Reintroduction of the European bison (*Bison bonasus)* in Almindingen on Bornholm

- Impact on biodiversity of vascular plants over a 5-year period from 2012-2017

Master's Thesis – 60 ECTS points

By: Bjarke Andersen Schäfer

Submitted on: 4 September 2018

60 ECTS point Master's thesis in Biology

Author:
Bjarke Andersen Schäfer

Title/Subtitle:
Reintroduction of the European bison (*Bison bonasus)* in Almindingen on Bornholm;
Impact on biodiversity of vascular plants over a 5-year period from 2012-2017.

Titel/Undertitel:
Reintroduktion af den europæiske bison (*Bison bonasus)* i Almindingen på Bornholm;
Effekt på biodiversiteten af karplanter over en 5-årig periode fra 2012-2017.

Subject description:
This study investigates the impact on biodiversity of wild plants in Almindingen on Bornholm, when exposed for grazing activity by the European bison (*Bison bonasus*) under semi-natural conditions for five years. Through vegetation analyses, the effect is determined and discussed in order to evaluate bison grazing as a management tool for maintaining and enhance biodiversity in a mosaic landscape.

Senior advisor:
Rita M. Buttenschøn, Senior advisor
Section for Forest, Nature and Biomass
Department of Geosciences and Natural Resource Management
University of Copenhagen

Academic advisor:
Lasse Gottlieb, PhD student
Section for Forest, Nature and Biomass
Department of Geosciences and Natural Resource Management
University of Copenhagen

Submitted:
4 September 2018

Place:
University of Copenhagen
Faculty of Science
Department of Geosciences and Natural Resource Management

Cover photo by:
Per Frost, the nature agency, Bornholm. European bison in Almindingen in 2014.

Abstract

Knowledge about the ecological effects from grazing by the European bison (*Bison bonasus*) is marginal. However, the successful reestablishment of the species has led to a curiosity of its potential as a management tool for nature restoration. This provides the opportunity to enlarge the methods of inhibiting the rising biodiversity crisis, while at the same time benefitting the further restoration of the species.

In 2012, seven European bisons were introduced to a 200 ha large enclosure in Almindingen, on the Danish island Bornholm. The intention was to investigate the effects on the biodiversity of vascular plants in the area over a 5-year period, in the hope of creating a more dynamic mosaic landscape with the formation of more light-open spaces into the closed woodland, to favour increased biodiversity.

This study was set to investigate these possible effects through registration of plant species composition, browsing, bark striping and vegetation structures in 130 circle plots, by three identical measurements in the summer of 2012, 2014 and 2017.

Overall, the investigated area contained 13 different habitat types, whereas three were considered open landscape and the remaining ten was forest habitat. Through statistical analyses, two open landscape habitats and three forest habitats showed a significant increase in species richness over the 5 years. Furthermore, seven of the habitats showed significant changes in Ellenberg indicator values of either light or nitrogen, whereas three of the affected habitats was open landscape and the remaining four was forest habitat. The study also revealed a measureable browsing activity of the most common and abundant tree species in the area, where the European bison preferred trees with the height of 0-50 cm. The study did not found any quantifiable bark striping activity in the circle plots.

Previous studies have investigated both food preferences and behaviour of the European bison, but this is the first to examine and document specific changes in flora biodiversity of vascular plants. The results evidences that the European bison can affect and alter both open land and forest vegetation after 5 years of continuous grazing. The results thereby forms a base for further discussion, whether the European bison can give a new contribute to the Danish nature management practices than the native herbivore fauna.

_____

Resume

Den nuværende viden omkring de økologiske effekter af græsning fra den Europæiske bison (*Bison bonasus*) er begrænset. Men den succesfulde reetablering af arten, efter den nær var uddød, har givet anledning til at undersøge dens potentiale i et naturforvaltnings perspektiv. Det kan give nye muligheder for at bremse den stigende biodiversitetskrise, samtidig med at man giver nye muligheder for den europæiske bison til at reetablere sig og øge den samlede population.

I 2012 blev syv Europæiske bisoner udsat i et 200 ha stort indhegnet område i Almindingen på den danske ø Bornholm. Formålet var at undersøge effekten på biodiversiteten af vilde planter over en 5-årig periode, i håbet om at skabe et mere dynamisk mosaik landskab, med flere lysåbne steder i det lukkede skovlandskab. Dette studie undersøgte udviklingen gennem arts-registrering, bidpåvirkning, barkskrælning og vegetationsstrukturer i 130 dokumentationscirkler, gennem tre identiske målinger i somrene 2012, 2014 og 2017. Det undersøgte område bestod af i alt 13 forskellige habitattyper, hvor tre af dem var lysåbne og de resterende ti var skovhabitat. Fem af disse områder viste en signifikant stigning i antal arter over den 5-årige periode, hvor to af disse områder var lysåbne arealer og tre var skovhabitat. Ydermere viste syv af områderne en signifikant ændring i Ellenberg indikator værdier for lys eller kvælstof, hvor tre af de påvirkede områder var lysåbne og de resterende fire var skovhabitat. Resultaterne påviste også en målbar bidpåvirkning af de mest hyppige løvfældende vedplanter i området, hvor træer med højden 0-50cm var mest udsat for bid. Studiet fandt ingen kvantificerbar barkskrælning på træerne i dokumentationscirklerne.

Tidligere studier har beskæftiget sig med både fødepræferencer og adfærd af den europæiske bison, men dette er det første studie til at undersøge og dokumentere ændringer i biodiversiteten af vilde planter. Resultaterne viser at den europæiske bison kan påvirke vegetationen i både lysåbne landskaber og skove efter 5 år kontinuerlig græsning.

Disse undersøgelser danner derfor et grundlag for den videre diskussion, om hvorvidt den europæiske bison kan give et nyt bidrag til den danske naturforvaltning i forhold til den nuværende fauna af herbivorer.

_____

Table of content

Preface and acknowledgements

This 60 ECTS-point Master's thesis serve as a finale evaluation of a 5-year project, set by the Danish nature agency. The study investigates the biodiversity of vascular plants in a closed mosaic landscape, in Almindingen on the Danish island Bornholm, when exposed to grazing by the European bison (*Bison bonasus*) over a 5-year period. By field surveys of plant and tree species and vegetation structures, the outcome are examined. With the existing knowledge of the subject in mind, the results are analyzed and discussed in order to give a novel contribution to the Danish nature management practices, whether the European bison could be a future management tool, for restoring the declining biodiversity.

How herbivores contribute to the ecological network of creating a more rich, dynamic and balanced nature have captivated me for a long time. I found it almost prosaic how nature can contribute to more nature by itself. This is also the reason why, I applied for the project, which gave me the opportunity to follow Europe's largest herbivore on Danish soil for the first time in more than 8.500 years.

The reintroduction of the European bison on Bornholm stands out as a groundbreaking project of this idealistic perception of nature management. Regardless of the outcome and effect from this majestic animal, the obtained knowledge will be of great importance, in the further project of fortifying biodiversity. Being a part of this project is therefore of great appreciation.

Introduction

In 1992, at the United Nations Conference on environment and development in Rio de Janeiro, 168 countries (at this day 189), including Denmark, signed the biodiversity convention text, and hereby committed to preserve biological diversity, defined as: *"The variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems"* (UN Rio convention, 1992).

The intent for conserving biodiversity has resulted in an increasing recognition, that the biological resources of the earth is vital for a sustainable future in many ways. This have created a growing interest, of how to maintain and enhance biological diversity (Reaka-Kudla, Wilson et al. 1996, De Groot, Wilson et al. 2002, Ravensbeck, Andersen et al. 2013).

At the same time, the threat of species extinction and ecosystem decline is greater than ever, due to increasing rates of human activities (Groom 2006). For the past century, the landscape of Denmark has changed significantly. This involves land use changes, increased population size, agriculture intensification and expansion of cities and roads. All factors that have resulted in a reduced area for natural habitats (Jensen, Boutrup et al. 2015).

The greater acknowledgement for preserving biodiversity has provided an enlarged understanding of the factors it catalyses. This includes ecosystem functions, which provides food, energy, materials, recirculation of nutrients, reduction of toxic particles etc. (Metera, Sakowski et al. 2010, Ravensbeck, Andersen et al. 2013). Improved biodiversity can also sustain a growing economy, both local and regional, through ecotourism (Gössling 1999). Furthermore, the valuation of biodiversity is almost impossible to define, since the existential value is a personal matter. One thing is for sure though, if the presumption is human existence, the value of biodiversity is infinite (Jacobsen 2013). The reasons for preserving biodiversity and natural habitats are therefore many, and management tools for the purpose are many as well.

Grazing by herbivores are suggested as a tool, for maintaining a diverse ecosystem and restoring biodiversity (Olff and Ritchie 1998, Vera 2000, Metera, Sakowski et al. 2010, Smit and Putman 2010). The grazing behavior of herbivores are therefore well used in many areas, in the attempt to maintain a certain level of the natural succession, since woody encroachment of the landscape is also considered a threat to biodiversity (Van Wieren 1995, Cremene, Groza et al. 2005, Bergmeier, Petermann et al. 2010, Smit, Ruifrok et al. 2015). Grazing can thereby prevent an up growth of woody plants and dominants species, and favor a more sensible environment, since the main mechanism of grazing is to increase the availability of the most limiting resources, which is most often light.

In May 2012, seven European bisons (*Bison bonasus)* from Poland*,* six females and one male, were introduced to Almindingen on the Danish island Bornholm. Almindingen is a forest located in the center of the island, covering approximately 6.400 ha (Orbitt 2018), but the seven bisons were released in an enclosed area of 200 ha, with the purpose of creating a more dynamic structure, with open places into the forest landscape (Brandtberg and Dabelsteen 2013).
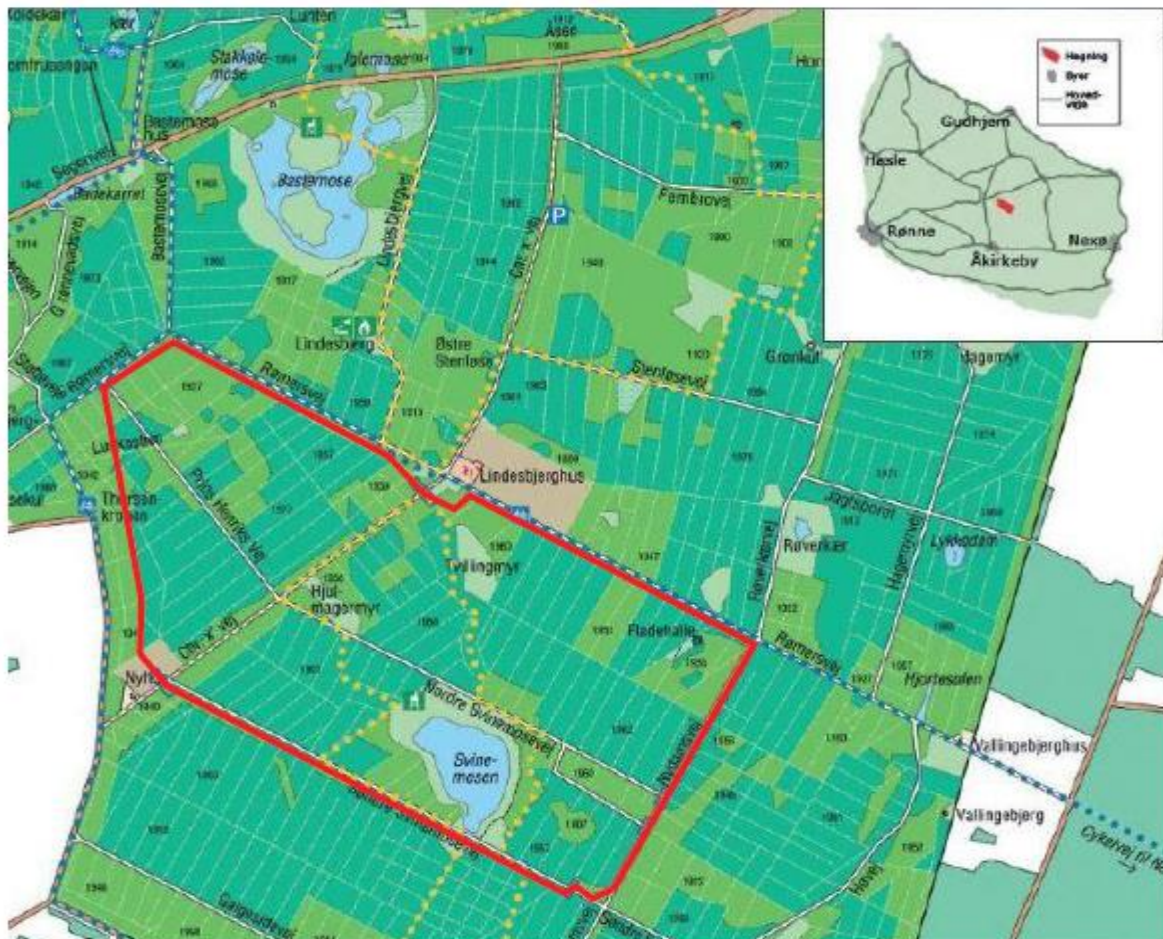
Figure 1. The red line marks the 200 ha large enclosure, where the European bisons were released in 2012. The main road 'Chr. X vej' is open for cars and 'Prins Henriks vej' is for walking only. Yellow dots represents smaller paths and tracks (Photo by NST, Bornholm 2015).

Since all herbivore species differ regarding food preferences, behavior and morphology, the way they shape the surrounding environment by alteration of species composition and structures also varies (Hester, Edenius et al. 2000, Vera 2000). The European bison stands out from the present grazing fauna of Denmark in several ways. Selective feeding is a key characteristic of herbivores, where all grazers are unique in their feeding behavior, and here the European bison differs compared to the native herbivore fauna (Gordon and Prins 2008, Cromsigt, Kemp et al. 2017). Furthermore, the mobility of the animal contributes to seed dispersal by endozoochory from faeces (Jaroszewicz, Pirożnikow et al. 2009).

The movement pattern and creation of footpaths also generates microhabitats with more disturbance, and thereby a more broad variation and niche establishment to the environment (Buttenschøn 2007). Additionally, the high body mass of the European bison adds another feature to its alteration abilities, since its tramp is more profound. The dusting baths and trampling of the European bison also forms more bare ground and vegetation gaps than any other grazer, which provide habitat for natural succession and pioneer species (Gordon and Prins 2008). Another specific effect on the vegetation from the European bison is the bark striping. Bark striping is documented on several tree species found in Almindingen, and is believed to affect the growth (Brender 2016).

The European bison once grazed the Danish landscape, but suffered extinction after the latest ice age (Aaris-Sørensen 1990). The loss of the European bison was not only constrained to Denmark, but also in the rest of Europe. In 1927, only twelve individuals were alive and hold in captivity in Europe, with the intention of restoring Europe's largest grazer.
Until this day, the revival has been a success, with a total population in Europe of more than 5.500 animals, with the majority being free living in nine different countries, most found in Poland, Belarus and Russia (Olech and Perzanowski 2014). When a population rise from only twelve individuals, genetic diversity remains low by natural causes, and the population is therefore still at risk this day (Tokarska, Pertoldi et al. 2011). Subsequently, the rewilding project in Almindingen does not only serve as a nature management project, but also contributes to increase genetic diversity and preserve the species.

Introduction of new species is a discussed topic, where critiques articulates concerns of the possible consequences and questions the benefit (Caro and Sherman 2009, Nogués-Bravo, Simberloff et al. 2016). Introduction of new species to a stable environment always involves risk, since new interactions will occur. Nevertheless, Denmark is obligated to improve the quality of protected nature areas, and the current effort from grazing projects by native species may not be enough to ensure the anticipated progress (Buttenschøn 2007). Intensified management is therefore needed and new methods probably as well.

The European Bison

The European bison, also known as the wisent, is Europe's largest terrestrial herbivore. It can reach a height over the shoulders of 2 meters, and the females can weigh 650 kg and the males 950 kg. It belongs to the mammal order of *Artiodactyla*, and the family *Bovidae*. The characteristics of the family is the permanent covering layer of keratin on the horns, and likewise being ruminants with a digesting system consisting of four stomachs.

The *Bovidaes* goes more than 20 million years back to the geological epoch of Miocene, but the specie *Bison bonasus* first originated 11.400 years ago, also with the formation of three subspecies, *B. b. bonasus, B. b. hungarorum* and *B. b. caucasicus* (Nowak and Walker 1999).
At present day, only the pure genus of *B. b. bonasus* and a mixed genus with the *B. b. caucasicus* exists in Europe, but mainly in northeast Europe (Pucek, Belousova et al. 2004).

In 1919 in Poland, the last population of free-living lowland bison *(B. b. bonasus)* suffered extinction, and in 1927 in the Caucasian mountains, the last herd of *B. b. caucasicus* underwent extermination. Even though the wild population of European bison were extinct, twelve individuals survived in captivity, of which seven (four males and three females) were the pure genus of *B. b. bonasus (*Olech and Perzanowski 2014) .
The surviving individuals were placed into zoos and breeding centers, in order to ensure the survival of the specie, which lead to an increase of the population to 54 individuals. After years of controlled conservation, the first rewilding of the European bison, after its extinction in the wild, happened in the Bialowieza forest in Poland in 1952 (Caboń-Raczyńska, Krasińska et al. 1987, Krasińska and Krasiński 2007).

In regarding of food sources, the European bison are not constraint to only one type of functional plant group, but can forage on both grasses, herbs, shrubs and trees, but will prefer understory vegetation (Cromsigt, Kemp et al. 2017). As a result of this, the European bison are considered to be a grazer with tendency to be partly browser (Hofmann 1989). It is important to remember, that this classification is not superior but only guiding, since the food preference and actual food intake is very much dependent on the accessible resources and local adaption (Cromsigt, Kemp et al. 2017). The food sources will therefore naturally differ dependent on habitat (Schmidt 2016), just like studies also shows, that European bisons which are being supplementary feed with hay, will have a decreased intake of bark and twigs (Kowalczyk, Taberlet et al. 2011).
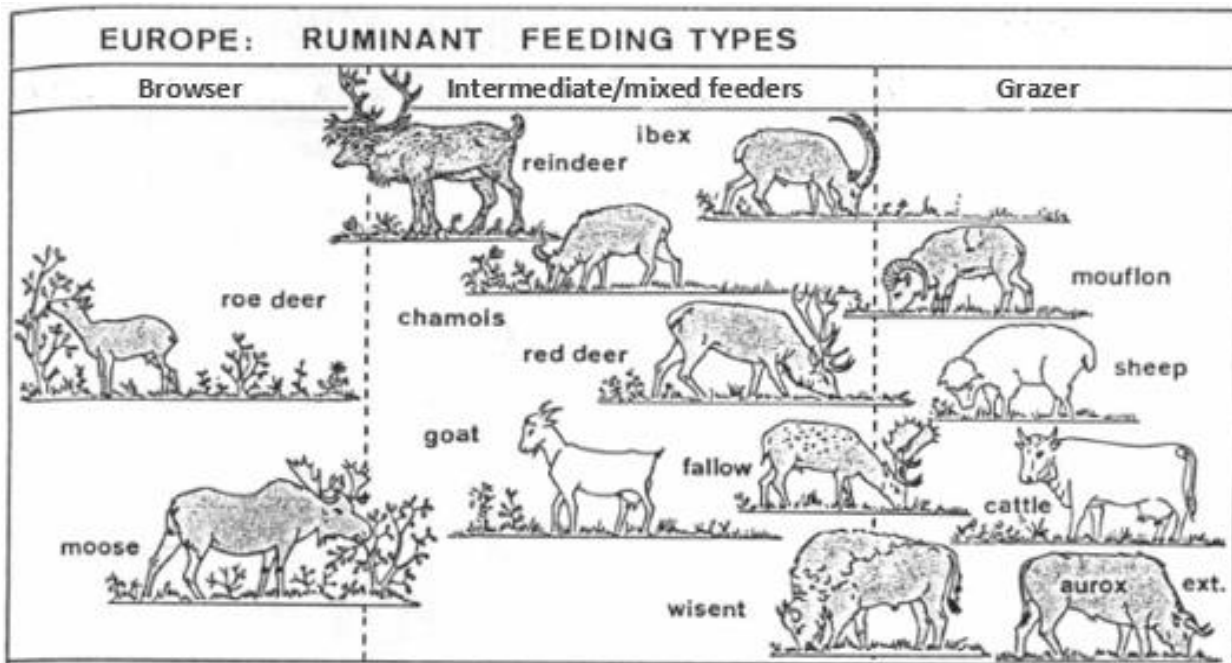
Figure 2. Classification of feeding types of European herbivores. The European bison (wisent) overlaps both grazers and intermediate feeders (Hofmann 1989).

A study of the food preferences of the European bison in the Polish forest Bialowieza during summer, showed that the rumen content consisted of 131 different plant species, of which 33% was trees and shrubs, and 67% was grasses and herbs (Borowski and Kossak 1972).

Another 5-year study by Cromsigt, Kemp et al. (2017) examines the food preferences during all four seasons in a coastal dune area in Netherland. The study evidences a temporal shift in food sources, mainly between summer and winter. During summer, the proportion of grasses in the diet was larger than during winter. At the same time, when the proportion of grasses in the diet decreases, the intake of woody material increases during winter. This supports the theory, that the diet of the European bison shifts towards more browsing during winter, most likely due to the absence of fresh grasses and herbs (Gębczyńska 1991, Krasińska and Krasiński 2007).

The shift in diets between seasons are a well-known phenomenon from other ungulates, and demonstrates an ability to adapt and cope with a variation in resources over seasons (Cederlund and Nyström 1981, Rayé, Miquel et al. 2011).

Schmidt (2016) performed a study in Almindingen on Bornholm in 2016, also with the intention to reveal how much different functional plant groups represents in the European bisons diet. From fecal samplings, the study revealed a diet consisting of 48% woody materials (30% tree species and 18% shrub species) during summer, which was a larger amount than expected. Furthermore was 40% of the diet herbaceous plants, 10% was allocated to grasses and 2% was undefined. This additionally supports the theory of a feeding plasticity and an ability to adapt local environments from the European bison, which furthermore support the European bison of being an intermediate feeder and making the categorization of being either grazer or browser for redundant, since it can vary in temporal and spatial scale.

Schmidt (2016) also found evidence, that the population on Bornholm consumed 6 new species and 25 new genera of plants, which was not present to the Bialowieza forest in Poland, where similar studies previously has been performed. This revealed an enlarged niche regarding food sources, also indicating an adaption to the local environment in Almindingen.

In Almindingen, five plant species or genera appeared in all fecal samplings, signifying a large contribution to the diet of the European bisons on Bornholm and a large abundance in the area. This regards the plants; *Calamagrostis sp., Descampsia flexuosa, Lysimachia sp., Rubus idaeus* and *quercus sp.*

Even though *Calamagrostis sp.* was not determined to species, it is reasonable to assume that the species found was *Calamagrostis epigejos.* This species was present in the area in large amounts in certain areas and no other species from the *Calamagrostis* genus was found. *Calamagrostis Epigejos* is considered an unwanted species in Almindingen, because of its ability to dominate and outcompete other species. Schmidt (2016) therefore suggested that the European bison could reduce the abundance of the species and contribute to a more diverse ecosystem in Almindingen.

These varying results regarding the food preferences for the European bison have also caused disagreements about the habitat preferences for the species. Even though most studies support a primary diet of grasses and herbs, and dental morphology and evolutionary background places the European bison as an open-land animal, most management practices for conservation of the European bison still places the animal as a forest specialist (Kerley, Kowalczyk et al. 2012). The distribution between open land and forest in the European bison's habitat preferences is a discussed topic and the species is also addressed as being a possible refugee specie (Kerley, Kowalczyk et al. 2012, Bocherens, Hofman-Kamińska et al. 2015). This theory submits that the European bison was forced into the forest by human predation and loss of habitat in earlier times, which therefore marks the forest as a sub-optimal habitat. However, it is almost for sure, that the European bison is dependent of both open land and forest in its habitat preferences, since the forest delivers alternative food resources during winter e.g. shelter and hiding places (Caboń-Raczyńska, Krasińska et al. 1987, Krasińska and Krasiński 2007, Brandtberg and Dabelsteen 2013).

The European bison are therefore reintroduced to Almindingen, which covers both open land and forest, with the purpose of being a bioengineer, and to create favorable habitat for an increased biodiversity. Its distinctive behavior and morphology together with its exclusive characteristics are hence expected to shape the current landscape differently, than the present grazers are able to.

14

Grazing and landscape history

Grazing has been an existing phenomenon, since the rise of herbivores. Since agriculture originated in Denmark for approximately 6.000 years ago, humans and their activity have shaped the expression of the landscape (Rowley-Conwy 1985). However, until then, herbivores and their behavior played a vital role of modifying the expression of the natural land, also before domestic grazers (Buttenschøn 2007, Nielsen & Buchwald 2010).

After the last ice age 15.000 years ago, the landscape evolved from open spaces and tundra steppe, towards a more dense and deciduous forest for the next 9.000 years. However, in this period, a different flora and fauna were also present, than the one characterizing the land today (Nielsen & Buchwald 2010). The colder climate and more light open areas resulted in a migration of grazers, which favored the current habitat. Therefore, in the for historic Denmark, in the transition between arctic tundra and primeval forest, large grazers such as woolly mammoth, reindeer, musk ox, woolly rhino and the European bison all existed (Aaris-Sørensen 1990). Subsequently, later as the forest slowly increased, leaving less herbaceous vegetation and open land, these species either migrated further north or underwent extinction due to both habitat fragmentation and human predation (Aaris-Sørensen 1990).

According to archaeological findings, the European bison became extinct in Denmark around 8.700 years B. P. when the landscape was preboreal, as illustrated in figure 3 (Aaris-Sørensen 1990). The extinction of European bison was most possible due to the shift from grasslands to more forest-dominated ecosystems, but more important, an increased hunting activity from a growing human population (Kuemmerle, Hickler et al. 2012).

Years B.P.	12.750	11.500	9.000	6.000	2.700
	Steppe, chaparral and tree tundra	Open birch, hazel and pine forest	Mixed deciduous forest	Beech forest	
Giant deer	━━━━━━				
Wolverine	━━━━				
Rein deer	━━━━				
Wolf	━━━━━━━━━━━━━━━━━				*
Dog	━━━━━━━━━━━━━━━━━━━				
Wild horse	━━━ ━━			━━	
Beer and moose	━━━━━━━━━━━━━				**
Beaver	━━━━━━━━━━━━━━				***
European bison		━━			****
Aurochs		━━━━━━━━━━━			
Lynx		━━━━━━━			
Boar		━━━━━━━━━━━			
Roe deer and red deer		━━━━━━━━━━━			
Domestic ox-, sheep-, goat and pig				━━━━	
Horse					━━━

Figure 3. Distribution of large mammals since the last ice age in Denmark. Black lines represents animals, which have disappeared from Denmark, while red lines represents animals still present (mod. a. Buttenschøn 2007). *The wolf immigrated to Denmark in 2011. **Mooses were reintroduced in Lille Vildmose in 2015. *** Beavers were reintroduced in Klosterheden in 1999. **** European bison were reintroduced to Almindingen on Bornholm in 2012.

One grazer, which are believed to have been of great importance in shaping the dynamics between the light open areas and the primeval forest, was the aurochs. Through archeological findings, the aurochs was found to be one of the most widespread grazers in the Danish landscape after the last ice age (Buttenschøn 2007). It is therefore reasonable to believe, that the aurochs had great merit in the open landscapes expression, before humans became farmers. The aurochs disappeared from Zealand 7.000 years ago, but was first extinct from Jutland 500 years B.C. The extinction was probably due to an increased hunting activity from humans (Buttenschøn 2007).

The aurochs is the genetic ancestor to the domestic cattle, and was also known to be a primary grazer. Former studies though suggests, that the Aurochs and the European bison shared several common traits in diet- and habitat preferences (Noe-Nygaard, Price et al. 2005, Buttenschøn 2007). Like the European bison, the aurochs was similarly dependent on a mosaic landscape with both forests and light open areas, with a rich herbaceous vegetation. The aurochs were probably also reliant on more dense forests for providing shelter, hiding places and supplementary feeding sources during winter.

Whit history in mind, the choice of reintroducing the European bison as an alternative grazer to Bornholm is not random. First, the European bison have previously thrived in the prehistoric Danish landscape, and according to pollen analyses, suitable habitats for European bisons can still be found in Denmark (Kuemmerle, Hickler et al. 2012).

Furthermore, since the aurochs were of great importance in forming the open landscape we want to maintain today, reintroducing the European bison could contribute to restore the prehistoric ecosystem dynamics we want to create, since they overlap in both behavior and food preferences with the aurochs.

Study objectives

This is the first rewilding project of the European bison in Denmark, and the results are therefore crucial for the further course, of a possible reintroduction of the species to what was once a native country, for the largest grazer in Europe.

The main purpose of the study is to monitor the biodiversity of vascular plants in the end of a 5-year project from the National nature agency, and to illuminate the development in species richness and selected Ellenberg indicator values, through the entire 5-year period. The bottom vegetation, abundance of selected species, trees and other vegetation structures are closely examined for a broad understanding of any possible changes during the 5-year period. In addition, browsing activity and bark striping are investigated to supplement the results if relevant. Hence, the study should contribute to a better understanding of the possible effects on plant species composition in different habitat types, when grazed by European bison, since Almindingen covers a broad and varied mosaic landscape.

The results of the study should also form a basis for the further discussion on whether the Danish nature can sustain a population of European bison, and whether the presence is at all rewarding in terms of enhancing the biodiversity.

Materials and methods

Over the 5-year period the study extended, three identical samplings was conducted in the years of 2012, 2014 and 2017. The first sampling from 2012 was made just before the bisons were released in the area, as a benchmark for the further project. Employees from the local unit of the national nature agency collected the data in 2012 and 2014, whereas I collected the data in 2017. In the three years, where measurements were made, the data was collected during summer, from mid-May to mid-August. Thirteen defined habitat types were investigated, all containing 10 circle plots each, resulting in a total of 130 plots. The habitat types were categorized either *forest* or *open landscape* based on vegetation structure. Ten of the vegetation types, was categorized *forest*, and the remaining three was then categorized *open landscape*. All 130 plots was marked with coordinates and a wooden pole, for the exactly location. To locate the plots, a handheld GPS with coded coordinates was used. Ten of the habitats were found inside the fencing and three was located outside the fencing as control fields.

Study site – Almindingen on Bornholm

The study site was located in a central place in Almindingen forest on the island Bornholm in Denmark (55.1031-55.1208 °N, 14.9317-14.96355 °E). The enclosed area contained a variety of closed forest, semi-closed forest and open land, which created a suitable habitat for the European bison. For out being one of Denmark's most appreciated forest areas, the range is also classified as a 'Natura2000' area, which means it is obliged to be conserved, in order to preserve the habitats and species associated with the area.

For the 5-year period the study persisted, the seven introduced individuals increased the population, which indicated that the animals thrived in the enclosure. The total population, and thereby grazing pressure, have varied over time, but has mostly been constant on approximately 15 individuals (Orbitt 2018). The population reached its maximum in 2017 with 21 individuals (Orbitt 2018). Over the 5-year period, some individuals suffered from liver fluke attack (*Fasciola hepatica)* with dead as consequence. Unfortunately, the parasitic attack is not an uncommon phenomenon, and are found in most studies dealing with the European bison (Karbowiak, Demiaszkiewicz et al. 2014). The herd was winterfeed weekly with hay, when the availability of grasses and herbaceous plants was considered low (Orbitt 2018).

The supplementary feeding was mostly constrained to the same spots all winter, and the activity in those specific areas probably increased, and nearby circle plots was possibly affected by a higher and more frequent activity. As previously described, winterfeeding may also affect the winter diet of the bisons. Furthermore, in the 5-year period, necessary and common forestry was implemented, both inside and outside the fencing (Orbitt 2018).

Regarding the climatic factors on the island of Bornholm, the precipitation and temperature varied through the germination- and growth season from year to year. Table 1 and 2 shows respectively monthly precipitation and temperature for comparing years of the study extent. These factors have an unquestionable effect on the facilitation of species germination and growth, and it is therefore rational to believe, that the same species have sprouted at different times over the 5 years.

However, some of the periods that stands out with deviation are found in the spring of 2013, where no registrations were made. The season were more dry and with colder temperatures than other years. The summer of 2015 was colder than the average temperatures from other years and the early summer in 2017 also showed a higher amount of precipitation than other years.

Table 1. Monthly amount of precipitation (mm) from 2012-2017 on Bornholm (Nexø). Reference: www.DMI.dk

Year	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.	Total amount
2012	113	57	12	38	35	79	94	39	48	50	51	58	674
2013	62	29	13	19	69	48	71	43	58	66	81	55	614
2014	42	36	30	30	31	26	33	200	104	61	27	113	733
2015	94	13	41	28	57	33	92	49	86	27	126	67	713
2016	33	33	9	34	7	39	77	33	50	77	75	33	500
2017	18	61	39	33	16	100	66	74	70	111	101	79	768

Table 2. Monthly average temperatures (C°) from 2012-2017 on Bornholm (Nexø). Reference: www.DMI.dk

Year	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.	Yearly average
2012	2.1	-0.4	5.8	5.8	10.6	13.7	16.0	16.8	13.8	8.9	6.3	0.6	8.2
2013	0.5	0	5.0	5.0	10.9	14.4	17.8	18.2	13.6	11.1	6.5	4.9	8.6
2014	0.6	3.2	7.1	7.1	11.2	14.4	19.4	17.4	15.4	12.4	7.9	3.5	9.8
2015	3.1	2.1	6.9	6.9	9.6	12.9	16.2	17.8	14.3	9.8	7.4	5.8	9.2
2016	-0.1	2.6	6.4	6.4	11.8	15.4	17.2	16.7	16.4	9.4	5.1	4.3	9.1
2017	1.0	1.5	5.7	5.7	10.8	14.5	15.7	17.2	14.2	10.9	6.2	3.8	8.9

Besides the European bison, two other ungulate species was found in the fence: fallow deer (*Dama dama*) and roe deer (*Capreolus capreolus*), which had entered the fencing by itself. The exact population size of these two herbivores were not known, but were believed to be minor than the population size of the European bison. The fallow deer and roe deer were therefore not considered, to affect the results of the study (Orbitt 2018).

Fallow deer (Dama dama) (left) and European bison (*Bison bonasus*) (right) inside the fencing. Photo: Bjarke Schäfer

Table 3 shows all 13 investigated habitat types and their field numbers. The enclosed area mainly consists of Norway spruce forest, which is also the primary harvest product (Orbitt 2018), and the bush grass dominated area was accounted for the smallest area.

Table 3. Habitat types in- and outside the fencing, named after the dominating vegetation type, with field-numbers. Data from Jønsson (2014)

Field	Vegetation type	Area (ha)	Cover of total area (%)
2.1 – 2.10	Alder swamp	3.04	1.52
3.1 – 3.10	Beech forest	12.41	6.20
5.1 – 5.10	Old oak forest	7.38	3.69
6.1 – 6.10	Middle age oak forest	20.26	10.13
7.1 – 7.10	Young oak forest	2.76	1.38
11.1 – 11.10	Old Norway spruce forest	63.78	31.89
12.1 – 12.10	Young Norway spruce forest	30.79	15.39
13.1 – 13.10	Rejuvenated area	17.64	8.82
16.10 – 16.10	Wet meadow	11.17	5.58
20.1 – 20.10	Bush grass dominated area	1.52	0.76
-	Other (lakes, roads, etc.)	29.27	14.63
4.1 – 4.10	Beech forest (ungrazed)	-	-
8.1 – 8-10	Old oak forest (ungrazed)	-	-
10.1 – 10.10	Old Norway spruce forest (ungrazed)	-	-
TOTAL		**200.02**	**100**

The 200 ha fenced area, contained 100 sampling plots, 70 of these was located in forest areas, and the remaining 30 were placed in open land. In addition to the plots in the fenced area, 30 control plots was located just outside the fencing, completing a total of 130 sampling plots. The sampling plots were distributed as illustrated in Figure 4.
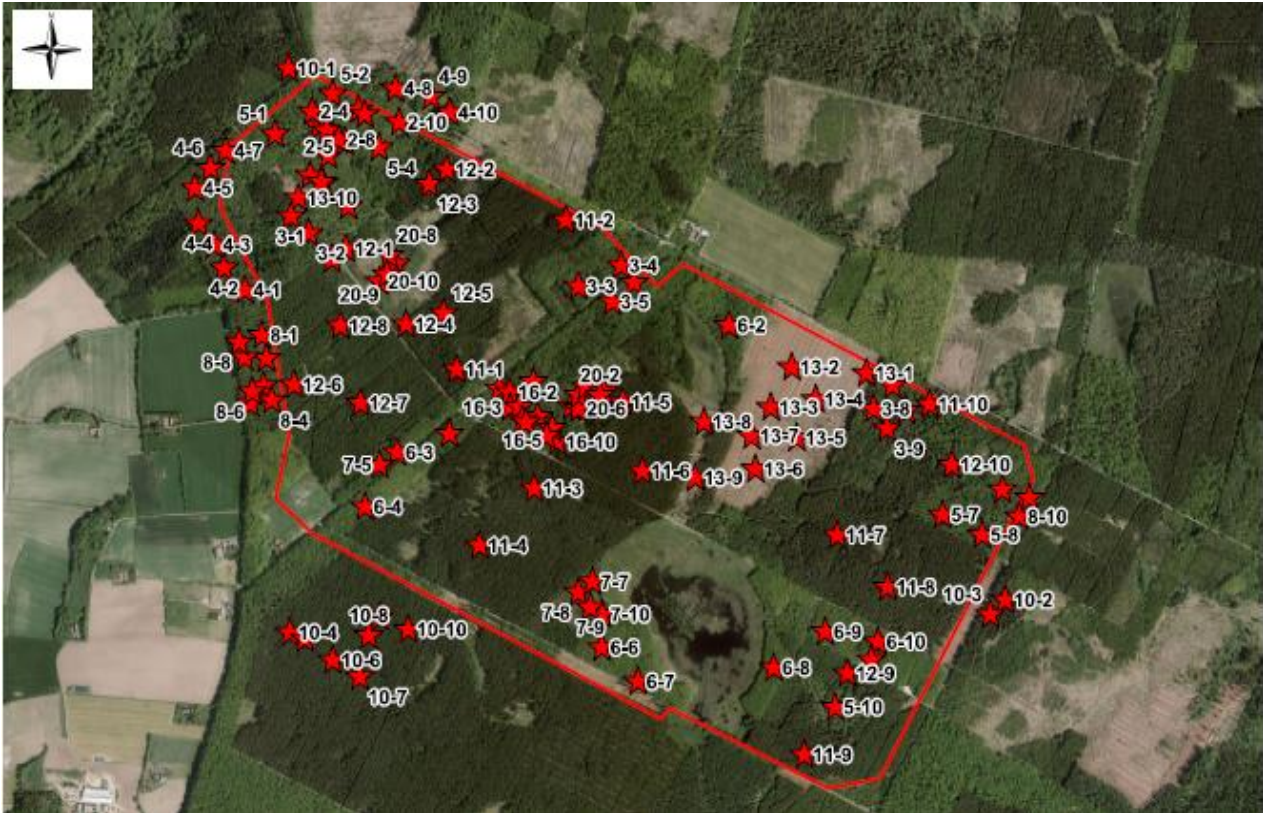
Figure 4. The study site, where the red line represent the fencing and red stars represent numbered fields. photo from NST Bornholm

The alder swamp (main field 2) was a semi-open forest with wet soil, where the majority of trees was *Alnus glutinosa* and *Alnus incana*. The alder swamp contained smaller wide spaces without a crown layer and a small river.

The beech forest (main field 3) was formed only by large even aged stands of *Fagus sylvatica*, with a minimum of light and vegetation in the ground layer.

The oak forest was divided into three main fields (main field 5, 6 and 7) based on age of the trees. The oak forest contained more variation regarding species composition and spaciousness, and hosted more tree species than only *Quercus robur*. *Sorbus aucuparia* and *Acer pseudoplatanus* was also found in smaller lots around the forest, as well as smaller open fields with a bottom vegetation conquered by graminoids.

The Norway spruce forest was likewise divided into two main fields (main field 11 and 12) also based on age of the forests. The Norway spruce forest was solitary dominated by a dense growth of *Picea abies*, with only small or none light heaps in the understory.

The rejuvenated area (main field 13), was clear-cutted from a dense growth of Norway spruce in 2011, leaving an open area with a poor bottom vegetation. According to a previous study by (Brandtberg and Dabelsteen 2013), the rejuvenated area was the main preferred habitat by the bison herd in the first years of reintroduction.

The wet meadow (main field 16) was a smaller isolated area in the fencing. The area did not include any trees, but only a rich assemblage of herbaceous species on the wet soil layer. The meadow was also the only area, which was previously grazed in recent times, both before and during the reintroduction of the European bison in Almindingen. Before the reintroduction of the bison herd in 2012, the meadow was fenced and summer grazed by horses (Orbitt 2018). The summer grazing by horses ended in 2012 when the bisons were released in the area, but from 2015 to 2016 the meadow was once again fenced, and grazed by domestic cattles (Orbitt 2018). Except from the period when grazed by cattles, the meadow was accessible for the bison herd.
It is reasonable to believe, that summer grazing by horses until the reintroduction, as well as the period when grazed by domestic cattle, has affected the results. A possible effect on biodiversity in this specific area can therefore not be attributed exclusively to the presence of the European bison.

The bush-grass dominated area (main field 20) was open areas with a compressed and dominating bottom vegetation of *Calamagrostis epigejos*. Few other species of graminoids and herbs was also found in the understory. The vegetation of trees was sparse, but smaller up growths of *Picea abies* was present in parts of the area.

The three control fields (main field 4, 8 and 10) was located outside the fencing and showed similarities with their corresponding grazed fields inside the fencing.

Vegetation registration in forest vegetation

For the ten forest vegetation types, species composition, vegetation structure, browsing activity and bark striping was analysed in all of the 100 plots. All of the circle plots were centered from the wooden pole, with a radius of either 5 meter (area of 78,5m^2) or 15 meter (area of 700m^2) dependent on registration type. The species registration, browsing activity and bark striping analyses will provide basis for specific results in this report, while the vegetation structure analysis will appear as supplementary material to support the results and discussion if relevant.

Species registration

The species registration of plants in the forest bottom layer was conducted in the 5-meter circle, by the Raunkjær circle method. A Raunkjær circle covers an area of 0,1 m^2 or 1000 cm^2, and for this study, the Raunkjær circle was divided into three zones (Damgaard 2015). *Zone 3* covering 10 cm^2, *zone 2* covering 100 cm^2 and *zone 1* covering the whole area of 1000 cm^2.

The Raunkjær circle anchored in the ground, with pink tape marking the zones. Photo: Bjarke Schäfer

The Raunkjær circle was thrown randomly in the 5-meter circle ten times, and the plant species covered by the circle was registered and determined by (Frederiksen 2012). The species registration started with investigation of the smallest zone, *zone 3,* and when all species was mapped, the registration continued to *zone 2* and in the end *zone 1.* All species was only marked once, by the smallest zone in which they appeared. Mosses was only noted do the division *Bryophyta* and lichens was neither determined further.

Registration of vegetation structures

For registration of vegetation structures, several measurements was conducted in both the 5-meter circle and the 15-meter circle. For every circle plot, a field chart containing the registrations was comprehensively completed. The field chart used can be found in appendix 1.

For the first registrations, the 15-meter circle was investigated. All trees, with a diameter bigger than 40 cm measured in breast height, were mapped and noted for species and diameter.
To determine the diameter of the trees, the circumference was measured by measuring tape, and the diameter was afterwards calculated by the circle equation.
After registration of trees, a variety of visual measurements were performed, also in the 15-meter circle. Table 4 shows a section of the field chart, with the categories that were estimated by counting.

Table 4: Cavities, dead wood, mosses, lichens and excrements was counted in the 15-meter circle.

	Number
• Trees with woodpecker holes	
• Trees with cavities (larger than woodpecker holes)	
• Trees with a dense growth of mosses and/or lichens	
• Standing dead trees higher than 2m and with a diameter larger than 25cm, measured in breast height	
• Lying dead trees higher than 5m and with a diameter larger than 25cm, measured in breast height	
• Bison dungs (excrements)	
• Deer droppings (excrements)	
• Other animal tracks	

The border between woodpecker holes and cavities was a visual estimate. Trees with a dense growth of mosses or lichens were categorized, when approximately more than 1/3 of the trees surface area was vegetated.
Dead trees were categorized as trees with rotten wood or without flower buds in the spring and summer period. The surface ground was screened for excrements and the vegetation was searched for other tracks as well.

The 15-meter circle measured around the wooden pole with measuring tape (left) and the circumference of a *Fagus sylvatica* measured with measuring tape (right). Photo: Bjarke Schäfer

The last vegetation structure registration in the 15-meter, circle was a classification and counting of all tree and shrub species and their size and development stage. All species were mapped according to table 5, where the size of the tree was divided into 5 stages.

Table 5: Counting and classification of tree and shrub species and their development stage.

Specie	1: seedling	2: <50 cm in height	3: 50-100 cm in height	4: 100-200 cm in height	5: >200 cm in height

The 15-meter circle was systematically investigated for all up growth to be counted and classified. For some plots though, the density and amount of seedlings was too high for a precise count (>200), and for those plots an estimate was done. The estimate was calculated from $1m^2$, which represented the growth of the circle plot as best as possible. The number of seedlings were counted and afterwards multiplied with the area of the circle plot. In some circle plots, the distribution of seedlings were clumped (e.g. when the area was uneven exposed to sunlight), in those plots the estimate was calculated from two or three fields of $1m^2$ for a better scale.

After these registrations, the documentary circle was decreased to the 5-meter circle, and the rest of the vegetation structure investigation was conducted in an area of 78,5m^2.

First, the coverage rates were determined. The coverage rates were divided into intervals, and determined by visual estimate in the percentage categories showed in table 6.

Table 6. Coverage rates intervals determined in the 5-meter circle in forest registrations by visual estimate.

Trees and shrubs lower than 1 meter				
<5%	5-10%	11-30%	31-75%	>75%
Trees and shrubs higher than 1 meter				
<5%	5-10%	11-30%	31-75%	>75%
Total crown cover				
<20%	20-50%	51-75%	76-90%	>90%
Total water surface				
<5%	5-10%	11-30%	31-75%	>75%
Proportion of bare ground				
<5%	5-10%	11-30%	31-75%	>75%

After the coverage rates were mapped, the 5-meter circle was assessed for rejuvenation, if present. The rejuvenation was categorised in three categories:

- Not present
- <2 seedlings per m^2
- >2 seedlings per m^2

In most cases, the category could be determined visually, but to support the precision, counting were used in some cases. In addition to one of the three categories, the specie was also noted.

Rejuvenation of Norway spruce in the category *'>2 seedlings per m$^{2'}$* in forest vegetation. Photo: Bjarke Schäfer

Browsing and bark striping

Browsing and bark striping registration was only performed in 2014 and 2017, since the first data collection was sampled just before the reintroduction in 2012. The registrations of browsing and bark striping were investigated in the 15-meter circle and could be done simultaneously with the classification of size and species of trees and shrubs in the 15-meter circle, when vegetation structures was conducted.

For all registrations of species and size of trees and shrubs, the individuals were also mapped for browsing if present. A four-step scale was used to evaluate the impact of the browsing activity on every individual, according to below standing categories:

- **0**: No browsing activity.
- **0.5**: Light browsing of a few buds and thin twigs – growth is not significantly affected.
- **1**: Browsing of many buds and twigs – growth is somewhat affected with no tendency to form twigs, but height growth occurs.
- **2**: Almost all buds and twigs are browsed and eaten – growth is highly affected

The classification of individuals was selected by a visual estimate.

Acer pseudoplatanus with light browsing (left) and *Quercus robur* with bark striping (right). Photo: Bjarke Schäfer

All individuals were also noted for bark striping activity, when mapped for specie and size. Individuals with visual bark striping were counted and noted, for the entire 15-meter circle. After the screening of all individuals, the total population inside the 15-meter circle was categorized into a four-step scale as well:

- **0:** no Bark striping
- **0.5**: Limited bark striping activity – less than 10% of all individuals
- **1**: medium bark striping activity – less than 50% of all individuals
- **2**: extensive bark striping activity – more than 50% of all individuals

Browsing and bark striping were both related to specie and size, and was therefore noted like in table 7.

Table 7. Registration scheme for evaluating any browsing or bark striping activity on the tree and shrub vegetation.

Browsing and bark striping			
Specie	Size (stage 1-5)	Browsing category	Bark striping present/absent

Vegetation registration in open land vegetation

Three open land habitats was located in the enclosing, and all 30 plots were analysed for species composition, vegetation structures, browsing and bark striping. All registrations were assessed within the 5-meter circle in the open land. The species registration, and browsing and bark striping analyses will provide basis for specific results, and the vegetation structure analysis will appear as supplementary material to support the results and discussion if relevant.

Species registration

The species registration of plants in bottom vegetation was conducted with the same procedure, as in the forest habitats. The three-zoned Raunkjær circle was stochastically thrown ten times within the 5-meter circle plot. Plants were then registered inside-out from the smallest zone (zone3, covering 10 cm^2) towards the biggest zone (zone1, covering 1000 cm^2). Species were only noted once, in the smallest zone they appeared in.
All species were thus assigned a number (1-3), corresponding to the zone they occurred in.

All species was again determined by (Frederiksen 2012), but mosses was only noted do the division *Bryophyta* and lichens was neither determined further.

Myosotis scorpioides (left) and *Lychnis flos-cuculi* (right) on the wet meadow. Photo: Bjarke Schäfer

Registration of vegetation structures

The registrations of vegetation structures were first assigned by a visual investigation of numerous structures, followed by a counting of trees and shrubs, all performed in the 5-meter circle. The field chart completed for every plot is found in appendix 2.

The covering rate in the circle plot of the following nine varieties, were initial determined into a percentage scale with five categories, ranging from; 0-5%, 5-10%, 10-30%, 30-75% and 75-100%. This was done by a visual estimate, while standing in the centre of the circle plot for best possible sight.

- Area without vegetation (bare ground, sand, water)
- Lichens
- Mosses
- Peat mosses (Sphagnum)
- Grass/herbaceous vegetation lower than 15cm
- Grass/herbaceous vegetation between 15-50cm
- Grass/herbaceous vegetation higher than 50cm
- Subshrubs

Afterwards, the covering rate of the following two varieties were determined into yet another 5-step percentage scale, ranging from; 0%, 1-10%, 10-25%, 25-50% and 50-100%. This was also assisted by a visual estimate.

- Crown cover from trees
- Occurrence of invasive species

Wet meadow (left) and a spot of bare ground in the rejuvenated area (right). Photo: Bjarke Schäfer

After the visual assessment of the vegetation structures, the grazing activity and operation of the area were evaluated. The same procedure was followed as in vegetation structure registration, and the covering rates of the three following factors was categorized:

- Grazing (clear sign of grazing in the circle plot)
- Mowing/clearing in the circle plot (biomass left in the circle plot)
- Clear signs of eutrophication (direct fertilization or supplementary feeding)

The two first factors (grazing and mowing) wer categorized after the following 5-step percentage scale; 0-5%, 5-10%, 10-30%, 30-75% or 75-100%. Eutrophication was categorized after the scale; 0%, 1-10%, 10-25%, 25-50% or 50-100%.

Subsequently, hydrology was explored and the drainage was considered for every circle plot, which was assigned to one of the following categories:

- No drainage (no ditches or sinks)
- Weak effect from drainage
- Clear effect from drainage
- Effect from drainage widely spread
- Completely drained

Afterwards, a counting of bison and deer excrements, along with other animal tracks were investigated and mapped in the 5-meter circle plot.

Finally, a counting and classification of every tree and shrub specie were completed, also in the 5-meter circle plot. Every single individual was mapped for specie and development stage, according to the same size intervals, as in the forest registration shown in table 3. The circle plot was systematically investigated for all individuals to be recorded.

Browsing and bark striping

The registration of browsing and bark striping was done in the 5-meter as well. These registrations were completed simultaneously with the counting and classification of tree and shrub individuals according to species and size.

The browsing and bark striping activity were measured and categorized with the same method, as in forest vegetation. The 4-step scale (0, 0.5, 1, 2) and description of the levels was identically used.

Browsing and bark striping was related to specie and size as well, and table 4 was therefore used when mapping the browsing and bark striping for the open land.

Pseudotsuga with bark striping (left) and the invasive moss *Campylopus introflexus* found in the rejuvenated area. Photo: Bjarke Schäfer

Statistics and data validation

The statistical calculations were performed in R Studio version 1.1.423. Data with observations from the Raunkjær circles and browsing and bark striping activity were imported as three separate CSV. files to the R studio, containing information about frequencies, weighted Ellenberg indicator values and presence/absence data for browsing and bark striping.

For the species richness and Ellenberg analyses, all mosses and lichens were removed from the Raunkjær circle observations, before a GLMM-model (general linear mixed model) from the lme4 package (version 1.1 – 17) was applied. This allowed responses to include both fixed and random effects and for me to investigate samples (rows) in relation to input factors (columns). The GLMM-model was poisson distributed with field as a random effect, since events happened in a fixed time interval. The models were validated based on residuals to find the best possible fit. Covariance was analyzed ass well (ANCOVA) with canopy cover from the fields as continuous variable and time as the categorical variable. The model validation allowed an interaction between these two variables in the species richness models for best possible fit, but not for the Ellenberg analyses. For out the continuous variable (canopy cover) and categorical variable (time), known inputs in the linear model also included area, grazing status, field and number of species in each field.

For browsing and bark striping activity, each individual's probability for being browsed or striped was investigated according to species and size. A GLMM model with binomial distribution was applied with logit link function, since only two outcomes were possible (browsed vs. non-browsed and striped vs. non-striped). Data from 'fields' was nested into 'area'. ANCOVA was also performed with canopy cover as continuous variable and time as categorical, but without interaction between the two variables also in order to obtain the desired model validation. Weighted values were taken into account in the model, since number of trees differed among samplings.

Frequency analyses of relative abundance of selected species were performed in excel by the 'figure tool' and based on presence/absence data from the 100 throws of the Raunkjær circle in each habitat. The biodiversity indices for all areas were calculated from the accumulated Raunkjær values, by the following formulas in excel:

Simpson dominance: $D_S = \sum p_i^2$
> The sum of the square of the relative abundance of all species

Shannon Wiener: $H = \sum p_i \cdot \ln(p_i)$
> The sum of the product of the relative abundance of all species multiplied
> with the natural logarithm of the relative abundance of all species

Pielou evenness: $J = H/\ln(S)$
> The Shannon Wiener value divided with the natural logarithm of species
> richness (number of different species present)

Results

The result section includes a data visualization of the relevant figures and results for discussing the main objective, on whether the grazing activity from the European bison affected the biodiversity of vascular plants for the past five years in the enclosure in Almindingen. This includes data of species richness and development in Ellenberg indicator values for light and nitrogen. Furthermore, a frequency analysis of the most abundant species for the last five years is included with related diversity indices for all areas. Finally, in order to support the discussion of the extensive topic, data from browsing and bark striping is presented and evaluated.

Species richness

The species richness concept is a simple way to describe community diversity, and represents number of species in a defined area. In this case, the species richness is the average number of species for the ten plots in each of the thirteen vegetation types. Quantifying species richness is important, not only for basic comparisons among sites and years, but also for determine the saturation of communities colonized from source pools. Maximizing species richness is often an explicit or implicit goal for conservation studies, since values are easy to compare and understand.

In the following species richness models, mosses and lichens are not included, since they were not determined to species in 2012 and 2014. In order to explain any possible significant changes in species richness, the models includes two variables; canopy cover (CC) and year (time). Canopy cover was extracted from the latest LiDAR scanning from 2014, and we thereby assume it was unchanged for the 5-year period. In those areas that contain both a grazed and ungrazed area, grazing activity is also included as a possible explanatory variable.

All P-values and changes are measured relative to the first measurement in 2012. In addition, measurements from ungrazed areas are relative measured to the corresponding grazed area from 2012.

Open land

Figure 5 shows the species richness of the three open land habitat types. Looking at the bush grass dominated area, species richness increased from 4.0 in 2012 to 4.1 in 2014 and furthermore to 6.0 in 2017. The development over time designated an increase in species richness, but not yet enough for a significant increase to be observed, leaving no substantial changes in this area.
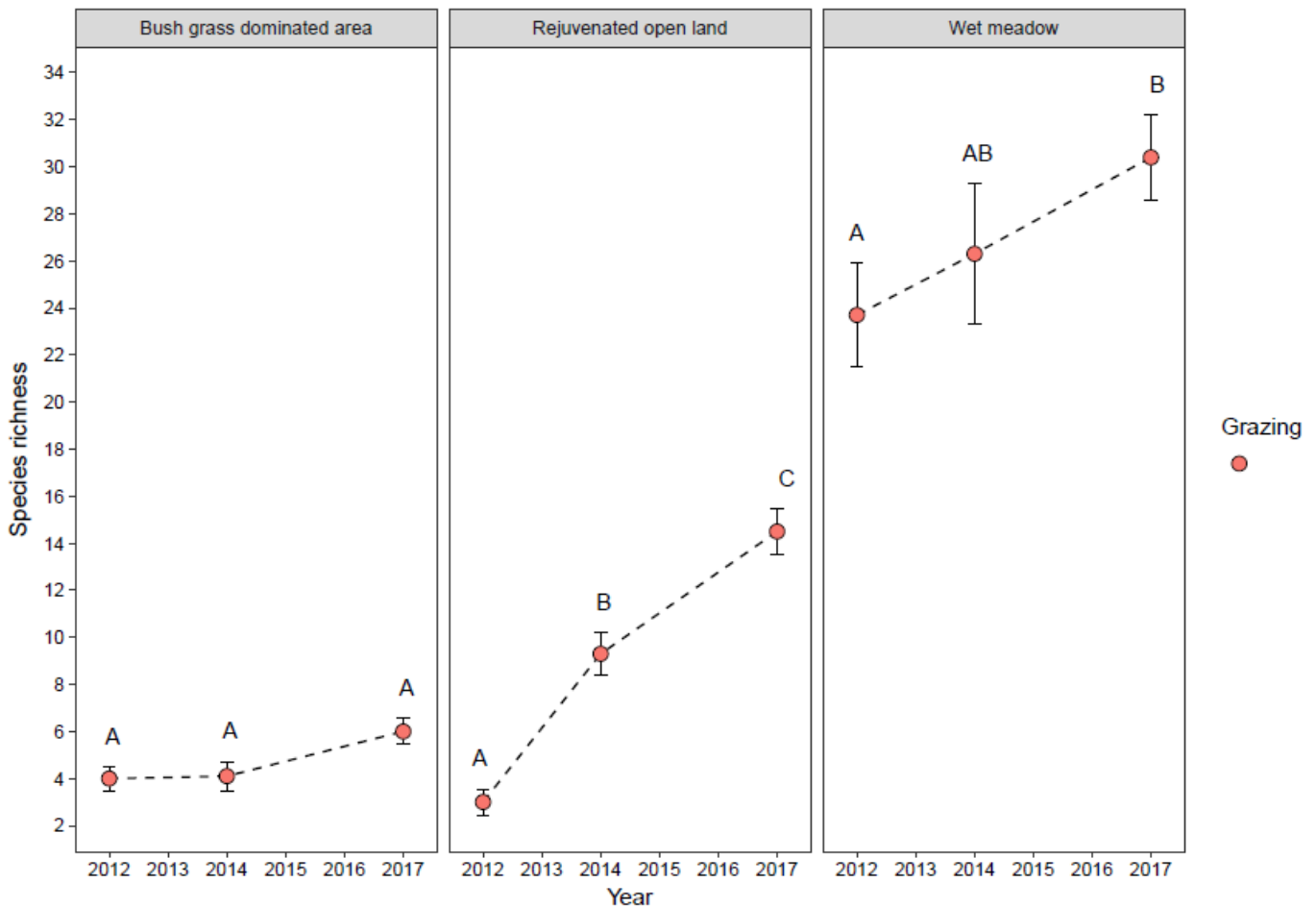
Figure 5. Average species richness for the ten plots (±SE) of the three open landscape vegetation types in the bison fencing. Different letters represents different significant groups. The species richness in the bush grass dominated area was in 2012 4.0 (SE=0.516) in 2014 4.1 (SE=0.622) and in 2017 6.0 (SE=0.557). The species richness for the rejuvenated area was in 2012 3.0 (SE=0.557), in 2014 9.3 (SE=0.907) and in 2017 14.5 (SE=0.957). The species richness for the wet meadow was in 2012 23.7 (SE=2.206), in 2014 26.3 (SE=2.977) and in 2017 30.4 (SE=1.814).

Looking at the rejuvenated area, species richness increased from 3.0 in 2012 to 9.3 in 2014 and additionally to 14.5 in 2017. The result in 2017 displayed a significant growth in species richness with time as an explanatory factor for the effect (P-value = <0.001). Observing the progression in figure 5, the largest increase happened in the first two years after clear-cutting, and afterwards a nearly equal progress for the next 3 years was observed.

The last open landscape habitat was the wet meadow, which increased in species richness over the 5 year period from 23.7 in 2012 to a maximum of 30.4 in 2017. The observed increase was significant with time as an explanatory factor (P-value=0.014). The species richness showed increases in both measurement after the baseline conduction in 2012, with a value of 26.3 in 2014 before reaching the highest measured value in 2017.

Comparing the three habitats, the highest species richness was found on the wet meadow for all five years. For all three habitats, the wet meadow also contained the highest baseline species richness in 2012. The lowest species richness for the wet meadow was also higher than the maximum of the other habitats. The rejuvenated area illustrated the highest increase in species richness over time with a rise of 383.3%, indicating the most cumulative tendency. The bush grass dominated area showed the lowest species richness and the nethermost increase compared to the two other habitats.

Beech forest

The species richness in the beech forest is showcased in figure 6, which exposed no significant changes through the 5-year period. In the grazed area, the species richness decreased from 2.8 in 2012 to 2.6 in 2014 and afterwards increased the last 3 years to 2.7 in 2017. These results directs an almost status quo in species richness after five years with grazing in the beech forest.
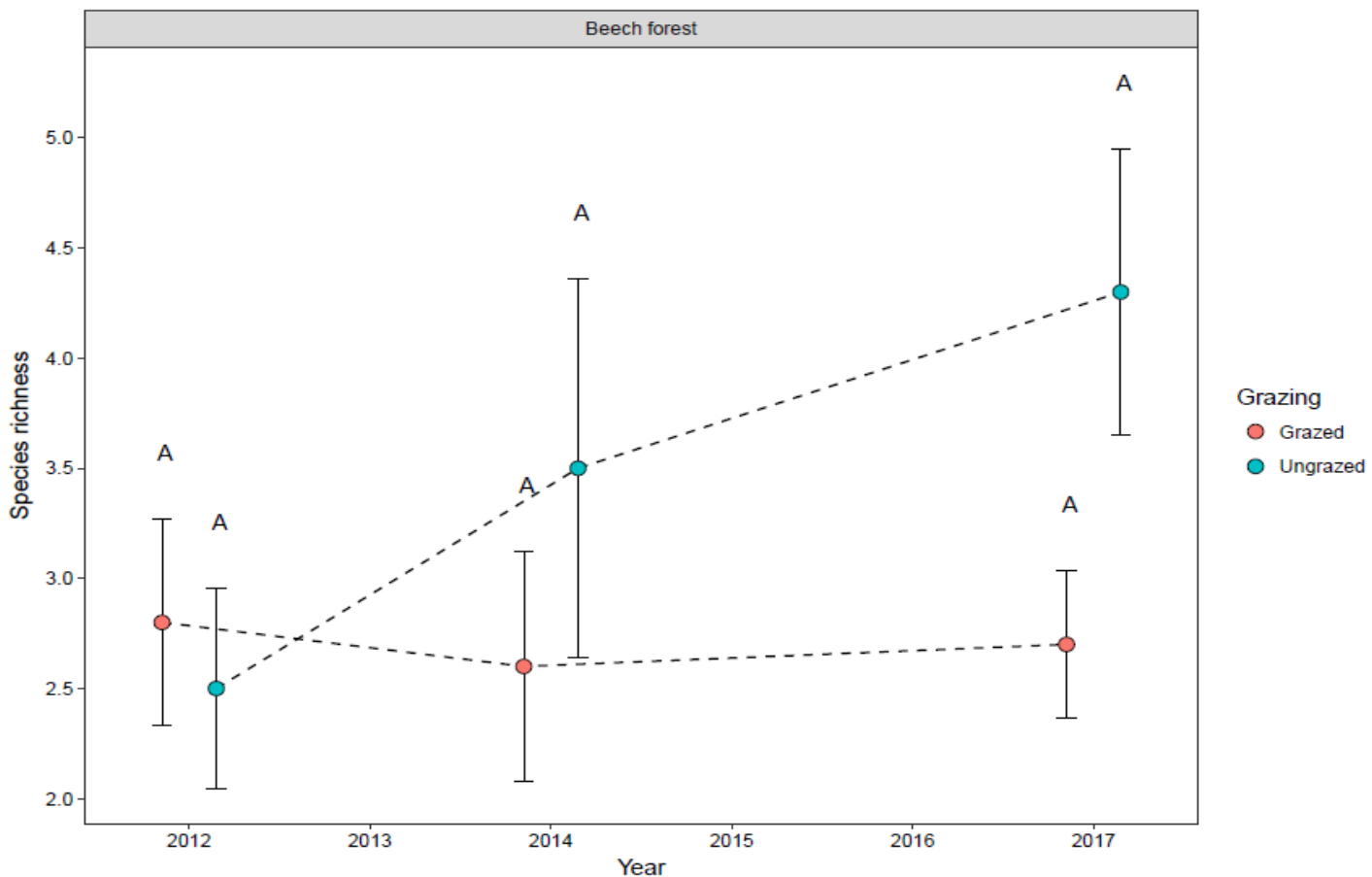
Figure 6. Average species richness (±SE) of the beech forest. Red dots represents species richness inside the bison fencing and blue dots represents species richness in the ungrazed area outside the fencing. Different letters represents different significant groups. The species richness in the grazed beech forest was 2.8 in 2012 (SE=0.466) 2.6 in 2014 SE=0.520) and 2.7 in 2017 (SE=0.334). In the ungrazed beech forest, the species richness was 2.5 in 2012 (SE=0.453), 3.5 in 2014 (SE=0.259) and in 2017 was it 4.3 (SE=0.650).

The species richness in the ungrazed beech forest did not show any significant changes neither. In the ungrazed area, the species richness though increased from 2.5 in 2012 to 3.5 in 2014 and furthermore to 4.3 in 2017. Even though the results showed no significant changes, the ungrazed area showed a higher species richness and a more increasing tendency compared to the grazed area. The species richness in 2012 was highest in the grazed area, but for the following two measurements, the highest values were found in the ungrazed area.

Alder swamp

Species richness of the alder swamp is presented in figure 7. This area endured a change in species richness from a minimum of 13.3 to a maximum of 18.0, though the baseline species richness was found to be 14.1 in 2012. The alder swamp experienced a decrease from 14.1 in 2012 to 13.3 in 2014. Afterwards, the species richness increased significantly from 2014 to 2017. Time was found as explanatory variable for the significant increase (P-value=0.0015). The results then illustrated a significant increase over a 3-year period from 2014 to 2017, since difference in species richness between 2012 and 2017 not would be sufficient to proclaim a significant change.
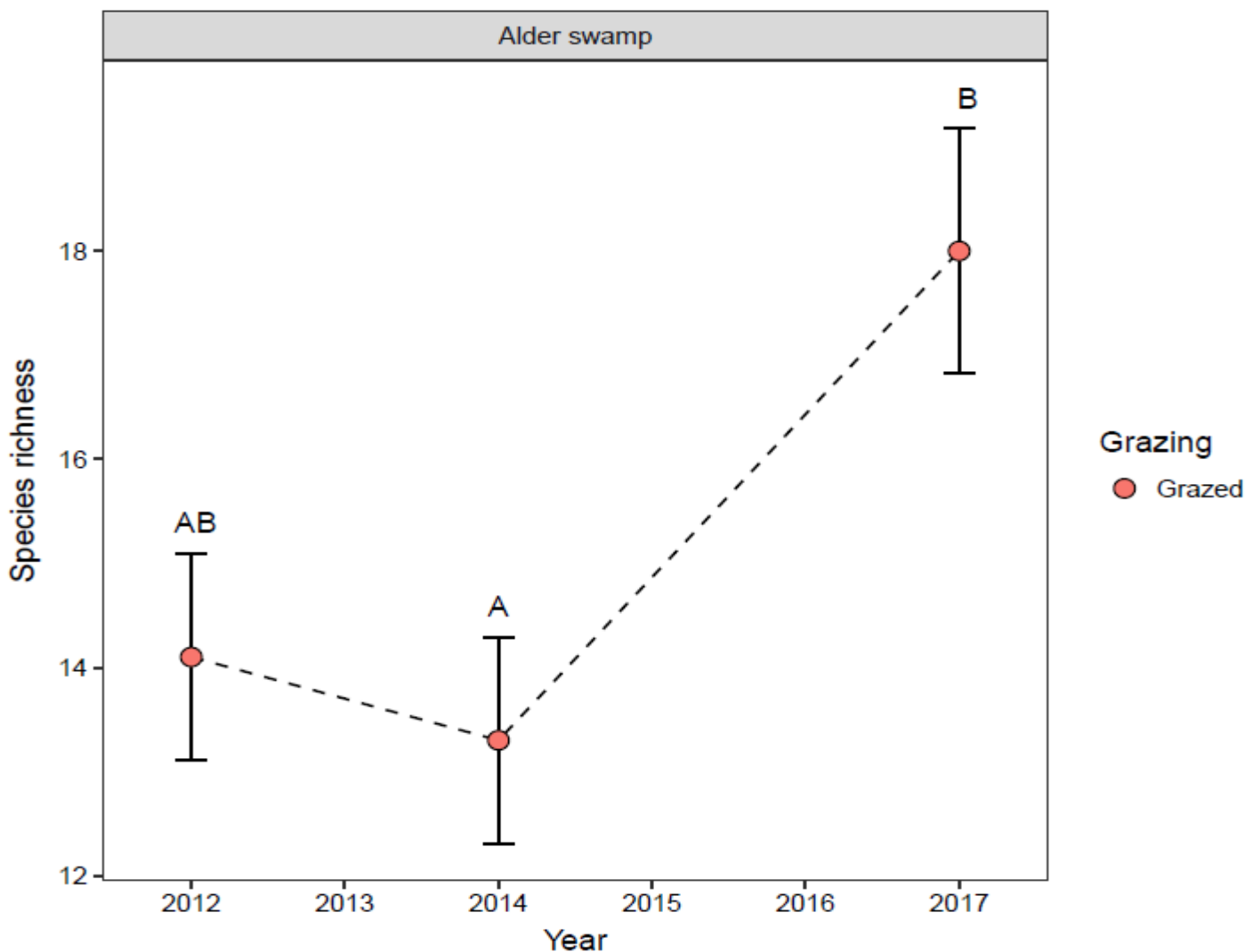
Figure 7. Average species richness (±SE) of the Alder swamp. Different letters represents different significant groups. The species richness was 14.1 in 2012 (SE=0.993), 13.3 in 2014 (SE=0.989) and 18.0 in 2017 (SE=1.173).

Oak forest

Species richness for the oak forest is illustrated in figure 8. The four defined oak forest areas all differed in species richness and development over the 5-year period. Regarding the middle-aged oak forest, no significant changes in species richness occurred over time, though an increase from 9.1 in 2012 to 9.4 in 2014 and furthermore to 11.9 in 2017 was observed.
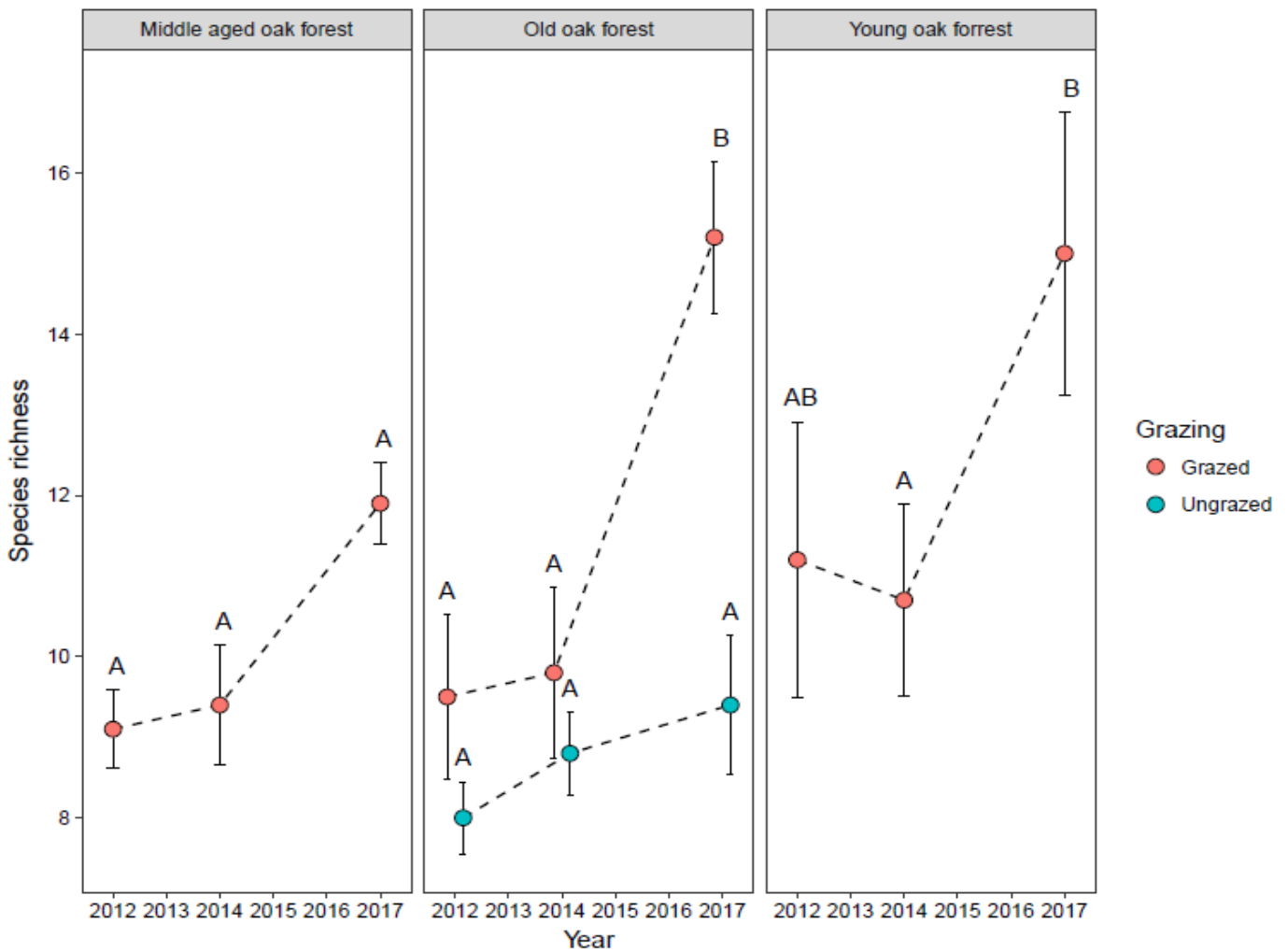
Figure 8. Average species richness for the ten plots (±SE) of the oak forest areas. Red dots represents species richness inside the bison fencing and blue dots represents species richness in the ungrazed area outside the fencing. Different letters represents different significant groups. The species richness in the middle-aged oak forest was in 2012 9.1 (SE=0.104) in 2014 was it 9.4 (SE=0.103) and in 2017 was it 11.9 (SE=0.091). The species richness in the grazed old oak forest was in 2012 9.5 (SE=0.102) in 2014 was it 9.8 (SE=0.101) and in 2017 was it 15.2 (SE=0.081). The species richness in the ungrazed old oak forest was in 2012 8.0 (SE=0.112) and in 2014 was it 8.8 (SE=0.107) and in 2017 was it 9.4 (SE=0.103). The species richness in the young oak forest was in 2012 11.2 (SE=0.211) and in 2014 was it 10.7 (SE=0.113) and in 2017 was it 15.0 (SE=0.200).

The old oak forest contained a grazed and ungrazed area. The grazed area showed a significant increase in species richness over the 5-yeaer period from 9.5 in 2012 to 15.2 in 2017, due to all three explanatory factors, time (P-value=<0.001), CC (P=0.001) and grazing activity (<0.001). The biggest increase was found from 2014 to 2017, where a rise from 9.8 to 15.2 was observed.

The ungrazed old oak forest did not display any significant changes in species richness during the 5-year period. Furthermore, the species richness was found to be lower in the ungrazed area than in the grazed for all three measurements. The results from the ungrazed area showed a species richness of 8.0 in 2012, 8.8 in 2014 and 9.4 in 2017. Even though no significant changes in species richness was observed, the results from the statistical model revealed, that when CC increases significantly, it will affect species richness to decrease significantly in this particular area (P-value=0.001).

In the young oak forest, the results showed a significant increase in species richness from 2014 to 2017, with both CC (P-value=<0.001) and time (P-value=<0.001) as explanatory factors. In 2012 the species richness was estimated to 11.2 which decreased to 10.7 in 2014. After a decrease the two first years, species richness then increased to 15.0 in 2017.

All four habitats showed increases in species richness over time, but the grazed old oak forest and the young oak forest demonstrated the highest increase during the five years. These two areas showed the highest estimate of species richness in the entire oak forest as well. The results evidences the ungrazed old oak forest as being the poorest oak forest habitat regarding species richness.

Norway spruce forest

Figure 9 illustrates the species richness for the Norway spruce forest. The Norway spruce forest contained three habitats, in which none revealed any significant changes during the 5-year period. The ungrazed old Norway spruce forest showed an almost even species richness with 1.8 in 2012, 1.6 in 2014 and 1.7 in 2017. The grazed old Norway spruce forest showed a higher and more increasing species richness compared to the ungrazed. The species richness increased from 2.3 in 2012 to 3.3 in 2014, where it also stagnated and showed similar value in 2017.
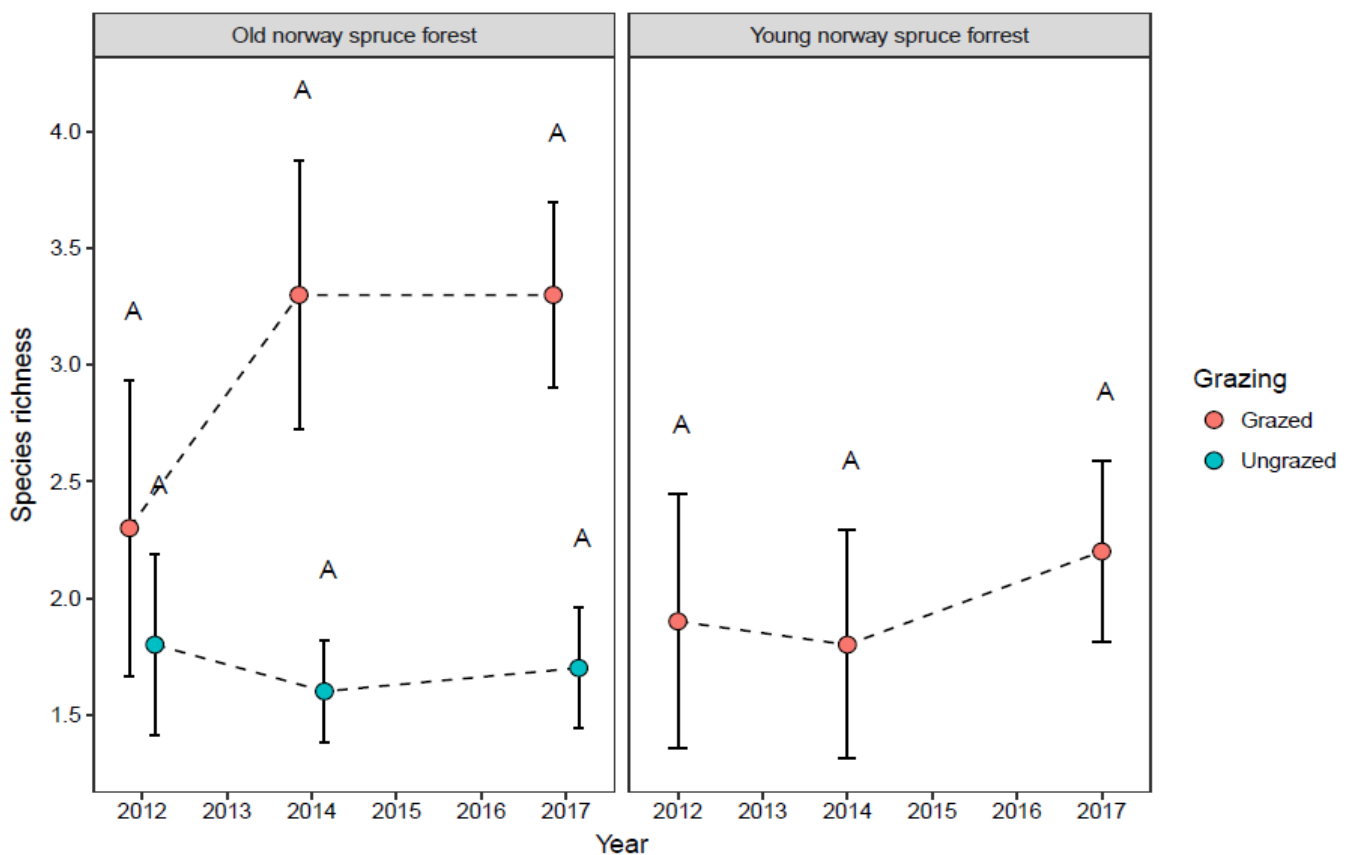
Figure 9. Average species richness for the ten plots (±SE) of the Norway spruce forest areas. Red dots represents species richness inside the bison fencing and blue dots represents species richness without grazing outside the fencing. Different letters represents different significant groups. The species richness in the grazed old Norway spruce forest was 2.3 in 2012 (SE=0.212), 3.3 in 2014 (SE=0.178) and 3.3 in 2017 (SE=0.178). The species richness in the ungrazed old Norway spruce forest was 1.8 in 2012 (SE=0.239), 1.6 in 2014 (SE=0.253) and 1.7 in 2017 (SE=0.246). The species richness in the young Norway spruce forest was 1.9 in 2012 (SE=0.288), 1.8 in 2014 (SE=0.288) and 2.2 in 2017 (SE=0.270).

The species richness in the young Norway spruce forest was found to be 1.9 in 2012, 1.8 in 2014 and 2.2 in 2017. Even though no significant changes in species richness occurred, CC was determined to affect the species richness significantly in the area (P-value= 0.015), whereas species richness decrease with increasing CC.

Compared to the grazed old Norway spruce forest, the young part showed both a lower richness and increase over time.

Ellenberg analyses

All plant species require a broad variety of environmental conditions to sustain growth and production. The levels of these conditions can be described through indicator values, and in this study, the Ellenberg indicator values are used. The Ellenberg indicator values (EIV) are an ordinal classification through an ecological gradient from 1-9. The scale places plant species along the environmental gradient based on their realized ecological niche, and thus provides knowledge regarding the ecological performance of species under competition. A plant species composition can therefore, through Ellenberg analyses, be an important tool of estimating abiotic key parameters of ecological conditions in any habitat.

The Ellenberg indicators applies more abiotic factors, but in this study we will only focus on two:

- Light value (L)
- Nitrogen value (N)

"1" on the L-scale refers to deep shade plant species, which require a minimum of light. "5" is intermediate plant species, and "9" are plant species, which are dependent of full light availability. "1" on the N-scale refers to plant species, which can only exist on the most nitrogen poor soils. "5" is intermediate and "9" is plant species, which demands an excessive supply of nitrogen to exist.

Based on current knowledge of grazing theory and the aim of this study, the most relevant values to investigate in this case is the light value and the nitrogen value. The purpose of grazing by is to enlarge the light availability for the wild flora, and it is therefore relevant to investigate this factor, to reveal any possible changes. Inhabiting of herbivores with the purpose of grazing is also likely to affect the nutrient cycle in the area (Schuman, Reeder et al. 1999, Pedersen, Buttenschøn et al. 2001, Buttenschøn 2007, Piñeiro, Paruelo et al. 2010). Besides the natural supply and removal of nitrogen in a closed environment, grazing by domestic or wild animals can affect this cycle by e.g. excrements, supply feedings, removal of plant litter, consuming of plants etc. Consequently, the nitrogen indicator is relevant to examine for any potential alterations.

It is not realistic to believe that this 5-year grazing experiment could force any changes in temperature, continentality, moist or soil reaction. Therefore, none of these factors are investigated.

For this analysis, the accumulated values from the Raunkjær specie analyses was used to determine the frequency of the plants and apply weighted EIV. The Ellenberg values used, comes from an internal look-up created by Torben Riis-Nielsen (Senior advisor at the University of Copenhagen) and Ib Johnsen (Associate professor emeritus at the University of Copenhagen), where most values are extracted from Ellenberg's indicator values for British plants (Hill, Mountford et al. 1999). Mosses and lichens are not included, since they were not determined to specie level.

Open land

The EIV, for the three open landscape types is illustrated in figure 10. Here it appeared that significant differences occurred in all three habitats. In the bush grass dominated area, no significant changes transpired for the EIV-L, which was found to be 6.77 in 2012, 6.73 in 2014 and 6.75 in 2017. Contrarily, the bush grass dominated area showed a significant increase in the EIV-N (P-value=0.016), with an assessment of 5.33 in 2012, 5.44 in 2014 and 5.60 in 2017.

In the rejuvenated area, the significant change was found for the EIV-L (P-value=0.014), with an increase from 4.97 in 2012 to 6.54 in 2014. After the increase towards 2014, the EIV-L decreased to 6.31 in 2017. For the EIV-N, no significant differences occurred during the 5-year period. Nevertheless, a decrease from 5.46 in 2012 to 5.09 in 2014 and furthermore to 4.59 in 2017 was observed.
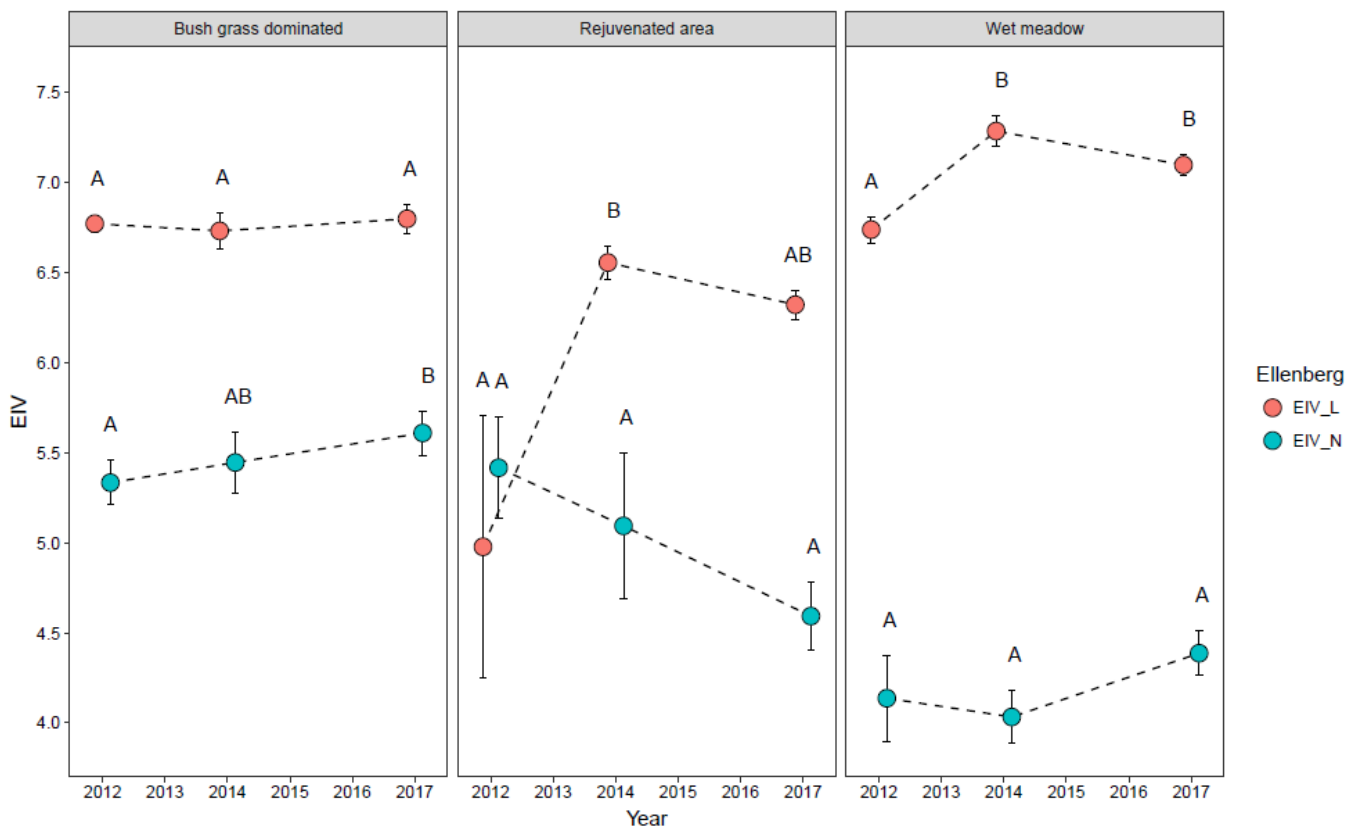
Figure 10. Weighted Ellenberg indicator values (EIV) for light and nitrogen (±SE), of the three open landscape vegetation types in the bison fencing. Different letters represents different significant groups. The EIV-L in the bush grass dominated area was in 2012 6.77 (SE=0.080) in 2014 was it 6.73 (SE=0.101) and in 2017 was it 6.75 (SE=0.088). The EIV-N in the bush grass dominated area was in 2012 5.33 (SE=0.140) in 2014 was it 5.44 (SE=0.195) and in 2017 was it 5.60 (SE=0.134). The EIV-L in the rejuvenated area was in 2012 4.97 (SE=0.864) in 2014 was it 6.54 (SE=0.105) and in 2017 was it 6.31 (SE=0.080). The EIV-N in the rejuvenated area was in 2012 5.46 (SE=0.314) in 2014 was it 5.09 (SE=0.456) and in 2017 was it 4.59 (SE=0.210). The EIV-L on the wet meadow was in 2012 6.73 (SE=0.073) in 2014 was it 7.27 (SE=0.098) and in 2017 was it 7.08 (SE=0.044). The EIV-N on the wet meadow was in 2012 4.13 (SE=0.236) in 2014 was it 4.03 (SE=0.189) and in 2017 was it 4.38 (SE=0.164).

On the wet meadow, the EIV-L increased significantly over time (P-value=<0.001), from 6.73 in 2012 to 7.27 in 2014. After the peak in 2014, it decreased to 7.08 in 2017. The EIV-N showed no significant change, but increased from 4.13 in 2012 to 4.03 in 2014 and furthermore to 4.38 in 2017.

Comparing the three habitats, both similarities and differences are exposed by the results. Except from the EIV-L found in the rejuvenated area in 2012, the other EIV-L values showed higher similarities across the three habitats than the EIV-N. Nevertheless, the results evidenced the wet meadow as the most light demanding community of the three habitats. Consequently, the wet meadow also showed the lowest EIV-N, while the bush grass dominated area demonstrated the highest EIV-N. The biggest differences in minimum and maximum values for both EIV's was found in the rejuvenated area, which displayed the biggest increase in EIV-L and the biggest decrease in EIV-N.

Beech forest

The EIV's for the beech forest is illustrated in figure 11, and demonstrated no significant changes in either the EIV-L or EIV-N during the 5 years, in neither the grazed or ungrazed area. In the grazed area, the EIV-L was estimated to 2.06 in 2012, 2.19 in 2014 and 2.17 in 2017. The EIV-N also demonstrated a flat curve with 6.37 in 2012, 6.43 in 2014 and 6.22 in 2017.

The ungrazed beech forest illustrated an EIV-L of 2.84 in 2012, 3.18 in 2014 before decreasing to the minimum of 2.78 in 2017. The EIV-L in the ungrazed area was subsequently higher than the grazed area, during the entire study. The EIV-N showed a more similar pattern compared to the grazed area regarding values. In 2012, the ungrazed area displayed an EIV-N of 6.53, in 2014 was it 6.23 and in 2017 was it 6.23.

Figure 11. Weighted Ellenberg indicator values (EIV) for light and nitrogen (±SE), of the beech forest. Different letters represents different significant groups. The EIV-L in the grazed beech forest was in 2012 2.06 (SE=0.310) and in 2014 was it 2.19 (SE=0.410) and in 2017 was it 2.17 (SE=0.221). The EIV-N in the grazed beech forest was in 2012 6.37 (SE=0.504) and in 2014 was it 6.43 (SE=0.511) and in 2017 was it 6.22 (SE=0.199). The EIV-L in the ungrazed beech forest was in 2012 2.84 (SE=0.731) and in 2014 was it 3.18 (SE=0.535) and in 2017 was it 2.78 (SE=0.412). The EIV-N in the ungrazed beech forest was in 2012 6.53 (SE=0.315) and in 2014 was it 6.11 (SE=0.269) and in 2017 was it 6.23 (SE=0.287).

Alder swamp

EIV's for the alder swamp is showcased in figure 12, where no significant changes occurred over time for neither of the EIV's. However, the EIV-L showed an development from 4.58 in 2012 to 4.38 in 2014 and 5.10 in 2017. Corresponding, the EIV-N showed an almost similar pattern, just with decreasing tendency, from 5.76 in 2012 to 5.93 in 2014 and 5.33 in 2017.

Both EIVs showed biggest change the last 3 years, with respectively the biggest increase and decrease.
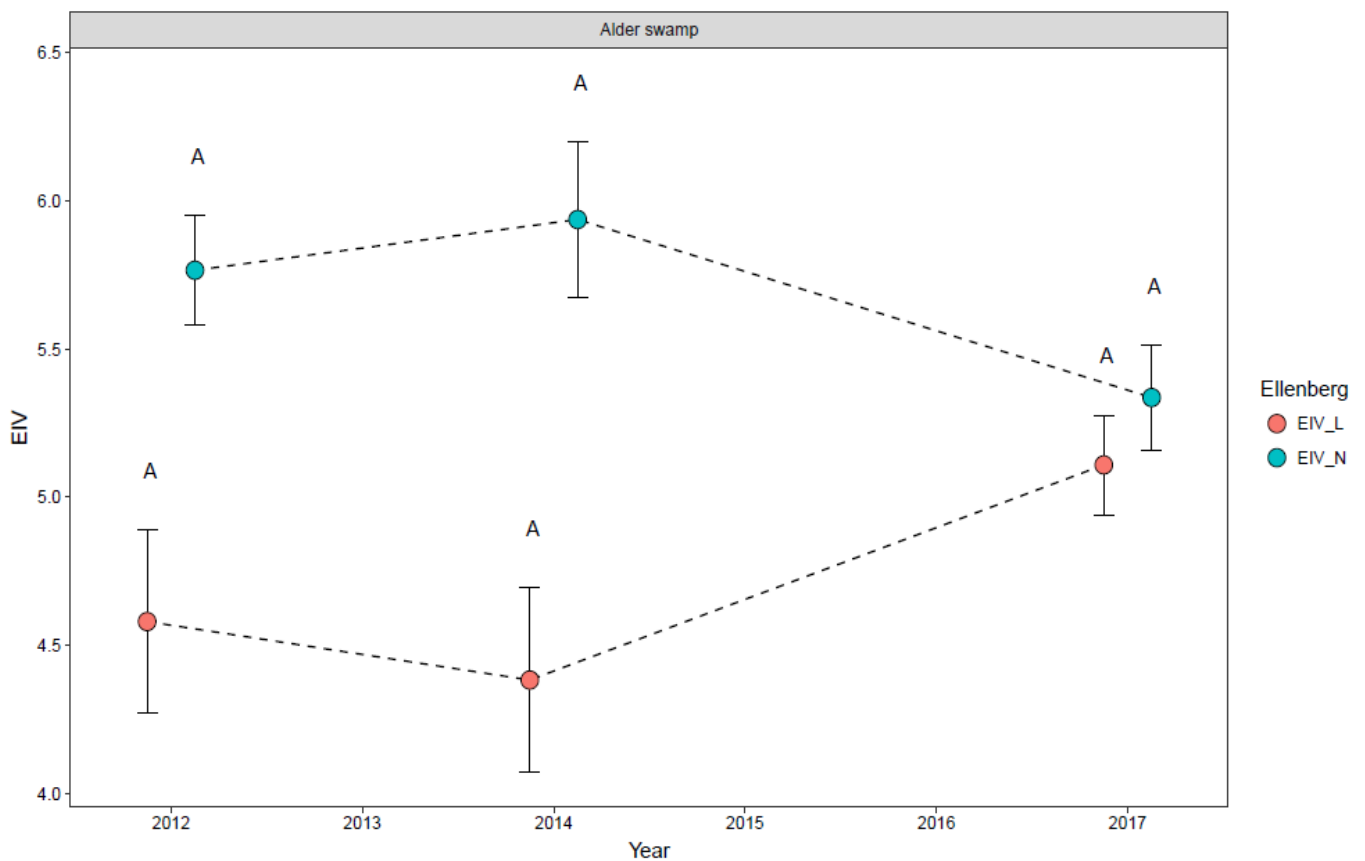
Figure 12. Weighted Ellenberg indicator values (EIV) for light and nitrogen (±SE) of the Alder swamp. Different letters represents different significant groups. The EIV-L was in 2012 4.58 (SE=0.270), in 2014 was it 4.38 (SE=0.306) and in 2017 was it 5.10 (SE=0.210). The EIV-N was in 2012 5.76 (SE=0.210), in 2014 was it 5.93 (SE=0.260) and in 2017 was it 5.33 (SE=0.208).

Oak forest

Figure 13 represent the EIVs for the oak forest habitats. Two of the four oak forest areas differed significantly in EIV-L over time, whereas the two other habitats did not show any significant changes at all.

In the middle-aged oak forest, we found a significant increase in EIV-L from 2014 to 2017 (P-value=0.001). In 2012 the EIV-L was found at 4.61 where it decreased to 4.31 in 2014. For the last three years, the EIV-L increased and reached the highest measured value of 4.77 in 2017. Regarding the EIV-N, the middle-aged oak forest did not show any significant changes over time.
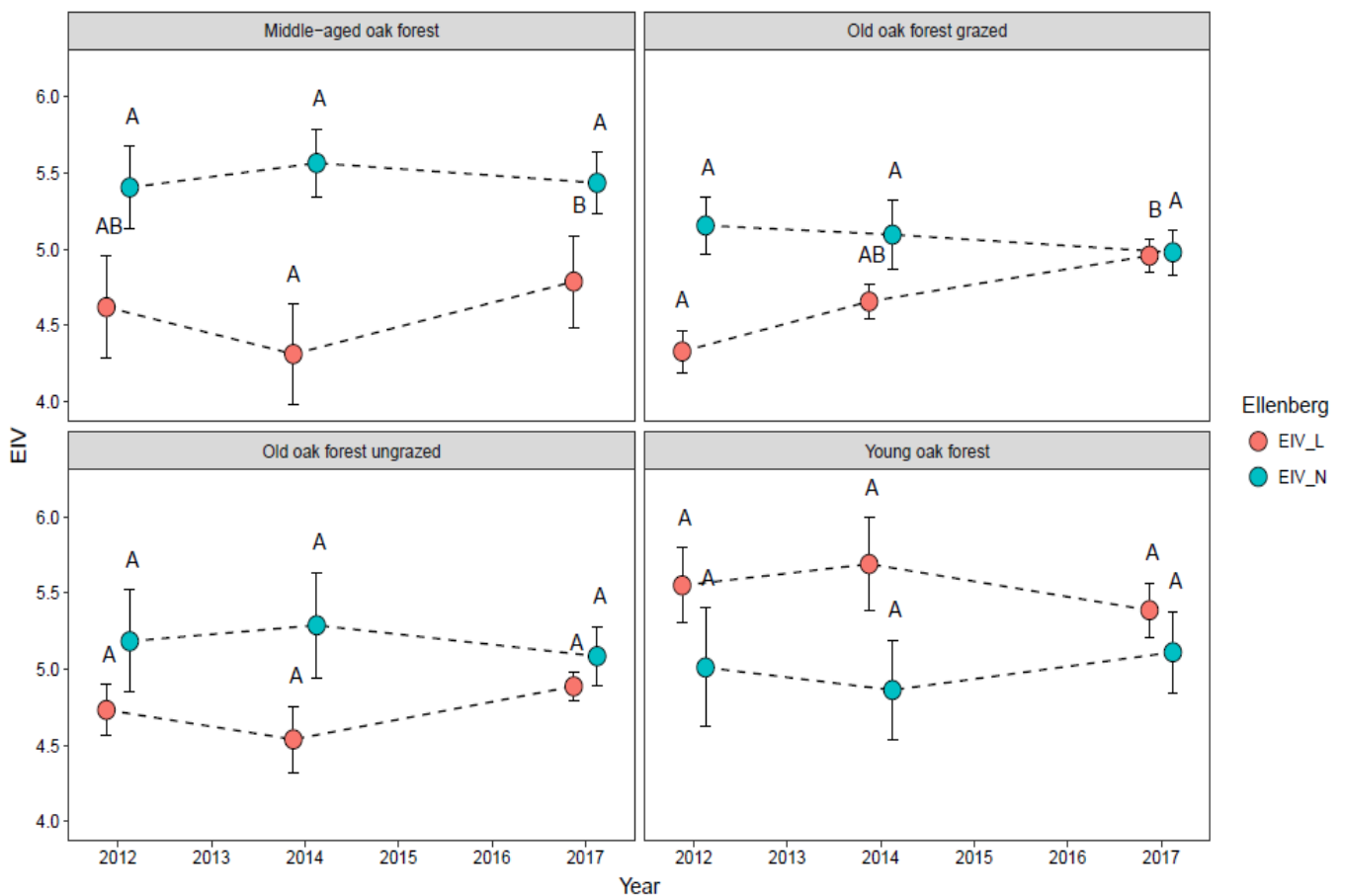
Figure 13. Weighted Ellenberg indicator values (EIV) for light and nitrogen (±SE), of the oak forest areas. Different letters represents different significant groups. The EIV-L in the middle-aged oak forest was in 2012 4.61 (SE=0.321) in 2014 was it 4.31 (SE=0.332) and in 2017 was it 4.77 (SE=0.310). The EIV-N in the middle-aged oak forest was in 2012 5.40 (SE=0.233) in 2014 was it 5.56 (SE=0.208) and in 2017 was it 5.43 (SE=0.200). The EIV-L in the grazed old oak forest was in 2012 4.32 (SE=0.118) in 2014 was it 4.64 (SE=0.095) and in 2017 was it 4.94 (SE=0.099). The EIV-N in the grazed old oak forest was in 2012 5.15 (SE=0.189) in 2014 was it 5.09 (SE=0.210) and in 2017 was it 4.98 (SE=0.120). The EIV-L in the ungrazed old oak forest was in 2012 4.73 (SE=0.168) in 2014 was it 4.54 (SE=0.2 11) and in 2017 was it 4.88 (SE=0.067). The EIV-N in the ungrazed old oak forest was in 2012 5.18 (SE=0.300) in 2014 was it 5.28 (SE=0.288) and in 2017 was it 5.09 (SE=0.191). The EIV-L in the young oak forest was in 2012 5.55 (SE=0.250) in 2014 was it 5.69 (SE=0.272) and in 2017 was it 5.39 (SE=0.188). The EIV-N in the young oak forest was in 2012 5.01 (SE=0.332) in 2014 was it 4.87 (SE=0.311) and in 2017 was it 5.11 (SE=0.241).

The grazed old oak forest increased significantly in EIV-L over the five years (P-value=0.017), from 4.32 in 2012 to 4.64 in 2014 and furthermore to 4.94 in 2017. The EIV-N decreased from 5.15 in 2012 to 5.09 in 2014 and further to 4.98 in 2017, but the change was not significant.

The ungrazed old oak forest did not display any significant changes in either the EIV-L or EIV-N. In 2012, the EIV-L was 4.73, in 2014 was it 4.54 and in 2017 was it 4.88. The EIV-N in the ungrazed old oak forest was 5.18 in 2012, 5.28 in 2014 and 5.09 in 2017.

No significant changes occurred in the young oak forest. The EIV-L was in 2012 estimated to 5.55, in 2014 was it 5.69 and in 2017 was it 5.39. The EIV-N showed a reverse pattern with an overall increase from 5.01 in 2012 to 4.87 in 2014 and then 5.11 in 2017.

In the middle-aged oak forest and both the grazed and ungrazed old oak forest, higher values of EIV-N was found compared to EIV-L. Looking at the young oak forest, the opposite pattern was observed, where a higher EIV-L compared to EIV-N was calculated for all three measurements. The EIV-N in the young oak forest resembled the values from the other oak forest habitats, but the EIV-L was found at higher levels. Both EIV-L and EIV-N were comparable in the middle-aged oak forest and the two old oak forest habitats.

Norway spruce forest

The EIV's for the Norway spruce forest are showcased in figure 14. In the Norway spruce forest, both the grazed and ungrazed old part showed significant changes regarding the EIV-L. The young Norway spruce forest showed no significant change over time.

In the grazed old Norway spruce forest, the EIV-L decreased significantly (P-value=0.004) from 5.84 in 2012 to 4.76 in 2014 and additionally to 3.83 in 2017. The EIV-N in the grazed old Norway spruce forest did not differ significantly, but increased from 4.58 in 2012 to 5.45 in 2014 and decreased to 5.11 in 2017.

In the ungrazed old Norway spruce forest, EIV-L increased significantly (P-value=<0.001) from 5.06 in 2012 to 5.57 in 2014 and furthermore to 5.76 in 2017. The EIV-N did not show any significant change, but decreased from 3.34 in 2012 to 2.97 in 2014 and increased to 3.00 in 2017.
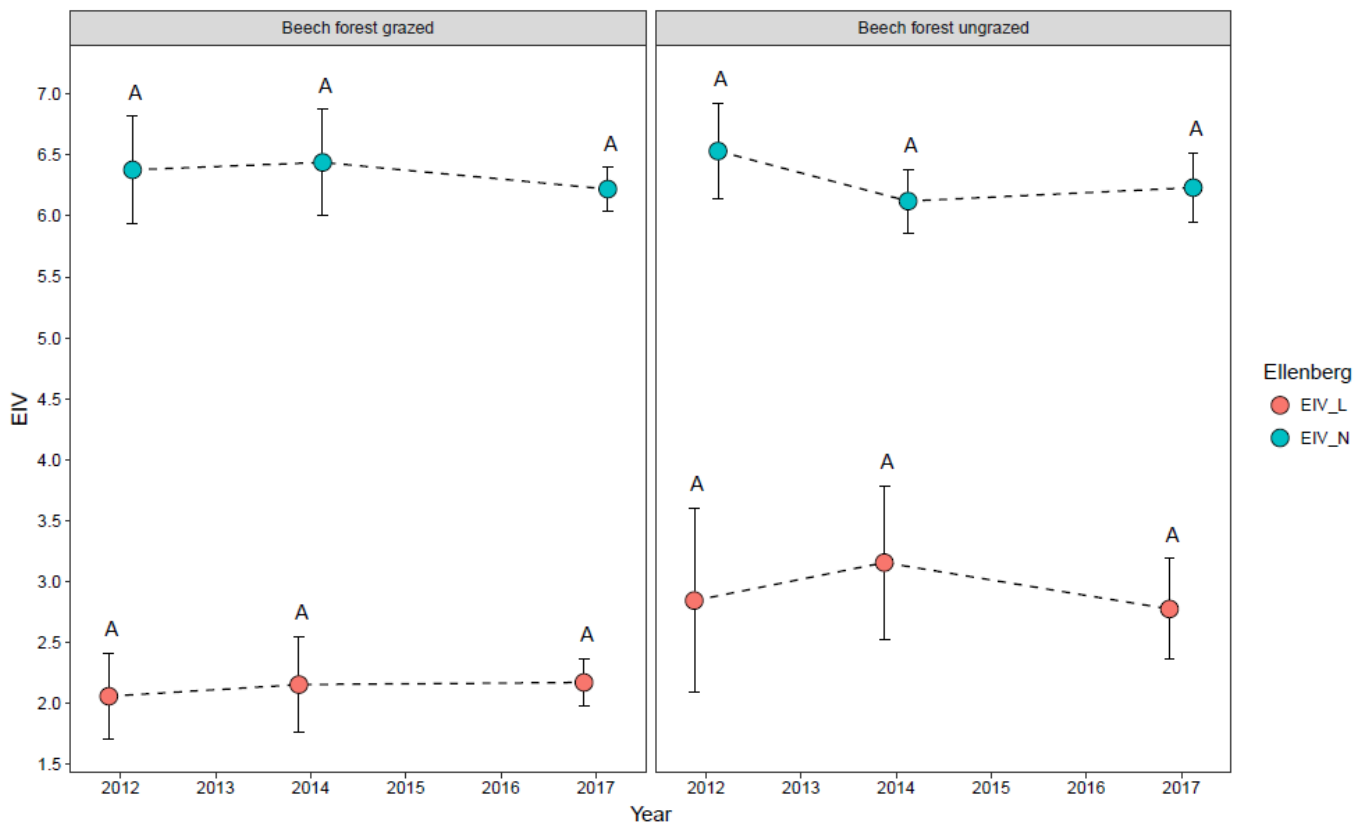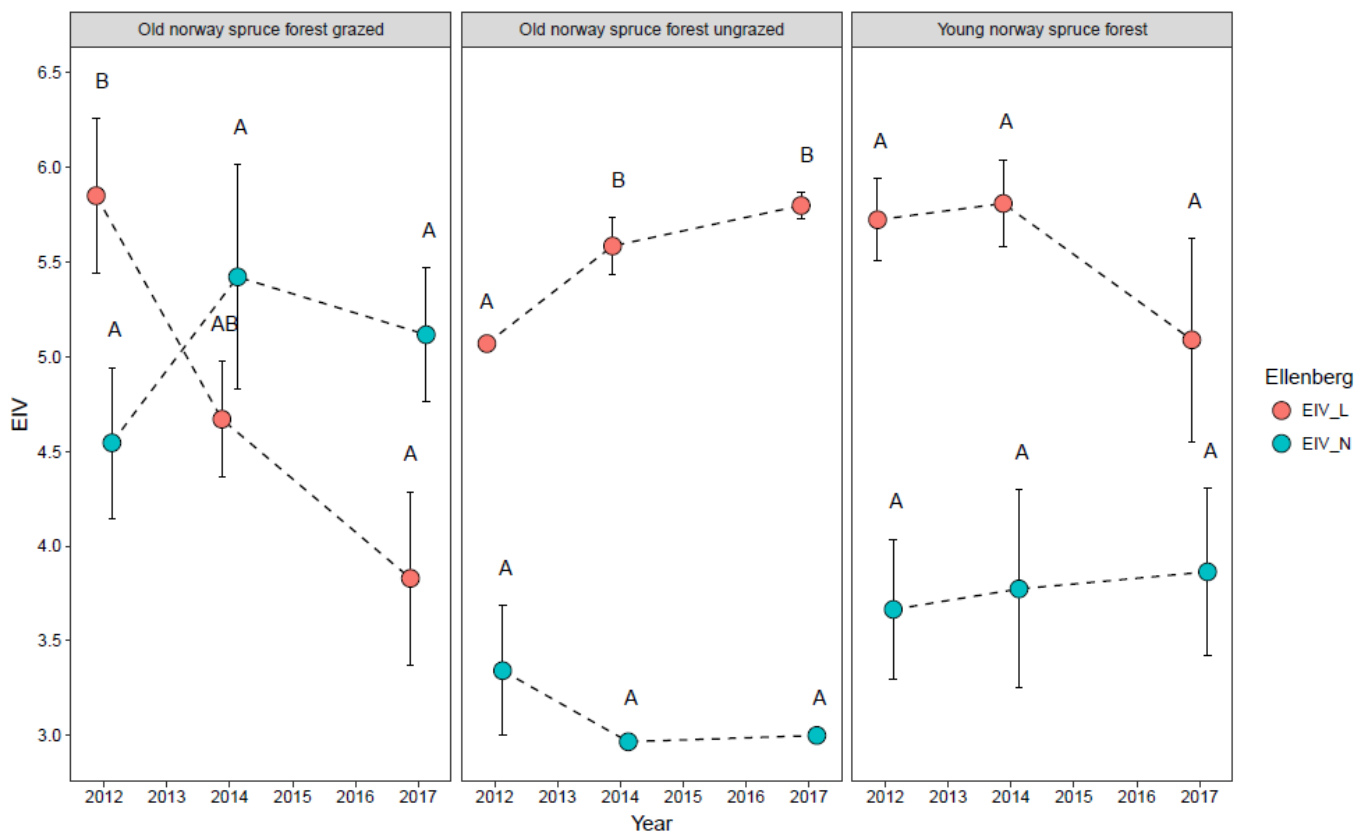

Figure 14. Weighted Ellenberg indicator values (EIV) for light and nitrogen (±SE), of the Norway spruce forest areas. Different letters represents different significant groups. The EIV-L in the grazed old Norway spruce forest was in 2012 5.84 (SE=0.451), in 2014 was it 4.76 (SE=0.373) and in 2017 was it 3.83 (SE=0.508). The EIV-N in the grazed old Norway spruce forest was in 2012 4.58 (SE=0.497), in 2014 was it 5.45 (SE=0.624) and in 2017 was it 5.11 (SE=0.490). The EIV-L in the ungrazed old Norway spruce forest was in 2012 5.06 (SE=0.096), in 2014 was it 5.57 (SE=0.104) and in 2017 was it 5.78 (SE=0.077). The EIV-N in the ungrazed old Norway spruce forest was in 2012 3.34 (SE=0.495), in 2014 was it 2.97 (SE=0.032) and in 2017 was it 3.00 (SE=0.000). The EIV-L in the young Norway spruce forest was in 2012 5.39 (SE=0.244), in 2014 was it 5.42 (SE=0.241) and in 2017 was it 5.14 (SE=0.595). The EIV-N in the young Norway spruce forest was in 2012 3.85 (SE=0.497), in 2014 was it 3.95 (SE=0.597) and in 2017 was it 3.86 (SE=0.502).

In the young Norway spruce forest, the results did not reveal any significant changes over time in any of the EIV's. However, EIV-L showed an increase from 5.39 in 2012 to 5.42 in 2014 and then a decrease to 5.14 in 2017. The EIV-N showed a change from 3.85 in 2012 to 3.95 in 2014 and 3.86 in 2017.

Looking at the grazed old Norway spruce forest, the EIV's and development through time differed compared to the ungrazed old Norway spruce forest and the young Norway spruce forest. The EIV-L in the grazed old Norway spruce decreased more than the two other areas and also demonstrated a lower EIV-L than EIV-N in the last measurement of 2017. The grazed old Norway spruce forest did also show a higher EIV-N for all three measurements, compared to the two other habitats. Even though EIV-L in the grazed old Norway spruce was found at lowest points in 2014 and 2017, the highest value across all three areas was found in 2012 in this area.

Species abundance and biodiversity indices

In this section, figures illustrating the most abundant plant species in all vegetation types is presented. For the habitats with both grazed and ungrazed areas, a comparison is illustrated. Dependent on habitat type, two or three species is illustrated, since richness and relevance differed among habitats. The relative abundance are based on presence/absence data from the 100 throws with the Raunkjær circle in each habitat. In appendix 3, more figures are showed, with focus on progressing and declining species in the most relevant areas. All species and related frequencies for the three years with measurements are found in appendix 4, where the accumulated values from the Raunkjær circle are listed.

For all habitats and years, the diversity indices *Pielou*, *Simpsons* and *Shannon Wiener* are also illustrated in tables – These calculations are based on the accumulated Raunkjær values:

- **Pielou (J):** Pielou describes evenness or equitability of assemblages of individuals among the species, and answers the question how evenly distributed are the species? The value of Pielou ranges between 0 and 1, where 0 represents a total dominance and 1 represents a total evenness of species in the sample.

- **Simpsons (D):** Simpson's diversity Index is a measure of diversity. In ecology, it is often used to quantify the biodiversity of a habitat. It takes into account the number of species present, as well as the abundance of each species. Simpsons index measures the probability that two individuals randomly selected from a sample will belong to the same species. In this index, 0 represents infinite diversity and 1 represents no diversity. The values is also often referred to as dominance, where a higher value indicates a higher dominance of single species within the sample.

- **Shannon Wiener (H):** The Shannon diversity index is another index that is commonly used to characterize species diversity in a community. Like Simpson's index, Shannon's index accounts for both abundance and evenness of the species present. The value describes how difficult it would be to correctly predict the species of the next individuals correctly. A lower value reflects a lower diversity, which is then easier to predict and vice versa.

Open land

In 2012 when the rejuvenated area was clear-cutted from a vegetation of Norway spruce, the abundance of any plant species was low, such as the species richness. Figure 15 illustrates the most abundant species in the rejuvenated area in 2017 and their development through the 5 years.

As evidenced by the figure, the three species all increased, where the relative abundance of *Calamgrostis epigejos* was estimated to 1 in 2012, 19 in 2014 and 52 in 2017. *Rubus idaeus* showed a relative abundance of 3 in 2012, 39 in 2014 and 66 in 2017. *Deschampsia flexuosa* increased to the highest level of all species, from 3 in 2012, to 51 in 2014 and 79 in 2017. The increasing pattern for the illustrated species, corresponds for the majority of the species in the concerned area, where most species increased.
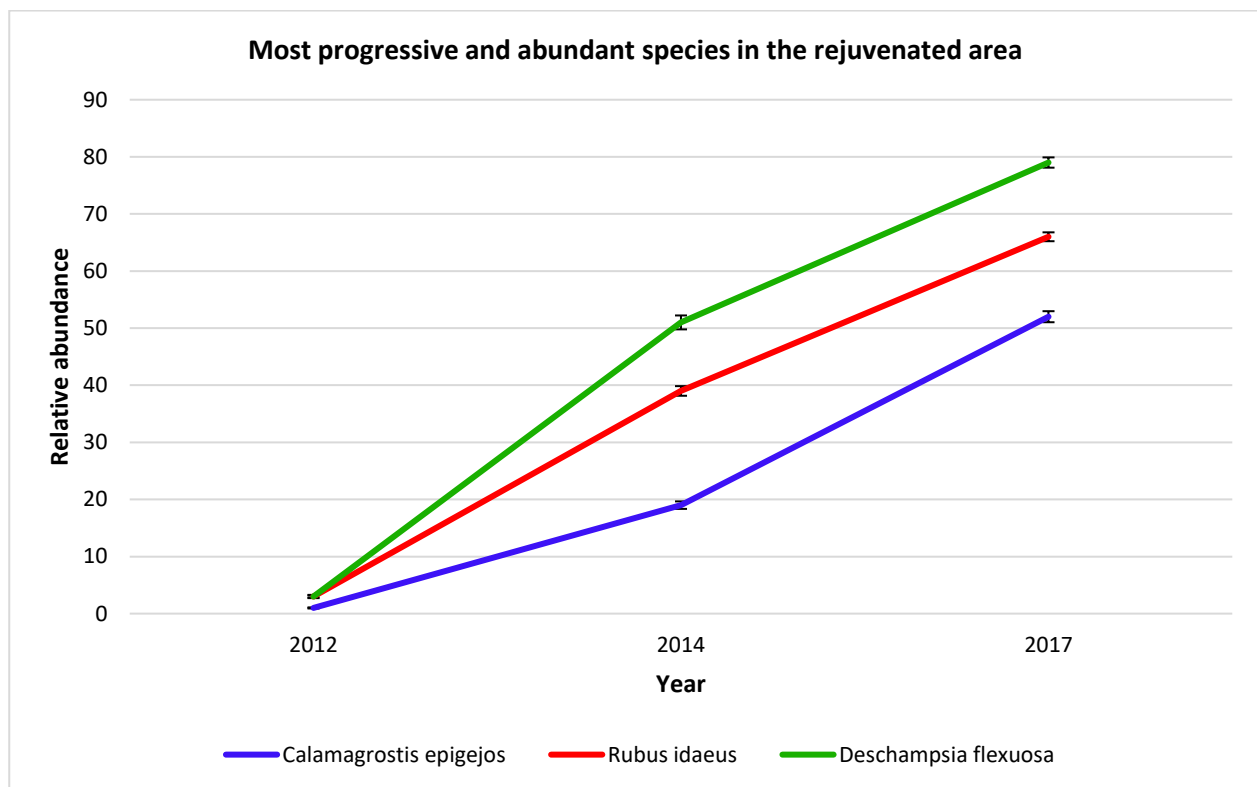
Figure 15. The most abundant species in the rejuvenated area over the 5-year period (±SE). Relative abundance of *Calamagrostis epigejos* was in 2012 1 (SE=0.094), in 2014 was it 19 (SE=0.654) and in 2017 was it 52 (SE=0.967). Relative abundance of *Rubus idaeus* was in 2012 3 (SE=0.202), in 2014 was it 39 (SE=0.842) and in 2017 was it 66 (SE=0.777). Relative abundance of *Deschampsia flexuosa* was in 2012 3 (SE=0.284), in 2014 was it 51 (SE=1.220) and in 2017 was it 79 (SE=0.899).

On the wet meadow, three species demonstrated an abundance above 60 in 2017, as illustrated in figure 16. *Mentha aquatica* was not initiated in 2012 and 2014, but was found in 65 out of 100 throws with the Raunkjær circle in 2017. *Equisetum fluviatile* appeared with a relative abundance of 84 in 2012, 70 in 2014 and 71 in 2017. *Equisetum palustre* was found at a stable level for the entire 5-year period with a relative abundance of 65 in 2012, 68 in 2014 and 66 in 2017.

Except from the *Mentha aquatica*, most species with a well-established presence above 40 in relative abundance in 2012, showed a steady level of abundance through the 5-year period with a minimum of variance, like the two species from the genus of *Equisetum.*
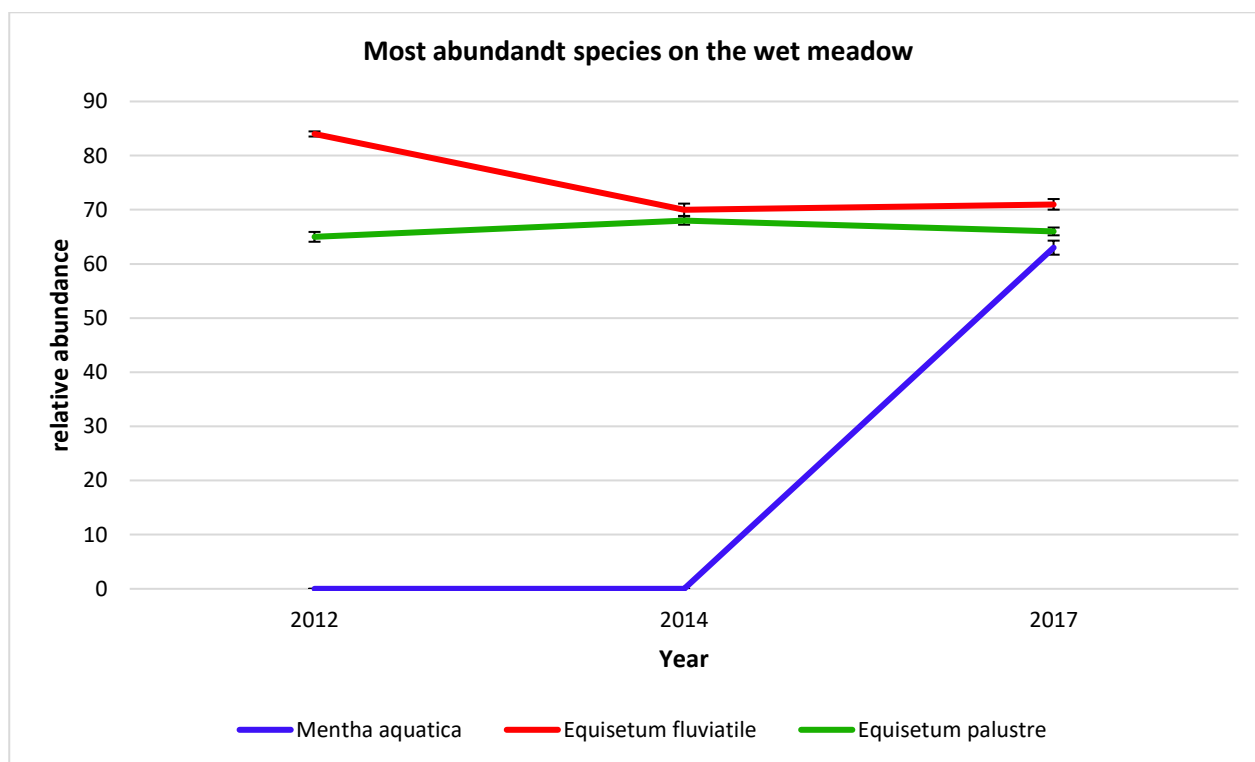
Figure 16. The most abundant species on the wet meadow over the 5-year period (±SE). Relative abundance of *Mentha aquatica* was in 2012 0 (SE=0.000), in 2014 was it 0 (SE=0.000) and in 2017 was it 65 (SE=1.296). Relative abundance of *Equisetum fluviatile* was in 2012 84 (SE=0.473), in 2014 was it 70 (SE=1.135) and in 2017 was it 71 (SE=0.984). Relative abundance of *Equisetum palustre* was in 2012 65 (SE=0.908), in 2014 was it 68 (SE=0.758) and in 2017 was it 66 (SE=0.723).

In the bush grass dominated area, *Calamagrostis epigejos* showed the highest abundance as shown in figure 17. The relative abundance was close to 100 during the entire 5-year period, but was 98 in both 2012 and 2014 before maximizing to 100 in 2017. The relative abundance of *Rubus idaeus* was 19 in 2012, 17 in 2014, before increasing to 43 in 2017. The relative abundance of the graminoid *Deschampsia flexuosa* was 40 in 2012, 41 in 2014 and 49 in 2017. All three species therefore showed an overall increase during the study extent.
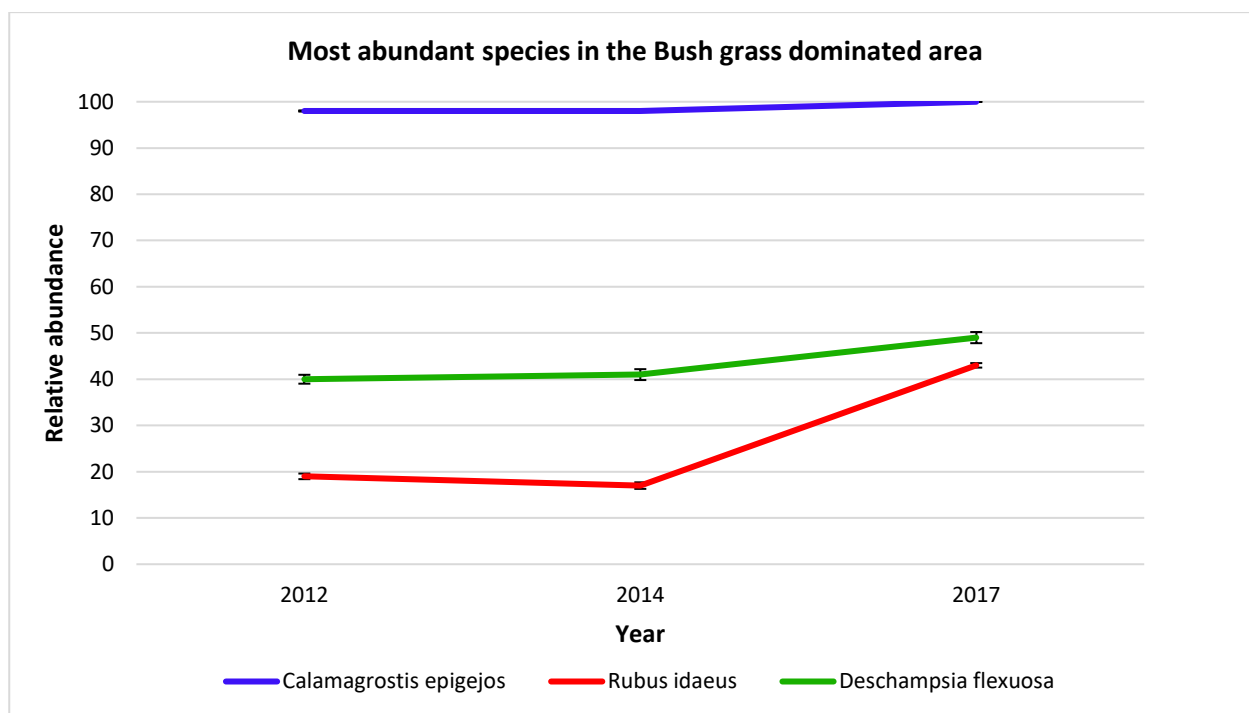
Figure 17. The most abundant species in the bush grass dominated area over the 5-year period (±SE). Relative abundance of *Calamagrostis epigejos* was in 2012 98 (SE=0.126), in 2014 was it 98 (SE=0.126) and in 2017 was it 100 (SE=0.000). Relative abundance of *Rubus idaeus* was in 2012 19 (SE=0.607), in 2014 was it 17 (SE=0.707) and in 2017 was it 43 (SE=0.485). Relative abundance of *Deschampsia flexuosa* was in 2012 40 (SE=0.959), in 2014 was it 41 (SE=1.178) and in 2017 was it 49 (SE=1.206).

In table 8, biodiversity indices for the three open land habitats is visualized and compared. The wet meadow contained the highest number of species and the highest rate of evenness reflected in the values of Pielou. The Simpson and Shannon Wiener values also indicated that the lowest dominance and most diverse habitat in the open land was found on the wet meadow.

Table 8. Biodiversity indices for the open land habitats.

	Rejuvenated area			Wet meadow			Bush grass dominated		
	2012	2014	2017	2012	2014	2017	2012	2014	2017
Number of species	17	25	42	65	69	66	13	16	15
Pielou	0.758	0.760	0.701	0.829	0.853	0.893	0.488	0.556	0.620
Simpson	0.184	0.117	0.115	0.048	0.039	0.032	0.418	0.328	0.278
Shannon Wiener	2.148	2.447	2.620	3.459	3.610	3.743	1.251	1.541	1.680

The bush grass dominated area showed the lowest number of species, and at the same time the highest dominance, due to the high abundance of the *Calamagrostis epigejos*. Though indicates the values for the three measurements a development over time going towards a lower dominance and a higher diversity and evenness in the bush grass dominated area. Both the Shannon Wiener and Pielou values increased over time while the Simpson value decreased.

The rejuvenated area showed an overall increase in number of species from 17 in 2012 to 42 in 2017. At the same time, dominance decreased as shown by the Simpson index indicating a higher diversity, also reflected by the increasing Shannon Wiener value. The evenness of the area did not differ between 2012 and 2014, even though the number of species increased, but instead decreased the following 3 years until 2017

For all three habitats, the highest Shannon Wiener values were found in 2017 and the lowest Simpson values likewise. These results therefore indicated that highest biodiversity of all three habitats occurred in the end of the 5-year period.

Beech forest

In figure 18, the relative abundance of *Oxalis acetosella* in the beech forest is showed, which was the most abundant species in the area. The relative abundance in the grazed area was 50 in 2012, 58 in 2014 and 53 in 2017. In the ungrazed area, the abundance was lower for all three years where measurements were made. In 2012, the abundance was 42, in 2014 was it 48, before it decreased to 30 in 2017. *Oxalis acetosella* increased in both areas from 2012 to 2014, before showing a decrease in both areas as well, but most profound in the ungrazed area.
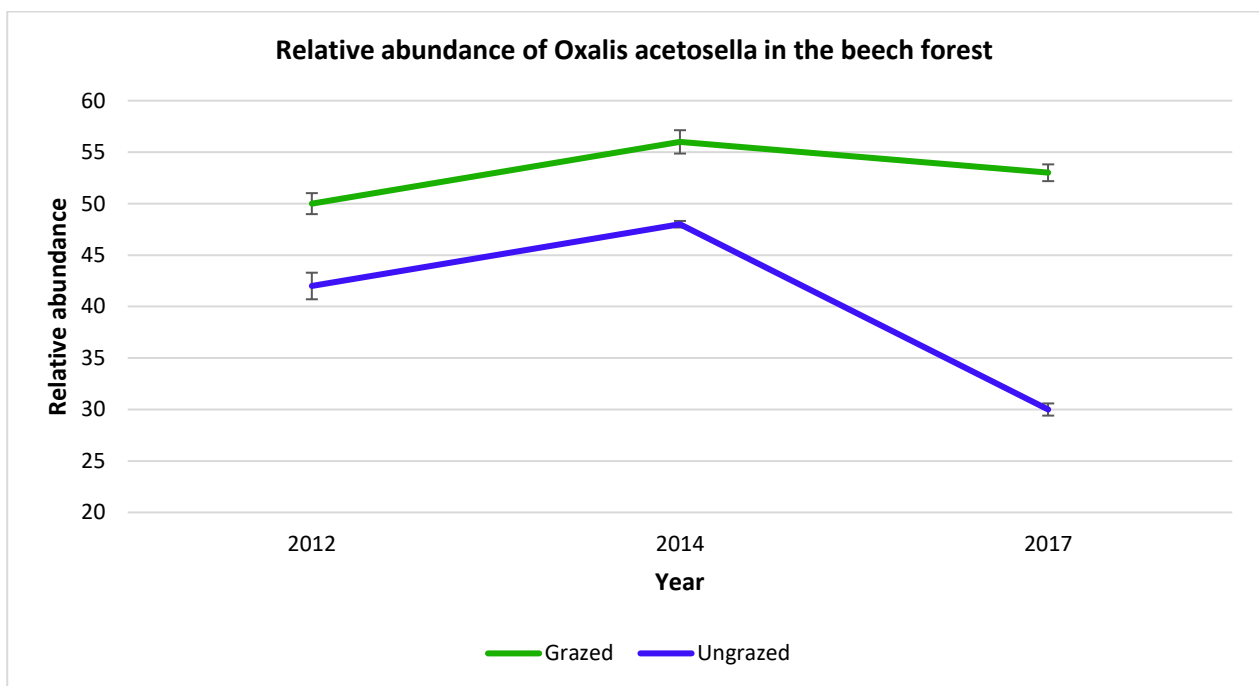
Figure 18. The relative abundance of *Oxalis acetosella*, in both the grazed and ungrazed beech forest (±SE). The relative abundance in the grazed beech forest was in 2012 50 (SE=1.019), in 2014 was it 58 (SE=1.133) and in 2017 was it 53 (SE=0.813). The relative abundance in the ungrazed beech forest was in 2012 42 (SE=1.291), in 2014 was it 48 (SE=0.317) and in 2017 was it 30 (SE=0.590).

The other species most observed in the beech forest, was *Deschampsia cespitosa*, which is showed in figure 19. Here it appears that the highest relative abundance was found in the ungrazed area, where it was estimated to 10 in 2012, 20 in 2014 and 16 in 2017. Looking at the grazed area, the abundance was more constant at a lower level. The abundance was here found to be 4 in both 2012 and 2014 and 3 in 2017.
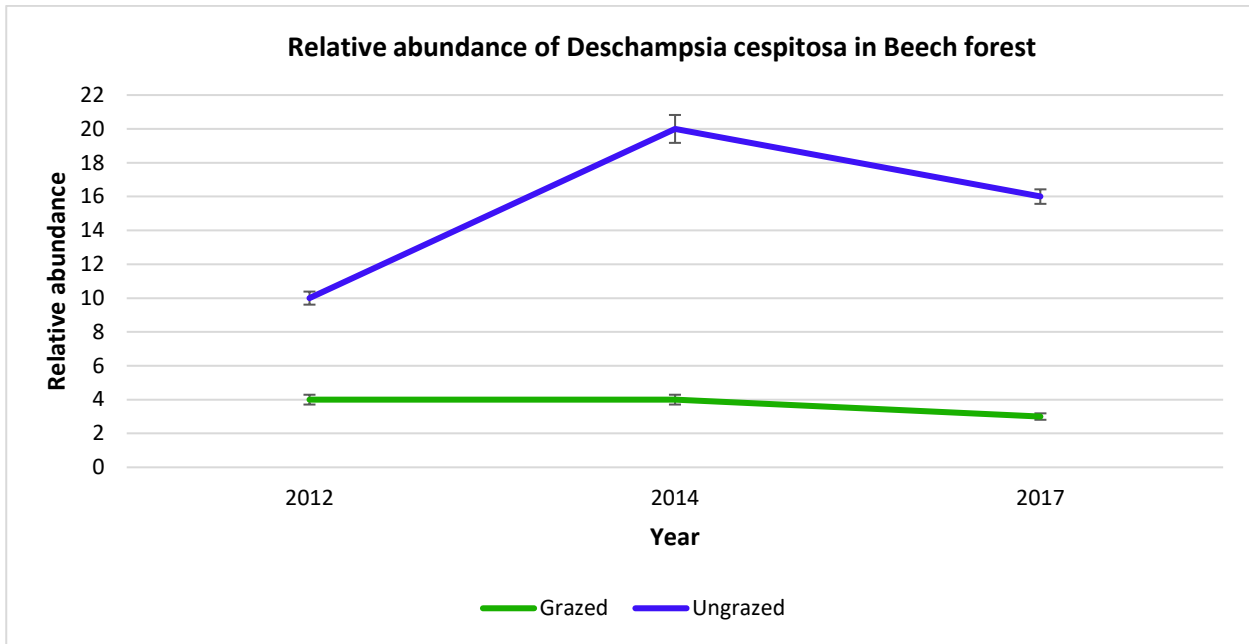
Figure 19. The relative abundance of *Deschampsia cespitosa* in both the grazed and ungrazed beech forest (±SE). The relative abundance in the grazed beech forest was in 2012 4 (SE=0.289), in 2014 was it 4 (SE=0.289) and in 2017 was it 3 (SE=0.189). The relative abundance in the ungrazed beech forest was in 2012 10 (SE=0.386), in 2014 was it 20 (SE=0.824) and in 2017 was it 16 (SE=0.428).

Looking at the biodiversity indices for the beech forest, table 9 displays differences between years and the grazed and ungrazed area. The biggest difference between the two areas are observed in the Simpson dominance index. From the table it appears, that the grazed area was more dominated by fewer species than the ungrazed, reflected by the Simpson values, where the biggest difference occurred in 2014 and 2017. In addition, the Shannon Wiener value from 2017 differed between the two areas. The value of 1.966 in the ungrazed area indicated a higher biodiversity compared to the 1.312 in the grazed area. The ungrazed area did also represent a more even species composition in 2017 than the grazed, reflected in the Pielou value.

Table 9. Biodiversity indices for the beech forest.

	Beech forest (grazed)			Beech forest (ungrazed)		
	2012	**2014**	**2017**	**2012**	**2014**	**2017**
Number of species	11	14	10	8	13	12
Pielou	0.535	0.497	0.601	0.688	0.652	0.791
Simpson	0.462	0.497	0.398	0.360	0.292	0.197
Shannon Wiener	1.283	1.312	1.384	1.430	1.672	1.966

Alder swamp

In the alder swamp, the three most abundant species was *Carex remota, Oxalis acetosella* and *Deschampsia flexuosa*, which is presented in figure 20. *Carex remota* showed a peak in abundance in 2014, where it was found in 45 throws with the Raunkjær circle out of 100.

Oxalis acetosella showed highest value in 2012 with a relative abundance of 63, but decreased to 52 in 2014 and furthermore to 44 in 2017. *Deschampsia flexuosa* showed the smallest fluctuations over time, with a relative abundance of 51 in 2012, 56 in 2014 and once again 51 in 2017.
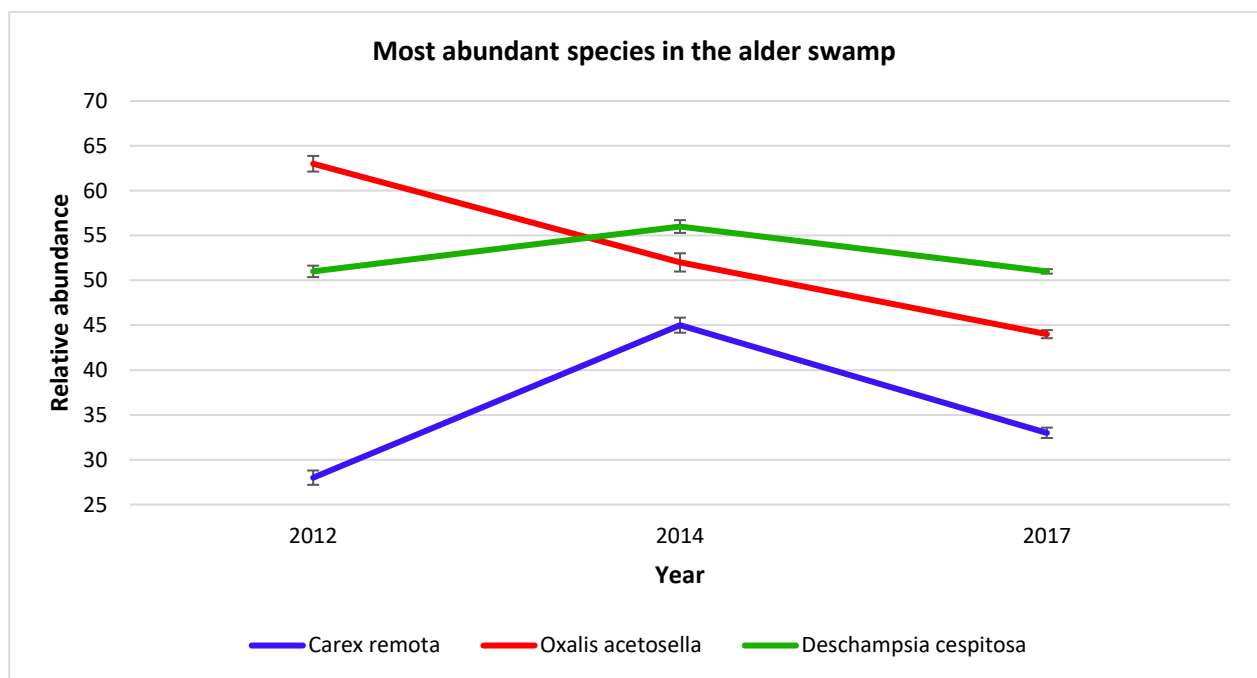
Figure 20. The most abundant species in the alder swamp over the 5-year period (±SE). Relative abundance of *Carex remota* was 28 in 2012 (SE=0.797), 45 in 2014 (SE=0.839) and 33 in 2017 (SE=0.579). Relative abundance of *Oxalis acetosella* was 63 in 2012 (SE=0.872), 52 in 2014 (SE=1.017) and 44 in 2017 (SE=0.451). Relative abundance of *Deschampsia cespitosa* was 51 in 2012 (SE=0.639), 56 in 2014 (SE=0.709) and 51 in 2017 (SE=0.246).

The diversity indices in table 10 reveals that most species was found in 2017. Evenness of the are also increased, showed by the highest Pielou value in 2017. The Shannon Wiener index also showed highest value in 2017 indicating highest level of biodiversity in the end of the project. The Simpson index followed the pattern and exposed lowest level of dominance in the end of the 5-year period.

Table 10. Biodiversity indices for the alder swamp.

	Alder swamp		
	2012	**2014**	**2017**
Number of species	45	46	49
Pielou	0.788	0.762	0.831
Simpson	0.077	0.085	0.059
Shannon Wiener	3.000	2.917	3.236

Oak forest

In the old oak forest, where both a grazed and ungrazed area was present, the two same species were most abundant, as illustrated in figure 21. *Oxalis acetosella* showed highest abundance in both areas, but the population was most dominant in the grazed area, where a stable curve above 90 was observed for all three measurements. *Deschampsia flexuosa* was found at a lower abundance than *Oxalis acetosella* in both areas, but showed a similar development across the grazed and ungrazed area. First, a decrease from 2012 to 2014 was observed, before an increase to the highest measured abundance was estimated in 2017 for both areas. The abundance of the species was highest in the ungrazed area in 2012 and 2014, but the pattern shifted towards 2017, where the grazed showed the highest abundance.
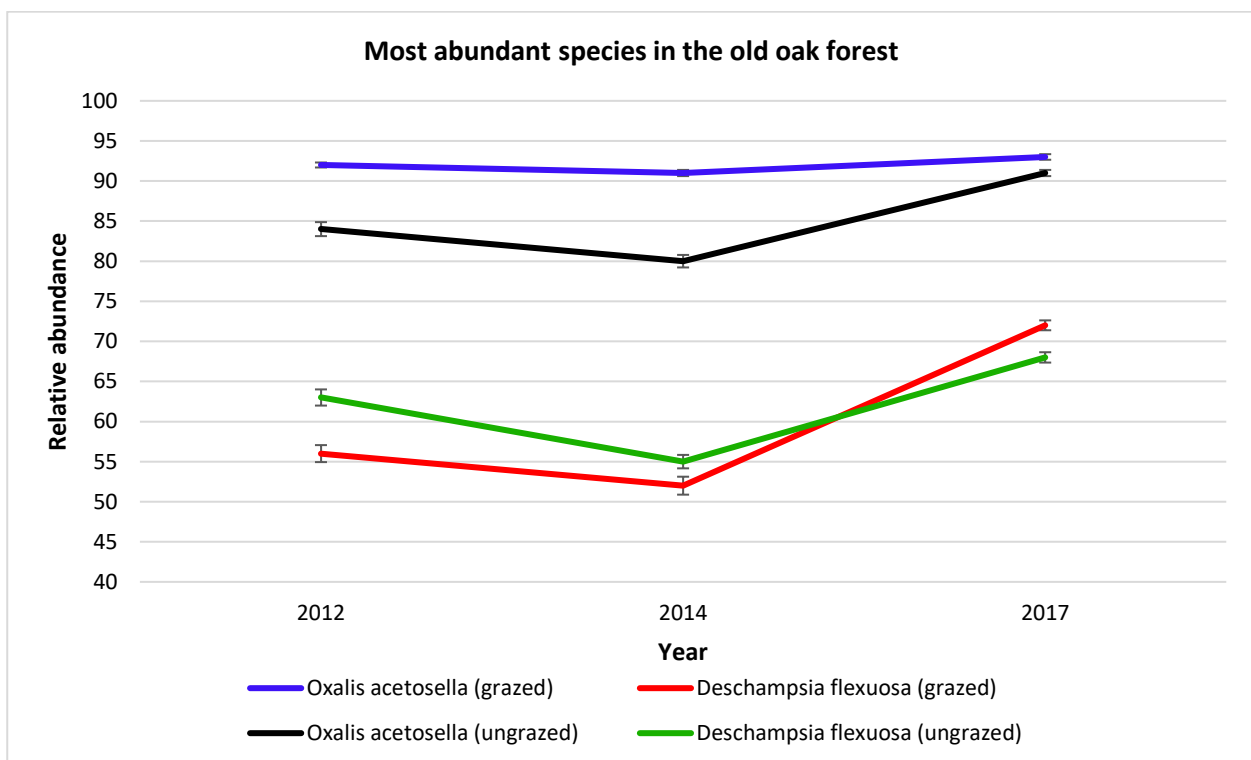
Figure 21. The most abundant species in the old oak forest, both grazed and ungrazed, over the 5-year period (±SE). Relative abundance of *Oxalis acetosella* in the grazed area was in 2012 92 (SE=0.309), in 2014 was it 91 (SE=0.386) and in 2017 was it 93 (SE=0.347). Relative abundance of *Oxalis acetosella* in the ungrazed area was in 2012 84 (SE=0.874), in 2014 was it 80 (SE=0.780) and in 2017 was it 91 (SE=0.386). Relative abundance of *Deschampsia flexuosa* in the grazed area was in 2012 61 (SE=1.060), in 2014 was it 49 (SE=1.120) and in 2017 was it 64 (SE=0.617). Relative abundance of *Deschampsia flexuosa* in the ungrazed area was in 2012 63 (SE=1.209), in 2014 was it 55 (SE=1.012) and in 2017 was it 68 (SE=1.007).

In the middle-aged oak, the three most abundant species in 2017 showed three different advances through the 5-year period. As illustrated in figure 22, the abundance of the species developed towards a more similar and even abundance in 2017. *Deschampsia flexuosa* decreased from the highest starting point in 2012, but ended with lowest abundance for all species in 2017. Deschampsia cespitosa showed an increasing pattern, starting with a relative abundance of 32 in 2012 but ended with a presence in 51 throws out of the 100 Raunkjær circles.

Lonicera periclymenum also showed an overall increase in relative abundance during the 5-year period. The relative abundance decreased from 47 in 2012 to 45 in 2014, but increased to 56 in 2017. *Lonicera periclymenum* was the most abundant species in the middle-aged oak forest in 2017.
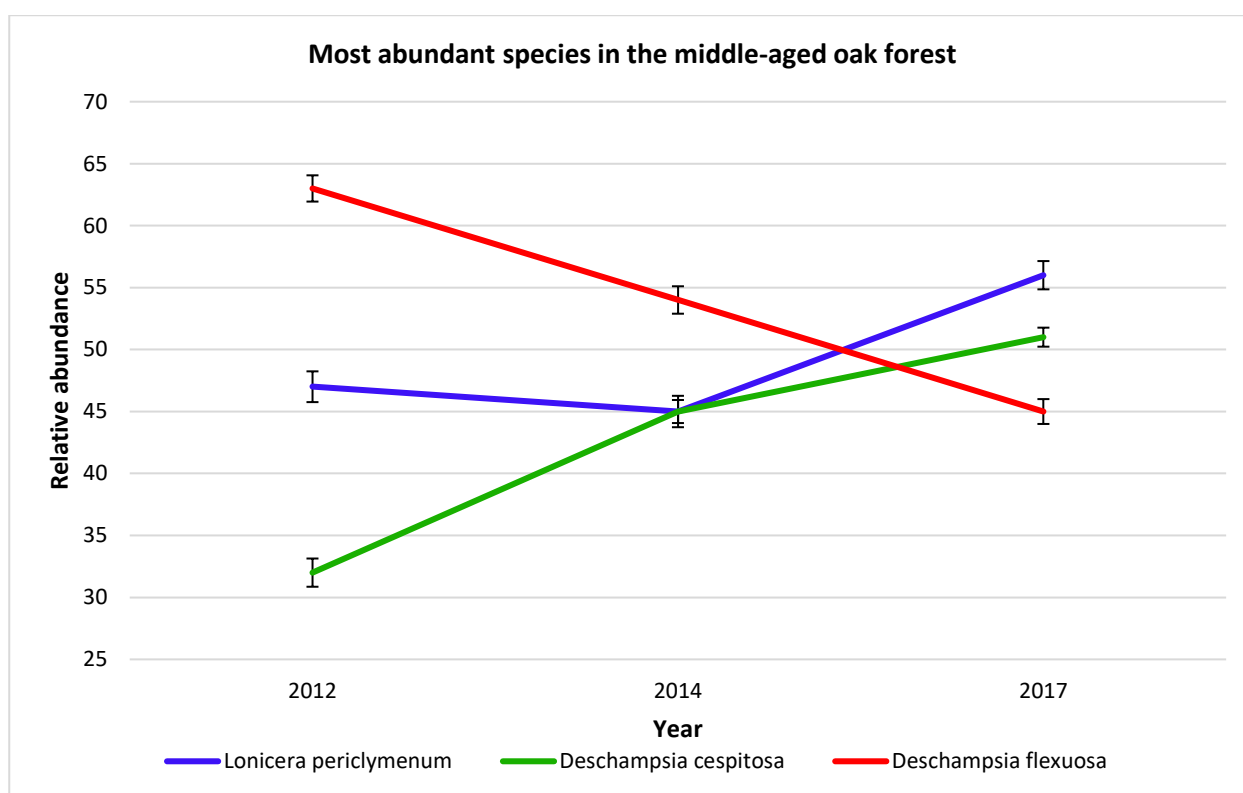
Figure 22. The most abundant species in the middle-aged oak forest over the 5-year period (±SE). Relative abundance of *Lonicera periclymenum* was 47 in 2012 (SE=1.241), 45 in 2014 (SE=1.266) and 56 in 2017 (SE=1.140). Relative abundance of *Deschampsia cespitosa* was 32 in 2012 (SE=1.138), 45 in 2014 (SE=0.930) and 51 in 2017 (SE=0.767). Relative abundance of *Deschampsia flexuosa* was 63 in 2012 (SE=1.058), 54 in 2014 (SE=1.106) and 45 in 2017 (SE=1.002).

In figure 23, the most abundant species in the young oak forest is illustrated. *Oxalis acetosella* and *Deschampsia cespitosa* both experienced a decrease from 2012 to 2014, but increased to highest measured abundance in 2017. *Oxalis acetosella* was found at the lowest relative abundance of all the three species in the two first measurements in 2012 and 2014, but showed the prime increase of all three species, ending at 56 in 2017. *Deschampsia flexuosa* was most stable in abundance over the five years, compared to the other two species. *Deschampsia flexuosa* showed the same decreasing pattern as the other species from 2012 to 2014, but did not experience the same increase for the last 3 years, resulting in an abundance of 39 in 2017, lower than the starting point of 41 in 2012.
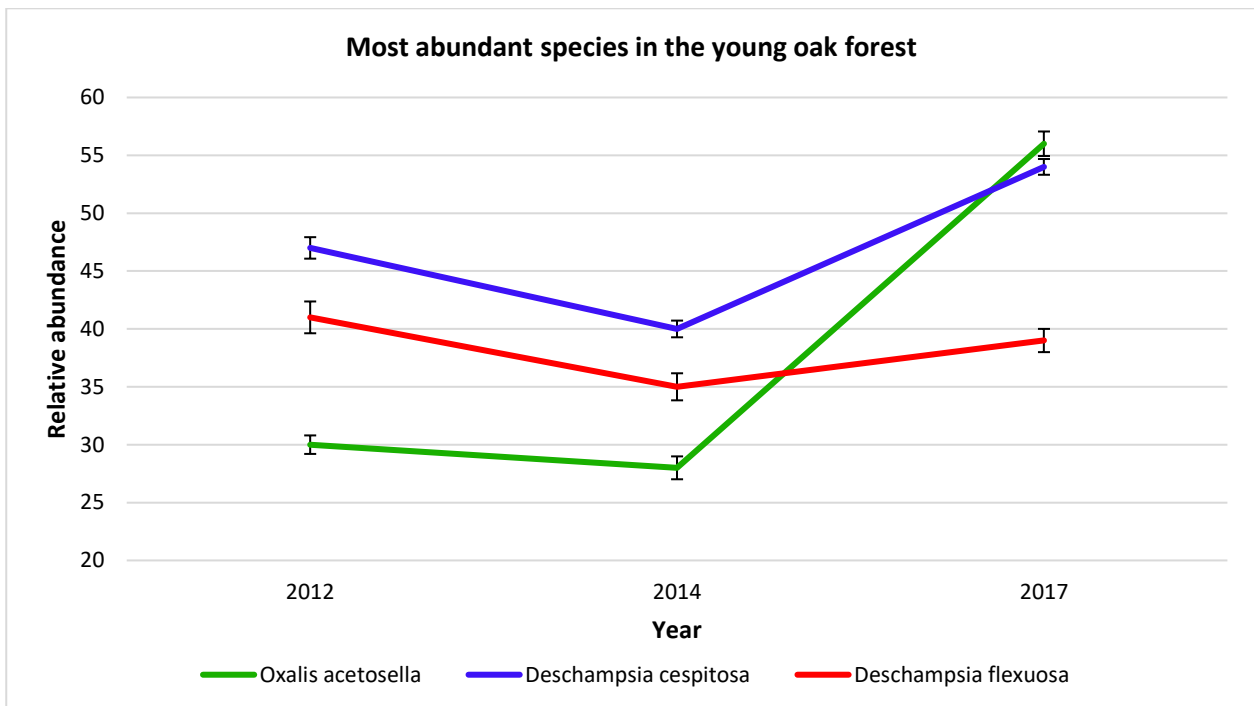
Figure 23. The most abundant species in the middle-aged oak forest over the 5-year period (±SE). Relative abundance of *Oxalis acetosella* was 30 in 2012 (SE=0.800), 28 in 2014 (SE=0.987) and 56 in 2017 (SE=1.060). Relative abundance of *Deschampsia cespitosa* was 47 in 2012 (SE=1.374), 40 in 2014 (SE=1.168) and 54 in 2017 (SE=1.004). Relative abundance of *Deschampsia flexuosa* was 41 in 2012 (SE=0.927), 35 in 2014 (SE=0.721) and 39 in 2017 (SE=0.681).

In table 11, the biodiversity indices for all oak forest areas is displayed and compared. These results revealed that highest number of species was found in the young oak forest for all three years of measurement, while the ungrazed old oak forest contained the lowest amount of different species. The highest evenness for the middle-aged- young- and grazed old oak forest were all found in 2017, where the ungrazed old oak forest showed the opposite results and decreased over time and ended with the lowest value within the area in 2017. The young oak forest demonstrated the highest evenness in all four areas where it peaked at 0.790 in 2014 and 2017. In contrast, the middle-aged oak forest showed the lowest evenness of all in 2014, where it plunged at 0.670.

Table 11. Biodiversity indices for the oak forest.

	Young oak forest			Middle-aged oak forest		
	2012	**2014**	**2017**	**2012**	**2014**	**2017**
Number of species	44	44	46	31	40	37
Pielou	0.781	0.790	0.790	0.710	0.670	0.753
Simpson	0.081	0.077	0.075	0.127	0.130	0.094
Shannon Wiener	2.957	2.988	3.026	2.437	2.472	2.720

	Old oak forest (grazed)			Old oak forest (ungrazed)		
	2012	**2014**	**2017**	**2012**	**2014**	**2017**
Number of species	36	33	38	21	21	22
Pielou	0.673	0.733	0.766	0.747	0.765	0.722
Simpson	0.134	0.112	0.084	0.139	0.134	0.135
Shannon Wiener	2.411	2.562	2.785	2.275	2.330	2.232

The Simpson index pointed out the ungrazed old oak forest, as being the most dominated area. This area demonstrated three values of the Simpson index which were higher than any other estimate from the oak forest area. The Simpson index was close to similar for the grazed- and ungrazed old oak forest in 2012, but the grazed area demonstrated a decrease in dominance over time, which highlighted the lowest value in 2017.

Lowest dominance for all oak forest areas was consecutive found in the young oak forest, which correlated the values from the Pielou index in the same area, and showed the lowest dominance in 2017.

The three grazed areas all showed increasing tendency regarding the Shannon Wiener index, which peaked in 2017, indicating highest level of biodiversity in the end of the 5-year period. The contradictory pattern was discovered in the ungrazed old oak forest, where the lowest value of Shannon Wiener was observed in 2017.

Norway spruce forest

In the ungrazed old Norway spruce forest, only one species was found repeatedly in the 3 years of measurements, whereas two was found in the grazed area, illustrated in figure 24. In the ungrazed area, *Deschampsia flexuosa* doubled its relative abundance from 9 in 2012 to 18 in 2014, before decreasing to 11 in 2017. In the grazed area *Deschampsia flexuosa* only showed an up going curve, where the abundance also was higher compared to ungrazed area, ending at a relative abundance of 39 in 2017. *Oxalis acetosella* also showed increasing tendency in both 2014 and 2017 and followed an abundance close to *Deschampsia acetosella* in the grazed area*.*
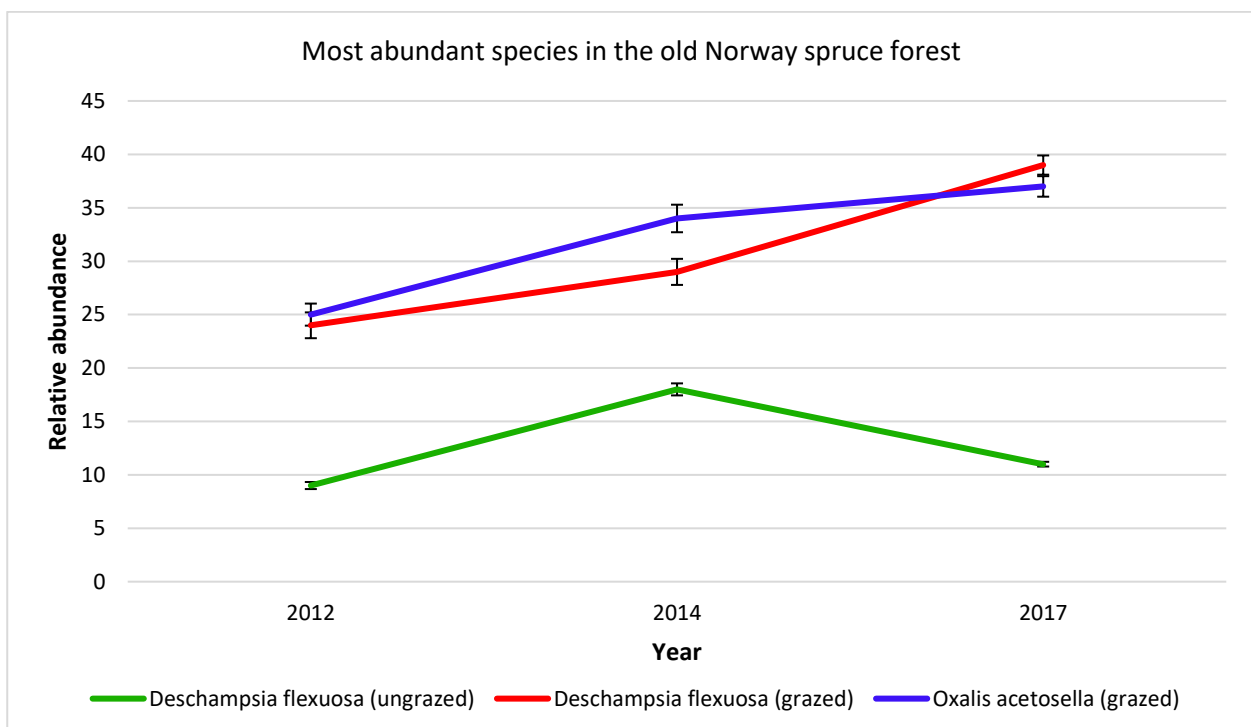
Figure 24. The most abundant species in the old Norway forest, both grazed and ungrazed, over the 5-year period (±SE). Relative abundance of *Deschampsia flexuosa* in the ungrazed area was 9 in 2012 (SE=0.330), 18 in 2014 (SE=0.562) and 11 in 2017 (SE=0.221). Relative abundance of *Deschampsia flexuosa* in the grazed area was 24 in 2012 (SE=1.208), 29 in 2014 (SE=1.220) and 39 in 2017 (SE=0.899). Relative abundance of *Oxalis acetosella* in the grazed area was 25 in 2012 (SE=1.031), 34 in 2014 (SE=1.289) and 37 in 2017 (SE=0.959).

In the young Norway spruce forest, two species showed an overall increase over the five years, while one suffered a decrease as shown in figure 25. *Deschampsia flexuosa* was found with the highest abundance in all three measurements compared to the other species, and at the same time, it showed an increase from 2012 to 2017. *Carex pilulifiera* decreased from 8 in 2012 to 4 in 2014 and raised again to 6 in 2017. *Oxalis acetosella* was the least abundant of the three species and appeared with a relative abundance of 2 in 2012 and 5 in 2017.
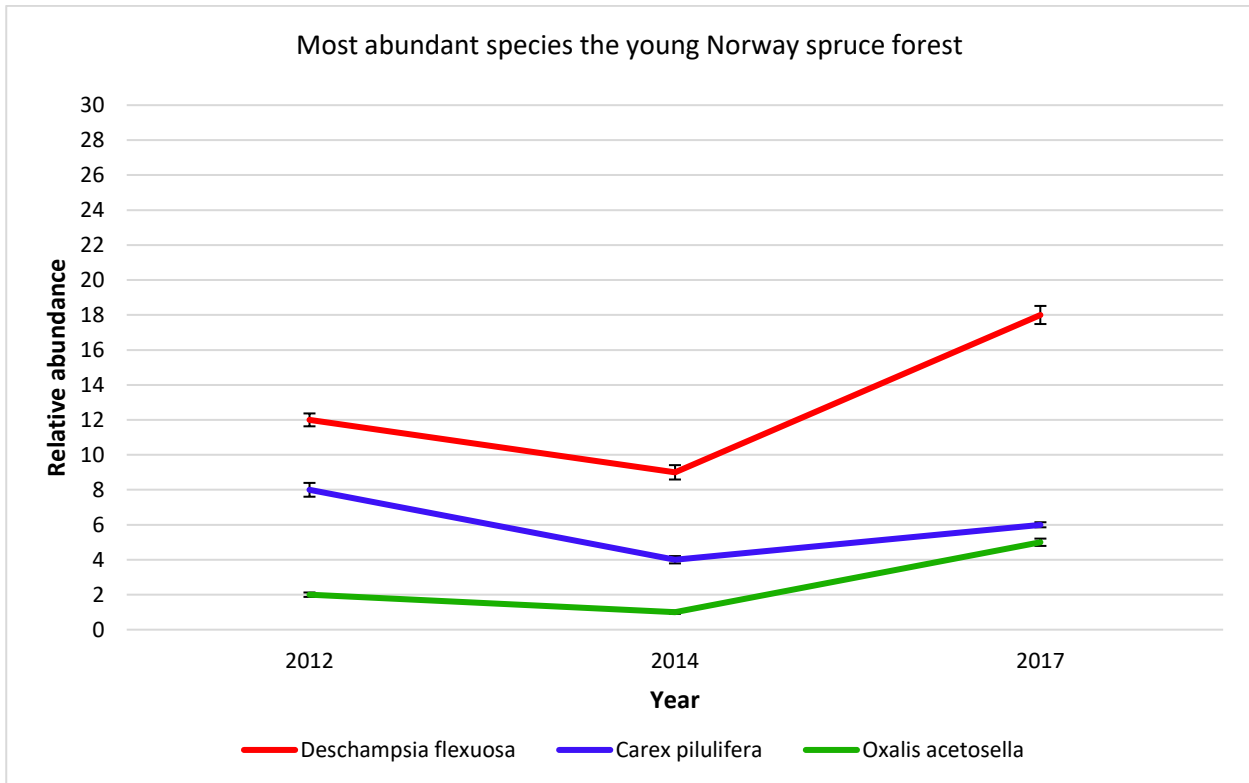
Figure 25. The most abundant species in the young Norway spruce forest over the 5-year period (±SE). Relative abundance of *Deschampsia flexuosa* was 12 in 2012 (SE=0.369), 9 in 2014 (SE=0.411) and 18 in 2017 (SE=0.518). Relative abundance of *Carex pilulifera* was 8 in 2012 (SE=0.394), 4 in 2014 (SE=0.209) and 6 in 2017 (SE=0.144). Relative abundance of *Oxalis acetosella* was 2 in 2012 (SE=0.126), 1 in 2014 (SE=0.094) and 5 in 2017 (SE=0.209).

The biodiversity indices for the Norway spruce forest is presented in table 12, which revealed that the lowest number of species was found in the ungrazed old Norway spruce forest, simultaneously with the lowest evenness, highest dominance and lowest diversity of the three areas. In addition, these values for the ungrazed area were obtained in 2017 which indicated a decrease in biodiversity over time in the ungrazed old Norway spruce forest.

In 2017, the highest evenness in the area was found in the grazed old Norway spruce forest, which also demonstrated the lowest dominance and highest Shannon Wiener index. The young Norway spruce forest obtained its highest number of species in 2014, equalizing the maximum of 11 in the grazed old Norway spruce forest. The young Norway spruce forest displayed a lower biodiversity in 2017 than in 2012, according to the Shannon Wiener index.

Table 12. Biodiversity indices for the Norway spruce forest.

	Young Norway spruce forest			Old Norway spruce forest (grazed)			Old Norway spruce forest (ungrazed)		
	2012	2014	2017	2012	2014	2017	2012	2014	2017
Number of species	7	11	8	11	11	7	6	5	5
Pielou	0.596	0.525	0.484	0.503	0.638	0.707	0.507	0.379	0.298
Simpson	0.475	0.484	0.539	0.411	0.276	0.318	0.449	0.710	0.789
Shannon Wiener	1.159	1.259	1.007	1.205	1.530	1.376	0.908	0.610	0.480

Browsing and bark striping

The browsing and bark striping activity was in this study conducted to serve as a supplementary source, for describing and explaining the possible changes in plant species biodiversity. Previous studies have focused more on only bark striping, but with other methods, for a more complete mapping to determine the influence of this habit (Jønsson 2014, Brender 2016).

These results does not rely on specific vegetation types, but covers the entire fenced area. The illustrated figure presents the probability for selected tree species, to be exposed for browsing. For the browsing results, only individuals of the size category 2 (cf. table 5) is included. Size category 1 was removed because browsed seedlings could not be determined to specie. Size category 5 was also removed because the size category could not distinguish between trees of 2.1 meter and 20 meter, which would be above the possible browsing-line and thereby create a more erratic result. For all species in the size category 3 and 4, the amount of individuals present, were not sufficient to make a reliable statistic data analyses. Consequently, these results only predict probability for browsing for the lowest stages of up growth encroachment.

Regarding the bark striping, not enough tree individuals were bark striped in the documentary circles to perform a statistical analyses. Only few individuals showed tracks of bark striping activity and results would therefore predict a probability for bark striping on all species close to 0 Therefore, no results for bark striping is present in the following section, and only browsing results will be evaluated.

Browsing

Figure 26 shows the most abundant tree species in the bison fencing, and the probability for being exposed to browsing when being the size of 0-50 cm. No species revealed significant changes in probability for being browsed between the two years 2014 and 2017, except one species*. The probability for *Betula pendula* in 2014 was 0.761, which was significant higher (P-value= <0.001) than the probability of 0.169 in 2017. *Picea abies* showed a probability of 0 in both 2014 and 2017, as the only species. For out *Picea abies* and *Betula pendula* in 2014, the rest of the species showed a probability between 0.106 and 0.383, whereas *Acer pseudoplatanus* being the minimum and *Fagus sylvatica* being the maximum.
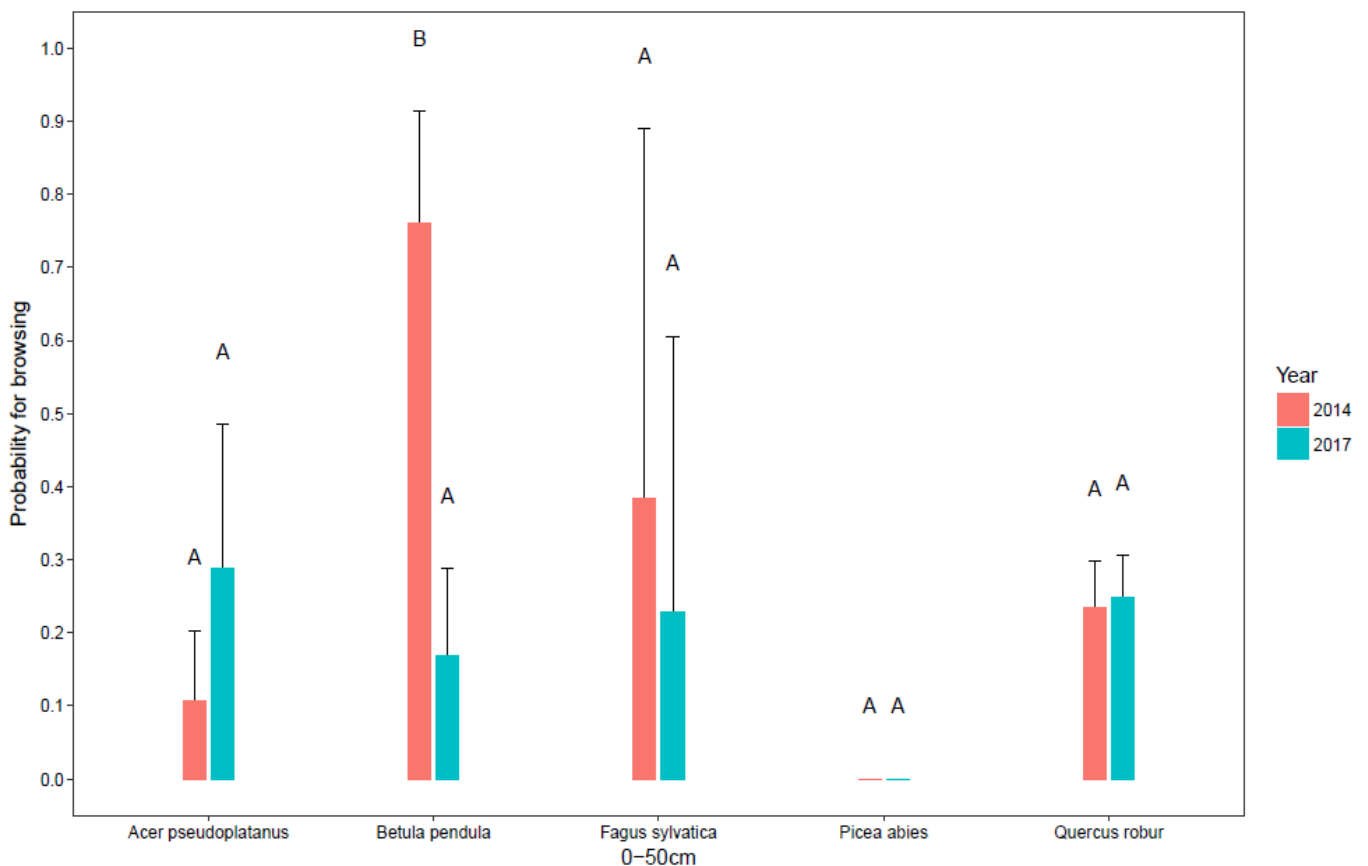
Figure 26. Probability for browsing of the most abundant tree species (+SE). *Acer pseudoplatanus* showed a probability for browsing of 0.287 (SE=0.197) in 2014 and 0.106 (SE=0.096) in 2017. *Fagus sylvatica* showed a probality for browsing of 0.383 (SE=0.505) in 2014 and 0.227 (SE=0.377) in 2017. *Betula pendula* showed a probability for browsing of 0.761 (SE=0.153) in 2014 and 0.169 (SE=0.119) in 2017. *Picea abies* showed a probability for browsing of 0 (SE=0.000) in both 2014 and 2017. *Quercus robur* showed a probability for browsing of 0.234 (SE=0.063) in 2014 and 0.247 (SE=0.057) in 2017

Discussion

In the following section, the previous described methods and results will be discussed in order to gain a larger perspective, of the realized effects from the project. The used methods and influential variables will be evaluated for a better understanding of the results, and to advance further knowledge for optimizing in the future.

The obtained results will be elaborated in order to explain possible reasons for the achieved outcomes. The results will furthermore be related to already known empiricism of the topic, to gain an even broader present understanding of grazing by the European bison. First, the species richness and weighted Ellenberg values will be evaluated, before concrete species and abundances will be discussed together with the biodiversity indices. The results will be compared on different levels for novel contribution to both ecology of the habitat types and grazing ecology of the European bison. Afterwards, the browsing results will be discussed, which will lead to the final discussion about future perspectives for nature management by grazing from the European bison.

Discussion of methods and sources of error

Nature is a changeable environment and field studies like this will therefore always have a sense of uncertainty, when conducted in free-living ranges. This experiment stretches over a 5-year period, with measurements in the years of 2012, 2014 and 2017, and abiotic factors was therefore most likely to vary. As seen in table 1 and 2, precipitation and temperature varied over time. These are only two factors of many affecting seed germination and plant growth (Akula and Ravishankar 2011).

The amount of collected data was extensive, and only assembled by 1-2 persons during all years. Therefore, the surveys extended over a 3-month period, which is a large time span for mapping plant species, when time of germination and flowering of species in mind. The 130 plots were therefore not investigated at exactly same time in years of measurements. The same plot could therefore potentially be assessed in mid-May in 2012 and in mid-August in 2017. This would most likely enlarge the variation in abiotic factors and also affect the obtained results. For example, *Anemone nemorosa* was found in the oak forest in 2012, which blooms in April-May, but fades out early due to shadowing from the forest. If the oak forest was investigated in august, *Anemona nemorsa* would probably not be present and was therefore not found in any areas in 2014 and 2017. The same problematic is cased for many other species such as *Primula elatior* or *Ajuga pyramidalis*, which also blooms early in April-May. These species were never mapped, but was seen in a relative large abundance in some areas during a visit to Almindingen in May 2018. Other species, like *Digitalis purpurea* and *Holcus mollis,* demonstrates a later blooming in July-August. These species could potentially be present in plots observed in May, but would therefore never be mapped, but possibly found in same areas the following years, if investigated later. This improbability was therefore likely to affect species composition in the areas.

The results of both species richness, Ellenberg analyses and frequencies of species depend on species composition, which is plausibly affected by this. Obviously, a certain amount of time are needed for a proper field survey when data is comprehensive. Nevertheless, in order to minimize sources of error from early and late blooming, the start of the investigations should be scheduled to initiate at the same time for all years. In addition, plots should be investigated in the same sequence for all years, also to limit temporal differences between investigated fields.

Ultimately, 130 documentary circles were made in the 200 ha large area. Species registration was in all cases made in the 5-meter circle, covering an area of 78.5 m^2. All circle plots joined together covered approximately 0.5% of the total area. It can be discussed whether this is representative enough for assessing the species composition. More importantly, the 130 plots were evenly distributed on the thirteen vegetation types with ten plots in each. This results in an uneven dissemination, since the habitat types differed in size. As shown in table 3, the bush grass dominated area covered less than 1% compared with the old Norway spruce forest which covered 31%. The smaller areas was therefore relatively more well-documented according to species registration.

For optimizing, more plots could have been created, but this would of course increase the demand for observer-resources if the survey extend should remain the same. Another discussion related to the circle plots, is the distribution inside the area. From figure 4, we found that the circle plots was uneven distributed. This created a minor view on specific areas, which could have contributed to a broader picture of the projects effect. Near the single lake in the area, no circle plots were present. According to Brandtberg and Dabelsteen (2013) and Orbitt (2018) the bison herd was frequently visiting the area near the water, since it delivers drinking resources. Schmidt (2016) also revealed a diet of the bison herd consisting of *Phragmites australis*, which is constrained to freshwater areas. It would have been relevant to add another vegetation type, consisting of the area near the water, to investigate possible influence in this range. Since plant species near the water was present in the diet of all individuals, it is reasonable to believe that species composition and richness was affected in this specific area (Schmidt 2016).

For most areas, the development through the years was measured relatively to the starting point of 2012. Nevertheless, control fields for the old Norway spruce forest, the beech forest and the old oak forest, was constructed outside the fencing for comparisons. This provided an opportunity to measure a possible effect and compare it with an area not exposed for grazing, to exclude change caused by other factors. For optimizing the study, establishment of control areas for all habitat types could have been created. A suggestion for control fields could be the formation of smaller fenced areas inside the large bison fencing. For example, ten fenced circle plots corresponding to the same area of the investigated documentary circles in all habitat types could be formed. Control fields inside the bison fencing would also minimize other influencing factors, compared to control areas outside the fencing. The control areas outside the fencing could be exposed to grazing by other herbivores or human activity.

Another benefit from having control fields inside the large fencing would be the minimizing of distance, between grazed and ungrazed fields. This would reduce variation in climatic and spatial fluxes and abiotic factors across the two areas.

Browsing activity was also measured in the circle plots, according to size category explained in table 5. Once again, we only covered a small part of the whole area and more widespread browsing activity would probably be present, if more area was covered. The method used, also made it difficult to register browsing activity on the smallest seedlings, since all above ground biomass was most likely removed. Other studies that investigated the proportion of browsing used physical monitoring, where observers followed the bison herd with binoculars and noted every bite (Cromsigt, Kemp et al. 2017). The investigation of bark striping was insufficient of showing any relevant results, when only very few individuals showed tracks of bark striping. The analyses were therefore rejected from the report, since they could not contribute with any relevant observations. Thus, a quick screening around the investigated habitat types in addition to the documentary circles, showed bark striping activity on some individuals. Bark striping was therefore present in the area, and previous studies also documented bark striping on pedunculate oak and Norway spruce, but found no significant effect on radial growth (Brender 2016). To investigate the bark striping more completely, focus should be on solo bark striping, so investigations could be expanded. More trees in larger areas should be investigated and other seasons should be added to monitoring the bark striping, since summer is probably the period with least bark striping, due to other accessible food sources, which are more preferable. The same applies for the browsing activity, where the results could probably differ if the investigations were made in other seasons, since temporal variations are likely to affect diet of the European bison (Cromsigt, Kemp et al. 2017).

A factor that could also possibly have affected the results of the bark striping and the browsing was the supplementary feeding in the fencing. If the bison herd is given extra hay, the intake of woody materials will decrease (Kowalczyk, Taberlet et al. 2011). The Supplementary feeding was therefore likely to disturb the obtained results, and studies without supplementary feeding are therefore necessary to investigate the effects under more natural conditions. The supplementary feeding could also be a factor to affect the local species composition within small areas. A dissimilarity between the seed bank in the supplementary feeding area and the standing vegetation more far away could be forced by this management practice, since the supplementary feeding material could source new species (Jaroszewicz, Kwiecień et al. 2017). Furthermore, probably the most important factor to sustain the most reliable results – is time. The study was conducted over a 5-year period, which in many cases would be assumed as insufficient. Especially, the desired effect of changed dynamics and structures of the forest vegetation is a long term process, where the succession stages of aging and breakdown takes more than 5 years (Hahn, Emborg et al. 2007). Additionally, population size of the European bison also varied over time, but the inequalities from the varying grazing pressure was most likely minimum.

Species richness

Species was significantly affected in two of the three open land habitats, as shown in figure 5. The rejuvenated area increased significantly in both 2014 and 2017, which was expected due to the clear cut of Norway spruce in 2011. Species richness in Norway spruce forests are expected to be low due to the lack of available light, which is crucial for plant growth. When the area then became totally exposed, a succession of pioneer species was expected (Bazzaz 1979, Glenn-Lewin, Peet et al. 1992). It is therefore reasonable to assume, that species richness would increase with or without grazing from the European bison in the rejuvenated area. The first five years of succession is primary growth of annuals, perennials and grasses, before fast growing trees and shrubs move into the area. Perhaps it is therefore more relevant to investigate if the grazing activity can prevent an up growth of trees, shrubs and unwanted species in the following years. Once again, control fields to compare the up growth and possible differences, would have been an advantage in determining the effect.

The wet meadow also showed a significant increase in species richness. Nevertheless, as described earlier this particular area was previously grazed by other herbivores before 2012, but also during the 5-year period. Therefore, any changes in this area cannot be attributed the grazing of European bison alone. The increase can therefore only be assigned to grazing in general, since the area was exposed of a combination from both horse- cattle- and bison grazing.

In the bush grass dominated area, no significant changes was observed. *Calamagrostis epigejos* is known to be a dominant and aggressive grass, which inhibit other species when present. That is probably the most plausible explanation for the low species richness compared to the other open sites. Previous studies in Denmark have also revealed, that in order to effectively reduce the abundance of *Calamagrostis epigejos*, two domestic cattle's per hectare is needed (Hansen, Kristensen et al.). One explanation of the missing changes in species richness could therefore be to low grazing pressure.

The fenced area contained a broad and varied vegetation, an alternative explanation could therefore also be selective food preferences. *Calamagrostis epigejos* was formerly found in the diet of the bison herd, but other food sources may be higher preferable.

In general, grazing are believed to enhance biodiversity in open land vegetation, which is supported by the results of the wet meadow. Species composition in the open land tends to express traits of both life-history and ecological functions, which are more resistant for disturbances comparable to grazing (Aerts 1999, Díaz, Noy-Meir et al. 2001). This especially includes grasses and herbaceous plants, where competition for light will be more even if grazed. Species adapted to the open land are therefore also more stress tolerant, since these areas in a historic perspective have been used to e.g. domestic livestock (Grime 1979, Buttenschøn 2007).

Looking at the forest habitats, we found increases in species richness in the young oak forest, the grazed old oak forest and the alder swamp. Of all forest habitats, these were also the areas with highest species richness. One characteristic they had in common compared to the other habitats, were a more open structure with higher light availability, which was supported by the registration of the vegetation structures. This is probably a causal factor for the higher richness and a plausible explanation for the observed increase, since more plots was comparable to open land. In these areas, a more dense growth of bottom vegetation was found compared to e.g. the Norway spruce forest or the beech forest.

Old oak forest with larger gaps and light shafts (left) and a smaller watercourse with rich vegetation of *Mentha aquatica* in the alder swamp (right). Photo: Bjarke Schäfer

Plant species in forest habitats are often adapted to a more stable environment with less disturbance, since changes occur more infrequently compared to the open land. Light is also more reduced, which creates basis for a more poor species richness. The availability of resources is a crucial factor for increasing of richness and biodiversity by grazing. If a e.g. nutrient poor environment are exposed to grazing, it can potentially decrease species richness, since the resources for restoring biomass are limited (Proulx and Mazumder 1998). A plausible explanation for the observations of species richness in the forest areas could therefore be the lack of open spaces and thereby the light gradient. The forest areas are maybe not able to sustain a higher richness due to lack of resources. The affected forest areas encloses a richer flora, which is probably more adaptive to disturbance. Thus, another explanation for the missing effect in the forest could be lack of presence, due to a different habitat selection. Nevertheless, the results suggested an effect in the open landscape and the semi-open forests, but in the most closed forest, species richness was not found to increase.

Ellenberg analyses

In the three open land habitats, we saw a significant increase in the demand of light in the rejuvenated area, from the species composition. Since the species composition in 2012 was adapted to the dark and more closed environment in a coniferous forest, the low starting value under 5.00, indicating more shade tolerant species, was expected. Since species richness also increased the following years, it is only natural, that the emerging species composition shows adaption to higher levels of light.

The wet meadow showed increase in EIV-L as well. This is probably also aligned with the significant increase in species richness. The grazing pressure from the shifting herbivore groups the past years probably lowered the average vegetation height in the area, which vegetation structure analysis from 2012 to 2017 also revealed. This would increase light availability in the bottom vegetation, and grazing pressure would at the same time prevent fast growing plants to overshadow. The increased species richness are therefore almost certainly species with a higher requirement of light, which then have altered the species composition towards more light demanding species.

In the bush grass dominated area, the EIV-N increased significantly, indicating a higher demand of nitrogen. Since number of species only differed by two in 2012 and 2017, this change is most likely caused by an alteration of the species composition. The bison herd was not able to reduce the abundance of *Calamagrostis epigejos*, but instead the production increased and the vegetation height growth further in the area. Smaller lots of young pine trees was also present in this area, and as the browsing-results discovered, coniferous species was not preferred. The production of biomass has therefore probably increased for the 5-year period, and other studies has previously emphasized that the EIV-N should progressively be interpreted to reflect productivity instead of simple nutrient content, which could be an explanation for the observed pattern (Hill and Carey 1997, Bartelheimer and Poschlod 2016).

Species adapted to a more effective intake of nutrients often express the trait of fast growing, which would be benefitted from the richer nutrient level (Aerts 1999). Some of these fast growing species with a higher demand of nutrients that occurred in the area after 2012 was e.g. *Epilobium parviflorum, Digitalis purpurea* and *Chamerion angustifolium*. The increase in more nutrient demanding species in the bush grass dominated area could therefore potentially be related to the increasing biomass production in the area. Thus, another explanation could be the previous described problematics regarding variations in sampling time. The three mentioned species with a higher value of EIV-N all flowers in late summer and could potentially be present in e.g. 2012, but not discovered if samplings of the bush grass dominated area were made in early summer.

The bush grass dominated area with small pine trees (left) and the beech forest with poor bottom vegetation (right). Photo: Bjarke Schäfer

In the beech forest, we found no significant changes in EIV in neither the grazed or ungrazed area. The grazed and ungrazed areas appeared almost similar and with only small variations over the years. The EIV-L in the ungrazed area performed a bit higher for all three measurements than the grazed area. Since no decrease or increase was later observed, the most reasonable explanation is a higher light availability in the ungrazed area from the beginning of the project, in at least the measured plots. The EIV-L was relatively low, and ranged between 2-3 for all years. This indicates a more shadow tolerant species composition, which is expected to associate with the beech forest. These results reflect the expectations, since the crown layer was dense and leaved a minimum of light to the ground. The EIV-N reflected a more competitive species composition and higher nutrient content in the area compared to e.g. the open area. This is probably due to the higher biomass production and slower decomposition time in forests (Persson 1980). The poor vegetation of grasses and herbs did probably not attract the bison herd, when more rich areas was available in the fencing, and this could also be a reason for the missing effect and activity in the beech forest.

In the alder swamp, no significant changes occurred, even though species richness increased significantly. Compared to the open landscape, EIV-L was slightly lower and EIV-N slightly higher, and vice versa compared to the beech forest. This is probably aligned to the vegetation structures in the alder swamp, being intermediate relative to especially light availability compared to the other areas. Since species richness increased in the area, these new species found in 2017 probably showed similar traits and preferences in EIV, with the also present flora.

In the oak forest, EIV-L increased significantly in the grazed old oak forest area. This area was characterized by wide-open spaces, with high grass vegetation in many spots. Since species richness in this area also increased, this could indicate a big enough activity in the area to reduce vegetation height and create more niches in the bottom for more light demanding species. Another interesting observation in the oak forest was the EIV's of the young area. In contrast to the other oak forest areas, EIV-L was higher than the EIV-N as shown in figure 13.

This is probably due to the lack of crown layer, which would thereby allow a more vulnerable and light demanding species composition in the understory layer. Furthermore, the total biomass production in the area was probably also lower than the rest of the oak forest, and the decomposition rate was most likely higher, which would favor a species composition of more nutrient-poor character.

The grazed old Norway spruce forest in 2017 with *Oxalis acetosella* (left) and the ungrazed Norway spruce forest (right). Photo: Bjarke Schäfer

The Norway spruce forest expressed some varied patterns over the 5-year period. The only significant change observed, was the decrease in EIV-L in the grazed old Norway spruce forest. To begin whit, the Norway spruce forest showed the lowest species richness of all areas, and in addition very few observations in some years. The EIV's was therefore very easily influenced by smaller changes and observations in the bottom vegetation, since sample sizes were low.

Nevertheless, the decrease in EIV-L in the grazed old Norway spruce forest could be correlated with the increasing frequency of *Oxalis acetosella,* which has an EIV-L of 1. The accumulated value of *Oxalis acetosella* from the Raunkjær circle plots was higher in 2017 than in 2012, and this was the only area with such a high frequency of the species. It is important to have in mind, that 50% of the observations was *Bryophyta*, which was not included in the statistical framework and the *Oxalis acetosella* therefore accounted for a large influence in this area.

In the ungrazed old Norway spruce forest and the young Norway spruce forest, the most abundant species were *Picea abies* (EIV-L=5 and EIV-N=none) and *Deschampsia flexuosa* (EIV-L=6 and EIV-N=3). The results are highly influenced by these species, but the ungrazed old Norway spruce forest also showed a significant increase in EIV-L.

When looking at the observations from 2012, the ungrazed area was highly influenced by *Picea abies*, which expressed an EIV-L of 5. In 2014 and 2017, the abundance of *Picea abies* was much lower, and instead *Deschampsia flexuosa* (EIV-L=6) was more relatively present and also *Molina caerulea* (EIV-L=7) in 2014 and *Sorbus aucuparia* (EIV-L=6) in 2017. The reason for the difference in abundance of seedlings of *Picea abies* could be many, including timing of observations, climatic factors etc. Of course, we cannot exclude grazing as a factor for the observed changes, but the browsing results did not reveal any activity on the *Picea abies* and grazing is not likely to affect flora biodiversity of a pine forest, simply because the environment cannot sustain a more rich flora. The analyses of vegetation structures did not reveal any changes either, levels of light available to the ground was therefore not expected to be higher in 2017 than 2012. The most plausible explanation for the signifanct increase in EIV-L in the ungrazed area is therefore the large decrease in relative abundance of *picea abies* seedlings from 2012 to 2017, which due to the low sample size of the area, resulted in a major impact on the EIV-L.

The results of EIV though evidences a profound effect in some habitats. Once again, the open landscapes and the semi-open forests showed the biggest changes, probably related to the correspondingly affected species richness. Also more closed forests expressed changes in EIV over time, but it is reasonable to question the reliability and stability of these results in e.g. the Norway spruce forest. Especially in these environments with very few observations, the time of sampling and the precision in determining the exactly species frequincies are crucial for the most optimal result.

Species abundance and biodiversity indices

In the rejuvenated area, we saw a progression of almost all species from 2012, due to the natural processes of succession. However, in figure 15 we saw that the three most progressive species in relative abundance was *Calamagrostis epigejos*, *Rubus idaeus* and *Deschampsia flexuosa*.

In the clear-cutted area, we expect a secondary succession, and studies normally address the traits of pioneer species to be tall, wind-pollinated capable of lateral spread and with the ability of high reproduction (Prach and Pyšek 1999). In other terms, the classic pioneer species are well adapted to changing environments and normally characterized by being r-strategists.

The *Calamagrostis epigejos* comprises all these traits and is thereby an ideal invader in the first stages of succession, if present in the seedbank. The root net is solid and deep, where the species creates long offshoots, so every clone can cover large areas. *Calamgrostis epigejos* was unwanted in the area, because of its height and almost carpet-dense growth, which creates an unfriendly environment for increased biodiversity. However, grazing management was not enough to prevent a colonization of the species, which only showed and up going tendency in abundance. As previously described, a field study from Denmark showed that in order to reduce the abundance of *Calamagrostis epigejos* significantly, a grazing pressure of 2 domestic cattle per ha was needed (Hansen, Kristensen et al.). The rejuvenated area covered an area of approximately 18 ha and the bison population consisted of an average of 15 individuals during the five years. Subsequently, even if the rejuvenated area was fenced with the whole bison herd inside it, the grazing pressure would probably still not be satisfactory to prevent the dominance of *Calamagrostis epigejos*.

The *Rubus idaeus* is a species with a high demand of both light and nutrients and with the formation of offshoots with both annuals and perennials sprouts. The conditions of the rejuvenated area was therefore optimal for the species, such as the traditional pioneer species *Deschampsia flexuosa*.

In figure 16, we saw that *Equisetum fluvitale* and *Equisetum palustre* dominated the wet meadow for all 5 years. These species are constrained to the wet meadow habitat and require a wet soil with high intensities of light, where they create a wide network of branched underground stems. In 2017, we also found a nearly identical abundance of *Mentha aquatica* compared to the species of the *Equisetum* genus, even though the abundance was estimated to 0 in the previous years. However, in 2012 and 2014 the results showed a notation of *Mentha sp.* with the abundance of 63 and 62 respectively. It is almost for sure, that *Mentha sp*. In 2012 and 2014 was *Mentha aquatica*, which would explain the similar abundance of the species in 2017. These results also indicates, that grazing in the area did not affect the abundance of the most abundant species in a noteworthy degree for the past five years, but grazing by other herbivores was a part of the area in earlier years, and previous changes to the current level might have been caused by this.

In the bush grass dominated area, we saw a continuing strong dominance of *Calamagrostis epigejos*, but also a slight increase of other species. Nevertheless, the most notable observation remains the absent effect on *Calamagrostis epigejos* as shown in figure 17, where the grazing activity did not seem to affect the species. Even though *Calamgrostis epigejos* was a part of the European bisons diet in Almindingen (Schmidt 2016), the grazing pressure was most likely not big enough to facilitate the desired effect in the area. However, the increase of other species could be assigned a positive progress in the area.

If the presumption for high biodiversity, is highest number of species with least possible dominance, the wet meadow was without doubt the area with highest biodiversity of the open landscape. The indices for the wet meadow remained close to similar for all years, where lowest and highest number of species only differed by four. This could indicate a temporal saturation of species, where the habitat has reached its maximum capacity. The high dominance in the bush grass dominated area was expected, due to the abundance of *Calamagrostis epigejos*, but the evenness increased, even though species number remained close to unchanged. Together with the increasing abundance in the area of other species, this could indicate enhanced conditions for other species, even though abundance of *Calamagrostis epigejos* did not decrease.

The rejuvenated area naturally showed a large increase in number of species, but the other indices showed high evenness and low dominance over all five years. These results therefore indicated a balanced environment, which not developed towards a more dominant area through the 5-year period.

Rubus idaeus in the bush grass dominated area (left) and *Calluna vulgaris* in the rejuvenated area (right). Photo: Bjarke Schäfer

When looking at the beech forest, a poor bottom vegetation was in general present during the entire project period. The two most abundant species in both the grazed and ungrazed area was *Oxalis acetosella* and *Deschampsia cespitosa*, which are showcased in figure 18 and 19.

Oxalis acetosella showed a slight increase in both areas from 2012 to 2014, before the abundance decreased considerably in the ungrazed area compared to the grazed, leaving a difference in relative abundance of 23 across the two areas in 2017. *Oxalis acetosella* is considered a shadow-species, which prefer a dark and moist environment. The ungrazed beech forest was in some years exposed for felling (Orbitt 2018), leaving a crown layer with greater gaps around the area. This could possibly explain the sudden decrease in abundance of *Oxalis acetosella* in the ungrazed area, since the increased light might have dried the soil and affected the *Oxalis acetosella*.

More difference was found between the grazed and ungrazed area, regarding the abundance of Deschampsia cespitosa. In the grazed area, the species was found in a low abundance through the whole study, while it increased in the ungrazed. It is unlikely that grazing affected the abundance of *Deschampsia cespitosa* to such an extent, since DNA-encoding of a previous study did not found any tracks of the species in the excrements (Schmidt 2016). The reason for the difference must therefore remain unclear, but could possibly be due to local differences in environment, seedbank or uncertainties from the used method.

In general, no observations in the beech forest gave reason to believe, that the presence of bison affected the biodiversity in the desired direction, which the biodiversity indices of the areas also indicated. However, the biggest changes in the diversity indices was found in the ungrazed area, which showed an increased biodiversity from 2012 to 2017 according to the Shannon Wiener index. As mentioned, these changes could probably be due to felling activity in the area, if harvest occurred inside or near the investigated circle plots.

Bison foot print in the alder swamp (left) and bison fur attached to a fallen tree (right). Photo: Bjarke Schäfer

The alder swamp did not present any notable changes for the most abundant species, but instead the two species *Melampyrum pratense* and *Melica uniflora* showed a large increase in abundance from especially 2014 to 2017. *Melampyrum pratense* is a semi parasitic plant species with a high seed dispersal, which prefer light open forest as habitat. The alder swamp thereby created an ideal habitat for the species, and the grazing activity could possibly have triggered the sudden increase, since species richness in general raised. The increase of *Melica uniflora* was possibly due to its well adapted morphology to grazing. With its crawling rootstocks and thin hairy leafs the *Melica uniflora* is not likely a preferred species for any herbivore, which would also explain the absence from the diet (Schmidt 2016). *Melica uniflora* requires a moist soil and can tolerate certain levels of shade, conditions that characterize the alder swamp well. The biodiversity indices furthermore supported the evidenced results of increased species richness and observed tendencies in EIV's, going through a less dominated area with higher levels of biodiversity, where the most optimal indices was found in 2017.

The old oak forest did not show any different trends between the grazed and ungrazed area regarding abundance of the most dominant species *Oxalis acetosella* and *Deschampsia flexuosa*. Even though the abundance of these species did not differ among the habitats or showed any decrease, the grazed area showed an increase in species richness and EIV-L compared to the ungrazed. The abundance of these species was therefore not a limiting factor for increased species richness in the grazed area. The low height of *Oxalis acetosella* and the narrow leafs of *Deschampsia flexuosa* are not traits which overshadows or outcompete other species, and an increase in biodiversity was perhaps possible without a decrease of the most abundandt species. Surprisingly, the abundance of *Deschampsia flexuosa* was found to decrease in the middle-aged oak forest, even though the other results indicated a less activity in this area. With the decrease of *Deschampsia flexuosa, Deschampsia cespitosa* demonstrated the reverse pattern, with an almost similar increase in abundance. The two species covers the same habitat preferences, which could possibly explain the symmetric shift, since *Deschampsia cespitosa* was not preferred as a food resource, and therefore most likely not affected by the grazing activity.
Of all the oak forest areas, the young forest showed highest levels of biodiversity, with highest number of species, highest evenness and lowest dominance as showed in table 11. In contrast, the ungrazed old oak forest showed lowest level of biodiversity, which was also expected. Nevertheless, the area did not show any remarkable changes in these indices over time, which could be a sign of a temporal saturation or a close level to optimal evenness for the certain habitats. For a further development in these forest areas, more years is probably necessary.

The Norway spruce forest revealed certain differences between the areas, where especially the ungrazed old Norway spruce forest showed lower levels biodiversity according to table 12, compared to the other areas. Differences reagrding both indices and plant species abundances were already observed in 2012, so the grazing activity from the European bison did not cause these differences among the habitats.

Browsing

Only browsing on trees and shrubs in the size category 2 (0-50cm) were present enough to provide a statistical framework. The favoring of woody material in this size category is supported by Jønsson (2014), who found evidence for biggest browsing activity on trees under 50cm.

Since the European bison is documented to be partly browser (Krasińska and Krasiński 2007, Kowalczyk, Taberlet et al. 2011, Cromsigt, Kemp et al. 2017), there can be several explanations for only enough observation in this category.

In any case, it would be hard to determine the browsing activity on the smallest seedlings, since most of the above ground biomass will be consumed, if exposed to browsing. We consequently have to assume, that browsing activity was higher than observed for the smallest seedlings.

In size category 5 (above 2 meter), many species would be unavailable for browsing, because branches would be above browsing height.

That still leaves us with size category 3 and 4. Looking at the observations for both 2014 and 2017, not many individuals in these categories were present in the entire area at all. One purpose of the grazing activity is to remove smaller shrubs and reduce the up growth of trees, and since the first measurement of browsing took place in 2014, a possible explanation might be, that the grazing the first two years have prevented up growth to this stage of succession. Unfortunately, no control fields were included in the investigated sites to compare and evaluate this hypothesis. Nevertheless, by visual observation of a similar rejuvenated area outside the fencing, a more dense growth of *Betula pendula* were present, and further studies to test this thesis could be relevant to determine the actual effect on up growth.

The browsing results showed a browsing probability for most species in the interval between 0.1 - 0.4. Only *Picea abies* showed no probability for browsing in both years, which indicate a selection against the species as food source. This is common for other browsers, where conifers often are deselected as food source (Buttenschøn 2007).

The other species that stands out is the *Betula pendula* in 2014. The probability for browsing was significant higher than in 2017 and much higher than the other species. The most obvious explanation could be a smaller amount of *Betula pendula* in the size of 0-50cm in 2014, where more than 70% then would be browsed. However, looking at the observations we found that 1.411 *Betula pendula* individuals of the size category 0-50cm was found in the investigated fields in 2014 against only 156 in 2017, and we can therefore reject this hypothesis. From the observations in 2014, we also found that more than 1.200 of the individuals present were found in the rejuvenated area. The population of *Betula pendula* were therefore distributed in a clumped pattern and gathered in a small area. This suggest that the bison herd have crossed the small area with the large population of *Betula pendula*, and browsed on almost every individual present, which would have caused the significant difference.

Future perspectives for the reintroduction

Since the reintroduction of free-living European bisons in 1952, only few studies have been conducted with the purpose of determine the potential of the species as a management tool in restoration projects. These studies mainly focused on habitat selection and food preferences, which made this 5-year project in Almindingen one of the first to investigate and evaluate effect on biodiversity. After this single project, extending over 5-years, our understanding of the ecological effects from the presence of the European bison is only enlarged in a limited amount. This should therefore only be one of further projects in the future, investigating the effect on species richness and biodiversity.

More studies in the future are needed, in order to determine whether the European bison can contribute as a new management tool to alter and affect the biodiversity in a new way, compared with the already present grazing fauna.

Not only are additional studies needed, but new study set-ups as well. The project in Almindingen covered a broad vegetation, where the bison herd had accessibility to both open land and closed forest. In the future, meadows without interfering from other grazers should be investigated, but also grassland and more deciduous forest should be included. The open landscape only covered 15% of the study area, also supporting the present tendency of managing the European bison as a forest specialist, even though its evolutionary background, dental morphology and diet addresses the species as a grazer (Cromsigt, Kemp et al. 2017).

This should also ease focus to investigate bisons effect and fitness in more open habitat, to contribute to the debated refugee-theory, which address the European bison as being marginalized to the forest in prehistoric time, as a result of human predation (Pucek, Belousova et al. 2004). This theory needs a further examination, since conservation practices in suboptimal habitats only enlarge the risk of refugee species to occur and forcing displacements in the fundamental niches of species (Braunisch, Bollmann et al. 2008).

However, even if the forest is a sub-optimal habitat for the European bison, Its behaviour and food preferences from previous findings suggest, that the species both need open land and forest habitat (Brandtberg and Dabelsteen 2013, Cromsigt, Kemp et al. 2017). Since domestic cattle is the only specie in Denmark, which are considered a large herbivore, this is also why the European bison should contribute to the future nature management practices. While there is a bit of controversy regarding habitat preferences for the European bison, the domestic cattle does definitely not belong in the forest, but is a distinctive grazing specialist. The results from this study also indicates that the European bison could affect forests areas, but it requires forest areas which also contains a certain amount of grasses and herbs like the alder swamp or young oak forest. If not, the migration availability to areas with more versatile food resources should be available.

Since extinction of species and biodiversity crisis is a wide-ranging matter, the possible effects of grazing by European bison should also be expanded, covering a more comprehensive understanding of increased biodiversity. As defined in the introduction, biodiversity includes variability among living organisms from all sources. This study only investigates the effects on vascular plants, but other organisms may also be relevant to survey in the future. It is reasonable to believe, that the bark striping activity possibly could create a new habitat, for an increased diversity of e.g. lichens or fungus on the striped trees, or the bisons dungs could assist an increased fauna of dung beetles. New studies also suggest that the activity of free ranging bison could decrease density of earthworms, due to compaction of the soil (Ivanova, Smirnov et al. 2018). Furthermore, if the species richness of vascular plants were increased, it would most likely facilitate an increased biodiversity of other organisms which are dependent on a more rich flora as habitat e.g. insects.

Increased biodiversity of vascular plants could possibly affect biodiversity of other organisms. *Stictoleptura sp*. (left) and *Lycaenidae sp.* (right). Photo: Bjarke Schäfer

Biodiversity associated with the vascular plants are a relevant factor to investigate, since e.g. pollinators are declining on a global scale (Potts, Biesmeijer et al. 2010). New herbaceous plants and increased abundance of already present species was found in this study after five years of grazing. Subsequently, increased habitat quality for pollinators would be generated by grazing of the European bison, since pollinators favours flowering plants (Potts, Vulliamy et al. 2003). This could be another subject to investigate, since this study only covers an isolated area of the wide-ranging biodiversity term.

Chorthippus sp (left) and *Bombus sp* (right). Photo: Bjarke Schäfer

The reintroduction of the European bison in Almindingen was planned as a 5-year project, from 2012-2017. After the five years, the project was evaluated to determine the further course of the European bisons in Almindingen. The most recent information about the project suggest an expansion of the fencing, but a scenario for the future propose a total release of the herd, creating a free-ranging population on the island of Bornholm. One of the prerequisites for releasing the herd is a documented effect of the species contribution to the nature management practices. However, as previously mentioned, a 5-year study covers a to short period to make any conclusion in long term effects of grazing by European bison in especially the forest areas. Nevertheless, the results though embraces tendencies of enhanced biodiversity of plants in more areas in Almindingen, which does not jeopardise a future release. Subsequently, some considerations are necessary before the discussion of a future release can transpire, one being the discussion of damage caused by the European bison. In Lithuania, a free-ranging population of 186 individuals caused damage in a anthropogenic agricultural landscape, with a damage compensation of 98.820 euros per year (Kibiša, Marozas et al. 2017). In addition, other factors needs to be considered e.g. interactions with present fauna and the general view from the public. Introduction of new species therefore involves many aspects and opinions, which needs to be evaluated and taken into account.

However, a reintroduction of the European bison would of course also be valuable in other ways. In the national forest program, many of the objectives would be benefitted from a reintroduction of the European bison (Miljøministeriet 2002). The European bison would e.g. develop and communicate a shift to more natural forestry, strengthen the possibilities for nature-experiences in the forests and improve the use of the state-owned forests to test and further develop management methods and operating principles.

Nevertheless, no matter how the further project of the European bison in Almindingen develops, the project have been of great importance and value for the last five years. The release of Europe's largest terrestrial animal have also functioned as a statement that makes it inevitable for many people to relate to the current situation of biodiversity on a larger scale. Public awareness have increased for both the reasons for the reintroduction, but also for the history of the European bison. Opinions in the local community have of course been many, but the majority of the local community on Bornholm have been proud of introducing the European bison for both locals and tourist (Orbitt 2018).

The further course of the European bison in Almindingen should therefore be followed with great interest, and not only in terms of the improving effects on biodiversity of vascular plants, but also the celebration of an infinite existential value of the wild and free living nature.

Conclusion

The European bison was introduced to Almindingen with the purpose of increasing biodiversity through ecological engineering, with its unique set of herbivore characteristics.

The reintroduction was successfully implemented, where the seven original individuals also managed to increase the population over time.

After five years of grazing, the results showed an increased species richness of vascular plants in five out of the thirteen habitat types. These five habitats were the alder swamp, the young oak forest, the old oak forest, the rejuvenated area and on the wet meadow. Furthermore, an increase was proven for the Ellenberg indicator value for light in five habitat types, in this case being the middle-aged oak forest, the old oak forest, the ungrazed old Norway spruce forest, the rejuvenated area and on the wet meadow. A decrease was observed for the Ellenberg indicator value for light in one habitat type, which was the grazed old oak forest. Furthermore, an increase was found for the Ellenberg indicator value for nitrogen in the bush grass dominated area. Browsing was preferred on deciduous tree species with the size of 0-50cm, while this study did not found any quantifiable amount of bark striping on any species in the fencing.

This study found evidence for increased biodiversity, since species richness and Ellenberg indicator values changed significantly through the 5-year period in both open land and forest vegetation. These results implies that the European bison could contribute to Danish nature management practices, by enhancing biodiversity of vascular plants in booth open land and forests. This intermediate behaviour in combination with its exclusive set of herbivore characteristics differs from the present fauna in the Danish nature. For a more complete picture of the European bisons effect on biodiversity, more studies should be conducted with different study set-ups to strengthen the results, but also to investigate other types of biodiversity. Furthermore, longer study extends should be investigated to get a more proper representation of long-term effects in especially the forest environments.

Epilogue

*"Skulle man nogensinde, træt af at omtumles
paa Livets tornfulde Vei, ønske sig et Tilflugtssted,
for i Ro at betragte Naturen og dens Værker, da
var her et passende Sted."*

- Ole Jørgen Rawert, historiker og kunster,
om Almindingen i 1821

References

Aerts, R. (1999). "Interspecific competition in natural plant communities: mechanisms, trade-offs and plant-soil feedbacks." Journal of Experimental Botany **50**(330): 29-37.

Akula, R. and G. A. Ravishankar (2011). "Influence of abiotic stress signals on secondary metabolites in plants." Plant signaling & behavior **6**(11): 1720-1731.

Bartelheimer, M. and P. Poschlod (2016). "Functional characterizations of E llenberg indicator values–a review on ecophysiological determinants." Functional ecology **30**(4): 506-516.

Bazzaz, F. (1979). "The physiological ecology of plant succession." Annual review of ecology and systematics **10**(1): 351-371.

Bergmeier, E., J. Petermann and E. Schröder (2010). "Geobotanical survey of wood-pasture habitats in Europe: diversity, threats and conservation." Biodiversity and Conservation **19**(11): 2995-3014.

Bocherens, H., E. Hofman-Kamińska, D. G. Drucker, U. Schmölcke and R. Kowalczyk (2015). "European bison as a refugee species? Evidence from isotopic data on Early Holocene bison and other large herbivores in northern Europe." PLoS One **10**(2): e0115090.

Borowski, S. and S. Kossak (1972). "Bisoniana LI. The natural food preferences of the European bison in seasons free of snow cover." Acta Theriologica **17**(13): 151-169.

Brandtberg, N. H. and T. Dabelsteen (2013). "Habitat selection of two European bison (Bison bonasus) on the Danish island Bornholm." European Bison Conservation Newsletter **6**: 73-80.

Braunisch, V., K. Bollmann, R. F. Graf and A. H. Hirzel (2008). "Living on the edge—modelling habitat suitability for species at the edge of their fundamental niche." Ecological modelling **214**(2-4): 153-167.

Brender, B. (2016). "The European bisons, Bison bonasus; impact on pedunculated oak and Norway spruce in Almindingen on Bornholm". Department of biology, University of southern Denmark.

Buttenschøn, R. (2007). Græsning og høslæt i naturplejen, Center for Skov, Landskab og Planlægning/Københavns Universitet.

Caboń-Raczyńska, K., M. Krasińska, Z. A. Krasiński and J. M. Wójcik (1987). "Bisoniana XCVII. Rhythm of daily activity and behavior of European bison in the Białowieża Forest in the period without snow cover." Acta Theriologica **32**(21): 335-372.

Caro, T. and P. Sherman (2009). "Rewilding can cause rather than solve ecological problems." Nature **462**(7276): 985.

Cederlund, G. and A. Nyström (1981). "Seasonal differences between moose and roe deer in ability to digest browse." Ecography **4**(1): 59-65.

Cremene, C., G. Groza, L. Rakosy, A. A. Schileyko, A. Baur, A. Erhardt and B. Baur (2005). "Alterations of steppe-like grasslands in Eastern Europe: a threat to regional biodiversity hotspots." <u>Conservation Biology</u> **19**(5): 1606-1618.

Cromsigt, J. P., Y. J. Kemp, E. Rodriguez and H. Kivit (2017). "Rewilding Europe's large grazer community: how functionally diverse are the diets of European bison, cattle, and horses?" <u>Restoration Ecology</u>.

Damgaard, C. (2015). "Revisiting the Böcher-modified Raunkiær method for estimating the frequency of plant species." <u>Ecological informatics</u> **26**: 1-5.

De Groot, R. S., M. A. Wilson and R. M. Boumans (2002). "A typology for the classification, description and valuation of ecosystem functions, goods and services." <u>Ecological economics</u> **41**(3): 393-408.

Díaz, S., I. Noy-Meir and M. Cabido (2001). "Can grazing response of herbaceous plants be predicted from simple vegetative traits?" <u>Journal of Applied ecology</u> **38**(3): 497-508.

Frederiksen, S. Rasmussen, F. N. Seberg, O. (2012). "Dansk flora". 2nd edition. Gyldendal

Gębczyńska, Z. (1991). "Bisoniana 105. Food eaten by the free-living European bison in Białowieża Forest." <u>Acta Theriologica</u> **36**(3-4): 307-313.

Glenn-Lewin, D. C., R. K. Peet and T. T. Veblen (1992). <u>Plant succession: theory and prediction</u>, Springer Science & Business Media.

Gordon, I. J. and H. H. Prins (2008). <u>The ecology of browsing and grazing</u>, Springer.

Grime, J. (1979). <u>Primary strategies in plants</u>. Transactions of the Botanical Society of Edinburgh, Taylor & Francis.

Groom, M. J. (2006). "Threats to biodiversity." <u>Principles of conservation Biology</u> **3**: 63-109.

Gössling, S. (1999). "Ecotourism: a means to safeguard biodiversity and ecosystem functions?" <u>Ecological economics</u> **29**(2): 303-320.

Hahn, K., J. Emborg, L. Vesterdal, S. Christensen, R. H. Bradshaw, K. Raulund-Rasmussen and J. B. Larsen (2007). "Natural Forest Stand Dynamics in Time and Space: Synthesis of Research in Suserup Skov, Denmark and Perspectives for Forest Management." <u>Ecological Bulletins</u>: 183-194.

Hansen, L. H., U. R. Kristensen and M. Mogensen "Naturforvaltning på Kalvebod Fælled."

Hester, A., L. Edenius, R. Buttenschøn and A. Kuiters (2000). "Interactions between forests and herbivores: the role of controlled grazing experiments." <u>Forestry</u> **73**(4): 381-391.

Hill, M. and P. Carey (1997). "Prediction of yield in the Rothamsted Park Grass Experiment by Ellenberg indicator values." <u>Journal of Vegetation Science</u> **8**(4): 579-586.

Hill, M. O., J. Mountford, D. Roy and R. G. H. Bunce (1999). Ellenberg's indicator values for British plants. ECOFACT Volume 2 Technical Annex, Institute of Terrestrial Ecology.

Hofmann, R. R. (1989). "Evolutionary steps of ecophysiological adaptation and diversification of ruminants: a comparative view of their digestive system." Oecologia **78**(4): 443-457.

Ivanova, N., V. Smirnov, L. Khanina, M. Bobrovsky and M. Shashkov (2018). "Changes in Vegetation and Earthworm Populations under Free Grazing European Bison (Bison bonasus) in Broad-Leaved Forests of the Kaluzhskie Zaseki State Nature Reserve." Biology Bulletin **45**(1): 100-109.

Jaroszewicz, B., K. Kwiecień, P. Czortek, W. Olech and E. Pirożnikow (2017). "Winter supplementary feeding influences forest soil seed banks and vegetation." Applied Vegetation Science **20**(4): 683-691.

Jaroszewicz, B., E. Pirożnikow and R. Sagehorn (2009). "Endozoochory by European bison (Bison bonasus) in Białowieża Primeval Forest across a management gradient." Forest Ecology and Management **258**(1): 11-17.

Jensen, P. N., S. Boutrup, J. R. Fredshavn, L. M. Svendsen, G. Blicher-Mathiesen, P. Wiberg-Larsen, R. Bjerring, J. W. Hansen, B. Søgaard and S. Pihl (2015). Vandmiljø og Natur 2013: NOVANA. Tilstand og udvikling-faglig sammenfatning, Aarhus Universitet, DCE-Nationalt Center for Miljø og Energi.

Jønsson, F. M. (2014). "Føde- og habitatpræferencer hos europæisk bison (Bison bonasus) på Bornholm". København: Det Natur- og Biovidenskabelige fakultet. Københavns Universitet - Institut for Geovidenskab og Naturforvaltning.

Karbowiak, G., A. W. Demiaszkiewicz, A. M. Pyziel, I. Wita, B. Moskwa, J. Werszko, J. Bień, K. Goździk, J. Lachowicz and W. Cabaj (2014). "The parasitic fauna of the European bison (Bison bonasus)(Linnaeus, 1758) and their impact on the conservation. Part 1 The summarising list of parasites noted." Acta parasitologica **59**(3): 363-371.

Kerley, G., R. Kowalczyk and J. Cromsigt (2012). "Conservation implications of the refugee species concept and the European bison: king of the forest or refugee in a marginal habitat?" Ecography **35**(6): 519-529.

Kibiša, A., V. Marozas, D. Talijūnas, R. Papšys, S. Gintarė and K. Šimkevičius (2017). "IMPACT OF FREE-RANGING EUROPEAN BISON TO ECOSYSTEMS IN FRAGMENTED LANDSCAPE, LITHUANIA." Balkan Journal of Wildlife Research **4**(2): 18-25.

Kowalczyk, R., P. Taberlet, E. Coissac, A. Valentini, C. Miquel, T. Kamiński and J. M. Wójcik (2011). "Influence of management practices on large herbivore diet—Case of European bison in Białowieża Primeval Forest (Poland)." Forest Ecology and Management **261**(4): 821-828.

Krasińska, M. and Z. A. Krasiński (2007). "European bison." The Nature Monograph. Mammal Research Institute PAS, Białowieża: 318.

Kuemmerle, T., T. Hickler, J. Olofsson, G. Schurgers and V. C. Radeloff (2012). "Reconstructing range dynamics and range fragmentation of European bison for the last 8000 years." <u>Diversity and Distributions</u> **18**(1): 47-59.

Metera, E., T. Sakowski, K. Słoniewski and B. Romanowicz (2010). "Grazing as a tool to maintain biodiversity of grassland-a review." <u>Animal Science Papers and Reports</u> **28**(4): 315-334.

Miljøministeriet. (2002). "Danmarks nationale skovprogram"

Noe-Nygaard, N., T. D. Price and S. U. Hede (2005). "Diet of aurochs and early cattle in southern Scandinavia: evidence from 15N and 13C stable isotopes." <u>Journal of Archaeological Science</u> **32**(6): 855-871.

Nogués-Bravo, D., D. Simberloff, C. Rahbek and N. J. Sanders (2016). "Rewilding is the new Pandora's box in conservation." <u>Current Biology</u> **26**(3): R87-R91.

Nowak, R. M. and E. P. Walker (1999). <u>Walker's Mammals of the World</u>, JHU Press.

Olech, W. and K. Perzanowski (2014). "Best practices manual for protection of European Bison." <u>Warsaw: Coordination Center for Environmental Projects</u>.

Olff, H. and M. E. Ritchie (1998). "Effects of herbivores on grassland plant diversity." <u>Trends in ecology & evolution</u> **13**(7): 261-265.

Orbitt, J. (2018). Local forest ranger from the National Nature Agency, Bornholm. Personal communications.

Pedersen, L. B., R. M. Buttenschøn and T. S. Jensen (2001). "Græsning på ekstensivt drevne naturarealer."

Persson, H. (1980). "Fine-root production, mortality and decomposition in forest ecosystems." <u>Vegetatio</u> **41**(2): 101-109.

Piñeiro, G., J. M. Paruelo, M. Oesterheld and E. G. Jobbágy (2010). "Pathways of grazing effects on soil organic carbon and nitrogen." <u>Rangeland Ecology & Management</u> **63**(1): 109-119.

Potts, S. G., J. C. Biesmeijer, C. Kremen, P. Neumann, O. Schweiger and W. E. Kunin (2010). "Global pollinator declines: trends, impacts and drivers." <u>Trends in ecology & evolution</u> **25**(6): 345-353.

Potts, S. G., B. Vulliamy, A. Dafni, G. Ne'eman and P. Willmer (2003). "Linking bees and flowers: how do floral communities structure pollinator communities?" <u>Ecology</u> **84**(10): 2628-2642.

Prach, K. and P. Pyšek (1999). "How do species dominating in succession differ from others?" <u>Journal of Vegetation Science</u> **10**(3): 383-392.

Proulx, M. and A. Mazumder (1998). "Reversal of grazing impact on plant species richness in nutrient-poor vs. nutrient-rich ecosystems." <u>Ecology</u> **79**(8): 2581-2592.

Pucek, Z., I. P. Belousova, Z. A. Krasiński and W. Olech (2004). European bison: status survey and conservation action plan, IUCN Gland, Switzerland and Cambridge.

Ravensbeck, L., P. Andersen, B. J. Thorsen and N. Strange (2013). Økosystemtjenester og deres værdisætning i tilknytning til et landsdækkende grønt naturnetværk i Danmark: eksisterende viden, potentiel anvendelse og vidensbehov, Institut for Fødevare-og Ressourceøkonomi, Københavns Universitet.

Rayé, G., C. Miquel, E. Coissac, C. Redjadj, A. Loison and P. Taberlet (2011). "New insights on diet variability revealed by DNA barcoding and high-throughput pyrosequencing: chamois diet in autumn as a case study." Ecological Research 26(2): 265-276.

Reaka-Kudla, M. L., D. E. Wilson and E. O. Wilson (1996). Biodiversity II: understanding and protecting our biological resources, Joseph Henry Press.

Rowley-Conwy, P. (1985). "The origin of agriculture in Denmark: a review of some theories." Journal of Danish Archaeology 4(1): 188-195.

Schmidt, E. N. B. (2016). "Meta-barcoding reveals high contribution of shrubs and trees in the diet of the European bison (Bison Bonasus) on Bornholm, Denmark - Local adaption og global misconception?" Department of geosciences and natural resource management, University of Copenhagen, Denmark.

Schuman, G., J. Reeder, J. Manley, R. Hart and W. Manley (1999). "Impact of grazing management on the carbon and nitrogen balance of a mixed-grass rangeland." Ecological Applications 9(1): 65-71.

Smit, C. and R. Putman (2010). "Large herbivores as' environmental engineers'." Ungulate management in Europe: problems and practices: 260-283.

Smit, C., J. L. Ruifrok, R. van Klink and H. Olff (2015). "Rewilding with large herbivores: The importance of grazing refuges for sapling establishment and wood-pasture formation." Biological Conservation 182: 134-142.

Tokarska, M., C. Pertoldi, R. Kowalczyk and K. Perzanowski (2011). "Genetic status of the European bison Bison bonasus after extinction in the wild and subsequent recovery." Mammal Review 41(2): 151-162.

Van Wieren, S. (1995). "The potential role of large herbivores in nature conservation and extensive land use in Europe." Biological Journal of the Linnean Society 56: 11-23.

Vera, F. (2000). "Succession, the climax forest and the role of large herbivores." Grazing ecology and forest history: 13-60.

Aaris-Sørensen, K. (1990). "Danmarks forhistoriske dyreverden."

__Supplementary materials and appendices__

Appendix 1. Registration scheme for forest vegetation

Felt: **Dato:**

Registrering I 15 m cirkel			
Store træer (dbh < 40 cm)		**Hulheder, dødt ved, mos og lav**	
Art	Dbh		Antal
		Med spæthuller	
		Med større hulheder (end spæthuller)	
		Kraftig bevoksning med lav/mos	
		Dødt stående træ (dbh <25 cm, højde < 2 m)	
		Dødt liggende ved (dbh <25 cm, længde <5 m)	
		Antal faldhobe (bison)	
		Antal faldhobe (rådyr)	
		Antal dyreveksler	

Registrering i 5 m cirkel, træer med diameter i brysthøjde større end 10 dm					
Art	Dbh	Art	Dbh	Art	Dbh

Registrering af dækning af træer, bar jord og vandflade i 5 m cirkel				
Træer og buske under 1 m				
<5%	5-10%	11-30%	31-75%	>75%
Træer og buske over 1 m				
<5%	5-10%	11-30%	31-75%	>75%
Samlet kronedække				
<20%	20-50%	51-75%	76-90%	>90%
Samlet vandflade				
<5%	5-10%	11-30%	31-75%	>75%
Bar jord				
<5%	5-10%	11-30%	31-75%	>75%

Registrering af foryngelse i 5 cirkel			
Art	Ingen	< 2pr/m²	> 2pr/m²

Art	1 (kimplante)	2 (<50 cm)	3 (50-100 cm)	4 (100-200 cm)	5 (>200 cm)

Bid og skrælning			
Art	Bid	Størrelse	Skrælning

Appendix 2. Registration scheme for open land vegetation

Vegetationsstruktur (i 5m. cirkel)					
Arealandel	0–5%	5–10%	10–30%	30–75%	75–100%
Uden vegetationsdække (bar jord, sand, vand)					
Laver					
Bladmosser					
Sphagnummosser					
Græs/urtevegetation under 15 cm					
Græs/urtevegetation 15 – 50 cm					
Græs/urtevegetation over 50 cm					
Dværgbuske					
	0%	1-10%	10-25%	25-50%	50-100%
Vedplanter (kronedække)					
Forekomst af invasive arter					
Afgræsning og drift					
	0-5%	5-10%	10-30%	30-75%	75-100%
Græsning (med tydelige tegn på græsning)					
Slåning/rydning (biomasse efterladt)					
Arealandel	0%	1-10%	10-25%	25-50%	50-100%
Tydeligt eutrofieret (direkte gødning el. tilskudsfodring)					
Hydrologi					
Afvanding (kun lavbundsjorde)	Ingen grøfter eller dræn	Afvanding (svag effekt)	Tydelig effekt	Udbredt effekt	Fuldstændig afvandet

Hulheder, dødt ved, mos og lav	
Antal faldhobe (bison)	
Antal faldhobe (rådyr)	
Antal dyreveksler	

Art	1 (kimplante)	2 (<50 cm)	3 (50-100 cm)	4 (100-200 cm)	5 (>200 cm)

Bid og skrælning			
Art	Bid	Størrelse	Skrælning

Appendix 3: Progressive and declining species in the different vegetation types
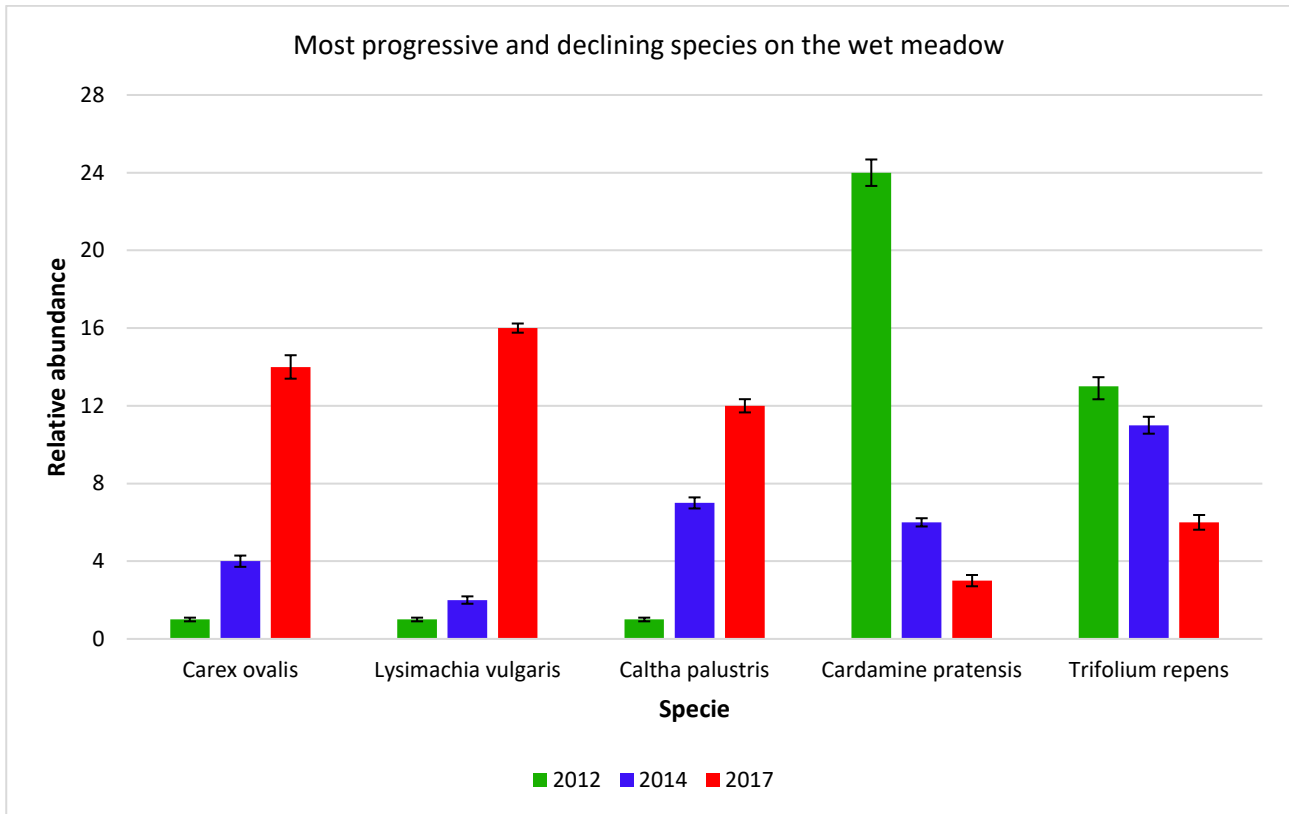
Figure 27. Relative abundance of the most progressive and declining species on the wet meadow (±SE). Relative abundance of *Carex ovalis* in 2012 was 1 (SE=0.094), in 2014 was it 4 (SE=0.289) and in 2017 was it 14 (SE=0.603). Relative abundance of *Lysimachia vulgaris* in 2012 was 1 (SE=0.094), in 2014 was it 2 (SE=0.189) and in 2017 was it 16 (SE=0.236). Relative abundance of *Caltha palustris* in 2012 was 1 (SE=0.094), in 2014 was it 7 (SE=0.284) and in 2017 was it 12 (SE=0.340). Relative abundance of *Cardamine pratensis* in 2012 was 24 (SE=0.681), in 2014 was it 6 (SE=0.212) and in 2017 was it 3 (SE=0.289). Relative abundance of *Trifolium repens* in 2012 was 13 (SE=0.664), in 2014 was it 11 (SE=0.434) and in 2017 was it 6 (SE=0.379).
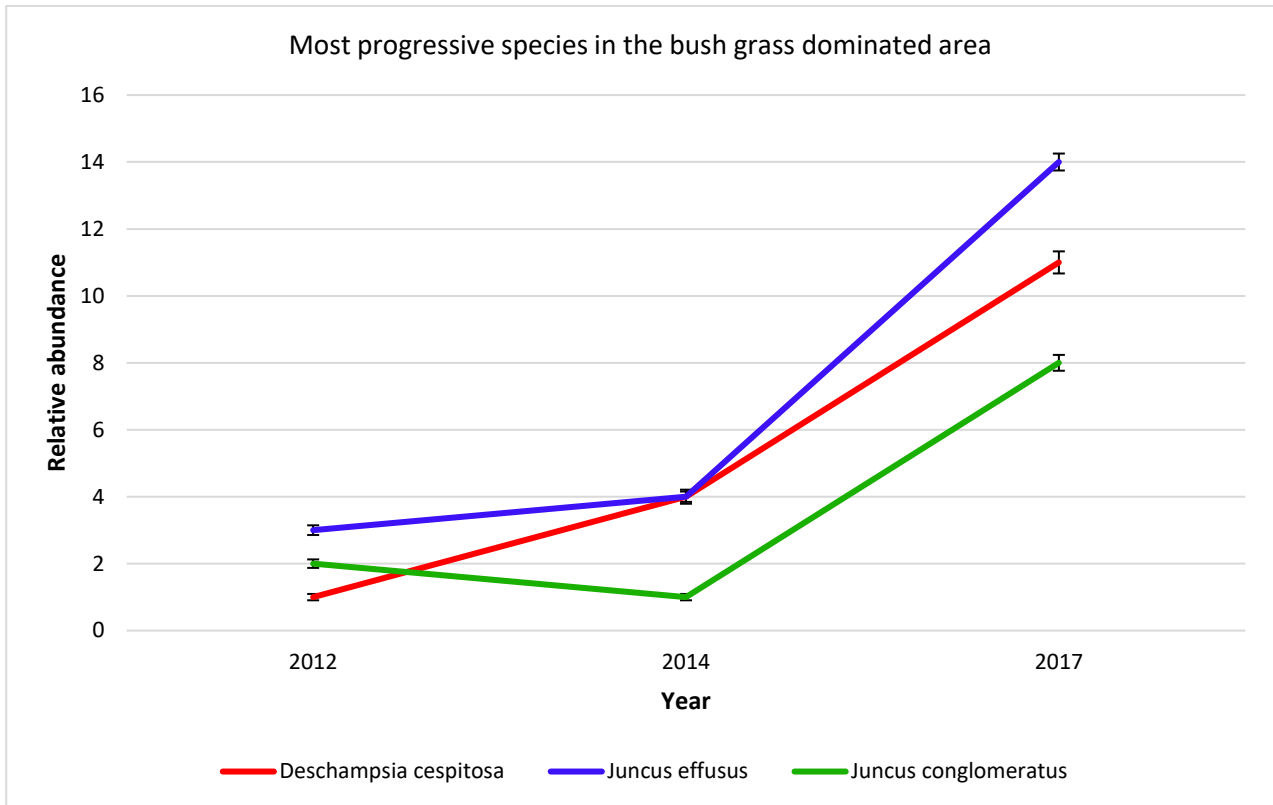
Figure 28. Relative abundance of the most progressive species in the bush grass dominated area (±SE). Relative abundance of *Deschampsia cespitosa* was in 2012 1 (SE=0.094), in 2014 was it 4 (SE=0.154) and in 2017 was it 11 (SE=0.330). Relative abundance of *Juncus effusus* was in 2012 3 (SE=0.144), in 2014 was it 4 (SE=0.209) and in 2017 was it 14 (SE=0.252). Relative abundance of *Juncus conglomeratus* was in 2012 2 (SE=0.126), in 2014 was it 1 (SE=0.094) and in 2017 was it 8 (SE=0.236).
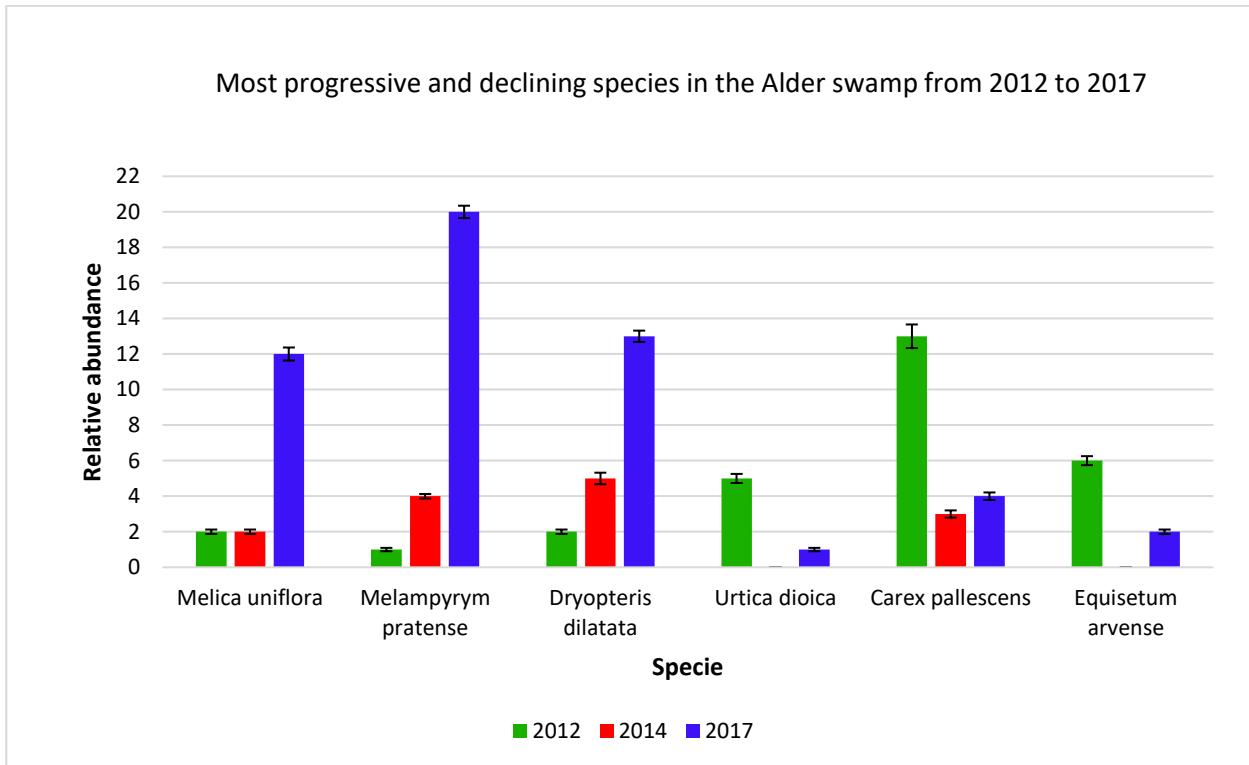
Figure 29. Relative abundance of the most progressive and declining species in the alder swamp (±SE). Relative abundance of *Melica uniflora* in 2012 was 2 (SE=0.126), in 2014 was it 2 (SE=0.126) and in 2017 was it 12 (SE=0.369). Relative abundance of *Melampyrum pratense* in 2012 was 1 (SE=0.094), in 2014 was it 4 (SE=0.126) and in 2017 was it 20 (SE=0.346). Relative abundance of *Dryopteris dilatata* in 2012 was 2 (SE=0.126), in 2014 was it 5 (SE=0.324) and in 2017 was it 13 (SE=0.317). Relative abundance of *Urtica dioica* in 2012 was 5 (SE=0.254), in 2014 was it 0 (SE=0.000) and in 2017 was it 1 (SE=0.094). Relative abundance of *Carex pallescens* in 2012 was 13 (SE=0.664), in 2014 was it 3 (SE=0.202) and in 2017 was it 4 (SE=0.212). Relative abundance of *Equisetum arvense* in 2012 was 6 (SE=0.252), in 2014 was it 0 (SE=0.000) and in 2017 was it 2 (SE=0.126).
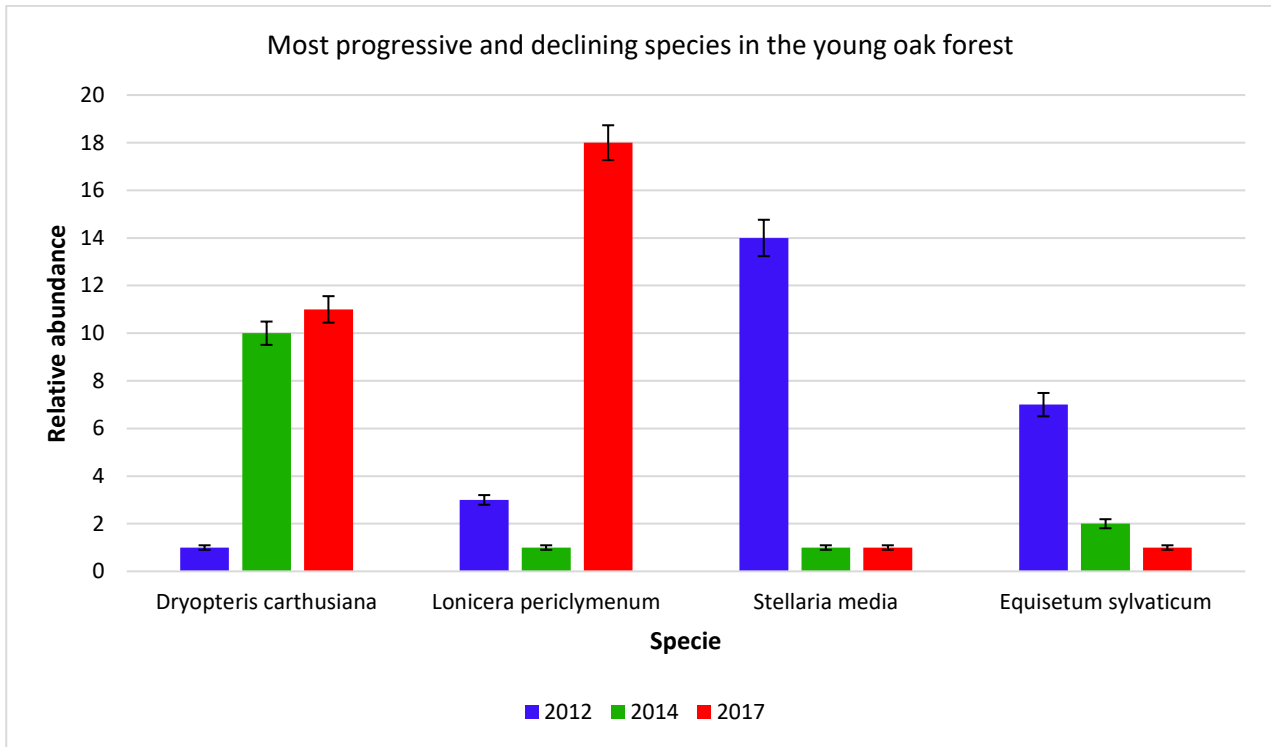
Figure 30. Relative abundance of the most progressive and declining species in the young oak forest (±SE). Relative abundance of *Dryopteris carthusiana* in 2012 was 1 (SE=0.094), in 2014 was it 10 (SE=0.489) and in 2017 was it 11 (SE=0.555). Relative abundance of *Lonicera periclymenum* in 2012 was 3 (SE=0.202), in 2014 was it 1 (SE=0.094) and in 2017 was it 18 (SE=0.732). Relative abundance of *Stellaria media* in 2012 was 14 (SE=0.764), in 2014 was it 1 (SE=0.094) and in 2017 was it 1 (SE=0.094). Relative abundance of *Equisetum sylvaticum* in 2012 was 7 (SE=0.490), in 2014 was it 2 (SE=0.189) and in 2017 was it 1 (SE=0.094).
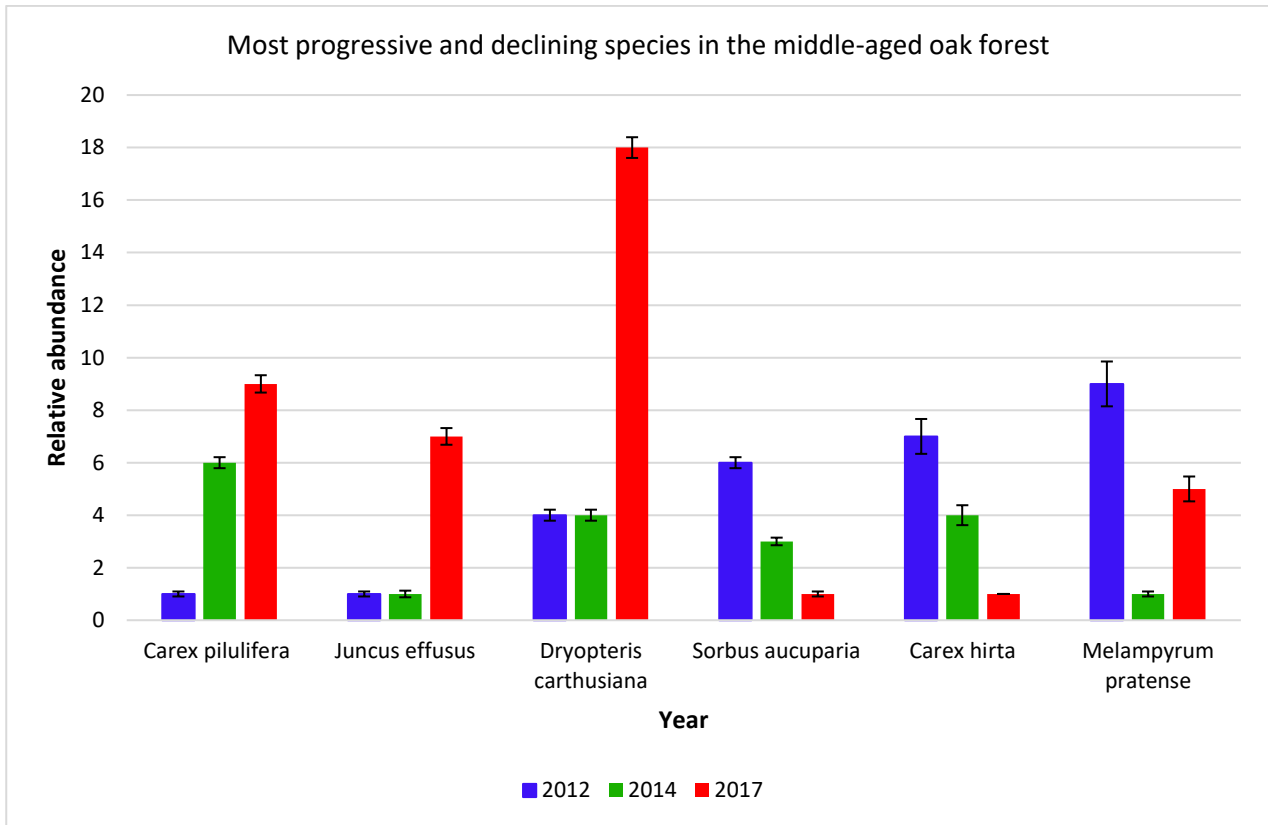
Figure 31. Relative abundance of the most progressive and declining species in the middle-aged oak forest (±SE). Relative abundance of *Carex pilulifera* in 2012 was 1 (SE=0.094), in 2014 was it 6 (SE=0.209) and in 2017 was it 9 (SE=0.330). Relative abundance of *Juncus effusus* in 2012 was 1 (SE=0.094), in 2014 was it 1 (SE=0.094) and in 2017 was it 7 (SE=0.317). Relative abundance of *Dryopteris carthusiana* in 2012 was 4 (SE=0.209), in 2014 was it 4 (SE=0.209) and in 2017 was it 18 (SE=0.304). Relative abundance of *Sorbus aucuparia* in 2012 was 6 (SE=0.209), in 2014 was it 3 (SE=0.144) and in 2017 was it 1 (SE=0.094). Relative abundance of *Carex hirta* in 2012 was 7 (SE=0.664), in 2014 was it 4 (SE=0.379) and in 2017 was it 1 (SE=0.094). Relative abundance of *Melampyrum pratense* in 2012 was 9 (SE=0.853), in 2014 was it 1 (SE=0.094) and in 2017 was it 5 (SE=0.473).
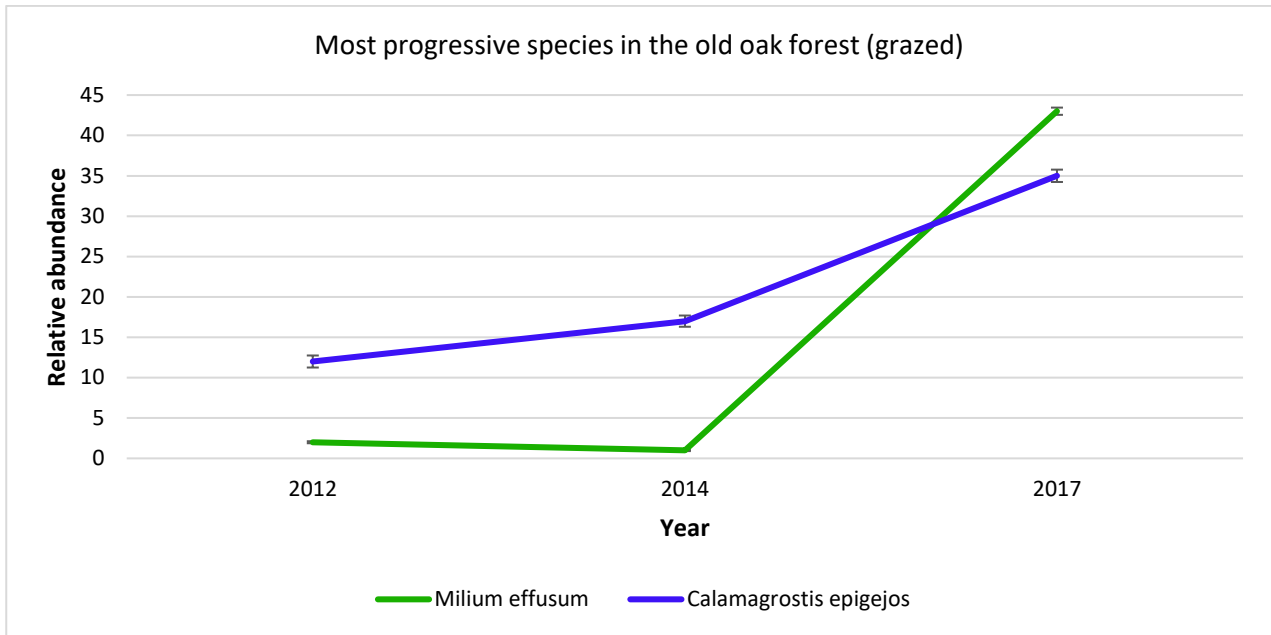
Figure 32. Relative abundance of the most progressive species in the grazed old oak forest (±SE). Relative abundance of *Milium effusum* was in 2012 2 (SE=0.126), in 2014 was it 1 (SE=0.094) and in 2017 was it 43 (SE=0.448). Relative abundance of *Calamagrostis epigejos* was in 2012 12 (SE=0.745), in 2014 was it 17 (SE=0.695) and in 2017 was it 35 (SE=0.764).
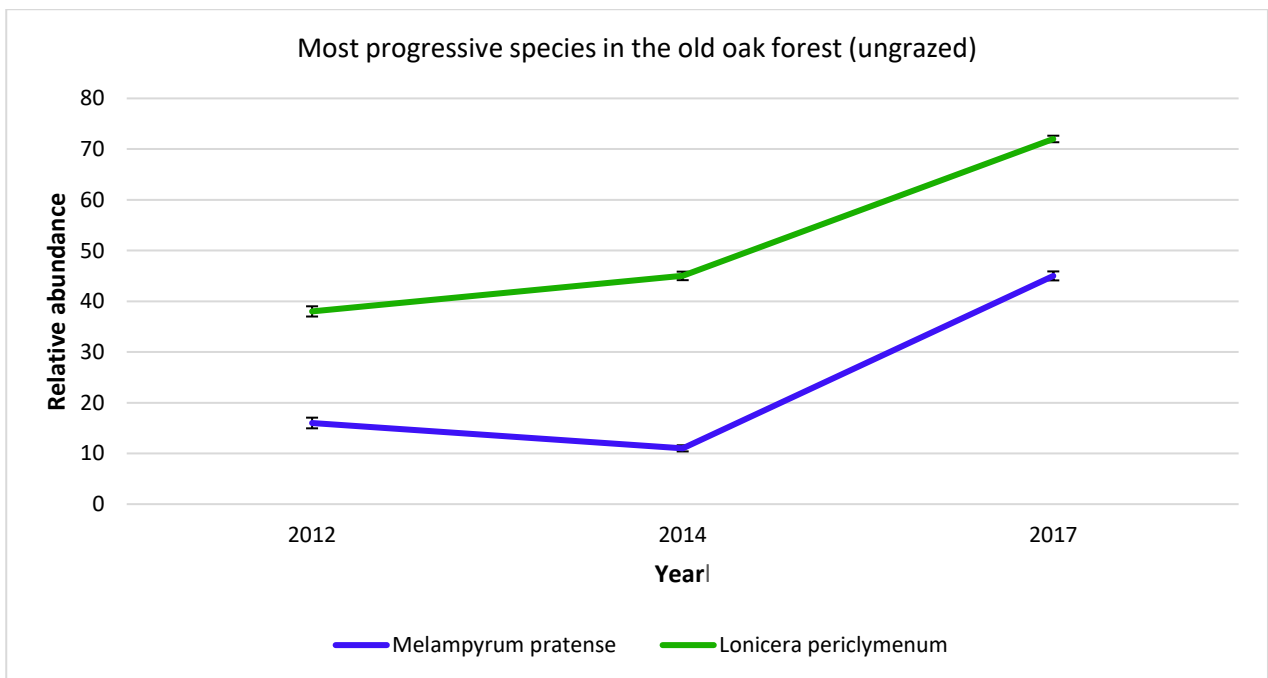
Figure 33. Relative abundance of the most progressive species in the ungrazed old oak forest (±SE). Relative abundance of *Melampyrum pratense* was in 2012 (SE=), in 2014 was it (SE=) and in 2017 was it (SE=). Relative abundance of *Lonicera periclymenum* was in 2012 (SE=), in 2014 was it (SE=) and in 2017 was it (SE=).

Appendices 4: Relative abundance and frequency of bottom vegetation for all 13 vegetation types in 2012, 2014 and 2017, determined from accumulated values from the Raunkjær circle

	Accumuluated value from Raunkjær			Frequency			Frequency progress from 2012 to 2017 (%)
Name	2012	2014	2017	2012	2014	2017	
Fraxinus excelsior	2	10	2	0,0029	0,01305	0,00249	-15,4
Molina caerulea	7	2	6	0,0103	0,00261	0,00748	-27,5
Scrophularia nodosa	1	1	2	0,0015	0,00131	0,00249	69,1
Deschampsia flexuosa	9	6	20	0,0133	0,00783	0,02494	87,9
Deschampsia cespitosa	105	95	95	0,1549	0,12402	0,11845	-23,5
Quercus robur	2	0	1	0,0029	0	0,00125	-57,7
Alnus glutinosa	1	0	4	0,0015	0	0,00499	238,2
Lathyrus pratensis	9	5	27	0,0133	0,00653	0,03367	153,6
Melica uniflora	2	3	22	0,0029	0,00392	0,02743	829,9
Luzula pilosa	3	0	0	0,0044	0	0	-100,0
Lonicera periclymenum	8	3	11	0,0118	0,00392	0,01372	16,2
Anthoxanthum odoratum	1	3	1	0,0015	0,00392	0,00125	-15,5
Holcus mollis	20	9	9	0,0295	0,01175	0,01122	-62,0
Rubus idaeus	27	39	33	0,0398	0,05091	0,04115	3,3
Humulus lupulus	3	0	0	0,0044	0	0	-100,0
Dactylis glomerata	19	37	34	0,0280	0,04830	0,04239	51,3
Hieracium sp.	2	1	0	0,0029	0,00131	0	-100,0
Cardamine sp.	1	8	0	0,0015	0,01044	0	-100,0
Juncus conglomeratus	9	1	13	0,0133	0,00131	0,01621	22,1
Melampyrum pratense	1	4	20	0,0015	0,00522	0,02494	1590,8
Juncus effusus	14	7	15	0,0206	0,00914	0,01870	-9,4
Dryopteris filix-mas	5	2	1	0,0074	0,00261	0,00125	.83,1
Dryopteris dilatata	2	9	25	0,0029	0,01175	0,03117	956,7
Dryopteris carthusiana	20	19	9	0,0295	0,02480	0,01122	-62,0
Dryopteris sp	2	0	0	0,0029	0	0	-100,0
Filipendula ulmaria	6	3	4	0,0088	0,00392	0,00499	-43,6
Mentha aquatic	3	4	4	0,0044	0,00522	0,00499	12,7
Geum rivale	14	12	7	0,0206	0,01567	0,00873	-57,7
Urtica dioica	7	0	1	0,0103	0	0,00125	-87,9
Equisetum arvense	8	0	3	0,0118	0	0,00374	-68,3
Equisetum fluviatile	1	0	8	0,0015	0	0,00998	576,3
Poa trivialis	38	18	11	0,0560	0,02350	0,01372	-75,5
Calamagrostis epigejos	7	10	15	0,0103	0,01305	0,01870	81,2
Oxalis Acetosella	105	114	84	0,1549	0,14883	0,10474	-32,4

Rumex obtusifolius	1	15	9	0,0015	0,01958	0,01122	660,8
Galium aparine	2	0	0	0,0029	0	0	-100,0
Carex remota	52	92	56	0,0767	0,12010	0,06983	-9,0
Carex pallescens	25	7	8	0,0369	0,00914	0,00998	-72,9
Carex vesicaria	19	16	19	0,0280	0,02089	0,02369	-15,5
Carex canescens	3	6	15	0,0044	0,00783	0,01870	322,7
Carex sylvatica	39	43	35	0,0575	0,05614	0,04364	-24,1
Festuca rubra	1	0	0	0,0015	0	0	-100,0
Rumex acetosa	2	1	4	0,0029	0,00131	0,00499	69,1
Viola riviniana	9	14	9	0,0133	0,01828	0,01122	-15,5
Bryophyta	61	113	99	0,0900	0,14752	0,12344	37,2
Lichens	0	0	0	0	0	0	0
Rumex sanguineus	0	1	2	0	0,00131	0,00249	-
Agrostis stolonifera	0	1	0	0	0,00131	0	-
Equisetum sylvaticum	0	1	0	0	0,00131	0	-
Viola sp	0	3	0	0	0,00392	0	-
Agrostis capillaris	0	2	24	0	0,00261	0,02993	-
Lysimachia nummularia	0	3	0	0	0,00392	0	-
Carex hirta	0	7	6	0	0,00914	0,00748	-
Festuca altissima	0	4	0	0	0,00522	0	-
Crepis paludosa	0	5	0	0	0,00653	0	-
Alnus incana	0	1	0	0	0,00131	0	-
Fragaria vesca	0	6	0	0	0,00783	0	-
Taraxacum sp	0	0	0	0	0	0	-
Potentilla erecta	0	0	13	0	0	0,01621	-
Eleocharis palustris	0	0	1	0	0	0,00125	-
Hippuris vulgaris	0	0	1	0	0	0,00125	-
Mycelis muralis	0	0	2	0	0	0,00249	-
Veronica officinalis	0	0	3	0	0	0,00374	-
Stellaria graminea	0	0	2	0	0	0,00249	-
Galium uliginosum	0	0	4	0	0	0,00499	-
Scutellaria galericulata	0	0	3	0	0	0,00374	-
Number of species, n	**45**	**46**	**49**				
Simpsons, D	**0,077**	**0,085**	**0,059**				
Shannon Wiener, H	**3,000**	**2,917**	**3,236**				
Pielou, J	**0,788**	**0,762**	**0,831**				

	Accumulated value from Raunkjær			Frequency			Frequency progress from 2012 to 2017 (%)
Name	**2012**	**2014**	**2017**	**2012**	**2014**	**2017**	
Acer pseudoplatanaus	12	3	0	0,0727	0,0197	0	-100,0
Deschampsia flexuosa	3	2	10	0,0182	0,0131	0,0649	257,1
Deschampsia cespitosa	6	7	3	0,0364	0,0460	0,0194	-46,4
Fagus sylvatica	12	1	0	0,0727	0,0065	0	-100,0
Quercus robur	1	0	1	0,0061	0	0,0064	7,1
Milium effusum	1	0	0	0,0061	0	0	-100,0
Oxalis Acetosella	110	106	93	0,6667	0,6973	0,6038	-9,4
Carex remota	3	3	2	0,0182	0,0197	0,0129	-28,6
Carex pilulifera	3	4	3	0,0182	0,0263	0,0194	7,1
Bryophyta	13	7	23	0,0788	0,0460	0,1493	89,6
Lichens	1	0	2	0,0061	0	0,0129	114,3
Dactylis glomerata	0	1	0	0	0,0065	0	-
Picea abies	0	10	9	0	0,0657	0,0034	-
Sorbus aucuparia	0	2	0	0	0,0131	0	-
Mycelis muralis	0	3	0	0	0,0197	0	-
Dryopteris carthusiana	0	2	8	0	0,0131	0,0026	-
Carex flacca	0	1	0	0	0,0065	0	-
Number of species, n	**11**	**14**	**10**				
Simpsons, D	**0,463**	**0,497**	**0,398**				
Shannon Wiener, H	**1,283**	**1,312**	**1,384**				
Pielou, J	**0,535**	**0,497**	**0,601**				

Table title (spanning header): **Beech forest – field 3.1-3.10**

Beech forest (ungrazed) – field 4.1-4.10						
	Accumulated value from Raunkjær			Frequency			Frequency progress from 2012 to 2017 (%)
Name	2012	2014	2017	2012	2014	2017	
Deschampsia flexuosa	7	1	8	0,0417	0,0047	0,0503	20,8
Deschampsia cespitosa	13	24	28	0,0774	0,1148	0,1761	127,6
Fagus sylvatica	12	11	11	0,0714	0,0526	0,0691	-3,1
Picea abies	3	1	0	0,0179	0,0047	0	-100
Rubus idaeus	3	2	5	0,0179	0,0095	0,0314	76,1
Oxalis Acetosella	95	104	57	0,5655	0,4976	0,3584	-36,6
Carex pilulifera	9	28	9	0,0536	0,1339	0,0566	5,6
Bryophyta	26	18	25	0,1548	0,0861	0,1572	1,6
Lichens	0	0	0	0	0	0	-
Larix decidua	0	4	4	0	0,0191	0,0251	-
Carex remota	0	11	1	0	0,0526	0,0062	-
Juncus effuses	0	2	0	0	0,0095	0	-
Dryopteris carthusiana	0	1	3	0	0,0047	0,0188	-
Luzula sylvatica	0	2	0	0	0,0095	0	-
Hieracium vulgatum	0	0	1	0	0	0,0062	-
Acer pseudoplatanaus	0	0	5	0	0	0,0314	-
Melica uniflora	0	0	2	0	0	0,0125	-
Number of species, n	8	13	12				
Simpsons, D	0,360	0,292	0,197				
Shannon Wiener, H	1,430	1,672	1,966				
Pielou, J	0,688	0,652	0,791				

Old oak forest – field 5.1-5.10						
	Accumulated value from Raunkjær			Frequency			Frequency progress from
Name	2012	2014	2017	2012	2014	2017	2012 to 2017 (%)
Anemone nemorosa	1	0	0	0,0012	0	0	-100,0
Rubus plicatus	2	0	0	0,0025	0	0	-100,0
Scrophularia nodosa	1	0	3	0,0012	0	0,0025	98,5
Deschampsia flexuosa	124	111	145	0,1565	0,1360	0,1211	--22,6
Deschampsia cespitosa	111	76	106	0,1401	0,0931	0,0885	-36,8
Fagus sylvatica	1	2	0	0,0012	0,0024	0	-100,0
Cardamene sp.	1	18	0	0,0012	0,0220	0	-100,0
Quercus robur	4	0	1	0,0050	0	0,0008	-83,5
Gymnocarpium dryopteris	2	0	3	0,0025	0	0,0025	-0,7
Lathyrus linifolius	1	0	1	0,0012	0	0,0008	-33,8
Stellaria graminea	4	0	5	0,0050	0	0,0041	-17,3
Stellaria holostea	10	12	15	0,0126	0,0147	0,0125	-0,7
Holcus lanatus	1	1	0	0,0012	0,0012	0	-100,0
Luzula pilosa	3	2	3	0,0037	0,0024	0,0025	-33,8
Stachys sylvatica	1	0	0	0,0012	0	0	-100,0
Lonicera periclymenum	65	81	116	0,0820	0,0992	0,0960	18,1
Holcus mollis	40	53	30	0,0505	0,0649	0,0250	-5+,4
Rubus idaeus	37	42	79	0,0467	0,0514	0,0659	41,3
Dactylis glomerata	49	32	42	0,0618	0,0391	0,0350	-43,3
Agrostis canina	2	0	0	0,0025	0	0	-100,0
Juncus conglomeratus	1	0	10	0,0012	0	0,0083	561,7
Melampyrum pratense	54	64	107	0,0681	0,0784	0,0893	31.1
Juncus effuses	5	11	12	0,0063	0,0134	0,0100	58,8
Dryopteris carthusiana	9	8	9	0,0113	0,0098	0,0075	-33,8
Milium effusum	6	3	73	0,0075	0,0036	0,0609	705,0
Filipendula ulmaria	1	3	6	0,0012	0,0036	0,0050	297,0
Prunus sp.	1	1	0	0,0012	0,0012	0	-100,0
Sorbus aucuparia	4	3	1	0,0050	0,0036	0,0008	-83,5
Calamagrostis epigejos	20	27	63	0,0252	0,0330	0,0526	108,4
Oxalis Acetosella	208	193	199	0,2626	0,2365	0,1662	-36,7
Carex sylvatica	5	0	2	0,0063	0	0,0016	-73,5
Rumex acetosa	1	2	3	0,0012	0,0024	0,0025	98,5
Viola riviniana	1	0	7	0,0012	0	0,0058	363,2
Viola reichenbachiana	3	0	0	0,0037	0	0	-100,0
Veronica sp	1	0	0	0,0012	0	0	-100,0
Bryophyta	12	40	70	0,0151	0,0490	0,0584	286,0
Lichens	0	1	1	0	0,0012	0,0008	-
Acer pseudoplatanaus	0	4	0	0	0,0049	0	-
Carex nigra	0	3	9	0	0,0036	0,0075	-
Festuca rubra	0	4	22	0	0,0049	0,0183	-
Carex pilulifera	0	2	7	0	0,0024	0,0058	-
Anthoxanthum odoratum	0	2	5	0	0,0024	0,0041	-
Fraxinus excelsior	0	1	0	0	0,0012	0	-

Carex remota	0	1	9	0	0,0012	0,0075	-
Viola sp	0	5	0	0	0,0061	0	-
Poa trivialis	0	4	5	0	0,0049	0,0041	-
Scutellaria galericulata	0	4	0	0	0,0049	0	-
Urtica dioica	0	0	0	0	0	0	-
Melica uniflora	0	0	1	0	0	0,0008	-
Veronica officinalis	0	0	5	0	0	0,0041	-
Potentilla erecta	0	0	14	0	0	0,0116	-
Lathyrus pratensis	0	0	6	0	0	0,0050	-
Agrostis capillaris	0	0	1	0	0	0,0008	-
Cardamine hirsuta	0	0	1	0	0	0,0008	-
Number of species, n	**36**	**33**	**38**				
Simpsons, D	**0,134**	**0,122**	**0,084**				
Shannon Wiener, H	**2,411**	**2,562**	**2,785**				
Pielou, J	**0,673**	**0,733**	**0,766**				

Middle aged oak forest – field 6.1-6.10						
	Accumulated value from Raunkjær			Frequency		Frequency progress from 2012 to 2017 (%)
Name	2012	2014	2017	2012	2014	2017	
Acer pseudoplatanaus	10	12	13	0,0251	0,0185	0,0163	-34,8
Anemone nemorosa	1	3	0	0,0013	0,0046	0	-100,0
Fraxinus excelsior	1	6	0	0,0013	0,0092	0	-100,0
Rubus plicatus	5	0	5	0,0066	0	0,0062	-4,8
Deschampsia flexuosa	148	96	76	0,1955	0,1483	0,0955	-51,1
Deschampsia cespitosa	68	79	84	0,0898	0,1221	0,1056	17,6
Fagus sylvatica	4	2	5	0,0053	0,0030	0,0062	19,0
Quercus robur	3	1	3	0,0040	0,0015	0,0037	-4,0
Luzula sylvatica	1	0	0	0,0013	0	0	-100,0
Lonicera periclymenum	100	75	97	0,1321	0,1159	0,1220	-7,6
Rubus idaeus	60	34	53	0,0793	0,0525	0,0666	-15,8
Dactylis glomerata	7	15	24	0,0092	0,0231	0,0301	226,5
Juncus conglomeratus	1	2	2	0,0013	0,0030	0,0025	90,4
Melampyrum pratense	16	1	7	0,0211	0,0015	0,0088	-58,3
Polygonatum multiflorum	2	0	0	0,0026	0	0	-100,0
Juncus effuses	1	2	11	0,0013	0,0030	0,0138	947,4
Dryopteris filix-mas	6	1	0	0,0079	0,0015	0	-100,0
Dryopteris carthusiana	6	4	29	0,0079	0,0061	0,0364	360,2
Milium effusum	14	3	24	0,0185	0,0046	0,0301	63,2
Equisetum arvense	1	0	0	0,0013	0	0	-100,0
Equisetum sylvaticum	3	3	3	0,0040	0,0046	0,0037	-4,8
Sorbus aucuparia	6	5	2	0,0079	0,0077	0,0025	-68,3
Calamagrostis epigejos	70	48	73	0,0925	0,0741	0,0918	-0,7
Oxalis Acetosella	163	167	151	0,2153	0,2581	0,1899	-11,8
Carex remota	5	0	9	0,0066	0	0,0113	71,4
Carex hirta	10	6	2	0,0132	0,0092	0,0025	-81,0
Carex pilulifera	3	7	11	0,0040	0,0108	0,0138	249,1
Viola riviniana	9	0	18	0,0119	0	0,0226	90,4
Viola reichenbachiana	5	3	0	0,0066	0,0046	0	-100,0
Bryophyta	15	39	57	0,0198	0,0602	0,0716	261,8
Lichens	4	0	0	0,0053	0	0	-100,0
Stellaria holostea	0	12	6	0	0,0185	0,0075	-
Viola canina	0	5	0	0	0,0077	0	-
Carex flacca	0	2	3	0	0,0030	0,0037	-
Fragaria vesca	0	1	1	0	0,0015	0,0012	-
Viola sp.	0	1	0	0	0,0015	0	-
Veronica chamaedrys	0	1	2	0	0,0015	0,0025	-
Rumex acetosa	0	1	0	0	0,0015	0	-
Agrostis capillaris	0	1	1	0	0,0015	0,0012	-
Lysimachia vulgaris	0	0	0	0	0	0	-
Galium palustre	0	1	1	0	0,0015	0,0012	-

Lathyrus linifolius	0	1	3	0	0,0015	0,0037	-
Alopecurus pratensis	0	2	0	0	0,0030	0	-
Scutellaria galericulata	0	1	5	0	0,0015	0,0062	-
Carex nigra	0	1	0	0	0,0015	0	-
Stellaria graminea	0	1	1	0	0,0015	0,0012	-
Holcus mollis	0	1	0	0	0,0015	0	-
cardamene sp.	0	1	0	0	0,0015	0	-
Potentilla erecta	0	0	5	0	0	0,0062	-
Digitalis purpurea	0	0	1	0	0	0,0012	-
Calamagrostis canescens	0	0	3	0	0	0,0037	-
Epipactis helleborine	0	0	2	0	0	0,0025	-
Galium valdepilosum	0	0	2	0	0	0,0025	-
Number of species, n	**31**	**40**	**37**				
Simpsons, D	**0,127**	**0,130**	**0,094**				
Shannon Wiener, H	**2,437**	**2,472**	**2,720**				
Pielou, J	**0,710**	**0,670**	**0,753**				

Young oak forest – field 7.1-7.10						
	Accumulated value from Raunkjær			Frequency		Frequency progress from 2012 to 2017 (%)
Name	2012	2014	2017	2012	2014	2017	
Anemone nemorosa	3	0	0	0,0056	0	0	-100,0
Deschampsia flexuosa	90	72	87	0,1695	0,1434	0,0768	-54,7
Deschampsia cespitosa	70	60	90	0,1318	0,1195	0,1032	-21,7
Fagus sylvatica	1	0	0	0,0019	0	0	-100,0
Lathyrus pratensis	2	2	14	0,0038	0,0039	0,0160	326,3
Stellaria graminea	4	0	10	0,0075	0	0,0114	52,2
Holcus lanatus	14	14	19	0,0264	0,0278	0,0217	-17,4
Luzula pilosa	2	0	3	0,0038	0	0,0034	-8,6
Luzula campestris	1	0	0	0,0019	0	0	-100,0
Stellaria media	17	1	1	0,0320	0,0019	0,0011	-96,4
Stellaria neglecta	3	0	0	0,0056	0	0	-100,0
Lonicera periclymenum	5	3	30	0,0094	0,0059	0,0344	265,4
Picea abies	2	0	0	0,0038	0	0	-100,0
Rubus idaeus	31	24	48	0,0584	0,0478	0,0550	-5,7
Hieracium sp	1	0	0	0,0019	0	0	-100,0
cardamene sp.	7	6	0	0,0132	0,0119	0	-100,0
Juncus conglomeratus	4	1	8	0,0075	0,0019	0,0091	21,8
Juncus effusus	14	13	22	0,0264	0,0258	0,0252	-4,3
Dryopteris carthusiana	1	12	17	0,0019	0,0239	0,0194	935,2
Milium effusum	16	1	16	0,0301	0,0019	0,0183	-39,1
Mentha sp	2	0	0	0,0038	0	0	-100,0
Taraxacum sp	1	0	1	0,0019	0	0,0011	39,1
Urtica dioica	8	16	14	0,0151	0,0318	0,0160	6,6
Equisetum arvense	10	9	15	0,0188	0,0179	0,0172	-8,6
Equisetum fluviatile	5	3	0	0,0094	0,0059	0	-100,0
Equisetum sylvaticum	10	2	2	0,0188	0,0039	0,0022	-87,8
Hypericum perforatum	1	0	0	0,0019	0	0	-100,0
Phleum pratense	4	1	2	0,0075	0,0019	0,0022	-69,5
Sorbus aucuparia	11	12	8	0,0207	0,0239	0,0091	-55,7
Calamagrostis epigejos	41	49	85	0,0772	0,0976	0,0974	26,2
Juncus articulatus	1	0	0	0,0019	0	0	-100,0
Oxalis Acetosella	56	53	124	0,1055	0,1055	0,1422	34,8
Galium palustre	2	1	10	0,0038	0,0019	0,0114	204,5
Galium uliginosum	2	2	3	0,0038	0,0039	0,0034	-8,6
Carex flacca	4	4	23	0,0075	0,0079	0,0263	250,1
Carex pilulifera	7	9	15	0,0132	0,0179	0,0172	30,5
Carex sylvatica	1	0	1	0,0019	0	0,0011	-39,1
Cirsium palustre	9	6	1	0,0169	0,0119	0,0011	-93,2
Potentilla erecta	5	2	17	0,0094	0,0039	0,0194	107,0
Vicia sp	1	0	2	0,0019	0	0,0022	21,8

Vicia cracca	1	1	5	0,0019	0,0019	0,0057	204,5
Viola riviniana	9	9	30	0,0169	0,0179	0,0344	103,0
Veronica chamaedrys	4	2	2	0,0075	0,0039	0,0022	-69,6
Bryophyta	48	59	119	0,0904	0,1175	0,1364	51,0
Lichens	0	1	1	0	0,0019	0,0011	-
Viola sp	0	6	0	0	0,0119	0	-
Epilobium sp	0	3	0	0	0,0059	0	-
Holcus mollis	0	7	2	0	0,0139	0,0022	-
Larix decidua	0	3	0	0	0,0139	0	-
Poa trivialis	0	5	4	0	0,0059	0,0045	-
Viola sp	0	1	0	0	0,0099	0	-
Polygonatum multiflorum	0	3	0	0	0,0019	0	-
Carex remota	0	1	0	0	0,0059	0	-
Betula pendula	0	2	0	0	0,0019	0	-
Galium boreale	0	2	0	0	0,0039	0	-
Quercus robur	0	8	2	0	0,0159	0,0022	-
Taxus baccata	0	1	0		0,0019	0	-
Alopecurus pratensis	0	3	7	0	0,0059	0,0080	-
Elymus repens	0	1	0	0	0,0019	0	-
Agrostis capillaris	0	6	1	0	0,0119	0,0011	-
Calamagrostis epigejos	0	0	2	0	0	0,0022	-
Athyrium Filix-femina	0	0	2	0	0	0,0022	-
Stachys sylvatica	0	0	1	0	0	0,0011	-
Epilobium palustre	0	0	6	0	0	0,0068	-
Melica uniflora	0	0	4	0	0	0,0045	-
Mentha aquatica	0	0	1	0	0	0,0011	-
Carex pallescens	0	0	3	0	0	0,0034	-
Scutellaria galericulata	0	0	2	0	0	0,0022	-
Epipactis helleborine	0	0	1	0	0	0,0011	-
Dactylis glomerata	0	0	3	0	0	0,0034	-
Melampyrum pratense	0	0	2	0	0	0,0022	-
Carex nigra	0	0	2	0	0	0,0022	-
Luzula multiflora	0	0	2	0	0	0,0022	-
Number of species, n	**44**	**44**	**46**				
Simpsons, D	**0,081**	**0,077**	**0,075**				
Shannon Wiener, H	**2,957**	**2,988**	**3,026**				
Pielou, J	**0,781**	**0,790**	**0,790**				

Old oak forest (ungrazed) – field 8.1-8.10						
	Accumulated value from Raunkjær			Frequency		Frequency progress from 2012 to 2017 (%)
Name	2012	2014	2017	2012	2014	2017	
Acer pseudoplatanaus	3	1	3	0,0048	0,0015	0,0036	-24,7
Deschampsia flexuosa	125	118	134	0,2029	0,1823	0,1638	-19,3
Deschampsia cespitosa	57	63	59	0,0925	0,0973	0,0721	-22,1
Quercus robur	7	4	4	0,0113	0,0061	0,0048	-57,0
Luzula pilosa	2	0	0	0,0032	0	0	-100,0
Lonicera periclymenum	44	68	132	0,0714	0,1051	0,1613	125,9
Picea abies	1	2	0	0,0016	0,0030	0	-100,0
Holcus mollis	8	15	2	0,0129	0,0231	0,0024	-81,2
Rubus idaeus	31	36	41	0,0503	0,0556	0,0501	-0,4
Dactylis glomerata	45	34	20	0,0730	0,0525	0,0244	-66,5
Melampyrum pratense	25	19	73	0,0405	0,0293	0,0892	119,9
Juncus effusus	2	0	0	0,0032	0	0	-100,0
Dryopteris carthusiana	4	7	12	0,0064	0,0108	0,0146	125,9
Poa nemoralis	14	0	0	0,0227	0	0	-100,0
Sorbus aucuparia	1	0	1	0,0016	0	0,0012	-24,7
Calamagrostis epigejos	60	40	115	0,0974	0,0618	0,1405	44,3
Oxalis Acetosella	154	166	174	0,2500	0,2565	0,2127	-14,9
Carex pilulifera	2	11	0	0,0032	0,0170	0	-100,0
Hedera helix	2	8	1	0,0032	0,0123	0,0012	-62.3
Viola sp	1	0	2	0,0016	0	0,0024	50,6
Bryophyta	28	19	19	0,0454	0,0293	0,0232	-48,9
Lichens	0	0	0	0	0	0	-
Stellaria holostea	0	29	3	0	0,0448	0,0036	-
Fagus sylvatica	0	1	16	0	0,0015	0,0195	-
Anthoxanthum odoratum	0	4	0	0	0,0061	0	-
Stellaria sp	0	1	0	0	0,0015	0	-
Potentilla erecta	0	1	1	0	0,0015	0,0012	-
Galium valdepilosum	0	0	3	0	0	0,0036	-
Juncus conglomeratus	0	0	2	0	0	0,0024	-
Hypericum maculatum	0	0	1	0	0	0,0012	-
Number of species, n	21	21	22				
Simpsons, D	0,139	0,134	0,135				
Shannon Wiener, H	2,275	2,330	2,232				
Pielou, J	0,747	0,765	0,722				

Old Norway spruce forest (ungrazed) – field 10.1-10.10

Name	Accumulated value from Raunkjær			Frequency			Frequency progress from 2012 to 2017 (%)
	2012	2014	2017	2012	2014	2017	
Deschampsia flexuosa	18	30	13	0,0401	0,0986	0,0646	61,0
Lonicera periclymenum	1	0	0	0,0022	0	0	-100,0
Picea abies	210	15	5	0,4687	0,0493	0,0248	-94,7
Dryopteris sp	3	0	0	0,0067	0	0	-100,0
Oxalis Acetosella	2	0	0	0,0044	0	0	-100,0
Bryophyta	214	254	178	0,4776	0,8355	0,8855	85,4
Lichens	0	1	1	0	0,0032	0,0049	-
Sorbus aucuparia	0	3	4	0	0,0098	0,0199	-
Molinia caerulea	0	1	0	0	0,0032	0	-
Number of species, n	**6**	**5**	**5**				
Simpsons, D	**0,449**	**0,710**	**0,789**				
Shannon Wiener, H	**0,908**	**0,610**	**0,480**				
Pielou, J	**0,507**	**0,379**	**0,298**				

Old Norway spruce forest – field 11.1-11.10

Name	Accumulated value from Raunkjær			Frequency			Frequency progress from 2012 to 2017 (%)
	2012	2014	2017	2012	2014	2017	
Deschampsia flexuosa	52	66	71	0,1721	0,1434	0,1694	-1,6
Deschampsia cespitosa	1	2	1	0,0033	0,0043	0,0023	-28,0
Lathyrus linifolius	1	0	0	0,0033	0	0	-100,0
Picea abies	12	97	58	0,0397	0,2108	0,1384	248,4
cardamene sp.	1	0	0	0,0033	0	0	-100,0
Dryopteris filix-mas	2	0	3	0,0066	0	0,0071	8,1
Calamagrostis epigejos	1	0	0	0,0033	0	0	-100,0
Oxalis Acetosella	48	71	63	0,1589	0,1543	0,1503	-5,4
Carex pilulifera	3	10	15	0,0099	0,02173	0,0357	260,4
Veronica officinalis	1	0	0	0,0033	0	0	-100,0
Bryophyta	180	199	208	0,5960	0,4326	0,4964	-16,7
Lichens	0	3	0	0	0,0065	0	-
Galium palustre	0	3	0	0	0,0065	0	-
Rubus idaeus	0	3	0	0	0,0065	0	-
Sorbus aucuparia	0	4	0	0	0,0086	0	-
Mycelis muralis	0	1	0	0	0,0021	0	-
Betula pendula	0	1	0	0	0,0021	0	-
Number of species, n	**11**	**11**	**7**				
Simpsons, D	**0,411**	**0,276**	**0,318**				
Shannon Wiener, H	**1,205**	**1,530**	**1,376**				
Pielou, J	**0,503**	**0,638**	**0,707**				

Young Norway spruce forest – field 12.1-12.10						
	Accumulated value from Raunkjær			Frequency			Frequency progression from 2012 to 2017 (%)
Name	2012	2014	2017	2012	2014	2017	
Deschampsia flexuosa	18	14	24	0,1208	0,0985	0,1481	22,6
Deschampsia cespitosa	7	2	2	0,0469	0,0140	0,0123	-73,7
Picea abies	7	6	0	0,0469	0,0422	0	-100,0
Oxalis Acetosella	3	3	5	0,0201	0,0211	0,0308	53,3
Carex pilulifera	11	6	9	0,0738	0,0422	0,0555	-24,7
Viola canina	3	2	2	0,0201	0,0140	0,0123	-38,7
Bryophyta	100	97	116	0,6711	0,6830	0,7160	6,7
Lichens	0	1	0	0	0,0070	0	-
Dryopteris carthusiana	0	1	2	0	0,0070	0,0123	-
Juncus conglomeratus	0	1	0	0	0,0070	0	-
Molinia caerulea	0	1	0	0	0,0070	0	-
Sorbus aucuparia	0	8	0	0	0,0563	0	-
Luzula pilosa	0	0	2	0	0	0,0123	-
Number of species, n	7	11	8				
Simpsons, D	0,475	0,484	0,539				
Shannon Wiener, H	1,159	1,259	1,007				
Pielou, J	0,596	0,525	0,484				

	Rejuvenated area – field 13.1-13.10						
	Accumulated value from Raunkjær			Frequency			Frequency progress from 2012 to 2017 (%)
	2012	2014	2017	2012	2014	2017	
Senecio sylvaticus	3	38	0	0,0270	0,0725	0	-100,0
Deschampsia flexuosa	7	116	205	0,0630	0,2213	0,2475	292.6
Deschampsia cespitosa	2	5	32	0,0180	0,0095	0,0386	114,5
gymnocarpium dryopteris	1	0	1	0,0090	0	0,0012	-86,6
Luzula multiflora	1	0	1	0,0090	0	0,0012	-86,6
Picea abies	4	22	12	0,0360	0,0419	0,0144	-59,8
Calluna vulgaris	1	3	10	0,0090	0,0057	0,0012	34,1
Rubus idaeus	4	60	85	0,0360	0,1145	01026	184,9
Juncus conglomeratus	2	5	9	0,0180	0,0095	0,0108	-39,7
Dryopteris carthusiana	1	2	4	0,0090	0,0038	0,0048	-46,4
Alopecurus pratensis	2	0	0	0,0180	0	0	-100,0
Rumex acetosella	8	32	20	0,0720	0,0610	0,0241	-66,5
Calamagrostis epigejos	1	27	92	0,0090	0,0515	0,1111	1133,3
Oxalis Acetosella	11	13	36	0,0990	0,0248	0,0434	-56,1
Carex pilulifera	40	66	56	0,3603	0,1259	0,0676	-81,2
Rumex acetosa	4	4	1	0,0360	0,0076	0,0012	-96,6
Bryophyta	19	79	111	0,1711	0,1507	0,1340	-21,7
Lichens	0	0	1	0	0	0,0012	-
Betula pendula	0	16	18	0	0,0305	0,0217	-
Juncus effusus	0	19	14	0	0,0362	0,0169	-
Holcus lanatus	0	0	36	0	0	0,0434	-
Agrostis sp	0	2	0	0	0,0038	0	-
Stellaria neglecta	0	2	0	0	0,0038	0	-
Carex ovalis	0	5	6	0	0,0095	0,0072	-
Carex distans	0	1	0	0	0,0019	0	-
Carex pairei	0	2	0	0	0,0038	0	-
Cirsium palustre	0	2	3	0	0,0038	0,0036	-
Poa annua	0	1	0	0	0,0019	0	-
Chamaenerion angustifolium	0	1	2	0	0,0019	0,0024	-
salix sp	0	1	0	0	0,0019	0	-
Luzula campestris	0	0	1	0	0	0,0012	-
Betula pubescens	0	0	5	0	0	0,0060	-
Fagus sylvatica	0	0	1	0	0	0,0012	-
Agrostis capillaris	0	0	32	0	0	0,0386	-
Stellaria graminea	0	0	3	0	0	0,0036	-
Rumex obtusifolius	0	0	1	0	0	0,0012	-
Hypochaeris radicata	0	0	4	0	0	0,0048	-
Sorbus aucuparia	0	0	4	0	0	0,0048	-

Carex echinata	0	0	3	0	0	0,0036	-
Veronica chamaedrys	0	0	2	0	0	0,0024	-
Festuca rubra	0	0	2	0	0	0,0024	-
Carex nigra	0	0	3	0	0	0,0036	-
Carex panicea	0	0	3	0	0	0,0036	-
Lysimachia nummularia	0	0	1	0	0	0,0012	-
Poa pratensis	0	0	1	0	0	0,0012	-
Carex hirta	0	0	3	0	0	0,0036	-
Carex remota	0	0	1	0	0	0,0012	-
Rubus plicatus	0	0	1	0	0	0,0012	-
Dactylis glomerata	0	0	1	0	0	0,0012	-
Quercus robur	0	0	1	0	0	0,0012	-
Number of species, n	**17**	**25**	**42**				
Simpsons, D	**0,184**	**0,117**	**0,115**				
Shannon Wiener, H	**2,148**	**2,447**	**2,620**				
Pielou, J	**0,758**	**0,760**	**0,701**				

Wet meadow – field 16.1-16.10						
	Accumulated value from Raunkjær			Frequency			Frequency progress from
Name	2012	2014	2017	2012	2014	2017	2012 to 2017 (%)
Lemna trisulca	1	0	0	0,0007	0	0	-100,0
Lemna minor	18	0	0	0,0126	0	0	-100,0
Prunella vulgaris	2	16	28	0,0014	0,0093	0,0152	984,6
Menyanthes trifoliata	22	13	12	0,0154	0,0076	0,0065	-57,7
Deschampsia cespitosa	5	13	9	0,0035	0,0076	0,0049	39,5
Lathyrus pratensis	19	29	33	0,0133	0,0169	0,0180	34,6
Holcus lanatus	8	25	29	0,0056	0,0145	0,0158	180,8
Myosotis scorpioides	6	0	0	0,0042	0	0	-100,0
Myosotis laxa	4	3	16	0,0028	0,0017	0,0087	209,9
Lysimachia vulgaris	1	2	24	0,0007	0,0012	0,0131	1759,4
Lysimachia nummularia	49	71	85	0,0344	0,0413	0,0462	34,4
Luzula multiflora	2	0	0	0,0014	0	0	-100,0
Picea abies	1	0	0	0,0007	0	0	-100,0
Anthoxanthum odoratum	19	16	20	0,0133	0,0093	0,0109	-18,4
Dactylis glomerata	1	1	0	0,0007	0,0006	0	-100,0
Agrostis capillaris	3	2	1	0,0021	0,0012	0,0005	-74,2
Agrostis stolonifera	40	87	76	0,0281	0,0506	0,0413	47,2
Cerastium fontanum	1	0	1	0,0007	0	0,0005	-22,5
Caltha palustris	1	8	20	0,0007	0,0047	0,0109	1449,5
Cynocurus cristatus	8	8	26	0,0056	0,0047	0,0141	151,8
Cardamine pratensis	32	8	4	0,0225	0,0047	0,0022	-90,3
Trifolium repens	19	118	12	0,0133	0,0105	0,0065	-51,1
Juncus conglomeratus	2	2	16	0,0014	0,0012	0,0087	519,8
Eriophorum angustifolium	18	15	26	0,0126	0,0087	0,0141	11,9
Juncus effusus	34	36	33	0,0239	0,0209	0,0180	-24,8
Larix decidua	7	0	0	0,0049	0	0	-100,0
Mentha sp.	130	127	0	0,0913	0,0738	0	-100,0
Geum rivale	2	6	5	0,0014	0,0035	0,0027	93,7
Equisetum arvense	11	23	29	0,0077	0,0134	0,0158	104,3
Equisetum fluviatile	190	176	146	0,1334	0,1023	0,0794	-40,5
Equisetum palustre	106	119	124	0,0744	0,0692	0,0675	-9,4
Persicaria amphibia	33	41	34	0,0232	0,0238	0,0185	-20,2
Lolium perrene	6	0	0	0,0042	0	0	-100,0
Ranunculus lingua	7	13	25	0,0049	0,0076	0,0136	176,7
Ranunculus repens	76	86	68	0,0534	0,0500	0,0370	-30,7
Poa trivialis	3	38	38	0,0021	0,0221	0,0207	881,4
Poa pratensis	42	23	36	0,0295	0,0134	0,0196	-33,6
Phleum pratense	17	11	18	0,0119	0,0064	0,0098	-18,0

Alopecurus pratensis	6	8	24	0,0042	0,0047	0,0131	209,9
Juncus articulatus	24	32	39	0,0169	0,0186	0,0212	25,9
Scutellaria galericulata	1	2	0	0,0007	0,0012	0	-100,0
Galium palustre	33	32	24	0,0232	0,0186	0,0131	-43,7
Galium uliginosum	6	14	30	0,0042	0,0081	0,0163	287,4
Carex disticha	20	42	34	0,0140	0,0244	0,0185	31,7
Carex nigra	60	106	82	0,0421	0,0616	0,0446	5,9
Carex pallescens	4	7	14	0,0028	0,0041	0,0076	171,2
Carex vesicaria	55	44	60	0,0386	0,0256	0,0326	-15,5
Carex elongata	4	0	0	0,0028	0	0	-100,0
Carex ovalis	1	4	26	0,0007	0,0023	0,0141	1914,4
Carex hirta	17	9	16	0,0119	0,0052	0,0087	-27,1
Carex rostrata	59	46	43	0,0414	0,0267	0,0234	-43,5
Carex sylvatica	5	0	0	0,0035	0	0	-100,0
Eleocharis palustris	31	41	35	0,0218	0,0238	0,0190	-12,5
Festuca pratensis	8	40	25	0,0056	0,0233	0,0136	142,1
Festuca rubra	4	33	29	0,0028	0,0192	0,0158	461,7
Lycopus europaeus	25	26	29	0,0176	0,0151	0,0158	-10,1
Rumex acetosa	12	18	19	0,0084	0,0105	0,0103	22,7
Glyceria fluitans	16	19	12	0,0112	0,0110	0,0065	.41,9
Cirsium palustre	6	17	15	0,0042	0,0099	0,0082	93,7
Lychnis flos-cuculi	10	8	10	0,0070	0,0047	0,0054	-22,5
Vicia cracca	2	6	10	0,0014	0,0035	0,0054	287,4
Veronica serpyllfolia	1	0	0	0,0007	0	0	-100,0
Veronica scutellata	39	9	21	0,0274	0,0052	0,0114	-58,3
Veronica chamaedrys	3	5	1	0,0021	0,0029	0,0005	-74,2
Bryophyta	26	34	12	0,0183	0,0198	0,0065	-64,2
Lichen	0	0	0	0	0	0	-
Trifolium pratense	0	1	9	0	0,0006	0,0049	-
Ranunculus acris	0	15	26	0	0,0087	0,0141	-
Rumex crispus	0	2	0	0	0,0012	0	-
Carex panicea	0	5	0	0	0,0029	0	-
Carex flacca	0	1	0	0	0,0006	0	-
Plantago lanceolata	0	4	0	0	0,0023	0	-
Taraxacum sp	0	12	3	0	0,0070	0,0016	-
Scorzoneroides autumnalis	0	2	0	0	0,0012	0	-
Vicia sativa ssp. Nigra	0	2	6	0	0,0012	0,0033	-
Potentilla anserina	0	1	3	0	0,0006	0,0016	-
Epilobium parviflorum	0	21	46	0	0,0122	0,0250	-
Stellaria media	0	3	0	0	0,0017	0	-
Filipendula ulmaria	0	2	3	0	0,0012	0,0016	-
Epilobium montanum	0	8	1	0	0,0047	0,0005	-
Typha latifolia	0	3	0	0	0,0017	0	-

Mentha aquatica	0	0	123	0	0	0,0669	-
Stellaria graminea	0	0	2	0	0	0,0011	-
Alopecurus geniculatus	0	0	1	0	0	0,0005	-
Veronica officinalis	0	0	2	0	0	0,0011	-
Carex pseudocyperus	0	0	3	0	0	0,0016	-
Ranunculus sceleratus	0	0	6	0	0	0,0033	-
Number of species, n	**65**	**69**	**66**				
Simpsons, D	**0,048**	**0,039**	**0,032**				
Shannon Wiener, H	**3,459**	**3,610**	**3,743**				
Pielou, J	**0,829**	**0,853**	**0,893**				

Bush grass dominated area – field 20.1-20.10							
	Accumulated value from Raunkjær			**Frequency**			**Frequency progression**
Name	**2012**	**2014**	**2017**	**2012**	**2014**	**2017**	**from 2012 to 2017 (%)**
Deschampsia flexuosa	91	87	91	0,2351	0,1847	0,1657	-29,5
Deschampsia cespitosa	1	11	13	0,0025	0,0233	0,0236	816,4
Holcus lanatus	1	2	1	0,0025	0,0042	0,0018	-29,5
Calluna vulgaris	2	0	0	0,0051	0	0	-100,0
Rubus idaeus	29	21	54	0,0749	0,0445	0,0983	31,3
Juncus conglomeratus	5	1	10	0,0129	0,0021	0,0182	41,0
Juncus effusus	6	6	14	0,0155	0,0127	0,0255	64,5
Dryopteris carthusiana	1	1	0	0,0025	0,0021	0	-100,0
Urtica dioica	2	0	1	0,0051	0	0,0018	-64,8
Equisetum arvense	8	0	0	0,0206	0	0	-100,0
Calamagrostis epigejos	231	245	256	0,5969	0,5201	0,4663	-21,9
Carex pilulifera	8	0	2	0,0206	0	0,0036	-82,4
Bryophyta	2	66	80	0,0051	0,1401	0,1457	2719,7
Lichens	0	0	0	0	0	0	-
Agrostis sp	0	1	0	0	0,0021	0	-
Poa pratensis	0	1	0	0	0,0021	0	-
Digitalis purpurea	0	7	4	0	0,0148	0,0072	-
Senecio sp	0	1	0	0	0,0021	0	-
Betula pendula	0	1	0	0	0,0021	0	-
Epilobium sp	0	1	0	0	0,0021	0	-
Equisetum sylvaticum	0	13	10	0	0,0276	0,0182	-
Chamaenerion angustifolium	0	6	6	0	0,0127	0,0109	-
Epilobium parviflorum	0	0	6	0	0	0,0109	-
Oxalis Acetosella		0	1	0	0	0,0018	-
Number of species, n	**13**	**16**	**15**				
Simpsons, D	**0,418**	**0,328**	**0,278**				
Shannon Wiener, H	**1,251**	**1,541**	**1,680**				
Pielou, J	**0,488**	**0,556**	**0,620**				

Appendix 5. Summary of species richness and Ellenberg indicator value results

The observed results after five years of grazing practice by European bison in Almindingen.

SI = Significant Increase
SD = Significant decrease
NSC = No Significant Change

Table 13. Summary of observed result for all thirteen habitat types, regarding species richness and Ellenberg indicator values (EIV). SI=Significant increase. SD= Significant decrease. NSC= No Significant Change

Habitat type	Species richness	EIV-L	EIV-N
Alder swamp	SI	NSC	NSC
Beech forest	NSC	NSC	NSC
Beech forest (ungrazed)	NSC	NSC	NSC
Old oak forest	SI	SI	NSC
Middle-aged oak forest	NSC	SI	NSC
Young oak forest	SI	NSC	NSC
Old oak forest (ungrazed)	NSC	NSC	NSC
Old Norway spruce forest (ungrazed)	NSC	SI	NSC
Old Norway spruce forest	NSC	SD	NSC
Young Norway spruce forest	NSC	NSC	NSC
Rejuvenated area	SI	SI	NSC
Wet meadow	SI	SI	NSC
Bush grass dominated area	NSC	NSC	SI