

DESERTIFICATION OF HIGH LATITUDE ECOSYSTEMS:  
CONCEPTUAL MODELS, TIME-SERIES ANALYSES AND EXPERIMENTS

A Dissertation

by

JOHANN THORSSON

Submitted to the Office of Graduate Studies of  
Texas A&M University  
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

December 2008

Major Subject: Rangeland Ecology and Management

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## ABSTRACT

Desertification of High Latitude Ecosystems:

Conceptual Models, Time-Series Analyses and Experiments. (December 2008)

Johann Thorsson, B.S., University of Iceland

Chair of Advisory Committee: Dr. Steven R. Archer

Ecosystem degradation in Iceland has been severe since man arrived 1100 years ago. Birch woodlands cover has declined from 25% of the land area, to only 1%. The deforestation is considered to be the initial stage in the land degradation process, followed by surface destabilization, and later erosion. The objective of this study was to quantify and evaluate factors that contribute to the early stages of land degradation in Icelandic ecosystems. Specific objectives were to improve our understanding of how livestock grazing might initiate early degradation stages, elucidate field-based landscape metrics useful for characterizing degradation stages, and to determine if landscape metrics obtained from remote sensing data can be used to detect landscape structure changes and identify degraded and at risk rangelands in real time over extensive and remote areas. A State-and-Transition conceptual model was constructed for the experimental area to identify potential key processes in the degradation sequence, and to formalize research questions. Experimental plots were established in five plant community types representing a space-for-time degradation sequence.

Birch seedling (*Betula pubescens* Ehrh.) growth and survival was reduced with repeated clipping treatment applied to simulate browsing, but the amount of decline

varied with plant community type. This suggests that continuous grazing may contribute to deforestation, as regeneration will be reduced over time.

Intense grazing treatments, simulating both grazing and trampling, increased surface instability and soil loss compared to grazing only or control, suggesting that intense grazing may contribute to surface destabilization and therefore to land degradation. Erosion appeared to be active in the most intense treatments, also within the woodlands. The data indicate that the woodlands may have lower resilience than the other plant communities as treatment effects appeared quicker there. The woodlands may thus be particularly vulnerable to intense grazing.

The landscape metrics used to quantify changes in landscape surface properties over a 51 year period yielded inconclusive results, either because of data limitations or because of non-detectable erosion activity.

The results do generally support the proposed S&T model for the experimental area. It is concluded that grazing may contribute to woodland decline, and intensify degradation processes.

DEDICATION

To my wife,  
for all her patience and love.

## ACKNOWLEDGEMENTS

I am most grateful to Dr. Steve Archer for his assistance, guidance, constructive criticism support and friendship during my stay at Texas A&M. The other members on my committee, Dr. Fred Smeins, Dr. X. Ben Wu, Dr. Charles Thomas Hallmark and Dr. Asa Aradottir were also always available and willing to provide advice when needed. Special thanks to Dr. Asa Aradottir with all her help on statistical analyses and support during my fieldwork and dissertation writing.

Many people helped me during the data collection in the field, data processing, lab work and discussion of ideas. I cannot name them all, but must mention the late Bergthor Johannsson for identification of mosses, Hordur Kristinsson for identification of lichens, Olafur Eggertsson for help with the dendrochronology, Einar Gretarsson, Sigmar Metusalemsson, Elin Asgeirsdottir and Bjorn Traustason for their assistance with my fieldwork and soil analyses in Iceland and Donna Prochaska for all her help with the soil analyses in Texas. Fridrik Aspelund and Dawn Browning must also be mentioned for the help they gave me, and special thanks goes to Andres Arnalds for his enthusiasm in my project and encouragements.

Sveinn Runolfsson and the Icelandic Soil Conservation Service provided generous support throughout the study, as well as Thorsteinn Tomasson and the Icelandic Agricultural Research Institute. For that I am very grateful. The people at Hofn and Motel Venus gave me free and unlimited access to their land for my field

experiments. My parents housed me and provided me with a car while I was collecting my field data, without their assistance this would not have been possible.

Olafur Arnalds and Asa Aradottir must also be thanked sincerely for both pushing me into doing this, and for all their support and friendship. This would not have happened without their encouragement. I would also not have been able to finish this without all the help my friend Tota gave me back in College Station while I was in Tucson.

At last, but not least, I must thank my wife, Berglind Orradottir for all her help with my fieldwork, data analyses and lab work. I would never have succeeded without her endless help, encouragement and support.

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## CHAPTER I

### INTRODUCTION

Few areas of the high-latitude regions of the Northern Hemisphere have experienced levels of ecosystem degradation as severe as Iceland. An estimated 65 % of the land was vegetated at the time of settlement (Arnalds et al., 1987), with birch (*Betula pubescens* Ehr.) woodlands covering 15-25 % of the total area (Sigurdsson, 1977; Gudjonsson and Gislason, 1998; Olafsdottir et al., 2001). Today, birch cover is 1% (Gudjonsson and Gislason, 1998) and herbaceous cover is estimated to have declined about 60 % (Thorsteinsson, 1978). Barren deserts are now estimated to cover 36 % of the country and additional 10-15 % are categorized as areas with limited plant cover (LMI, 1993).

Iceland's barren deserts are the combined result of natural and human-induced erosion. Ólafsdóttir and Gudmundsson (2002) have suggested that climate is the main driving force. They point out that at least two extensive geologic erosion periods coinciding with cooler temperatures occurred before settlement. However, the current erosion episode appears more extensive than the two previous, suggesting anthropogenic activities have accelerated erosion associated with natural geologic forces. That observation is supported by Thorarinsson (1961) who pointed to an increase in soil

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This dissertation follows the style of *Arctic, Antarctic, and Alpine Research*.



thickening rates in lowland areas, following the arrival of humans, and suggested this was caused by eolian material from soils eroding in the area.

Land degradation and desertification of the proportions and scale observed in Iceland are virtually unknown in the surrounding western world (Arnalds, 2000). The Icelandic erosion processes are often characterized by a total loss of the soil profile down to the glacial till substrate. Icelandic soils are predominantly of the Andisol order, formed from volcanic ejecta (Arnalds, 1990). Andisols have very high water holding capacities, low aggregate cohesion and low bulk density (Wada, 1985); all properties that make them highly susceptible to wind and water erosion once exposed (Maeda and Soma, 1986; Arnalds, 1990). Good vegetation cover is therefore critical in order to minimize erosion.

The introduction of domestic herbivores in 900 A.D. may have altered the dynamic equilibrium present since the end of Pleistocene. Combined defoliation, trampling and hoof action have changed plant community composition, structure, biomass and root density to potentially increased soil susceptibility to erosion. Our understanding of the process is limited, however. While the end results are obvious, it is not clear how erosion processes are initiated or how they proceed. What specific role do grazers play in the process? What vegetation properties are important for reducing erosion risk? Are some plant community types more vulnerable to soil erosion than others? Are there thresholds in plant cover and soil properties beyond which positive feedbacks are initiated and rates of change are disproportionately accelerated? Answers to such questions are critical if effective conservation plans are to be developed for

Icelandic landscapes. An understanding of the degradation process will help land managers to target landscapes at risk, prioritize mitigation activities, and create management plans specific to landscapes at various stages of retrogression.

This study had four specific goals: 1) quantify and evaluate factors that contribute to the early stages of land degradation in Icelandic ecosystems, 2) improve our understanding of how livestock grazing might initiate early degradation stages, 3) elucidate field-based landscape metrics useful for characterizing degradation stages, and 4) determine if landscape metrics obtained from remote sensing data can be used to detect landscape structure changes and identify degraded and at-risk rangelands in real time over extensive and remote areas.

Chapter II describes the weather, characteristics and land use history of the study site. A State-and-Transition model constructed used to define research questions is presented in Chapter III. Chapter IV addresses recruitment of birch (*Betula pubescens* Ehrh.) in the context of quantifying seedling response to browsing in different plant communities. The effect of plant communities and grazing intensity on surface stability is addressed in Chapter V. Chapter VI summarizes the results of a remote sensing study that uses various landscape metrics to identify landscapes at risk for erosion. The final chapter, Chapter VII, summarizes the overall findings.

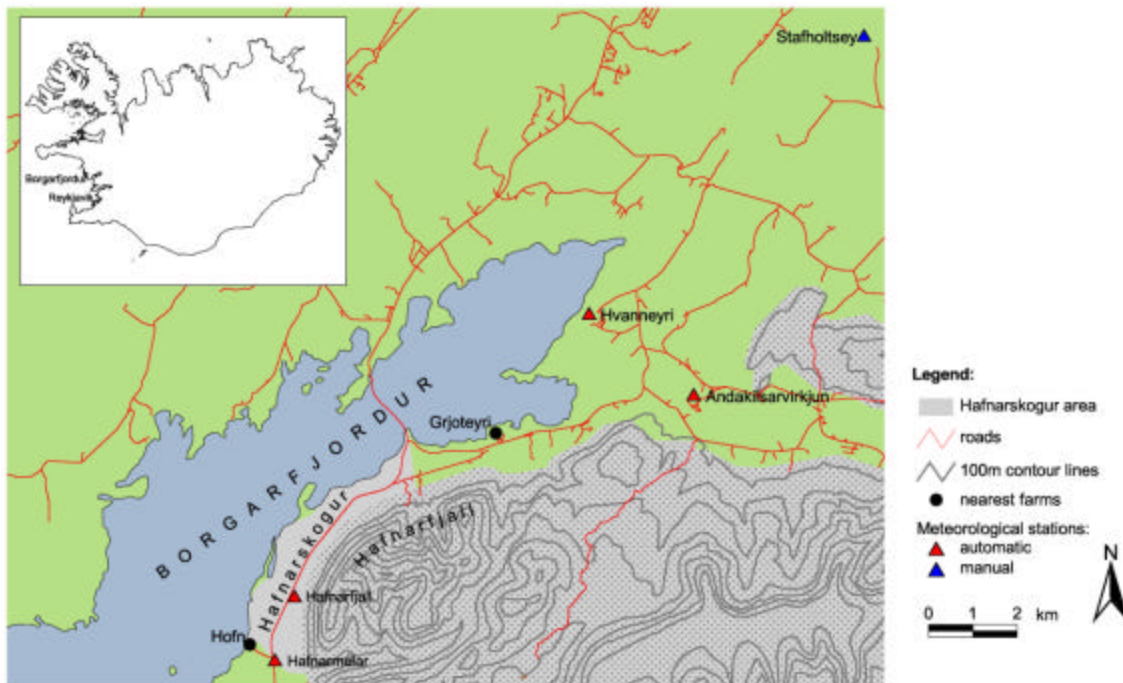
## CHAPTER II

### SITE DESCRIPTION

#### **Physiognomy, Historic Vegetation and Land use of Hafnarskogur**

The area chosen for this study was Hafnarskogur in the lowlands of west Iceland. Hafnarskogur is a 430 ha area (10 km × 1 - 1.5 km) area between Mt. Hafnarfjall and the Borgarfjordur fjord (64°30'N, 21°38'W) (Figure 2.1). Elevation ranges from 2 m in the southern part to 80 m in the northern part. The topography is mostly flat, but toward the north, the terrain slopes gently to the ocean (NW aspect, ~ 3 - 5°). The bedrock is late Tertiary basalt transgressed by the ocean after the last ice age (Einarsson, 1980). Soils have formed in eolian and tephra materials deposited on a 10 000 y old sandy and gravelly shoreline. The soils are Andisols, the dominant soil order in Iceland (Arnalds, 2004). Eroded portions of the landscape are characterized by shallow Vitricryands, with either bare or lag-gravel desert pavement surfaces (Arnalds and Kimble, 2001). Ground cover ranges from woodlands to savannas, wetlands, heathlands, and grasslands.

Hafnarskogur belongs to the Hofn farm, one of the oldest known farmsteads in Iceland (Thorgilsson, 1968). It was settled between 874 - 930 A.D. and has been farmed continuously since. Hafnarskogur was historically characterized by birch (*Betula pubescens* Ehrh.) woodland, as were many other Icelandic lowland areas at the time of settlement (Sigurdsson, 1977). The assumption that the lowlands were dominated by birch woodlands is supported by pollen analyses (Einarsson, 1962; Hallsdottir, 1987),



**FIGURE 2.1.** Hafnarskogur study area in west Iceland. It is located 40 km north of Reykjavík (see insert), between Mt. Hafnarfjall and Borgarfjörður fjord. The farm Hofn can be seen in bottom left.

historical records and woodland remnants (e.g. Bjarnason, 1942; Thorgilsson, 1968; Gudbergsson, 1996), land descriptions dating from the 16<sup>th</sup> century (N.N., 1949), and old place names (Helgason, 1950; Gislason, 1975). Pollen records from a site 15 km north of Hafnarskogur, indicate woodlands were more widespread before settlement than they are today (Hallsdottir, 1995). Also, the name Hafnarskogur (skogur = forest) suggests woodlands once covered areas that are now denuded (Helgason, 1950; Magnusson and Vidalin, 1982). A land description of the Hafnarskogur area from 1707 mentions the declining woodlands, ostensibly the result of overgrazing and fuel harvesting (Magnusson and Vidalin, 1982). Woodlands with large trees (12 - 15 cm stem diameter, 4 m height) were still present at the beginning of the 19<sup>th</sup> century, but had largely vanished by the first half of the 1880s (Thoroddsen, 1913).

Icelandic ecosystems evolved without large herbivores from the end of Pleistocene until the arrival of sheep, goats, pigs and horses with Norse emigrants in the 9<sup>th</sup> century (Adalsteinsson, 1981). Farming required clearing of the woodlands for pasture and haymaking, typically in close proximity to the homesteads. The cleared woodlands were maintained through yearlong grazing and in some cases, burning of grass-litter (Fridriksson, 1978). The remaining woodlands were used for fuel, charcoal and grazing (Thorsteinsson and Olafsson, 1967). Limited haymaking capacity and long winters required farmers to utilize the rangelands for grazing in all seasons, and winter grazing was commonplace until the 20<sup>th</sup> century. The northern part of the Hafnarskogur area was a grazing common for sheep and horses from the time of settlement until it was fenced off in the 1980's. The southern part of the area has been fenced since the middle

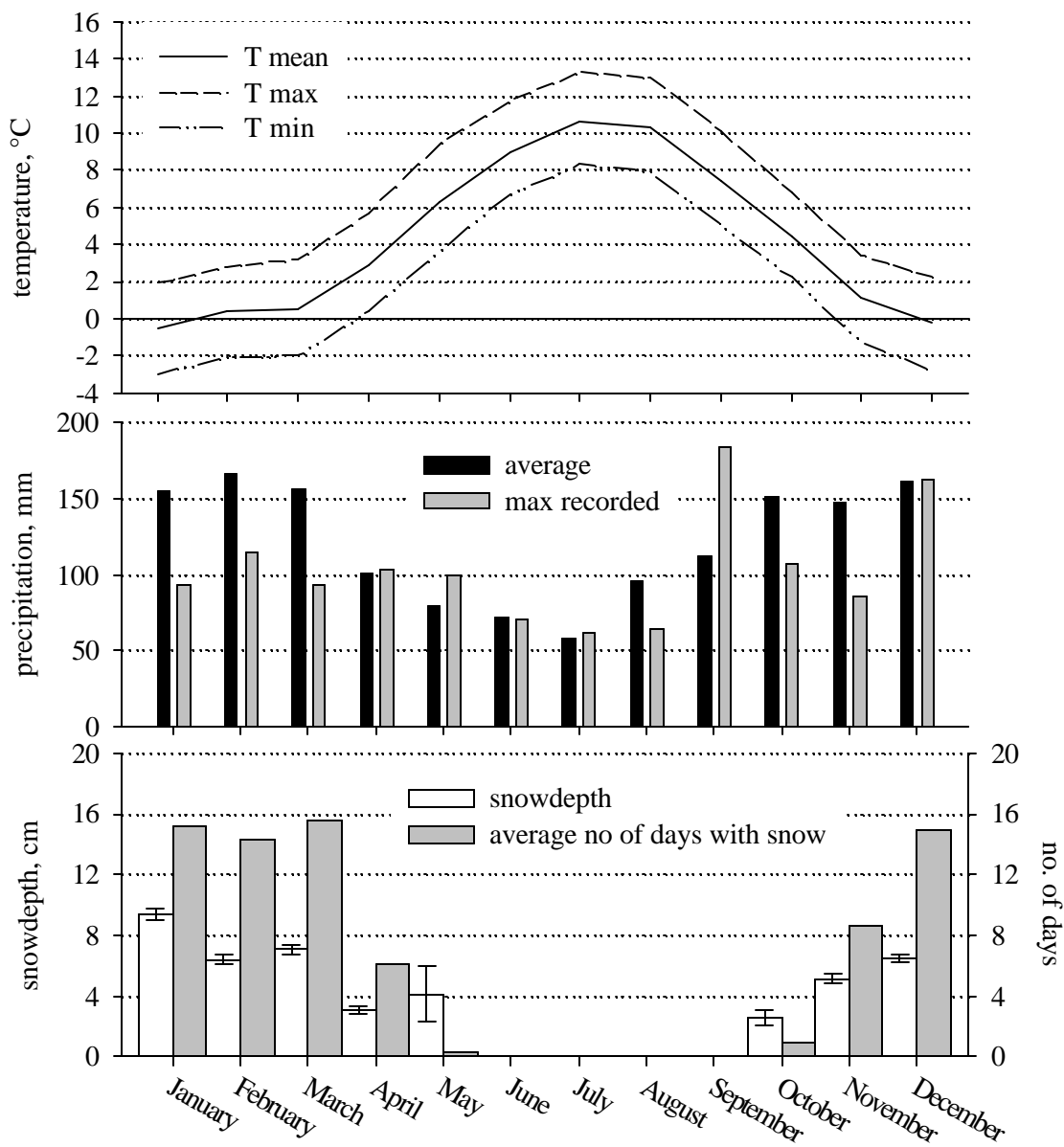
of the 20<sup>th</sup> century, but was grazed intensively by sheep and horses until ca. 1985; and since then only by horses through the time of this writing (G. Jonsson and G. Olafsson, personal communications).

At present, native birch woodlands remain in a portion of the study area and large areas have been severely eroded (Figure 6-3). The most intense erosion appears to have occurred closest to the farmstead, and around an old sheep barn ca. 1 km north of the Hofn farm, suggesting a piosphere grazing effect (Phelps and Bosch, 2002).

The land cover changes and land degradation at the Hofn farm are representative of what is considered to have happened throughout much of Iceland since the onset of human settlement. Therefore, a knowledge of the processes underlying the changes at the Hofn farm will have broad relevance.

### **Weather**

The climate in Iceland is maritime, characterized by cool summers and mild winters (Einarsson, 1984). The Hafnarskogur area has a temperate rainy climate according to the Köppen's classification scheme (Köppen, 1931). Mean temperatures are -0.5 °C and 10.6 °C for January and July respectively (Figure 2.2). Mean annual precipitation is 1460 mm, with monthly precipitation  $\geq$  150 mm in October through March (Icelandic Meteorological Office, IMO; temperature from Reykjavik [1961 - 1990] 40 km South of Hafnarskogur; precipitation from Andakilsarvirkjun [1961 - 2000] 12 km ENE of Hafnarskogur). The winters are characterized by air temperature fluctuations around 0 °C and shallow and ephemeral snow cover (Figure 2.2). Freeze-thaw cycles are thus pronounced and frequent during winter (Einarsson, 1984).



**FIGURE 2.2.** Average climatic conditions in the Hafnarskogur study area. Top panel: Average mean, maximum and minimum monthly air temperature. Middle panel: Monthly precipitation and maximum-recorded daily precipitation in each month. Bottom panel: Average monthly snow depth and number of days recorded having snow each month. Data are from the Icelandic Meteorological Office (IMO): 1961 - 1990 temperature data from Reykjavik, 40 km S of Hafnarskogur; 1961 - 2000 precipitation and 1964 - 1998 snow data from Andakilsarvirkjun, 12 km ENE of Hafnarskogur.

Hafnarskógur is notorious for high winds (Agústsson and Ólafsson, 2005; Thordarson and Ólafsson, 2008), which bring salt over the area in storms and can cause abrasion of vegetation by wind-driven snow and ice particles in winter [commonly known as ‘skaraveður’ (Akerman, 1973)].

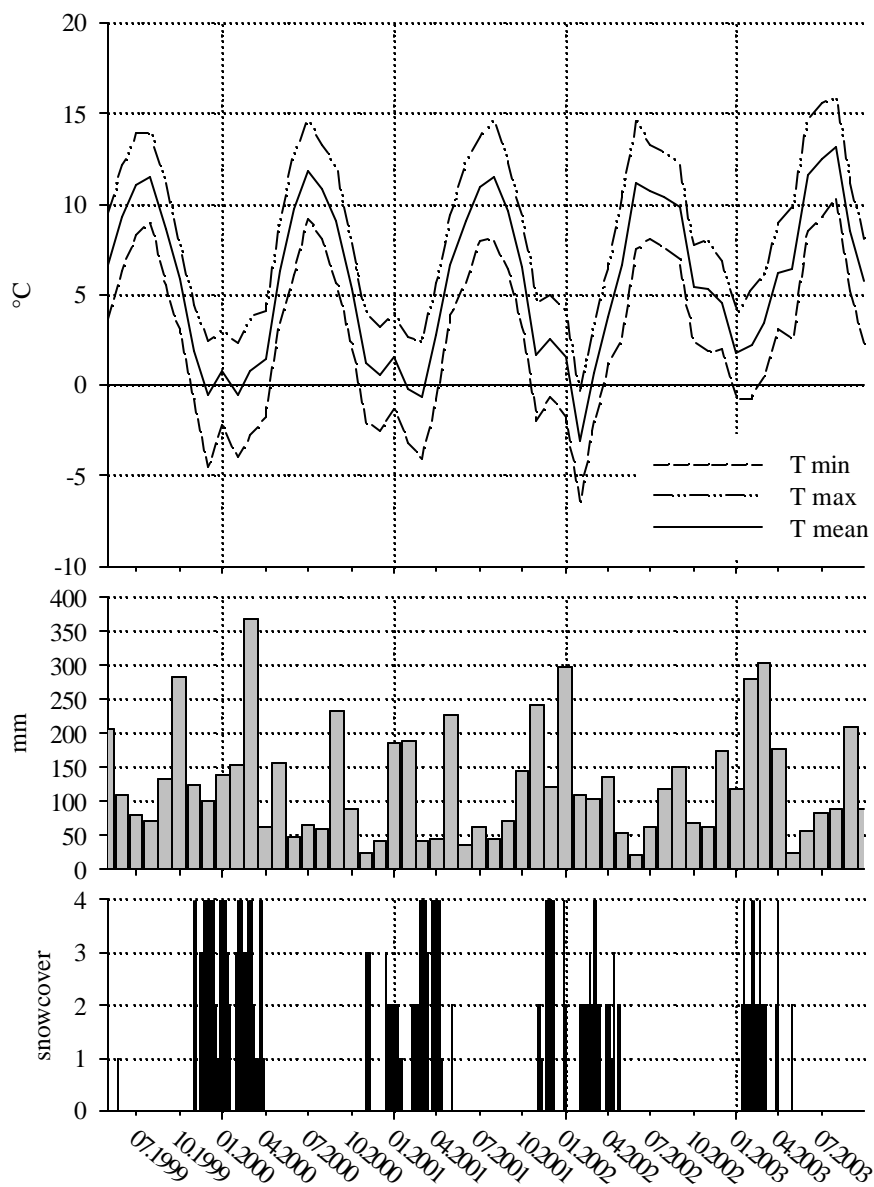
During the study years 1999 - 2003, mean annual temperature (MAT) at Hafnarskógur (weather station Hafnarmelar 5 km SSW of site) was 0.2 °C higher than that in Reykjavík, and 0.9 °C higher than the 30 y (1960 - 1990) Reykjavík average. For the five study years, MAT was highest in 2003 (6.3 °C) and lowest in 1999 (4.5 °C) (Figure 2.3). Annual precipitation ranged from 1360 mm in 2002 to 1774 mm in 2003. Snow cover was greatest during the 1999 - 2000 winter (Figure 2.3), which was colder (Nov - Feb temperatures = 0.4 °C) than the other winters (0.8 °C in 2000 – 01; 0.7 °C in 2001 – 02; 3.5 °C in 2002 - 03). Snow cover was 100 % for 62 days in the 1999 - 2000 winter, but the following winters had only in 25, 19 and 7 days with 100 % cover, respectively. The 2002 - 2003 winter was the third warmest since measurements began in 1920 in Reykjavík, and annual temperature was also a record high in Reykjavík (IMO, 2003).

### **Vegetation Characteristics**

The vegetation can be categorized into three main plant community types:

- i) woodlands, with birch trees (> 2 m) dominating the overstory with a ground layer comprised of graminoids (*Deschampsia flexuosa*, *Festuca vivipara*, *Agrostis capillaris* and *Anthoxanthum odoratum*) ferns (*Gymnocarpium dryopteris*) and mosses (*Racomitrium lanuginosum*); ii) grasslands, dominated by *D. flexuosa*, *D. caespitosa* and





**FIGURE 2.3.** Average mean, maximum and minimum monthly temperatures, monthly precipitation, and daily snow cover at Hafnarskogur during the study: May 1999 to September 2003 (data from IMO). Snow measured as percentage cover: 0 = no snow, 1 = 25 %, 2 = 50 %, 3 = 75 %, 4 = 100 % cover. Temperature data from weather station Hafnarmelar (5 km SSW of site); precipitation and snow cover data from weather station Andakilsarvirkjun (12 km ENE of site). Vertical grid lines separate years.

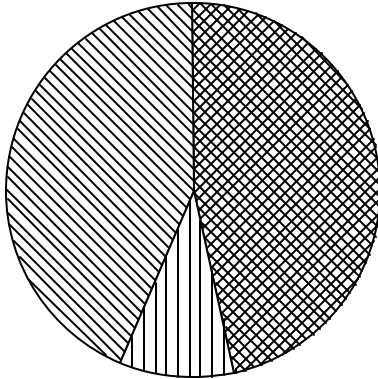
*Agrostis capillaris*, and iii) heathlands characterized by few and widely scattered birch plants (< 1 m), graminoids such as *D. flexuosa*, *D. caespitosa*, *Agrostis capillaris*, *Anthoxanthum odoratum*, *F. vivipara*, *F. richardsonii* and *C. bigelowii*, and the dwarf shrubs *Empetrum nigrum* and *Vaccinium uliginosum*. Parts of the area are wetlands, dominated by graminoids such as *Eriophorum angustifolium*, *Carex nigra*, *C. chordorrhiza* and *Calamagrostis stricta*. Field experiments were not conducted in the wetlands.

Within these three main plant community types, five sub-categories, also referred to as plant communities in the subsequent chapters, can be defined, based on their physiognomy. They are, from north to south: dense woodlands (woodlands), woodland heathlands (also referred to as w heathlands), grasslands, savanna heathlands (also called s heathlands), and open birch savanna, belonging to the woodland category (i) above. Soils of the woodlands and grasslands have been preliminarily classified (Soil Survey Staff, 1999), as Typic Fulvicryands and Histic Cryaquands respectively (Orradottir, 2002). In these five communities, studies on growth and survival of birch seedlings were conducted (Chapter IV); and studies on surface stability, surface strength, and frost heaving (Chapter V), to evaluate livestock potential grazing and trampling effects. Therefore, these five vegetation types are described in detail here below.

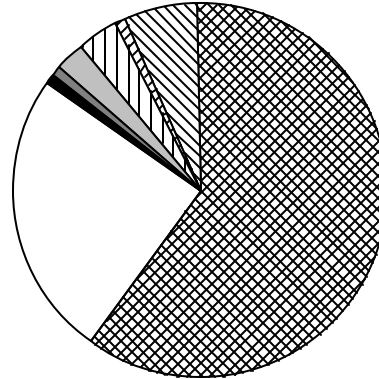
Ground cover in the woodlands, w heathlands, grasslands, s heathlands and savanna differed markedly (Figure 2.4). Vascular plants, mosses and litter were the only cover categories observed in the woodland, savanna and grassland. Vascular plants comprised 61 % and 60 % of the total cover in savanna and grassland communities,

**FIGURE 2.4.** Braun-Blanquet ground cover categories [1 < 1 %, 2 = 1 - 5 %, 3 = 6 - 10 %, 4 = 11 - 15 %, 5 = 16 - 25 %, 6 = 26 - 50 %, 7 = 51 - 75 %, 8 = 76 - 100 %; (Pandeya et al., 1968)] of vascular plants, mosses, lichens, litter, stones and bare soil in the (A) woodlands, (B) w heathlands, (C) grasslands, (D) s heathlands and (E) savanna plant communities in Hafnarskogur. Data are means of visually estimated cover in six 0.5 ´ 0.5 m plots along three transects in each community type (total 18 plots per community), in late July and early August 2002. Braun-Blanquet data were transformed to midpoint percentages before calculating average cover. The vegetation categories used here were selected based on properties considered to be important for surface stability, i.e. presence of roots, and above ground structure.

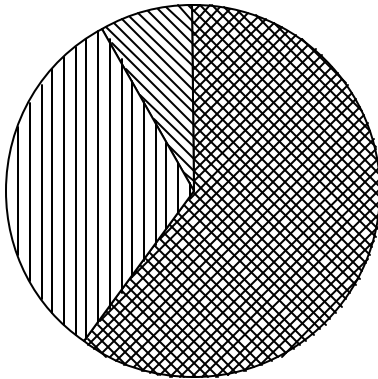
A: woodlands



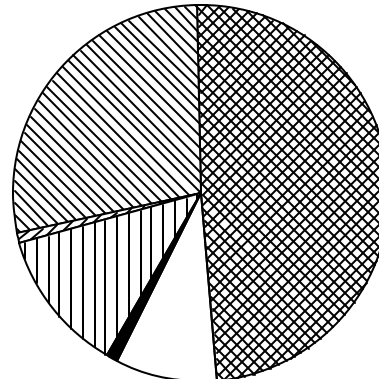
B: woodland heathlands



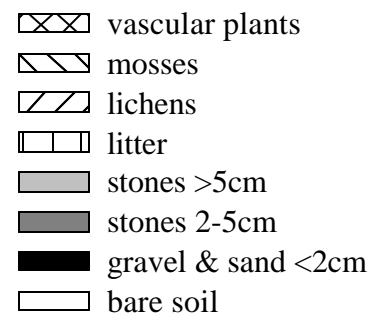
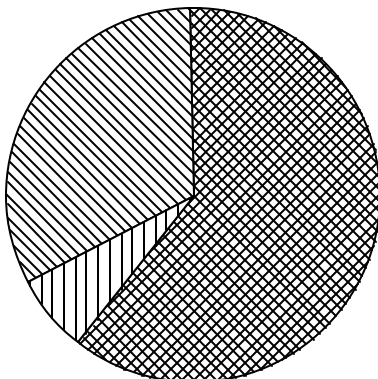
C: grasslands



D: savanna heathlands



E: savanna

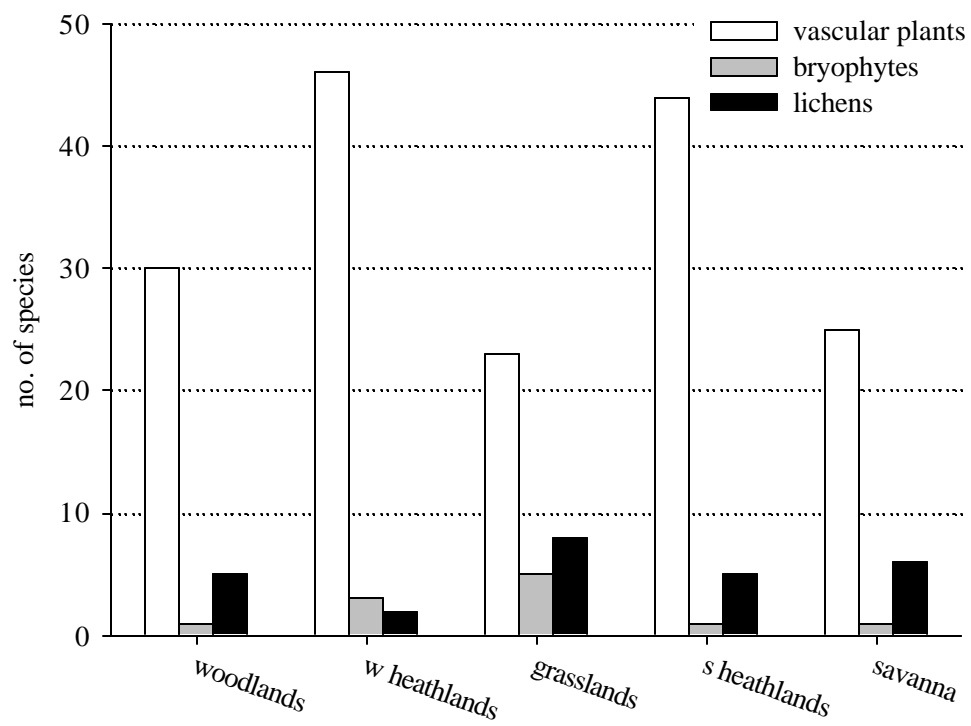


respectively, but only 46 % in woodland communities, where moss cover was more abundant (44 %) than in the savanna (32 %) and grassland (8 %). Bare soil cover was 25 % in woodland heathland, 9 % in savanna heathland, but not observed in other communities. Stones, and gravel and sand cover was 4 % in woodland heathland, 0.9 % in savanna heathland, and was not observed in other communities. Lichens were only observed in the heathlands, but were < 1 % cover.

All vascular plants, bryophytes and lichens observed in the five plant communities were recorded to species, and species diversity was computed as richness (number of species). Vascular plant richness was highest in the w heathland (46) and s heathland (44), and lowest in the savanna (25) and grassland (23) (Figure 2.5). Lichens and bryophytes richness were greatest in the grasslands (Figure 2.5).

### **Woodland Characteristics in the Hafnarskogur Area**

The woodlands in the Hafnarskogur area were characterized by low stature (average height/length is < 260 cm), and multistemmed, flat and shrubby growth forms (Table 2.1). These characteristics are common in Icelandic woodlands, but are particularly widespread in woodlands under strong oceanic conditions (Aradóttir and Eysteinnsson, 2005). The woodland community type had highest tree densities, the tallest trees, greatest stem diameter and canopy diameter (Table 2.1). Furthermore, the trees in the woodlands had on average 1.6 stems per tree, and were characterized by spherical or flat crown shape. Average number of shoots per tree varied from 7.3 to 8.0 for the savanna and s heathland respectively, emphasizing their vegetative renewal from old



**FIGURE 2.5.** Species richness of vascular plants, bryophytes and lichens in the Hafnarskogur woodlands, w heathlands, grasslands, s heathlands and savanna. Note: w heathlands = woodland heathlands, s heathland = savanna heathlands

**TABLE 2.1**

**Averages ( $\pm$  SE) from the *Betula pubescens* survey in the woodland, w heathland, s heathland and savanna in Hafnarskogur. Data were collected from all birch trees in three 10  $\times$  10 m macroplots in each community in late June 2000 and early July 2001; n = 125 for woodland, 85 for w heathland, 9 for s heathland and 77 for savanna.**

Community type	Live trees, m <sup>-2</sup> mean	Dead trees, m <sup>-2</sup> mean	Avg. number of stems per tree mean $\pm$ SE	Avg. number of shoots per tree mean $\pm$ SE	Avg. stem diameter at 0.5 m per tree (cm) mean $\pm$ SE	Avg. height or length per tree (cm) mean $\pm$ SE	Max. canopy diameter per tree (cm) mean $\pm$ SE	Avg. canopy cover per tree (%) mean $\pm$ SE
woodland	0.42	0.04	1.6 $\pm$ 0.09	7.8 $\pm$ 1.42	23.6 $\pm$ 0.74	258.4 $\pm$ 4.18	188.4 $\pm$ 7.30	32.7 $\pm$ 1.40
w heathland	0.28	-	6.0 $\pm$ 1.70	-	1.0 $\pm$ 0.11	43.1 $\pm$ 1.68	72.8 $\pm$ 6.64	37.1 $\pm$ 1.69
s heathland	0.03	-	2.4 $\pm$ 0.75	8.0 $\pm$ 1.00	13.8 $\pm$ 2.95	59.8 $\pm$ 21.98	64.8 $\pm$ 19.79	20.6 $\pm$ 3.38
savanna	0.26	0.20	3.6 $\pm$ 0.57	7.3 $\pm$ 1.41	9.8 $\pm$ 1.10	127.2 $\pm$ 7.81	134.4 $\pm$ 6.96	33.1 $\pm$ 2.60

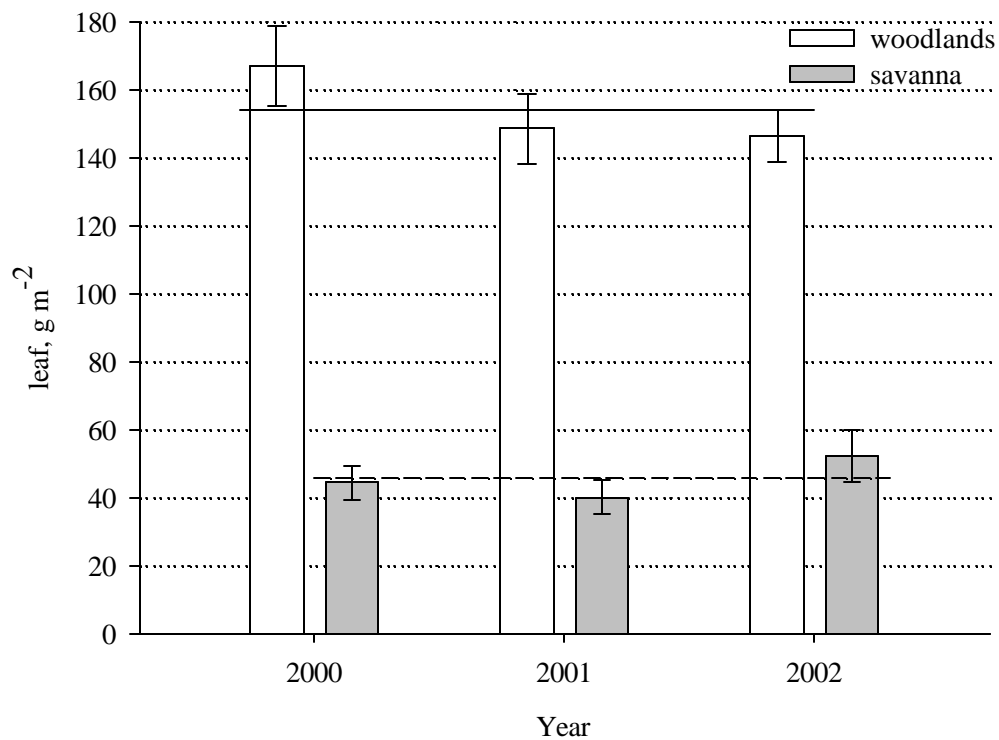
Dashes indicate no data.

roots. In the savanna, trees had more stems (averaged = 3.6) stems; and were shorter-statured than those in woodland communities. The trees in s heathland averaged 60 cm in height and had crown shapes more flat than shrubby, whereas the woodland heath trees were shrubby, multi-stemmed and < 50 cm tall. Flat crown shapes indicates nearly stagnant growth (Aradottir et al., 2001).

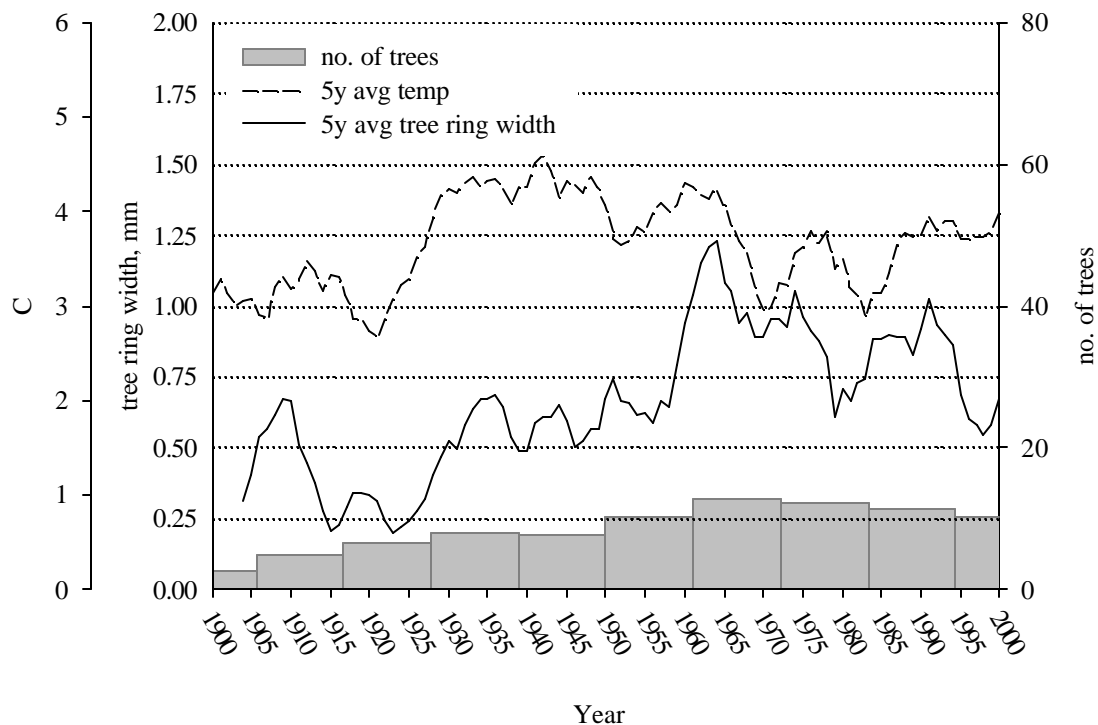
Birch leaf litter was collected in the woodland and savanna communities in late fall 2000, 2001 and 2002. There was about three times more leaf litter in the woodland than the savanna (Figure 2.6), which reflects greater birch tree density in the woodland (Table 2.1). Year to year variability within communities was small.

The age of the Hafnarskogur birch and growth patterns were determined using dendrochronology. Trees in the woodlands, savanna (n = 45 in each community) and the most northern area of Hafnarskogur (n = 17) were cored at 50 cm height in summer and fall 2003, and 5 cross-sections from fall 2002 were collected. Annual rings were narrow in many of the cores, limiting the number of trees that could be used for age determination. Average tree age in the woodland and savanna was 74 and 64 years respectively, indicating that these trees germinated in the 1920's and 1930's or earlier. Given that the renewal of birch trees within stands may be predominantly through formation of new shoots (cf. Aradottir et al., 2001), these trees may be much older than indicated by annual rings. Site chronology for the 20<sup>th</sup> century was built from six cores from the northern area, three cores from the woodland and all the cross-sections (14 in total). Ring widths ranged from a minimum of 0.12 mm to a maximum of 1.65 mm. A five year running average of the ring widths was calculated for the century (Figure 2.7)





**FIGURE 2.6.** Mean ( $\pm$  SE) birch leaf litter biomass in the Hafnarskogur woodlands and savanna communities. Means are from 20 (woodlands) and 17 (savanna) traps placed in three 10  $\times$  10 m plots (macroplots) in each community. Horizontal lines represent three year mean for each community. The traps collected leaves in late summer and autumn 2000, 2001 and 2002. Birch leaves were collected in late autumn each year, then dried at 60  $^{\circ}$ C for 48 hours and weighted.



**FIGURE 2.7.** Five year average tree ring widths in Hafnarskogur for the 20<sup>th</sup> century, based on means from variable number of birch trees (gray bars) that could be cross-dated. Tree rings were counted, and measured under magnification with 0.0001 mm precision using LINTAB measuring table (Accurate Technology Inc.). Five year average air temperature from Stykkisholmur weather station (data from IMO).

to get an estimation of tree growth. Average ring widths were smallest in 1913 - 1927, but largest from 1960 - 1978 and 1984 - 1994 (Figure 2.7). The high growth rates in the 1960 - 1978 period takes place during cold years, but during the warm years of the 1930's and 1940's the growth was about average for the whole 20<sup>th</sup> century. Oscillations in ring growth did, however, track air temperature fairly well (Figure 2.7) but other factors apparently influence the growth.

## CHAPTER III

### HIGH LATITUDE DESERTIFICATION: A STATE-AND-TRANSITION MODEL

#### **Introduction**

Ecosystems are characterized by dynamic fluctuations around nominal means, disturbances, and their interactions. Human-induced disturbances, including various types of land use, are of special interest due to their potential impacts. Predicting and managing changes caused by human activities are critical to resource conservation and sustainability. Around the world, livestock grazing is a predominant land use on 'rangelands', which are landscapes not suitable for row-crop agriculture or forestry. The discipline of 'range management' evolved in response to the widespread degradation of rangelands by livestock grazing in late 1800s and early 1900s (Sampson, 1923; Stoddart et al., 1955; Holechek et al., 2003). Assessing and predicting vegetation response to livestock grazing has long been a concern to rangeland managers. Dyksterhuis (1949) developed a widely-used conceptual model of vegetation management for grazed rangelands based on Clements (1916) theory of climax communities. That model described both grazing-induced retrogression and the successional changes that would be expected to occur subsequent to relaxation of grazing. However, this model did not robustly represent the dynamic nature of ecosystems, especially in drylands; and it did not account for multiple pathways for change nor hysteresis effects (Lauenroth and Laycock, 1989). An alternative approach, now widely referred to as the State-and-Transition (S&T) model, was proposed by Westoby et al. (1989).

As their name implies, S&T models consist of two basic elements: states or plant communities that are discrete and distinguished by differences in structure and rates of ecological processes; and the transitions between them. The transitions are pathways of community change, and qualitative, heuristic assessments of the resilience and resistance of the states. S&T models accommodate discontinuous, reversible and non-reversible vegetation change (Briske et al., 2003) and can be readily constructed for various spatial or temporal scales due to their flexible nature. Like all models, they are limited by the degree of understanding of interaction between ecosystem components and data availability (Herrick et al., 2005). S&T models are flexible and have proven useful for organizing existing ecological information and for representing the current understanding of ecosystem processes in the context of disturbance and land use (Bestelmeyer et al., 2003). As such, S&T models summarize and integrate the best available information into a framework that articulates underlying assumptions, that proposes hypotheses which can be addressed by research, and that serves as a guide for management. Furthermore, S&T models can be readily updated as new information becomes available.

To date, S&T models have been developed primarily for dryland systems in tropical, subtropical and temperate regions (Milton and Hoffman, 1994; Pivello and Coutinho, 1996; Frasier et al., 1998; Oba et al., 2000; Stringham et al., 2001; Asefa et al., 2003). In this chapter I propose a S&T model for a 430 ha area in west Iceland with a long history of land use. Known as Hafnarskogur, this area was historically characterized by birch woodlands. However, since settlement, land cover has shifted

from woodlands to a mosaic of open grasslands, heathlands and wetlands. The land cover transitions in Hafnarskogur mirror what is believed to have occurred in many parts of Iceland since settlement.

### **Study Site**

Iceland is a 103,000 km<sup>2</sup> island on the Mid-Atlantic Ridge in the North Atlantic Ocean, just below the Arctic Circle. It is geologically young and active as is evident by frequent volcanic activity. The dominant soils are Andisols (Arnalds, 2004), which derive their physical properties from volcanic materials (Wada, 1985; Brady and Weil, 1998). Andisols are characterized by low bulk density and low aggregate cohesion, which makes them highly vulnerable to eolian and fluvial erosion (Wada, 1985).

It is commonly believed that the Icelandic lowlands (i.e. < 400 m.a.s.l.) were dominated by birch woodlands (*Betula pubescens* Ehrl.) when Norse farmers first arrived in the 9<sup>th</sup> century (Einarsson, 1963; Olafsdottir et al., 2001). Since the end of Pleistocene, Icelandic ecosystems evolved without large grazers. With the advent of the settlement, sheep, goats, pigs and horses were introduced (Simpson et al., 2004), birch woodlands began to decline (Hallsdottir, 1987, 1992) and severe soil erosion began to occur (Thorarinsson, 1961; Olafsdottir and Gudmundsson, 2002).

Today, almost 40 % of the total land area of Iceland is classified as having considerable to extremely severe erosion; and 10 - 15 % is categorized having limited plant cover and thus at-risk for erosion (LMI, 1993). Less than 5 % of the pre-settlement woodlands remain (Gudjonsson and Gislason, 1998; Aradottir et al., 2001). The drivers of these changes are debated and center around changes in climate (Olafsdottir and

Juliusson, 2000), land use (Thorarinsson, 1961), volcanic activity and Andic soil properties (Arnalds, 2001).

### **Hofn Farm and Hafnarskogur**

Hofn in Hafnarskogur west Iceland is one of the oldest known farmsteads in the country; it dates back to the period of settlement and has been farmed continuously since then (Thorgilsson, 1968). Based on current woodland remnants and pollen records from a site 15 km north of Hafnarskogur, it appears that woodlands were more widespread in the area before settlement than they are today (Hallsdottir, 1995). Contemporary records from the 18<sup>th</sup> century describe woodlands in areas where no birch is found today; and the name Hafnarskogur (skogur = forest) suggests woodlands once covered areas that are now denuded (Helgason, 1950; Magnusson and Vidalin, 1982). Most of the Hofn farm appears to have been part of an extensive grazing commons (Magnusson and Vidalin, 1982). There are thus many similarities between the Hofn farmland and what is considered to have happened in Iceland since the arrival of humans. Thus, an understanding of land cover change and land degradation at the Hofn farm may help us to understand what has occurred elsewhere in Iceland.

Today, the Hofn farmland consists of a mosaic of open grass and heathlands, woodlands, wetlands and severely eroded areas. How this vegetation pattern came to be is open to speculation. Based on elevation and current water levels, it appears that prior to the arrival of man, the area may have been comprised of two main plant community types: woodlands and wetlands. The wetlands would likely have been of two types: lacustrine (or palustrine) fens and slope fens. Topogenic lacustrine or palustrine fens are

characterized by water accumulation in depressions where water movement is very slow or almost stagnant, whereas slope fens are fed by surface flow and interflow from higher elevations (Cowardin et al., 1979). Topogenic fens would likely have been in the southern part of the area, west of an imaginary centerline drawn from north to south, where the elevation is only a few meters above sea level. Today, ponds and water channels with slow-running and stagnant water, and in some cases tide water characterize the topogenic fen area. Landforms classified as topogenic fens were excluded from S&T model.

To the east, closer to Mt. Hafnarfjall, are dry ridges with relatively well-drained depressions between them. The ridges are often eroded down to gravelly substrates. The worst erosion appears to have occurred closest to the farmstead, and around an old sheep barn ca. 1 km north of the Hofn homestead location (Helgason, 1950; Gislason, 1975), suggesting a piosphere grazing effect (Phelps and Bosch, 2002). Birch woodlands and a large slope fen fed by Mt. Hafnarfjall are situated in the northern part of the area, where elevation is higher and landscapes slope gently toward the ocean. The slope fen is well-drained in the upper portion near the mountain and is wettest in the low-lying portion.

### **Model Development**

The ecological and resource management communities are currently grappling with standardizing approaches for developing and using S&T models. Nuances and ambiguity in concepts and terminology contribute to these challenges (Bestelmeyer, 2006; Bestelmeyer et al., 2006; Briske et al., 2006). Development of a S&T model for



the Hofn and Hafnarskogur sites follows the nomenclature, terminology and approach articulated in Bestelmeyer et al. (2003) and Stringham et al. (2003), customized for high-latitude Icelandic ecosystems. Below is a brief overview of S&T terms and concepts that will be used in the Hafnarskogur model.

**States (S)** describe the physiognomy at a given point in time and are distinguished from each other by relatively large differences in functional group or species composition. They consist of two components, the soil and vegetation, which both are integrated through ecosystem processes, and expresses themselves in the physiognomy at any given time (Stringham et al., 2003). States are thus a vegetation-soil complex representing the outcome of interactions between climate, soils, vegetation and land management. States are sometimes defined solely based on the plant community structure; but in cases of total denudation, it may be more appropriate to define states based on surface features other than vegetation. Icelandic ecosystems, as an example, are prone to total denudation or desertification (Arnalds et al., 2001) and referring to such areas as ‘plant communities’ is not logical. It may therefore be more appropriate to recognize ‘surface types’ and their properties. By definition, states are relatively stable (Westoby et al., 1989).

**Phases.** Community phases or seral stages occur within states and are defined based on dominant species (Stringham et al., 2003). Phases typically occur at smaller scales and are governed by different processes than states (Allen and Starr, 1982; King, 1993). They represent dynamic fluctuation in response to external factors, which lack the

intensity or duration needed to cause state shifts (e.g. climate variability, or small-scale, short-term, low intensity disturbances).

**Community pathways** are transitions between phases within states and represent shifts in ecosystem structure (e.g., in plant composition or relative abundance).

Community pathways are reversible within a state, such that altering the intensity of various driving processes will reverse the pathway direction with minimal hysteresis effects, e.g. increasing grazing pressure will lead to increase in unpalatable functional groups; reducing grazing pressure will enable palatable functional groups to regain dominance (Moretto and Distel, 1997; Altesor et al., 1998).

**Transitions (T)** are the trajectories between proposed ecosystem states under the current management and environmental conditions. Halting or reversing state-transitions may require a significant change in management, environmental conditions or substantial cultural energy inputs. Transitions may be gradual and cumulative, or non-linear and characterized by abrupt thresholds. Thresholds have been defined as the point at which one or more key ecological processes change, such that continuation to a new state is likely to occur despite management adjustments. Once a threshold is crossed, soils, seed banks and ecosystem processes will have been altered such that substantial inputs of resources will be required to halt and reverse the trajectory (Archer, 1989; Tausch et al., 1993). At this point, the processes driving change must be modified for the ecosystem to return to its previous state (Stringham et al., 2003). In S&T models, changes in ecosystem function are typically coupled with changes in structure (Briske et al., 2005), but the latter is often overlooked (King and Hobbs, 2006). Changes in

ecosystem structure may affect the capture, retention and processing of water, nutrients and energy and thus influence function in ways that feedback to further impact structure (Ludwig et al., 1997). These structure–function feedbacks may induce non-linearities in state-changes. Thus, it is important that S&T models move beyond descriptions of states, but rather identify and represent processes that drive and result from state changes. In addition to structure–function thresholds, consideration should also be given to resource damage and economic thresholds (Brown et al., 1999).

### **Conceptual background**

Aradottir et al., (1992) developed a conceptual model of birch woodlands degradation. Their model proposes a grazing-induced chronosequence of six states where woodlands are replaced by heathlands, which then transition to degraded barren areas (Figure 3.1 A). The rate of change between these six states is hypothesized to vary. The initial shift between States I and II, where woodlands yield to dwarf shrubs or heathlands, is thought to be gradual and slow; and biotic processes buffer geophysical forces to maintain soil stability. However, this transition is accompanied by an increase in the number of small, bare soil patches or spots [Soil Erosion Spots, SES (Arnalds, 2001)] and loss of woody cover (Figure 3.1 B, I - II). With time and continued grazing, the SES density increases and small SES begin to coalesce. The coalesced SES have greater exposed surfaces and escarpments, making them increasingly susceptible to wind/water erosion. A positive feedback is now initiated, whereby rates of SES expansion increase with increasing size. As these eroded surfaces expand and coalesce, active ‘erosion fronts’ (Arnalds, 2000) develop, further increasing the rate of soil loss

and SES expansion (Figure 3.1 B, III - IV). Geophysical processes now drive the soil erosion; and biotic processes on the remaining vegetated patches cannot halt the march of erosion fronts across the landscape. The degradation process now proceeds unabated to the final state of barren, eroded surfaces (Figure 3.1 B, V – VI). Aradottir et al. (1992) recognize two major factors as the potential triggers for the land degradation sequence in the model: disruption of ground cover due to trampling and grazing, followed by a grazing driven shift in vegetation functional groups that alters site productivity and thermal balance. In turn, these changes are thought to have amplified soil freeze-thaw dynamics that decrease soil stability and promote hummock formation (the latter appearing to have increased since settlement; Ahronson, pers. comm.), both of which render sites more susceptible to losses of soils and nutrients via erosion.

This model proposes functional threshold between States II and III (Figure 3.1 A), where the rate of change shifts from being relatively low and inconspicuous to high and readily apparent, with corresponding soil and nutrient losses. As this threshold is passed, of erosion fronts across the landscape. The degradation process now proceeds unabated to the system shifts from being under the control of biotic processes to one controlled by geophysical processes. Once geophysical processes predominate, the probability of returning to States I or II is greatly reduced even if grazing pressure is relaxed. Furthermore, restoration efforts beyond this point would need to be much more aggressive and hence more expensive to implement; and their success rate would be lower.

**FIGURE 3.1.** A conceptual model of landscape degradation in Icelandic rangelands grazed by livestock. The model proposes a decline from birch woodland with high vegetation cover, high levels of soil nutrients and high levels of nutrient conservation, to barren desert with low vegetation cover, low levels of nutrients and nutrient conservation. From Aradottir et al. (1992) as modified by Archer and Stokes (2000).

**A:** Land cover change (solid line), and the associated restoration cost or energy required for restoring previous states (broken line). Initial stages of degradation are buffered by biotic processes up to a point; but beyond this point, geophysical processes overwhelm biotic processes and lead to accelerated wind and water erosion. The resulting loss of soils and nutrients severely reduces probabilities of recovery.

**State I, Closed-Prime.** Undisturbed or lightly grazed vegetation characterized by high proportion of palatable plant species. Vegetation consists of deciduous shrubs and graminoids. Bare patches not present.

**State II, Closed-Altered.** With moderate grazing, vegetation cover is continuous, but species composition is dominated by grazing avoidance-type species of low productivity, such as small shrubs and mosses. Soil erosion spots (SES) begin to appear.

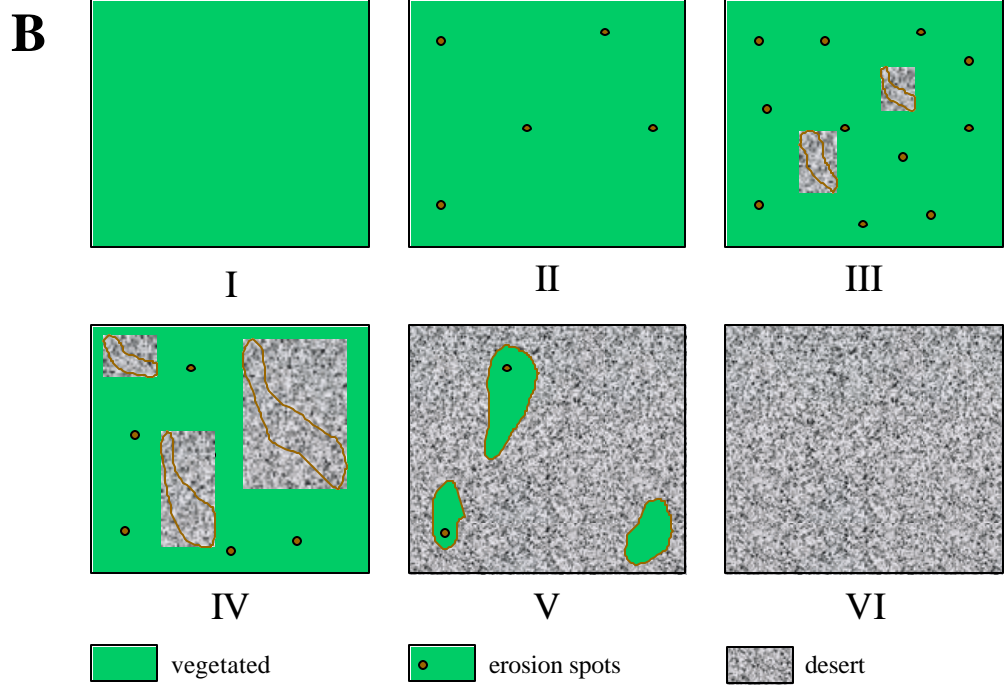
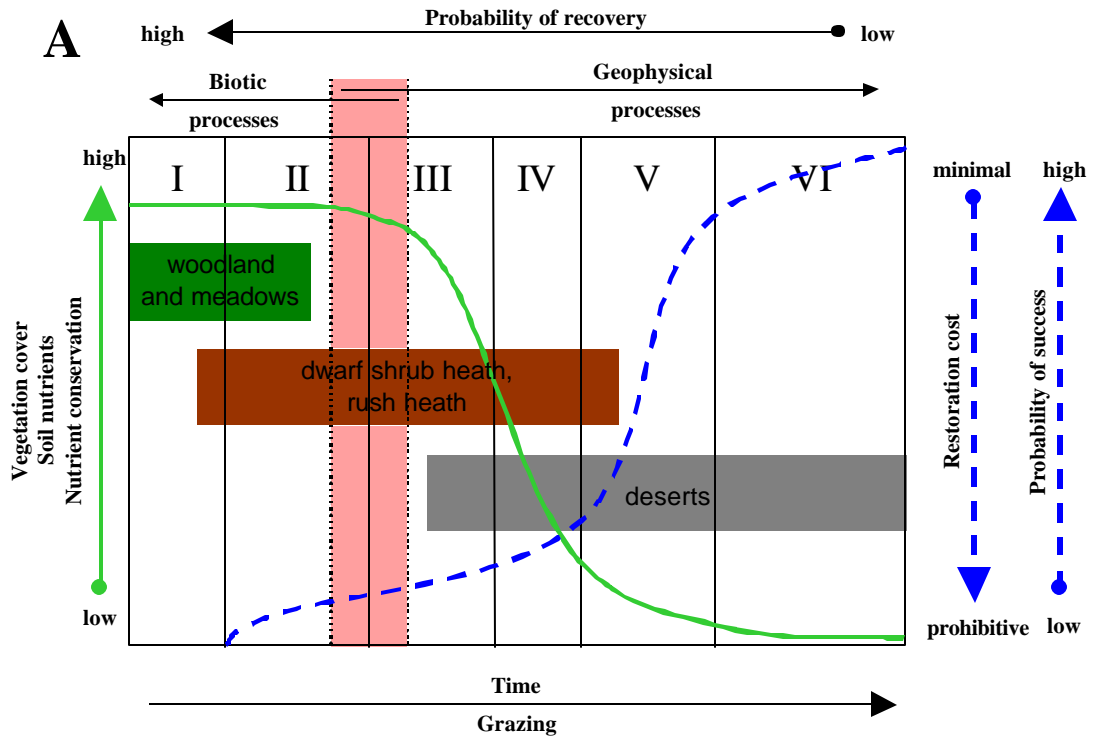
**State III, Spot Erosion.** Botanical composition similar to State II, but soil fertility is reduced. Plant productivity decreases and high rates of plant mortality associated with defoliation and trampling have created unoccupied gaps. SES density increases and their size begins to increase.

**State IV, Bank Erosion.** Vegetative cover ranges from 25 - 85 %. Rills and gullies and slope failures occur across the landscape. SES continue to expand and coalesce.

**State V, Vegetated Remnants.** Land cover has transitioned from a vegetated matrix with erosion spots to a matrix of lithic-barren soils with vegetated remnants dotting the landscape (5 - 25 % coverage). Vegetated remnants highly susceptible to wind erosion along the exposed soil face [rofaboard escarpments, (Arnalds, 2000)] defining their perimeter.

**State VI, Barren.** The final degradation stage; barren land (humid desert). Plant cover is < 5 %, consisting of solitary plants or isolated, small patches.

**B:** Schematic overview of land cover changes as the degradation progresses. Roman numbers correspond to A above.



The sequence of changes proposed in Figure 3.1 is the result of biotic and abiotic disturbances interacting with Andic soil properties and relatively short sub-arctic growing season. It assumes landscapes were dominated by birch woodlands with a nearly continuous ground cover of herbaceous vegetation and mosses prior to the settlement (Figure 3.2, state 1). With the advent of tree cutting and browsing/grazing by sheep and horses, plant communities would shift from birch woodland to communities characterized as open savanna and grassland (Figure 3.2, state 2). The resulting new communities would likely have been dominated by grasses (e.g. *Deschampsia flexuosa*, *Agrostis capillaries* and *Festuca richardsonii*), mosses (*Racomitrium* sp.) and heath-type vegetation (e.g., *Vaccinium uliginosum* L., *Empetrum nigrum* L.) (Figure 3.2, state 3). Livestock browsing of seedlings and shoots of palatable woody species, such as *Betula pubescence*, would have limited woodland regeneration. Opening of the woodland canopy would promote nighttime radiative heat loss and reduce snow depth (McKay and Gray, 1981) with corresponding insulation loss (Hinkel and Hurd, 2006); and grazing and trampling by livestock would similarly reduce the insulative capacity of the ground layer vegetation (Cole and Monz, 2002). Combined with a reduction in snow cover in

**FIGURE 3.2.** A schematic overview of the proposed changes in land cover and land degradation for a high latitude ecosystem.

**State 1.** Pristine birch woodlands with well-developed ground cover consisting of palatable plant species. Following clearing, the land follows one of two trajectories, based on soil hydrology, producing either wetlands (State 2) or dry grasslands (State 3).

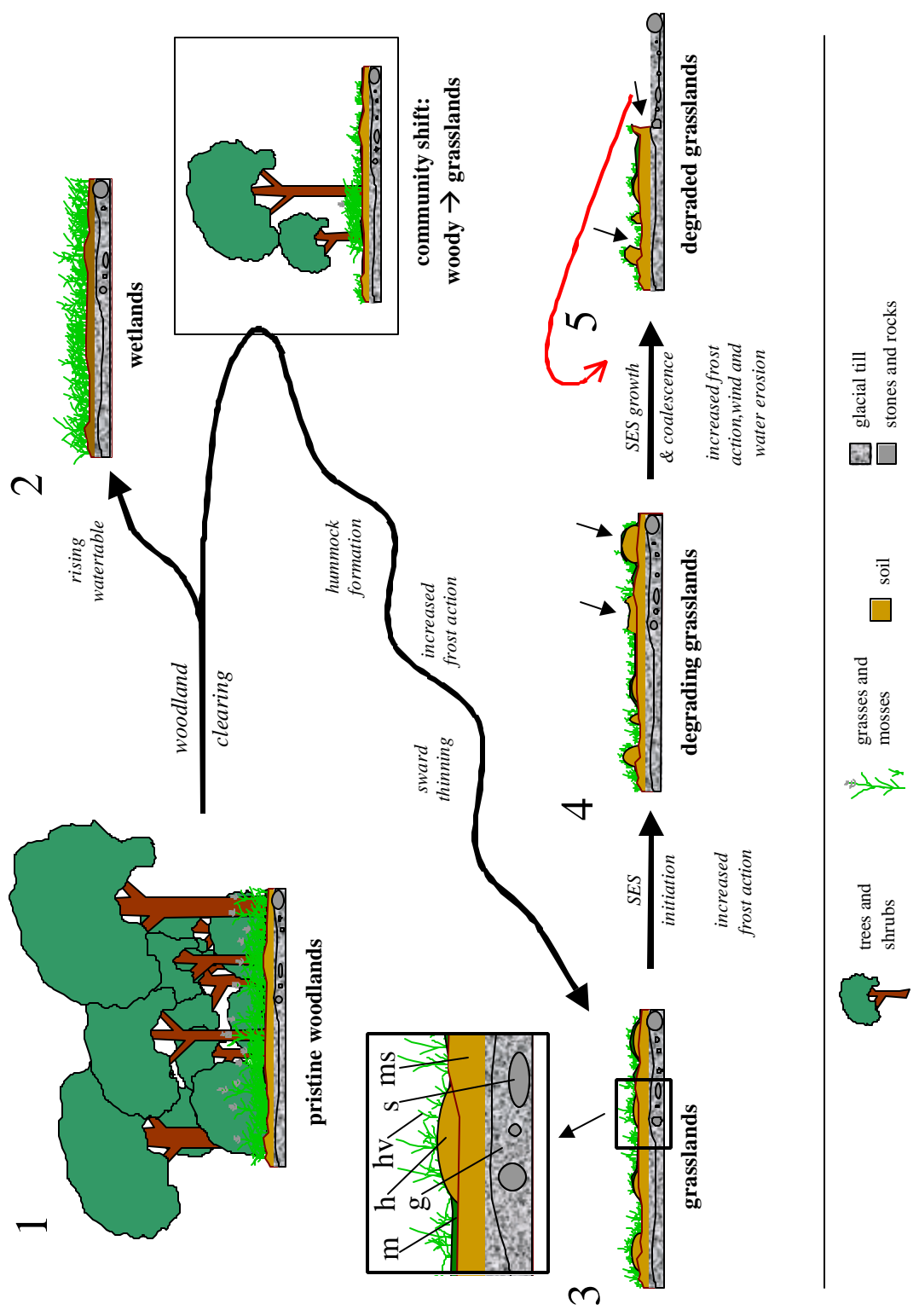
**State 2.** Removal of woody plants in certain landscape locations enables the water table to rise follows causing for formation of wetlands (slope bogs, in this case; see text). In well-drained soils, open birch savannas develop.

**State 3.** Continued cutting and browsing of birch leads to formation of grassland; and continued grazing and trampling reduces ground layer productivity and continuity. This disruption of shrub and herbaceous cover reduces snow accumulation and amplifies diurnal radiant energy flux so as to increase the frequency and magnitude of freeze-thaw events promoting SES and hummock formation, Insert: an enlargement showing different surface and soil profile features: m=mosses, g=gravel, h=hummock, hv=herbaceous vegetation, s=stones and rocks, ms=mineral soil.

**State 4.** SES and hummocks increase in size and density driven by cryoturbic processes accentuated by declines in ground cover and biomass under continued grazing. Trampling by livestock or frost damage may expose mineral soil on hummock (Figure 3.3, arrow).

**State 5.** SES begin to coalesce and thus accelerating the erosion process. At this stage the system will enter a positive feedback loop where SES will grow and merge at an increasing rate as the SES perimeters form distinct escarpments, thus providing more exposed surface area for wind and water erosion (arrows). These escarpments represent erosion fronts or rofabard (see text). Once these erosion fronts form abiotic processes (wind speed and direction, rainfall intensity, freeze-thaw cycles) will dictate rates and patterns of erosions regardless of management on the remaining vegetated portions of the landscape.





afforested areas, these would be expected to promote and intensify cryoturbic processes that adversely affect seedling establishment and soil stability (e.g. needle-ice, frost boils (McCarthy and Facelli, 1990; Goulet, 1995; Defosse et al., 1997; Oddsdottir et al., 1998; Loffler, 2000)) and therefore both reduce probability for woodland regeneration or SES recovery (Shimano and Masuzawa, 1998).

Intensifications of cryoturbation processes and direct grazing impacts associated with trampling and hoof action would combine to promote the formation of the small SES (Figure 3.2, state 4, arrows). Soils in these bare patches will be unstable and frost heaving may be intensified in the vegetation mat surrounding the SES, potentially destabilizing plants near the SES perimeter and reducing their ability to persist and bind soil. The patches may thus begin to expand in size, fueled by small-scale wind and water erosion.

Hummocks are a striking feature of many Icelandic landscapes. They have come to be known by their Icelandic name 'thufur' (þúfur) in the literature (Schunke, 1977; Schunke and Zoltai, 1987; Van Vliet-Lanoe et al., 1998; Grab, 2005). Hummocks are the product of interactions between soil texture, water content, and frost activity (Grab, 2005). Declines in thermal barriers provided by woody plants and ground cover would be expected to intensify cryoturbation and promote the formation of more, larger and steeper hummock forms.

I hypothesize that hummock formation is an important stage in the proposed degradation sequence, as they may increase vulnerability of the community to disturbance (Arnalds, 1994). Initiation of SES can result from biotic and abiotic forces.

Wind-blown ice particles from frozen snow surfaces (commonly known as ‘skaraveður’) are an example of an abiotic agent of disturbance. These ice particles can abrade the vegetation mat, thereby exposing the mineral soil (Olafsdottir and Juliusson, 2000). By virtue of their elevated stature, hummocks would be more likely to intercept blowing ice particles than surrounding vegetation; and when desiccated in winter, they would be more vulnerable to this disturbance (e.g., Akerman (1973). Herbivory is a well-known example of a biotic disturbance factor. In high latitude systems, large herbivores can damage hummocks and expose soils through ‘side-stepping’ (Figure 3.3), as is often evident along sheep trails in grazed, hummocky landscapes. As new SES form the probability of coalescence with existing, expanding SES will increase. As the perimeter length and the steepness/height of the perimeter ‘face’ increase, escarpments form. At this point, SES become hypersensitive to wind and water erosion; and their expansion will be driven by geophysical processes that cannot be mitigated by the remaining plant cover (Figure 3.2, state 5).

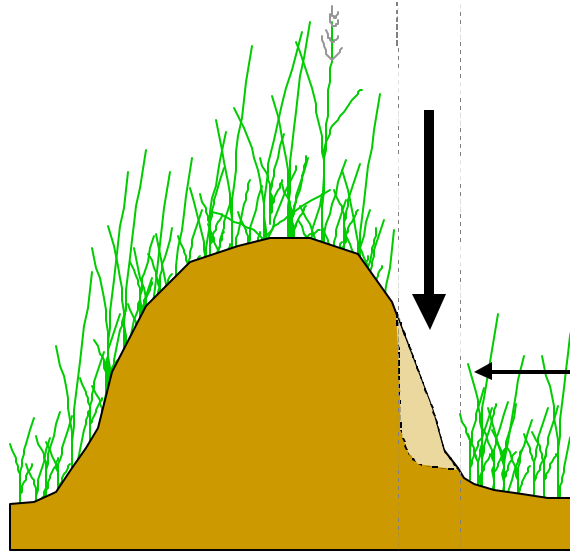
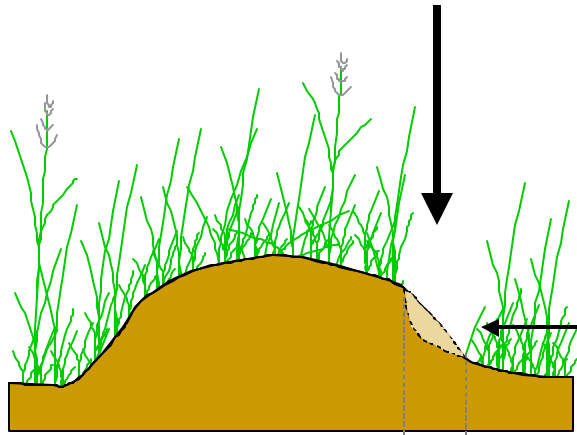
The conceptual models in Figures 3.1 and 3.2 represent a host of hypotheses based on field observations, space-for-time substitution and limited quantitative data. The next section describes a S&T model for desertification of lowlands in SW Iceland based on these graphic models. The proposed S&T model is offered as a first step toward formalizing a spatially explicit process model of high latitude desertification that can guide and prioritize research and management.

**FIGURE 3.3. Side-stepping - disturbance that results when large grazers step on the steep sides hummocks and disrupt plant cover to expose mineral soil. The potential damage caused by this depends on the hummock shape.**

**A. Upper schematic: Low, oval hummocks will suffer a relatively little damage due to their lower profile and because the vegetation cover is relatively thick relative to their height, which adds additional surface strength. Lower schematic: comparatively larger side-stepping scars form when hummock is taller with steeper sides. The resulting disturbances are more orthogonal to wind/water forces and are thus more susceptible to erosion.**

**B: Typical unobvious hummock scars caused by side-stepping.**

**A**



### **State-and-Transition Model for Hafnarskogur**

The purpose of constructing a S&T model for Hafnarskogur is to (a) help us to understand the mosaic of land cover types on present-day landscapes; (b) identify research needs and priorities; (c) codify a basis for a future quantitative model of plant community dynamics; and (d) provide a tool which can be used to anticipate changes likely to occur under specific management regimes or future environmental conditions. The model is based on field surveys in an area that was used as a common grazing land for sheep and horses grazing until in the 1980's when it was fenced and has been used by 30 – 40 horses since (Aspelund, Pétursdóttir, pers. comm.).

#### **States**

Five states (S) are recognized in the proposed model (Table 3.1; Figures 3.4 and 3.5), but not all are necessarily present at a given time. S1 exists where most of the existing woodlands are open to grazing and they can be quite heavily grazed before deforestation takes place. They can thus be somewhat degraded at this stage, however, they probably have high resilience, hence belong to S1. The other four are considered a degradation sequence associated with livestock grazing, tree cutting, land clearing, fuel harvesting, or combination of these.

**S1: Birch communities.** Phases in S1 (Figure 3.5) are dominated by birch woodlands. Historically considered as the climax plant communities in the Iceland lowlands (Bjarnason, 1942; Thorarinsson, 1974; Bjarnason, 1979), they typically have lush herbaceous ground cover which is an impediment to birch seedling establishment (Kinnaird, 1974; Magnusson and Magnusson, 1990; Aradottir, 1991).

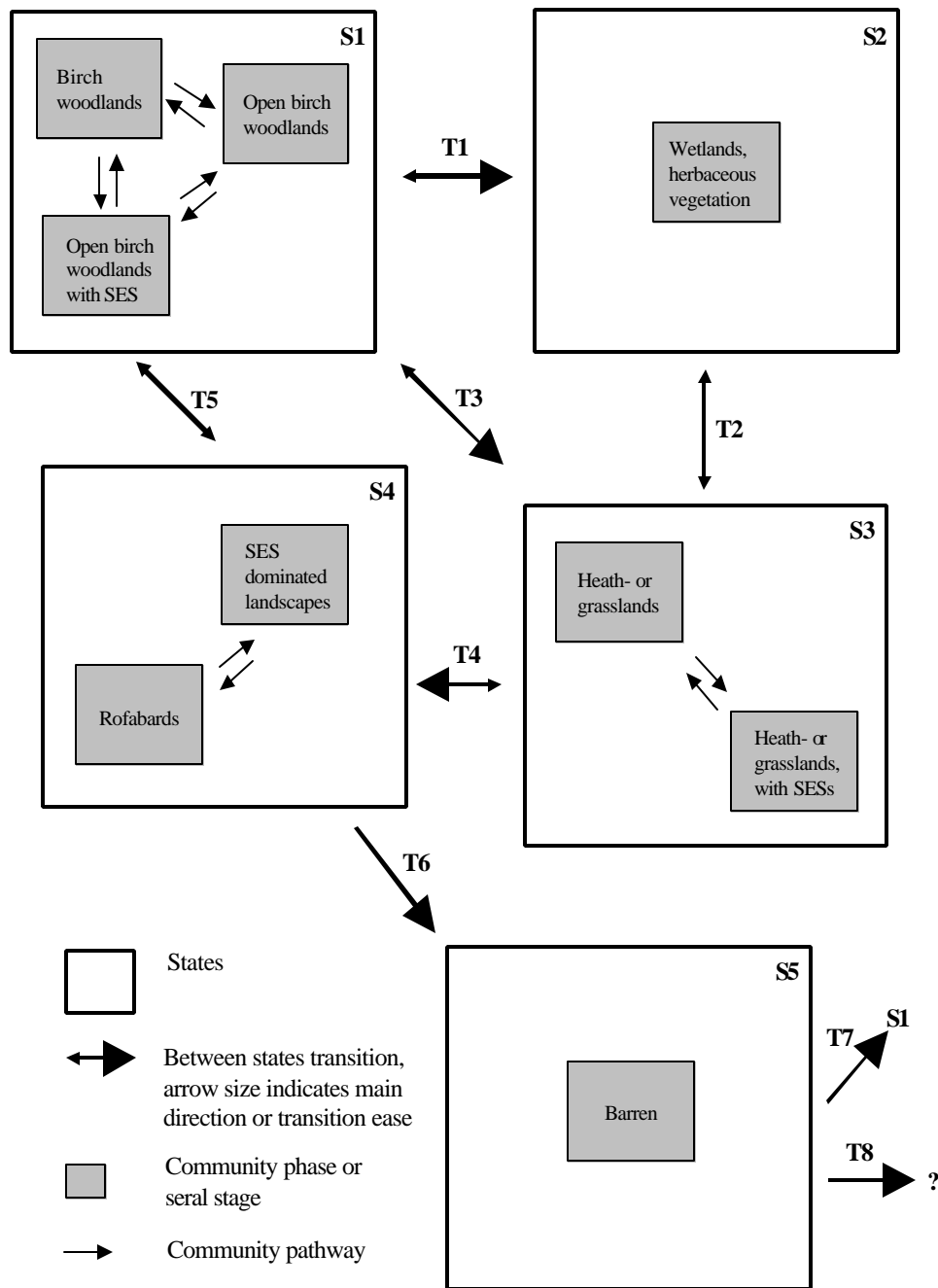
TABLE 3.1

**Definitions of vegetation states and main species occurring in the Hafnarskogur state-and-transition model. See Figure 3-5 for further clarification.**

State	Definition and typical plant species
S1	Birch ( <i>Betula pubescens</i> ) woodlands; areas with trees taller than 1.5m on average, and over 50% canopy cover. Typical groundcover species: <i>Deschampsia flexuosa</i> , <i>Agrostis capillaris</i> , <i>Gymnocarpium dryopteris</i> and <i>Anthoxanthum odoratum</i> . <i>Hylocomium</i> sp. moss is common.
S2	Wetland communities; vegetation dominated by sedges, rushes and grasses. Typical species: <i>Carex nigra</i> , <i>C. chordorrhiza</i> , <i>Eriophorum angustifolium</i> , and <i>Calamagrostis stricta</i> .
S3	Heath- or grassland communities; vegetation dominated by perennial grasses, heath and occasionally mosses Typical species: <i>Empetrum nigrum</i> , <i>Deschampsia flexuosa</i> , <i>Vaccinium uliginosum</i> , <i>Carex bigelowii</i> and <i>Agrostis capillaris</i> . <i>Betula pubescens</i> is present. <i>Racomitrium</i> sp. moss is common.
S4	Soil erosion spot (SES) cover > 30%, some SES 5m <sup>2</sup> . Typical species: Composition contains elements of S3 and S5.
S5	Barren or denuded areas surface types with very limited vegetation cover (<5%). Typical species: <i>Cardaminopsis petraea</i> , <i>Armeria maritima</i> , <i>Silene uniflora</i> , <i>Oxyria digyna</i> and

**FIGURE 3.4. State-and-transition (S&T) model for Hafnarskogur, west Iceland. The model consists of five states (S1 to S5) and eight transitions (T1 to T8). States are distinguished from each other by differences in functional group composition or surface types (Figure 3.4, Table 3.1). Within each state, community phases represent sub-states or seral stages, which can occur depending on external factors. At time-scales relevant to land management, transitions can potentially be uni- or bi-directional, as indicated by arrows. Reversing between state transitions typically requires aggressive intervention (Figure 3.1).**







**FIGURE 3.5.** Birch (*Betula pubescens* Ehr.) woodlands in Hafnarskogur, representing state 1 (Figure 3.2). Note the dense ground cover and total absence of SES.

**Insert:** Generalized characterization of the resistance (degree of displacement along the Yaxis) and resilience (time required to gain new steady state) of this community to environmental stress or disturbance (D). Dashed line indicates the hypothetical functional threshold between biotic and abiotic process domains.

Birch regeneration is thus primarily vegetative (Pigott, 1983; Aradottir and Arnalds, 2001). As individual birch plants die, the herbaceous layer in the resulting gaps may experience harsher environmental conditions (warmer and drier in summer; colder in winter), which may result in the local formation of small SES.

Birch litterfall creates layers on the forest floor every autumn. Leaf-litter layers have been observed to delay soil thawing in the spring (Sartz, 1957) and thus provide thermal insulation that may reduce the frequency and magnitude of freeze-thaw events during the winter and hence stabilize surfaces. Grass-litter or old hay is also known to have similar effects and is often used by the Icelandic Soil Conservation Service in land reclamation projects (Svavarsdottir et al., 2006). Litterfall from birch trees may stabilize SES formed in canopy gaps and thereby play an important role in the open birch woodland ↔ birch woodland phase shift. Deciduous woodlands also tend to trap snow (McKay and Gray, 1981), thus providing an additional thermal layer compared to open landscapes lacking an arborescent strata, with increased probability for seed and seedling survival (Shimano and Masuzawa, 1998).

**S2: Wetland communities.** The slope fen wetland state (S2) consists of a single phase dominated by herbaceous species (Figure 3.6). Drylands are preferred over wetlands by livestock (Thorsson, unpublished data). Wetlands are thus exposed to less grazing disturbances than drylands. As such, SES seldom form and the state is dynamically stable. Furthermore, S2 sites may act as a sink that accumulate soil particles eroding from surrounding areas (Thorarinsson, 1961). Small (< 0.5 m in height) birch plants, occur in these slope fen areas, and seem to have persisted for a long



**FIGURE 3.6.** The slope fen in Hafnarskogur, representing state 2 (Figure 3.2). Small birch shrubs and lack of SES are apparent.

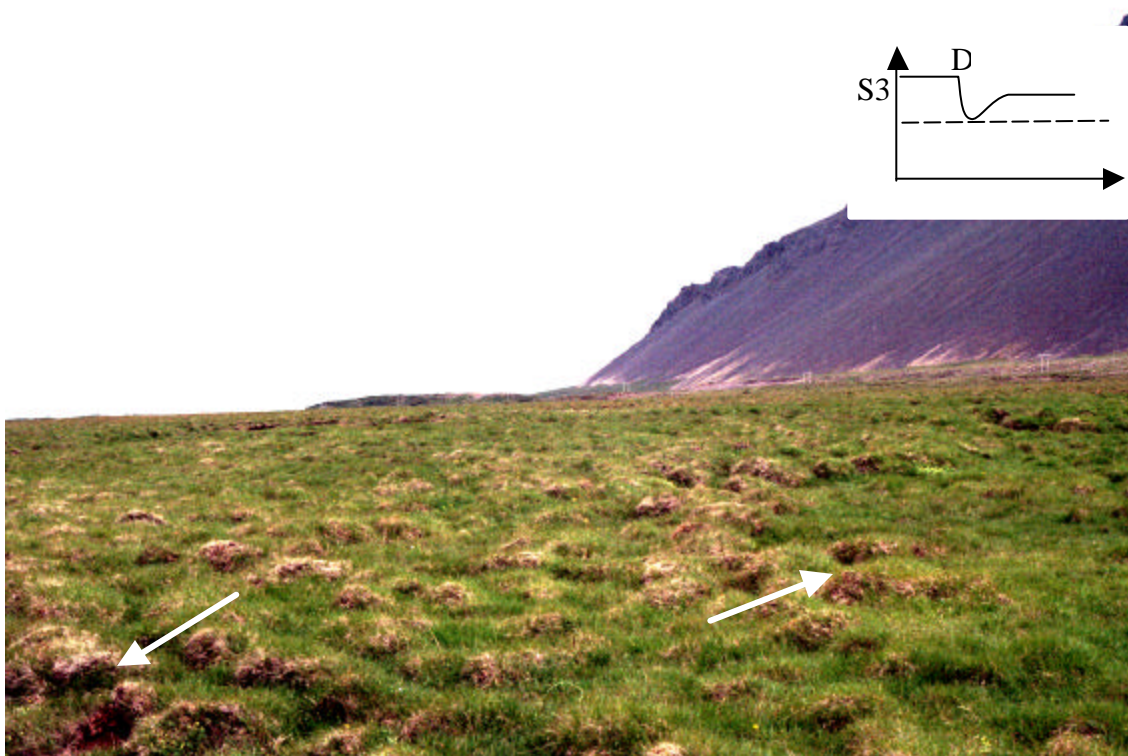
**Insert:** Resistance and resilience of this state are deemed similar to those for state 1. See Figure 3.5 legend for insert explanation.

time, suggesting this area may have been forested previously, as historical annals suggest (Aspelund pers. comm.). It is therefore postulated that deforestation caused the water table to rise, creating hydrologic regime that now favors graminoids and presents a barrier to the development of woodlands (Weltzin et al., 2000). A shift from dryland vegetation types to species favoring wetlands would have followed the changing ground water level (Jauhiainen et al., 2002). This state is considered very stable (see T2 and T3 below).

**S3: Heathland or grassland communities.** This state consists of phases with and without SES (Figure 3.7, and State II in Figure 3.1). The SES are the product of grazing impacts that promote cryoturbic processes as a result of biomass removal and trampling (Figure 3.2, state 3 and Figure 3.3). They may also be inherited from the communal phase in S1, thus not forming exclusively at the S3 state. With a reduction or exclusion of grazing, SES will heal and decrease in number and size and may eventually disappear.

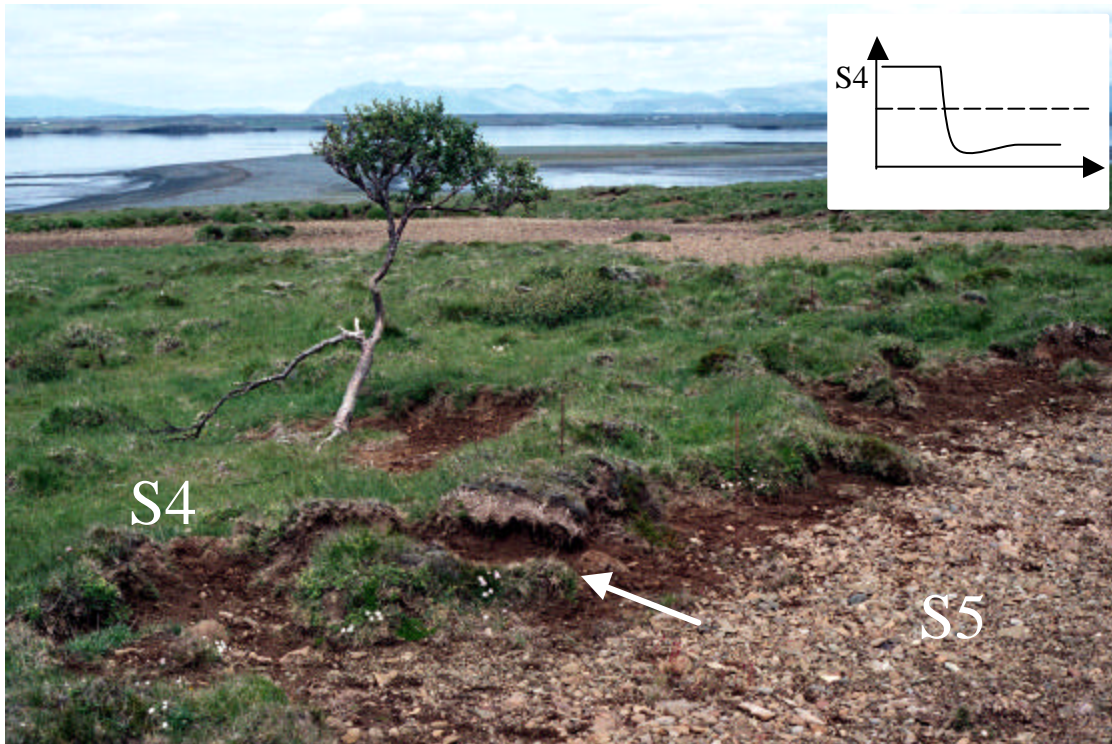
**S4: SES surface types.** This state is characterized by its abundance of SES and active erosion (Figure 3.8, Figure 3.1 III - IV, Figure 3.2, state 3). SES are actively expanding and coalescing, creating both larger eroded surfaces and perimeters, which may develop into erosion pedestals, or 'rofabards' where the entire soil profile is exposed and both wind and water erosion are active (Arnalds, 1990, 2000). The dynamics of this state are dominated by geophysical forces (wind, water, temperature) interacting with Andisol physiochemical properties to amplify the frequency and magnitude of freeze-thaw dynamics. Grazing of S4 further accelerates the degradation





**FIGURE 3.7.** Grassland in Hafnarskogur, representing state 3 (Figure 3.2). Small SES are starting to form and damaged hummocks, possibly due to side-stepping (Figure 3.3), are evident. Right-hand arrow points to a small side damage; the left arrow denotes more extensive damage and formation of a soil erosion spot (SES).

**Insert:** This state has lowered resistance and resilience compared to S1 and S2, and will not return to the same state if exposed to disturbance D. This community is at risk of crossing the functional threshold (dashed line). See Figure 3.5 legend for insert explanation.



**FIGURE 3.8.** Degraded grassland in Hafnarskogur, representing state 4 (Figure 3.2). Note remnant birch plant, the large SES to its right, and the extensive SES in the lower right and background portions of the image that have formed from the growth and coalescence of small SES such as those depicted in Figure 3.7. The arrow points to an escarpment constituting an erosion front or ‘rofabard’, where erosion is active and ongoing. Note the dead vegetation along the escarpment where erosion exposes roots, leading to plant desiccation which reduces erosion resistance considerably. Inset: These sites have crossed the functional threshold (dashed line) and geophysical processes are now driving land cover change. Vegetated areas in S4 will eventually succumb to massive erosion unless rofabards and SES are somehow stabilized.

See Figure 3.5 legend for insert explanation.

process by facilitating SES appearance, expansion and coalescence, and rofabard destabilization (Arnalds, 2000).

**S5: Barren or denuded areas surface types.** This single-phase state is the outcome of the accelerated erosion initiated in S4, and is characterized by total denudation (Figure 3.9, States V–VI in Figure 3.1, Figure 3.2, state 5). On S5 sites, vegetation cover has been totally removed and only the mineral surfaces, glacial till or frost-heaved gravel remain.

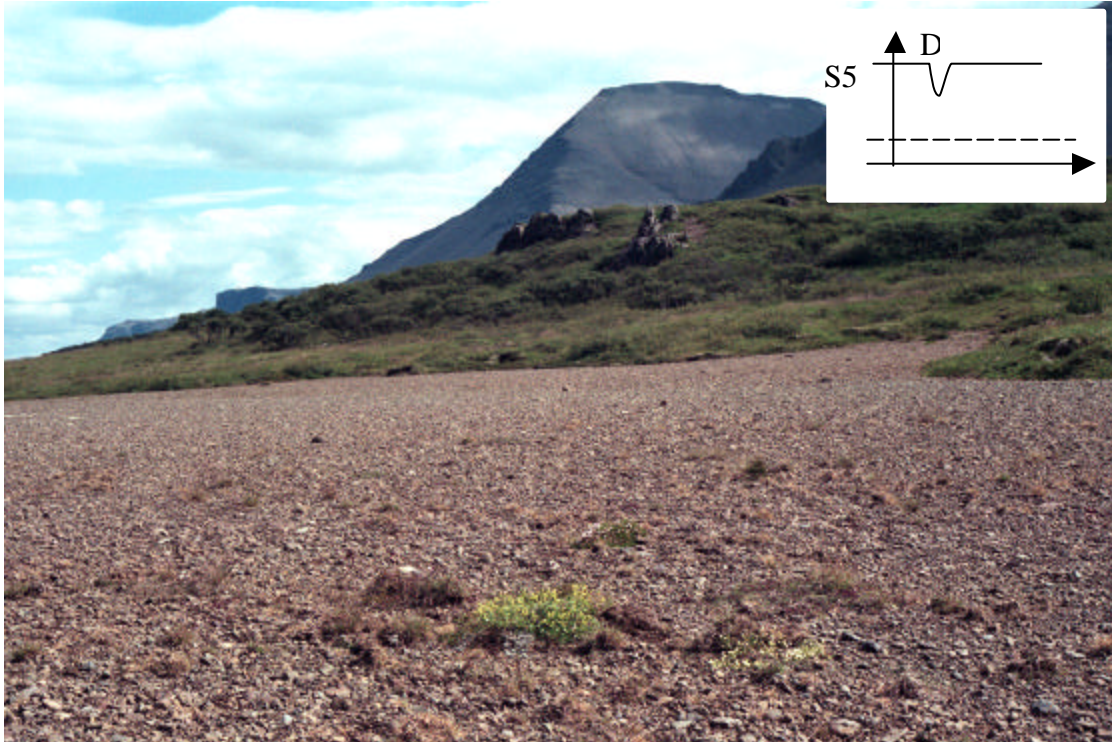
### **Transitions**

The model includes eight transitions (T1 to T8) between the five proposed states. Some transitions will have a higher probability of occurrence than others; some may proceed more rapidly than others; and some may be linear and others non-linear. The nature of these transitions will be mediated by climate, environmental conditions and management activities. For example, T1 (birch woodlands to wetlands), may be related to spatial variability in the intrinsic depth of the water table and contingent upon ground water levels being actively suppressed by woody vegetation transpiration, and a thinning of woodlands sufficient to enable water levels to rise. On the other hand, T3 and T4 (shift from S1 woodlands to S3 heathland or grasslands, or S4 SES landscapes) may be more probable, as they require only chronic grazing disturbance and periods of adverse climate.

The general mechanisms driving transitions are both biotic and abiotic and may operate across co-occurring states within an ecological site (Table 3.2). For example,



the transitions from S1 woodlands to S2 wetlands or S3 heathland/grasslands (T1, T3) are



**FIGURE 3.9.** Barren areas in Hafnarskogur, representing state 5 (Figure 3.4) where land degradation and erosion has resulted in loss of much of the mineral soil and nutrients, leaving a frost-heaved, gravely, oligotrophic substrate behind. High cryoturbic activity on these sites hinders plant establishment on the one hand, but creates potential microsites for seed germination and seedling establishment on the other hand. S5 sites are highly resistant to change; and their restoration is expensive and high risk. See Figure 3.5 legend for insert explanation.

driven by biotic disturbances (tree removal, heavy continuous grazing), accentuated by concomitant abiotic factors such as water table rise and late spring thaw (wetlands), and augmented soil temperature fluctuation and impermeable soil frost formations (heathland/grasslands) (Orradottir et al., 2008). By contrast, geophysical forces drive T4 and T5 transitions and the geophysical forces operating at S5 may have a direct impact on neighboring S4 elements (Figure 3.8).

**T1: birch woodlands (S1)  $\leftarrow$  wetlands (S2).** The main direction of this transition is S1  $\rightarrow$  S2. T1 occurs when ground water levels, suppressed by woody vegetation, rise after the trees are removed (Williams and Lipscomb, 1981; Walker et al., 1993; Sun et al., 2000). The reverse transition is much less likely and may require management intervention (e.g., draining) and cultural energy inputs to enable reestablishment of birch woodland.

Although *B. pubescens* is most abundant on well-drained uplands, it also colonize on hydric soils (Magnusson and Magnusson, 1990) and commonly occurs on moderately well-drained wetlands in southern Iceland (Thorhallsdottir, unpublished data). However, while *B. pubescens* can tolerate such conditions, they are uncommon in

TABLE 3.2

**Transitions between vegetation states defined in Table 3.1 with an indication of the transition occurring under current management regime. See Figure 3.4 for further clarification.**

Transition	Cause and estimated probability
T1 S1 @ S2	<i>Cause:</i> changes in soil water as birch ( <i>Betula pubescens</i> Ehr.) density decreases following cutting, burning or grazing, thus raising the water table to, or above, the surface. <i>Probability:</i> medium for areas exposed to grazing only, high when trees are actively harvested or burned in the grazed areas; very low on gravelly ridges.
S2 @ S1	<i>Cause:</i> lowering of soil water table, either due to draining or increased birch density <i>Probability:</i> very low.
T2 S2 @ S3	<i>Cause:</i> lowering of soil water table caused by draining <i>Probability:</i> very low.
S3 @ S2	<i>Cause:</i> elevation of soil water table caused by water re-channeling <i>Probability:</i> very low.
T3 S1 @ S3	<i>Cause:</i> expansion of open areas due to birch removal or grazing induced die-off, thus changing the vegetation from woodlands to grass- or heathlands. <i>Probability:</i> medium for areas exposed to grazing only, high when trees are actively harvested or burned in the grazed areas.
S3 @ S1	<i>Cause:</i> decreased grazing intensity coinciding with available birch seed sources, safe sites for the seed to germinate and favorable climate. <i>Probability:</i> low.
T4 S3 @ S4	<i>Cause:</i> increase in soil erosion spot formations (SES) due to continuous grazing and cryoturbations. <i>Probability:</i> high.
S4 @ S3	<i>Cause:</i> grazing removal or maintained at very low intensity coinciding with favorable climate and sufficient seed availability. <i>Probability:</i> medium.
T5 S4 @ S1	<i>Cause:</i> decreased grazing intensity, especially when it coincides with favorable climate conditions (increased annual average temperatures and sufficient precipitation). Approximate seed source of <i>B. pubescens</i> must be present. <i>Probability:</i> low.
S1 @ S4	<i>Cause:</i> deforestation and intense grazing. <i>Probability:</i> low.
T6 S4 @ S5	<i>Cause:</i> SES expansion and coalescence due to intensified cryoturbations and wind and water erosion, facilitated by continuous grazing. <i>Probability:</i> medium to high.
S5 @ S4	Does not exist as SES or rofabards do not form in already barren areas.
T7 S5 @ S1	<i>Cause:</i> very low grazing intensity, coinciding with favorable climate conditions (increased annual average temperatures and sufficient precipitation). Approximate seed source of <i>B. pubescens</i> must be present for birch to colonize. <i>Probability:</i> low.
S1 @ S5	Does not exist as SES are always an intermediate stage in the degradation process.
T8 S5 @ ?	<i>Cause:</i> establishment of plant species capable of surviving in eroded areas, and thus starting primary succession. <i>Probability:</i> low.
? @ S5	Not considered to exist.

wetlands (Kristinsson, 1989). The original birch woodlands may thus have maintained drier soils than would have occurred otherwise. Deforestation would have caused the water table to rise, thus creating hydrologic regimes favoring graminoids over trees or shrubs (Weltzin et al., 2000). A shift from dryland vegetation types to communities characterizing wetlands would have followed elevation of the water table (Jauhiainen et al., 2002). The opportunity for birch to become dominant in wetlands may be confined to periods when temperatures are warmer and precipitation lower than average (Einarsson, 1963; Caseldine et al., 2003). Conversions of wetlands into drylands have been noticed in recent times as land has been afforested (Bragason, 1998), hence the reverse should be true if the forest is removed, given that the soil hydrology has not been altered permanently. A consequence of such change would be soil alterations, where histic epipedons could form over time in the waterlogged areas.

**T2: Wetlands (S2) « heathland or grassland communities (S3).** T2 is possible, albeit unlikely. S2 → S3 would require prolonged drought conditions or management intervention (e.g. draining). The S3 → S2 transition is also considered unlikely and may require a number of years of high rainfall and alteration in surface or subsurface water flow and soil hydrology.

**T3: Birch woodlands (S1) « heathland or grassland communities (S3).** This transition originates in birch woodlands on well-drained soils, which excludes T1 from occurring. It is driven by deforestation and/or grazing (see S1 above). Deforestation would constitute a ‘pulse disturbance’ and effect a rapid transition, whereas continuous livestock grazing would be ‘press disturbance’ and effect a slower,

more gradual transition (Bender et al., 1984); and the two operating simultaneously may produce a synergistic, novel transition dynamic. Reversal of the trend would require relaxation of grazing, a birch seed source, safe sites for birch seed germination and favorable climatic conditions (Kullman, 1990; Kullman, 2002).

**T4: Heathland or grassland communities (S3) « SES surface types (S4).**

T4 originates in heathland or grassland community types with the widespread development SES. It is triggered by factors that facilitate SES expansion and coalescence, i.e., continuous livestock grazing and frequent freeze-thaw cycles.

The reverse transition requires minimal or no grazing and favorable climatic conditions (snow cover in winters to reduce the frequency of freeze-thaw events, warm summers with sufficient precipitation). Birch seedlings can potentially establish in disturbed areas and birch plants may thus act as pioneers in the new plant community (Persson, 1964; Magnusson and Magnusson, 1990; Aradottir, 1991). It is thus suggested that if grazing pressure is relaxed, the system may slowly move toward the heathland/grassland State 3, depending on availability of seed sources.

This transition is characterized by a shift from the biotic process domain and to the abiotic process domain. It is therefore a functional threshold (Briske et al.) and is not easily reversed. The S4 → S3 transition is therefore improbable.

**T5: SES surface types (S4) « birch communities (S1).** The S4 → S1 transition requires considerably reduced grazing and climatic conditions favorable for birch seedling establishment. Birch seedlings can colonize disturbed areas (Persson, 1964; Aradottir, 1991), if seed sources are available. The S1 → S4 transition may have

been widespread in the past, as farmers actively cleared woodlands to obtain grazing land, wood and fuel and livestock were grazed year-round. Under the current land management, woodlands are no longer cut; and are often fenced to exclude grazing. Both transitions are thus considered improbable as indicated by the arrow sizes in Figure 3.4. Today, a more gradual change, where the system first passes through S3 is more probable. This transition also includes a functional threshold, so while the S1 → S4 transition may take place as described above, the reversal is very unlikely unless grazing is removed, ample birch seed sources are in the vicinity and climate is favorable (mild winters with good snow cover, warm, moist summers). Birch leaf litterfall in autumn may increase the probability of the reverse transition by reducing cryoturbation and stabilizing the soil surface (Chambers et al., 1990; Groeneveld and Rochefort, 2005). These transitions may be more likely on sites with sandy or gravelly soils that are less prone to cryoturbic disturbances.

**T6: SES dominated surface types (S4) ® denuded (S5).** This transition may occur rapidly once S4 has been reached and disturbances continue. This transition is uni-directional (Figure 3.4) at decadal time-scales. The predominance of geophysical drivers originating with S4 makes it highly unstable and vulnerable to complete degradation, especially when S4 communities are contiguous with S5 landscapes (Figure 3.8). The fact that geophysical drivers are now dictating rates and patterns of erosion means the remaining vegetation is of little consequence in affecting this transition. Prevention of this transition would require cessation of livestock grazing and cultural inputs to stabilize SES and erosion fronts (rofabards).

**T7 and T8: denuded (S5) ® ?**. In effect, primary succession is required to restore vegetation on the denuded areas. What direction it takes depends on the nature and proximity of seed sources, the availability of safe sites for seed germination/seedling establishment, and climate. Based on the work of Steindorsson (1964; 1980), the species composition of the Hafnarskogur area and species observed to establish in SES there, the following vascular plants would be candidates for colonizing S5 in Hafnarskogur:

*Armeria maritima*, *Cardaminopsis petraea*, *Equisetum arvense*, *Eriophorum angustifolium*, *Festuca rubra*, *F. vivipara*, *Luzula spicata*, *Oxyria digyna*, *Plantago maritima* and *Silene uniflora*. In the absence of external inputs (e.g., seeds, nutrients), plant establishment will be extremely slow (Gretarsdottir et al., 2004). The outcome of the succession process will depend on site-specific seed availability and environmental conditions (Magnusson, 1994). As noted earlier, birch can pioneer denuded areas if seed sources are in the vicinity and environmental conditions are suitable (Aradottir, 1991), but survival is highly correlated with seedling size (Magnusson and Magnusson, 1990) as small seedlings are easily disturbed by cryoturbation. There is thus the potential for birch woodlands to develop on S5 sites [T7 on Figure 3.4 (Aradottir, 1991; Magnusson, 1994)]. The question mark at the end of T8 in Figure 3.4 indicates the unpredictability of the primary succession processes and our limited understanding of how it proceeds.

S5 represents a harsh and stressful environment for plant re-establishment. Survival is inheritably low due to unstable surfaces (Decker and Ronningen, 1957; Aradottir, 1991; Magnusson, 1994) and plant abrasion by eroding particles (Magnusson, 1994). Despite these adverse conditions, the abiotic processes responsible for the land

degradation and erosion up to this point in the degradation sequence, may now play an important role in providing safe sites for plant reestablishment. Over time, frost will heave small pebbles and stones to the surface (Arnalds and Kimble, 2001). This creates a complex mosaic of micro-sites providing favorable moisture and temperature conditions (Pérez, 1987), and soil stability. Such gravelly and rocky sites have been shown to increase the probability of seedling establishment (Pérez, 1987; Aradottir, 1991; Arnalds and Kimble, 2001; Elmarsdottir et al., 2003), e.g. by suppressing needle-ice formation (Jumpponen et al., 1999). It is thus possible that heaved gravel and stones may eventually create safe sites that provide opportunities for seedling establishment, and thus drive the T7 or T8 transitions.

### **Discussion**

Predicting and managing land cover change is critical to conservation and sustainability. However, we are frequently hampered by poor understanding of the underlying autogenic and allogenic processes governing the ecosystem responses to disturbance and land use, and may not know the management actions most appropriate for a given situation. This may be due to lack of overview, if we do know where the pieces fit in the big puzzle we are trying to assemble, or because we lack the insight to identify key gaps in understanding. An important first step in charting the way forward is to systematically organize existing and often fragmented information. The strength of S&T models lies in their formal articulation of the circumstances under which vegetation and land cover changes can be expected. Their structural presentation and emphasis on process driven changes between alternate states, does at the same time point out



knowledge gaps in how and under what conditions change occurs. They are therefore valuable in focusing research efforts, and for opportunistic land management.

The S&T model proposed in this chapter describes land changes in Hafnarskogur in west Iceland, where birch woodlands degrade into dysfunctional landscapes characterized by eroded surfaces. It consists of states, which are defined based on ecosystem processes and alterations driving their formation and transition to other states. The states themselves are relatively easy to identify as they have distinctive surface features.

It is suggested that the arrival of man in Iceland triggered the degradation sequence through deforestation and the introduction of domestic herbivores, and the process was then exacerbated by interactions between land management, climate and soil properties, driving the system into what could also be described as a degradation spiral (see Chapter 2 in Whisenant, 1999). Initially the Hafnarskogur birch woodland (S1) was pushed in one of two directions based on the soil hydrology. In situations where ground water levels were suppressed by the woody cover, loss of birch woodlands caused a rise in the water table and led to the development of wetland vegetation (S2). On sites that naturally had low ground water table (not tied to the present woodlands), loss of birch plants led to the development of open single-strata plant communities (S3) dominated by heathland or grassland vegetation (Figure 3.4). Continuous grazing prevented the transition, back to birch woodlands and led to the appearance and expansion of SES. The increasing SES cover then triggered a positive feedback where diminishing grazing area is exposed to constantly increasing grazing pressure. Defoliation caused a reduction

in aboveground biomass, root density and a change in plant functional group composition; and the trampling disturbed the sward surface, thus exposing the mineral soil to wind and water erosion. These grazing-induced changes concomitantly diminished the vegetative thermal barrier and promoted cryoturbation processes that promoted hummock formation and further de-stabilized soils. This chronic livestock disturbance regime promoted formation of new SES while at the same time preventing the recovery of existing SES. Under these conditions, ecosystems transitioned into S4 (Figure 3.4), across a functional threshold and into an abiotic process domain where erosion forces dominate. At this point, reversal to S3 is almost impossible; and continued transition to S5 is almost inevitable, unless aggressive management inputs are implemented to simultaneously reduce grazing impacts and stabilize erosion. Once in S5, the restoration cost and the probability of failure are very high and natural succession may take decades or centuries. By establishing S&T models for landscapes at risk for such degradation, the land manager can identify sites at risk for undesirable transitions; and take steps to prevent them from occurring.

Maintaining vegetative cover is important in all land management. Vegetative cover reduces erosion risk directly by providing a sheltering barrier between the mineral soil surface and the elements, and dampens damage caused by trampling. Vegetation also reduces erosion risk indirectly by reducing surface flow (Thurow et al., 1986; Orradottir, 2002; Orradottir et al., 2008), and by increasing soil organic matter content which improves soil structure and stability through better aggregate cohesion (Brady and Weil, 1998; Whisenant, 1999). These ecosystem services provided by the vegetation are

important in any ecosystem, but especially in Sub-Arctic ecosystems such as Iceland where the soil is especially fragile and erodible (Arnalds, 1999, 2004). Land management in such systems should focus on two primary things: 1) maintaining good vegetative cover with as much biomass present at the end of the growing season as is possible, and 2) reducing grazing pressures when SES start to appear, and aggressively stabilizing small SES to prevent them from becoming large SES. Management efforts should therefore be concentrated on areas that are approaching the S3→S4 transition, thereby allowing for a dynamic land management where land is kept in a healthy and sustainable state.

While S&T models have been widely developed for temperate, sub-tropical and tropical rangelands, there is an urgent need to develop S&T models for high latitude systems. The global importance of high latitude ecosystems has become clear over the past decades as it has become apparent that northern sub-arctic and arctic ecosystems may significantly contribute to and be affected by the current global warming trends (Miller, 1981; Chapin et al., 2005; Houghton, 2005). This will affect processes on a global scale through positive feedback and may exacerbate the warming trend. The effect of climate change on high latitude ecosystems, and thus the global effect, are hard to predict however, as these systems may not only work as carbon sources on a large scale, but also as sinks due to changes in vegetation composition (Marion et al., 1997). Nevertheless critical ecosystem processes such as decomposition, soil nutrient mineralization, photosynthesis, and thus vegetation growth, are affected by temperature (Nadelhoffer et al., 1991; Hobbie, 1996; Koch and Mooney, 1996; Rustad et al., 2001).

Temperature increases are likely to be greater at higher latitudes following reduced sea ice and snow (Houghton, 2005), and thus the effect on arctic and sub-arctic ecosystems can be expected to be proportionally greater than at lower latitudes, with unknown effects on the vegetation (Wahren et al., 2005) and unknown global implications. These are pressing questions, which we may have a short time to answer. S&T models may be very helpful under such scenarios.

In arctic systems increased shrub abundance and rise in range-margins of shrubs, triggered by warmer climate, have been observed since the middle of the last century (Sturm et al., 2001; Kullman, 2002). In the Swedish Scandes (63°26'N; 13°06'E) range-margins of *Betula pubescens* have advanced upwards about 300 m under low grazing pressure (Kullman, 2002). This large climb in range-margins was mostly attributed to high growth and colonization in the exceptionally warm 1990's. In Iceland, increased abundance of deciduous and evergreen dwarf shrubs under moderate experimental warming has been observed (Jonsdottir et al., 2005), and reduced erosion and greater vascular plant abundance in rangelands, between 1997/98 and 2005, has been attributed to reduced grazing pressures and climate warming (Magnusson et al., 2006). This, and the positive correlation of birch growth and summer temperatures (Levanic and Eggertsson, 2008; Eggertsson and Gudmunsson 2002) indicates that the historic trend for loss of birch in Iceland may be more readily reversed in current climate situation with the reduction or exclusion of grazing.

## **Next steps**

The Hafnarskogur S&T model identifies many gaps in our understanding of the degradation process, and is therefore based on several assumptions. It assumes that the degradation starts with land cover changes centered around the loss of birch woodlands. We do not know if that transition is as critical as the model implies, or if this assumption is predicted on the perception that such dramatic change in functional group composition must be accompanied by changes in processes. That remains an open question. There are little data comparing ecosystem processes (e.g. primary production, water and nutrient cycles, land surface-atmosphere interactions, etc.) in Sub-Arctic-Andic woodlands versus grasslands derived from woodlands. Current research does indicate that critical system processes, such as hydrological processes differ between these community types, especially during the winter (Orradottir, 2008). It is likely that tree canopy does add an extra thermal layer when compared to open lands (Sartz, 1957) and the loss of canopy results in loss of the extra thermal layer. It is therefore reasonable to expect some differences in temperature-dependent processes if such plant community shift occurs. However, even though we know that rates of decomposition, mineralization and photosynthesis are likely affected by vegetation mediation of temperature, we do not know how changes in microclimate associated with changes in vegetation structure might impact system resilience or resistance via influences on the frequency and magnitude of freeze-thaw cycles that influence soil stability and seedling establishment.

Our understanding of the formation, maintenance and expansion of the SES is also very limited. Insight into these processes is the key to preventing their expansion and coalescence, which is critical to avoid crossing the functional threshold between S3 and S4. Is hoof action and grazing important factors in their formation or is it a natural phenomenon driven by oscillating climate and simply exacerbated by grazing? Or are weather extremes the main driving force in SES formation and thus the critical initial step in the land degradation sequence? What level of grazing is required to initiate or maintain SES; and is there a time or size/density threshold at which abiotic erosion processes begin to override biotic soil stabilization processes? Is there a critical annual minimal temperature or critical number of soil freeze-thaw cycles required to initiate SES formation? Or, does the lowered soil temperature in bare soils, compared to vegetated areas, decrease seedling root growth to such an extent that survival is severely affected (Weih and Karlsson, 2002) and seedling mortality thus higher in the bare areas? Is SES maintenance a function of available microsites for seed germination followed by water availability (Bell and Bliss, 1980; Jones and del Moral, 2005)? Is the shift from woodlands to open areas in the presence of grazing, simply so drastic that we cannot expect reversion of states due to lack of facilitation (e.g. Miller and Halpern, 1998; Rousset and Lepart, 2000), and increased extremes in microclimate (e.g. Carlson and Groot, 1997). If that is the case - is this perhaps what initiates the degradation sequence?

The questions are many and reflect our current level of understanding. The proposed S&T helps articulate potential change pathways, proposes critical processes that may be driving change, and pinpoints processes unique to each stage of the

degradation sequence. At the same time, the S&T model for Hafnarskogur identifies key intervention points where management could be adjusted to prevent costly, and potentially irreversible degradation from occurring.

The S&T models summarize and integrate the best available knowledge into a framework to guide management and to link pattern-process, structure-function, and cause-effect. As such S&T models are an invaluable tool for guiding management (Walker, 1993) and for improving communication between scientists and land managers (Grice and MacLeod, 1994). S&T models are equally important as a research tool and research should be considered as an integrated factor in the S&T model development and application - and as the first step in research planning. The reward will be a framework that integrates the practical experience of land managers and scientific understanding of ecosystem properties and processes. This framework can be readily updated to accommodate insights generated as results from new research comes on line and are tested by land managers confronting new challenges and specific conditions.

CHAPTER IV  
THE EFFECT OF LATE-SUMMER BROWSING ON BIRCH  
(*Betula pubescens* EHRH.) SEEDLING SURVIVAL

**Introduction**

Deforestation, the temporary or permanent clearance of forest (Grainger, 1993), is well-documented (FAO, 2001; Achard et al., 2002; Arnalds and Stahr, 2004; FAO, 2007) and is fueled by the growing human population's need for wood, timber and pulp, and agricultural land. Deforestation is commonly followed by soil erosion, landslides, soil nutrient loss (R. C. Derose, 1993; Dai et al., 2002; Zheng et al., 2005) and flooding (Khalequzzaman, 1994). The effects of, deforestation have received the greatest attention in tropical and temperate regions, where it has widespread social and economic impacts (Barbier and Burgess, 2001; Geist and Lambin, 2001). However, it has also occurred in sub-arctic areas (Arnalds and Stahr, 2004). Iceland, a 103,000 km<sup>2</sup> island in the North Atlantic Ocean, is one such example. Birch (*Betula pubescens* Ehrh.) woodlands that dominated most of the Icelandic lowlands began disappearing soon after the settlement in the 9<sup>th</sup> century AD (Hallsdottir, 1992, 1995; Aradottir and Eysteinnsson, 2005). Today it is estimated that over 95 % of these woodlands have been lost (Sigurdsson, 1977; Gudjonsson and Gislason, 1998; Olafsdottir et al., 2001). Land degradation and soil erosion, which followed the woodland disappearance have been and continue to be a serious problem (Arnalds, 1999; Arnalds et al., 2001).



Why were Icelandic woodlands so dramatically affected, when comparable ecosystem have persisted in the other Nordic countries (Wielgolaski, 2001)? In contrast to other Nordic regions, Icelandic birch evolved without large grazing animals, and with few invertebrate herbivores (Arnthorsdottir and Olafsdottir, 2001; Neuvonen et al., 2001). Icelandic birch is more palatable than birch populations that evolved with large grazers and browsers (Bryant et al., 1989) and may have been more vulnerable to herbivory than conspecific woodlands e.g. in Scandinavia, especially during years of exceptional cold weather (Haukioja and Neuvonen, 1985; Raitio et al., 1994; Lappalainen et al., 2000; Neuvonen et al., 2001). The introduction of livestock may also have exacerbated poor regeneration (e.g. low seed production, germination and/or establishment) of the birch woodlands. Research on the critical establishment phase of the birch life cycle has focused primarily on germination safe sites (Aradottir, 1991; Magnusson, 1994), seedling growth and survival (Kullman, 1986; Weih, 2000) and the impacts of leaf and bud herbivory (Arnthorsdottir and Olafsdottir, 2001). Less is known of birch seedling response to browsing, which removes both meristems and photosynthesis tissue; and how browsing might affect seedling recruitment in plant community types representing a degradation chronosequence.

Winter sheep grazing was a common practice until early or mid 20<sup>th</sup> century (Thorsteinsson, 1986) and has been partially blamed for the woodland decline in Iceland. The fall clipping treatments applied in this study were selected based on this farming practice. The goal of this study was to assess birch seedling tolerance to browsing and thus advance our understanding of how livestock affect birch recruitment. In addressing

this goal, I quantified *B. pubescens* seedling growth and survival, under three defoliation regimes in five plant community types, ranging from dense woodlands to open grasslands. I predicted that the highest mortality and lowest growth would occur in the most intense defoliation treatment, and that seedling sensitivity to simulated browsing would vary from one community to another owing to differences in microclimate and competition. For example, the chances of *B. pubescens* seedling survival under browsing might be greater in woodland communities than in more open communities owing to a lower risk of desiccation and frost damage associated with snow accumulation and the amelioration of cold, dry winter winds and nighttime radiative heat loss. Alternatively, shading by adult *B. pubescens* plants during the growing season may limit seedling growth and make seedlings more sensitive to browsing. In more open communities, competition from herbaceous vegetation during the growing season may operate in conjunction with a harsher winter microenvironment to limiting recruitment and increase seedling vulnerability to defoliation; and the effects of herbaceous competition on seedling establishment and response to browsing may depend on species composition.

### **Material and Methods**

The study was conducted in Hafnarskogur, west Iceland (64°30'N; 21°55'W) which belongs to the Hofn farm that was settled in the 9<sup>th</sup> and 10<sup>th</sup> centuries (Thorgilsson, 1968). The area was a common grazing land used by sheep and horses until it was fenced in the 1980's, (Aspelund and Olafsson, pers. comm.) and has since been grazed by 30 - 40 horses year round. Common plant community types, on the site

include dense woodlands, sparse woodlands (savanna), woodland heathlands (w heathlands), savanna heathlands (s heathlands), and open grasslands. Tree cover consists solely of *Betula pubescens* (Ehrh.), with *Deschampsia flexuosa* and *Agrostis capillaris* dominating the ground cover, while *Agrostis capillaris* and *Deschampsia caespitosa* dominate in the open grasslands. See Appendix A for a more detailed description of each plant community type.

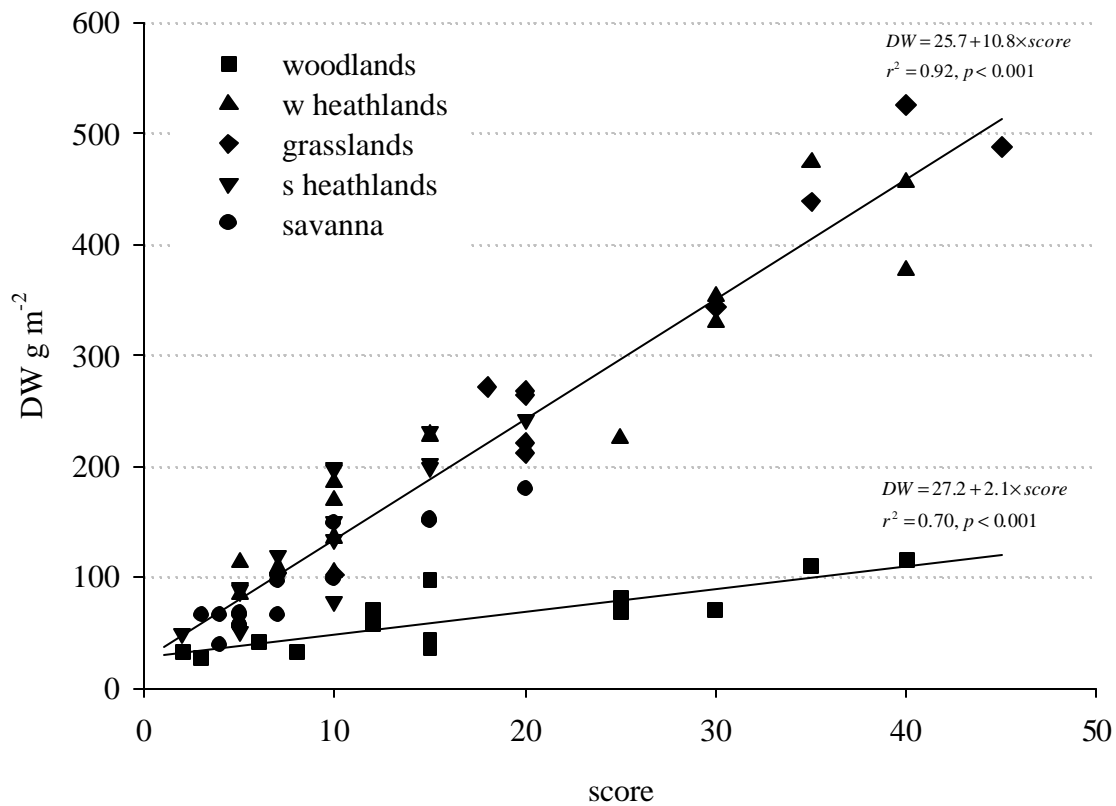
The soils in the area are Andisols and have tentatively been classified, based on Soil Taxonomy (Soil Survey Staff, 1999), as Typic Fulvicryands and Histic Cryaquand,s (Orradottir et al., 2008) (see Chapter II). Mean January, July and annual temperatures are -0.5, 10.6 and 4.5 °C, and mean annual precipitation totals 1460 mm (data from the Icelandic Meteorological Office, IMO; temperature from Reykjavik [1961 – 1990] 40 km N of site; precipitation from Andakilsarvirjkjun [1961 – 2000] 12 km ENE of site). Snow cover was 100 % in 25, 19 and 7 days in winter 2000 – 2001, 2001 – 2002 and 2002 – 2003, respectively. The 2002 - 2003 winter was the third warmest since measurements began in 1920 (data from the Icelandic Meteorological Office, IMO).

Three plots were established within each of the five plant community types on flat surfaces between hummocks (see Appendix A for plot for locations). Thirty one-year-old container grown *B. pubescens* seedlings were transplanted in each of these plots on 4 July 2000 (hence a total of 90 seedlings per community, or a total of 450 seedlings). Browsing treatments (none = controls; and removal of 25 % or 75 % of the distal primary and secondary shoots) were randomly applied to the seedlings (n = 150 seedlings/treatment) on 17 – 18 August 2000; and were repeated on 6 August 2001 and

18-19 August 2002. Ground layer biomass within  $10 \times 10$  cm sub-plot surrounding each seedling was non destructively estimated by double-sampling on 19 - 20 June 2003 (Campbell and Arnold, 1973). Vegetation from the calibration plots was dried at  $105^{\circ}$  C and weighed. Correlations between visual estimates and true biomass were determined and used to correct ocular estimates (Figure 4.1)

Seedling responses were assessed on 17 - 18 August 2000, 21 - 26 June 2001, 6 August 2001, 20 June 2002, 18 - 19 August 2002 and 19 - 20 June 2003. The following data were collected: height (mineral soil surface to most distal point), crown depth (base of lowest side branch to tip of most distal branch) number of leaves, total number of shoots, number of  $1^{\circ}$  and, length of  $1^{\circ}$  shoots (top shoot), and length of frost damage (dead twig ends). The number of dead plants and the number of plants exhibiting signs of frost damage were also recorded. Insect herbivory was scored on a 0 - 4 scale (0 = no visible leaf damage, 1 = < 25 % of leaves damaged, 2 = 25 – 50 % of leaves damaged, 3 = 50 – 80 % of leaves damaged, and 4 = > 80 % of the leaves damaged). Vigor was estimated on a 1 – 4 scale (1 = crown cover < 10 % of stem length and leaves very small, or leaves small and < 5; 2 = crown cover < 11 - 30 % of stem length. leaves < 10; 3 = crown 31-50% of stem length,  $1^{\circ}$  branches present; and 4 = crown length >50% of stem length,  $1^{\circ}$  branches present and plant has uniform leafing). All scalar estimates were made by the same person.

Seedling descriptors represent three types of data: continuous data (total height, crown length, number of leaves, number of active buds, number of  $1^{\circ}$  branches, length



**FIGURE 4.1.** Simple regressions relating ocular estimates of aboveground biomass (Dry Weight) to harvested biomass. The birch woodland community had a lower and smaller range of biomass values than the other four communities and is shown separately.

of 1° branches); binary categorical data (number of dead plants, number of plants with frost damage); and ordinal categorical data (level of browsing, insect damage, vigor).

### **Statistics**

The influence of plant communities and treatments on frost damage was tested using Chi-Square when sample sizes were equal and Crosstab contingency tables when they were not (Ott and Longnecker, 2001; Dytham, 2003). When Chi-Square assumptions were violated (i.e., if > 20 % of the cells had values < 5), a Fisher's Exact test was performed (Ott, 1993). The overall effect of communities on seedling vigor and response to level of insect damage was evaluated with Kruskal-Wallis tests by comparing controls across the five communities for each year (Sokal and Rohlf, 1981; Dytham, 2003). When main effects were significant, post-hoc Mann-Whitney U tests were conducted (Dytham, 2003). Effects of communities and treatments on mortality were tested with two-way Analysis of Variance (ANOVA) on the proportional mortality of seedlings per plot.

Spearman's  $\rho$  correlation was calculated for all the measured variables in order to check for variable relationship. This was deemed necessary as many of the variables measure similar properties, e.g. crown length and total length. Correlation was used to reduce redundant variable from the data analysis. The correlation revealed strong relationship within each of these three groups ( $p < 0.001$ ). Due to this, a new variable was calculated:

$$\text{total growth} = (\text{height} + \text{length of side branches})$$

and used as test parameter in subsequent growth analysis. Dead seedlings were not included in the growth calculations. Overall differences in total growth between the three browsing treatment groups were tested separately for each year using GLM ANOVA. The community effect on total seedling growth was tested by comparing controls in the five community types with GLM ANOVA for each year separately. Community differences in ground layer biomass in the neighborhood of the seedlings were also ascertained with GLM ANOVA and linear regression used to determine the direction and strength of relationship with seedling growth and mortality for the June 2003 data. When differences were significant the Bonferroni post-hoc test was used for pairwise comparisons. Box-Plots were used for initial screening for extreme outliers. Normality and multivariate normality of the residuals was tested with normal Q-Q plots, Kolmogorov-Smirnov or Shapiro-Wilk tests, and equality of variances, both univariate and multivariate, with Levine's test (Neter et al., 1996; Ott and Longnecker, 2001). Kruskal-Wallis tests followed by pairwise Mann-Whitney U post-hoc tests were applied to main effects if parametric test assumptions could not be met with logarithmic, Square-Root or Box-Cox data transformations (Sokal and Rohlf, 1981).

Significance levels were set at  $\alpha < 0.05$ ; and  $\alpha$  error levels for post-hoc tests were adjusted for multiple comparisons ( $\alpha_{\text{adj}} = \alpha / [g (g - 1) / 2]$ , where  $g$  = number of groups). Data were analyzed with SPSS v.13.0 (SPSS Inc., 2004).

## Results

### Seedling growth and neighborhood biomass

Figure 4.2 shows the total cumulative growth over time. Cumulative seedling growth was comparable between communities in June 2001, the first spring after planting (Kruskal-Wallis,  $n = 145$ ,  $df = 4$ ,  $C^2 = 9.10$ ,  $p = 0.059$ ), but by August 2001 growth of seedlings in the woodland, savanna and grassland communities were significantly greater than that of seedlings in the two heath communities (GLM ANOVA,  $n = 134$ ,  $df = 4$ ,  $F = 12.5$ ,  $p < 0.001$ ). In August 2002 seedlings in the savanna had significantly more cumulative growth than did seedlings in the other four communities (GLM ANOVA,  $n = 135$ ,  $df = 4$ ,  $F = 6.8$ ,  $p < 0.001$ ). By June 2003 the growth of seedlings was greatest in savanna and grassland communities and lowest in the woodland and heath communities (GLM ANOVA,  $n = 124$ ,  $df = 4$ ,  $F = 10.7$ ,  $p < 0.001$ ). Figure 4.3 shows the mean cumulative total growth of seedlings in the browsing treatments. No significant differences were found in June 2001, but by August 2001 seedlings in the 25 % clipping treatment exhibited lower cumulative growth than either control or the 75 % removal treatment (GLM ANOVA,  $n = 422$ ,  $df = 2$ ,  $F = 6.5$ ,  $p < 0.01$ ). These differences persisted through June 2002 (Kruskal-Wallis;  $n = 387$ ,  $df = 2$ ,  $C^2 = 14.319$ ,  $p < 0.01$ ). By August 2002 seedling growth was comparable between browsing treatments (Kruskal-Wallis;  $n = 362$ ,  $df = 2$ ,  $C^2 = 5.638$ ,  $p > 0.05$ ), but in June 2003 control seedlings had considerably higher growth rates than seedlings in the two shoot removal treatments (Kruskal-Wallis;  $n = 306$ ,  $df = 2$ ,  $C^2 = 13.782$ ,  $p < 0.01$ ).



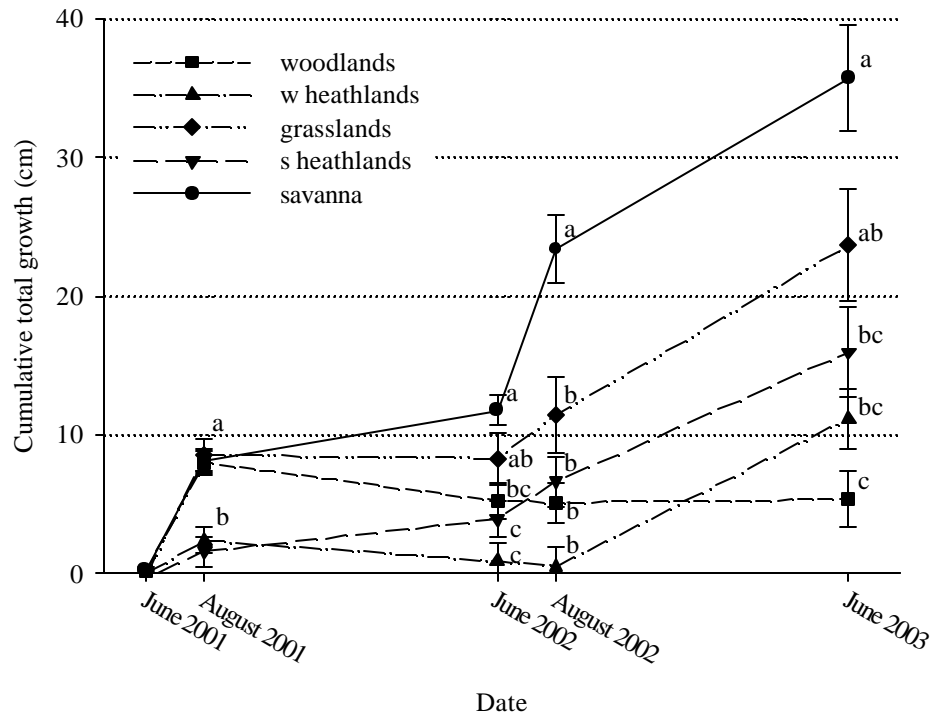
Aboveground herbaceous biomass in the neighborhood of birch seedlings varied significantly between plant communities (GLM ANOVA,  $n = 444$ ,  $df = 4$ ,  $F = 193.5$ ,  $p < 0.001$  Figure 4.4) and was lowest in birch woodland communities and highest in grassland and w heathlands communities. However, regression between estimated neighborhood biomass and seedling growth explained very little of the variation in seedling growth ( $r = 0.152$ ), although significant.

### **Insect herbivory**

Seedlings in woodland and the grassland communities experienced the highest levels of insect herbivory; and those in w heathland the least (Figure 4.5 A) (Kruskal-Wallis;  $n = 708$ ,  $df = 4$ ,  $C^2 = 148.5$ ,  $p < 0.001$ ). Birch seedlings subjected to 75 % defoliation experienced less insect herbivory than non- and 25 % defoliated plants (Figure 4.5 B; Kruskal-Wallis;  $n = 1972$ ,  $df = 2$ ,  $C^2 = 53.6$ ,  $p < 0.001$ ).

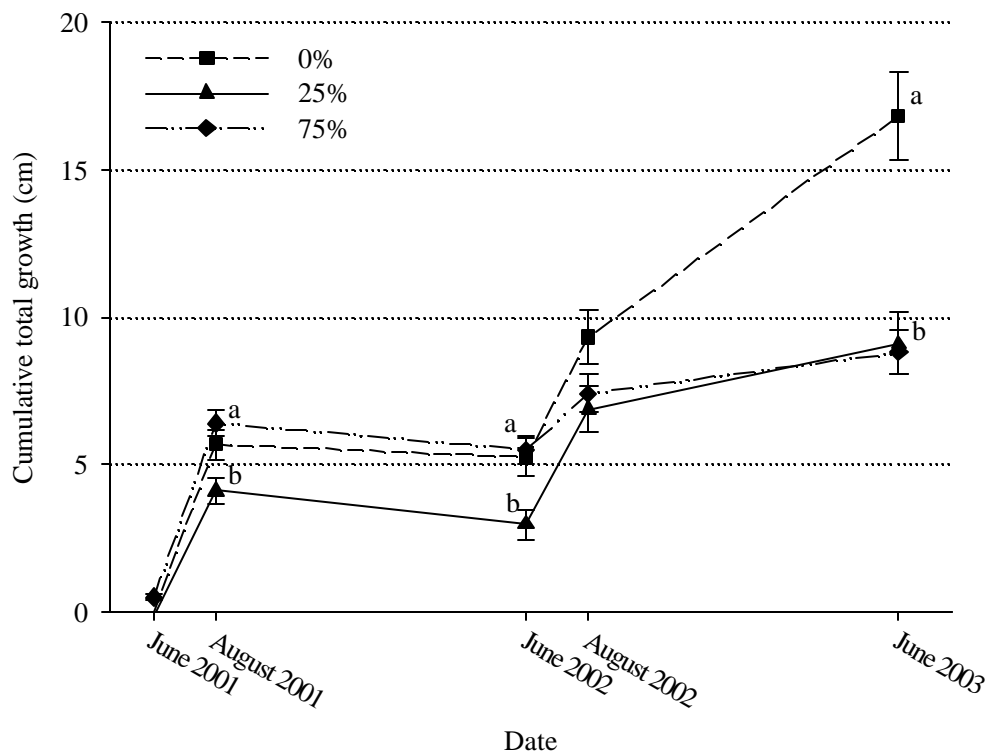
### **Vigor**

Vigor score of non-defoliated (control) birch seedlings differed between communities (Kruskal-Wallis;  $n = 135$ ,  $df = 4$ ,  $C^2 = 10.7$ ,  $p > 0.05$ ) in June 2003, but no pairwise differences were observed. Birch seedlings subjected to 75 % defoliation had significantly lowest vigor score, the non- and 25 % defoliated plants also differed significantly in vigor, the non-defoliated plants having the highest score (Kruskal-Wallis;  $n = 324$ ,  $df = 2$ ,  $C^2 = 46.3$ ,  $p > 0.001$ ; Figure 4.6).



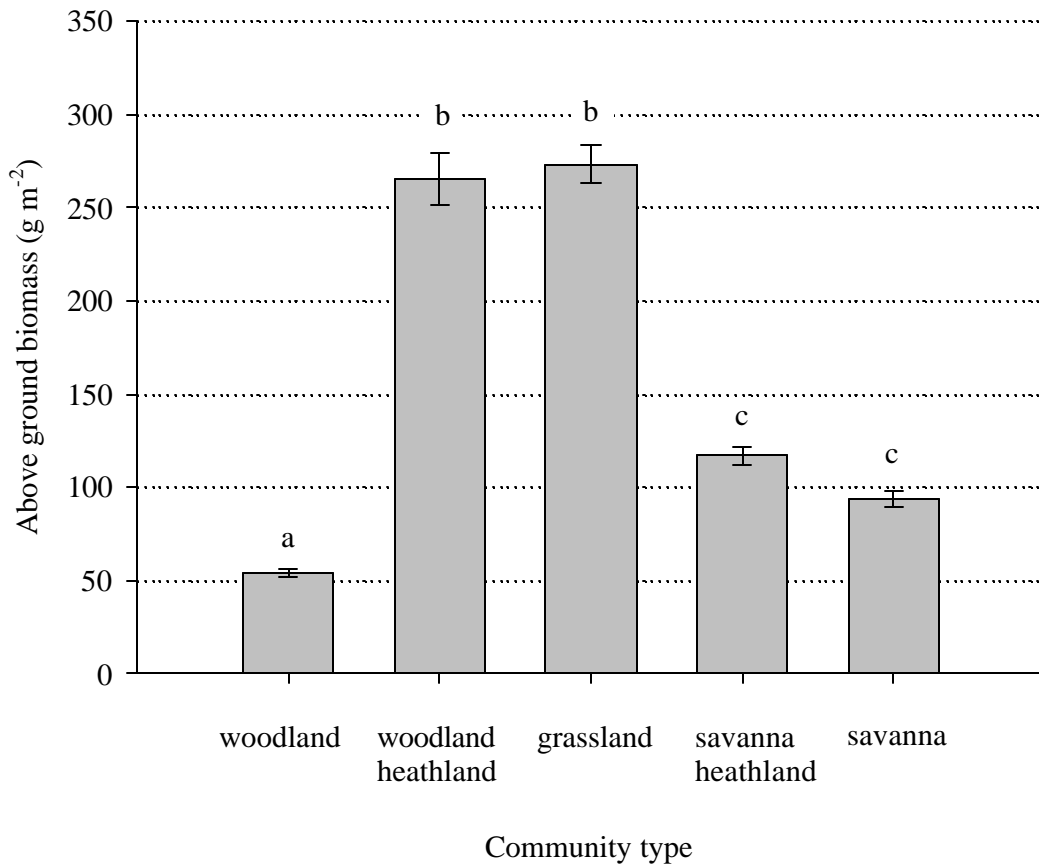
**FIGURE 4.2.** Mean ( $\pm$  SE) cumulative growth of non-browsed (control) birch seedlings in five plant community types. Different letters show statistical differences within each month ( $\alpha = 0.005$ ).

Note: w heathlands = woodland heathlands, s heathland = savanna heathlands

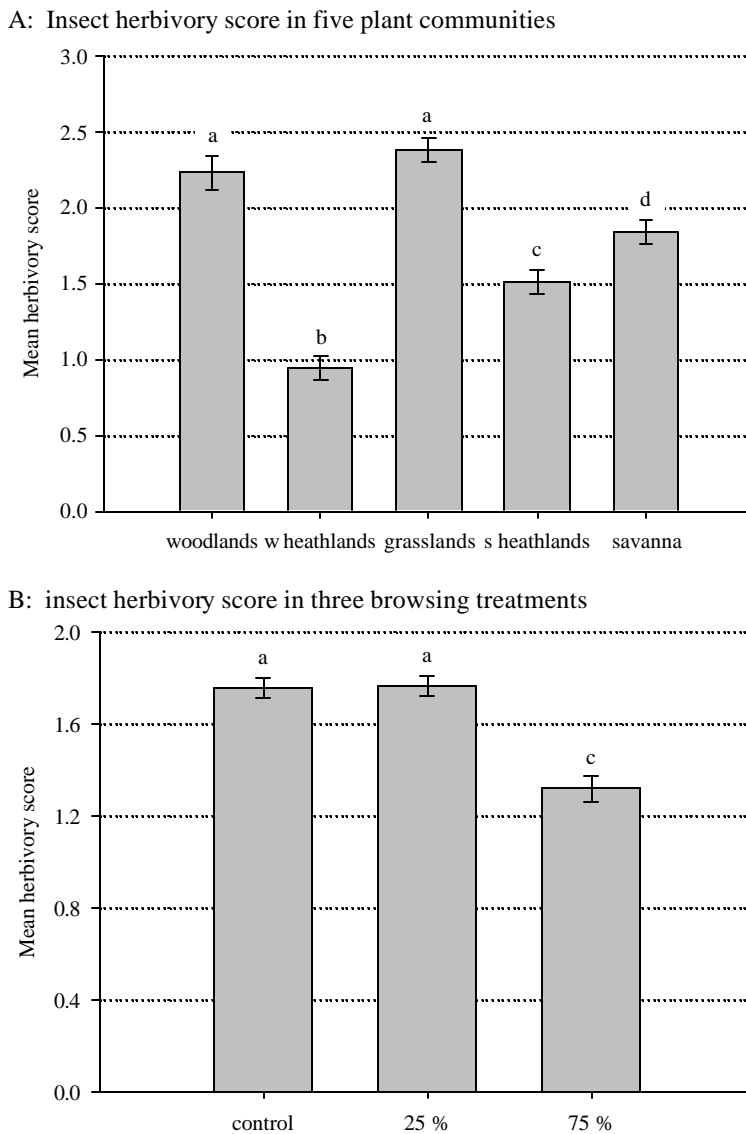


**FIGURE 4.3.** Mean ( $\pm$  SE) cumulative growth of control (non-browsed) and defoliated (25% and 75%) birch seedlings pooled across the five plant community types. Different letters show statistical differences on each date ( $\alpha = 0.017$ ). Missing letters indicate non-significant differences between treatments.

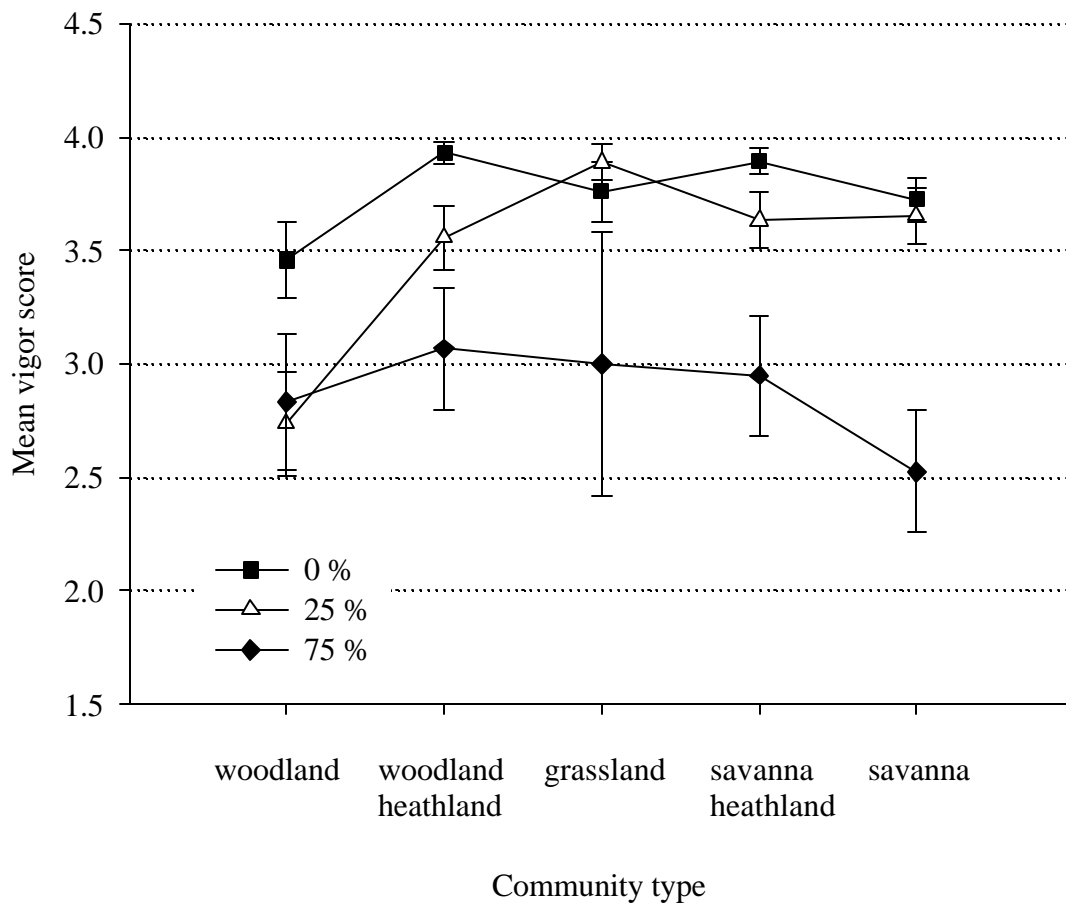
Note: w heathlands = woodland heathlands, s heathland = savanna heathlands



**FIGURE 4.4.** Mean ( $\pm$  SE) aboveground biomass in five plant community types in June 2003. Different letters show statistical differences between communities.



**FIGURE 4.5.** Mean ( $\pm$  SE) insect herbivory score on (A) non-browsed (control) birch seedlings in five plant communities, and (B) birch seedlings in browsing treatments, pooled across communities, [see text for scoring rating codes range from 0 (no impact) to 4 (>80% of leaves impacted); see text for details]. Scores are pooled across all measurement dates between June 2001 and June 2003. Different letters indicate statistical differences between communities ( $\alpha = 0.005$ ).



**FIGURE 4.6.** Mean ( $\pm$  SE) vigor score on non-defoliated (0%) and defoliated (25%, 75%) birch seedlings in five plant communities in June 2003.

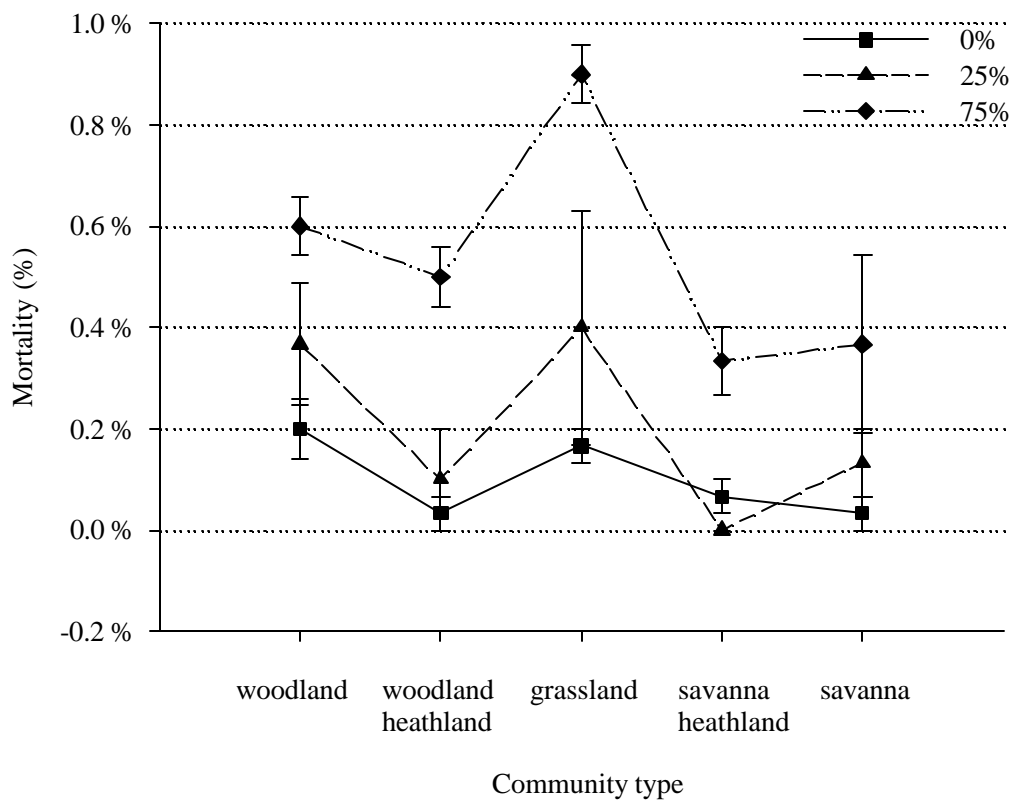
## **Mortality**

Mortality of birch seedlings differed significantly between both communities (GLM ANOVA;  $n = 45$ ,  $df = 4$ ,  $F = 7.7$ ,  $p < 0.001$ ) and treatments (GLM ANOVA;  $n = 45$ ,  $df = 2$ ,  $F = 29.4$ ,  $p < 0.001$ ), but the interaction was not significant ( $p = 0.407$ ) (Figure 4.7). Seedling mortality was higher in the grassland than in the savanna and heath communities, but comparable to birch woodlands that only differed from the savanna heathland community. Mortality of seedlings in the 75 % defoliation treatment (54 %) was higher than that in control (10 %) and 25 % defoliation treatments (20 %). The incidence of mortality in birch seedlings appeared to increase with increasing seedling age, the proportional number of mortality increased markedly over time in the 75 % defoliation treatment, whereas the increase was subtler in the control and 25 % defoliation treatment (Figure 4.8). From June 2002 through June 2003 mortality was significantly higher in the 75 % defoliation treatment compared to both the non- and 25 % defoliation treatment (GLM ANOVA;  $n = 45$ ,  $df = 2$ ,  $p < 0.001$  for June 2002 to June 2003).

The neighborhood biomass (Figure 4.4) did not explain seedling mortality ( $n = 44$ ,  $F = 1.3$ ,  $R^2 = 0.031$ ,  $p > 0.05$ ).

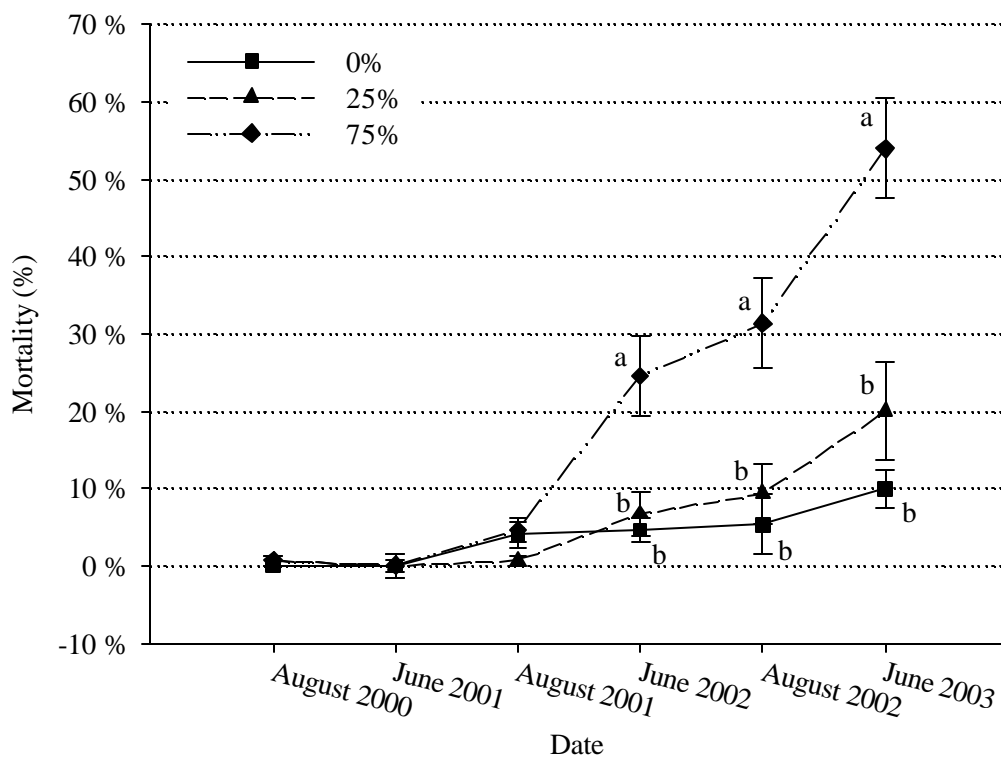
## **Frost damage**

The incidence of frost damage in non-defoliated seedlings appeared to decrease with increasing seedling age, the patterns varying among seedlings in the different plant communities (Figure 4.9). Non-defoliated seedlings in savanna and woodland

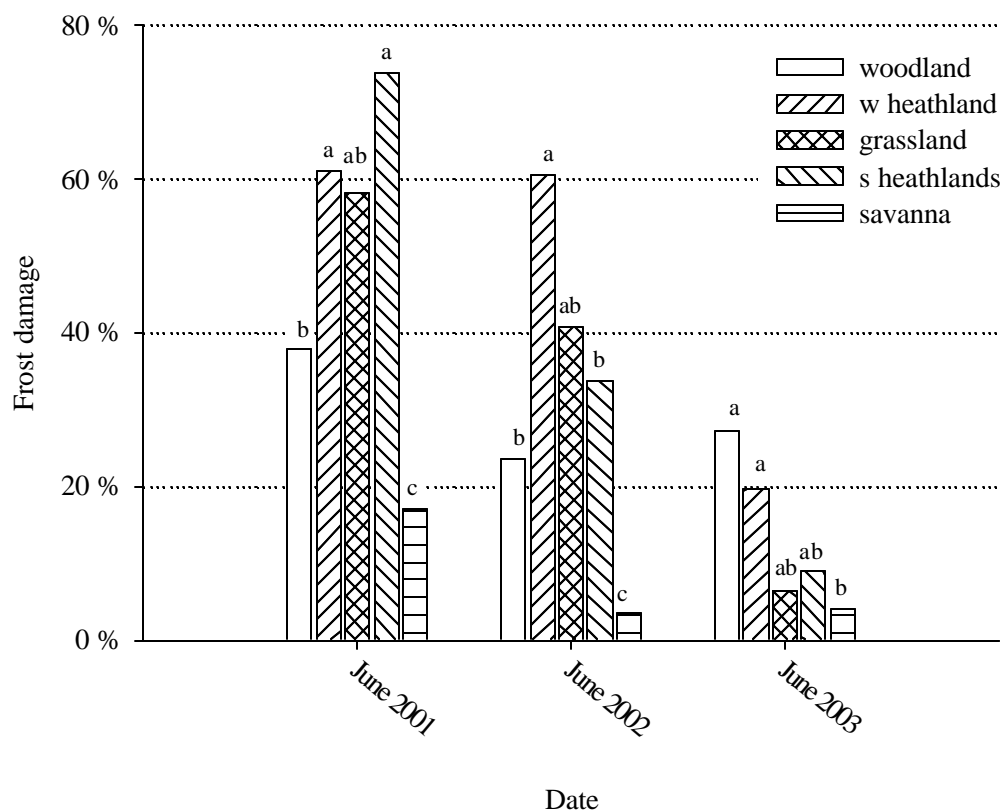


**FIGURE 4.7.** Mortality (%) of non-defoliated (control) birch seedlings and seedlings clipped 25% and 75% in five plant communities in June 2003 ( $n = 90$  initial seedlings/community;  $n = 150$  initial seedlings/treatment). Data are proportional mortality of seedlings per plot.





**FIGURE 4.8.** Mortality ( $\pm$  SE) of control (non-browsed) and defoliated (25% and 75%) birch seedlings pooled across five plant community types from August 2000 to June 2003. Data are proportional mortality of seedlings per treatment per plot.

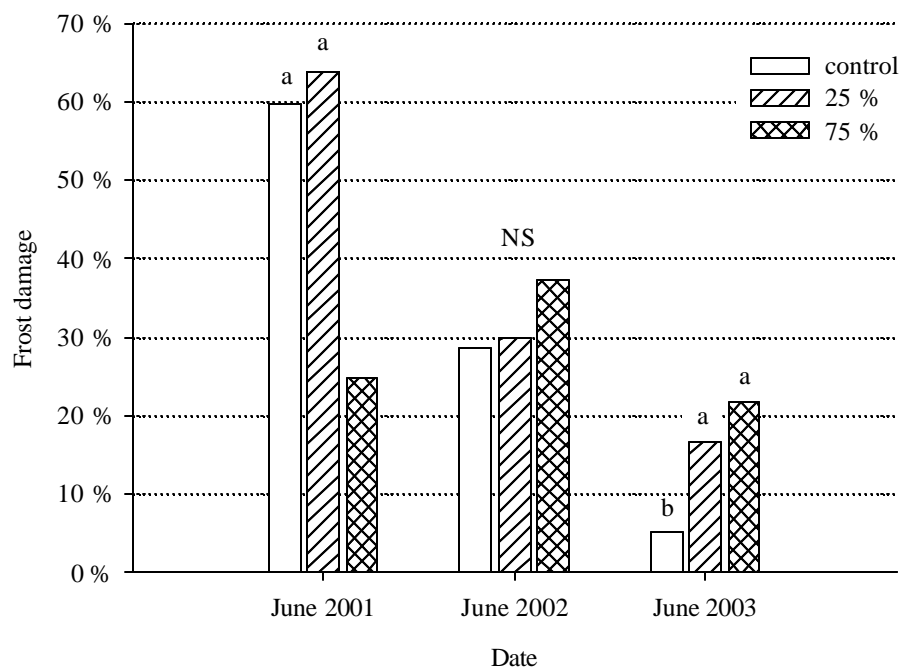


**FIGURE 4.9.** Percentage of non-defoliated birch seedlings in five plant communities exhibiting signs of frost damage on three dates. Different letters show statistical differences between communities on each date [pairwise Chi-Square tests and Fisher's Exact Tests when Chi-Square assumptions were violated ( $\alpha = 0.005$ )].

communities exhibited the least amount of frost damage over the three year observation period, whereas seedlings in the grass and heath communities exhibited higher levels of damage in 2001 and 2002; but had levels approximating that of seedlings in savanna and woodland communities by 2003. When data were pooled across sites, the proportional number of frost-damaged seedlings in the control and 25% defoliation treatment decreased markedly over time; whereas the incidence of frost damage in seedlings subjected to 75% defoliation fluctuated around 30% (Figure 4.10). Although differences in the number of seedlings experiencing frost damage were observed in the various communities and defoliation treatments, the proportion of tissue damage experienced by seedlings was not significant in any case.

### **Discussion**

In this study I sought to compare birch (*Betula pubescens* Ehrh.) seedling growth rates and survival between five plant community types, ranging from dense woodlands to open grasslands, under two simulated autumn grazing regimes using twig-clipping. It was hypothesized that growth rates and survival would be highest in the woodlands due to amelioration of harsh winter conditions; and lowest in the grasslands where competition from herbaceous vegetation and a harsher winter microclimate would combine to constrain seedling establishment. Contrary to expectations, seedlings in the woodland community were less productive (Figure 4.2) than those in the grassland and savanna communities. Mortality was also high in the woodlands, and significantly higher than in the s heathland (Figure 4.7). High seedling mortality rates under natural



**FIGURE 4.10.** Percentage of birch seedling in control (0%) and 25% and 75% defoliation treatments exhibiting signs of frost damage in June of 2001, 2002 and 2003. Different letters indicate statistical differences between treatments within each year (pairwise Chi-Square tests;  $\alpha = 0.017$ ).

grazing conditions are well documented (Pigott, 1983; Lehtonen and Heikkinen, 1995), but it is not clear if the mortality is caused by browsing, trampling or both.

Herbaceous biomass was very low in the woodlands compared to the other four community types (Figure 4.4), so competition with herbaceous vegetation is not a likely explanation for the poor performance of birch seedlings in birch woodlands. Frost damage, an index of winter desiccation, was intermediate in the woodlands despite the proposed sheltering effect (Figure 5.13). Milder microclimate in the sheltered woodlands thus appears to play only minimal role in the seedling growth and survival. Others have found greater frost heaving damage of *Picea abies* seedlings with increasing size of forest gaps in multistoried *Pinus* and *Picea* forest (Hanssen et al., 2007) but forest gaps provide both shelter and more light levels whereas the Hafnarskogur birch woodland only provided shelter. The most striking difference between the communities is the amount of light available in the communities. The woodland, with its dense canopy does not only provide shelter from wind, but light as well. *B. pubescens* requires good light for growth, thus the high mortality in the woodlands may be caused by insufficient light levels. Herbivory was high in both the woodlands and the grasslands, but their growth rates differed markedly. That makes herbivory unlikely to be the driving force behind the increased woodland mortality. Insect herbivory is known to increase mortality levels (Neuvonen et al., 2001), especially under adverse climate condition (Kallio and Lehtonen, 1973; Haukioja et al., 1985). The fact that no relationship appears to be between herbivory and mortality in this study may suggest that climate was not a critical factor during the experiment.

Both clipping treatments reduced seedling vigor and growth, and mortality increased markedly with time from the first treatment application. The clipping effect thus became more pronounced with time, suggesting a carryover treatment effect, possibly reflecting a depletion of energy and nutrients stored in the seedling tissues. The first clipping treatments were applied in August 2000, and then reapplied in August 2001. In June 2002 the mortality rate had risen significantly, suggesting that a tolerance limit had been reached. Repeated intense clipping is therefore detrimental for the seedlings. In contrast to insect defoliation, which seldom lasts more than 2-3 years (Tenow, 1972), livestock browsing in Iceland was continuous and increasingly severe in harsh years, thus likely resulting in repeated depletion of resources that has detrimental effects on seedlings although mature trees can tolerate browsing. In Scotland, less browsing intensity of birch saplings adjacent to tall vegetation and good quality forage has been observed (Pollock et al., 2005). This might be explained by the livestock preference of good quality forage if available over birch, or that the tall vegetation protects the saplings. Both scenarios support the theory that intense grazing pressure would lead to more intense browsing or trampling damage of the birch. Long-term or continuous grazing would thus be expected to cause higher seedling mortality.

This experiment was conducted to assess the potential tolerance of birch seedlings to repeated browsing under natural field conditions in order to enhance our understanding of the historic trends in the birch woodland decline. The results indicate that natural regeneration will be slow under continuous grazing, and use of birch seedlings for land reclamation might not be successful unless browsing is absent.

CHAPTER V  
GRAZING, SURFACE STABILITY AND CRYOTURBIC PROCESSES IN  
A HIGH-LATITUDE ECOSYSTEM

**Introduction**

The physiognomy of Iceland has changed dramatically since the settlement in the 9<sup>th</sup> and 10<sup>th</sup> centuries. Pollen data and historical evidence suggest that Icelandic lowlands were dominated by birch (*Betula pubescens* Ehrh.) (Thorgilsson, 1968; Hallsdottir, 1992, 1995) and 15 - 25% of the country had woodland cover (Sigurdsson, 1977; Gudjonsson and Gislason, 1998; Olafsdottir et al., 2001). Today birch cover is only 1 % (Gudjonsson and Gislason, 1998), and herbaceous cover is estimated to have declined about 60 % (Thorsteinsson, 1978). Erosion of the extensive Andisols (Arnalds, 2004) has created barren deserts, now estimated to cover 36 % of the country, and an additional 10 - 15 % of the land area is categorized as having limited plant cover (LMI, 1993). Birch woodlands appear to have declined soon after humans arrived, at which time ecosystems that evolved to cope with harsh climate and volcanic activity were subject to intense biotic disturbances related to agriculture and farming.

The woodland disappearance is regarded as a precursor to land degradation, (Chapter III) as it is often followed by soil erosion (Carson, 1985; R. C. Derose, 1993; Olafsdottir and Gudmundsson, 2002; Rosenmeier et al., 2002). Openings created in the tree canopy resulting from clearing and grazing promotes radiative heat loss and attenuates snow accumulation (McKay and Gray, 1981) with corresponding insulation

loss (Hinkel and Hurd, 2006). Grazing and trampling by livestock may similarly reduce the insulative capacity of the ground layer vegetation (Cole and Monz, 2002). In Iceland, where the climate is maritime with winters characterized by temperature fluctuations around 0 °C, these changes in energy balance potentially increase the frequency and intensity of freeze-thaw cycles (Williams and Smith, 1989) and amplify cryoturbic disturbances (frost heaving, needle ice formation), with adverse consequences for plant recruitment (Goulet, 1995; Aradottir and Arnalds, 2001; Nagamatsu et al., 2002). Hummocks, a ubiquitous landscape feature in high latitude systems (Van Vliet-Lanoe et al., 1998), are likely an expression of such cryoturbation processes (Schunke and Zoltai, 1987). It is hypothesized that early stages in the degradation sequence are characterized by intensification of cryoturbation processes (see Chapter III). Andisols, the dominant soil order in Iceland (Arnalds, 2004) are characterized by low aggregate cohesion and high water holding capacity (Maeda and Soma, 1986), two properties that make them particularly unstable when exposed to freeze-thaw cycles. Changes in surface microtopography should therefore be symptomatic of the initial phases in a degradation chronosequence. However, the extent to which soil surfaces might be destabilized by freeze-thaw events may depend on the nature of the ground layer vegetation, which provides insulation (Decker and Ronningen, 1957) and a network of roots and mycorrhizae that bind and stabilize soil particles.

Chapter IV focused on the direct effects of grazing on birch seedlings, hence their ability to regenerate and persist when defoliated. The specific goals of the studies summarized in this chapter were to improve our understanding of how vegetation



changes accompanying deforestation and livestock grazing affect cryoturbic disturbances.

To address this goal, experiments were conducted in five plant community types representing a degradation chronosequence (see Chapter II): woodlands, woodland heathlands (w heathlands), grasslands, savanna heathlands (s heathlands), and savanna. It was hypothesized that mineral soil surface stability would be high in woodlands compared to more open community types such as grasslands and heathlands, owing to lower convective and night-time radiative heat losses (Jordan and Smith, 1995) and greater snow accumulation (Bunnell et al., 1985; Essery et al., 1999; Pfister and Schneebeli, 1999). As a result, the frequency and intensity of frost heaving are predicted to increase with decreases in birch cover and decreases in soil strength. Accordingly, grazing disturbances (defoliation and trampling of the herb layer) were hypothesized to have the greatest impacts on surface soil stability and soil strength in open grassland and heath communities, and the least impact on these properties in woodland communities.

### **Material and Methods**

This study was conducted at two sites in west Iceland: Hafnarskogur and Keldnaholt. At Hafnarskogur experiments were performed in five plant community types: birch (*Betula pubescens* Ehrh.) woodlands, woodland heathlands (w heathlands), grasslands, savanna heathlands (s heathlands), and savanna. The two heathland communities represent a transitional zone between the woodlands and savanna, and the grasslands. See Chapter 2 “Study area” and Appendix A for more detailed descriptions of climate, soils and vegetation of the Hafnarskogur study site. At Keldnaholt an

experiment was set up in a heathland plant community type, similar to the heathland communities found at Hafnarskogur. The Keldnaholt site was chosen due to its proximity to the Agricultural Research Institute in Reykjavik (now the Agricultural University of Iceland), which allowed easy monitoring of data loggers used in the study.

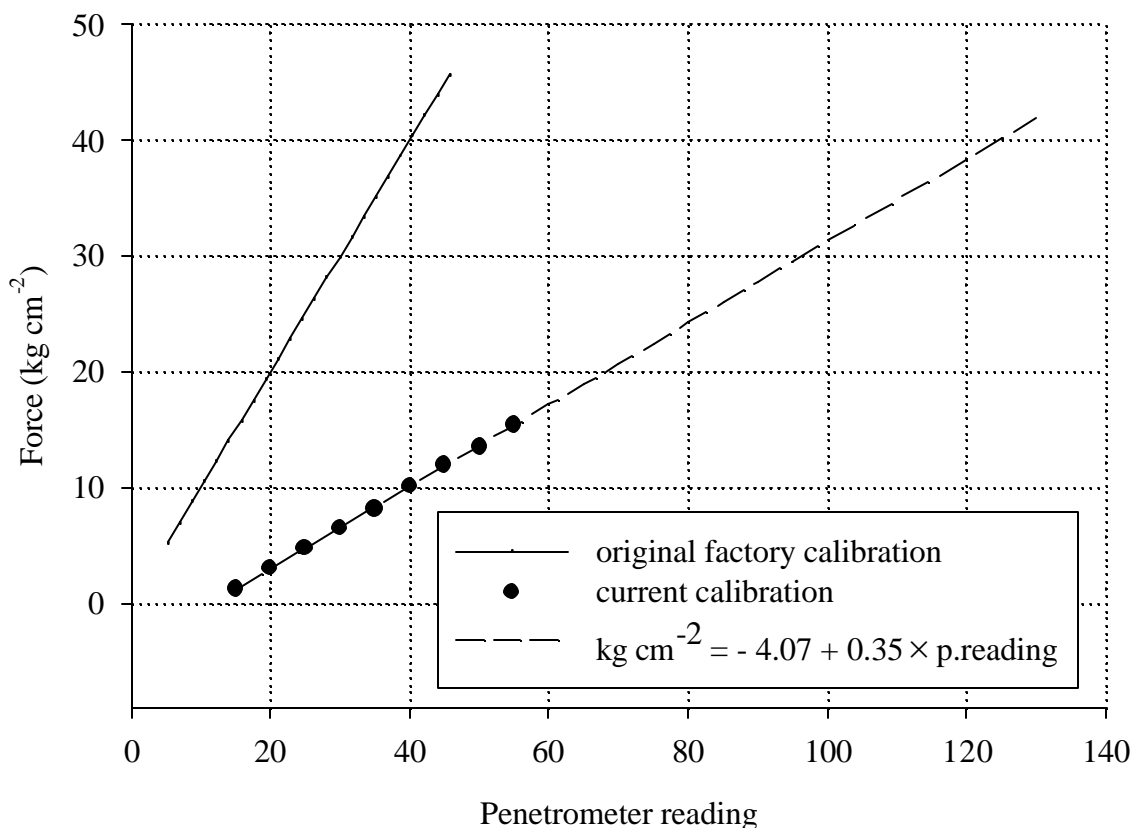
Three experiments were conducted: (i) a peg frost heaving experiment examined the influence of sward properties and surface strength on frost heaving of pegs, assuming that displacement of the pegs is an indicator of frost heaving potential (Portz, 1967; Johnson and Hansen, 1974; Péres, 1997); (ii) a soil surface strength and surface microtopography experiment sought to quantify surface strength and microscale surface movements in different plant communities, and how these are influenced by different levels of simulated livestock grazing disturbances; and (iii) an insulation experiment quantified the effect of sward insulation on soil frost and surface microtopography. Data in all experiments were analyzed with SPSS v.11.0 - v13.0 (SPSS Inc., 2001b, 2004).

Soil strength is the “property of the soil that causes it to resist deformation” (Brady and Weil, 1998). It is typically quantified by the force needed to push a pin, of certain surface area, into the soil. Here surface strength is used as a measure of the resistance to deformation provided by the mineral soil, plant roots, mycorrhizae and organic matter of the sward layer. The sward layer is defined here as all mineral and organic (both living and dead) materials extending from the land surface to the mineral soil surface. This would include portions of mosses, grasses, herbs and litter.

### **Frost heaving**

Frost heaving potential was assessed by quantifying displacement of wooden pegs (cylindrical, flat point, 3 mm diameter, 300 mm long, 1.4 g dowels). Two plots (0.5 × 0.5 m) were established in each of the five community types at Hafnarskogur in late July 2002. Permanent markers (metal rods, driven > 1 m into the ground) were installed in three plot corners to serve as permanent reference points. A perforated plate with a grid of holes at 5 cm intervals was positioned at corner markers and leveled. Pegs were then inserted into the soil through the holes (n = 100 pegs per plot) until 15 mm protruded above the plate. Vertical displacement of pegs was recorded June 2003 by re-positioning the plate at the markers and measuring peg height to the nearest mm. In June 2003, sward thickness was measured to the nearest cm at six random locations in each plot, and surface strength was measured with a soil penetrometer [with a circular flat point (13 mm diameter); Proctor Model CN-419, Soiltest, Inc.] at six random points near each peg plot. The penetrometer was calibrated prior to data collection to adjust for changes in spring tension (Figure 5.1). Braun-Blanquet cover classes (1 < 1 %, 2 = 1 - 5 %, 3 = 6 - 10 %, 4 = 11 - 15 %, 5 = 16 - 25 %, 6 = 26 - 50 %, 7 = 51 - 75 %, 8 = 76 - 100 %) (Pandeya et al., 1968) of vascular plant species, bryophytes, lichens, litter, stones and bare soil were visually estimated within each plot in late July 2002.

Pegs were categorized as undisturbed [peg movement < 2 mm; or 'heaved' (Orradottir, 2002)]. Chi-square tests were used to determine whether the observed frequencies of heaved pegs varied by community. Mean peg heaving for the heaved pegs was calculated from displaced pegs (hence undisturbed pegs excluded).



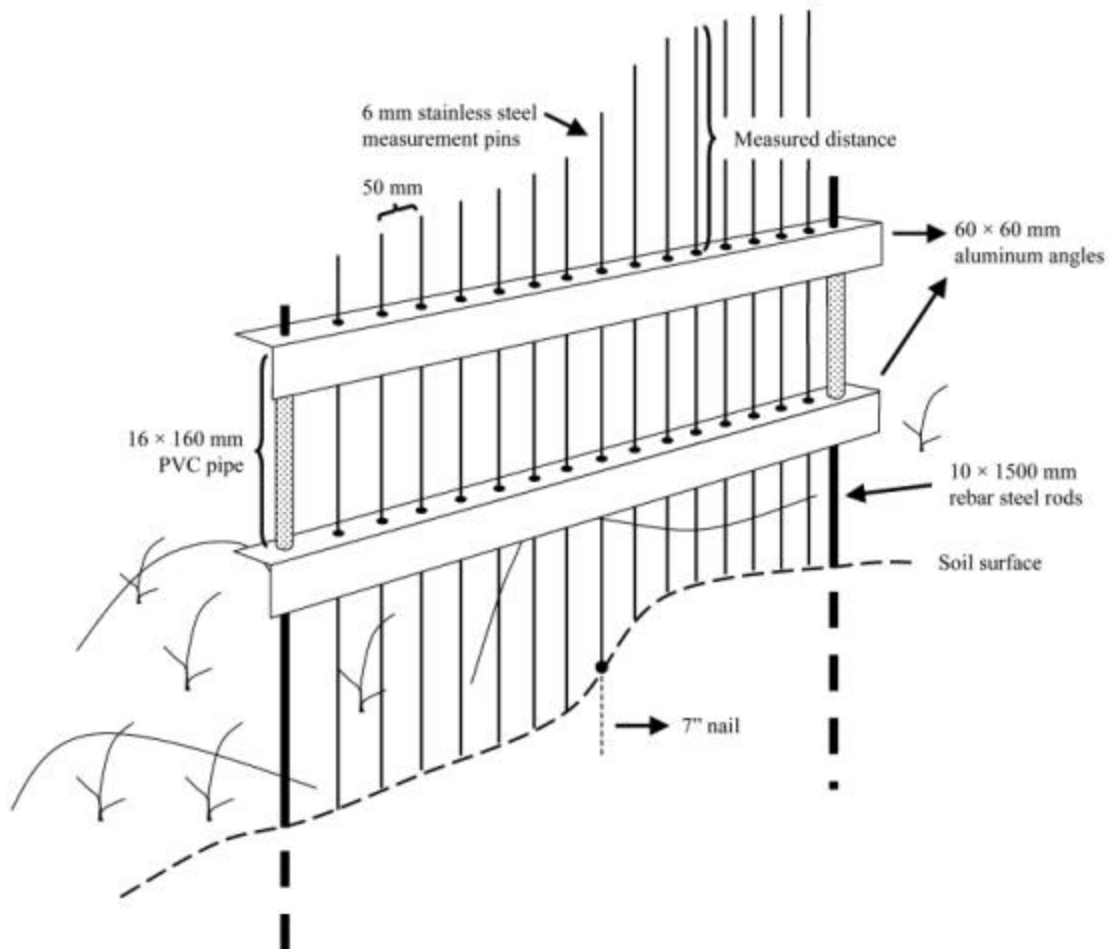
**FIGURE 5.1. Calibration curves for the Proctor Model CN-419 (Soiltest, Inc.) penetrometer.** Values on the Y-axis are calculated for a 1.29 cm<sup>2</sup> needle (1/5 sq. in.). The original factory calibration is based on a supplied penetrometer datasheet (Soiltest, Inc.). Current calibration curve was obtained by pressing the penetrometer against a toploading scale (25.0 kg capacity; Mettler, Inc.). Each calibration point is the average of six penetrometer scale readings (15, 20, 25, [...], 55 [n = 54]). The data were then used to calculate the correlation between the penetrometer reading and applied force, corrected for needlepoint area. Standard error for the current calibration data is omitted from the image, but ranged from 0.031 - 0.228.

Differences between communities were assessed with Kruskal-Wallis tests; and Mann-Whitney U tests were used for pairwise comparisons, as ANOVA assumptions were not met. Braun-Blanquet cover classes data were transformed to midpoint percentages before calculating average cover. Spearman's  $\rho$  was computed to determine the degree and direction of association between peg displacement and sward thickness, surface strength and plant cover.

### **Soil surface strength and surface microtopography**

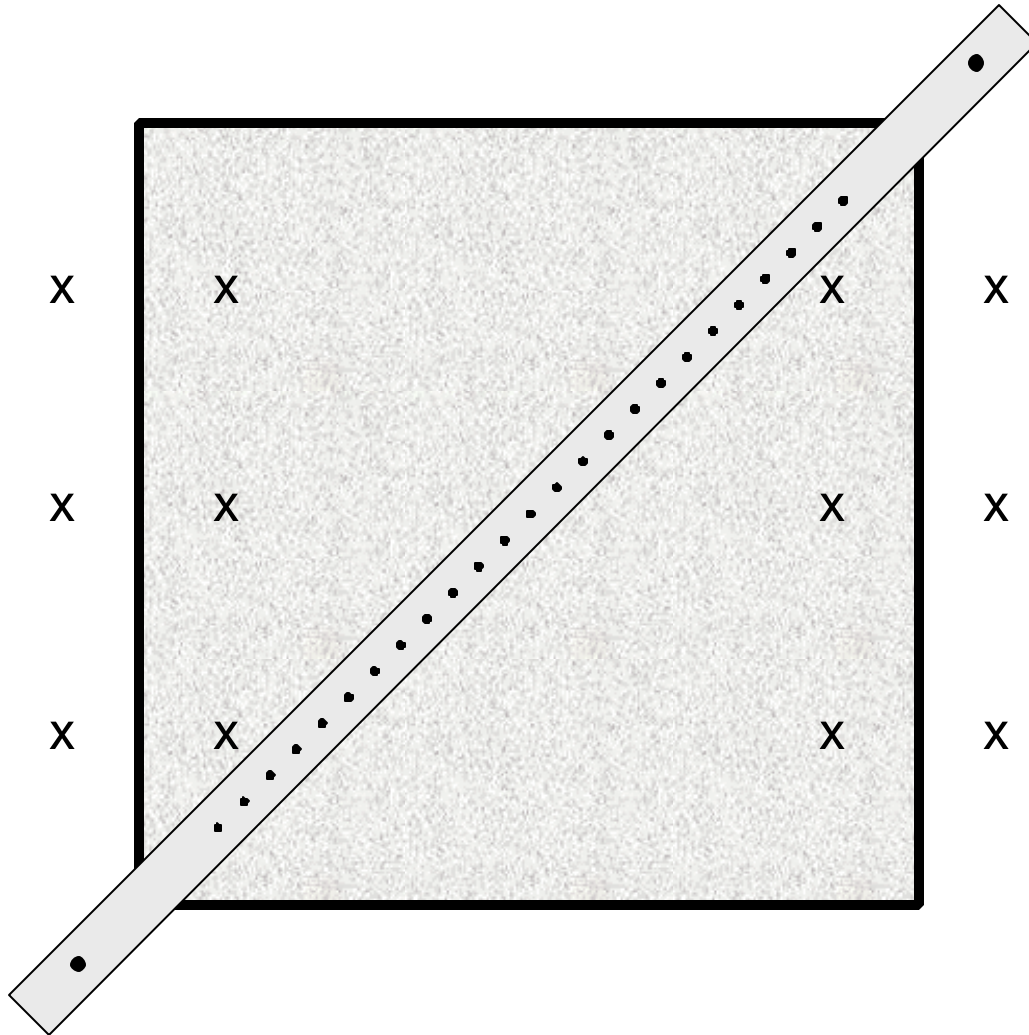
Three  $10 \times 10$  m macroplots were established in each of the five community types at Hafnarskogur in 1999. Three treatments and a control were established in 12 randomly selected  $0.5 \times 0.5$  m subplots within each macroplot (3 treatment replicates per macroplot) in 1999 and repeated in the same plots in 2000. The treatments simulate different livestock grazing disturbances in each community ( $n = 9$  replicates): (1) clipping (all vegetation trimmed down to  $\sim 1$  cm height), (2) trampling (sward pounded with a hammer (60 mm diameter head, weight  $\sim 1.4$  kg) and then compressed by human foot traffic), and (3) clipping and trampling. The treatments were applied in August of each year, prior to the first frost. The trampling treatments were intended to simulate severe disturbances as might occur with high concentrations of large domestic herbivores (e.g., sheep, reindeer, horses). Measurements on soil microtopography and sward were made in spring and autumn each year, through 2003; surface strength was quantified every spring; and ground cover recorded every spring from 2001 to 2003.

Soil Erosion Bridges [SEB; (Shakesby et al., 1991; Shakesby, 1993; White and Loftin, 2000)] were used to quantify changes in surface microtopography (Figure 5.2). The SEB consisted of two permanent stakes driven deep ( $> 1$  m) into the soil. Two horizontal bars were fitted to the stakes at a fixed position, and vertical pins ( $n = 14$ ; stainless steel, flat point, 4 mm in diameter, 680 mm long, 67.7 g) guided through aligned holes at 5 cm intervals along the parallel bars. This allows repeated measurements of the distance between the bars and the surface below, and is well suited for detecting change in surface microtopography over time. The SEB readings were obtained from permanent locations along the diagonal of 12 randomly selected  $0.5 \times 0.5$  m subplots (Figure 5.3) randomly located within three  $10 \times 10$  m macroplots in each plant community. Pin height differences between seasons (fall to spring, and spring to fall) over five years were computed from SEB, and two surface microtopographic metrics were calculated for each SEB placement: 1) *mean mineral surface height* (i.e. lowering/rising of surface), calculated by averaging the pin height differences for each SEB; negative values indicate lowering of the surface as with soil loss or compaction; positive values denote heaving of soil; and 2) *mean absolute movement*, calculated by averaging the absolute pin height differences for each SEB (all measurement differences denoted as positive). This is an assessment of soil stability [as opposed to net changes in height; (Shakesby et al., 2002)]. Pin height differences  $\geq \pm 5$  cm were considered extreme outliers, likely measurement errors, and were excluded. When this resulted in exclusion of more than 3 pins per SEB, that SEB was compared to bridges in the same community and treatment and excluded if it showed abnormal numbers. One bridge was



**FIGURE 5.2.** A schematic diagram (not to scale) of the soil erosion bridge (SEB) used in this study [based on Shakesby et al. (1991)]. Pairs of 1.5 m stakes separated by 95 cm were driven at least 1 m into the ground along a transect line. Two aluminum 90° angle bars were mounted on the rebars and used to guide pins to the same spot every time measurements were made. The lower bar rests on a pair of hose clamps (not shown) left on rebars between measurements and is leveled. The upper bar sits on a pair of 160 mm PVC pipes, thus ensuring a constant distance between the two parallel bars.

To ensure stationary between measurements, a 7" nail was driven into the ground under one of the central measurement pins. If the pin hit the nail, and the bars were level at subsequent measurement dates, then the bridge setup was considered to be intact and the measurement spots under each pin thus the same as measured previously.



**FIGURE 5.3.** A schematic diagram of a 0.5 × 0.5 m micro plot. X's indicate approximate locations of penetrometer readings (six inside and six outside the plots). The diagonal bar shows SEB placement. Not to scale.



excluded after such comparisons. The average number of pins used per SEB (out of 13 possible) was  $12.9 \pm 0.09$  for the winter (fall to spring) dataset and  $12.5 \pm 0.1$  for the summer (spring to fall) dataset. In addition to the pin height measurements, sward thickness was measured under each SEB pin in the control and clipped treatments, and ground cover (vegetation life form groups: dwarf shrubs, graminoids and herbs, bryophytes and lichens; litter; bare cover: soil and stones) under each pin was recorded every spring from 2001 to 2003.

Surface strength in and just outside each subplot ( $n = 6$  readings inside,  $n = 6$  readings outside; Figure 5.3) was quantified every spring with a soil penetrometer (described earlier). In spring 2000 only two of the three macroplots were measured in each community. One soil sample (0 - 5 cm and 5 - 15 cm depths) was collected from each subplot in autumn 2003. Samples from treatment subplots within macroplots were pooled and analyzed for soil organic carbon (SOC; %) by dry combustion (LECO CR-12 carbon analyzer; (Nelson and Sommers, 1982) of sieved fine fractions ( $< 2$  mm) dried at  $60$  °C.

Time constraints were such that SEB readings were only determined in one macroplot for each community in fall 1999. These macroplots proved comparable to the other two macroplots in each community (tested with ANOVA on the fall 00 to spring 01 dataset) therefore the fall 1999 to spring 2000 (f99 to s00) data are presented with the other years. Only one macroplot was established in the grassland community hence, it was not included in statistical analysis. Statistical analyses were done separately on winter and summer datasets. Statistical differences in mineral surface

height and absolute soil surface movement between communities and treatments were determined with partially nested ANOVA models, where communities were tested with “macroplot (community)”; and treatments and community  $\times$  treatment interactions were tested with the “treatment  $\times$  macro (community)” term (Neter et al., 1996). When differences were significant, Bonferroni comparisons were used to ascertain pairwise differences. When ANOVA assumptions were violated, Kruskal-Wallis tests were used to test community and treatment effects separately, their interaction inferred from plots, and pairwise differences were then ascertained with Mann-Whitney U tests.

Frequencies of ground cover categories were calculated for each treatment, across plant communities, and Chi-Square tests were used to determine whether a relationship existed between treatment and the frequency distribution of cover classes. Effects of two cover categories (grass + herbaceous vegetation and moss) on surface metrics (height change and absolute movement) were tested with Kruskal-Wallis on the combined control and clipped treatments data from f02 to s03. Spearman’s  $\rho$  was computed to determine correlations between each of the two surface metrics and the number of vegetation functional layers (dwarf shrubs, graminoids and herbs, bryophytes and lichens, and litter) and the sward thickness. Averages of treatment subplots were used for these correlation calculations.

Differences in SOC were tested with two-way ANOVA with communities and treatments as main effects; and with one-way ANOVA to check for community differences in control plots. Regression analysis was used to test whether absolute movement of the surface predicted changes in SOC content.

Surface strength in untreated subplots in each community were compared with ANOVA across years. When main effects were significant, Bonferroni comparisons were used to ascertain pairwise differences. Surface strength was compared between years, communities and treatments using partially nested repeated measures ANOVA model on the differences between inside and outside plot readings (*outside – inside + 11* to avoid negative numbers) averaged for each subplot. Differences between inside and outside subplots readings were used to adjust the treatment effect for the surface strength observed at each subplot. Measurements were not complete for the grasslands and year 2000, therefore they were not included in the repeated measure ANOVA. Interactions between year, community and treatments were significant therefore separate ANOVAs were conducted for each year. That model included community, clipped and trampled as main effects and all possible interaction terms. Communities were tested with the “macroplot (community)” term but other factors were tested with the model error. Clipped and trampled were used as main effects (instead of treatment) to test whether one or both of these treatment effects cause the interaction with communities. Spearman’s  $\rho$  was used to test for associations between surface strength measured in spring 03 and the two surface microtopographic metrics from fall 02 to spring 03. These dates were chosen because clear treatment effects were apparent by these times.

### **Simulated sward insulation**

The effect of sward insulation on surface stability was experimentally evaluated in a field trial initiated in autumn 2000 in a heathland community at the Keldnaholt site,

Reykjavík. Twelve  $0.5 \times 0.5$  m plots were established and set up for SEB measurements as described above. Vegetation was removed down to the mineral soil surface in nine plots; and three plots were left intact as controls. Vegetation removal plots received one of three levels of insulation: none, 25 mm and 100 mm ( $n = 3$  for each). The material used for insulation was a hydrophobic mineral mat ('Rockwool'). The mats were covered with 8 mm mesh to exclude mice. Initial SEB measurements of surface microtopography were made in fall 2000, again in spring 2001, autumn 2002 and spring 2003.

One frost tube (Rickard and Brown, 1972) was installed in the center of each plot to measure maximum soil frost penetration. These frost tube consisted of an outer black PVC tube (20 mm inside diameter [ID]; 24 mm outer diameter [OD]) and an inner removable transparent tube (12 mm ID; 15 mm OD) containing 0.05 % potassium permanganate solution. As the solution freezes, salt is expelled from solution and the frozen portion becomes transparent. The boundary between the transparent and colored solutions persists after thawing, thus allowing determination of maximum frost depth next spring. Maximum soil frost depths (cm) were recorded on June 16<sup>th</sup> 2001 and sward thickness (cm) in the vicinity of frost tubes was measured on control plots. For the insulation treatments sward thickness was regarded as the thickness of the Rockwool mats. Soil temperature (soil thermometers model 107, connected to 21X datalogger from Campbell Scientific Inc.) was recorded every 30 minutes at 5 cm below the mineral soil surface in each plot from 7 November 2000 through 5 March 2001.

Pin height differences between autumn and spring were computed for each pin in each SEB, and the two surface microtopographic metrics computed as described in the preceding section. Variances between treatments were not homogenous, thus the Kruskal-Wallis test was used to test treatment differences in *mean mineral surface height* and *mean absolute movement*. When differences were significant, Mann-Whitney U tests were used to ascertain pairwise differences. Correlations between the two surface microtopographic metrics and sward thickness, maximum soil frost depth and mean daily soil temperature (during the measurement period) were assessed using Spearman's  $\rho$ .

## Results

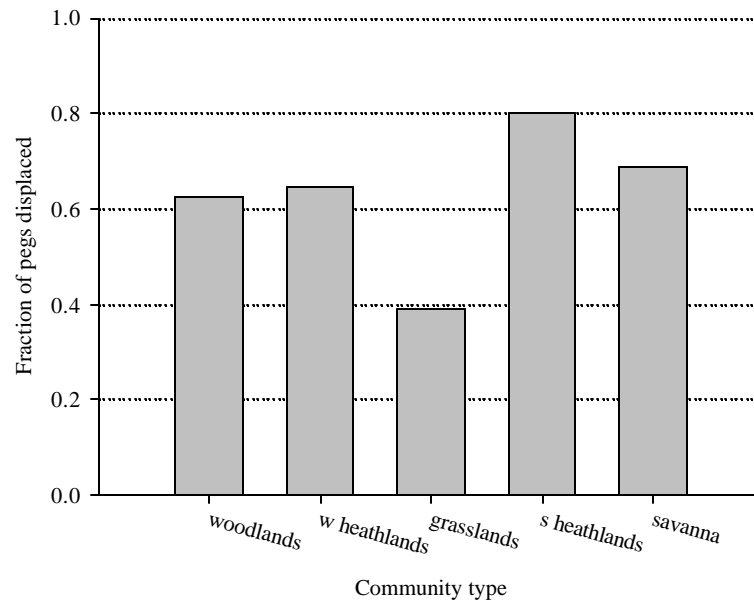
### Frost heaving

Peg displacement frequency varied by community ( $C^2 = 26.9$ ,  $df = 4$ ,  $p < 0.001$ ; Figure 5.4 A), and was lowest in grasslands, and highest in savanna heathlands. Community differences in the magnitude of peg displacement were also significant (Kruskal-Wallis:  $p < 0.001$ ), being least in woodlands and grasslands, and greatest in savanna heathlands and savannas (Figure 5.4 B). No significant correlations were found between the extent of peg displacement, soil surface strength, sward thickness and vegetation cover.

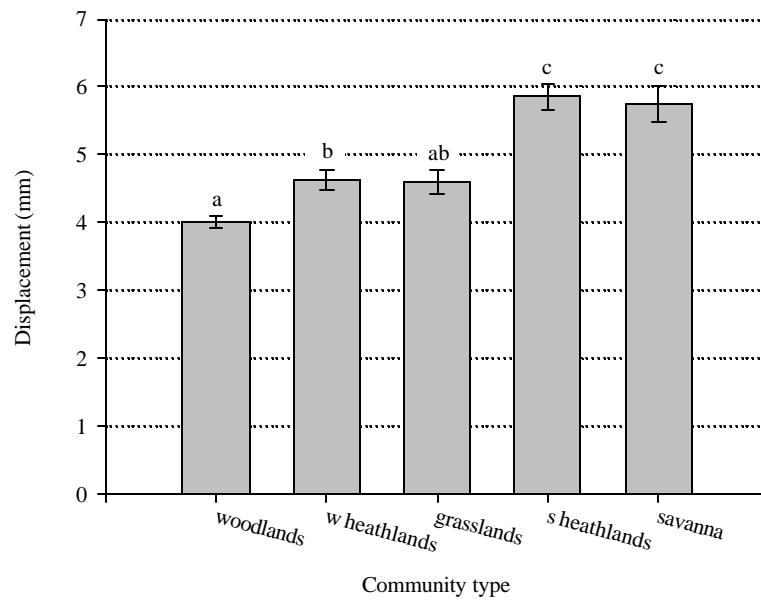
**FIGURE 5.4. (A) Fraction of pegs displaced and (B) mean ( $\pm$  SE) displacement of pegs in five plant communities at Hafnarskogur, in the 2002 - 2003 winter. Different letters in panel B show statistical differences between communities ( $p < 0.05$ ).**

**Note: w heathlands = woodland heathlands, s heathland = savanna heathlands.**

A: Fraction of pegs displaced



B: Mean displacement of pegs



## **Soil surface strength and surface microtopography**

### *Soil/surface microtopographic metrics*

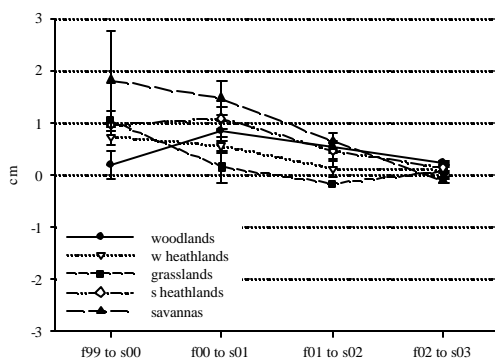
Temporal changes in ground surface height and absolute surface movement in the five plant communities and four treatments are summarized in Figure 5.5 for four winters, and in Figure 5.6 for three summers. High absolute surface movements generally lead to a large rising or lowering of the soil surface. Table 5.1 summarizes the statistical results for mean changes in soil surface height. Significant differences were found between communities in the first two winters. The woodland and savanna communities were significantly different in f00 to s01, but other communities were comparable. Changes in soil surface height differed significantly between treatments in all winters.

Communities  $\times$  treatment interactions were significant in f99 to s00 and f01 to s02. When the interaction was not significant (f00 to s01 and f02 to s03), clipped and control plots were comparable and significantly different from trampled and clipped + trampled plots (the latter being comparable in f02 to s03 but different in f00 to s01). Mean absolute surface movement was comparable between communities in all winters; and differences were significant between treatments in all winters (Table 5.2). Communities  $\times$  treatment interactions were significant in all winters except f02 to s03. During that period, absolute surface movement in clipped and control plots was comparable and significantly different from trampled and clipped + trampled which were comparable. Results from statistical tests on the two surface metrics in summers are in Tables 5.3 and 5.4. Treatment effects were significant for both surface metrics in all



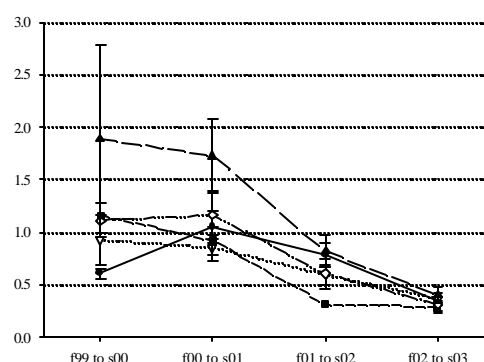
## Soil surface height

A: control

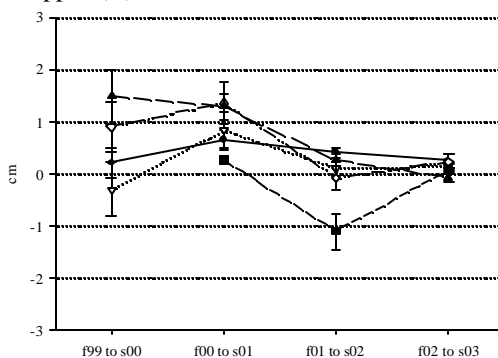


## Absolute movement

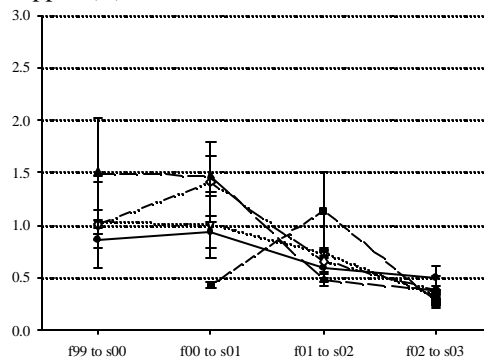
B: control



C: clipped (cl)



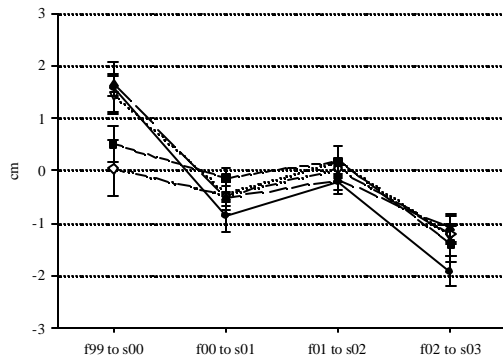
D: clipped (cl)



**FIGURE 5.5.** Temporal changes in mean ( $\pm$ SE) soil surface height (A, C, E, G) and mean ( $\pm$ SE) absolute surface movement (B, D, F, H) for five plant communities and four grazing/trampling treatments in Hafnarskogur over four winters. Data are mean pin height differences (A, C, E, G) and mean absolute pin height differences (B, D, F, H) between fall and spring measurements for each SEB;  $n = 40$  for f99 - s00;  $n = 127$  for f00 - s01;  $n = 104$  for f01 - s02;  $n = 126$  for f02 - s03. Note: different scales in left and right column; w heathlands = woodland heathlands, s heathland = savanna heathlands.

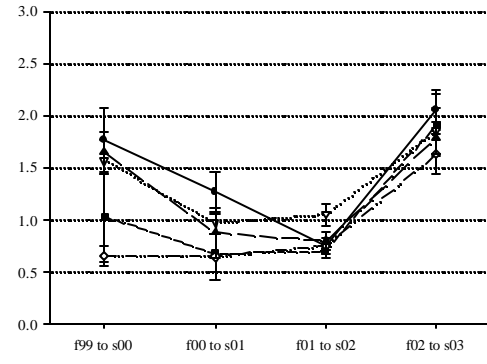
Soil surface height

E: trampled (tr)

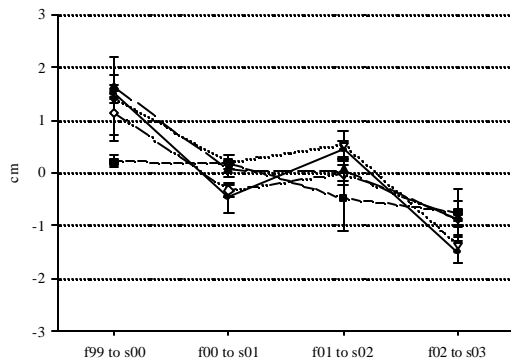


Absolute movement

F: trampled (tr)



G: cl & tr



H: cl & tr

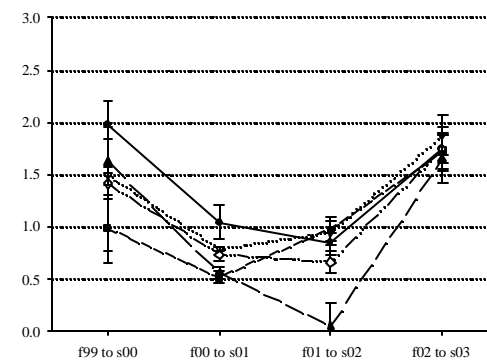
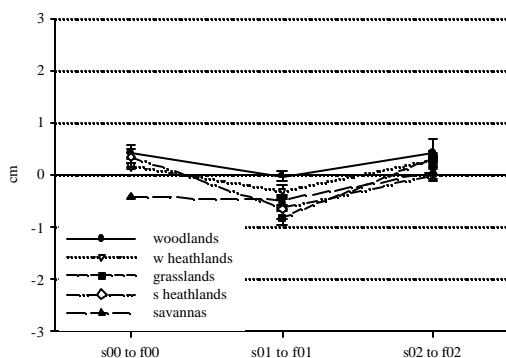


FIGURE 5.5. Continued.

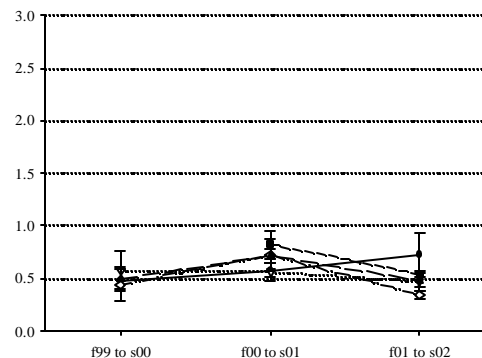
## Soil surface height

A: control

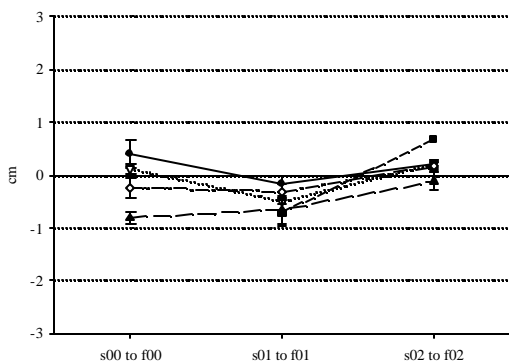


## Absolute movement

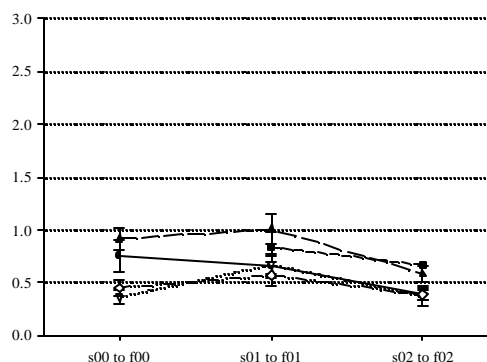
B: control



C: clipped (cl)



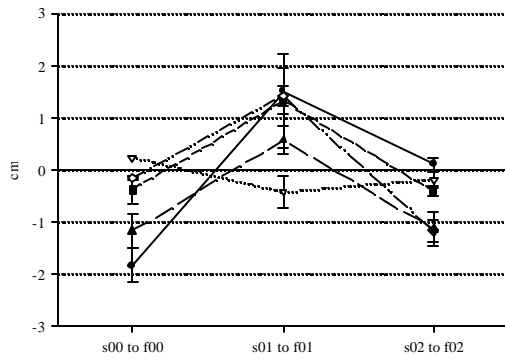
D: clipped (cl)



**FIGURE 5.6.** Temporal changes in mean ( $\pm$ SE) soil surface height (A, C, E, G) and mean ( $\pm$ SE) absolute surface movement (B, D, F, H) for five plant communities and four grazing/trampling treatments in Hafnarskogur over three summers. Data are mean pin height differences (A, C, E, G) and mean absolute pin height differences (B, D, F, H) between spring and fall measurements, for each SEB:  $n=38$  for s00-f00;  $n=124-125$  for s01-f01;  $n=100$  for s02-f02. Note: different scales in left and right column; w heathlands = woodland heathlands, s heathland = savanna heathlands.

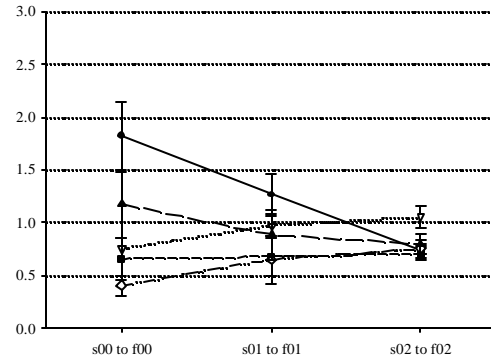
Soil surface height

E: trampled (tr)

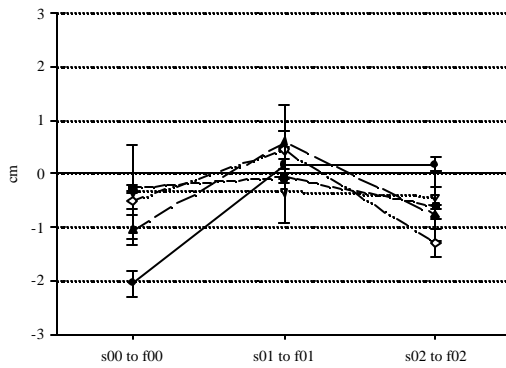


Absolute movement

F: trampled (tr)



G: cl & tr



H: cl & tr

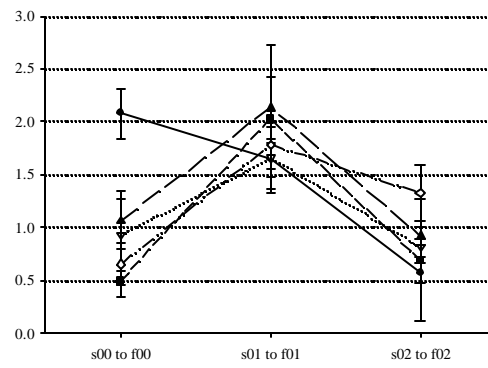


FIGURE 5.6. Continued.

**TABLE 5.1**

**Results from statistical tests on mean soil surface height measured by SEBs in four communities (grassland not included) and four treatments in Hafnarskogur over four fall (f) to spring (s) periods. Analyses performed on data means from each SEB (n = 40 for f99 to s00; n = 127 for f00 to s01; n = 104 for f01 to s02; n = 126 for f02 to s03), with Kruskal-Wallis and interaction terms evaluated from plots, except f00 to s01 data were analyzed with ANOVA.**

	Community	Treatment	Community × treatment
Winter	p	p	p
f99 to s00	0.038	0.046	present
f00 to s01 <sup>φ</sup>	0.035	0.001	0.804
f01 to s02	0.519	0.002	present
f02 to s03 *	0.813	0.001	absent

<sup>φ</sup> Bonferroni comparisons: savanna and woodland different; all treatments different except clipped and control comparable

\* Mann-Whitney U test comparisons: clipped and control different from trampled and clipped and trampled.

**TABLE 5.2**

**Results from statistical tests on mean absolute movement of the ground surface measured by SEBs in four communities (grassland not included) and four treatments in Hafnarskogur over four fall (f) to spring (s) periods. Analyses performed on data means from each SEB (n = 40 for f99 to s00; n = 127 for f00 to s01; n = 104 for f01 to s02; n = 126 for f02 to s03), with Kruskal-Wallis and interaction terms evaluated from plots, except f00 to s01 data were analyzed with ANOVA.**

	Community	Treatment	Community × treatment
Winter	p	p	p
f99 to s00	0.137	0.036	present
f00 to s01	0.559	0.002	0.001
f01 to s02	0.382	0.018	present
f02 to s03 <sup>φ</sup>	0.819	0.001	absent

<sup>φ</sup> Mann-Whitney U test comparisons: clipped and control different from trampled and clipped and trampled.

**TABLE 5.3**

**Results from statistical tests on mean soil surface height measured by SEBs in four communities (grassland not included) and four disturbance treatments at Hafnarskogur over three summers. Analyses performed on data means from each SEB (n = 38 for s00 to f00; n = 124 for s01 to f01; n = 100 for s02 to f02) with ANOVA, except s02 to f02 were analysed with Kruskal-Wallis and interaction terms evaluated from plots.**

	Community	Treatment	Community × treatment
Summer	p	p	p
s00 to f00	0.001	0.001	0.001
s01 to f01 <sup>φ</sup>	0.050	0.001	0.095
s02 to f02 *	0.001	0.001	absent

<sup>φ</sup> Bonferroni comparisons: savanna and woodland different; clipped and control comparable but different from trampled, clipped also different from clipped and trampled, but clipped and trampled comparable to both trampled and control.

\* Mann-Whitney U test comparisons: woodland different from w heathland and savanna, and savanna also different from w heathland; clipped and control different from trampled and clipped and trampled.

**TABLE 5.4**

**Results from statistical tests on mean absolute movement of the ground surface measured by SEBs in four communities (grassland not included) and four treatments at Hafnarskogur over three summers. Analyses performed on data means from each SEB (n = 38 for s00 to f00; n= 124 for s01 to f01; n = 100 for s02 to f02) with Kruskal-Wallis and interaction terms evaluated from plots, except s01 to f01 were analysed with ANOVA.**

	Community	Treatment	Community × treatment
Summer	p	p	p
s00 to f00	0.006	0.009	present
s01 to f01 <sup>φ</sup>	0.221	0.001	0.627
s02 to f02	0.090	0.001	present

<sup>φ</sup> Bonferroni comparisons; clipped and control different from trampled and clipped and trampled.



summers, and community effects were significant in all summers for soil surface height, and for the first summer for absolute surface movement.

In control plots and clipped treatments surface rise was generally observed in winter although of variable amount, but in the last winter all communities were crowded around zero and variances were small (Figure 5.5 A, C). Absolute movement of the surface was greater in the first winters than the last one, when movements were limited to the 0.3 to 0.5 cm range in all communities (Figure 5.5 B, D). However, the clipped grassland responded differently (perhaps reflecting the fewer SEB in this community). For the trampled and clipped + trampled treatments, the mineral surface height was generally lower than in the control and clipped treatments, except in the first winter, and the surface declined with time (Figure 5.5 E, G). In the fourth winter surfaces declined considerably; the greatest decline was observed in the trampled and clipped + trampled woodland (-1.9 and -1.5 cm) but the least decline in the clipped and trampled grassland (-0.7 cm). This large surface decline was accompanied by a large increase in absolute surface movement (1.6 to 2.1 cm) (Figure 5.5 F, H).

As treatment effects differed among the communities (Tables 5.1 and 5.2), each community response to the treatments was examined closely. The trampling and clipped + trampling treatments had immediate effect in the woodlands that differed from those in the control and clipped treatments: absolute surface movements were more than 2 times larger and rise of the surface 7- to 8-fold larger than in the clipped and control plots in the first winter. Treatment influences were more obscure in the other

communities, and the pattern among communities in the fourth winter was different from the first winter (Figure 5.5).

In control plots, surface height changes of  $-0.8$  to  $+0.4$  cm were observed in summer in all communities, and similar values were observed in the clipped treatments (Figure 5.6). Surface level changes were greater and more variable in the trampled and clipped + trampled treatments, and were generally accompanied by greater absolute surface movements than that observed in clipped and control plots. Treatment effects were clear in the woodlands in the first summer; surfaces were lowered in trampled and clipped + trampled plots and elevated in clipped and control plots.

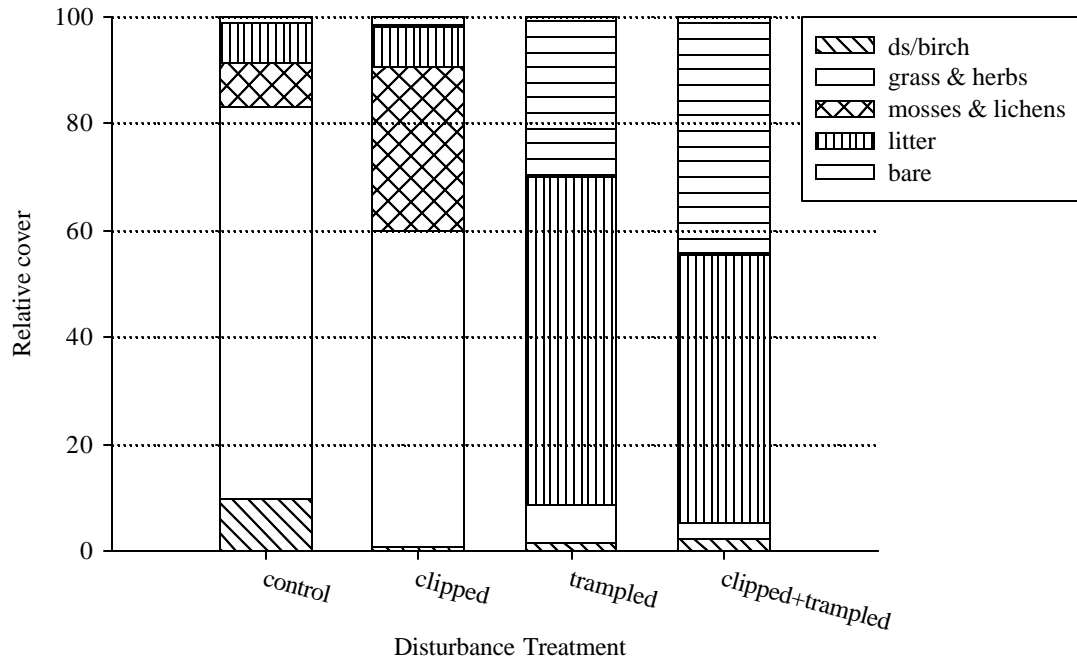
Frequency distribution of the five ground cover classes for the control and treatments plots are presented in Table 5.5. The large increase in moss and lichen cover in the clipped woodlands compared to the control is noteworthy, as is the large decline in grass and herb cover in the clipped treatments compared to the control plots in the woodlands.

Frequency distribution of the five ground cover classes varied with treatment (Chi-Square:  $C^2 = 1549.5$ ,  $df = 12$ ,  $p < 0.001$ ) (Figure 5.7), with no two treatments having comparable cover class distributions. Grasses and herbs declined from 73.4 % in control plots to 59.3 % in clipped plots, and fell to 7.1 % and 2.7 % in trampled and clipped + trampled plots, respectively. Mosses and lichens increased from 8 % cover in control plots to 30.7 % in clipped plots; and virtually disappeared from trampled and clipped + trampled plots. Effect of grass + herb vs. moss cover on surface microtopographic metrics were comparable in the combined control and clipped data

TABLE 5.5

Frequency distribution (%) of cover classes for the control (c) and the clipped (cl) treatment in five plant communities at Hafnarskogur, measured in fall 2003 (n = 448 for control and 469 for the clipped treatment). The high moss/lichens cover for the clipped treatments are due to the fact that the herbaceous layer has been removed. Note: w heathlands = woodland heathlands, s heathland = savanna heathlands.

Community types	Treatment	Dwarf shrub /birch	Grass/ herbs	Moss/ lichens	Litter	Bare	Total
woodlands	c	0	68.3	8.7	23.1	0	100
	cl	0.8	33.3	52.0	13.8	0	100
w heathlands	c	21.6	66.4	6.4	1.6	4	100
	cl	1.6	57.3	33.9	4.0	3.2	100
grasslands	c	0	90.5	7.1	2.4	0	100
	cl	0	89.3	3.6	7.1	0	100
s heathlands	c	0	89.3	6.3	4.5	0	100
	cl	0	71.1	16.5	7.2	5.2	100
savanna	c	24.6	56.9	13.8	4.6	0	100
	cl	0	74.2	21.7	4.1	0	100



**FIGURE 5.7.** Relative ground cover (%; autumn 2003; n = 448 - 491 for each treatment) for clipping/trampling treatments pooled across five plant communities in Hafnarskogur. Data are cover type recorded under each pin in all SEB.

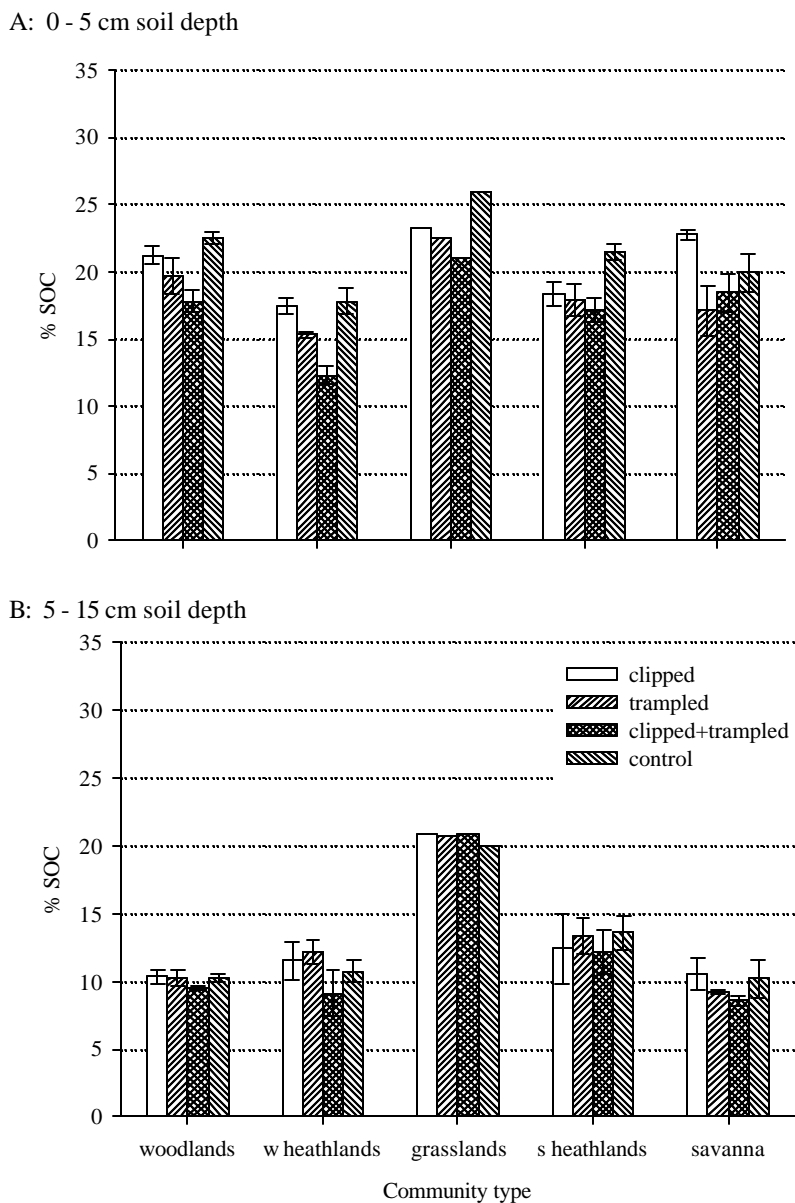
from fall 02 to spring 03 (Kruskal-Wallis:  $n = 418$ ,  $p = 0.761$  for height of surface;  $n = 418$ ,  $p = 0.215$  for absolute movement of surface).

Changes in surface height were positively correlated with the number of functional vegetation layers in f00 to s01 (Spearman's  $\rho = 0.543$ ,  $n = 52$ ,  $p < 0.001$ ) and f02 to s03 (Spearman's  $\rho = 0.664$ ,  $n = 52$ ,  $p < 0.001$ ). Absolute movement of the surface was negatively correlated with the number of functional layers in f02 to s03 (Spearman's  $\rho = -0.715$ ,  $n = 52$ ,  $p < 0.001$ ). There was a negative correlation between sward thickness and surface height in f00 to s01 (Spearman's  $\rho = -0.830$ ,  $n = 26$ ,  $p < 0.001$ ).

SOC content in control plots differed between communities at both soil depths (Figure 5.8) (ANOVA:  $p < 0.001$ ; grassland not included), but pairwise comparisons were only significant for the 0 - 5 cm depth where SOC in woodlands was significantly higher than that in wooded heathlands which had the lowest SOC content. Both communities and treatments had significant (ANOVA:  $p < 0.001$ ) effect on SOC content at 0 – 5 cm depths (Figure 5.8 A). Treatments had the same effect across communities as the treatment  $\times$  community interaction was not significant ( $p > 0.05$ ). SOC in control and clipped treatments was comparable and significantly higher than that in trampled and clipped + trampled treatments. SOC at 5 – 15 cm depths differed between communities (ANOVA:  $p < 0.001$ ), but treatments had no effect ( $p > 0.05$ ) (Figure 5.8 B).

Absolute surface movement explained 25 % of the variance in SOC at 0 – 5 cm depths and the following regression model ( $n = 52$ ,  $p < 0.001$ ,  $R^2 = 0.25$ ,  $R^2_{\text{adj}} = 0.23$ ) shows the best fit line:

$$\% \text{ SOC} = 21.25 - 2.127 \times \text{surface movement (cm)}$$



**FIGURE 5.8.** Mean ( $\pm$  SE; %) soil organic carbon (SOC) content at A) 0- 5 cm and B) 5- 15 cm soil depths in five plant communities at Hafnarskogur. Means are from three samples, each a composite of 3 cores (one core per subplot in each macroplot) taken in autumn 2003. Data for grasslands are comprised of only one sample per treatment. Note: w heathlands = woodland heathlands, s heathland = savanna heathlands.

### *Soil surface strength*

Surface strength differed between communities (ANOVA:  $p < 0.01$ ; only control plots) across the years. The surface strength was lowest in woodlands ( $9.6 \pm 0.5 \text{ kg cm}^{-2}$ ), highest in savanna heathlands ( $16.3 \pm 0.5$ ) and intermediate and comparable in wooded heathland ( $13.8 \pm 1.3$ ) and savanna ( $14.5 \pm 0.6 \text{ kg cm}^{-2}$ ). Surface strength in grasslands averaged  $15.1 \pm 0.5 \text{ kg cm}^{-2}$  (Figure 5.9 A).

Repeated measure ANOVA indicated significant differences in surface strength between years ( $p < 0.001$ ), communities ( $p < 0.01$ ), treatments ( $p < 0.001$ ), and for their interactions ( $p < 0.05$ ). Results from the analysis for the separate years, are summarized in Table 5.6. Fewer interaction terms became significant in the later years and in 2003 only the trampled  $\times$  community interaction was significant. Surface strength declined with time in the trampled and clipped + trampled treatments compared to the control and clipped treatments (Figure 5.9). Exception was the clipped treatment in the woodlands, which behaved similarly to the trampled treatments. There was a positive and significant correlation between surface strength in spring 03 and soil surface height from f02 to s03 (Spearman's  $\rho = 0.501$ ,  $n = 136$ ,  $p < 0.001$ ), but a negative correlation

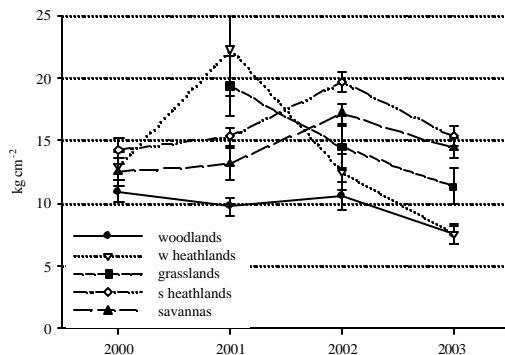
TABLE 5.6

**Results from ANOVAs on surface strength in four communities (grassland not included) and four treatments in Hafnarskogur. Analyses performed on mean penetrometer measurements from each subplot (n = 94 for 2000; n = 140 for 2001; n = 144 for 2002; n = 139 for 2003). Communities referred to as Comm, treatments effect summarized as clipped = Cl and trampled = Tr.**

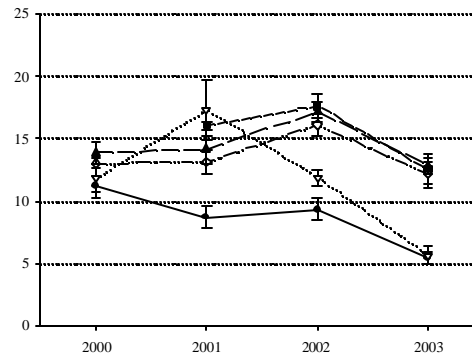
	Comm	Cl	Tr	Cl×Tr	Cl×Comm	Tr×Comm	Comm×Cl×Tr
Year	p	p	p	p	p	p	p
2000	0.040	0.031	0.069	0.003	0.231	0.001	0.495
2001	0.122	0.004	0.001	0.007	0.004	0.022	0.018
2002	0.002	0.393	0.001	0.354	0.073	0.001	0.520
2003	0.001	0.006	0.001	0.369	0.589	0.001	0.679



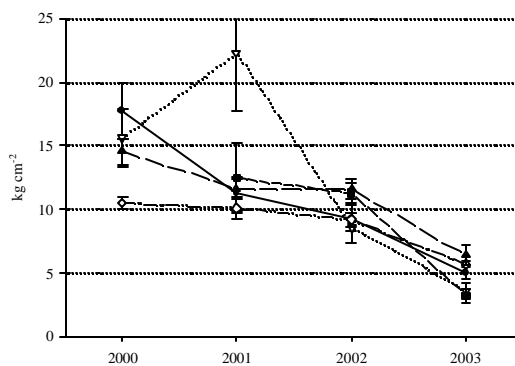
A: control



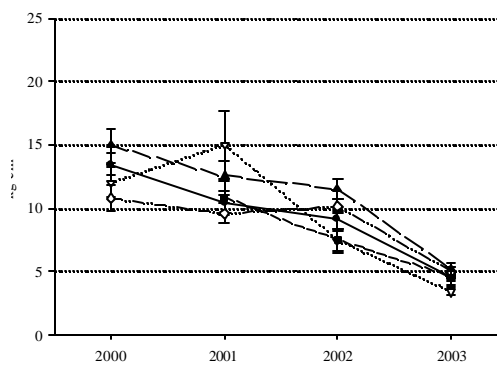
B: clipped



C: trampled



D: clipped &amp; trampled



**FIGURE 5.9.** Temporal changes in mean ( $\pm$  SE) surface strength in five plant communities at Hafnarskogur as affected by disturbance treatments ( $n = 94$  for 2000;  $n = 140$  for 2001;  $n = 144$  for 2002 and  $n = 139$  for 2003). Note: w heathlands = woodland heathlands, s heathland = savanna heathlands.

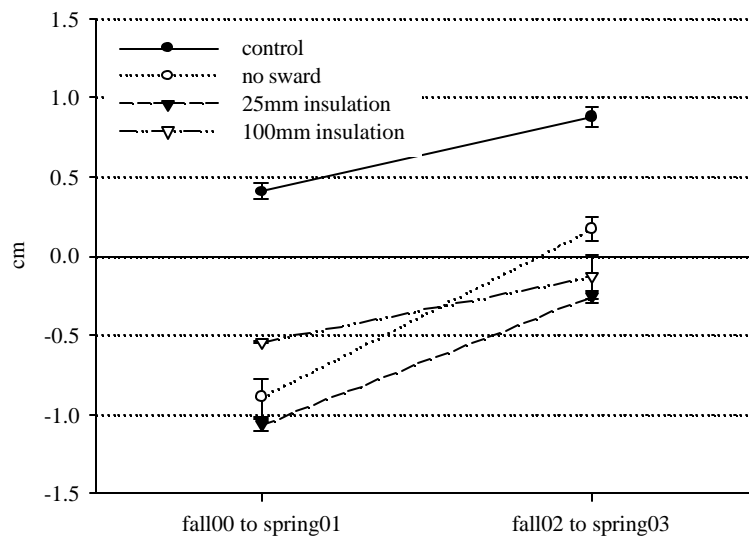
with mean surface movement from fall 02 to spring 03 (Spearman's  $\rho = -0.608$ ,  $n = 136$ ,  $p < 0.001$ ).

### **Sward insulation**

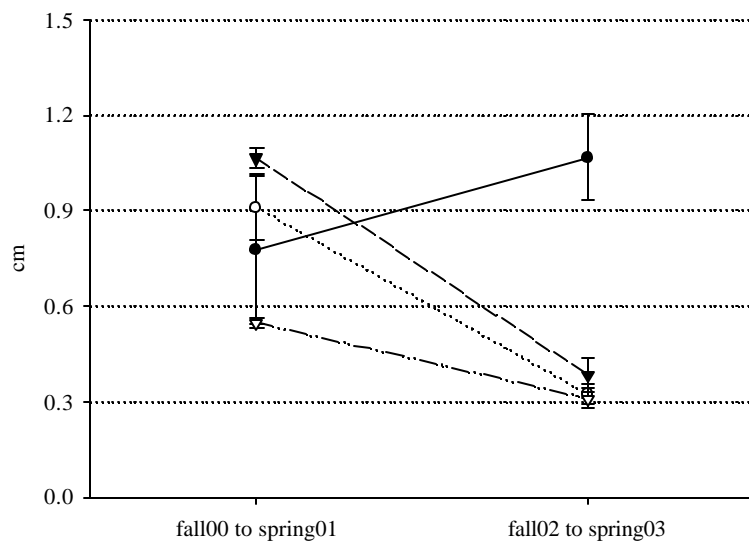
The surface rose in control plots in both winters and, with the exception of a slight rise in the no sward treatment in the latter winter, surface lowered in response to treatments (Figure 5.10). Changes in surface height differed significantly between treatments in both years (Kruskal-Wallis:  $p < 0.05$ ), but no pairwise comparisons were significant. Absolute movement of the ground surface was statistically comparable among treatments in both winters. Soil frost depths were greatest in the no sward treatment but the least in the 100 mm insulation treatment (Table 5.7). Mean daily soil temperatures from November 7<sup>th</sup> 2000 to Mars 5<sup>th</sup> 2001 were lowest in the no sward treatment, and highest in the 100 mm insulation treatment.

Sward thickness was correlated with maximum soil frost depth (Spearman's  $\rho = -0.907$ ,  $n = 12$ ,  $p < 0.001$ ; Figure 5.11 A), and mean soil temperature (Spearman's  $\rho = 0.766$ ,  $n = 12$ ,  $p < 0.01$ ). Mean daily soil temperature and maximum frost depth were also correlated (Spearman's  $\rho = -0.800$ ,  $n = 11$ ,  $p < 0.01$ ; Figure 5.11 B; soil probe in one 100 mm insulation plot malfunctioned). The correlation between mean movement of surface and maximum frost depth was positive and significant (Spearman's  $\rho = 0.595$ ,  $n = 12$ ,  $p < 0.05$ ); and sward thickness was negatively correlated with surface movement (Spearman's  $\rho = -0.501$ ,  $n = 12$ ,  $p = 0.097$ ).

A: Soil surface height



B: Absolute movements

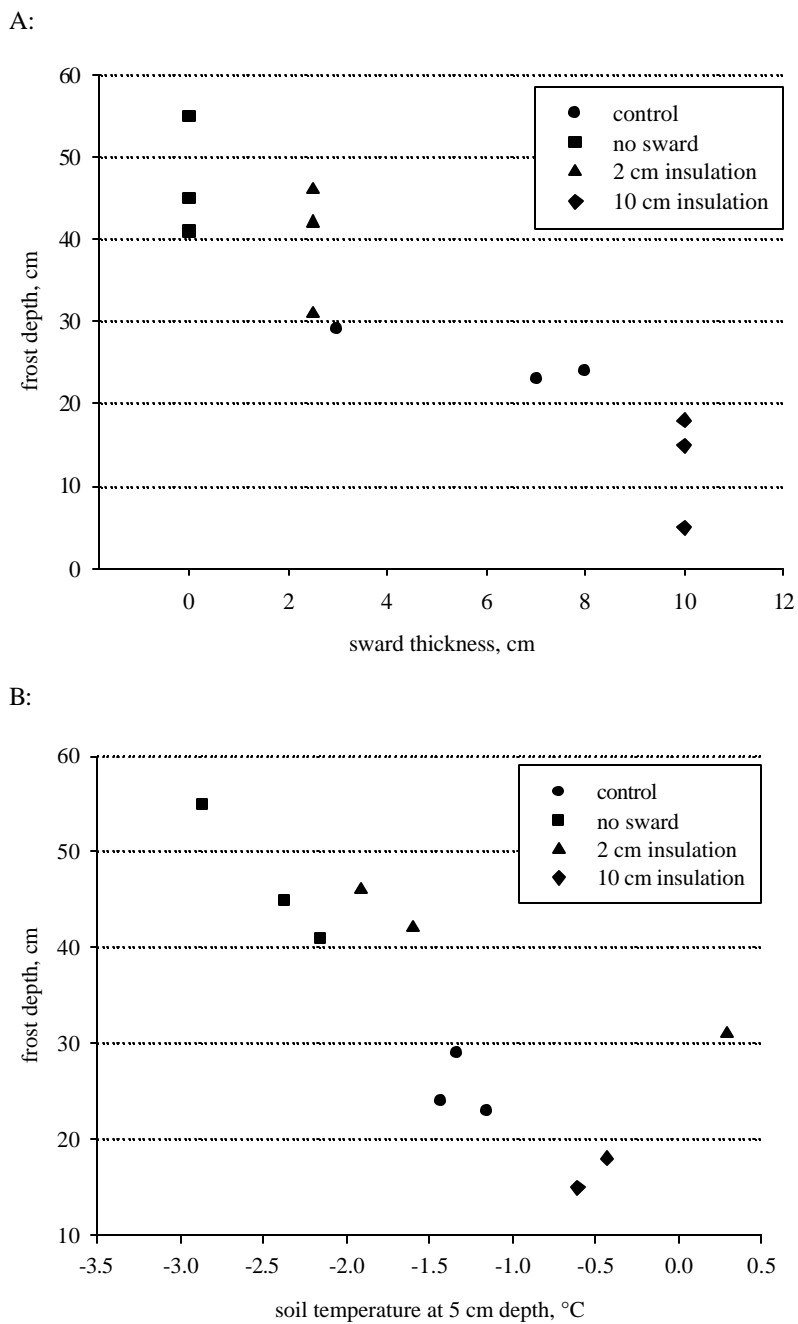


**FIGURE 5.10.** Changes in (A) soil surface height and (B) mean absolute surface movement for four insulation treatments in the heathland community at Keldnaholt over two winters. Data are (A) mean SEB pin height differences and (B) mean absolute pin height differences between fall and spring measurements;  $n = 12$  for both winters.

**TABLE 5.7**

**Mean ( $\pm$  SE; n= 3) sward thickness measured in spring 2001, maximum soil frost depth during the 2000 - 2001 winter, and daily soil temperature at 5 cm depth from November 7<sup>th</sup> 2000 to Mars 5<sup>th</sup> 2001, in a heathland community at Keldnaholt.**

Year	Sward thickness, cm			Max soil frost depth, cm			Soil temperature, °C		
	Mean $\pm$ SE			Mean $\pm$ SE			Mean $\pm$ SE		
control	6.0 $\pm$ 1.5			25.3 $\pm$ 1.9			-1.3 $\pm$ 0.1		
no sward	0.0			46.7 $\pm$ 4.3			-2.5 $\pm$ 0.2		
25 mm insulation	2.5			39.3 $\pm$ 4.3			-1.1 $\pm$ 0.7		
100 mm insulation	10.0			12.2 $\pm$ 3.9			-0.5 $\pm$ 0.1		



**FIGURE 5.11. Relationship between (A) the maximum frost depth and sward thickness and (B) the maximum frost depth and soil temperature at 5 cm depth (B).**

## Discussion

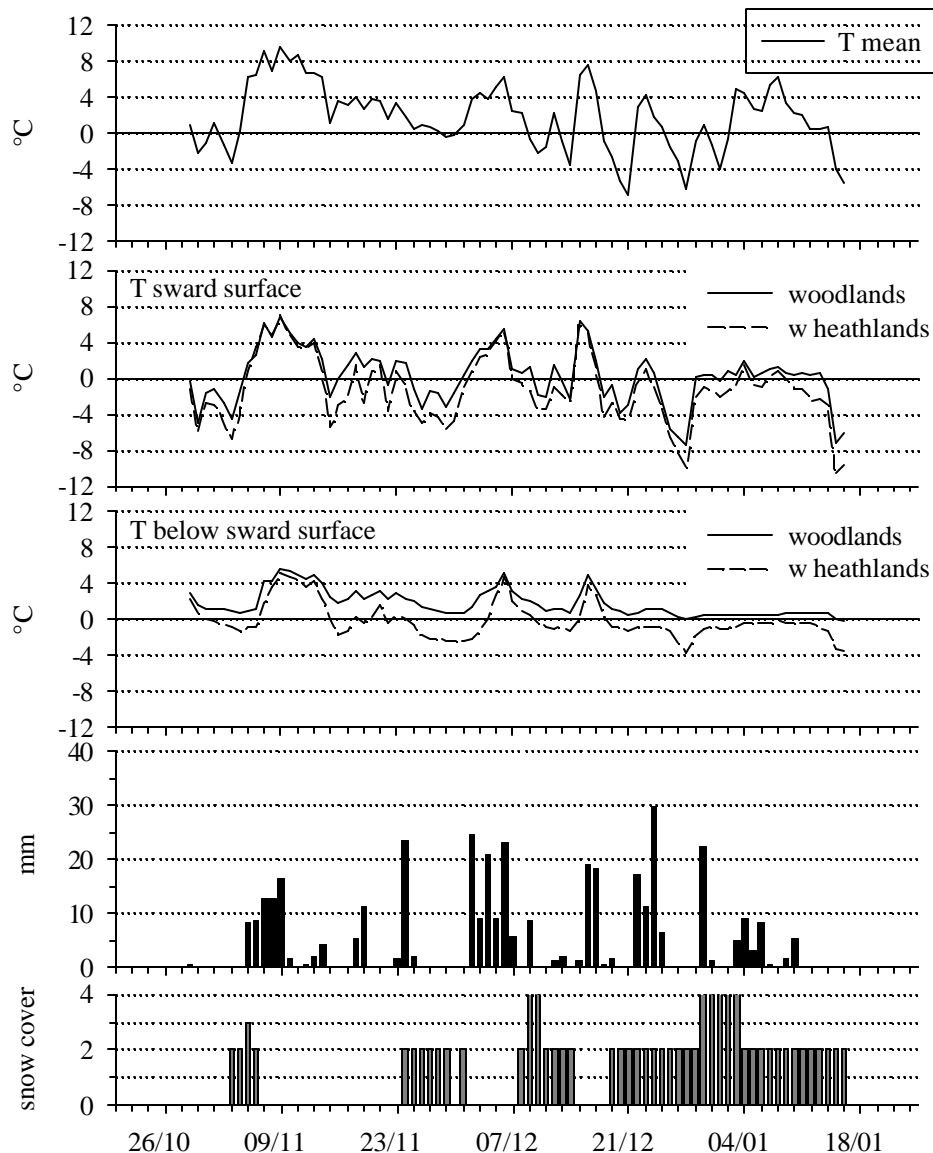
This experiment was conducted to test five hypotheses, all related to soil surface stability – or movement - when exposed to freeze-thaw cycles, i.e. cryoturbic disturbances:

1. Frequency and intensity of frost heaving would increase from dense woodlands to open communities;
2. Woodland mineral soil surfaces are more stable than mineral soil surfaces in open plant communities;
3. Mineral soil surface stability in different communities would respond differently to simulated grazing, due to differences in microclimate;
4. Grazing would reduce stability the most in open grasslands and heathlands, but less in woodlands; and
5. Mineral soil strength would be correlated with grazing, such that under grazing regimes surface strength would be low but higher when absent.

Wooden pegs were used to compare frost heaving between communities. Fewer pegs heaved in the grasslands than in the other communities (Figure 5.4 A) and the magnitude of heaving was also low, but comparable to the woodlands (Figure 5.4 B). The savanna heathland and savanna both had the highest number of heaved pegs and the pegs there also heaved the most. The lack of correlation between sward thickness and surface strength versus peg heaving indicates that other factors influence the heaving. The thicker sward had more moss and the thinner sward more vascular plants. The mosses provide greater insulation (Gornall et al., 2007), but the vascular plants provide

root cohesion (Abernethy and Rutherford, 2001). Sward with both these components may thus provide the best protection to heave. Other parameters than just the sward thickness may thus explain heaving better.

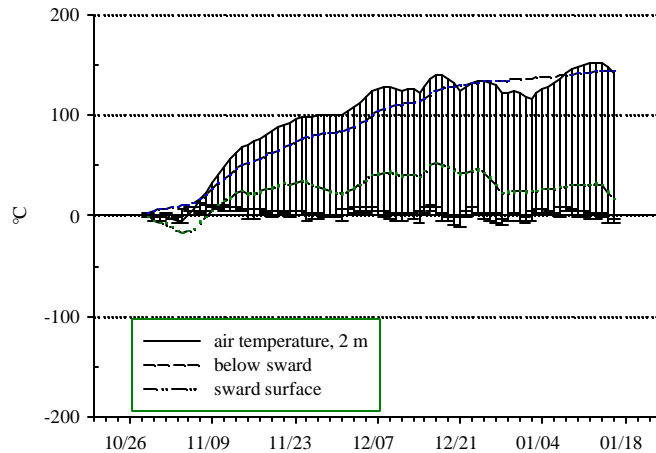
The insulation properties of the woodland and w heathland communities and ground layers were explored with temperature measurements on the sward surface and below the sward (on the mineral soil surface), from October 29, 2003 through January 16, 2004. The sward layer is defined here as all mineral and organic (both living and dead) materials extending from the land surface to the mineral soil surface. This would include portions of mosses, grasses, herbs and litter. Mean daily air temperatures were mostly above freezing until December the 9, at which time daily air temperatures were characterized by oscillations between +6.0 °C and -6.0 °C (Figure 2.6). Temperatures on the sward surface tracked air temperature, but were on average 1.6 °C and 3.1 °C lower in the woodland and w heathland, respectively (Figures 5.12 and 5.13). Temperature differences between woodland and w heathland may reflect reduced wind speed, reduced outgoing long wave radiation at night, and greater snow accumulation in the woodlands (Figure 5.12). The 1.5 cm thicker woodland sward also insulated the mineral soil surface better than the thinner w heathland sward (sward thickness = 5.3 cm and 3.8 cm respectively) (Figure 5.12). The greater insulation of the woodland community was clearly reflected in the number of freeze-thaw cycles (defined as a decline in temperature below 0 °C lasting  $\geq 24$  hr, followed by an increase above 0 °C) observed below the sward, which were five in the w heathland, but none in the woodlands (Figure 5.5).



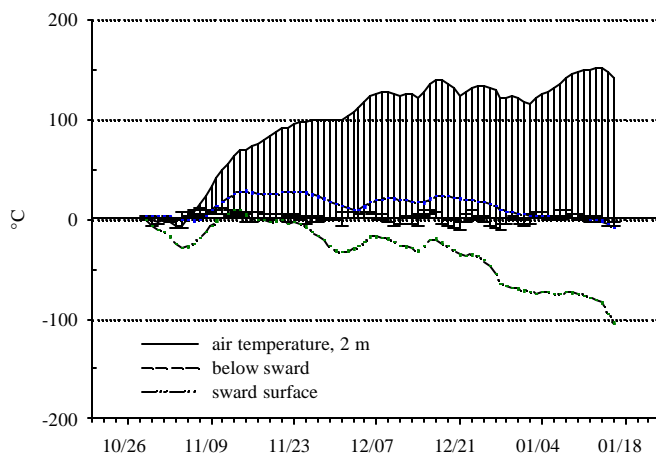
**FIGURE 5.12.** Daily air temperature (2 m), sward surface and below sward surface temperatures in the Hafnarskogur woodlands and woodland heathlands, and daily precipitation and snow cover, from October 29<sup>th</sup> 2003 to January 16<sup>th</sup> 2004. Weather data are from the Icelandic Meteorological Office (IMO): air temperature from Hafnarmelar, 5 km SSW of site; precipitation and snow cover from Andakilsarvirkjun, 12 km ENE of Hafnarskogur. The daily air temperature was obtained from Hafnarmelar IMO weather station, but above and below sward temperatures were collected at two adjacent locations in each community with WatchDog data loggers (Spectrum Technologies, Inc; n = 1 per depth).



A: woodlands



B: woodland heathlands



**FIGURE 5.13.** Cumulative daily air temperature, and sward surface and below sward surface temperatures in the Hafnarskogur (A) woodlands and (B) w heathlands, measured from October 29<sup>th</sup> 2003 to January 16<sup>th</sup> 2004. The woodlands sward between the loggers was 80 % moss and 20 % grass cover, of 5.3 cm mean thickness. The wheathlands sward between the loggers was 85 % moss, 12.5 % grass and dwarf shrub and 2.5 % shrubby birch cover, of 3.8 cm mean thickness. The daily air temperature was obtained from Hafnarmelar IMO weather station, but above and below sward temperatures were collected at two adjacent locations in each community with WatchDog data loggers (Spectrum Technologies, Inc; n = 1 per depth).

Surface levels generally rose for controls in winter (Figure 5.7 A), but differed between years probably reflecting different climatic conditions. The f02 to s03 with little variability, however, stands out as there is both a considerably less variability in the mean height values and the absolute movement, possibly because of the favorable winter of 2002 – 2003 when mean monthly temperature never fell below 0°C (Figure 2.3). Corresponding values for the summer tended to be negative, hence reflecting subsidence in summer (Figure 5.8A) after the winter heaving. These results demonstrate that surface movements are the norm in these communities which experience many freeze-thaw cycles each winter. Lack of comparable research in similar environments halt comparisons with other studies, however others have observed lowering of surfaces after disturbances by ploughing and wildfire (up to 2.7 cm yr<sup>-1</sup> and 1.8 cm yr<sup>-1</sup> respectively) in Atlantic-Mediterranean climate (Shakesby et al., 2002).

With regard to surface movement during winter (Figure 5.7), it stands out that the clipped treatment and the control on one hand, and clipped and trampled and trampled on the other hand, form two distinct groups, which each shows similar response over time, possible due to the insulation properties of the sward (Figure 5.6). This is hardly a surprise; the trampling is such an intense disturbance that an additional herbaceous layer clipping adds little to it. The differences between these two groups can further be seen in Figure 5.8, as both the trampled treatments do frequently display greater summer subsidence than the clipped treatment or control.

The trampled and the clipped and trampled treatments show a declining trend for mean surface height with time. It is not unreasonable to conclude that it reflects the

intense treatment effect. The trampling stirs up the surface, roots are torn and eventually detached from the plants, vegetation is buried and over time the plots became homogenized. Such impact damages soil structure and is expected to increase bulk density, and hence impair water permeability. The almost total absence of vascular plants in the trampled plots (Figure 5.9) also suggests that the soil structural support, which roots would provide, is absent. Root density was not estimated in the treatment plots, but surface strength can be taken as an indirect indicator of root density as well as the cover. Figure 5.11C&D shows clearly how the surface strength is reduced over time for the trampled treatments. Neither the clipped treatments, nor the controls do show a similar reduction trend. The observed lowering of the plot surfaces may thus be because of soil compaction caused by deterioration of soil structure, rain-splash erosion or even wind. Cryoturbic processes also loosen up the soil and make the surface more vulnerable to detachment. However, in these small plots wind and water erosion are expected to be less intense than on a larger scale. The small scale in these experiments reflects processes similar as expected in small erosion spots, when the mineral soil surface is exposed to the elements. The surface lowering may also be related to carbon loss as  $\text{CO}_2$ . It was noted that the soil in the trampled plots became more pliable with time, suggesting that root and plant fragments were decaying. Figure 5.10 shows SOC in each treatment within plant community at the end of the experiment. The SOC was highest in the control and clipped, but significantly lower in the clipped and trampled treatments, reflecting erosion of organic matter, which is a sign of soil nutrient decline.

Others have documented decline of SOC as a result of grazing (e.g. Podwojewski et al., 2002; Zhao et al., 2007).

The absolute movements of the trampled and clipped and trampled treatments are of special interest, as they show a quite different pattern than the control and clipped treatments (Figure 5.7 B, D, F & H), and reflect a high degree of surface movement, especially in the last winter after repeated treatment application. Figure 5.12 (T below sward surface) shows the mineral soil surface temperature, below an intact sward, in the fall and early winter of 2002. Five freeze-thaw cycles can be identified from November through January 2003, so albeit the winter of 2002 - 2003 was mild and mean temperature above 0°C, temperatures did fluctuate around the freezing point even below intact sward. Intense freeze-thaw actions in the unprotected treatment plots are therefore evident despite the warm winter. Thus, even in mild winters, freeze-thaw action cause extreme surface heaving and decline if the vegetative cover is impaired, resulting in observed high surface instability. This cannot be seen for the clipped treatment or the control, suggesting that the sward cover reduced the number of freeze-thaw cycles, or reduced their intensity.

When the treatments are compared within the communities, an interesting difference can be seen (Figure 5.9). There is little or no clear separation between treatments until the last year, except for the woodland where this occurs during the first winter. From that point and onward the clipped treatment and control, and trampled and clipped and trampled follow similar separate trajectories. The woodland community differed from the other communities by having the highest moss and lichen cover (52 %)

and the lowest grass and herbaceous cover (33 %), and by having the lowest surface strength (Figure 5.11 A). This means that there is little root reinforcement to be expected in the topsoil, and the moss cover will be vulnerable towards any trampling due to its growth form. The untreated woodland also had the highest SOC (at 0 – 5 cm depth), which is known to correlate with good soil structure (Brady and Weil, 1998) and high soil infiltration rates, as has been shown for the Hafnarskogur woodlands (Orradottir et al., 2008). Despite the high SOC content and good hydrological properties, the woodlands seem to be more sensitive to grazing disturbances than the other communities due to the low surface strength and high moss and lichen cover. This is reflected by the fact that there was a clear treatment response in the woodland two years before it is observed in any of the other communities. The woodlands do thus appear to have low resilience to trampling, and the herbaceous layer may have low resistance if the forest is removed.

The fact that the woodlands, with low root structural support and having the lowest surface strength, did not show greater mean change in the height of the mineral surface (Figure 5.7 A) opens up the question why more surface movements were not observed. The answer may partially be found in the thermal barriers provided by the woody canopy and the sward layer. Figure 5.13 compares the cumulative temperature on and below the sward in the woodlands (panel A) and the adjacent w heathland (panel B). The woodland community had higher cumulative temperature on top of the sward than the open w heathland reflecting the ameliorating effect of the trees on the microclimate; hence there should be fewer freeze-thaw cycles in the woodland than in

the other communities. The woodland does thus appear to provide a stable environment, but have low resilience to disturbance. This can be supported by an example from northern Finland, where a rapid disappearance of birch woodland was initiated by caterpillar outbreak and followed by reindeer browsing, which resulted in the replacement of the woodland by heath vegetation (Chapin III et al., 2004).

The effect of the sward thermal properties was tested in the simulated sward insulation experiment (Figure 5.12, Table 5.7) where soil temperature data were collected in the SEB plots. SEB control data showed the same heaving trend as was observed in Hafnarskogur w heathland, and there was a significant positive correlation between mean absolute surface movements and frost depth, and a negative correlation between mean absolute surface movements and sward thickness. This shows that the sward has a direct effect on the soil freezing process, the thicker the sward, the less is the frost intensity (Figure 5.13A) which results in more stable surfaces.

The woodlands with their double thermal barriers, the tree canopy, and the sward layer, will thus reduce frost action compared to open areas, and thus provide greater surface stability during the winter, and this may explain lower frost heaving in the woodlands.

The results of this experiment show that the woodlands may provide a more stable environment than open community types with regards to peg heaving. Peg heaving is less in the woodlands, but many pegs do heave, hence the disturbance is there, but the intensity appears to be low. The surface movements, as measured with SEB however, are not conclusive in this respect. The communities do not respond differently

to the selected grazing treatments, with the exception of more rapid changes in the woodlands. Surface stability was not reduced more in the open community types than in the woodlands. However, the simulated grazing resulted in reduced surface strength as originally hypothesized.

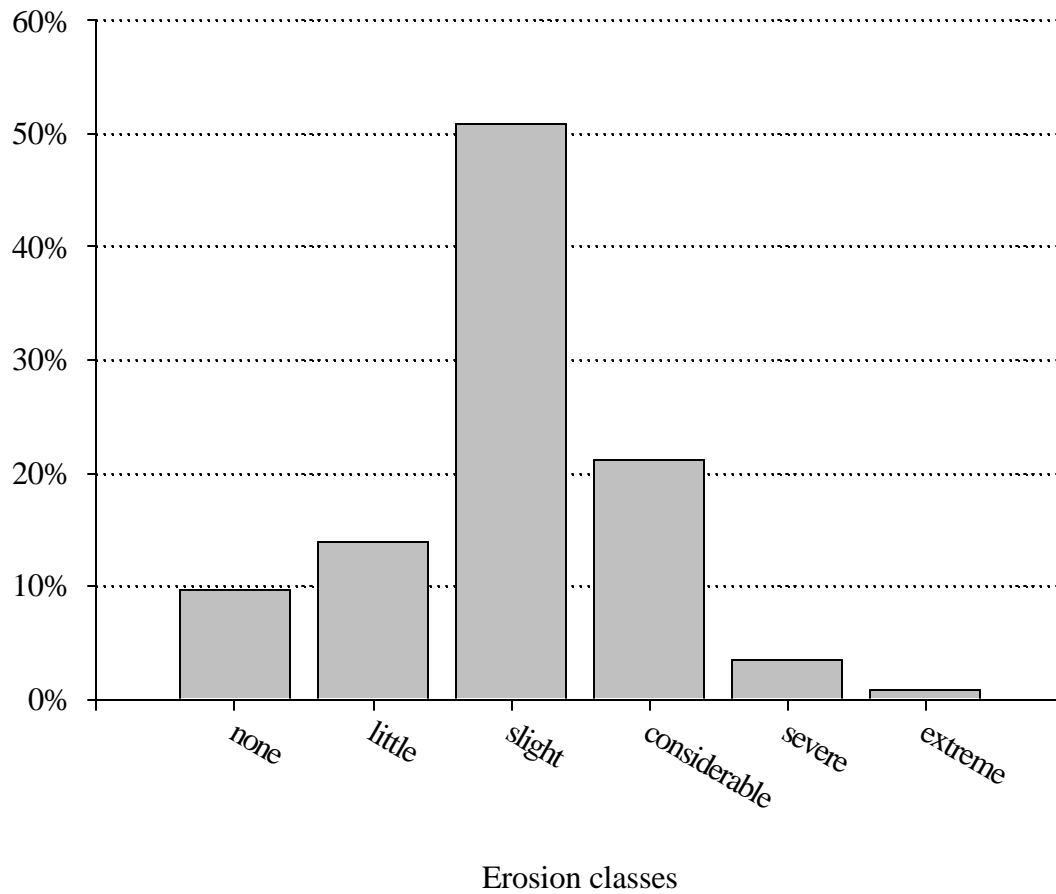
CHAPTER VI  
HAFNARSKOGUR, WEST ICELAND – A CASE STUDY OF LANDSCAPE  
FRAGMENTATION OVER TIME

**Introduction**

It has been stated that few areas of the high-latitude regions of the Northern Hemisphere have experienced levels of ecosystem degradation as severe as Iceland (Arnalds, 2000). Almost 80% of the areas categorized as “well vegetated”, “vegetated” or “sparsely vegetated” are classified as having slight to extremely severe erosion (Figure 6.1) (Arnalds et al., 2001). The erodable volcanic soils, cold unstable maritime climate and periodic volcanic eruptions make the ecosystems particularly vulnerable to disturbance and a challenge to manage. Despite widespread land degradation and erosion, relatively little is known about the underlying processes.

It has long been assumed that the current land degradation episode was triggered as man arrived and traditional contemporary farming practices were introduced (e.g. Thorarinsson, 1974; Einarsson, 1995). No large grazers were present prior to the arrival of man. Farming required open land for grazing and haymaking, so the existing woodlands were cleared around homesteads. The open landscapes were maintained through yearlong grazing and even grass-litter burning (Fridriksson, 1978). The remaining woodlands were used for fuel harvesting and grazing (Thorsteinsson and Olafsson, 1967).

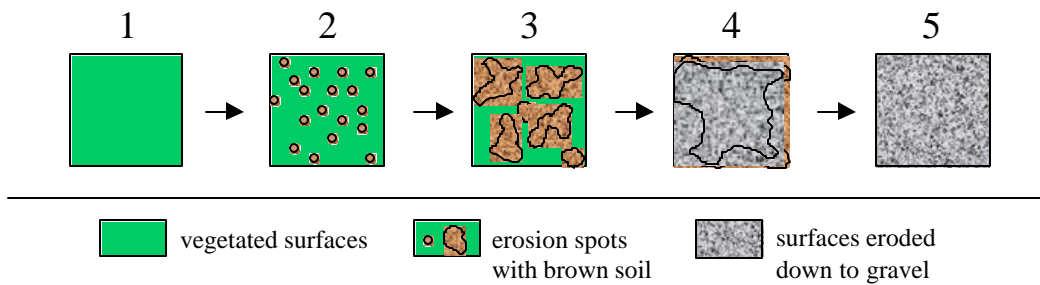




**FIGURE 6.1.** Combined erosion classes for vegetated areas classified as "well vegetated", "vegetated" and "sparsely vegetated", excluding high mountains, glaciers and rivers/lakes. Based on Arnalds et al. (2001).

This apparent transition from woody regimes to open plant community types coinciding with the introduction of contemporary Norse farming traditions (Hallsdottir, 1987). Some of the earliest noticeable signs of the degradation process are the appearances of small bare soil spots (Soil Erosion Spots; SES) in the vegetation cover, where the mineral soil surface is exposed (Arnalds, 1990; Aradottir et al., 1992; Arnalds et al., 2001). Such SES can occur at various spatial scales, ranging from less than 1 m<sup>2</sup> to patches as large as hundreds or even thousands of square meters. They can be seen as a spatial hierarchy (Allen and Starr, 1982) with the finest spatial unit being a single SES, caused by grazing or harsh weather (see Chapter III and Figure 6.2). The next hierarchical unit is a group of small SES occurring together in a relatively small area. The third spatial unit can be defined as coalesced SES. These are larger and may represent a shift in domain of scales (Wiens, 1989), as surface area and perimeter length per SES increases, and therefore more bare are is exposed. The potential for increased erosion rates are thus higher as SES size increases. They should also be more irregular in shape than spots occurring at lower hierarchical levels, as a result of unevenly distributed SES merging. The fourth hierarchical level represents collection of large irregular SES. This hierarchy is acknowledged in the conceptual model describing the degradation process in Iceland, originally published by Aradottir et al. (1992) (refer to Figure 3.1).

Spatial structures affect ecological processes (Turner, 1989; Gustafson, 1998; Turner et al., 2001). Such hierarchy of landscape features related to the degradation process suggests that it might be possible to identify landscapes at risk by applying



**FIGURE 6.2.** Schematic overview over the soil erosion spot (SES) expansion and coalescence process. **State 1:** pristine land with continuous vegetation cover. **State 2:** small regular SES in the otherwise dominating vegetation cover. **State 3:** early SES expansion and coalescence state. **State 4:** late expansion and coalescence state. **State 5:** barren denuded surfaces.

States 2-4 can be viewed as a spatial hierarchy (Allen and Starr, 1982) where the smallest unit is a single SES, followed by a group of SES (state 2). State 3 represents the third and fourth level represented by coalesced SES and group of such SES. State 4 represents the fifth hierarchical level, which occurs when the erosion features have merged so they dominate large parts of the landscape (i.e. 'coalescence of coalesced' SES). Level five also indicates that a shift has occurred in the degradation process from biotic to abiotic process domain.

methods suitable for detecting and quantifying landscape features if we know what kind of patterns to seek. The importance and usefulness of using land geometric change to monitor and evaluate landscape changes over time are becoming increasingly clear and feasible as availability of suitable data increases. In 1998 the Organization of Economic Co-operation and Development (OECD) suggested that monitoring trends in land cover was viable to evaluate changes in land use, in addition to traditional indicators that have been used or suggested to monitor rangelands (Committee on Rangeland Classification, 1994).

In this study I applied selected landscape metrics to landscapes considered to be at different degradation stages, and thus test the feasibility of using remote sensing for monitoring and categorizing landscapes. I evaluated the suitability of 12 landscape metrics in classification of an Icelandic landscape over 51-year period. The objective was to test if simple landscape metrics obtained with unsupervised classification could be used to classify and categorize landscapes, which may be at risk for entering an accelerated erosion phase.

## **Methods**

### **Site description**

The study site is in Hafnarskógur, a 10 km long and 1-1.5 km wide area between Mt. Hafnarfjall and Borgarfjörður fjord (64°30'N, 21°38'W) (see Figure 2.1). Elevation ranges from 2 m in the south to 80 m in the north. The topography is mostly flat, but towards north the terrain slopes into the ocean (NW aspect, ~3-5°). The soils are Typic

Fulvicryands (woodlands) and Histic Cryaquands (grasslands) (Orradottir, 2002) and  $\geq 1$  m thick.

Hafnarskogur belongs to Hofn, a farm settled between 874 – 930 A.D. and is among the oldest farmsteads in the country (Thorgilsson, 1968). The Hofn landscapes are comprised of diverse plant community and surface types, ranging from birch woodlands, grasslands, heathlands and wetlands, to areas eroded down to the gravelly substrate. It is generally assumed that Hafnarskogur, as in many other Icelandic lowland areas, was dominated by birch woodlands (*Betula pubescens* Ehrh.) at the time of settlement. The assumption that the lowlands were dominated by birch woodlands is supported by pollen analyses (Einarsson, 1962; Hallsdottir, 1987), historical records and woodland remnants (e.g. Thorgilsson, 1122-32; Bjarnason, 1942; Gudbergsson, 1996), land descriptions dating from the 16<sup>th</sup> century (N.N., 1949), and old place names (Helgason, 1950; Gislason, 1975). Today, noteworthy woodlands remain only in small areas of the traditional Hafnarskogur area, and SW of Hafnarfjall mountain. Large areas are eroded, especially on the dry ridges between wetlands in the southernmost part. In many ways, the Hafnarskogur area as it is today represents Iceland, but on a smaller scale. Not only are the plant communities representative, with woodlands, grasslands, heathlands and wetlands, but the erosion features are comparable to what is found in many Icelandic landscapes. The Hafnarskogur area is ideal for research dealing with land changes believed to have occurred since the arrival of man. Refer to Chapter II and Appendix A for a more detailed description of the area.

## Maps

Black and white aerial photos and RGB color aerial photo imagery were obtained for the area from the National Land Survey and Loftmyndir ehf. respectively (Table 6.1). The black and white images were obtained as 10 × 10 in. negatives. The color imagery were delivered as georeferenced jpg files with a scale of 1:2,000 and 0.5 m pixel resolution.

### *Black and white image data processing*

The black and white images were scanned at 0.5 m resolution (1600 dpi) using an Epson Expression 836XL (Seiko Epson Corp.) and saved as TIFF files for further processing. This resolution was considered necessary due to the relatively small nature of some of the erosion features we sought to quantify in this study (Turner et al., 1989). The resulting files were then imported into ArcView 3.2a (ESRI, 2000a) and georeferenced to the color images using the ImageWarp extension (McVay, 1999), followed by conversion to grid format using the Image Analysis extension (ESRI, 2002). A minimum of 250 points was used for each image.

In order to reduce classification errors and data anomalies, a low-pass filter (3 × 3 nearest neighbor grid) was applied to the grid data using ArcView's Spatial Analysis extension (ESRI, 2000b) prior to classification. The resulting grids were classified using an unsupervised classification to categorize the resulting grids into 40 spectral classes (Lillesand and Kiefer, 2000). The classes were then categorized as either vegetated or eroded. Further classification was not possible as it was impossible to distinguish between different plant communities from the black and white images.

**TABLE 6.1****Imagery properties and total precipitation five days prior to image date.**

Year	Date	Type	Original resolution	Precipitation mm <sup>1</sup>		Notes
1946	August 26 <sup>th</sup>	B&W	~1:35,000	-	-	Multiple small speckles on film
1960	July 8 <sup>th</sup>	B&W	~1:35,000	4.5	-	
1977	August 7 <sup>th</sup>	B&W	~1:35,000	1.9	1.4	
1989	August 25 <sup>th</sup>	B&W	~1:35,000	3.5	6.3	
1997	August 11 <sup>th</sup>	B&W	~1:35,000	23.4	98.2	
1999	August 17 <sup>th</sup>	RGB	~1:2,000	8.3	15.4	

<sup>1</sup>Total precipitation five days prior to image. Left column: Reykjavik, 40 km south of Hafnarskogur, right column: Andarkill 10 km west of Hafnarskogur. Dash indicates no data.

Categorization into vegetated and eroded classes was fairly straightforward in most cases. Many of the eroded areas have much higher reflectance than the darker vegetation due to accumulation of light colored frost heaved stones or gravel on the bare surfaces. However, surfaces with exposed brown soils, such as recently eroded areas and the SES perimeters, are difficult to distinguish from vegetated areas on black and white imagery, as the higher soil moisture in these recently exposed soils will appear very similar to the dark green vegetation. This may cause a systematic overestimation of vegetated areas in the classification and must be taken into account when the results are interpreted. Images obtained after a recent rainfall pose a specific problem, as they cause the exposed soil surfaces to darken even further and adds to this overestimation problem of the vegetated surfaces. This is especially noticeable for the 1997 data (Table 6.1).

#### *Color image data processing*

The color images were imported into ArcView 3.2a (ESRI, 2000a) and categorized into 256 classes using unsupervised classification using all three color bands [Image Analysis extension (ESRI, 2002)]. The categorize command uses 'Iterative Self-Organizing Data Analysis Technique' (ISODATA) clustering. Unsupervised classification has advantages over supervised classification. The algorithm defines and groups distinct spectral classes present in the image data, some which may not be visible otherwise, and might thus be overlooked in supervised classification (Lillesand and Kiefer, 2000). Unsupervised classification on the other hand assumes that spectral



classes represent true land cover types, which may not always be true, as different vegetation may have similar spectral classes.

The resulting clusters were then matched with the original color imagery and the black and white imagery from 1997, and subsequently assigned as eroded or vegetated classes. This method proved good in most cases. However, in some cases the brownish eroded surfaces and the reddish wetland vegetation turned out to be indistinguishable, as the sedges and rushes tend to have red-brownish hue, which is similar to the spectrum representing some of the eroded areas. It was possible to circumvent this problem by excluding the problematic areas from the classification. Because erosion is absent in the wetland plant communities, it was decided to combine them with the grasslands in the subsequent analysis. They are referred commonly to as the “grasslands” plant community type.

Trees and woodlands were not easily distinguishable with this method. To improve tree classification, the Image Analysis extension was used to transform the color image dataset using ‘Histogram Equalize Stretching’. This algorithm reassigns and stretches the current range of pixel values over a range of 256 values, but takes into account their frequency of occurrence (Lillesand and Kiefer, 2000). The woodland areas were then delineated and selected woodland areas used as ground truth data or training areas for a supervised ‘Find Like Areas’ command in the Image Analysis Extension. The procedure uses a parallelepiped maximum likelihood classifier to find and group areas similar to the ones defined as the decision region in the training process (Lillesand and Kiefer, 2000; Campbell, 2002).

Supervised classification is accurate, as the patch type to be classified is known to the user. The main problem, however, is that unlike unsupervised classification, some of the spectral range present in the data, may not be present in the training data, thus causing an underestimation of this category. In this dataset this was not considered to be a problem, as the current location and extent of woodlands was known after spending five summers in the area, and thus easy to delineate which minimizes errors due to area exclusion. In general, supervised classification methods have been found to produce better maps than unsupervised classification, given that good training data or good knowledge of the area are available (Schowengerdt, 1997). In this case it helped that the area is small, making it possible to scan it afterwards to find overlooked tree patches, but some of the smallest patches were undoubtedly overlooked causing this class to be underestimated. The supervised classification was repeated and the results compared to the color and black and white images, until satisfactory results had been achieved.

The third class of interest was obtained by subtracting the other two classes, eroded areas and woodlands from the total area. This class was defined as “grassland”. These three classes were then used to define plant community types on the black and white imagery.

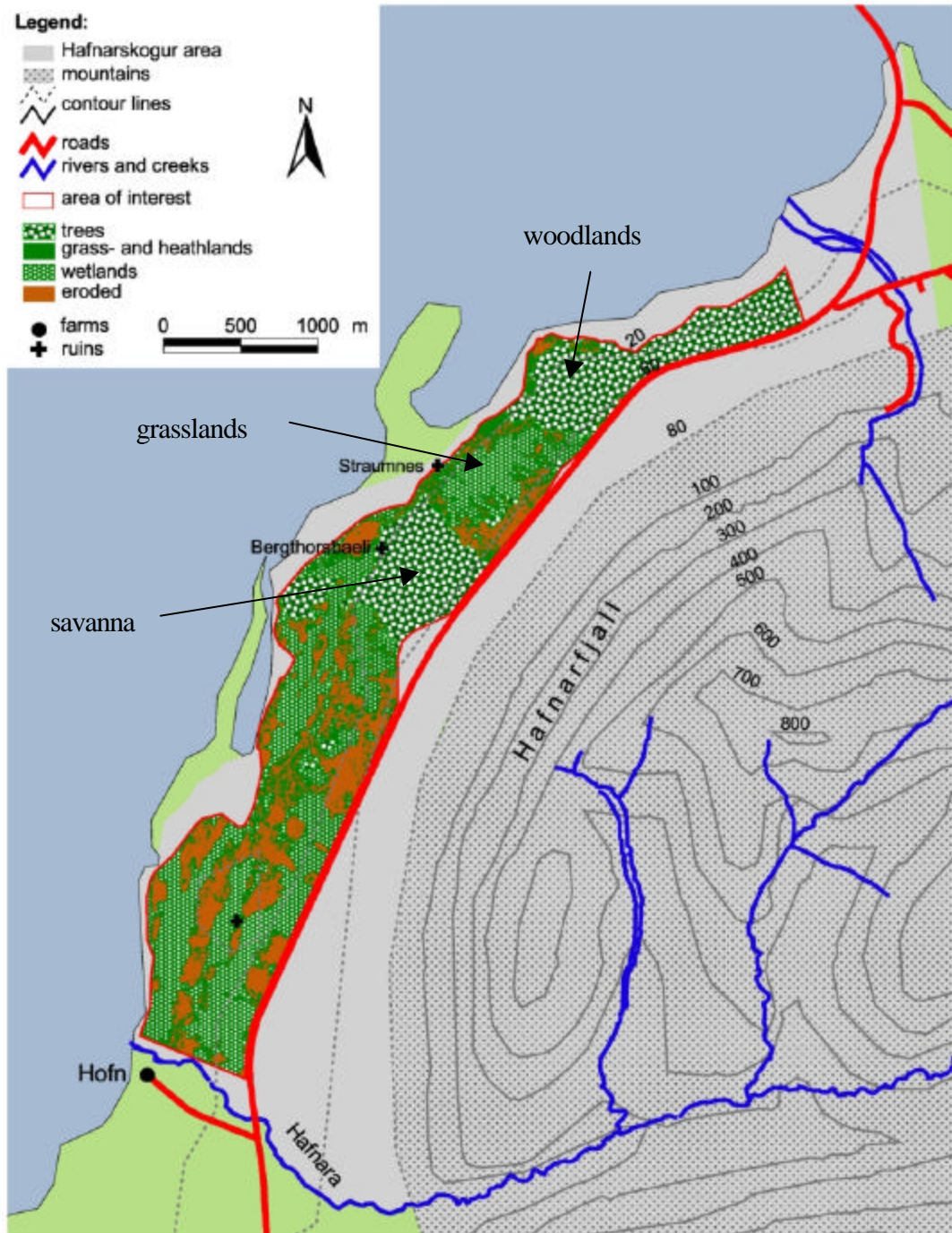
The classification of the color images was evaluated by selecting 73 points from the categorized dataset. The coordinates for the points were then uploaded into a GPS unit (Garmin GPS III+; Garmin Ltd.), and each point visited in summer of 2005 to obtain ground truth data. The accuracy for grasslands, woodlands and eroded areas was 80.8%, but 98.6% when vegetated and eroded surfaces were only considered.

### *Plant community type masking*

The surface type data created from the color imagery was applied to the black and white images for all years as separate layers, thus acting as masks, which could then be used to define plant community types of interest in the black and white data, assuming that these had not changed over time.

Woodlands were first defined by creating a 30 m buffer around the tree patches, followed by applying a -60 m buffer, thus creating a buffer 30 m within the outermost boundary. By doing this, the buffer excluded single trees, which can be found outside the larger patches, and would have caused a considerable overestimation of the actual woodland cover. The heathland community type was then defined by applying a 60 m buffer around the woodlands. The sparse birch trees found around the woodland perimeter were thus included in this buffer. The 60 m pick was not an arbitrary choice, but based on field observations when plots for other experiments in the area were established (see Chapters IV and V). Finally the remaining portions of the landscape were defined as grasslands, including both grasslands and wetlands, as explained earlier. Thus, there were three surface type categories defined: woodlands, heathlands and grasslands. For the purpose of this research, five plant community types were used, based on these three surface type categories: woodlands, grasslands and savanna; and w heathlands (woodland heathlands) and s heathlands (savanna heathlands) (Figure 6.3; see also Chapter II and Appendix A).

**FIGURE 6.3. Surface type categories in Hafnarskogur. The main plant community types defined in the area, woodlands, grasslands and savanna, are shown, but two are omitted: w heathlands and s heathlands. They form a perimeter around the woodlands and heathlands, respectively.**



### *Metric selection*

The maps which produced from the black and white aerial photographs consisted of only two classes, vegetated and bare, or eroded. They are thus relatively simple. The purpose of the research was to quantify surface configuration changes over time, specifically how the size and shape of vegetated and bare patches change. Those research goals require metrics that describe vegetation cover and density and patch shape.

Selecting appropriate landscape metrics is not straightforward. Despite common use, no holistic system has been developed for selecting appropriate metrics to measure and quantify landscape patterns (Bogaert, 2003). Different metrics designed to describe similar parameters may yield ambiguous results (Gustafson and Parker, 1992), and it can be unclear what they actually represent, e.g. because of their sensitivity to spatial resolution (McGarigal and Marks, 1995; Turner et al., 2001). All maps used for this analysis are of the same spatial scale, so this should not be a problem with the current dataset. Another problem related to metric selection is redundancy, as many metrics are highly correlated and add little to the descriptive information (O'Neill et al., 1988; Riitters et al., 1995; Robert H. Giles, 1999).

Metrics used for quantifying landscape patterns fall into three general categories: metrics of landscape composition, metrics of spatial configuration, and fractal metrics (Turner et al., 2001). Metrics of each category provide insight into different aspects of the underlying processes creating the current landscape pattern being analyzed. Ideally, each of these three categories should be represented in the metrics

selected to obtain the widest range of ecological information from the data. This is reflected in Turner et al. (2001), who suggest selecting metrics that are considered to be of ecological importance for the landscape properties of interest. There are a few rules about how to select the best metrics or which are considered to be best tools for a given situation. The selection of each metric should be based on what is estimated to be a good descriptor for the landscape or the features of interest, rather than using a 'shotgun' approach where multiple metrics are calculated blindly, and then the most promising ones selected afterwards, often by applying multiple correlation or multivariate statistical approaches. A 'shotgun' approach is illogical, as correlation does not necessarily imply causality (Kenny, 1979), even though the reverse may be true, i.e. that causation implies correlation (Shipley, 2000), and because of the risk of Type II statistical error (Sokal and Rohlf, 1981; Gardner and Urban, 2007). In addition, many of the currently available metrics are still poorly understood in terms of their ecological importance (McAlpine and Eyre, 2002) and their use should therefore be limited. McAlpine and Eyre (2002) also suggest that statistical methods (correlation or ordination) may lead to metric selection which fails to preserve subtle information in landscape data. Such blind searches for patterns, and thus metric selection, should be avoided.

Despite the lack of a general consensus on metric selection, some have become accepted as critical descriptors of certain processes. That applies to Patch Density (PD), Mean Patch Size (MPS) and Largest Patch Index (LPI), which are considered critical to describe landscape fragmentation and usually included when landscape fragmentation processes are of interest (McGarigal and Marks, 1995; Yang and Liu, 2005).

Other metrics were selected based on the *a priori* hypotheses on the expected landscape behavior as it goes through a land degradation sequence and active soil erosion. My hypothesis predicts that the landscape becomes fragmented as bare soil erosion spots emerge in the vegetated patches (Figure 6.2), later to expand and later coalesce, thus forming a gradient of simple to complex forms in the process. At later stages, the vegetated surfaces have become so fragmented that instead of a landscape consisting of vegetated surfaces with eroded areas, it is composed of eroded surfaces with small, shrinking vegetated islands, becoming more simple in shape as their size decreases. Metrics which describe cover, patch numbers and shapes are thus needed.

Twelve metrics were selected for image analysis and interpretation. Their description, the rationale behind the selection and expected metric behavior for the bare patch class can be seen in Table 6.2. In addition to the four already mentioned, the selected metrics were *Percentage of Landscape* (PLAND), *Number of Patches* (NP), *Edge Density* (ED) *Landscape Shape Index* (LSI), *Mean Shape Index* (MSI), *Shannon's Diversity Index* (SHDI), *Contagion Index* (CONTAG), and *Patch Cohesion Index* (COHESION).

Misclassification is always a problem and is hard to overcome. It is caused by the quality of the data (or lack thereof), resolution and the classification itself, i.e. complex classification increases error probability. However, metrics representing spatial distribution do not appear to be amplified by land cover misclassification (Wickham et al., 1997). Landscape metrics also differ in terms of sensitivity to classification



errors. The metrics selected here are relatively insensitive to such errors (Yu and Ng, 2006) and should thus reflect spatial configuration with acceptable accuracy.

The landscape metrics were calculated using Fragstats v3.3 (McGarigal et al., 2002) with the black and white grids described above as input data. Analysis type was “standard”, as opposite to “moving window”, due to hardware memory constraints, using an 8-cell rule for patch neighbors.

Regression lines for metrics vs. time were calculated in SigmaPlot v8.0 (SPSS Inc., 2001a) to show trends. No further statistical analysis was performed on the data, as it reflects an entire landscape of interest.

## **Results**

The results show clearly that simple classification on black and white imagery can be problematic. Andisols, the dominating soils in the area, are very dark when they are wet. This means that images taken shortly after rainfall will cause a systematic overestimation of vegetated surfaces, as both will appear equally dark. This problem was present in the image from 1997 (Table 6.1). The 1946 image quality did also create problems. The film had multiple speckles, which were interpreted as bare spots, hence overestimating both bare soil cover and bare spot number and density (Figure 6.4). The problems with the 1997 image were confined to a relatively small area, the savanna and the s heathlands, and did not appear to influence the results considerably, so it was

**TABLE 6.2**

**Landscape metrics selected and used in this study. Refer to McGarigal et al. (1995) and Gustafson (1998) for a more detailed descriptions.**

Acronym	Name	Unit and range	Description, rationale and expected bare class behavior	Category	Justification
PLAND	Percentage of Landscape	% 0 < and =< 100	The sum of all patches of a patch type, divided by total landscape area. PLAND is a good absolute descriptor of total landscape dominance or class cover. PLAND is expected to increase until the bare patch cover approaches the total area	Landscape composition	Dominance index
NP	Number of Patches	None >= 1	The number of patches in the landscape. NP is useful for quantifying landscape fragmentation. NP is expected to increase rapidly until coalescence starts, and then declines again.	Landscape composition	Fragmentation index
PD	Patch Density	No. per 100 ha > 0	Equals the number of patches of the corresponding patch type. PD is identical to NP if area is constant, but becomes critical for comparison of landscapes or areas of different sizes as it is defined on per unit area.	Spatial configuration	Fragmentation index
MPS	Mean Patch Size	ha > 0	Mean size of patches, equal to the sum of the areas (m <sup>2</sup> ) of all patches of the corresponding patch type divided by the number of patches of the same type, divided by 10,000. MPS is expected to be useful for detecting patch expansion and coalescence in conjunction with other metrics. MPS is expected to behave similarly to PLAND.	Spatial configuration	Fragmentation index
LPI	Largest Patch Index	% 0 < and =< 100	The area of the largest patch in the landscape divided by total landscape area. LPI is the proportional cover of the largest patch of the corresponding class in the landscape. LPI measures dominance as PLAND, but yields specific information on the largest patch within a class. It may thus be helpful in determining state of fragmentation. LPI is expected to behave similarly to PLAND	Spatial configuration	Dominance Index

**TABLE 6.2****Continued.**

Acronym	Name	Unit and range	Description, rationale and expected bare class behavior	Category	Justification
ED	Edge Density	m per ha > 0	The sum of the lengths of all edge segments in the landscape, divided by the total landscape area. ED is 0 when there is no class edge in the landscape (i.e. there is only one class). ED is expected to increase rapidly as new patches form, and then decline as they expand and coalesce	Spatial configuration	Fragmentation index
LSI	Landscape Shape Index	None ≥ 1	The total length of edge in the landscape divided by the minimum total length of edge possible. LSI = 1 when the landscape consists of only one class and it is circular (vector data) or square (raster data). It increases without limit as the shape becomes irregular or as the total edge increases, or both. LSI quantifies shape complexity influenced by ED. Low LSI should thus be sensitive towards small irregular shapes, as is expected at the early degradation stages when coalescence has just begun. Rapid ED growth is therefore expected to cause a rapid LSI increment, followed by a decline once ED declines.	Spatial configuration	A measure of patch aggregation
MSI	Mean Shape Index	None ≥ 1	MSI describes shape complexity. It equals 1 when all patches are circular or square, but increases without limit as the shape becomes more irregular. MSI describes mean shape complexity for each class type, and is expected to behave in a similar way as LSI described above. It differs however as it does not include edge density (McGarigal and Marks, 1995), only shape, and should thus yield helpful information in addition to LSI, especially when coalescence is occurring, but increased shape complexity is expected while the patches are expanding and coalescing. When MSI is used it must be kept in mind that it appears to be sensitive towards different spatial resolutions, and it has thus been suggested that MSI should not be used in studies where data with different spatial resolutions is used (Saura, 2002; Frohn and Hao, 2006). That is not the case in this study.	Spatial configuration	Fragmentation index

**TABLE 6.2**

**Continued.**

Acronym	Name	Unit and range	Description, rationale and expected bare class behavior	Category	Justification
MPFD	Mean Patch Fractal Dimension	None 1 =< and =< 2	MPFD estimates shape complexity as MSI, but is based on fractals. It approaches 1 when average shape geometry is very simple (circles, squares), but 2 as they become more irregular with highly convoluted plane-filling perimeters. Same behavior is expected as for MSI.	Fractal dimension	Fragmentation index
CONTAG	Contagion Index	% 0 < and =< 100	CONTAG approaches 0 when the distribution of adjacencies among unique patch types becomes increasingly uneven, but equals 100 when all patch types are equally adjacent to all other patch types. CONTAG measures physical connectedness between patches of different classes. It should therefore be sensitive towards initial degradation states where many small bare patches are forming. CONTAG should be low for landscapes consisting of only one patch type, but highest when the landscape is highly fragmented. e.g. early in the coalescence process.	Spatial configuration	Fragmentation index
COHESION	Patch Cohesion Index	None 0 =< and < 100	COHESION measures physical connectedness of patches within a class type. It ranges from 0 to 100; lower numbers indicate subdivided landscapes with little connection between the patches, e.g. isolated and small eroded patches. This means that a landscape early in the predicted degradation sequence should have low cohesion values for the bare class, which should then increase as the bare class area increases. The reverse should be true for the vegetated class as the degradation sequence progresses.	Landscape composition	A measure of physical connectedness

**TABLE 6.2****Continued.**

Acronym	Name	Unit and range	Description, rationale and expected bare class behavior	Category	Justification
SHDI	Shannon's Diversity Index	None $\geq 0$	SHDI equals, minus the sum, across all patch types, the proportional abundance of each patch type by that proportion. SHDI equals 0 when all patches are of the same type (no diversity), but increases as number of different patch types increases or the area between the different patch types becomes more equitable. SHDI measures patch diversity and is thus useful for estimating landscape composition (O'Neill et al., 1988; Turner, 1990). It is more sensitive to richness than evenness, rare patch types should thus have disproportionately greater influence on this metric (McGarigal and Marks, 1995). This sensitivity makes it more feasible than e.g. Simpson's diversity index, and this property is expected to be helpful at detecting changes during the initial degradation stages, when small and initially few eroded patches may form. SHDI is expected to increase gradually as bare patches increase in number, but decrease once the bare patch class becomes dominant..	Landscape composition	Diversity index

decided to include it in the results and discussion. The 1946 image data however was included in comparisons across years (Figure 6.5) but excluded in comparisons across plant communities (Figure 6.6). Regression lines were added to graphs where appropriate to show trends (Figure 6.5).

### **Total area (landscape level)**

*Total dominance (PLAND, LPI), Figure 6.5 A&E.*

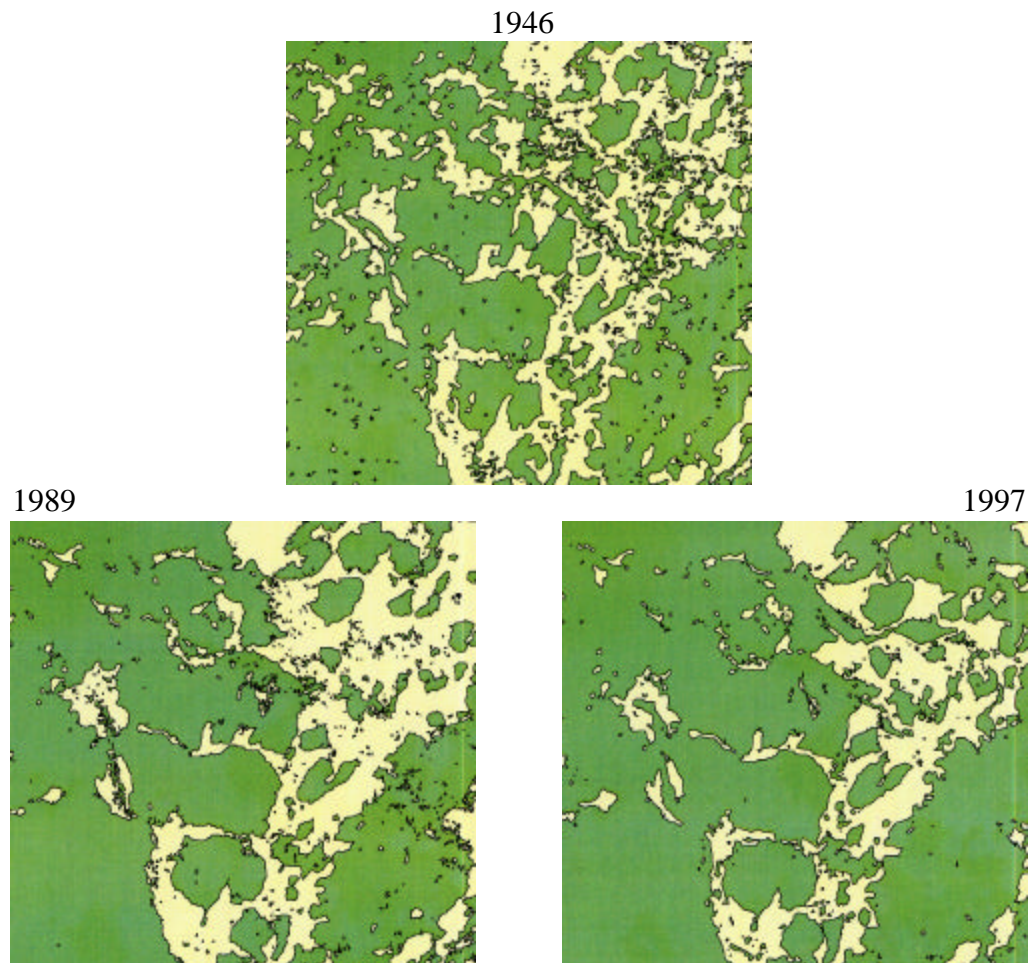
PLAND<sub>b</sub> (PLAND bare; this notation is used in the following text, b for bare patches [eroded] and v for vegetated) and PLAND<sub>v</sub> show no directional trend over the period. The same is true for LPI<sub>v</sub>, but LPI<sub>b</sub> increases slightly with time.

*Total fragmentation (PD, MPS, ED, MSI, MPFD, CONTAG), Figure 6.5 B, C, E, G, H & J*

NP and PD are identical for all years, as the landscape area is constant. Bare patches decline steadily, but the trend is heavily influenced by the 1946 data with very high NP<sub>b</sub> and PD<sub>b</sub>. A declining trend is also noticed for the vegetated areas for the same reason, but is far less noticeable due to lower values.

MPS<sub>v</sub> and MPS<sub>b</sub> increase over the period, but at different rates and the variability is high for both classes as the SE indicates.

ED shows a steady decline over the 51 years, mostly due to high edge densities in 1946, but no such strong directional trend is visible if that year is excluded.



**FIGURE 6.4.** Classified landscape from Hafnarskogur. Top: 1946 image showing multiple small bare spots, due to poor film quality. Bottom left and right: 1989 and 1997 images, respectively. A much higher proportion of vegetated (green) are is obvious in 1997 when compared to 1989. This is due to wet bare soils, which cannot be distinguished from vegetation cover, hence causing a systematic overestimation of the vegetated area.

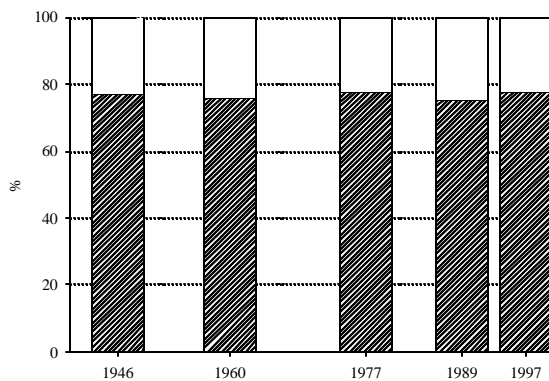
**FIGURE 6.5.** Landscape metrics averaged over the woodland, savanna, grassland, w heathland and s heathland plant community types in Hafnarskogur for five selected years ranging from 1946 - 1997. Diagonal (▨) and eroded (□) bars represent vegetated and eroded surfaces respectively. Cross-hatched bars (⊠) either stand for metrics at the landscape level, or metrics with identical values for the vegetated or eroded surface classes. Number of patches (NP) are omitted as they equal patch density at the landscape scale.

The regression lines represent best fit for vegetated and bare surfaces. Two lines were calculated for the bare surface data, with and without 1946.

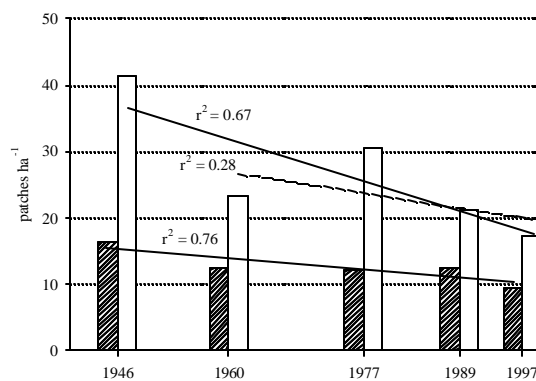
Whiskers stand for  $\pm$  SE for the mean where applicable. Please note variable Y-scales and refer to Table 6.2 for full list of acronyms and metric ranges.



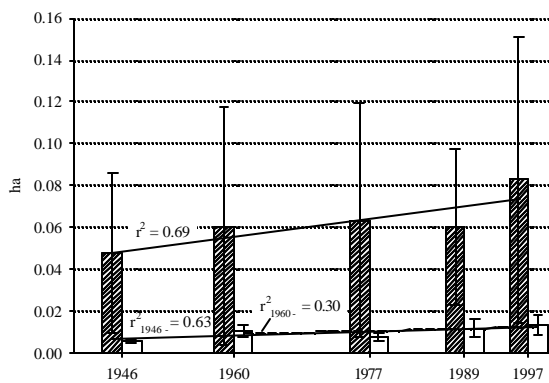
A: percentage of landscape (PLAND)



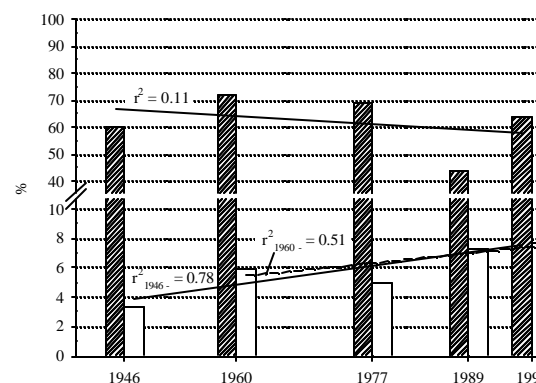
B: patch density (PD)



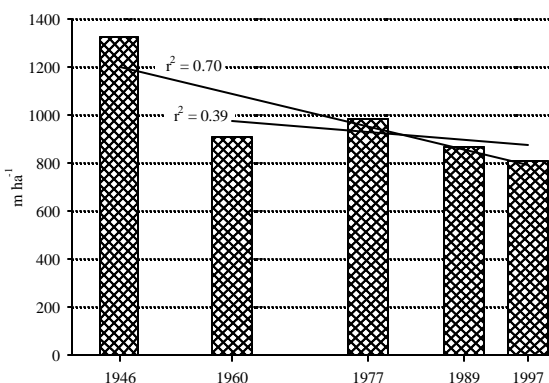
C: mean patch size (MPS)



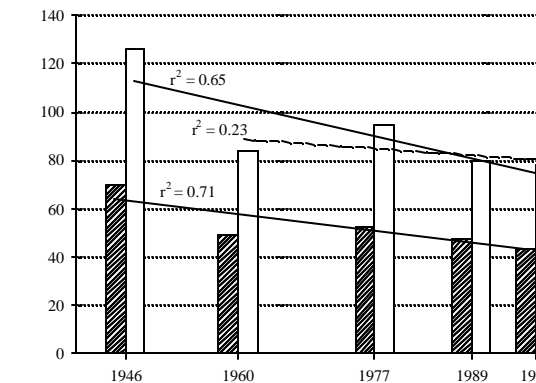
D: largest patch index (LPI)



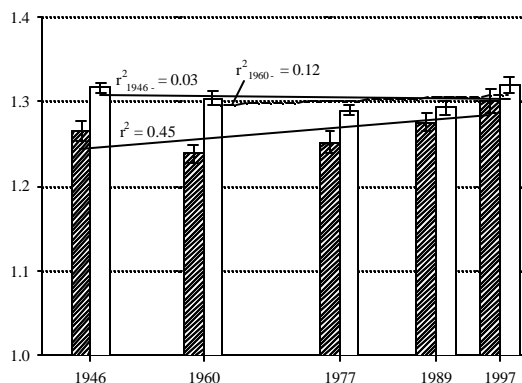
E: edge density (ED)



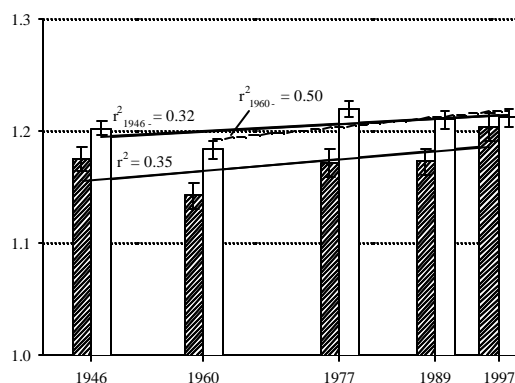
F: landscape shape index (LSI)



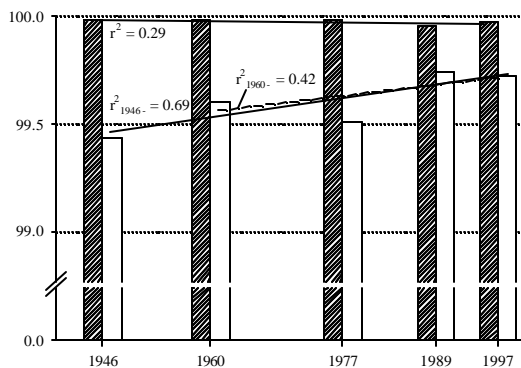
G: mean shape index (MSI)



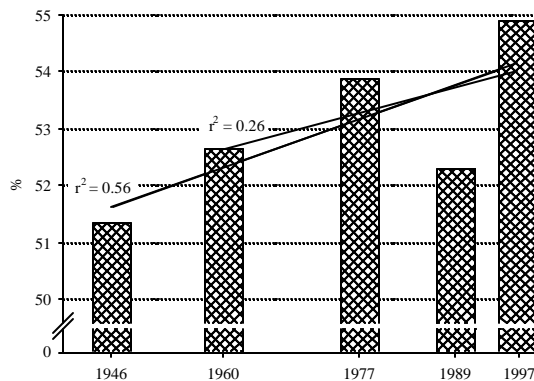
H: mean patch fractal dimension (MPFD)



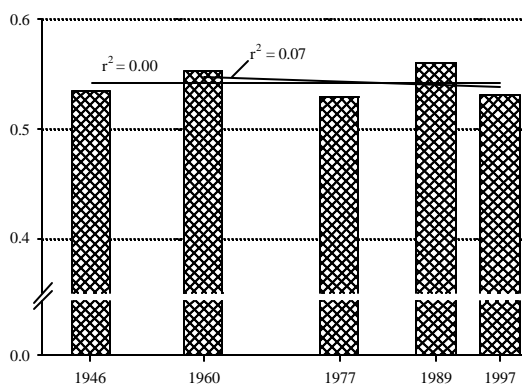
I: patch cohesion index (COHESION)



J: contagion index (CONTAG)

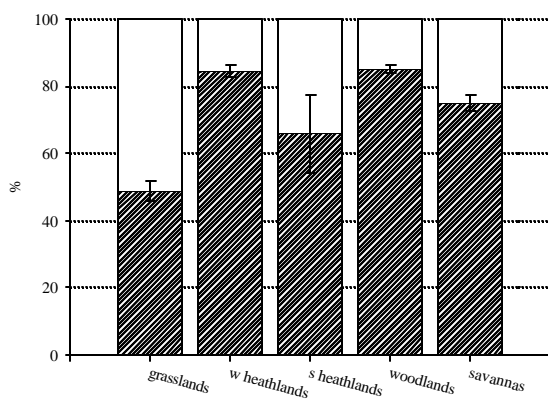


K: Shannon's diversity index (SHDI)

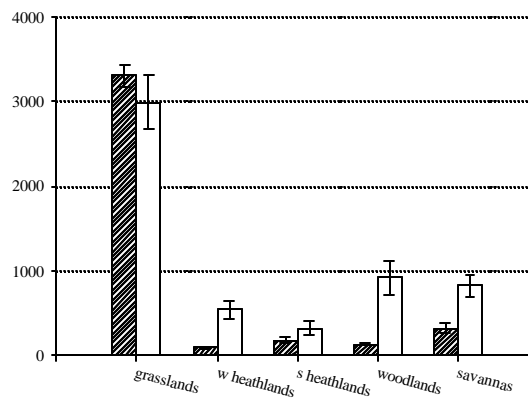


**FIGURE 6.6.** Landscape metrics describing surface types for plant community types in Hafnarskogur. Each bar represents five-year average for 1946, 1960, 1977, 1989 and 1997 for vegetated (▨) and eroded (□) surfaces. Cross-hatched bars (⊠) represent either metrics at the landscape level, or metric with identical values for the vegetated or eroded surface classes. Whiskers stand for  $\pm$  SE for the mean. Please note variable Y-scales and refer to Table 6.2 for full list of acronyms and metric ranges.

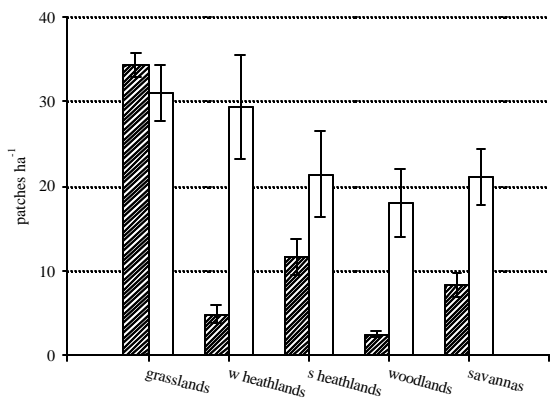
A: percentage of landscape (PLAND)



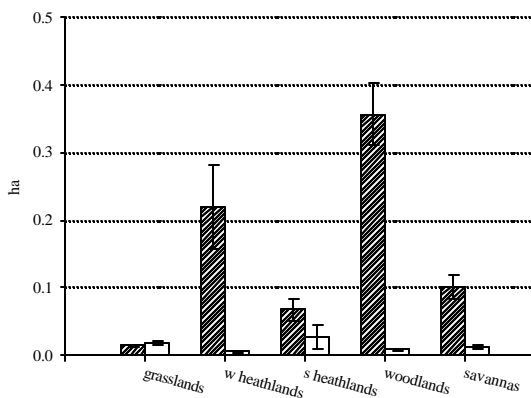
B: number of patches (NP)



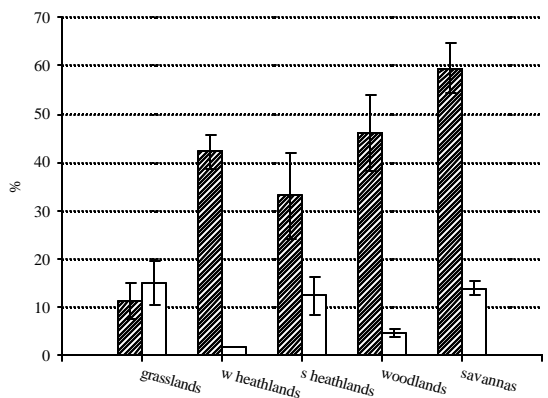
C: patch density (PD)



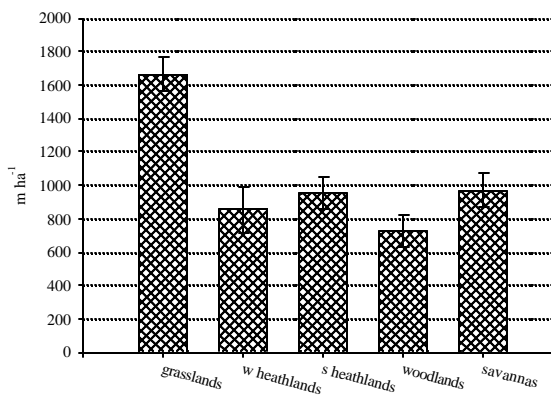
D: mean patch size (MPS)



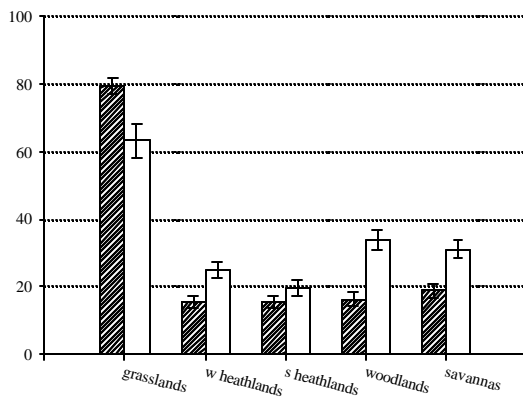
E: largest patch index (LPI)



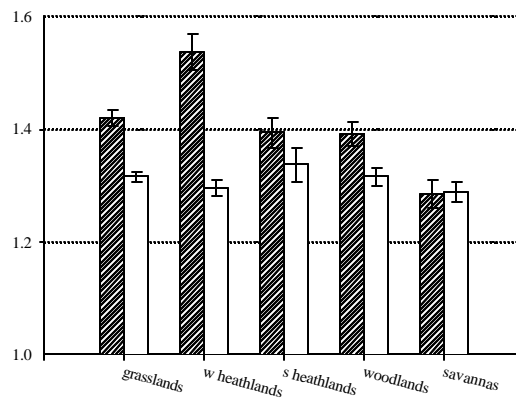
F: edge density (ED)



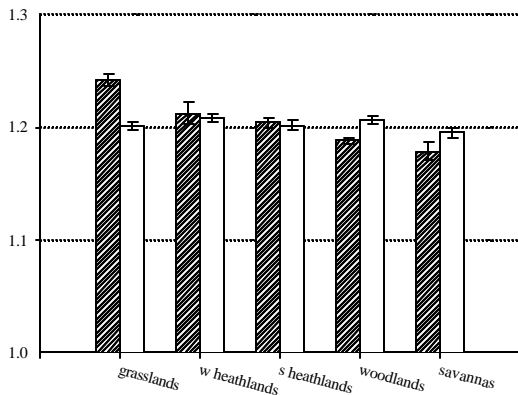
G: landscape shape index (LSI)



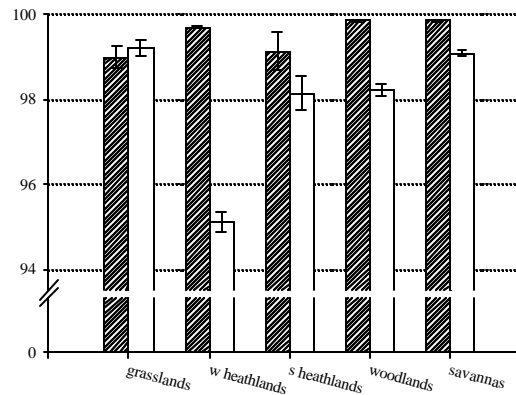
H: mean shape index (MSI)



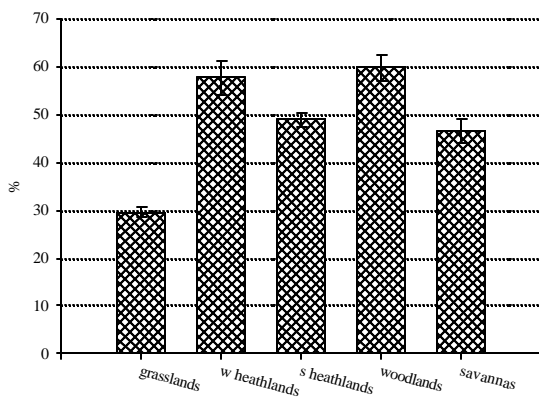
I: mean patch fractal dimension (MPFD)



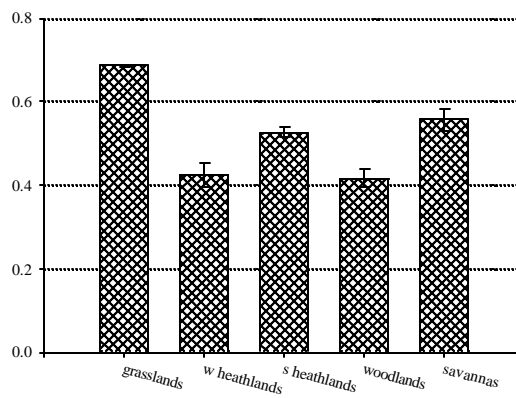
J: patch cohesion index (COHESION)



K: contagion index (CONTAG)



L: shannon's diversity index (SHDI)



MSI and MPFD both describe shape complexity and show similar trends for the total landscape over the 51-year period. It is noteworthy that both  $MSI_v$  and  $MPFD_v$  increase steadily if 1946 is excluded, but the trend is not nearly as pronounced if it is included.

Contagion increases steadily over the time period, but the range is low.

*Total aggregation (LSI, COHESION), Figure 6.5 F & I*

Simple shapes such as circles or squares are reflected in low LSI, whereas more convoluted and complex geometry yields higher values. LSI is also affected by ED, low ED results in smaller LSI (McGarigal and Marks, 1995). This metric does thus not only reflect shape complexity as MSI and MPFD, but also patch size.

$LSI_v$  is consistently lower than  $LSI_b$ , which suggests that the vegetated patches are more regular than their eroded counterparts. Both classes decline over time, but the data for 1946 influences the trend. By excluding that year the rate of decline becomes lower. The  $LSI_b$  metric also indicates a steady decline over the time period, but again this is heavily influenced by the 1946 data.

$COHESION_v$  shows little changes over time, whereas  $COHESION_b$  increases slightly.

*Diversity (SHDI), Figure 6.5 L*

SHDI is low and shows no directional trend for the landscape over the time period.

### **Plant community types**

*Total dominance (PLAND, LPI), Figure 6.6 A & E*

PLAND indicates that the grasslands have proportionally lowest vegetated surfaces compared to the other community types, and it is noteworthy that the two woodland-related community types (woodlands and w heathlands) both have the highest vegetation cover and the lowest variability, whereas the s heathland community type stands out when compared to the other three woodland related community types (woodlands, w heathland and savannas) as being both lowest and with considerably higher data variability.

There is a gradual increase in  $LPI_v$  going from grasslands, to heathlands to woodlands community types, the woodland types having the highest values but grasslands the lowest. The same is not true for  $LPI_b$  however. There the grasslands, s heathlands and savannas are similar, and at least two times higher than the corresponding values for the woodlands and w heathland communities, reflecting wrong classification due to wet soils.

*Total fragmentation (NP, PD, MPS, ED, MSI, MPFD, CONTAG) Figure 6.6 B, C, D, F, H, I & K*

The grasslands stand out by having the highest NP for both the vegetated and eroded classes.  $PD_v$  in the grassland plant community type is also very high compared to the other community types, but  $PD_b$  is more in line with the other plant community types. The ED shows a similar trend as  $PD_v$  and is highest for the grasslands but a non-directional trend is observed for the other four plant community types.

MPS varies considerably.  $MPS_v$  is highest in the woodlands, followed by the woodland heathlands, but lowest in the grasslands.  $MPS_b$  is highest for the savanna heathlands, followed by the grasslands and savannas, but lowest in the woodlands and woodland heathlands.

The two shape metrics, MSI and MPFD differ in their results.  $MSI_b$  and  $MPFD_b$  reveal relatively small differences between the plant community types, but greater differences exist for the vegetated patches, and considerably greater for the  $MSI_v$  than  $MPFD_v$ . The woodland heathlands and grasslands are highest, but the savannas lowest.

Contagion is lowest for the grassland communities, but highest for the two woodland types, woodland heathlands and woodlands.

*Total aggregation (LSI, COHESION), Figure 6.6 G & J*

LSI reveals high values for both cover classes for the grasslands communities. The values for the vegetated classes for the other four plant community types are similar, but greater differences exist for the bare cover classes. There the woodlands and savannas types stand out as being higher than the corresponding class for the two heathland community types.

The COHESION metric behaves very differently for the two cover classes. The vegetated class is relatively uniform across all five plant community types, with the woodlands and savannas community types being the highest, but the grasslands the lowest. Little difference exists between the grasslands and the savanna heathlands, however, the variability is higher for the savanna heathlands communities than the grasslands communities. The bare cover class varies considerably more across the



community types. The woodland heathland type is noticeably lower than any other bare class, followed by the savanna heathlands cover class and the woodlands. The highest values were obtained for the grasslands and the savannas community types.

*Diversity (SHDI), Figure 6.6 L*

SHDI is low but the grasslands are higher than the other plant community types, which reflects the high NP values for this community.

### **Discussion**

Changes in metrics over time are small, but it appears as the bare class patches are increasing in size, albeit very little. The results do thus indicate a small overall change with time. PLAND and LPI measure total dominance. PLAND and LPI<sub>v</sub> reflect no changes over time. LPI<sub>b</sub> indicates a slight increase in largest patch size, which could suggest bare patch coalescence. This may well be the case, as MPS<sub>b</sub> increases slightly, and PD<sub>b</sub> increases, while ED decreases, hence bare area appears to stay constant, although the bare patches increase in size, but decrease in numbers.

The shape indices, LSI, MSI and MPFD are conflicting. LSI<sub>b</sub> is relatively high, suggesting that complex bare patch shapes are present in the landscape, but the value decreases with time, which should indicate a transition from complex shapes to more regular. The decline trend however, is mostly due to the 1946 data. If it is omitted, then there is no such decline trend, and it can be compared to both MPFD<sub>b</sub> and MSI<sub>b</sub>, which increase with time, suggesting that shape complexity is increasing. The change is relatively small however.

The  $COHESION_b$  shows very little change with time. It can range from 1 to 100, with high values characteristic of landscapes with many large patches of the same class. The slightly increasing trend for  $COHESION_b$  may suggest that the bare patches are coalescing and growing, but the difference is so small that it must be considered almost negligible, despite that the slight growth trend does exist. The same trend is displayed for the  $CONTAG$  metric. The slight increase may suggest that one patch type, in this case bare, is increasing in size.

The subtle overall changes detected for the entire landscape may suggest that some parts of the landscape are behaving differently than others, i.e. that erosion is more active in some parts than others. Figure 6.6 A-L shows the landscape metrics across the five plant communities defined in the landscape, but there are no clear trends.  $PLAND_b$  shows that grasslands have the highest bare cover, hence the largest eroded areas, but the woodlands, w heathland and savanna the smallest. This may reflect a fundamental difference between these community types, that the woodlands are less likely to suffer from erosion than the open plant communities. High  $NP_{b\&v}$  and high  $PD_{b\&v}$  suggest high fragmentation within a community, and low  $MPS_{b\&v}$  and low  $LPI_{b\&v}$  suggest that it is dominated by relatively small vegetated and bare patches. However, there is little difference between the grasslands and the other community types when shape complexity is considered ( $MSI$  and  $MPFD$ ), with the exception of  $LSI$ .  $LSI$  is based on  $ED$ , so high  $ED$  values are reflected in the  $LSI$  metric.

The only plant communities that stands out from the others, are the savanna and s heathlands. It was observed during fieldwork in 1999 – 2003 that the eroded areas in

those two plant community types, especially the savanna, were different from the eroded areas in the other community types, the grasslands, woodlands and w heathlands. In those three the eroded surfaces were commonly covered with gravel and stones, resulting from frost heaving, but this was uncommon in the savanna. This might be an indicator of younger erosion features in the savanna. Classification of the black and white images was problematic in that area (Figure 6.4) as it was impossible to distinguish between wet soils and vegetation, and this problem may have been attenuated by the lack of gravelly surfaces in that area. This underlines the importance of using imagery with infrared spectral bands in areas dominated with Andisols, black and white images should be avoided.

The purpose of this study was to use a relatively simple and straightforward image data analysis method to test how 12 selected landscape metrics describe a degraded landscape considered to be at different degradation stages (Chapter III). The metrics were selected based on how well they were suited to quantify the degradation landscape features. Their expected behavior was listed in Table 6.2. Table 6.3 shows the expected metric behavior (graphs) and the observed metric trend for each of the five plant community types. The observed data for each metric rarely fit the expected trend more than 50 %, and often less. This suggests that the metrics are either poorly suited for detecting the changes, or the landscape does not behave as expected.

Tables 6.4 and 6.5 show Spearman's rank correlation coefficient matrix for the landscape metrics used in this study. Many of them show high correlation with other metrics. This indicates that despite selecting the metrics carefully based on their

properties and the expected behavior of the landscape, many of them are redundant. It has been pointed out that many landscape indices are improperly used (Li and Wu, 2004). Spatial heterogeneity is scale dependent, but the understanding of scaling relationships of spatial patterns are often lacking. Better metric selection can be achieved by understanding their behavior across scales (Wu et al., 2002), e.g. by applying them to neutral landscapes with known properties (Li et al., 2005; Gardner and Urban, 2007). This approach should ensure that the “best” metrics are selected at any given time. This study underlines the importance of understanding and knowing and understanding the expected patterns for a given process and landscape prior to the metric selection.

The subtle changes over time observed in this study does not support the hypothesis that the landscape, or different plant communities within the communities, are on a degradation trajectory, driven by grazing and extreme climatic events, as is suggested in Chapter III. It is possible that the land degradation occurs in episodic events, and such events have not happened over the last 51-years. It may also be that this landscape, or parts of it, has crossed the hypothesized threshold between S3 → S4 presented in Figure 3.4, and changes cannot be expected under the current conditions. It can not be stated, based on these results that this landscape, under the current conditions, is at degradation risk. The subtle differences observed also mean that the metrics cannot be properly evaluated as intended. The results are thus inconclusive with respect to the feasibility of the selected 12 metrics.

TABLE 6.3

Expected (see Table 6.2) and observed behavior of landscape metrics for bare surface patches 1946–1997. The letters p, e, i, l and d stand for pristine land, early, intermediate, late degradation stage, and denuded land, respectively.

The last column indicates how well the observed trend fits with expected values.

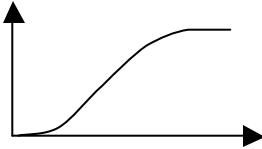
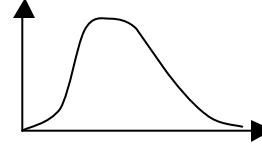

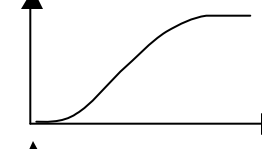
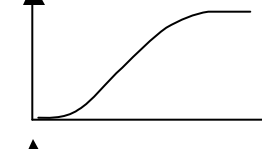
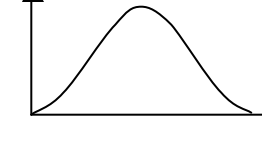



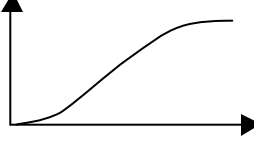
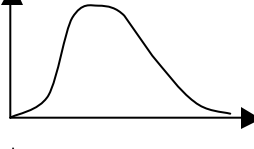
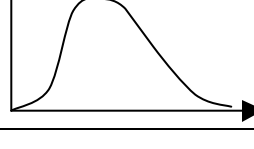
Acronym	Community	Expected					Observed <sup>1</sup>				
		p	e	i	l	d	'46-'60	'60-'77	'77-'89	'89-97	% fit
PLAND	grasslands		+	-	+	-	+	-	+	-	50
	w heathlands		-	-	-	-	-	-	-	-	0
	s heathlands		+	-	-	-	+	-	-	+	50
	woodlands		-	-	-	-	-	-	-	-	0
	savannas		+	-	-	-	-	+	-	-	-
NP	grasslands		-	-	+	-	-	-	+	-	25
	w heathlands		-	-	-	+	+	+	+	50	
	s heathlands		-	-	-	+	+	+	+	25	
	woodlands		-	-	-	+	+	+	+	50	
	savannas		-	-	-	-	+	+	+	25	
PD	grasslands		-	-	+	-	-	-	+	25	
	w heathlands		-	-	+	+	+	+	50		
	s heathlands		-	-	-	-	+	+	+	25	
	woodlands		-	-	+	+	+	+	50		
	savannas		-	-	-	-	+	+	+	25	
MPS	grasslands		+	-	+	-	+	-	+	50	
	w heathlands		+	-	+	+	+	+	75		
	s heathlands		+	-	-	-	+	+	+	50	
	woodlands		+	-	+	+	+	+	75		
	savannas		+	-	-	-	+	+	+	25	
LPI	grasslands		+	-	+	+	+	+	75		
	w heathlands		-	-	-	-	+	+	+	50	
	s heathlands		+	-	+	+	+	+	75		
	woodlands		-	-	+	+	+	+	50		
	savannas		+	-	-	-	-	-	-	25	
ED	grasslands		-	+	+	-	+	-	+	50	
	w heathlands		-	+	+	+	+	+	75		
	s heathlands		-	+	-	-	-	-	-	25	
	woodlands		-	+	+	+	+	+	75		
	savannas		-	+	+	+	+	+	75		

TABLE 6.3

Continued.

Acronym	Community	Expected p e i l d	Observed <sup>1</sup>				% fit
			'46-'60	'60-'77	'77-'89	'89-97	
LSI	grasslands		-	+	+	-	50
	w heathlands		-	+	+	+	75
	s heathlands		-	+	-	+	50
	woodlands		-	+	+	+	75
	savannas		-	+	-	-	25
MSI	grasslands		-	-	-	+	25 <sup>3</sup>
	w heathlands		+	-	-	+	50 <sup>3</sup>
	s heathlands		+	-	+	-	50 <sup>3</sup>
	woodlands		+	-	-	+	50 <sup>3</sup>
	savannas		+	-	+	-	50 <sup>3</sup>
MPFD	grasslands		-	+	+	-	50 <sup>3</sup>
	w heathlands		+	-	-	+	50 <sup>3</sup>
	s heathlands		+	-	+	+	75 <sup>3</sup>
	woodlands		+	+	+	-	75 <sup>3</sup>
	savannas		-	-	+	-	25 <sup>3</sup>
COHESION	grasslands		+	-	+	+	75 <sup>3</sup>
	w heathlands		+	-	-	-	25 <sup>3</sup>
	s heathlands		+	-	-	+	50 <sup>3</sup>
	woodlands		-	-	+	+	50 <sup>3</sup>
	savannas		+	-	+	-	50 <sup>3</sup>
CONTAG <sup>2</sup>	grasslands		+	+	-	+	75
	w heathlands		+	+	-	-	50
	s heathlands		+	-	-	-	25
	woodlands		+	+	-	-	50
	savannas		+	-	-	-	25
SHDI <sup>2</sup>	grasslands		+	+	+	-	75 <sup>3</sup>
	w heathlands		-	-	+	+	50
	s heathlands		+	+	+	-	75
	woodlands		-	+	+	-	50
	savannas		+	+	+	+	100

<sup>1</sup> Observed scores are the comparison between actual data and expected metric behavior. Positive (+) signs indicate that the data follows the expected metric behavior for that time interval, but negative (-) signs indicate the opposite trend.

<sup>2</sup> Metrics at the landscape level.

<sup>3</sup> Metric change is less than 10% for the corresponding community type over the time period, and thus inconclusive despite apparent high or low fit.

**TABLE 6.4**

**Spearman's rank correlation coefficient matrix for the landscape metrics used in this study. Only bare surfaces in all years are included in the computation. Only the coefficients in the lower diagonal part are presented.**

	PLAND	NP	PDha <sup>1</sup>	MPS	ED	LPI	LSI	MSI	MPFD	SHDI	CONTAG	COHESION
PLAND	1.00											
NP	0.42	1.00										
PDha	0.27	0.65	1.00									
MPS	0.66	0.10	-0.44	1.00								
ED	0.80	0.75	0.76	0.19	1.00							
LPI	0.79	0.14	-0.18	0.77	0.42	1.00						
LSI	0.39	0.98	0.59	0.13	0.69	0.12	1.00					
MSI	0.10	-0.29	-0.53	0.53	-0.27	0.04	-0.18	1.00				
MPFD	-0.41	-0.05	0.13	-0.38	-0.22	-0.62	0.00	0.35	1.00			
SHDI	0.91	0.58	0.45	0.54	0.86	0.67	0.55	-0.03	-0.30	1.00		
CONTAG	-0.89	-0.64	-0.55	-0.44	-0.92	-0.59	-0.61	0.08	0.23	-0.98	1.00	
COHESION	0.76	0.38	-0.19	0.85	0.43	0.89	0.38	0.15	-0.57	0.65	-0.59	1.00

<sup>1</sup>Patch Density per hectare.

**TABLE 6.5**

**Spearman's rank correlation coefficient matrix for the landscape metrics used in this study. Only vegetated surfaces in all year are included in the computation. Only the coefficients in the lower diagonal part are presented.**

	PLAND	NP	PDha <sup>1</sup>	MPS	ED	LPI	LSI	MSI	MPFD	SHDI	CONTAG	COHESION
PLAND	1.00											
NP	-0.87	1.00										
PDha	-0.87	0.84	1.00									
MPS	0.90	-0.84	-0.98	1.00								
ED	-0.80	0.79	0.85	-0.80	1.00							
LPI	0.61	-0.45	-0.59	0.58	-0.61	1.00						
LSI	-0.79	0.83	0.75	-0.72	0.92	-0.65	1.00					
MSI	0.18	-0.39	-0.07	0.13	0.08	-0.48	0.00	1.00				
MPFD	-0.46	0.38	0.65	-0.60	0.57	-0.71	0.48	0.51	1.00			
SHDI	-0.91	0.85	0.84	-0.81	0.86	-0.51	0.77	-0.13	0.44	1.00		
CONTAG	0.89	-0.85	-0.85	0.81	-0.92	0.53	-0.84	0.07	-0.48	-0.98	1.00	
COHESION	0.64	-0.39	-0.72	0.72	-0.55	0.86	-0.45	-0.45	-0.77	-0.52	0.52	1.00



In general, the results show a landscape where relatively little change is occurring, and little directional trend is observed over the 51 years this study covers; neither landscape composition, nor configuration changes in noticeable fashion over the time period covered by the aerial images.

Despite the fact that landscape metrics have been used to monitor land surface changes over time for various purposes, including rangeland monitoring (Bastin et al., 2002), watersheds (Yang and Liu, 2005), urban sprawl (Jat et al., 2008) and land degradation (Kepner et al., 2000), not many studies have been published where surface features have been monitored which are driven by similar degradation processes and conditions as are found in Iceland. It is thus hard to find comparable data and examples to evaluate the results presented here. This is further confounded by the fact that there appear to be little changes occurring, the degradation processes are not active, and therefore it is impossible to evaluate how well the selected metrics would represent changes driven by them. We do thus not know if these metrics are applicable, further research is needed in this field, and they should start with hypothetical modeling, e.g. neutral landscapes.

## CHAPTER VII

### SUMMARY AND CONCLUSIONS

Ecosystem degradation during 1100 years of human settlement in Iceland has resulted in extensive soil erosion and altered vegetation composition. Most of the lowlands were covered with woodlands at the time of settlement, but they have declined and cover only about 1% of the total land area today, less than 5 % of the pre-settlement woodlands (Gudjonsson and Gislason, 1998; Aradottir et al., 2001). The dominant soils are Andisols (Arnalds, 2004), which derive their physical properties from volcanic materials (Wada, 1985; Brady and Weil, 1998). Andisols are characterized by low bulk density and low aggregate cohesion, which makes them highly vulnerable to eolian and fluvial erosion (Wada, 1985). Icelandic climate is characterized by cool summers and mild winters. This causes temperatures to fluctuate around 0°C during the winter, hence causing frequent freeze-thaw cycles that may destabilize soils and vegetated surfaces and contribute to land degradation and erosion. The woodland disappearance is regarded as a precursor to land degradation (Carson, 1985; R. C. Derose, 1993; Olafsdottir and Gudmundsson, 2002; Rosenmeier et al., 2002). Woodland openings promote radiative heat loss and attenuates snow accumulation (McKay and Gray, 1981) with corresponding insulation loss (Hinkel and Hurd, 2006). Grazing and trampling by livestock may similarly reduce the insulative capacity of the ground layer vegetation (Cole and Monz, 2002).

The research presented here addresses several questions related to deforestation as it may have occurred following the settlement: how it may have affected birch (*Betula pubescens* Ehrh.) seedling survival, how the deforestation may have increased cryoturbic disturbances and thus decreased surface stability, and how the land degradation and erosion, which followed the deforestation, manifests itself at larger scales, by applying remote sensing and landscape metrics.

The research questions were approached by constructing a State-and-Transition model (S&T) (Chapter III). S&T models are conceptual models widely used by resource management professionals to organize current knowledge and identify key gaps in knowledge and understanding. To date, S&T models have been developed primarily for dryland systems in tropical, subtropical and temperate regions. The S&T model presented in Chapter III proposes a degradation sequence driven by continuous grazing and climate, where woodlands will transit into open heathlands or grasslands. If the grazing persists, soil erosion spots (SES) will form, expand and coalesce, causing the system to transit into a state dominated by abiotic processes causing SES to further expand and coalesce, eventually resulting in total denudation and desertification (Figure 3.4). The key transitions suggested by the model are T3, linking woodlands and heathlands and grasslands, and T4 linking heathlands and grasslands, and SES dominated landscapes. T3 marks the initial degradation process as plant communities shift from woodlands to open plant community types, whereas T4 represents an ecosystem threshold where states dominated by biotic processes enter a new state

dominated by abiotic processes. T3 is driven by grazing, but T4 by grazing and intensified cyoturbation.

Chapter IV addresses questions related to the T3 transition, how repeated grazing may have restricted woodland regeneration, and thus contributed to their degradation and disappearance. The results showed that seedling growth was dependent on the plant community type, as open woodlands (savanna) and grasslands showed significantly higher total growth than seedlings in dense woodlands (Figure 4.2). Clipping also reduced total growth significantly when compared to controls, but there was no statistical difference between the two clipping treatments at the end of the experiment (Figure 4.3). Clear effects of the treatments did not appear until after 3 years, suggesting a carryover treatment effect, possibly reflecting a depletion of energy and nutrients stored in the seedling tissues. Seedling mortality was high in the woodlands and significantly higher than in the grasslands (Figure 4.7). Mortality rates were highest for the most intense treatment (75% of the total crown length removed) but no difference were observed between the control and the low browsing treatment (25% of the total crown length removed). The results do thus suggest that continuous grazing may lead to reduced seedling growth and increased mortality. Grazing may therefore have harmful effect on woodland regeneration and thus contribute to deforestation over time.

Chapter V addresses questions related to the T4 transition, how vegetation changes accompanying deforestation and livestock grazing affect cryoturbic disturbances, or surface stability. Frost heaving was lowest in the woodland and grasslands (Figure 5.4 B), suggesting that they provide stable environments, possibly

due to high herbaceous biomass in the grasslands (Figure 4.4) and fewer freeze thaw cycles in the woodlands (Figure 5.12). The simulated grazing treatments, clipped, trampled and clipped and trampled, showed clear effects on the surface stability after four years, with the exception of the woodlands, where clear trends were apparent after only two years. This may suggest that the woodland communities have low resilience, and the herbaceous layer present in woodlands may have low resistance if the forest is removed.

Surface movement in treatments where mineral soil was exposed (trampled) was significantly greater than for clipped treatments or controls (Figure 5.5 B, D, F, H), and surface levels declined for the trampled treatments, compared to the clipped and controls (Figure 5.5 A, C, E, G), suggesting that erosion was active in the trampled plots. The presence of vegetation cover does thus reduce both absolute surface movement, and surface decline. The surface strength decreased with time for the trampled treatments (Figure 5.9) which may have contributed to the increased surface movements, but the loss of vegetation cover in the trampled treatments also intensified cryoturbic processes as can be seen in Figure 5.11 and Figure 5.12. The results do thus suggest that intense grazing may intensify cryoturbic processes, and therefore can contribute to the T4 transition.

Chapter VI focuses on the feasibility of applying selected landscape metrics to assess landscapes at different degradation stages. Spatial structures affect ecological processes (Turner, 1989; Gustafson, 1998; Turner et al., 2001), which suggests that it might be possible to identify landscapes at risk by applying methods suitable for

detecting and quantifying landscape features if we know what kind of patterns to seek. The results were inconclusive. Little or no directional trend was observed in the data, either due to data limitation or because little changes did occur. No conclusions on the feasibility of using landscape metrics can thus be drawn, but further research is needed, both on metric behavior using simulated landscapes, but also on areas currently experiencing active land degradation and erosion.

The S&T model presented in Chapter III suggested that two critical transitions, T3 and T4 were driven by grazing and grazing and abiotic processes, respectively. The results show that browsing may reduce growth and cause increased seedling mortality. Such disturbances would be able to initiate the T3 transition over time. Simulated intense grazing disturbances also appear to intensify cryoturbic disturbances and erosion, as the surface level decline suggests. Continuous grazing does thus appear to be able to drive the T4 towards the SES dominating state, hence pushing the ecosystem across the ecological threshold present in the T4 transition. The deforestation may thus have had greater consequences than appears at first. It is worth emphasizing that the woodlands differ from the other plant communities in several ways. Seedling growth was low in the woodlands and seedling mortality was high. Tree regeneration would thus have been low compared to other community types. Woodlands also seem to have lower resilience than the other plant communities. Deforested areas may thus be very sensitive for grazing disturbances.

Land use is thus likely to have been a factor in triggering land degradation and erosion following the arrival of man to Iceland 1100 years ago. The results emphasize

the importance of good land management, and how important it is to understand the underlying ecosystem processes. A land manager armed with such knowledge will be able to confront new challenges, avoid undesirable thresholds and prevent costly degradation from occurring.

Our understanding of the degradation processes, especially the initial stages before an ecological threshold has been crossed, is critical for sustainable land use and restoration of degraded areas. Future studies should emphasize on quantifying the variables, which drive state transitions. That would provide land managers with information they need to improve land use and develop effective restoration and land management plans. Further research should also focus on improving landscape classification methods using suitable data and metrics, and how they relate to the degradation processes.

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## APPENDIX A

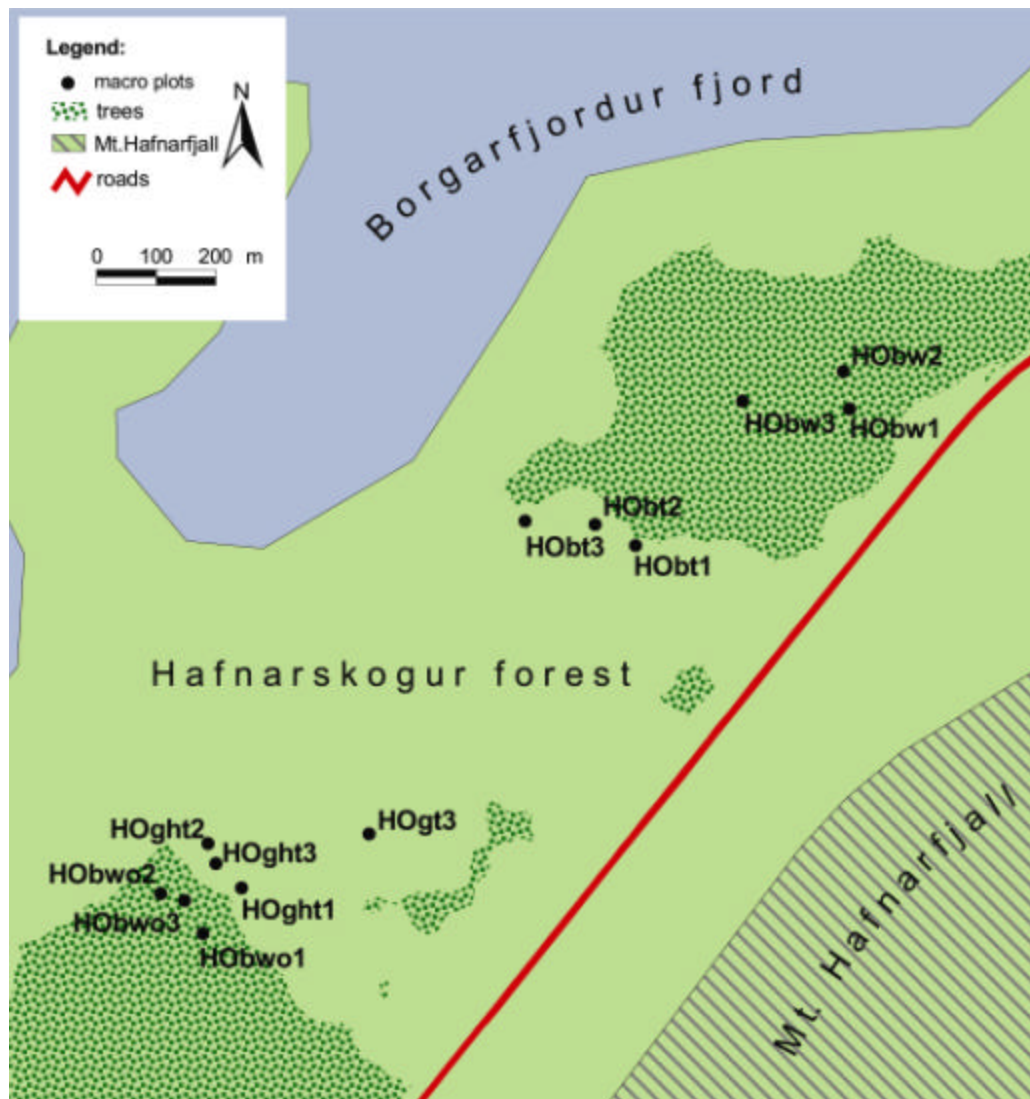
## HAFNARSKOGUR FOREST PLOT DESCRIPTIONS

Thirteen  $10 \times 10$  m plots (macro plots) were established in the Hafnarskogur within the five plant community sub types defined in the area: birch (*Betula pubescens* Ehrh.) woodlands, woodland heathlands, grasslands, birch savanna heathlands and birch savannas. In each of these community types, three macro plots were established, except for the grassland type, which only had one plot. Within each macro plots, twelve  $0.5 \times 0.5$  m subplots (micro plots) were installed with three treatments; three clipped, three trampled, three clipped and trampled and three controls, a total of 12 micro plots. Both location and treatments were assigned randomly for each subplot.

This section contains descriptions of these thirteen macro plots. Refer to Figure A.1 below for a map of the area and plot locations.

The following notation is used in this appendix (*X* represents numbers from 1 - 3):

Plot IDs	Community type
HObw <i>X</i>	woodlands
HObt <i>X</i>	woodland heathlands (w
HOgt <i>X</i>	grasslands
HOght <i>X</i>	savanna heathlands (s
HObwo <i>X</i>	savannas



**FIGURE A.1.** Plot locations in Hafnarskogur. HObw1, 2 and 3 are in the birch woodlands, HOgt1, 2 and 3 are in the grasslands, HOgt3 is in the sgrasslands, HOgt1, 2 and 3 are in the sgrasslands, and HObwo1, 2 and 3 are in the open savannas.



**Plot ID:** HObw1  
**Community type:** Woodlands

**Location:** Borgarfjordur, West Iceland  
64°30'55.4N; 21°55'31.2W, approx. 30 m a.s.l.

**Description:** Dense birch woodland, tree density about 0.44 m<sup>-2</sup>, average height 2.3 m. Canopy cover per tree 29%, mostly continuous. Ground is level, but hummocky. Well drained. This area has a long history of sheep grazing until about ten years ago.

**Vegetation:** *Betula pubescens* dominates the overstory. Ground cover is dominated by *Deschampsia flexuosa*, *Agrostis capillaris*, *Gymnocarpium dryopteris* and *Anthoxanthum odoratum*. *Hylocomium* sp. moss is common.



**FIGURE A.2 A. Overview of the HObw1 plot, facing SE.**





**FIGURE A.2 B.** A schematic drawing of the HObw1 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HObw2
<b>Community type:</b>	Woodlands
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'58.7N; 21°55'33.7W, approx. 30 m a.s.l.
<b>Description:</b>	Moderately dense birch woodland, tree density about 0.32 m <sup>-2</sup> , average tree height 2.8 m. Canopy cover per tree 28%, mostly continuous. Ground is level, but hummocky. Well drained. The area has a long history of sheep grazing until about ten years ago.
<b>Vegetation:</b>	<i>Betula pubescens</i> dominates the overstory. Ground cover is dominated by <i>Deschampsia flexuosa</i> , <i>Agrostis capillaris</i> , <i>Gymnocarpium dryopteris</i> and <i>Anthoxanthum odoratum</i> . <i>Hylocomium</i> sp. moss is common.



**FIGURE A.3 A. Overview of the HOBw2 plot, facing SE.**



**FIGURE A.3 B.** A schematic drawing of the HObw2 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.



**Plot ID:** HObw3  
**Community type:** Woodlands

**Location:** Borgarfjordur, West Iceland  
64°30'55.3N; 21°55'54.0W, approx. 25 m a.s.l.

**Description:** Dense birch woodland, tree density about 0.49 m<sup>-2</sup>, average tree height 2.4 m. Canopy cover per tree 31%, mostly continuous. Ground is level, but hummocky. Well drained. The area has a long history of sheep grazing until about ten years ago.

**Vegetation:** *Betula pubescens* dominates the overstory. Ground cover is dominated by *Deschampsia flexuosa*, *Agrostis capillaris*, *Gymnocarpium dryopteris* and *Anthoxanthum odoratum*. *Hylocomium* sp. moss is common.



**FIGURE A.4 A.** Overview of the HObw3 plot, facing SE.



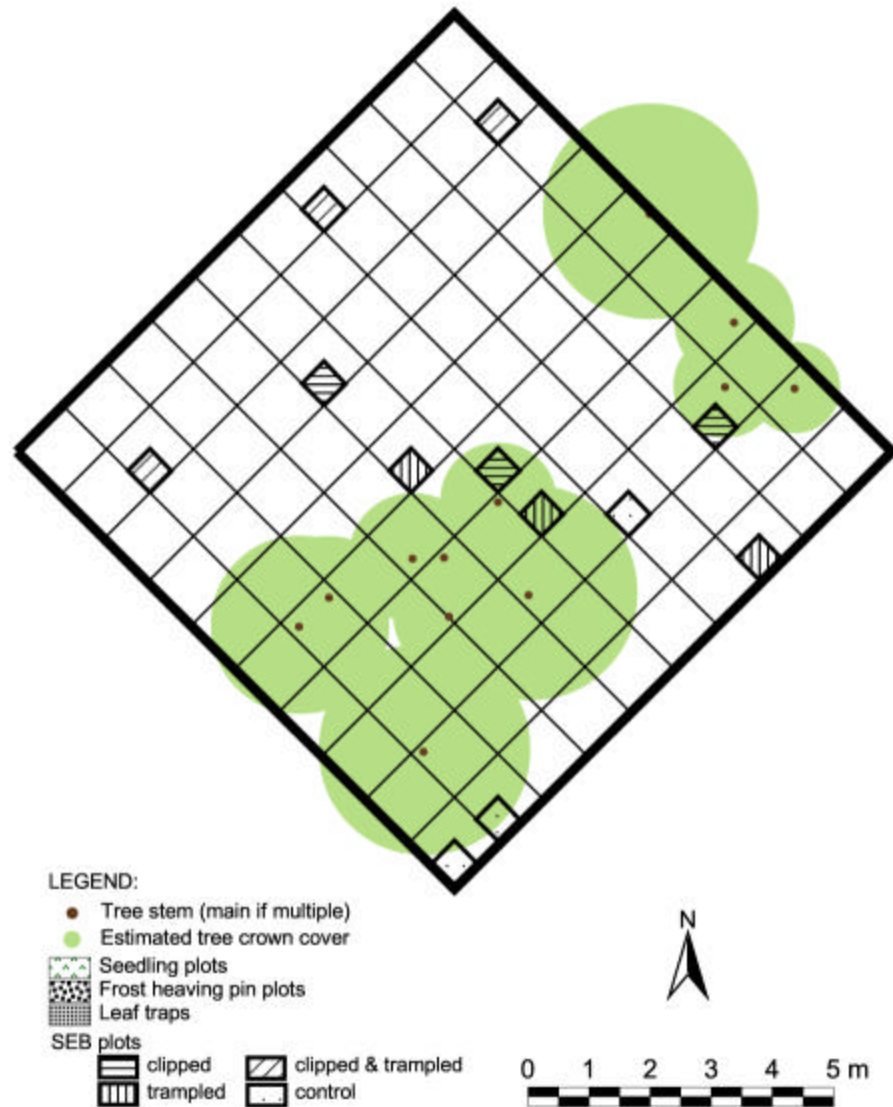
**FIGURE A.4 B.** A schematic drawing of the HObw3 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HObt1
<b>Community type:</b>	Woodland heathlands (w heathlands)
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'42.1N; 21°55'75.1W, approx. 30 m a.s.l.
<b>Description:</b>	Heathland with small birch shrubs, West of the birch woodlands. Shrub density about 0.12 m <sup>-2</sup> , average height 0.6 m. Canopy cover per shrub 37%, continuous where present. Approximately 5% slope, facing East, moderately hummocky. Well drained. The area has a long history of sheep grazing until about ten years ago.
<b>Vegetation:</b>	<i>Empetrum nigrum</i> and <i>Deschampsia flexuosa</i> dominate the community, followed by <i>Vaccinium uliginosum</i> and <i>Agrostis capillaris</i> . <i>Betula pubescens</i> present. <i>Racomitrium</i> sp. moss is common.



**FIGURE A.5 A.** Overview of the HObt1 plot, facing SE.





**FIGURE A.5 B.** A schematic drawing of the HObt1 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes.

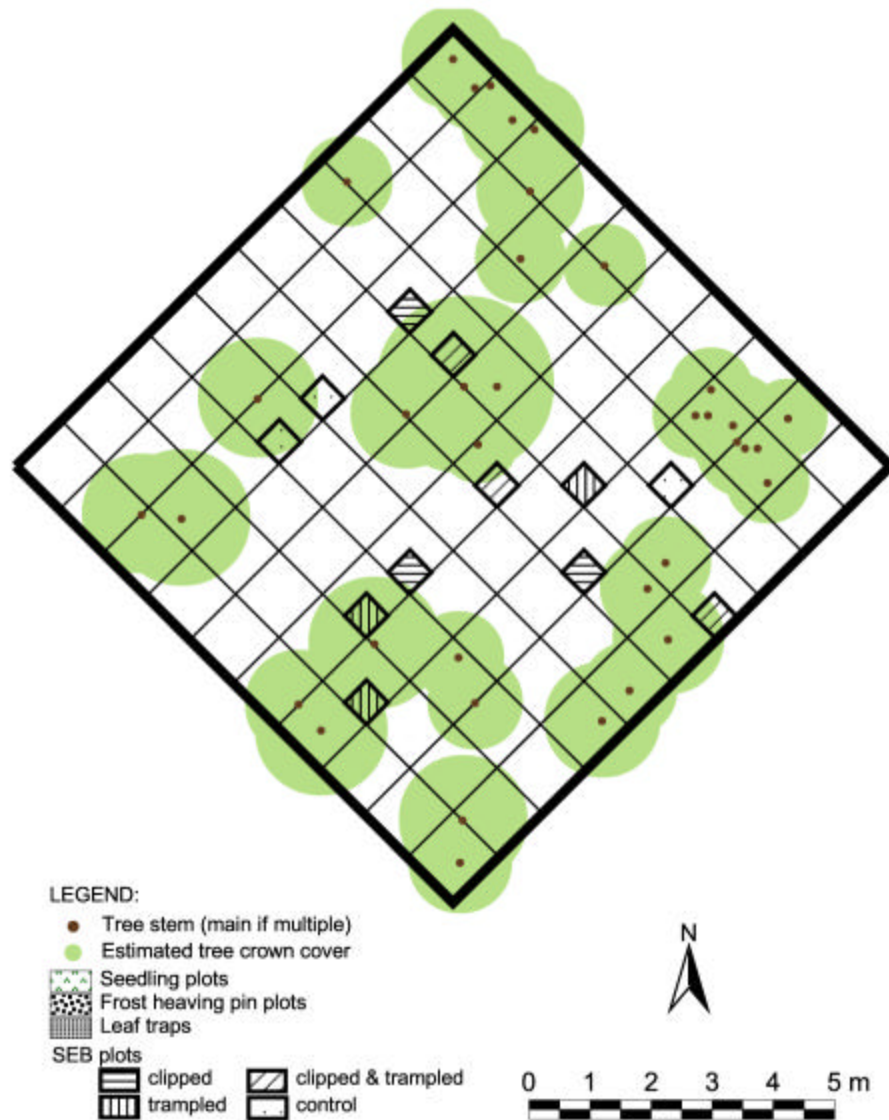
Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HObt2
<b>Community type:</b>	Woodland heathlands (w heathlands)
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'43.8N; 21°55'84.2W, approx. 25 m a.s.l.
<b>Description:</b>	Heathland with small birch shrubs, West of the birch woodlands. Shrub density about 0.37 m <sup>-2</sup> , average height 0.4 m. Canopy cover per shrub 36%, continuous where present. Approximately 15% slope, facing East, moderately hummocky. Well drained. The area has a long history of sheep grazing until about ten years ago.
<b>Vegetation:</b>	<i>Empetrum nigrum</i> and <i>Deschampsia flexuosa</i> dominate the community, followed by <i>Vaccinium uliginosum</i> and <i>Agrostis capillaris</i> . <i>Betula pubescens</i> present. <i>Racomitrium</i> sp. moss is common.



**FIGURE A.6 A.** Overview of the HObt2 plot, facing SE.





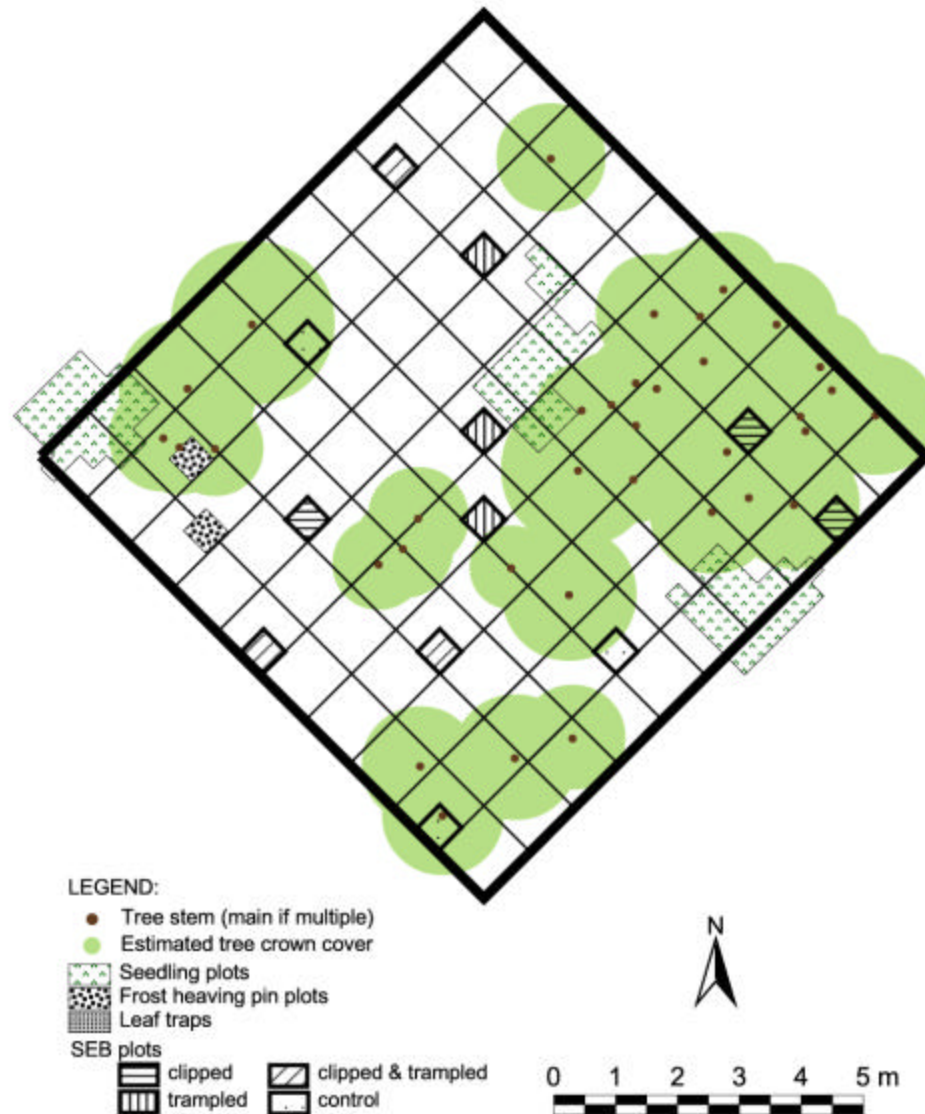
**FIGURE A.6 B.** A schematic drawing of the HObt2 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes.

Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HObt3
<b>Community type:</b>	Woodland heathlands (w heathlands)
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'43.5N; 21°55'99.1W, approx. 25 m a.s.l.
<b>Description:</b>	Heathland with small birch shrubs, West of the birch woodlands. Shrub density about 0.36 m <sup>-2</sup> , average height 0.4 m. Canopy cover per shrub 39%, continuous where present. The surface is level and hummocky. Well drained. The area has a long history of sheep grazing until about ten years ago.
<b>Vegetation:</b>	<i>Empetrum nigrum</i> and <i>Deschampsia flexuosa</i> dominate the community, followed by <i>Vaccinium uliginosum</i> and <i>Agrostis capillaris</i> . <i>Betula pubescens</i> present. <i>Racomitrium</i> sp. moss is common.



**FIGURE A.7 A.** Overview of the HObt3 plot, facing SE.



**FIGURE A.7 B.** A schematic drawing of the HObt3 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes.

Note that not all micro plot types are present in all macro plots.

**Plot ID:** HOgt3  
**Community type:** Grasslands

**Location:** Borgarfjordur, West Iceland  
64°30'14.6N; 21°56'28.2W, approx. 50 m a.s.l.

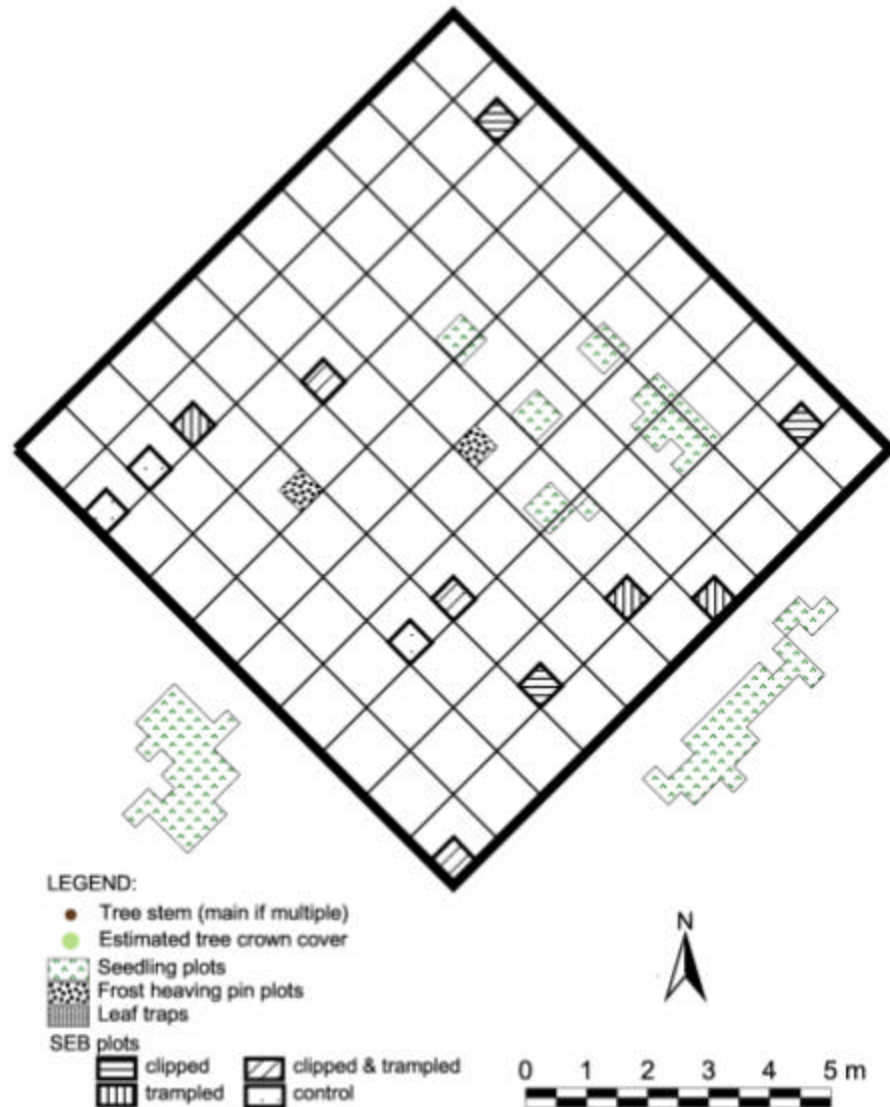
**Description:** Rich grassland in the middle of the experimental site. No trees or shrubs present. Surface is very hummocky and slopes gently towards North (2%). Well drained and south of the wetlands in the area. The area has a long history of sheep grazing until about ten years ago.

**Vegetation:** *Agrostis capillaris*, *Deschampsia caespitosa* and *Taraxacum* spp.



**FIGURE A.8 A.** Overview of the HOgt3 plot, facing SE.





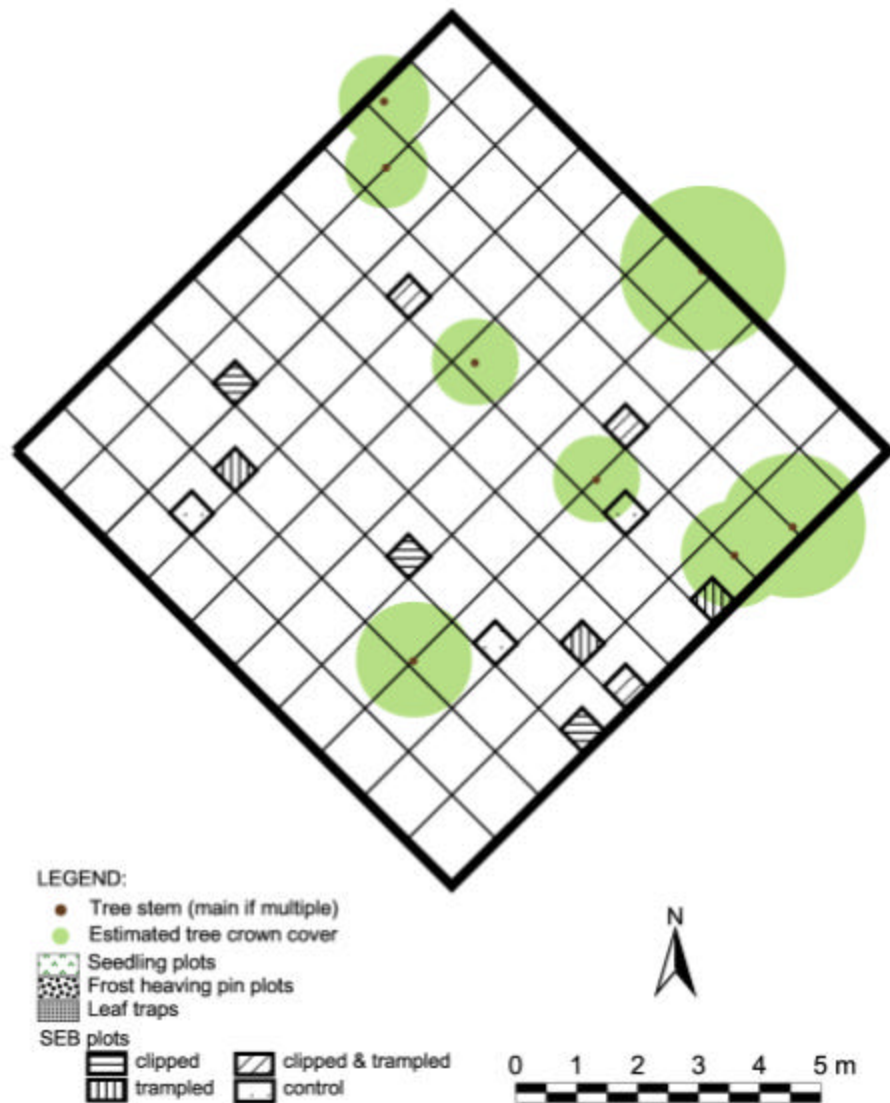
**FIGURE A.8 B.** A schematic drawing of the HOgt3 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes.

Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HOght1
<b>Community type:</b>	Savanna heathlands (s heathlands)
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'09.2N; 21°56'54.9W, approx. 40 m a.s.l.
<b>Description:</b>	Open grassland just East of the open birch woodland community. Few birch trees and shrubs are present. Density is 0.08 m <sup>-2</sup> , average height 0.6 m. Canopy cover per tree 21%. Surface is moderately hummocky and slopes gently towards North (5%). Well drained. The area has a long history of sheep grazing until about ten years ago.
<b>Vegetation:</b>	<i>Agrostis capillaris</i> , <i>Carex bigelowii</i> , <i>Festuca richardsonii</i> and <i>F. vivipara</i> . <i>Racomitrium</i> sp. mosses are common.



**FIGURE A.9 A.** Overview of the HOght1 plot, facing SE.



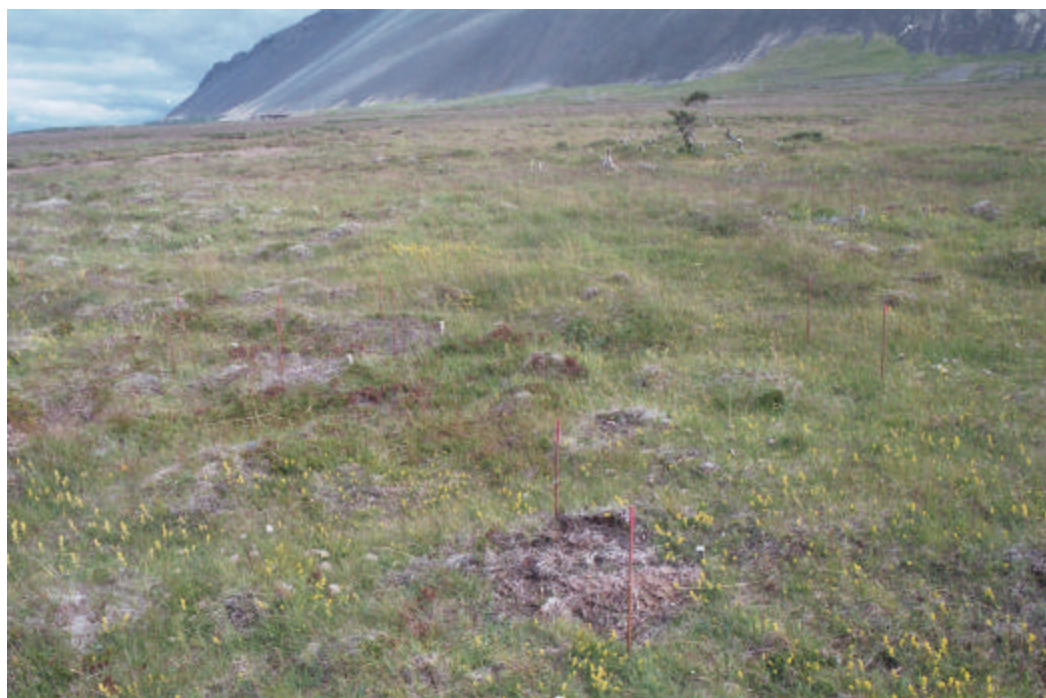
**FIGURE A.9 B.** A schematic drawing of the H0ght1 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

**Plot ID:** HOght2  
**Community type:** Savanna heathlands (s heathlands)

**Location:** Borgarfjordur, West Iceland  
64°30'13.3N; 21°56'62.0W, approx. 40 m a.s.l.

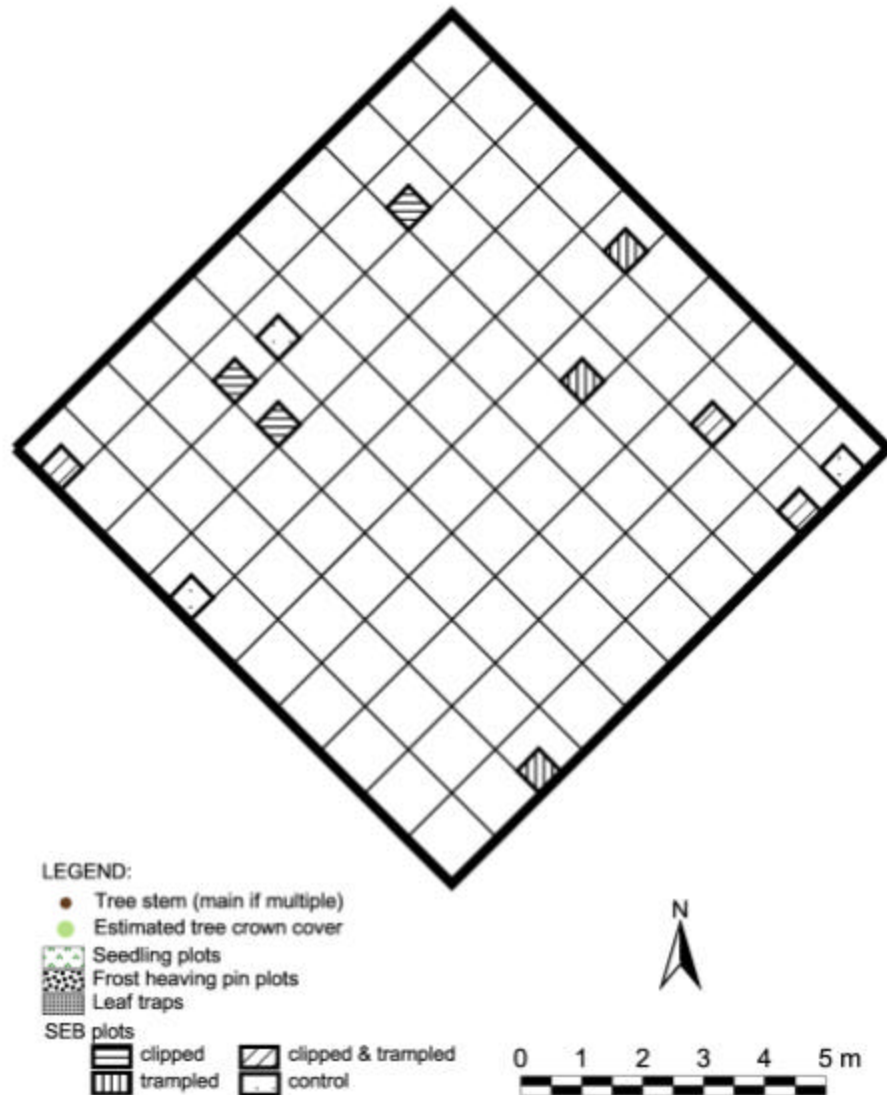
**Description:** Open grassland just East of the open birch woodland community. Surface is moderately hummocky and slopes gently towards North (5%). Well drained. The area has a long history of sheep grazing until about ten years ago.

**Vegetation:** *Agrostis capillaris*, *Carex bigelowii*, *Festuca richardsonii* and *F. vivipara*. *Racomitrium* sp. mosses are common.



**FIGURE A.10 A.** Overview of the HOght2 plot, facing SE.



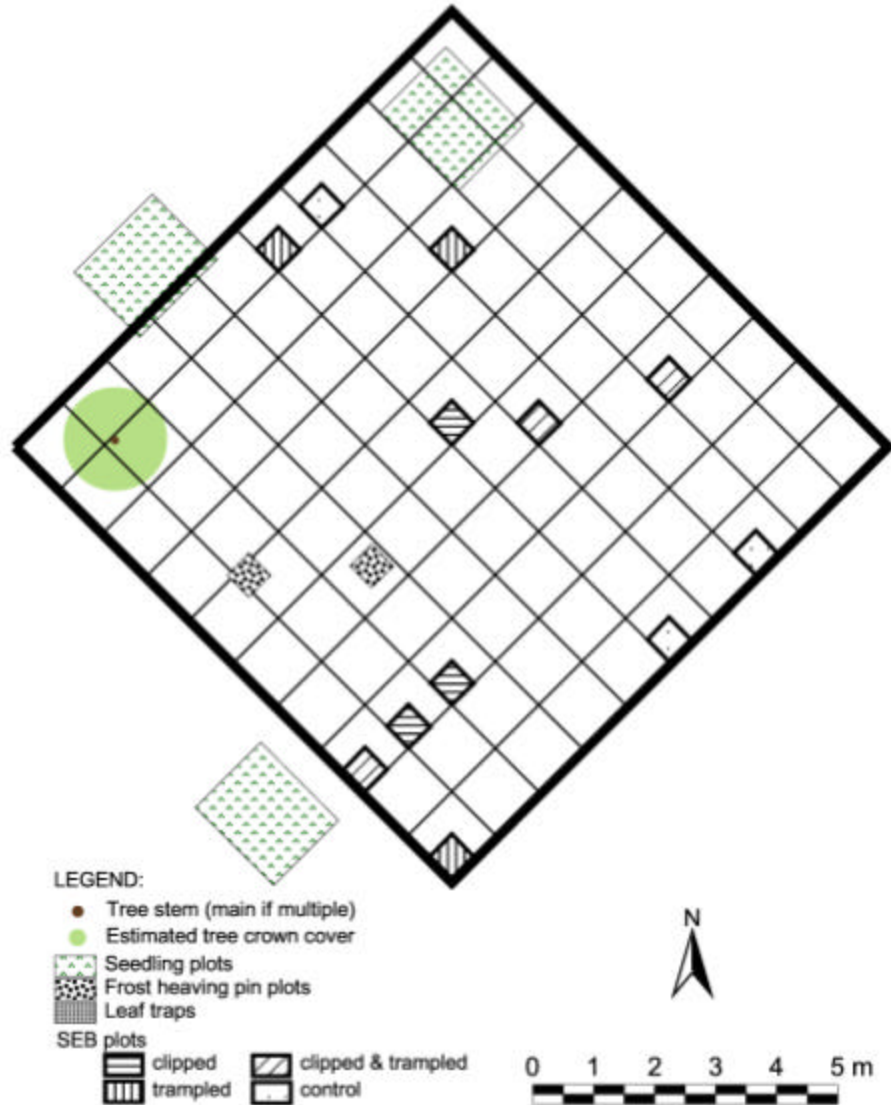


**FIGURE A.10 B.** A schematic drawing of the HOght2 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HOght3
<b>Community type:</b>	Savanna heathlands (s heathlands)
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'11.3N; 21°56'60.3W, approx. 40 m a.s.l.
<b>Description:</b>	Open grassland just East of the open birch woodland community. One birch shrub present, 0.4 m tall. Canopy cover is shrub 20%. Surface is moderately hummocky and slopes gently towards North (5%). Well drained. The area has a long history of sheep grazing until about ten years ago.
<b>Vegetation:</b>	<i>Agrostis capillaris</i> , <i>Carex bigelowii</i> , <i>Festuca richardsonii</i> and <i>F. vivipara</i> . <i>Racomitrium</i> sp. mosses are common.



**FIGURE A.11 A. Overview of the HOght3 plot, facing SE.**



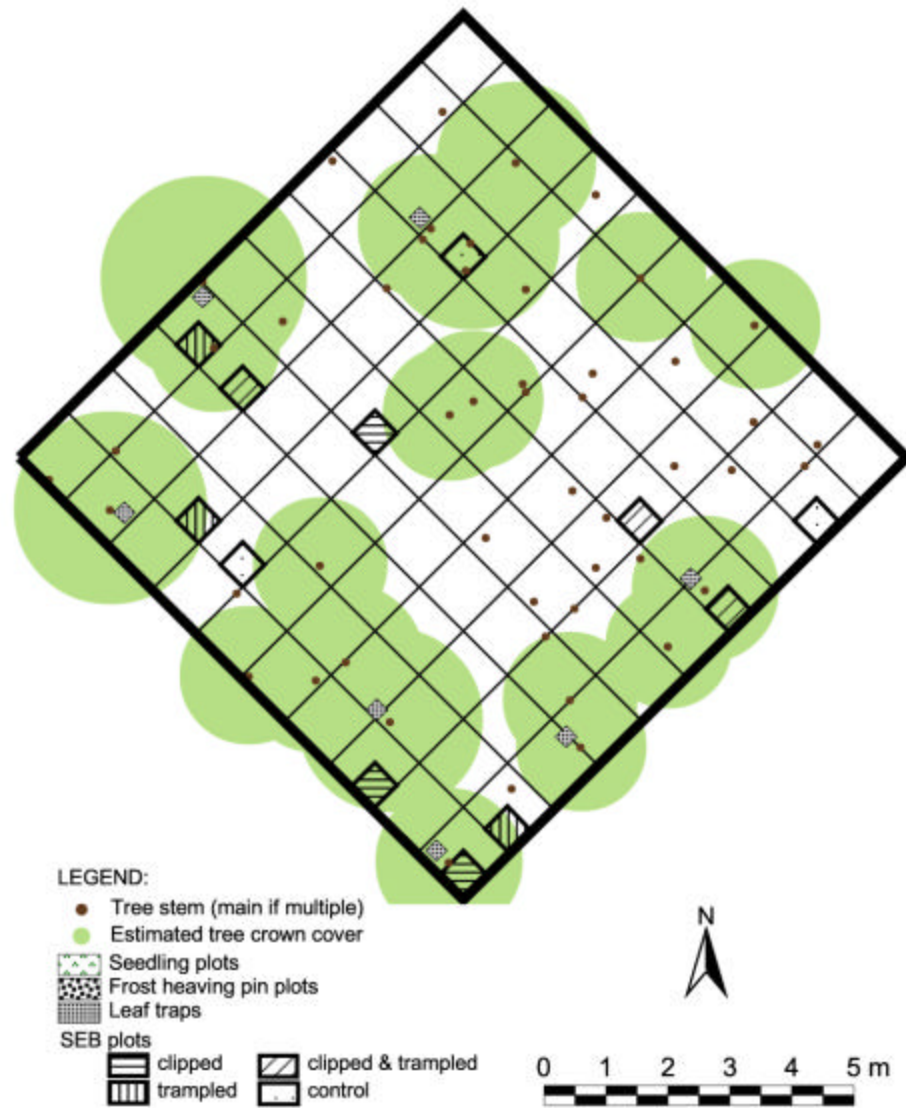
**FIGURE A.11 B.** A schematic drawing of the HOght3 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HObwo1
<b>Community type:</b>	Savannas
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'05.1N; 21°56'62.4W, approx. 30 m a.s.l.
<b>Description:</b>	Open birch woodland with considerable grass undergrowth. Birch shrubs and dead trees are common. Live tree density is about 0.20 m <sup>-2</sup> , average height 1.4 m. Canopy cover per tree is 58%, mostly continuous. Ground is level, but hummocky. Well drained. This area has been excluded from grazing for the last 20 years.
<b>Vegetation:</b>	<i>Betula pubescens</i> dominates the overstory as in the birch woodlands, but the trees are much scarcer and smaller. Ground cover is dominated by <i>Deschampsia flexuosa</i> , <i>Agrostis capillaris</i> and <i>Anthoxanthum odoratum</i> . <i>Hylocomium</i> sp. moss is common.



**FIGURE A.12 A. Overview of the HObwo1 plot, facing SE.**





**FIGURE A.12 B.** A schematic drawing of the HObwo1 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

<b>Plot ID:</b>	HObwo2
<b>Community type:</b>	Savannas
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'08.3N; 21°56'71.9W, approx. 30 m a.s.l.
<b>Description:</b>	Open birch woodland with considerable grass undergrowth. Birch shrubs and dead trees are common. Live tree density is about 0.33 m <sup>-2</sup> , average height 1.0 m. Canopy cover per tree is 97%, mostly continuous. Ground is level, but hummocky. Well drained. This area has been excluded from grazing for the last 20 years.
<b>Vegetation:</b>	<i>Betula pubescens</i> dominates the overstory as in the birch woodlands, but the trees are much scarcer and smaller. Ground cover is dominated by <i>Deschampsia flexuosa</i> , <i>Agrostis capillaris</i> and <i>Anthoxanthum odoratum</i> . <i>Hylocomium</i> sp. moss is common.



**FIGURE A.13 A.** Overview of the HObwo2 plot, facing SE.



**FIGURE A.13 B.** A schematic drawing of the HObwo2 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

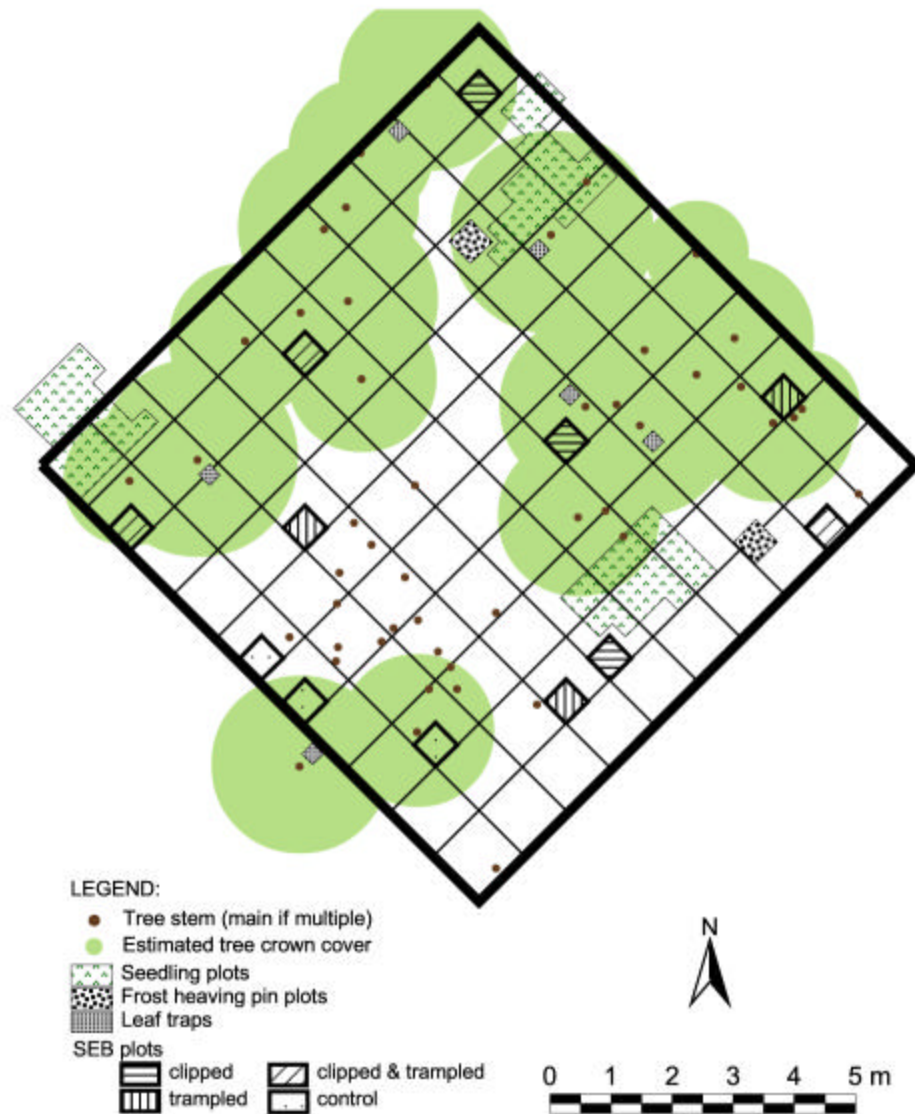


<b>Plot ID:</b>	HObwo3
<b>Community type:</b>	Savannas
<b>Location:</b>	Borgarfjordur, West Iceland 64°30'07.9N; 21°56'66.2W, approx. 30 m a.s.l.
<b>Description:</b>	Open birch woodland with considerable grass undergrowth. Birch shrubs and dead trees are common. Live tree density is about 0.24 m <sup>-2</sup> , average height 1.1 m. Canopy cover per tree is 79%, mostly continuous. Ground is level, but hummocky. Well drained. This area has been excluded from grazing for the last 20 years.
<b>Vegetation:</b>	<i>Betula pubescens</i> dominates the overstory as in the birch woodlands, but the trees are much scarcer and smaller. Ground cover is dominated by <i>Deschampsia flexuosa</i> , <i>Agrostis capillaris</i> and <i>Anthoxanthum odoratum</i> . <i>Hylocomium</i> sp. moss is common.



**FIGURE A.14 A.** Overview of the HObwo3 plot, facing SE.





**FIGURE A.14 B.** A schematic drawing of the HObwo3 macro plot, showing location of micro plots (SEB plots), tree stems, seedling plots, frost heaving pin plots and leaf traps. Crown cover is estimated based on maximum crown diameter and is thus overestimated in this figure due to their irregular shapes. Note that not all micro plot types are present in all macro plots.

## VITA

Johann Thorsson obtained his B.S. degree in biology from the University of Iceland in 1990. As an undergraduate, he was involved in assessing condition and extension of wetlands in south Iceland. He was a research scientist at the Agricultural Research Institute in Reykjavik from 1990 to 1998, where he was involved in research on animal grazing, land use and land reclamation. He received his Doctor of Philosophy degree in December 2008. His research interests include processes active in early land degradation stages.

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