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Research article

Recovery of vegetation on former alpine roads: how long does it take?

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Alpine areas worldwide are under heavy land-use pressure and degradation. Active restoration treatments can contribute to accelerating recovery of degraded areas. However, monitoring data are needed to understand the contribution of restoration treatments to long-term management and to predict time to recovery (TR). In this study, we used monitoring data on removed roads in an alpine area in Norway to investigate TR of three vegetation-based indicators. Four restoration treatments were tested: 1) removal of added gravel down to original terrain surface, and stirring of topsoil; 2) adding fertilizer to the stirred topsoil; 3) adding seeds to the fertilized topsoil; and 4) no removal of added gravel, but stirring of top layer (gravel and soil). The restoration of roads took place in 2002, and monitoring of permanent plots was carried out in 2004, 2009, 2014, and 2019. Reference plots in intact vegetation next to removed roads were monitored in 2014 and 2019. We used species composition and species richness of vascular plants as well as total vegetation cover as indicators of restoration outcome and investigated predicted TR for these indicators under different restoration treatments. Species composition changed significantly with time since restoration in all treatments, approaching that of the reference vegetation. The recovery of species composition was slowest in fertilized and seeded plots, where estimated TR was 2–3 times longer (> 45 years) than in the other treatments (< 20 years). Species richness of vascular plants was restored quickly (< 5 years) within all restoration treatments, whereas recovery of vegetation cover varied more (20–30 years). Our study confirms that vegetation recovery in alpine environments is a long-term process, but that adding seeds and nutrients is unnecessary for, and even inhibits, the recovery of narrow, disturbed sites such as former roads.

Keywords: alpine ecosystems, indicators, restoration, seeding, time to recovery

Introduction

Alpine ecosystems are under heavy pressure from land-use changes in combination with climate change (IPBES 2018). Although mountain and tundra areas still contain larger proportions of undisturbed land compared to other regions (IPBES 2018), impacts from development of infrastructure, renewable energy, military training, and recreation facilities cause continuous increase in disturbed areas (Norwegian Environment Agency 2015, EEA 2016). Recovery after disturbance is slow in alpine ecosystems, due to abiotic factors such as low temperatures, a short growing season, and low water and nutrient availability (Billings 1987, Willard et al. 2007, Rydgren et al. 2011). Efficient restoration treatments to enhance recovery rates is therefore particularly needed in alpine ecosystems (Forbes and Jefferies 1999, Krautzer et al. 2012, Hagen et al. 2022). To restore alpine areas after disturbance, three main groups of restoration techniques have traditionally been applied: 1) remediation of soil and terrain, 2) adding nutrients, and 3) seeding or planting (Urbanska and Chambers 2002, Hagen and Evju 2013).

Systematic and regular monitoring is essential to assess the progress of restoration following interventions (McDonald et al. 2016), and to evaluate if the ecosystem approaches the desired reference state. Because of the slow recovery processes in alpine ecosystems, it is vital to understand recovery trajectories long before recovery is complete, to be able to adjust restoration interventions if results suggest that the development is on the wrong track (Suding 2011). To this end, defining a relevant comparison, i.e. a reference vegetation (undisturbed, intact vegetation towards which the restored vegetation should develop), is useful (McNellie et al. 2020, Atkinson et al. 2022). The monitoring should therefore also include the reference vegetation, to be able to detect ongoing dynamics in undisturbed vegetation during the time of recovery.

There is a need for linking a project's restoration goal, ecological targets, and measurable indicators, to be able to evaluate the outcome of restoration efforts (Zedler 2007, Suding 2011, Prach et al. 2019). The diversity of indicators used in restoration projects is vast (Evju et al. 2020), and which metric to use to monitor the restoration outcome is under considerable debate (Brudvig et al. 2017, Abella et al. 2018, Rydgren et al. 2020). Species composition, i.e. which species are present and how abundant, is a key ecological attribute of ecosystems (McDonald et al. 2016) and particularly relevant for assessing restoration outcome (Rydgren et al. 2020). Nevertheless, other vegetation-based indicators, such as vegetation cover or biomass, structural diversity or species richness, which are independent of species identity, may be relevant – and may respond at other time scales than species composition. Using a varied set of indicators may shed light on recovery rates of different properties of an ecosystem, and guide management decisions on the need for supplemental interventions.

Restoration outcomes often vary, both between and within restoration projects (Brudvig 2017, Mehlhoop et al. 2018). To be able to plan and implement efficient restoration and monitoring it is crucial to understand why different restoration efforts give varying outcomes, the relative importance of site effects, and how different indicators of restoration outcome respond in a given context (Brudvig 2017, Brudvig et al. 2017, Abella et al. 2018).

In this study, we investigated the effect of different restoration treatments on vegetation recovery on former roads that have been removed and restored in an alpine ecosystem in central Norway. The goal of the large-scale restoration project, covering 165 km^2 in a former military training area, was to restore the vegetation to a 'natural state' (Norwegian Ministry of Defence 1998), i.e. similar to the alpine vegetation occurring in undisturbed parts of the area (Hagen and Evju 2013, Hagen et al. 2022). Monitoring of vegetation recovery in restored sites was central to evaluating this goal. To evaluate restoration outcomes, specific targets were defined: 1) to initiate and promote natural recovery of native species, 2) to approach the species richness and plant cover of adjacent undisturbed sites, and 3) to ensure that treatments would not facilitate the establishment of non-native species (Hagen and Evju 2013). To this end, we defined three vegetation-based indicators: species composition, assessing the first and third targets; and vegetation cover and species richness, both addressing the second target. We investigated 1) the effects of restoration treatments on recovery of the vegetation-based indicators; 2) the estimated time to recovery (TR) – that is, a satisfying restoration outcome, for each combination of indicator and restoration treatment; and 3) the variability of restoration treatment effect on indicator performance.

Material and methods

Study site

The former Hjerkinn military training area is situated at Dovrefjell, central Norway (Fig. 1), between 1000 and 1700 m a.s.l., in one of the last largely intact high mountain ecosystems in Europe (Norwegian Environment Agency 2021). The area covers 165 km^2 and was used for military purposes from 1923 to 2008.

The geology in the area is characterized by calcium-poor glacial till overlying Precambrian metamorphic and igneous bedrock (Geological Survey of Norway 2019). The average annual temperature (2002–2019) at the nearest weather station (Fokstugu, 973 m a.s.l.) was 1.0°C, with an annual precipitation of 436 mm (Norwegian Meteorological Institute 2023). Mean summer temperature (June–July–August) was 10.0°C in the study period, with 2018 being particularly warm (11.4°C) and 2012 being particularly cold (8.6°C). Summer precipitation ranged between 133 mm (2002) and 310 mm (2011), with an average of 184 mm in June–August. The study sites were situated in dry and medium-dry alpine heath vegetation dominated by lichens, dwarf shrubs, and some graminoids and forbs (Hagen and Evju 2013).

In 1999, the Norwegian Parliament decided to close down the military training area and restore the area to its 'original, natural state' (Norwegian Ministry of Defence 1998). To reach this goal, restoration activities were carried out by the Norwegian Defence Estates Agency from 2009 to 2020. The restoration included removing technical infrastructure, including 75 km of roads (Hagen et al. 2022).

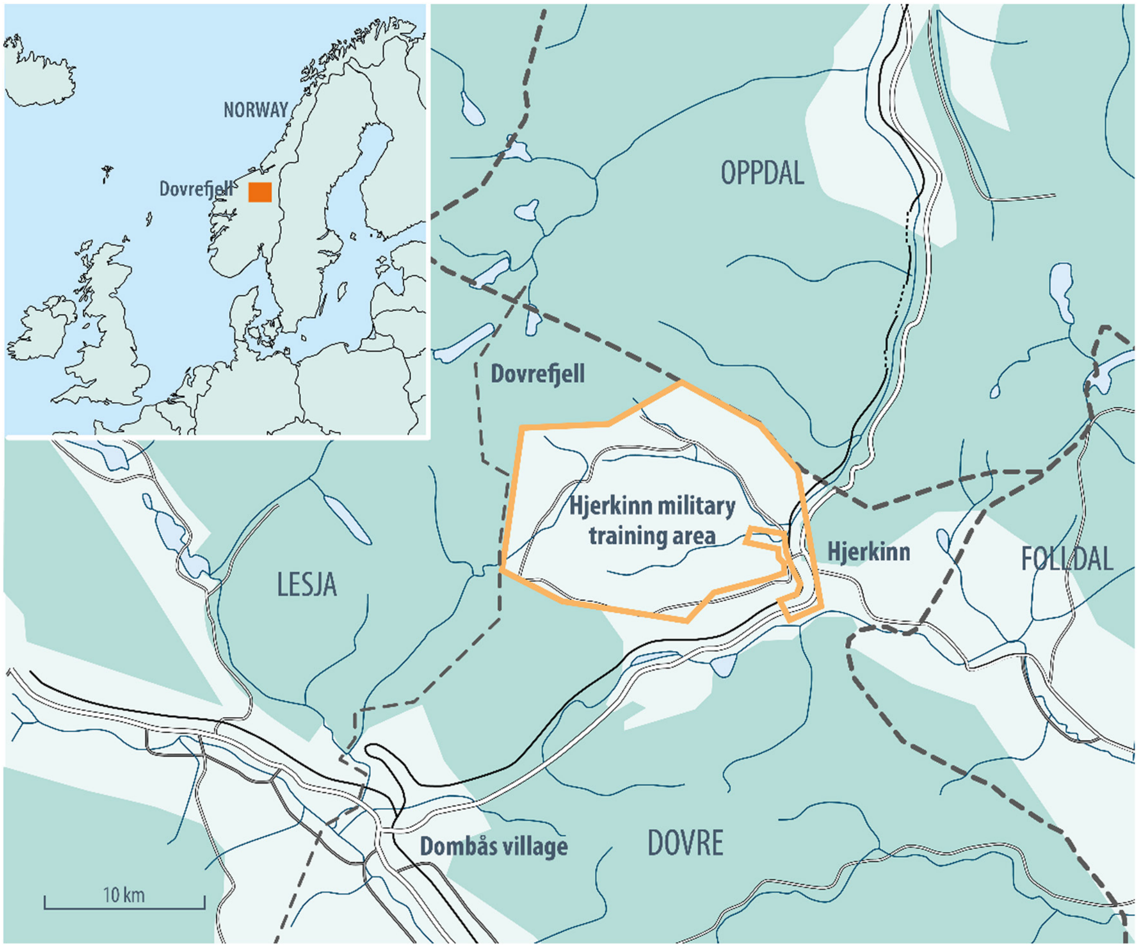

Figure 1. The Hjerkinn military training area in Dovrefjell, Innlandet county, central Norway. Before the restoration the training area was surrounded by protected areas (dark colors), including national parks, landscape protected areas, and nature reserves (Hagen et al. 2022).

Restoration methods

In 2002, a pilot project removing 1.2 km of roads was initiated to explore safety, logistics, and ecological methods as well as economic calculations prior to the large-scale restoration. An ecological monitoring program was established to evaluate the recovery of the vegetation following removal using different restoration treatments (Hagen and Evju 2013).

The pilot project was established along three road sections (hereafter denoted sites). The roads were constructed during the 1960s by adding crushed stone and gravel on top of undisturbed terrain and vegetation. To remove the roads in August 2002, a shell-proof excavator first removed the crushed stone down to the original terrain surface. Then the upper soil layer was stirred down to 20 cm as the excavator grab lifted the compressed surface.

Four restoration treatments were tested: 1) soil preparation by stirring as described above (hereby denoted *Soil*), 2) *Fertilization*, where 20 g m^{-2} of granulated N–K–P fertilizer was added to the stirred (treatment 1) topsoil, 3) *Fertilization and seeding*, where 7 g m^{-2} of commercial seeds of *Festuca rubra* were added to the stirred and fertilized (as in treatment 2) topsoil. In addition, a fourth restoration treatment, in which the added gravel was not removed, but the topsoil and gravel were stirred together, was tested (*No removal of gravel*) at two of the three sites. At all sites, vegetation turfs ($\leq$ 1 m^2 in size) were transplanted from nearby road margins, at a 5–10 m planting distance. The planting density was equal in all sites, and the effect of turf transplants was not included in this study (for further details see Hagen et al. 2022).

The road section in each site was divided into three blocks (four at the two sites where the *No removal of gravel* treatment was included) approximately 100 m long, and restoration treatments were assigned randomly to the blocks. To monitor effects of restoration treatments, 55 permanent plots of 0.5 × 0.5 m were established in 2004, five for each restoration treatment at each site (Fig. 2). The plots were randomly placed within blocks and were permanently marked with aluminum poles in the corners and with marker sticks, and location was recorded with a hand-held GPS.

In 2014, we established 15 monitoring plots in intact vegetation in close proximity (10–20 m) to the road section sites to collect data on reference vegetation, i.e. the target for the

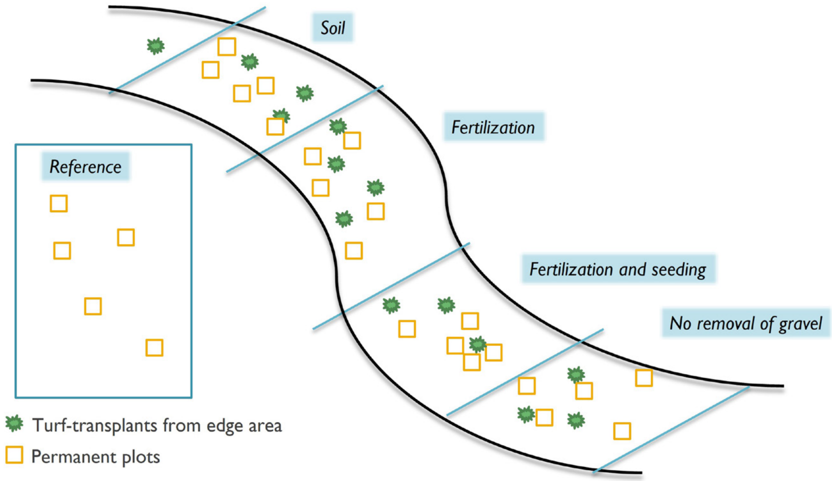

Figure 2. Pilot project study design. Each site was divided into blocks, each block allocated to one restoration treatment. Five permanent monitoring plots were established randomly within each block in 2004. Five reference plots were established in intact vegetation between 10 and 20 m from the removed roads, at each site in 2014. The '*No removal of gravel*' treatment was replicated in two of three sites. *Soil* = removal of added gravel down to the original terrain surface, and stirring of topsoil; *Fertilization* = addition of fertilizer to the stirred topsoil; *Fertilization and seeding* = addition of seeds to the stirred and fertilized topsoil; and *No removal of gravel* = added gravel stirred with the topsoil, but not removed.

restoration. At each site, a block of 100 m^2 was established in undisturbed vegetation next to the road, and five plots were placed randomly within the block (Fig. 2).

The vegetation in the treatment plots was monitored in 2004, 2009, 2014, and 2019, i.e. two, seven, 12, and 17 years after restoration, respectively, and reference plots were monitored in 2014 and 2019, giving a total of 250 plot-time observations. Three plots were not relocated in 2009 (one in each of treatments *Soil*, *Fertilization and seeding* and *No removal of gravel*), but were found again in 2014.

In each plot we recorded the abundance of all vascular plants as sub-plot frequency (i.e. presence–absence in 16 subplots per plot). In addition, we recorded total vegetation cover (visual estimate of percent cover, continuous scale), including cover of bryophytes and lichens.

Statistical analyses

Before carrying out statistical analyses, we removed plots with missing data ($n = 3$), and for the multivariate analysis also plots with no observed taxa ($n = 5$). We included taxa determined to species or genus level. This resulted in a dataset of 247 plots (242 for multivariate analyses) with 94 observed vascular plant taxa. All statistical analyses were carried out in R Studio ver. 4.03 (www.r-project.org).

To investigate TR for species composition, we used the ordination-regression-based approach (ORBA) (Rydgren et al. 2019, Rydgren et al. 2020). This approach consists of the following components (cf. Rydgren et al. 2019): species composition data for restored plots analyzed at several points in time, species composition data for an adequate reference community, a proxy for the successional gradient obtained by ordination, a regression model which relates the species composition data (ordination axes) from restored plots to the temporal gradient and the reference plots, and a predictor for TR.

To extract the gradient of the species composition data, we followed the recommendations by Rydgren et al. (2019) and used two ordination methods in parallel: detrended correspondence analysis (DCA; Hill and Gauch 1980), and global nonmetric multidimensional scaling (GNMDS; Minchin 1987), as implemented in the 'vegan' package ver. 2.5-7 (Oksanen et al. 2020, see Supporting information for details) and calculated pairwise Kendall's rank correlation coefficients τ between pairs of ordination axes from the two ordination methods to investigate if the two ordination methods revealed similar gradient structures. We investigated gradient structure using the full dataset, including all treatments and plots ($n = 242$), and on subsets of the datasets including only one treatment and reference plots (four subsets). The ordinations of the four subset datasets revealed clear successional gradients by use of GNMDS (Supporting information). The gradients using the full dataset also revealed successional gradients; however, they were more divergent (Supporting information). To calculate TR, we thus proceeded with four GNMDS ordinations, one for each restoration treatment.

To investigate the successional gradient of the species composition, we ran linear mixed-effect models, as implemented in lmer4 (Bates et al. 2015), of ordination axis scores as a function of time and treatment, including reference plots. We used plot nested in site as random variables to account for the nested sampling design and repeated measurements of plots. We calculated the successional distance, i.e. the distance along the successional gradient for each plot j at time t to the reference vegetation as:

$$d_{jt,0} = x_0 - x_{jt}$$

following Rydgren et al. (2019), where x_0 was the centroid of reference plots in the ordination space, pooled over the years 2014 and 2019 (Supporting information). We modelled successional distance as a function of time since restoration with two approaches: a linear model (untransformed response variable, see Supporting Information), and a log-linear asymptotic model (logarithmically transformed response variable), both with plot nested in site as random factors. In cases where $d_{jt,0} < 0$ (i.e. the restored plots had higher axis score than the centroid of the reference plot scores, occurring for four and one plot in 2019, for treatments *Soil* and *No removal of gravel*, respectively), data were omitted from the asymptotic model (c.f. Rydgren et al. 2020).

We predicted TR as the predicted number of years since restoration which corresponded to the reference vegetation. We used two thresholds for successful restoration: a fixed distance of 0.01 off the value of the centroid of the reference plot scores, as the asymptotic model can never reach 0, and one standard deviation off the centroid of reference plot scores (cf. Rydgren et al. 2019, 2020). The TR estimates for the asymptotic models were labeled $TR_{A0.01}$ and TR_{ASD} (and $TR_{L0.01}$ and TR_{LSD} for the linear models, see Supporting Information).

To calculate TR for the univariate indicators, we modelled vegetation cover and species richness per 0.25 m^2 plot as functions of time since restoration for each treatment separately, not including reference plots in the analyses. Vegetation cover was logit-transformed before being modelled with identity-link and Gaussian errors. Species richness was modelled with log-link and Poisson errors, with the predictor variable transformed as 1/time since restoration. We used plot nested in site as random variables to account for the nested sampling design and repeated measurements of plots. We used these models to predict TR, i.e. the number of years to reach reference levels. We used two thresholds for reference levels: 1) TR_0: mean value for the indicator pooled over reference plots, and 2) TR_{SD}: one standard deviation off the mean of the reference plots.

Last, to estimate the predictability of indicators, we modelled each indicator as a function of treatment (treatment and reference), year, and their interactions, using linear mixed-effect models, where site and plot were used as random factors. To investigate the relative importance of restoration treatment compared to site and plot effects, we calculated the variation explained by fixed factors (R^2_m) and the total variation explained (R^2_c), including random factors, following Laughlin et al. (2017), using the *r.squaredGLMM* function in the 'MuMIn' package (Nakagawa and Schielzeth 2013, Barton 2015). Comparing R^2_m and R^2_c–R^2_m allows us to investigate the spatial and temporal variability in responses relative to intervention effects. For the species composition indicator, we used the full dataset ordination scores of both the DCA and GNMDS ordinations, to be able to investigate treatment versus site and plot effects in one analysis (Supporting information).

Results

A total of 94 vascular plant taxa, of which 78 were determined to the species level and the remaining 16 were determined to genus level, were identified during the 15 year time-period of the study (Supporting information). A total of 45 vascular plant species were found in the reference plots (Supporting information), most of which were also found in the restored plots. However, except for the *No removal of gravel* treatment, substantially more species were recorded overall in restored plots (64, 67, 58, and 28, in *Soil, Fertilization*, *Fertilization and seeding* and *No removal of gravel*, respectively; Supporting information). The reference plots were dominated by typical species from alpine heath vegetation, such as *Betula nana*, *Empetrum nigrum* and *Vaccinium vitis-idaea* (Supporting information). Species recorded only in restored plots included several forbs and graminoids which occur less frequently in undisturbed heath vegetation.

The most abundant species in all treatments including reference plots was *Festuca ovina*, except in the *Fertilization and seeding* treatment, where the seeded *F. rubra* was clearly most abundant. All species recorded are native to the Norwegian flora, although the seeded *F. rubra* is a commercial cultivar of non-native origin.

Time to recovery

Species composition

In all ordinations, there was no difference in plot scores of reference plots between 2014 and 2019 (Supporting information), i.e. species composition was stable in the reference vegetation in the five-year period. The first axis of all four GNMDS ordinations represented a successional gradient, with a gradual shift in species composition towards the reference vegetation plots (Supporting information).

There was a significant effect of time since restoration on species composition in all treatments, i.e. species composition approached that of the reference vegetation (Supporting information; Fig. 3). The shift in species composition was slowest in the *Fertilization and seeding* treatment, where estimated TR was 2–3 times longer than in the other treatments (Table 1, Supporting information). Choice of recovery threshold affected estimated TR considerably; predicted TR was three times longer if species composition were to reach the mean of reference plots (i.e. $TR_{A0.01}$) compared to reaching one standard deviation off the mean (i.e. TR_{ASD}; Table 1).

Vegetation cover

The total vegetation cover in the reference plots, including vascular plants, bryophytes, and lichens, was 82.8 ± 12.2% (mean ± SD). Vegetation cover was somewhat higher in 2014 (85.8 ± 11.3%) than in 2019 (79.9 ± 12.7%), but this difference was not significant (paired t-test of logit-transformed data: $t = 1.618$, $df = 14$, $p = 0.128$). Vegetation cover increased with time since restoration in all treatments (Fig. 4, Supporting information), but temporal trends varied between treatments. The *Fertilization and seeding*

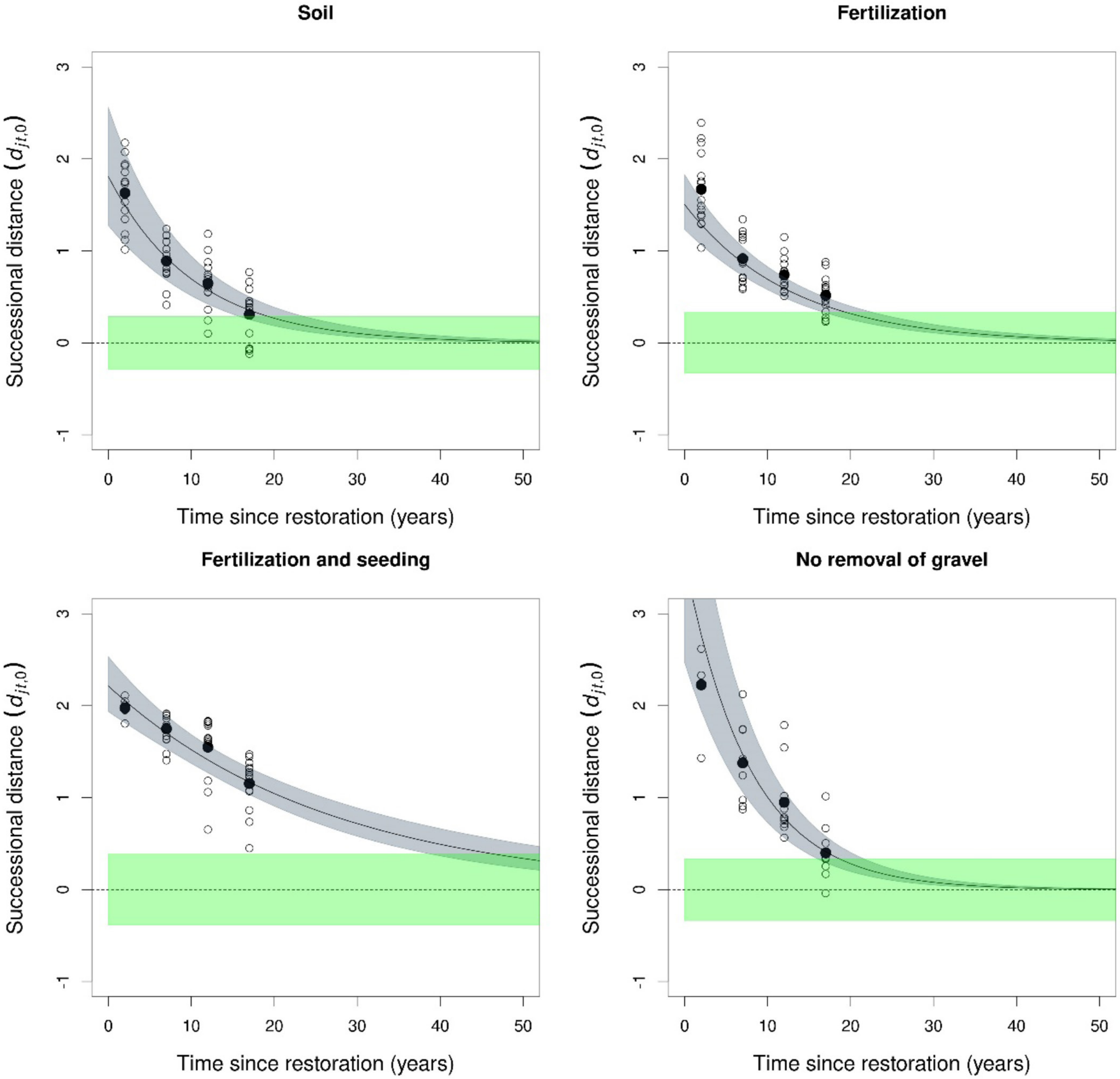

Figure 3. Models for successional distance (i.e. distance along the successional gradient represented by the first global nonmetric multidimensional scaling axis; see Supporting information) as functions of time since restoration (number of years), obtained separately for each of the four restoration treatments. Open circles show plot values, and closed circles show mean values for each year. Black lines show model predictions, and grey shaded areas indicate 95% confidence intervals. Green dotted lines show the centroid of reference plot scores in the ordination space (pooled over 2014 and 2019), whereas green shaded areas represent one standard deviation off the centroid. *Soil* = removal of added gravel down to the original terrain surface, and stirring of topsoil; *Fertilization* = addition of fertilizer to the stirred topsoil; *Fertilization and seeding* = addition of seeds to the stirred and fertilized topsoil; and *No removal of gravel* = added gravel stirred with the topsoil, but not removed.

treatment gave a rapid development of a vegetation cover, with mean cover being 44.7% (± 17.7%) in 2004, but subsequent development of vegetation cover development was slow (Fig. 4). Vegetation cover developed extremely slowly in the *No removal of gravel* treatment (Fig. 4, Supporting information), and was on average only 14.2% (± 12.1%) in 2019. In the *Soil* and *Fertilization* treatments, a more gradual increase was found.

The estimated TR depended on the choice of threshold level, particularly for the *Fertilization and seeding* treatment.

Table 1. Estimated time to recovery (TR) in years obtained from models for species composition, the total vegetation cover, and species richness as a function of time since restoration for each of the four restoration treatments separately. For species composition, TR $_{A0.01}$ estimates the number of years for reaching the centroid of plot scores of the reference plots (pooled over years) + 0.01, and TR_{ASD} estimates the number of years for reaching one standard deviation (SD) off reference plot centroid, using log-linear models. For vegetation cover and species richness, TR_0 estimates the number of years for reaching the mean value of the reference plots, pooled over years; whereas TR_{SD} estimates number of years for reaching the mean minus one standard deviation of reference plots. Numbers in parentheses give the 95% confidence intervals of estimates. *Soil* = removal of added gravel down to the original terrain surface, and stirring of topsoil; *Fertilization* = addition of fertilizer to the stirred topsoil; *Fertilization and seeding* = addition of seeds to the stirred and fertilized topsoil; and *No removal of gravel* = added gravel stirred with the topsoil, but not removed.

	Species composition		Total vegetation cover		Species richness	
	$TR_{A0.01}$	TR_{ASD}	TR_0	TR_{SD}	TR_0	TR_{SD}
Soil	54.4 (50.7–58.0)	19.2 (15.5–22.8)	22.9 (16.9–29.0)	19.0 (12.9–25.0)	5.1 (3.9–7.4)	1.9 (1.7–2.1)
Fertilization	64.5 (62.0–67.0)	19.5 (16.9–22.0)	23.6 (18.2–29.4)	17.8 (12.0–23.6)	4.4 (3.5–6.2)	1.5 (1.4–1.6)
Fertilization and seeding	143.8 (140.2–147.3)	46.5 (42.9–50.0)	31.8 (17.1–46.4)	18.6 (5.0–32.2)	9.7 (5.9–27.9)	2.5 (2.3–3.3)
No removal of gravel	46.3 (43.4–49.2)	18.6 (15.7–21.5)	31.1 (26.3–35.9)	28.0 (23.2–32.8)	48.4 (15.9-inf.)	4.4 (3.7–5.4)

To reach a vegetation cover of 82.8% (TR_0) was predicted to take approximately 20 years in the *Soil* and *Fertilization* treatments, and 30 years for the two other treatments (Table 1). However, reaching 70% cover (TR_{SD}) was predicted to take < 20 years for all treatments, except for *No removal of gravel*.

Species richness

Mean species richness was 10.2 per 0.25 m^2, but the species richness varied substantially between plots (1SD = 6.9). There was no difference in vascular plant species richness between reference plots in 2014 and 2019 (paired t-test of log-transformed data; $t = 1.436$, $df = 14$, $p = 0.173$).

Species richness increased with time since restoration in all treatments (Fig. 5, Supporting information). The mean species richness found in reference plots was reached within five years for the *Soil* and *Fertilization* treatments, but reaching 10 species per plot was predicted to take almost 50 years in plots where gravel was not removed (Fig. 5, Table 1). In terms of one standard deviation off the mean, reference levels were reached 2–4 years after restoration for all treatments.

Predictability of indicators

The variation explained by restoration treatment and time was high for all indicators of restoration outcome (Fig. 6, Supporting information), and generally much higher than plot and site (random) effects. The combination of treatment and time was a better predictor for species composition (86.0 and 78.8% for DCA and GNMDS, respectively) than for species richness and total vegetation cover (74.4 and 71.6%, respectively; Fig. 6). Most variation between plots and sites was found for total cover (15.9%), followed by species richness and species composition by GNMDS (11.1 and 10.7%, respectively).

Discussion

Long-term monitoring of restoration outcome is important to assess the effectiveness of restoration efforts, and understand the relative importance of restoration treatments, landscape and site effects, and historical contingencies (Brudvig and Damschen 2011, Brudvig 2017). In this study we used 15 years of monitoring data to investigate the recovery of vegetation after removal of roads in an alpine military training area.

Which indicators should be used to evaluate restoration outcome?

Ecological restoration is described as to 'assist (…) the recovery of an ecosystem that has been degraded, damaged or destroyed', aiming to move the degraded ecosystem into a trajectory towards a reference ecosystem (Gann et al. 2019). The relevance of species composition as an indicator for restoration outcome is evident: species composition, including both the identity and abundance of the species present, is a key ecosystem attribute (McDonald et al. 2016), and to restore an ecosystem, recovery of the species composition is crucial (Rydgren et al. 2020). Our results suggest the recovery of vascular plant species composition was substantial in the 17 years after restoration at our alpine site, although notably slower in the *Fertilization and seeding* treatment. The species of the undisturbed reference vegetation were mainly present in the treatment plots, and their abundances were approaching those of the undisturbed vegetation. This suggests that the restoration treatments carried out were indeed efficient, for example as opposed to restoration treatments of alpine spoil heaps in similar habitats and altitudes (Rydgren et al. 2020, Sulavik et al. 2021).

Using species composition alone as an indicator of recovery would, however, give an overly optimistic view on recovery, particularly when using a relaxed threshold for recovery (the TR_{ASD} threshold; Table 1). In the *No removal of gravel* treatment, although composition was predicted to recover within ca 20 years, the development of vegetation cover was extremely slow: the raw data showed a stable low cover of < 15% in 2014 and 2019, compared to ca 80% in the reference plots.

Species richness has previously been found to be a more predictable indicator of restoration outcome than species composition (Laughlin et al. 2017), as the metric is independent of the identity of the species present. In our study

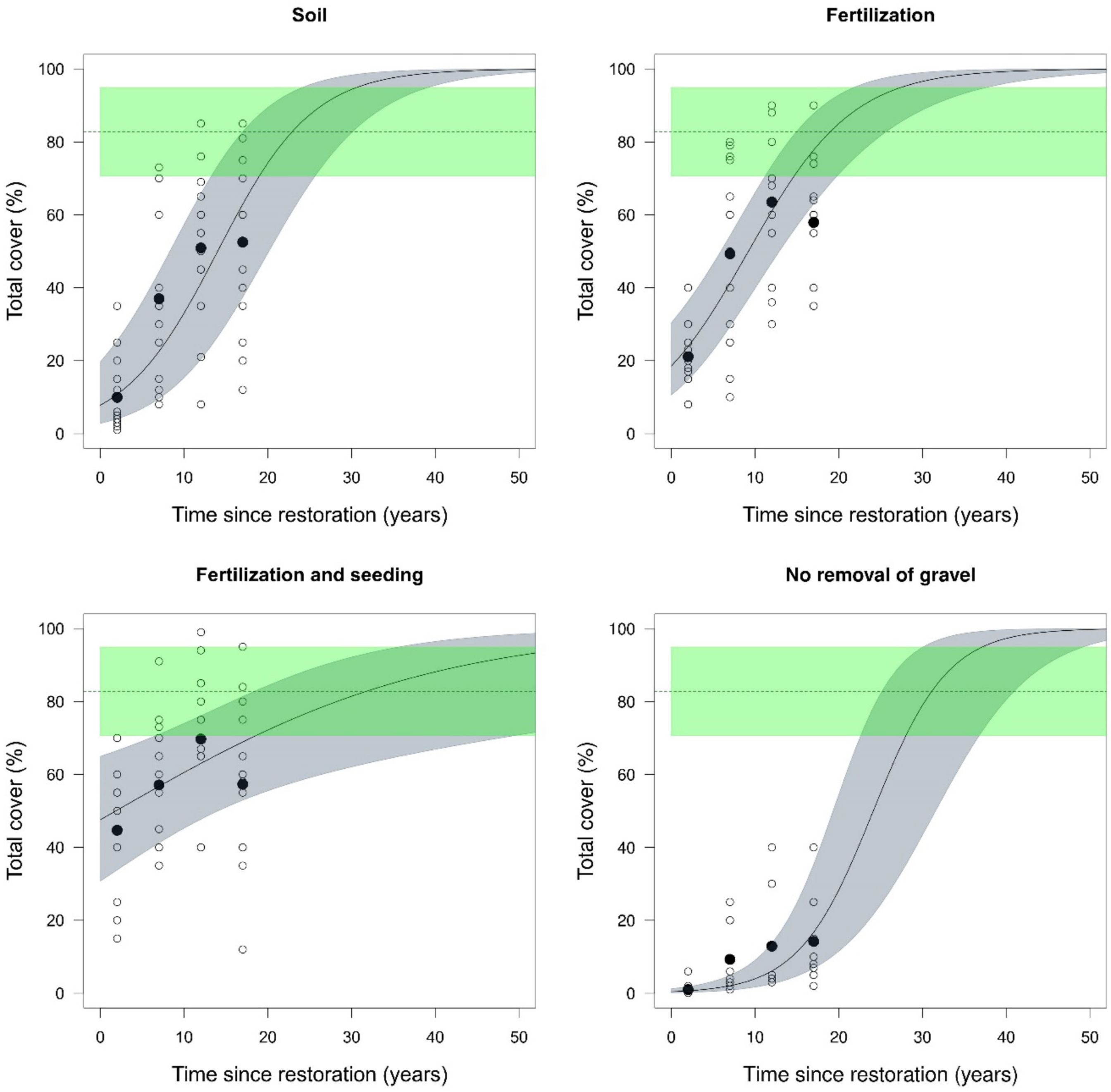

Figure 4. Models for the total vegetation cover as functions of time since restoration (years), obtained separately for each of the four restoration treatments. Open circles show observed plot values, and closed circles show mean observed values for each year. Black lines show model predictions, and grey shaded areas indicate 95% confidence intervals. Green dotted lines show mean of reference plots pooled over years, whereas green shaded areas represent one standard deviation off the mean. *Soil* = removal of added gravel down to the original terrain surface, and stirring of topsoil; *Fertilization* = addition of fertilizer to the stirred topsoil; *Fertilization and seeding* = addition of seeds to the stirred and fertilized topsoil; and *No removal of gravel* = added gravel stirred with the topsoil, but not removed.

plots, species richness increased rapidly after restoration, and the mean richness of the reference vegetation was quickly reached, except in the *No removal of gravel* treatment in which substantially fewer species were recorded in total and per plot. Nevertheless, our results show that species richness varied more among plots and sites (random error) than species composition. Our results thus do not support the hypothesized hierarchy of predictability of indicators for restoration outcome, which ranks vegetation-based indicators of restoration outcomes from most to least predictable in this order: vegetation structure > species richness > species composition (Brudvig et al. 2017, Laughlin et al. 2017). Hence, our findings are rather in line with Abella et al. (2018), who found no systematic increase in predictability

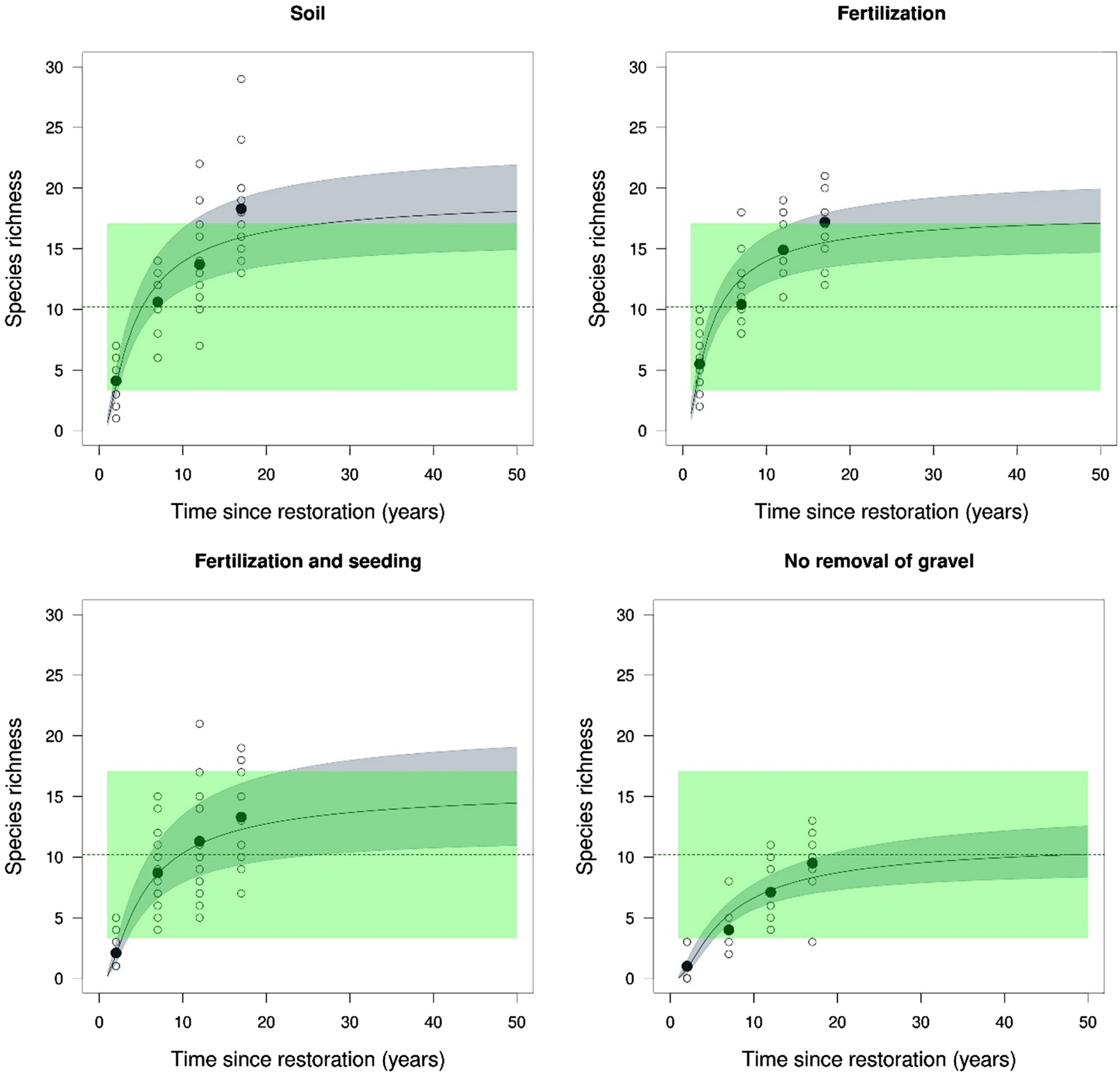

Figure 5. Models for the vascular plant species richness (number of species per 0.25 m^2 plot) as functions of time since restoration (years), obtained separately for each of the four restoration treatments. Open circles show observed plot values, and closed circles show mean observed values for each year. Black lines show model predictions, and grey shaded areas indicate 95% confidence intervals. Green dotted lines show mean of reference plots, whereas green shaded areas represent one standard deviation off the mean. *Soil* = removal of added gravel down to the original terrain surface, and stirring of topsoil; *Fertilization* = addition of fertilizer to the stirred topsoil; *Fertilization and seeding* = addition of seeds to the stirred and fertilized topsoil; and *No removal of gravel* = added gravel stirred with the topsoil, but not removed.

of a range of indicators representing structure, richness, and function. Our study site is characterized by harsh environmental conditions and a relatively limited species pool. Such factors may increase the predictability of restoration outcomes, as recovery trajectories are less influenced by factors such as chance dispersal events and priority effects (Brudvig et al. 2017).

Rather than predicting the wrong TR (Rydgren et al. 2020), the use of a broad set of indicators that describe several aspects of the ecosystem highlights the varied recovery trajectories of different ecosystem attributes. Hence, we would advocate using a diverse set of vegetation-based indicators to evaluate restoration outcome (Brudvig et al. 2017), also for the sake of communicating recovery times to stakeholders.

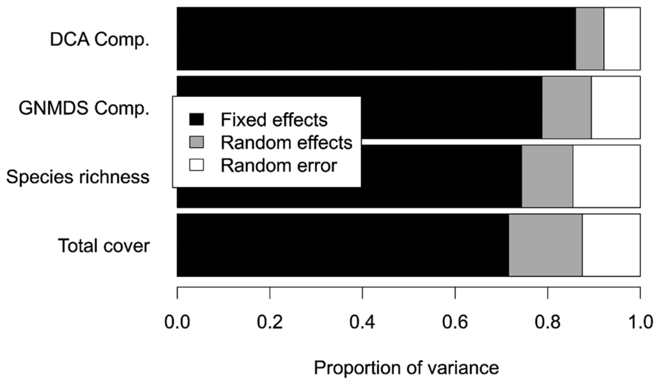

Figure 6. Proportion of variance explained by the fixed effects (restoration treatment, year, and their interactions), and the random effects (plots nested in sites), sorted by increasing variation explained by fixed effects. Detrended correspondence analysis (DCA) Comp. and global nonmetric multidimensional scaling (GNMDS) Comp. refer to species composition data of all plots (n = 242) with DCA and GNMDS, respectively (Supporting information).

What are efficient restoration treatments in alpine sites?

We found relatively high variation explained by treatment and year, and comparably low variation explained by site and plot, for all indicators of restoration outcome. Our results thus suggest that the restoration treatments are highly important and control the outcome more than site-specific factors (Grman et al. 2013, Laughlin et al. 2017).

The short-term effects of the restoration treatments differed substantially. Plots under the *Fertilization and seeding* treatment rapidly developed a vegetation cover. However, the vegetation cover seemed to stabilize at ca 60%, well below that of the reference vegetation, and species composition recovered more slowly than in the other treatments, with a high abundance of the seeded species after 17 years. The use of seed mixtures may reduce the number of species that are able to establish (Densmore 1992, Grman et al. 2013, Hagen et al. 2014), and this effect may be promoted by fertilizing, as increased productivity facilitates the dominance of competitive species (Mittelbach et al. 2001). Also, short-term monitoring at our study sites revealed this potential negative effect on long-term recovery of seeding *Festuca rubra* (Hagen and Evju 2013), and a greenhouse experiment revealed that germination and survival of *Betula nana* were suppressed by *F. rubra* (Hagen et al. 2014). Therefore, in subsequent restoration efforts of removing roads in our study area, the *Fertilization and seeding* treatments were abandoned as restoration treatments (Hagen et al. 2022).

In the *No removal of gravel* treatment, species composition appeared to recover at the same rate as the *Soil* and *Fertilization* treatments. However, the development of a vegetation cover was very slow, and the road sections where gravel was not removed were still visually striking after 17 years (Fig. 7, Supporting information). This treatment also implies that road construction is still present, and the terrain differs from the undisturbed surroundings. Several studies, both in our study area and in other alpine sites, demonstrate the importance of restoring soil and terrain conditions in order to promote vegetation recovery (Mehlhoop et al. 2018, Rydgren et al. 2020, Sulavik et al. 2021, Hagen et al. 2022). In alpine spoil heaps where no soil remediation has been carried out, the vegetation cover, soil properties, and species composition recovered extremely slowly (Rydgren et al. 2020). Although being the cheapest option, our results demonstrate that not removing gravel before stirring the soil severely impedes recovery of the ecosystem.

Our studies demonstrate that recovery trajectories in the *Soil* and the *Fertilization* treatments were similar. The monitoring thus reveals that adding fertilizer was unnecessary for assisting vegetation recovery in our study sites, and that minimal intervention is a good restoration option in this site,

Figure 7. The same study plot in the *No removal of gravel* treatment in 2009 and 2019. See Supporting information for landscape view photos. Photos: The authors.

where site conditions reflect intermediate productivity and stress (Prach and Walker 2011).

What is the expected time to recovery?

The ultimate goal of restoration, in the ecological sense, is to create a self-maintaining and resilient ecosystem (Ruiz-Jaen and Aide 2005). Full recovery after restoration is achieved when all key ecosystem attributes are very similar to those of the reference ecosystem (Gann et al. 2018). However, restored ecosystems will almost never fully reach the function and appearance of original undisturbed vegetation (Palmer and Stewart 2020). Hence, the choice of recovery threshold, i.e. the level of an indicator at which the restoration outcome is deemed satisfactory, has implications for the evaluation of the restoration project (Ehrenfeld 2001).

In this study, we followed the advice of Rydgren et al. (2019) and investigated two restoration thresholds: 1) the mean, and 2) one standard deviation off the mean, of indicator values in undisturbed reference vegetation. For total vegetation cover, choice of restoration threshold had minor effects on predicted times to recovery; recovery was generally fast. This was also partially the case for species richness, but not for species composition.

Successional change in species composition is normally a non-linear process, where successional rates gradually decline over time (Walther et al. 2002, Rydgren et al. 2019). Choice of restoration threshold had a large impact on predictions: using the centroid (mean) of reference plots increased TR approximately threefold, compared to the more relaxed criterion of one standard deviation off the centroid, for all treatments. Using a fixed reference level, i.e. mean of all reference plots, seems overly rigid, and allowing for variation around the mean, e.g. one standard deviation, as we used here, seems a more pragmatic approach. Such an approach is further supported by the observed variation in indicator values among plots in undisturbed vegetation in this study.

Our results suggest that by using the minimal intervention of the *Soil* treatment, recovery will be reached within a 20-year period. Species richness is the indicator with the quickest recovery, and our findings are in line with Mehlhoop et al. (2018) who, based on another study on removed roads in the same area, suggested that species richness was restored after 14 years, but that the time needed to reach reference levels of vegetation cover was longer. Nevertheless, the species richness indicator gives less information on whether or not a restored ecosystem is recovering along a wanted trajectory, as it discards information on species identity. In addition, a higher species richness is to be expected at early stages of succession than in intact, undisturbed vegetation; and, in our study sites, restored plots contained a larger diversity of forbs and graminoids than the alpine heath vegetation dominating in the reference sites. Rydgren et al. (2020) found recovery rates to be much slower in alpine spoil heaps in Norway. They suggest that species composition should be expected to recover more slowly than total cover and species richness. We find, however, that this depends on the restoration treatment applied. Applying fertilizers and adding seeds result in a rapid development of vegetation cover, but the species composition recovers more slowly. The spoil heaps studied by Rydgren et al. (2020) are present as artificial features in the alpine landscape, while in our study and that of Mehlhoop et al. (2018) the artificial landscape features (the roads) have been removed and the terrain is restored. Even the *No removal of gravel* treatment in our study promotes the same recovery rates of species composition as the *Soil* and *Fertilization* treatments although, as already stated, the development of vegetation cover is extremely slow.

The relatively optimistic predictions of TR could be influenced by our choice of only including vascular plants in the species composition indicator. In alpine heath vegetation, bryophytes and lichens contribute significantly to biomass and biodiversity (Cornelissen et al. 2007). Lichens have low inherent growth rates and low recovery rates (MacGillivray et al. 1995, den Herder et al. 2003), and thus restoring the full species composition could be expected to take longer. Excluding bryophytes and lichens from the species composition and richness indicators was, however, a pragmatic decision: determining bryophytes and lichens to the species level – particularly shortly after the restoration treatment – is time consuming and requires specific competence, and thus would increase costs of monitoring substantially. Our predictions on total vegetation cover are not biased by this choice. Nevertheless, our results on vascular plant species composition suggest that under such harsh environmental conditions, the rather small regional species pool ensures a relatively fast colonization of most vascular plant species from the reference site.

The roads studied in this project are narrow sites in a homogenous and undisturbed surrounding landscape, allowing for frequent supply of seeds and plant fragments from the surroundings. This is in contrast to larger disturbed landscapes and landscape features (Gretarsdottir et al. 2004, Rydgren et al. 2020). Such large and highly degraded sites are also present next to our study sites (Vloon et al. 2022) and may call for other or more comprehensive revegetation treatments, as well as site-specific evaluations of TR. In large-scale restoration projects with multiple restored sites, different times to recovery can be expected within a given landscape, even for the same indicators (Hagen et al. 2022).

Future management

A monitoring plan should be an obvious and integrated part of any restoration project (Nilsson et al. 2016), in order to measure the ecological outcome and develop cost-efficient solutions. Good monitoring depends on including the most relevant indicators, and our results suggest that species richness is not as good as vegetation cover and species composition for predicting the time for the ecosystem to recover, i.e. for key ecosystem attributes to approximate those of the reference ecosystem (Gann et al. 2018). We therefore suggest recording species composition in combination with vegetation cover in restored alpine sites.

Our study demonstrates the varying outcomes of different restoration treatments. Adding seeds and nutrients seems to be superfluous in narrow disturbed sites such as roads, even when the site is entirely vegetation-free before the restoration. Despite the rapid development of a vegetation cover, the longer-term ecological advantage of seeding is absent, compared to the minimal intervention of just stirring the topsoil. The very slow recovery in the *No removal of gravel* treatment demonstrates that the removal of added gravel and reshaping of the terrain is essential for vegetation recovery. These findings can be used to formulate specific guidelines for the technical performance of future restoration in northern alpine vegetation.

Finally, our study confirms that vegetation recovery in alpine environments is a long-term process, although depending on the target. A rapid establishment of any vegetation cover can sometimes be a desired outcome, e.g. to prevent dust flow and provide erosion control (Gretarsdottir et al. 2004). However, a preferred goal for future management of these vulnerable and valuable alpine areas (IPBES 2018) should be to restore a long-term and well-functioning ecosystem that supports biodiversity and ecosystem services. The predictions of TR are important to inform managers that the system is on the right trajectory, and to explain for politicians, funders, and the general public that despite slow processes, active restoration is an efficient and sustainable way to spend money (BenDor et al. 2015, Stange et al. 2022).

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Author contributions

Marianne Evju: Conceptualization (supporting); Data curation (lead); Formal analysis (lead); Investigation (equal); Methodology (lead); Writing – original draft (lead); Writing – review and editing (lead). **Dagmar Hagen**: Conceptualization (lead); Data curation (supporting); Formal analysis (supporting); Funding acquisition (lead); Investigation (equal); Methodology (supporting); Project administration (lead); Writing – original draft (supporting); Writing – review and editing (supporting). **Siri Lie Olsen**: Formal analysis (supporting); Investigation (equal); Methodology (supporting); Writing – original draft (supporting); Writing – review and editing (supporting). **Anne Catriona Mehlhoop**: Formal analysis (supporting); Investigation (equal); Methodology (supporting); Writing – original draft (supporting); Writing – review and editing (supporting).

Data availability statement

Data are available from the Living Norway Network Ecological Data Network: https://doi.org/10.15468/grq33j (Evju and Vang 2023).

Supporting information

The Supporting information associated with this article is available with the online version.

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