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Perspective Article

Conceptual and methodological issues in estimating the success of ecological restoration

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ABSTRACT

Ecological restoration (ER) of terrestrial ecosystems has become widespread in past decades. However, assessing its success is complex mainly due to the diversity of objectives pursued, actions undertaken but also statistical methods for treating data. We demonstrate here that, due to the heterogeneity of collected data, the success of restoration actions can be overestimated in meta-analyses. We advocate analyzing distinctly two types of actions in ER, those aiming at increasing an ecosystem attribute (e.g. species richness of a native plant species, ER^{+}), and those aiming at decreasing it (e.g. invasive species cover, ER^{-}). We also suggest that only one index for assessing the success of a restoration action is not enough. We propose here to complete RR (Remaining Recovery) by a novel index informing on '*what has been restored by comparison to what should have been recovered*': the '*Achieved Restoration*' index (AR).

1. Introduction

The 21st Century is witnessing widespread awareness on the need to effectively restore degraded ecosystems (Bullock et al., 2011; UNDP, 2014; Corlett, 2016). Ecological restoration (ER) – the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (Society for Ecological Restoration definition) - is now solicited to deliver proven and scalable actions coping with the loss of biodiversity and ecosystem services (Menz et al., 2013; Possingham et al., 2015; Kaiser-Bunbury et al., 2017; Comín et al., 2018; Jones et al., 2018). It turned into a global priority after the 2010 Aichi Convention on Biological Diversity, intending to restore at least 15% of degraded ecosystems globally, the 2011 Bonn Challenge (to restore 150 million hectares of lost or degraded forest), the 2014 New York Declaration in which the parties committed to restore a staggering 350 million hectares of forest land by 2030 (UNDP, 2014) and, more recently, the declared UN Decade (2021–2030) on Ecosystem Restoration (UNEP, 2019) and the European Green Deal and its EU Biodiversity Strategy for 2030 (European Commission, 2018). However, ER actions are not magic bullets for instantly recovering the composition and functions of ecosystems (Menz et al., 2013). Uncertainty is to be expected in dealing with the recovery of ecological functions, which will not match exactly the reference ecosystem characteristics and will, frequently, support lower ecological integrity (Miller and Bestelmeyer 2016). Moreover, the

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willingness of recovering a 'pristine' reference ecosystem is nowadays no longer relevant. As reported in the *International Principles and Standards for the Practice of Ecological Restoration* (Gann et al., 2019) 'appropriate reference models for ecological restoration are not based on immobilizing an ecological community at some past point in time, but rather increasing potential for native species and communities to recover and continue to reassemble, adapt, and evolve'. In other words, the aim is to manage the multiple intermediate states defined by Suding and Gross (2006) and the possible restoration trajectories to optimize the restoration potential.

In this quest for a balance between optimizing rendered ecological functions and ecosystem services, without persisting in trying to recover what is no longer there, practitioners implementing ER currently hit against a 'glass ceiling' beyond which it generally seems difficult, if not impossible, to restore the desired functions and/or services. Such glass ceiling was defined as the 'colonization credit' (Cristofoli et al., 2010) or the 'recovery debt' by Moreno-Mateos et al. (2017), i.e. an interim reduction of ecological integrity that restoration generally cannot overcome (Moreno-Mateos et al., 2012; Rey-Benayas et al., 2017; Jones et al., 2018). The level of this glass ceiling is influenced by the actions engaged, but also by both the monitoring indicators chosen and their related reading grids used *a posteriori* to evaluate the success of the operations. Today, a wide variety of indicators are available to evaluate the success of ecological restoration operations. Among them, species richness and abundance are the most commonly used (Jaunatre et al., 2013). However, indicator relevance needs to be reassessed in a multi-criteria evaluation framework, as the Society for Ecological Restoration (SER) has done with its universal standards and practices (Mc Donald et al., 2018, Gann et al, 2019), or Prach et al (2019) in their "primer on choosing goals and indicators to evaluate ecological restoration success". These authors distinguish "general indicators" as those usable in almost every project: structural characteristics (e.g. canopy cover), processes (e.g. carbon sequestration), or biodiversity (e.g. the number or composition of target species) from "specific indicators", which inform specific restoration targets for particular projects (e.g. monitoring of trace elements concentrations in biomass).

In its "Standards", the SER also reminds that specific tools are necessary to identify the levels of recovery aspired to and to track progress. Among such tools, meta-analyses offer a reading grid which allows assembling disparate results into quantitative and reproducible outcomes (Crouzeilles et al., 2016, 2017; Jones et al., 2018a, 2018b; Meli et al., 2014, 2017; Moreno-Mateos et al., 2012, 2015, 2017; Rey-Benayas et al., 2009, 2017). Meta-analyses may however also suffer from methodological biases which lead to generating approximations in conclusions (Koricheva and Gurevitch, 2013; Lajeunesse, 2015; Gurevitch et al., 2018). As an example, in most of the studies dealing with meta-analyses for analyzing ecological restoration outcomes, results are expressed as LnRR (ln *Restored/Reference*). But, as pointed out by Craven (post in Craven blog, 2014), such representation may also be a source of misinterpretation since it exists an asymmetry between response ratios and log-response ratios in graphic renderings. Among these biases, we detail here that mixing case studies where the level of degraded condition is below baseline (e.g. native species richness) and cases where the degraded condition is above baseline (e.g. invasive species cover) results in an overestimation of the average size of the overall effect. Also, the success or failure of ecological restoration is often estimated based on the comparison between the restored state of the ecosystem and a reference state (e.g. by calculating the log response-ratio of the restored vs. reference state, LnRR, Rey-Benayas et al., 2009). Such an approach only partially explains ER outcomes since it only considers "*what remains to be done*" as a measure of the restoration success, the less the better. Another option when estimating the different achieved objectives in restoration programs is an ER assessment by accounting for '*what has been done*', in a complement of '*what remains to be done*' and more precisely for '*what has been done by comparison to what was expected for achieving a complete restoration*'.

Here, we demonstrate that only accounting for '*what remains to be done*' leads to an incomplete assessment of the ER success. In response to this issue, we emphasize on the importance of completing the '*remaining recovery*' (RR) by accounting for '*what has been done relative to what was expected to be done*' for satisfactorily quantify the ER success. To do so, we screened the scientific literature and assembled a large database of published case reports on degraded, restored, and reference states of a broad range of ecosystems undergoing ecological restoration. We propose a novel index – the *'Achieved Restoration'* index (AR) - to be used in association with RR for better quantifying the ER success. We tested the influence of a series of moderators (Type of restoration, habitat, target, etc.) on both AR and RR. The complementarity between AR and RR indices should make it possible to better assess the successes and failures at the end of ecological restoration operations.

2. Method

The research literature was systematically screened using Scopus (https://www.scopus.com) and Web of Science (http://apps.webofknowledge.com). The search was performed between December 2016 and February 2017 using the three following keyword strings independently: 1. (rehabilit* OR restor*) AND (degrad* AND reference), 2. (rehabilit* OR restor*) AND (reference) AND (forest OR grassland OR savan* OR steppe OR wetland OR woodland), and 3. (rehabilit* OR restor*) AND (BACI OR "Before After Control Impact"). All unique references or different primary studies identified by the three keyword strings were compiled into a single collection. We retained only articles, book chapters, reviews, and abstracts published in English between 1940 and 2017. We additionally searched for relevant references in recent reviews on restoration ecology as well as in co-authors' databases. The PRISMA flow chart shows the total number of primary papers retrieved from each source and eventually retained in the meta-analysis **(Appendix A)**.

We examined titles, abstracts, full text, tables, and figures of peer-reviewed papers and reports (2,300 references). Among the initially selected articles, we only retained studies providing mean values, sample size, and variability (i.e. variance, standard deviations, or standard errors) for any observation used to describe ecosystem attributes (e.g., species richness, soil respiration, etc.) measured at the reference (i.e., non-degraded), degraded (i.e., pre-restoration), and current/restored (i.e., post-restoration) ecosystem states. The degraded ecosystem represents the restoration starting point, while the reference ecosystem represents the desired theoretical end-point of ER (Rey-Benayas et al., 2009). Studies providing data only on one or two of these states were excluded. In the end, 87 primary studies encompassing 871 observations from a broad range of ecosystems worldwide and published between 2000 and 2017 were retained **(Appendices A-D)**.

We started analyzing the complete dataset by mixing all case studies. First, we calculated the effect size of each case study as the log-response ratio of the restored state as compared to the reference state $LnRR = ln\left(\frac{Restored}{Reference}\right)$ from which we derived the remaining recovery (RR), that is the proportion (%) of restoration that "*remains to be done*": $100 \times (1 - e^{LnRR})$ or $100 \times (1 - Restored/Reference)$. Then, we calculated the grand mean effect size as the estimate of an intercept on linear mixed effect model in which we declared the primary study (ID) as a random factor to account for the non-independence of multiple comparisons drawn from the same primary study (package LME4, Bates et al., 2015).

In the aftermath, from our complete dataset, we distinguished between two action types: those aiming at increasing the level of an ecosystem attribute (ER^+, e.g. the species richness of native plant species) and those aiming at decreasing the level of an ecosystem attribute (ER^-, e.g. the invasive plant species cover). ER^+ corresponds to case studies such that, in theory, the level of ecosystem attributes rank as follows: *Degraded* < *Restored* < *Reference*. On the contrary, ER^-

corresponds in theory to ones for which the level of ecosystem attributes are lower in the reference state than in the restored state, and even lower than in the degraded state (i.e., *Degraded* > *Restored* > *Reference*). We obtained 689 ER^{+} and 182 ER^{-} case studies, respectively. We then computed the grand mean effect size and total heterogeneity for ER^{+} and ER^{-} case studies, separately. Analyses were performed using the R statistical software (version 3.5.0 R Core Team, 2019), with both the metafor (Viechtbauer 2010) and the orchaRd (Nakagawa et al., 2020) libraries for meta-analysis calculations.

Last, we suggest integrating the three states of the ecosystem (Reference, Degraded, and Restored) in a synthetic index to assess the success of ecological restoration actions: The '*Achieved Restoration*' index (AR).

$$AR = \frac{Restored - Degraded}{Reference - Degraded}$$

The AR index includes the level of the ecosystem attribute in the degraded state (*Degraded*) at both the numerator and the denominator level in its calculation. Thus, to avoid possible biases due to variance distortion when computing average AR, the use of (G)LMMs was favored over the use of a meta-analysis in this case. The average AR for each level of moderators were estimated using both the lmerTest (Kuznetsova, Brockhoff & Christensen 2017) and ggeffects (Lüdecke, 2018) libraries.

3. Results and discussion

3.1. Caution against bias in meta-analyses in restoration ecology

The use of Log-ratio in ecological meta-analyses is nowadays used routinely, mainly because it is normally distributed around zero, with a value of zero representing no effect between the groups compared (post in Craven blog, 2014). But, as reported by Craven in his blog, the way of calculating or even representing ecological meta-analysis outcomes may be a source of interpretation biases. Here, we point out that a misuse of the original dataset when implementing the meta-analysis is also a source of biases in the interpretation. For instance, by using our complete initial dataset, the grand mean effect size estimated (±SE) was -0.29 ± 0.09 (k = 871), which corresponded to a remaining recovery (RR) of 25%. However, this overall mean hid large differences between ecological restoration operations, some aiming at increasing and others at decreasing targeted ecosystem attributes. In ER^{+} case studies, those aiming at increasing the ecosystem attributes, the grand mean effect size was -0.58 ± 0.08 (n = 689), resulting in a RR of 44%. On the contrary, in ER^{-} case studies, the grand mean effect size was 0.63 ± 0.22 (k = 192), giving a RR of −87.7% (Fig. 1). Therefore, by aggregating data from ecological restoration operations aimed at increasing the attributes of an ecosystem with those from actions aimed at decreasing them, LnRR artificially tends towards zero, which leads to underestimating the resulting RR and thus overestimating the potential of ecological

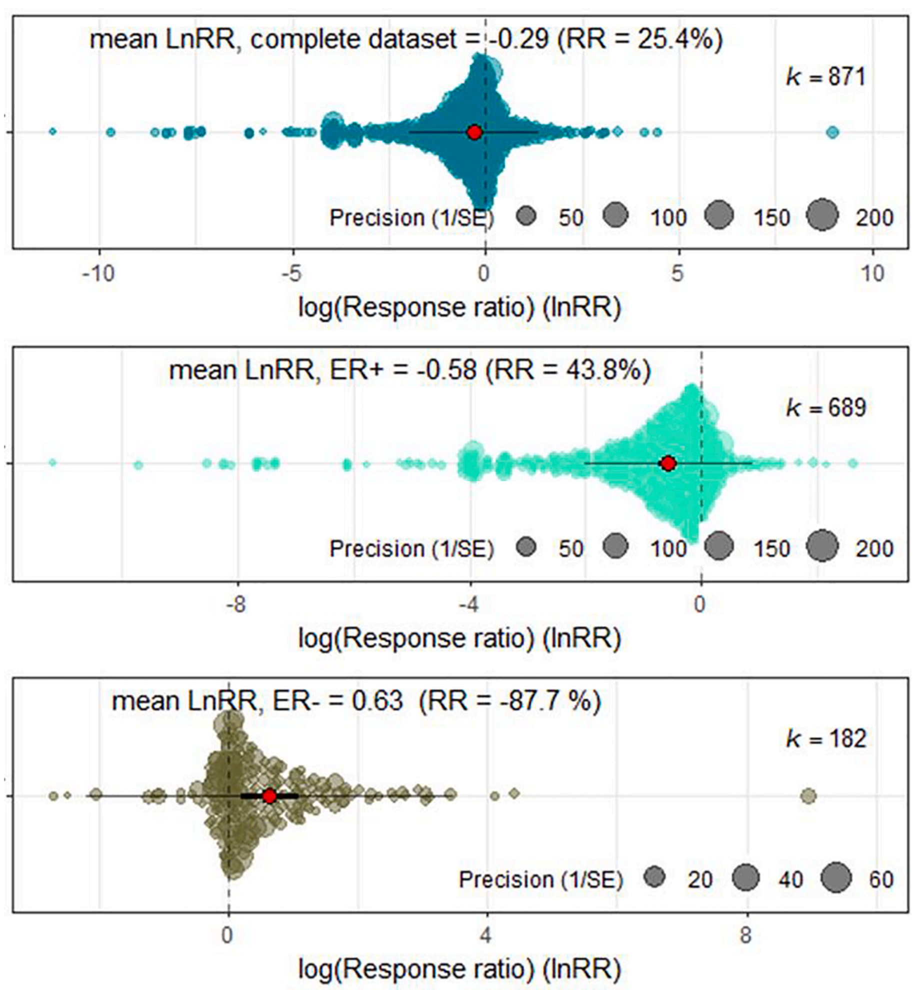

Fig. 1. Distribution of effect sizes estimating the success of ecological restoration actions as the log-response ratio between ecosystem attributes in restored vs. reference states. From top to bottom: Whole dataset (n = 871), ecological restoration actions aiming at increasing (ER^{+}, n = 689) or decreasing (ER^{-}, n = 182) the levels of ecosystem attributes. The x-scale is not always centered in the same way between the three panels.

restoration operations.

By splitting our dataset into two subsets we estimated that, on average, >40% of the work remains to be done to reach the reference state after ecological restoration operations aiming at increasing the level of an ecosystem attribute. This figure reaches 88% when the goal of the action is to diminish the targeted attribute (the minus sign indicates here that the value of the restored state was initially higher than that of the degraded state). Such values exceed those previously reported for a worldwide range of ecosystem types (Meli et al., 2014, 2017; Barral et al., 2015; Rey-Benayas et al., 2017). In wetlands, biological structure and biogeochemical functioning values were evaluated, on average, 26% and 23% lower than those in reference sites by Moreno-Mateos et al. (2012). Similarly, losses in restored ecosystems, as compared to reference ecosystems, were estimated to range between 27% and 33% for species diversity, 32% and 42% for the carbon cycle, and 31% and 41% for the nitrogen cycle (Moreno-Mateos et al., 2017). All these values were lower than our reported values of remaining recovery, except the organism abundance value which was estimated to reach 46% to 51% (Moreno-Mateos et al., 2017).

From now on, for the sake of clarity, both subsets are considered separately to avoid misleading in grand mean LnRR calculation, and we will focus our discussion on the ER^+ subset (79% of the whole dataset).

3.2. The achieved recovery index (AR)

Even when calculated correctly, LnRR (or its derived RR) only provides partial information since it only tells about '*what remains to be done*', but not on '*what has been done*', both information being not correlated (**Appendix E,** Fig. 2). Indeed, a success or a failure after an ER operation can provide the same RR value, since it only informs on the remaining work to do and not on the work performed (Fig. 2). We thus introduce a novel index, the '*Achieved Restoration*' index (AR), that simultaneously accounts for the degraded, restored, and reference states.

The AR index provides information not only on '*what has been done*', but more precisely on '*what has been done compared to what should be done*'. It does not replace the RR index but completes it (Fig. 2). The AR and RR index are not correlated (**Appendix F**). Conversely to LnRR (or RR), AR allows to distinguish failures (e.g. in ER^+ subset: AR < 0, LnRR < 0) from successes (e.g. in ER^+ subset: 0 < AR < 100, LnRR < 0) after ER operations (table 1). Such an index is in line with principles 5 and 6 recently stated by Gann et al. (2019) about the use of tools, indices, and methods allowing to better track progress after ER operations.

3.3. What does AR bring to the debates around ecological restoration

The influence of the type of implemented restoration actions (namely passive vs. active restoration) is still in debate (Prach and Hobbs, 2008; Jones et al., 2018). Both types of restoration provide contradictory remaining recovery (Meli et al., 2014, 2017). Moreover, as reported by Jones et al. (2018), few studies have directly compared different restoration actions in the same location after the same disturbance (Mench et al., 2018). In this context, recent meta-analyses (Crouzeilles et al., 2017; Jones et al., 2018) have favored passive versus active restoration. In our meta-analysis, the remaining recovery was overall clearly higher when implementing passive than following active restoration (Fig. 3), which tends to argue in favor of active restoration. However, although the difference was not significant, the fraction recovered (i.e. the work performed) was greater in the case of passive restoration (AR = 60%) by comparison to the active restoration (45%). Both seemingly contradictory results can be explained to the light of the degradation intensity. It emerges from our results, even if here again differences were not significant, that the greater the initial degradation, the more practitioner's resort to passive restoration. Therefore, even though passive restoration would be globally more efficient than active restoration, the higher RR associated with it is probably partly related to the intensity of the initial degradation, which is globally higher in passive than in active restoration.

Moreover, the relevance of the restoration type should not be

Table 1
Range of expected values for the response ratio (LnRR) and Achieved Recovery (AR, %) indices for the three scenarios observed following ecological restoration (ER) operations. REF: Reference.

		ER failed	ER worked	ER exceeds the expected REF state
ER^+	*LnRR*	<0	<0	>0
	AR	<0	0–100	>100
ER^-	*LnRR*	>100	>100	<0
	AR	<0	0–100	>100

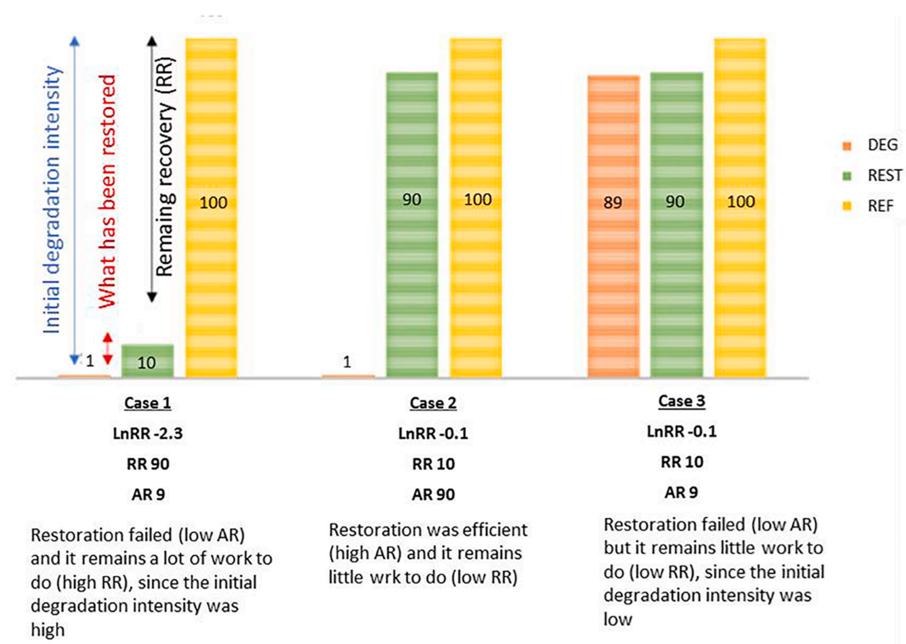

Fig. 2. Graphical illustration of how the AR index (Achieved Restoration, %) complements the LnRR index (Remaining Recovery). Case studies correspond to theoretical situations. Red, green and yellow bars, respectively, represent the degraded (Deg), Restored (Rest) and Reference (Ref) states. LnRR corresponds to Ln(Rest/Ref), RR (%) is calculated as 100 * (1 - e^{LnRR}), and AR as 100*[(Rest-Deg)/(Ref-Deg)]. The three case studies correspond to situations where the restoration operation (here ER +) correctly worked (see Table 1). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

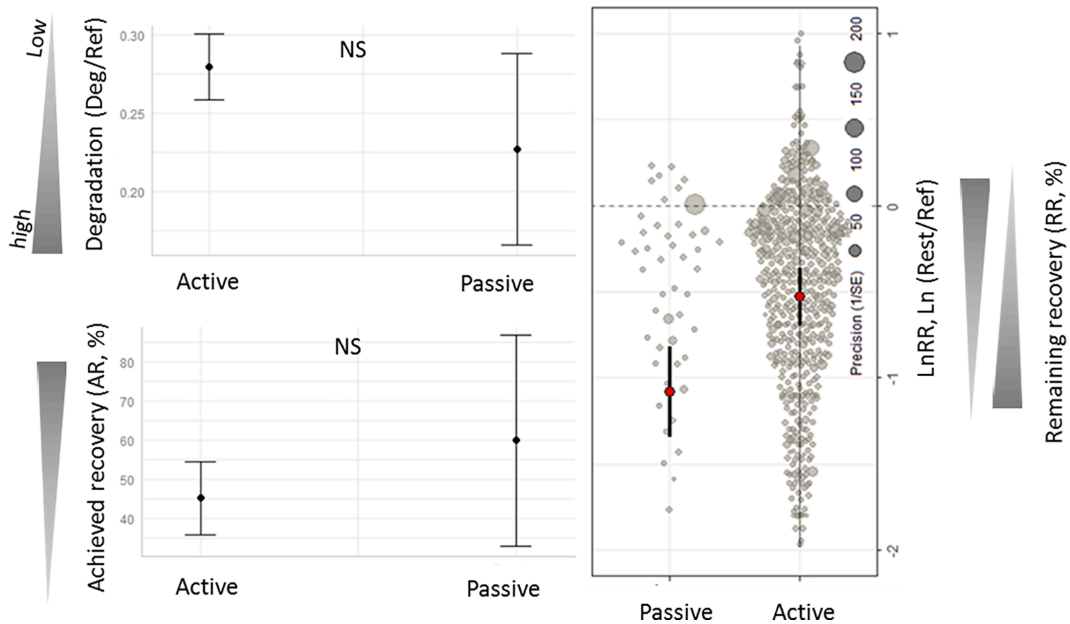

Fig. 3. Coefficient parameter estimates (±95% CI represented by bars) of models decoupling active and passive restorations. Top left: initial degradation level (glm model), bottom left: Achieved recovery, AR (glm model), right: Log ratio of Remaining recovery, LnRR (rma.mv model). Grey arrows indicate the level of the three indices. For LnRR, a fourth arrow informs on the resulting remaining recovery (RR). NS: non-significant.

considered alone, but in the light of the time since restoration took place. For instance, up to decades are left to a forest system for being restored, but not more than a few years to sites such as opencast mines, independently of the restoration type. Here, the remaining recovery slowly decreased with the elapsed time since restoration was undertaken, but time was not the most relevant (Fig. 4). Such a result agrees with other meta-analyses showing a positive correlation between time since restoration started and recovered levels (Cole et al., 2014; Crouzeilles et al., 2016; Meli et al., 2017). Conversely, other studies highlighted an absence of correlation between the recovery and the elapsed time (Meli et al., 2014; Barral et al., 2015; Jones et al., 2018). Here, we reported no influence of the elapsed time on the AR index. The role of time is intimately linked to the nature of the restoration operations implemented and the targeted ecosystem. Time is sometimes decisive in the success of an ER operation (e.g. restoration of a forest cover), but it can also be of more relative importance by comparison to other moderators (e.g. the management mode in the case of grassland restoration). Additionally, the restoration trajectories are not necessarily linear, and exhibit slowdowns, plateaus and reversals (Suding & Hobbs, 2009). Thus, although active restoration may reduce the time to achieve plateaus compared to passive restoration, active restoration can also be involved in a longer duration of these plateaus, partly due to an incomplete implementation of biotic interactions after active restoration (Pocock et al., 2012).

Based on our dataset, ER efficiency tends to be lower in wetlands than in grasslands or forests, even if the differences observed were not significant. One explanation is that in wetlands, where turnover times for species and nutrient pools are quicker compared to forest and grasslands, restoration may promote one or a few functional groups without necessarily benefit the others. This may lead to unbalanced gains in biodiversity (Pocock et al., 2012), and thus to a lower achieved recovery by comparison to what was expected. However, in our study wetlands displayed similar overall RR by comparison to forests; grasslands being the habitat where most work remains after the completion of ER operations. This pattern can be explained by the initial degradation intensity before ER implementation. Indeed, in this study, it is the lowest in wetlands, followed by forests and grasslands ($p < 0.01$, Fig. 5). We propose the following scenarios: wetlands are generally less degraded

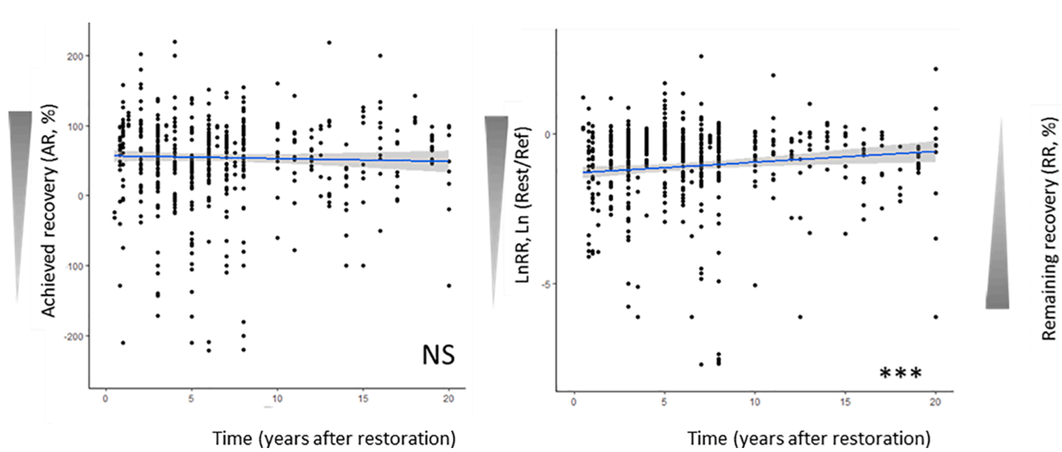

Fig. 4. Relationship (linear regression, ± 95% CI) between the achieved recovery, AR (left), or the Log ratio of Remaining recovery, LnRR (right), and the time elapsed since the restoration operation started (x-axis). Grey arrows indicate the level of both indices. For LnRR, a fourth arrow informs on the resulting remaining recovery (RR). NS: non-significant, ***: $p < 0.001$.

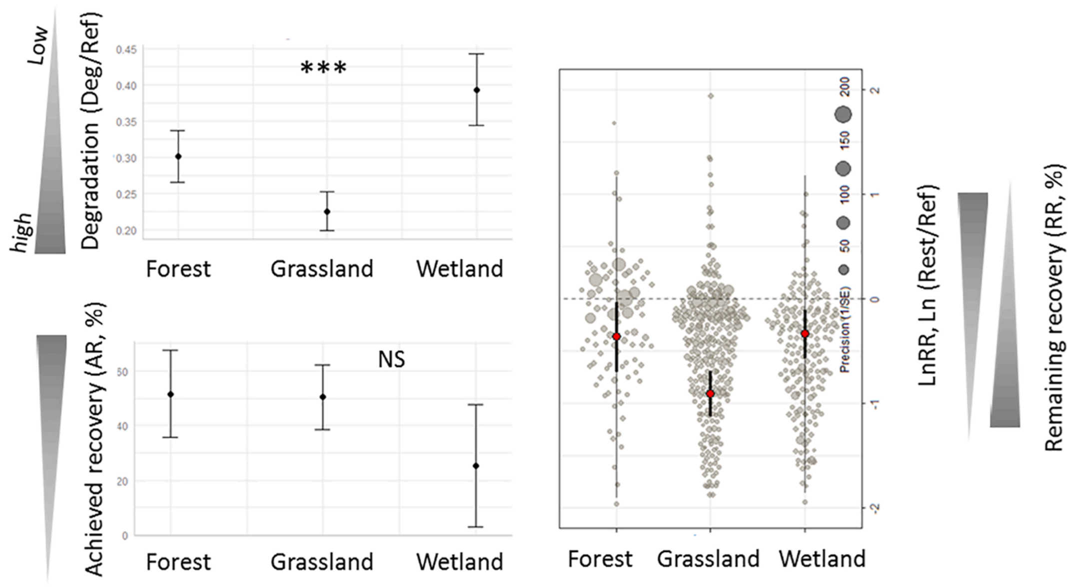

Fig. 5. Coefficient parameter estimates (±95% CI represented by bars) of models decoupling the habitat types where restoration was implemented. Top left: initial degradation level (glm model), bottom left: Achieved recovery, AR (glm model), right: Log ratio of remaining recovery, LnRR (rma.mv model). Grey arrows indicate the level of the three indices. For LnRR, a fourth arrow informs on the resulting remaining recovery (RR). NS: non-significant, ***: $p < 0.001$.

than forests at the beginning of an ER operation, but as restoration efficiency is lower, the level of restoration achieved in the end is similar in both habitats (similar RR). Regarding grasslands, although the overall restoration efficiency is high there, their level of degradation being initially higher than in forests and wetlands, their RR remains the strongest. The difficulty of predicting the results of ER in grasslands is often highlighted (*sowing gets you something but, it may not be the target vegetation*, Smith et al., 2017). We show here that grassland restoration can sometimes work well, but as it is implemented on the most degraded habitat compared to forests and wetlands, it is more difficult to reach an equivalent RR level.

The influence of biomes and the ecological compartments targeted (plants, invertebrates, etc.) were also investigated in this study (**appendix G and H**), but none of them displayed relevant effects on both AR and RR. The effects of these moderators are probably overshadowed by the stronger effect of moderators, such as the type of restoration or the habitat.

4. Conclusion

As we pointed out here, mixing results from two distinct restoration actions in meta-analyses (increasing vs. diminishing an attribute level of the degraded ecosystem) leads to overestimating their success. We recommend treating separately restoration operations aiming at increasing ecosystem attributes from those aiming at decreasing them in future meta-analyses. We also suggest completing the conventional RR approach by a novel index, the AR index, which informs on '*what was done by comparison to what should be done*'. Our meta-analysis allows identifying the intensity of the initial degradation as a pivotal driver leading to success or failure after restoration. Initial degradation encompasses the local site history, the sign, and the strength of the degradation of communities and interaction networks that structure ecosystems. Consequently, the characterization of initial degradation is essential to support practitioners when designing restoration projects and estimating their outcomes. Here, passive restoration appears to work better than active restoration (higher AR index). However, *'what remains to be done'* remained higher in the cases of passive restoration, due to a stronger initial degradation intensity. The habitat would also be a driver of the success or failure in ER operations, being grasslands more difficult to restore than wetlands and forests. Once again, such outcomes are related to the initial degradation, higher in grasslands. We thus suggest doing a priori identification of degradation impact on community structure and on species interactions before selecting the restoration type, and them, to complete RR by AR when assessing the success or failures of ER operations.

CRediT authorship contribution statement

Lilian Marchand: Conceptualization, Writing - original draft, Supervision, Funding acquisition. **Bastien Castagneyrol:** Conceptualization, Writing - original draft. **Juan J. Jiménez:** Conceptualization, Writing - original draft. **Jose M. Rey Benayas:** Conceptualization, Writing - original draft. **Marie-Lise Benot:** Writing - review & editing. **Carolina Martínez-Ruiz:** Writing - review & editing. **Josu G. Alday:** Writing - review & editing. **Renaud Jaunatre:** Writing - review & editing. **Thierry Dutoit:** Writing - review & editing. **Elise Buisson:** Writing - review & editing. **Michel Mench:** Writing - review & editing. **Didier Alard:** Writing - review & editing. **Emmanuel Corcket:** Writing - review & editing. **Francisco Comin:** Conceptualization, Writing - original draft, Funding acquisition.

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.

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Appendix A. Supplementary data

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