Wetland restoration on Öland

- a short-term evaluation of vascular plant community composition after topsoil removal and reinstated grazing

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Swedish University of Agricultural Sciences, SLU
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Keywords: wetland restoration, vegetation, biodiversity, topsoil removal, Ellenberg indicator values

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Abstract

Wetlands are recognized as important types of ecosystems, inhabiting rich biodiversity and performing multiple ecosystem services. However, many have been lost because of anthropogenic activities and a majority of the remaining wetland habitats are in bad conditions, displaying both loss of functionality and biodiversity. Despite large scale global incentives and extensive conservation programs, there are large gaps of knowledge in how to restore and maintain wetlands, where variable success rates have been reported. Here, a short term evaluation was conducted in Petgärdeträsk and Djustadträsk on Öland, after restorations had been initiated three years prior, focusing on the vascular plant community. Patches where vegetation had previously been completely removed were here compared to neighbouring untreated vegetation in order to study potential effects of restoration measures on biodiversity and plant community composition. As patches of different ages existed, these were further compared to evaluate the regeneration of vegetation and formation of community composition over time. In addition, two sites were evaluated in this study accordingly to the standardized national guidelines based on the European habitat protection directive. Studied either as an alkaline fen (EU-code 7230) or a potential Molinia meadow (EU-code 6410), the purpose was to evaluate if and what effect the restoration has had on the vegetation and the trajectory goal of reaching favourable conditions. As an alternative to functional groups, Ellenberg indicator values were applied to analyse the community composition and to study if the existing vegetation would indicate favourable ecological conditions.

Results showed no clear pattern of regrowth or change in species composition over time after vegetation had been removed, although species unique to the scraped patches were observed. However, due to the short time since the initiation of the restoration, it remains unclear whether these species will persist and lead to a shift in community composition or are more opportunistic species that eventually will be outcompeted as the old vegetation establish further. For the Natura 2000-habitats, no evident improvement of the conditions were seen. The differences in the number of characteristic species and in vegetation height that were observed in the vegetation here, could be due to seasonal fluctuations and can therefore not be determined as effects of the restoration only.

Three years is a relatively short time to evaluate effects of restorations, and despite inconclusive results hopefully this material can provide insight to the further development of the vegetative community if the same sites would to be revisited in the future. As discussed lastly, the need for thorough evaluations persists, where adopting wider perspectives and including analyses of functional diversity could improve our understanding of how vegetative communities function and hopefully facilitate the maintenance of valuable wetland habitats.

Keywords: wetland restoration, vegetation, biodiversity, topsoil removal, Ellenberg indicator values.

Table of contents

1. Introduction

Wetland habitats are associated with high biodiversity and provides a multiple ecosystem services such as providing fresh water and food. While always of importance, historically the human use of wetlands have shifted over time, shaping the ecosystems and leaving traces behind.
By the previous turn of the century and throughout the 20th century, the use was intensified and wetlands were commonly converted to arable land for agriculture or forestry. With that, many of the productive wetlands were damaged or completely lost (Gunnarsson & Löfroth 2009).
According to recent estimations, 50% of the wetland area within EU is degraded (Tanneberger et al. 2021). In Sweden, based on the national wetland inventory, 20% of recorded wetlands are in natural, undisturbed conditions while the remaining 80% are affected by some sort of encroachment, exhibiting various levels of damage (Gunnarsson & Löfroth 2009). In Europe, Sweden is the country with the largest diversity of wetland habitats and have second largest mire areal, after Russia (Naturvårdsverket 2006). With that comes a global responsibility of maintaining and restoring these ecologically valuable habitats.

To distinguish wetlands from other types of habitats the Swedish Environmental Protection Agency use a definition where water in wetlands should be close to the surface, in or above ground level over a large part of the year (Löfroth 1991). The vegetation should also be composed by at least 50% of hydrophilic species. With the variation allowed for in this definition, closer to 50 different types of wetlands can be found in Sweden (Naturvårdsverket 2006, Gunnarsson & Löfroth 2009). Hydro-morphology, geochemistry, buffer capacity, nutrient availability and vegetation are factors that shape and can be used to further differentiate wetland habitats from each other (Lamers et al. 2015). The wetlands included in this study are of a few different types (described more in detail later), but are all commonly associated to the agricultural landscape and more nutrient rich conditions.

1.1 Encroachments and implications

During the intense use of wetlands, wetlands were commonly drained to increase the areal suitable for agriculture and forestry. This alteration of hydrology increased productivity where the often more diverse wetland vegetation could be replaced with monocultural crops. In Sweden regulations for draining of wetlands were introduced first in 1986, becoming stricter over the coming decades, to be very limited today. However, being permanent encroachments, many wetland habitats are still affected by drainage, having an altered hydrology.

Alongside the physical encroachments done to convert wetlands, forestry on its own is, according to the national wetland inventory (Gunnarsson & Löfroth 2009), one of the main threats to wetland habitats as deforestation has severe implications on neighbouring ecosystems. Agricultural practices also severely alters the ecology within wetlands, affecting the ecosystem functionality. Carbon sequestration, biodiversity on local, regional and landscape level, buffer capacity, water retention, regulation of water quality, are some examples on services that has been negatively affected (Foley et al. 2005, Gómez-Baggethun et al. 2019). Other anthropogenic activities and infrastructures as sourcing peat for fuel, urbanisation and expansion of the waterpower grid system has also contributed to the pressure that is put on wetland ecosystems (Gunnarsson & Löfroth 2009).

Implications

The drought that comes from drainage not only affects the species composition causing a shift towards vegetation favoured by more dry conditions, but further increase the decomposition and therewith lowers the carbon retention in the system (Lamers et al. 2015). Peat forming wetlands, such as fens, function as carbon sinks when in good, natural conditions, while damaged wetlands have been found to act as carbon sources (Pfadenhauer & Klötzli 1996). Although wetlands are not the largest carbon sink in the world, the effect is globally not negligible according to the end report for the National wetland inventory from 2009.

Other common problems in wetlands are eutrophication and overgrowth. The historical use helped develop the vegetative community of wetlands and allowed for high biodiversity. Mowing and grazing kept the productive communities open and exposed to sunlight, but as these practises are no longer common, closing vegetation is a threat to smaller, light dependent species in the community (Gunnarsson & Löfroth 2009). The use of fertilizer in agriculture has severely increased the availability of primary nutrients in wetland habitats, causing eutrophication and resulting in increased productivity. The additional nutrients reach wetlands through direct application of fertilizer, surface run off, ground water or come from atmospheric deposition and is still deposited in large amounts (Gunnarsson & Löfroth 2009).

Increased nutrient availability, in combination with ceased management, contributes to shifts in vegetation where fast growing species become dominant and out compete slower growing species, thus leading to a loss of diversity (Smolders et al. 2008, Lamers et al 2015, Schnoor et al. 2015, Klimkowska et al. 2019). Of course individual species vary in their requirements and limiting factors, so the change in vegetation is dependent on the availability of certain nutrients. In combination with other factors such as moisture, sun exposure, grazing etc., populations of species adapted to wetland are threatened to various levels of extinction, both at local regional and landscape levels. The vegetation in turn, further influences which fauna the ecosystem can house and support.

With eutrophication, the nutrient cycling in the ecosystem is also altered. A shift in species composition can affect decomposition rates and peat formation in the ecosystem, as these are processes influenced by litter type (Lamers *et al.* 2002, Straková *et al.* 2011). Increased decomposition can in turn, further increase nutrient availability, mineralization and high productivity can decrease the buffering capacity as cations are depleted, contributing to acidification (Lamers *et al.* 2002). This exemplifies the cascade effect that comes with change, when many processes in an ecosystem are linked by complex interactions and are dependent on multiple environmental factors. With this we see a variation in the response to encroachments caused by both direct and indirect effects, where the decomposition rate and productivity etc. differs between wetland habitats.

1.2 Conservation

To remedy the effects of draining, the hydrology needs to be restored. This process can be difficult and is dependent on the extent of the damages. Closing ditches might be sufficient to rewet habitats that have only been mildly affected by drainage whereas, in more severe cases of drainage, restoring hydrology might require a larger effort. Rewetting is in general a complex process that needs to be adapted after the ecosystem and successful outcomes cannot be guaranteed (Large et al. 2007, Lamers et al. 2015, Kreyling et al. 2021).

Restoring the hydrology does not either solve the problem of eutrophication, as nutrients still persists in wetland habitats preventing the vegetation from returning to a more natural state (Smolders et al. 2008, Schnoor et al. 2015). To some extent, the additional nutrients from agriculture can be limited by redirecting the surface run off. However, for the nutrients that do reach the wetlands, few

alternatives to efficiently decrease nutrient availability exists today.
Removing nutrients from the ecosystem by harvesting vegetation and removing it from the site, is often applied on constructed wetlands as a method of wastewater treatment, however, is depended on fast growing species and require a quite long term investment. Harvesting of vegetation is therefore not necessarily compatible when, at the same time, trying restore a specific vegetation.

If an excess of phosphorous is available, one option of restoration is to remove the top soil layer. Although this might not be an intuitive method when restoring vegetation, this removes the phosphorus stored in the peat, which otherwise could have allowed for hypertrophic conditions for decades to come (Smolders et al. 2008). The method basically restarts the ecosystem by creating a more favourable condition where the desired biodiversity can re-establish (Smolders et al. 2008, Klimkowska et al. 2019). Top soil removal is therefore also a method to decreasing productive competitive species that has some reported success (Klimkowska et al. 2019).

As many wetland species are favoured by disturbance and a long historic use of clearing and grazing, reinstating similar measures are commonly practiced to aid a development of characteristic wetland flora. However, studying the response of vegetation to this type of restoration efforts, various levels of success have been reported where the level of disturbance is influencing the outcome together with other environmental factors such as pH and nutrient availability (Sundberg 2012, Schnoor et al. 2015, Zhu et al. 2021).

1.3 Vegetation as ecological indicators

Vegetation is not only an important part of the biodiversity and ecosystem, but as plants often have quite strict requirements, their presence/absence can be used as indicators of the environmental conditions. As a continuation of this, assigning ecological indicator values (hence referred to as EIVs) to plant species has become a method commonly applied in ecology. Although their application has been discussed, multiple studies has found that EIVs can be highly accurate in describing environmental condition if used correctly (Diekmann 2003, Large *et al.* 2007, Andersen *et al.* 2013, Schnoor *et al.* 2015). As reported by Scherrer and Guisan (2019), the explained variation for some measured ecological traits can as much as double when instead using EIVs, suggesting that EIVs are as good or better in predicting conditions as conventional measuring methods, especially regarding traits that are hard to estimate, such as soil moisture.

Although not widely used to evaluate the conservation statues, EIVs can be excellent indicators of the conditions in wetlands and therewith a tool when assessing the progress in response to restoration (Andersen *et al.* 2013). In a long term study evaluating conservation efforts, Large *et al.* 2007, observed a strong correlation between average EIV and the moisture gradient and succession of plants that developed over time after initiated restoration. Further, Andersen *et al.* (2013) found a strong correlation between typical wetland species and Ellenberg indication of nutrient conditions, suggesting that Ellenberg indicators could be used to reflect eutrophication. Which further enhanced what a strong threat added nutrients in wetlands can be to the presence of characteristic plant species. Andresen et al. (2013) concluded that vegetative indicators can be a useful tool in management by supplementing abiotic/structural environmental parameters and sometimes be more cost efficient and accurate in describing conservation statuses in alkaline fens.

1.4 Challenges

Although vegetation shifts have been reported following restoration efforts numerous times, the vegetation often quickly return to the previous composition or develop into something other than the target vegetation (Smolders *et al.* 2008, Schnoor *et al.* 2015, Kreyling *et al.* 2021). So in addition to restoring favourable conditions of wetland habitats, another concern is to get the desired vegetation, often species that are characteristic for a specific type of habitat, to re-establish. It is easier to increase an already existing population by improving favourable conditions, but in many cases species have been locally or even regionally lost. If long time has passed, there is an increased probability of the seedbank being degraded. For many of the characteristic wetland species, the longevity of the seedbank is relatively short, which after long time would have reduced the chances to recolonization of vegetation by seed dispersal or vegetative regrowth (Dijk *et al.* 2007). As not all species are able to disperse over long distances, this affects the reestablishment of target vegetation (Sundberg 2012, Morimoto *et al.* 2017, Klimkowska *et al.* 2019). With that, a larger landscape perspective including, habitat fragmentation comes in to play in restoration of characteristic wetland vegetation. The vegetation adjacent to a restored site is further influential in how the vegetation respond or re-establish after restoration (Morimoto *et al.* 2017).

Alongside topsoil removal preventing competitive species to dominate, a more active management, especially initially, has been suggested to potentially improve the recovery of characteristic wetland flora, assuming that the environmental conditions are also restored to be favourable (Large *et al.* 2007, Smolders *et al.* 2008).

1.5 Prospects and expectations

With the complexity of ecosystems and the various levels of success seen in restoration projects, there is an uncertainty surrounding the conservation of wetlands, and one might question if it is even possible to restore these habitats to their former states. Multiple long-term, large-scale studies of wetlands subjected to various types of restorative actions, have reported little progression towards restoring functionality and diversity similar to that of natural, undisturbed wetlands (Large *et al.* 2007, Klimkowska *et al.* 2019, Gómez-Baggethun *et al.* 2019, Kreyling *et al.* 2021).

Research suggests that restoration, to some extent, contributes to recovery but does not ensure that the ecosystem return to a state similar to natural wetlands in regards to biodiversity and functional diversity (Klimkowska *et al.* 2019). Following rewetting of fens across Europe, Kreyling *et al*. (2021) saw a reestablishment of wetland species, however, lacking the diversity and ecosystem functionality similar to habitats in near natural states. Examining over 30 years of data, time seemed to have little effect on the recovery. In other cases, the conservation effort made has not been enough to counteract the extensive damage and prevent further loss of ecosystem services (Gómez-Baggethun *et al.* 2019).

Despite, or perhaps because of, the slow progress, methods of improving conservation are a popular topic of discussion. Suggested are amongst other, a more active approach, directing efforts towards restoring multiple stressors, and tailoring restoration to specific projects where goals have clearly defined (Large *et al.* 2007, Lamers *et al.* 2015, Morimoto *et al.* 2017, Klimkowska *et al.* 2019). According to Hambäck *et al.* (2023), multifunctionalty and the synergy between individual functions, is another aspect of wetland ecosystems where our management of wetlands would benfit of a better understanding.
A challenge today is that relatively little information is available on how to practically restore and maintain wetland habitats. In the past, much of the conservation effort to restore wetlands have been performed without evidence based knowledge and, with varying levels of success, it is hard to extract information that can be applied in other conservation scenarios. We need to include a wider perspective and better evaluate the effects of conservational efforts in order to optimize the future management of the valuable wetlands.

1.6 Aims

The intent with this paper was to address the gap of knowledge surrounding wetland conservation, by visting two nature reserves on Öland where a wetland restoration project has been ongoing since in 2019. The aim was here to study the effects of top soil removal, and hence also vegetation removal, on the vascular plant community in wetlands. As previously mentioned, top soil removal is a management effort of which effects we know relatively little of. Therefore multiple aspects of the vascular plant community were studied, including biodiversity levels and community composition, with the purpose of gathering a deeper understanding of the recovery of the vegetation and its functionality.

Further inventories of two Natura 2000-classified habitats were also conducted according to the standardized methods for evaluation (Götbrink & Haglund 2010), to provide a short time evaluation of potential changes in vegetation as a response to the ongoing restoration, which also included grazing and clearing of vegetation. Here previous data collected in 2019, to some extent, allowed for a comparison over time.

Now in 2022, three years have past, and although being a relatively short time, vegetation removal was hypothesised to have affected the plant community composition where sites more recently removed of vegetation, were expected to display a lower diversity of species as signs of not yet being fully recovered. Scraping of the top soil layer, in general, might have allowed for new opportunistic species to establish that are not normally found in wetland habitats. As a progression towards recovering the former flora, which seems likely when revising available literature, a decrease of the opportunistic species might be observed over time as the community settles. It was also believed that the plant community composition might show a shift towards containing more species favoured by soil disturbance or other environmental factors that could be associated to the ongoing restoration.

In regards to the Natura 2000-habitats, little change was expected to be noticed in terms of the number of characteristic species that are found. The number of negative indicator species were not either expected to have changed, as negative species, such as *Phragmites australis*, are normally quite persistent once established in a community. However, the vegetation height was expected to be lower because of the now reinstated, continuous grazing.

Lastly, this paper hopes to continue the discussion on the challenges of conservation and the need to further investigate the effects of individual measures and methods in order to be able to tailor them to the purpose and optimize the outcome of restorations.

2. Material and methods

2.1 Location

The data collection was conducted between the 18th of July and 20th of august 2022, on the north east of Öland, in two closely adjacent nature reserves; Petgärde- and Djurstadträsk. Only divided by a small road and some pastures, the two reserves have a shared hydrology and are both rich in bird life. Since both reserves has been a part of the same restoration project and undergone the same conservatory measures, inventories were conducted in both reserves to evaluate the progress of the restoration that was initiated three years prior, in 2019.

2.1.1 Wetland habitat types

Together the two nature reserves includes multiple wetland types. Of concern here are mainly three, which all are included in the Natura 2000-network and there defined either as Molinia meadow, Alkaline fen or Calcareous fen habitats (EU codes: 6410, 7230 and 7210). According to the guidance documents provided by the Swedish Environmental Protection Agency for these habitat types (Naturvårdsverket 2011a, 2011b, 2011c) the biggest threat is different types of drainage causing hydrological and hydrochemical changes, with effects such as dehydration, increased erosion and overgrowth.

Alkaline fens are minerothropic and retain the nutrients from the ground or surface water. Due to being nutritious, fens a have often been converted to agricultural land or productive forests, both nationally and internationally. Now strongly threatened, alkaline fens are to a larger extent included in the mire protection plan (Naturvårdsverket 1994). Alkaline fens are characterized by stress tolerant species, sensitive to increased nutrient levels (Andersen et al. 2013).

Calcareous fens (full name; Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae*) are distinguished by its vegetation that can almost entirely be dominated by Saw grass/saw sedge, *Cladium mariscus*. If managed, these habitats can be more diverse containing multiple species of *Carex* and *orchidae*. This habitat type is prioritised in the habitat protection directive and is most commonly found on Gotland and Öland (Naturvårdsverket 2011c).

Molinia meadows are habitats that have developed under strong management and are variable depending on the geographic location and geochemical properties. Moisture levels are further fluctuating. Molinia meadows can house a rich vegetation, of which some species are uncommon and should further contain species dependent on either grazing or similar forms of management (Naturvårdsverket 2011b).

2.1.2 Restoration efforts

To maintain biodiversity and prevent overgrowth, the practical restoration efforts includes an initial clearing of vegtation and continuous grazing by cattle. The hydrology in the area has been impaired, and what once was a small lake, has been drained to a small watercourse. Since 2019, the soil top layer in certain areas has also been scraped, removing all vegetation, with the main purpose being to improve the hydrology in the area. With little focus on vascular plants, the effect of the scrapings on the vegetation had not yet been studied here.

To maintain important ecological functions within the reserves and not cause too great of a disturbance to wildlife during restorations, the effort to remove vegetation have been conducted in stages, where smaller patches have been scraped over the last few years. For the calcareous fen, the nature reserve regulations limited the area that could be disturbed to a third of the habitat, so that at least 20 ha of the characteristic saw sedge vegetation is over 4 years old (Länsstyrelsen i Kalmar Län 2005). Distributing the effort of scraping over time has resulted in smaller patches with different age regrowth.

2.2 Evaluation

With the purpose of evaluating the plant community response to complete vegetation removal, three of the scraped patches were here chosen in advance, with similar size (~6800-9000 m^2) but different age, scraped either in 2019, 2020 or as late as last year in 2021 (see figure 1).
The first two patches were located in Djurstadträsk, while the latter was located in Petgärdeträsk. Although dry at the time of the study, all sites were closely situated to a stream/water course that periodically contains water. I randomly distributed 30 sample points in each patch with a minimum distance of 2 meters to each other, using the “random points in polygon function” in QGis (3.16.5 with Grass

7.8.5). To act as a reference, another 30 control points were distributed in each reserve, this was to avoid any difference in plant community composition between the two reserves regarding habitat type and environmental conditions. In Djurstadträsk the two scraped patches were located in a Molinia meadow and Calcareous fen habitat, while the site in Petgärdeträsk was defined as an alkaline fen.

The control points were distributed with the same criteria, in the immediate areas surrounding the scraped patches. At each sample point, I placed a 50x50 cm quadrant and all species, as well as their abundances, were noted by counting their occurrence in percent.

Grazing animals were present in both nature reserves with the only exception of the patch scraped of vegetation in 2019, in Djurstadträsk, which was noted first when the inventories were to be conducted. Which is something to consider when interpreting the result from the analysis.

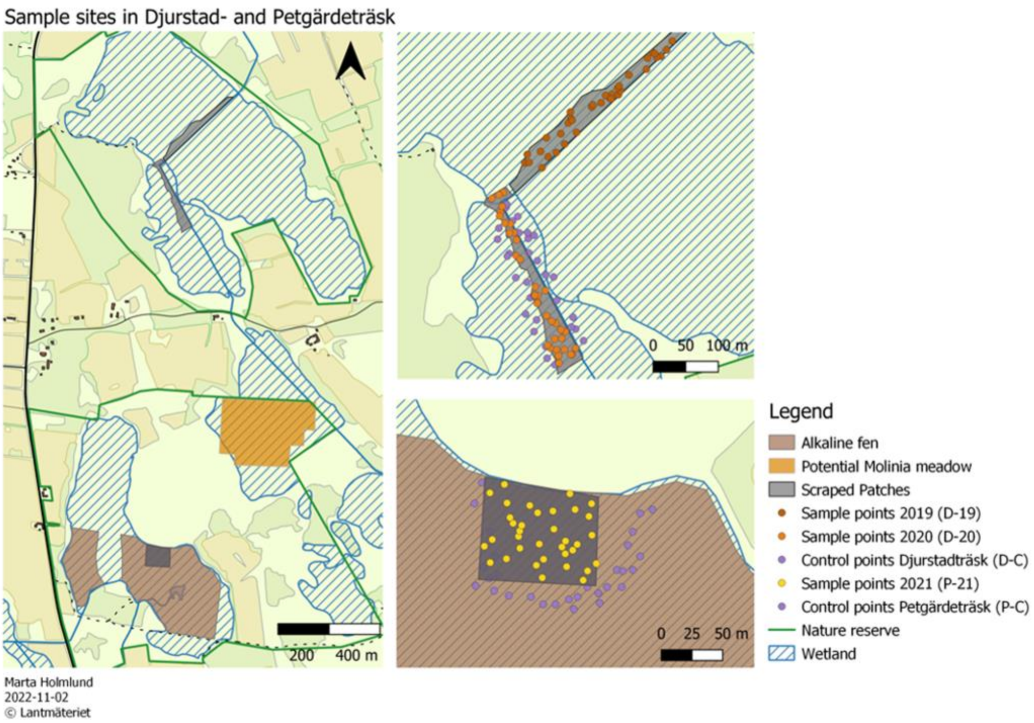

Figure 1. Overview of the area containing the two nature reserves, with Djurstadträsk located in the north, above Petgärdeträsk. All samples sites included in the study are included with sample points for the scraped patches visible in the enlargements to the right. Sample sites have been given abbreviation after their location, either Petgärdeträsk (P) or Djurstadträsk (D), followed by either the year when the top soil layer was removed (-19,-20 or -21) or if control (C). Only the scraped patches included in this study are displayed in the map.

With a particular interest in evaluating how the restoration had affected certain classified Natura 2000 habitats, another 80 points were inventoried as a part of a monitoring program of restored Natura 2000 habitats in wetlands.
Here, Länsstyrelsen in Kalmar län provided the sample points, in a grid system, in accordance with the nationally implemented standardized methods for follow-up surveys (Götbrink & Haglund 2010, Haglund & Vik 2010) (see figure 2). These sample sites had not been directly subjected to vegetation removal but to grazing and clearing. However, a change in hydrology as an effect of the removal of vegetation could not be excluded as scrapings had been performed within the same habitat, in close proximity to some of the sample points.

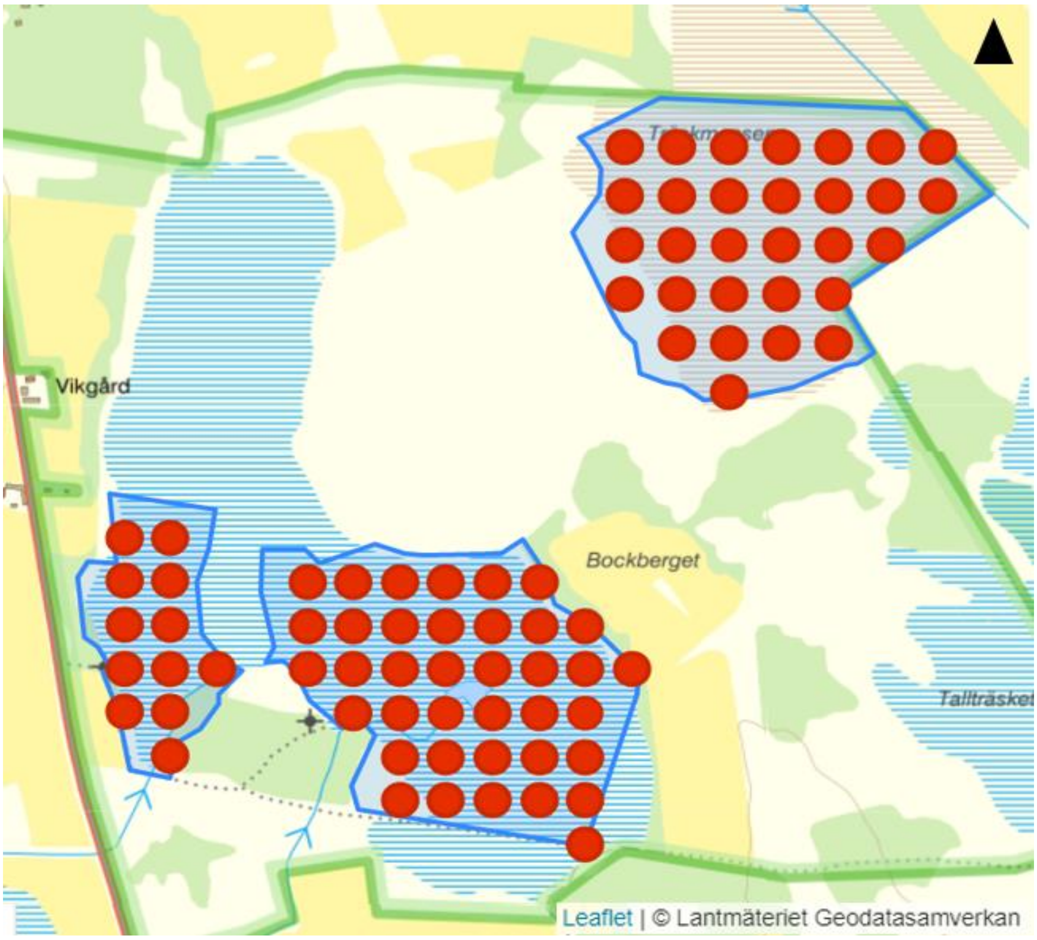

Figure 2. Overview of the sample locations provided by Kalmar länsstyrelse. Noticeable is that the alkaline fen to some extent overlap with the scraped patches in Petgärdeträsk.

All plots were located in Petgärdeträsk where 50 of them were found in a habitat classified as an alkaline fen (EU-code: 7230). In the alkaline fen, a previous data collection had been executed in connection to the restoration in 2019, following the methods found in the guide provided by the Swedish Elänanvironmental Protection Agency (Götbrink & Haglund 2010, Naturvårdsverket 2011a).

The remaining 30 samples were located in a wet meadow area and as the goal is to see a progression towards a Molinia meadow on calcareous, peaty or clayey-silt laden soil, EU-code 6410, these points were therefore inventoried as such. The two habitat types were inventoried after lists of characteristic and negative indicator species provided by länsstyrelsen i Kalmar län. Vegetation data such as height, occurrence of mosses (sphagnum, brown mosses) were also noted along with additional species. In the inventory in 2019, species of brown moss were identified, something that was not done in this study.

In the case of the potential Molinia meadow habitat, no previous data was available for analysis which restricted the evaluation of how the restoration had affected the vegetation to instead being a comparison of how well the current state of the habitat corresponds to what is typically expected in Molinia meadow habitats. During the inventory, all additional species that were not included in the given lists and their abundances, were noted down to a few grasses where identification were deemed impossible. So although no vegetation change could be studied in response to restoration, the recorded species abundance could be used in the analysis in an attempt to provide a more detailed picture of the status of the habitat and its resemblance to a Molinia meadow.

2.3 Analyses

All collected data was processes and assembled in Excel before undergoing analysis conducted in R. General packages used were “picante”, “vegan”, “ggplot2”, “tidyverse” and “writxl”. Additional packages required for specific functions or tests are mentioned below.

2.3.1 Scrapings: Removal of vegetation

Since the scraping had removed all vegetation, the recovery of the vegetation were analysed by comparing the patches of different age. The control sites were used to analyse if and how the community composition had changed due to the scrapings, in terms of species and plant family composition as well as diversity.

Biodiversity

Firstly after being loaded into R, the data was converted to relative abundance by dividing the recorded percentage abundance for each species and sample with the total sample abundance.

To assess if the sample effort was enough to be representative for the area of the study, an accumulation curve and an estimate of species were produced using the functions “speccurve” (method= “random”, permutation=1000) and “specpool” available in the vegan package in R (R Core Team 2021). Included species richness estimates in the “specpool” function are:

Chao	$S_P = S_0 + a1^2/ (2*a2) * (N-1)/N$
Chao bias-corrected	$S_P = S_0 + a1*(a1-1)/ (2*(a2+1)) * (N-1)/N$
First order jackknife	$S_P = S_0 + a1*(N-1)/N$
Second order jackknife	$S_P = S_0 + a1*(2*N-3)/N - a2*(N-2)^2/ N/ (N-1)$
Bootstrap	$S_P = S_0 + Sum (1-p_i)^N$

S_P is the estimated richness and S_O refers to the observed species richness. N equals the sample size, p_i is the frequency of species, where i refers to a species. In the equations a1 and a2 are species with only single or double occurrences. The Chao biased corrected estimate is only applied by the function when a2=0 and was not used here (R Core team 2021).

In continuation, the diversity levels in form of Species richness, Shannon and Simpson’s diversity index were calculated. Because the equations for the Shannon and Simpson’s indices of diversity differs in their sensitivity to evenness and rare species in a sample, both were included in the analysis to capture potential differences in community composition between sites.

When comparing the variables of diversity for individual years against their respective controls, student’s t-tests were performed when the criteria were met. In many cases however, the normal distribution and the variance did not meet the criteria, even after multiple attempts of transforming the data. In those cases, the data was instead tested using a non-parametric Wilcoxon rank sum test. When comparing the three different years against each other, a Kruskal-Wallis test was instead was opted for, to allow for an inclusion of more than two groups. This non-parametric correspondence of an Anova was followed by a Dunn’s multiple comparison test with p-values adjusted using the Benjamin Hochberg method, which is a common post hoc test (Mangiafico 2016). To perform the Dunn’s test in R, the package “FSA” was required. Due to a heterogeneous variance a Welch’s Anova was also performed. However, providing similar results as the Kruskal-Wallis test, no following post hoc test was performed, although the results themselves are included below.

Community composition

To further evaluate the potential effect that the removal of vegetation might have had on the plant communities, Bray-Curtis dissimilarity tests were conducted comparing the species- and plant family community compositions. This was done using the function “vegdist” that is available in in the vegan package in in R, specifying the method as Bray-Curtis.

Ecological indicator values

In continuation, as an alternative to compare functional groups, ecological indicator values (hereafter called EIVs) were applied to study if the restoration had caused a noticeable shift in community composition.

For each site, species were grouped based on their EIV for different environmental traits, after which the relative abundance was calculated and used to create histograms, allowing a visual comparison of how the abundance in each site was distributed over the EVI scale.
The EIVs could for example indicate to what degree a site contain species that are commonly associated with disturbed habitats or point toward a preference in environmental factors on a scale, such as pH or moisture. By comparing sites the intent was to hopefully be able to link potential differences in species distribution along the EIV scale to the conducted restorative measures. To further compare the community composition, Bray-Curtis dissimilarity tests were conducted on this groping, providing a numerical value of how differently species were distributed over the EIV scale for the ecological traits studied.

Ecological traits included were; Biodiversity relevance, Moisture, pH, nutrients (nitrogen and phosphorous), salinity, longevity, seed bank, grazing/mowing and soil disturbance. These factors were selected by their relevance to describe environmental conditions related wetland habitats in general or the restorative measures performed. Here the EIVs used were those provided by Tyler et al. in 2021, which have been adjusted for vascular plants in Scandinavia. Not all species are suitable ecological indicators and species not included in EIV system by Tyler et al. (2021) were here also excluded. In total 5 species were excluded, ranging between 1-3 species in the individual sites.

The trait of biodiversity relevance reflects the amount of biodiversity that a certain species support, where supporting many species return a higher value on a scale from 1-8. Moisture reflects the average moisture/water niche ranging from a preference in very dry conditions to species standing permanently in deep water. Soil reaction (pH), refers to the soil pH measured in the soil where returned values range between 1 and 8. The traits regarding nutrients also refers to the amount of nutrient available in the soil and values indicate preference ranging from low to high nutrient content. The indicator values of salinity reflects a preference of salinity and are based on the species presence along costal lines. Here it is includes as the two nature reserves are quite closely located to the eastern cost of Öland. For grazing and soil disturbance the indicator values reflect how well species thrive under these conditions, which could be comparative with the restoration efforts. The trait of longevity and seed bank refers to the plant's life cycle, ranging from annual to perennial plant species, or in the case of seed bank, the lifespan of the seed in outdoor conditions. All the information of the mentioned ecological traits and how they were calculated are provided in more detail by Tyler et al. (2021).

If a pattern was observed in the histograms, that could suggest that the removal of vegetation might have influenced the community composition, and if supported by the Bray-Curtis test suggesting a substantial dissimilarity, then the difference between groups would be statistically tested. That could be the case if, for example when looking at the trait soil disturbance, scraped sites would show a comparatively higher species distribution at the end of the EIV scale, as it is a potential indication of an alteration in species composition favoured by soil disturbance as a response to the restoration. Statistical testing would either be conducted through an anova or corresponding non-parametric Kruskal-Wallis test.

2.3.2 The Natura 2000 habitats

Initially the intention was to conduct full comparisons on the vegetation recorded in 2019 and 2022, similar to the one performed for the scraped patches. However, due to the limitations in the previously collected data, more simplistic analyses had to be made when studying how the restoration might have affected the two areas of Alkaline fen and potential Molinia meadow habitat. With presence/absence data based on list of characteristic species and species with negative association to the particular habitat types, the data sets were compared by the number of species found on the list for each site.

In the case of the alkaline fen habitat, this would allow us to see whether or not the number of characteristic and negative species had increased or decreased as a result of the reinstated grazing and clearing in the reserves. Moss coverage and vegetation height were however recorded in percentage and centimetres respectively, which in both cases, allowed a Wilcoxon rank sum tests comparing the two time points. The data did not meet the criteria for parametric tests as the normal distribution and homogenous variance were lacking despite efforts of transformation.

For the moist meadow habitat, no previous data was available, so the results from the inventory could only be used as a guide to how much the vegetation correspond with what typically expected for Molinia meadows (EU-code 6410). However, the lack of comparative data removed the limitation of using presence/absence data. With the recorded species abundances both Shannon and Simpson's diversity indices could be calculated in addition to species richness and the average vegetation height and moss coverage. Presented in the results, this will be compared to relevant literature in the discussion.

Ecological indicator values

The EIV system was also applied on the analysis of the two Natura 2000-network habitats, in order to see if, and how well, the recorded vegetation would correspond to the environmental conditions that is characteristic to the specific habitats when in good conditions.

For the potential Molinia meadow it was simply an addition to evaluate if the indicated ecological values would be comparable to the ecology of typical Molinia meadow habitats, while the previously collected data from the alkaline fens, to some extent, also allowed for a comparison to evaluate a potential change in vegetation since the initiation of the restoration.

By assigning EIVs to the species, the recorded community composition in the potential Molinia meadow was compared to the list of characteristic species used in the inventory. As a high presences of characteristic species could be interpreted as a sign that conditions are favourable, their collected range on the EIV scale were here used as a reference of acceptable environmental conditions. In the instance where characteristic specie provides a narrow range of EIVs, this could suggest very specific environmental demands.

If the recorded vegetation largely deviates from this, it could be an indication that the site is far from reaching conditions favourable for Molinia meadow habitats, and perhaps identify environmental factors of specific interest.

As the recorded data and the characteristic list of species are not identical in species numbers, a relative distribution of species was calculated for each value on the EIV scale. Species were grouped by their EIV and counted. To obtain the relative distribution over the EIV scale, the species sum of each group were divided by the total species number. This was repeated for every environmental factor studied.

Histograms were created comparing the relative distribution of species on the EIV scale for the recorded species and the species included in the list of characteristic species. The same procedure was done for the alkaline fen as well, including the previously collected data and also the list of characteristic species as a sort of reference.

By choosing to plot relative species richness against the EIV scale we, to some extent, disregard the large difference in number of recorded species, especially referencing to the different extent of which additional species were recorded between years in the alkaline fen. The relative occurrence would make the two data sets more comparable within individual EIVs, while the number of species recorded could still influence the range of EIVs found at the site each time. Of course the sample method also affects how accurately the data describes the reality.

These problems, although not compensated for in the histogram, were not disregarded in the analysis and is also the reason to why no further statistical analyses were performed. Knowingly, a comparison of these two data sets could not provide reliable results, which will be further discussed later.
However, the intended purpose of the histogram was to provide a visualization of how the plant community is composed that could hopefully give some insight to the current state of the alkaline fen.

In addition to the traits studied in connection to the scraped patches, EIVs for specific habitat types included in the adapted Ellenberg indicator values by Tyler et al. (2021) were applied. Habitat types included were; moist meadow, moist calcareous meadow and rich/calcareous fen. Although these habitat descriptions might not be identically to the Natura2000 definitions, it could be of interest to see how well the vegetation correspond to the alkaline fen and moist meadow described by Tyler et al. (2021).

3. Results

3.1 Scrapings: removal of vegetation

The regrowth in the scraped patches varied, displaying clear edge effects with taller and denser vegetation along the sides, which was especially true for the patch scraped in 2019. Much of the sand layer was still exposed, often in the centre or along the stream bed of the patches. The control sites were fully covered with vegetation that bore signs of grazing. Late in the summer, the habitats were quite dry.

3.1.1 Species accumulation and sampling effort

In total, 43 species were found in the previously scraped areas and their environmental controls. The species accumulation curves were relatively similar between sites, and although not perfect, showed signs of levelling out, suggesting a sufficient sampling effort, even though all species were not likely recorded during this time (See figure 3).

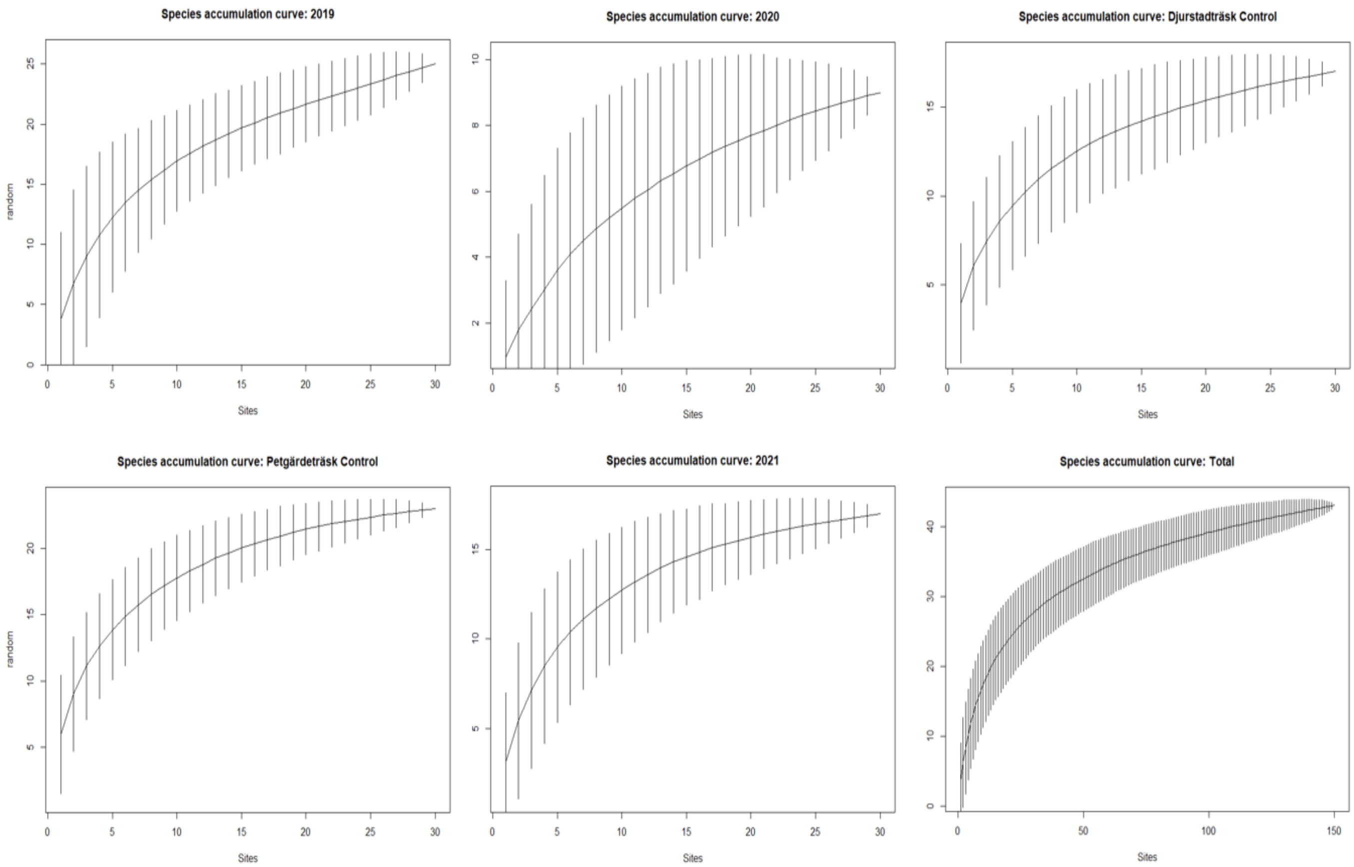

Figure 3. Species accumulation curves of all scraped patches and their controls, as well as a curve based on the compilation of all sites together. With graphs having a similar form between sites and showing signs of levelling out, the results suggests a sample effort that are comparable between sites and has captured much of the plant diversity within the area.

The species number were also quite similar between sites, with the exception of the area that was scraped in 2020, in Djurstadträsk, that had only about half of the species richness, as the other areas. Looking at the estimated species richness, all sites fell close to the lower end of the estimated range that was provided by the species pooling (table 1).

Table 1. Recorded and estimated species richness- *both in total and for each individual site (scraped patch or control). Sites are named with the initial letter indicating place, either Petgärdeträsk (P) or Djurstadträsk (D), followed with the year of scraping or a C for control. n equals sample size.*

Site	Species No	Chao*	Jack1*	Jack2*	Boot*	n	Estimated range SR**
All	43	67.83 (24.08)	52.93 (3.1411)	60.84	47.28 (1.72)	150	~45-92
P 21	17	21.35 (6.96)	19.9 (1.67)	21.80	18.43 (1.12)	30	~17-28
P C	23	24.09 (1.72)	25.9 (1.67)	25.09	24.77 (1.14)	30	~24-26
D 19	25	68.5 (29.2)	34.67 (3.63)	44	28.72 (1.73)	30	~27-97

D 20	9	11.18 (3.3)	11.9 (1.67)	12.90	10.42 (1.31)	30	~9-13
D C	17	20.87 (5.13)	20.87 (1.93)	22.80	18.80 (1.32)	30	~17-26

**Chao, Jack1, Jack2 and boot are functions used to calculate estimated species richness, as previously described in the method. As estimations, the functions does return exact values of species richness, which is why the results presented here are followed by the standard error given in brackets.*

***As the results from the different estimates varies, Estimated range SR (Species Richness) here shows the range of estimated species richness considering the lowest and highest species richness returned of all the functions used in the species pooling.*

3.1.2 Biodiversity

Species richness

The patches scraped in 2021 and 2020 had a lower total species richness (see table 1) and a significantly lower species richness per sample site than their respective controls given the results of the Wilcoxon rank sum tests and the Students T-test (see table 2 and figure 4). In contrary, the 2019 scraping had a higher total species count than the control, with a wider range of the number of species present in the separate samples (see table 1 & figure 4). The control appeared, however, to have a higher general species richness per sample, although this difference lacked statistical significance (table 2).

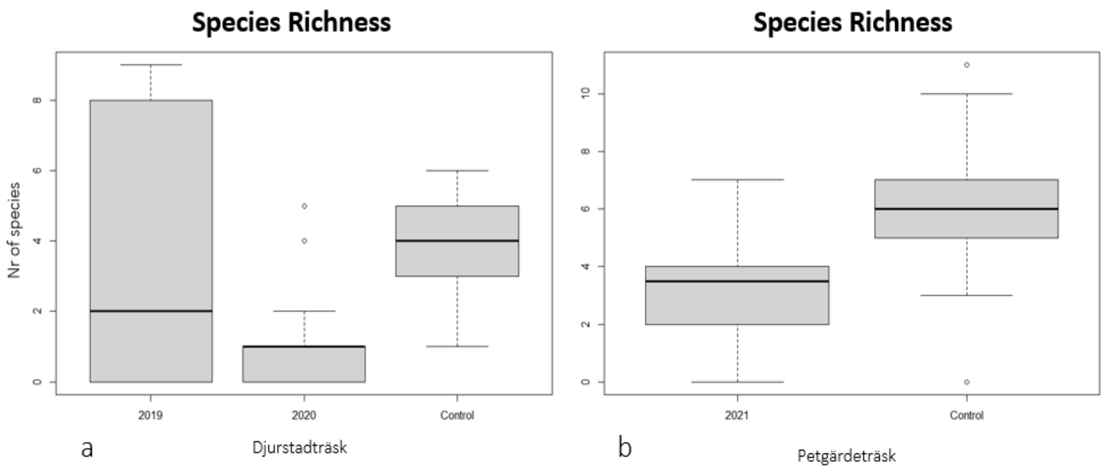

Figure 4. Vizualisation of the varaition in species richness between sites (a-Djurstadträsk, b-Petgärdeträsk), where the year indicate when the scraping was conducted. With one control site per nature reserve, the scraped patches has their corresponding cotrol site located to the right. n=30. The variation seen, especially in the patch scraped in 2019, is most likely reflecting the variation of vegetative recovery within sites, where some samples still had the sand layer exposed exibithing no vegetation, while other samples had multiple plant species established.

Table 2. Species Richness- *Results of statistical tests comparing species richness between scraped patches and control sites.*					
Group	Test	w*	95%-confidence interval	Sample estimates	*P-value*
2021: Control	Student's T-test	t=4.797 df=57.459	1.5149 - 3.6850	mean scraped site:3.23 mean control:5.83	< 0.001
2020: Control	Wilcoxon rank sum test	828	2 - 4	3.0	< 0.001
2019: Control	Wilcoxon rank sum test	507	-1.999 - 2.999	0.999	0.4003

**w- sum of ranks*

When comparing the different years to each other, both the Kruskal-Wallis test (table 3) and the Welch's anova (table S1, appendix 1) indicated a significant different in species richness between sites. But as visualized in figure 4, we don't see a change in species richness that change in one direction, increase or decrease, over time as the patch scraped in 2020 had a lower species richness than both year 2019 and 2021 (figure 4 and table 1). The post hoc test (Dunn's multiple comparison test) showed a significant difference between 2019 and 2020 as well as between 2020 and 2021. Between the first and the last year, no significant difference was found (see table 4).

Table 3. Differences in diversity indices between the years 2021, 2021 and 2019 *- containing the results from the Kruskal-Wallis test comparing scraped patches against each other, studying the recovery of diversity over time. In each patch n equal 30.*			
Variable	Chi-squared	df	*P-value*
SR	16.26	2	< 0.001
Shannon	19.377	2	< 0.001
Simpson	3.5759	2	0.1673

Table 4. Dunn's Multiple comparison test. *Post hoc test accompanying the statistically significant results presented in table 3. The scraped patches were compared to identify where the difference in diversity levels resided. In each patch, n equal 30.*			
Species richness			
Group	z*	*P-value* unadjusted	*P-value* adjusted
2019:2020	3.078	0.002	0.003
2019:2021	-0.717	0.473	0.473
2020:2021	-3.795	0.0001	< 0.001
Shannons index of diversity			
Group	z*	*P-value* unadjusted	*P-value* adjusted
2019:2020	3.415	6.36e-04	< 0.001
2019:2021	-0.697	0.485	0.486
2020:2021	-4.113	3.91e-05	< 0.001

The patches compared are specified under Group, indicated by the year of their scraping.

**z-test statistic*

Shannon and Simpson's index of diversity

For the patch scraped in 2020, both the Shannon and the Simpson's index values were significantly lower than in the corresponding control (see table 5). The patch scraped in 2021 did not differ significantly from its control regarding these measures of diversity.

Looking at the Simpson's diversity index, the Wilcoxon rank sum test provided statistical evidence of a lower diversity for the control in Djurstadträsk when compared to the patch scraped in 2019. For the Shannon index, the diversity between the two sites did not differ significantly (table 5).

By studying the levels of diversity within sites (presented in figure 5 & 6) we see a higher returned value in the case of the Shannon index, as opposed to the Simpsons index of diversity, for the sites sampled in Petgärdeträsk as well as in the control site in Djurstadträsk. This would indicate that more rare species are influencing the level of diversity, rather than a high evenness in the community. For the sites in Petgärdeträsk, we also see a higher variation between samples for the Shannon diversity measure.

This variation is also observed for the patch scraped in 2019, however, the general diversity is higher for the Simpson's index. This would be suggesting that here, although some samples have a diversity of rare species, the diversity level is influenced by an evenness in the community as well.

For the patch scraped in 2020, still displaying the lowest level of diversity among the sites, the Shannon index returned a value close to zero, with a few exception compared to a slightly higher returned value when using the Simpson's index of diversity. This is reflecting the low species number found, indicating a more even community composition of a few more dominant species, with few rare species present contributing to the diversity.

Comparing the diversity between the differently age patches, both the Kurskal-Wallis test and the Welch's anova returned a significant difference (see table 3), with a similar pattern as in the case of Species richness, where the patch scraped in 2020 were showing the lowest level of diversity (visualized in figure 5 & 6). Post hoc testing (Dunn's multiple comparison test) confirmed this and indicated a difference between year 2020 and the other two scraped patches regarding the Shannon index, but again did not find a difference between the patches scraped in 2019 and 2021 (see table 4). Studying the diversity using the Simpson's index, the statistical testing did not provide support that the time passed since scraping has had affected the diversity levels and the null hypostasis of similarity was accepted.

Overall, the differently aged patches show varying results regarding the levels of diversity when compared to their respective controls and each other, not showing a clear pattern in the vegetative regrowth that has occurred since the areas were scraped.

Table 5. Diversity indices- Shannon and Simpson's measure of diversity
Effects of restoration on vegetation. Results of statistical tests comparing the level of diversity between patches and control sites.

Group	Test	w*	95%- confidence interval	Sample estimate	*P-value*
Shannon index of diversity					
Group	Test	w*	95%- confidence interval	Sample estimate	*P-value*
2021:Control	Wilcoxon rank sum test	558	-0.0436 0.5840	0.2309	0.110
2020:Control	Wilcoxon rank sum test	797.5	0.6914-1.1015	0.8872	< 0.001
2019: Control	Wilcoxon rank sum test	497	-0.335 0.7087	0.1679	0.486
Simpson's index of diversity					
Group	Test	w*	95%- confidence interval	Sample estimate	*P-value*
2021: Control	Wilcoxon rank sum test	415.5	-0.1251 0.0905	-0.022	*0.626*
2020: Control	Wilcoxon rank sum test	443.5	-0.311 0.3494	-2.572e-05	< 0.001
2019: Control	Wilcoxon rank sum test	242.5	-0.352 -0.0948	-0.233	0.00214

Group specifies which patch is tested and a significant p-value supports a difference in diversity. n equal 30.
**w- sum of ranks*

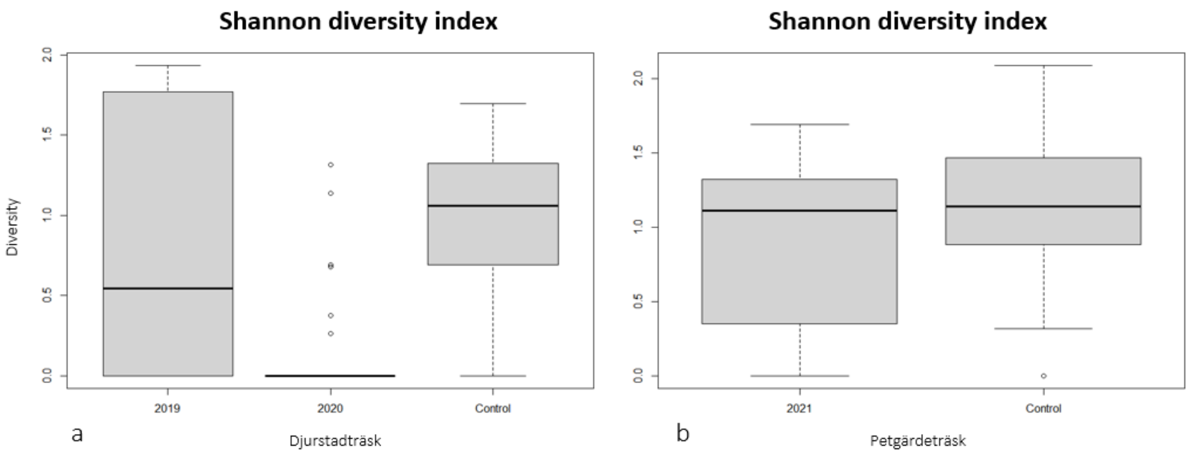

Figure 5. Vizualisation of the varaition in diversity between sites (a-Petgärdeträsk, b-Djurstadträsk), according to the Shannon index of diversity, where the year indicate when the scraping was conducted. Each patch has their corresponding cotrol site located to the right. n=30. Again, the varaition seen here is probably related to the of stage regrowth seen in the different samples. As the Shannon index is more senstive to the number of rare species in a sample, the extremly low value and lack of variation for the patch scraped in 2020 (a), is most likely due to the low number of recorded species there.

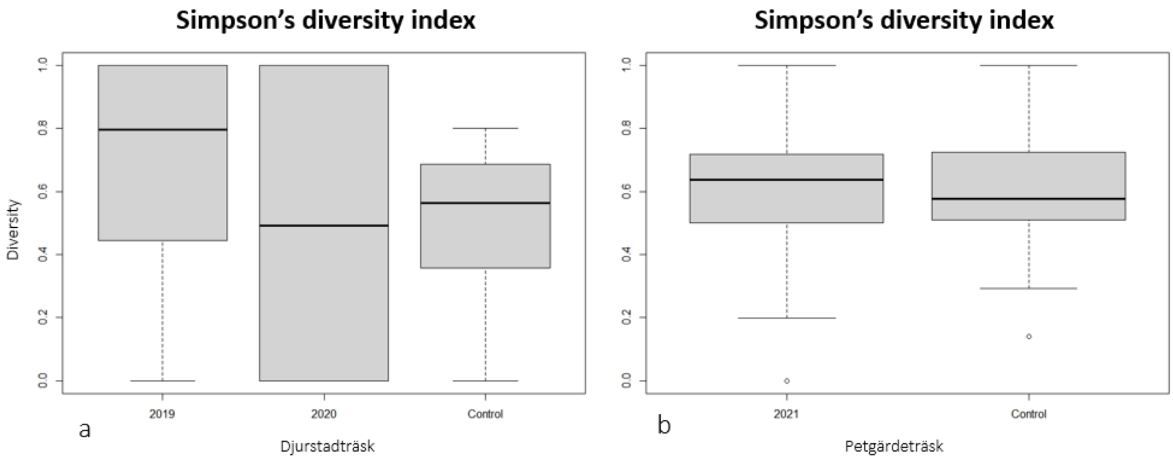

Figure 6 . Vizualisation of the varaition in diversity between sites (a-Petgärdeträsk, b-Djurstadträsk), according to the Simpson's index of diversity, where the year indicate when the scraping was coducted. Each patch has their corresponding cotrol site located to the right. n=30.

3.1.3 Community composition

The Bray-Curtis dissimilarity tests were conducted in order study and compare other aspects of the plant community between patches, such as species and plant family composition. The results can be seen in table 6 with the species and families unique to one sample site collected in table 7.

Table 6. Bray-Curtis dissimilarity test
Indicated difference in community composition (species and family composition) between sample sites, provided by the Bray-Curtis dissimilarity test. Values can range between one and zero, increasing the more dissimilar two communites are. A zero would indicate that two samples are identical.

Species composition				
	P-21	P-C	D-20	D-C
P-C	0.6903323			
D-20	0.8233890	0.8918455		
D-C	0.7726839	0.7200223	0.8581081	
D-19	0.6872964	0.7108578	0.6981627	0.6589928
Plant family composition				
	P-21	P-C	D-20	D-C
P-C	0.67522			
D-20	0.7804	0.8643		
D-C	0.7306	0.5013	0.822	
D-19	0.6482	0.6004	0.68503	0.5266

The table functions as a matrix comparing sites. The patches are identified by their location, Petgärdeträsk (P) or Djurstadträsk (D), followed by the year they were scraped (-19,- 20 or -21) or a C for control.

The Bray-Curtis test returns a number between zero and one, where a one indicates that two communities are very different. So in terms of species composition all patches have a moderately high dissimilarity. Some species, such as *Phragmites australis*, were commonly found throughout the reserves which would explain some of the similarity, but all patches had at least one species that were unique to that site, contributing to a dissimilarity as well. Difference in relative abundance also contributes to the dissimilarity seen. When grouped in plant families we saw a small decrease in dissimilarity but it was occurring across all sites and is thus relatively proportional to the species composition. Fewer families than species were unique to specific patches, with the patch scraped in 2019 having the highest number of exclusive species and plant families (table 7).

Table 7. Community composition- *Compilation of vegetation data, with species and plant families unique to a sample site presented. Marked in bold are species and families with single sample occurrences.*

Number of unique species		
Site	No	Species
P-21	3	*Chenopodium album,* ***Cirsium sp.,*** *Oxybasis rubra*
P-C	6	*Agrostis canina,* ***Carex flacca,*** *Eleocharis palustris, Odonites vulgaris,* ***Sparganium erectum, Taraxacum sp.***
D-20	1	*Utricularia vulgaris*
D-C	3	*Carex panacea,* ***Filipendula ulmaria,*** *Selinum carvifolia*
D-19	7	***Alisma plantago-aquatica, Iris pseudacorus, Menyanthes trifoliate,*** *Lathyrus palustris,* ***Poaceae sp,*** *Thylpteridaceae sp., Thalictrum flavum*
Number of unique plant families		
Site	No	Family name
P-21	1	*Amaranthaceae*
P-C	1	*Orobanchaceae*
D-20	1	*Lentibulariaceae*
D-C	0	
D-19	6	***Alismataceae,*** *Fabaceae,* ***Iridaceae, Menyanthaceae,*** *Ranunculaceae, Thelypteridaceae*

The sites are identified by their location, Petgärdeträsk (P) or Djurstadträsk (D), followed by the year they were scraped (-19,- 20 or -21) or a C for control.

3.1.4 Ecological indicator values

As an alternative to studying functional diversity, the relative species abundance was studied by assigning species EIVs and comparing the distribution over the EIV scale between sites. In most cases, the scraped patches, based on their recorded vegetation, showed a similar range of distribution over the EIV scale when compared to their respective controls. The results for the environmental factors of soil moisture, nitrogen, grazing/mowing and soil disturbance are visible in figure 7 & 8. Histograms of the remaining ecological traits (biodiversity relevance, salinity, pH, phosphorous, seed bank and longevity) and results from the dissimilarity tests are found in the appendix.

Regarding nitrogen, the vegetation in all the scraped patches covered a slightly wider range of EIVs, compared to their respective controls, which was further supported by a higher dissimilar reported from the Bray-Curtis dissimilarity test (table S2, appendix 1). For the patches located in Djurstadträsk, the wider range came from recorded species abundance in the lower end of the EIV scale, while the patch in Petgärdeträsk had recorded species with higher EIV values.

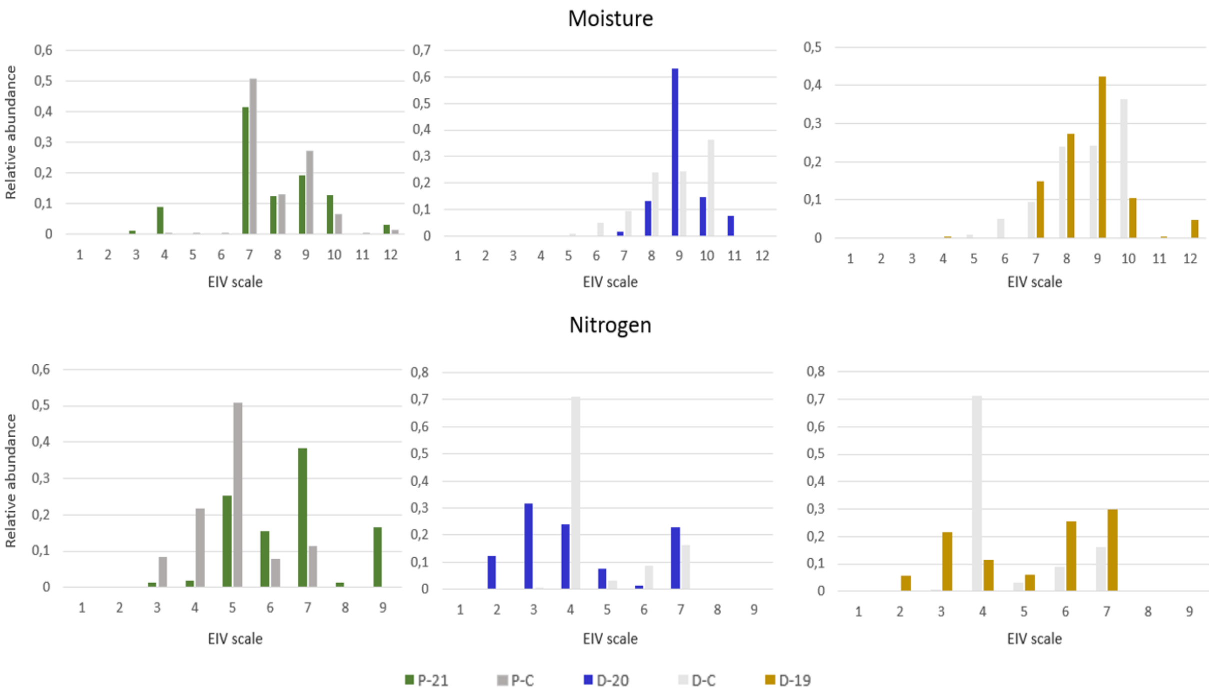

Figure 7. The relative abundance of species recorded in scraped patches (coloured) distributed over the EIV scale of the environmental traits of soil moisture and nitrogen. For moisture the distribution was relatively similar, when compared to their control (grey), suggesting a similar functional composition. For nitrogen the scraped patches showed a wider distribution over the scale, perhaps indicating that the re-established vegetation is not as niched as the in the control site.

Comparing the scraped patches against each other (figure 9), the distribution of species abundance over the EIV scale were quite similar with the occasional exception where the patch scraped in 2021 showed a larger range, as in the case of moisture, or a similar range but distributed further towards the end of the scale, as in the case of nitrogen indication. The dissimilarity test did not point towards large difference in distribution and as no patterns were seen that indicated a consistent shift in vegetation (table S2, appendix 1) composition over time, these were not further investigated.

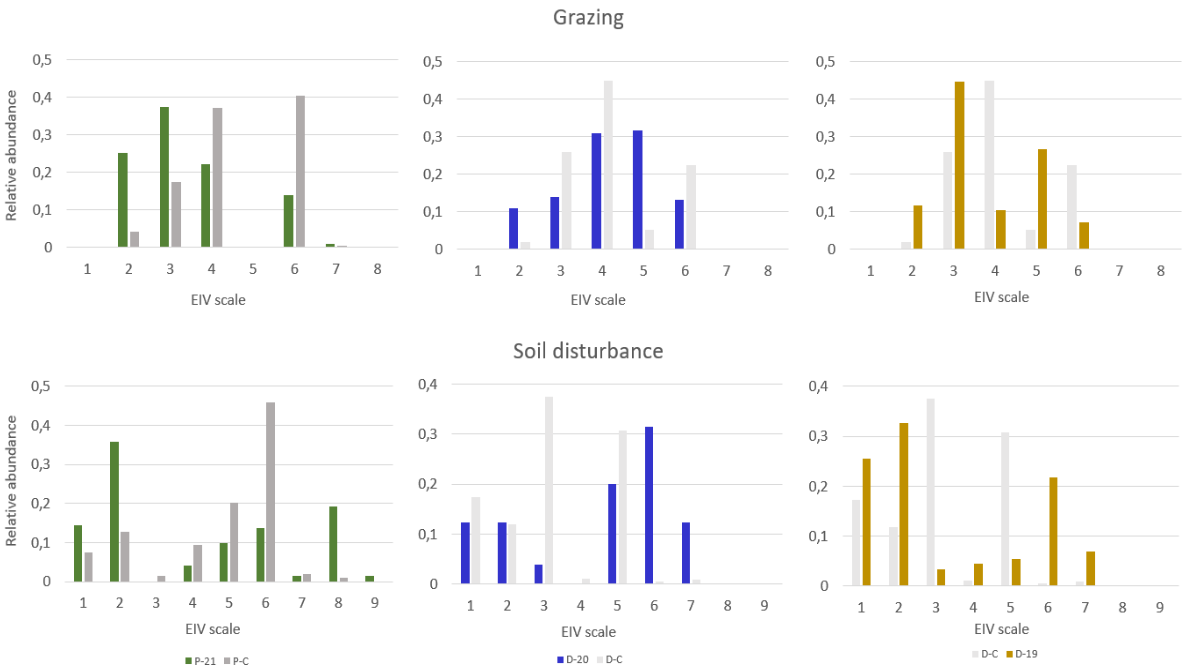

Figure 8. The relative abundance of species recorded in scraped patches (colour) distributed over the EIV scale of the environmental traits of grazing/mowing and soil disturbance. For both traits, although the relative abundance sometime differed, the range of distribution were quite similar when compared to the reference vegetation in corresponding controls (grey).

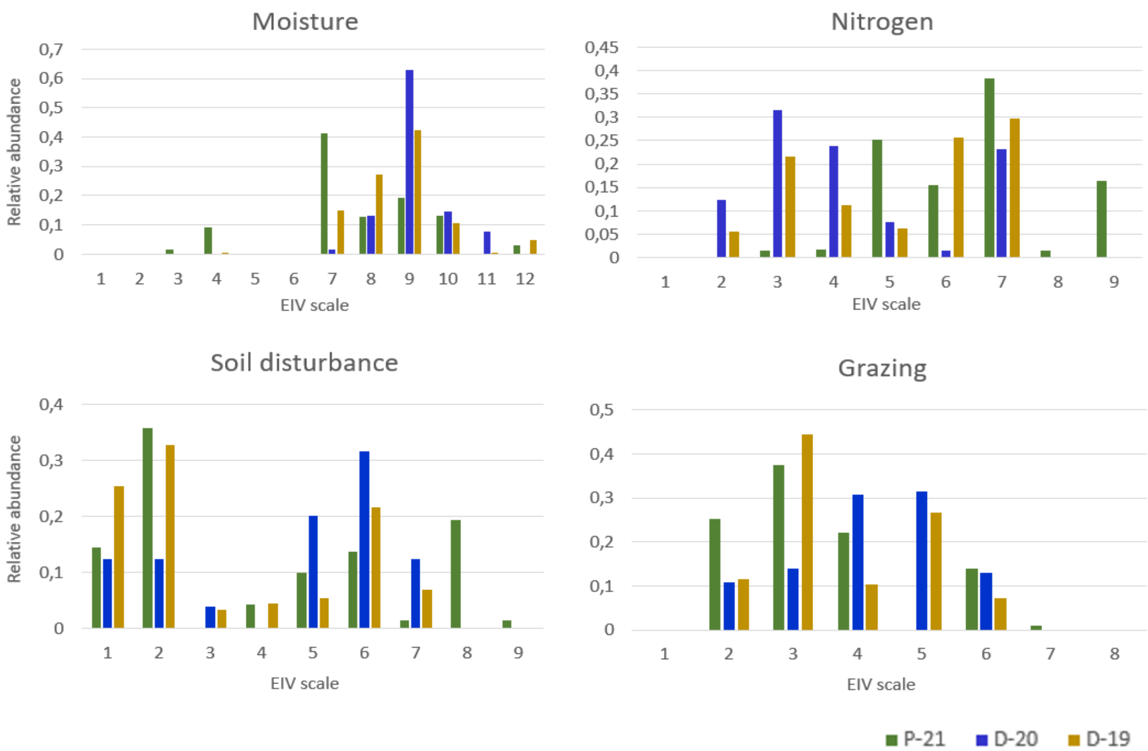

Figure 9. A comparison of the scraped patches where the relative abundances of species recorded were distributed over the EIV scale of four environmental traits. Both the range and the relative

distribution was similar in the case of Soil moisture, disturbance and grazing. For nitrogen the distribution varied some, especially between the patch scraped in 2021 (P-21) and 2020 (D-20).

3.2 Natura 2000-habitats

3.2.1 The Alkaline fen (EU-code 7230)

The same negative indicator species recorded in 2019 were found in 2022 without the species number having changed. However, no characteristic species were recorded, including those observed in 2019. In total, 46 vascular plant species were observed, 43 being additional species (table 8). In the previous inventory in 2019, brown mosses were also identified to species levels, something that was not done in 2022. Moss coverage, was however recorded both times.

Table 8. Alkaline fen- *Current and previous results of the inventory of the alkaline fen. The list of negative and characteristic species were provided by Länsstyrelsen in Kalmar län. Numbers given within brackets include moss species.*

Year	Nr of negative indicator species recorded	Nr of characteristic species recorded	Total nr of species recorded inc. additional species
2019	3 /7	4 (5)/ 28(56)	7 (9)
2022	3/7	0/ 28 (56)	46

Comparing change in vegetation, a significant increase of both brown moss coverage and vegetation height were detected (table 9 and figure 10 below). To no surprise, the statistical test found no real change in the recorded negative indicator species from 2019 to 2022. The 3 characteristic species found in the previous inventory was enough to provide a statistically significant difference in the Wilcoxon rank sum test. However, the p-value was only 0.023, without including the previously recorded mosses.

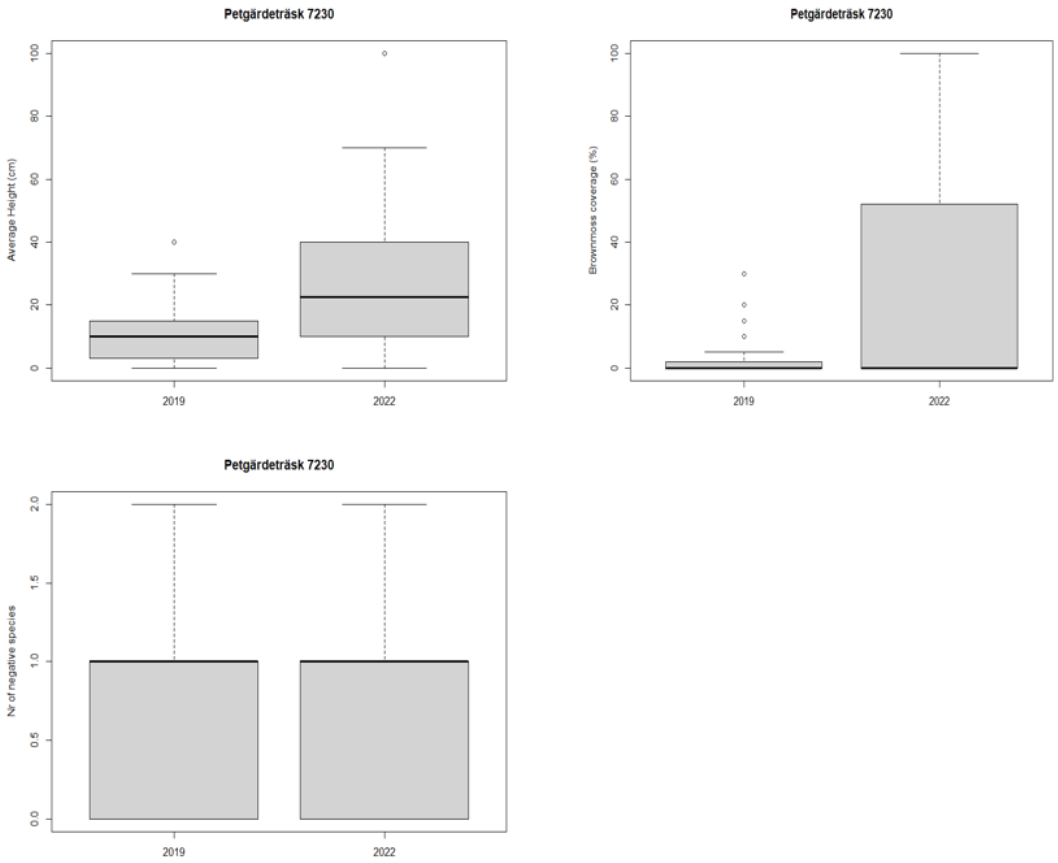

Figure 10. Visualization of parameters of the vegetation recorded in 2019 and 2022. Parameters include the number of characteristic and negative species recorded, vegetation height as well as moss coverage, where both moss coverage and height had increased since 2019, while the presence of negative indicator species had remained.

Table 9. Vegetation change in Alkaline fen from 2019 to 2022

Statistical testing of the vegetative parameters confirming the increase of moss and height from 2019 to 2022. As expected, no statistical difference was found regarding the recoded number of negative species as it remained unchanged since the previous inventory. No characteristic species was recorded in 2022 which resulted in a significant decrease, here seen with a P-value of 0.023. n equal 50.

Variable	Test	w*	95 % confidence interval	Sample estimate	*P-value*
Brown moss coverage (%)	Wilcoxon rank sum test	869	-1.199e+01 - -4.060e-05	-1.611338e-05	0.0026
Height (cm) Sq transformation	T-test	t = 5.016 df =89.083	-2.899 - -1.254	mean in 2019 2.833548 mean in 2022 4.910350	< 0.001
Height	Wilcoxon rank sum test	641.5	-20.00 - -5.00	-13.00004	< 0.001
Negative species	Wilcoxon rank sum test	1239.5	-3.334e-05 - 3.163e-06	-3.813546e-06	0.9392
Charecteristic species	Wilcoxon rank sum test	1375	0 - 0	0	0.023

**w- sum of ranks*

Ellenberg indicator values

When studying the distribution of species over the EIV scale, the recorded plant community was often widely dispersed as seen in figure 11. In comparison, the characteristic species range was a bit more narrow, although still having quite a wide distribution for many ecological traits. The largest discrepancy between recorded and characteristic species was observed in the case of pH indication. All characteristic species prefer and indicate more alkaline soil conditions, while the recorded species was dispersed along the scale, with the highest relative species distribution closer to values indicating neutral conditions. In the case of nitrogen, characteristic species then to indicate and be favoured by less nutrient rich environments, while some the recorded species indicate more nutrient rich conditions, perhaps being an indication of eutrophication. Regarding soil disturbance and grazing the distribution pattern of species were similar, with the recorded vegetation not quite indicating disturbance/ grazing to the same extent as characteristic species. The remaining environmental traits can be found in the appendix.

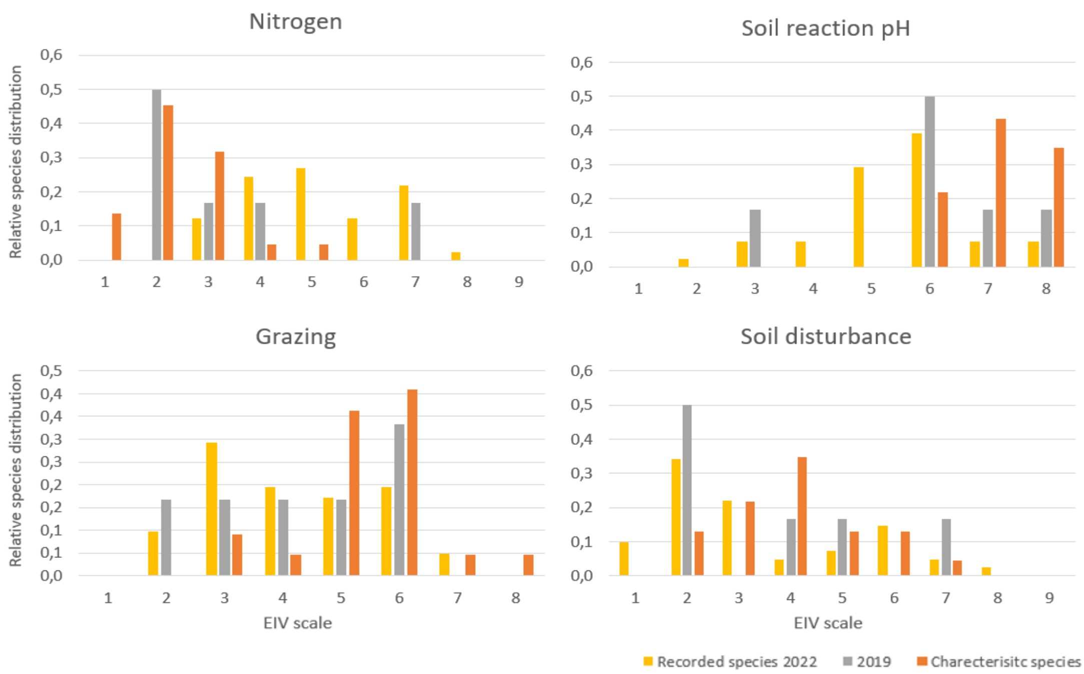

Figure 11. Results from the relative distribution of species over the EIV scale of various ecological traits. Provides an indication of the current state of the alkaline fen, how it might differ from 2019, and by comparison to the list of characteristic species, also how it may differ from conditions that could be considered favourable. As seen the range is quite similar between groups with the exception for Nitrogen and soil pH, where the recently recorded species had a much wider range of distribution.

3.2.2 The potential Molinia Meadow (EU-code 6410)

In total 31 species pf plants were recorded, but only one were included in the list of characteristic species, while 11 negative indicator species were found (table 10). As presented in figure 12, the average species richness was calculated to around 4.4, with a maximum of 7 species being recorded in one sample. Studying the Shannon and Simpson's diversity indices, Shannon returned a higher average at around 1, emphasizing richness rather than evenness amongst the species. The average vegetation height was 74.5 cm with the lowest and highest value recorded being 40 and 145 cm. No mosses were observed in any of the samples and were therefore not included in the table below.

Table 10. Molinia meadow (EU-code 6410)
Collected vegetation data for the potential Molinia meadow, where closer to half of the recorded species are considerd to be negative indicator species and only one characteristic species were observed.

Year	Negative indicator species recorded	Nr of characteristic species recorded	Total nr of species recorded inc. additional species	Average Vegetation height
2022	11/35	1/41	31	74.5 cm (40-145cm)

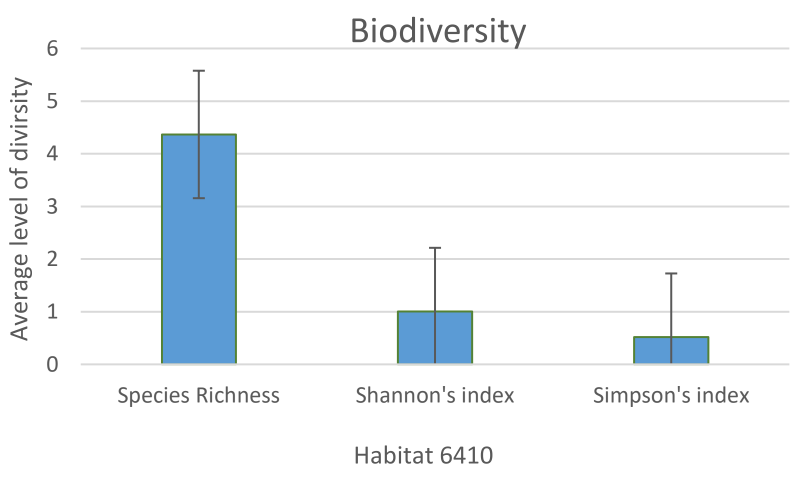

Figure 12. Average level of diversity per sample for the vegetation recorded in the potential Molinia meadow.

Ellenberg indicator values

When compared to the list of characteristic species, the vegetation recorded in the potential Molinia meadow did differ some in how the relative species abundance

was distributed over the EIV scales (Figure 13). The characteristic species often displayed a species distribution over a wide range of the scale, for instance in the case of nitrogen, suggesting that the level of nitrogen might not be the largest determining factor to why these certain species are absent or why the condition of habitat could be called unfavourable.
However, in the case of nitrogen, we also see a higher relative distribution at the lower end of the scale, indicating that a majority of the characteristic species prefer less nitrogen rich environments. Meanwhile the recorded vegetation was more distributed on the middle/higher end of the EIV scale suggesting a species composition in favour of more nitrogen rich conditions. This could be an indication that nutrient levels are a little above the optimum for Molinia meadow habitats, although interpretations of ecological indicators and their applied use will have to be carefully considered before conclusions can be drawn.

A similar pattern, where the relative species distribution of the recorded vegetation is dispersed differently than the characteristic species collection, is also seen for the ecological trait of Grazing. Here a large proportion of the recorded species are collected at the lower/ middle of the EIV scale, while most characteristic species are collected at higher EIV values, perhaps indicating that the habitat in not yet effected by grazing/mowing to the extent characteristic species prefer. Although soil disturbance is also an ecological trait to which grazing cattle contributes, this pattern was not repeated in the case of soil disturbance. Ecological factors not included in figure 13 are found in the appendix.

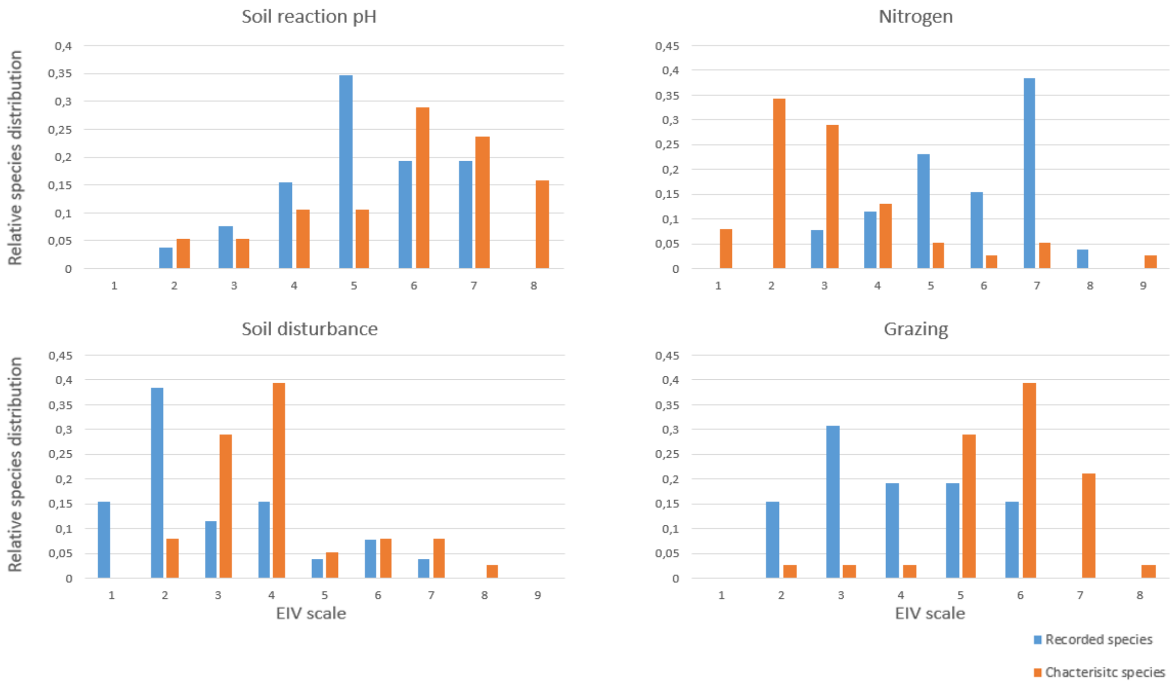

Figure 13. Comparison of vegetation and their indication of ecological traits. The x-axis represents the EIV scale of an ecological trait, with the relative distribution of species from either the recorded vegetation (blue) or the list of species characteristic species for Molinia meadow habitats (orange).

4. Discussion

4.1 Scrapings: removal of vegetation

As indicated by the Wilcoxon rank sum tests of the three indices of diversity, it was only the patch scraped 2020 that showed a significantly lower level of diversity in all three cases when compared to its control. In the other two patches, scraped in 2019 and 2021, the results showed both higher and lower levels of diversity to no significant difference at all, depending on what index you study. Due to this, removal of vegetation cannot be said to have a certain influence, neither positive nor negative, on the level of plant diversity in this case.
The highest species count was found in the patch scraped in 2019, despite an insignificant result in the Wilcoxon rank sum test when compared to the control site in Djurstadträsk. This could likely be explained by the high number of samples in the scraped patch with no recorded vegetation at all. If only looking at samples where vegetation was recorded, it is possible that the patch scraped in 2019 would show to inhabit a more rich and diverse plant community.

With a purpose of evaluating the regrowth of vegetation over time, the patches scraped in different years were also compared against each other. Here, the patch scraped in 2020 showed a significantly lower level of diversity in comparison to the other two patches, for both the species richness and the Shannon diversity index. No significant difference was found between 2019 and 2021 for either of those diversity indices. For the Simpsons measure of diversity none of the scraped areas differed significantly. Again, the results does not show a clear regrowth that would suggest a succession of vegetation over time. For that you would expect species count and diversity increasing with time or perhaps develop to be more similar to the control patches. However, land use might of course influence the regeneration of vegetation and as three years is not a very long time, a progression as hypothesized might be more evident in time.
However, the overall lack of a pattern in how the vegetation had re-established might also be explained by chance, as only one patch of each age was studied. In this case there were other patches available, but not always of the same size and

age. When also wanting to study a time line of regrowth with a balanced sample effort in regards of area, I was therefore limited to only one sample site per year. Preferably, would have been to visit multiple patches of the same age and habitat type in order to draw more reliable conclusion and to rule out random occurrences.

In addition to the statistical analyses, observations were made during the inventory, which made this patch stand out in comparison to the other two scraped surfaces studied. Firstly, there was a clear succession and gradient in vegetation height from the longer side edge towards the other side, where the water stream would normally be (patch was rectangular, see map in figure 1). This is probably linked to dispersal patterns where the distance to the remaining population commonly is correlated to the re-establishment of the plant community (Sundberg 2012, Morimoto *et al.* 2017).

Being a long but narrow patch, this would be most visible on the longer sides, however, the watercourse might prevent regrowth on the one side, while also contributing to a moisture gradient or a condition that is too moist for some of the species. All samples that were recorded having zero vegetation were closer to where the watercourse would normally run earlier in the year. A lot of dried up algae, *Charophyceae*, were covering the ground here, to some extent continuing in the patch scraped in 2020. *P. australis, Schoenoplectus lacustris,* and *Utricularia vulgaris* were the only species recorded in the actual streambed in Djurstadträsk.

The succession was also observed in the patches scraped in 2020 and 2021, however more subtle and without a large height difference.

One explanation to this could be the presence of grazing cattle could which of course tend to lower vegetation levels but further disturbance from the hooves on the sandy soil might also have affected the regeneration to some extent by trampling young seedlings.

In Petgärdeträsk, it was noted that the cow herd particularly favoured the patch scraped in 2021 and spend much of the days resting in the relatively small surface. With the amount of cattle present, it likely added pressure and an element of disturbance in addition to only grazing which might contribute to damage to the emerging vegetation and perhaps in the long run slow down regrowth. In Djurstadträsk, cow prints were found in the patch scraped in 2020, but the surface did not seemed to be used to the same extent as in Petgärdeträsk. The oldest patch, were however not accessible for the cattle, and was thus not subjected to grazing, something that could partially have contributed to the higher number of species counted for.

I would also want to further discuss the patch scraped in 2019 as it, to some extent, differed from its control. This patch was located in an area classified as a calcareous fen. Due to a very dense and inaccessible vegetation, the control points were instead set surrounding the patch scraped in 2020 as seen in map 1. Although connected, a shift in vegetation occurred between the sites and directly adjacent to the calcareous fen is a Molinia meadow habitat. Still being adjacent to the patch scraped in 2019 and containing many of the same species, this from a practical stand point, seemed as an acceptable alternative. However, given density and knowing that the vegetation of calcareous fens can be quite homogenous, it is possible that a control site set in this habitat type would have a lower biodiversity that the control recorded in Djurstadträsk. If comparing the vegetation in the patch scraped 2019 to a more suitable reference, it is possible that a significantly higher level of biodiversity would be found as a response to restoration effort.

When restoring habitats one might also have to define what makes a good reference. Here of course, we want to use a comparable habitat but althought not scraped, the nearby vegetation might still be affected by restoration efforts indirectly by, for example, an alteration in hydrology. If the goal is to see an improvement in vegetation, a damaged, but untreated wetland habitat with similar ecological conditions, might be a better reference. Although, such references are not always available.

Community composition

The results from the Bray-Curtis dissimilarity tests indicated a difference in community composition although some species were commonly shared between all sample sites. These results were also comparable on a plant family level. Species unique to one patch were present in all sample sites, which is one explaination to the dissimilarity seen between communities. As the Bray-Curtis test further includes the relative abundance of species, this also contributed to the results as a species common in one patch, although not unique, might have been occurring rarely in other sites.

When disturbance is commonly known in ecology to create openings for opportunistic species to establish, it would not have be surprising to see an increase of species that is not normally found in wetland habitats. Over time these species would be expected to be out competed by species more suited for the local conditions (Zedler & Kercher 2005, Kreyling et al. 2021).
Here some species, such as *Oxybasis rubra* and *Chenopodium album* were found in the patch scraped in 2021 that are not normally associated to wetlands and that were not present elsewhere in the nature reserves. However, despite their absence in the other two patches, it is unlikely that this would be due to being out competed,

as the sand layer was still partially exposed in all sites with uncomplete vegetative regrowth. The unique species and plant families found in the other two scraped patches were most often common to wetland habitats. The hypothesis that more opportunistic species, favoured by disturbance, would be found in the patches that were scraped more recently was therefore not supported by the evidence in this study.

Restoring vegetation: outlook

Although it might be too soon to say how exactly the plant communities will develop after the ongoing restoration, the vegetation, when looking at the recoded species, seems to re-establish similar to that adjacent untreated communities. According to current literature, this would not be unexpected (Zedler & Kercher 2005, Schnoor *et al.* 2015, Morimoto *et al.* 2017). With the main purpose being to restore hydrology, it is hard to say whether or not such a reestablishment would be positive in this case. On the one hand, the patches are only small portions of their individual habitat types and will alone not determine the condition of the habitat and, if favourable, reestablishment of adjacent vegetation should not be considered negative. On the other hand, if the adjacent vegetation contain competitive dominant wetland species, then perhaps a more active management is needed.

In the patch scraped in 2020, *P. australis*, was commonly recorded in the samples. Although likely, it is unclear if this species was located along the watercourse of the scraped patch before restoration or if it has established after the topsoil removal. Either way, being a negative indicator species for Molinia meadows further establishment should perhaps be avoided. In the alkaline fen in Petgärdeträsk, including the patch scraped in 2021, common rush was comparatively more common, and although heavily grazed, preventing reestablishment in scraped patches and decrease the already existing population might require even larger efforts. In the patch scraped in 2019, located in a calcareous fen type habitat, common rush was already established to some extent. However as the vegetation in calcareous fens is dominated by saw sedge, also a tall growing species, the habitat quality might be relatively unaffected by the implications that normally associated to rush establishing, for instance in the regard of light availability. At least as long as the rush does not outcompete the sedge and other characteristic features that comes with that type of vegetation.

In addition to the effects of top soil removal, restoring the hydrology alone can induce regeneration of wetland species (Large *et al.* 2007, Kreyling *et al.* 2021). Therefore, any results from this study could also be a response to changes in the moisture gradient.

As seen by Kreyling *et al*. (2021), restorative measures does often lead to shifts in vegetation but does not guarantee a development towards the vegetation and functionality targeted. Regarding the aim of promoting characteristic wetland species, it is impossible to say if the target vegetation will ever be recovered naturally, as establishment can be limited by dispersal issues in addition to the environmental requirements. Current restoration efforts normally only apply to the latter (restoring environmental conditions). To reach the conservation goals set for vegetation, a more active management might be required to avoid establishment of the "wrong" flora and where longevity, seed bank and dispersal possibilities should be incorporated and considered when planning restorative actions.
However, in some cases when reviewing literature, more drastic measures might be a solution. Due to the challenges with restoring vegetation, Smolders *et al.* (2008) argue that topsoil removal might be essential to reset vegetation allowing for the target species to establish while also removing additional nutrients that is otherwise very hard to combat.

4.2 The Natura 2000 habitats

4.2.1 Alkaline fen

When evaluating the conditions of the alkaline fen, there was a significant difference in the number of characteristic species, with fewer species found in 2022. However, the timing of the inventory might have influenced the probability of finding the previously reported species characteristic to alkaline fens. *Gymnadenia canopsea, Epipactis palustris, Carex oederi* and *Carex heleonastes* were registered in the inventory 2019, which was conducted in May. In the beginning of August, which was when the inventory was carried most recently, it would have been past the flowering stage making it harder to detect and identify these species. Without complementary research earlier in the season, these characteristic species cannot be assumed lost. Seasonal fluctuations might also explain the difference in vegetation height seen between 2019 and 2022. Vegetation height is of course dependent on grazing pressure and species composition, but many species might have had time to grow taller later in the season. During the inventory in 2019, restorations had already begun, so grazing and clearing of vegetation could further have resulted in a relatively low height being recorded.
Moss coverage had instead increased, although during the last inventory much of the coverage was made up of dry, less dense moss. An increased moss coverage could be a positive indicator of restoration, however, considering the other results it is too soon to make assumptions in that regard.

4.2.2 Potential Molinia meadow

With only one characteristic species and 11 negative indicator species observed, the habitat evaluated as a Molinia meadow have a long road ahead before the condition could be called favourable. As Molinia meadows are habitats strongly affected by grazing, longer time might have to pass before we see a development in the right direction. During the weeks of field work it struck me that the cattle never visited this area of the reserve, and little traces of them could be seen. So although grazing is technically carried out, it is uncertain how much grazing pressure is put in this particular area and how it varies over the season. If not enough, additional clearing or mowing might be necessary to aid the restoration resulting in the desired vegetation shift.
Looking at the distribution of species on the EIV scale regarding soil disturbance and grazing further supports the idea that more grazing is needed before characteristic species find the habitat suitable.

4.3 Ecological indicator values

The intention of using EIVs was to further investigate community composition and how it might have responded to the restoration efforts made. The application of Ecological indicators, such as Ellenberg values, is a thoroughly discussed topic, which will not addressed here in great extent. However, based on available literature and guidelines provided by Diekmann (2003) and Tyler *et al.* (2021) amongst others, weighted averages is the most common way to use EIVs. Without a full species inventory, the recorded vegetation would in this case not have provided true average values. In addition to also lacking comparative data, this method was therefore opted against for both the Natura 2000-habitats evaluated and the scraped patches. The method used instead, studying the relative species distribution along the EIV scale, provided an insight in community composition. This was seen as an alternative to functional diversity, as species on different ends on the EIV spectrum probably have quite different niches.

As characteristic species are already used as reference of good conditions in Natura 2000 habitats, I don’t see a problem in employing that idea in combination with EIVs, although it is important to understand that this is a theoretical scenario and that in reality, vegetation in favourable conditions would be comprised of additional species which would affect the relative distribution of species over the EIV scale. Therefore one should be careful interpreting these results as evidence and view them more as indications to support other observations. Further I would think that narrow range of distribution among target species would be a better indicator of the environmental condition as a wide range contributes with uncertainty and would require additional research.

The fact that vegetation recorded in both of the evaluated Natura 2000-habitats showed similar patterns of distribution, regarding the traits of grazing and nitrogen when compared to the characteristic species, could suggest that the state of the nature reserve has not yet reached optimal conditions. When revising literature, it could be decades before environmental demands are met and recovery of the vegetation is seen (Smolders *et al.* 2008, Schnoor *et al.* 2015, Kreyling *et al.* 2021). It could also be that there is a delayed response in vegetation, as suggested by Diekmann (2003) and that a community shift will eventually be apparent, better reflecting species favoured by the conservation. As time pass the predictive powers of the plant community will improve and time will further have to tell if the restoration efforts are successful.

In addition to the ecological traits, the vegetation's correspondence to certain habitat types were studied using the EIV system provided by Tyler *et al*. (2021). The recorded vegetation matched poorly to these habitat types. Due to the low correspondence to both the habitat types described by Tyler *et al*. (2021) and those included in Natura 2000-network, no effort will be made discussing the potential difference in these definitions.

As for the scraped patches, community composition and distribution along the EIV scale were similar for most of the ecological traits when compared to their respective controls. Signs that the distribution would change over time was not evident either. This was supported by the Bray- Curtis dissimilarity tests, where high dissimilarity were rarely and inconsistently indicated. Overall, the dissimilarity was greater when comparing species and family composition between the sample sites than when studying the species distribution along the EIV scales. The lower dissimilarity in this case might suggest that the sites had quite a similar functional diversity.
Although the sample sites were located in a quite close proximity, due to differences in habitat types, ecological traits such as pH and nutrients, should be compared carefully as variation may occur naturally. Factors, such as soil disturbance, that could reflect effects of the restoration effort would perhaps be more informative. However, as this study did not produce any results in this regard, a long term evaluation would perhaps be more insightful.
Because new vegetation take time to establish after fast environmental changes, and species might linger some time in condtions that are no longer suitable, EVIs can be less accurate in young habitats (Diekmann 2003). Again, with this in mind, more time would also benefit the predictions and use of ecological indicator values.

4.4 General discussion

With specific environmental requirements vegetation can be a good indicator of ecological status. However, as discussed by Schnoor *et al*. (2015), reestablishments of the vegetative community is influenced by multiple factors and stochastic events. So, to fully understand the effects of restoration it is crucial to perform evaluations that include a broader perspective of the ecosystem and its functionality that might extend beyond out point of interest. On this topic I want to shortly address the standardized methods provided for evaluation of habitats included in the Natura 2000-network.
By only using a select few species, as negative or positive indicators, we limit the evaluation and our understanding of how conservation affects functionality and diversity beyond the target vegetation. Biodiversity and functionality could potentially shift without affecting the number of characteristic or negative indicator species. With this, we risk a having changes in the ecosystem go undiscovered, contributing an uncertainty in regards to the effects of restoration on a larger scale. Including a larger perspective have, amongst others, been adressed by Hambäck *et al.* (2023) discussing the importance of including multiple functions and different groups of taxa when studying the effects of wetland restoration.

By saying this, the advantages with the standardized methods used today are not disregarded. A standardized method, as the one applied for the Natura 2000-network, allows for long term, frequent comparisons to be made all over Europe, which are practically feasible in terms of budgeting and work effort etc. However, a supplement with total species inventories could contribute with valuable information that would be otherwise missed. Further, I would personally want to insist on conducting a more extensive evaluation before initiating restoration. In this case, no inventories were conducted previous to the restoration and therefor the true response to the conducted measures will never be known. This study well exemplifies where the limitation that come with insufficient data collection in itself highlights the problems of producing reliable evidence.

Conducting multiple conservation actions simultaneously also makes it more difficult to assess and evaluate the effect of individual restorative measures. Here, to some extent, grazing could be over looked as both the control and the scraped site were subjected to the treatment. Preferably the patch scraped in 2019 would have been included in this as well. Observed shifts in vegetation as a response to scraping of the top soil layer would however be an additive effect to the response of the other restoration efforts. Extracting information about effects observed in the field, to apply to other conservation situations becomes more difficult as multiple restorative measures are combined. Research investigating individual measures could be hugely beneficial to future management of wetlands.

With the results from long-term studies, such as Klimkowska *et al.* (2019), Gómez-Baggethun *et al.* (2019) and Kreyling *et al.* (2021), one could potentially argue that restoring wetlands to former states of natural conditions is impossible. If true, we should perhaps ask ourselves what it is we want to conserve. High biodiversity is generally considered a positive feature, but if the target species composition cannot be sustained in the current wetland ecosystems, would it not be better to focus on maintaining any kind of biodiversity or perhaps instead focus on preserving functionality?
The lack of success in past and current restoration projects might also be reflecting inadequate conservation efforts where we either need to direct more resources towards conservations or revaluate our goals and better tailor the measures after realistic goals.
Either way, our lack of knowledge surrounding the practical conservation measures and their effects on the ecosystem, presents a challenge when planning and conducting successful conservational work.

4.5 Conclusion

In order to make specific improvements in the condition of wetland habitats more efficiently and to better predict the outcome of restorations, more focus needs to be directed towards investigating the effect of individual measures and their effects on the ecosystem as a whole.

In this study, three years after initiated restoration, no apparent shift in vegetation had occurred in favour of the characteristic species of the target vegetation and the occurrence of negative indicator species was still prominent. Further, the vascular plant communities establishing after top soil removal, showed similarities to that of the adjacent vegetation in terms species composition and functionality, although not being fully recovered and showing variation in the level of diversity and community composition. No general patterns were discovered that would suggest a vegetative shift as a direct response to the restorative measures conducted. Time will instead have to tell if restorations are successful. Although inconclusive results, this study could provide material for future evaluations at this site, something that is often missing in conservation projects.

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Appendix 1

Species lists:

Scrapings:
Vegetation removal
Agrostis canina
Agrostis stolonifera
Alisma plantago-aquatica
Argentina anserina
Bidens tripartita
Bulboschoenus maritimus
Carex elata
Carex flacca
Carex oederi
Carex panicea
Chamaenerion angustifolium
Chenopodium album
Cirsium sp.
Cladium mariscs
Comarum palustre
Eleocharis palustris
Filipendula ulmaria
Galium palustre
Hydrocotyle vulgaris
Iris pseudacorus
Juncus articulatus
Lathyrus palustris
Lycopus europaeus
Lysimachia vulgaris
Lythrum salicaria
Mentha aquatica
Menyanthes trifoliata
Odonites vulgaris
Oxybasis rubra
Phragmites australis
Plantago major
Poaceae sp.
Salix sp.
Schoenoplectus lacustris
Scutellaria galericulata
Selinum carvifolia
Sium latifolium
Sparganium erectum
Taraxacum sp.
Teucrium scordium
Thalictrum flavum
Thylpteridaceae sp.
Utricularia vulgaris

The Alkaline fen:
Achillea millefolium
Agrostis canina
Agrostis stolonifera
Anthriscus sylvestris
Argentina anserina
Bolboschoenus maritimus
Briza media
Bromopsis inermis
Calamagrostis canescens
Carex elata
Carex flacca
Carex panicea
Carex pseudocyperus
Carex sp.
Centaurea jacea
Chamaenerion angustifolium
Cladium mariscs
Comarum palustre
Eleocharis palustris
Galium palustre
Hydrocotyle vulgaris
Iris pseudacorus
Juncus articulatus
Lotus corniculatus
Lycopus europaeus
Lysimachia vulgaris
Lythrum salicaria
Mentha aquatica
Molina caerulea
Odonites vulgaris
Phalaris arundinacea
Phleum pratense
Phragmites australis
Plantago major
Poa pratensis
Poaceae sp.
Potentilla reptans
Ranunculus sp.
Rumex crispus
Salix sp.
Schoenoplectus lacustris
Sesleria uliginosa
Sparganium erectum
Teucrium scordium
Typha angustifolia

The Molinia meadow:
Achillea millefolium
Agrostis canina
Alopecurus pratensis
Argentina anserina
Calamagrostis canescens
centaurea jacea
Chamaenerion angustifolium
Cirsium arvense
Dactylis glomerata
Daucus carota
Deschampsia cespitosa
Filipendula ulmaria
Galium palustre
Lysimachia vulgaris
Mentha aquatica
Milium effusum
Phalaris arundinacea
Phleum pratense
Phragmites australis
Plantago lanceolata
Potentilla reptans
Ranunculus sp.
Rubus sp.
Salix sp.
Selinum dubium
Taraxacum sp.
Thalictrum flavum
Thalictrum simplex
Urtica dioica
Valeriana officinalis

Scrapings: complete vegetation removal

Table 1S. Results from the Welch's Anova analysing the differences in diversity indices between the years 2021, 2021 and 2019*. Complementary to table 3, presenting the results from the Kruskal-Wallis test studying vegetation recovery over time.*				
Variable	*P-value*	F	numerator df	denominator df
SR	8,122e-07	18.76	2.000	50.421
Shannon	2.088e-07	20.992	2.000	51.901
Simpson	0.2267	1.525	2.000	55.91

Ecological indicator values

By assigning recorded species their Ellenberg values for various ecological traits, the relative abundance distribution over the EIV scale was compared between sample sites. Below are the graphs containing the remaining ecological traits investigated that were not included in the results section above.

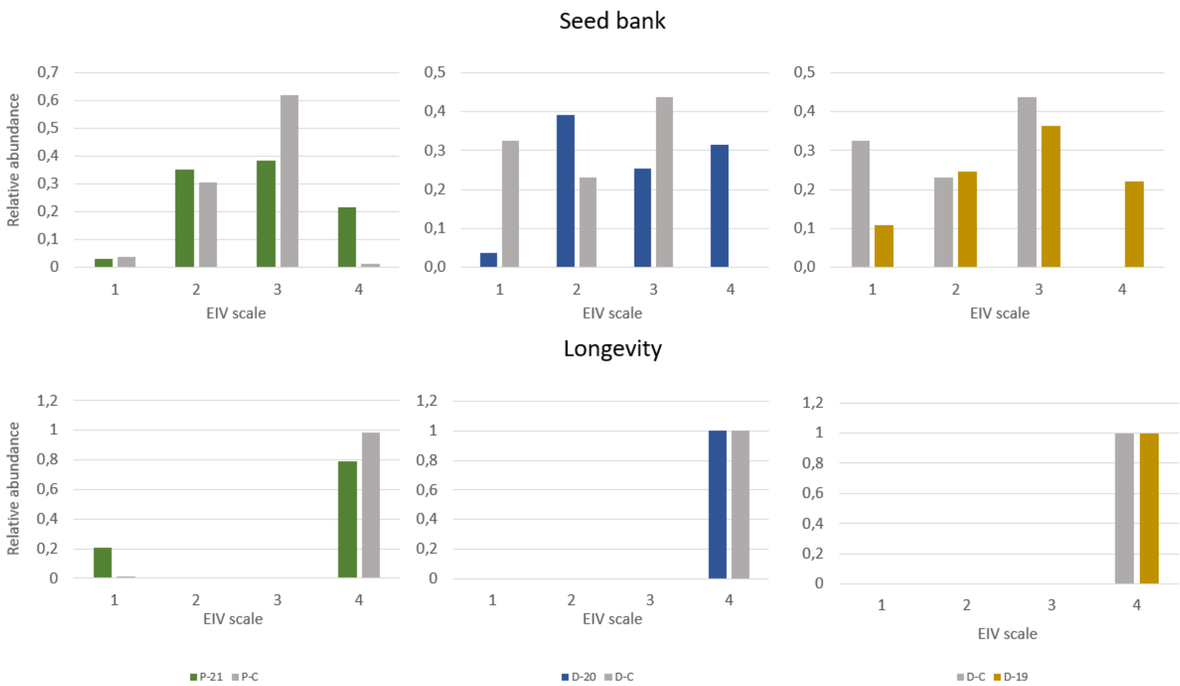

Figure S3. Regarding the ecological factors of Longevity and Seed bank, all sample sites had a similar range and relative distribution.

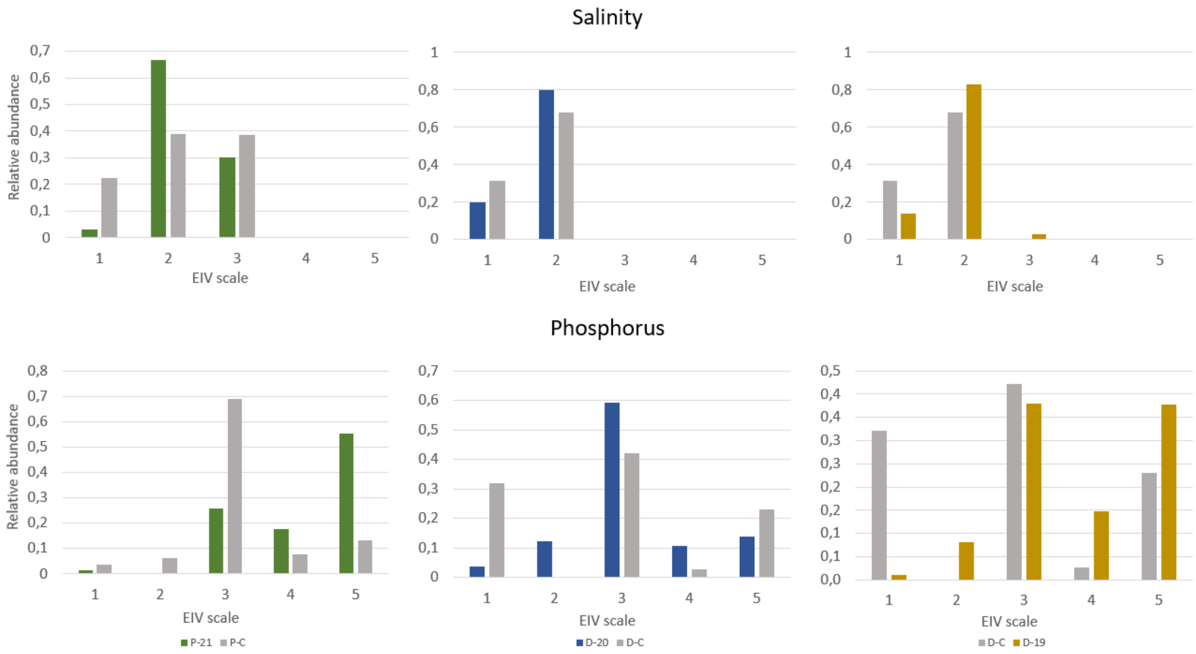

Figure S4. In the case of salinity and Phospohrus content, all sample sites had a similar range of distribution while the relative abundance had variation in the categorical EIVs, especially when looking at phosphorus. Comparing the scraped patches, the patches scraped in 2019 (yellow), and 2020 (Blue) had the most similar distribution patterns.

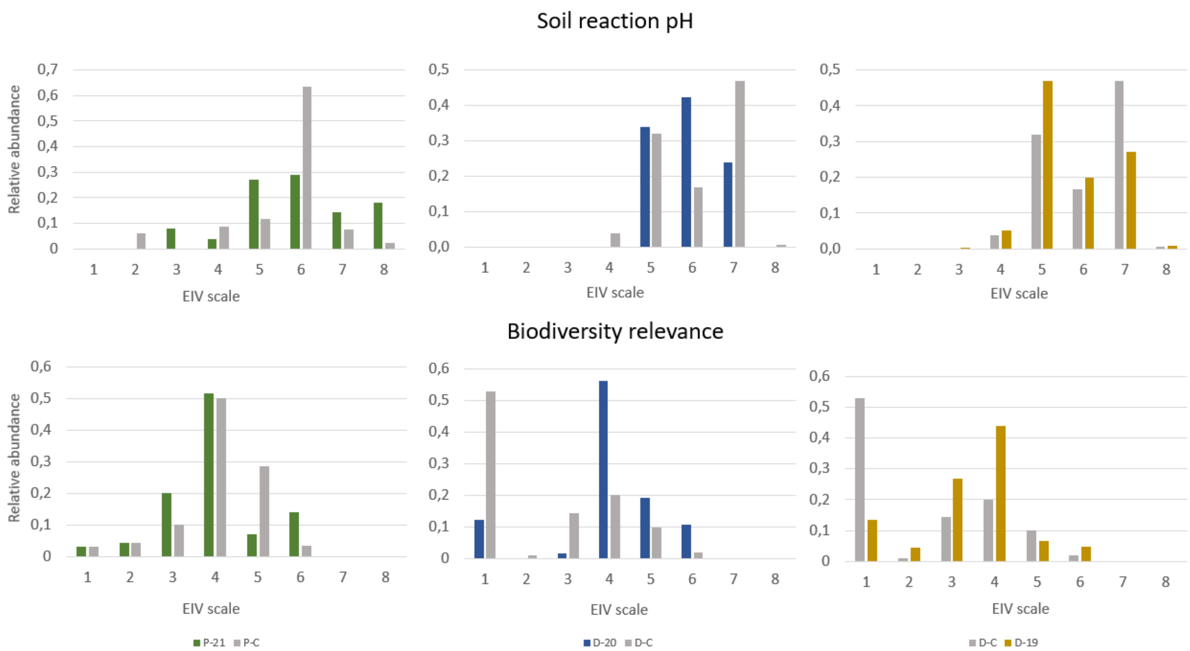

Figure S5. Once again the patches scraped in 2019 (yellow) and 2020 (blue) had quite similar distribution patterns that also corresponded quite well to the control. The patch scraped in 2021 (green) and its control (P-C) had a wider range of distribution in the predictions of pH, while sharing a similar patterns to the other sample sites regarding Biodiversity relevance.

Table 2S. Distribution of the Relative species abundance along the EIV scale *Results of the Bray-Curtis dissimilarity testing between sample sites, based on the community composition when species had been assigned Ellenberg indicator values. A value closer to 1 indicates higher dissimilarity. Communities with dissimilarities greater than 0.5 are marked in bold*			
Seedbank			
	P-21	D-20	D-C
P-C	0,250		
D-20	0,140		
D-C		0,475	
D-19	0,108	0,218	0,271
Moisture			
	P-21	D-20	D-C
P-C	0,182		
D-20	**0,535**		
D-C		0,465	
D-19	0,393	0,323	0,318
Soil Disturbance			
	P-21	D-20	D-C
P-C	0,497		
D-20	0,485		
D-C		0,485	
D-19	0,283	0,355	**0,596**
Grazing/Mowing			
	P-21	D-20	D-C
P-C	0,417		
D-20	0,402		
D-C		0,354	
D-19	0,335	0,314	0,498
Nitrogen			
	P-21	D-20	D-C
P-C	**0,525**		
D-20	**0,645**		
D-C		**0,548**	
D-19	0,455	0,307	**0,600**
Biodiversity relevance			
	P-21	D-20	D-C
P-C	0,218		
D-20	0,259		
D-C	0,527	**0,543**	
D-19	0,175	0,310	0,430
Soil reaction pH			
	P-21	D-20	D-C
P-C	0,456		
D-20	0,298		
D-C	0,375	0,274	
D-19	0,339	0,224	0,199
Phosphorus			
	P-21	D-20	D-C
P-C	**0,520**		
D-20	0,484		

D-C	0,472	0,374	
D-19	0,205	0,281	0,351
Salinity			
	P-21	D-20	D-C
P-C	0,277		
D-20	0,302		
D-C	0,296	0,118	
D-19	0,271	0,061	0,174
Longevity			
	P-21	D-20	D-C
P-C	0,197		
D-20	0,207		
D-C	0,207	0,000	
D-19	0,207	0,002	0,002

The Natura 2000-Habitats

Alkaline fen (EU-code: 7230)

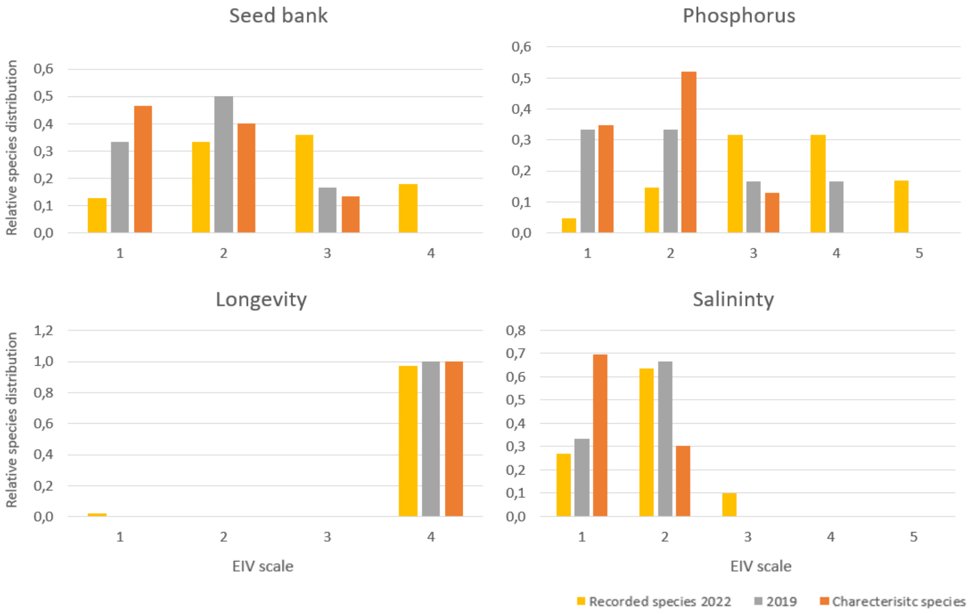

Figure S6. Results from the relative distribution of species over the EIV scale of various ecological traits. Provides an indication of the current state of the alkaline fen, how it might differ from 2019, and by comparison to the list of characteristic species, also how it may differ from conditions that could be considered favourable. As seen the range is quite similar between groups with the exception for Phosphorus where the recorded vegetation have a wider distribution range that the listed characteristic species.

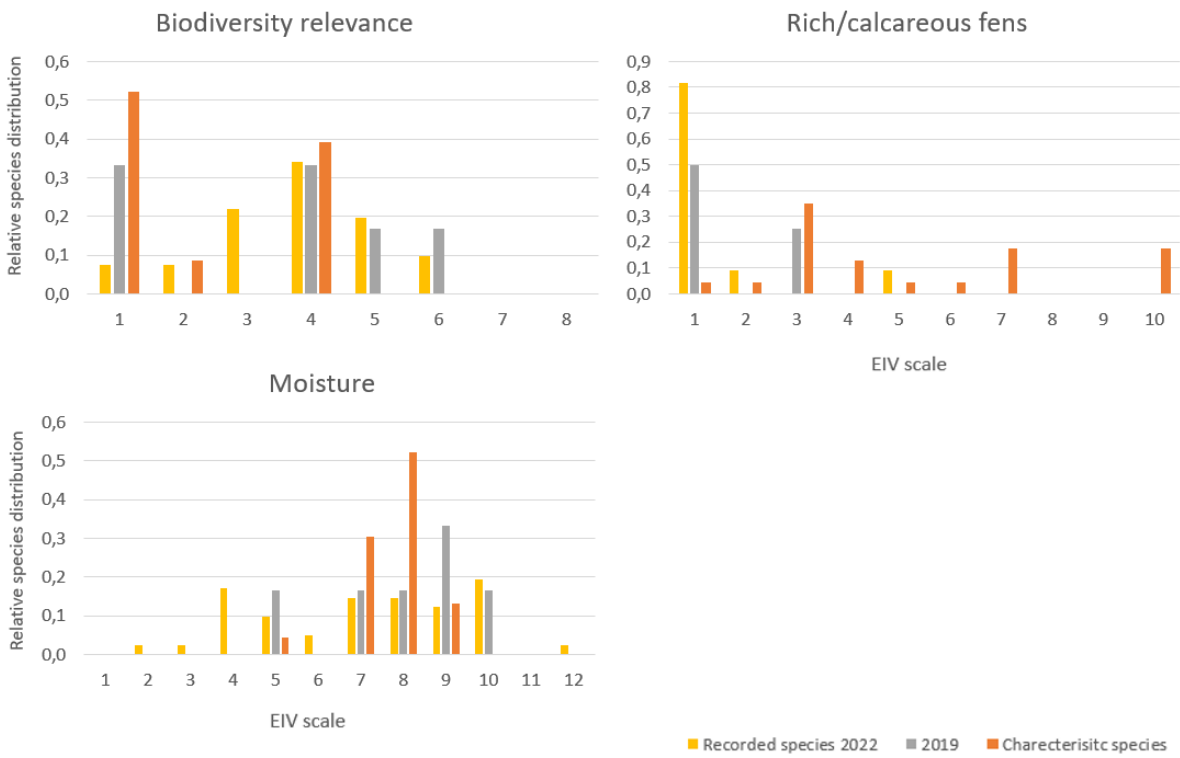

Figure S7. Results from the relative distribution of species over the EIV scale of ecological traits and a habitat type. Provides an indication of the current state of the alkaline fen, how it might differ from 2019, and by comparison to the list of characteristic species, also how it may differ from conditions that could be considered favourable. The recorded vegetation corresponded poorly to the distribution of characteristic species. The habitat type did not correspond well to either of the tested species distributions.

Potential Molinia meadow (EU-code: 6410)

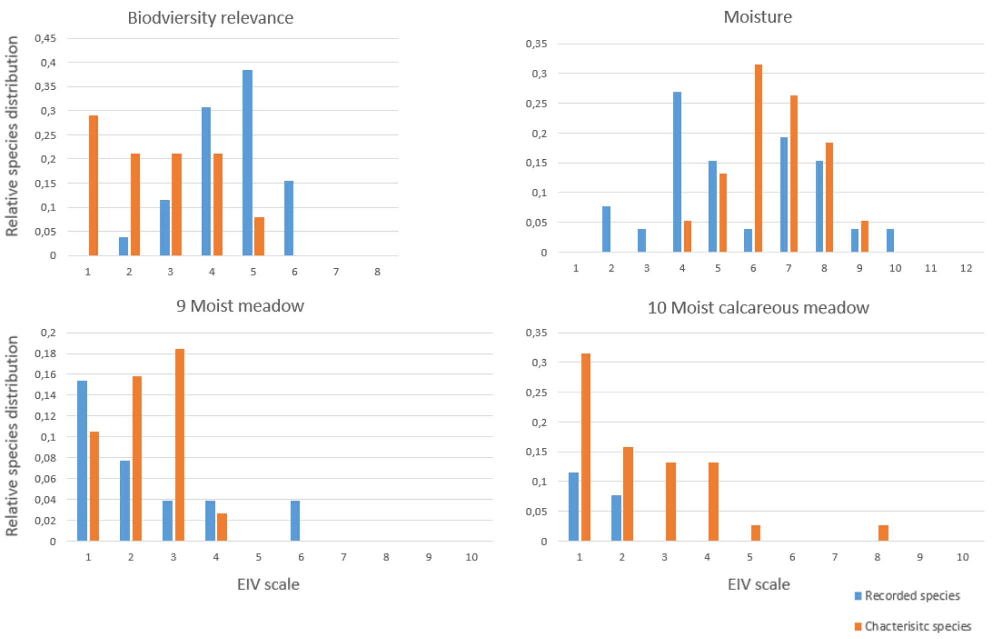

Figure S8. Comparison of vegetation and their indication of ecological traits. The x-axis represents the EIV scale of an ecological trait, with the relative distribution of species from either the recorded vegetation (blue) or the list of species characteristic species for Molinia meadow habitats (orange). In addition to the two ecological traits are also an indication of correspondence to two types of habitats. Both the recorded vegetation and the characteristic species corresponded poorly to the habitat types.

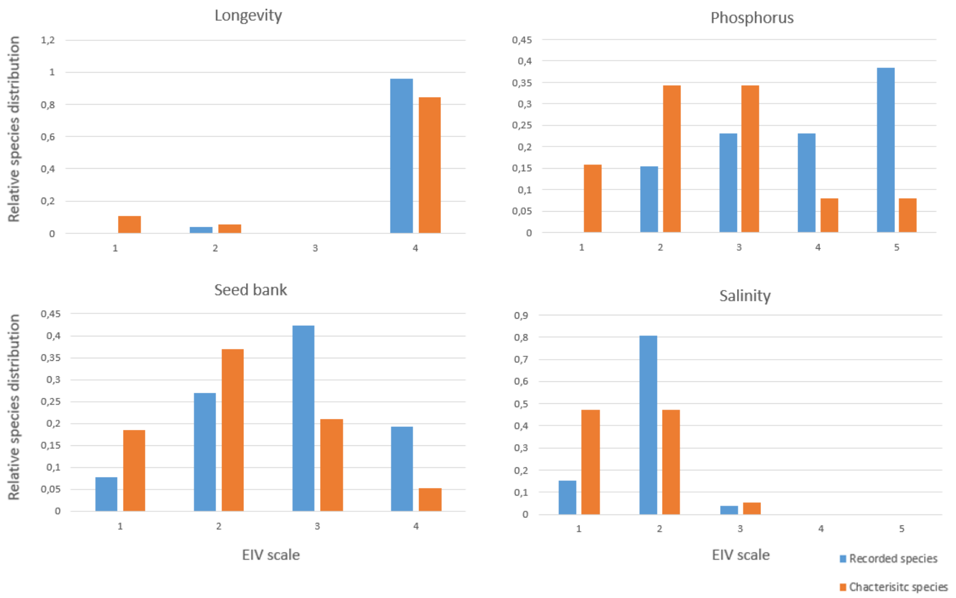

Figure S9. Comparison of vegetation and their indication of ecological traits. The x-axis represents the EIV scale of an ecological trait, with the relative distribution of species from either the recorded vegetation (blue) or the list of species characteristic species for Molinia meadow habitats (orange). Both the groups showed a similar range of distribution although the relative species abundance varied some.

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