Received: 22 August 2022 | Accepted: 19 December 2022

DOI: 10.1111/1365-2664.14364

Journal of Applied Ecology BRITISH ECOLOGICAL SOCIETY

RESEARCH ARTICLE

Synthesis on the effectiveness of soil translocation for plant community restoration

Gijs M. Gerrits[1,2] | **Rik Waenink**[3] | **Asa L. Aradottir**[4] | **Elise Buisson**[5] | **Thierry Dutoit**[5] | **Maxmiller C. Ferreira**[6] | **Joseph B. Fontaine**[7] | **Renaud Jaunatre**[5,8] | **Paul Kardol**[9] | **Roos Loeb**[10] | **Sandra Magro Ruiz**[11] | **Mia Maltz**[12] | **Meelis Pärtel**[13] | **Begona Peco**[14] | **Julien Piqueray**[15] | **Natasha A. L. Pilon**[16] | **Ignacio Santa-Regina**[17] | **Katharina T. Schmidt**[18] | **Philip Sengl**[19] | **Rudy van Diggelen**[20] | **Daniel L. M. Vieira**[21] | **Wolfgang von Brackel**[22] | **Pawel Waryszak**[23,24] | **Tim J. Wills**[25] | **Rob H. Marrs**[26] | **E. R. Jasper Wubs**[2]

[1]Mathematical and Statistical Methods Group (Biometris), Wageningen University and Research, Wageningen, The Netherlands; [2]Department of Terrestrial Ecology, Netherlands Institute of Ecology (NIOO-KNAW), Wageningen, The Netherlands; [3]Plant Ecology and Nature Conservation Group, Wageningen University and Research, Wageningen, The Netherlands; [4]Faculty of Environmental and Forest Sciences, Agricultural University of Iceland, Reykjavik, Iceland; [5]Mediterranean Institute of Biodiversity and Ecology (IMBE), Avignon University, Aix Marseille University, CNRS, IRD, Avignon, France; [6]Ecology Graduate Program, Institute of Biological Sciences, University of Brasília, Brasília, Brazil; [7]Environmental and Conservation Sciences, Murdoch University, Perth, Australia; [8]University of Grenoble Alpes, INRAE, LESSEM, St-Martin-d'Hères, France; [9]Department of Forest Ecology and Management, Swedish University of Agricultural Sciences (SLU), Umeå, Sweden; [10]B-WARE Research Centre, Nijmegen, The Netherlands; [11]Creando Redes NatCap S.L., Madrid, Spain; [12]Riverside, Center for Conservation Biology, University of California, Riverside, California, USA; [13]Institute of Ecology and Earth Sciences, University of Tartu, Tartu, Estonia; [14]Terrestrial Ecology Group (TEG), Department of Ecology, Institute for Biodiversity and Global Change, Autonomous University of Madrid, Madrid, Spain; [15]Natagriwal ASBL, Site de Gembloux., Gembloux, Belgium; [16]Institute of Biology, University of Campinas (UNICAMP), Campinas, Brazil; [17]Institute of Natural Resources and Agrobiology of Salamanca, Salamanca, Spain; [18]Center for Environmental Biology, University of California, Irvine, California, USA; [19]Engineering office for Biology, Sankt Anna am Aigen, Austria; [20]Department of Biology, Ecosystem Management Research Group, University of Antwerp, Antwerp, Belgium; [21]Brazilian Agricultural Research Corporation, Embrapa Genetic Resources and Biotechnology, Brasília, Brazil; [22]Büro für Vegetationskundlich-Ökologische Gutachten & Lichenologie, Röttenbach, Germany; [23]Deakin University, Burwood, Victoria, Australia; [24]University of Southern Queensland, Toowoomba, Queensland, Australia; [25]The Ecology Office Pty Ltd, Melbourne, Victoria, Australia and [26]School of Environmental Sciences, University of Liverpool, Liverpool, UK

Correspondence
Gijs M. Gerrits
Email: geiger-1@hotmail.com

Funding information
British Ecological Society, Grant/Award Number: LRB19/1004; Estonian Research Competency Council, Grant/Award Number: PRG609; European Regional Development Fund; Koninklijke Nederlandse Akademie van Wetenschappen, Grant/Award Number: 019.181EN.01

Handling Editor: Jin-tian Li

Abstract

1. Many degraded ecosystems need active restoration to conserve biodiversity and re-establish ecosystem function, both highlighted targets of the UN Decade on Ecosystem Restoration and the proposed EU Nature restoration law. Soil translocation, where both plant propagules and their associated soil biota are co-introduced, has increasingly been proposed as a powerful restoration technique for terrestrial ecosystems. However, a synthesis of the effectiveness of this method across ecosystems is lacking.
2. To address how soil translocation affects restoration success, we performed a meta-analysis synthesizing data from 46 field experiments and their respective

---

Gijs M. Gerrits and Rik Waenink contributed equally to the study.

reference ecosystems in 17 countries across four continents. In each experiment, vegetation composition was recorded in response to soil translocation treatments and the resultant vegetational changes (diversity and composition) were quantified.

3. We found that soil translocation leads to plant community development further away from the control and more towards the reference plant communities compared with treatments where only plant propagules were introduced. However, the variability of effect sizes among experiments was large, suggesting strong dependence of restoration success on restoration context. We found that restoration success was more likely on loamy soils and when translocation treatments were implemented over larger spatial areas (>180 m^2).
4. Furthermore, we found that restoration success either consistently increased or decreased over time depending on the experiment. Not only is this congruent with positive feedbacks between plant and soil communities driving plant community development, but it also suggests that the composition of the translocated plant and soil communities, and initial starting conditions, are critical for long-term restoration success.
5. *Synthesis and applications*. Our analysis highlights soil translocation can be a successful restoration method across a broad range of ecosystems. However, its implementation needs to depend on a thorough evaluation of local conditions and the potential added value. Further refinement of soil translocation techniques is needed to increase success rates.

KEYWORDS
above-ground–below-ground interactions, degraded soils, environmental filters, meta-analysis, restoration thresholds, soil inoculation, soil transfer

1 | INTRODUCTION

Biodiversity is declining world-wide due to human activities, which are causing ecosystem degradation (IPBES, Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, 2019). In response, the UN launched the 'Decade on Ecosystem Restoration' to 'prevent, halt and reverse' ecosystem degradation, with one of its main aims is to identify good restoration practices (UNEA, 2019). Ecological restoration, however, remains a difficult and often unpredictable process where success rates remain below expectations; suggesting strong context dependence on specific starting points and creating large recovery debts (i.e. the difference between restoration objective and result; Eviner & Hawkes, 2008; Van Diggelen et al., 2001). Hence, there is an urgent need for both new and innovative ecological restoration techniques as well as rigorous assessments of the effectiveness of existing methods (Cooke et al., 2019; Young & Schwartz, 2019). Traditionally, restoration techniques have focussed on restoring above-ground communities with the aim of bringing back a full complement of target species. Here abandonment of degraded land with no further restoration measures remains one of the most widely applied restoration strategies (Prach et al., 2021). This approach, however, is not always effective. Ecological research suggests that in terrestrial ecosystems there is an intimate link between above- and below-ground biodiversity (Bardgett & van der Putten, 2014; Ohgushi et al., 2018) and restoration practises could benefit from leveraging these interactions (Kardol & Wardle, 2010).

Plant community development is thought to depend on three main community assembly filters (Grman et al., 2015; Hobbs & Norton, 2004) that shape local species composition contingent on the regional species pool. These filters are as follows: (1) dispersal, (2) abiotic conditions (e.g. hydrology, soil type) and (3) biotic interactions, such as competition, herbivory and facilitation (Aavik & Helm, 2018; Zobel et al., 1998). It is increasingly clear that biotic interactions, particularly plant–soil interactions, can strongly affect community composition (Heinen et al., 2020; Ohgushi et al., 2018; Radujković et al., 2020). Plants and their associated soil biota interact via multiple feedback mechanisms and these plant–soil feedbacks can be important drivers of community assembly (Bever et al., 2015; Van der Putten et al., 2013). Plants interact directly with herbivores, parasites and mutualists such as mycorrhizal fungi that affect establishment and maintenance in the community (Brown & Gange, 1990; Neuenkamp et al., 2019), but there are also indirect feedbacks via soil structure and nutrient cycling (Wardle et al., 2004). After abandonment of agricultural land, early-successional plants were shown to generate negative plant–soil feedback, whereas later-successional species, the typical targets for restoration, generate more positive feedback (Carbajo et al., 2011; Kardol et al., 2006). Hence, the recovery of plant–soil interactions may alleviate restoration thresholds and effectively steer plant community restoration (Wubs et al., 2016).

If certain abiotic or biotic thresholds have been crossed to such an extent that simple abandonment of degraded land will no longer develop towards restoration objectives (Harris & van Diggelen, 2008), active intervention is needed (Perring et al., 2015; Suding et al., 2004). This includes active restoration of abiotic conditions such as increasing water-tables and topsoil removal to reduce excessive nutrient levels (Emsens et al., 2015; Geissen et al., 2013). Dispersal limitation can be overcome by introducing plant propagules through seeding or hay addition (Edwards et al., 2007; Freitas et al., 2019). Soil translocation as a restoration technique also seems to have originated mainly as a way of transferring plant propagules (Pärtel et al., 1998; Pywell et al., 1995). Soil translocation consists of transferring an amount of topsoil (either intact as turfs or broken up as spreadable crumbs or soil slurry) from a donor site to a restoration site, combining the transfer of plant propagules (defined here as any plant material that functions in propagating an individual, that is, seeds and resprouting structures such as rhizomes, bulbs, etc.) with concomitant transfer of soil biota (Harris & van Diggelen, 2008). It, therefore, incorporates the potential benefits of a functional soil community because it co-introduces plant propagules and their associated soil organisms. Soil translocation has been used as a restoration tool in a variety of habitats, with varying degrees of success and across a broad set of goals, ranging from restoring vegetation cover on heavily degraded soils (e.g. former mine and quarry waste; Chenot et al., 2017; Ferreira et al., 2015) to inoculating soil biota and plant propagules for restoring rare and highly protected plant communities such as lowland *Nardus*-grasslands (Loeb et al., 2017). However, a thorough assessment of field studies testing the impact of soil translocation on vegetation has been lacking up to now.

Here, we report on a synthesis of soil translocation experiments from a wide range of environments (boreal to tropical, grasslands to forests) across four continents using plant-propagule-only treatments (hay addition and seeding) as a frame of reference with which to compare the effects of soil translocation. We asked the following questions: (1) Does soil translocation accelerate vegetation development away from the control and towards the reference communities in terms of diversity and composition? (Figure 1) (2) Is soil translocation more effective than introducing plant propagules without soil (i.e. seeding and hay addition)? (3) How does effectiveness of soil translocation vary across environments, time since intervention and translocation method? We tested the following hypotheses: (1) soil translocation results in plant communities that are distinct from the control in terms of their species composition and have become more similar to their respective donor plant communities than the controls; (2) because both plant propagules and their associated soil organisms are co-introduced, we expected soil translocation to be more effective than hay or seed addition, and more effective when soil was translocated as intact turfs (turfing) than as crumbs (spreading) and (3) effects of soil translocation were expected to be stronger when larger volumes of soil were introduced (deeper layer or larger area) due to higher probability of survival of translocated species. For all three questions, we expected that differential effectiveness would be amplified over time due to positive plant–soil feedback. This means that vegetations could develop either in the direction of their reference communities ('hit') or alternatively develop in another direction ('misfit'). We also explored if initial misfits could turn into hits later on and vice versa.

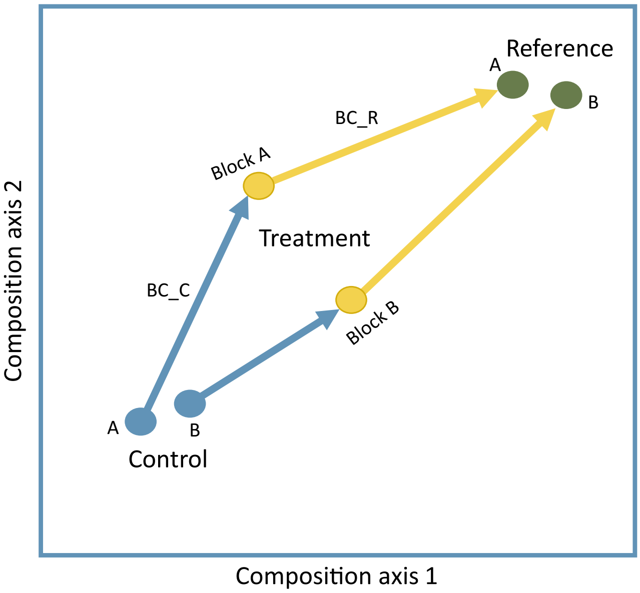

FIGURE 1 Conceptual figure of the analyses used for analysing the available vegetation data. We distinguish control plots (blue), treatment plots (yellow) and reference (green) plots that differ in vegetation composition. For each study, we calculated the compositional dissimilarity between control and treatment plots as the Bray–Curtis distance (BC_C). For those studies that had data from reference plots, we also calculated the similarity between treatment and reference as 1 - Bray–Curtis distance (BC_R). Reference plots can be either plots in the soil donor area or in nearby vegetation that is the desired community. Typically, restoration treatments are expected to shift the vegetation in the direction of the reference vegetation (blue arrows), compared with the controls. It is also expected that with time, treatment plots will develop towards reference vegetation types (yellow arrows).

2 | MATERIALS AND METHODS

We conducted a meta-analysis of soil translocation experiments for biodiversity restoration and conservation. Here, we provide a condensed description of our methods, full details can be found in the

Supplementary Methods. Using Web of Science, Scopus and other sources we found 1377 unique records. Records were screened for the following selection criteria: studies had to report (1) field experiments where the goals were nature restoration or conservation; (2) topsoil from another location was introduced in some form; (3) experiments had to have onsite controls and (4) the entire vascular plant community composition was recorded (i.e. not only a few target species). This filtering produced a selection of 39 studies (Figure S1). To assess the treatment effects on plant community composition, we asked the original authors to provide their primary species composition data collected during field trials. We obtained data from the authors of 32 studies, providing data from 46 experiments for inclusion in the meta-analysis (Data S1).

For each experiment, we compared the vegetation in treated plots with controls (Figure 1). Controls were those experimental plots that were equal to the treatments in all respects but the treatment itself (that is soil or plant material addition) so that we examined the *additional* effect of the treatments. When there were other treatments (e.g. topsoil removal, liming), or blocks in the experimental set-up, comparisons were conducted within these groupings. Where records were available, the vegetation of the donor (i.e. the plant community at the site where the donor soil material came from) or reference (i.e. plant community that serves as a reference for where the treatment ideally develops itself into) sites were also compared with the treated plots and controls (Figure 1). Plots were compared in terms of total plant abundance (cover or biomass), diversity (Shannon index), and composition (using Bray–Curtis distances). We distinguished between two forms of soil translocation (Bullock, 1998): (1) *soil spreading*, that is, the transfer of fragmented and homogenized donor topsoil material and (2) *turfing*, that is, the transfer of intact turfs (monoliths). For experiments that also included treatments where plant material was translocated, we made the same comparisons as for soil translocation. Here, we classified these treatments as (3) *hay addition*, that is, introduction of above-ground plant material that contains seeds and (4) *seeding*, that is, addition of specific reference seed material.

For each study, we extracted information on a range of experiment meta-data (see Data S1) directly from published papers or via primary authors. We recorded the initial and target habitats, the climate, hydrology, soil acidity and texture as environmental metadata (Table S1). In addition, we recorded the treated area and amount of soil that was translocated and whether topsoil was removed prior to soil translocation as methodological factors. While we considered more factors (e.g. soil fertility), these were the variables that we could consistently extract across the wide range of experiments.

2.1 | Data analysis

The experiments form time series that were observed over different lengths of time (<1–30 years, median = 3) and in which records were made for a different number of years (range 1–21 years, median = 3 years, not necessarily consecutive). To deal with this heterogeneity, we created three datasets. First, we took only the final year of observation post-treatment from each study (final year dataset, 46 experiments). In this way, we could include all studies (largest coverage spatially and ecosystem types) and assess the longer-term effects of soil translocation. For the second dataset, we selected all observations made 2 years post-treatment (2-year dataset, 37 experiments), so we could assess the restoration effectiveness at a fixed time-scale. Finally, for the third dataset, we selected studies that monitored the vegetation in at least three different years (time-series dataset, 14 experiments) to study temporal plant community dynamics in response to soil translocation.

All three datasets were analysed with linear mixed models accounting for hierarchical and repeated sampling designs as appropriate. We linked the effects of environmental and methodological predictors (Table S1) with changes in plant community composition using boosted regression trees (BRTs), a form of machine learning. We then assessed the relative influence for each predictor as a percentage of the total variance explained. We fitted linear mixed models with the predictors that had a relative influence >15% to evaluate their statistical significance. Since BRTs produce nonlinear response curves for continuous variables, we simplified those into categorical variables based on visually observed cut-off points in the fitted curves.

We used graphical methods to assess publication bias (Supplementary Methods) and model assumptions (normality and homoscedasticity of errors).

3 | RESULTS

3.1 | Overall analyses

Reference plant communities had on average greater plant abundance (cover or biomass), species richness, but not diversity (Shannon) than the respective controls (Table S2). Of the restoration treatments, only soil spreading and turfing led to greater plant abundance and species richness (log-response ratios >0). Only turfing significantly increased plant diversity (Shannon).

We found that all four experimental treatments strongly impacted vegetation composition relative to the control (Figure 2a, Table S3a,b), with mean Bray–Curtis dissimilarities in the range 0.48–0.58 (possible range 0–1). The reference plant communities were vastly different from the controls (Bray–Curtis dissimilarity = 0.88, 95% CI: [0.83:0.92]), as is to be expected in restoration studies. On average turfing led to the largest changes in vegetation composition and hay addition to the smallest changes. Seeding and soil spreading were intermediate and not significantly different from either turfing or hay addition (Table S3a,b).

In terms of similarity to the reference vegetation (Figure 2b, Table S3c,d), turfing was the most similar, followed by soil spreading and hay addition. The controls were less similar to the reference vegetation, and seeding had the lowest similarity. When comparing only those experiments where both hay addition and soil spreading treatments were implemented ($n = 9$), soil spreading led to a higher similarity to the reference vegetation (Figure 2c, mean difference = +40.4%,

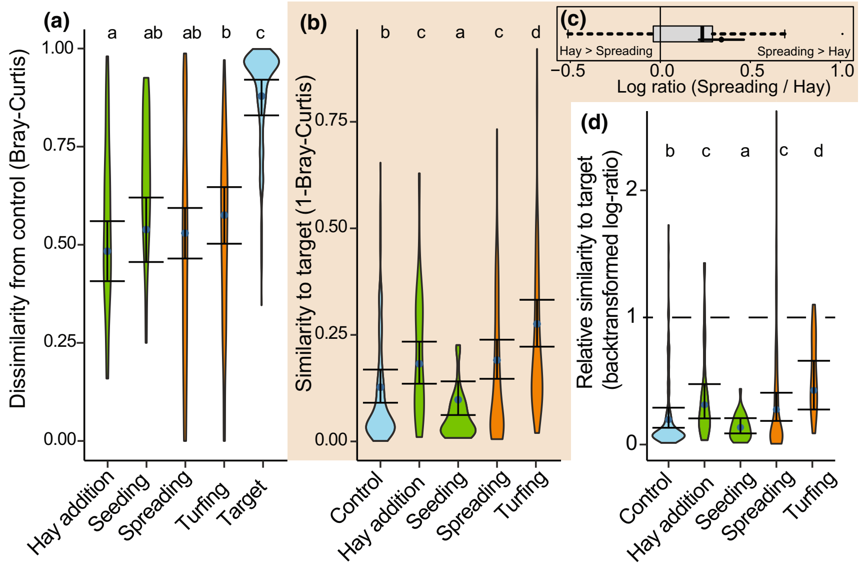

FIGURE 2 Mean effect sizes for treatments in the final year dataset. Shown are Bray–Curtis distances relative to control (a), and Bray–Curtis similarity (1-BC) to the reference vegetation (b). Shown are means (dots), 95% confidence intervals (error bars) and observed data (violin plots). Post-hoc comparisons are given by different letters (see Table S3). (c) Log-ratio of similarity to reference in soil spreading relative to hay addition treatments per experiment where both treatments were implemented (boxplot). A log-ratio of zero (vertical black line) indicates both treatments were equally similar to reference. The black dot and line indicate the model mean ± 95% CI. (d) Vegetation similarity to the reference vegetation relative to the mean similarity among reference vegetation plots (i.e. 100% relative similarity, dotted line at 1, equals the mean similarity among plots surveyed in the reference vegetation).

95% CI = [+23.9%:+58.9%], t_8 = 6.27, p = 0.0002). However, there was a large degree of variation among experiments (range −40% to +175%). Overall, the mean similarity in each treatment was still well below complete similarity (Bray–Curtis similarity = 1), covering a range of 0.10–0.28, indicating that none of the restoration treatments led to a good degree of rehabilitation of the intended reference vegetation within the duration of the experiment. When rescaling the similarity to reference plant community by the mean similarity among reference vegetation plots, this range increased to 0.19–0.42 (Figure 2d).

Compared with the final year dataset, in the second year, the vegetation had changed much more strongly away from the control in seeding treatments (Table S3b). Other treatments showed less pronounced changes relative to final year data. In terms of similarity to the reference vegetation, the effect sizes were on average slightly smaller than in the final year dataset, and there was no longer a significant difference between control and seeding treatments (Table S3d). The overall results were robust to publication bias (Supplementary Methods).

3.2 | Time series

In the 14 experiments in the time-series dataset, the analysis confirms that both soil translocation and plant material addition can have strong effects on plant community composition and that these effects can persist for years to decades (Figure S2). However, the analyses showed that the temporal patterns for both dissimilarity from the control and similarity to the reference vary significantly across experiments (Table S4).

For most experiments, control plots became more similar to the reference plots over time (Figure 3a,c,e–g,i,j), but in two experiments, controls became less similar (Figure 3d,k). In eight out of 11 experiments, soil translocation increased the similarity to reference (Figure 3a–h). Plant material addition had a positive effect in four out of eight experiments (Figure 3a,c,e,f). In four out of eight experiments, soil translocation led to a greater similarity to the reference vegetation than plant material addition (Figure 3a,e,f,i,j), and it reduced similarity to the reference vegetation in one experiment (Figure 3k).

Soil translocation often led to higher similarity to the reference vegetation than the controls early on in the time series (Figure 3a,e,f,g,h) and sometimes to a higher similarity than addition of plant material (Figure 3a,e,f). Later on, plant-only addition treatments tended to catch up after about 10 years (Figure 3a,e,f), although turfing was typically the best treatment over the entire period of monitoring (Figure 3a,c,e). In two cases, soil translocation initially increased similarity to the reference vegetation but was subsequently overtaken by the control treatments (Figure 3g,j). In two other cases, soil translocation lowered the similarity to the reference for long periods of time (Figure 3i,k).

3.3 | Predictors of effectiveness

Notwithstanding the average differences in vegetation changes reported above, for both dissimilarity from control plots and similarity to the reference vegetation, we found large variations among experiments (Figures S3 and S4, Table S3). The application of BRT helped

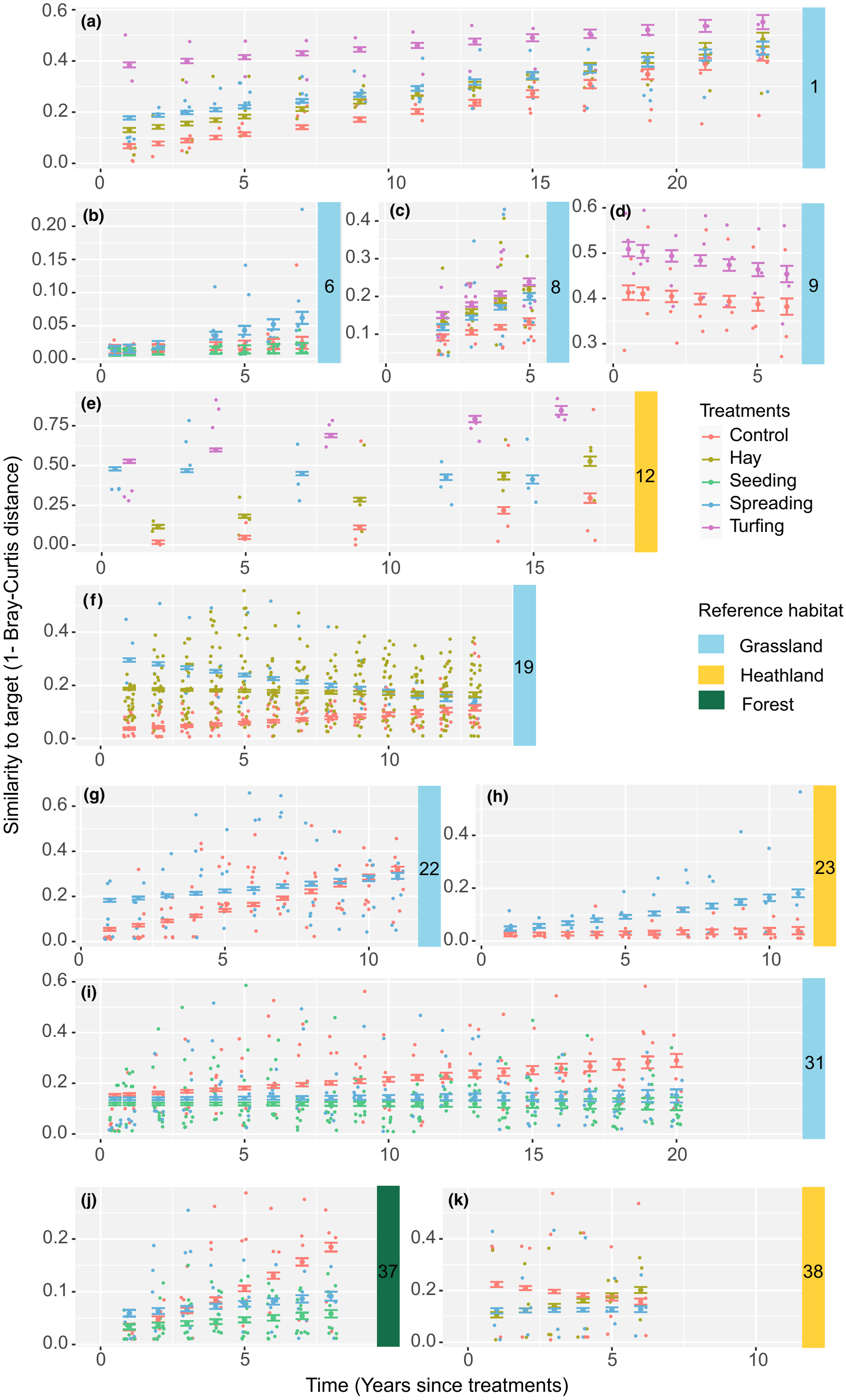
(a)
0.6
0.4
0.2
0.0
0
5
10
15
20
1
(b)
0.20
0.15
0.10
0.05
0.00
6
(c)
0.4
0.3
0.2
0.1
8
(d)
0.6
0.5
0.4
0.3
9
(e)
0.75
0.50
0.25
0.00
12
(f)
19
(g)
22
(h)
23
(i)
31
(j)
37
(k)
38
Similarity to target (1- Bray-Curtis distance)
Time (Years since treatments)
Treatments
Control
Hay
Seeding
Spreading
Turfing
Reference habitat
Grassland
Heathland
Forest

FIGURE 3 Temporal patterns of similarity to the reference in 11 experiments (panels a–k, experiment IDs on the vertical bars, their colours refer to the reference habitat of the experiment) with ≥3 post-treatment years of observations. Shown are raw data (small dots) and model output (Table S4) shown as treatment means (large dots) with their 95% confidence intervals (only accounting for uncertainty in fixed effect parameters).

to reveal which variables (Table S1) were the best predictors of plant community responses (Figure S5). The BRT models explained a significant part of the variation (Table S5), with the model for similarity to reference (R^2 = 62.5%) doing better than the model for dissimilarity from control (R^2 = 27.7%). We found for the linear mixed models with predictors that had a relative influence >15% in the BRT models (Figure S5a,b), that for dissimilarity from the control, a thicker layer of donor soil led to a larger change in vegetation composition (Figure 4a; Figure S6a). However, the model marginal R^2 was quite low (<2%). For similarity to the reference vegetation, the mixed model had a higher marginal R^2 (25.4%), and both predictors identified with BRT were significant (Table 1). Experiments with larger treatment plot sizes (>180 m^2) had a higher similarity to the reference than smaller ones (Figure 4c; Figure S6c). In addition, responses differed by soil texture of the recipient site, with loamy soils leading to higher similarity to the reference plant community (Figure 4b; Figure S6b).

4 | DISCUSSION

Our results show that soil translocation was on average a more effective restoration technique compared with plant propagule-only treatments, supporting our hypothesis on the augmentative role of soil (biota) transfer in restoration outcomes (Figure 5). However, this synthesis also clearly shows that soil translocation, as well as the plant propagule-only treatments, led to strong compositional restoration of plant communities in only a few cases. Similarity to reference in most cases remained well below 50%, meaning that large recovery debts remained at the end of experiments. Moreover, despite significant differences in the mean effect sizes of plant community development between treatments, studies varied widely in the similarity of the resultant vegetation to its reference. This variation was strongly experiment-driven, that is, the effect size of treatments differed across experiments much more than within experiments, suggesting strong site-specific conditionality on restoration success.

Soil translocation treatments showed higher plant abundance as well as species richness than their respective controls. For heavily degraded sites, for example, mine tailings or construction sites, higher plant abundance in the soil translocation treatments could be due to the fact that the added soil provides a substrate for plant establishment and growth. The translocated soil adds organic matter, as well as other chemical and physical properties (e.g. water retention) necessary for germination and survival of plants. Turfing had significantly greater similarity to reference plant communities than soil spreading, presumably because of the transfer of more intact root systems as well as soil mycelial networks due to lower physical disturbance compared with soil spreading (Bullock, 1998). Physical soil disturbance is known to affect soil meso- and macrofauna (Cole et al., 2006) and soil microbes (e.g. tillage effect on fungi; Tsiafouli et al., 2015). In addition, turfing also translocates adult plants that are already established in the vegetation, thereby bypassing the vulnerable seedling stage.

We studied the experiment-level factors influencing restoration effectiveness. The results showed larger spatial treatments to be a strong predictor, leading to greater similarity to the reference vegetation. This might be explained by the fact that a larger treatment surface leads to more soil organisms and/or plant propagules that survive the translocation, leading to a higher chance of establishment and survival, but may also be explained by edge effects playing a larger role in smaller treatment plots. In contrast, while the thicker the soil layer (range 0.02–45 cm) that was added did increase the dissimilarity to the control, it increased the similarity to the reference only marginally (Bulot et al., 2014). Note, however, that donor

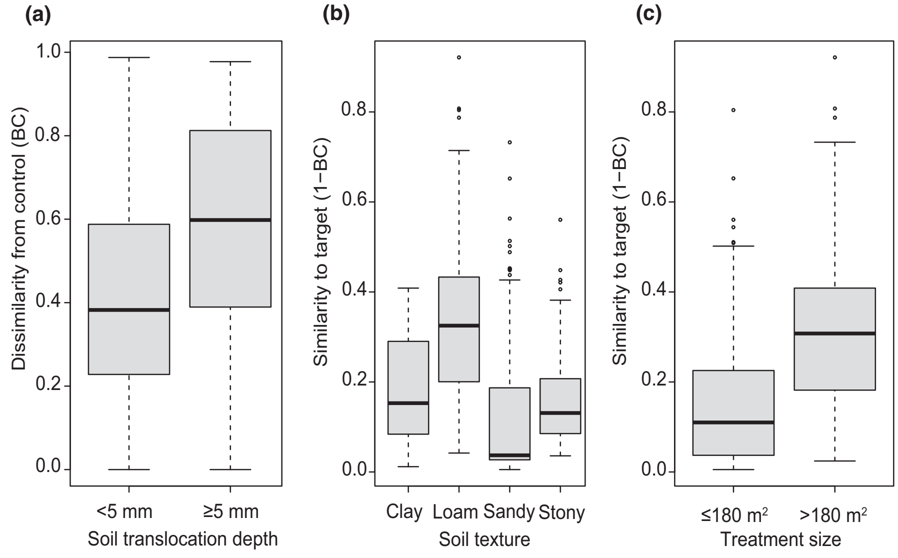

FIGURE 4 Effects of the experiment-level predictors with the highest relative influence (Figure S5) on dissimilarity from control (a) and similarity to reference vegetation (b, c). Continuous predictors (a, c) were recoded to categorical variables based on major changes in the nonlinear boosted-regression tree functions (Figure S6).

TABLE 1 Mixed model results for the effects of selected predictors on plant community responses to soil translocation.

Dissimilarity from control				**Similarity to reference**			
R^2_{marginal}	0.020			R^2_{marginal}	0.254		
$R^2_{\text{conditional}}$	0.746			$R^2_{\text{conditional}}$	0.834		
Predictor	*d.f.*	*F*	*p*	Predictor	*d.f.*	*F*	*p*
Soil translocation depth	1,78	5.63	0.0201	Treatment area class	1,19	5.75	0.0269
				Soil texture class	1,8	9.47	0.0152

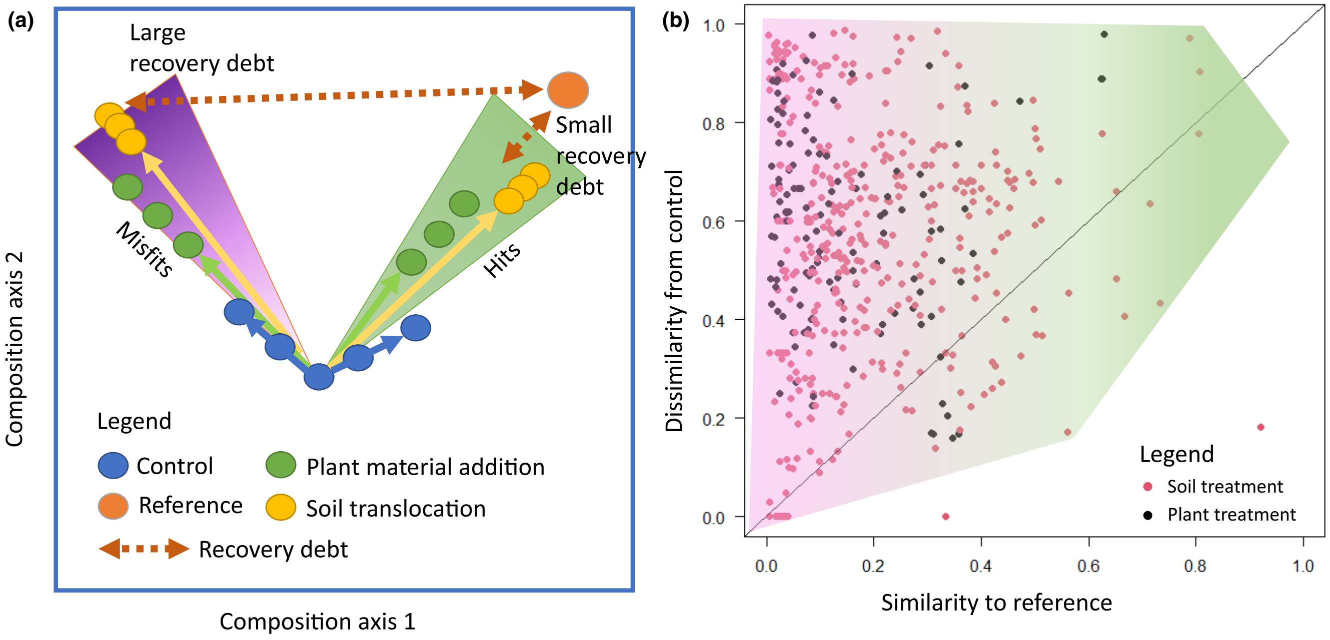

FIGURE 5 Synthesis of results. (a) Both plant material addition and soil translocation lead to strong changes in plant community composition away from the controls that develop more slowly. Soil translocation methods cause more rapid changes, but plant material transfer plots can catch up over time. However, both plant and soil treatments generate long-term plant community trajectories t either develop in the direction of the reference vegetation (the hits) or away from it (the misfits). These compositional legacies (orange arrows) can persist for years to decades and can be positive or negative from a restoration perspective. (b) Data from the final year dataset showing the relationship between the similarity to reference and the dissimilarity from the control. Treatments are generally effective in moving vegetation away from the controls, but many stay very different from the reference plots as well (purple part of the gradient area, misfits from a). Others move away from the controls and become more similar to the reference in proportion (green part of the gradient area, hits from a). The black line is the 1:1 relationship.

soil volume does not explain the higher success rate of turfing, since turfing was done over smaller spatial areas and thus used less net volume than soil spreading. Contrary to our expectation soil texture, but not habitat type, was a strong predictor for restoration effectiveness. It is known that soil texture affects soil restoration trajectories and plant–soil interactions (Bach et al., 2010), and our data suggest this translates into different amenability to soil translocation. Finally, our dataset had substantial missing values for soil acidity and our analysis may thus have underestimated its importance.

4.1 | Time series

We found evidence for strong directionality in the restoration trajectories, with steady gradual temporal developments in a particular successional direction that persisted in most experiments and even on the decadal scale, and in both types of treatments (soil translocation and plant propagule-only). Trajectories were either successful, what we call 'the hits', developing on a trajectory towards reference plant communities, or unsuccessful, termed 'the misfits', remaining on a divergent trajectory relative to their reference vegetation (Figure 5). We had expected to see curvilinear successional developments, where the initial establishment of target plant communities was successful and subsequently diverged away from the reference, or, alternatively, initial slow establishment of target species accelerating after a certain period (Matthews & Spyreas, 2010). However, comparisons of AIC scores for linear and nonlinear (additive) models strongly favoured linear models.

There are strong indications that restoration of target plant communities can take decades to centuries (e.g. Nerlekar &

Veldman, 2020; Rozendaal et al., 2019). Therefore, it was important to analyse successional development in the long-term studies contained in the dataset. Our analyses of plant community development after restoration showed that, most often, both the restored communities as well as the controls increased in similarity to references over time. This was most likely due to controls 'catching up' with the treatments, partly due to natural colonization (Matthews & Spyreas, 2010). However, this convergence between treatment plots and their controls in the experiments could in part also result from the exchange of plant propagules between spatially-close treatment and control plots. This suggests that the differences we report between treatments and control are likely to be underestimated over longer time-scales.

4.2 | Choosing the right technique

Soil translocation, and especially turfing, is a drastic and destructive technique for the donor ecosystems (Pywell et al., 1995; Von Brackel, 2010). Furthermore, it can be logistically complicated and expensive, especially at large scales. The results from our synthesis indicate that soil translocation is not always necessary for successful restoration. In fact, there are many scenarios when less far-reaching techniques are an effective approach to ecological restoration (Lohbeck et al., 2021), that is, when the abiotic conditions are appropriate and source populations of the target vegetation are found close to the restoration site. The added value of soil translocation clearly is case-dependent. Therefore, a thorough assessment of goals, risks and potential benefits must be made before deciding whether or not to implement soil translocation into restoration plans. When the goal is to restore functionality (i.e. restoring vegetation cover) and large volumes of donor material would be needed, sacrificing well-developed donor vegetation is probably not a worthwhile strategy. However, when one is trying to rescue rare habitats from extinction (Bullock, 1998; Le Stradic et al., 2016) and the objective is to recreate intact phytosociological plant communities then the effort and cost that comes with turfing might well be justified.

Soil translocation, in particular, will be more effective than plant propagule-only techniques for vegetation types in which seed- and bud-banks contain starch-storing stems, rhizomes, tubers or roots, which are major mechanisms of persistence and regeneration in certain ecosystems. In tropical grasslands, for instance, resprouting from underground structures by plants after disturbance events is a much more common ecological strategy than re-establishment from seed. In fact, a large percentage of seeds in these seedbanks are dormant or infertile (Pilon et al., 2021; Salazar et al., 2012). Furthermore, both availability and requirement of soil material can be hugely different and dependent on case-specific circumstances. Typically, turfing schemes have been implemented where infrastructural developments would anyhow have destroyed the donor vegetation. The amount of topsoil that is needed is similarly variable. In case of restoring soil in former quarries or road embankments, large amounts are needed. When adding soil biota is the main purpose of soil translocation, small volumes of soil crumbs may suffice.

Our synthesis clearly shows a need for further refinement and testing of soil transfer techniques. For instance, it will be important to test the efficacy of donor material collected via non-destructive methods (e.g. raking). Furthermore, the resilience of donor vegetation to soil collection should be better studied. Some evidence suggests that sod extraction is not invariably detrimental to donor vegetation (Pilon et al., 2019), and some vegetations recover rapidly from this disturbance, but others do not. An important goal of the UN Decade on Ecosystem Restoration is expanding the restoration of degraded ecosystems to much larger surface areas. This means that the possibilities of scaling-up soil translocation programmes need to be tested, for example, by exploring the potential for ex situ cultivation of soil inocula (FAO, ITPS, GSBI, SCBD and EC., 2020).

5 | CONCLUSIONS

This quantitative synthesis shows that soil translocation can be an effective technique for restoring plant community diversity and composition. In addition, soil translocation was more effective than plant-propagule-only treatments, such as sowing and hay addition. However, there was a large variation in the effectiveness across experiments. Based on our data, we highlight that loamy soils are more conducive to soil translocation treatments than others. Moreover, soil translocation had a substantially higher chance of success when applied to larger spatial areas. Nevertheless, more research into the environmental and technical factors for success is needed to achieve the goals of the UN Decade of Ecosystem Restoration. Particularly important are estimating and minimizing the impact on donor ecosystems and scaling up the area of application. In addition, decision-support frameworks should be developed to help determine which restoration techniques are to be used for different restoration end goals.

AUTHOR CONTRIBUTIONS

E. R. Jasper Wubs, Rik Waenink and Rob H. Marrs conceived the ideas and designed the methodology; Asa L. Aradottir, Elise Buisson, Thierry Dutoit, Maxmiller C. Ferreira, Joseph B. Fontaine, Renaud Jaunatre, Paul Kardol, Roos Loeb, Sandra Magro Ruiz, Mia Maltz, Meelis Pärtel, Begona Peco, Julien Piqueray, Natasha A. L. Pilon, Ignacio Santa-Regina, Katharina T. Schmidt, Philip Sengl, Rudy van Diggelen, Daniel L. M. Vieira, Wolfgang von Brackel, Pawel Waryszak and Tim J. Wills collected the data; E. R. Jasper Wubs, Rik Waenink and Gijs M. Gerrits analysed the data; Gijs M. Gerrits and Jasper Wubs led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

CONFLICT OF INTEREST STATEMENT

All authors declare that they have no conflict of interest.

DATA AVAILABILITY STATEMENT

The data are archived in *Figshare* https://doi.org/10.6084/m9.figshare.21777083.v1 (Wubs et al., 2022). A list of data sources used in the study is provided in the Supplementary Information (Data S1).

ORCID

Gijs M. Gerrits https://orcid.org/0000-0002-2315-9677
Elise Buisson https://orcid.org/0000-0002-3640-8134
Thierry Dutoit https://orcid.org/0000-0001-9461-9215
Maxmiller C. Ferreira https://orcid.org/0000-0003-1104-6365
Joseph B. Fontaine https://orcid.org/0000-0002-6515-7864
Renaud Jaunatre https://orcid.org/0000-0001-6970-8304
Paul Kardol https://orcid.org/0000-0001-7065-3435
Sandra Magro Ruiz https://orcid.org/0000-0002-5432-7872
Mia Maltz https://orcid.org/0000-0001-9714-3566
Meelis Pärtel https://orcid.org/0000-0002-5874-0138
Begona Peco https://orcid.org/0000-0003-2149-1438
Julien Piqueray https://orcid.org/0000-0002-5930-8785
Ignacio Santa-Regina https://orcid.org/0000-0002-7312-3605
Katharina T. Schmidt https://orcid.org/0000-0001-9130-9414
Philip Sengl https://orcid.org/0000-0001-6556-4446
Rudy van Diggelen https://orcid.org/0000-0001-8053-945X
Daniel L. M. Vieira https://orcid.org/0000-0002-2709-5308
Pawel Waryszak https://orcid.org/0000-0002-4245-3150
E. R. Jasper Wubs https://orcid.org/0000-0002-2925-9222

REFERENCES

Aavik, T., & Helm, A. (2018). Restoration of plant species and genetic diversity depends on landscape-scale dispersal. *Restoration Ecology*, *26*, S92–S102. https://doi.org/10.1111/rec.12634

Bach, E. M., Baer, S. G., Meyer, C. K., & Six, J. (2010). Soil texture affects soil microbial and structural recovery during grassland restoration. *Soil Biology and Biochemistry*, *42*, 2182–2191. https://doi.org/10.1038/s43017-022-00366-w

Bardgett, R. D., & van der Putten, W. H. (2014). Belowground biodiversity and ecosystem functioning. *Nature*, *515*, 505–511. https://doi.org/10.1038/nature13855

Bever, J. D., Mangan, S. A., & Alexander, H. M. (2015). Maintenance of plant species diversity by pathogens. *Annual Review of Ecology, Evolution, and Systematics*, *46*, 305–325. https://doi.org/10.1146/annurev-ecolsys-112414-054306

Brown, V. K., & Gange, A. C. (1990). Insect herbivory below ground. *Advances in Ecological Research*, *20*, 1–58.

Bullock, J. M. (1998). Community translocation in Britain: Setting objectives and measuring consequences. *Biological Conservation*, *84*(3), 199–214.

Bulot, A., Provost, E., & Dutoit, T. (2014). A comparison of different soil transfer strategies for restoring a Mediterranean steppe after a pipeline leak (La Crau plain, South-Eastern France). *Ecological Engineering*, *71*, 690–702. https://doi.org/10.1016/j.ecoleng.2014.07.060

Carbajo, V., den Braber, B., van der Putten, W. H., & de Deyn, G. B. (2011). Enhancement of late successional plants on ex-arable land by soil inoculations. *PLoS ONE*, *6*(7), e21943. https://doi.org/10.1371/journal.pone.0021943

Chenot, J., Jaunatre, R., Buisson, E., & Dutoit, T. (2017). Long-term effects of topsoil transfer assessed thirty years after rehabilitation of dry alluvial quarries in southeastern France. *Ecological Engineering*, *99*, 1–12. https://doi.org/10.1016/j.ecoleng.2016.11.010

Cole, L., Bradford, M. A., Shaw, P. J. A., & Bardgett, R. D. (2006). The abundance, richness and functional role of soil meso- and macrofauna in temperate grassland-a case study. *Applied Soil Ecology*, *33*(2), 186–198. https://doi.org/10.1016/j.apsoil.2005.11.003

Cooke, S. J., Bennett, J. R., & Jones, H. P. (2019). We have a long way to go if we want to realize the promise of the "Decade on Ecosystem Restoration.". *Conservation Science and Practice*, *1*(12), 1–5. https://doi.org/10.1111/csp2.129

Edwards, A. R., Mortimer, S. R., Lawson, C. S., Westbury, D. B., Harris, S. J., Woodcock, B. A., & Brown, V. K. (2007). Hay strewing, brush harvesting of seed and soil disturbance as tools for the enhancement of botanical diversity in grasslands. *Biological Conservation*, *134*(3), 372–382. https://doi.org/10.1016/j.biocon.2006.08.025

Emsens, W. J., Aggenbach, C. J. S., Smolders, A. J. P., & van Diggelen, R. (2015). Topsoil removal in degraded rich fens: Can we force an ecosystem reset? *Ecological Engineering*, *77*, 225–232. https://doi.org/10.1016/j.ecoleng.2015.01.029

Eviner, V. T., & Hawkes, C. V. (2008). Embracing variability in the application of plant-soil interactions to the restoration of communities and ecosystems. *Restoration Ecology*, *16*(4), 713–729. https://doi.org/10.1111/j.1526-100X.2008.00482.x

FAO, ITPS, GSBI, SCBD and EC. (2020). *State of knowledge of soil biodiversity -status, challenges and potentialities, report 2020*. FAO. https://doi.org/10.4060/cb1928en

Ferreira, M. C., Walter, B. M. T., & Vieira, D. L. M. (2015). Topsoil translocation for Brazilian savanna restoration: Propagation of herbs, shrubs, and trees. *Restoration Ecology*, *23*(6), 723–728. https://doi.org/10.1111/rec.12252

Freitas, M. G., Rodrigues, S. B., Campos-Filho, E. M., do Carmo, G. H. P., da Veiga, J. M., Junqueira, R. G. P., & Vieira, D. L. M. (2019). Evaluating the success of direct seeding for tropical forest restoration over ten years. *Forest Ecology and Management*, *438*, 224–232. https://doi.org/10.1016/j.foreco.2019.02.024

Geissen, V., Wang, S., Oostindie, K., Huerta, E., Zwart, K. B., Smit, A., Ritsema, C. J., & Moore, D. (2013). Effects of topsoil removal as a nature management technique on soil functions. *Catena*, *101*, 50–55. https://doi.org/10.1016/j.catena.2012.10.002

Grman, E., Bassett, T., Zirbel, C. R., & Brudvig, L. A. (2015). Dispersal and establishment filters influence the assembly of restored prairie plant communities. *Restoration Ecology*, *23*(6), 892–899. https://doi.org/10.1111/rec.12271

Harris, J. A., & van Diggelen, R. (2008). Ecological restoration as a project for global society. In J. van Andel & J. Aronson (Eds.), *Restoration ecology* (pp. 3–15). Blackwell Publishing.

Heinen, R., Hannula, S. E., De Long, J. R., Huberty, M., Jongen, R., Kielak, A., Steinauer, K., Zhu, F., & Bezemer, T. M. (2020). Plant community composition steers grassland vegetation via soil legacy effects. *Ecology Letters*, *23*(6), 973–982. https://doi.org/10.1111/ele.13497

Hobbs, R. J., & Norton, D. A. (2004). Ecological filters, thresholds, and gradients in resistance to ecosystem reassembly. In V. M. Temperton, et al. (Eds.), *Assembly rules and restoration ecology* (pp. 72–95). Island Press.

IPBES, Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. (2019). *Report of the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on the work of its seventh session IPBES/7/10/Add.1*. https://ipbes.net/events/ipbes-7-plenary

Kardol, P., Bezemer, T. M., & Van Der Putten, W. H. (2006). Temporal variation in plant-soil feedback controls succession. *Ecology Letters*, *9*(9), 1080–1088. https://doi.org/10.1111/j.1461-0248.2006.00953.x

Kardol, P., & Wardle, D. A. (2010). How understanding aboveground-belowground linkages can assist restoration ecology. *Trends in Ecology & Evolution*, *25*(11), 670–679. https://doi.org/10.1016/j.tree.2010.09.001

Le Stradic, S., Séleck, M., Lebrun, J., Boisson, S., Handjila, G., Faucon, M.-P., Enk, T., & Mahy, G. (2016). Comparison of translocation methods to conserve metallophyte communities in the Southeastern D.R. Congo. *Environmental Science and Pollution Research*, *23*, 13681–13692. https://doi.org/10.1007/s11356-015-5548-6

Loeb, R., Bobbink, R., Frouz, J., Vogels, J., Benetková, P., Van der Bij, A., & Van Diggelen, R. (2017). Ontwikkeling van heischrale graslanden op voormalige landbouwgronden. Technical report VBNE.

Lohbeck, M., Rother, D. C., & Jakovac, C. C. (2021). Editorial: Enhancing natural regeneration to restore landscapes. *Frontiers in Forests and Global Change*, *4*, 13–16. https://doi.org/10.3389/ffgc.2021.735457

Matthews, J. W., & Spyreas, G. (2010). Convergence and divergence in plant community trajectories as a framework for monitoring wetland restoration progress. *Journal of Applied Ecology*, *47*(5), 1128–1136. https://doi.org/10.1111/j.1365-2664.2010.01862.x

Nerlekar, A. N., & Veldman, J. W. (2020). High plant diversity and slow assembly of old-growth grasslands. *Proceedings of the National Academy of Sciences of the United States of America*, *117*(31), 18550–18556. https://doi.org/10.1073/pnas.1922266117

Neuenkamp, L., Prober, S. M., Price, J. N., Zobel, M., & Standish, R. J. (2019). Benefits of mycorrhizal inoculation to ecological restoration depend on plant functional type, restoration context and time. *Fungal Ecology*, *40*, 140–149. https://doi.org/10.1016/j.funeco.2018.05.004

Ohgushi, T., Wurst, S., & Johnson, S. N. (Eds.). (2018). *Aboveground-belowground community ecology*. Springer International Publishing.

Pärtel, M., Kalamees, R., Zobel, M., & Rosén, E. (1998). Restoration of species-rich limestone grassland communities from overgrown land: The importance of propagule availability. *Ecological Engineering*, *10*(3), 275–286. https://doi.org/10.1016/S0925-8574(98)00014-7

Perring, M. P., Standish, R. J., Price, J. N., Craig, M. D., Erickson, T. E., Ruthrof, K. X., Whiteley, A. S., Valentine, L. E., & Hobbs, R. J. (2015). Advances in restoration ecology: Rising to the challenges of the coming decades. *Ecosphere*, *6*(8), art131. https://doi.org/10.1890/ES15-00121.1

Pilon, N. A. L., Assis, G. B., Souza, F. M., & Durigan, G. (2019). Native remnants can be sources of plants and topsoil to restore dry and wet cerrado grasslands. *Restoration Ecology*, *27*(3), 569–580. https://doi.org/10.1111/1365-2745.13456

Pilon, N. A. L., Cava, M. G. B., Hoffmann, W. A., Abreu, R. C. R., Fidelis, A., & Durigan, G. (2021). The diversity of post-fire regeneration strategies in the cerrado ground layer. *Journal of Ecology*, *109*, 154–166. https://doi.org/10.1111/1365-2745.13456

Prach, K., Fajmon, K., Řehounková, K., & Jongepierová, I. (2021). Hierarchy of environmental factors driving restoration of dry grasslands: A multi-site analysis. *Applied Vegetation Science*, *24*(2), e12576. https://doi.org/10.1111/avsc.12576

Pywell, R. F., Webb, N. R., & Putwain, P. D. (1995). A comparison of techniques for restoring heathland on abandoned farmland. *The Journal of Applied Ecology*, *32*(2), 400. https://doi.org/10.2307/2405106

Radujković, D., van Diggelen, R., Bobbink, R., Weijters, M., Harris, J., Pawlett, M., Vicca, S., & Verbruggen, E. (2020). Initial soil community drives heathland fungal community trajectory over multiple years through altered plant–soil interactions. *New Phytologist*, *225*(5), 2140–2151. https://doi.org/10.1111/nph.16226

Rozendaal, D. M. A., Bongers, F., Aide, T. M., Alvarez-Dávila, E., Ascarrunz, N., Balvanera, P., Becknell, J. M., Bentos, T. V., Brancalion, P. H. S., Cabral, G. A. L., Calvo-Rodriguez, S., Chave, J., César, R. G., Chazdon, R. L., Condit, R., Dallinga, J. S., De Almeida-Cortez, J. S., De Jong, B., De Oliveira, A., … Poorter, L. (2019). Biodiversity recovery of Neotropical secondary forests. *Science Advances*, *5*(3), eaau3114. https://doi.org/10.1126/sciadv.aau3114

Salazar, A., Goldstein, G., Franco, A. C., & Miralles-Wilhelm, F. (2012). Seed limitation of woody plants in Neotropical savannas. *Plant Ecology*, *213*(2), 273–287. https://doi.org/10.1007/s11258-011-9973-4

Suding, K. N., Gross, K. L., & Houseman, G. R. (2004). Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology & Evolution*, *19*(1), 46–53. https://doi.org/10.1016/j.tree.2003.10.005

Tsiafouli, M. A., Thébault, E., Sgardelis, S. P., de Ruiter, P. C., van der Putten, W. H., Birkhofer, K., Hemerik, L., de Vries, F. T., Bardgett, R. D., Brady, M. V., Bjornlund, L., Jørgensen, H. B., Christensen, S., Hertefeldt, T. D., Hotes, S., Hol, W. H. G., Frouz, J., Liiri, M., Mortimer, S. R., … Hedlund, K. (2015). Intensive agriculture reduces soil biodiversity across Europe. *Global Change Biology*, *21*(2), 973–985. https://doi.org/10.1111/gcb.12752

UNEA. (2019). United Nations Environment Agency, The United Nations Decade on Ecosystem Restoration Strategy. https://www.decadeonrestoration.org/strategy

Van der Putten, W. H., Bardgett, R. D., Bever, J. D., Bezemer, T. M., Casper, B. B., Fukami, T., Kardol, P., Klironomos, J. N., Kulmatiski, A., Schweitzer, J. A., Suding, K. N., Van de Voorde, T. F. J., & Wardle, D. A. (2013). Plant-soil feedbacks: The past, the present and future challenges. *Journal of Ecology*, *101*(2), 265–276. https://doi.org/10.1111/1365-2745.12054

Van Diggelen, R., Grootjans, A. P., & Harris, J. A. (2001). Ecological restoration: State of the art or state of the science? *Restoration Ecology*, *9*(2), 115–118. https://doi.org/10.1046/j.1526-100X.2001.009002115.x

Von Brackel, W. (2010). Neuanlage von Magerrasen auf Ausgleichsflächen der Stadt München – Vergleich verschiedener Techniken. *Anliegen Natur*, *34*, 24.

Wardle, D. A., Bardgett, R. D., Klironomos, J. N., Setälä, H., Van Der Putten, W. H., & Wall, D. H. (2004). Ecological linkages between aboveground and belowground biota. *Science*, *304*(5677), 1629–1633. https://doi.org/10.1126/science.1094875

Wubs, E. R. J., Van Der Putten, W. H., Bosch, M., & Bezemer, T. M. (2016). Soil inoculation steers restoration of terrestrial ecosystems. *Nature Plants*, *2*(8), 16107. https://doi.org/10.1038/NPLANTS.2016.107

Wubs, E. R. J., Waenink, R., & Gerrits, G. M. (2022). Data from: Synthesis on the effectiveness of soil translocation for plant community restoration. *Figshare*. https://doi.org/10.6084/m9.figshare.21777083.v1

Young, T. P., & Schwartz, M. W. (2019). The decade on ecosystem restoration is an impetus to get it right. *Conservation Science and Practice*, *1*(12), e145. https://doi.org/10.1111/csp2.145

Zobel, M., Van der Maarel, E., & Dupré, C. (1998). Species pool: The concept, its determination and significance for community restoration. *Applied Vegetation Science*, *1*(1), 55–66. https://doi.org/10.2307/1479085

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Gerrits, G. M., Waenink, R., Aradottir, A. L., Buisson, E., Dutoit, T., Ferreira, M. C., Fontaine, J. B., Jaunatre, R., Kardol, P., Loeb, R., Magro Ruiz, S., Maltz, M., Pärtel, M., Peco, B., Piqueray, J., Pilon, N. A. L., Santa-Regina, I., Schmidt, K. T., Sengl, P. … Wubs, E. R. J. (2023). Synthesis on the effectiveness of soil translocation for plant community restoration. *Journal of Applied Ecology*, *00*, 1–11. https://doi.org/10.1111/1365-2664.14364