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Restoration of peatland by natural revegetation from indigenous soils along E10 Lofast II, Northern Norway

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Preface

This thesis is the final work of my master's degree in Natural Resource Management at the Norwegian University of Life Science (NMBU). The Norwegian Public Road Administration (Statens vegvesen) supported the project financially.

I would like to thank my supervisors Kari Klanderud at NMBU and Astrid Brekke Skrindo at Norwegian Public Road Administation for help and guidance. Thank you for taking your time to come with us to Lofoten and always answering our questions.

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Norwegian University of Life Sciences

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Marte Dalen Johansen

Abstract

Human degradation of peatlands causes large carbon emissions, loss of biodiversity and reductions in ecosystem services. Ecological restoration is one of the practices trying to mitigate the damages through assisting the recovery of degraded ecosystems. This study was conducted on restored peatland on roadsides along E10 Lofast II, Northern Norway. During roadside restoration, traditional methods of sowing seeds poses a risk of spreading alien species to adjacent areas. The sites in this study were revegetated using indigenous topsoil as the only restoration method. This is the first study of restoration success using this method in peatland ecosystems.

Vegetation analysis was conducted in 108 plots of 1x1m in restored and undisturbed peatland, where the undisturbed peatland was used as the target vegetation type. Additionally, biotic and abiotic environmental factors were recorded for each plot. A Canonical Correspondence Analysis (CCA) tested the effect of restoration on species composition.

The ordination showed differences in species composition between restored and undisturbed sites, indicating an incomplete restoration. Soil moisture, pH, slope and microtopography were recognized as the most important environmental drivers for species composition. Additionally, the ordination and linear regression showed that character peatland species decreased in abundance with increased depth of *Polytrichum* spp. cushion.

The dominance of especially *Eriophorum vaginatum, Polytrichum* spp. and *Carex rostrata* in the restored peatland indicates that the site is still in an early successional stage. This is confirmed by previous studies that show a longer restoration time in peatland than in other ecosystems. The low soil moisture level is most likely limiting the establishment of *Sphagnum* spp. at the restored site. This might explain the poor establishment of other peatland species, as *Sphagnum* spp. is a key genus in forming the self-regulating peatland environment. Suggested improvements for future projects include shorter storage time of topsoil, storage in larger piles, redistributing of soil with respect to the natural microtopography of the area and rewetting strategies.

Sammendrag

Menneskelig ødeleggelse av myr forårsaker store utslipp av karbon, tap av biologisk mangfold og svekker myrens økosystemtjenester. Økologisk restaurering forsøker å redusere skadene ved slike inngrep. Denne studien undersøker restaurert myr langs E10 Lofotens fastlandsforbindelse II. Et problem med tradisjonelle restaureringsmetoder i veikanter er bruken av fremmede frø som øker risikoen for spredning av fremmede, uønskede arter til nærliggende natur. Det restaurerte området i denne studien ble revegetert utelukkende fra stedlige toppmasser. Dette er det første studiet av restaurering ved bruk av denne metoden i myr.

Vegetasjonsanalyser ble utført i 108 ruter av 1x1m i restaurert og urørt myr, hvor urørt myr ble brukt som mål for restaureringen. I tillegg ble abiotiske og biotiske miljøvariabler registrert i hver rute. En «Canonical Correspondence Analysis» (CCA) ble brukt for å teste effekten av restaurering på artssammensetning.

Ordinasjonen viste at det var forskjeller i artssammensetning mellom restaurert og urørt myr, en indikasjon på ufullstendig restaurering. Jordfuktighet, pH, helning og mikrotopografi var de viktigste miljøvariablene som drev artssammensetningen i de ulike sonene. I tillegg viste ordinasjonen og lineær regresjon at det var en negativ sammenheng mellom dybden på bjørnemosetuer (*Polytrichum* spp.) og tilstedeværelsen av typiske myrarter.

Dominansen av spesielt torvull (*Eriophorum vaginatum),* bjørnemose *(Polytrichum* spp.) og flaskestarr (*Carex rostrata)* i restaurert myr indikerer at området fortsatt er i et tidlig stadie av suksesjonen. Dette blir bekreftet av tidligere studier som viser at restaurering av myr tar lang tid sammenlignet med andre økosystemer. Den lave jordfuktigheten begrenser med stor sannsynlighet utbredelsen av torvmoser (*Sphagnum* spp.) i det restaurerte området. Dette kan forklare den dårlige etableringen av typiske myrarter, da torvmoser er viktige for å skape det spesielle selvregulerende miljøet som finnes i myra. Forslag til forbedringer av metoden for fremtidige prosjekter inkluderer kortere lagringstid av toppmasser, lagring i større hauger, tilbakelegging av toppmasene med hensyn til den naturlige mikrotopografien i området og vanningstiltak.

Contents

Terms and definitions

Ecological restoration is defined by the International Society for Ecological Restoration (SER 2004, p. 3) as "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed". This definition assumes that the goal for restoration is a full recovery of the ecosystem. **Rehabilitation** is a term used for restoration with a goal to repair ecosystem processes, but not necessarily to the previous state (SER 2004).

Restoration ecology is the scientific background for ecological restoration based on studies and experiments (van Andel & Aronson 2012). While ecological restoration provides the practical approach, restoration ecology is the theoretical background for these measures.

Revegetation is the phase of ecological restoration involving regeneration of vegetation. The term is commonly used for establishment of a new vegetation cover. In **natural revegetation**, the area is left to recover by natural regrowth (Hagen & Skrindo 2010a).

Peatland is defined by Wieder et al. (2006, p. 1) as "a terrestrial environment where over the long term, on an areal basis, net primary production exceeds organic matter decomposition, leading to the substantial accumulation of deposit rich in incompletely decomposed organic matter, or **peat**."

1. Introduction

1.1 Disturbance and succession

One of the biggest threats against global biodiversity today is human degradation and changes in land use (Convention on Biological Diversity 2010). Road construction is a major anthropogenic disturbance that degrades natural habitats. As roads and their adjacent road verges consume large areas, biodiversity is lost both directly through habitat loss and indirectly caused by fragmentation and isolation of populations (Andrews 1990).

Anthropogenic disturbance must however be distinguished from natural disturbances. Ecological communities are dynamic and heterogeneous, changing in structure over time (Sousa 1984). Disturbance is the major agent causing these changes. Disturbance is defined by Smith and Smith (2006, p. 411) as “any relatively discrete event that disrupts community structures and functions”. Disturbance occurs in all ecosystems and may include both physical events, like flood and fires, and biological disturbance, like grazing and predation (Sousa 1984). Such events facilitate succession. Primary succession is succession on uninhabited sites, while secondary succession follows when a previously habited site has been disturbed (Smith & Smith 2006). During human degradation, natural disturbance processes are mimicked and secondary succession induced, often to a much larger extent than under natural disturbance events.

Ecological restoration is one of the practices trying to reduce the negative impact of anthropogenic disturbance. Although a young academic field, the importance of restoration ecology has never been greater in this fast developing world (Young et al. 2005). The link between ecological restoration and succession is strong, as restoration can be described as a manipulation of natural succession. While successional theory can offer restoration insight in various ecosystem functions, studies of ecological restoration provide practical tests to these theories. Successional studies and studies of ecological restoration operates on different timescales, as successional studies tend to last over longer periods than restoration. Ecological restoration is therefore dependent on a scientific fundament gained from successional research (Walker et al. 2007).

1.2 Restoration of boreal peatlands

Peatlands are important ecosystems on a global scale. The distribution of peatland mosses are closely linked to climatic factors like mean annual precipitation and temperature and peatlands are mainly associated with the northern hemisphere (Wieder 2006). Approximately 87 % of the world’s peatland are found in boreal and subarctic regions (Vitt 2006). Despite covering only a small fraction of the earth’s surface, they have profound ecological impacts. In addition to being habitat for a large number of flora and fauna, peatlands also provide both industrial materials and recreational values. Of current interest is also their role as a provider of ecosystem services. Their ability to reduce floodings and store carbon is some of the important roles (NOU 2013:10). Because of their water storing abilities and slow decomposition rate, peatlands make up a huge carbon and water reservoir. Northern peatlands alone store about a quarter of the world's total soil carbon (Roulet et al. 2007). A key genus in the fixation of carbon is *Sphagnum* mosses. Given that they can cover the entire peatland surface, they contribute with most of the carbon fixation in these ecosystems (Gunnarsson 2005). The ability of *Sphagnum* mosses to form peatlands is closely linked to their anatomically traits, and the link between *Sphagnum* traits and ecosystem functions in peatlands is strong (Rydin et al. 2006). The physiology of *Sphagnum* makes them more resilient to decomposition than other plants. Firstly, they tolerate and create an acidic, humid and nutrient and oxygen poor environment. Specialized hyaline cells can absorb and store water against suction pressure, limiting the chances of desiccation. In addition, chemical properties of the cell walls create an acidic environment (van Breemen 1995). Secondly, they are resistant to decay, resulting in an accumulation of dead organic material known as peat (Wieder 2006). Lastly, the genus has a large number of species specialized to different parts of the peatland, allowing them to colonize large areas (Rydin et al. 2006). This gives these mosses a unique ability to positively feedback themselves by using their own dead tissue to create a desirable environment, resulting in the dominance in peatland ecosystems.

However, peatlands are suffering under modern development. Over the past hundred years, approximately half of the world's wetlands have been degraded (IUCN 2000) and in Europe it is believed that less than half of the natural peatlands are left intact (Joosten & Clarke 2002). In addition, the main peatland regions are expected to undergo future climate changes, resulting in higher mean temperatures and precipitation (Roulet et al. 2007). This will add to the effect of human induced degradation that peatlands already experience.

Due to its specialized hydrological characteristics, peatlands are especially vulnerable to changes in the hydrological regime. In ecosystems where peat is accumulating, the system consists of two layers. The layers are not distinct, but rather a transition from one environment to another (Rydin & Jeglum 2013). The upper layer, the acrotelm, is aerobic. Despite being waterlogged, both conductivity of water and decay is high in this layer. Beneath this layer is the catotelm, an anaerobic layer with low conductivity and low rate of decay. Disturbances to this specialized system might be hard to restore and for this reason it is expected that peatland vegetation will have a slower recovery than other vegetation types (Grootjans et al. 2012). Restoration successes on peatland have however been documented. Lavoie et al. (2003) found that it was possible to restore species composition and other studies show successful restoration of carbon accumulation (Tuittila et al. 1999; Waddingston et al. 2001).

To evaluate the success of restoration, a set of goals and a reference system is necessary (van Andel & Aronson 2012). Rochefort (2000) suggests that the main objective for peatland restoration in the northern hemisphere should be to reestablish a plant community of *Sphagnum* and other brown mosses. Associated with this goal is the restoration of the hydrological layers which are characteristic of active peatlands.

1.3 Restoration at E10 Lofast II

This study focuses on ecological restoration of peatlands affected by the road project E10 Lofast II. The construction of E10 Lofast II was a controversial project due to the considerable impacts on the natural environment in the area. The road passes through previously unexploited areas and borders Møysalen National Park (Kongsbakk & Skrindo 2009). Although the final alignment route avoided some of the most valuable areas, large negative environmental impacts were still expected. In an attempt to reduce these damages, much attention was given to compensatory methods such as restoration of the road verges. Revegetation from indigenous soils was chosen as the restoration method. This is the largest project of its kind to date in Europe (Solvoll et al. 2014).

Restoration from indigenous soils focuses on using the upper part of the soil profile, the topsoil, as a basis for revegetation. This part of the soil has the highest portion of organic matter, propagules and microfauna, and is the best basis for natural revegetation of a degraded ecosystem (Skrindo 2005). Revegetation from topsoil is hence based on germination from the propagule bank. The composition of species on the restored site will depend largely on the species present in the transferred soil in addition to dispersal from adjacent areas.

Invasion of alien species is recognized as a major threat to global biodiversity (Gederaas et al. 2012). Traditional revegetation practice tends to use fast growing, not necessarily indigenous species as the main source of plant material. The Nature Conservation Act's (2009) concerns the prevention of introducing alien species to Norwegian nature. In contrast to traditional revegetation methods, revegetation from indigenous soils does not introduce any non-native organisms and risk of spread of alien species is minimized, fulfilling the act's directions.

Much research has been conducted on restoration from indigenous topsoil in other ecosystems, for example shrub- and woodlands (Fowler et al. 2015; Holmes 2001; Rockich et al. 2000; Skrindo & Pedersen 2004), meadows (Vécrin & Muller 2003) and arid grasslands (Golos & Dixon 2014). In North America restoration of mined peatlands has been extensively studied (Girard et al. 2002; González & Rochefort 2014; Lavoie et al. 2005; Price & Whitehead 2001; Price et al. 2003). In Europe both mined (Lanta et al. 2004; Soro et al. 1999; Triisberg et al. 2011; Tuittila et al. 2000) and drained peatlands (Haapaletho et al. 2011; Jauhainen et al. 2002) have been the object of several studies, but research on the restoration of peatlands using indigenous topsoil has not yet been conducted. Many questions were therefore raised prior to the project on how the peat would withstand the treatment and how the species would regenerate in this type of soil. Of special concern was how the hydrological regime would be affected (Kongsbakk & Skrindo 2009).

Ecological restoration is a young academic field, that dates back only to the late 1980's (Young et al. 2005). In Norway, early practice was limited to simple practical measures, but during the 1990's the number of scientific projects began to rise (Hagen & Skrindo 2010b). Evaluating the success of a restoration project is a key step towards a science-based management. Previous studies along E10 Lofast II, showed good plant establishment during the first years after restoration, but with large changes in the species composition (Kongsbakk & Skrindo 2009; Nystad 2006).

This study investigates the success of restoration almost one decade after the project was finalized through analyses of vegetation and possible environmental drivers. I considered restoration successful if plant community properties in the restored plots were similar to control plots and investigate restoration success by asking; (1) whether restoration changes plant community properties in the investigated peatlands; (2) what are the main environmental factors driving restoration; and (3) whether there is a need for improving the method in future restoration management.

2. Materials and methods

2.1 Study area

The study site is located along the road stretch E10 Lofast II in Hadsel, Lødingen and Kvæfjord municipalities in Nordland and Troms Counties, Northern Norway (68° N, 15° S) (Fig. 1). The road stretch is in total 29.5 km long, with approximately 10 km going through four tunnels. The road stretch is a continuation of Lofast I which was opened for the public in 1998. When Lofast II was opened in 2007 the road was connected to the existing mainland road (Kongsbakk, E. & Skrindo, A.B. 2009). The study site is situated in a typical alpine coastal landscape type with fjords, mountain peaks, valleys and steep mountain sides (Kongsbakk & Skrindo 2009). Located in the oceanic part of the north boreal vegetation region, the area is characterized by high annual precipitation, low summer temperatures and fairly short growing seasons (150-160 days with an average temperature above 5°) (Moen et al. 1998) (Table 1). The peatlands in the area are poor to intermediate fens dominated by *Sphagnum* mosses, graminoides, heather and a sparse cover of herbs. In addition, humid alpine birch forest dominated by ferns were common to the area.

Table 1. The 18 transect with their site location, UTM coordinates, altitude mean annual temperature and precipitation and bedrock. Mean annual temperatures and mean annual precipitation are provided by Meterologisk institutt from the weather station closest to the respective site (eklima.met.no). Geological data are provided by NGU (2015).*Stokmarknes LH- Skagen weatherstation, **Borkenes weatherstation, ***Kanstadbotn VI weatherstation, ****Raftsund-Ulvøy weatherstation.

Site	*Transects*	*Coordinate (UTM zone 33)*	*Altitude (m asl)*	*Mean annual temperature (°C)*	*Mean annual precipitation (mm)*	*Bedrock*
Oceanic						
Storåa	1 - 4	510434 7594248	40	5.6*	1925****	Mangerite
Ingelsfjordeidet	5-6	517015 7594889	17	4.0**	1925****	Mangerite
Inland						
Sørdalen	7-18	530602 7600668 – 531237 7601924	40-50	4.0**	2015***	Banded gneiss

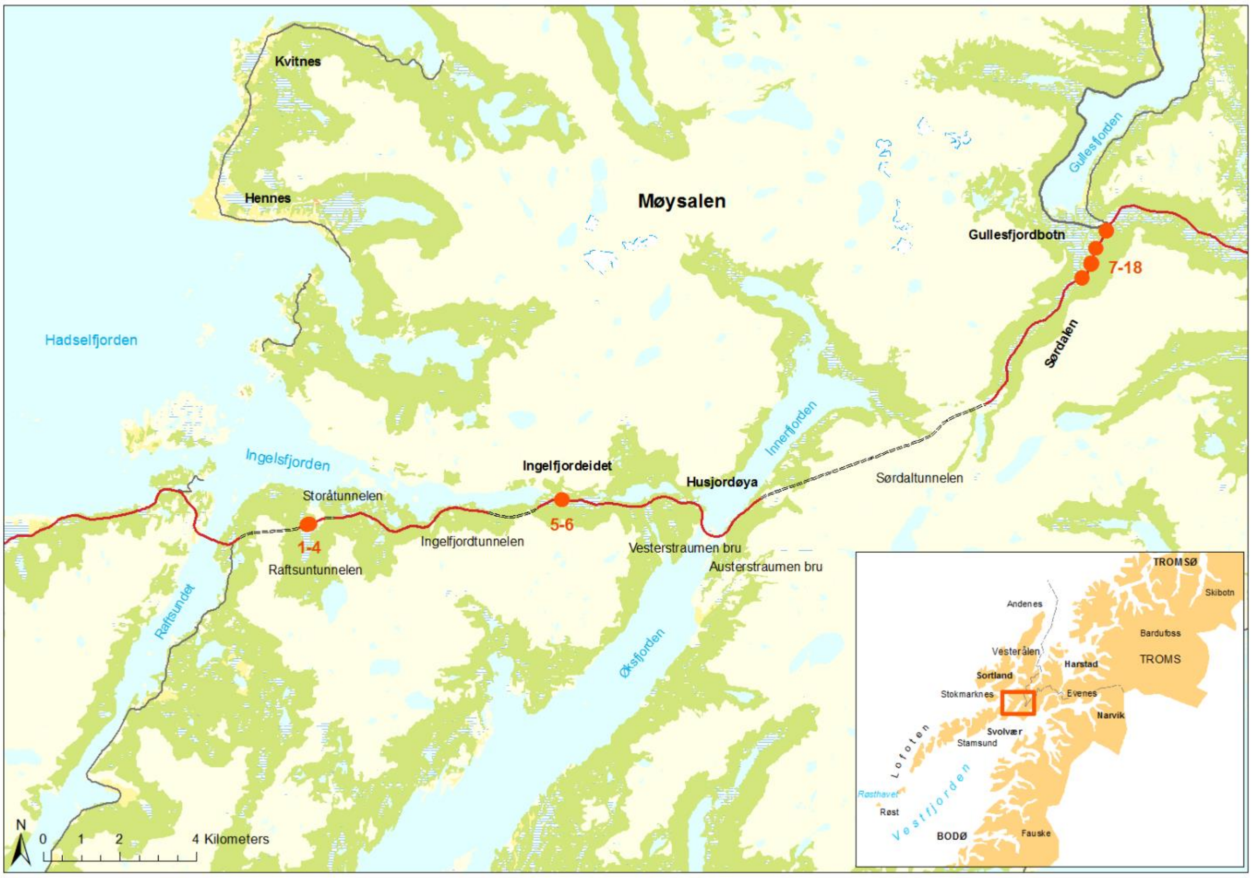

Fig. 1 The 18 transect located along E10 Lofast II in Nordland and Troms County, Northern Norway. The transects are clustered in three groups located in Storåa (1-4), Ingelsfjordeidet (5-6) and Sørdalen (7-18). Maps were created using ArcMap version 10.2.2 (ESRI 2014) with N50 geographical data from Norwegian map authorities (Kartverket 2014).

2.2 Revegetation from indigenous topsoils

Revegetation from indigenous soils was used as the only restoration method during construction of E10 Lofast II. Using this method, topsoil is stockpiled and stored in piles of 2-3 m during construction before being redistributed on the degraded site. Topsoil was defined as the upper 30 cm of the soil profile. About 10 - 20 cm of this was later redistributed. During construction some focus points were carried out; (1) the sub- and topsoil were kept separated, (2) a serrated edge was created between the encroachment and the unaffected areas, making the contact surface as large as possible to assist dispersal of species from natural vegetation and (3) the topsoil was redistributed loosely creating microhabitats and good aeration (Fig. 2). After redistribution of the topsoil, the site was left to develop by the forces of natural succession with no further assistance (Kongsbakk & Skrindo 2009).

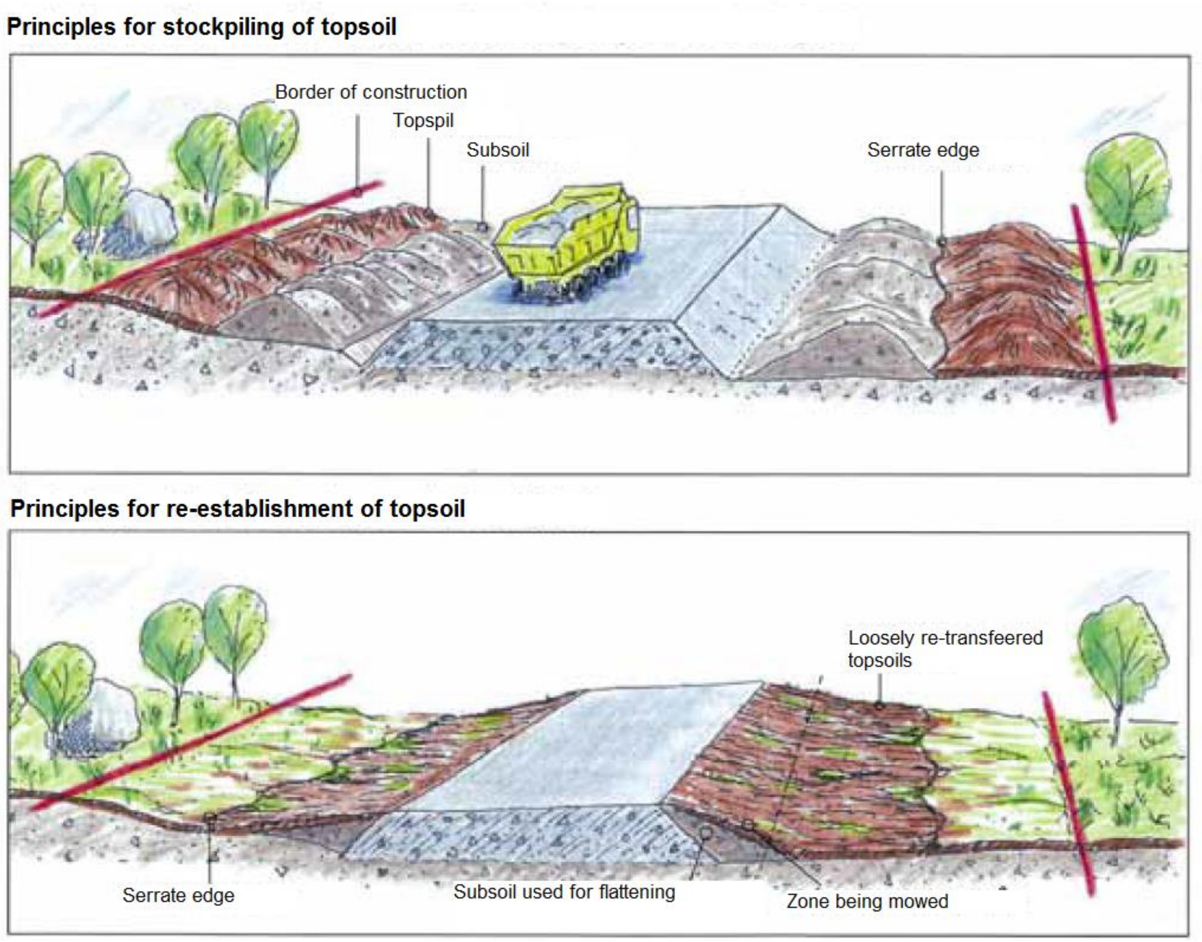

Fig. 2 Principles for restoration by topsoil of indigenous soils. Illustrations by Kongsbakk in Kongsbakk and Skrindo (2009).

2.3 Study design

Data were collected during July 2014. Eighteen transects were selected along the road stretch from Sørdalen to Storåa (Fig. 1) following a set of criteria. Each transect should (1) be a continuous line from the road, uninterrupted by any open water bodies, streams or unnatural roughness caused by road construction, (2) have control plots of intact peat land, (3) be located within natural ground slope (not in slope created through construction work). The transects were divided into zones based on their treatment during restoration. Two main zones distinguish between disturbed and undisturbed peatland; "restored" and "control", where control plots represented the undisturbed peatland and target vegetation type. The "restored" zone was further divided into a "road verge" zone in the innermost five meters of the transect. This area is affected by mowing and traffic. From each zone one pair of 1 x 1 m plots was randomly chosen. This gave a total of 108 plots (36 road verge, 36 restored and 36 control plots) (Fig. 3)

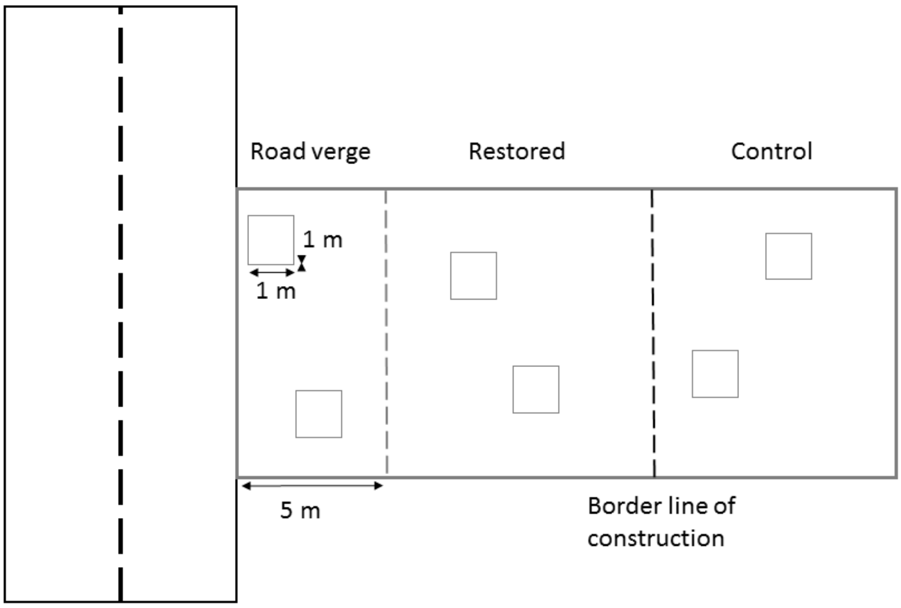

Fig. 3. Each transect had three zones: road verge, restored and control. The road verge covers the area one meter from the road and five meter inwards. The restored zone covers the area five meters from the road to the borderline of construction. The control zone consist of undisturbed peatland. Two random plots of 1x1m were placed in each zone in each transect. This gave a total of 108 plots.

2.4 Species and environmental data

To study differences in vegetation, I recorded species richness and percent cover of single species for each plot. All vascular plants were identified to species level, while some bryophytes and lichens were identified to family or genus. From the genus *Polytrichum* spp. most of the recorded specimens were *Polytrichum commune,* but also *Polytrichum strictum*, *Polytrichastrum alpinum, Polytrichum juniperinum* and *Polytrichum piliferum* could have been recorded. For *Sphagnum* spp. no species were specified. All vascular plant names follow nomenclature by Lid et al. (2005), bryophyte names by Hallingbäck and Holmåsen (1981) and lichen names by Holien and Tønsberg (2008). Overlapping vegetation were taken into account, meaning a total cover above 100% was possible for each plot. Total cover of bryophytes and vascular plants was recorded for each plot. In addition, cover of stones and gravel, bare soil and litter was recorded for each plot. To predict productivity, I recorded vegetation height for each plot. The individual closest to the plot corner was measured from ground to shoot. Where the individual was branched in the top, the uppermost bud was measured. An individual from each plot corner was recorded, making an average for each plot.

Potential important environmental variables affecting species composition was recorded in each plot. Microtopography was recorded on a relative scale; (1) flat, (2) uneven and (3) very uneven. Slope and aspect were recorded as degrees with a Silva Expedition 15 compass. During surveys prior to the fieldwork we observed a higher coverage of *Polytrichum* spp. in restored areas than undisturbed, giving reasons to investigate the influence of this genus on the plant community. For *Polytrichum* spp. I therefore measured depth of cushions from ground to the top of the gametophyte (in cases where the sporophyte had developed, it was not measured). Four measurements were taken as close to the plot corners as possible, making an average for each plot. The transects in Storåa and Ingelsfjordeidet (1-4 and 5-6) were characterized as oceanic, while the transects in Sørdalen (7-18) were characterized as interior due to their location in the mouth and head of the fjord respectively (Fig. 1). Time since restoration differed by one year with approximately half of the transects being restored in 2005 and the rest in 2006.

Soil samples were taken from all plots to examine differences in pH and soil moisture between the control, restored and road verge zone. The samples were taken at approximately 30 cm depth from each plot during two days of similar weather conditions. The samples were stored frozen to avoid further biological activity.

2.4.1 Soil analysis

The soil samples were thawed in a fridge at 4°C overnight the day before analysis. The samples were placed in an aluminum box and weighed using a Sartorius ED Analytical Balance ED224S weight with SartoConnect software. The samples were then dried at 105 °C for 24 hours and cooled in a desiccator before being re-weighed, in order to calculate volumetric soil moisture (%). To measure pH, 10 ml of dry soil were taken from the dried samples and added to 25 mL of distilled water and mixed thoroughly. The samples were stored overnight (approximately 24 hours) before pH was measured using a WTW Series inoLab pH/Cond 720 pH-meter.

2.5 Data management and statistical analysis

Data management was carried out using Excel (2013). A group of species, named character species, were defined based on the collected data. This group includes the most important species of vascular plants and bryophytes found in the control plots and represented the typical peatland vegetation in some of the analysis. Character species were defined as species with a frequency greater than 27.7 % (10 out of 36 plots) in control plots. Community properties were presented visually in boxplots to show differences between the three zones. Medians were calculated for each explanatory variable. The zones were used as the response variable, while cover of bare soil, soil moisture content, vegetation height, character species cover, *Sphagnum* spp. cover and *Polytrichum* spp. cover were used as explanatory variables.

In order to investigate variations in the species data for further analysis, a Detrended Correspondence Analysis (DCA) was run to check for axis lengths. The DCA showed long gradient lengths (first axis = 4.698), indicating a large variation in the species data. A Canonical Correspondence Analysis (CCA) was chosen based on the first DCA axis length (> 4.0), as recommended by Lepš and Šmilauer (2003). Transects were used as conditioning variable to control for variation between transects. A one- way analysis of variance (ANOVA) was run on the CCA to test the effect of restoration on species composition. The species with a total cover above 150 % for all plots were plotted to give a visual impression of the distribution of the most dominant species. Soil moisture, depth of *Polytrichum* cushions, slope, pH, microtopography, aspect, oceanity and time since restoration were used as explanatory variables and fitted to the ordination using the env.fit function in the vegan package (Oksanen et al. 2015). Since *Polytrichum* was used as an explanatory variable in this analysis, it was taken out of the species data when environmental variables were fitted to avoid false correlation. The variable time since restoration was also taken out of the data due to correlation with oceanity. The remaining environmental gradients were scaled to equal relative units. To find the variables with significant effect, I used backward selection based on p- values. The backward selection were performed by fitting all non-correlating variables to the ordination and removing the variable with the highest p- value before fitting the variables again. This was done until all variables were significant. Based on backward selection aspect and oceanity were removed. All significant variables were plotted in the ordination plot.

The frequency of species that occurred in more than 41.6% (15 out of 36 plots) of the plots in each of the three zones was presented in a barplot, with the frequency of species as the explanatory variable and zone as the response variable.

In order to investigate species diversity in the different zones, I performed a one-way analysis of variance (ANOVA) using species number as response variable and zones as explanatory variable. The response variable was tested for normal distribution before the analysis. To test for effect of cushion depth of *Polytrichum* on cover of character species I performed a simple linear regression, using total cover of character species as response variable and depth of *Polytrichum* as explanatory variable. The response variable was squared root transformed to achieve normal distribution.

All analyses were performed in R version 3.2.1 (The R foundation for Statistical Computing Platform 2014) using RStudio version 0.98.1102 (RStudio Inc., Boston, Massachusetts, USA). The package vegan (Oksanen et al. 2015) was used for the multivariate analysis and plotrix package (Lemon 2014) was used for ordination graphics.

3. Results

Boxplots for community properties showed variance between the zones (Fig. 4). The cover of bare soil was higher in the restored zone, although it varied greatly between plots (Fig. 4 a). Soil moisture also varied greatly within the restored and road verge zone, while the control zone generally was wetter with a median of 82.2 % (n= 36) volumetric soil moisture (Fig. 4 b). Short vegetation dominated in the control zone, while the restored and road verge zone consisted of taller vegetation (Fig. 4 c). The bryophyte communities in the zones different greatly. An decrease in character species was apparent with distance from the control zone, with the road verges and restored zone having a considerable lower coverage of these species than control plots (Fig. 4 d). *Sphagnum* spp. had a large coverage in the control plots (median of 98.5 % (n= 36) cover), but were almost absent from restored and road verge plots (Fig. 4 e). *Polytrichum* spp. was abundant in the road verge and restored plots, although it varied between plots (Fig. 4 f).

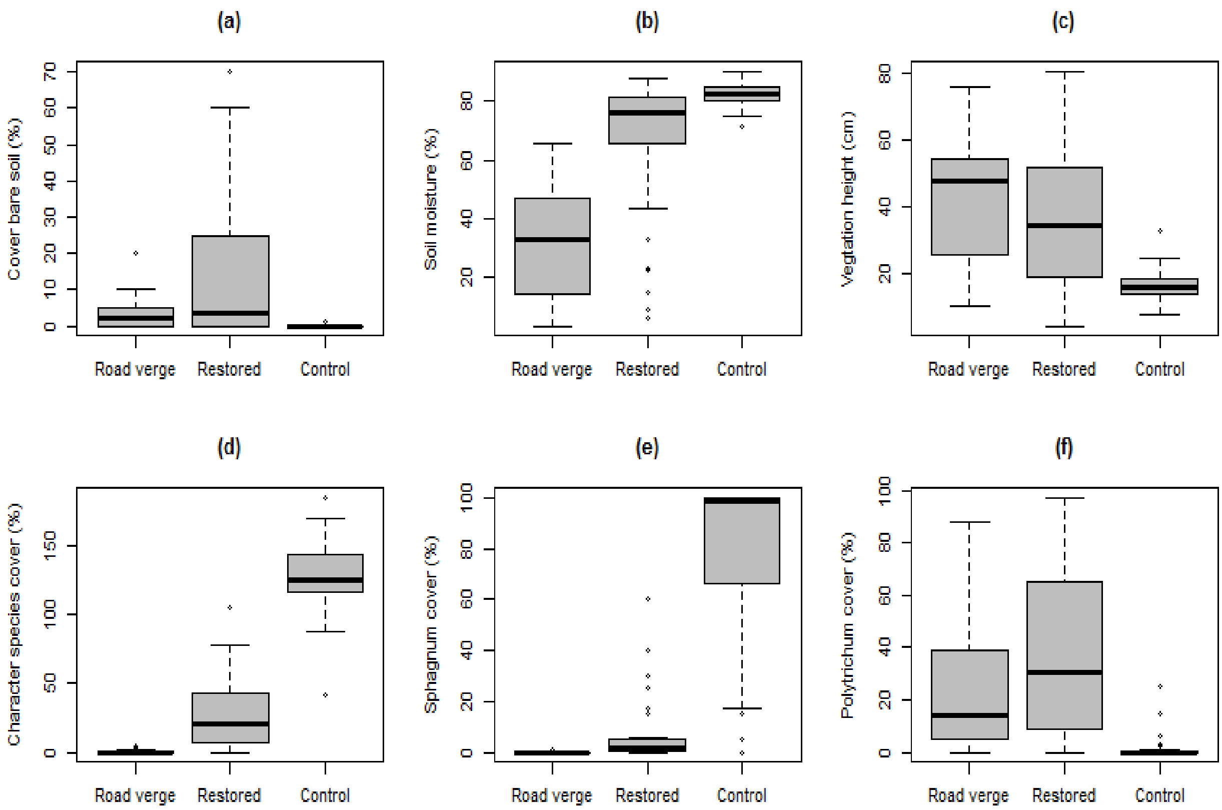

Fig. 4. Boxplots showing differences between the three zones for (a) cover of bare soil (%), (b) soil moisture (%) (c) vegetation height (cm), (d) character species cover (%), (e) Sphagnum cover (%) and (f) Polytrichum cover (%). Character species were defined as species occurring in control plots with a frequency greater than 21.7 % (10 out of 36 plots) which included *Andromeda polifolia, Calluna vulgaris, Carex pauciflora, Dicranum spp., Drosera rotundifolia, Empetrum nigrum, Eriophorum angustifolium, Eriophorum vaginatum, Oxycoccus microcarpus, Pleurozium schreberi, Ptilidum spp., Rubus chamaemorus, Sphagnum spp., Trichophorum cespitosum* and *Vaccinium uliginosum.*

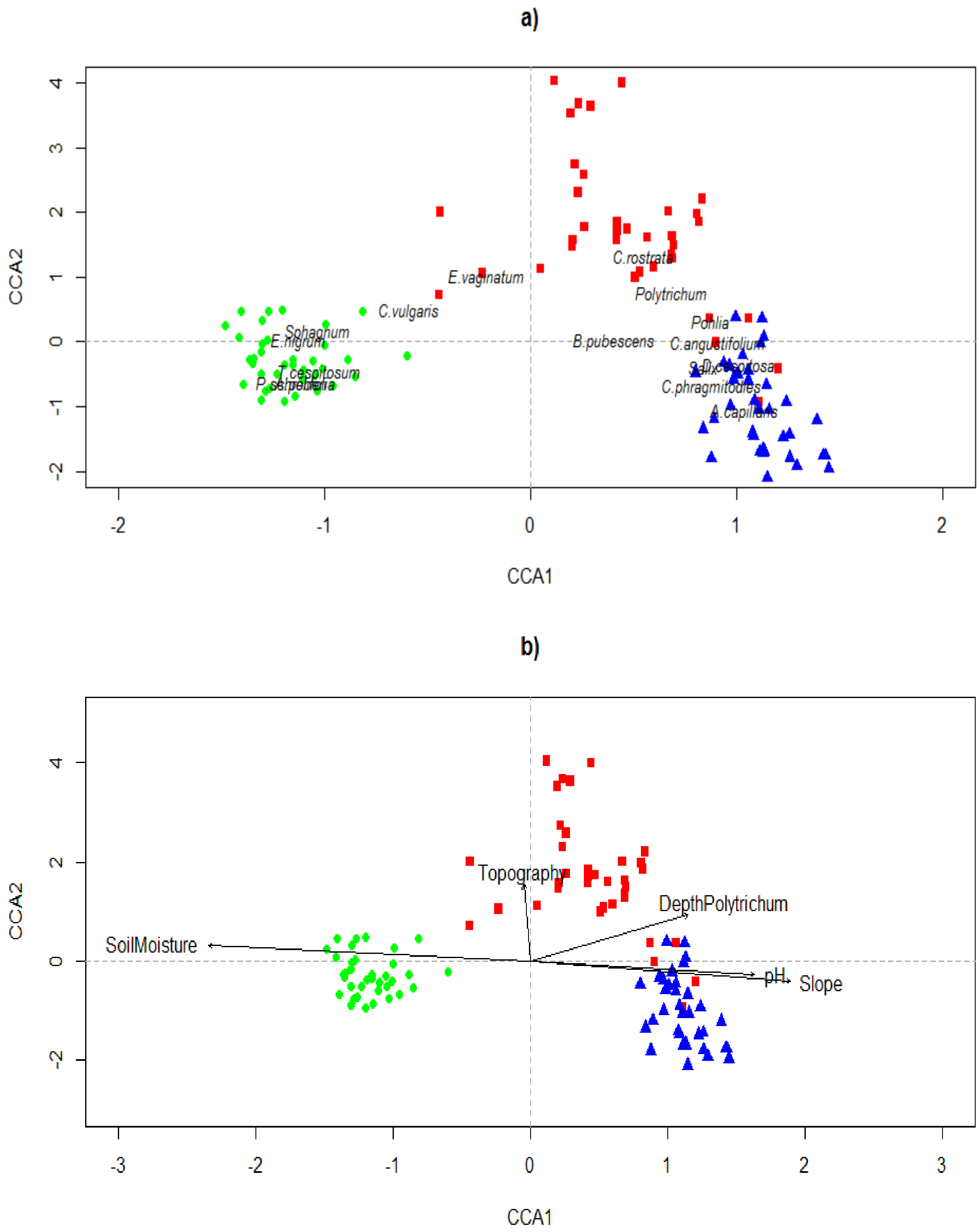

Fig. 5. CCA plots for the 108 plots distributed across 18 transects along E10 Lofast II. Blue (Δ), red (□) and green (O) colors represents road verge, restored and control plots respectably. a) The most abundant species and their associated with the different zones. A total of 88 species were identified, but only species with a total cover above 150 % are shown. From the left: *Pleurozium schreberi, Andromeda polifolia, Empetrum nigrum, Sphagnum* spp., *Trichophorum cespitosa, Calluna vulgaris, Eriophorum vaginatum, Betula pubescens, Carex rostrata, Polytrichum* spp., *Calamagrostis phragmitoides, Chamerion angustifolium, Salix* spp., *Pohlia* spp., *Deschampsia cespitosa and Agrostis capillaris.* b) The environmental variables and their relation to the species data. The direction of the arrows show which axis the variable is most correlated with, and the length indicate the strength of the correlation. Environmental variables were scaled to equal relative units before being fitted to the model.

A total of 88 species or taxa was recorded for all zones. No significant difference was found in number of species between the different zones (f^{ANOVA}= 1.735, p^{ANOVA} = 0.181) and the species number averaged around 10 species for each zone. However, the difference in species composition between the three zones were large. The CCA model incorporating the different treatments in the three zones explained approximately 15 % of the variation in species composition (f^{ANOVA} = 10.22, p^{ANOVA} >0.001). Road verge and control plots show a clustered distribution in the ordination and the plots within these zones have a high resemblance in their species composition. The restored plots are more scattered and species composition differ greatly between plots (Fig. 5 a).

Soil moisture, depth of *Polytrichum* cushions, slope, pH and microtopography were significantly correlated with species distribution in the CCA model, while aspect and oceanity were not (Table 2). The first ordination axis has a strong positive correlation with pH and slope and a strong negative correlation with soil moisture, indicating these variables as the most important drivers for species composition (Table 2, Fig 5 b). Hence, the plots in the restored zone that had the best restored species composition were moister and with a lower pH. Flat sites also tended to restore better than sloped ones. Microtopography were strongly positively correlated with the second axis, while cushion depth of *Polytrichum* was equally positively correlated with both axes (Table 2, Fig 5 b). High microtopography resulted in poorer restoration in the restored plots. *Polytrichum* cushions were deeper in road verge and restored plots.

Table 2. Correlation values with the two first axis, r-squared and significance values for the fitted environmental variables before backward selection.

	CCA1	CCA2	r^2	p - value
Oceanity	-0.26959	0.96298	0.0115	0.671
Aspect	-0.95793	-0.28699	0.0243	0.430
Soil moisture	-0.99475	0.10235	0.5663	0.001 ***
pH	0.99872	-0.05059	0.3697	0.001 ***
Polytrichum cushion depth	0.79331	0.60881	0.1243	0.010 **
Slope	0.99671	-0.08108	0.2891	0.001 ***
Microtopography	-0.02993	0.99955	0.3582	0.001 ***

Based on percent cover of the species recorded, *Sphagnum* spp., *Empetrum nigrum, Pleurozium schreberi, Andromeda polifolia, Calluna vulgaris* and *Trichophorum cespitosum* were the most abundant species in the control zone. *Carex rostrata, Eriophorum vaginatum* and *Polytrichum* spp. were characteristic for the restored zone. *Pohlia* spp., *Calamagrostis phragmitoides, Chamerion angustifolium, Betula pubescens* and *Salix* spp. were abundant in both road verge and restored plots, while *Agrostis capillaris* and *Deschampsia cespitosa* were more abundant in the road verges (Fig. 5 a).

Some species occurred with a low cover, but a high frequency. In the control plots, some typical peatland shrubs and herbs like *Vaccinium uliginosum, Oxycoccus microcarpus, Drosera rotundifolia* and *Rubus chamaemorus* were frequent. Few species recorded in the control zone had been able to establish with high frequency in the restored and road verge zone. The road verge and restored zone were more similar in species composition, but differed greatly from the control zone (Fig. 6). *E. vaginatum* and *Sphagnum* were the only two recorded species with a high frequency in both control and restored zone (Fig. 6). *Polytrichum* spp. was found in approximately 97 % of the plots in the restored zone, and were also frequent in the road verges. *C. angustifolium* and *D. cespitosa* seems to be species associated with road verges, having a higher frequency in this zone than other zones.

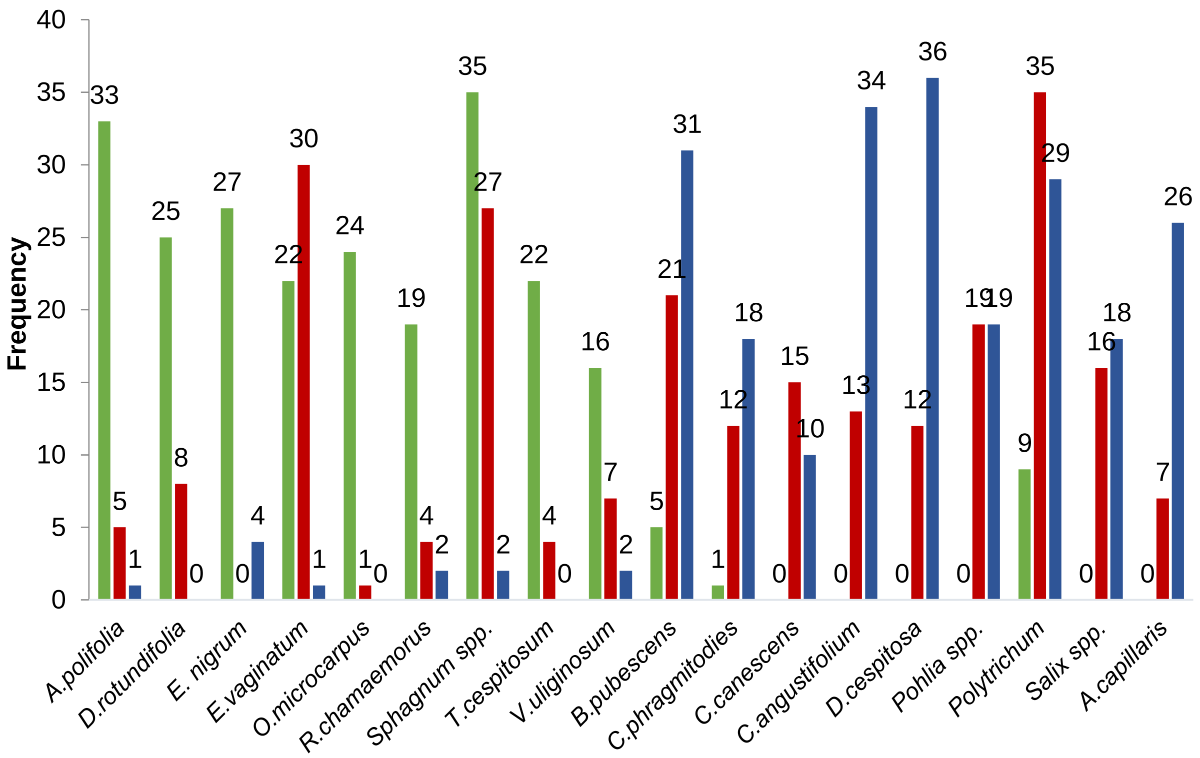

Fig. 6. Distribution of the species with a frequency greater than 41.6 % (15 out of 36 plots). Green bars indicate control plots, red bars restored plots and blue bars road verge plots.

The depth of *Polytrichum* cushions was correlated with species composition in road verge and restored plots (Fig. 5 b), indicating a negative relationship with the survival of peatland species during restoration. The depth of *Polytrichum* cushions tested against cover of character species in the linear regression showed a significant negative effect (f =10.04, r^2 = 0.205, p =0.003) (Fig. 7).

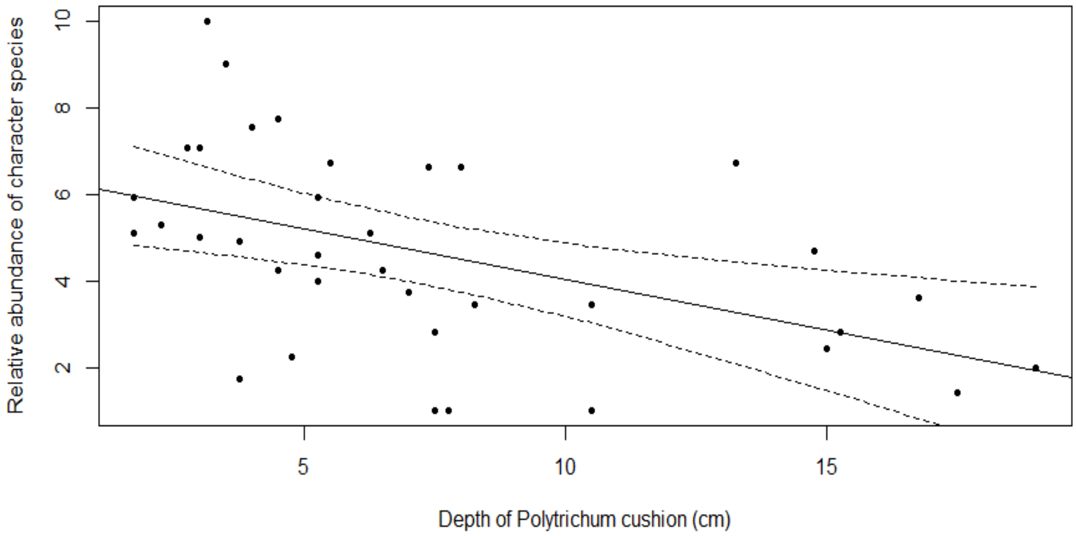

Fig. 7. The relationship between abundance of character species and the depth of *Polytrichum* cushions in the restored zone. Character species were defined as species occurring in control plots having a frequency greater than 21.7 % (10 out of 36 plots) which included *Andromeda polifolia, Calluna vulgaris, Carex pauciflora, Dicranum spp., Drosera rotundifolia, Empetrum nigrum, Eriophorum angustifolium, Eriophorum vaginatum, Oxycoccus microcarpus, Pleurozium schreberi, Ptilidum spp., Rubus chamaemorus, Sphagnum spp., Trichophorum cespitosum* and *Vaccinium uliginosum.*

4. Discussion

4.1 Restoration success

Restoration of peatland after road construction along E10 Lofast II resulted in a vegetation that differed greatly in community properties from undisturbed peatland. Firstly, the restored zone generally had more bare soil, lower soil moisture and taller vegetation than the control zone, indicating that the restored peatland lack key characteristics. Secondly, there was a sparse cover of typical peatland bryophytes, especially *Sphagnum* spp., in the restored zone. This must be considered a restoration failure if restoration success on peatlands is evaluated from the cover of this genus as proposed by Rochefort (2000). Lastly, the ordinations showed that the three zones had different plant communities, indicating that the goal to restore species composition has not been reached.

This is the first study of natural revegetation from indigenous topsoil in a peatland ecosystem, which gives little literature to compare with. However, much research has been conducted in restored mined peatlands. These restoration projects have many similarities, which makes it possible to compare the ecological processes. In both methods, peat from the acrotelm is removed from the site. Additionally, none of the projects involves primarily drained peatlands, except for some drainage caused by the mechanical work. The major difference between the projects is how the acrotelm is used after removal. In this study the acrotelm was returned to the site, whereas in mined peatlands the acrotelm is permanently removed.

Studies on mined peatlands confirm that their restoration takes more than a decade. Studies based on natural regeneration of mined peatlands showed that even fifty years after abandonment, the ecosystem had not recovered to its original state (Girard et al. 2002; Soro et al. 1999), and Girard et al. (2002) suggested that it might require as much as a century to do so. Konvalinková and Prach (2010) however, found high resemblance between the undisturbed and restored peatlands, but the sites had all been abandoned for more than fifty years and some even as long a century.

In studies where mitigating measures were put in place, a faster recovery would be expected. However studies conducted one to two decades after restoration showed no goal achievement (González & Rochefort 2014; Haapaletho et al. 2011). An exception is found in restoration of peatlands that have been drained for forestry, where species composition had recovered well two decades after restoration (Jauhainen et al. 2002). The fast recovery on drained peatland compared to mined is partly explained by the presence of vegetative parts and dormant buds that react to changes in moisture conditions. Regardless, the slow recovery of peatland is clear from previous studies and has been confirmed in this study.

4.2 Species composition

Species composition differed between the three zones. The majority of species in the restored zone could be characterized as pioneer plants, such as *Chamerion angustifolium, Betula pubescens, Salix* spp., *Polytrichum* spp., *Eriophorum vaginatum* and *Carex rostrata*, which indicates that succession is still in an early phase. Three species characterized the restored plots by their abundance; *E. vaginatum, C. rostrata* and *Polytrichum* spp. These are all typical early successional species favored by regular disturbance. Species from the genus *Polytrichum* are considered pioneer mosses in other ecosystems. *Polytrichum commune* is dominant during early succession in heathlands (Clément & Touffet 1990; Corradini & Clément 1999; Maltby et al. 1990) and several *Polytrichum* spp. are pioneer species in boreal coniferous forests with frequent logging (Parker et al. 1997). Studies on abandoned mined peatlands showed that *Polytrichum strictum* was the first species to colonize bare peat (González et al. 2013; González & Rochefort 2014; Groeneveld et al. 2007; Lavoie et al. 2003; Lavoie et al. 2005). This is probably due to the high dispersal potential of *P. strictum*, which has small spores that are carried by wind over long distances (Campbell et al. 2003). An extensive cover of *Polytrichum* could also indicate a failure in the recovery of hydrological properties. This genus can withstand relatively dry conditions better than many other bryophytes due to leaves adapted to store water under dry conditions (Bayfield 1973).

E. vaginatum already characterized the restored peatland a few years after restoration (Kongsbakk & Skrindo 2009), and has continued to expand. The fast establishment of *E. vaginatum* could be due to a higher generation from seeds (Salonen et al. 1992), as peatland species are often limited by vegetative dispersal (Jauhainen 1998). Several studies demonstrate the role of *E. vaginatum* as an early colonizer in restored peatlands, as it is opportunistic and grow vigorously under the newly created environment (Jauhainen et al. 2002; Lavoie et al. 2005; Tuittila et al. 2007). Both *E. vaginatum* and *C. rostrata* have some physiological traits favorable for colonizing disturbed peatlands. Due to their deep rooting system they can tolerate a wide range of moisture conditions (Visser et al. 2000; Wein 1973), making them less dependent on stable moist conditions than other peatland species.

Whether the species have germinated from the redistributed topsoil or dispersed from the surroundings is hard to say. The control plots consisted of typical poor peatland species demanding little nutrients, such as *Andromeda polifolia, Empetrum nigrum, Rubus chamaemorus* and *Trichophorum cespitosum*. These species are present in the restored plots, but not abundant. Some species were frequent in both restored and control plots, such as *E. vaginatum* and *Sphagnum*. However, many of the species in the restored plots were not found in the control plots and must have been dispersed from elsewhere. Some of these species could have come from dormant seeds in the topsoil seed bank and germinated when the environment changed. This is especially valid for the grass and shrub species found in the restored plots, which require a drier habitat. Most of the germination in boreal peatlands come from roots or buried propagules rather than seeds, and vegetative clonal growth from rhizomes or stolons is considered the most important reproductive strategy (Jauhainen 1998). Assuming that establishment of peatland species is largely dependent on vegetative growth suggests that regeneration will be slow, which might be one of the explanation to the poor regeneration of peatland species after restoration.

The road verges differed greatly from natural peatland and were dominated by grasses like *Deschampsia cespitosa, Agrostis capillaris* and *Calamagrostis phragmitoides*. Their specialized anatomy with several apical meristems divided by nodes makes them resilient to mowing, and the species composition in the road verges is therefore strongly affected by this. The effects of cutting in road verges are generally seen as an increase in graminoids together with a decrease in shrubs and trees (Parr & Way 1988). The road verges were constructed differently from the rest of the site in order to create gentle slopes from the road to adjacent areas, and drainage was therefore much higher here compared to the rest of the restored area. This, in combination with the effect of frequent cutting, results in a vegetation type that was so different that a complete restoration is not likely. Also in restored forest along the road these species are dominating the innermost road verges (Aker 2015), demonstrating their survival under such treatment. On the other hand, based on guidelines from the road administration the vegetation in the road verges should preferably consist of grasses or short vegetation that do not hinder visibility for drivers (Statens vegvesen 2011). Based on this, the vegetation in the road verges today is desired.

4.3 Interspecies interactions

There was a negative relationship between *Polytrichum* spp. and the abundance of character peatland species in the restored zone. Most previous studies show a positive effect of *Polytrichum* spp. during peatland restoration as it functions as a nurse plant for establishment of peatland species (Groeneveld et al. 2007; Grosvernier et al. 1995; Rochefort et al. 2003). A nurse plant facilitates the growth of other plants by offering more suitable microhabitats for germination and recruitment than the surroundings (Ran et al. 2008). Of special interest is the effect on *Sphagnum* establishment. In peatland restoration, a shift towards a self-sustaining system where *Sphagnum* is the moss that creates and builds the ecosystem is essential (Robert et al. 1999). This will establish an acrotelm were other peatland species will have a competitive advantage due to the specific hydrological regime. Of special concern in this study is therefore the sparse establishment of *Sphagnum* in the restored zone. *Polytrichum* has shown to benefit the establishment of *Sphagnum* in several studies conducted on mined peatland, by creating a more moist and cool microclimate (Groeneveld et al. 2007; Lavoie et al. 2003). Other studies show no effect of *Polytrichum* on *Sphagnum* establishment (Ferland & Rochefort 1997). However, as suggested by Callaway and Walker (1997), the nurse plant syndrome might only be valid for a certain period of time and may shift from a beneficial relationship to a competitive one over time. González et al. (2013) found that with a cover above 29 % *Polytrichum* reaches a threshold, not providing the benefits of a nursing plant, but instead being a competitor for *Sphagnum* and other typical peatland species. *Polytrichum* was not dominant in the time immediately following restoration in this project. A sparse establishment was observed in some places after two years (pers. obs, Astrid Brekke Skrindo), but it has probably expanded rapidly after establishment. With an average cover of almost 40 % in the present study, this might indicate that *Polytrichum* has taken the role of a competitor, by forming large and deep cushions.

Sphagnum spp. was observed more often in association with *E. vaginatum* tussocks than *Polytrichum* spp. (pers. obs) (Fig. 8). *Eriophorum* spp. is also believed to nurse the colonization and dispersal of other plants in restored peatlands (Farrell & Doyle 2003; Ferland & Rochefort 1997; Grosvernier et al. 1995; Tuittila et al. 2000). Both *E. angustifolium* (Lanta et al. 2004) and *E. vaginatum* (Grosvernier et al. 1995; Lavoie et al. 2003; Tuittila et al. 2000) have been recognized as nurse plants for recolonization of *Sphagnum* and other peatland species. This is explained by the tussock formation of *E. vaginatum.* The tussocks create a microclimate with lower temperature, higher moisture levels and decreased evaporation, favorable for peatland species (Grosvernier et al. 1995).

Fig. 8. *Sphagnum* growing in *Eriophorum vaginatum* tussocks in a restored plot.

4.4 Environmental drivers

Soil moisture, pH, slope and microtopography were the most important environmental gradients for the distribution of species after restoration. The sites with the best restoration were moister, more acidic and had little slope or microtopography. Soil moisture is connected to all other variables, and therefore seems like a key factor for restoration success. High slope and heterogeneous microtopography creates a drier environment through local drainage. pH is raised when the peat is dried and aerated and the nutrient levels are often raised to levels not representative for this type of poor peatland (Wind-Mulder et al. 1996; Wind-Mulder & Vitt 2000). Some variables might be unique for restoration in peatland. Slope and pH did not show any effect on restoration success in restored forest in the area (Aker 2015), which indicates that these variables are more important in peatlands.

A challenge for peatland species is their strong dependency on the peat environment, which was altered during construction. Most of the acrotelm was removed, leaving only the underground peat (catotelm) to store water for longer periods. The acrotelm provides a high and stable water table by its large pores and high conductivity of water. The acrotelm can shrink and swell in response to changes is moisture, which helps keep the water table close to the surface in natural peatlands (Price & Ketcheson 2009). When the acrotelm was stripped and stored, the large pores were compacted and conductivity reduced. In addition the acrotelm might have been mixed with the naturally more compacted catotelm (Kongsbakk & Skrindo 2009). After redistribution, the acrotelm might have been so compacted that the high water table could not be restored. Cagampan and Waddington (2008) conducted a similar restoration project, where the top part of the acrotelm was stored and redistributed. Large variability was observed in soil moisture, possibly explained by damages to the peat matrix structure during restoration. Buttler et al. (1998) also showed the negative effect of small pore size on *Sphagnum* growth. *Sphagnum* spp. was frequent, but had a low cover in restored plots. There is a strong connection between *Sphagnum* and soil moisture. Studies by Lavoie et al. (2003) and Price and Whitehead (2001) show that regeneration of *Sphagnum* is strongly associated with soil moisture and soil water pressure, generally demanding volumetric soil moisture above 50 %, which Price and Whitehead (2001) suggest as a threshold for establishment. This illustrates the importance of soil moisture during peatland restoration.

Peatland species were associated with low slope and microtopography. High micro topographic heterogeneity is generally thought to increase species diversity and colonization of species due to a higher variety of habitats and niches (Ricklefs 1977). The positive effect of high microtopography was also shown in forested areas in this project (Aker 2015). However, in peatlands, the distribution of species has shown that peatland species preferred only some parts of the topographic spectrum and were mostly associated with ditches and depressions, causing an overall reduction in peatland species (Ferland & Rochefort 1997; Price et al. 1998; Triisberg et al. 2011). Microtopography cause high local drainage which might have prevented peatland species from establishing except from in depressions and ditches. *Carex rostrata* is known for its preference for submerged conditions under stagnant water (Triisberg et al. 2011) and might be one of the species that has established only in the depressions, as it occurred with a high total cover, yet a low frequency. Price et al. (1998) found no correlation between creation of artificial microtopography and establishment of *Sphagnum,* and Girard et al. (2002) confirm that distribution is driven more by the higher soil moisture created in ditches than the microtopography itself. High slope might inhibit peatland restoration (Price & Whitehead 2001), and it seems that more microhabitats and niches cannot compensate for the local drainage caused by higher microtopographic variation.

4.5 Recommendations for restoration practice

The most dominant species in the restored plots were early successional species. Succession will proceed, and the outcome of this restoration is not possible to predict with certainty at this point. Many findings suggest that time will enhance restoration. Firstly, some typical peatlands species had established in the restored plots, most prominently *Eriophorum vaginatum,* but also *Calluna vulgaris* and *Drosera rotundifolia.* Especially the presence of *E. vaginatum* might accelerate regeneration of other species by providing a more suitable microhabitat. Secondly, *Sphagnum* is frequent, although it has a low coverage. If *Sphagnum* continues to disperse over the next years, it will eventually self-reinforce its own establishment due to increased moisture level and decreased pH. In the studies of González and Rochefort (2014) most of the restored sites dominated by *Polytrichum* in early successional stages develop into *Sphagnum* dominated communities over time. Lastly, peatlands require more time to restore, and the northern latitude with short growth periods makes an additional challenge for plant growth (Forbes & Jefferies 1999). However, the oceanic climate in the area with high summer and autumn precipitation might favor restoration success as suggested by González and Rochefort (2014). Further investigations should be carried out in the future, preferably within 10 to 20 years. This will give useful information on whether a self-regulating peatland habitat will be able to establish or if the ecosystem has reached an alternative state dominated by species from other ecosystems.

Restoration success is dependent on what is defined as the goal. In many terms, the project was successful. There were no alien species recorded during the survey, although some *Rumex longifolius* individuals were observed in other parts of the road stretch and young *Picea* spp. were recorded in the forest (Aker 2015). To limit the spread of alien species was one of the most important arguments for using this method. Additionally, the species diversity was high in the road verge and restored zone, indicating a well-established vegetation cover in most sites. However, additional measures could have been carried out to further enhance restoration. The findings of this thesis suggests that soil moisture levels is the key factor limiting restoration success and should be the main focus for improvement in future projects. In the following, I will present some further recommendations.

4.5.1 Storage

Storage causes large changes to the properties of the soil. In this study, topsoil was stored in runners maximum three meter in height. Most of the soil were stored in this way for one year, while some was stored for two. Storage time may reduce seed germination in redistributed topsoil, as shown by Rockich et al. (2000) and Rivera et al. (2012). They both found reductions in germination caused by longer storage time in topsoil from wood- and grassland respectively. Although there was no effect of storage time in my results, the minimum storage time of one year might already have been too long. A shorter storage time is recommended, but could be hard to influence with respect to the ongoing construction work. However, Rivera et al. (2012) also found a positive effect of burial depth on the survival of seeds, indicating advantages of storing soil in larger piles. Although storing the topsoil in larger piles offers both advantages and challenges, I will recommend testing this method in future projects. In a larger pile, more of the seeds will be buried, protected from sunlight, which might reduce germination. Surface transpiration will also be reduced simply because a larger pile would have less surface area than a small one. A large pile can also reduce oxidation of the peat since most of the peat will be stored inside the pile where the environment is anaerobic. On the other hand, transporting the topsoil away from its original location will pose a risk of mixing different types of soil together in addition to increasing the total encroachment time for the soil. Regardless, covering the soil with a non-transparent cover during storage is recommended for any pile size in order to reduce surface germination and water loss.

4.5.2 Redistribution of soil

In this project, one of the focuses was to create an uneven surface by redistributing the soil loosely. This was done to increase aeration and permeability of water, but in addition it created microhabitats with different hydrological conditions between ridges and depressions. This might have limited the establishment of peatland species to the depressions with higher soil moisture, as microtopography resulted in an overall decrease of peatland species. Although artificial creation of microhabitats have been used as restoration method, also in wetlands (Vivian-Smith 1997), it might be unnecessary and in the worst cause do more harm than good. I therefore recommend taking into account the topographical regime of the original habitat when planning the redistribution of soil. Where possible, a low slope should be attempted in order to reduce runoff from the site.

4.5.3 Rewetting strategies

Water availability depends on the temporal distribution of rainfall, the ground water level and evapotranspiration. Evapotranspiration is a key component of the water balance of peatlands (Rydin & Jeglum 2013). Precipitation was probably not a limiting factor for reestablishing the water balance here due to its location, so high transpiration might be a more likely explanation.

Different rewetting strategies have been tested out in order to reduce evapotranspiration, such as creating water reservoirs and pumping up runoff water (Price et al. 2003). However, these measures are expensive, time-consuming and might be unapplibable. In addition, there is a risk of problems with frost heaving, and hence reduced recruitment, when using exterior rewetting in boreal regions (Groeneveld & Rochefort 2002). A more applicable method is to apply a protective cover immediately after soil redistribution. Different types of protection have been tested, such as various types of plastic sheets (Bugnon et al. 1997; Buttler et al. 1998) and straw mulch (Cobbaert et al. 2004; Price 1997), all showing increased soil moisture levels and increased establishment of peatland species. There are many advantages with a straw mulch compared with other types of covers. The straw will not block out water from precipitation, allowing water to penetrate to the surface. Straw mulch creates a more stable temperature regime by controlling the heat flow both over and under the cover, making the substrate more resilient to temperature changes (Petrone et al. 2004). However, the effect of straw mulching decreases with time and it is most effective at storing heat in the first months after application since it will start to degrade over time (Petrone et al. 2004).

However, the effect of protection must be evaluated primarily with regards to shading, since *Sphagnum* is sensitive to shade (Buttler et al. 1998). A plastic cover allows most light to penetrate to the surface (Buttler et al. 1998), while a straw mulch might prevent some of the light to reach the surface. On the other hand, using natural material as straw mulch may be ecologically better and is therefore recommended over plastic covers. This however assumes that the straw is taken from the surrounding areas and consists of indigenous species. If this is done, it might even contribute to the spread natural seeds and increase the plant establishment.

Findings from this thesis demonstrate that low soil moisture levels in combination with increased pH, slope and microtopography are the main constraints for peatland restoration along E10 Lofast II. Further research must be carried out on the outcome of species composition and suggested improvements of the method must be tested in order to conclude on the use of revegetation from indigenous soils in peatland ecosystems. The methods should focus on maintaining moisture levels in the stripped topsoil in order to optimize growth conditions for *Sphagnum* and other peatland species. Optimal storage time and storing methods for topsoil should be tested in addition to alternative strategies for redistribution. Different rewetting strategies should be tested out after the soil has been redistributed to limit transpiration from the soil.

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Appendix

Appendix 1. Species list with frequencies and total cover for each zone.

	Control		Restored		Roadverge	
Species	Frequency	Total cover (%)	Frequency	Total cover (%)	Frequency	Total cover (%)
Agrostis capillaris	0	0	7	30	26	190
Andromeda polifolia	33	213	5	6	1	2
Asteraceae spp.	0	0	0	0	1	1
Avenella flexuosa	0	0	1	1	0	0
Barbilophozia spp.	0	0	1	1	0	0
Betula nana	5	55	1	12	0	0
Betula pubescens	5	28	21	131	31	146
Calamagrostis phragmitodies	1	2	12	73	18	241
Calluna vulgaris	12	168	12	139	2	6
Carex atrata	0	0	1	1	0	0
Carex canescens	0	0	15	65	10	18
Carex echinata	0	0	3	5	0	0
Carex spp.	2	2	0	0	1	1
Carex nigra	0	0	2	2	0	0
Carex pauciflora	12	41	3	12	0	0
Carex paupercula	5	20	6	29	1	1
Carex rariflora	0	0	1	1	0	0
Carex rostrata	6	15	8	168	1	1
Cerastium alpinum	0	0	0	0	1	1
Cerastium spp.	0	0	1	1	2	2
Cerastium vulgare	0	0	0	0	1	1
Chamerion angustifolium	0	0	13	66	34	141
Cicerbita alpina	0	0	0	0	0	0
Cladina spp.	7	19	2	2	0	0
Cladonia spp.	4	5	6	6	2	2
Cornus suecica	8	27	1	1	0	0
Deschampsia cespitosa	0	0	12	272	36	1630
Dicranum spp.	10	87	7	16	1	1
Drosera longifolia	4	4	0	0	0	0
Drosera intermedia	1	1	0	0	0	0
Drosera rotundifolia	25	64	8	10	0	0
Empetrum nigrum	27	311	0	0	4	7
Epilobium palustre	1	1	1	1	1	1
Equisetum arvense	1	1	0	0	0	0
Equisetum sylvaticum	2	3	7	55	6	15
Eriophorum angustifolium	10	23	4	62	0	0
Eriophorum vaginatum	22	161	30	370	1	3
Festuca ovina	0	0	0	0	2	13
Festuca rubra	0	0	1	1	0	0
Festuca vivipara	0	0	1	3	1	5
Filipendula ulmaria	0	0	1	1	0	0
Geranium sylvaticum	0	0	0	0	1	1

Gymnocarpium dryopteris	0	0	1	2	1	3
Hieracium spp.	0	0	0	0	1	1
Huperzia selago	1	1	0	0	0	0
Hylocomium splendens	5	21	3	4	1	1
Juncus filiformis	0	0	11	86	5	30
Luzula frigida	0	0	4	4	7	13
Luzula multiflora	0	0	0	0	1	1
Trientalis europaea	5	14	5	15	5	15
Matteuccia struthiopteris	0	0	1	2	0	0
Melampyrum pratense	2	3	0	0	0	0
Menyanthes trifoliata	2	30	1	2	0	0
Mnium spp.	0	0	9	19	1	1
Molinia caerulea	2	5	1	1	0	0
Oxycoccus microcarpus	24	53	1	1	0	0
Picea abies	0	0	2	2	4	5
Pleurozium	10	176	1	1	0	0
Poa alpina	0	0	0	0	1	1
Pohlia spp.	0	0	19	122	19	151
Polytrichum spp.	9	55	35	1381	29	814
Potentilla erecta	0	0	0	0	2	2
Ptilidum spp.	10	91	5	15	1	1
Racomitrium lanuginosum	6	129	3	6	0	0
Ranunculus acris	0	0	0	0	3	4
Ranunculus spp.	0	0	0	0	2	2
Rhinanthus minor	0	0	0	0	7	15
Rhytidiadelphus spp.	4	6	7	22	1	1
Rubus chamaemorus	19	131	4	11	2	3
Rumex acetosa	0	0	5	15	10	34
Rumex acetosella	0	0	0	0	1	5
Rumex longifolius	0	0	0	0	4	10
Sagina procumbens	0	0	0	0	1	1
Salix glauca	0	0	1	1	0	0
Salix spp.	0	0	16	71	18	98
Salix lapponum	0	0	1	1	0	0
Salix phylicifolia	0	0	0	0	1	10
Solidago virgaurea	0	0	0	0	4	11
Sorbus aucuparia	0	0	0	0	2	3
Sphagnum spp.	35	2808	27	242	2	2
Stellaria graminea	0	0	0	0	1	1
Trichoporum cespitosum	22	194	4	15	0	0
Trifolium repens	0	0	0	0	2	3
Vaccinium myrtillus	6	17	1	1	2	2
Vaccinium uliginosum	16	31	7	7	2	2
Vaccinium vitis-idaea	2	3	0	0	0	0
Veronica spp.	0	0	1	1	1	1
Viola palustris	0	0	2	3	5	8
Unknown bryphytes	3	5	1	1	1	1

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