UNIVERSITY OF COPENHAGEN
FACULTY OF SCIENCE

Master's Thesis

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Effects of management practices on the habitat quality of Danish dry heathlands

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Handed in: 21st March 2016

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Faculty:	Faculty of Science
Name of department:	Department of Geosciences and Natural Resource Management
Program:	MSc Nature Management (Landscape, Biodiversity and Planning)
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Title:	Effects of management practices on the habitat quality of Danish dry heathlands
Cover picture:	Randbøl Hede in summer 2015. Area used as control.
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Submitted:	21st March 2016

Table of contents

List of tables

List of figures

Abbreviation

CBD: Convention on Biological Diversity

ELC: European Landscape Convention

ES: Ecosystem Services

UNESCO: United Nations Educational, Scientific and Cultural Organization

MA: Millennium Ecosystem Assessment

TEK: Traditional Ecological Knowledge

Abstract

Heathlands have traditionally been used for agricultural and farming purposes as grazing, in combination with other secondary practices as cutting or burning. A bad habitat quality status of heathlands has been recognized by the European Commission. Therefore, member states of EU should aim at improving their habitat quality. Nowadays, management is conducted to reduce the effects of atmospheric Nitrogen deposition and regenerate dwarf shrubs species. However, there is no aim at creating a mosaic of habitats for fauna. In order to determine the effects of management on the habitat quality of heaths, data on soil structure and vegetation community were collected from three dry heathlands located in Jutland, Denmark. Moreover, habitat quality assessments were carried out. We compared areas that have been under cutting or burning regimes with others in which no management has been applied for a long period. Our results revealed that the application of cutting and burning in heaths enhances functional diversity and improves their habitat quality status. We concluded that there is potential to improve the methodology for the assessment of habitat quality of dry heathlands in Denmark. However, substantial data remains to be filled before a fully integrated and complete habitat quality assessment can be carried out. We recommend that future actions taken by decision-makers should address such major challenges as Ecosystem Services (ES) in combination with biodiversity. We suggest management aiming to create a mosaic of habitats for fauna and flora.

Key-words: habitat quality, heathland, fauna, functional diversity, management practices

Preface

When I first landed in Denmark in summer 2013, I had no idea about what this master would bring me. After my Erasmus in Helsinki in 2012, I was motivated to gain experience and knowledge in nature conservation and sustainable management of European landscapes from a Scandinavian perspective. Now and still, I am not totally aware of the consequences this might have in my professional career and at a personal level.

I started the master thesis with the celebration of Sankt Hans in 2015. Now, almost a year later, birds are back, plants are blooming and the air smells like fresh grass. Nature is out there, waiting to be admired, loved and enjoyed.

If I am here at this point, it is because I have been shaped of the influence of admirable human beings, who selflessly dedicated their life to protect nature and promote environmental education in Spain. Somehow, this is a tribute to those who contributed along the last few years in raising my appreciation for nature, as for instance, Carlos de Aguilera Salvetti.

I feel lucky and fortunate to reach this stage of studying in a Danish context, a totally unfamiliar environment, far from family and friends. This process has been not a mere learning process to varied facets, but also a major challenge and a step towards my dream. For instance, the field work with researchers and students provided the opportunity to determine danish flora of heathlands. Further, the working environment at the Department contributed creating such as amazing atmosphere to work on the thesis and get to know different projects. This experience tough me to turn insurmountable problems into exciting challenges. At the end, what matters is the attitude in life, whether positive or negative, is what makes the difference.

There is a saying, written by a Spanish poet, that I like a lot. It is as follows:

"Caminante, no hay camino, se hace camino al andar." Antonio Machado

This saying, means that there is no a single right path in life, but it is created step by step.

Acknowledgments

I would like to acknowledge everyone that has helped me through this process. Special thanks to my supervisor Inger kappel Schmidt, who has greatly contributed through guidance and suggestions. When I met her for the first time, she introduced me to the threatened European heathlands and, soon, I realized that studying them it would contribute with some insights to my career. I would also like to thank to my co-supervisor, Sebastian Kepfer-Rojas, for his inputs, Muchas gracias!!

Furthermore, to the students Jesus Muñoz Serrano, Romain Baudet and Fabian Gutzat for their useful and valuable assistance in field work and for the provision of data. Additionally, to Gert Hougaard Rasmussen and Inken Breum Larsen from the Danish Nature Agency, who provided detailed information on the management of Randbøl Hede.

Special thanks to the Erasmus crew (Alex, Leo, Liisa and Bea) for sharing your time and experiences with me and highly contributing in this time lapse of our lives. To conclude, I express my gratitude to my family and friends (specially Rafa, Marta G., J. Martin and Marta M.) who kindly contributed with moral support through the last two years. I feel really fortunate to have you around.

1. Introduction

In agreement with UNESCO (2008), cultural landscapes represent the "combined works of nature and man" and are characteristic of the evolution of human society and settlement over time. The European Landscape Convention (ELC) (Council of Europe 2000) defined landscapes as "an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors" (Plieninger and Bieling 2012).

Cultural landscapes are mosaics of varied ecosystems and therefore, deliver ecosystem services. Ecosystem services were defined by Millennium Ecosystem Assessment (MA) in 2006 as "the benefits people obtain from ecosystems". MA recognized four different categories of Ecosystem Services (ES): provisioning, regulating, cultural and supporting services. Between their many essential functions and services cultural landscapes are, specially, acknowledged for allowing sustainable uses, serving as habitats for fauna and flora, providing economic benefits and supporting cultural heritage (Plieninger and Bieling 2012).

Interactions between cultural and natural forces have caused constant change throughout centuries. However, the speed, scale and magnitude of landscape and ecosystem change since the 1940s have been remarkable in Europe (Plieninger and Bieling 2013). In the particular case of Denmark, the extension of heathlands (cultural landscape) has decreased since 1800 due to agriculture practices and conifer plantations (Fig. 1). Danish heathlands accounted for 40% of whole Jutland in 1850, but declined to 3.4% by 1965. In only two centuries, the cover of Danish heaths has been reduced from 1 million ha to 84.000 ha, which accounts for 2% of total Danish land (Buttenschøn and Schmidt 2015). Strong drivers such as globalization, agricultural expansion and intensification, land abandonment and urbanization have affected on many cultural landscapes at European level (Plieninger and Bieling 2012). Those drivers have led to a decrease and, in some cases, a loss of traditional uses in cultural landscapes (Plieninger et al. 2014). As a consequence, an ample range of ecosystem services are on threat, as biodiversity, water purification or nonmaterial values (MA 2005).

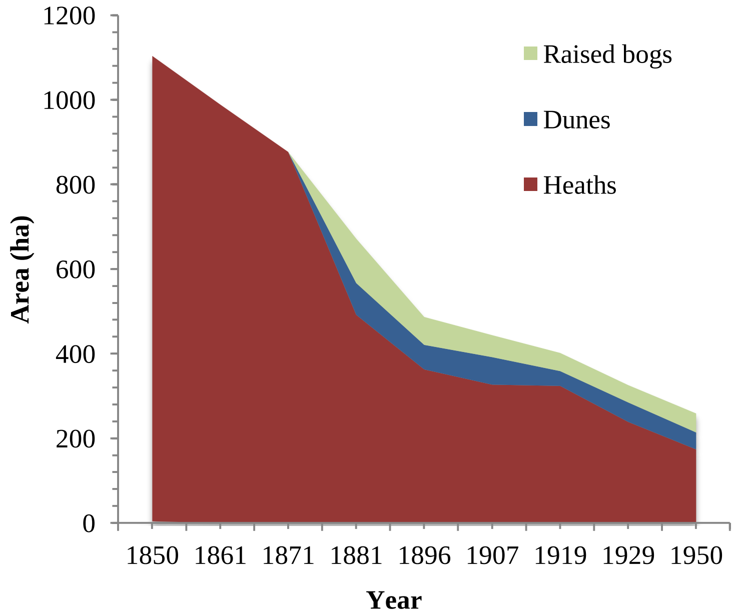

Fig. 1. Change in the coverage (ha) of heathlands in Jutland from 1850 to 1950. Taken from Nielsen (1953).

Adapting the traditional uses to the current environmental and economic situation is crucial to allow the continuity of cultural landscapes in the future. According to that, a deeper understanding of the natural dynamics of these landscapes is a requirement. In this work, I place an emphasis on the heathland European cultural landscape.

Within the scope of the thesis, I proceed with a review of the existing literature on heathland ecology, ecosystem services, traditional management and their impacts on biodiversity and nutrient pools. I will then follow with a description of the research conducted herein as well as the results. Taken together, I will present a discussion on the results and the future implications for heathlands conservation.

2. Heathlands ecosystems

Heathland is a semi-natural nature type characterized by dwarf shrubs and occurs on nutrient poor soils. It is restricted to North Atlantic regions in a European temperate context (Fig. 2), between Scandinavia and Cantabrian mountains (Fagundez 2013). Heathlands are dominated by dwarf shrubs in the family Ericaceae such as heather (*Calluna vulgaris*), cross-leaved heather (*Erica tetralix*), bell heather (*Erica cinerea*), bilberry/blaeberry (*Vaccinium myrtillus*), cowberry (*Vaccinium vitis-idaea*), with other shrubs such as *Juniperus communis* and the leguminous *Cytisus scoparius*, gorse (*Ulex europaeus*), dwarf gorse (*Ulex minor*), and *Genista anglica*. In the north and west of Europe, berry producing dwarf shrubs are of relevance, as *Empetrum nigrum* and *Vaccinium* sp (Gimingham 1992).

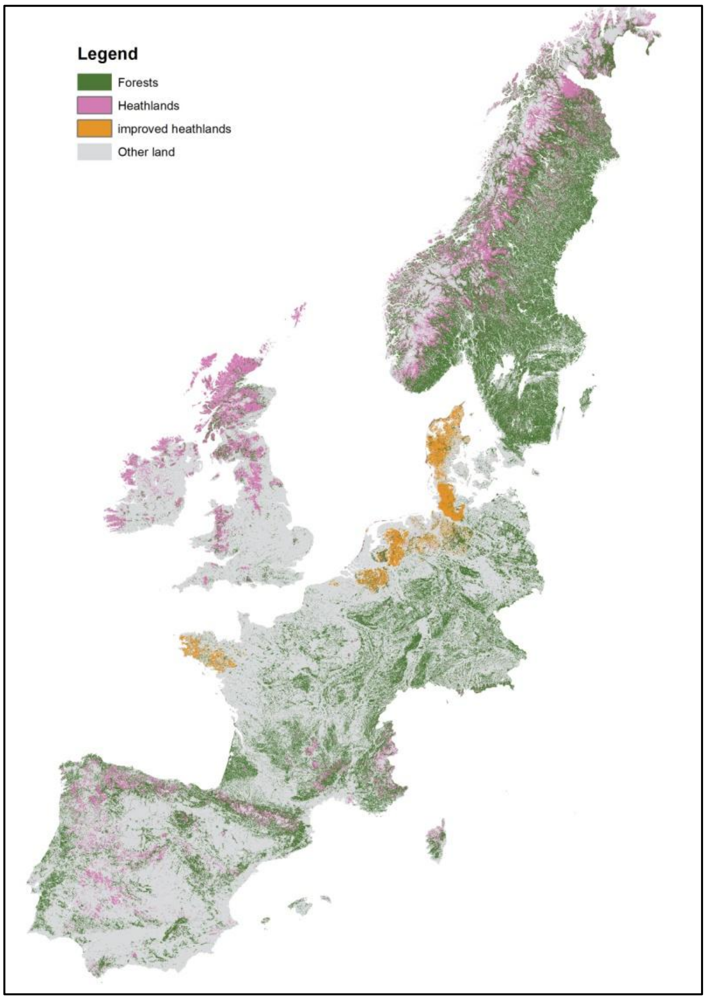

Fig. 2. Distribution of heathland ecosystems across Europe at the end of the 19th century. The legend shows the cover by forests (green), heathlands (pink) and the improved heathlands (orange) based on historic maps. There is high abundance of improved heathlands in Northern Germany and Western Denmark. Taken by Diemont et al. (2013).

Heaths support a range of characteristic plants and animals some of which are rare, protected and many are in decline. Heaths are ecosystems rich in insects as caterpillars, grasshoppers, moths and butterflies. They host species of lizards and snakes as well as very specialized birds e.g. grouse (Webb 2010). They also host moss and lichen species, such as cup lichens (*Cladonia* sp.).

Heathlands are associated with nutrient poor, acid mineral soils with sandy texture and gravels (Gimingham 1992). This type of soil is named podzol. The formation of a podzol includes mobilization, eluviation and illuviation. The structure of the podzol is characterized by an Ah-horizon which consists of a dark grey mixture or organic material and mineral matter (Fig. 3). The Ah-horizon is then followed by a bleached Ae-horizon where the leaching of Fe and Al cations and organic matter occur. Beneath of Ae-horizon, a thin iron-pan can be formed in areas where there is periodic water stagnation. This iron-pan is part of the B horizon (spodic horizon), characterized by a dark subsurface horizon with illuvial amorphous alumino-organic substances (FAO 2016).

Fig. 3. Soil profile showing the structure of a podzol from Randbøl Hede (control area). From top to bottom: Ho, horizon O; Ah, A-horizon; B, B-horizon; C, parental material.

Many authors have recognized leaching of nutrients as a process affecting the functioning and structure of heathlands (Schmidt et al. 2004, Niemeyer et al. 2005a, Hardtle et al. 2006). Leaching is mainly due to dwarf shrubs leaching organic acids (Hardtle et al. 2007). Leaching is also influenced by the application of treatments. Treatments increase the water percolation and mineralization rates due to increased soil surface (Niemeyer et al. 2005a). The increased leaching is also explained by the increase of NH_4^+ after fires, which replaces cations in the soil (Mohamed et al. 2007). The higher concentration of inorganic N (nitrate and ammonium) contributes to acidification (Houdijk et al. 1993, McGovern et al. 2014). Thus, leaching has effects on the nutrient balances and, therefore, determines the species enable to survive in this habitat (Houdijk et al. 1993).

2.1. Heathlands as European cultural landscapes

The loss of cultural landscapes with high biodiversity levels, dependent on extensive farming, is a clear problem all over Europe (Plieninger and Bieling 2012). Heathland is one of the principal cultural landscapes present in Europe (Webb 1998). It is also considered as semi-natural areas because it has been shaped by farming and agricultural practices through thousands of years. Semi-natural areas are the cornerstone of High Nature Value farming (HNV), consisting of ecosystems with low-intensity animal husbandry (Oppermann et al. 2011). Semi-natural areas are of conservation relevance when they are characterized by a typical vegetation association, by high species-richness, rare species or by high proportion of European populations of these species. In the case of heathlands, their relevance as semi-natural areas is because it has a typical vegetation association and it is a species poor habitat. Moreover, the species are specialized in stress-tolerance and not found outside heathlands.

The loss of traditional farming systems is a key threat to semi-natural ecosystems, which is motivated by land abandonment or agriculture intensification. The loss of traditional farming systems is affected by a combination of social, economic, political and environmental factors (Keenleyside and Tucker 2010), by which certain areas of farmland cease to be viable under existing land use and socio-economic structures (Felipe-Lucia et al. 2015).

Cultural landscapes also appear to behave as large-scale social–ecological systems in terms of the dynamics and synergies associated with economic production (Plieninger and Bieling 2012). Both systems, which are a combination of social (governmental, economic, human, built) and ecological (biotic, physical) subsystems, have the potential to shift into alternative stable states over time in relatively unpredictable ways as a consequence of internal and external change drivers and are influenced by variation across multiple spatial and temporal scales in relevant variables.

Extensive research has been accomplished regarding ecology of heathlands ecosystem (Fagundez 2013). Nevertheless, very few studies have been carried using the Ecosystem Services Framework to deeply study the varied range of ecosystem services delivered by heathlands, and as a consequence, ignoring interactions between social and ecological components at different spatio-temporal levels. Moran-Ordonez et al. (2013a) highlighted the relevance of studying ES in order to better understand the dynamics of this landscape and design more appropriate management strategies. The existence of grazing in different areas of Europe is recognized as an element having repercussion in the delivery of ecosystem services, and specially, soil fertility, seeds dispersion, cultural identity and traditional ecological knowledge (TEK) (Oteros-Rozas et al. 2013b). Oteros-Rozas has acknowledged that TEK contributes to maintain the capacity of systems to cope with disturbances in changing conditions, as it has been for many centuries (Oteros-Rozas et al. 2013a).

Moran-Ordonez et al. (2013a) concluded that in the Cantabrian mountains (Northern Spain) there has been a shift from the provisioning of local products (as for instance, food, fuel or wood) to provisioning of genetic resources (mainly at species level) for the accomplishment of national and international conservation measures. In these mountains, there has been a change in the landscape uses for the past 60 years. This means that cultural services have experienced an adjustment. Local-villagers are no longer dependent on the provisioning of goods from grazing and hay cutting. Heritage from shepherds' culture (songs, tales, handicrafts, etc) and TEK are decreasing in landscapes associated to transhumance (seasonal movement of shepherds with herds looking for good pastures) (Oteros-Rozas et al. 2013b). On the other hand, recreational and aesthetic values have increased, generating a new demand for business activities, as tourism recreation. (Moran-Ordonez et al. 2013a).

The Millennium Ecosystem Assessment states that the importance of cultural services and values is not currently recognized in landscape planning and management (MA 2005). Moreover, it recognizes that policy making could benefit from a better understanding of the way in which societies manipulate ecosystems and then relate that to cultural, spiritual and religious belief systems. MA also states that the ecosystem approach implicitly recognizes the importance of a socio-ecological system approach. Further, MA states that policy formulations should empower local people to participate in managing natural resources as part of a cultural landscape, integrating local knowledge and institutions (MA, 2005). However, at the moment there is no framework addressing the role of different stakeholders mediating ecosystem services of heathlands. Integrating food production and landscape use together with the maintenance and enhance of ecosystem functions and biodiversity is essential if one needs to stop and reverse declining tendencies of ecosystem services (O'Farrell and Anderson 2010).

2.2. Drivers of change

As population grew all over Europe in the 20th century, the demands for food have greatly increased. Population change is the main factor driving the land use (MA 2005).

Heathlands occur in places where both climate and soils are suitable for the forests as the potential vegetation. It has been shown that the majority of these heathlands were initially forest covered (Webb 1998). Human cleared the forest on need of areas for agriculture and grazing and, later on the areas with nutrient poor soils and were over-exploited, gradually developed into heahtlands. This scenery started during the Neolithic period (3000 BC) and continuing up to the 19th century (Gimingham 1992).

The demands of the humans were accountable for the reduction of the forest surface. The consequence of progressive forest clearance was the expansion of grasslands and heaths. These uses prevented the return to woodland (Gimingham 1992).

The Danish Heath Society, founded in 1866, encouraged the conversion of heaths to more profitable farmland and forests. Since then, the area of heathlands has decreased by more than 80% (Buttenschøn and Schmidt 2015). Nowadays, the remained area of heathlands is approximately 84.000 ha.

The reduction of heath areas is attributed to agricultural practices, due to land-use change and an increase in soil pollution. Nevertheless, atmospheric deposition of nitrogen and lack of nutrient-removal from traditional agricultural practices are allowing other species to grow (Power et al. 2001). As a consequence, the vegetation composition structure is changing (Brys et al. 2005b). To revert this situation, it is important to implement appropriate management practices (Haerdtle et al. 2006).

Main drivers of biodiversity loss at worldwide level as described by Convention of Biological Diversity (CBD) are: land-use changes, biological invasions, climate change, pollution and eutrophication, over-exploitation of natural resources (Fagundez 2013). Nevertheless, Sala et al. (2000) calculated the expected shifts for the year of 2100 in the main drivers of biodiversity change, showing N deposition as the principal driver for temperate regions. In accordance to CBD (2010) and Sala et al. (2000), a deeper understanding of how these drivers affect the natural dynamics of heathland vegetation is of particular relevance to establish prescribe conservation and management measures.

At the moment, fragmentation and loss of habitats are the main threats to biodiversity internationally recognized by the CBD. This led to the adoption of the Strategic Plan for Biodiversity 2011-2020 by the European Union (EU). This plan includes different strategic goals. Some goals of this plan are to address the causes of biodiversity loss or to reduce the direct pressures on biodiversity and promote sustainable use. This strategy includes six targets and 20 action points covering the full implementation of the EU nature legislation. Target 2 aims to maintain and enhance ecosystems and their services by establishing green infrastructures and restoring 15% of degraded ecosystems. Three of the 20 actions are determined to fulfill target point 2. The first one (action 5) aims at improving the knowledge base on ecosystems and ecosystem services. The second one (action 6) sets priorities to restore ecosystems and promotes the use of green infrastructure. The third one (action 7) launches an initiative to ensure no net loss of biodiversity and ecosystem services. This was adopted by the Council of the EU and as a consequence it should be implemented by the member states (CBD 2010).

EU legislation on protection of habitats is articulated through the Council Directive 92/43/EEC of May 1992 on the conservation of natural habitats and of wild flora and fauna (European Commission 1992). Article 2 gathers the aim: to contribute the insurance of biodiversity through maintaining or restoring natural habitats and species of European relevance. Article 3 helps to accomplish with this aim: the EU created an ecological network of special areas of conservation through Bird Directive and Habitat Directive, named “Natura 2000”. In the EU, 18% of the land is now included in the

Natura 2000 network of protected areas. Natura 2000 accounts for 8.32% of total Danish coverage (European Commission 2013). Moreover, under Article 6 the European Commission recognizes that all member states shall establish the necessary conservation measures to avoid the deterioration of the natural habitat types and the species in Annex I and II. (reference to Council Directive). Further, under Articles 8-11, the Commission recognizes that is up to the Member States to identify all measures essential for the maintenance or re-establishment at a favorable conservation status of habitat species and priority species. (European Commission 1992). According to Council Directive 92/43/EEC, Member States shall undertake surveillance of the conservation status of the natural habitats and the species of community interest (Art. 11). Hence, management measures to reach favorable conservation status and cost-efficient monitoring are required to evaluate the status.

Temperate heaths are listed in Annex I of Council Directive 92/43/EEC. The European Commission has recognized different heathlands habitat types, based on different climate conditions (oceanic, Mediterranean or continental), edaphic conditions (acidic or basic soils), water logging (wet or dry heaths), altitudinal range (alpine, upland or lowland) and other ecological variables conditioning structure and composition of the ecosystem (Fagundez 2013).

In Denmark, a few European heathlands can be found. For instance, habitat type 2140 (Decalcified fixed dunes with Empetrum nigrum), 4010 (Northern Atlantic wet heaths with *Erica tetralix*), 4030 (European dry heaths), 5130 (*Juniperus communis* formations on heaths or calcareous grasslands).

2.3. Habitat quality status

The concept of favourable conservation status is a subject of intensive research and discussion (Louette et al. 2015). This term is referred to the optimal and long-term ecological functioning and conservation of habitats and species. A habitat quality status implies the establishment of reference values, minimal requirements. These reference values are based on crucial elements for preserving habitats (structures and functions) and species (range, population, habitat). These values are based on scientific knowledge. In many circumstances, data is insufficient or models lead to unrealistic values (Louette et al. 2015).

The assessment of the conservation status is conducted through an evaluation of the habitat quality and other components as area and natural range, soil pH and soil nutrients content (Søgaard et al. 2007). Habitat quality refers to the ability of the ecosystem to provide conditions appropriate for individual and population persistence (McDermid et al. 2005). It is considered a continuous variable in the model, ranging from low to medium to high (values range from 0 to 1), based on resources availability for survival, reproduction, and population persistence, respectively (Hall et al. 1997).
Every member state shall undertake surveillance of conservation status (Art. 11). This also allows the implementation of conservation measures to revert the negative aspects and improve the conservation status. However, monitoring every aspect of the conservation status at different spatial scales is impossible due to lack of time and economic reasons (Metzger and Brancalion 2013). Data collection can be reduced to a minimum required to allow an appropriate assessment of the conservation status.
Some authors claim the fact that a habitat with a high quality score is more likely to cope with disturbances and maintain its identity and functions (Vogiatzakis et al. 2015).

2.4. Habitat resilience

The drivers of change mentioned before are pushing landscapes towards a reduction in the conservation status and a loss or shift on the delivery of ES (Moran-Ordonez et al. 2013a). Therefore, it is suitable to study how cultural landscapes respond to different levels of disturbance. I herein consider the term resilience as defined by Walker et al. (2007) “the capacity of a system to experience shocks while retaining essentially the same function, structure, feedbacks and therefore identity”. Hence, integrating resilience and cultural landscape together would enable to understand how a system is able to cope with perturbations without changing functions and structures until they cross thresholds. Beyond these thresholds, disturbances cannot longer be buffered and the system shifts towards another state. Combining the resilience and the cultural landscapes approaches may help dealing with landscape change and implementing suitable management policies (Plieninger and Bieling 2012).
Heathlands will be affected by climate change along coming years and it might alter the competitiveness between dwarf shrubs and grasses. Climate change experiments in multiple European heathlands show higher vulnerability of plant communities when

they are in early succession stages, leading to a shift in plant community composition (Ransijn et al. 2015a). The future of cultural landscapes is subjected to risk.

2.5. Management practices

As heathlands have always been human-shaped ecosystems, in the absence of management, the natural succession leads to woodland (Webb 1998). The gradual abandonment of traditional practices (e.g. extensive grazing, transhumance, controlled burning, etc.) has led to the natural succession of these ecosystems, as it happened in the Cantabrian mountains together with a loss of cultural ES (Moran-Ordonez et al. 2013b).

The traditional land uses and management practices of this ecosystem are: grazing, burning, cutting, turf-cutting, farming (Webb 1998). These practices have been applied to heathlands along centuries, and some times, local communities used a combination of them to increase the productivity of the ecosystem and obtain more benefits (Gimingham 1992). The lack of management results in a mature to degenerative phase of *Calluna vulgaris*, which may affect the regenerative potential and biodiversity (Calvo et al. 2007). As mentioned before, management is then a relevant issue for heathland maintenance and improvement (Fagundez 2013).

Optimal heathland management requires taking in consideration different aspects. First of all, the life cycle of *Calluna vulgaris*. This species has four growth phases and typically die at an age of 30-40 years (Gimingham 1992). These phases are known as: pioneer phase, building phase, mature phase and degenerate phase (Fig. 4). An ecosystem should include different stages of *C. vulgaris* life cycle because many species are associated with different life cycle stages and, specially, arthropods (Buchholz et al. 2013). A study conducted by Usher (1992) showed an influence of the growth-phase of the heather on the presence of particular species of spiders and beetles in a British heathland.

These phases are summarized as follows:

- Pioneer phase. Establishment takes part from seedlings or sprouts from stem bases that survived after fire. On the second year, flowering begins. This phase lasts for six years.
- Building phase. During this phase, which takes up to 10-15 years from the first sprouts, the plant has the highest ratio of biomass production.

- Mature phase. This phase develop up to 20-25 years after the first sprouts. The growth is less vigorous and gaps may form in the canopy.
- Degenerate phase. Gaps in the canopy increase and eventually the whole plant may die.

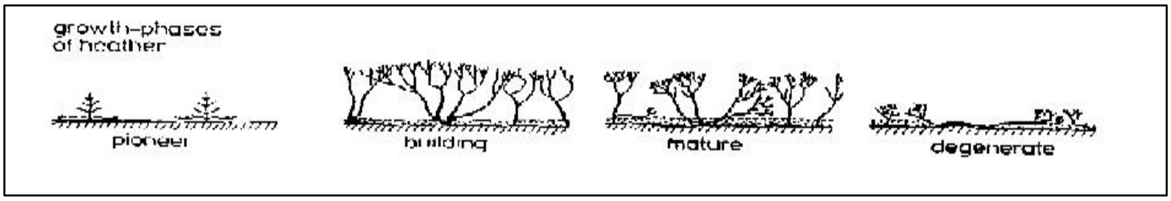

Fig. 4. The four growth phases of heather (from left to right: pioneer, building, mature and degenerate). From Gimingham (1992).

The second aspect is the functional diversity. At the same time as variation on age of *C. vulgaris* promotes species diversity (Haysom and Coulson 1998), different functional types influence the presence of many arthropods (Bell et al. 2001a; Schirmel et al. 2010). A heathland with high functional types diversity is an ecosystem with a large number of potential niches for fauna and flora (Schirmel et al. 2011). There are many species which find shelter in trees (e.g. *Caprimulgus ruficollis, Tetrao tetrix*), require temporary lakes (e.g. *Numenius arquata*) or is favoured by a mixture of grass and dwarf shrubs (Webb et al. 2010).

The third aspect is the increased deposition of atmospheric N as a threat to heathland dynamics. The increasing amount of atmospheric N deposition in recent decades and the abandonment of traditional land use has led to increased N:P ratios. The resulting shift from N to P leads to changes in the competitive balance between species (Marrs et al. 1993, Uren et al. 1997, Kirkham 2001, Roem et al. 2002). As a result, there has been a transition from *Calluna* dominated heathland to grassland dominated by the grasses *Deschampsia flexuosa* and *Molinia caerulea,* which have low P-requirements. (Fagundez 2013).

The last aspect is bare soil. Bare soil is another potential niche to be filled by a new generation of plants (Diemont and Linthorst Homan 1989). It is an opportunity for other species, as scrubs or grasses, to establish and especially, for *C. vulgaris* (Britton et al. 2000). Moreover, it is the habitat for different invertebrates. These gaps are of high relevance when considering the management and conservation of this ecosystem (Gimingham 1992).

Habitat management can mitigate the consequences of atmospheric N deposition and can regenerate *Calluna vulgaris* (Hardtle et al. 2006, Mohamed et al. 2007). Nevertheless, one only management practice is not able to guarantee all the benefits obtained by a combination of two or more practices (Sedlakova and Chytry 1999, Niemeyer et al. 2005a). For example, Niemeyer et al. (2005a) concluded in a study conducted in Northern Germany that prescribed fire is not sufficient to compensate the atmospheric N-deposition. It cannot compensate the effects when the frequency of prescribed fire is ca. 10 years. Each management practice has similar effects, but also different results. In view of this, I present a description of different management practices and how they influence on the ecosystem.

2.5.1. Grazing

Grazing is an appropriate method for heathland management and is the most common used in Denmark (Buttenschøn and Schmidt 2015). Grazing has effects in vegetation composition, soil structure and biodiversity (Bokdam and Gleichman 2000, Britton et al. 2005, Garcia et al. 2009, Jauregui et al. 2009).

Grazing alters vegetation community, shifting from a shrub dominated heath towards a grassy heath (Bullock and Pakeman 1997, Newton et al. 2009). Species which benefit from reduced grazing pressure are *Calluna vulgaris, Carex nigra, Deschampsia flexuosa* and *Molinia caerulea* (Hulme et al. 2002). Grazing at low densities has a positive impact on dwarf shrubs by promoting annual growth, preventing them from passing into the degenerate phase (Hardtle et al. 2009). However, overgrazing of heaths is a key threat to the habitat (Garcia et al. 2013). Overgrazing can bring on the loss of dwarf shrubs, facilitating the development of grassland species that are already present or uncommon (Hulme et al. 2002, Garcia et al. 2009, Celaya et al. 2010). Garcia et al. (2009) revealed in a study in the Cantabrian heathlands that increasing grazing pressure led to an increase in herbaceous cover, height, and biomass and opposite effects for shrubs. Heather biomass and cover may decrease, allowing grasses as *Nardus stricta* and *Agrostis curtisii* and to propagate (Hartley and Amos 1999, Celaya et al. 2010).

Different types of livestock and its levels of pressure generate variations in community composition (Celaya et al. 2010). However, heathlands have its limitations to feed livestock over winter, making more difficult and expensive to farmer the feeding of their livestock. Cattle and sheep are highly competitive for herbaceous species with high nutrient content (Ferreira et al. 2013). Cattle is the preferred grazing animal in Denmark, but there is little documentation of the effect. A study conducted by Fottner et al. (2007) argued that sheep grazing can compensate for atmospheric N deposition. In contrast, goats can be used to tackle areas where *Juncus* spp dominate (Gimingham 1992). Goats play an important role, as their preference is associated with shrubland (Garcia et al. 2011), although its use is not traditional from many countries. Appropriate stocking levels should be considered by taking into consideration its habitat quality status, other practices as well as wild herbivores to maintain heathlands in the pioneer and building phases (Buttenschøn and Schmidt 2015).

Soil structure is also affected by grazing. Bell et al. (2001b) outlined an increase in the size of litter layer. Different grazers create dissimilar microhabitats, as cattle trampling shapes topography and increase heterogeneity, encouraging some invertebrate groups (Gimingham 1992).

A study conducted by Garcia et al. (2009) revealed that grazing is a driver of arthropod community structure. Garcia et al. (2011) confirmed that goats increase environmental heterogeneity, providing a wider variety of microhabitats for invertebrates. However, goats should be combined with other animals (Ferreira et al. 2013). For example, carabids and lycosids were enhanced under high stocking rates, whereas Opiliones were promoted by low stocking rates.

In general, grazing effects depend on varied factors (Hulme et al. 2002), there is no grazing pressure that will be appropriate across countries, and for all fauna (Jauregui et al. 2008, Celaya et al. 2010). Hence, the optimal grazing regime will depend on the management objectives and local conditions (Garcia et al. 2013). Grazing is an adequate practice for heathlands but, it needs to be combined with other practices, as burning, because grazing may cause a P-deficiency (Fottner et al. 2007, Hardtle et al. 2009).

2.5.2. Burning

It has been acknowledged that controlled burning has many effects on heathland dynamics (Niemeyer et al. 2005a). Burning has traditionally been a common used practice due to low economic cost and fast effects. It is complex due to diverse factors (e.g. weather, topography, season, water content),which might result in undesirable outcomes (Gimingham 1992). Uncontrolled burning and prescribed burning can influence negatively on vegetation composition and nutrients balance in soil and plants.

In Denmark, burning is not allowed through summer periods because temperatures can attain high values (Michael 1996). Further, windy periods should be avoided due to the drying effect on vegetation litter (Gimingham 1992). In some regions as in the Cantabrian Mountains, burning has been banned under legislation for the protected areas. As consequence of this conservation policy, a shrub and trees encroachment is going on (Moran-Ordonez et al. 2013b).

Prescribed burning has effects on vegetation community, soil structure and fauna assemblages. Davies et al. (2010b) inferred, in a study conducted in the Scottish Highlands, that the growth phase of *C. vulgaris* affects the regeneration by shoots. Regeneration is lower in the mature phase compared to the building one after applying fire. This means that growth phase should be considered when prescribing burning at local scale (Davies et al. 2010a, Velle et al. 2012). Two management recommendations are to burn the vegetation on a 10-20 years rotation or when *C. vulgaris* is 20-30 cm tall (Michael 1996). When vegetative regeneration is not effective, seed germination may be successful (Gimingham, 1992; Alvarez-Alvarez et al. 2013).

Prescribed burning promotes an increased deficiency of nutrients in *Deschampsia flexuosa* and, therefore, facilitating the competitive capacity of *Calluna* against *Deschampsia* (Mohamed et al. 2007). A negative consequence is that prescribed burning also promotes seed germination and biomass production of *Molinia caerulea*, and therefore, increasing its competitiveness and influencing heathland community (Brys et al. 2005b)

Burning has also consequences on soil structure (Niemeyer et al. 2005b, Mohamed et al. 2007, Davies et al. 2010b). An increase in soil pH was reported after fire due to ash deposits and destruction of organic matter (Green et al. 2013). Moreover, leaching rates changes with the application of fire. The leaching rates were particularly high for N, Ca and K after controlled burning in a German experiment and, as a consequence, it improved the competitiveness of *Calluna* (Mohamed et al. 2007). Intense fires are more likely to ignite peat and, therefore, causing carbon losses and poor regeneration, decreasing the effects of carbon sequestration of heathlands (Davies et al. 2010b). Further, O-horizon can be affected at different rates by fire depending on the presence of cryptogams and their cover, which act protecting the O-horizon (Niemeyer et al. 2005a). Mosses, lichens and litter are generally destroyed and, subsequently, bare

ground is exposed, giving an opportunity for *Calluna* seedlings to contribute in the recovery and promoting empty niches (Sedlakova and Chytry 1999).
To ensure the maintenance of invertebrate's population and composition through recolonization, it is important to account for some factors as vegetation patch size, frequency and intensity (Bell et al. 2001b, Bargmann et al. 2015). Bargmann et al. (2015) concluded that burning is specially relevant to maintain carabid diversity in Norwegian heathlands. They suggested the creation of a mosaic of varied burnt years, to allow dispersal from nearby patches. Bell et al. (2001b) highlighted that unburned areas are necessary to allow the recolonization of other patches by spiders.
However, prescribed burning is not enough to revert the effect of the atmospheric N deposition (Niemeyer et al. 2005a). Different studies suggest combining burning with another management practice, as cutting or grazing (Sedlakova and Chytry 1999, Niemeyer et al. 2005a).

2.5.3. Cutting

Cutting management has been recently intensified due to modernization of machinery. For that reason, this tool can be applied at different levels of intensity. On one hand, the low-intensity mowing, in which only the aboveground biomass is affected. On the other hand, the high-intensity sod-cutting, in which the soil organic layer is also affected (Hardtle et al. 2006). Cutting affects vegetation community, soil structure and fauna assemblages (Sedlakova and Chytry 1999, Barker et al. 2004, Mohamed et al. 2007).
The regeneration of heathland species after cutting depends on five different factors. It depends on the condition of the previous heather canopy, of the abundance and behavior of herbivores, on the climatic conditions, on the intensity of cutting and on the persistence of the seed bank (Calvo et al. 2007). A study carried out in Northern Spain, showed that grasses can become established, and those already present can increase in cover and, eventually, become very abundant (Calvo et al. 2007). Moreover, in an experiment carried out in Czech Republic by Sedlakova and Chytry (1999), cutting management resulted with three consequences. First, a slow process of vegetation dynamics. Second, a rapid spread of grasses as *Deschampsia flexuosa* and *Festuca ovina*. And, finally, a slow vegetative regeneration of *Calluna vulgaris*. Moreover, cutting reduces vegetation height and biomass.
The growth phase of *Calluna vulgaris* also affects the process of regeneration of the heather. Old heather stands show poor regeneration (Mohamed et al. 2007; Gimingham

1992). If a heather stand is on a degenerate phase is less likely that will regenerate in comparison with a stand on a building phase. Moreover, if the cutting is of high-intensity, then *Calluna* root systems can be damaged and may hamper the vegetative regeneration. However, if the cutting is at low-intensity, it could lead to a vigorous vegetative regeneration (Barker et al. 2004).

Cutting also affects the soil structure. It causes large changes in relative humidity and light exposure (Diemont and Linthorst Homan 1989). As a consequence, there is an increased mineralization of organic matter and the A-horizon. That could be the reason for the increase in N leaching (Haerdtle et al. 2006). Moreover, bare ground is created when cutting is applied.

At the same time, cutting has a drastic impact on the abundance and diversity of arthropods (Bell et al. 2001b). For example, many carabid beetles are associated to different growth-phases of *Calluna vulgaris* (Schirmel and Buchholz 2011). Moreover, a study conducted in Germany identified the loss of 35 species due to succession towards a grassy-heath and tree encroachment (Buchholz et al. 2013). The effects of cutting practice over arthropods community are also influenced by the season when the treatment is applied (Bell et al. 2001a).

In general, cutting contributes to the regeneration of heather. However, cutting is less effective in depleting nutrients from the soil in comparison to fires (Webb 1998).

3. Problem statement

The extension of Danish heathlands has been highly reduced in the last two centuries mainly due to land-use change and pollution. For millennia, humans have played a role in the maintenance of heathlands by using grazing associated with other management practices such as periodic burning or cutting, in order to obtain benefits such as food or gravel and sand for building.

The change in the management practices, intensification or abandonment, increases the risk of disappearance of heath species. As consequence, heathlands have experienced a decline in habitat quality, which affects to community structure, ecosystem functions and biodiversity (Fagundez 2013). Thus, heathlands are on threat and on need for enhancement of habitat quality. Therefore, the adoption of appropriate measures (including traditional practices) is essential for heathland conservation. According to the Habitat Directive, all member states of the EU shall establish the necessary conservation measures to avoid the deterioration of the natural habitat types and the species (European Commission 1992).

To avoid further deterioration and improve heathlands conservation status, member states have been applying the traditional management practices with the objective of tackling the effects of atmospheric nitrogen deposition and regenerate dwarf shrubs species (Power et al. 2001). The increase of nitrogen in the soil causes a shift from N limitation towards P-limitation, increase the competitiveness which favours grasses as *Molinia caerulea* and *Deschampsia flexuosa*. *C. vulgaris* has four growth phases and its life cycle last for 20-25 years (Gimingham 1992). The development of *Calluna* towards the mature and degenerate phases implies negative consequences on the nutrients balance and biodiversity.

The European Commission recognizes that the member states shall accomplish an evaluation of the conservation status of habitats types included in the Habitat Directive every six years (European Commission 1992). In agreement with the Habitat Directive, the Danish Nature Agency developed a habitat quality assessment to evaluate the status of dry heahtlands (habitat type 4030). The last habitat assessment, carried out by the member states in 2013, concluded that many heathlands in Denmark and, in other European countries, have a bad habitat quality (European Commission 2015; Danish Nature Agency 2014).

To achieve an improvement of the habitat quality status, the focus of managers and member states should be to create a mosaic of different structures, including scrubs, trees, bare ground, temporary water systems, etc. These structures are the habitat for many animal species, specially invertebrates.

This study aims to understand the effect of the application of management strategies such as cutting and burning assisting the establishment of heathland communities. For that reason, we analyze the effects on biodiversity, abundance of functional groups and soil structure, produced by: 1) cutting, and 2) prescribed burning. We believe that studying the effects of grazing on heathlands dynamic and habitat quality is interesting. However, limitations due to the design made impossible to achieve this goal. Further, this study aims to evaluate the habitat quality of three heathlands after the application of prescribed burning and cutting as management practices. These practices have been applied to different areas in two heathlands. Therefore,

In order to answer these aims, two hypotheses have been addressed in this study:

- The application of management practices enhances diversity of functional plant groups.
- The habitat quality of heathlands is improved by the application of management practices.

4. Methodology

4.1. Study area

Three different sites were selected in western Denmark to study the effects of different management practices on the habitat quality of European dry heathlands (Habitat type 4030). These heathlands are Nørholm Hede, Randbøl hede and Trehøje Hede, allocated in Jutland peninsula (Fig. 5).

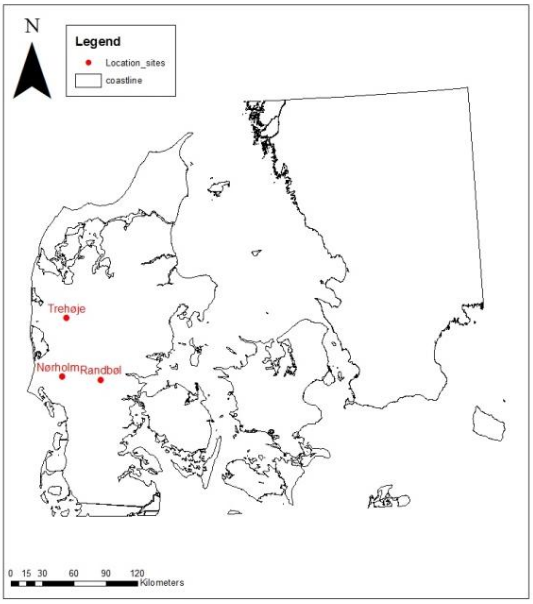

Fig. 5. Location of the three areas of study: Nørholm, Randbøl and Trehøje hede. They are allocated in Western Jutland, Denmark.

Nørholm hede (55º 67'N, 8º 59' E) is a private-owned area located 10 km northeast of Varde. It is part of a Natura 2000 area named Nørholm Hede, Nørholm Skov og Varde Å øst for Varde (site code: DK00AX175), which has an extension of 991ha. It was declared Natura 2000 in 1998, although, it is protected at national level since 1913. This heath (Fig. 6), which covers 350ha, was part of a traditional farming system until 1890. Natural succession has occurred since then and it has been an object of research along 20th century (Riis-Nielsen, et al. 2005).

Between 1000 BC-1900 AD, Nørholm hede was managed under traditional practices, consisting of "infield/outfield system". This practice consisted of areas for agriculture (infields) and areas for low intensity grazing (outfields). At night, the animals were kept in the stables and their manure were collected from the outfields, to be applied in the infields as fertilizer, transferring nutrients to the agricultural fields (Riis-Nielsen, 2005).

This agricultural system was maintained until 1865 and the grazing continued until 1895 when Nørholm hede was left to free

succession. This system had outstanding influence on the environment due to alteration of soil structure, removal of vegetation and mobilization of nutrients (Kepfer-Rojas et al. 2014). As a consequence, it generated a nutrient poor area in the outfields and a more nutrient rich area in the infields.

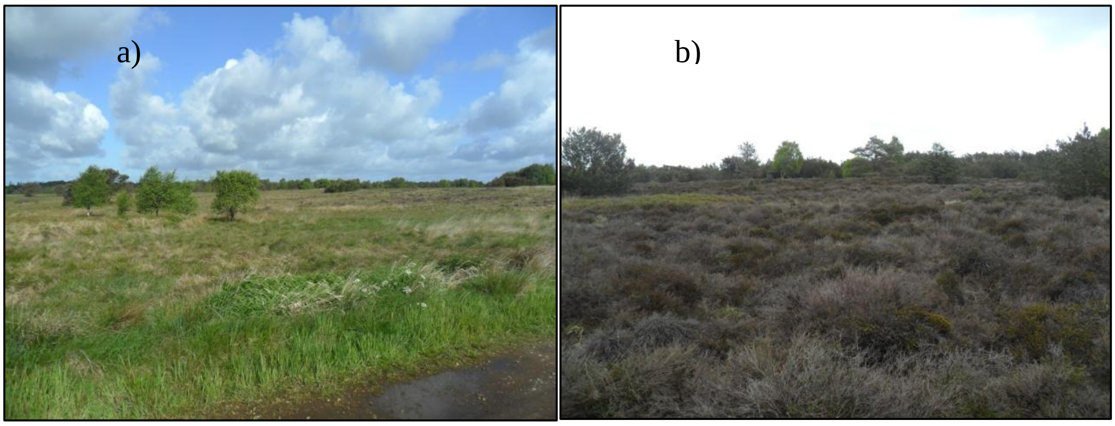

Fig. 6. Different treatment areas within Nørholm Hede. Differences in the vegetation communities can be appreciated: a) the cultivated treatment, b) the control, (or unmanaged). Pictures were taken in summer 2015.

Randbøl hede is a protected area situated ca. 25 km west from Vejle (Fig. 5). It is protected at national level since 1932 and, as Natura2000 (site code: DK00BY171) since 1998. This Natura2000 (55º 61'N, 9º 15'E) covers 958ha, including circa 750ha of heathlands, one of the biggest heathland found upcountry. The recent history of Randbøl hede is linked to Germans since 18th century. The Danish government established measurements in 1760 to promote the cultivation of land. German families started cultivating potatoes and soon, sand drifting become a great problem as a consequence of intense cultivation. A measure to revert this effect was the forestation of 4.5km^2 in 1804. The forest was named Frederikshåb Plantation and took more than 100 years to develop due to the harsh conditions with frost, sand drift, drought and pests. The recent activity that shaped this heath was the Second World War under the influence of German soldiers. Randbøl hede was used to hide planes and, nowadays, the resulting structures of that activity can be observed (Naturstyrelsen 2015).

Randbøl hede is a big and diverse heath and it is protected due to the presence of numerous species cataloged by the Habitat and Birds Directive. This heath was the first location for nesting of the Eurasian curlew (*Numenius arquata*) in 1934, after the extinction of this species due to hunting in 1800. Nowadays, it is present in many locations in Jutland. Moreover, the common crane (*Grus grus*) is increasing its visits to Randbøl hede and in a near future, this site may become a common area for its breeding (Naturstyrelsen 2015). The rare ladybird spider (*Eresus sandaliatus*) finds its habitat in this heath.

One of the most relevant problems in this heathland is the excessive abundance of *Molinia caerulea.* Recent experiments have been applied to determine which management practices (i.e. cutting, burning, grazing by cattle and grazing by sheep) promotes a decrease of its abundance. Additionally, Randbøl hede has been an object of several projects, as for example, LIFE projects with the aim of habitat restoration and dissemination of information (Naturstyrelsen 2015).

The area has been divided into sections with varied treatments. There are two well defined areas: one with grazing by cattle and, another with grazing by sheep. Moreover, there is another section without management for more than 70 years. Further, there are other areas where cutting and burning are the management practices applied for long periods (Fig. 7).

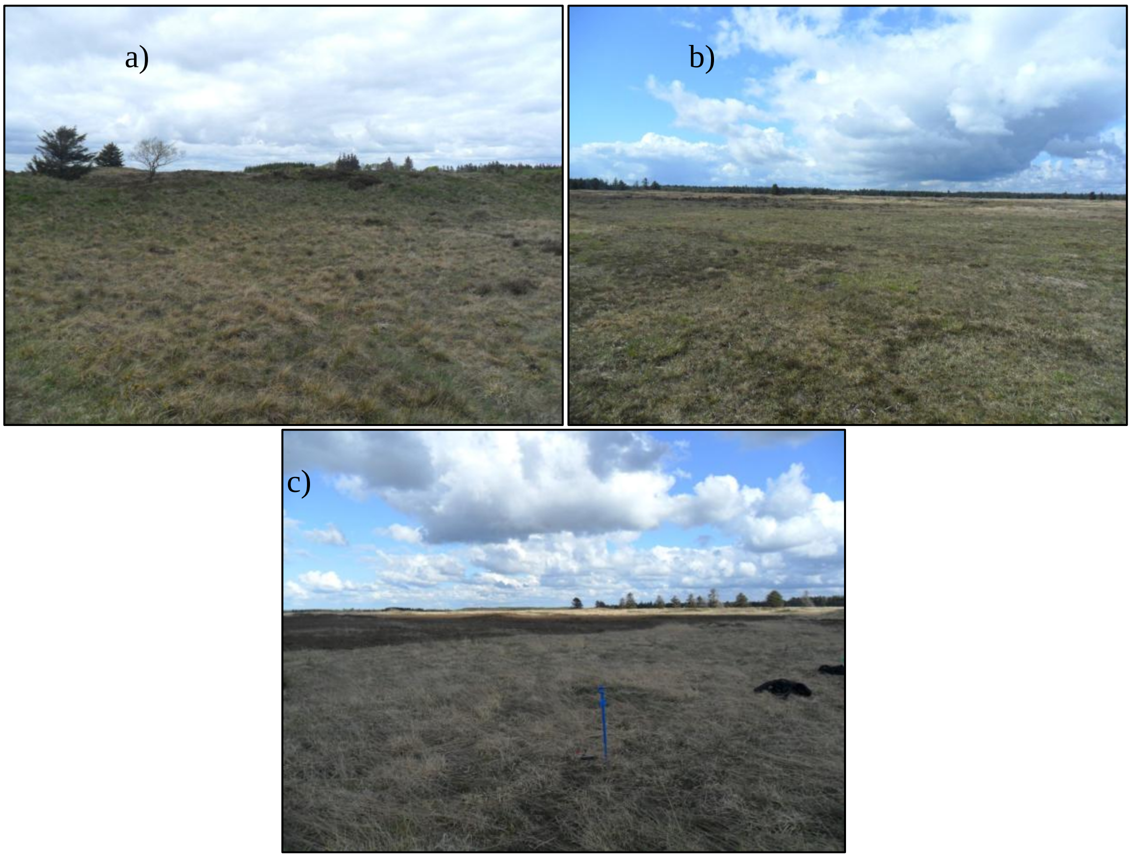

Fig. 7. Different treatments areas within Randbøl Hede. a) Control area, b) cut treatment, c) burnt treatment. Images were taken in July 2015.

Trehøje hede (56º10'N, 8º39'E) is a heath with an area of circa 260ha, located 15 km west from Herning (Fig. 5). It is not protected as Natura2000. This heath has been exposed to different management practices. Currently, there are two main areas that can be distinguished: from one side, an area with grazing; from the other side, an area without grazing activities. Grazing by sheep has recently been applied by managers. Moreover, other treatments have been applied over similar areas as cutting or burning. This heathland is of relevance for science because there are no many studies on combined management.

Climate

Denmark falls into temperate climate zone, characterized by cool summers and mild winters. The data herein is an average for the period 1961 -1990 in Middle and West Jutland. There was an average annual temperature 7.7ºC. However, the month with the highest average temperature (20.1ºC) was August, in which drought could affect negatively to some plant species. The lowest average temperatures (T=-2.9ºC and T=-3.1ºC) were registered for January and February, respectively, in which some plant species can suffer damages. The average annual precipitation is 781 mm. The sun is shining during 1395 hours every year on average (Danish meteorological station, 2016).

4.2. Experiment Design

Areas subjected to only one type of management for a long period were selected for establishment of the experimental units. In Nørholm hede, we selected a control area, which has been without management, and a cultivated area, which has not been used for agriculture since 1890 (Appendix A: Fig. A1). We used the term control for the unmanaged area and, cultivated, for the area that was used for agricultural purposes until the end of nineteenth century. In Randbøl hede, two treatments (cut and burnt) and a control were selected (Appendix A: Fig. A2). The selection of the experimental units was determined based on staff from the Danish Nature Agency (Table 1). There is no information available for this heathland about the nineteenth century. The control, in this case, it is an unmanaged area (for more than 70 years) of Randbøl and it might have been used for agricultural purposes during 18^{th} and 19^{th} century. In Trehøje hede, the experimental units were set up following a similar methodology as for Randbøl hede

(Appendix A:Fig. A3). We selected the treatments in Trehøje as similar as possible to the ones in Randbøl, although the frequency of application of treatments was lower (Table 2). The control has been cut two times in the last sixty years and no as recent as the treatments.

Table 1. Management history of the areas sampled in Randbøl hede. Hyphen (-) indicates absence of management in the correspondent year. No data was available on the period 1995-2012. Based on Larsen (summer 2015).

	Past management		
Year	Control area	Cut area	Burnt area
2015	-	-	Burnt
2014	-	Cut	Burnt
2013	-	Cut	Burnt
2012	-	Burnt, Cut	Burnt
1947	-	Burnt	Burnt
1929	Burnt	-	-
1927	-	-	Burnt

As mentioned before, this heath was used by germans for the cultivation of potatoes in the 18th century.

Table 2. Management history of the areas sampled in Trehøje hede. Hyphen (-) indicates absences of management in this year. Period from 1954-2012 based on Danmarks miljøportal. Period from 2013 based on Buttenschøn, R.M., Schmidt, I.K. (2015).

	Past management		
Year	Control area	Cut area	Burnt area
2013	-	-	Burnt
2012	-	-	-
2011	-	-	-
2010	-	Cut	-
2008	Cut	-	-
2006	-	-	-
2004	-	-	-
2002	Cut	Cut	Cut
1999	-	-	-
1995	-	Cut	-
1954	-	-	-

4.3. Soil sampling and analysis

To identify how the management practices influence the soil structure, we collected samples for different parameters. Soil samples were taken in Randbøl hede by Fabian Gutzat and in Trehøje Hede by Jesus Muñoz Serrano and posterior analysis in laboratory, for their respective master thesis. They kindly shared their data for the current study. Their designs consisted of 3 circular experimental units of 5 meters radius for each treatment (in total, 3 replicates). In each replicate, 3 soil samples were taken at a distance of 5 meters from the center point. The resultant soil core was divided into three samples based on the soil horizon (O, A, B) in the case of Randbøl Hede, and only for O and A horizons in Trehøje Hede. The depth of the O-horizon was determined. Further, lab analysis was conducted by Fabian Gutzat and Jesus Muñoz to determine C, N, pH (Gutzat, 2015; Muñoz, 2015).

Regarding Nørholm hede, the soil data was obtained from Van Steerteghem Evelien, with permission from Inger Kappel. Based on GPS data, we selected the data from the 3 closest transects (for each area; established by Evelien in 2012) to the transects we set up in 2015. For each transect (each transect consisted of 10 plots with a size of $1m^2$ quadrants) set up by Evelien, we selected soil data from three plots (Van Steerteghem 2012).

4.4. Vegetation sampling

In order to identify how the management practices affects the vegetation community, we recorded the coverage of functional types. In each treatment of Nørholm hede and Randbøl Hede, three transects were randomly marked with a GPS (Appendix A). The selection of the locations for the replicates in Randbøl hede was built according to the knowledge of staff from the Danish Nature Agency who kindly provided information about management activities.

Each transect was 50 meters long and consisted of 25 plots with two meters interval along a fixed line. In every plot, vegetation and abiotic variables were recorded. These plots were 0,1 m^2 circular. We recorded the following abiotic variables: thickness of Organic Matter layer (OMD), thickness of Litter Layer (LL), soil temperature (T) and soil moisture (SM). OMD and LL were estimated in cm; T was estimated in Fahrenheit (and posterior adjustment to Celsius) and; SM in percentage. OMD and LL were measured by cutting 5x5 cm of soil with a depth of 15cm using a knife and, subsequently, discarded during the field work. T and SM were recorded at 5cm depth using a soil digital thermometer and Theta Probe device.

All species of vascular plants present in Nørholm hede and Randbøl hede were recorded for each single transect in June and July 2015. However, mosses and lichens were not identified at species level. Moreover, within every plot we measured the coverage of four dominant species (*Calluna vulgaris, Empetrum nigrum, Deschampsia flexuosa and Molinia caerulea*), and functional types (other grasses, other dwarf shrubs, forbs, shrubs, trees, lichens and mosses). Further, we measured the maximum height of *Calluna vulgaris*, vegetation height and percentage of bare soil for the plots. The vegetation height is an average of 5 samples taken within each plot and without taking in consideration the inflorescence of the plant species.

In Trehøje Hede, the vegetation analysis was conducted by Rita Buttenschøn in July 2015 and kindly shared by Jesus Muñoz Serrano (Muñoz, 2015). In this case, the vegetation was analyzed in 6 different treatments with 3 replicates each. Within each replicate, 20 randomly placed 0.1 m2 circular plots were marked. Presence-absence data was recorded for every species, including lichens and mosses, occurring within the plots. As there were 20 plots for each replicate, the percentage of plot was used as a proxy for the species or functional group abundance within each replicate (i.e. dwarf shrubs, grasses, herbs, lichens). From these 6 treatments, we selected three of them to compare with the other locations (control, cut and burnt). In order to be able to compare the data from the three different locations, the proportion of species was calculated on base to the frequency for replicates as a relative abundance.

4.5. Habitat Quality Assessment

In order to determine how the management practice influences the habitat quality of heathlands, a habitat quality assessment was conducted for each treatment (Appendix D). The methodology was developed by the National Environmental Research Institute (at the request of the Danish Nature Agency), which was located in Aarhus (Naturstyrelsen 2010). This assessment has different sections referred to vegetation analysis and other characteristics of heathlands. It is a visual method based on qualitative data. It includes a section for the analysis of 4 positive structures (Age variation in *Calluna vulgaris*, age variation in *Erica tetralix*, dominance of dwarf shrubs and presence of lichens) and 4 negative structures (Old dead areas with *Calluna vulgaris*, dominance of grasses and *Molinia caerulea*, presence of the invasive moss

Campylopus introflexus, the invasive shrub *Cytisus scoparius,* and conifers (except for *Juniperus communis*)). It has another section for the cover of different functional groups, differentiating 5 groups in relation to the percentage cover (i.e. 0-5%, 5-10%, 10-30%, etc). Moreover, it has a section for the cover of agriculture area and the one for grazing or cutting. It also includes a description of hydrology system and management recommendations (Naturstyrelsen 2010).

4.6. Data analysis

Statistical analyses was performed using R 3.2.3 (The R Foundation for Statistical Computing 2014).

The diversity of vegetation communities was estimated using different diversity indices: richness (S), Shannon-Wiener (H'), Pielou (J) and Simpson (D) diversity indices were calculated. We decided to include all this indices because S per se offers limited amount of information about the biodiversity of the area. Heterogeneity indices such as Shannon-Wiener and Simpson provide information in both species richness and evenness (Buckland et al. 2012). Shannon-Wiener (a) is defined as:

a) $$H' = -\sum \ pi \times ln \ (pi)$$

Where *pi* is the proportion of individuals in the *i*-th species. H' takes values between 0 and 5. This index is commonly used. Its inclusion can facilitate comparisons with other studies. Other studies advocates for the use of Simpon's diversity index (b), defined as follows:

b) $$D = pi^2$$

And Pielou (J), which is defined as:

c) $$\mathrm{J} = \frac{\mathrm{H'}}{\ln(\mathrm{S})}$$

Where H' is the Shannon-Wiener previously calculated and ln(S) is the natural logarithm of species richness. Biodiversity was then analyzed with species richness by each transect. Shannon, Pielou and Simpson were analyzed using the coverage of functional groups found in each plot in the case of Nørholm and Randbøl.

However, biodiversity indices do not always provide enough information. To gain insight, species richness should be complemented by measures of e.g. as functional coverage, rarity or habitat quality when evaluating management practices (Pottier et al. 2007, Massant et al. 2009).

The influence of treatment on soil parameters (i.e. thickness of litter layer, thickness of organic matter layer, soil moisture), biodiversity indices and the cover of different functional groups was analyzed using one-way ANOVA.

Prior to analyses, we classified the study in two sections: two or three samples (treatments) location. When the variables had two samples, they were checked for the Student's t-test assumptions of independency, normality and homogeneity of variance (the non-parametric Mann–Whitney *U* test was performed when the data did not meet the assumptions). Alternatively, when the variables had three samples, they were tested for the ANOVA assumptions of equal sample size (N), independency, normality and homogeneity of variance. Samples were always considered independent. Variables were checked for normality with the Shapiro-Wilk W test and, homogeneity of variance with the Barlett test. When the data did not meet the assumptions for one-way ANOVA, the non-parametric Kruskal-Wallis (KW) by ranks test was performed for three or more groups.

From one hand, when one-way ANOVA resulted in significant differences, multiple comparisons test was performed (Tukey'HSD post-hoc test). The reasons to use Tukey'HSD was that is the most common test and it is a multiple comparison test used after ANOVA to find means that are significantly different from each other (Mangiafico 2015). By the other hand, when KW test resulted in significant differences, KW post-hoc Nemenyi test was applied to identify differences among treatments. Nemeny test was performed using the "*PMCMR"* R package. I used Nemenyi test because groups had equal number of samples (Mangiafico 2015).

The influence of treatment on soil parameters, vegetation structure (vegetation height and *C. vulgaris* height), vegetation composition and biodiversity was studied by using raw data with one-way ANOVA or Student's t-test for Randbøl and Nørholm, respectively.

We also used one-way ANOVA to find differences for soil parameters, biodiversity indices and functional groups coverage between the controls across locations. In this case, we used the controls from Nørholm, Randbøl, which have been unmanaged ad least for seventy years, together with the control from Trehøje, which has been without management for 8 years. Hence, we expect to find similarities and differences.

To determine the possible links among the studied parameters, Spearman's rank correlations were applied between functional groups, soil parameters and biodiversity indices.

We also calculated the Ellenberg indicator values for each treatment (Stevens et al. 2010). We only used the most relevant ones for our study: soil fertility (N), soil humidity (F) and soil acidity (R), as these parameters are the main ones driving the community composition in heathlands (Van Landuyt et al. 2008, Mantilla-Contreras et al. 2012). Ellenberg indicator values were calculated as unweighted means of indicator values for species present in each transect using a spreadsheet (Riis-Nielsen 2006). Transect mean values were compared for Nørholm and Randbøl hede, separately.

Vegetation community assemblages and soil parameters were explored by Non-metrical multidimensional scaling (NMDS) analysis. NMDS provided graphical ordination of the community grouping, using the function *metaMDS* in vegan R package. We used the NMDS analysis instead of other ordination technique (i.e. PCA, CA, DCA) since its use its widely extended in ecology to identify patterns among multiple samples and allows the correlation with the environmental variables (Brunbjerg et al. 2015). Bray-Curtis distances were used as a measure of dissimilarity among different treatment areas. The *envfit* function was used to plot the vectors of variables with the assemblage of vegetation communities and identify the main drivers of change. Two dimension graphs were finally presented.

In the case of the habitat quality assessment, the structure assessment from field work was used to calculate a value by granting a scored value (0-100) for each of the parameters. Second, a species bio-indicator index with scored values was used. A scored value was given to each species found in the treatments according to their relevance for the ecosystem. Values from -1 to 7, where -1 is attributed to a negative species (for instance, *Pinus mugo* and *Molinia caerulea*), zero means no relevance and, positive values are positive indicator species. A negative species is a species representing a threat to the habitat, as for instance, an invasive species. The sum of the values was used together with the species richness to calculate the final bio-indicator value. Finally, these two values (structure index and bio-indicator index) were used to calculate the final value with an excel spreadsheet facilitated by Inger Kappel.

5. Results

This chapter is structured in six different sections: soil parameters, vegetation height, biodiversity, coverage of functional groups, vegetation composition and habitat quality. This structure will facilitate the reader to understand the analysis.

5.1. Soil parameters

This section is divided into three subsections. First, a subsection presenting the results for the analysis across locations, where only the areas defined as controls are included in the test. Second, an analysis of treatment effect in Nørholm hede. And last one, an analysis of treatment effect for soil parameters in Randbøl hede.

Differences among controls:

There were significant differences in all the soil parameters among controls of the three studied heathlands (Table 3), except for Nitrogen (N) in the O-horizon (Appendix B: Table B1).

The thickness (cm) of the O-horizon showed very significant difference ($F=13.81$, $p=0.005$) among the locations. A post hoc analysis showed differences between Randbøl and the other locations (Table 3). The lowest values were found in Randbøl and the highest ones in Trehøje hede.

Table 3. The means ± standard errors of the soil variables for each control in the three locations (n=3). Means with different lowercase letters (a, b, c) represent a significant difference between treatments. Ho is referred to organic horizon and 0-10 is referred to the mineral layer.

Soil variable	Nørholm hede	Randbøl hede	Trehøje hede	Differences
Thickness (Ho)	5.3 ± 0.69 (a)	1.8 ± 0.15 (b)	7.3 ± 1.08 (a)	1 = 3 > 2**
% C (0-10)	2.5 ± 0.45 (a)	2.6 ± 0.27 (a)	4.1 ± 0.37 (a)	-
% C (Ho)	27.6 ± 8.10 (a)	19.4 ± 2.63 (a)	44.2 ± 0.32 (b)	1 = 2 < 3**
% N (0-10)	0.103 ± 0.01 (a)	0.160 ± 0.01 (b)	0.100 ± 0.01 (a)	1 = 3 < 2*
% N (Ho)	1.2 ± 0.34 (a)	1.1 ± 0.12 (a)	1.8 ± 0.02 (a)	-
C:N (0-10)	23.6 ± 2.64 (a)	16.3 ± 0.58 (a)	42.6 ± 1.38 (b)	1 = 2 < 3***
C:N (Ho)	22.8 ± 0.39 (a)	17.0 ± 0.58 (b)	25.0 ± 0.43 (c)	2 < 1 < 3***
pH (0-10)	3.08 ± 0.07 (a)	3.9 ± 0.04 (b)	3.0 ± 0.04 (a)	1 = 2 < 3***

Significance is tested with ANOVA (or Kruskal-Wallis for non-normal). Numbers in the last column are referred to location: 1 (Nørholm hede), 2 (Randbøl hede), 3 (Trehøje hede). Pair-wise comparisons (tested with Tukey's HSD test) are given for $P < 0.05$ (*), $P < 0.01$ (**), $P < 0.001$ (***).

The percentage of Carbon (C) in the O-horizon was significantly different (F=6.63, p=0.03) among the locations, with the highest values in Trehøje hede compared to the other two locations (Table 3).

N content in the mineral horizon is generally low and Nørholm and Trehøje had very significantly lower values compared to Randbøl hede (Table 3).

High significant differences were found in the C:N ratio in the O-horizon (F=76.07, p<0.001) and in the mineral horizon (F=59.90, p<0.001). A post hoc analysis showed differences across all locations. Lowest values were found in Randbøl and higher values were found in Trehøje hede (Table 3).

The pH in the mineral horizon showed significant differences (F=86.64, p<0.001) across locations. Randbøl hede showed the highest values (Table 3).

In general, Trehøje hede presented higher N and C concentration, and thickness in the organic horizon in the upper part of the mineral horizon.

Second, we focus in the soil parameters estimated in Nørholm hede:

In Nørholm, significant differences were registered between the different management practices for all the soil parameters studied (Appendix B: Table B3), except by temporary water surfaces and bare ground coverage (Table 4).

Table 4. The effects of treatment on the soil variables in Nørholm hede. Means and standard errors are shown. Means with different lowercase letters (a, b) represent a significant difference between treatments. OMD and LL are the acronyms chosen for the thickness of organic layer and litter layer, respectively.

Soil variable	Control	Cultivated
Moisture (%)	77.9 ± 7.80 (a)	37.1 ± 3.06 (b)
T (ºC)	11.7 ± 0.07 (a)	11.0 ± 0.31 (b)
OMD (cm)	5.1 ± 0.16 (a)	1.5 ± 0.11 (b)
LL (cm)	3.7 ± 0.93 (a)	7.7 ± 1.21 (b)
Bare ground (%)	20.3 ± 2.64 (a)	26.5 ± 4.38 (a)

Difference was tested with t-test (or Mann-Whitney U test for non-normal distribution) and given for P < 0.05.

Significant differences were found in the soil moisture (W=217.5, p<0.05) and soil temperature (W=1072.5, p<0.05) in the cultivated plots. The highest values were found in the control plots (Table 4).

Significant differences (W=342.5, $p<0.05$) were found in the thickness of the organic matter layer and the litter layer (W=4375.5, $p<0.05$). Values were higher in the control plots (Table 4).

There was no significant differences in the cover of bare ground (Table B9). And, temporary water surfaces were only found in the unmanaged (1.9%).

In summary, soil moisture, soil temperature and thickness of organic matter layer showed a general decrease with cultivation, contrary to thickness of litter layer which increased.

Soil parameters of Randbøl hede:

In Randbøl hede, significant differences were found in all the soil parameters studies due to treatment effect in Randbøl hede (Table 5; Appendix B: Table B3).

A very significant difference (F=6.79, p=0.001) was found in soil moisture. A post hoc analysis showed differences among control and treatments (Table 5). Moisture was similar in the treatment, while higher values were found in the control.

Table 5. The effects of treatment on the soil variables in Randbøl hede. Means and standard errors are shown. Means with different lowercase letters (a, b, c) represent a significant difference between treatments and control. OMD and LL are the acronyms chosen for the thickness of organic layer and litter layer, respectively.

Soil variable	Control	Cut	Burnt	Differences
Moisture (%)	27.7 ± 2.95 (a)	21.7 ± 3.02 (b)	23.7 ± 1.78 (b)	1 > 2 = 3**
T (ºC)	15.9 ± 0.72 (a)	18.1± 0.36 (ab)	19.2 ± 0.61 (b)	1 < 3*
OMD (cm)	2.4 ± 0.24 (ab)	2.8 ± 0.36 (b)	2.0 ± 0.19 (a)	2 < 3*
LL (cm)	3.3 ± 0.70 (a)	1.1 ± 0.10 (b)	0.8 ± 0.17 (b)	1 > 2 = 3**
Bare ground (%)	5.2 ± 1.30 (a)	7.4 ± 2.28 (a)	17.70 ± 3.49 (b)	1 = 2 < 3***

Significance due to treatment effect is tested with ANOVA (and Kruskal-Wallis for non parametric tests). Numbers in the last column are referred to location: 1 (Control), 2 (Cut), 3 (Burnt). Pair-wise comparisons (tested with Tukey's HSD test or Nemenyi test) are given for $P < 0.05$ (*), $P < 0.01$ (**), $P < 0.001$ (***).

Soil temperature differed between treatments (W=112.3, $p<0.001$). Values were lower in the control while they were higher in the burnt area (Table 5).

There was significance in the thickness of the organic matter layer (W=12.11, p=0.002). Values were similar in the control and cut, while higher values were found in the burnt (Table 5).

The thickness of the litter layer presented highly significance (W=83.57, $p<0.05$). Higher values were found in the control, while the treatments presented low values.

There was a significant effect in the cover of bare ground ($W=32.86$, $p<0.05$). Low values were found in the control and cut, while the highest values were found in the burnt.

In summary, soil moisture and thickness of litter layer decreased with the application of treatment, while soil temperature experienced an increase. Moreover, the cover of bare ground and the thickness of organic matter layer presented differences in relation to the specific treatment applied (Table 5). There were no temporary water surfaces.

5.2. Vegetation height

Two variables were considered to measure the vegetation height in Nørholm and Randbøl hede : the height of *C. vulgaris* and the height of the rest of plant species.

In Nørholm hede, values in vegetation height were higher in the cultivated area (29.5) in comparison with the control (25.4) ($W=3498$, $p<0.05$). However, *C. vulgaris* was absent in the control and no test was performed.

In Randbøl hede, vegetation height (Table B3) was higher in the control (24.3) in comparison to the treatments (cut: 14.5; burnt: 13.0) ($W= 82.07$, $p<0.05$). *C. vulgaris* height was higher in the treatments (cut: 9.1; burnt: 10.8) than in the control (3.0) ($W= 34.89$, $p<0.05$).

5.3. Biodiversity

This section is divided into four subsections. First, a subsection presenting the results for the analysis of biodiversity across locations, only considering the areas defined as controls. Second, an analysis of treatment effect on biodiversity of Nørholm hede. Third, an analysis for biodiversity in Randbøl hede. And last one, and analysis of Trehøje hede.

In general, biodiversity indices differed between locations (Table 6). Species richness (S) showed significance ($F=9.18$, $p<0.05$) due to location effect. Species richness values were similar Nørholm ($S=6.67$) and Trehøje ($S=6$) while, Randbøl hede showed the highest mean values of S (13.67). Shannon index (H’) and Pielou (J) also showed significance (Appendix B: Table B1). Trehøje hede presented the highest values.

A highly significant difference (F=39.02, p<0.001) was found across locations for Simpson (D) index. A post hoc analysis showed differences among all groups. Higher values were found in Nørholm hede.

Table 6. Means and standard errors for species richness (S), Shannon- Wiener (H'), Pielou (J) and Simpson (D) within controls. Means with different lowercase letters (a, b, c, ab) represent a significant difference among locations.

Biodiversity Index	Nørholm hede	Randbøl hede	Trehøje hede	Differences
S	6.67 ± 1.76 (a)	13.67 ± 1.20 (b)	6.00 ± 1.15 (a)	1 = 3 < 2*
H'	1.10 ± 0.01 (a)	1.13 ± 0.01 (a)	1.44 ± 0.10 (b)	1 = 2 <3*
J	0.63 ± 0.09 (ab)	0.43 ± 0.01 (a)	0.84 ± 0.10 (b)	2 < 3*
D	0.59 ± 0.00 (a)	0.46 ± 0.01 (b)	0.30 ± 0.04 (c)	1 > 2 > 3***

Significance for location effect is tested with ANOVA (or Kruskal-Wallis for non-normal). Numbers in the last column are referred to location: 1 (Nørholm hede), 2 (Randbøl hede), 3 (Trehøje hede). Pair-wise comparisons (tested with Tukey's HSD test or Nemenyi test) are given for $P < 0.05$ (*), $P < 0.001$ (***).

Nørholm hede

Biodiversity indices did not differ between cultivated and control areas (Table 7, B4). On average, there was a tendency for higher values in the cultivated. Considering the variability of species among areas, we realized that only four species were present in both areas (Appendix B: Table B8). There are differences between the species that were found in the cultivated in comparison with the Control. Twelve plant species from the control were not found in the cultivated, although some of them are considered invasive species as *Pinus mugo*. Moreover, seven species were only found in the cultivated.

Table 7. The effects of treatment on the alpha-diversity indices in Nørholm hede are shown in this table. Means and standard errors for species richness (S), Shannon- Wiener (H'), Pielou (J) and Simpson (D). Means with different lowercase letters (a, b) represent a significant difference within management practices. Significance of treatment effect is tested with t-test ($P < 0.05$).

	Control	Cultivated
S	6.67 ± 1.76 (a)	10.33 ± 2.03 (a)
H'	1.10 ± 0.01 (a)	1.09 ± 0.02 (a)
J	0.63 ± 0.09 (a)	0.48 ± 0.05 (a)
D	0.59 ± 0.00 (a)	0.42 ± 0.01 (a)

Randbøl hede

In general, biodiversity indices did not differ between treatments in Randbøl hede, except for D index (Appendix B: Table B4). Higher values of D were found in the control (Table 8). Considering the pool of species in this location, we found 3 species (*Carex arenaria*, *Empetrum nigrum*, *Frangula alnus*) present only in the control and 9 species found only in the treatments (Appendix B: Table B8). In the case of mosses, we did not identify the species. However, no lichens were found in the plots.

Table 8. The effects of treatments on the alpha-diversity indices in Randbøl hede are shown in this table. Means and standard errors for species richness (S), Shannon- Wiener (H'), Pielou (J) and Simpson (D). Means with different lowercase letters (a, b, c) represent a significant difference within management practices.

	Control	Cut	Burnt	Differences
S	13.67 ± 1.20 (a)	13.67 ± 0.33 (a)	12.67 ± 0.33 (a)	-
H'	1.13 ± 0.01 (a)	1.14 ± 0.00 (a)	1.12 ± 0.01 (a)	-
J	0.43 ± 0.01 (a)	0.43 ± 0.00 (a)	0.44 ± 0.01 (a)	-
D	0.46 ± 0.01 (a)	0.31 ± 0.02 (b)	0.33 ± 0.02 (b)	1 > 2 = 3**

Significance difference for treatment effect is tested with ANOVA. Numbers in the last column are referred to treatment: 1 (Control), 2 (Cut), 3 (Burnt). Pair-wise comparisons (tested with Tukey's HSD test) are given for $P < 0.01$ (**).

Trehøje hede

In general, biodiversity indices did not differ among treatments of Trehøje hede, except for H' and D (Table 9, B4). Considering the variability of species in this heath, Rita Buttenschøn found 4 plant species in the control (including *Quercus robur* and *Picea abies*) but were not present in the treatments. Moreover, eleven species were present in one or two treatments, but not in the treatment (Buttenschøn, unpublished data).

Table 9. The effects of treatments on the biodiversity indices in Trehøje hede are shown in this table. Means and standard errors for species richness (S), Shannon- Wiener (H'), Pielou (J) and Simpson (D). Means with different lowercase letters (a, b, c) represent a significant difference among locations.

Biodiversity Index	Control	Cut	Burnt	Differences
S	6.00 ± 1.15 (a)	9.67 ± 1.45 (a)	9.33 ± 0.33 (a)	-
H'	1.44 ± 0.10 (a)	1.84 ± 0.08 (b)	1.70 ± 0.01 (ab)	1 < 2*
J	0.84 ± 0.10 (a)	0.82 ± 0.03 (a)	0.78 ± 0.01 (a)	-
D	0.30 ± 0.04 (a)	0.18 ± 0.01 (b)	0.20 ± 0.02 (ab)	2 < 1*

Difference among treatments obtained with ANOVA. Numbers in the last column are referred to treatment: 1 (Control), 2 (Cut), 3 (Burnt). Pair-wise comparisons (tested with Tukey's HSD test) are given for $P < 0.05$ (*).

5.4. Coverage of functional types

This section is divided into five subsections. First, a subsection presenting the results for the analysis across locations, where only the areas defined as controls are included in the test. Second, an analysis of treatment effect in the coverage of functional types in Nørholm hede. Third, an analysis of treatment effect in the coverage of functional types in Randbøl hede. The last two sections are focused in Ellenberg values and correlations.

Differences in coverage of functional types across controls:

In general, significant differences have been found across locations (Table B1). High significant differences have been found within the coverage of *C. vulgaris* and *E. nigrum* (Table B1).

There was found a very significant difference (F=14.47, p<0.01) in the coverage of *other dwarf shrubs* and *D. flexuosa* (W=15.26, p<0.01). Trehøje hede presented the highest values of other dwarf shrubs and Randbøl hede presented the highest values respect the other two locations (Table 10, B7).

No significative differences were found in the coverage of *M. caerulea*, the *other grasses* and the mosses (Table B1).

The cover of lichens and trees functional groups did not differ between controls (Table B1). Shrubs were absent in the plots established during field work (Table 10).

In general, most of the functional groups presented the highest values in the control of Randbøl hede.

Table 10. Table. Means and standard errors for the coverage of functional groups in the controls of each heathland. Means with different lowercase letters (a, b, c) represent a significant difference between treatments.

Functional types	Nørholm hede	Randbøl hede	Trehøje hede	Differences
C. vulgaris	0.0 ± 0.00 (a)	3.2 ± 0.17 (b)	23.2 ± 0.16 (c)	1 < 2 < 3***
E. nigrum	38.2 ± 0.61 (a)	19.4 ± 0.36 (b)	9.5 ± 0.48 (b)	1 > 2 =3***
Otero dwarf shrubs	2.3 ± 0.46 (a)	2.8 ± 0.25 (a)	17.1 ± 0.56 (b)	1 = 2 > 3**
D. flexuosa	0.0 ± 0.00 (a)	25.6 ± 1.23 (b)	2.3 ± 0.25 (a)	1 = 3 < 2**
M. caerulea	0.9 ± 0.19 (a)	4.8 ± 0.59 (a)	0.4 ± 0.08 (a)	-
Other grasses	0.5 ± 0.10 (a)	1.8 ± 0.15 (a)	0.7 ± 0.08 (a)	-
Forbs	0.0 ± 0.01 (a)	16.4 ± 0.78 (b)	1.1 ± 0.13 (a)	1 = 3 < 2**
Moss	45.0 ± 0.62 (a)	53.9 ± 1.44 (a)	44.0 ± 1.02 (a)	-
Trees	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	0.7 ± 0.15 (b)	1 = 2 < 3*
Lichens	0.4 ± 0.00 (a)	0.0 ± 0.00 (a)	0.8 ± 0.16 (a)	-
Total cover	87.6 ± 0.82 (a)	128.0 ± 1.47 (b)	100.0 ± 0.00 (c)	1 < 3 < 2**

Significance for location effect is tested with ANOVA (or Kruskal-Wallis for non-normal). Numbers in the last column are referred to location: 1 (Nørholm hede), 2 (Randbøl hede), 3 (Trehøje hede). Pair-wise comparisons (tested with Tukey's HSD test or Nemenyi test) are given for P < 0.05 (*), P < 0.001 (***).

Nørholm hede:

No significant difference was found in the total cover of vegetation (t=-1.26, p=0.21) within the plots due to treatment effect in Nørholm hede (Table B2). However, there were found significant differences in functional groups among the cultivated plots and the control ones in Nørholm (Table B2), except for the functional type's trees, lichens and shrubs (Fig. 8).

There was a significant difference (Table B2) in the coverage of *C. vulgaris and E. nigrum. C.vulgaris* presented the highest values in the cultivated, while *E. nigrum* presented higher values in the control plots (Table B5).

There were not found significant differences (W=2838.5, p=0.832) within *other dwarf shrubs*. Higher values were registered in the control (Fig. 8).

Significant differences were found in *D. flexuosa* (W=5250, p<0.05) and M. caerulea (W=2397.5, p=0.0014). *D. flexuosa* presented higher values in the cultivated, while *M. caerulea* presented slightly higher values in the control.

Significant differences were found in *other grasses* (W=5175, p<0.05) and forbs (W=5466, p<0.05). Values were higher in the cultivated plots (Fig. 8).

Significant differences were found in the coverage of mosses (W=2150.5, p=0.0128) and lichens (W=2400, P=0.0039). Higher values were found in the control area (Table B5).

In summary, there was an increase of forbs, *D. flexuosa*, *other grasses* and *C. vulgaris* in the cultivated area. By the contrary, *M. caerulea, E. nigrum* and mosses experienced a decrease in the cultivated.

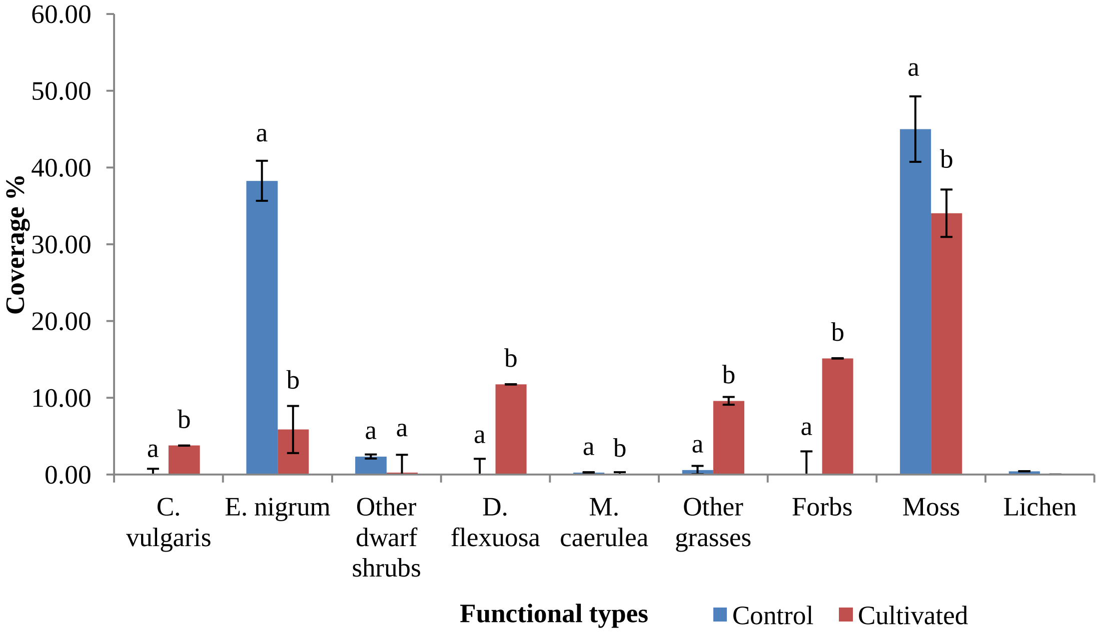

Fig. 8. The effects of treatment on the vegetation functional groups and comparison with the control in Nørholm hede. Error bars represent the SE of the means. Means with different letters (a, b) represent significant difference between treatments. The horizontal axis represents the functional groups and the vertical axis the coverage.

Randbøl hede

The coverage of many functional types presented significant differences among treatments (Fig. 9). Moreover, there were found high significant differences in the total vegetation coverage of the plots ($F=24.693$, $p<0.001$) among treatments. Higher values of total vegetation coverage were found in the control (Appendix B: Table B6).

A high significant difference was found in the percentage of coverage of *C. vulgaris*, *E. nigrum* and other dwarf shrubs (Appendix B: Table B2). A post hoc analysis showed significant differences among control and treatments (Fig. 9). Values were higher for *C. vulgaris and other dwarf shrubs* and lower *E. nigrum* for the treatments (Appendix B: Table B6).

There was not found significant difference ($W=0.878$, $p=0.645$) in the coverage of *D. flexuosa* among the control and the treatments (Appendix B: Table B6). There was a decrease of coverage in the treatments with respect of the control.

A significant difference ($W=28.78$, $p<0.05$) was found in the cover of *M. caerulea*. There was an increase of coverage in the treatments in comparison to the control.

High significant difference was found in *other grasses* due to treatment effect (Appendix B: Table B6). There was an increase of coverage of *other grasses* in the treatments (Fig. 9).

Very significance difference (W=12.08, p=0.002) was found in forbs among sites. A general decrease of coverage of forbs was registered due to the application of treatments (Appendix B: Table B6).

High significant differences (W=44.31, p<0.001) were registered in mosses across areas. A post hoc analysis showed differences among the areas with the lower values in the treatments (Appendix B: Table B6).

In summary, there was an increase of dwarf shrubs and grasses, except for *Empetrum nigrum* and *Deschampsia flexuosa*, which showed a general decrease. Forbs, mosses and the total vegetation cover also showed a decrease with the application of treatments. No lichens were found in the plots of this heath (Appendix B: Table B6).

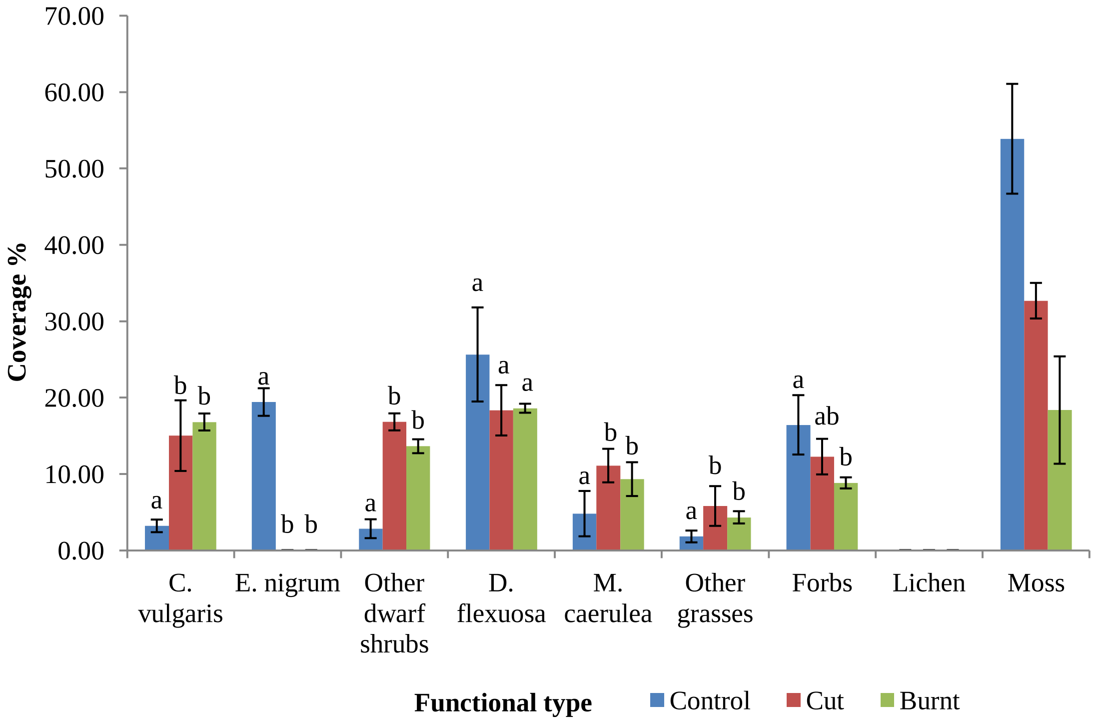

Fig. 9. The effects of treatments on the functional groups and comparison with the control in Randbøl hede. Error bars represent the SE of the means. Means with different letters (a, b, c) represent significant difference among management practices (legend). The horizontal axis represents the functional groups and the vertical axis the coverage.

Ellenberg indices

In Nørholm, a maximum value of 2.37 was registered for N in the unmanaged and a minimum value of 2.27 in the cultivated (Appendix C: Table C1, C3). R values ranged from 3.35 in the control to 2.52 in the cultivated. A maximum value of 5.67 was found in the F, while a minimum value of 5.35 was registered in the cultivated.

In Randbøl hede, N values ranged from 2.72 in the control to 2.66 in the treatments (Appendix C: Table C2-3). R values ranged from 2.91 in the control to 3.45 in the cut treatment. And, F ranged from 5.38 in the control to 5.67 in the burnt.

Correlations

Spearman rank correlations were applied to the soil variables in relation to functional groups and biodiversity indices. Low to moderate significant correlations were found for some of the variables (Appendix B: Table B9-10).

In Nørholm, a positive correlation was found between the thickness of the organic horizon and *E. nigrum* (r=0,57), and negative correlations between the thickness of organic horizon with *D. flexuosa* (r=-0.63) and other grasses (-0.67) and forbs (r=-0.67). The presence of mosses was negatively correlated (r=-0.76) with bare ground. Moreover, soil moisture was negatively correlated with *D.flexuosa* (r=-0.68), other grasses (r=-0.66) and forbs (r=-0.65) and, positively correlated with *E. nigrum* (r=0.6). These correlations were calculated across treatments. I believe that results can vary considering only one area in Nørholm or only the treatments in Randbøl hede.

In Randbøl hede, mosses presented a negative correlation with bare ground (r=-0.69) and a positive correlation with the thickness of litter layer (r=0.44).

5.5. Vegetation composition community

The NMDS across locations shows differences in the community composition regarding the *locations* and *treatments* (Fig. 10). There is a clearly separations between different treatments and, therefore, there will be 3-5 clusters.

In relation to the environmental factors, the NMSD ordination plot shows a compositional change driven by higher pH values in the mineral layer. In addition, higher thickness of the organic layer, C concentration in the organic layer, C:N ratio in the organic layer and C:N ratio in the mineral layer are driving the community in opposite direction.

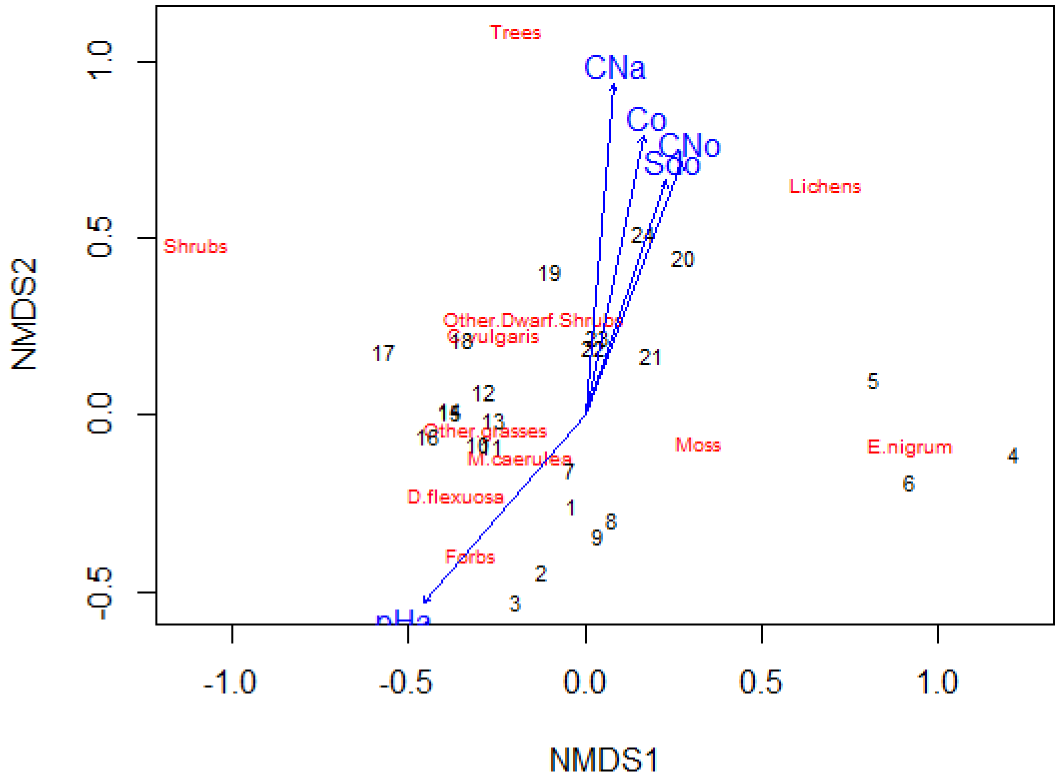

Fig. 10. Non-metric multidimensional scaling (NMDS) ordination plot of the vegetation communities in the three heathlands. Plant communities are presented in black, species and functional types in red and, environmental factors in blue. The communities from Nørholm hede are as follow: from 1 to 3 correspond to the *cultivated* and, from 4 to 6 to the *unmanaged*. The communities from Randbøl hede are as follow: from 7 to 9 the *control*; from 10 to 12 the *cut* and, from 13 to 15 correspond to the *burnt*. The communities from Trehøje hede are as follow: from 16 to 18 the *burnt*, from 19 to 21 the *control* and, from 22 to 24 the *cut*. Environmental factors names have been abbreviated for graphical purposes (pHa= pH in the mineral layer, Sdo = Thickness of the organic layer, CNa = C:N in the mineral layer, CNo= C:N in the organic layer, Co= Carbon in the organic layer).

The NMDS for Nørholm hede showed differences in the community composition in respect of the applied treatments (Fig. 11). There are two defined clusters, previously specified. The seventy-five plots from the cultivated area clustered together in a group and the 75 plots from the unmanaged area clustered in another group. The communities that developed in the treatment and the control advanced in opposite directions.

In relation to the environmental factors, the NMDS shows a compositional change forced by one factor comprising higher thickness of organic layer, driving the community in one direction. Furthermore, the thickness of litter layer and bare soil appear to be driving the community to some degree in the opposite direction (with certain deviation).

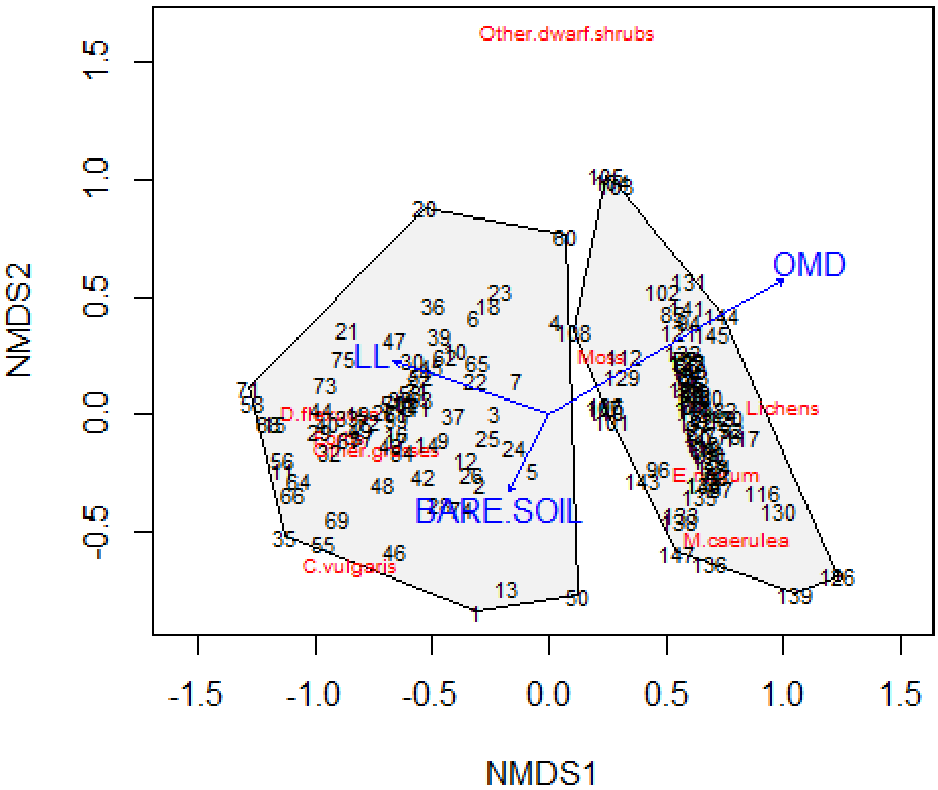

Fig. 11. Non-metric multidimensional scaling (NMDS) ordination plot of the vegetation communities in Nørholm hede. Plant communities are presented in black, species and functional types in red and, environmental factors in blue. Communities from 1 to 75 correspond to the cultivated; and, communities from 76 to 150 correspond to the unmanaged area. Environmental factors names have been abbreviated for graphical purposes (BARE.SOIL=Bare Soil, LL = thickness of Litter Layer, OMD = thickness of organic matter layer,).

5.6. Habitat quality

The habitat quality of the three heathlands presented a bad status (Appendix D). However, there was found a tendency for a slight improvement on the habitat quality after the application of management practices (Table 11). A minimum value of 0.26 was found in the control of Randbøl hede and a maximum value of 0.38 in the burnt treatment. Moreover, values in Trehøje hede ranged from 0.22 in the control to 0.26 in the treatments. In the case of Nørholm, a minimum value of 0.19 was found in the unmanaged area and a maximum value of 0.22 in the cultivated.

6. Discussion

This chapter is divided into five different sections with the intention to prove the two hypotheses. These sections are: soil parameters, biodiversity, vegetation height, vegetation composition and habitat quality.

My aims were to assess whether current management practices for Danish heathlands and the habitat quality assessment developed by the Danish Nature Agency are likely to remain fully appropriate under future scenarios, where heathlands are preserved and their conservation status improved.

This study compared the management practices of three Danish heathlands and their effects on the vegetation, soil and conservation status. As mentioned before, heathlands are no longer exploited by shepherds and farmers. Together with the increasing atmospheric deposition of N, the lack of management has become a problem, leading to a shift from *C. vulgaris* dominated heathland towards a grassland dominated by *D. flexuosa* and an encroachment of trees, respectively (Niemeyer et al. 2005a). Nowadays, heathlands conservation is the responsibility of public administrations and environmental agencies.

Our study found a number of similarities but also key differences, which may have important implications for designing appropriate habitat quality assessments and, as a consequence, effective biodiversity conservation strategies.

The coming sections are focused to demonstrate my first hypotheses: The application of management practices enhances diversity of functional plant groups.

Differences among locations highlight the relevance of local context (f.e. historical, biophysical and temporal context) and the requirement to adapt the biodiversity conservation measures to the local conditions (Velle and Vandvik 2014). For example, Nørholm hede is remarkable because its natural succession is unique in Denmark.

The differences on vegetation communities across controls are shown in the Figure 12. Besides the treatments which have recently been applied (Table 1, 2) in Randbøl and Trehøje, we can already appreciate significative tendencies (Appendix B: Table B3). In general, the application of management practices to heathlands increases the diversity and coverage of functional groups. For instance, the coverage of *other grasses* experienced a general increase in the treatments respect of their controls. The presence of lichens was found in only one treatment, the *cut* in Trehøje hede. We established our transect in areas without trees and shrubs, but none of these groups were found in the treatments of Randbøl hede. This is in contrast to Gimingham (1992), who concluded that prescribed burning promotes the growth of new trees.

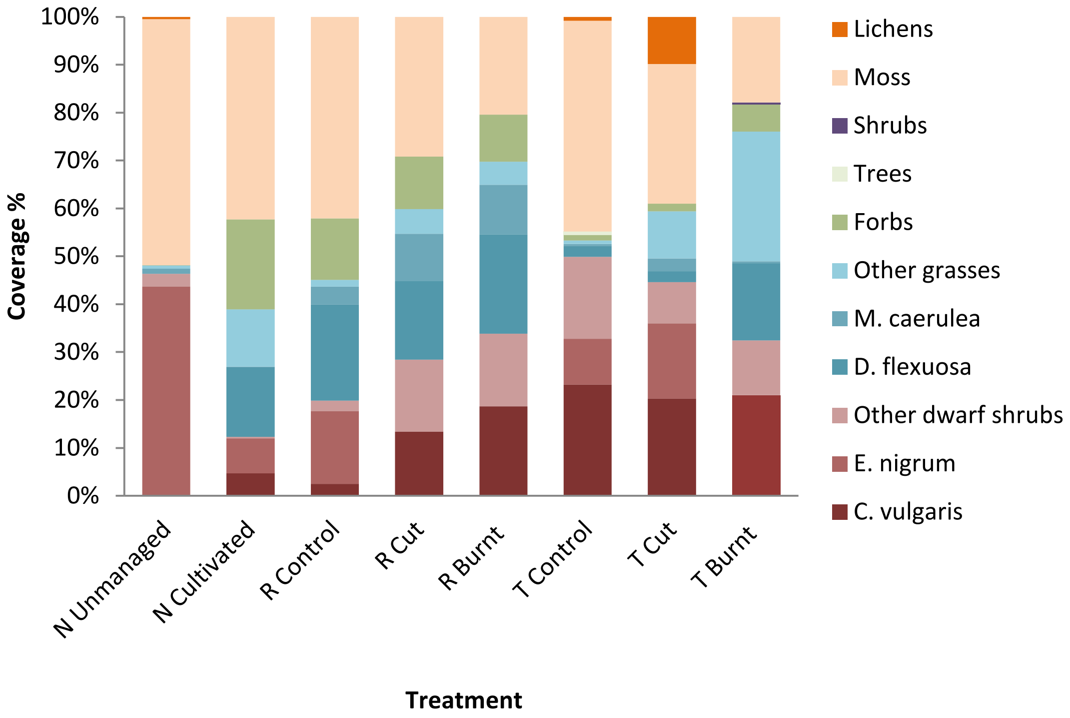

Fig. 12. Coverage of functional groups for each treatment and control across locations. N, R and T are the acronyms used for the name of the heathlands: Nørholm, Randbøl hede and Trehøje hede. Horizontal axis represents the different treatments and, vertical axis represents the coverage (100%).

The change in the vegetation is also the consequence of removal of the litter and organic layers by cutting and burning. This removal (specially caused by prescribed burning) changes the availability of nutrients in the humus horizons. However, a limited nutrient uptake by plants after burning increases the leaching and loss of many nutrients as N, Ca, K, and Mg (Mohamed et al. 2007).

6.1. Soil parameters

In general, there are some differences in soil parameters among the controls of each heathland, which might be explained by different reasons. There were no differences in N content in any of the layers neither in Carbon content of the mineral layer due to location effect. However, if we consider each heathland and their management practices, there are some differences.

In the case of Randbøl hede, as we hypothesized, the total cover of functional groups has been reduced due to the application of management practices, which increases the exposure of soil to the sun. As a consequence, we would expect an increase in the soil temperature and a reduction of the soil moisture. Different studies consider moisture and vegetation encroachment (or shadow) as main environmental factors driving spiders and carabid beetles communities as well as promoters of potential habitats for plant species (Gimona and Birnie 2002, Schirmel and Buchholz 2011). However, these two factors, soil moisture and temperature, were not taken into consideration in our analysis because the data was collected only along one week field work in July. In order to find significant results, soil moisture and temperature data should be collected different times along the year. Although, the thickness of O-horizon is a good indicator of higher moist conditions.

Further, as expected, the thickness of litter and organic layers have generally been reduced with the application of treatments, which means a reduction in the content of nutrient pools from the system (Frouz et al. 2009). Topsoil removal is likely to have major effects on vegetation and soil communities (Kardol et al. 2009). A study conducted by Green et al. (2013) demonstrated that soil microbial communities experienced a high dynamic change with the application of fire and, therefore, rapidly re-established by different species. Soil communities (bacteria and fungi species) affect to the vegetation communities by the establishment of specific relationships, like symbiosis.

A higher thickness of the organic layer was found in the *cut* of Randbøl hede (Table) in comparison to *burnt* and *control*. This could indicate that the organic material was deposited after the cutting process due to partial removal of vegetation (Kopittke et al. 2013). As a consequence of the removal of vegetation, an increase in the leaching process is expected (due to a reduced nutrient uptake, percolation and increased evaporation), contributing to the removal of nutrients from the system (Hardtle et al. 2006). It is likely that prescribed burning will remove low quantities of K, Ca and Mg. This could be because of the high levels of these nutrients in the ash (Hardtle et al. 2006). In contrast, cutting promotes higher removal of these nutrients. Yet, it is likely that elevated N leaching will take place over a year or more after application of prescribed burning (Niemeyer et al. 2005a). Nonetheless, the quantities of N removed by cutting and prescribed burning were only able to compensate for a short period of time in a study conducted by Hardtle et al. (2006). It is of special interest in relation to N and P budgets. Strong effects of management have been showed in P budgets, causing a shift from N to P limitation of plant growth (Hardtle et al. 2007).

As a consequence of the management practices, there has been an increase of bare ground. Bare ground is a potential habitat for the establishment of new plant individuals through sprouting from stem base and germination (i.e. *Calluna vulgaris*) as well as habitat for varied life stages of some invertebrates (Davies et al. 2010a).

Temporary water systems were recorded in the survey, however, we did not find any, except in the unmanaged area of Nørholm hede. I believe that the presence of this system is a factor influenced by seasonality as well as soil structure (i.e. iron-pan). Therefore, their absence during our field work does not necessarily mean that they are absent all the year around. In fact, the intact iron pan in Nørholm hede blocks the percolation of water.

6.2. Biodiversity

In general, the majority of biodiversity indices were not significant among the treatments and the control in any of the locations. However, if we consider beta diversity, we can appreciate that some species growing in areas with treatments are not present in areas without management (Appendix: Table B4). For instance *Empetrum nigrum* and *Dryopteris dilatata* are only found in the control of Randbøl hede, but not in its treatments. In Trehøje hede, *Salix repens*, *Trientalis europaea* and *Vaccinium uliginosum* are only present in the *cut* treatment for that location. The fact of having a heathland with different management practices and areas with no management is relevant because there is variability in species requirements (Anderson et al. 2006), and therefore, an increase of habitats availability for species.

It is widely accepted that heathland requires some degree of disturbance to reduce the effects of N deposition, avoid natural succession and maintain heathland structure (Power et al. 2001). Beneficial effects of disturbances are at landscape level where a combination of practices can occur in a long time span and moderate disturbance (Niemeyer et al. 2005a, Hardtle et al. 2006, Hardtle et al. 2007). However, a recent study showed that Nørholm hede still maintains characteristics of heathlands and there is low tree and shrub encroachment (Kepfer-Rojas et al. 2014).

Roem and Berendse (2000) have acknowledged that there is a relation between plant species diversity and soil acidity. Roem et al. (2002) points that germination of several heathland species was reduced when pH values were below 5. This author also indicates that the concentration of aluminium in the upper soil determines species richness because this element influences the process of acidification. In our study, there was a linear regression between these two parameters of $r^2 = 0.658$ (Fig B1). There is a tendency for lower values in the controls, even though if the area has not been managed for 8 years, there is a difference.

Thus, we cannot conclude that the application of disturbances increases plant species richness in the case of Trehøje hede. Nevertheless, species richness is not necessarily a good indicator of the conservation status of heathlands (Fagundez 2013, Brunbjerg et al. 2015). For that reason, it is also relevant to study the habitat structure, which includes vegetation height and vegetation composition.

6.3. Vegetation height

The vegetation height has changed due to the application of treatments. Vegetation height experienced an increase in the cultivated area of Nørholm. This could be due to the presence of functional groups as grasses. In the case of Randbøl hede, the vegetation height has been reduced.

The height of *C. vulgaris* presented higher values in the treatments of Randbøl hede, compared to the control. This could be because the control presented young and scattered individuals of this species.

An increase in vegetation height lead to lower surface temperatures, which may prolong development time of grasshopper eggs and juveniles with potential lethal effects (Borchard et al. 2013). We may also expect a change in arthropods species composition driven by vegetation height (Moranz et al. 2012). In overall, having patches of heathlands where the vegetation height varies, results in habitat heterogeneity (Oberndorfer and Lundholm 2009).

6.4. Vegetation composition

Based on coverage values, as well as on their position within the NMDS plots, most of functional groups are favoured due to the application of treatments. The only group that it seems to decrease after the application of treatments is mosses.

First of all, I discuss the possible clusters that can be formed due to a treatment effect (Fig. 10). A clear cluster is formed by the Nørholm hede' *control* (communities: 4-6; Fig. 10). A second clear cluster is composed by the *cultivated* in Nørholm and the *control* in Randbøl (communities: 1-3 and 7-9; Fig. 10). A third cluster is represented by the *burnt* treatments (communities: 13-18; Fig. 10). And the last cluster is formed with the *cut* treatments, including the *control* of Trehøje (communities: 10-12, 19-24; Fig. 10). Although, the last cluster is not very clear because the *cut* (10-12) from Randbøl has similarities with the *cut* (22-24) from Trehøje and, the *burnt* (13-15) from Randbøl. A cluster analysis is recommended to find out what are the clusters formed due to treatment effect and not in relation to history of management.

Empetrum nigrum experienced a decrease in coverage along with the application of disturbance, as well as seen in the position of their NMDS plots. This fact is supported by Ransijn et al. (2015b), who indicated that this species might be favoured by the absence of management. Moreover, as found in the correlations in Table B6-7 (Appendix B), *E. nigrum* hampers the development of different species as forbs and grasses, which may explain the low species number at Nørholm control where the cover of *E. nigrum* is high.

Calluna vulgaris experienced an increase in the treatments, except for the case of Trehøje hede, which may take a few years more to experience a higher encroachment. As seen in field work in the treatments, there were many young shouts, which mean that the application of treatments has improved the status of the species, and there is a continuity of the heathland community.

Based on the coverage of *other dwarf shrub* group and its position in the NMDS plot, this group also experienced a slight increase along with disturbances (Fig. 10).

Molinia caerulea experienced an increase in Randbøl hede in the treatments (Fig. 9). Different studies concluded that fire benefits the spread of *M. caerulea* (Brys et al. 2005a). Hardtle et al. (2006) found in a study conducted in Northern Germany that low-intensity management can not compensate for atmospheric N loads in the long term. In sense with that, species as *M. caerulea* would be favoured by P-limited conditions. In our study, this finding could be related to the limitations of P in the soil of the control (or unmanaged) areas.

Deschampsia flexuosa experienced an increase in the treatments of Nørholm hede and Trehøje hede. In the case of Nørholm, D. flexuosa is favoured in cultivated plots due to the high content of P, even 100 years after the cessation of farming activities (Evelien 2012). Besides there were no significant differences in the abundance of *D. flexuosa* in Randbøl hede due to a treatment effect, we can appreciate a pattern of reduction. As indicated in the correlations (Appendix B: Table B6-7), there is a negative relation (r=-0,23) between *D. flexuosa* and *C. vulgaris*. This is in accordance with Ransijn et al. (2015a), who suggested a competitive interaction between both species. After a disturbance, *D. flexuosa* is able to take advantage of the new openings in the canopy of *C. vulgaris*, increasing quickly its abundance. However, years later *C. vulgaris* could dominate again (Niemeyer et al. 2005a). This could be the reason for Trehøje hede, where the breakup of the shrub canopy promoted the germination of *D. flexuosa*.

The coverage of *other grasses* group experienced an increase in the treatments and together with the position in the NMDS plots, it seems that this functional group is favoured by the application of disturbances. This fact is also supported by different studies, who indicate that grasses are favoured by disturbance from intensified management, fire or heather beetle attacks (Power et al. 1998, Brys et al. 2005a, Fagundez 2013). Those factors can break up the canopy of shrubs and promote the germination of grasses (Terry et al. 2004). This is because the species included in this group (Appendix B: Table B4) are specific of heathlands and, therefore, adapted to nutrient-poor soils and they will experience a slow-growing process. However, *D. flexuosa* and *M. caerulea* are favoured by high nutrients content and are fast-growing species.

Natural succession tends to trees and shrubs encroachment and, therefore, the formation of forests (Gimingham 1992). Different authors concluded that prescribed burning promotes the development of trees (Borghesio 2009, Velle et al. 2012). In accordance with these authors, trees and shrubs might colonize the areas in few years. However, Randbøl has been under periodic management and I did not find any trees in the treatments. However, not all the tree species have positive impacts. *Picea abies* and *Pinus mugo*, and the shrub *Cytisus scoparius* were identified in some of the heathlands, although not inside the plots. These species are considered as having a negative impact and indicator species of low habitat quality in heaths ecosystems and, moreover, might change ecosystem functions and alter the structure (Fagundez 2013). For instance, *Cytisus scoparius* is a legume and, therefore, fixes nitrogen from the atmosphere in the soil (Watt et al. 2003).

Another invasive species found was the moss *Campylopus introflexus*. Mosses and lichens were not determined to species level due to lack of knowledge. Thus, only coverage of mosses was recorded. On one hand, the presence of mosses might hamper the germination by seeds (Davies et al. 2010a) and, therefore, a decrease in mosses coverage is seen as positive effect. On the other hand, lichens were scarce or not found in the plots, which is in agreement with the results of the NMDS plots. This could be because disturbances contribute to the extinction of the species in the area (Boch et al. 2015) and the establishment of lichens could take a few years to develop again.

In accordance with the above results, the NMDS plots revealed that lower thickness of organic layer and litter layer are factors promoting a richer vegetation community. The removal of organic and litter layer produce a reduction of N availability in the soil and an increase in the leaching of nutrients due to fire (Niemeyer et al. 2005a). In contrast to the case of N, small proportions of P, Ca, Mg and K from the aboveground biomass pool are expected to be lost in the smoke and, therefore, retained in the ash (Niemeyer et al. 2005a). The role of nutrients on the species composition is complicated and the species composition depends on many other factors, as growth phase of dominant species and stress (Calvo et al. 2005). As a consequence of the leaching, there is a reduction in the nutrients uptake by plants, which may hamper the colonization and growth of some species in the first years (Mohamed et al. 2007). These conditions promote the development and establishment of *C.* vulgaris (Sedlakova and Chytry 1999, Nilsen et al. 2005).

6.5. Habitat quality

The second of our aims was to evaluate the habitat quality of these heathlands after the application of treatments. Thus, our hypotheses was that there is an improvement of habitat quality due to treatment effect and herein, I present a discussion on the obtained results. The overall habitat quality (Table 11) is in accordance with the last national conservation status assessment conducted by the Danish government (Danish Nature

Agency 2014), which indicates an unfavourable-bad quality status. Comparing to another European country as UK, we realized that the habitat quality status is similar (JNCC 2013; JNCC 2012). And a study at European level indicates that no more than 20% of heaths are under favourable status (European Commission 2015; Maes 2013). However, the EU Commission (2015) indicates that the general trend for this habitat is an improvement of conservation status if appropriate management is targeted.

Table 11. Habitat quality and structure index measured for each treatment and location following the methodology established by the Danish Nature Agency. Supplementary data available in Appendix C.

Location	Treatment	Habitat quality	Structure index
Nørholm hede	Unmanaged	0.19	0.41
Nørholm hede	Cultivated	0.22	0.45
Randbøl hede	Control	0.26	0.50
Randbøl hede	Cut	0.34	0.70
Randbøl hede	Burnt	0.38	0.79
Trehøje hede	Control	0.22	0.51
Trehøje hede	Cut	0.26	0.54
Trehøje hede	Burnt	0.26	0.55

In general, the application of different management practices to a particular heathland brings into it some degree of disturbance, creating niches for specific species adapted to heathlands (Garcia et al. 2009, Schirmel et al. 2011). Nonetheless, invertebrate responses to fire are conflicting in the literature (Buchholz et al. 2013). Invertebrates may take a few years to colonize new areas (Bargmann et al. 2015). Another factor to consider is that the habitat preference of specific species may change along with life cycle stage (Wunsch et al. 2012), and for that reason, mosaic management is also recommended.

According to Webb et al. 2010, a heathland with diverse structures is a heathland capable to host populations of different animal species, species that are to high degree directly dependent and adapted to heath conditions, as seen in figure 13. In that study, 133 animal species are associated with heathlands, from which 60% are invertebrates. Webb recognized a variety of niche requirements for these species: shelter, bare ground, grasslands, dwarf-shrubs, seasonal water bodies, scrub/trees and a mosaic of different vegetation patches. They also calculated the percentage of each species that requires each particular habitat in any of their life phases. However, there is no such as study for Danish heaths.

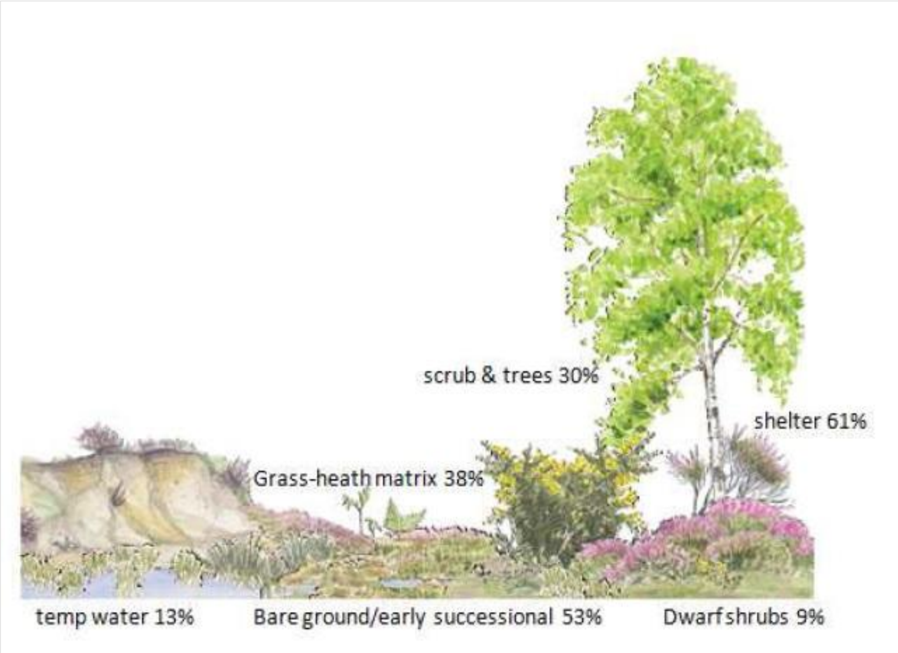

Fig. 13. This figure shows the habitat requirements of 133 species associated to heathlands in England. Many species are depending on several habitats as their life phases may develop in different habitats. Illustrated by Isabel Alonso/Natural England published in Flora og Fauna by Buttenschøn and Schmidt (2015).

Far from the habitat requirements presented by Webb et al (2010), the current management of heathlands does not create a matrix of different habitats. In contrast, they are focus on the reduction of N content from the system and promote the presence of plant heath specialists (Hardtle et al. 2006). Still, the conservation status of Brittish heaths are unfavourable or bad (JNCC 2012). Current practices as prescribed burning often destroy all the trees and scrubs (Ascoli et al. 2013), a potential habitat for endangered species as invertebrates and birds (Webb et al. 2010).

The use of this assessment during our field work and the consideration of the research by Webb et al (2010) open up for a question: is this assessment appropriate for the analysis of conservation status if there is a lack of information on how the management practices affect to the danish fauna assemblages?

The assessment developed by the Danish Nature Agency to evaluate the habitat quality of heathlands (habitat type 4030) protected under the Habitat Directive considers a scored index including information about vegetation coverage, the presence of positive and negative structures, details about management practices and an index based on vascular plants (Appendix D). However, this habitat quality assessment does not acknowledge the relevance of different functional groups, as for example, mosses and soil microorganisms. Mosses often cover the soil and surface of shrubs and trees and, moreover, it is a potential habitat for invertebrates, diminish water evaporation and soil erosion (Gimona and Birnie 2002). Another functional group disregarded in this assessment is the soil microorganisms (i.e. bacteria and fungi), which has been identified as playing a valuable role in nutrient cycling and decomposition (Emmett et al. 2004).

Management of heathlands is mainly focused in the removal of N, in an attempt to counteract the negative effect of eutrophication on *Calluna vulgaris* (Power et al. 2001), Per contra, only 9% of the animal species are associated with dwarf shrubs in British heahtlands. Shelters (i.e. woodland edge) could host up to 61% of the animal diversity as well as bare ground and early succession stages could host 53% of the 133 species. Further, temporary water systems are habitats for 13% of species. There were found temporary water systems in Nørholm hede. We should also consider that the field work was carried out between May and July in different locations. There could be temporary water systems in autumn or winter seasons. Scrubs and trees can host about a third of the species (Webb et al. 2010). Some species of trees and scrubs were found in the areas of study.

A study conducted by Haysom and Coulson (1998), concluded that Lepidoptera species are directly depending on *Calluna* height, which means that the species richness of this group increases along with the growth of *C. vulgaris*. Moreover, some Lepidoptera species have their larvae phase in the litter layer, which is reduced or absent after the application of prescribed burning and cutting. Nonetheless, whether the diversity of other groups shows patterns comparable to that of plants after the application of burning or cutting is uncertain (Schirmel et al. 2011). For example, a study conducted in Norway by Bargmann et al. (2015), shows that carabids richness increases with time after prescribed fire and there is spatial dependence between patches in species composition, although only two species were specialist from heathlands. Another study conducted in Germany shows that Orthoptera species have special requirements for bare soil to ovoposite (Borchard et al. 2013). Therefore, burning should be applied at small-patch scale and rotational-based to allow the development and growth of *C. vulgaris* and the colonization of disturbed areas by invertebrates.

Different studies conducted by Schrimel et al. (2011) in coastal heathlands of Northern Germany found dissimilar results for the habitat requirements of various invertebrate groups. By one hand, species richness of Orthoptera was higher in grassy heath than in dwarf-shrub heath. This might be explained by microclimate conditions, as the broad food availability in that stage of succession is higher, in which there is high plant biomass (Schirmel et al. 2011). By the other hand, carabid and spiders presented lower values of species richness in the grassy-heath stage in a similar research (Schirmel and Buchholz 2011). Thus, different stages of heath succession offer habitat to varied endangered and specialized species (Buchholz et al. 2013).

The season of fire application is also of relevance in relation to the life cycle stage of invertebrates. For example, if fire is applied when a protected invertebrate is under the larvae stage, it’s likely that the population during first years will be low or scarce unless it is able to disperse easily and colonize the disturbed area (Moranz et al. 2012).

Although low, our results suggest positive effects on the habitat quality of Danish heathlands after the application of management practices. Nevertheless, due to the complexities of ecosystems functioning, we advise a carefully interpretation of our findings.

In overall, the Danish heathlands studied herein do not accomplish with the habitat requirements of all species (Danish Nature Agency 2014). Thus, these heaths are not able to host the potential fauna associated to heathlands (Webb et al. 2010). As a consequence, the habitat quality of these Natura2000 areas (Nørholm and Randbøl) can be improved. An improvement could be achieved by considering the ES framework (Eastwood et al. 2016; Moran-Ordonez et al. 2013). A research recently performed in UK showed that there is space for trade-offs between biodiversity and ecosystem services in lowland heathlands as a result of a landscape-scale approach (Cordingley et al. 2016). Therefore, targeting management interventions on different aspects might integrate conservation and economic development (Carboni et al. 2015).

We suggest the use of an indicator framework for the assessment of ecosystem services developed by Maes et al. (2016). Combining data has a high degree of complexity, which can hinder the assessment. Still, simplifying this index could lead to a wrong conclusion.

Herein, I suggest the integration of data from the habitat requirements of fauna with vegetation structure, in order to create a new habitat quality assessment more appropriated for the evaluation of Danish heathlands.

In general, a heathland which combines areas with different treatments and stages of succession will be an ecosystem enable to create a complex trophic system (Webb et al. 2010) and increase the delivery of ecosystem services (Moran-Ordonez et al. 2013a) And therefore, it would increase the resilience of the system, helping the species to cope with disturbances and allowing them to adapt under adverse scenarios (Plieninger and Bieling 2012). Future research including a more complete habitat quality assessment and a long term monitoring are recommended to prove these findings.

7. Conclusion

This study shows that the coverage of plant functional groups and soil parameters changes when cutting and burning practices are applied to lowland heathlands communities. In consequence, a certain range of potential habitats are available for specialist plant and animal species.

Our study illustrates an attempt to assess the habitat quality of heathlands. In general, the habitat quality of the studied Danish heaths are under bad quality status. Nevertheless, the application of traditional practices contributes to the improvement of habitat quality.

This document demonstrates that there is potential to improve an existing habitat quality assessment based on existing data if they are combined in an appropriate way. For instance, the animal species are not considered within the objectives of management practices neither the assessment of habitat quality. We may therefore conclude that substantial data gaps remain to be filled before a fully integrated and complete habitat quality assessment can be carried out.

I believe that studying the delivery of ES is crucial if we wish to gain a more holistic understanding of how heathlands are affected by interventions. The use of that data would thus facilitate the mainstreaming of biodiversity and ecosystem services, which is embedded in EU 2020 Biodiversity Strategy.

8. Recommendations

During centuries, farmers and shepherds have been using these areas to produce benefits and survive. The abandonment of heathlands, due to low productivity, together with other drivers of biodiversity loss, such as pollution, have driven this system to a bad habitat quality. To preserve this habitat and improves its quality, traditional management practices should be re-established and adapted to current situation. Successful management practices implies that stakeholders are taken in consideration. Current practices should aim at creating mosaic of habitats for varied species of plants and animals.

Moreover, the conservation of cultural landscapes should take in account the different stakeholders to design management practices according to the requirement of the local situations (Plieninger et al. 2015). A way to involve the stakeholders in the planning and decision-process is through the PPGIS tool. This tool is used in different regions to gain knowledge on the perception of local stakeholders on their landscapes and on a better understanding of their uses (Brown et al. 2012, Brown and Kytta 2014). Different studies across Europe and other countries as Australia and USA, showed that PPGIS is effective in the mapping of cultural and provisioning ES indicators among different stakeholders groups (Darving and Lindo 2015; Garcia-Nieto et al. 2015). The methodology consists normally of questionnaires/interviews designed to receive responses on how stakeholders use their landscape and where they find benefits (ES). Additionally, there is a map, printed or online version, where stakeholders can point landscape practices and cultural values. The role of stakeholders (power) and their relationships in the landscape matters is relevant because it limits the access of some stakeholders to some ecosystem services (Felipe-Lucia et al. 2015). In addition to the role of stakeholders, different studies identified that a sense of place has implications in people’s attitude towards the successful management of nature resources (Larson et al. 2013; Stuart and Knapp. 2015).

9. Future perspectives

Our results contribute to opening up for new research. There is a need to focus on new approaches to assess heathlands using more appropriate methods based on knowledge from local stakeholders and experts. There is little knowledge on the delivery of ES by heathlands, because studies have focused on the effects of the drivers of biodiversity loss, vegetation community and soil structure. Few studies have targeted the delivery of cultural services and how they have changed along with the abandonment of heathlands. Further research should include an integrative framework combining an assessment of biodiversity and ecosystem services, and identifying the possible trade-offs between them. In a long term, this will enable to develop a more appropriate habitat quality assessment and to adopt more suitable conservation measures.

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Appendix A. Maps

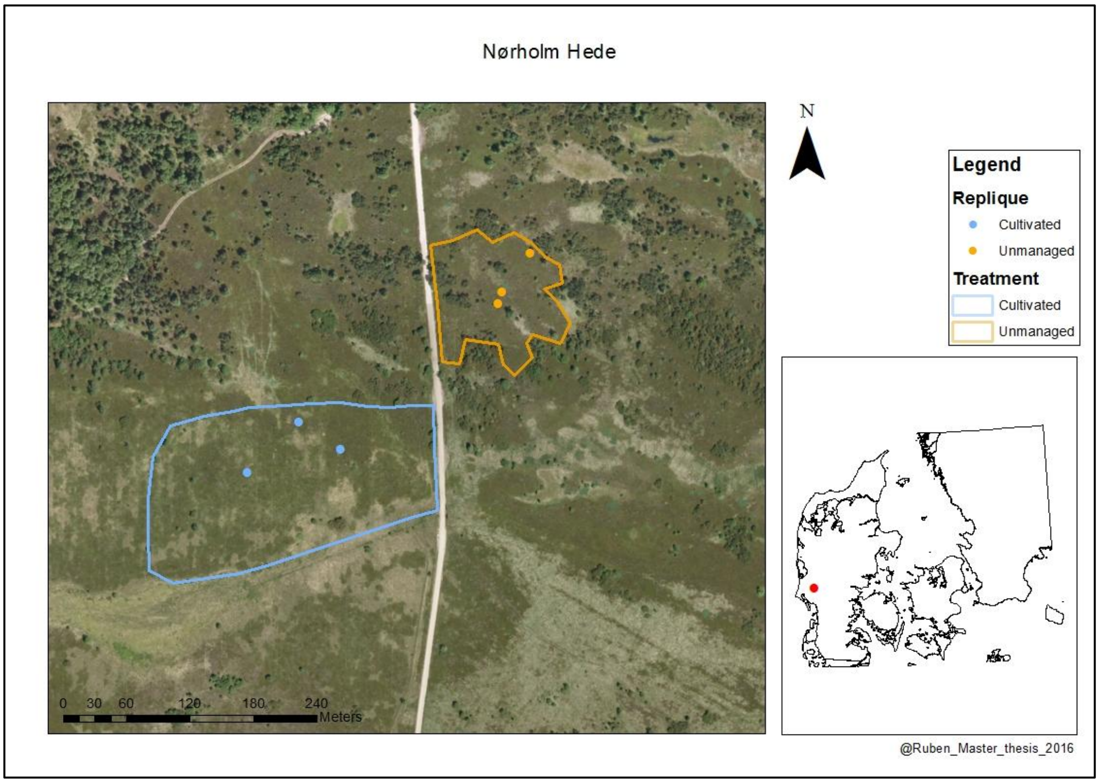


Fig. A1 Location of points in Nørholm. The points indicate the start of the transects.

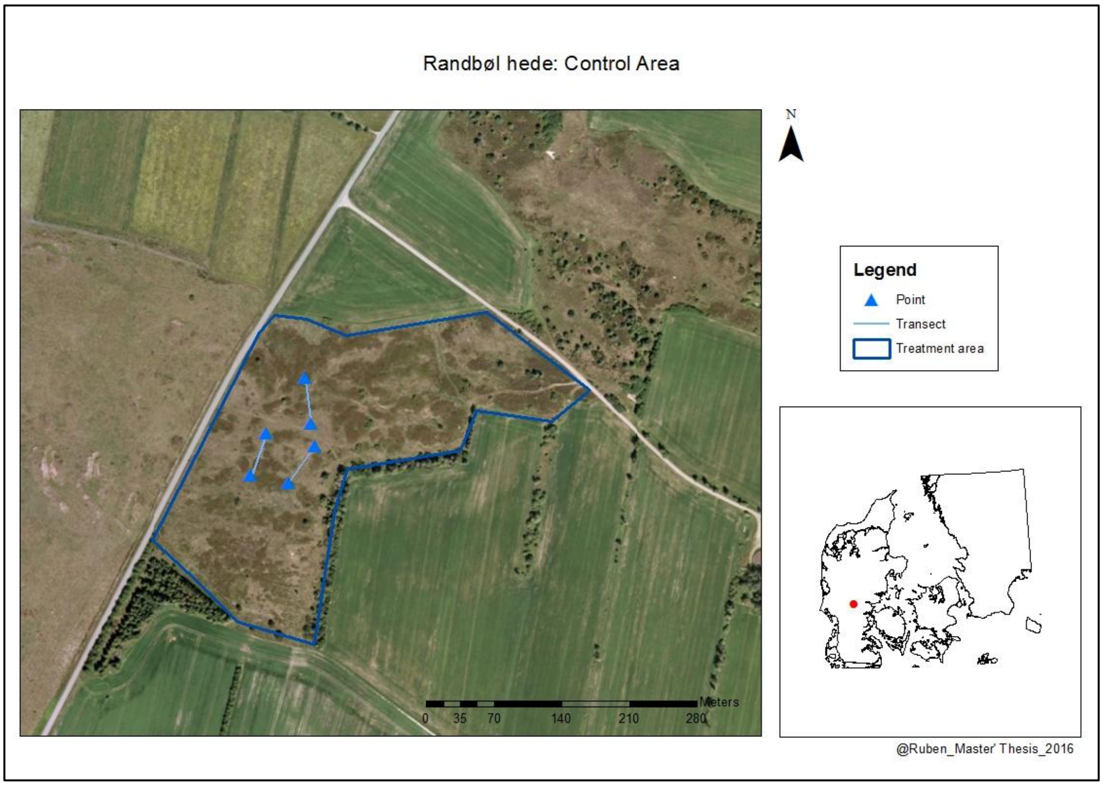


Fig. A2 Location of control area and the transects established in Randbøl heath.

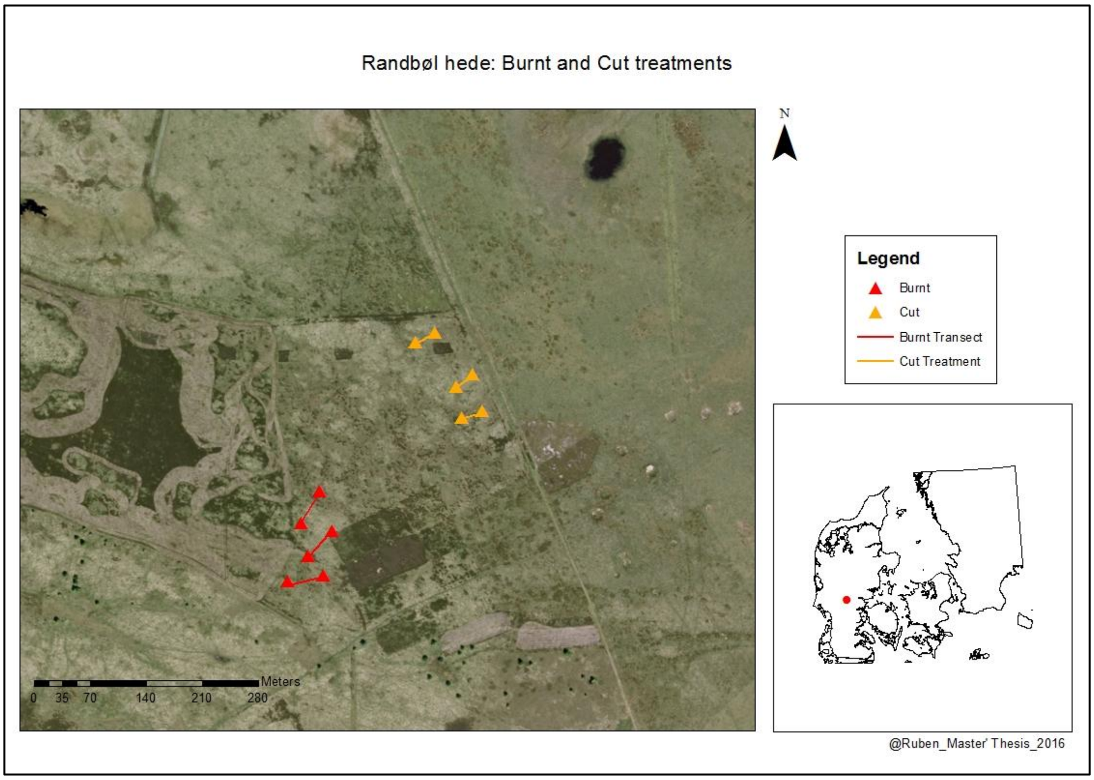


Fig. A3 Location of cut and burnt transects in Randbøl heath.

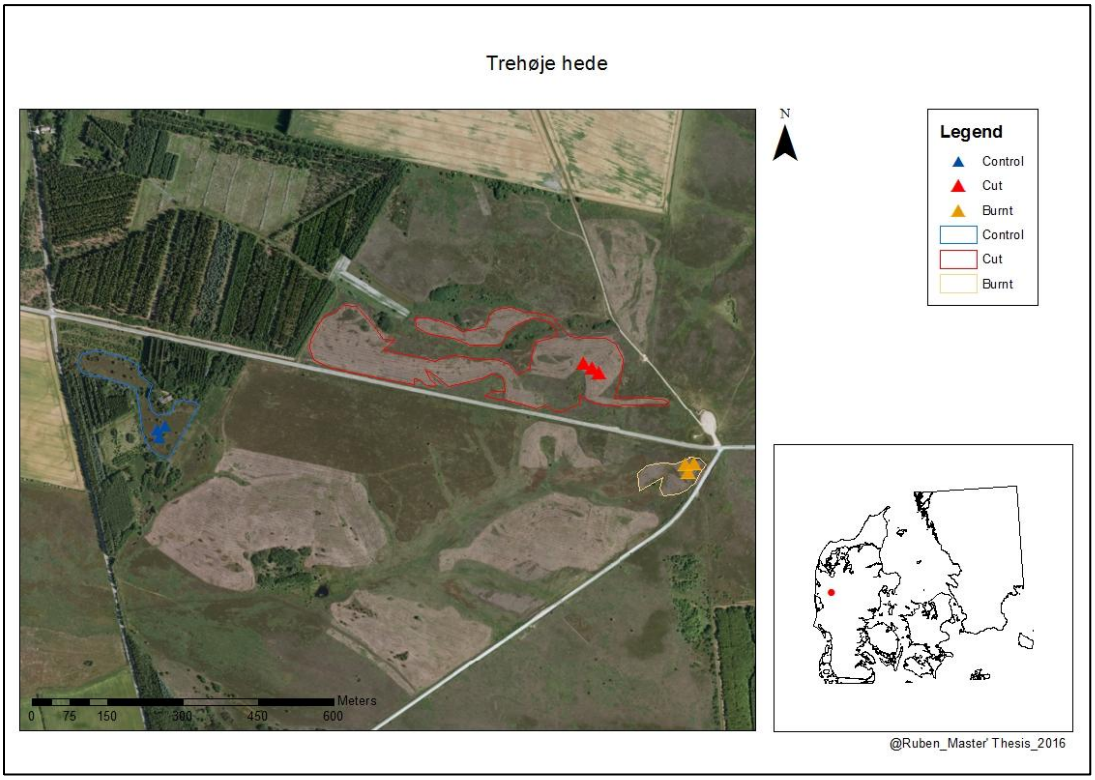


Fig. A4 Location of treatments and plots in Trehøje heathland.

Appendix B. Functional groups

Table B1. F and p-values for one-way ANOVA of location effect across controls for functional types, biodiversity indices and soil parameters. Significance of treatment effect is given for p-value * (P < 0.05), ** (P < 0.01) and *** (P < 0.001).

Functional type	F	p-value
C. vulgaris	363.11	5.50e-7***
E. nigrum	34.83	0.0005***
Other dwarf shrubs	14.47	0.0050**
D. flexuosa	15.26	0.0044**
M. caerulea	17.38	0.249
Other grasses	1.33	0.331
Forbs	16.25	0.0038**
Moss	1.01	0.417
Lichens	0.75	0.512
Total cover	18.23	0.0028**
Soil parameter	F	p-value
Thickness Ho	13.81	0.0057**
C Ho	6.63	0.0302*
C Ha	5.63	0.0419*
N Ho	2.82	0.137
N Ha	10.96	0.0099**
C:N Ho	76.07	5.46e-5***
C:N Ha	59.90	0.0001***
ph Ha	83.64	4.15e-5***
Biodiversity index	F	p-value
S	9.19	0.0149*
H’	9.69	0.0132*
J	6.81	0.0286*
D	39.02	0.0003***

Ha is referred to the mineral layer, while Ho is referred to the organic layer.

Table B2. Statistic results for treatment effect on functional types, soil parameters and vegetation height. Man-Whitney test and Kruskal-Wallis for functional types and soil parameters and vegetation height of Nørholm and Randbøl hede, respectively. *D. flexuosa* and bare ground were calculated with t-test for Nørholm. Soil moisture for Randbøl was calculated with one-way ANOVA.

	Nørholm		Randbøl	
Functional type	W	p-value	W	p-value
C. vulgaris	3337.5	9.35e-5	41.53	9.58e-10***
E. nigrum	536	2.2e-16	63.30	1.79e-14***
Other dwarf shrubs	2838.5	0.832	91.42	2.2e-16***
D. flexuosa	-1.26	2.2e-16	0.88	0.6447*
M. caerulea	2397.5	0.0014	28.78	5.62e-7*
Other grasses	5175	2.95e-13	35.44	2.01e-8***
Forbs	5466	2.2e-16	12.08	0.0024**
Moss	2150.5	0.0128	44.31	2.38e-10***
Lichens	2400	0.0039	-	-
Total cover	-1.26	0.21	42.45	6.05e-8***
Soil parameter	W	p-value	F	p-value
Soil moisture (%)	217.5	2.2e-16	6.79	0.0014**
Soil Temperature (ºC)	1072.5	3.82e-14	112.32	2.2e-16***
Organic matter depth (cm)	342.5	2.2e-16	12.11	0.0023*
Litter layer depth (cm)	4675.5	2.05e-12	83.57	2.2e-16***
Bare ground (%)	-1.75	0.131	32.87	7.29e-8*

Table B3. The effects of treatment on the vegetation height of Randbøl hede. Means and standard error shown. Means with different lowercase letters (a, b) represent a significant difference between treatments and control. VH and CH are the acronyms chosen for vegetation height and *Calluna* height, respectively.

Variable	Control	Cut	Burnt	Differences
VH	24.3 ± 0.06 (a)	14.5 ± 2.10 (b)	13.0 ± 1.41 (b)	1 > 2 = 3 ***
CH	3.0 ± 1.00 (a)	9.1 ± 2.57 (b)	10.80 ± 1.33 (b)	1 < 2 = 3 ***

Significance due to treatment effect is tested with ANOVA (and Kruskal-Wallis for non parametric tests). Numbers in the last column are referred to location: 1 (Control), 2 (Cut), 3 (Burnt). Pair-wise comparisons (tested with Tukey's HSD test or Nemenyi test) are given for $P < 0.001$ (***).

Table B4. T-test and one-way ANOVA for treatment effect on biodiversity indices of Nørholm, Randbøl and Trehøje hede. Significance of treatment effect is given for p-value *** ($P < 0.001$).

	Nørholm		Randbøl		Trehøje	
Index	t	p-value	F	p-value	F	p-value
S	-1.36	0.245	0.6	0.579	1.88	0.232
H'	0.72	0.535	2.54	0.158	6.81	0.028
J	1.44	0.246	0.23	0.797	0.18	0.837
D	14.42	0.002	18.79	0.003***	7.07	0.026

Table B5. Means and standard errors for the coverage of functional groups in the two different management practices of Nørholm hede. Means with different lowercase letters (a, b) represent a significant difference between treatments.

Functional type	Control	Cultivated
C. vulgaris	0.0 ± 0.00 (a)	3.7 ± 0.75 (b)
E. nigrum	38.2 ± 3.07 (a)	5.8 ± 2.61 (b)
Other dwarf shrubs	2.3 ± 0.27 (a)	0.2 ± 0.27 (a)
D. flexuosa	0.0 ± 0.00 (a)	11.7 ± 2.05 (b)
M. caerulea	0.2 ± 0.27 (a)	0.04 ± 0.04 (b)
Other grasses	0.6 ± 0.51 (a)	9.6 ± 0.54 (b)
Forbs	0.03 ± 0.03 (a)	15.1 ± 2.99 (b)
Moss	45.0 ± 3.09 (a)	34.0 ± 4.27 (b)
Trees	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)
Lichen	0.4 ± 0.01 (a)	0.0 ± 0.00 (a)
Shrubs	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)
Total cover	87.6 ± 4.10 (a)	80.5± 5.55 (a)

Significance due to treatment effect is tested with Mann-Whitney U test ($P < 0.05$).

Table B6. The effects of treatment on the coverage of functional groups of Randbøl hede. Means and standard errors for the coverage of functional groups in the three different management practices of Randbøl hede. Means with different lowercase letters (a, b, c, ab) represent a significant difference between treatments.

Functional type	Control	Cut	Burnt	Differences
C. vulgaris	3.2 ± 0.83 (a)	15.0 ± 4.62 (b)	16.8 ± 1.11(b)	1 < 2 = 3**
E nigrum	19.4 ± 1.81(a)	0.0 ± 0.00 (b)	0.0 ± 0.00 (b)	1 > 2 = 3**
Other dwarf shrubs	2.8 ± 1.24 (a)	16.8 ± 1.11 (b)	13.6 ± 0.91 (b)	1 < 3 = 2**
D. flexuosa	25.6 ± 6.16 (a)	18.3 ± 3.30 (a)	18.6 ± 0.59 (a)	-
M. caerulea	4.8 ± 2.97 (a)	11.0 ± 2.20 (b)	9.3 ± 2.21 (ab)	1 < 3 < 2***
Other grasses	1.8 ± 0.77 (a)	5.8 ± 2.60 (b)	4.3 ± 0.80 (b)	1 < 3 = 2**
Forbs	16.4 ± 3.89 (a)	12.2 ± 2.33 (ab)	8.8 ± 0.73 (b)	1 > 2 > 3*
Moss	53.8 ± 7.19 (a)	32.6 ± 0.90 (b)	18.3 ± 7.03 (c)	1 > 2 > 3**
Trees	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	-
Lichen	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	-
Shrubs	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	0.0 ± 0.00 (a)	-
Total cover	128.00 ±7.33 (a)	112.0 ± 2.92 (b)	89.8 ± 6.83 (c)	*1 > 2 > 3*

Significance due to treatment effect is tested with Kruskal-Wallis test. Post-hoc numbers are referred to the treatment type: Control (1), Cut (2), Burnt (3). Pair-wise comparisons using Nemenyi test are given for P < 0.001(***), P < 0.01(**) and P < 0.05 (*).

Table B7. Means and standard errors for the coverage of functional groups in the different management practices of the three locations: Nørholm, Randbøl and Trehøje hede. All the values are in percentage 0-100%.

Functional Types	Nørholm Control	Cultivated	Randbøl Control	Cut	Burnt	Trehøje Control	Cut	Burnt
C. vulgaris	0.0 ± 0.00	3.7 ± 0.75	3.2 ± 0.17	15.0 ± 4.62	16.8 ± 1.11	23.2 ± 0.16	20.3 ± 1.82	21.0 ± 0.61
E. nigrum	38.2 ± 0.61	5.8 ± 2.61	19.4 ± 0.36	0.0 ± 0.00	0.0 ± 0.00	9.5 ± 0.48	15.7 ± 1.45	0.0 ± 0.00
Other dwarf shrubs	2.3 ± 0.46	0.2 ± 0.27	2.8 ± 0.25	16.8 ± 1.11	13.6 ± 0.91	17.1 ± 0.56	8.6 ± 3.20	11.4 ± 0.87
D. flexuosa	0.0 ± 0.00	11.7 ± 2.05	25.6 ± 1.23	18.3 ± 3.30	18.6 ± 0.59	2.3 ± 0.25	2.3 ± 0.68	16.1 ± 0.52
M. caerulea	0.9 ± 0.19	0.0 ± 0.04	4.8 ± 0.59	11.1 ± 2.20	9.3 ± 2.21	0.4 ± 0.08	2.7 ± 1.55	0.3 ± 0.36
Other grasses	0.5 ± 0.10	9.6 ± 0.54	1.8 ± 0.15	5.80 ± 2.60	4.3 ± 0.80	0.7 ± 0.08	9.8 ± 3.23	27.1 ± 1.52
Forbs	0.0 ± 0.01	15.1 ± 2.99	16.4 ± 0.78	12.2 ± 2.33	8.8 ± 0.73	1.1 ± 0.13	1.6 ± 0.81	5.7 ± 3.17
Moss	45.0 ± 0.62	34.0 ± 4.27	53.9 ± 1.44	32.7 ± 0.90	18.3 ± 7.03	44.0 ± 1.02	29.1 ± 1.46	17.9 ± 4.53
Trees	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.7 ± 0.15	0.0 ± 0.00	0.0 ± 0.00
Lichens	0.4 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.8 ± 0.16	10.0 ± 1.62	0.0 ± 0.00
Shrub	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.0 ± 0.00	0.3 ± 0.35
Total cover	87.6 ± 0.82	80.5 ± 5.55	128.0 ± 1.47	112.0 ± 2.92	89.8 ± 6.83	100.0 ± 0.00	100.0 ± 0.01	100.0 ± 0.0

Species	Functional group	N Control	N Cultivated	R Control	R Cut	R Burnt
Achillea millefolium	Forbs	x		x		x
Agrostis tenuis	Other grasses	x				
Agrostis vinealis	Other grasses					
Arnica montana	Forbs				x	x
Betula sp	Tree		x			x
Calluna vulgaris	*	x		x	x	x
Carex arenaria	Other grasses	x	x	x		
Carex nigra	Other grasses		x		x	
Carex panicea	Other grasses			x	x	x
Carex pilulifera	Other grasses		x		x	
Cytisus scoparius	Shrub					
Dactylorhiza dilatata	Forbs			x		x
Danthonia decumbens	Other grasses					
Deschampsia flexuosa	*	x	x	x	x	x
Dryopteris dilatata	Fern			x		x
Empetrum nigrum	*	x	x	x		
Epilobium angustifolium	Forbs					
Erica tetralix	Other dwarf shrub		x			
Festuca ovina	Other grasses	x		x	x	x
Frangula alnus	Shrub	x		x		
Galium saxatile	Forbs	x		x	x	x
Genista anglica	Other dwarf shrub	x		x	x	x
Hieracium pilosella	Forbs					
Holcus lanatus	Other grasses	x				
Hypochoeris radicata	Forbs				x	x
Lonicera caprifolium	Forbs	x				

Species	Functional group	N Control	N Cultivated	R Control	R Cut	R Burnt
Luzula multiflora	Other grasses	x		x	x	x
Molinia caerulea	*	x	x	x	x	x
Pice abies	Tree					
Pinus mugo	Tree		x			
Plantago sp	Forbs					x
Potentilla erecta	Forbs	x		x	x	x
Quercus robur	Tree					
Rumex acetosa	Forbs			x	x	x
Salix repens	Other dwarf shrub				x	x
Solidago virgaurea	Forbs				x	x
Sorbus aucuparia	Tree	x				
Trientalis europaea	Forbs	x	x	x	x	x
Trichophorum cespitosum	Other grasses		x			
Trifolium campestre	Forbs					x
Vaccinium uliginosum	Other dwarf shrub		x			
Vaccinium vitis-idaea	Other dwarf shrub			x	x	x
Viola silvestris	Forbs	x				

Table B8. The presence of each plant species in each of the treatments and controls and its matching functional group. N, is referred to Nørholm and; R, is referred to Randbøl. Cross (x) means presence of the species and empty species means that the species is absent in the corresponding treatment. Asterisk (*) means that the species was considered as separated per se in another category. Mosses and lichens are not included in this table because we did not determine to the species level.

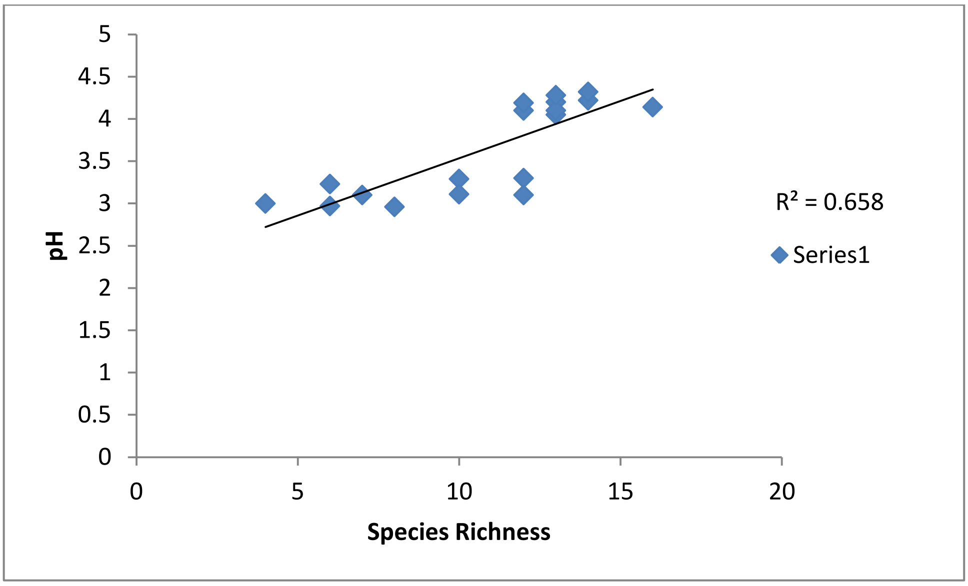
Table B9. Correlations between functional groups and soil parameters for Nørholm hede. Only values with significative differences are included (P < 0.05).

Variable	VH	CH	LL	OMD	*C.vulgaris*	*E. nigrum*	Other dwarf shrubs	*D. flexuosa*	*M. caerulea*	Other grasses	Forbs	Mosses
LL	0.35	0.25	-	-0.38	0.25	**-0.46**	-	0.43	-0.25	0.41	0.4	-
OMD	0.17	-0.2	-0.38	-	-0.2	**0.58**	-	**-0.63**	0.16	**-0.67**	**-0.67**	0.39
Bare ground	-	-	-	-	-	-0.27	-	-	-	-	-	**-0.76**
C.vulgaris	0.18	0.99	0.25	-0.2	-	-	-	0.22	-	0.25	0.19	-
E. nigrum	0.31	0.16	**-0.47**	**0.58**	-	-		**-0.6**	0.16	-0.58	**-0.67**	-
Other dwarf shrubs	0.28	0.09	0.16	-	-	-	-	-	-	-	-	-
D. flexuosa	0.31	0.22	0.43	**-0.63**	0.22	**-0.6**	-	-	-0.22	**0.64**	**0.77**	-
M. caerulea	-	-	-0.26	0.17	-	-	-	-0.24	-	-	-0.22	-
other grasses	0.23	0.25	0.41	**-0.68**	0.25	**-0.58**	-	**0.64**	-0.24	-	**0.71**	-0.19
Forbs	0.26	0.2	0.4	**-0.68**	0.20	**-0.67**	-	**0.77**	-0.22	**0.71**	-0.18	-

Table B10. Correlations between functional groups and soil parameters for Randbøl hede. Only values with significative differences are included (P < 0.05).

Variable	VH	CH	LL	OMD	*C.vulgaris*	*E. nigrum*	Other dwarf shrubs	*D. flexuosa*	*M. caerulea*	Other grasses	Forbs	Mosses
LL	-	-	-	-	-0.37	0.21	**-0.43**	-	-0.19	-0.23	0.22	**0.44**
OMD	-	-0.13	-	-	-0.13	0.25	-	-	-	-	-	-
Bare ground	-	-	-	-	-	-0.06	-	-0.23	-	-	-0.31	**-0.69**
C.vulgaris	-0.27	0.93	0.37	-0.13	-	-0.25	0.20	-0.23	-	-	**-0.37**	-0.21
E. nigrum	0.3	-0.21	0.21	0.248	-0.245	-0.35	-0.14	-0.13	-0.26	-0.355	-	-
Other dwarf shrubs	**-0.53**	0.17	**0.43**	-	0.2	-0.35	-	-	0.13	-	-	-0.25
D. flexuosa	-	-0.23	-	-	-0.23	-0.13	-	-	-0.23	-	0.25	0.17
M. caerulea	-	-	0.19	-	-	-0.26	0.13	0.13	-	0.13	-0.14	-
other grasses	-0.32	-	0.23	-	-	-0.35	0.25	-0.13	0.13	-	0.22	-
Forbs	-	-0.33	0.22	-	**-0.37**	-	-	0.25	-0.14	-	-	0.35
Mosses	0.17	-	0.44	-	-0.21	-	-0.25	0.17	-0.15	-	0.35	-

Fig B1. Correlations between species richness and soil acidity (pH) for Randbøl and Trehøje hede. Only values from the mineral layer are considered. Taken from Muñoz (2015) and Gutzat (2015). Values ranged from 2.96 to 4.32.



Appendix C. Ellenberg values.

Table C1. The effect of treatment on the Ellenberg values. Means and standard error for the Ellenberg values of each management practice in Nørholm hede. Means with different lowercase letters (a, b) represent a significant difference ($P < 0.05$) between treatment and control.

Ellenberg	Control	Cultivated
N	2.37 ± 0.08 (a)	2.27 ± 0.18 (a)
R	3.35 ± 0.83 (a)	2.52 ± 0.33 (a)
F	5.67 ± 0.51 (a)	5.35 ± 0.59 (a)

Table C2. Means and standard error for the Ellenberg values of each management practice in Randbøl hede. Means with different lowercase letters (a, b, c) represent a significant difference ($P < 0.05$) between treatments.

Ellenberg	Control	Cut	Burnt	Differences
N	2.72 ± 0.36 (a)	2.66 ± 0.05 (a)	2.66 ± 0.19 (a)	-
R	2.91 ± 0.39 (a)	3.45 ± 0.13 (a)	3.29 ± 0.21 (a)	-
F	5.38 ± 0.15 (a)	5.53 ± 0.15 (a)	5.67 ± 0.22 (a)	-

Table C3. T-test and one-way ANOVA for Ellenberg values (N, F, R). T and p-value for Nørholm and F and p-value for Randbøl hede.

	Nørholm		Randbøl	
Index	t	p-value	F	p-value
N	0.55	0.623	0.02	0.977
F	3.98	0.705	2.54	0.158
R	0.91	0.435	0.65	0.554

Appendix D. Description of Habitat Quality Assessment.

The habitat quality assessment includes different points: area and natural range; characteristic species and; structure and function. Here, I include the method for field work. This method describes positive and negative habitat structures, cover of functional groups and a section for management recommendations.

Form available at: http://naturstyrelsen.dk/media/nst/70876/pgf3-hede104_040210.pdf

Teknisk anvisning til besigtigelse af naturarealer omfattet af Naturbeskyttelseslovens. Version 1.04, Juni 2010: Available online at : http://bios.au.dk/fileadmin/Resources/DMU/Dyr%20og%20planter/Naturtilstand/TA-besigtigelse_af_naturarealer-104.pdf

STRUCTURE. It is referred to habitat structures. There are two types: positive and negative structures. Each structure has a value: 1 (Not present), 2 (in between scattered) or 3 (widely widespread).

- Positive structures (P):
  - P1: Age variation in *Calluna vulgaris*
  - P2: Age variation in *Erica tetralix*
  - P3: Dominance of dwarf shrubs
  - P4: Presence of lichens
- Negative structures (N):
  - N1: Old dead areas with *Calluna vulgaris*
  - N2: Dominance of grasses and *Molinina caerulea*
  - N3: Invasive moss *Campylopus introflexus*
  - N4: Conifers except for *Juniperus communis* and *Cytisus scoparius*

COVER: Coverage of functional groups are considered in this section. It includes vegetation, grasses/forbs, dwarf shrubs, woody plants and invasive species. Values range from 1 to 5 in relation to the percentage of coverage (%): 1 (0-5), 2 (5-10), 3 (10-30), 4 (30-75), 5 (75-100).
The areas that have been subject to agriculture or grazing/cutting management are classified following the same method as for cover of functional groups (1-5).

INVASIVE PLANT SPECIES (0-100%): This category is mainly referred to *Pinus mugo*, although there are other invasive species: *Cytisus scoparius*, *Campylopus introflexus* and *Rosa rugosa*. Values range from 1 to 5 as for coverage of functional groups.

Table D1. Habitat quality assessment for each location and treatment/control. *N*, is referred to negative structure and *P*, referred to positive structure. The values range from 1 to 3 in the case of structure and from 1 to 5 in vegetation parameters and management. This table has been used to calculate the index presented in the discussion for the habitat quality (Table), which is in accordance with Table C2. The index is calculated following a formula which includes characteristic species (indicators), structure and function, habitat type score. Based on methodology developed by the Danish Nature Agency.

Location	Nørholm	Nørholm	Randbøl	Randbøl	Randbøl	Trehøje	Trehøje	Trehøje
Treatment	Cultivated	Control	Control	Cut	Burnt	Control	Cut	Burnt
Structure								
N1	1	3	1	1	1	2	2	1
N2	2	3	3	3	2	1	1	2
N3	1	1	1	1	1	1	1	1
N4	3	2	2	1	1	2	1	1
P1	2	2	1	2	3	1	1	2
P2	1	1	1	1	1	1	1	1
P3	2	2	2	3	2	3	3	2
P4	2	1	2	1	1	2	2	1
Coverage								
Vegetation	5	5	5	5	5	5	5	4
Grasses/Forbs (<15cm)	1	4	3	3	4	1	1	2
Grasses/Forbs (15-50cm)	2	4	5	3	5	1	1	1
Grasses/Forbs (>50cm)	3	3	1	1	1	1	1	1
Dwarf shrubs	2	1	2	4	4	5	5	4
Woody plants	4	4	2	1	1	2	1	1
Invasive species	1	3	2	1	1	3	2	1
Management								
Agriculture	1	1	1	1	1	1	1	1
Grazing/Cutting	1	1	1	5	5	1	3	2

Table D2. Index table for the habitat quality of habitat types 4030 in Denmark (Habitat Directive). Values range from 0 to 1 with five different categories, from really bad to good status. The index is calculated following a formula which includes characteristic species (indicators), structure and function, habitat type score. Based on methodology developed by Danish Nature Agency.

Habitat quality	Status
0- 0.2	Really bad
0.2-0.4	Bad
0.4-0.6	Moderate good
0.6-0.8	Fairly good
0.8-1	Good