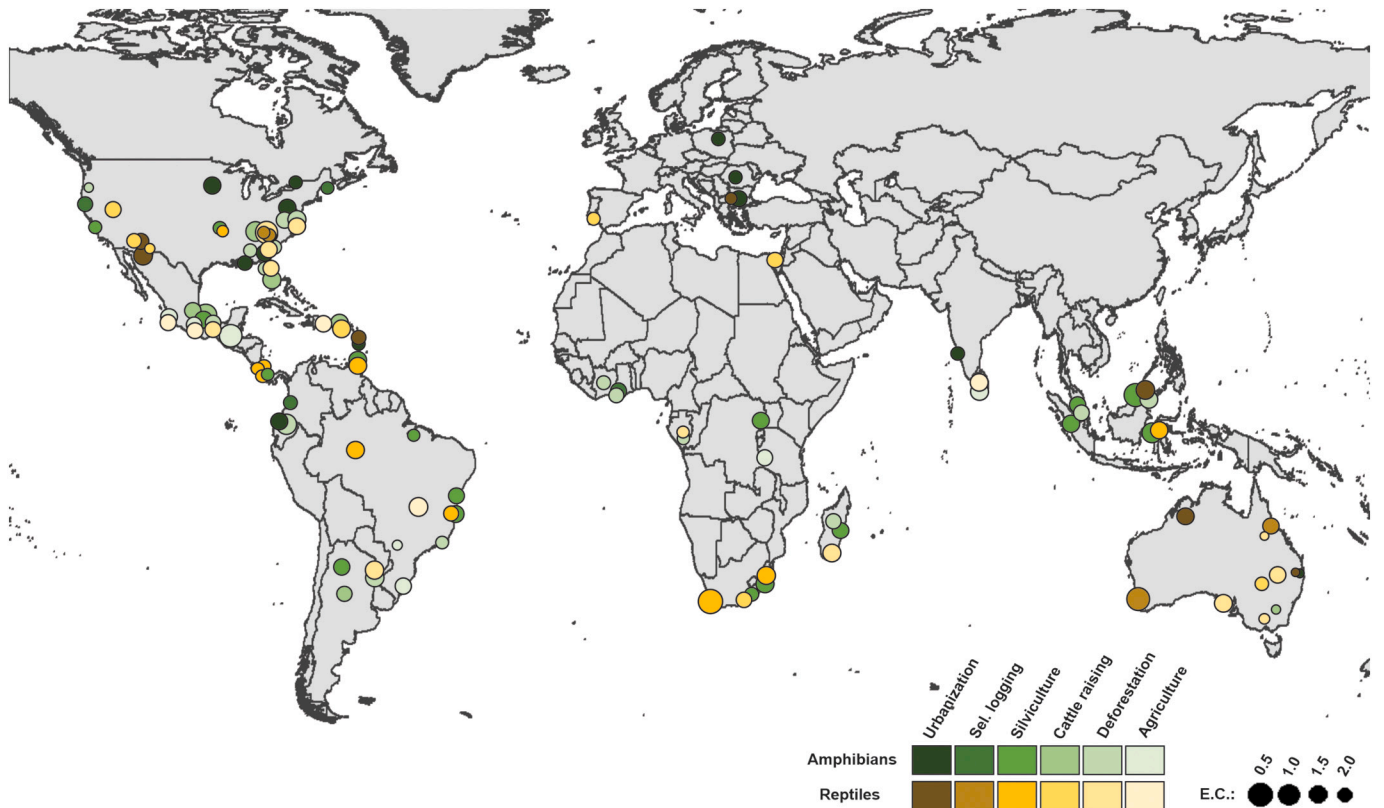


Fig. 1. Geographical coverage: each point indicates the geographical location of the records incorporated in the analyzes. The color of the dot represents both the vertebrate group, and the land-use category. The dot size represents the estimated confidence (E.C.) based on its sampling intensity. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

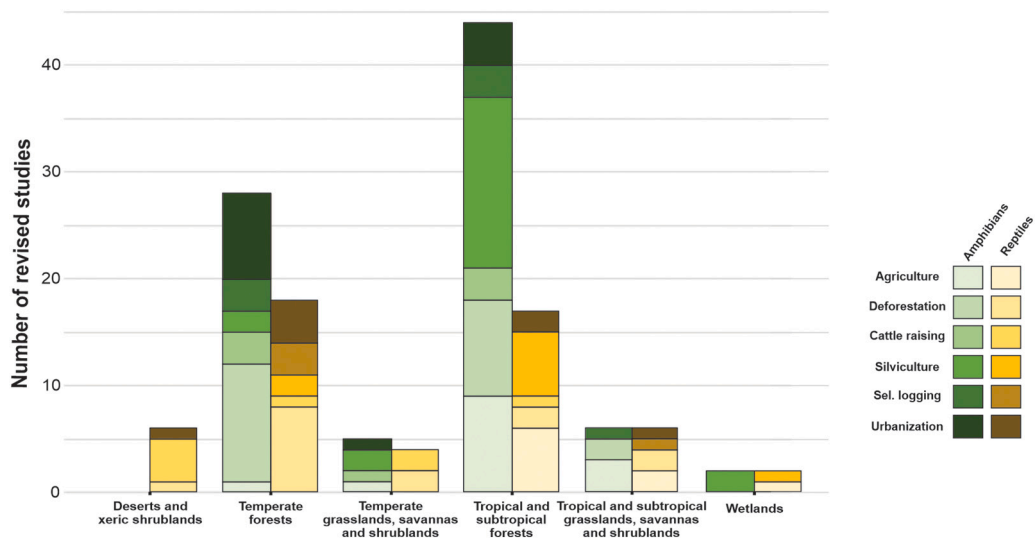
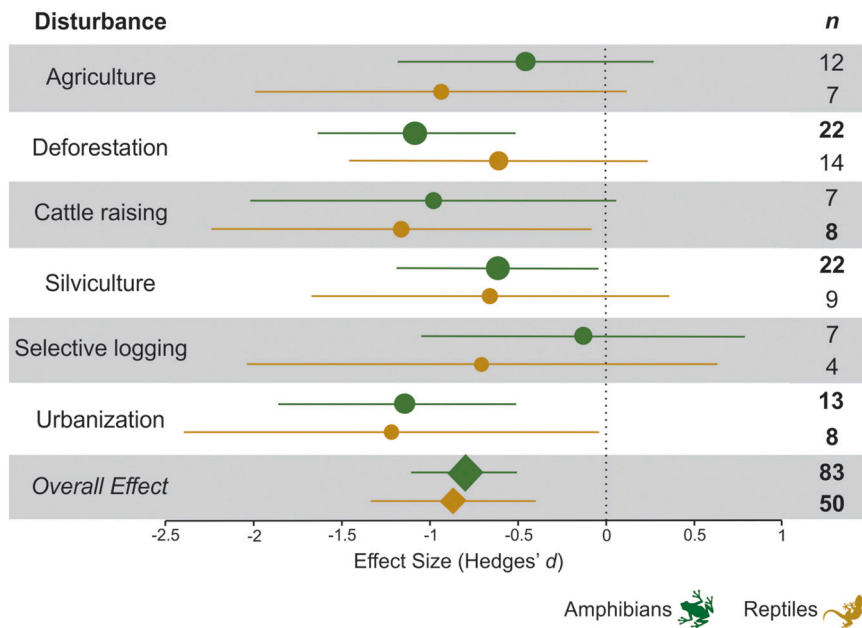


Fig. 2. Stacked histograms of the analyzed land-use changes, showing the overall number of records within each land-use category, separated by biome groups. The color references represent both the vertebrate group, and the land-use category. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

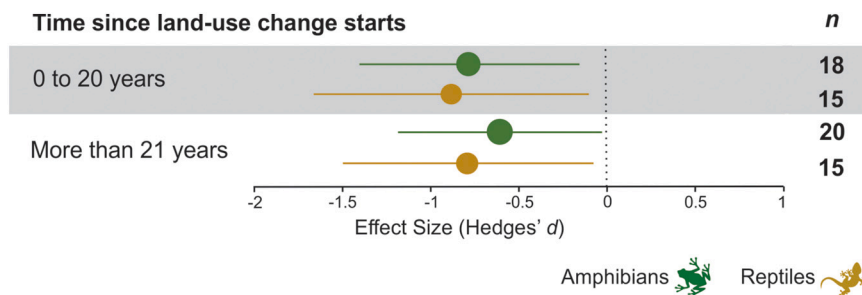
= -0.0011,  $P$ -value = 0.8963,  $N$  = 49; Fig. 5).

When studies were compared across biomes, we found significant heterogeneity in the effects of land-use changes among them ( $QM$  = 28.43,  $df$  = 5,  $P$  = 0.001 for amphibians;  $QM$  = 18.57,  $df$  = 6,  $P$  = 0.005 for reptiles). For amphibians, we found pervasive negative effects in Temperate forests, Tropical and subtropical forests and Tropical and subtropical grasslands, savannas and shrublands. In other two biomes

(Temperate grasslands, savannas and shrublands, and Wetlands) we also found similar but non-significant negative response patterns (i.e., 95% CIs overlapping zero effect size values (Fig. S1)). For reptiles, negative effects were found in Temperate forests, Temperate grasslands, savannas and shrublands, as well as in Tropical and subtropical forests, and in Tropical and subtropical grasslands, and savannas and shrublands, and a slight positive non-significant effect for Wetlands. Despite the low



**Fig. 3.** Overall weighted-mean effect sizes and 95% bias-corrected confidence intervals of land-use changes on amphibians' and reptiles' assemblages. Sample sizes of each category are represented by *n*. The dashed vertical line shows Hedge's *d* = 0.



**Fig. 4.** Overall weighted-mean effect sizes and 95% bias-corrected confidence intervals of time since the start of land-use changes on amphibians' and reptiles' assemblages. Sample sizes of each category are represented by *n*. The dashed vertical line shows Hedge's *d* = 0.

number of studies to draw any conclusion, reptiles inhabiting in hot and dry deserts and xeric shrublands appear to be strongly negatively affected by the anthropic changes (Fig. S1).

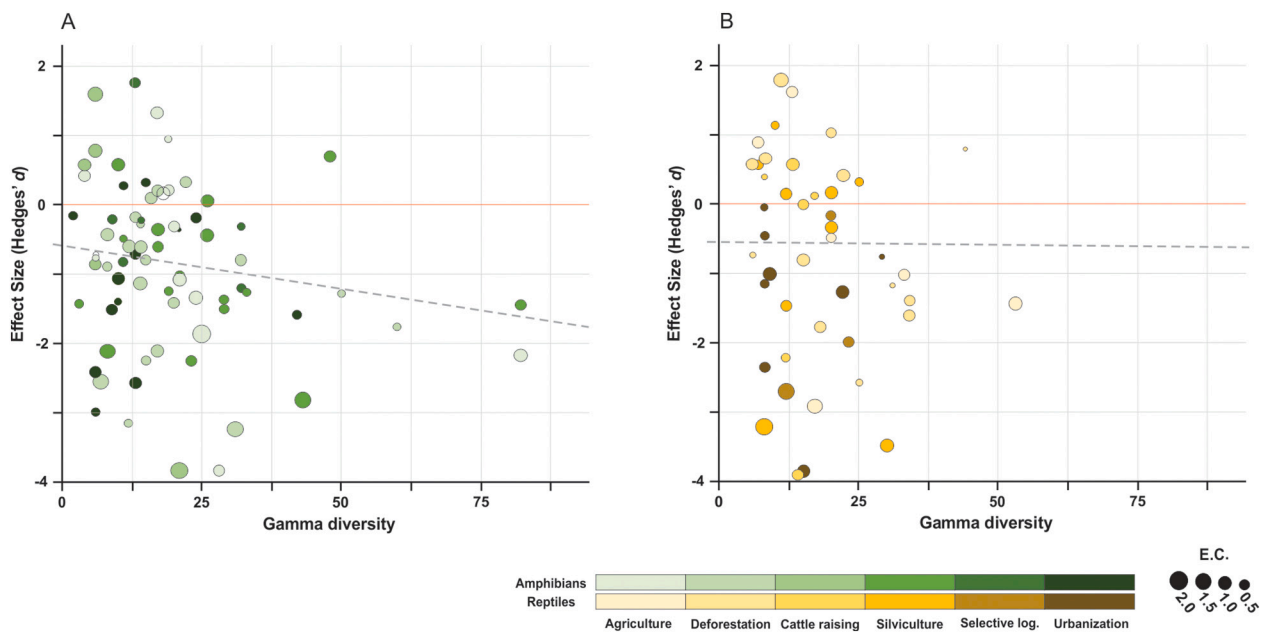
The rank correlation tests between sample size and effect sizes indicated no potential presence of publication bias in our dataset (Table S1). Also, the calculated fail-safe numbers were always higher than  $5n + 10$ , indicating that the overall results obtained here are robust, regardless of publication bias (Amphibians  $2135 > 435$ ; Reptiles  $413 > 265$ ). Finally, the corrections introduced by the Trimm and Fill method determined that the addition of new studies is not necessary to achieve correction of the potential bias (Table S1). Thus, our systematic research synthesis is not influenced by publication bias and our results capture and reflect the nature of the phenomenon appropriately. The comprehensive statistics summary for the performed analyzes is presented in Table S2.

#### 4. Discussion

Our results provide empirical evidence to the fact that most amphibians and reptile species are strongly affected by land-use change, causing significant local extinction in species assemblages. While amphibians and reptiles are commonly treated as one group (the herpetofauna), they have many differences such as habitat requirements, life histories, and even different evolutionary origins, which imply

potentially different responses to land-use changes. Here, we observed that amphibian species richness was adversely affected by deforestation, silviculture and urbanization, whereas reptile richness significantly decreased with cattle raising and urbanization. The remaining human land-use change types assessed here showed negative trends in species for both groups. These non-significant trends are likely due to the lower number of studies (i.e. meta-analytical replicates) of these types of disturbances, which imply low statistical power.

Urbanization was the only land-use change showing a strong negative effect on species richness in both groups (Fig. 3). Urbanization probably represents the most drastic human land-use change type, as it leads to a dramatic transformation and simplification of natural habitats, where there is not only loss of suitable habitat in terms of resources and shelter, but also large changes in abiotic micro environmental conditions. For example, compared to natural habitats, urbanizations show increased noise and air pollution as well as increased soil temperature, luminosity, and decreased humidity, conditions that are extended through the night by artificial illumination (e.g. Brasfield et al., 2004; Neuman-Lee et al., 2015; Sievers et al., 2019; Snodgrass et al., 2008). All these aspects imply that only a small number of amphibians and reptile species are able to thrive in these new urban conditions (Cornelis and Hermy, 2004; Helden and Leather, 2004; McKinney, 2008). Urbanization is also known to be a driver to other threats, such as the introduction of non-native species (Bellard et al.,



**Fig. 5.** Meta-regressions between gamma diversity and land-use change effect sizes (Hedges'  $d$ ) on species richness. Red horizontal line shows Hedge's  $d = 0$ . Dashed line represents the slope of meta-regression. E.C.: estimated confidence. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

2016), the second most significant threat for biodiversity in continents and the first one in islands (Spatz et al., 2017). Invasive species can directly affect native amphibian and reptile species by competing for resources and by the effects of diseases (Doan et al., 2019).

In the case of amphibians, deforestation showed a similar strong and negative response as urbanization (Fig. 3). Such similar negative effects are not surprising as clear-cutting resembles many aspects of urbanization. Deforestation alters the structure and function of forests. The removal of the canopy exposes the lower layers of the forest to increased sunlight and wind (Semlitsch et al., 2009), generating warmer and dryer surfaces (Zheng et al., 2000), all of which reduces leaf litter dynamics and food resources for amphibians. In general, these drastic changes imply high mortality of most native organisms (Rittenhouse et al., 2008; Todd and Andrews, 2008), and amphibians are particularly susceptible considering their complex life cycle. Thus, worryingly, when a drastic change like deforestation occurs, these regions cannot be recolonized (Graeter et al., 2008; Todd and Andrews, 2008). Similarly, silviculture is also a harmful practice for amphibians (Fig. 3). Silviculture is a heterogeneous practice, which can be highly variable regarding its intensity and thus it is rather challenging to analyze. Additionally, the management practices required to develop silvicultural activities are incredibly harmful to most taxa (Riffell et al., 2011) and no less for amphibians (Haggerty et al., 2019; Rittenhouse et al., 2008). In general, these management practices include timber harvest and clearcutting (Agee and Skinner, 2005) affecting microhabitat features, such as leaf litter depth, shading, and coarse woody debris, among other indispensable factors for the subsistence of native amphibian species (Riffell et al., 2011). As in the case of deforestation, once microclimatic conditions are altered, the recolonization success of these organisms is highly unlikely (Fig. 4). In short, both deforestation and silviculture can drastically reduce the formation and permanence of ponds and other temporary watercourses (Koralay and Kara, 2018; Panday et al., 2015) that are fundamental for amphibians. Unfortunately, amphibian species inhabiting the few remaining pristine or undisturbed areas are also affected by other threats such as chytridiomycosis, the most important emerging disease in amphibians, which is mainly dispersed by humans (O'Hanlon et al., 2018) and is causing several species and population extinctions worldwide (Fisher et al., 2009). Paradoxically, while chytridiomycosis is

spread by humans, its highest prevalence in amphibians has been observed in well-preserved areas (Becker and Zamudio, 2011), posing a major threat to highly susceptible amphibian species that are unable to cope or survive in human-disturbed habitats.

Similarly, species richness of reptiles showed negative effect trends of deforestation and silviculture. Reptiles have particular adaptations or life-history traits (e.g. low evapotranspiration rate, low caloric consumption, tolerance to food shortages, etc.), which could confer them certain tolerance to specific types of land-use changes, and thus they are not expected to respond as amphibians do to the same types of land-use changes. Nevertheless, the negative but non-significant trends observed here are likely the result of reduced statistical power, due to the low number of studies assessing deforestation and silviculture effects on reptile richness. Thus, we stress the need of increasing research on these human disturbance factors on reptile communities.

In contrast, cattle-raising showed strong significant negative effects on reptile richness, despite the low number of replicates (Fig. 3). Reptiles commonly use vegetation as a refuge to evade predators (Castellano and Valone, 2006), foraging, nesting, and thermoregulation sites. Under cattle-raising pressure, the vegetation cover decreases, limiting the availability of prey, and increasing the probability of being predated. Additionally, large proportions of bare soil are exposed, thus generating high temperatures at ground-level, that affect reptile species by impairing thermoregulation (Nowakowski et al., 2018). This deviation from its thermal optima will reduce the efficiency of foraging, predator escape and reproduction (Huey et al., 2009). This is likely to be exacerbated in some biomes, like deserts and xeric shrublands, where there are continuous overgrazing activities (Pfeiffer et al., 2019; Sandhage-Hofmann et al., 2015), causing a drastic reduction of the already scarce vegetation cover. Reptile species from hot and dry deserts and xeric shrublands are highly dependent on their possibilities of thermoregulation (availability of light/shadow), to adjust their activity periods and to prevent water loss (Sinervo et al., 2010). In short, the reptile assemblages are sensitive to cattle-raising impacts, especially in habitats conditioned by low rainfall and high temperatures. While this activity also affects amphibians (Fig. 3), cattle-raising may increase the availability of nesting sites (e.g. ponds for watering cattle) for amphibians, showing in some cases a compensatory effect (Knutson et al., 2004),

which could explain the lesser effect in this last group.

We found overall negative but non-significant effects of agriculture on species richness of both groups (Fig. 3). Such lack of significance is mostly due to the yet low number of studies assessing this type of disturbance (Fig. 2). However, there may be some “profitable” aspects for herps in certain agricultural environments, which should be highlighted. For example, farming practices -especially those traditionally managed- generate human-made ponds, which can provide an adequate environment for many amphibian species (Hartel et al., 2010; Knutson et al., 2004; Lescano et al., 2015). For reptiles, some evidence indicates that species richness of assemblages is not significantly modified in agricultural landscapes (Suazo-Ortuño et al., 2008), and such response maybe related to a potential increase in the abundance of prey for reptiles (Rotem et al., 2013).

Similarly, selective logging showed no significant effects on the richness of both groups, and nearly a nil effect for amphibians. In concordance, a meta-analysis of Verschuyf et al. (2011) pointed out a neutral and positive impact of selective logging on amphibians. Although this kind of land-use change is highly variable on its intensity, the removal of a limited number of trees per hectare would not have significant effects on soil’s microclimatic conditions. Also, selective logging can change soil moisture conditions favoring amphibian species, by enhancing the productivity of herbaceous and undergrowth forest (Zheng et al., 2000). It was observed that some types of selective logging are compatible with the maintenance of reptile diversity (Russell et al., 2004; Todd and Andrews, 2008). For example, forest thinning can benefit many reptile species, which are not adapted to closed-canopy forest nor clear-cut areas (Todd and Andrews, 2008). In summary, some selective logging practices can be compatible with the maintenance and permanence of the amphibian and reptile assemblages (Greenberg and Waldrop, 2008; Semlitsch et al., 2009), and in some cases may even produce local increases of species richness (Ofori-Boateng et al., 2013). However, due to the scarcity of studies assessing logging effects on amphibian and reptile communities, we cannot draw any robust conclusions. Increasing studies on these effects should help us understand their relative influence on herpetofauna.

To investigate whether species richness can be recovered in altered areas across time, it is important to analyze the impact of these human-made alterations within a temporal frame. Our findings showed that the time elapsed after the onset of land-use change does not alter the magnitude of negative effects on species richness in both amphibians and reptiles (Fig. 4). There was only a slight trend of lower effect size magnitude in studies where land-use changes have occurred over more than two decades (Fig. 4). This same pattern of long-lasting impacts was also observed by Haddad et al. (2015) for global biodiversity patterns. In general, human-altered habitats show larger losses of habitat specialist species and the permanence of habitat generalist species (Thompson et al., 2016). A relative increase of habitat generalist species of wider distribution, generates a noticeable impact regarding the conservation value of these areas, beyond the general loss in species richness.

The response of each community to land-use changes was highly variable depending on the biome where the study was conducted. Such a variable response among climatic biomes may be initially correlated to the climatic and topological conditions that also favor the development of certain human activities. Not surprisingly, despite their relative low number of studies, those biomes with the greater aptitude for intensive agriculture -specifically tropical and subtropical grasslands, savannas, and shrublands- also showed greater negative effects for amphibians and reptiles. On the other hand, all forest biomes, which hold the highest richness and abundance of amphibians and reptiles and are therefore much more studied, showed a more consistent negative response.

In agreement to this, we observed that richer amphibian communities showed larger negative effects of human disturbances. The meta-regressions reveal a significant negative relationship between gamma diversity and the response to land-use changes for amphibians, increasing the magnitude of negative land-use change effects on species

richness as the number of species in the assemblage increase (Fig. 5-A). Richer assemblages (those with a gamma diversity >75 species) typically belong to rainforest biomes, where the percentage of habitat-specialist species is very high (Vallan, 2002). Given the lower capacity of habitat-specialist species to tolerate human disturbances, these assemblages could be particularly sensitive to land-use changes reflecting a high conservation value (Wanger et al., 2010; Faruk et al., 2013). On the other hand, we found that gamma diversity did not influence the magnitude of land-use changes effects for reptiles: all assemblages seem to be negatively affected by land-use changes regardless of the number of species they have (Fig. 5-B). These differential responses must be taken into account when dealing with conservation planning strategies for both groups. Our results suggest that all forest biomes (tropical, subtropical, and temperate) represent the most susceptible biomes for the conservation of amphibians. Finally, the important but under represented biomes for reptiles such as desert and xeric shrublands, showed the most extreme negative responses (Fig. S1). Therefore, our review highlights the importance of increasing research on these poorly studied biomes, to determine their relative susceptibility to land-use changes on reptiles. Should the patterns found here prevail, immediate conservation actions are needed before it is too late.

This quantitative synthesis represents an important step toward understanding the current status of amphibian and reptile communities under widespread land-use changes around the world. One of them is related with the extensive cattle-raising in deserts and xeric shrublands (Attum et al., 2006; Kutt and Woinarski, 2007). Since cattle raising generally involves large land extensions, it is essential to increase research on these areas and plan strategies on the conservation of desert and xeric shrublands reptile communities, considering endemism centers and micro endemic distributions, to minimize the impact of these local activities on this important and restricted conservation sites. Additionally, we highlight that despite not finding a general significant effect of agricultural practices (Fig. 3), there were negative effects of land-use changes in those biomes where this activity predominates: grasslands, shrubs and savannas (Fig. 2, Fig. S1). Likewise, it is also necessary to develop local policies to encourage other activities (e.g. selective logging) that are more compatible with the maintenance of local assemblages of herps (Fig. 3).

Our findings also underline that much more additional research is still needed. There are entirely political and eco-geographical regions that are not represented in our synthesis such as central and northern Africa or central and northern Asia, (Fig. 1, Fig. S1), evidencing important gaps of knowledge. This type of research bias (sensu Gurevitch and Hedges, 1999), in which particular biomes are more frequently selected by different authors represents a problem when trying to attain broad generalizations with meta-analysis, as it circumscribes the response patterns to the particular biomes analyzed. We strongly recommend increasing research in non-forests biomes (grasslands, shrublands, savannas, wetlands and deserts). Moreover, in general both reptiles and amphibians still continue to be understudied in the conservation literature (Bonnet et al., 2002). In this regard, our results are also evidence of the yet underestimated great repercussions of knowledge shortfall for biodiversity conservation (Diniz-Filho et al., 2013; Hortal et al., 2015).

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.biocon.2020.108863>.

#### Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

#### Acknowledgements

AB, DM and JMC and research is possible thanks to CONICET

doctoral grants; AB, DM, JMC, JN and JL research is funded by SECyT (336 201801 01120 CB); JL research is also funded by FONCyT (PICT 2017-2666); RL research is funded by CNPq (grant #306694/2018-2). This paper is a contribution of the INCT in Ecology, Evolution and Biodiversity Conservation founded by MCTIC/CNPq/FAPEG (grant 465610/2014-5); RA research is funded by FONCyT (PICT 2016-0764) and CONICET (PIP 2015-0371).

## References

- Adum, G.B., Eichhorn, M.P., Oduro, W., Ofori-Boateng, C., Rödel, M.O., 2013. Two-stage recovery of amphibian assemblages following selective logging of tropical forests. *Conserv. Biol.* 27, 354–363. <https://doi.org/10.1111/cobi.12006>.
- Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. *For. Ecol. Manag.* 211, 83–96. <https://doi.org/10.1016/j.foreco.2005.01.034>.
- Aguilar, R., Ashworth, L., Galetto, L., Aizen, M.A., 2006. Plant reproductive susceptibility to habitat fragmentation: review and synthesis through a meta-analysis. *Ecol. Lett.* 9, 968–980. <https://doi.org/10.1111/j.1461-0248.2006.00927.x>.
- Arnqvist, G., Wooster, D., 1995. Meta-analysis: synthesizing research findings in ecology and evolution. *Trends Ecol. Evol.* 10, 236–240. [https://doi.org/10.1016/S0169-5347\(00\)89073-4](https://doi.org/10.1016/S0169-5347(00)89073-4).
- Attum, O., Eason, P., Cobbs, G., Baha El Din, S.M., 2006. Response of a desert lizard community to habitat degradation: do ideas about habitat specialists/generalists hold? *Biol. Conserv.* 133, 52–62. <https://doi.org/10.1016/j.biocon.2006.05.017>.
- Badillo-Saldaña, L.M., Ramírez-Bautista, A., Wilson, L.D., 2016. Effects of establishment of grazing areas on diversity of amphibian communities in tropical evergreen forests and mountain cloud forests of the Sierra Madre Oriental. *Rev. Mex. Biodivers.* 87, 133–139. <https://doi.org/10.1016/j.RMB.2015.09.019>.
- Barrett, K., Guyer, C., 2008. Differential responses of amphibians and reptiles in riparian and stream habitats to land use disturbances in western Georgia. *USA. Biol. Conserv.* 141, 2290–2300. <https://doi.org/10.1016/j.biocon.2008.06.019>.
- Becker, C.G., Zamudio, K.R., 2011. Tropical amphibian populations experience higher disease risk in natural habitats. *Proc. Natl. Acad. Sci.* 108, 9893–9898. <https://doi.org/10.1073/pnas.1014497108>.
- Beever, E.A., Brussard, P.F., 2004. Community- and landscape-level responses of reptiles and small mammals to feral-horse grazing in the Great Basin. *J. Arid Environ.* 59, 271–297. <https://doi.org/10.1016/j.jaridenv.2003.12.008>.
- Bellard, C., Cassey, P., Blackburn, T.M., 2016. Alien species as a driver of recent extinctions. *Biol. Lett.* 12, 20150623. <https://doi.org/10.1098/rsbl.2015.0623>.
- Blackburn, T.M., Petchey, O.L., Cassey, P., Gaston, K.J., 2005. Functional diversity of mammalian predators and extinction in island birds. *Ecology* 86, 2916–2923. <https://doi.org/10.1890/04-1847>.
- Bonnet, X., Shine, R., Lourda, O., 2002. Taxonomic chauvinism. *Trends Ecol. Evol.* 17, 1–3. [https://doi.org/10.1016/S0169-5347\(01\)02381-3](https://doi.org/10.1016/S0169-5347(01)02381-3).
- Borenstein, M., Hedges, L.V., Higgins, J.P.T., Rothstein, H.R., 2009. Introduction to Meta-analysis, Introduction to Meta-analysis. John Wiley and Sons, Chichester, UK. <https://doi.org/10.1002/9780470743386>.
- Brasfield, S.M., Bradham, K., Wells, J.B., Talent, L.G., Lanno, R.P., Janz, D.M., 2004. Development of a terrestrial vertebrate model for assessing bioavailability of cadmium in the fence lizard (*Sceloporus undulatus*) and in ovo effects on hatchling size and thyroid function. *Chemosphere* 54, 1643–1651. <https://doi.org/10.1016/j.chemosphere.2003.09.030>.
- Cano, P.D., Leynaud, G.C., 2010. Effects of fire and cattle grazing on amphibians and lizards in northeastern Argentina (Humid Chaco). *Eur. J. Wildl. Res.* 56, 411–420. <https://doi.org/10.1007/s10344-009-0335-7>.
- Castellano, M.J., Valone, T.J., 2006. Effects of livestock removal and perennial grass recovery on the lizards of a desertified arid grassland. *J. Arid Environ.* 66, 87–95. <https://doi.org/10.1016/J.JARIDENV.2005.10.016>.
- Ceballos, G., Ehrlich, P.R., Barnosky, A.D., Garcia, A., Pringle, R.M., Palmer, T.M., 2015. Accelerated modern human-induced species losses: entering the sixth mass extinction. *Sci. Adv.* 1, e1400253. <https://doi.org/10.1126/sciadv.1400253>.
- Cornelis, J., Hermy, M., 2004. Biodiversity relationships in urban and suburban parks in Flanders. *Landsc. Urban Plan.* 69, 385–401. <https://doi.org/10.1016/j.landurbplan.2003.10.038>.
- Cushman, S.A., 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. *Biol. Conserv.* 128, 231–240. <https://doi.org/10.1016/j.biocon.2005.09.031>.
- Davidson, N.C., 2014. How much wetland has the world lost? Long-term and recent trends in global wetland area. *Mar. Freshw. Res.* 65, 934. <https://doi.org/10.1071/MF14173>.
- Diniz-Filho, J.A.F., Loyola, R.D., Raia, P., Mooers, A.O., Bini, L.M., 2013. Darwinian shortfalls in biodiversity conservation. *Trends Ecol. Evol.* 28, 689–695. <https://doi.org/10.1016/j.tree.2013.09.003>.
- Doan, T.M., Devlin, B.G., Greene, K.C., 2019. Malaria infection is lower in invasive anoles than native anoles in Central Florida. *USA. J. Herpetol.* 53, 22. <https://doi.org/10.1670/18-056>.
- Echeverría-Londoño, S., Newbold, T., Hudson, L.N., Contu, S., Hill, S.L.L., Lysenko, I., Arbeláez-Cortés, E., Armbrecht, I., Boekhout, T., Cabra-García, J., Dominguez-Haydar, Y., Nates-Parra, G., Gutiérrez-Lamus, D.L., Higuera, D., Isaacs-Cubides, P.J., López-Quintero, C.A., Martínez, E., Miranda-Esquivel, D.R., Navarro-Iriarte, L.E., Noriega, J.A., Otavo, S.E., Parra-H, A., Poveda, K., Ramírez-Pinilla, M.P., Rey-Velasco, J.C., Rosselli, L., Smith-Pardo, A.H., Urbina-Cardona, J.N., Purvis, A., 2016. Modelling and projecting the response of local assemblage composition to land use change across Colombia. *Divers. Distrib.* 22, 1099–1111. <https://doi.org/10.1111/ddi.12478>.
- Faruk, A., Belabut, D., Ahmad, N., Knell, R.J., Garner, T.W., 2013. Effects of oil-palm plantations on diversity of tropical anurans. *Conserv. Biol.* 27, 615–624. <https://doi.org/10.1111/cobi.12062>.
- Fisher, M.C., Garner, T.W., Walker, S.F., 2009. Global emergence of Batrachochytrium dendrobatidis and amphibian chytridiomycosis in space, time, and host. *Annu. Rev. Microbiol.* 63, 291–310. <https://doi.org/10.1146/annurev.micro.091208.073435>.
- Flynn, D.F.B., Gogol-Prokurat, M., Nogeire, T., Molinari, N., Richers, B.T., Lin, B.B., Simpson, N., Mayfield, M.M., DeClerck, F., 2009. Loss of functional diversity under land use intensification across multiple taxa. *Ecol. Lett.* 12, 22–33. <https://doi.org/10.1111/j.1461-0248.2008.01255.x>.
- Fredericksen, N.J., Fredericksen, T.S., 2002. Terrestrial wildlife responses to logging and fire in a Bolivian tropical humid forest. *Biodivers. Conserv.* 11, 27–38. <https://doi.org/10.1023/A:1014065510554>.
- Futuyma, D.J., Moreno, G., 1988. The evolution of ecological specialization. *Annu. Rev. Ecol. Syst.* 19, 207–233. <https://doi.org/10.1146/annurev.es.19.110188.001231>.
- Gagné, S.A., Fahrig, L., 2007. Effect of landscape context on anuran communities in breeding ponds in the National Capital Region, Canada. *Landsc. Ecol.* 22, 205–215. <https://doi.org/10.1007/s10980-006-9012-3>.
- Gardner, T.A., Barlow, J., Peres, C.A., 2007. Paradox, presumption and pitfalls in conservation biology: the importance of habitat change for amphibians and reptiles. *Biol. Conserv.* 138, 166–179. <https://doi.org/10.1016/j.biocon.2007.04.017>.
- Glor, R.E., Flecker, A.S., Benard, M.F., Power, A.G., 2001. Lizard diversity and agricultural disturbance in a Caribbean forest landscape. *Biodivers. Conserv.* 10, 711–723. <https://doi.org/10.1023/A:1016665011087>.
- Graeter, G.J., Rothermel, B.B., Gibbons, J.W., 2008. Habitat selection and movement of pond-breeding amphibians in experimentally fragmented pine forests. *J. Wildl. Manag.* 72, 473–482. <https://doi.org/10.2193/2006-330>.
- Greenberg, C.H., Waldrop, T.A., 2008. Short-term response of reptiles and amphibians to prescribed fire and mechanical fuel reduction in a southern Appalachian upland hardwood forest. *For. Ecol. Manag.* 255, 2883–2893.
- Guerra, C., Aráoz, E., 2015. Amphibian diversity increases in an heterogeneous agricultural landscape. *Acta Oecol.* 69, 78–86. <https://doi.org/10.1016/J.ACTAO.2015.09.003>.
- Gurevitch, J., Hedges, L.V., 1993. Meta-analysis: combining the results of independent experiments. In: Scheiner, S., Gurevitch, J. (Eds.), *Design and Analysis of Ecological Experiments*. Chapman and Hall.
- Gurevitch, J., Hedges, L.V., 1999. Statistical issues in ecological meta-analyses. *Ecology* 80, 1142–1149. [https://doi.org/10.1890/0012-9658\(1999\)080\[1142:SIEMA\]2.0.CO;2](https://doi.org/10.1890/0012-9658(1999)080[1142:SIEMA]2.0.CO;2).
- Gurevitch, J., Curtis, P.S., Jones, M.H., 2001. Meta-analysis in ecology. *Adv. Ecol. Res.* 32, 199–247. [https://doi.org/10.1016/S0065-2504\(01\)32013-5](https://doi.org/10.1016/S0065-2504(01)32013-5).
- Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R.M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., Melbourne, B.A., Nicholls, A.O., Orrock, J.L., Song, D.-X., Townshend, J.R., 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. *Sci. Adv.* 1, e1500052. <https://doi.org/10.1126/sciadv.1500052>.
- Haggerty, C.J.E., Crisman, T.L., Rohr, J.R., 2019. Effects of forestry-driven changes to groundwater and soil moisture on amphibian desiccation, dispersal, and survival. *Ecol. Appl.* e01870. <https://doi.org/10.1002/eap.1870>.
- Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stepanov, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O., Townshend, J.R.G., 2013. High-resolution global maps of 21st-century forest cover change. *Science* 342 (80), 850–853. <https://doi.org/10.1126/science.1244693>.
- Hartel, T., Schweiger, O., Öllerer, K., Cogălniceanu, D., Arntzen, J.W., 2010. Amphibian distribution in a traditionally managed rural landscape of Eastern Europe: probing the effect of landscape composition. *Biol. Conserv.* 143, 1118–1124. <https://doi.org/10.1016/J.BIOCON.2010.02.006>.
- Helden, A.J., Leather, S.R., 2004. Biodiversity on urban roundabouts—Hemiptera, management and the species-area relationship. *Basic Appl. Ecol.* 5, 367–377. <https://doi.org/10.1016/J.BAAE.2004.06.004>.
- Herrera-Montes, A., Brokaw, N., 2010. Conservation value of tropical secondary forest: a herpetofaunal perspective. *Biol. Conserv.* 143, 1414–1422. <https://doi.org/10.1016/J.BIOCON.2010.03.016>.
- Hodgkinson, S.C., Hero, J.-M., Warnken, J., 2007. The conservation value of suburban golf courses in a rapidly urbanising region of Australia. *Landsc. Urban Plan.* 79, 323–337. <https://doi.org/10.1016/j.landurbplan.2006.03.009>.
- Hoekstra, J.M., Boucher, T.M., Ricketts, T.H., Roberts, C., 2004. Confronting a biome crisis: global disparities of habitat loss and protection. *Ecol. Lett.* 8, 23–29. <https://doi.org/10.1111/j.1461-0248.2004.00686.x>.
- Hooke, R.L.B., Martin-Duque, J.F., Pedraza, J., 2012. Land transformation by humans: A review. *GSA Today* 22, 4–10. <https://doi.org/10.1130/GSAT151A.1>.
- Hortal, J., de Bello, F., Diniz-Filho, J.A.F., Lewinsohn, T.M., Lobo, J.M., Ladle, R.J., 2015. Seven shortfalls that beset large-scale knowledge of biodiversity. *Annu. Rev. Ecol. Syst.* 46, 523–549. <https://doi.org/10.1146/annurev-ecolsys-112414-054400>.
- Howell, H.J., Mothes, C.C., Clements, S.L., Catania, S.V., Rothermel, B.B., Searcy, C.A., 2019. Amphibian responses to livestock use of wetlands: new empirical data and a global review. *Ecol. Appl.* 29. <https://doi.org/10.1002/eap.1976>.
- Huey, R.B., Deutsch, C.A., Tewksbury, J.J., Vitt, L.J., Hertz, P.E., Álvarez Pérez, H.J., Garland, T., 2009. Why tropical forest lizards are vulnerable to climate warming. *Proc. R. Soc. B Biol. Sci.* 276, 1939–1948. <https://doi.org/10.1098/rspb.2008.1957>.
- Hunter, M., 2002. *Fundamentals of Conservation Biology*, 2nd edn. Blackwell Science, Malden, MA, pp. 448–452.



- IUCN, 2015. The IUCN Red List of Threatened Species. International Union for Conservation of Nature and Natural Resources. <https://www.iucnredlist.org>.
- IUCN, 2020. The IUCN Red List of Threatened Species. International Union for Conservation of Nature and Natural Resources. <https://www.iucnredlist.org>.
- Jantz, P., Goetz, S., Laporte, N., 2014. Carbon stock corridors to mitigate climate change and promote biodiversity in the tropics. *Nat. Clim. Chang.* 4, 138–142. <https://doi.org/10.1038/nclimate2105>.
- Jennions, M.D., Lortie, C.J., Rosenberg, M.S., Rothstein, H.R., 2013. Publication and related biases. In: *Handbook of Meta-analysis in Ecology and Evolution*, pp. 207–236.
- Karraker, N.E., Welsh, H.H., 2006. Long-term impacts of even-aged timber management on abundance and body condition of terrestrial amphibians in Northwestern California. *Biol. Conserv.* 131, 132–140. <https://doi.org/10.1016/j.biocon.2006.02.013>.
- Kissinger, G., Herold, M., De Sy, V., 2012. Drivers of Deforestation and Forest Degradation. A Synthesis Report for REDD+ Policymakers. Lexeme Consulting.
- Klein Goldewijk, K., Beusen, A., Van Drecht, G., De Vos, M., 2011. The HYDE 3.1 spatially explicit database of human-induced global land-use change over the past 12,000 years. *Glob. Ecol. Biogeogr.* 20, 73–86. <https://doi.org/10.1111/j.1466-8238.2010.00587.x>.
- Knutson, M.G., Richardson, W.B., Reineke, D.M., Gray, B.R., Parmelee, J.R., Weick, S.E., 2004. Agricultural ponds support amphibian populations. *Ecol. Appl.* 14, 669–684. <https://doi.org/10.1890/02-5305>.
- Koralay, N., Kara, Ö., 2018. Forestry activities and surface water quality in a rainfall watershed. *Eur. J. for. Eng.* 4, 70–82. <https://doi.org/10.33904/ejfe.438621>.
- Koricheva, J., Gurevitch, J., Mengersen, K.L., 2017. Handbook of Meta-analysis in Ecology and Evolution, Handbook of Meta-analysis in Ecology and Evolution. Princeton University Press. <https://doi.org/10.23943/princeton/9780691137285.001.0001>.
- Kutt, A.S., Woinarski, J.C.Z., 2007. The effects of grazing and fire on vegetation and the vertebrate assemblage in a tropical savanna woodland in north-eastern Australia. *J. Trop. Ecol.* 23, 95–106. <https://doi.org/10.1017/S0266467406003579>.
- Kutt, A.S., Vanderduys, E.P., Ferguson, D., Mathieson, M., 2012. Effect of small-scale woodland clearing and thinning on vertebrate fauna in a largely intact tropical savanna mosaic. *Wildl. Res.* 39, 366. <https://doi.org/10.1071/WR11171>.
- Laurance, W.F., Carolina Useche, D., Rendeiro, J., Kalka, M., Bradshaw, C.J.A., Sloan, S.P., Laurance, S.G., Campbell, M., Abernethy, K., Alvarez, P., Arroyo-Rodriguez, V., Ashton, P., Benítez-Malvido, J., Blom, A., Bobo, K.S., Cannon, C.H., Cao, M., Carroll, R., Chapman, C., Coates, R., Cords, M., Danielsen, F., De Dijn, B., Dinerstein, E., Donnelly, M.A., Edwards, D., Edwards, F., Farwig, N., Fashing, P., Forget, P.-M., Foster, M., Gale, G., Harris, D., Harrison, R., Hart, J., Karpanty, S., John Kress, W., Krishnaswamy, J., Logsdon, W., Lovett, J., Magnusson, W., Maisels, F., Marshall, A.R., McClearn, D., Mudappa, D., Nielsen, M.R., Pearson, R., Pitman, N., van der Ploeg, J., Plumptre, A., Poulsen, J., Quesada, M., Rainey, H., Robinson, D., Roetgers, C., Rovero, F., Scatena, F., Schulze, C., Sheil, D., Struhsaker, T., Terborgh, J., Thomas, D., Timm, R., Nicolas Urbina-Cardona, J., Vasudevan, K., Joseph Wright, S., Arias-G, Carlos, Arroyo, L.J., Ashton, M., Auzel, P., Babaasa, D., Babweteera, F., Baker, P., Banki, O., Bass, M., Bila-Isia, I., Blake, S., Brockelman, W., Brokaw, N., Brühl, C.A., Bunyavejchewin, S., Chao, J.-T., Chave, J., Chellam, R., Clark, C.J., Clavijo, J., Congdon, R., Corlett, R., Dattaraja, H.S., Dave, C., Davies, G., de Mello Biesiegl, B., de Nazaré Paes da Silva, R., Di Fiore, A., Diesmos, A., Dirzo, R., Doran-Sheehy, D., Eaton, M., Emmons, L., Estrada, A., Ewango, C., Fedigan, L., Feer, F., Fruth, B., Giacalone Willis, J., Goodale, U., Goodman, S., Guix, J.C., Guthiga, P., Haber, W., Hamer, K., Herbinger, I., Hill, J., Huang, Z., Fang Sun, L., Ickes, K., Itoh, A., Ivanauskas, N., Jackes, B., Janovec, J., Janzen, D., Jiangming, M., Jin, C., Jones, T., Justiniano, H., Kalko, E., Kasangaki, A., Killeen, T., King, H., Klop, E., Knott, C., Koné, I., Kudavidanage, E., Lahoz da Silva Ribeiro, J., Lattke, J., Laval, R., Lawton, R., Leal, M., Leighton, M., Lentino, M., Leonel, C., Lindsell, J., Ling-Ling, L., Eduard Linsenmair, K., Losos, E., Lugo, A., Lwanga, J., Mack, A.L., Martins, M., Scott McGraw, W., McNab, R., Montag, L., Myers Thompson, J., Nabe-Nielsen, J., Nakagawa, M., Nepal, S., Norconk, M., Novotny, V., O'Donnell, S., Opiang, M., Ouboter, P., Parker, K., Parthasarathy, N., Piscicotta, K., Prawiradilaga, D., Pringle, C., Rajathurai, S., Reichard, U., Reinartz, G., Renton, K., Reynolds, G., Reynolds, V., Riley, E., Rödel, M.-O., Rothman, J., Round, P., Sakai, S., Sanaïotti, T., Savini, T., Schaab, G., Seidensticker, J., Siaka, A., Silman, M.R., Smith, T.B., de Almeida, S.S., Sodhi, N., Stanford, C., Stewart, K., Stokes, E., Stoner, K.E., Sukumar, R., Surbeck, M., Tobler, M., Tschamtké, T., Turkalo, A., Umaphathy, G., van Weerd, M., Vega Rivera, J., Venkataraman, M., Venn, L., Vereca, C., Volkmer de Castilho, C., Waltert, M., Wang, B., Watts, D., Weber, W., West, P., Whitacre, D., Whitney, K., Wilkie, D., Williams, S., Wright, D. D., Wright, P., Xiankai, L., Yonzon, P., Zamzani, F., 2012. Averting biodiversity collapse in tropical forest protected areas. *Nature* 489, 290–294. <https://doi.org/10.1038/nature11318>.
- Lescano, J.N., Bellis, L.M., Hoyos, L.E., Leynaud, G.C., 2015. Amphibian assemblages in dry forests: multi-scale variables explain variations in species richness. *Acta Oecol.* 65–66, 41–50. <https://doi.org/10.1016/j.actao.2015.05.002>.
- Leynaud, G.C., Bucher, E.H., 2005. Restoration of degraded Chaco woodlands: effects on reptile assemblages. *For. Ecol. Manag.* 213, 384–390. <https://doi.org/10.1016/j.foreco.2005.04.003>.
- McKinney, M.L., 2006. Urbanization as a major cause of biotic homogenization. *Biol. Conserv.* 127, 247–260. <https://doi.org/10.1016/j.biocon.2005.09.005>.
- McKinney, M.L., 2008. Effects of urbanization on species richness: a review of plants and animals. *Urban Ecosyst.* 11, 161–176. <https://doi.org/10.1007/s11252-007-0045-4>.
- Møller, A.P., Jennions, M.D., 2002. How much variance can be explained by ecologists and evolutionary biologists? *Oecologia* 132, 492–500. <https://doi.org/10.1007/s00442-002-0952-2>.
- Neuman-Lee, L.A., Carr, J., Vaughn, K., French, S.S., 2015. Physiological effects of polybrominated diphenyl ether (PBDE-47) on pregnant gartersnakes and resulting offspring. *Gen. Comp. Endocrinol.* 219, 143–151. <https://doi.org/10.1016/j.ygcen.2015.03.011>.
- Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L., Bennett, D.J., Choimes, A., Collen, B., Day, J., De Palma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M.J., Feldman, A., Garon, M., Harrison, M.L.K., Alhussaini, T., Ingram, D.J., Itescu, Y., Kattge, J., Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D.L.P., Martin, C.D., Meiri, S., Novosolov, M., Pan, Y., Phillips, H.R.P., Purves, D.W., Robinson, A., Simpson, J., Tuck, S.L., Weiher, E., White, H.J., Ewers, R.M., Mace, G.M., Scharlemann, J.P.W., Purvis, A., 2015. Global effects of land use on local terrestrial biodiversity. *Nature* 520, 45–50. <https://doi.org/10.1038/nature14324>.
- Nori, J., Lemes, P., Urbina-Cardona, N., Baldo, D., Lescano, J., Loyola, R., 2015. Amphibian conservation, land-use changes and protected areas: a global overview. *Biol. Conserv.* 191, 367–374. <https://doi.org/10.1016/j.biocon.2015.07.028>.
- Nowakowski, A.J., Watling, J.L., Thompson, M.E., Bruschi, G.A., Catenazzi, A., Whitfield, S.M., Kurz, D.J., Suárez-Mayorga, Á., Aponte-Gutiérrez, A., Donnelly, M.A., Todd, B.D., 2018. Thermal biology mediates responses of amphibians and reptiles to habitat modification. *Ecol. Lett.* 21, 345–355. <https://doi.org/10.1111/ele.12901>.
- Ofori-Boateng, C., Oduro, W., Hillers, A., Norris, K., Oppong, S.K., Adum, G.B., Rödel, M.-O., 2013. Differences in the effects of selective logging on amphibian assemblages in three West African forest types. *Biotropica* 45, 94–101. <https://doi.org/10.1111/j.1744-7429.2012.00887.x>.
- O'Hanlon, S.J., Rieux, A., Farrer, R.A., Rosa, G.M., Waldman, B., Bataille, A., Kosch, T.A., Murray, K.A., Brankovics, B., Fumagalli, M., Martin, M.D., 2018. Recent Asian origin of chytrid fungi causing global amphibian declines. *Science* 360, 621–627. <https://doi.org/10.1126/science.aar1965>.
- Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'Amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P., Kassem, K.R., 2001. Terrestrial ecoregions of the world: a new map of life on earth. *Bioscience* 51, 933. [https://doi.org/10.1641/0006-3568\(2001\)051\[0933:teotwa\]2.0.co;2](https://doi.org/10.1641/0006-3568(2001)051[0933:teotwa]2.0.co;2).
- Ordeñana, M.A., Crooks, K.R., Boydston, E.E., Fisher, R.N., Lyren, L.M., Siudyla, S., Haas, C.D., Harris, S., Hathaway, S.A., Turschak, G.M., Miles, A.K., Van Vuren, D.H., 2010. Effects of urbanization on carnivore species distribution and richness. *J. Mammal.* 91, 1322–1331. <https://doi.org/10.1644/09-MAMM-A-312.1>.
- Östergård, H., Ehrlén, J., 2005. Among population variation in specialist and generalist seed predation—the importance of host plant distribution, alternative hosts and environmental variation. *Oikos* 111, 39–46. <https://doi.org/10.1111/j.0030-1299.2005.13902.x>.
- Panday, P.K., Coe, M.T., Macedo, M.N., Lefebvre, P., Castanho, A.D. de A., 2015. Deforestation offsets water balance changes due to climate variability in the Xingu River in eastern Amazonia. *J. Hydrol.* 523, 822–829. <https://doi.org/10.1016/J.JHYDROL.2015.02.018>.
- Pfeiffer, M., Langan, L., Linstädter, A., Martens, C., Gaillard, C., Ruppert, J.C., Higgins, S. I., Mudongo, E.I., Scheiter, S., 2019. Grazing and aridity reduce perennial grass abundance in semi-arid rangelands – insights from a trait-based dynamic vegetation model. *Ecol. Model.* 395, 11–22. <https://doi.org/10.1016/J.ECOLMODEL.2018.12.013>.
- Pimm, S.L., Raven, P., 2000. Extinction by numbers. *Nature* 403, 843–845. <https://doi.org/10.1038/35002708>.
- Pimm, S.L., Jenkins, C.N., Abell, R., Brooks, T.M., Gittleman, J.L., Joppa, L.N., Raven, P. H., Roberts, C.M., Sexton, J.O., 2014. The biodiversity of species and their rates of extinction, distribution, and protection. *Science* 344 (6187). <https://doi.org/10.1126/science.1246752>.
- Popescu, V.D., Patrick, D.A., Hunter, M.L., Calhoun, A.J.K., 2012. The role of forest harvesting and subsequent vegetative regrowth in determining patterns of amphibian habitat use. *For. Ecol. Manag.* 270, 163–174. <https://doi.org/10.1016/J.FORECO.2012.01.027>.
- Raudenbush, S.W., 2009. Analyzing effect sizes: random-effects models. In: *The Handbook of Research Synthesis and Meta-analysis*, pp. 295–316.
- Riffell, S., Verschuyt, J., Miller, D., Wigley, T.B., 2011. Biofuel harvests, coarse woody debris, and biodiversity – a meta-analysis. *For. Ecol. Manag.* 261, 878–887. <https://doi.org/10.1016/J.FORECO.2010.12.021>.
- Rittenhouse, T.A.G., Harper, E.B., Rehard, L.R., Semlitsch, R.D., 2008. The role of microhabitats in the desiccation and survival of anurans in recently harvested Oak-Hickory forest. *Copeia* 2008, 807–814. <https://doi.org/10.1643/CH-07-176>.
- Roll, U., Feldman, A., Novosolov, M., Allison, A., Bauer, A.M., Bernard, R., Böhm, M., Castro-Herrera, F., Chirio, L., Collen, B., Colli, G.R., Dabool, L., Das, I., Doan, T.M., Grismer, L.L., Hoogmoed, M., Itescu, Y., Kraus, F., Lebreton, M., Lewin, A., Martins, M., Maza, E., Meirte, D., Nagy, Z.T., Nogueira, C.D.C., Paulwels, O.S.G., Pincheira-Donoso, D., Powney, G.D., Sindaco, R., Tallowin, O.J.S., Torres-Carvajal, O., Trape, J.F., Vidan, E., Uetz, P., Wagner, P., Wang, Y., Orme, C.D.L., Grenyer, R., Meiri, S., 2017. The global distribution of tetrapods reveals a need for targeted reptile conservation. *Nat. Ecol. Evol.* 1, 1677–1682. <https://doi.org/10.1038/s41559-017-0332-2>.
- Rotem, G., Ziv, Y., Giladi, I., Bouskila, A., 2013. Wheat fields as an ecological trap for reptiles in a semiarid agroecosystem. *Biol. Conserv.* 167, 349–353. <https://doi.org/10.1016/J.BIOCON.2013.08.028>.
- Russell, K.R., Wigley, T.B., Baughman, W.M., Hanlin, H.G., Ford, W.M., 2004. Responses of southeastern amphibians and reptiles to forest management: A review. *Gen. Tech. Rep. SRS-75*. Asheville, NC U.S. Dep. Agric. For. Serv. South. Res. Station. Chapter 27, 319–334.
- Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Hueneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M.,

- Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. *Science* 287, 1770–1774.
- Sandhage-Hofmann, A., Kotzé, E., van Delden, L., Dominiak, M., Fouché, H.J., van der Westhuizen, H.C., Oomen, R.J., du Preez, C.C., Amelung, W., 2015. Rangeland management effects on soil properties in the savanna biome, South Africa: a case study along grazing gradients in communal and commercial farms. *J. Arid Environ.* 120, 14–25. <https://doi.org/10.1016/J.JARIDENV.2015.04.004>.
- Scheffers, B.R., Paszkowski, C.A., 2013. Amphibian use of urban stormwater wetlands: the role of natural habitat features. *Landsc. Urban Plan.* 113, 139–149. <https://doi.org/10.1016/J.LANDURBPLAN.2013.01.001>.
- Scott, D.M., Brown, D., Mahood, S., Denton, B., Silburn, A., Rakotondraparany, F., 2006. The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar. *Biol. Conserv.* 127, 72–87. <https://doi.org/10.1016/J.BIOCON.2005.07.014>.
- Semlitsch, R.D., Todd, B.D., Blomquist, S.M., Calhoun, A.J.K., Gibbons, J.W., Gibbs, J.P., Graeter, G.J., Harper, E.B., Hocking, D.J., Hunter, M.L., Patrick, D.A., Rittenhouse, T.A.G., Rothermel, B.B., 2009. Effects of timber harvest on amphibian populations: understanding mechanisms from forest experiments. *Bioscience* 59, 853–862. <https://doi.org/10.1525/bio.2009.59.10.7>.
- Sievers, M., Hale, R., Swearer, S.E., Parris, K.M., 2019. Frog occupancy of polluted wetlands in urban landscapes. *Conserv. Biol.* 33, 389–402. <https://doi.org/10.1111/cobi.13210>.
- Sinervo, B., Mendez-De-La-Cruz, F., Miles, D.B., Heulin, B., Bastiaans, E., Villagrán-Santa Cruz, M., Lara-Resendiz, R., Martínez-Méndez, N., Calderón-Espinosa, M.L., Meza-Lázaro, R.N., Gadsden, H., 2010. Erosion of lizard diversity by climate change and altered thermal niches. *Science* 328, 894–899. <https://doi.org/10.1126/science.1184695>.
- Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A., Haberl, H., Harper, R., House, J., Jafari, M., Masera, O., 2014. Agriculture, forestry and other land use (AFOLU). In: *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, pp. 811–922.
- Snodgrass, J.W., Casey, R.E., Joseph, D., Simon, J.A., 2008. Microcosm investigations of stormwater pond sediment toxicity to embryonic and larval amphibians: variation in sensitivity among species. *Environ. Pollut.* 154, 291–297. <https://doi.org/10.1016/J.ENVPOL.2007.10.003>.
- Spatz, D.R., Zilliacus, K.M., Holmes, N.D., Butchart, S.H.M., Genovesi, P., Ceballos, G., Tershy, B.R., Croll, D.A., 2017. Globally threatened vertebrates on islands with invasive species. *Sci. Adv.* 3, e1603080 <https://doi.org/10.1126/sciadv.1603080>.
- Stevens, J.R., Taylor, A.M., 2009. Hierarchical dependence in meta-analysis. *J. Educ. Behav. Stat.* 34, 46–73. <https://doi.org/10.3102/1076998607309080>.
- Stuart, S.N., 2004. Status and trends of amphibian declines and extinctions worldwide. *Science* 306, 1783–1786. <https://doi.org/10.1126/science.1103538>.
- Suazo-Ortuño, I., Alvarado-Díaz, J., Martínez-Ramos, M., 2008. Effects of conversion of dry tropical forest to agricultural mosaic on herpetofaunal assemblages. *Conserv. Biol.* 22, 362–374. <https://doi.org/10.1111/j.1523-1739.2008.00883.x>.
- Team, R.C., 2016. A language and environment for statistical computing. R Foundation for statistical computing, 2015; Vienna, Austria. <https://www.R-project.org>.
- Thompson, M.E., Nowakowski, A.J., Donnelly, M.A., 2016. The importance of defining focal assemblages when evaluating amphibian and reptile responses to land use. *Conserv. Biol.* 30, 249–258. <https://doi.org/10.1111/cobi.12637>.
- Todd, B.D., Andrews, K.M., 2008. Response of a reptile guild to forest harvesting. *Conserv. Biol.* 22, 753–761. <https://doi.org/10.1111/j.1523-1739.2008.00916.x>.
- Vallan, D., 2002. Effects of anthropogenic environmental changes on amphibian diversity in the rain forests of eastern Madagascar. *J. Trop. Ecol.* 18, 725–742. <https://doi.org/10.1017/S026646740200247X>.
- Verschuyll, J., Riffell, S., Miller, D., Wigley, T.B., 2011. Biodiversity response to intensive biomass production from forest thinning in North American forests - a meta-analysis. *For. Ecol. Manag.* 261, 221–232. <https://doi.org/10.1016/j.foreco.2010.10.010>.
- Viechtbauer, W., 2010. Conducting meta-analyses in R with the metafor package. *Journal of Statistical Software* 36, 1–48. <https://doi.org/10.18637/jss.v036.i03>.
- Wake, D.B., Vredenburg, V.T., 2008. Colloquium paper: are we in the midst of the sixth mass extinction? A view from the world of amphibians. *Proc. Natl. Acad. Sci. U.S.A.* 105 (Suppl), 11466–11473. <https://doi.org/10.1073/pnas.0801921105>.
- Wanger, T.C., Iskandar, D.T., Motzke, I., Brook, B.W., Sodhi, N.S., Clough, Y., Tscharnkte, T., 2010. Effects of land-use change on community composition of tropical amphibians and reptiles in Sulawesi, Indonesia. *Conserv. Biol.* 24, 795–802. <https://doi.org/10.1111/j.1523-1739.2009.01434.x>.
- Zheng, D., Chen, J., Song, B., Xu, M., Sneed, P., Jensen, R., 2000. Effects of silvicultural treatments on summer forest microclimate in southeastern Missouri Ozarks. *Clim. Res.* 15, 45–59. <https://doi.org/10.3354/cr015045>.