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RESEARCH ARTICLE

# Best practice—Is natural revegetation sufficient to achieve mitigation goals in road construction?

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## Abstract

**Aims:** The area influenced by road construction is large, and measures to re-establish vegetation in disturbed areas are routinely carried out to reduce impacts on biodiversity. However, goals of mitigation measures are often unclear, and the effects on biodiversity of mitigation measures is rarely monitored. We assessed the effects of different revegetation treatments (natural revegetation, seeding, planting) on vegetation development along highways, and on wildlife crossings of different age.

**Location:** Highways in southeast Norway.

**Methods:** We collected data on vascular plant species, vegetation cover and height, soil grain size and organic matter content, and compared the species composition, richness, and diversity of the restored sites with reference plots in the adjacent target vegetation (mature forest).

**Results:** Our results show a significantly higher richness and diversity in restored plots compared to reference plots, and an increased similarity of species composition over time. Species composition was most similar to reference plots in naturally revegetated plots and seeding seemed to reduce both species and functional trait composition similarity.

**Conclusions:** It is unrealistic that the defined target vegetation will develop on restored sites. Defining a realistic and achievable target vegetation for each road construction project in relation to land use, adjacent vegetation type and successional stage, as e.g., forest edge instead of forest, would be useful. While this may require more effort for management it will translate to higher mitigation success.

KEYWORDS

ecological restoration, mitigation measures, natural recovery, revegetation, road construction, vegetation development

## 1 | INTRODUCTION

The increasing human mobility of the 21st century leads to rising demands on infrastructure worldwide (van der Ree et al., 2015). Road construction projects and the area influenced by their construction work are usually large. Consequently, substantial areas of natural habitat are disturbed, degraded and destroyed. Habitat loss due to anthropogenic land-use changes is one of the major causes for biodiversity loss (IPBES, 2019). One approach to combat the negative impacts from new land use, such as road construction, on biodiversity, is to apply the principles and ideas of ecological restoration within the development and application of mitigation measures (IPBES, 2019; Mckenney & Kiesecker, 2010).

The ecological effects of roads—and consequently, the required mitigation measures—are diverse and range from small-scale impacts close to the road itself, to large-scale landscape impacts that extend away from the road, depending on traits characteristics of the road, traffic, landscape, wind, and species (Coffin, 2007; van der Ree et al., 2015). Road effects on wildlife and vegetation are numerous and mostly negative, contributing to habitat loss and degradation. For wildlife, roads constitute barriers to movement and can lead to increased mortality (Coffin, 2007). Specific road effects on vegetation include changes in light availability, soil and hydrology, and available land, as well as pollution and impacts on dispersal (Coffin, 2007; van der Ree et al., 2015). The latter can be both positive and negative (Bignal et al., 2008; Oldén et al., 2021).

Mitigation measures at different spatial scales include soil restoration, vegetation restoration and the construction of wildlife crossing structures. To restore roadside vegetation, seeding (with native or commercial seed mixtures) is frequently applied (Brekke Skrindo & Anker Pedersen, 2004; Hagen et al., 2014; Krautzer et al., 2011). Other measures include natural revegetation (possibly supported by topsoil addition), planting of trees or shrub species, the use of vegetation turfs ("dispersal islands"), moving entire plant communities (e.g. small grasslands) in large turfs, and spreading of plant material (e.g. hay or chopped-up plant parts) (cf. Aradottir & Hagen, 2013; Auestad et al., 2016; Baasch et al., 2012; Gann et al., 2019; Johansen et al., 2017). The choice of mitigation measures typically depends on the goals of individual road projects, often encompassing aspects of traffic safety, technical issues, and aesthetics, as well as biodiversity. Further, a given project can have many biodiversity aims, for example, facilitation of connectivity for wildlife, establishment of wildlife habitat, and development of a target vegetation type or a functional ecosystem. For the construction of large wildlife crossing structures, development of a vegetation cover and structure to facilitate wildlife use is crucial (Denneboom et al., 2021).

To define undisturbed surrounding vegetation as reference or target vegetation may be unrealistic. Restoration of roadside or wildlife-crossing vegetation to resemble undisturbed vegetation types, as for example mature forests (cf. Gann et al., 2019), may be unlikely, due to severely changed environmental conditions at the site. Further, restrictions, such as road safety zones that prevent the development of forests, or management choices, such as the minimizing of management costs, may preclude the use of undisturbed or mature reference vegetation as restoration target. In such cases, formulating alternative goals, e.g. the establishment of a self-sustainable substitute ecosystem, or temporary replacement ecosystems, may be an option (Aronson et al., 2017).

During the last decade the number of peer-reviewed studies about mitigation measures and restoration of large construction projects has increased (Glista et al., 2009; Stokes et al., 2014). However, existing documentation is still often in the form of gray literature, frequently written in the local language, and rarely publicly accessible. This lack of documentation of measures carried out, combined with a general lack of evaluation of outcomes, restricts the opportunities for assessment of mitigation measures and prevents the establishment of methods to evaluate restoration success systematically (Nilsson et al., 2016), hampering the development of nature-friendly and cost-efficient mitigation solutions. To be able to upscale results of small restoration projects and experiments to use in large-scale restoration, systematic documentation of measures, and of outcomes, is highly needed (Rieger et al., 2014).

In this study, we use road construction projects in Norway as a case to investigate the effectiveness of active revegetation measures commonly used in road construction.

The Norwegian Public Roads Administration (NPRA) is responsible for national highway construction, with commitments including biodiversity and landscape restoration of areas degraded during road construction (Norwegian Public Roads Administration, 2018). Mitigation measures, such as revegetation of roadsides, restoration of temporary rig-areas and access roads, are routinely carried out (Norwegian Public Roads Administration, 2014). However, project-specific aims are rarely formulated, and effects of implemented revegetation measures not systematically evaluated. Consequently, the success of revegetation measures is to be determined.

We investigated the effects of three commonly used mitigation measures: seeding, natural revegetation (with added topsoil) and a combination of planting and natural revegetation (natural/planting). Whereas seeding and natural revegetation is common on roadside sites, the combination of planting and natural revegetation is often used on wildlife crossing structures, to facilitate the development of vegetation acting as shelter for crossing wildlife species. A wide variety of indicators for assessing impacts of mitigation measures are frequently in use (Evju et al., 2020; Wortley et al., 2013). In this study, we included species richness, diversity and composition, as well as indicators of ecological functions by including plant functional traits (Carlucci et al., 2020), while taking into account environmental variation as well as time since construction. The study was carried out at 27 sites in three study areas in southeastern Norway, encompassing road construction projects implemented between 2003 and 2016. The target vegetation, as expressed in the project descriptions, was adjacent forest communities. We therefore evaluated effects of mitigation measures in terms of similarity with this reference community.

We investigate the effectiveness of revegetation treatments on vegetation development in restored plots, and hypothesize that: (1)

revegetation treatments have a significant impact on vegetation development in restored plots, more specifically, that naturally revegetated plots are more similar to reference plots in terms of species richness, diversity and composition, compared to plots that are seeded or planted; (2) the similarity between restored plots and reference plots increases with time since construction/restoration; and (3) time since restoration is more important for the more complex indicator species composition. We expect, however, large differences between restored plots and reference plots, as the target vegetation in our study sites is mature forest. Therefore, we also investigate the functionality of restored plant communities compared to the reference vegetation. Specifically, we ask: (4) to what extent do the different revegetation treatments increase functional similarity of restored versus reference vegetation.

## 2 | METHODS

### 2.1 | Restoration approach

We investigated the effect of three revegetation treatments: (1) commercial seeding (roadsides, access roads); (2) natural revegetation with added topsoil (roadsides, access roads); and (3) natural revegetation with additional planting (on wildlife crossing structures). These are common treatments applied in road construction projects in our study areas (Appendix S1; Norwegian Public Roads Administration, 2014; Norwegian Public Roads Administration, 2018). A more detailed summary of these methodologies can be found in Appendix S2.

### 2.2 | Study areas and study sites

The study was conducted in three study areas located in southeastern Norway (area 1: E6, E16 east and north of Oslo airport; area 2: E6, E18 southeast of Oslo; area 3: E18 southwest of Oslo), in the counties Viken and Vestfold og Telemark in the boreonemoral and south boreal vegetation zones (59°38′30.85″N 10°38′29.81″E, Figure 1). We chose the study areas due to the high density of wildlife crossing structures within road systems, and the frequent use of revegetation measures during construction. The study areas were characterized by small forests, agriculture and urban areas. Estimated annual precipitation in the areas for the normal period 1991–2020 ranged from 868 to 1028 mm, and mean July temperature was 17°C for all areas (Appendix S3). All climate data were retrieved from the local meteorological stations closest to the study sites which had sufficiently long running times for temperature and precipitation. The stations were located between 3 and 30 km from the study sites (The Norwegian Centre for Climate Services, 2022).

We established 27 study sites along highways and on wildlife crossings, nine for each revegetation treatment. A set of requirements was established for the selection of sites (Appendix S4). Study sites were situated along highways and on wildlife crossing structures in sections with wildlife fences. The highways were constructed between 2003 and 2016, and the study sites thus varied in time since restoration (2–15 years). We selected study sites located in or close to a forest, to incorporate wildlife crossing structures that are usually located in such surroundings. Study sites did not include the road safety zone, where vegetation is managed for road maintenance and safety issues, but were established on the outside of wildlife fences away from the road. Forest was the target vegetation type for restoration in all sites. Forests in the area were dominated by *Pinus sylvestris* and *Picea abies*, except in two sites in study area 3, where forests were dominated by *Fagus sylvatica*. Forests in the study areas are mostly managed by forestry, but mature, with a continuous field and bottom layer of vegetation.

### 2.3 | Study design

To sample the vegetation, we chose a special transect design (Appendix S5), to include apparent topographical variation in plant communities due to differences in terrain, such as roadside slopes. The transects were placed in a diagonal cross, with length and angle modified to the study site (Appendix S5). Eight plots of 1 m × 1 m were systematically placed in each transect cross (Figure 2). Four reference plots were established in the adjacent forest at each site (Figure 2). For natural-planting sites, located on wildlife crossing structures located beside roadside sites (either natural revegetation or seeding), we used the same set of reference plots for both treatments. If no roadside sites were established close to a wildlife crossing site, we established reference plots for the crossing structure sites in the adjacent forest. In total we established 296 plots: 216 in restored vegetation (72 per treatment) and 80 in reference vegetation.

### 2.4 | Data collection

Field work was carried out in June–July 2018. In each plot (1 m$^2$), the presence and cover (percentage) of all vascular plant species was recorded and identified to species level if possible, whereas bryophytes and lichens were only recorded as such (nomenclature after Norwegian Biodiversity Information Centre, 2020b). We also identified red- and alien list species (Norwegian Biodiversity Information Centre, 2020a; Norwegian Biodiversity Information Centre, 2020c). We further recorded cover (percentage) and maximum height of the tallest individual for each vegetation layer (herb, shrub, tree layer). The shrub layer was divided into two height classes (50–100, 100–200 cm) and the tree layer into three (200–500, 500–990, >1000 cm).

We measured soil properties for each plot by taking four samples from the plot corners, which were then merged to one sample per plot. We determined soil grain size by touch and classified it into four classes of grain size (modified after Halvorsen et al., 2020, Appendix S6). Soil from the upper 5 cm of the O-horizon was sampled and analyzed with loss on ignition (at 590°C, min. 3 h), as a

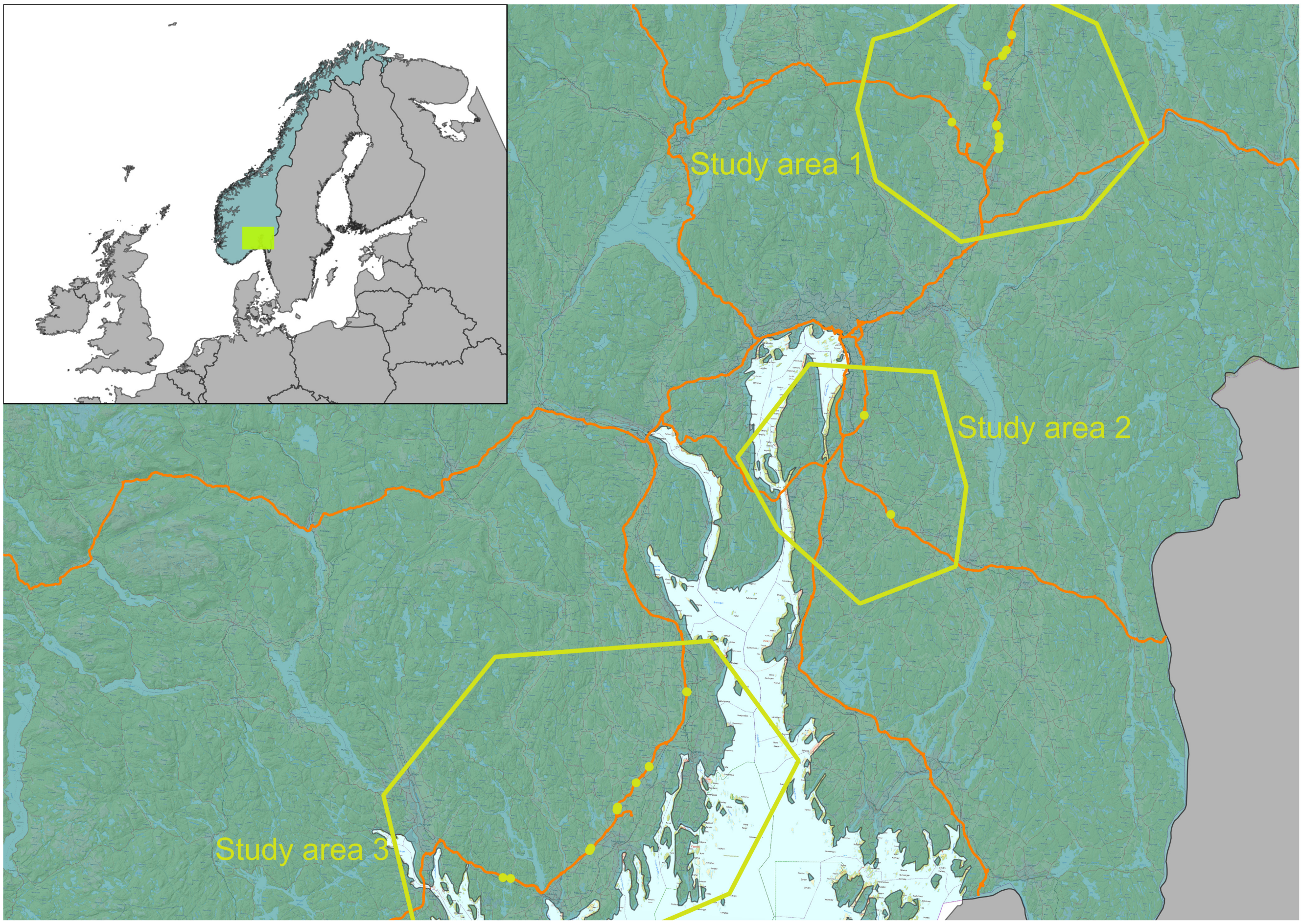


**FIGURE 1** The study areas in southeastern Norway shown as polygons, with points indicating study sites. Major roads are shown in orange

measure of amount of soil organic matter (SOM, varied from 1.71% to 95%, with a mean ± SD of 13.75 ± 18.61) (laboratory: www.nibio.no/en).

Slope and exposure of the plots were recorded with a compass, and canopy cover was measured with the Gap Light Analysis Mobile Application software (GLAMA app, version 3.0), using the Canopy Cover index (CaCo) (Tichý, 2016). Distance to intact vegetation for each plot was measured in Google Earth using the GPS position of each plot (accuracy 3–5 m).

To investigate if functional community composition varied between restored and reference plots, and according to revegetation treatment, we selected plant functional traits with relevance for ecosystem services (Carlucci et al., 2020; De Bello et al., 2010) and restoration (Kollmann et al., 2016). The traits included represented habitat requirements, nutrient acquisition strategies, competitive abilities, importance for pollinators and survival and recovery after disturbance (Table 1). Trait values for competition–stress–ruderal (CSR) strategy were downloaded from the BiolFlor trait base, while trait values for specific leaf area and growth form were downloaded from the LEDA trait base (Table 1). Maximum height was downloaded from the Ecological Flora Database (Table 1). We used the "tr8" function from the *TR8* R-package (Bocci, 2015) to download the data. Indicator values for light, moisture and nitrogen, as well as nectar production, were extracted from Tyler et al. (2021) (Table 1).

We included only plant observations with identification to the species level ($n = 141$ species) in the traits data set. Missing species-by-trait combinations ($n = 33$) were replaced by the median value for the given trait for species from the same genus present in the trait dataset ($n = 3$) or present in another literature source ($n = 29$; www.floraweb.de). For missing species–trait combinations without a suitable replacement ($n = 1$), we used the median value for the trait for all species present in the trait dataset (cf. van Son et al., 2013).

## 2.5 | Statistical analysis

First, to extract gradients in species composition, we subjected the species-by-plot matrix to two ordination methods, as recommended by Son and Halvorsen (2014): detrended correspondence analysis (DCA; Hill & Gauch, 1980) and global non-metric multidimensional scaling (GNMDS; Minchin, 1987). We ran two-, three- and four-dimensional GNMDS and compared stress levels and axes scores with the DCA ordination (see Appendix S7 for details). The three-dimensional GNMDS and the DCA provided three similar

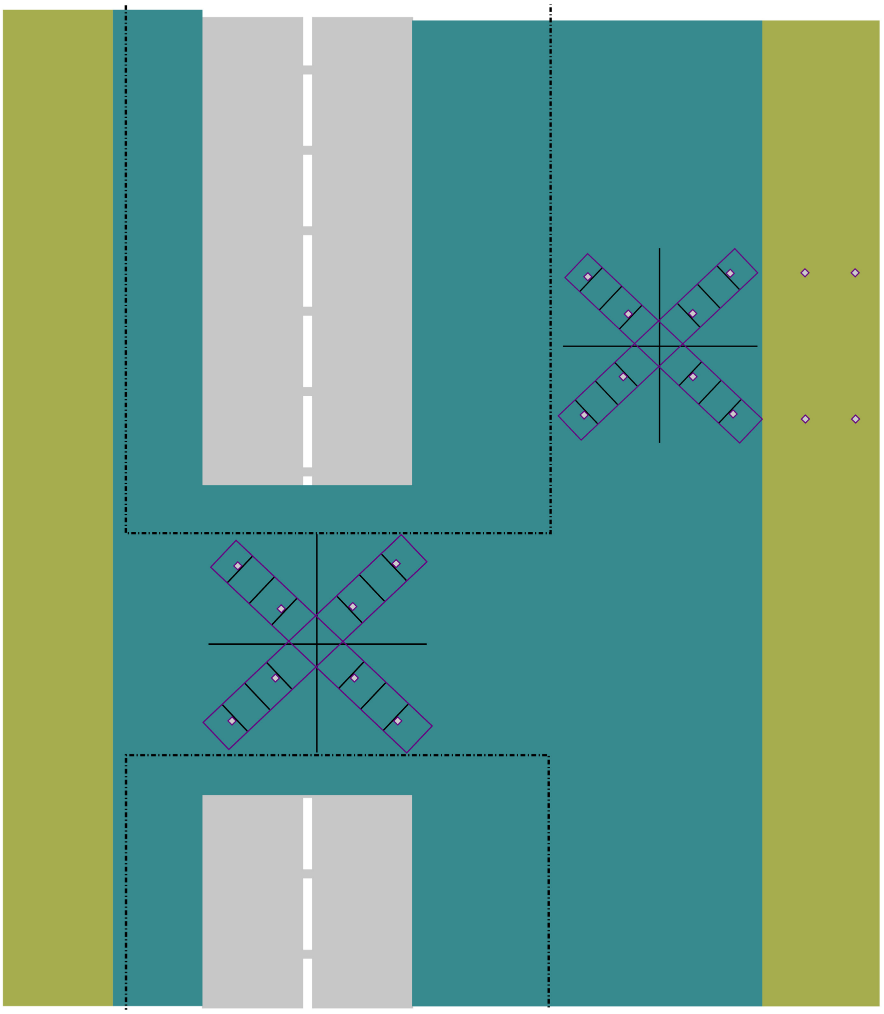

**FIGURE 2** The study design on a wildlife crossing (center) and on the site of a road (upper right) with transects (purple outlines) and restored plots (gray quadrats, 1 m$^2$), with four reference plots in the intact vegetation on the right (light green). Dotted lines indicate wildlife fences

ordination axes (Appendix S7), and the GNMDS was selected for further interpretation. For more information on parameter choice for the GNMDS see Appendix S7.

We investigated the importance of revegetation treatment and environmental variables for the species composition using used linear mixed-effect models (cf. Zuur et al., 2009) with GNMDS axis scores for the plots as response variable, and revegetation treatment, distance to intact vegetation, SOM, canopy cover, and grain size as predictor variables, using site as a random factor. Time since restoration was applicable to revegetated plots. To investigate the importance of time since restoration on species composition, we first extracted the centroid of reference plot scores for each site separately (cf. Rydgren et al., 2019, 2020). We then calculated the successional distance for each restored plot as the difference between the plot score and the centroid of the reference plot scores for the given site. We subsequently modeled successional distance (i.e. difference in species composition) as a function of revegetation treatment, time since restoration, and their interaction, using linear mixed-effect models with site as a random factor (Rydgren et al., 2019).

Next, we used linear mixed-effect models to examine whether plot-level species richness (no. of species per plot) and diversity (calculated as the Shannon diversity index) varied in relation to revegetation treatments and environmental variables. We then explored effect of time since restoration by modeling species diversity and species richness functions of revegetation treatment, time since restoration, and their interaction, using linear mixed-effect models, but excluding reference plots, as time since restoration was not applicable for these plots.

In order to analyze functional trait composition, all continuous traits were centered and standardized by subtracting the mean and dividing by SD, and subsequently scaled to have values between 0 and 1 (cf. Sulavik et al., 2021). Categorical traits were arranged as binary traits, having values of 0 or 1 for each trait category. For categorical traits where more than one category was applicable for a species, values were distributed among categories to sum to 1 (e.g. a species with CS-strategy would have: R = 0, S = 0.5, and C = 0.5). We then calculated the community-weighted means for each trait and plot. To investigate functional trait composition, we ran multiple parallel ordinations (as described above, see Appendix S7 for details). The four-dimensional GNMDS and the DCA provided four similar ordination axes (Appendix S7), and the GNMDS was selected for further interpretation. For more information on parameter choice for the GNMDS see Appendix S7. We investigated both the

**TABLE 1** Traits used in the functional trait composition analysis, with a description and characteristics of the trait, associated (ecosystem) services/purpose and source of data

| Trait | Description | Type | Associated (ES) services | Source |
|---|---|---|---|---|
| Light | Optimal light/shade conditions of the species | Ordinal, seven-degree scale: 1, deep shade; 2, moderate–deep shade; 3, half shade–moderate shade; 4, half shade; 5, sun–half shade; 6, sun; 7, always full sun | Light requirements differ for species in forests and more open landscapes | Ecological indicator and traits values for Swedish vascular plants (Tyler et al., 2021) |
| Moisture | Mean realized moisture/water niche of the species | Ordinal, 12-degree scale: 1, very dry; 2, dry; 3, dry–mesic; 4, mesic; 5, mesic–moist; 6, moist; 7, moist–wet; 8, wet; 9, wet–temporarily inundated; 10, temporarily inundated; 11, shallow permanent water; 12, deep permanent water | Water requirements | |
| Nitrogen | Mean realized soil (water) nitrogen availability niche of the species | Ordinal, nine-degree scale: 1, very N-poor; 2, moderately–very N-poor; 3, moderately N-poor; 4, moderately N-poor–moderately N-rich; 5, moderately N-rich; 6, moderately–very N-rich; 7, very N-rich; 8, confined to the naturally most N-rich soils; 9, mostly on artificially N-enriched soils | Nutrient cycle | |
| Nectar production | Nectar production per species per year and area (g sugar/$m^2$/year) and amount of collectable pollen | Ordinal, seven-degree logarithmic scale: 1, no; 2, insignificant; 3, small; 4, modest; 5, rather large; 6, large; 7, very large | Pollination | |
| Strategy | C-S-R strategy of the species | Three variables: C, competitive ability; S, stress tolerance; R, ruderality. For species with mixed strategies; the sum of all variables = 1 | Erosion and disturbance prevention; soil formation, carbon dynamics | BiolFlor (Klotz et al., 2002) |
| SLA | Specific leaf area; leaf area/leaf dry mass ($mm^2$/mg) | Continuous | Nutrient cycle | LEDA traitbase (Kleyer et al., 2008) |
| Growth form | Growth form of species | Categorical: chamaephyte; geophyte; hemicryptophyte; phanerophyte; therophyte | Erosion and disturbance prevention | |
| Maximum height | Maximum plant height (cm) | Continuous | Position of the species in the vertical light gradient of the vegetation | The Ecological Flora Database (Fitter & Peat, 1994) |

importance of revegetation treatment and environmental variables, and the effect of time since restoration, on functional composition, as described for species composition.

Plots with missing data for one or several variables were excluded from analyses, and the number of plots included varies. For all linear-mixed models, models were reduced using the Akaike information criterion (AIC) as selection criterion (Burnham et al., 2011). All processing and analyses were done in the R environment (R Core Team, 2020) and RStudio (RStudio Team, 2020). Linear and generalized linear mixed-effect models were run with *lme4* (lmer; Bates et al., 2015) and *lmerTest* (Kuznetsova et al., 2017), and *AICcmodavg* (Mazerolle, 2020) was used for the AIC selection (Burnham et al., 2011). Community-weighted means were calculated with the *stats R* package (R Core Team, 2020). Figures for Shannon diversity and species richness were made with *ggplot2* (Wickham, 2016). The DCA and GNMDSs were run with *vegan* (Oksanen et al., 2020) and *MASS* (Venables & Ripley, 2002).

## 3 | RESULTS

### 3.1 | Species composition

A total of 188 vascular plant taxa were recorded, with 8.47 ± 3.66 taxa (mean ± SD) observed per plot (Appendix S9). Of these, 161 were identified to species level, 24 to genus and one to family level. We recorded four alien species (*Lupinus polyphyllus*, *Myrrhis odorata*, *Sambucus racemosa*, *Solidago canadensis*); all of which are classified as having a severe ecological impact (SE) in Norway. No red-listed species were found.

Tree cover was slightly higher, and canopy cover considerably higher on planted–natural plots compared to the other revegetation treatments (Appendix S8). Furthermore, planted–natural plots were the only study plots that had developed a tree layer height class 2 (500–990 cm; Appendix S8). Bryophytes were very abundant in the reference plots and naturally revegetated plots (Appendix S9). In naturally revegetated plots, *Betula pendula* was among the four most dominant species, along with *Agrostis capillaris*, *Festuca ovina* and *Rubus idaeus* (Appendix S9). Beside grasses (*Festuca rubra*, *Deschampsia cespitosa*, *Agrostis capillaris*), planted–naturally revegetated plots were dominated by *Cirsium arvense* (Appendix S9). Seeded plots were dominated by the grasses *Festuca rubra*, *Agrostis capillaris* and *Festuca ovina*, and herbaceous species such as *Tussilago farfara* and the alien species *Lupinus polyphyllus* (Appendix S9). The four most dominant species in the reference plots were *Vaccinium myrtillus*, *Picea abies*, *Avenella flexuosa* and *Betula pendula* (Appendix S9).

The species composition GNMDS showed variation in both the first (range −2.40 to 2.60 half change units), second (range −2.93 to 1.70 half change units) and third axis (range −1.85 to 1.83 half change units) (Appendix S10).

Reference plots were located on the negative end of the first axis and seeded restored plots on the positive end (Figure 3b, Appendix S10). Natural restored plots were located around the center but overlapped with the reference plots on the negative end (Figure 3b, Appendix S10). Planted–natural restored plots were also located around the center, but with more plots located toward the

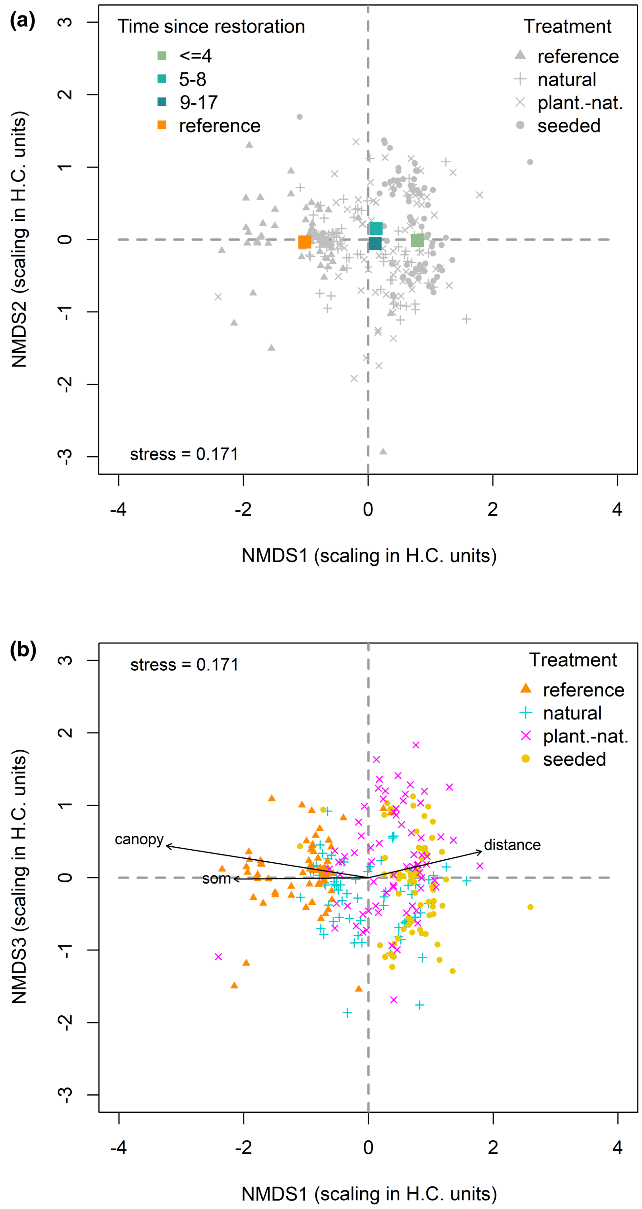


FIGURE 3 Ordination biplots based on global non-metric multidimensional scaling (NMDS; with Bray–Curtis dissimilarity, three dimensions) of the species composition in the sample plots. Panel (a) shows axes 1 and 2 with centroids for sample plots with varying time since restoration (≤4, 5–8, 9–17 years) and reference plots in different colors, with each sample plot as a gray shape in the background. Panel (b) shows sample plot placement along axes 1 and 3 with each revegetation treatment (natural, planted–natural, seeded, and reference) represented by different colors and symbols. Arrows indicate correlations between the ordination and environmental variables, with the length of the arrows proportional to the correlation strength. Only fitted environmental variables with a significant relationship ($p < 0.05$) with the ordination axes are shown in the figure. Canopy = canopy cover (%); som = soil organic matter (%); distance = sample plot distance to intact vegetation (m). H.C., half change

positive end of the axis (Figure 3b, Appendix S10). Revegetation treatments significantly explained variation in species composition on axes one and three (Figure 3b); however, variation explained by fixed predictor variables was low for axes two and three (Appendix S10). On axis one, there was a significant difference between planted–natural and seeded plots and reference plots, while naturally revegetated plots were not significantly different from reference plots (Appendix S10).

In addition, species composition varied significantly with a gradient of SOM content and canopy cover on the first axis, but not on the second or third axis (Appendix S10). Typical forest species (e.g., *Vaccinium myrtillus*, *Calluna vulgaris*, *Picea abies*, *Equisetum sylvaticum*) had low scores on axis 1, whereas grasses such as *Agrostis canina*, *Festuca ovina*, *F. rubra*, and *Agrostis capillaris* had high scores (Appendix S10).

There was a significant positive effect of time since restoration on distance from reference plots (Figure 3a, Table 2), showing that plots increased in similarity with reference plots as time since restoration increased. The significant interaction between the seeded treatment and time (Table 2) showed a different trajectory over time for seeded plots, indicating that similarity did not increase with time.

## 3.2 | Species richness and diversity

Both species richness and diversity were significantly higher in restored plots than in the reference plots, but there was very little difference between the revegetation treatments (Figure 4, Table 3). For species richness, there was a small, but significant, negative effect of SOM content, and species richness was lower in plots with coarse substrate (grain size 3; Table 3). Diversity was lower in plots with high canopy cover and coarse substrate (Table 3). There was no main effect of restoration treatment in the analysis of time since restoration by treatment, for species richness, and the effect of time since restoration was the only retained variable in the model for species richness, albeit unsignificant (Appendix S11). For diversity, the analysis revealed a somewhat lower diversity in natural–planted plots, and a significant interaction of years and treatment indicated somewhat varying trajectories between the treatments. However, the variation explained by the models was low (Appendix S11).

**TABLE 2** Parameter estimates and confidence intervals from the linear mixed-effects models of the distance from reference plots on axis 1 (species composition GNMDS) as a function of time since restoration × treatment

| | Distance from reference plots | |
|---|---|---|
| **Predictors** | **Estimates** | **CI** |
| Intercept | −2.29*** | −3.46 to −1.13 |
| Planted–natural | −0.18 | −1.51 to 1.16 |
| Seeded | 0.50 | −0.91 to 1.90 |
| Time since restoration | 0.15* | 0.03 to 0.27 |
| Planted–natural × years | −0.02 | −0.15 to 0.12 |
| Seeded × years | −0.15* | −0.30 to −0.01 |
| Random effects | | |
| $\sigma^2$ | 0.22 | |
| $\tau_{00\ \text{site}}$ | 0.15 | |
| ICC | 0.41 | |
| $N_{\text{site}}$ | 19 | |
| Observations | 203 | |
| Marginal $R^2$/conditional $R^2$ | 0.413/0.651 | |

*Note*: The intercept represents average values for naturally revegetated plots. Significant *p*-values are noted as stars behind the estimates.

Abbreviations: CI, confidence interval; GNMDS, global non-metric multidimensional scaling; ICC, intraclass correlation coefficient.

*$p<0.05$; **$p<0.01$; ***$p<0.001$.

## 3.3 | Functional trait composition

The functional trait composition GNMDS showed some variation in both the first (range −0.62 to 0.64 half change units), second (range −0.58 to 0.7 half change units), third (range −0.42 to 0.40 half change units) and fourth axis (range 0.72–0.69 half change units) (Appendix S12).

Reference plots were mainly separated from restored plots on GNMDS axes 1 and 2 (Appendix S12). In addition, canopy cover and SOM contributed to explaining variation in trait composition (Appendix S12). Variation on the third and fourth axis was further significantly explained by distance to intact vegetation (Appendix S12). Reference plots, on the high end of axis 1, were more dominated by chamaephytes and phanerophytes, and tall plants, than restored plots (Appendix S12).

There was no significant main effect of time since restoration on distance from reference plots in the GNMDS (Appendix S12). However, significant time by treatment effects revealed that functional composition developed in different ways over time in the three revegetation treatments: whereas functional composition in the natural treatment increased in similarity with reference plots over time, this was not the case for the planted–natural and seeded treatments.

# 4 | DISCUSSION

In this study, we found species richness and diversity to restore rapidly in road construction projects, and even exceed the observed values in reference vegetation, regardless of revegetation method. However, the restoration of species composition and functional composition depended more on revegetation treatment, as well as on time.

## 4.1 | Revegetation treatment has a significant impact on vegetation development of restored plots

In opposition to our expectations, the effect of revegetation treatment (i.e., natural revegetation, planted–natural, seeding) on

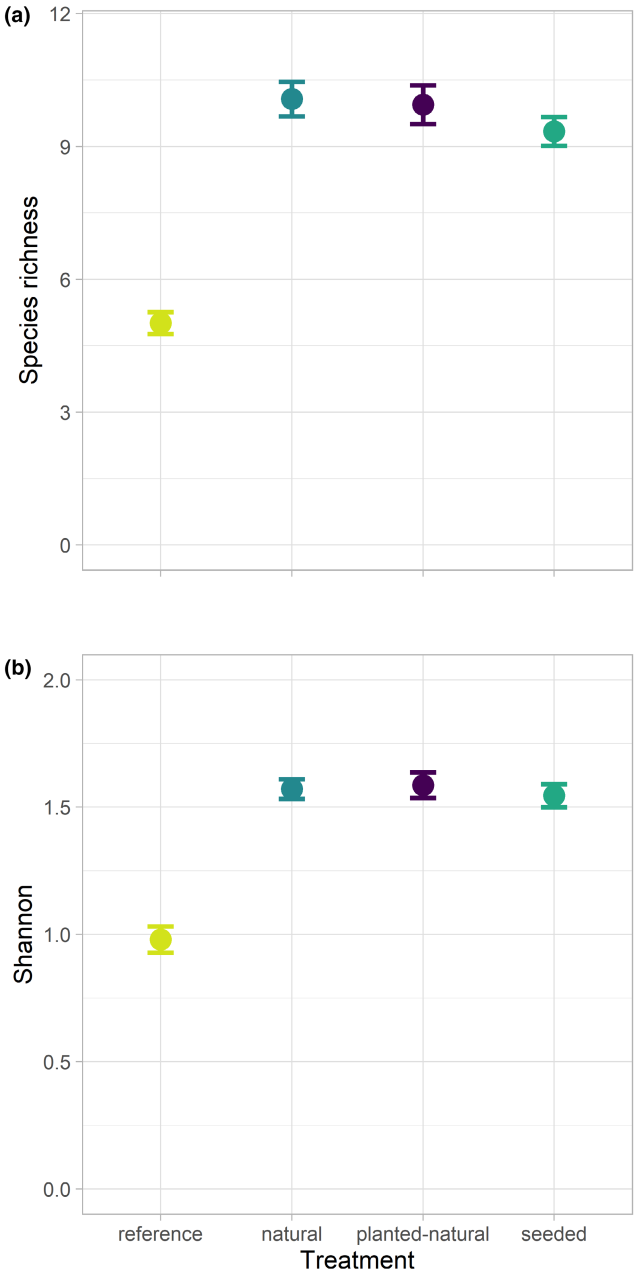


FIGURE 4 Mean species richness (a) and mean Shannon diversity index values (b) for reference plots and the different revegetation treatments. Vertical bars show the standard error. Note that the *Y*-axes are on different scales

species richness and diversity was small, suggesting that choice of restoration treatment is of minor relevance. However, several authors have pointed to the lack of usefulness of species richness and diversity for measuring restoration outcome (Rydgren et al., 2020), as these indicators ignore species identities (Brudvig et al., 2017). We found that revegetation method mattered more for restoring species composition: reference plots were significantly more similar in species composition to naturally revegetated plots than seeded or planted–natural plots, and similarity decreased from naturally revegetated plots over planted–natural plots to seeded plots, which has also been shown in other studies (Hagen et al., 2014). Topsoil, which was used for natural revegetation, usually contains seeds or other propagules of local native species, and also allows colonization from the adjacent intact vegetation with implications for good ecological recovery (Alday et al., 2011; Aradottir & Hagen, 2013; Farrell et al., 2020; Skrindo & Halvorsen, 2008). While seeding of commercial grass mixtures often leads to a fast recovery, it may also lead to a dense vegetation cover of highly competitive grass species. This impedes colonization by local native species, lowering species richness and diversity, as well as altering species composition (Aradottir & Hagen, 2013; Hagen et al., 2014; Rydgren et al., 2016). This was also indicated by the highest field layer cover in the seeded sites in our study, compared to the other revegetation measures. We also found that seeded plots did not approach reference plots over time, as opposed to the two other treatments, suggesting that seeding may be efficient for establishing a rapid vegetation cover, but not for re-establishing natural vegetation composition. The planted–natural treatment, as well as vegetation structure components connected to forest (tree and bush cover, canopy cover), were more similar to reference plots compared to the seeding treatment. Planting was used to enhance the establishment of a bush and tree cover on wildlife crossing structures to increase their attractivity for animals. Our results show that tree cover was slightly higher, and canopy cover considerably higher on planted–natural plots compared to the other revegetation treatments. Furthermore, planted–natural plots were the only study plots that had developed a tree layer height class 2 (500–990 cm). Both results suggest that planting in addition to natural revegetation facilitates a faster establishment of trees on wildlife crossing structures, something the NPRA aims for as a mitigation measure.

With increasing time, the diversity and composition of reference plots and restored plots became more similar, which has been shown by other studies and is connected to ecological succession (Connell & Slatyer, 1977; Mehlhoop et al., 2018; Prach et al., 2016; Swanson et al., 2011). Nevertheless, and in accordance with our expectations, we found large differences in species richness, diversity and composition between reference plots and restored plots, reflecting different successional stages (Chang & Turner, 2019; Swanson et al., 2011). Naturally, restored plots will take a long time to recover (Liebsch et al., 2008; Matzek et al., 2016; Walker & Wardle, 2014) or may never reach the species composition of the reference plots due to other factors such as vegetation management or changed environmental conditions (Farrell et al., 2020). Restored plots had, however, a much higher species richness and diversity than the reference plots, which is in line with ecological succession theory where species richness is highest in intermediate successional stages (Connell & Slatyer, 1977; Swanson et al., 2011).

### 4.2 | Similarity in functional community composition (community-weighted means) as mitigation goal

We explored whether functional community composition, as an alternative to species composition, could describe the achievements of

**TABLE 3** Parameter estimates and confidence intervals from the linear mixed effects models of species richness and Shannon diversity as a function of revegetation treatment and environmental variables

| | Species richness | | Shannon diversity | |
|---|---|---|---|---|
| **Predictors** | **Estimates** | **CI** | **Estimates** | **CI** |
| Intercept | 6.76*** | 5.14 to 8.39 | 1.24*** | 1.03 to 1.46 |
| Natural | 3.67*** | 2.08 to 5.26 | 0.38*** | 0.16 to 0.60 |
| Planted–natural | 3.45*** | 1.99 to 4.92 | 0.41*** | 0.21 to 0.61 |
| Seeded | 3.51*** | 1.84 to 5.17 | 0.40*** | 0.17 to 0.63 |
| Soil organic matter | −0.03* | −0.05 to −0.00 | 0.00 | −0.01 to 0.00 |
| Grain size—unsorted | −0.77 | −1.95 to 0.42 | −0.13 | −0.29 to 0.03 |
| Grain size—coarse gravel | −1.89* | −3.58 to −0.20 | −0.25* | −0.47 to −0.03 |
| Canopy cover | −0.02 | −0.04 to 0.01 | −0.00* | −0.01 to −0.00 |
| Random effects | | | | |
| $\sigma^2$ | 7.19 | | 0.14 | |
| $\tau_{00}$ | $1.39_{site}$ | | $0.01_{site}$ | |
| ICC | 0.16 | | 0.07 | |
| $N$ | $20_{site}$ | | $20_{site}$ | |
| Observations | 285 | | 285 | |
| Marginal $R^2$/conditional $R^2$ | 0.358/0.462 | | 0.327/0.374 | |

*Note*: The intercept represents average values for the restored plots with fine substrate (grain size – fine and medium coarse). Significant *p*-values are noted as stars behind the estimates.
Abbreviations: CI, confidence interval; ICC, intraclass correlation coefficient.
$^*p < 0.05$; $^{**}p < 0.01$; $^{***}p < 0.001$.

mitigation goals (Kollmann et al., 2016). Our study indicates similarity of most ecological functions between restored plots and reference plots, such as light, moisture and nitrogen requirements, productivity (specific leaf area), and nectar production. Strategy and growth form showed some variation along the axis, with ruderal species being more common in restored plots, opposed to woody species in reference plots. By using a small plot size (1 m$^2$), our data represent the field layer rather than the tree layer in the forest (Gjerde et al., 2007). Thus, our results suggest that the field layer vegetation of restored plots is approaching the functional traits composition of reference plots over time, although the seeding treatment may preclude this trajectory. Incorporating also ecosystem function into restoration or mitigation goals could improve the evaluation of success and facilitate the choice of appropriate restoration methods, ensuring that success is also evaluated in terms of ecological processes such as pollination, decomposition or trophic interactions (Kollmann et al., 2016). Furthermore, functionality can be related to ecosystem services and by associating certain traits with certain ecosystem services, provide information about which ecosystem services have changed from before the construction/ mitigation to after (Carlucci et al., 2020; De Bello et al., 2010).

### 4.3 | Setting realistic and precise goals for mitigation in road construction

The mitigation aims in road construction are diverse (Gann et al., 2019; van der Ree et al., 2015; Whisenant, 2002). Areas closest to roads need to have a vegetation that ensures road safety while enabling easy maintenance, whilst further away from the road it is possible to aim at a former state or adjustment to the surroundings. In our study sites the aim was to restore mature forest while maintaining and facilitating biodiversity along roads, and to establish a forest-like vegetation on wildlife crossings to provide cover for animals when crossing (Norwegian Public Roads Administration, 2014; Norwegian Public Roads Administration, 2018).

Our results suggest that more realistic goal setting is needed for restoration in road construction projects, with relevance for long-term management. Realistic goal setting must address the fundamental challenge of defining an appropriate target state for restoration.

Currently restoration goals are clearly defined in the NPRA handbooks, but appear unrealistic. The development of a forest next to a high-traffic road is undesirable (due to security and management purposes) and unachievable within the lifetime of road projects. The development of a mature forest takes decades if not centuries, depending on the forest type (Liebsch et al., 2008; Matzek et al., 2016), which is possible only for the areas furthest away from the road. Considering these points, achieving realistic restoration goals might be facilitated by setting successive subgoals over a certain time period (Gann et al., 2019; Nilsson et al., 2016). These subgoals could be, for example, different successional states; a temporary replacement ecosystem followed by forest development where possible; or a substitute ecosystem (Aronson et al., 2017; Gann et al., 2019, Higgs, 2017). Instead of having one general approach for all projects, goals should be clearly stated for each individual project on the basis of habitats/ecosystems surrounding the construction area, expected management measures and habitat use (Gann et al., 2019; Nilsson et al., 2016). For our study area, we suggest that "forest edge habitat" is a better restoration target than "forest habitat." Forest edges have higher light availability and can be either dryer or wetter than forest interiors, which reflects well the conditions of a recently disturbed/restored area, even after several years (Harper et al., 2015).

### 4.4 | Management implications

Mitigation measures after road construction are crucial to minimizing impacts on diversity and maintaining ecosystem services. By restoring areas along roads, a natural buffer zone is created that reduces long-term negative impacts on adjacent undisturbed areas, including habitat loss, air and light pollution, sun and wind exposure, and microclimate disturbances (Coffin, 2007; van der Ree et al., 2015). Wildlife crossing structures contribute to a network of green infrastructure that increases landscape connectivity, provides habitat for plants and animals, facilitating their movement, dispersal and gene flow (Denneboom et al., 2021). Natural revegetation with local topsoil, as in our study, utilizes the local propagule bank and limits the need for introduced plant material or seeds.

However, defining realistic restoration goals and selecting the most appropriate mitigation measures is challenging, and greatly hampered by a lack of long-term monitoring data, incomplete descriptions of how mitigation measures have been implemented and/or missing information for a given measure in a specific habitat. In order to make better decisions regarding mitigation measures and improve restoration work, we first require improved assessment of the effectiveness of the mitigation measures currently employed. For example, the potential for soil engineering to improve mitigation measures is high, because the propagule bank and characteristics such as nutrient content, density, grain size, etc. have strong influences on revegetation processes (Aradottir et al., 2013). However, to date there is insufficient information about the prior soil characters at restored sites, and the duration and location of stockpiling to allow assessment and optimization of mitigation measures. Strict documentation procedures and long-term monitoring of restoration efforts should be included in the contracts in the planning phase of projects (Gann et al., 2019) in order to improve further mitigation work and advance the methods for future projects.

## AUTHOR CONTRIBUTIONS

Anne Catriona Mehlhoop and Dagmar Hagen conceived the idea, with support from Marianne Evju. Anne Catriona Mehlhoop collected and compiled the data. Anne Catriona Mehlhoop performed statistical analysis, with input from Marianne Evju. Anne Catriona Mehlhoop wrote the manuscript, with comments and contributions from all co-authors (Dagmar Hagen, Marianne Evju, Astrid Brekke Skrindo).

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## DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in the Open Science Framework at https://osf.io, reference number DOI 10.17605/OSF.IO/6H2VJ.

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## SUPPORTING INFORMATION

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

**Appendix S1.** A list of the Norwegian Public Roads Administrations (NPRA) commitments to biodiversity and landscape restoration of areas degraded during road construction

**Appendix S2.** Detailed English summary of the methodologies Norwegian Public Roads Administrations (NPRA) mainly advises and that were used in our study sites

**Appendix S3.** Annual precipitation and mean July temperature in the study areas

**Appendix S4.** Requirements for study sites, and list of sites included

**Appendix S5.** A description and sketch of the transect and survey plot setup

**Appendix S6.** Table of grain size classes and their descriptions

**Appendix S7.** Detailed information on the ordinations

**Appendix S8.** Results of vegetation structure

**Appendix S9.** Tables of the species list of the vegetation analysis and the most dominant species

**Appendix S10.** Global non-metric multidimensional scaling (GNMDS) biplot of the species composition

**Appendix S11.** Table with parameter estimates from linear mixed effects models of species richness and diversity

**Appendix S12.** Global non-metric multidimensional scaling (GNMDS) biplot and table of GNMDS scores of the trait composition

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