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Evaluating the efficacy of alpine restoration actions at Elgsjøveien, Norway

Eline Risberget
Master of Science in Natural Resource Management

Preface

This master thesis is the final product in my MSc in Natural resource management at the Norwegian University of Life Sciences (NMBU). Completing this research project has been challenging, but mostly it has been fun to create a study based on my own interest, carry out fieldwork in beautiful Dovre, meet new people and learn from my wonderful supervisors. Most of all, I would like to thank my supervisor Erik Aschehoug for all of the guidance, feedback, support, and, not least, time. I would also like to thank my co-supervisor Dagmar Hagen for good guidance, text feedback, and the opportunity to use the restoration project at Elgsjøveien for my thesis. Thanks to Simen Olafsen and Bernhard Askedalen for contributing to the fieldwork, to Siri Lie Olsen for help with the statistics, to Tore Sollibråten from Hafslund Eco Vannkraft Innlandet AS for background information and support, and to Øystein Dahl for help with excel. Finally, I wish to thank my family and friends for patiently supporting me through the entire process.

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Eline Risberget

Norwegian University of Life Sciences

Abstract

Human induced land-use change degrades natural ecosystems and results in habitat loss, fragmentation, and patch isolation with important consequences for biodiversity and ecosystem functions. Human pressure on natural ecosystems is expected to increase because of the construction of roads and energy production facilities (e.g. hydropower). This makes ecological restoration an important management tool to conserve biodiversity, ecosystems, landscape qualities, and combat climate change. Adoption of international principles and standards for planning and assessing the success of restoration projects can improve outcomes and contribute to the transfer of knowledge from one project to another. However, the application of principles and standards to restoration actions is rare and most restoration projects lack formal evaluation. Thus, the objective of this thesis is to evaluate the efficacy of restoration actions on an alpine plant community.

I monitored vegetation recovery along an access road (Elgsjøveien) that was built for upgrading a hydropower dam within the Knutshø landscape protected area. Based on monitoring from 2016, 2018 and 2021, I used a statistical hypothesis test to compare cover of vegetation and abiotic variables in the restored plots (referred to as disturbed plots in this thesis) against intact plots, and multivariate ordination to compare community composition in disturbed plots against intact plots within habitat types.

One vegetation type, wetland, had similar cover of functional types and abiotic variables within disturbed and intact plots. The other two vegetation types, ridge and willow, had high cover of graminoids and low cover of shrubs in disturbed plots versus intact plots. Additionally, the ridge habitat had low cover of lichens and higher cover of bryophytes in disturbed plots versus intact plots. These findings correspond with the results of total community composition. Total community composition in disturbed zones was similar to nearby intact zones within the wetland habitat, while the disturbed zones in ridge and willow habitats had a community composition that was significantly different than nearby intact communities. This suggests that disturbed wetland habitat is on a trajectory of recovery, but that recovery of other plant community types in the study area may require long periods of time or, in some cases, may not be possible because of the emergence of alternative stable states. This study underscores the importance of applying

principles and standards, including assessment, that provides information about ecosystem recovery and the need for dynamic management post-restoration.

Contents

Introduction

Human-induced changes have significantly altered natural systems globally (IPBES, 2018). For example, at the beginning of the 20th century, approximately half of the Norwegian mainland was more than 5 km from human developments and therefore classified as wilderness. However, areas of wilderness and pristine nature in Norway have declined dramatically. Today, due to the construction of roads and energy production projects just 11,5 percent of Norway's wilderness remains (Miljødirektoratet, 2022).

Construction of roads and energy production facilities (e.g. hydropower) can result in degradation and fragmentation of pristine natural systems. The direct consequences of human induced land-use changes are habitat loss and removal of populations, but accelerating effects may occur through isolation and reduction of remnant patches (Haddad et al., 2015). These impacts have been shown to drive declines in biodiversity and ecosystem functions for more than two decades following fragmentation (Haddad et al., 2015). Thus, preserving large, intact natural areas is important for biodiversity, ecosystems, landscape qualities and climate adaptation (IPBES, 2018; Miljødirektoratet, 2022).

Hydropower is the largest source of electricity in Norway (ca. 90%). Hydropower facilities need upgrading and retrofitting approximately every 50 years. Facility upgrades, including the construction of new support roads, cause substantial disturbance to both terrestrial and aquatic systems. Nearly half of all hydropower facilities were upgraded during the last 20 years, and the remaining facilities are expected to be upgraded in the next 20 years (NVE, 2022). The construction, maintenance and operation of hydropower facilities causes ecosystem degradation, and recovery can take decades or centuries, especially in high latitude, and high elevation natural systems (Campbell & Bergeron, 2012; Nilsson et al., 2016).

Ecological restoration, i.e. "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Gann et al., 2019, p.7) , can prevent species extinctions, increase overall ecosystem function, and combat climate change (McDonald et al., 2016). This makes restoration an important management tool for mitigating ecosystem damage in development projects, such as hydropower facility upgrades. Restoration has previously been neglected in Norway because of the large areas of nature (Hagen & Skrindo, 2010; Stange et al., 2021). However, ecological restoration has been identified as a key management tool to help Norway reach its national goal relating to ecosystem condition and ecosystem services (Goal 1.1;Miljødirektoratet, 2021). Because Norway has a short history of restoration, principles and standards that underpin restoration (including evaluation) can contribute to the transfer of knowledge from one project to another. Despite this obvious need, restoration often lacks formal evaluation (Hagen & Evju, 2013; Nilsson et al., 2016; Suding, 2011).

To ensure that restoration actions result in the recovery of intended ecosystem attributes, restoration practitioners should apply basic principles and standards during the restoration planning phase (Gann et al., 2019; Hobbs & Norton, 1996). In addition, a reference model that has characteristics of the target ecosystem should be established prior to restoration actions. Reference models are useful to inform 1) target ecosystem attributes 2) medium- or long-term restoration goals, and 3) intermediate objectives necessary to reach long-term goals (Gann et al., 2019; McDonald et al., 2016; Prach et al., 2019). During restoration projects, adjustments to actions may be necessary based on changes in ecosystem attributes, and therefore, an adaptive management framework that leverages best available knowledge should be adopted (Prach et al., 2019).

Alpine areas are defined as the life zone that includes vegetation above the natural high elevation tree line (Körner, 2021, p. 23) Here, plant communities are primarily influenced by climatic conditions of low temperature, strong wind, long periods of snow cover, short growing seasons, slow decomposition, and low available soil nutrients (Framstad et al., 2022). At high elevations, plants are characterized broadly as slow growing, stress-tolerant species dominated by positive interactions, in which some species may protect neighboring species against temperature, sun, drought, and herbivores (Callaway et al., 2002).

In this thesis, I study the effectiveness of restoration actions in three alpine plant community types along a newly constructed gravel road. In 2010 permission was given to construct a 2 km road in order to upgrade a hydropower dam within Knutshø landscape protection area. Post construction restoration along the road verges was required to minimize the negative impacts on disturbed plant communities. Restored vegetation communities were monitored in 2016 and 2018 (Hagen et al., 2017; Hagen et al., 2019). Building upon previous efforts to assess restoration success in 2016 and 2018, I sought to answer the following general questions:

1) Are plant communities within restored zones similar in functional type and composition to plant communities in adjacent intact zones?
2) Is the trajectory of recovery within restored zones heading towards mature, intact communities within the area?

Based on the results of data collection from 2016 and 2018, I had no *a priori* hypotheses regarding restoration success. Instead, based on existing literature, I established appropriate methodologies to assess what constitutes restoration success at Elgsjøveien and whether there are any barriers to full recovery of the restored communities.

Materials and Methods

STUDY AREA

This study was conducted in Oppdal municipality in the alpine regions of central Norway. The study site, Elgsjøveien, is a two km road between Bekkelægeret and Elgsjøen within the Knutshø landscape protection area. Elgsjøveien is situated at 62°22′34″N 9°47′09″Ø, and the elevation is approximately 1133 meters above sea level. Mean temperature is about -4 °C in January and 11 °C in July. Precipitation during the growing season ranges from 3mm to 145mm (NCCS, 2022). The bedrock is calcareous and consists of tuffite and phyllite (Norwegian Geological Survey, 2011).

The study area is situated above the tree line in the low alpine zone on the transition between C1 weak continental and OC transition vegetation region (Moen, 1998). Dwarf birch (*Betuna nana*) and willows (*Salix*) are dominating the vegetation cover. Common species are downy willow (*Salix lapponum*), grey willow (*Salix glauca* var *glauca*), common juniper (*Juniperus communis*), heather species like crowberry (*Empetrum nigrum*), bilberry (*Vaccinium myrtillus*), and blue heath (*Phyllodoce caerulea*). The area is used by grazing animals such as domestic sheep (*Ovis aries*) and wild reindeer (*Rangifer tarandus*).

HISTORY OF THE SITE, INCLUDING RESTORATION ACTIONS

The hydropower company Glommens og Laagen Brukseierforening (GLB) granted permission in 2010 to build a new road to upgrade the hydropower dam at Elgsjøen. Because of the location inside landscape protected area, permission was contingent upon specific requirements regarding the placement of the new road and ecological restoration following the construction of road (Glommen og Laagen Brukseierforening, 2014). Accordingly, construction practices and mitigating measures were focused on minimizing the negative impact on landscape and vegetation. This included removal and storage of vegetation for placement following construction and monitoring the effects of the restoration actions (Hagen & Erikstad, 2007; Hagen et al., 2017; Hagen et al., 2019). Specifically, vegetation, including the upper soil layer and plant material (including roots) were removed as “turfs” and stored along the road for two seasons and reapplied to promote re-vegetation and narrow the footprint of the road. The soil was re-applied without compression to allow for establishment of surrounding seeds. After completion of construction work in 2013, a monitoring program to follow the recovery over time was established, and the vegetation recovery was recorded in 2016 and 2018 (Hagen et al., 2017; Hagen et al., 2019).

Based on existing literature and international principles and standards for ecological restoration, I established appropriate methodologies with an emphasis on six key ecosystem attributes to determine whether the restoration actions at Elgsjøveien meet basic ecological recovery standards at the time of sampling in 2021 (Table 1) (Gann et al., 2019):

Table 1*: Targets (Gann et al., 2019, p.7), goals (Gann et al., 2019, p.14), and objectives based on six key ecosystem*

	Attribute	**Description**
Target	Community composition	To move the community composition in the disturbed plots to a trajectory of recovery informed by the reference model (intact zones and reference sites)
Goals	Absence of threats Physical conditions	Direct threats to the ecosystem such as invasive species are absent Environmental conditions (including conditions of soil moisture, pH and local temperatures) required to sustain the target ecosystem are present
	Species composition	Native species characteristics of reference ecosystem are present
	Species diversity	Appropriate diversity of key structural components is present
	Ecosystem function	Appropriate levels of decomposition, nutrient cycling, and species interactions
	Landscape	The ecosystem is appropriately integrated into its larger landscape context
Objectives	Absence of threats	Zero percent cover of identified high risk species (Artsdatabanken, 2018)
	Physical conditions	Soil moisture, pH and local temperatures in disturbed plots are not significantly different from intact plots within habitat types
	Species composition and diversity	Native species composition and diversity in disturbed zones are not significantly different from reference model within habitat types
	Ecosystem function	Cover of dead vegetation in disturbed zones is not significantly different from intact zones within habitat types
	Landscape	Percentage cover of vegetation increase along Elgsjøveien with time and reduce the visible scope of the intervention

PREVIOUS DATA COLLECTION

In 2016, 15 transects were established to monitor vegetation along the road (Fig.1) (Hagen et al., 2017; Hagen et al., 2019). The 15 transects were positioned in three vegetation types including ridge (six transects), willow (six transects) and wetland (three transects) (Fig.2). Maximum and minimum distance between the transects is 30 meters and 10 meters, respectively. Each transect was positioned perpendicular to Elgsjøveien and into vegetation on both sides of the road (Fig. 1). Thus, all transects consist of road in the middle, disturbed zones on both sides of the road where restoration was carried out, and intact zones at the beginning and end of the transect. The length of the transects was 17 meters.

Point-intercept (PO) method was applied in 2016 and 2018 to monitor vegetation within the disturbed zone and the road. All species above or below a measuring tape were recorded. Individuals were recorded if they stretched out within the area of the tape. Species that covered less than five cm were registered as a point, while species covering more than five cm were recorded as points in a sequence of five cm. To illustrate, species covering the measuring tape from cm 13-25 were registered as three points (15, 20, 25). Lichens and bryophytes were registered as groups, not at the species level.

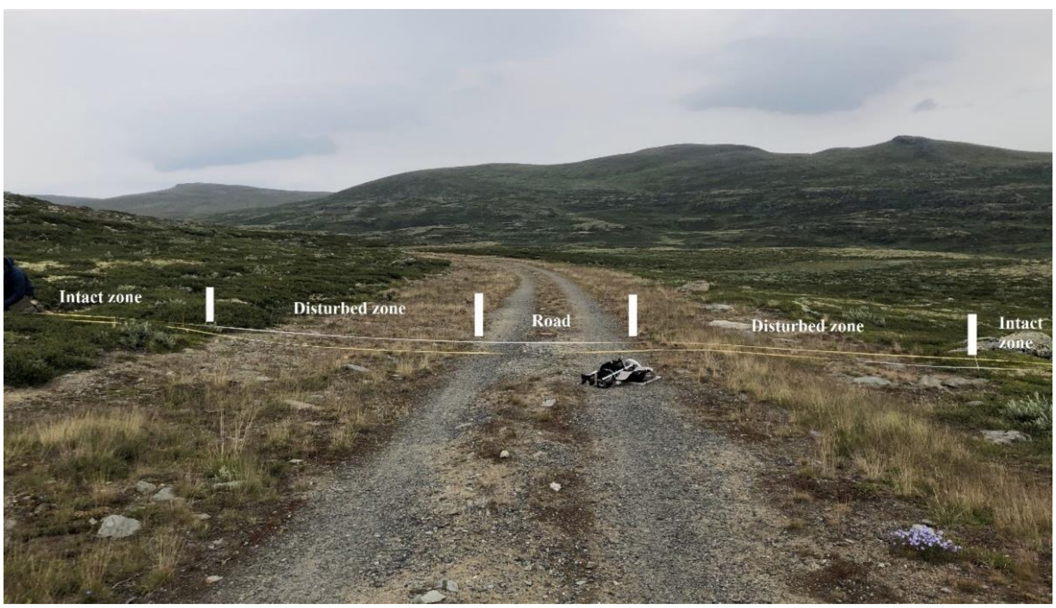

Figure 1: *The transects are positioned across the road, and into vegetation (disturbed zone and intact zone, respectively) on both sides of the road.*

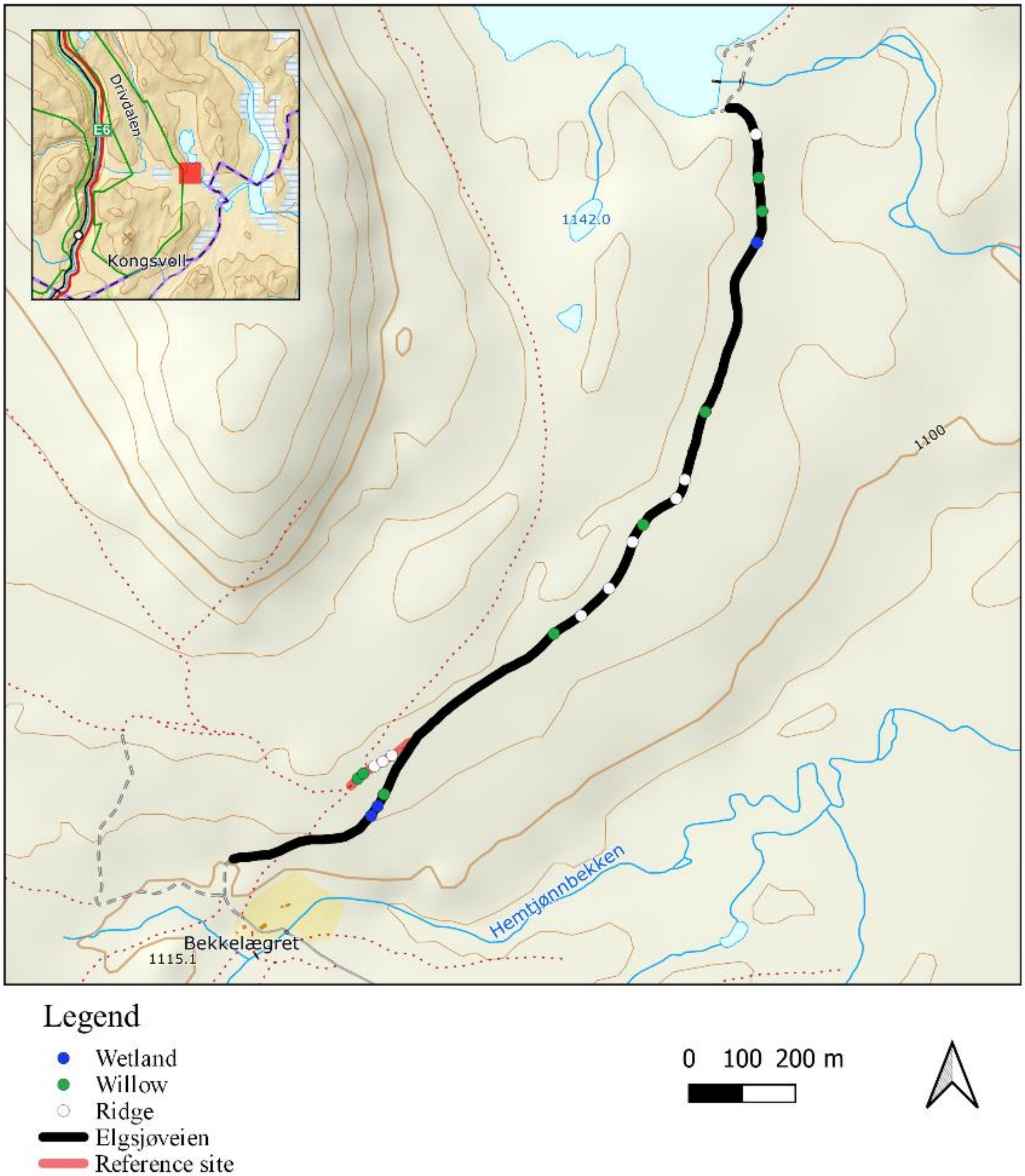

Figure 2: *Map of study site. Elgsjøveien is marked in black and located between Bekkelægret and Elgsjøen. The reference site is marked in red. Background map of Topografisk norgeskart from Geonorge.no (made by Kartverket), retrieved 05.04.22.*

DATA COLLECTION 2021

Environmental Data Collection

Environmental data was collected in 2021 to compare intact and disturbed zones. Tiny Tag temperature loggers (Intab Interface-Teknik AB, Sweden) were placed on the uphill, north-west side of Elgsjøveien. Fifteen temperature loggers were placed in the different vegetation types: wetland (four loggers), willow (six loggers), and ridge (five loggers). Within each vegetation type, one logger was placed in the middle of each of the intact zone and disturbed zone. Each logger recorded temperature every 30 minutes for 11 days. One logger malfunctioned and was not used in the analysis.

A Delta-T SM150 soil moisture sensor kit was used to record soil moisture. For each transect, soil moisture was recorded in four places within disturbed and intact zone on both sides of the road, at approximately half a meter from the markings of the unique zones. All data for soil moisture was collected on the same day and rain was absent ten days prior to data collection.

Soil was collected for laboratory analysis. Approximately 250ml was sampled from upper/north-west side of all transects on Elgsjøveien. Soil was collected from the O horizon (humus or organic) half a meter from the road in the disturbed zone and half a meter from the transition to intact zone. The soil was stored at four ºC for approximately 30 days prior to analysis. For pH measurement, soil mixed with demineralized water in a 1:2,5 ratio was centrifuged for 10 minutes (standard lab procedures). A two-pint calibration was used to measure pH, with a pH four buffer and pH seven buffer.

Vegetation Sampling

The PO method employed in 2016 and 2018 was replicated in 2021, to compare between years (Fig. 3). Some adjustments to the protocol were made. Lichens were registered as individual species or genus in 2021, while only as a group in previous years. Species were recorded every one cm instead of every five cm. Data was sampled in all three zones (road, intact zone, and disturbed zone) in 2021, while no data was sampled from intact zones in 2016 and 2018.

Percent cover of vegetation was estimated using quadrat sampling. Squares were placed at every meter mark along all transects (Fig. 3). Within the quadrat frame (0.5 x 0.5 meter), I recorded percent cover of species, bare ground, dead vegetation, gravel, rocks, and dead wood. Percent cover was standardized to 0.01 % to indicate presence, 1%, 3%, 5%, and rounded to closest 5% intervals thereafter.

Figure 3: *Point intercept method to the left, quadrat sampling to the right.*

A nearby, old, simple "two-track" road was used as a reference site in 2021 (Fig.2; Fig.4). Even though it differs in road quality to Elgsjøveien, it is useful for comparison because it is located near Elgsjøveien and represents a later successional stage that the disturbed plots are directed towards. Five transects were established within willow (two transects) and ridge (three transects) vegetation types. All reference transects are made up of road and intact zone with a total length of 1700 cm and are permanently marked with GPS locations. Environmental variables were not recorded for reference sites.

Figure 4*: Transects of different vegetation types from year 2021. Wetland (transect 5) topleft, willow heath (transect 15) topright, ridge (transect 7) bottom left, and willow bottom right (reference site 1).*

Statistical analyses

All statistical analysis were performed in R-studio version 4.1.1 (R core team, 2021).

ENVIRONMENTAL DATA

I tested whether the data met the assumptions for the specific analyses performed. Because some of the data did not conform to a normal distribution, I did a Wilcoxon Rank Sum test to study differences in environmental data between intact zones and disturbed zones. Additionally, the environmental data is presented as mean and standard error (SE) in a bar chart via the “rmisc” (Hope et al., 2013) and “ggplot2” package (Wickham, 2016).

VEGETATION DATA

Species data from quadrat sampling was grouped and summarized into functional types. Because the data did not conform to normal distribution, I did a Wilcoxon Rank Sum test to study differences between intact zones versus disturbed zones of functional types within similar vegetation types sampled in 2021. The p-values were adjusted using Bonferroni correction to counteract the issue of multiple comparisons. Differences between intact zones and disturbed zones were calculated as mean and standard error (SE) using the “rmisc” package (Hope et al., 2013) and presented in a bar chart using the “ggplot2” package (Wickham, 2016).

Multivariate ordination techniques were based on two datasets from the point intercept method. The first dataset includes data sampled in 2021 in which lichens were registered as individual species or genus, bryophytes were registered as a group, and vegetation was registered every 1cm. To allow for comparison across years, a second dataset was produced with data from all 3 years (2016, 2018 and 2021). In the second dataset, 2021 data from the point intercept method was resampled to every 5cm and lichens and bryophytes were registered as groups rather than species to conform with previous year datasets. Data from intact zones was only collected in 2021, however, there is little reason to believe that intact community composition changed considerably during the 8-year period. Thus, data from intact communities recorded in 2021 is assumed to be comparable for 2016, 2018 and 2021. Furthermore, the multivariate ordination is based on data from the transect method, with the exception of cover of dead wood, rocks, bare ground, gravel and dead vegetation, which is from the quadrat method.

I used the ordination technique global non-metric multidimensional scaling (GNMDS) on the full vegetation data matrix (20 transects) to study differences in species composition between disturbed and intact zones between habitat types. The GNMDS was two-dimensional with 100 initial configurations, maximum 200 iterations, stress tolerance 10^{-7}, and run with Bray-Curtis dissimilarity measure. I used Procrustes permutation test to compare solutions. To calculate the correlation coefficient between GNMDS ordination axis scores and environmental variables, nonparametric Kendall's τ was used. I tested for treatment effects (disturbed zone/ intact zone, disturbed zone/ reference site, intact zone/ reference site) using redundancy analysis (RDA). I used the packages "vegan" (Oksanen et al., 2012) and "MASS" (Ripley et al., 2012) for the multivariate analysis.

Results

ENVIRONMENTAL DATA

Within wetland and willow habitats, there were no significant differences in air temperature, soil moisture, and pH between disturbed plots and intact plots (Fig.5; Table 2). In ridge habitats, air temperature recorded in July was approximately 1,2°C higher in disturbed plots versus intact plots (p<0,05; Table 2). Furthermore, pH was significantly higher in disturbed plots in ridge habitats (4,8) compared to intact plots (4,5) (p<0,05; Table 2). Soil moisture did not differ between treatment plots in ridge habitats.

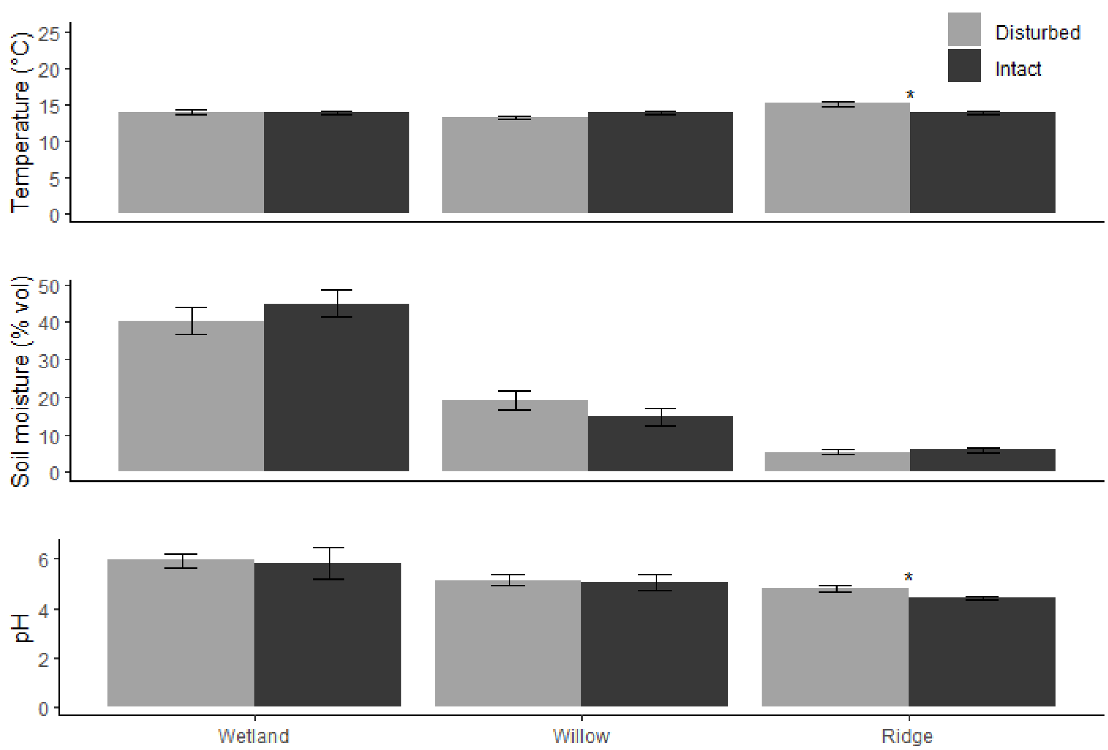

Figure 5. *Mean temperature, soil moisture, and pH of disturbed and intact plots ± SE. Stars above the bars denote significant relevance (*** P ≤ 0,001, ** P ≤ 0,01, * P ≤ 0,05).*

Table 2: *Results from Wilcoxon Rank Sum test, mean, and SE of temperature, soil moisture and pH within habitat types. Highlighted p-values are significant below a 5% test level.*

				Wetland				
	Disturbed			Intact			W	p-value
	N	Mean	SE	N	Mean	SE		
Temperature	1102	13,905	0,276	1102	13,778	0,264	605645,000	0,917
Soil moisture	18	40,289	3,547	18	44,883	3,596	136,000	0,420
pH	3	5,927	0,301	3	5,820	0,639	5,000	1,000
				Willow				
	Disturbed			Intact			W	p-value
	N	Mean	SE	N	Mean	SE		
Temperature	1653	13,173	0,201	1653	13,739	0,228	1349084,000	0,533
Soil moisture	36	19,114	2,538	36	14,683	2,417	792,000	0,106
pH	6	5,147	0,226	6	5,043	0,312	22,000	0,575
				Ridge				
	Disturbed			Intact			W	p-value
	N	Mean	SE	N	Mean	SE		
Temperature	1102	15,068	0,306	1102	13,831	0,276	643092,000	**0,016**
Soil moisture	36	5,242	0,582	36	5,864	0,727	612,500	0,693
pH	6	4,797	0,118	6	4,453	0,064	32,000	**0,030**

COVER OF VEGETATION WITHIN HABITAT TYPES

Within the *wetland habitat*, cover of rocks was significantly higher in disturbed plots (14%) versus intact plots (3%; Fig.6; Table 3). Cover of bryophytes in the intact zones of wetland was approximately three times higher than disturbed plots, although this was not significant ($p=0,062$; Table 3). There were no other significant differences in cover between disturbed versus intact plots within the wetland habitat type for any of the other functional types or abiotic characteristics (Table 3).

Within the *willow vegetation* type, cover of graminoids in disturbed plots was two times higher than in nearby intact plots ($p<0,05$; Table 3). The average cover of berries and shrubs were approximately six times higher ($p<0,01$) and three times higher ($p<0,001$) respectively in intact versus disturbed plots. Cover of lichens, bryophytes, forbs, bare ground, dead vegetation, gravel, rocks, and dead wood were not significantly different in disturbed versus intact zones (Figure 6; Table 3).

Within the *ridge habitat*, average cover of graminoids was approximately 26% in disturbed plots compared to less than 1% in intact plots (<0,001; Table 3). Cover of berries, lichens and shrubs were approximately 7, 13 and 4 times higher in intact versus disturbed plots, respectively (p<0,001). Cover of bryophytes, bare ground, dead vegetation, gravel, rocks, and dead wood were significantly higher in disturbed versus intact plots. However, cover of forbs was not significantly different between plot types within ridge habitat (Figure 6; Table 3).

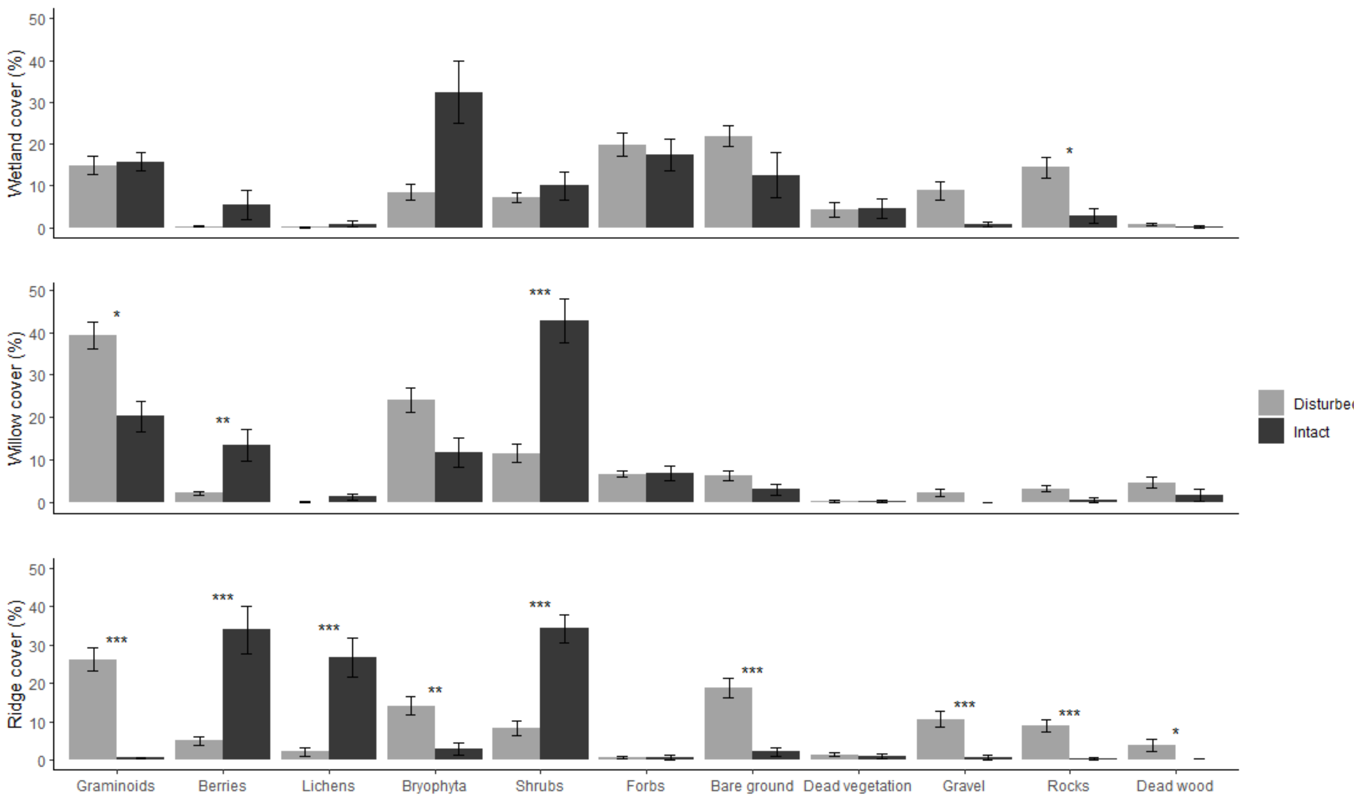

Figure 6: *Mean cover of all functional groups and abiotic cover types within each habitat type. Error bars represent ± SE, and stars above the bars denote significant relevance (*** P ≤ 0,001, ** P ≤ 0,01, * P ≤ 0,05).*

Table 3. *Descriptive statistics for cover. Results from Wilcoxon Rank Sum test with Bonferroni correction, mean, and SE of cover within habitat types. Highlighted p-values are significant below a 5% test level.*

	Wetland							
	Disturbed			Intact			W	p-value
	N	Mean cover	SE	N	Mean cover	SE		
Graminoids	30	0,149	0,021	12	0,158	0,022	145,500	11,329
Berries	30	0,003	0,002	12	0,054	0,035	153,000	8,438
Lichens	30	0,002	0,001	12	0,009	0,008	172,500	24,707
Bryophyta	30	0,085	0,018	12	0,324	0,075	69,000	0,062
Shurbs	30	0,071	0,012	12	0,101	0,033	158,500	18,407
Forbs	30	0,199	0,028	12	0,174	0,038	195,000	22,641
Bare ground	30	0,218	0,025	12	0,126	0,055	249,500	1,712
Dead vegetation	30	0,042	0,017	12	0,047	0,024	172,000	26,357
Gravel	30	0,089	0,022	12	0,008	0,006	284,000	0,079
Rocks	30	0,144	0,024	12	0,029	0,018	302,500	**0,017**
Dead wood	30	0,007	0,003	12	0,003	0,003	207,000	9,280
	Willow							
	Disturbed			Intact			W	p-value
	N	Mean cover	SE	N	Mean cover	SE		
Graminoids	58	0,394	0,032	26	0,204	0,036	1119,000	**0,014**
Berries	58	0,023	0,005	26	0,135	0,038	376,000	**0,005**
Lichens	58	0,002	0,001	26	0,014	0,006	623,000	3,184
Bryophyta	58	0,242	0,028	26	0,118	0,034	1056,000	0,110
Shurbs	58	0,117	0,022	26	0,428	0,051	260,500	**0,000**
Forbs	58	0,068	0,007	26	0,069	0,017	868,000	8,956
Bare ground	58	0,064	0,011	26	0,032	0,014	1010,000	0,285
Dead vegetation	58	0,003	0,002	26	0,005	0,003	708,500	10,870
Gravel	58	0,024	0,008	26	0,001	0,001	938,000	0,368
Rocks	58	0,034	0,007	26	0,007	0,004	1000,500	0,147
Dead wood	58	0,048	0,014	26	0,019	0,014	1007,500	0,204
	Ridge							
	Disturbed			Intact			W	p-value
	N	Mean cover	SE	N	Mean	SE		
Graminoids	60	0,262	0,030	24	0,006	0,003	1377,000	**0,000**
Berries	60	0,051	0,012	24	0,341	0,062	175,500	**0,000**
Lichens	60	0,022	0,010	24	0,269	0,051	129,000	**0,000**
Bryophyta	60	0,141	0,024	24	0,030	0,015	1096,000	**0,004**
Shurbs	60	0,084	0,019	24	0,343	0,037	190,500	**0,000**
Forbs	60	0,007	0,003	24	0,007	0,006	843,500	3,594
Bare ground	60	0,188	0,025	24	0,022	0,011	1247,500	**0,000**
Dead vegetation	60	0,015	0,006	24	0,010	0,006	790,000	10,778
Gravel	60	0,107	0,021	24	0,006	0,006	1129,000	**0,000**
Rocks	60	0,091	0,016	24	0,004	0,002	1154,500	**0,000**
Dead wood	60	0,039	0,015	24	0,002	0,002	1011,000	**0,019**

COMMUNITY COMPOSITION

In ordination for global nonmetric multidimensional scaling (GNMDS), soil moisture and pH were the most important variables explaining community composition along axis one (Fig. 7; Table 4). The first axis correlated with soil moisture, pH, bare ground, gravel, rocks, and dead wood, while the second axis correlated with pH and weakly with soil moisture and temperature (Fig. 7). To illustrate, the results show higher soil moisture for wetland and willow vegetation type as compared to ridge vegetation type which is characterized by wind and drier conditions.

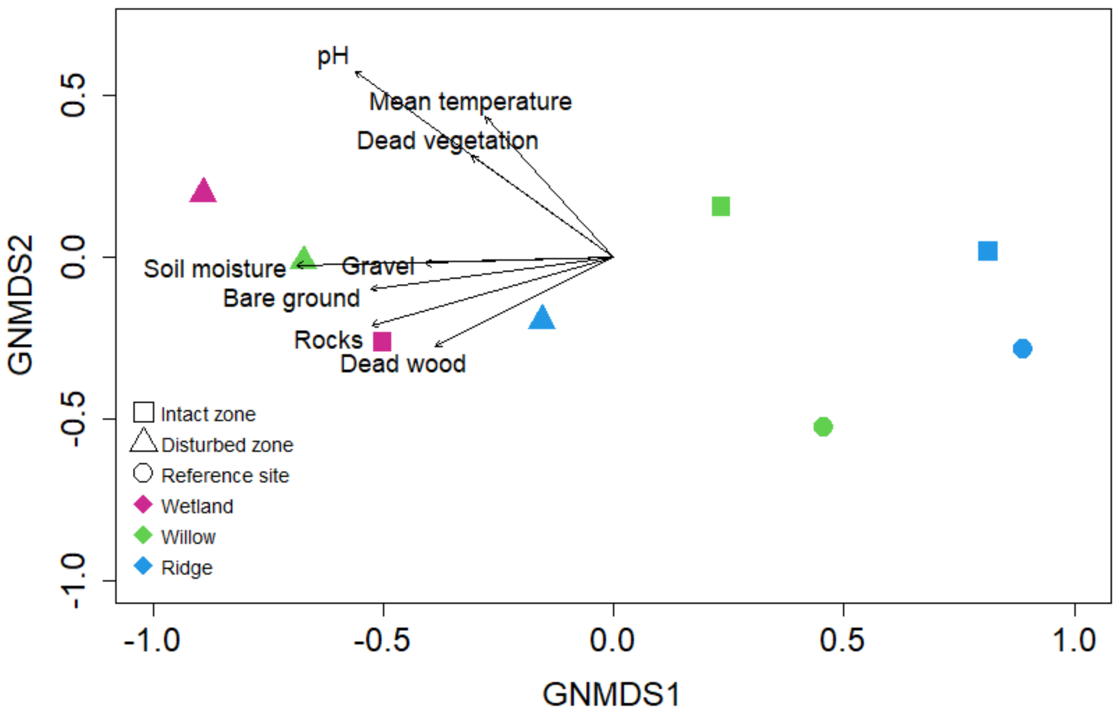

Figure 7*: GNMDS ordination plot of total plant community composition in intact, disturbed, and reference sites (symbol shape) for wetland, willow, and ridge vegetation type (symbol colour). Arrows indicate correlations between the ordination and environmental variables. Arrow length indicates correlation strength. All data was collected July 2021.*

Table 4*: Kendall's rank correlation tau testing the correlation coefficient between GNMDS ordination axis scores and environmental variables. Highlighted values are significant below a 5% test level.*

Variable	GNMDS1	GNMDS2
Soil moisture	**0,000**	0,085
Temperature	0,111	0,074
pH	**0,000**	**0,039**
Bare ground	**0,000**	0,525
Dead vegetation	0,176	0,140
Gravel	**0,000**	0,458
Rocks	**0,000**	0,273
Dead wood	**0,001**	0,566

For the *wetland habitat*, the GNMDS indicated similarities in total community composition within disturbed and intact plots measured in 2021 (Fig.8). Accordingly, the redundancy analyses (RDA) showed no significant difference in total plant community composition between intact plots and disturbed plots for wetland during 2021 (p= 0,2; Table 5). Ordination from all three years (2016, 2018 and 2021) showed signs of wetland recovery as community composition within disturbed zones are displaced towards intact plots over time (Fig. 9).

For the *willow vegetation* type, total community composition within disturbed plots appears to be different from intact plots and reference sites, as indicated by separation along axis 1 (Fig.8). The results from the RDA showed that community composition in 2021 was significantly different in disturbed plots versus nearby intact plots (p<0,01; Table 5), and disturbed plots versus reference site (p<0,05; Table 5). However, displacement of disturbed plots towards the intact plots in 2021 may suggest positive signs of recovery (Fig.9).

Total community composition within the *ridge habitat* appears to be different in disturbed plots compared to intact plots and reference sites in 2021 (Fig.8). Accordingly, the RDA showed a significant difference in plant community composition within disturbed plots versus intact plots (p<0,01; Table 5), and disturbed plots versus reference site (p<0,05; Table 5). Total community composition within ridge showed no signs of recovery eight years after completion of restoration, as indicated by the displacement of disturbed plots away from intact plots in 2021 (Fig. 9).

The RDA of intact zones versus reference site showed a significant difference in plant community composition within ridge habitats (p<0,05; Table 5). Intact zones and reference site within the willow habitat was weakly correlated but showed no significant difference (p=0,063; Table 5).

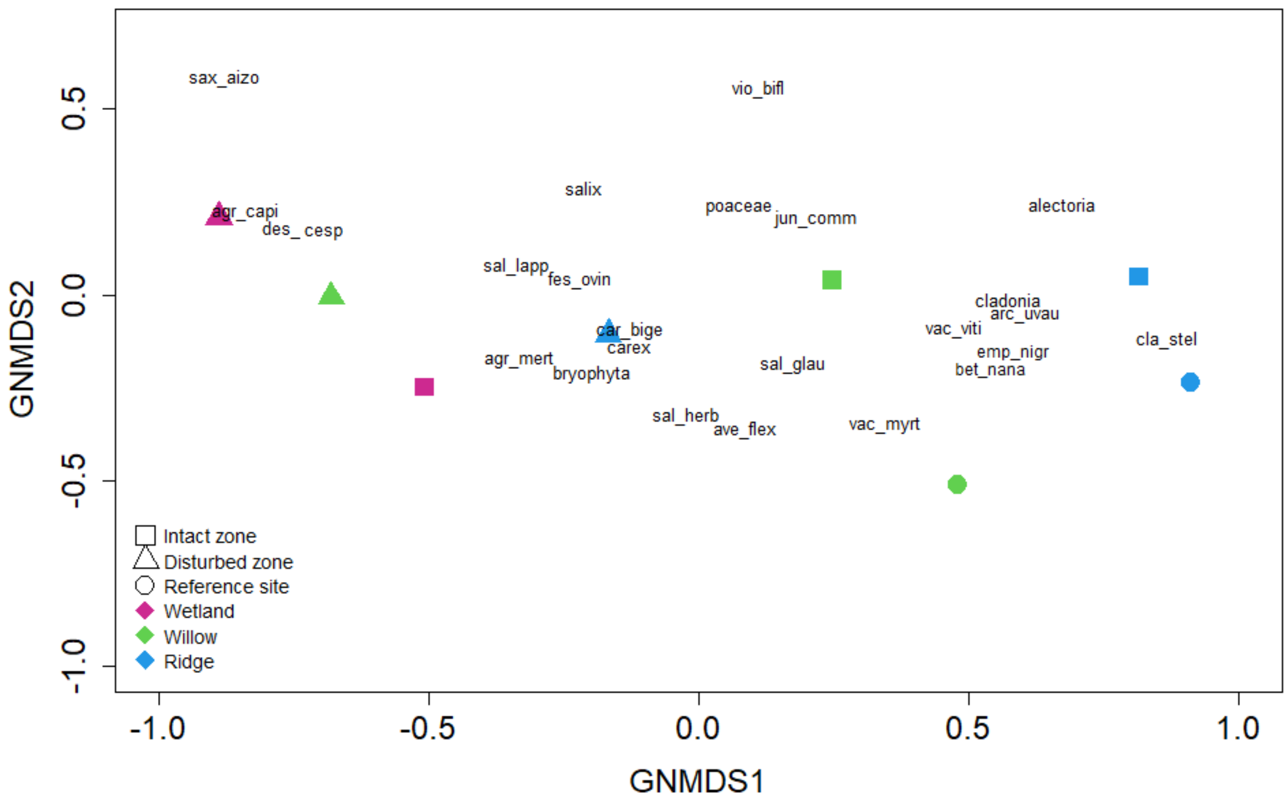

Figure 8: *Ordination for global nonmetric multidimensional scaling (GNMDS) of total plant community composition in intact, disturbed, and reference sites (symbol shape) for wetland, willow, and ridge vegetation type (symbol colour). Only species with abundance >150 are shown, and all data was collected July 2021. Species are registered every one cm, and species of bryophytes are collectively referred to as bryophytes. For species abbreviations, see Appendix 1.*

Table 5*: P-values of redundancy analyses (RDA) testing the effects of disturbed zones versus intact zones, disturbed zones versus reference site, and intact zones versus reference sites on total community composition from 2021. Highlighted p-values are significant below a 0,05% test level.*

	Total plant community composition		
	Wetland	Willow	Ridge
Disturbed zone/ intact zone	0,2	**0,002**	**0,003**
Disturbed zone/ reference site	NA	**0,034**	**0,014**
Intact zone/ reference site	NA	0,063	**0,012**

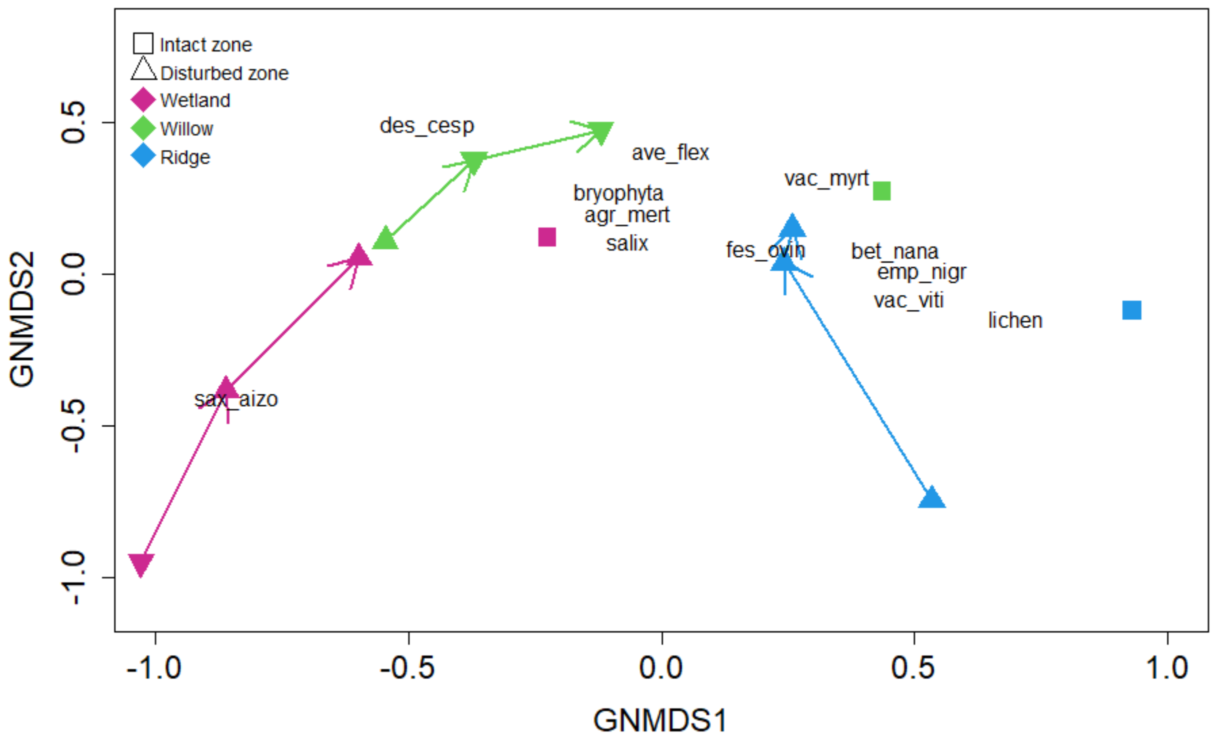

Figure 9*: GNMDS ordination plot of total plant community composition in intact- and disturbed plots (symbol shape) for wetland, willow, and ridge vegetation type (symbol colour). Arrows show the trajectory of wetland, willow and ridge community from 2016 to 2018 to 2021. Only species with abundance >200 are shown. To allow for comparison, species cover is rounded to nearest 5 cm for all 3 years, and species of lichens are collectively referred to as lichens. For species abbreviations, see Appendix 1.*

Discussion

Of the three major vegetation types, only wetland showed similar cover of functional types and abiotic variables within disturbed and intact zones. Ridge and willow vegetation types had significantly higher cover of graminoids and significantly lower cover of shrubs in disturbed than in intact zones. In ridge plots, lichens had not recovered from disturbance, and cover of bryophytes was high in disturbed plots compared to nearby intact plots. These results of cover correspond with the results of total community composition. Wetland community composition in disturbed zones was similar to intact zones, while the disturbed zones in ridge and willow habitats had a community composition that was significantly different than nearby intact communities. The results suggests that disturbed wetland habitat is on a trajectory of recovery, but that recovery of other plant community types in the study area may require more time or, in some cases, may not be possible because of the emergence of alternative stable states.

COVER OF VEGETATION WITHIN HABITAT TYPES

Restoration actions can facilitate recovery, but after eight years there are still clear differences in cover of functional types between disturbed and intact plots within the study area (Fig. 6). Establishment of alpine vegetation is slow because of the cold climate and short growing seasons (Framstad et al., 2022). Thus, recovery of communities from human degradation may take a long period of time (Madeline et al., 2018). Vegetation turfs can function as sites for plant establishment and dispersal because they may be sources of mature individuals of locally native species, soil seed bank, soil biota, and mycorrhizae (Conlin & Ebersole, 2001; Klimeš et al., 2010; Mehlhoop et al., 2018). By facilitating plant establishment, turf transplantations can decrease vegetation recovery time. This is in line with other studies that found increased species richness and vegetation cover relating to turfs transplantations (Aradóttir & Óskarsdóttir, 2013; Hagen & Evju, 2013), and of time as the primary factor explaining species establishment and vegetation cover (Hagen et al., 2022; Mehlhoop et al., 2018).

Wetland Habitat Type

Similar cover of functional types and abiotic variables in disturbed and intact plots within wetland habitat may be related to the characteristics of the turf transplantations (Fig.6). The turfs that were collected from the wetland habitat along Elgsjøveien consisted of more organic mass and a deeper soil layer than the other two major vegetation types. Thus, the turfs that were transplanted back after construction likely contained a diverse seed bank and plant parts that improved the trajectory of wetland recovery. The removed turfs were stored on cloths for two growing seasons, leaving them vulnerable to drought. However, soil characteristics of the wetland habitat, particularly high soil moisture, may have resulted in fewer negative effects of storage for wetland vegetation turfs, than the other two vegetation types with lower levels of soil moisture (Madeline et al., 2018).

Ridge and Willow habitat types

Graminoids were the most common functional group within disturbed plots of ridge and willow habitats, but not in nearby intact plots (Fig.6). Increased cover of graminoids after turf transplantation have been reported by other studies (Aradottir, 2012; Aradóttir & Óskarsdóttir, 2013; Bay & Ebersole, 2006). Soil nutrients are generally low in alpine ecosystems (Walker & Del Moral, 2003) and nitrogen is the most limiting nutrient to alpine plant productivity (Körner, 1999). Turf transplants may enhance nutrient supply, including nitrogen, (Bruelheide, 2003) via decomposition of dead plant biomass that results from the intense stress of turf removal and transplantation. Grasses respond positively to increases in nutrients (Jägerbrand et al., 2009; Klanderud, 2008) and may be causing a type conversion from a system dominated by stress tolerant alpine species towards competitive graminoid species in disturbed plots within ridge and willow habitats. Furthermore, increased soil surface temperature within disturbed ridge plots (Fig.5) may also favour grasses. This is in line with other studies that found increased graminoid abundance and biomass in response to warming (Walker et al., 2006), and warming combined with nutrient addition in alpine and arctic environments (Jägerbrand et al., 2009; Klanderud, 2008; Olsen & Klanderud, 2014).

The response to increased soil nutrients and warming differ for individual species and functional types, and may favour graminoids in disturbed ridge and willow habitats (Fig.6). Grasses are strong competitive species because of their fibrous roots, big root/shoot ratio, and high nitrogen use efficiency (Caldwell & Richards, 1986; Shaver et al., 1997). Accordingly, altered abiotic conditions can increase dominance of nutrient-demanding competitors like grasses at the expense of other functional types (Jägerbrand et al., 2009; Klanderud & Totland, 2005; Olsen & Klanderud, 2014). For example, increased abundance of graminoids affect competition for light and can decrease cryptogam abundance because of shade-effects and burial from litter (Walker et al., 2006). Furthermore, when two species require the same resources competitive exclusion may occur (Gause, 1934). Thus, increased dominance and competition from grasses at Elgsjøveien can exert strong influence on community structure. Additionally, the low water availability within the ridge vegetation type may further increase the competitive effects of grasses (Michalet et al., 2016).

Within ridge habitats, lichens have not recovered and showed significantly lower cover in disturbed plots compared to intact plots (Fig.6). Patterns of primary succession show slow establishment of lichens compared to bryophytes (Hagen et al., 2019; Rydgren et al., 2013). Bryophytes had significantly higher cover in disturbed plots versus intact plots (Fig.6). Bryophytes are considered pioneer species that typically establish soon after disturbance (Rydgren et al., 2020), which may contribute to their rapid recovery in ridge habitats. Another factor explaining the recovery of bryophytes is the turf transplants. Aradottir (2012) tested the effect of different turfs transplants and found that moss cover increased with time in all treatment plots, but the increase was the fastest in shredded turfs. Shredded turfs consist of bryophyte fragments that act as effective propagules (Mälson & Rydin, 2007). Similarly, the turf transplants at Elgsjøveien were fragmented and shredded into smaller pieces, favouring bryophytes (Hagen et al., 2019).

Cover of shrubs has not been restored to the pre-disturbed state within ridge and willow vegetation types (Fig. 6). Many shrubs in the turf transplants died, and many shrubs growing underneath the stored turfs for two seasons were damaged or died (Hagen et al., 2019). Drought and damage to turfs (and roots particularly) may limit vegetative propagation and shrub recovery. Long turf storage time can hinder establishment because of limited willow (*Salix*) seed viability (Raven, 1992). Thus, if willows are dispersed from surrounding areas later, competition from already established species may constrain shrub establishment. Aradottir (2012) found that shrubs were less tolerant to division into small turfs compared to other functional types like grasses and bryophytes. This is relevant to Elgsjøveien because the turfs were fragmented and shredded into smaller pieces (Hagen et al., 2019).

Shrubs can act as nurse plants in restoration because their allocational patterns and architecture allow for niche partitioning and facilitate other species at minimum competitive costs (Gómez-Aparicio, 2009). For example, facilitation by shrubs in alpine ecosystems can alleviate the high stress of exposed areas and enhance diversity (Ballantyne & Pickering, 2015). Accordingly, lack of recovery of shrubs in disturbed plots within ridge and willow habitats may have consequences for community structure because positive interactions can increase diversity, promote co-existence and reduce the negative effects of competitive exclusion (Aschehoug et al., 2016).

Herbivore selectivity and plant tolerance are linked to species-specific traits. Plants with resistance/ avoidance strategy and tolerance strategy (regrowth capacity) in alpine ecosystems tend to increase in abundance under enhanced grazing pressure (Evju et al., 2009). Grazers such as sheep are present within the study site at Elgsjøveien, although the grazing pressure is unknown. However, studies have shown that low sheep densities increase shrub abundance (*Salix* ssp.), while high sheep densities increase lichens and bryophytes (Austrheim et al., 2014) and promote tolerant graminoid species (Austrheim et al., 2008; Van der Wal & Brooker, 2004). Additionally, herbivores can alter ecosystem processes like decomposition with effects on aboveground plants and indirect effects on below-ground systems (Augustine & McNaughton, 1998). Faeces and urine from herbivores can benefit grasses by way of added nutrients and contribute to positive feedback loops in which proliferation of grasses by herbivores attract grazers and thus further promote grass abundance (Van der Wal & Brooker, 2004). As a consequence, recovery of willow and ridge types may be further hindered by the presence of grazers.

COMMUNITY COMPOSITION

Wetland Habitat

I found that wetland community composition was similar in disturbed zones and intact zones (Fig.8). Soil characteristics, particularly soil moisture, play an important role in forming local community composition. Other studies have shown that soil moisture had a positive effect on plant diversity (Madeline et al., 2018), and that recovery of wetland communities can happen in relatively short periods of time (10-30 years) compared to drier habitat types (Forbes & Jefferies, 1999). This may help explain why wetland community composition in disturbed zones was similar to intact zones, while the drier habitat types, ridge and willow, had a community composition that was significantly different from nearby intact communities.

Willow Habitat

Community composition within the willow habitat was different in disturbed zones versus intact zones and reference sites (Fig.8), but the disturbed communities may be on a trajectory to recovery (Fig.9). Although the abiotic variables measured in disturbed zones were sufficient for recovery, willow habitats are dominated by shrubs and woody species that dominate later in succession (Alday et al., 2011) and shrubs may facilitate other species that are typical for communities in willow habitats (Gómez-Aparicio, 2009). Thus, it is too soon to determine whether succession of willow communities will result in restoration of original community composition.

Ridge Habitat

Ridge community composition in disturbed zones was displaced away from intact plots and does not show a trajectory towards recovery over time (Fig.9). Although restoration of many degraded systems can follow directional change from pioneer stage towards stable climax states (Suding et al., 2004), communities can also shift between multiple stable states because of changes in ecological processes, disturbance, or attributes of the community (Petraitis, 2013). Olsen & Klanderud (2014) found that shifts in dominance hierarchies by competitive grasses at the expense of other functional types altered alpine community composition and are difficult to reverse. While plant recovery in alpine ecosystems is inherently slow (Hagen & Evju, 2013) and long term-responses of alpine plant communities are uncertain, it is possible that a shift in dominance hierarchies is taking place at Elgsjøveien within disturbed ridge zones, moving the community into an alternative stable state dominated by graminids. However, to test whether a degraded system is characterized as an alternative stable state is normally beyond the scope of restoration efforts (Suding et al., 2004).

WHAT CONSTITUTES RESTORATION SUCCESS

There is a need for all restoration projects to adhere to basic principles and standards in order to increase effectiveness and success (Gann et al., 2019). As such, I established methodologies and success criteria consistent with the international principles and standards set forth in Gann et al. (2019). For example, the reference model in this project is not based on a community at some past point in time but is derived from multiple sources (intact zones and reference sites). Furthermore, I used community level data on plant species for evaluating restoration outcomes, as advocated by Rydgren et al. (2020). Accordingly, the overall target at Elgsjøveien is to move community composition in the disturbed zones to a trajectory of recovery as informed by the reference model, while also allowing for adaptation to changes (Gann et al., 2019). I found that one habitat type, wetland, is on target for recovery based on vegetation cover and community composition data. However, ridge and willow habitats have significantly different community composition in disturbed plots versus nearby intact plots and the reference site.

The goals and objectives used to assess recovery progress at Elgsjøveien include six key ecosystem attributes (Table 1) (Gann et al., 2019). The different vegetation types show different status and recovery. The first goal, absence of threats from invasive species was fulfilled, as no invasive species have been registered. Invasive species can threaten natural plant communities and exacerbate native species decline when introduced to new ranges because they bring mechanisms of interactions to native plant communities that can prevent recovery (Callaway & Aschehoug, 2000). Specifically, the competitive effects of introduced invasive species can disrupt interactions among species resulting in the collapse of native communities.

The second goal relates to fundamental assumptions about whether the abiotic conditions within the disturbed zones can support plant community restoration. The environmental conditions measured are sufficient to restore disturbed wetland and willow habitats, but after eight years post-restoration, temperature and pH within the ridge habitat are still significantly different from nearby intact zones, which may be a barrier to recovery.

Multivariate ordination incorporates species diversity and community composition into the analysis and thus provide insight into the third and fourth goal. Native species diversity and community composition of the disturbed wetland habitat have similar characteristics to the reference model. However, the disturbed zones of willow and ridge habitats are significantly different from the reference model at the time of monitoring in 2021.

While assessment of community composition is based on clear and measurable indicators, assessment of ecosystem functions (goal 5; decomposition, nutrient cycling, and species interactions) is more complex. For example, the measure of indicators like litter, rather than the functions themselves, gives an understanding of decomposition within disturbed plots versus intact plots. The cover of dead vegetation in disturbed zones is within appropriate levels relative to intact zones for all habitat types, while cover of dead wood is significantly higher in disturbed versus intact plots within the ridge habitat. Because of slow decomposition rates, dead wood can affect nutrient cycling in the future. Furthermore, accumulated litter from grasses can return nitrogen to the soil, with positive feedbacks that may favour grasses (Klanderud, 2008). Graminoid domination in disturbed plots of ridge and willow habitats provides insight into the last part of Goal 5, namely species interactions. Grasses can exert strong influence on community composition by way of competition, which may prevent successful restoration.

The visual impression of the restoration site and its integration into the larger landscape is also an important outcome of restoration actions (Goal 6). This attribute can be assessed in two ways; as a direct output from the construction when the road was immersed in the terrain and constructed with minimal land-use change (Glommen og Laagen Brukseierforening, 2014), and the visible increase in cover of vegetation along the road from 2016 to 2021 (Appendix 2). The reference site can be useful for comparison of the visible cover increase because soil was reapplied in the middle section of the road at Elgsjøveien to resemble a simple “two-track” road over time. The visible impression of the road is reduced, and Elgsjøveien is better incorporated into the remaining natural areas in Knutshø landscape protection area as a result of restoration actions.

MANAGEMENT DECISIONS

Adaptive Management, informed by short-term data and assessment of initial ecosystem recovery, can be used to update knowledge and adjust restoration practices (Gann et al., 2019; Hagen & Evju, 2013; Schaaf et al., 2011). Although the distinction between alternative stable states and ongoing successional change can be difficult (Petraitis & Latham, 1999), especially within the initial recovery phase, lack of management actions can result in unwanted restoration outcomes. It is unclear whether the dominance of grasses within the disturbed ridge habitats represents an emerging alternative stable state, but grasses are highly competitive species that may be constraining recovery of ridge habitats (Jägerbrand et al., 2009; Shaver et al., 1997). Often, removal of problematic species is necessary before the system can recover or respond to any other management actions (Suding et al., 2004). Thus, removal of graminoids within disturbed ridge sites may help restore original community composition, particularly if the restored community at Elgsjøveien exhibits hysteresis, i.e. an inability to return to the original community after restoration actions (Petraitis, 2013).

Although transplant of vegetation turfs can promote more rapid vegetation establishment (Hagen & Evju, 2013), a number of the willow plants (*Salix*) were damaged or killed during this process at Elgsjøveien. Some of the willow plants underneath the stored turfs were damaged because of lack of sunlight, while the willows in the turfs may have been damaged because of a long storage time (two seasons) and/or a thin soil layer in the turf (Hagen et al., 2019). If possible, future restoration should decrease the turf storage time, or implement active measures to restore willow populations. The use of locally adapted individuals may be important for restoration success because individuals from local sites may have higher fitness than individuals from distant sites (Menges, 2008), therefore, additional turfs of willows (including soil layer and roots) can be translocated from local sources to the restoration site. Here, site preparations that include the presence of soil organic matter and removal of foreign materials are important for vegetation recovery (Mehlhoop et al., 2018).

The impact of land-use change (e.g. fragmentation) can be inter-related to other agents of change like invasive species (Didham et al., 2007). In addition to the degrading effects on biodiversity and ecosystem function caused by fragmentation (Haddad et al., 2015), roads facilitate and increase human use, which may unintentionally introduce and disperse invasive species (Ansong & Pickering, 2014). Although no invasive species were found in the study sites along Elgsjøveien, the new road will likely attract more human use. Thus, information boards at access points to the Knutshø protected area that encourage self-inspection and cleaning can decrease the likelihood of human activities dispersing invasive species to vulnerable areas.

ECOLOGICAL RESTORATION MOVING FORWARD

Ecosystems are dynamic entities and often do not respond predictably to management efforts (Hobbs & Harris, 2001). Accordingly, finding appropriate reference models for restoration sites is challenging, as demonstrated by the significant difference between total community composition in intact zones versus reference site within ridge habitats (Table 5). While local scientific knowledge can inform the design and implement restoration projects, how ecosystems may respond to management treatments and climate change over time represent crucial knowledge gaps and uncertainty (Gann et al., 2019). For example, biodiversity contributes to ecosystem resilience under predicted warming because it is positively correlated with community stability, i.e. the ability to return or remain in the original structure and function after a disturbance (Elton, 1958; McCann, 2000; Tilman & Downing, 1994). Thus, ecological restoration when implemented with a basis in ecological theory, restoration standards, and an adaptive management framework can support not just restoration outcomes, but long-term biodiversity and climate change mitigation (Gann et al., 2019).

SOURCES OF UNCERTAINTY AND THE ECOLOGICAL SIGNIFICANCE OF THIS STUDY

Selecting a suitable reference model from diverse sources is crucial to assess ecosystem recovery (Gann et al., 2019; Prach et al., 2019). However, it was challenging to find appropriate wetland reference sites near the study area that operate under similar local environmental conditions as Elgsjøveien. Reference sites in ridge and willow habitats are located near one another and were, at times, difficult to distinguish from each other. However, it can be beneficial to select reference sites that capture the full breadth of habitat variation in the local landscape because it gives insight to all of the possible outcomes of ecosystem recovery.

Restoration of degraded alpine systems take long periods of time, and it is important to continually assess restoration success throughout the process of succession to obtain reliable results (Auestad et al., 2016). The data collection that forms the basis for this thesis was carried out during different time periods, by different people, and with different methodologies. This may have introduced bias or increased error in the dataset.

The understanding of the trajectory of restoration could be improved by having employed more frequent sampling and appropriate methods during all years of sampling. For example, data from intact zones was not collected in 2016 and 2018, and no data was collected prior to the initiation of road construction, which limits the quality of the reference model for the study. Furthermore, recording all individuals to species level rather than major groups could also improve the understanding of the trajectory.

Conclusion

Restoring ecosystem processes, biodiversity, and landscape qualities can help mitigate the effects of human development, combat climate change, and contribute to the green transition. However, restoration should not be considered a substitute for protecting pristine ecosystems because the quality of restored ecosystems may never be equal to undisturbed wilderness (McDonald et al., 2016) and recovery, if it occurs, may take decades (Campbell & Bergeron, 2012; Nilsson et al., 2016). Use of international standards within an adaptive management framework can improve restoration success, however most restoration projects fail to establish recovery objectives and conduct any form of evaluation. As a result, little is known about how successful restoration actions are at achieving meaningful ecosystem recovery.

This study serves as a model of how evaluation can provide valuable feedback on restoration success, even when the initial project design did not include specific restoration objectives. Here, evaluation based on data sampling carried out at three different times over an eight-year period post-restoration showed that one habitat type, wetland, is on a trajectory towards full recovery, while two other habitat types, willow and ridge, are dominated by graminoids that may prevent full recovery. This demonstrates the need for repeated assessment to identify how human disturbance may have altered abiotic and biotic factors, and whether negative feedbacks that inhibit recovery have emerged.

As human pressure on natural systems increases, politicians, policymakers and the general public should prioritize the preservation of intact wilderness areas. In addition, as Norway adopts a policy of restoration, there must be an explicit agreement to include future evaluation and management as crucial components of restoration process. Importantly, because of the known limits of restoration, future human induced land-use change should be carefully considered.

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Appendix

Appendix 1. Full species list from all transects (2016, 2018 & 2021)

Scientific name	**Species abbreviation**
Agrostis capillaris	agr_capi
Agrostis mertensii	agr_mert
Agrostis	agrostis
Alchemilla alpina	alc_alpi
Alchemilla	alchemilla
Alectoria	alectoria
Andromeda polifolia	and_poli
Antennaria alpina	ant_alpi
Antennaria dioica	ant_dioi
Anthoxanthum nipponicum	ant_nipp
Anthoxanthum odoratum	ant_odor
Arctous alpina	arc_alpi
Arctostaphylos uva-ursi	arc_uvau
Artemisia norvegica	art_norv
Avenella flexuosa	ave_flex
Bartsia alpina	bar_alpi
Betula nana subsp. nana	bet_nana
Bistorta vivipara	bis_vivi
Bryocaulon divergens	bry_dive
Bryophytes	bryophyta
Campanula rotundifolia	cam_rotu
Carex atrata	car_atra
Carex atrofusca	car_atro
Carex bigelowii	car_bige
Carex capillaris	car_capi
Carex dioica	car_dioi
Carex lachenalii	car_lach
Carex saxatilis	car_saxa
Carex vaginata	car_vagi
Carex	carex
Caryophyllaceae	Caryophyllaceae
Cerastium alpinum	cer_alpi
Cerastium fontanum	cer_font
Cetraria	cetraria
Cladonia stellaris	cla_stel
Cladonia	cladonia

Deschampsia cespitosa subsp. cespitosa	des_cesp
Diphasiastrum alpinum	dip_alpi
Empetrum nigrum	emp_nigr
Epilobium davuricum	epi_davu
Epilobium	epilobium
Equisetum arvense	equ_arve
Equisetum palustre	equ_palu
Equisetum	equisetum
Eriophorum angustifolium	eri_angu
Eriophorum vaginatum	eri_vagi
Eriophorum	eriophorum
Euphrasia	euphrasia
Festuca ovina	fes_ovin
Festuca rubra	fes_rubr
Festuca	festuca
Flavocetraria nivalis	fla_niva
fungi	fungi
Galium boreale	gal_bore
Gentiana nivalis	gen_niva
Geranium sylvaticum	ger_sylv
Graminid	graminid
Hieracium alpinum	hie_alpi
Juncus biglumis	jun_bigl
Juncus castaneus	jun_cast
Juniperus communis	jun_comm
Juncus filiformis	jun_fili
Juncus trifidus	jun_trif
Juncus triglumis	jun_trig
Juncus	juncus
Kalmia procumbens	kal_proc
Koenigia islandica	koe_isla
Luzula confusa	luz_conf
Luzula multiflora subsp. multiflora	luz_mult
Luzula spicata	luz_spic
Luzula	luzula
Lysimachia europaea	lys_euro
Melampyrum sylvaticum	mel_sylv
Nardus stricta	nar_stri
Omalotheca norvegica	oma_norv
Omalotheca supina	oma_supi
Omalotheca	omalotheca

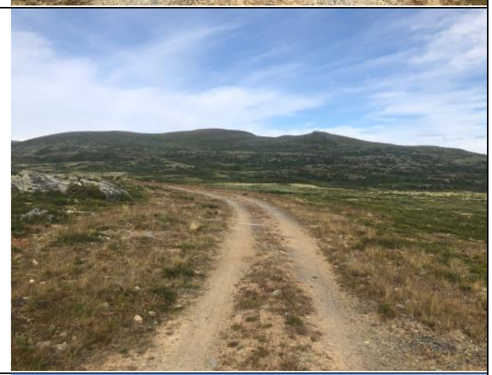
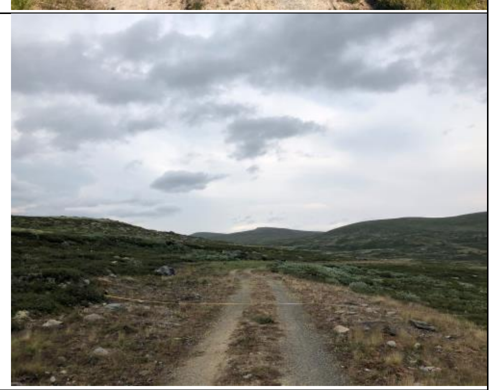
Oxyria digyna	oxy_digy
Parnassia palustris	par_palu
Parmelia	parmelia
Pedicularis lapponica	ped_lapp
Pedicularis palustris	ped_palu
pedicularis	pedicularis
Peltigera	peltigera
Phleum alpinum	phl_alpi
Phyllodoce caerulea	phy_caer
pinguicula vulgaris	pin_vulg
Poa	poa
Poa alpina	poa_alpi
poaceae	poaceae
Potentilla crantzii	pot_cran
Primula	primula
Pyrola grandiflora subsp. norvegica	pyr_gran
pyrola	pyrola
Ranunculus acris subsp. Acris	ran_acri
Rumex acetosa var. Acetosa	rum_acet
Rumex	rumex
Sagina nivalis	sag_niva
Sagina saginoides	sag_sagi
Sagina	sagina
Salix arbuscula	sal_arbu
Salix glauca subsp. Glauca	sal_glau
Salix herbacea	sal_herb
Salix lanata	sal_lana
Salix lapponum	sal_lapp
Salix phylicifolia	sal_phyl
Salix reticulata	sal_reti
Salix	salix
Saussurea alpina	sau_alpi
Saxifraga aizoides	sax_aizo
Saxifraga oppositifolia	sax_oppo
Schedonorus pratensis	sch_prat
Scorzoneroides autumnalis	sco_autu
Scorzoneroides autumnalis var. pratensis	sco_autu_prat
Selaginella selaginoides	sel_sela
Sibbaldia procumbens	sib_proc
Solidago virgaurea	sol_virg
Stellaria borealis	ste_bore

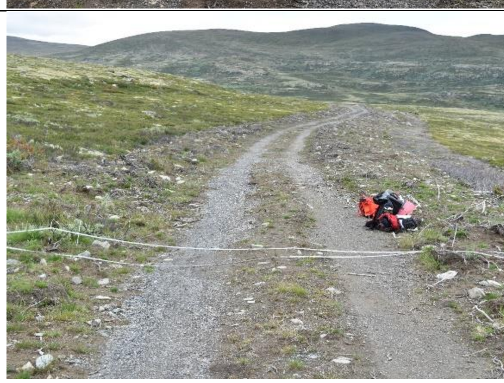
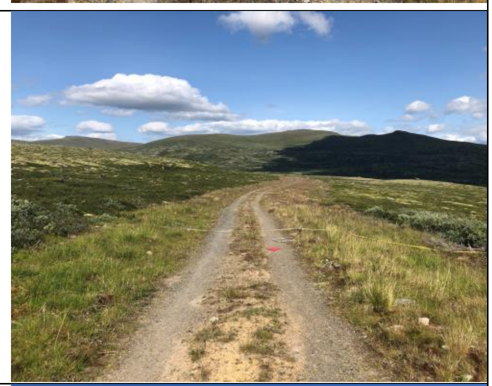
Stellaria	stellaria
Stellaria graminea	ste_gram
Stereocaulon	Stereocaulon
Taraxacum	taraxacum
Thalictrum alpinum	tha_alpi
Thalictrum	thalictrum
Tofieldia pusilla	tof_pusi
Trichophorum cespitosum	tri_cesp
Trichophorum cespitosum subsp. cespitosum	tri_cesp_cesp
Vaccinium myrtillus	vac_myrt
Vaccinium uliginosum	vac_ulig
Vaccinium vitis-idaea	vac_viti
Veronica alpina subsp. alpina	ver_alpi
Veronica	veronica
Viola biflora	vio_bifl
Viscaria alpina	vis_alpi

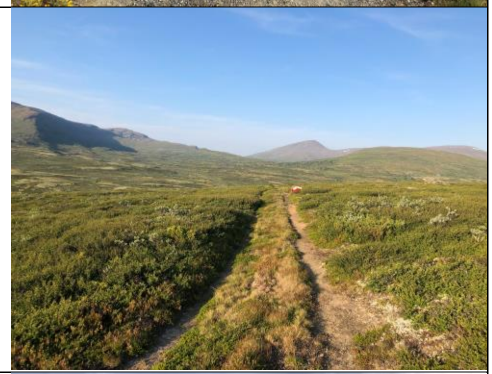
Appendix 2. Pictures of all 15 transects in 2016 and 2021, and reference sites in 2021.

	2016 (Hagen et al., 2017)	**2021**
Transect 1 wetland		
Transect 2 ridge		

Transect 3 willow		
Transect 4 willow		
Transect 5 wetland		
Transect 6 willow		

Transect 7 ridge		
Transect 8 ridge		
Transect 9 willow		
Transect 10 ridge		

Transect 11 ridge		
Transect 12 ridge		
Transect 13 willow		
Transect 14 wetland		

Transect 15 willow		
Reference site 1 willow		
Reference site 2 willow		
Reference site 3 ridge		

Reference site 4 ridge		
Reference site 5 ridge		

Norges miljø- og biovitenskapelige universitet
Noregs miljø- og biovitskapelege universitet
Norwegian University of Life Sciences

Postboks 5003
NO-1432 Ås
Norway