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Re-inventory of alpine heath vegetation after three decades

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Summary

To assess possible effect of global warming, we re-visited two sub-Arctic heath sites which were previously part of a long-term intensive monitoring programme (from 1982-1994) and performed a new inventory of the vegetation by retracing permanent monitoring plots. This allowed an assessment of how vegetation species and communities have been affected by recent decades of rapidly changing climate, making use of the rare opportunity afforded by the existence of high-quality data from the earlier period. Changes in vegetation community were found since the earlier period, but these were largely shifts in the relative abundances of the same species as previously found rather than substantial species turnover.

1 Introduction

The Arctic and subarctic regions of the world are currently experiencing rapid changes in climate, exceeding those experienced in most other areas (AMAP, 2017; Gilg et al., 2012; Post et al., 2009), with a trend of earlier snow melt (Rumpf et al., 2014) and tree line advance (Kullman, 2002; Sundqvist et al., 2008). Climate change has been more rapid in the Arctic than at lower latitudes. Air temperature has increased at more than double the global rate (IPCC, 2019). While the future development of these changes depends in part upon political and economic decisions that are made in relation to emissions of greenhouse gases, there is a consensus that further warming is very likely over the coming decades, and a virtual certainty in predictions that the Arctic region will warm more rapidly than the global mean (IPCC, 2021).

Clearly, such major shifts in the climate have impacts on vegetation, including increased shrub height and cover (Myers-Smith et al., 2019), increased graminoid cover and decreases in mosses and lichens (Elmendorf et al., 2012), and shifts in functional traits (Bjorkman et al., 2018). Most plants found in the Arctic and sub-Arctic are slow growing species, which may lack the capacity to respond rapidly to environmental changes (Kapfer et al., 2012). Although there is local habitat heterogeneity in tundra, the same species are found across very large areas (Scheiner & Rey-Benayas, 1994). Changes in vegetation communities that reflect factors affecting some of these very widespread plant species could therefore have an impact across the biome and are potentially of much more than local concern. The slow growth of Arctic vegetation also means that to investigate changes, long time periods are needed, and the use of historical data is one way to achieve the necessary time frame. Studies contrasting the present situation with the past are a useful method that can avoid some of the limitations of manipulation experiments and (especially where they can be combined with reliable climate data) provide another source of information for predictions, complementing manipulation studies (Vellend et al., 2013). While space-for-time approaches to contrasting past and present are often necessary for practical reasons, they suffer from limitations that do not affect long-term monitoring data (Pickett, 1989).

Two sites in sub-Arctic/mountain areas of Sweden (Abisko and Ammarnäs), which were part of the PMK intensive monitoring programme were regularly inventoried from the mid-1980s to 1994. The sites were abandoned in 1994 (1996 Ammarnäs) but all data are kept in the database for the succeeding program ICP Integrated Monitoring (IM). A site at Abisko located below the treeline was also re-surveyed.

A re-survey of vegetation at these locations presented a rare opportunity to investigate changes that have occurred in the intervening decades characterised by rapid shifts in environmental conditions. We assess how taxonomic and functional diversity have changed in this sensitive environment, examine which species or groups of species have increased at the expense of others, and contribute to the wider assessment of long-term trends in high latitude vegetation communities.

2 Methods

2.1 Locations

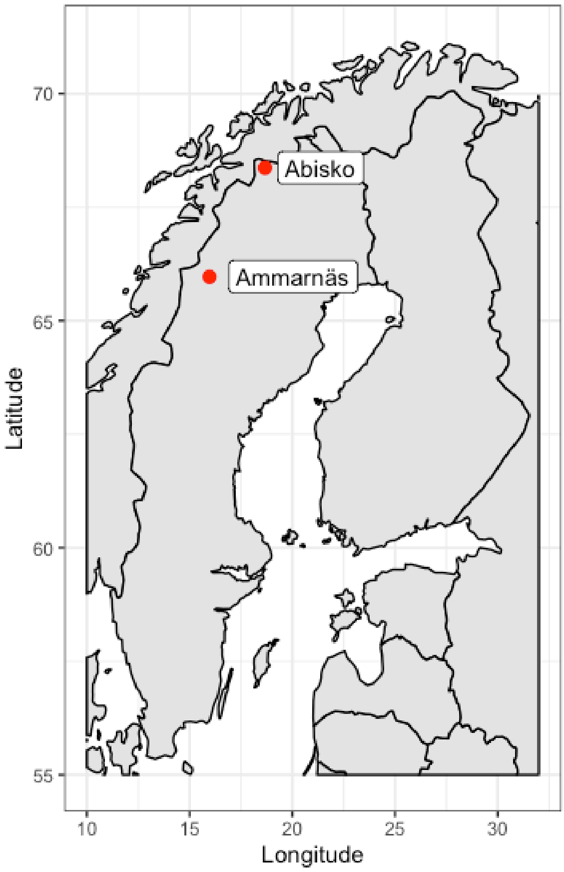

Figure 1 : Map showing site locations

Two sites in sub-Arctic/mountain areas of Sweden (Abisko and Ammarnäs, Fig. 1, Table 1), which were part of the PMK intensive monitoring programme were regularly inventoried from the mid-1980s to 1994. The sites were abandoned in 1994 but all data are kept in the database for the succeeding program ICP Integrated Monitoring (IM) (ICP IM, 2022). A site at Abisko located below the treeline was also re-surveyed.

Table 1 : Main characteristics and approximate locations of the monitored sites. Precise co-ordinates suitable for re-locating the plots can be found in the appendix

Site	**lat.**	**long.**	**alt, m**	**zone**	**dominant vegetation**
Abisko heath	68°22´	18°40´	915	alpine	alpine heath
Abisko forest	68°21´	18°45´	440	subalpine	subalpine birch forest
Ammarnäs heath	65°58´	15°58´	1034	alpine	alpine heath

2.2 Intensive monitoring plots

The intensive monitoring plots were originally selected subjectively within the reference area, with the aim of representing one or more typical vegetation communities for the area. Choosing an area on as flat ground as possible, a 40x40 m area was measured, and the corners and centre marked with an aluminium marker peg. Within this area, 16 or 32 small squares of 50 x 50 cm were distributed according to a random stratified pattern and inventoried. The south-west corner of each small square was marked with an aluminium plate buried in the ground to allow identification in later inventories with the use of a metal detector.

The sites were revisited in the first two weeks of August 2021. The intensive plots were initially located with the help of the approximate co-ordinates available, photographs taken during the previous inventories and the use of a metal detector to find the corner pegs. Once the corners were located the edges of the large 40 x 40 m square were marked out using tape measures and the smaller 50 x 50 cm squares identified using a metal detector.

While the two heath sites were almost intact, with corner and centre pegs and almost all small square markers being found, the forest site was not so well preserved (only 23 of the 32 small plots from the last inventory were able to be found). The reasons for this are unclear but whether due to animal disturbance, frost heaving, surface water or some other cause, the corner pegs and many of the marker plates seem to have been lost in the intervening period. Consequently, the results presented for this site may be affected by bias in the surviving small plots.

2.3 Analysis

Changes in vegetation community composition were investigated using nonmetric multidimensional scaling (nMDS), using the R package vegan 2.5-7 (Oksanen et al., 2020). The nMDS analysis was applied to a Bray–Curtis dissimilarity matrix created from the species abundances (Faith, Minchin, & Belbin, 1987). However, as there was a change in observers between the re-inventory and the original surveys, we also used a Jaccard index based on presence/absence data as input to an nMDS. The rationale here was that since a greater degree of subjectivity and inter-personal variation is involved in estimating species cover than in ascertaining species identity, it would be useful to check if the same patterns are seen in both approaches. However, multiple studies have shown that a warmer climate may lead to changes in species dominance rations on alpine heaths (#REF) and changes in recorded species cover may well be true changes and not a bias due to change of field staff. For all nMDS ordinations performed, a minimum stress value of 0.2 was required. Detrended correspondence analysis (DCA) was also performed in all cases to confirm the nMDS results using an independent method, but as these were in all cases very similar to the nMDS they are not shown here.

We tested for differences in community composition over time, by using year as a grouping factor in permutational multivariate analysis of variance (Permanova (Anderson, 2001)) using the adonis2 function of the R package vegan 2.5-7 (Oksanen et al., 2020). This is a test for difference in the group centroids, applied to Bray-Curtis distances. The BETADISPR function of vegan was used to test for homogeneity of multivariate dispersion, an assumption of Permanova (although Permanova is robust to heterogeneity (Anderson & Walsh, 2013)).

Where significant differences in community composition were found, the species most responsible for any changes were analysed using similarity percentage analysis (using the Simper function in the vegan package). While this method outputs a ranked list of species contributions to dissimilarities it can confound within group and between group differences. For example, the most abundant species often have high variance within a group of samples, and thus have a high contribution to differences between groups even they do not really differ greatly between groups. A solution is permutation tests (permuting between groups) to find the species where difference between groups is actually important in their contribution to dissimilarities, and this is used here. As the results are ranked in order of importance, only the first three significant species are presented rather than including species that may be significant but have very small effects. Where the same species are present in the same order in all contrasts (e.g., between 2021 -1990 and 2021 – 1986) only one set of results is presented.

Changes in community preference for abiotic variables (light, pH, nutrient levels, temperature, and moisture levels) was investigated using community weighted mean Ellenberg values (Ellenberg et al. 2001). These are based on the classification of species on a scale according to the location of their ecological optimum along a gradient. We calculated the community weighted mean values using the vegdata R package (Jansen & Dengler 2010) and the BryForTrait database (Bernhardt-Römermann et al. 2018), thus producing an Ellenberg index for the community. Changes were assessed for significance using ANOVA. We also calculated the Shannon diversity index for each subplot, allowing a comparison of values between years (again using ANOVA).

3 Results

3.1 Abisko Heath

The first nMDS analysis, based on Bray-Curtis distance shows a high degree of overlap between the three older surveys (Fig 2), and some divergence from these in the 2021 re-resurvey (although there is still considerable overlap. Note that the ellipses drawn in this section are all based on the standard deviation of point scores and do not encompass all points for a given year).

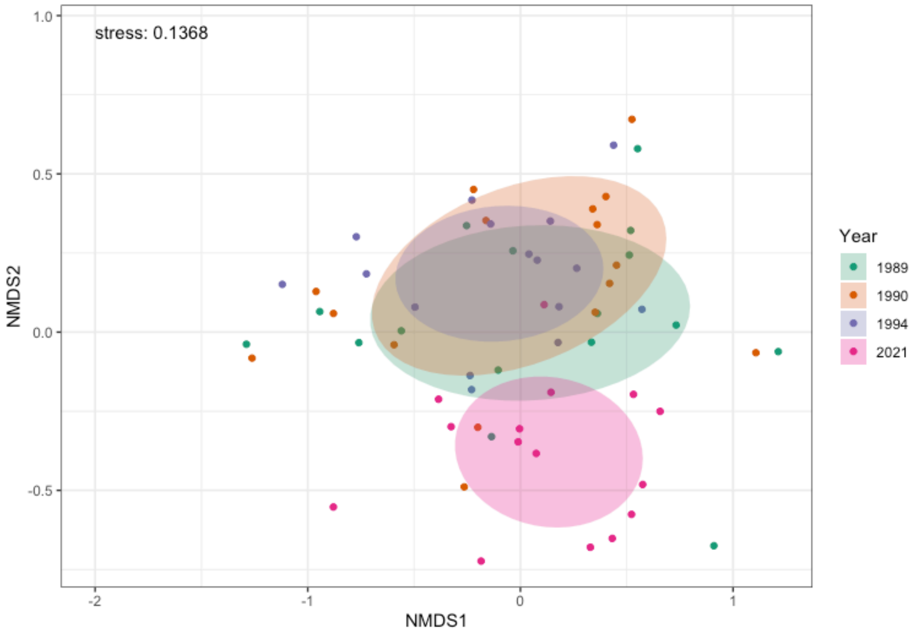

Figure 2: Abisko heath, NMDS based on Bray-Curtis distance. Ellipses based on standard deviation of point scores. Each point represent a 50 × 50 cm vegetation monitoring plot.

The analysis based on presence-absence data is very similar, suggesting that variation in estimation of cover between observers is not an important factor in this case (Fig 3).

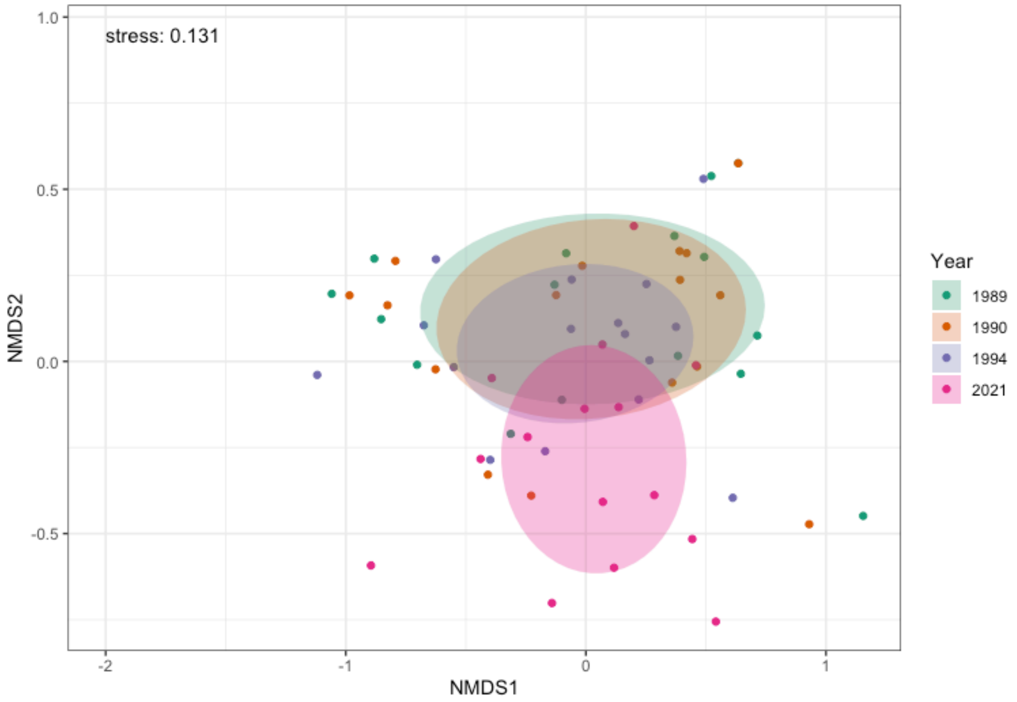

Figure 3: Abisko heath, NMDS based on Jaccard distance (presence/absence). Ellipses based on standard deviation of point scores. Each point represent a 50 × 50 cm vegetation monitoring plot.

The Permanova test for difference between years indicates significant differences between 2021 and both 1989, 1990 and 1994, differences which involve a shift in the group centroid rather than only greater dispersion (Table 2).

Table 2: Permanova test for difference between years and test for change in dispersion, Abisko heath site.

pairs	Df	SumsOfSqs	F.Model	R2	p.value	p.adjusted	Sig	Beta disp.
1989 vs 1990	1	0.1467	0.5876	0.0192	0.776^{ns}	1	ns	ns
1989 vs 1994	1	0.8515	4.0933	0.1200	0.002^{ns}	0.012	ns	ns
1989 vs 2021	1	0.7789	3.6567	0.1119	0.007**	0.042	*	ns
1990 vs 1994	1	0.5107	2.4655	0.0759	0.029^{ns}	0.174	ns	ns
1990 vs 2021	1	1.0858	5.1198	0.1500	0.001**	0.006	**	ns
1994 vs 2021	1	1.7562	10.3965	0.2638	0.001**	0.006	**	ns

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, ns non-significant

The species most responsible for the above changes can be seen in the Simper results (Table 3), which show that the dwarf shrubs *Betula nana* and *Empetrum hermaphroditum* and the sedge *Carex bigelowii* are the top significant results. In all cases the change was an increase in their abundance.

Table 3: Simper analysis showing the species most responsible for changes between years. First three significant results shown. Ratio is Average to sd ratio, ava and avb are average abundances per group, cumsum is ordered cumulative contribution to contrasts.

Contrast	Species	average	sd	ratio	ava	avb	cumsum	p
1994-2021	Betula nana	0.1598	0.1150	1.3897	6.1250	27.333	0.221	0.002**
1994-2021	Empetrum hermaphroditum	0.1089	0.0891	1.2222	6.1875	19.000	0.372	0.002**
1994-2021	Carex bigelowii	0.0304	0.0238	1.2738	0.9375	5.600	0.553	0.002**

** $p < 0.01$

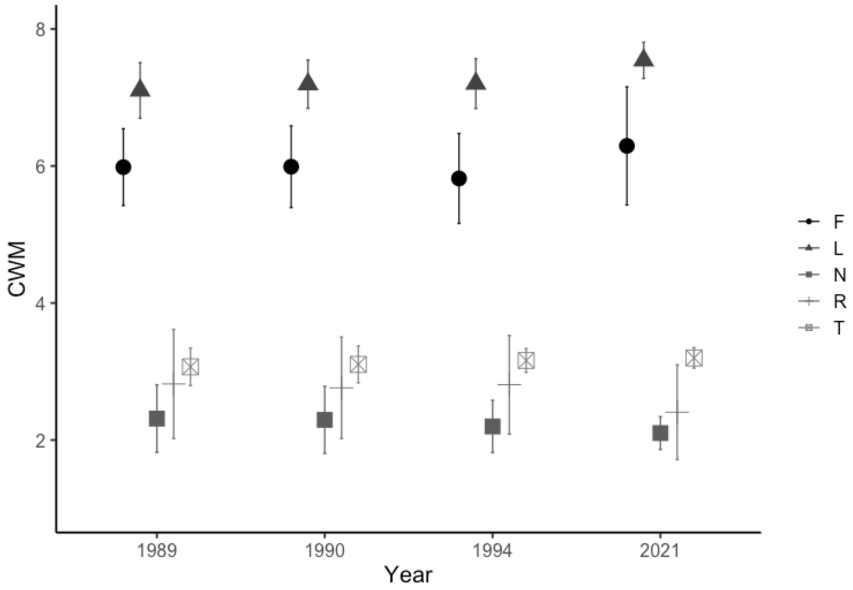

Figure 4 : Community weighted mean Ellenberg values (CWM) for preferences in moisture (F), light (L), nitrogen (N), reaction(R) and temperature (T) at the Abisko heath site.

Changes in Ellenberg index can be seen in Fig 4, but only the increase in mean light preference between the 1989-1994 surveys and the 2021 re-survey was significant (Table 4). Between year changes confirmed for significant results using Tukey HSD tests.

Table 4: ANOVA test for significant difference between years for community weighted mean Ellenberg values and Shannon diversity at the Abisko heath site.

	F	L	N	R	T	Shannon div.
Abisko heath	ns	**	ns	ns	ns	***

*** $p < 0.001$, ** $p < 0.01$, ns non-significant

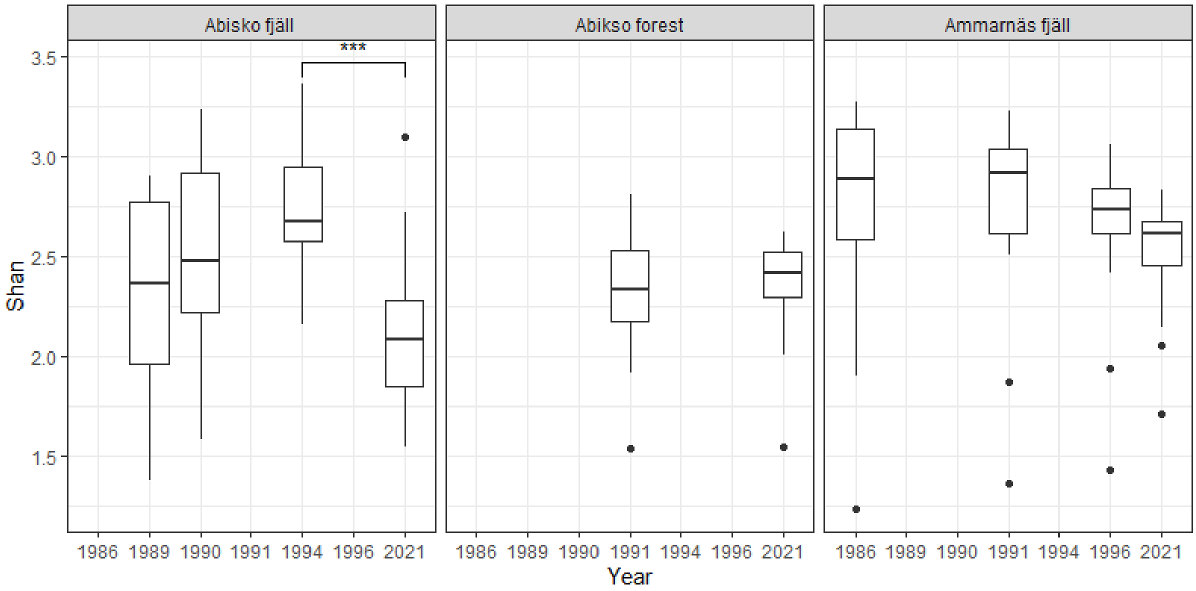

*Figure 5: Changes in Shannon diversity index across years at all sites. ANOVA/Tukey HSD, significance codes, *** p < 0.001, ** p < 0.01, * p < 0.05*

Shannon diversity decreased significantly at the Abisko heath site between the last (1994) survey and the 2021 re-survey (Fig 5), from a mean of 2.74 to a mean of 2.1. Diversity apparently also decreased at Ammarnäs, but this was not significant.

3.2 Ammarnäs Heath

For the Ammarnäs site, a similar pattern to Abisko can be seen, with some separation between the 2021 re-survey and the older surveys, but a large degree of overlap (Fig 6).

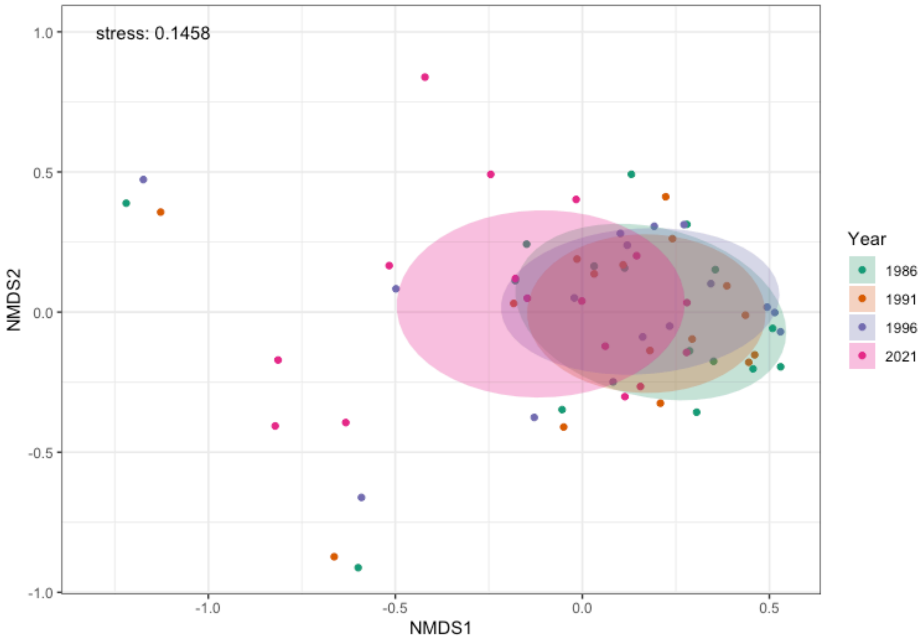

Figure 6: Ammarnäs heath, NMDS based on Bray-Curtis distance. Ellipses based on standard deviation of point scores. Each point represents a 50 × 50 cm vegetation monitoring plot.

The presence-absence ordination (Fig 5) shows that the ellipse for the 2021 survey more closely overlays the earlier surveys, which suggests that observer estimation may account for some of the separation seen in Fig 6.

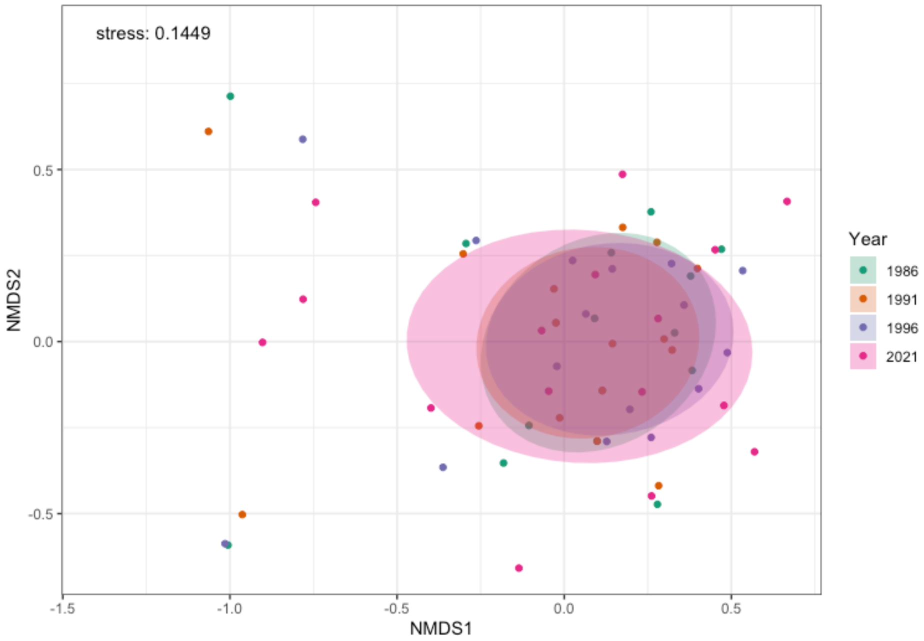

Figure 7: Ammarnäs heath, NMDS based on Jaccard distance (presence/absence). Ellipses based on standard deviation of point scores. Each point represents a 50 × 50 cm vegetation monitoring plot.

However, the Permanova test for difference between years (Table 5) indicates significant differences between 2021 and both 1986, 1991 and 1996, differences which involve a shift in the group centroid rather than only greater dispersion.

*Table 5: Permanova test for difference between years, and test for change in dispersion, Ammarnäs heath site. Significance codes, *** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$*

pairs	Df	SumsOfSqs	F.Model	R2	p.value	p.adjusted	sig	Beta disp.
1986vs1991	1	0.0553	0.3214	0.01059	0.975	1	ns	ns
1986vs1996	1	0.3984	2.3511	0.07267	0.013	0.078	ns	ns
1986vs2021	1	0.9248	5.6371	0.15818	0.001	0.006	**	ns
1991vs1996	1	0.2560	1.5462	0.04901	0.1	0.6	ns	ns
1991vs2021	1	0.8424	5.2582	0.14913	0.001	0.006	**	ns
1996vs2021	1	0.8292	5.2707	0.14943	0.001	0.006	**	ns

The species most responsible for these changes are shown in the Simper results (Table 6), which show that the sedge *Carex bigelowii*, the dwarf shrub *Empetrum hermaphroditum* and the grass *Festuca ovina* are the top significant results, with increases in all cases

*Table 6: Simper analysis showing the species most responsible for changes between years. First three significant results shown. Significance codes, *** p < 0.001, ** p < 0.01, * p < 0.05*

Contrast	Species	average	sd	ratio	ava	avb	cumsum	p
1996-2021	Carex bigelowii	0.0454	0.0423	1.0715	2.875	8.250	0.35	0.0479 *
1996-2021	Empetrum hermaphroditum	0.0288	0.0544	0.5304	0.500	5.375	0.45	0.0019 **
1996-2021	Festuca ovina	0.0268	0.0327	0.8188	1.625	5.437	0.54	0.0019 **

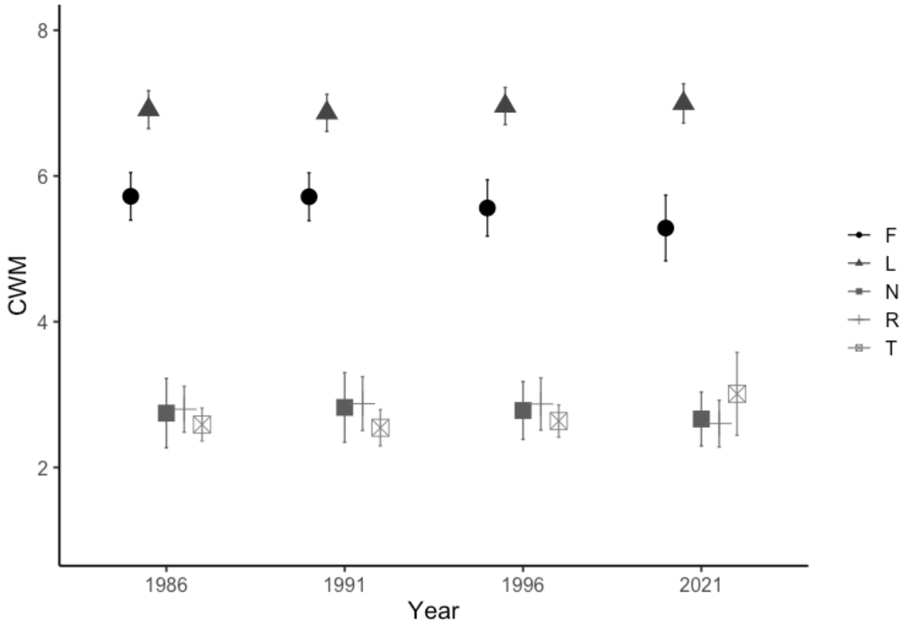

Figure 8: Community weighted mean Ellenberg values (CWM) for preferences in moisture (F), light (L), nitrogen (N), reaction(R) and temperature (T) at the Ammarnäs heath site.

At Ammarnäs the mean Ellenberg index for moisture decreased significantly from the 1986 and 1991 surveys to the 2021 re-survey, while the mean temperature preference increased significantly between the 1986-1996 surveys and the 2021 re-survey (Fig 8, Table 7).

*Table 7: ANOVA test for significant difference between years for community weighted mean Ellenberg values and Shannon diversity at the Ammarnäs heath site. Significance codes, *** p < 0.001, ** p < 0.01, * p < 0.05. Between year changes confirmed for significant results using Tukey HSD tests.*

	F	L	N	R	T	Shan
Ammarnäs heath	**	ns	ns	ns	**	ns

3.3 Abisko Forest

At the Abisko forest site, the two surveys are quite distinct (Fig 9), suggesting substantial changes in the vegetation community here, although interpretation is complicated by the loss of around a third of the subplots present in the first survey.

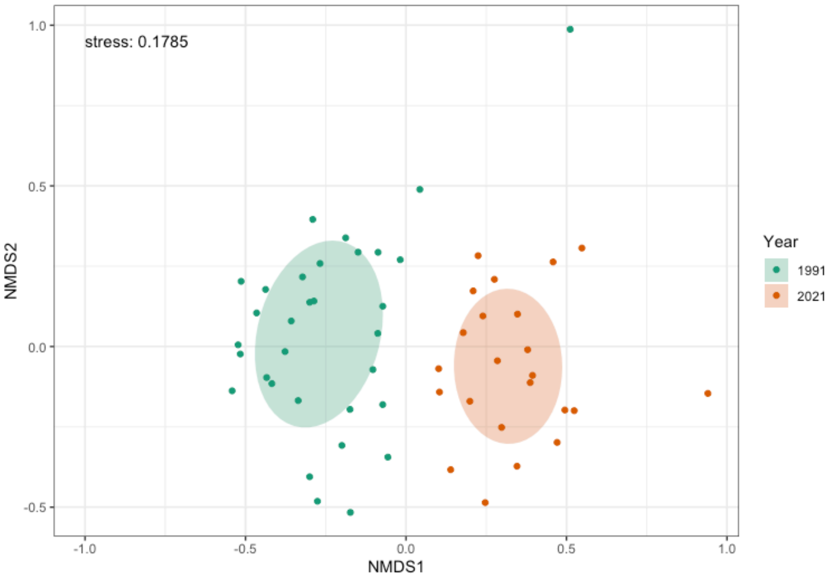

Figure 9: Abisko Forest, nMDS, Bray-Curtis distance. Ellipses based on standard deviation of point scores. Each point represents a 50 × 50 cm vegetation monitoring plot.

The presence-absence version of the ordination (Fig 10) again reduces the degree of separation found between surveys, but distinct clusters of points based on survey year remain.

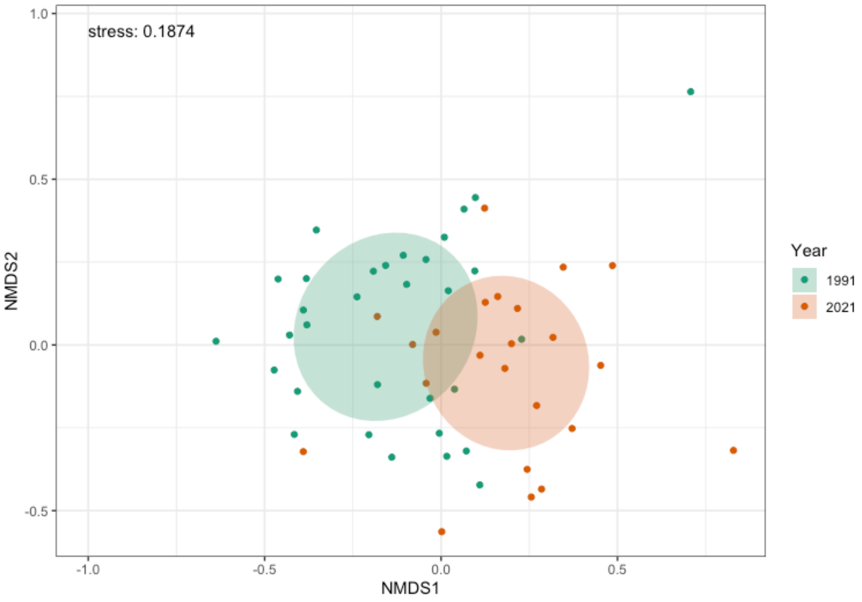

Figure 10: Abisko Forest, NMDS based on Jaccard distance (presence/absence). Ellipses based on standard deviation of point scores. Each point represents a 50 × 50 cm vegetation monitoring plot.

The Permanova test for difference between years (Table 8) indicates significant differences in centroids between 2021 and both 1991, although there is also a significant difference in dispersion. While this indicates that either only a change in dispersion, or both a change in centroid and a change in dispersion has occurred, the nMDS plot (Fig 10) strongly suggests that both have occurred.

*Table 8: Permanova test for difference between years, Abisko forest. Significance codes, *** p < 0.001, ** p < 0.01, * p < 0.05*

	Df	SumOfSqs	R2	F	Pr(>F)	Beta disp.
Year	1	3.715	0.315	24.378	0.001	0.012*
Residual	53	8.077	0.685	NA	NA	
Total	54	11.793	1	NA	NA	

The species most that best account for the changes seen above according to the Simper analysis (Table 9) are the forb *Geranium sylvaticum* (decrease) the grass *Deschampsia flexuosa* (increase) and the fern *Gymnocarpium dryopteris* (increase).

*Table 9: Simper analysis showing the species most responsible for changes between years. Top three significant results shown. Significance codes, *** p < 0.001, ** p < 0.01, * p < 0.05*

Contrast	Species	average	sd	ratio	ava	avb	cumsum	p
1991-2021	Geranium sylvaticum	0.084	0.074	1.136	19.91	4.69	0.113	0.014*
1991-2021	Deschampsia flexuosa	0.083	0.048	1.695	1.28	15.87	0.223	0.002 **
1991-2021	Gymnocarpium dryopteris	0.077	0.063	1.222	1.38	13.65	0.326	0.002**

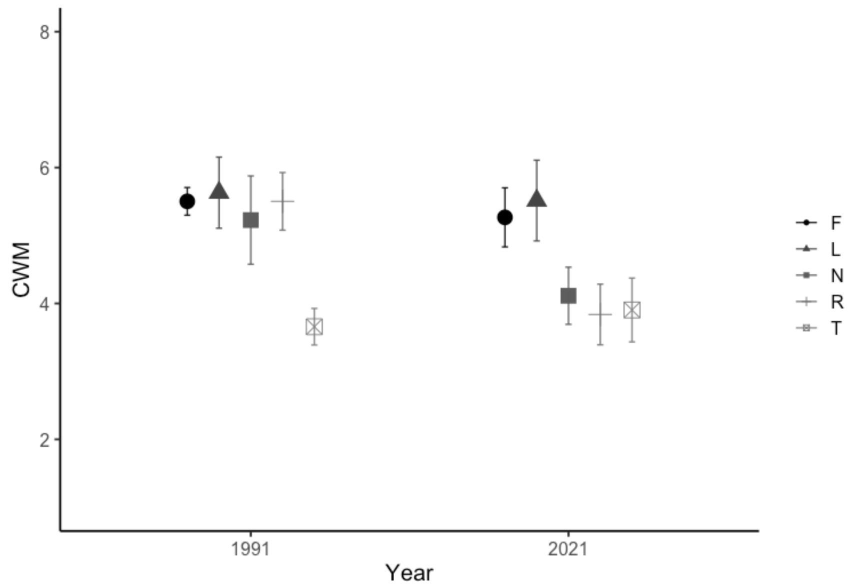

Figure 11: Community weighted mean Ellenberg values (CWM) for preferences in moisture (F), light (L), nitrogen (N), reaction(R) and temperature (T) at the Abisko forest site.

Ellenberg moisture index decreased significantly between 1991 and 2021, nitrogen and pH preference also decreased, while temperature preference increased. Note however that around a third of subplots present in 1991 were not found in 2021, and the missing plots may not be randomly distributed in terms of species preferences.

*Table 10: ANOVA test for significant difference between years for community weighted mean Ellenberg values and Shannon diversity at the Abisko heath site. Significance codes, *** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$*

	F	L	N	R	T	Shan
Abisko forest	*	ns	***	***	*	ns

Table 11: Frequency table for longevity of species found at all sites, where 1 = strictly annual, 2 = biennial, 3 = short lived perennial, 4 = long-lived perennial. Values not available for all species (Tyler et al., 2021).

Longevity	**Number of species**
1	2
2	1
3	2
4	60

4 Discussion

At the Abisko heath site, the vegetation community in 2021 has shifted from the composition seen in the earlier surveys (Figs 2, 3), as confirmed by Permanova testing (Table 2). Increases in the abundances of two dwarf shrub species and a grass are most responsible for this shift (Table 3). Despite the change in composition, there are few changes in Ellenberg index, with only an increase in light preference (Fig 4). Shannon diversity has however decreased since the earlier period (Fig 5).

At the Ammarnäs heath site, the ordination of community composition again shows a difference between 2021 and the earlier years (Fig 6), although this is less apparent for the analysis based on presence/absence (Fig 7). The Permanova analysis shows that 2021 is significantly different from the earlier period (Table 5), mainly as a result of increases in a dwarf shrub, a grass and a sedge (Table 6). The Ellenberg index indicates a decrease in moisture and an increase in temperature preference, suggesting that the site has become warmer and drier since the 1990s (Fig 8).

Analyses of the Abisko forest site is unfortunately complicated by the loss of around one third of the small plots, which may not be randomly distributed in terms of the response variables used here (if the markers were lost due to herbivore disturbance for example, losses may be heavily focused on plots with an abundance of certain palatable species). This may explain (at least partly) why the differences between the previous survey and the current re-survey are more pronounced here than at the two heath sites. The separation in the ordinations is very distinct (Figs 9, 10) and is significant in Permanova testing (Table 8), with decreases in a forb and increases in a grass and a fern being highlighted by Simper analysis (Table 9). Community preference for moisture, nitrogen and pH all declined, while temperature preference increased (Table 10). It is unfortunately not possible to have high confidence in these results as a result of the lost plots, but a warmer, drier forest with increased abundance of ferns and grasses is at least consistent with the climatic changes that the area has experienced since the last survey in 1991.

The species lists (Appendix 2) show no signs of invasive species or indeed important changes in the local species pools. The only new species found in this re-survey are those that would very likely have been present in vicinity of the intensive plots in the 1980's and 1990's (for Abisko *Dryas octopetala, Ranunculus nivalis, Ranunculus pygmaeus* and *Rhododendron lapponicum*, for Ammarnäs *Betula nana* and *Salix glauca*). In the case of Ammarnäs, we noted that both these newly occurring shrub species were very abundant near the intensive plot that was surveyed, to a degree that suggests they were unlikely to have been absent nearby in the earlier period. In terms of species loss, the species present in the previous survey but missing in 2021 were in all cases present earlier in only a few subplots. They were in some cases small species that may have been missed or misidentified (*Barbilophozia floerkei, Cladonia coccifera* (Ammarnäs) *Sphaerophorus globosus* (Abisko) or were actually observed at or very near the site but not in the measured subplots (*Luzula multiflora, Polytrichum piliferum* (Ammarnäs), *Thalictrum alpinum, Trollius europaeus* (Abisko). *Luzula spicata* and *Ranunculus acris* were also absent at Abisko in 2021.

An interesting comparison is a recent re-survey study very close to the sites in Abisko. MacDougall et al. (2021) re-surveyed a 900m transect of 20 monitoring plots on a strong elevational gradient, which were last surveyed 100 years ago. They also found no loss of species present in the earlier period, but the transect design involving multiple plots allowed detection of substantial species migration. This was mostly upwards, including the expansion of some boreal woodland species not found in the first survey (although some moisture favouring snowbed flora shifted downwards).

Although species loss and colonisation were not important in our re-survey, the relative abundances of species have shifted to produce a change in community composition. This is driven by increases in some common shrubs and grasses/sedges at both heath sites. These changes are consistent with numerous studies. A recent literature review of 560 observational and experimental warming studies at Arctic and sub-Arctic sites (Bjorkman et al., 2020) found that graminoids and shrubs were the most likely groups to respond positively and increase in abundance.

Some caution in interpreting the results is worth considering. While the analyses based on presence/absence data (Figs 3,7,10) generally align with those based on relative abundance, the fact that groupings based on year are more clearly separated in the former suggests that variation in observer visual estimation of percentage cover by species may account for some proportion of the differences found. Indeed, this is widely recognised as an issue in vegetation monitoring (e.g., Futschik et al., 2020), and a commonly advocated solution of using the same observers is hardly possible over multi-decadal time frames. However, we attempted to minimise variation by using two experienced observers who independently estimated species cover before comparing estimates and agreeing a final value. Another factor is the limited spatial extent, with only 16 or 32 50cm x 50cm monitoring plots within an area of 40m x 40m at each location. While this allows for detailed comparisons to be made at the location in question, results may not be simply extrapolated to larger spatial scales even though it is known that alpine heath vegetation is or at least used to be homogenous over a larger spatial extent (Scheiner & Rey-Benayas, 1994).

While we found changes in the vegetation community it is notable that both this study and the recent Abisko re-survey (MacDougall et al., 2021) found no or very little loss of species present in the earlier periods. One likely reason for limited changes found despite very substantial warming in the regions where the sites are located is that the species found there are typically long-lived and slow growing (over 90% of species present for which longevity values are available have the highest value of 4, indicating long-lived perennials, Table 11). Despite the three decades between surveys the re-survey likely involved many of the same individuals, at least for some species (Bennington et al., 2012). This implies that even where conditions have become less favourable for some taxa, it will take considerably longer for this to be fully reflected in the vegetation community.

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7 Appendix 1: Site co-ordinates

Not all of the marker pegs outlining the 40 x 40 m areas were found, but this was mainly a problem in the Abisko Forest site. The two heath sites should be straightforward to find and mark out based on a visual and metal detector search of the following co-ordinates.

Table 12: Plot marker locations

Abisko heath	Latitude	Longitude
South-east corner marker	68.354792	18.711122
North-east corner marker	68.355037	18.711184
North-west corner marker	68.355150	18.710350
Centre marker	68.354953	18.710816
Ammarnäs heath	Latitude	Longitude
South-east corner marker	65.991797	15.955533
North-east corner marker	65.992163	15.955519
North-west corner marker	65.992168	15.954633
South-west corner marker	65.991805	15.954656
Centre marker (not found, measured location based on corners)	65.991980	15.955073
Abisko forest	Latitude	Longitude
Centre marker (not found, approx. location)	68.355562	18.762955

8 Appendix 2: Species Lists

Ammarnäs	**Abisko Heath**	**Abisko Forest**
Alchemilla glomerulans	Antennaria alpina	Angelica archangelica
Anthoxanthum odoratum	Anthoxanthum odoratum	Anthoxanthum odoratum
Barbilophozia lycopodioides	Arctous alpina	Anthriscus sylvestris
Betula nana	Astragalus alpinus	Astragalus frigidus
Brachythecium reflexum	Aulacomnium Sp.	Barbilophozia barbata
Bryum sp.	Barbilophozia Sp.	Barbilophozia lycopodioides
Carex bigelowii	Bartsia alpina	Barbilophozia Sp.
Carex brunnescens	Betula nana	Betula pubescens ssp. tortuosa
Carex lachenalii	Calamagrostis lapponica	Brachythecium reflexum
Carex vaginata	Carex bigelowii	Brachythecium salebrosum
Cassiope hypnoides	Carex vaginata	Brachythecium sp.
Cerastium alpinum	Cassiope hypnoides	Bryum Sp.
Cetraria islandica	Cassiope tetragona	Calamagrostis purpurea
Cladina arbuscula	Cetraria cucullata	Carex vaginata
Cladonia cornuta	Cetraria islandica	Cerastium fontanum
Cladonia crispata	Cetraria nivalis	Cirsium helenioides
Cladonia deformata	Cladinia spp.	Deschampsia flexuosa
Cladonia fimbriata	Cladonia fimbriata	Dicranum Sp.
Cladonia gracilis	Cladonia gracilis	Elymus mutabilis
Cladonia pyxidata	Cladonia spp.	Epilobium angustifolium
Cladonia sp.	Cladonia uncialis	Equisetum Sp.
Cladonia squamosa	Dicranum spp.	Festuca ovina
Cladonia uncialis	Drepanocladus Sp.	Filipendula ulmaria
Deschampsia flexuosa	Dryas octopetala	Geranium sylvaticum
Dicranum fuscescens	Empetrum hermaphroditum	Gymnocarpium dryopteris
Dicranum scoparium	Euphrasia frigida	Juniperus communis
Empetrum hermaphroditum	Festuca ovina	Linnaea borealis
Euphrasia frigida	Gnaphalium supinum	Luzula pilosa
Festuca ovina	Hieracium alpinum	Lycopodium annotinum
Gnaphalium norvegicum	Hylocomium splendens	Melampyrum sylvaticum
Gnaphalium supinum	Juncus trifidus	Melica nutans
Gymnomitrion concinnatum	Juniperus communis	Milium effusum
Gymnomitrion coralloides	Linnaea borealis	Myosotis decumbens
Hieracium alpinum	Lycopodium annotinum	Paris quadrifolia
Juncus trifidus	Minuartia biflora	Plagiomnium cuspidatum
Juniperus communis	Nephroma arctictum	Plagiomnium Sp.
Lophozia sp.	Pedicularis lapponica	Poa Sp.
Nephroma arctictum	Peltigera aphthosa	Polygonatum verticillatum
Nephroma resupinatum	Peltigera Sp.	Polytrichum commune
Phleum alpinum	Phyllodoce caerulea	Potentilla erecta
Pleurozium schreberi	Pleurozium schreberi	Pyrola minor

Poa alpina
Pohlia nutans
Polygonum viviparum
Polytrichum commune
Polytrichum juniperinum
Pyrola minor
Ranunculus acris
Rumex acetosa
Sagina saginoides
Salix glauca
Salix herbacea
Sanionia uncinata
Sibbaldia procumbens
Solidago virgaurea
Solorina crocea
Sphaerophorus fragilis
Stereocaulon paschale
Taraxacum croceum
Trientalis europaea
Vaccinium myrtillus
Vaccinium vitis-idaea
Veronica alpina
Viola biflora
Barbilophozia floerkei
Cladonia coccifera
Luzula multiflora sub. Frigida
Polytrichum piliferum

Polygonum viviparum
Polytrichum Sp.
Potentilla crantzii
Ptilidium ciliare
Pyrola minor
Ranunculus nivalis
Ranunculus pygmaeus
Rhododendron lapponicum
Salix herbacea
Salix polaris
Saussurea alpina
Selaginella selaginoides
Sibbaldia procumbens
Solidago virgaurea
Stereocaulon Sp.
Trientalis europaea
Vaccinium myrtillus
Vaccinium uliginosum
Vaccinium vitis-idaea
Veronica alpina
Viola biflora
Luzula spicata
Ranunculus acris
Sphaerophorus globosus
Thalictrum alpinum
Trollius europaeus

Ranunculus acris
Rhodiola rosea
Rubus saxatilis
Sanionia uncinata
Saussurea alpina
Silene dioica
Solidago virgaurea
Sorbus aucuparia
Stellaria nemorum
Trientalis europaea
Trollius europaeus
Vaccinium vitis-idaea
Valeriana officinalis ssp. sambucifolia
Viola biflora

(Species highlighted in green were not present in the earlier survey, species highlighted in red were present in the previous survey but not the 2021 resurvey. In some cases, taxonomy may have changed since the earlier period, names used in the older surveys are given here).