EVD Research Paper

The impact of ecological restoration on reptile conservation:
A global meta-analysis

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Abstract

To mitigate biodiversity loss, ecological restoration has been promoted as an effective tool. Governments from around the world have made many commitments to restore previously degraded habitat to meet their domestic and international conservation goals and address the biodiversity crisis. In this systematic review and meta-analysis, I compile and analyze information on the effects of ecological restoration on reptiles, a relatively understudied group of organisms in the field of restoration ecology. Restored habitats had a significantly higher reptile population density compared to degraded habitats, but there was no difference in species richness. Most of the studies captured by my review were from Australia and the United States of America and examined restoration sites that were between 0-9 years old. A diversity of restoration interventions were implemented, but the most common were prescribed burning, vegetation thinning, passive regrowth, invasive control, or some combination of the above. My review highlights the need for longer-term restoration monitoring data, better data reporting, and increased global representation of studies. It would also be of great benefit to the research community to collaborate with governments, private sector companies, and not-for-profit organizations to compile restoration project monitoring data and further improve our understanding of the effects of ecological restoration on biodiversity.

Introduction

To mitigate biodiversity loss, ecological restoration has been promoted as an effective tool (Aronson et al., 2006). Ecological restoration can partly reverse the environmental degradation caused by humans and increase ecosystem biodiversity (Benayas et al., 2009). habitat restoration Notably, however, restoration is often incapable of sustaining the same levels of biodiversity and ecosystem services as an intact habitat (Wortley et al., 2013). The mitigation hierarchy, an international standard for managing impacts to biodiversity, gives lower priority to restoration measures compared to avoidance measures (which prevent impacts from occurring at the outset) and minimization measures (which reduce the duration, intensity, and extent of the impacts) (Arlidge et al., 2018). Nonetheless, restored habitats provide a means to recover benefits such as improved soil quality, cleaner water, and ecotourism that have been lost in degraded ecosystems due to the pursuit of short-term economic gain, for example (IPBES, 2019). As such, ecological restoration has been supported as a promising practice in areas where preliminary habitat protection has failed (Wortley et al., 2013).

The practice of ecological restoration is an attempt to return a habitat to pre-settlement conditions by assisting the recovery of a degraded, damaged, or destroyed ecosystem (Gann et al., 2019). Restoration projects aspire to achieve the highest level of recovery possible to reestablish the native biota and ecosystem functions, and restore the habitat to a state that is self-sustaining and resilient in the long-term (Gann et al., 2019). Restoration projects can also be designed with the primary goal of reestablishing or enhancing populations of specific plants and animals (Thompson and Donnelly, 2018; Volis, 2019). Examples of common restoration practices include the removal of invasive species, seeding and planting of native flora, the reintroduction of natural grazers, and passive rewilding (Wortley et al., 2013). The field of restoration ecology is relatively young, but is growing quickly as demand from decision-makers for this research increases (Suding et al., 2015).

Governments from around the world have made many commitments to restore previously degraded habitat (Suding et al., 2015). At an international level, ecosystem restoration has been integrated within the Strategic Plan for Biodiversity 2011-2020 as part of the multilateral treaty of the Convention on Biological Diversity (which now has 196 signatories; United Nations, 2010). Included within the plan are the Aichi Biodiversity targets 14 and 15 that “ecosystems providing services are restored and safeguarded” and “15% of degraded ecosystems are restored” (Convention on Biological Diversity, 2013). The rationale for these targets was explained through the capacity of restored landscapes and seascapes to address biodiversity loss through habitat creation as well as increase ecosystem resilience to future perturbations (Convention on Biological Diversity, 2013). Restoration continues to be a key theme within the Post-2020 Global Biodiversity Framework as the 2050 vision is stated as, “By 2050, biodiversity is valued, conserved, restored, and widely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people” (CBD, 2020). The Bonn Challenge, a global goal to restore 350 million ha of degraded and deforested habitat by 2030 has gained international traction since its launch in 2011 with 210 million ha having been pledged by 61 countries, surpassing past commitment projections (IUCN, 2020). In addition, habitat restoration is an important practice for fulfilling several of the 2015 United Nations’ Sustainable Development goals (IPBES, 2019) with 2021- 2030 declared the United Nations Decade on Ecosystem Restoration (United Nations, 2021). Unilaterally, several countries have established national policies involving habitat restoration. For instance, the government of Canada has recently proposed to invest CAD$ 631 million over 10 years to restore and enhance degraded habitats (Government of Canada, 2019). In China, more than US$ 379 billion has been invested in ecological restoration projects from 1979 to 2015 (Li et al., 2021). As countries mobilize restoration efforts in the coming decades, it will be important to research the efficacy and potential of ecological restoration as a conservation strategy for Earth's biodiversity and ecosystems.

My study focuses on the impact that ecological restoration has had for reptile conservation. Despite being one of the most speciose groups of terrestrial vertebrates (Pincheira-Donoso et al., 2013), reptiles are consistently understudied in ecology and conservation (Fitzgerald et al., 2017). In addition, reptiles currently face a severe decline in numbers (IUCN, 2016). Using a random sample of 1500 species of reptiles, roughly one in five species is facing extinction (Böhm et al., 2013). Indeed, there is a need for more research on reptile conservation (Roll et al., 2017). The field of restoration ecology is no exception to this plea as only 0.4% of ecological restoration studies in Brazilian biomes had some indicator of restoration success for reptiles (Guerra et al., 2020). The majority of systematic reviews and meta-analyses on ecosystem restoration success either focus on the impacts for biodiversity in general (Benayas et al., 2009; Shimamoto et al., 2018), impacts on vertebrates as a whole (Meli et al., 2017) or, when distinctions are made between taxa, data on reptiles and amphibians are combined and presented under herpetofauna (Atkinson and Bonser, 2020; Crouzeilles et al., 2017). Notably, a recent meta-analysis on passive secondary forest restoration concluded that population density and species richness recovery is less pronounced for reptiles, suggesting a reduced effectiveness of restoration for reptiles compared to amphibians, leading the authors to conclude that reptiles and amphibians should be considered separately for restoration ecology research (Thompson and Donnelly, 2018). There is a lack of information regarding the efficacy and impact of ecological restoration as a tool for reptile conservation. If countries are committed to including ecological restoration as a major tool to meet conservation goals, it will be important to understand how most principal taxa will respond to these efforts.

In my study, I tested the hypothesis that the restoration of ecological attributes provides benefits to reptiles living in relatively degraded habitats. Thus, I predicted that reptiles should increase in population density and species richness in restored habitats compared to degraded habitats.

Material and Methods

Literature Search and Data Extraction

I searched Web of Science (Clarviate Analytics, 2022) from January 17th, 2022 until February 1st, 2022 to identify relevant studies. I used the following string of terms:

First Line (Topic): reptile* OR snake* OR lizard* OR turtle* OR crocodile* OR tuatara* OR herpeto*;

Second line (Topic): (restor* OR rehabilit* OR reclamat* OR revitaliz* OR renatur* OR regenerat* OR habitat enhanc* OR habitat management*).

The string of terms captures all major groups of reptiles in the first line and all variation of wording for restoration in the second line. The search includes all years prior to and including 2022. I did not apply restrictions or filters. The search yielded 4986 results. I evaluated all the search results by reading the title and abstract of the article to determine its relevance. I took a conservative approach when excluding articles in the first level of screening. The first scan eliminated articles that were not relevant to the inclusion/exclusion criteria (Table 1) since the article search had a low specificity to avoid excluding relevant studies. If there was any doubt about whether to include a study, I retained the study for the second scan. At the end of the first scan, there were 306 studies after I removed any duplicates.

This search does not follow the guidelines published by the Collaboration for Environmental Evidence (Collaboration for Environmental Evidence, 2018) primarily due to time constraints. I did not include a pre-review or a publication of the protocol, nor an explicit strategy for searching the "grey literature" (unpublished government reports, etc.). I conducted the search in English only and I did not review the bibliographies of included studies for relevant studies.

Table 1. Inclusion/Exclusion criteria designed for the systematic review.

Criteria	Include	Exclude
Ecosystem	Any ecosystem where reptiles are present	Ecosystems unsuitable for naturally-occurring reptiles
Location	Global	None
Project objectives	Restoration, rehabilitation, mitigation	None
Progress	Implemented	Planned
Monitoring design	Before/After, Control/Impact, or BACI	No monitoring data or only data after restoration
Monitoring data	Quantitative data on at least one biological metric of one organism group that is a measure of abundance or species richness	No quantitative biological data or biological data that are not a measure of abundance or species richness

Environmental data	Basic ecosystem and project characteristics reported (*e.g.*, location and restoration technique used)	Irrelevant or missing details
Effects	Irrespective of effects (*i.e.*, negative, non, and positive effects)	None
Data	Empirical data based on a set of sampling units (*e.g.*, experimental replicates) and sufficient statistical information (e.g., mean with some estimate of precision)	

The second scan consisted of a full text review. If I excluded a study during the second scan, I had to provide a reason (see Supplementary Material – M2 for the second scan database and rationales for exclusion). If I included a study, I extracted the data into a separate database (see Supplementary Material – M1 for the data extraction database). If a study only presented its results in a figure, I extracted the data using PlotDigitizer which allows users to extract numerical data from images such as XY plots (pOrbital, 2022). By the end of the second scan, fifty-four studies met the inclusion criteria (Table 1). I extracted: geographic location, ecosystem type, target population, restoration intervention, age of restoration, study design type, general methodology, sample size, mean abundance and/or richness, and a measure of variance from each study if available (see Supplementary Material – M1). I conducted counts for the country of study, age of restoration, and restoration intervention for each data point. For studies that provided a range of values for the age of restoration, I classified the study based on its youngest restoration age.

Calculation and analysis of effect sizes

Data analysis was carried out using OpenMEE, an open-source, cross-platform software for ecological and evolutionary meta-analysis (Viechtbauer, 2010; Wallace et al., 2012; Wallace et al., 2017). I organized data from the data extraction database, and I inserted into OpenMEE. For studies with multiple categorical treatments, I inserted multiple datapoints, comparing each test variable (e.g., prescribed burning and herbicide treatment, etc.) to the degraded and/or reference (relatively intact) condition. If a study used a before/after control/impact study design, I used separate datapoints for comparisons of control (after) to impact (after) and impact (before) to impact (after). The effect sizes across studies were calculated using the log-transformed ratio of means (Hedges et al., 1999) using the mean, sample size, and standard deviation. I calculated effect sizes using the natural log of the ratio of mean species richness or mean abundance in degraded or reference habitats to the mean species richness or mean abundance in restored habitat. I also calculated effect sizes for degraded versus reference habitats if a study provided richness or abundance measures for comparable, degraded and reference sites.

I conducted a subgroup meta-analysis using the log ratio of mean species richness and mean abundance, respectively, using OpenMEE. I classified the subgroups as “degraded” (comparing degraded sites to restored sites), “reference” (comparing reference sites to restored) and “degraded vs reference” (comparing degraded sites to reference sites). For abundance there were: degraded habitats, n = 87; for reference habitats, n = 50; for restored sites n = 119; for degraded vs reference, n = 18. For richness there were: degraded habitats, n = 34; for reference habitats, n = 38; for restored sites n = 59; for degraded vs reference, n = 13.

All statistical analyses were conducted using the package metaphor (Viechtbauer, 2010) in OpenMEE (Wallace et al., 2017). For each response variable (abundance and richness), a continuous random-effects model (DerSimonian-Laird method) compared subgroups. A funnel plot (a scatter plot of effect sizes against a measure of variance) for each subgroup meta-analysis evaluated whether publication bias significantly influenced the dataset. I calculated Rosenberg's weighted fail-safe number (Rosenberg, 2005), an estimate of the number of unpublished studies with an effect size of zero that would need to be added to make the observed effect size non-significant ($p > 0.05$).

Results

Characteristics of included studies

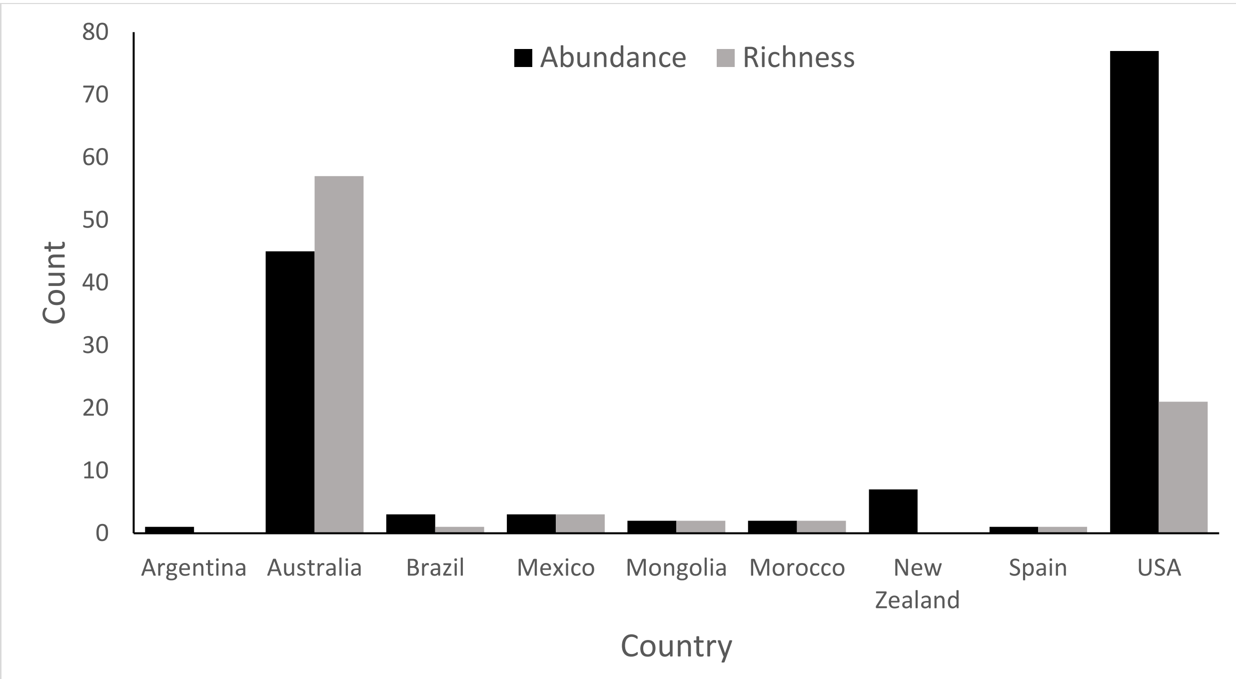

Figure 1. Histogram of datapoints per country for abundance and richness datasets. Total datapoints: Abundance = 141; Species Richness = 87.

An overwhelming majority of the studies in the meta-analysis were from the United States of America (USA) and Australia (Figure 1). Studies from the USA were more likely to include only measures of mean abundance compared to studies from Australia which were more likely to include both measures of mean abundance and mean species richness (Figure 3).

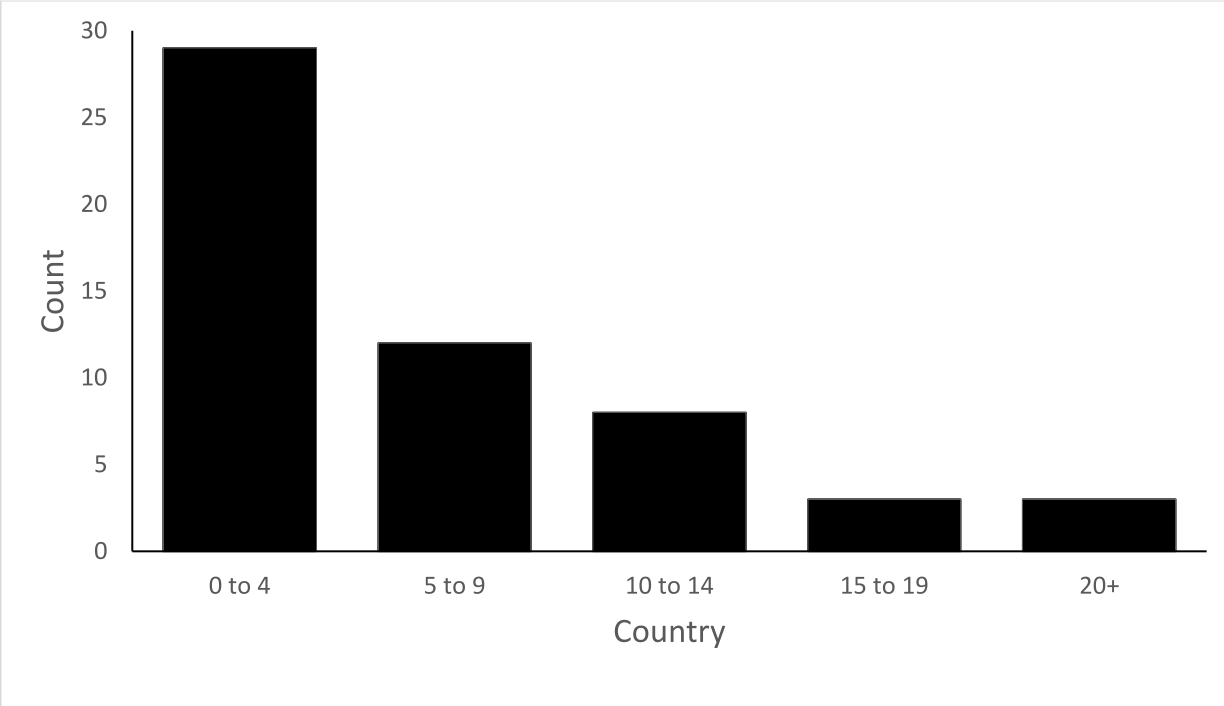

Figure 2. Age distribution of restored sites studied in the articles included in the meta-analysis. Three studies did not provide information on the age of restoration. For the studies which provided a range of values for the age, the youngest age of restoration was tallied; therefore, this table provides an underestimate of the actual average age of restored sites for this meta-analysis.

Approximately 50% of the studies included in the meta-analysis examined restored sites which were between 0-4 years, around 75% were between 0-9 years old, and 5% of studies examined sites that were 20+ years old (Figure 2).

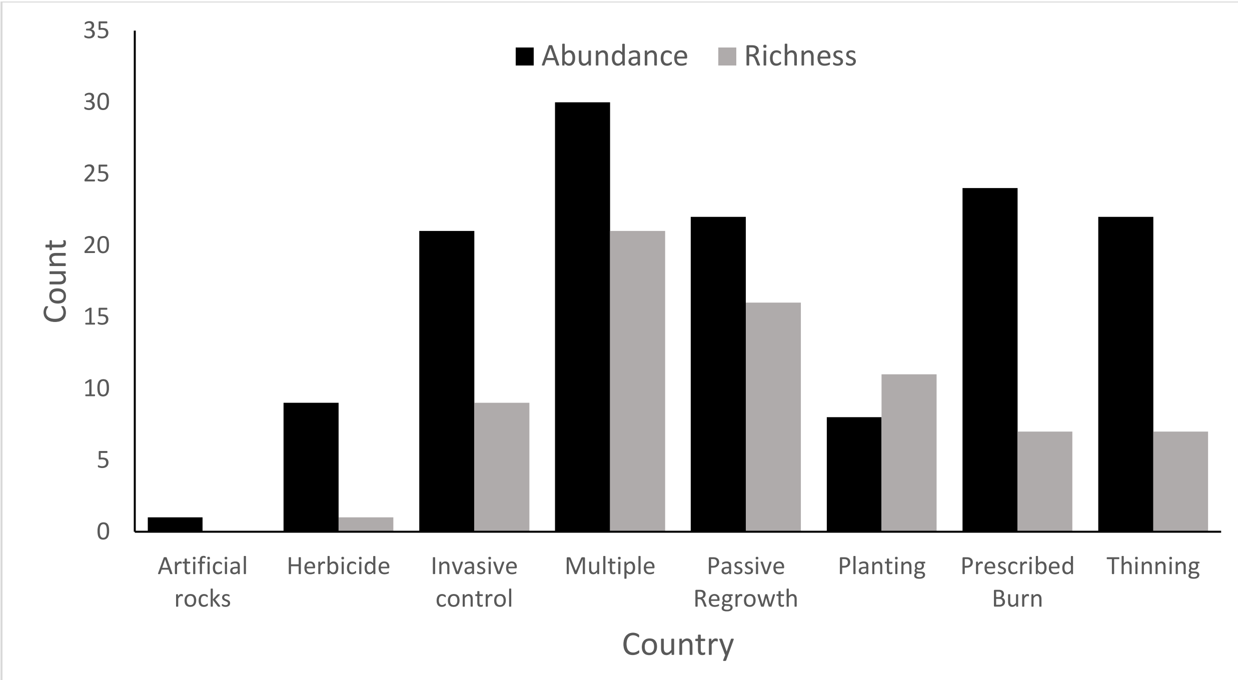

Figure 3. Histogram of datapoints per restoration intervention for abundance and richness dataset. More than one restoration intervention used per datapoint was classified as “Multiple”. Total datapoints: Abundance = 137; Richness = 72.

Most studies examined the effects of a single restoration intervention, but a sizable portion studied the effects of multiple restoration interventions within a single site (Figure 3). Studies which implemented multiple techniques either used a combination of the single interventions presented in Figure 3 or a combination of impact-specific interventions (see Supplementary Material – M1 for details on restoration interventions). Studies that implemented prescribed burning, thinning, invasive control, or passive regrowth were more likely to only measure reptile abundance, whereas studies that implemented planting were more likely to measure reptile species richness (Figure 3).

Most studies included within the meta-analysis examined lizards, snakes and/or turtles (see Supplementary Material – M1). There were a few studies which included tuatara and there were no studies which included species from the order Crocodylia.

Many studies had to be rejected due to flawed study designs (see Supplementary Material – M2). Several studies had insufficient or missing controls, had no replicated sites, and/or did not provide a measure of variance. Incomplete reporting was also an issue, with several studies omitting sample sizes, raw data, or missing key methodological details.

Effect of restoration

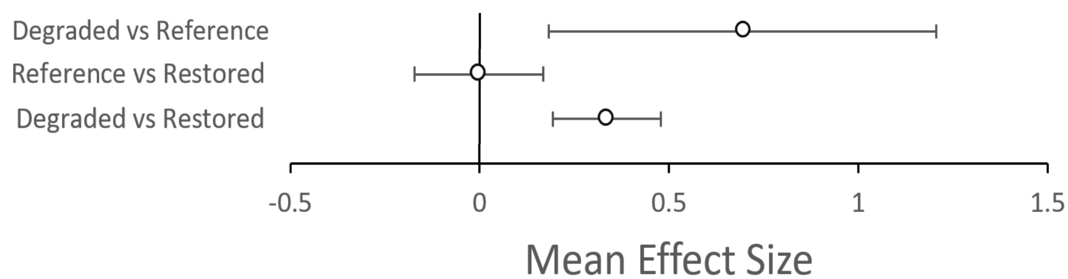

Figure 4. Forest plot showing the mean effect sizes (ln response ratio) and 95% confidence intervals for the comparison of mean abundance in degraded and reference habitats (relatively undisturbed sites), in reference and restored habitats, and degraded and restored habitats. The points represent the mean effect size for all studies within the group based on a continuous random-effects model (DerSimonian-Laird method). Response ratios were calculated as the natural log of the ratio of the average abundance in a restored habitat to the average abundance in a degraded or restored habitat. A positive value means that abundance was higher in restored habitats or that abundance was higher in reference habitats. For degraded habitats, n = 87; for reference habitats, n = 50; for restored sites n = 119; for degraded vs reference, n = 18.

Restoration had a positive effect on reptile population density compared to degraded sites (Figure 4; n = 87, mean effect size = 0.335, 95% CI 0.192 to 0.479, $p < 0.001$) and restored sites showed no difference in reptile abundance compared to relatively pristine reference sites (Figure 4; n = 50, mean effect size = -0.002, 95% CI -0.172 to 0.168, p = 0.985). Degraded sites had a significantly lower abundance compared to reference sites (Figure 4; n = 18, mean effect size = 0.707, 95% CI 0.184 to 1.229, p = 0.008).

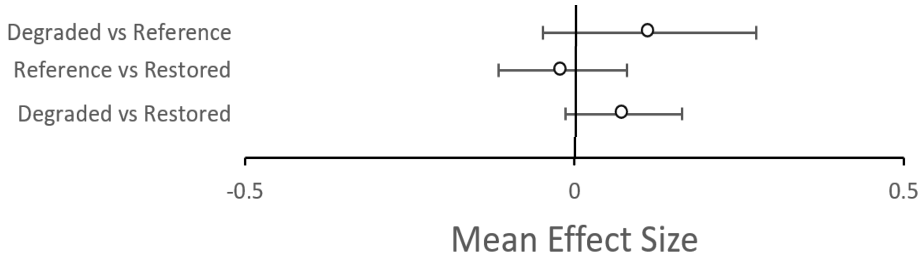

Figure 5. Forest plot showing the mean effect sizes (ln response ratio) and 95% confidence intervals for the comparison of mean richness in degraded and reference habitats (relatively undisturbed sites), in reference and restored habitats, and degraded and restored habitats. The points represent the mean effect size for all studies within the group based on a continuous random-effects model (DerSimonian-Laird method). Response ratios were calculated as the natural log of the ratio of the average richness in a restored habitat to the average richness in a degraded or restored habitat. A positive value means that richness was higher in restored habitats or that richness was higher in reference. For degraded habitats, n = 34; for reference habitats, n = 38; for restored sites n = 59; for degraded vs reference, n = 13.

Restoration had no effect on reptile species richness compared to degraded sites (Figure 5; n = 34, mean effect size = 0.073, 95% CI -0.015 to 0.162, p = 0.104) nor compared to reference sites (Figure 5; n = 38, mean effect size = -0.020, 95% CI -0.117 to -0.078, p = 0.694). Degraded sites did not differ in reptile species richness compared to reference sites (Figure 5; n = 13, mean effect size = 0.112, 95% CI -0.050 to 0.274, p = 0.175).

Publication Bias of ecological restoration studies on reptiles

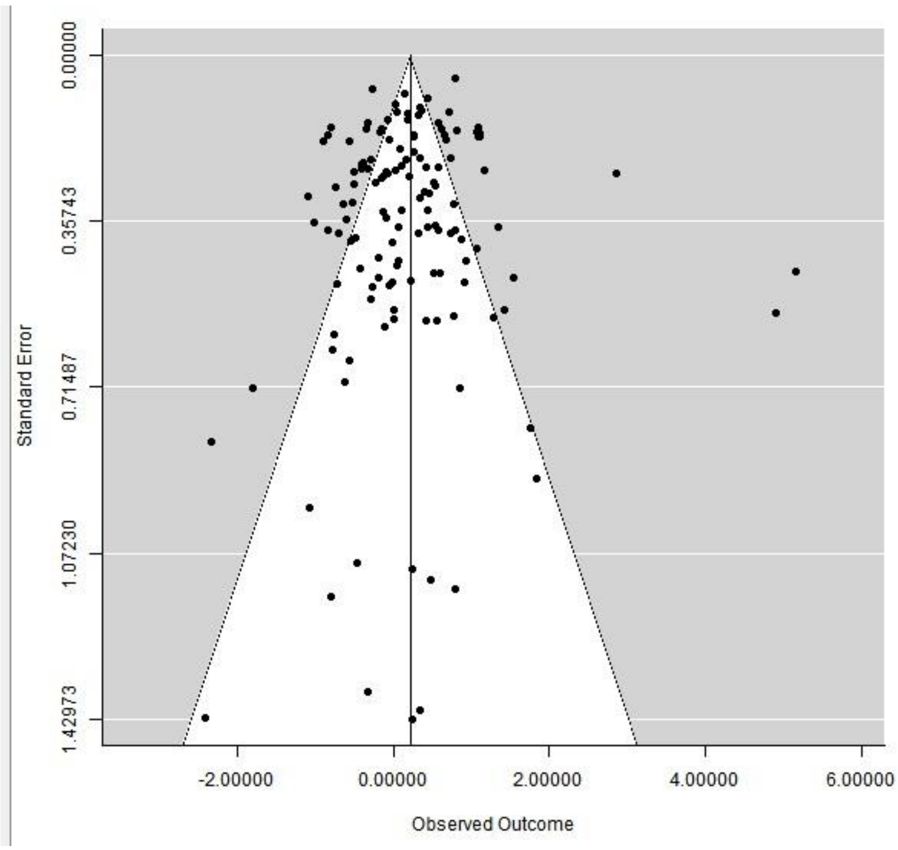

Figure 6. Funnel plot showing the mean effect sizes (Observed Outcome) plotted against the standard error (95% confidence intervals delineated by the dotted lines) for the comparison of mean abundance in degraded and restored habitats, and in reference (relatively undisturbed sites) and restored habitats. Response ratios were calculated as the natural log of the ratio of the average abundance in a restored habitat to the average abundance in a degraded or restored habitat. The overall mean effect size was calculated using a random effects model encompassing all studies from all subgroups and is represented by the solid black line in the middle of the funnel. For degraded habitats, n = 87; for reference habitats, n = 50; for restored sites n = 119; for degraded vs reference, n = 18 (total number of datapoints = 155).

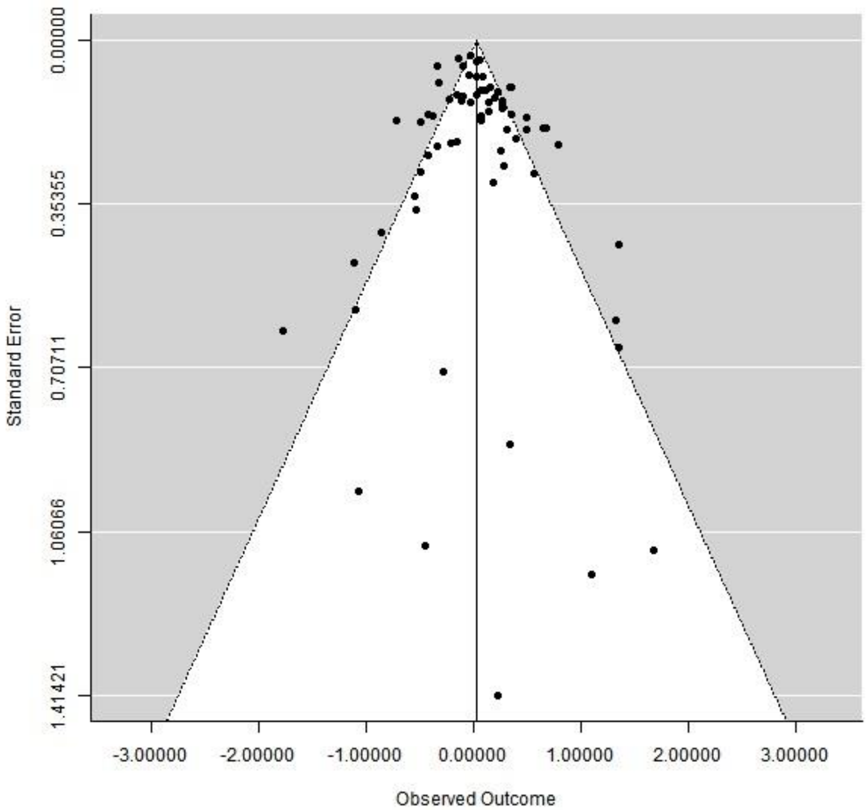

Figure 7. Funnel plot showing the mean effect sizes (Observed Outcome) plotted against the standard error (95% confidence intervals delineated by the dotted lines) for the comparison of mean richness in degraded and restored habitats, and in reference (relatively undisturbed sites) and restored habitats. Response ratios were calculated as the natural log of the ratio of the average richness in a restored habitat to the average abundance in a degraded or restored habitat. The overall mean effect size was calculated using a random effects model encompassing all studies from all subgroups and is represented by the solid black line in the middle of the funnel. For degraded habitats, n = 34; for reference habitats, n = 38; for restored sites n = 59; for degraded vs reference, n = 13.

Funnels plots display the spread of the datapoints and evaluate data asymmetry. The funnel plots for both the mean effect size for population density (Figure 6) and richness (Figure 7) displayed a symmetrical spread and suggest that publication bias did not significantly skew the mean effect size for either meta-analysis. The fail-safe N calculation using the Rosenberg Approach indicated that 7282 studies with null effects would render $p = 0.05$ from $p < 0.001$ for the effect of restoration on mean population density (when compared to degraded sites).

Therefore, this positive effect of restoration is robust to the possibility that the systematic review did not capture all relevant studies or the possibility of publication bias.

Discussion

I compiled current research on the effects of ecological restoration on reptiles, which have been relatively understudied compared to other vertebrate groups (Fitzgerald et al., 2017). My hypothesis was supported: restoration interventions can significantly improve the abundance of reptiles within degraded sites and can allow to achieve population densities similar to that of relatively intact reference sites. In a meta-analysis on the efficacy of restoration on reptiles in forested habitats, by Thompson and Donnelly (2018), there was no difference in reptile population density between passive secondary forest succession and human-modified land or old growth forest. Although I included passive regrowth in my meta-analysis, many more active restoration interventions were also included. Perhaps active restoration interventions that are applied to benefit populations of reptiles specifically are more effective at increasing abundance compared to less targeted approaches such as passive regrowth. Further, there were fewer studies captured by the systematic review of Thompson and Donnelly (2012), with a sample size ranging from n = 5-10, and the authors note that there was variation in the reptile abundances reported in the studies. It is also possible that the lack of effect noted by Thompson and Donnelly (2012) is specific to forest succession and forested habitats since the scope of the present meta-analysis is much broader.

Contrary to my hypothesis, there were no significant differences in reptile species richness between degraded, restored, and reference sites. Crucially, there was no difference between the mean species richness of degraded and reference sites meaning that it is unlikely that restored sites would have an effect given this result. Compared to population density, reptile species richness has a lower sensitivity since there is a narrower range of values. As well, population density for a given species would have to reach zero before richness was affected. Furthermore, only a minority of studies included within the meta-analysis provided richness data for both degraded and reference sites (n = 13) and amongst these studies, degraded sites varied from highly disturbed such as a heavily urbanized site (Banville et al., 2012) to less disturbed, such as sites which had lost fire regimes (Steen et al., 2013). Thompson and Donnelly (2012) also found no difference in the reptile species richness of secondary forest succession compared to old-growth forest or human-modified habitat. Acevedo-Charry and Aide (2019) found that reptile species richness was significantly lower in early succession and young secondary forest compared to mature tropical, an effect that was absent in mid-successional forest secondary forest and old secondary forest. In contrast, reptile species compositional similarity in all four secondary successional stages never reached the same level as mature tropical forest. Although there exists other meta-analyses on habitat restoration, reptile-specific data are combined with amphibians (Atkinson and Bonser, 2020; Crouzeilles et al., 2017), other vertebrates (Meli et al., 2017), or lumped into biodiversity generally (Benayas et al., 2009; Shimamoto et al., 2018); therefore, comparisons will not be made between the present meta-analysis and those which are not reptile-specific.

As reported in this study and others dealing with ecological restoration (see review by Wortley et al., 2013), most restoration projects lack long-term monitoring and so, data are often limited to the first few years of a project when the restoration has not been fully developed and populations may not have had the chance to respond. Conversely, it is possible that the restoration intervention fails post-monitoring and the positive short-term benefits are lost. Longer term monitoring and monitoring made accessible to the public and the private sector is needed to test effectively the efficacy of mitigation measures for conserving biodiversity.

Study design limitations

Perhaps the most significant limitation of this study is that there is considerable heterogeneity within the data. Each study varies in the target population of reptile, the geographic location, the habitat type, the restoration intervention applied, and the age of restoration. Certainly, the needs of a lizard will vary from that of a freshwater turtle. Heterogeneity of data in meta-analyses on ecological restoration leads to an overestimation of restoration success (Lilian et al., 2021). Despite this, there are simply not enough data in the literature to isolate the effect of each variable appropriately. This study attempts to improve upon the currently available meta-analyses on the effects of ecological restoration, which have not yet isolated reptiles as a study group. For a smaller scope analysis, there are a few meta-analyses dealing with the effects of habitat restoration on reptiles in specific habitats (Acevedo-Charry and Aide, 2019; Thompson and Donnelly, 2018) and indeed there are many reptile species-specific articles (see Appendix for list of studies included in this meta-analysis).

The skewed global representation of the studies within this meta-analysis is possibly an effect of restricting the search language of the systematic review to English only. It is unsurprising that most of the included studies are from primarily English-speaking countries such as the USA and Australia. Despite this, there were no studies from Canada, England, and many other English-speaking countries meaning that there is likely a gap in published studies on the effects of ecological restoration on reptiles. With more time, the bibliographies of reviews from other countries could be searched to expand global representation. An English-only review on ecological restoration also found a skewed global representation (Wortley et al., 2013) while two other reviews which did not specify the search language, but focused on tropical forests, found a more diverse global representation (Acevedo-Charry and Aide, 2019; Crouzeilles et al., 2017). It would be of benefit to collaborate with researchers from other countries to capture a more representative picture of the data on reptiles and habitat restoration. Until then, the results of this study are largely restricted to the ecosystems found within the USA and Australia.

I did not perform a grey literature search, but it is likely that there exists a significant amount of data on ecological restoration and reptiles due to the applied nature of the topic. Indeed, countries such as the USA, Australia, New Zealand, South Africa, and Brazil have developed government-led offsetting programs that require ecological restoration. It is also possible that data exist in the private sector and with not-for-profit institutions. To date, there is no central directory or database for the public to access data on ecological restoration monitoring and, so, it would demand considerable effort to track down this information in a systematic fashion.

I acknowledge that abundance and species richness are limited in their representation of the effects of habitat restoration on reptiles. Including measures of reproductive success, survival, and growth would provide a more complete story but these measures would still be closely tied to population density. Measures of community composition and similarity are more sensitive to age of restoration and could provide a more detailed analysis of the effect of ecological restoration (Crouzeilles et al., 2017) but may not reveal major changes due to a reduced sensitivity from a limited pool of potential species (reptiles-only). Lilian et al. (2021) advocated for separating actions that aim at increasing an ecosystem attribute such as native tree planting from actions that aim to decrease an ecosystem attribute such as invasive species cover. Failing to do so can lead to an overestimation of restoration success (Lilian et al., 2021). Due to the limited availability of data and the heterogeneity of the study populations, it was not possible to include the aforementioned variables in this meta-analysis.

The calculated effect size in this study represents a measure of "*what remains to be done*" in the comparisons between reference sites only. Lilian et al. (2021) explained that this presents an interpretation bias since the initial degree of degradation of each site is likely to be different and so, the degree of "*what has been done*" is neglected in those studies which present data for only reference sites and not for degraded sites. If more time was available, the *Achieved Restoration Index* presented in Lilian et al. (2021) would be calculated to mitigate the interpretation bias.

Conservation Implications

This study provides support for the use of restoration interventions to increase reptile population density, but not for increasing reptile species richness. It is quite possible that based on the study results, limited conservation resources may be better spent on more aggressive pursuits of avoidance and minimization measures than on-site restoration measures. Land managers will have to continue evaluating case-specific needs of reptile populations as the scope of this study is far too broad to be applied on a local scale. Most importantly, this study adds to the narrative that data on restoration interventions are still lacking for long-term applied effectiveness, global representation, and species representation. The greatest source of restoration monitoring data compiled by governments and private sector companies remains inaccessible to the public and would be an invaluable resource to the research community.

Supplementary Material

M1. Data Extraction Database. Contains all extracted information that was used in the meta-analyses.

M2. Level 2 Screening Database. Contains bibliographic information on included studies (also see Appendix) and reasons for excluding studies which passed the Level 1 screening.

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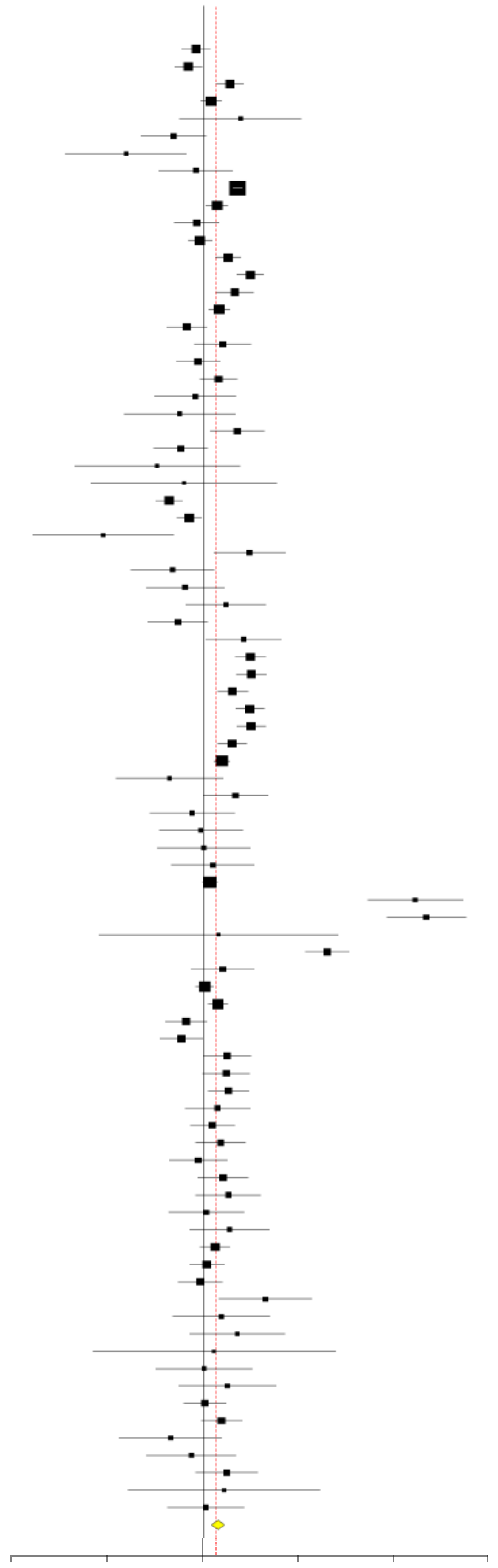
Appendix

Studies	Estimate (95% C.I.)
Michael 2014	-0.174 (-0.502, 0.153)
Michael 2014-3	-0.354 (-0.668, -0.039)
Michael 2014-5	0.607 (0.292, 0.922)
Michael 2014-7	0.176 (-0.073, 0.425)
Radke 2008	0.851 (-0.554, 2.257)
Brunjes 2003	-0.693 (-1.447, 0.061)
Brunjes 2003-2	-1.792 (-3.194, -0.389)
Brunjes 2003-3	-0.182 (-1.037, 0.672)
Baecher 2018	0.789 (0.690, 0.888)
Zeng 2014-2	0.316 (0.065, 0.567)
Banville 2012-2	-0.154 (-0.671, 0.362)
Sirois 2014	-0.076 (-0.348, 0.196)
Sirois 2014-2	0.569 (0.284, 0.853)
Mosher 2016	1.080 (0.772, 1.388)
Mosher 2016-2	0.726 (0.294, 1.158)
Perry 2009	0.363 (0.126, 0.600)
Bateman 2015	-0.387 (-0.842, 0.069)
Bateman 2015-2	0.446 (-0.207, 1.100)
Virkki 2012	-0.122 (-0.637, 0.393)
Virkki 2012-2	0.348 (-0.086, 0.783)
Leynaud 2005	-0.188 (-1.126, 0.750)
Christen 2022	-0.555 (-1.843, 0.734)
Bateman 2008	0.780 (0.148, 1.411)
Bateman 2008-2	-0.532 (-1.152, 0.087)
Greenberg 2016	-1.073 (-2.985, 0.840)
Greenberg 2016-2	-0.455 (-2.598, 1.687)
Valentine 2009	-0.790 (-1.093, -0.488)
Valentine 2009-2	-0.330 (-0.613, -0.048)
Wolf 2015	-2.325 (-3.954, -0.695)
Azor 2015	1.067 (0.248, 1.886)
Halstead 2019	-0.720 (-1.686, 0.246)
Halstead 2019-2	-0.423 (-1.325, 0.478)
Foster 2016	0.519 (-0.403, 1.441)
Schlesinger 2020	-0.596 (-1.290, 0.097)
Schlesinger 2020-2	0.930 (0.064, 1.796)
Wilson 2017	1.087 (0.740, 1.433)
Wilson 2017-2	1.110 (0.767, 1.454)
Wilson 2017-3	0.676 (0.318, 1.033)
Wilson 2017-4	1.073 (0.747, 1.400)
Wilson 2017-5	1.102 (0.773, 1.432)
Wilson 2017-6	0.661 (0.324, 0.998)
Moseley 2003	0.432 (0.251, 0.613)
Gonsalves 2018-4	-0.786 (-2.026, 0.455)
Gonsalves 2018-5	0.741 (-0.009, 1.490)
Gonsalves 2018-6	-0.265 (-1.240, 0.710)
Kilpatrick 2010	-0.059 (-1.029, 0.911)
Kilpatrick 2010-2	0.009 (-1.064, 1.082)
Kilpatrick 2010-3	0.218 (-0.732, 1.168)
Luja 2008-2	0.147 (-0.017, 0.310)
Kanowski 2006-2	4.898 (3.808, 5.988)
Kanowski 2006-4	5.153 (4.237, 6.070)
Kanowski 2006-6	0.348 (-2.420, 3.117)
Kanowski 2006-8	2.865 (2.364, 3.365)
Hromada 2018	0.440 (-0.284, 1.163)
Chergui 2019	0.023 (-0.182, 0.228)
Chergui 2019-2	0.334 (0.113, 0.555)
Sutton 2013	-0.404 (-0.883, 0.075)
Sutton 2013-2	-0.510 (-1.000, -0.020)
Sutton 2013-3	0.539 (-0.009, 1.088)
Sutton 2013-4	0.523 (-0.018, 1.063)
Sutton 2013-5	0.575 (0.102, 1.048)
Sutton 2013-6	0.330 (-0.424, 1.084)
Sutton 2013-7	0.204 (-0.306, 0.714)
Sutton 2013-8	0.397 (-0.180, 0.975)
Sutton 2013-9	-0.121 (-0.782, 0.541)
Sutton 2013-10	0.452 (-0.131, 1.034)
Steen 2013-4	0.574 (-0.168, 1.316)
Steen 2013-5	0.067 (-0.799, 0.934)
Steen 2013-6	0.600 (-0.319, 1.519)
Steen 2013-7	0.268 (-0.075, 0.611)
Steen 2013-8	0.078 (-0.318, 0.474)
Steen 2013-9	-0.074 (-0.576, 0.429)
Steen 2013-13	1.427 (0.355, 2.500)
Steen 2013-14	0.413 (-0.710, 1.536)
Steen 2013-15	0.780 (-0.323, 1.883)
Steen 2013-19	0.241 (-2.561, 3.044)
Steen 2013-20	0.012 (-1.102, 1.126)
Steen 2013-21	0.553 (-0.566, 1.672)
Bruton 2013-2	0.029 (-0.460, 0.519)
Bruton 2013-4	0.413 (-0.058, 0.884)
Cosentino 2013-2	-0.760 (-1.940, 0.421)
Cosentino 2013-3	-0.281 (-1.313, 0.750)
Cosentino 2013-4	0.536 (-0.183, 1.255)
Cosentino 2013-5	0.470 (-1.746, 2.686)
Cosentino 2013-6	0.051 (-0.839, 0.942)
Subgroup Degraded (I^2=88.26 % , P=0.000)	**0.335 (0.192, 0.479)**



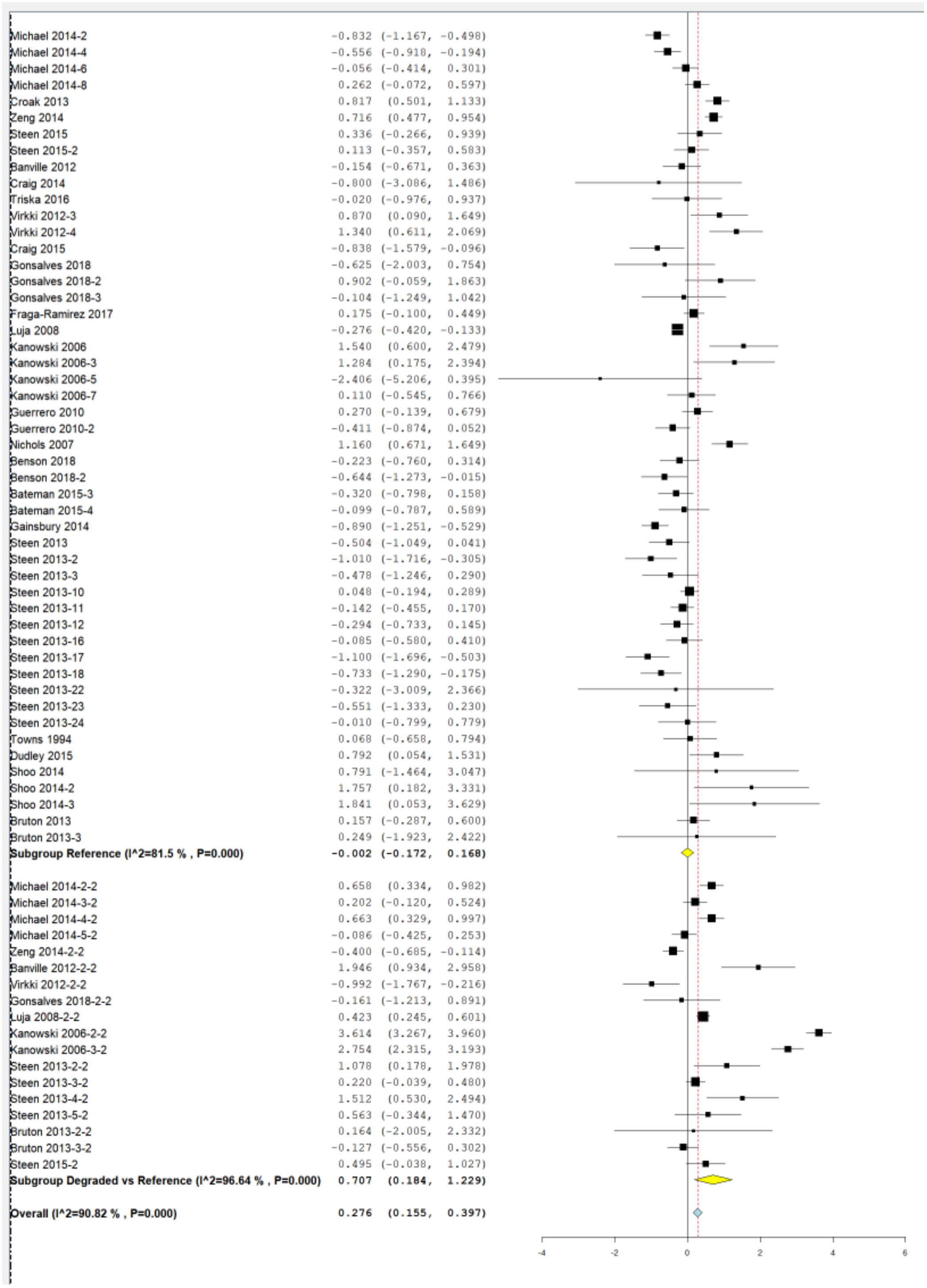


Figure S1. Forest plot showing the mean effect sizes (and 95% confidence intervals in brackets) for the comparison of mean abundance in degraded and restored habitats (Subgroup Degraded), in reference (relatively undisturbed) and restored habitats (Subgroup Reference), and degraded and reference habitats (Subgroup Degraded vs Reference). The diamonds represent the mean effect size for all studies within the group based on a continuous random-effects model (DerSimonian-Laird method). Response ratios were calculated as the natural log of the ratio of the average abundance in a restored habitat to the average abundance in a degraded or restored habitat. A positive value means that abundance was higher in restored habitats (for Subgroups Degraded and Reference) or that abundance was higher in reference habitats (for Subgroup Degraded vs Reference). The random effects model encompassing all studies from all subgroups is represented by the blue diamond and the red dotted line. For degraded habitats, n = 87; for reference habitats, n = 50; for restored sites n = 119; for degraded vs reference, n = 18.

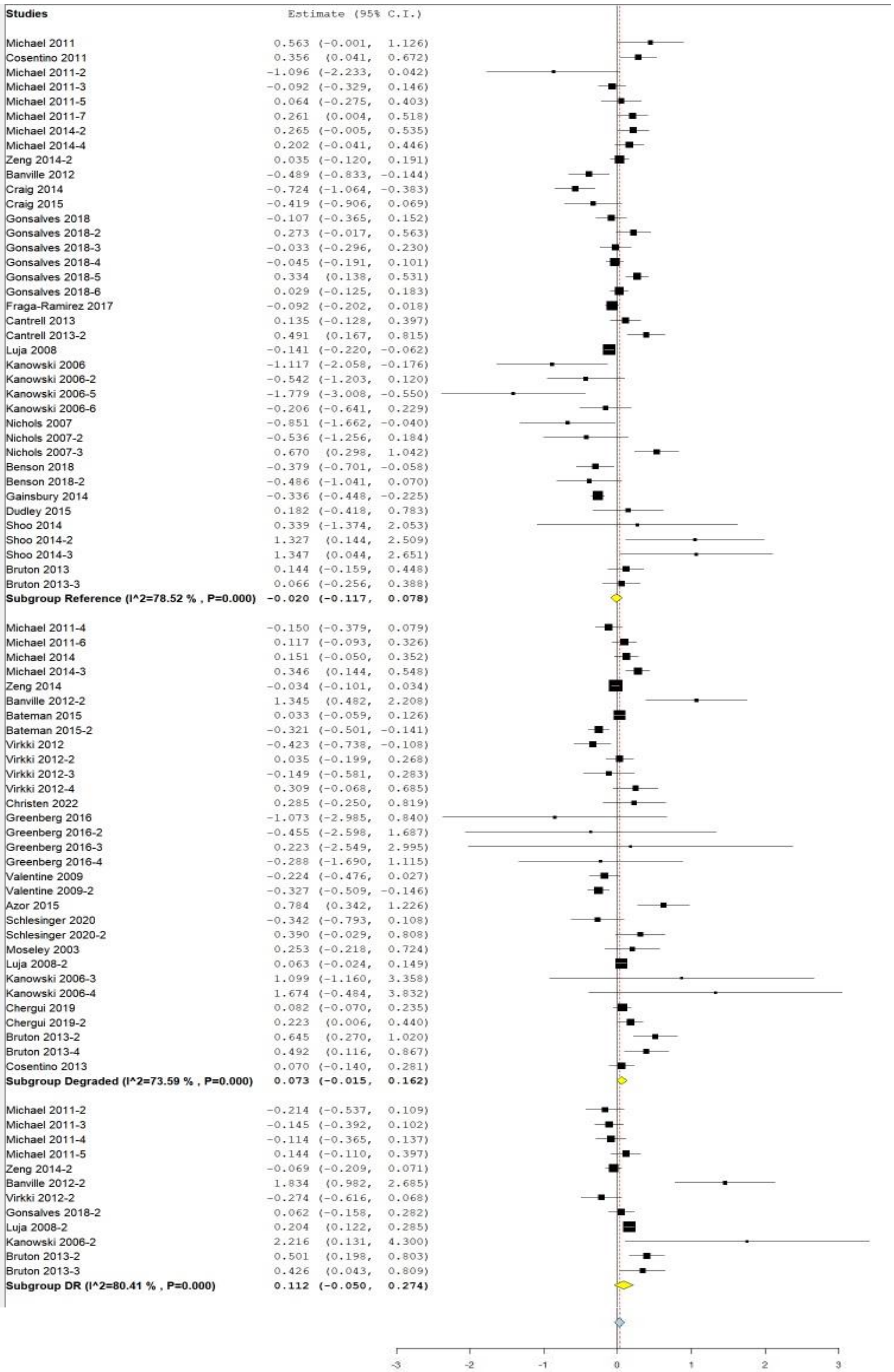

Studies
Estimate (95% C.I.)
Michael 2011 0.563 (-0.001, 1.126)
Cosentino 2011 0.356 (0.041, 0.672)
Michael 2011-2 -1.096 (-2.233, 0.042)
Michael 2011-3 -0.092 (-0.329, 0.146)
Michael 2011-5 0.064 (-0.275, 0.403)
Michael 2011-7 0.261 (0.004, 0.518)
Michael 2014-2 0.265 (-0.005, 0.535)
Michael 2014-4 0.202 (-0.041, 0.446)
Zeng 2014-2 0.035 (-0.120, 0.191)
Banville 2012 -0.489 (-0.833, -0.144)
Craig 2014 -0.724 (-1.064, -0.383)
Craig 2015 -0.419 (-0.906, 0.069)
Gonsalves 2018 -0.107 (-0.365, 0.152)
Gonsalves 2018-2 0.273 (-0.017, 0.563)
Gonsalves 2018-3 -0.033 (-0.296, 0.230)
Gonsalves 2018-4 -0.045 (-0.191, 0.101)
Gonsalves 2018-5 0.334 (0.138, 0.531)
Gonsalves 2018-6 0.029 (-0.125, 0.183)
Fraga-Ramirez 2017 -0.092 (-0.202, 0.018)
Cantrell 2013 0.135 (-0.128, 0.397)
Cantrell 2013-2 0.491 (0.167, 0.815)
Luja 2008 -0.141 (-0.220, -0.062)
Kanowski 2006 -1.117 (-2.058, -0.176)
Kanowski 2006-2 -0.542 (-1.203, 0.120)
Kanowski 2006-5 -1.779 (-3.008, -0.550)
Kanowski 2006-6 -0.206 (-0.641, 0.229)
Nichols 2007 -0.851 (-1.662, -0.040)
Nichols 2007-2 -0.536 (-1.256, 0.184)
Nichols 2007-3 0.670 (0.298, 1.042)
Benson 2018 -0.379 (-0.701, -0.058)
Benson 2018-2 -0.486 (-1.041, 0.070)
Gainsbury 2014 -0.336 (-0.448, -0.225)
Dudley 2015 0.182 (-0.418, 0.783)
Shoo 2014 0.339 (-1.374, 2.053)
Shoo 2014-2 1.327 (0.144, 2.509)
Shoo 2014-3 1.347 (0.044, 2.651)
Bruton 2013 0.144 (-0.159, 0.448)
Bruton 2013-3 0.066 (-0.256, 0.388)
Subgroup Reference (I^2=78.52 % , P=0.000) -0.020 (-0.117, 0.078)
Michael 2011-4 -0.150 (-0.379, 0.079)
Michael 2011-6 0.117 (-0.093, 0.326)
Michael 2014 0.151 (-0.050, 0.352)
Michael 2014-3 0.346 (0.144, 0.548)
Zeng 2014 -0.034 (-0.101, 0.034)
Banville 2012-2 1.345 (0.482, 2.208)
Bateman 2015 0.033 (-0.059, 0.126)
Bateman 2015-2 -0.321 (-0.501, -0.141)
Virkki 2012 -0.423 (-0.738, -0.108)
Virkki 2012-2 0.035 (-0.199, 0.268)
Virkki 2012-3 -0.149 (-0.581, 0.283)
Virkki 2012-4 0.309 (-0.068, 0.685)
Christen 2022 0.285 (-0.250, 0.819)
Greenberg 2016 -1.073 (-2.985, 0.840)
Greenberg 2016-2 -0.455 (-2.598, 1.687)
Greenberg 2016-3 0.223 (-2.549, 2.995)
Greenberg 2016-4 -0.288 (-1.690, 1.115)
Valentine 2009 -0.224 (-0.476, 0.027)
Valentine 2009-2 -0.327 (-0.509, -0.146)
Azor 2015 0.784 (0.342, 1.226)
Schlesinger 2020 -0.342 (-0.793, 0.108)
Schlesinger 2020-2 0.390 (-0.029, 0.808)
Moseley 2003 0.253 (-0.218, 0.724)
Luja 2008-2 0.063 (-0.024, 0.149)
Kanowski 2006-3 1.099 (-1.160, 3.358)
Kanowski 2006-4 1.674 (-0.484, 3.832)
Chergui 2019 0.082 (-0.070, 0.235)
Chergui 2019-2 0.223 (0.006, 0.440)
Bruton 2013-2 0.645 (0.270, 1.020)
Bruton 2013-4 0.492 (0.116, 0.867)
Cosentino 2013 0.070 (-0.140, 0.281)
Subgroup Degraded (I^2=73.59 % , P=0.000) 0.073 (-0.015, 0.162)
Michael 2011-2 -0.214 (-0.537, 0.109)
Michael 2011-3 -0.145 (-0.392, 0.102)
Michael 2011-4 -0.114 (-0.365, 0.137)
Michael 2011-5 0.144 (-0.110, 0.397)
Zeng 2014-2 -0.069 (-0.209, 0.071)
Banville 2012-2 1.834 (0.982, 2.685)
Virkki 2012-2 -0.274 (-0.616, 0.068)
Gonsalves 2018-2 0.062 (-0.158, 0.282)
Luja 2008-2 0.204 (0.122, 0.285)
Kanowski 2006-2 2.216 (0.131, 4.300)
Bruton 2013-2 0.501 (0.198, 0.803)
Bruton 2013-3 0.426 (0.043, 0.809)
Subgroup DR (I^2=80.41 % , P=0.000) 0.112 (-0.050, 0.274)
-3
-2
-1
0
1
2
3

Figure S2. Forest plot showing the mean effect sizes (ln response ratio) and 95% confidence intervals (in brackets) for the comparison of mean richness in degraded and restored habitats (Subgroup Degraded), in reference (relatively undisturbed) and restored habitats (Subgroup Reference), and degraded and reference habitats (Subgroup Degraded vs Reference). The diamonds represent the mean effect size for all studies within the group based on a continuous random-effects model (DerSimonian-Laird method). Response ratios were calculated as the natural log of the ratio of the average richness in a restored habitat to the average richness in a degraded or restored habitat. A positive value means that richness was higher in restored habitats (for Subgroups Degraded and Reference) or that richness was higher in reference habitats (for Subgroup Degraded vs Reference). The random effects model encompassing all studies from all subgroups is represented by the blue diamond and the red dotted line. For degraded habitats, n = 34; for reference habitats, n = 38; for restored sites n = 59; for degraded vs reference, n = 13.

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