Effects of environmental factors on inducing soil erosion and shaping the plant community in Icelandic habitats

Sóldögg Rán Davíðsdóttir

Master's project in Conservation Biology
60 credits
BION02
Faculty of Science
Department of Biology
Lund University

Supervisor: Bryndís Marteinsdóttir
Co-supervisor: Pål Axel Olsson

Abstract

Land degradation and soil erosion are considered one of the main threats to ecosystem services and functions, soil stability and activity, plant community structure and biodiversity, and overall life on Earth. Soil erosion is a problem occurring all over Iceland, with an immediate urgency on the Icelandic highlands and rangelands. Currently, over 40% of the country is eroded, steadily causing loss of vegetation cover with imminent consequences for species, structure and productivity. Erosion is caused by environmental and anthropogenic factors, such as climate change with changing precipitation patterns and temperature regimes, distribution of invasive species, and unsustainable land use and agricultural practices. This research project uses data from an extensive vegetation and soil monitoring scheme in Iceland to examine the effects of specific environmental factors on controlling the level of soil erosion in various natural habitats in Iceland, and on how soil erosion may shape the environment and plant community structure. The aim was to examine which factors induce erosion, how erosion influences species richness and community composition, and how two different methods for estimating erosion compare when evaluating the severity of erosion levels in Icelandic habitats, one being a calculated measurement, erosion %, and the other a visual estimation, erosion level. Elevation, habitat type, and soil type all significantly affected erosion. Furthermore, erosion significantly affected species richness and the number of plant functional groups present within an area. The two erosion estimation methods provided similar results. However, erosion % provided more precise information regarding the actual erosion of each plot, allowing for a more accurate representation of changes over time, whereas erosion level is less efficient in observing changes over time, whilst being an accurate representation of the severity of erosion at a specific time. This research provided novel and valuable information for an ongoing long-term monitoring project, confirming the accuracy of the methods being performed and highlighting the importance of evaluating and monitoring the level or soil erosion in natural Icelandic habitats.

Acknowledgements

I would like to express my gratitude towards my supervisors for their involvement, expertise, valuable feedback, and immense support throughout this research project, as well as further increasing my interest in the subject. Additionally, I would like to thank Land and Forest Iceland, previously the Soil Conservation Service of Iceland, for their collaboration and providing me with extensive amount of data and information needed for this project, and Lund University for giving me the opportunity to see this thesis through.

I would like to extend my sincere thanks to my classmates and teachers throughout my studies, as well as my co-workers and other research assistants that helped me gather all the necessary data I needed. I would also like to acknowledge my family and friends for any further assistance, editing help, moral support, and always keeping my spirits up and motivation high during the whole process.

Table of contents

1. Introduction

Soil is considered to be one of the fundamental factors of all life on Earth. It provides the bases for various energy and nutrient cycles and is an important resource for all organisms and their ecosystems (Arnalds, 2015). Most of the terrestrial ecosystems of the world are based on soil, which is one of the main factors maintaining ecosystem functions. A healthy soil is essential for the sustainability and productivity of ecosystem services and a key factor when it comes to sustaining biodiversity within habitats (Stockmann *et al.*, 2015). The soil contains one of the most diverse and complex communities ranging from micro-niches to microorganisms in entire landscapes playing major roles in nutrient cycling, and interacting with above ground vegetation (Dubey *et al.*, 2019). Therefore, it is very important to maintain and improve soil properties in order to sustain healthy ecosystems and habitats, preserve biodiversity and environmental quality (Blanco-Canqui *et al.*, 2013).

Large repositories of carbon can be found in the soil, which makes it one of the most fundamental carbon storages on Earth (Carter & Stewart, 1995). The earth's soil can store up to a few thousand gigatons (Gt) of organic carbon (Lal *et al.*, 1997). This enormous amount of carbon can increase the level of atmospheric carbon dioxide (CO_2) if released, *e.g.,* as a consequence of soil erosion and poor land use. However, storage of organic carbon in soil can also reverse the impact of greenhouse effects with proper use of land, natural resources and with less strain on soils in various areas (Lal *et al.*, 1997). The quality of soil can be determined by cohesion abilities, productivity, and general function within the environment (Doran *et al.*, 1996). Soils can be classified according to their dominant particle size. Soil consists of and is constructed from soil particles of various sizes, such as sand, silt and clay (Finch *et al.*, 2014).

Land degradation is the decline in land quality through processes driven mainly by human activities (Bridges, 2001). The degradation processes include erosion, compaction, reduction in soil organic matter, landslides, salinization, contamination, and biodiversity loss (Montanarella, 2016). Generally, the degradation processes are initiated by formation of isolated erosion areas that result in fragmentation of the vegetation cover (Magnusson, 1997). Land degradation can be the result of a mismatch between land quality and land use. Mechanisms initiating land degradation include chemical processes such as nutrient depletion, physical processes such as erosion, and biological processes such as overgrazing and loss of diversity (Bridges, 2001).

Soil erosion is one of the main indicators for land degradation (Arnalds *et al.*, 2001b). Drylands all over the world are being heavily affected by erosion as a result of alterations in vegetation cover, plant community composition, hydrologic cycles, and overall soil characteristics (D'Odorico *et al.*, 2013). External environmental forces can initiate chemical weathering, where minerals degrade through chemical reactions upon exposure to water and other substances (Macheyeki *et al.*, 2020). These changes, *e.g.,* caused by climate change and shifts in temperature and precipitation patterns, lead to a decline in ecosystem function and the services they provide, which is the basis of a sustainable life (D'Odorico *et al.*, 2013). Furthermore, alterations in soil condition can impact its biological activity and structure (Arnalds, 2015).

Soil erosion is driven by climatic and environmental factors, alongside anthropogenic influence, *e.g.*, grazing and agricultural practices (Montanarella, 2016). Soil erosion has been changing ecosystem functioning and the appearance of landscapes at a concerning rate over the past decades and is today considered to be one of the most serious threats to life on earth (Arnalds *et al.*, 2001a). Deforestation, overgrazing, and excessive land use has been leading to the degradation of ecosystems with loss of biological productivity and diversity, resulting in severe erosion and desertification on a global scale. In 1996, the United Convention to Combat Desertification (UNCCD) was formed to raise awareness about ongoing erosion and try to combat further impact (Zonn *et al.*, 2017).

Soil erosion causes loss of soil stability due to the removal of the amount of soil organic matter (SOM) and other organic nutrients being most prominent in the upper layers of the soil profile (Arnalds *et al.*, 2001b), in addition to loss of important soil functions within the ecosystem increasing with induced erosion (Oskarsson *et al.*, 2004). Soil erosion reduces the soil's water holding capacity and accumulation of nutrients, inhibiting regeneration and succession of vegetation (Jiao *et al.*, 2009). Erosion further destroys plant root systems due to geomorphological processes, resulting in reduced root formation, seed retention, and overall plant establishment. Seed germination and establishment are key factors for the growth and development of plants. Destress to these factors can severely impact the plant community structure and population (Jiao *et al.*, 2009). When land lacks sufficient vegetation cover to protect its soil, it becomes susceptible to erosion by wind, precipitation, and water. The potential for soil regeneration then hinges on the intensity and frequency of natural erosion forces. It can take decades and even centuries for eroded land to regain its former function and ecosystem services again (Arnalds *et al.*, 2001a).

Environmental monitoring can provide important data, which is critical for transforming land use policies to mitigate environmental threats. Environmental monitoring depend on informative measurable parameters to estimate the state, trends, and conditions of habitats (Lovett *et al.*, 2007). Vegetation data can be used to disentangle the different environmental drivers across various gradients, as well as analyzing differences within and between plant groups (Wiegmann & Waller, 2006). Plant communities can also provide valuable information regarding shifts and changes in species composition over a certain period of time, *e.g.,* as a response to environmental changes or external pressures (Wiegmann & Waller, 2006). Based on the current understanding of vegetation composition and characteristics in Nordic regions, the fundamental environmental factors and drivers are light availability, soil characteristics, *e.g.,* soil pH and texture, moisture and levels of disturbance and erosion (Tyler *et al.*, 2021).

The ecological niches of plant species can differ across their distribution areas and environmental conditions and some species tend to have narrow ecological niches while others much wider (Wasowicz *et al.*, 2013). With rapidly changing climate, soil erosion and various land-use practices, there is an increase in the general loss of ecosystem services and biodiversity (Tyler *et al.*, 2018). Therefore, the need for further research regarding environmental factors and indicator values, as well as edaphic and climatic factors, is becoming more important when understanding the processes shaping vegetation composition within habitats (Tyler *et al.*, 2018).

This thesis will focus on some of the effects of specific environmental factors on inducing the level of soil erosion in various natural habitats in Iceland, and how soil erosion may shape the environment, plant community, and diversity. The aim of this study was to examine how soil erosion in Iceland is influenced by various environmental variables and how erosion effect plant community structure in Iceland. In addition, it compared the two methods used in GróLind, a long-term monitoring project, for estimating soil erosion. The specific questions asked were:

i) How do different measurements of erosion compare when estimating land degradation in Iceland?
ii) Which measured environmental factors are inducing soil erosion in Icelandic habitats?
iii) How does soil erosion affect and influence the plant community structure, species composition and diversity in Icelandic habitats?

1.1. Icelandic habitats & vegetation

The Arctic is defined as an area within the Arctic circle, located north of the potential treeline, about 66.5° north of the Equator (Walker *et al*., 2005). It is characterized by a cold climate and short growing seasons (Walker *et al.*, 2008), low productivity with reduced mineralization processes and slow decomposition rates (Callaghan *et al.*, 2005). Environmental conditions within Arctic regions create microclimates that make up the structure of diverse ecosystems, various habitats and their corresponding functions (ACIA, 2004). Ecosystems are biological systems composed of all the various organisms present within a particular physical environment, interacting not only with their environment but also among each other (Tsujimoto *et al.*, 2018). A habitat refers to those areas that are utilized by specific organisms that meet all the environmental conditions they may need to survive and reproduce (Fath, 2019). For vegetation, habitats must be able to provide the plant communities with a suitable combination of light source, shelter, water regulation, air quality, and good nutritious soil (Ottosson *et al.*, 2016).

Iceland is a volcanically active island located in the sub-Arctic where the climate has been described as oceanic-subarctic (Olafsson *et al.*, 2007). The northern peninsulas of the country are considered to be a part of the Arctic and the rest falls under the Subarctic region (Meltofte, 2013). Furthermore, Iceland is located where the North Atlantic current of the Gulf stream meets the cold air from the polar systems, resulting in a cold temperate, yet humid climatic environment (Einarsson, 1976). The weather regimes are characterized by strong winds, frequent precipitation, mild winters and relatively cold summers (Olafsson *et al.*, 2007). The growing season in Iceland usually last from the beginning of May till the end of August and beginning of September (Leblans *et al.*, 2017). Approximately a third of the country's area is located above 600 m and a quarter below 200 m above sea level (Einarsson, 1976). The mean annual temperatures range from 0°C to -3°C during winter in the lowlands (below 400 m above sea level) and overall, the average temperature is about -5°C. During the summer season, the temperatures in lowland areas range from 8°C to 10°C with the overall average being 7°C (Bjornsson, 2007).

Around 40% of the area of Iceland is considered vegetated, thereof, roughly 1% being occupied by forest areas, about 15% is covered in glaciers, waterbodies and man-made surfaces, and around 45% is estimated to have little or no vegetation (Arnalds, 2015; Bjornsson, 2016). The vegetated areas are dominated by heathlands and grasslands. Mosses and a variety of lichens also dominate in various areas, *e.g.*, extensive lava fields (Thorhallsdottir, 1991). Unvegetated areas are most commonly present in the highlands above elevations of 700 m, which delimit where a continuous cover of vegetation can be found. Great parts of the highland areas are composed of sub-arctic deserts characterized by scattered vegetation patches and/or singular plant that overall cover about 2-5% of the total surface of the highland areas (Thorhallsdottir, 1991).

Based on the EUNIS-classification system, which describes all habitats within Europe (Chytrý *et al.*, 2020; EEA, 2024), 105 habitat types have been described in Iceland, with 64 of them being terrestrial (Ottosson *et al.*, 2016). Terrestrial habitats refer to the non-aquatic and natural habitats located more inland from the main coastlines (Beraldi-Campesi, 2013). The different types of terrestrial habitats found in Iceland are mainly categorized depending on their varying levels of vegetation cover, coverage percentage of all main plant groups, cover of individual plant species, mean vegetation height, and soil characteristics, such as soil depth, soil carbon content and pH values (Ottosson *et al.*, 2016). Icelandic terrestrial habitat types range from rich wetlands to dry lava fields, from areas characterized by high geothermal activity to areas surrounding glaciers, and from lowlands to mountainous highlands. The 64 terrestrial habitats have then been further assigned to one of 12 main habitat groups (Ottosson *et al.*, 2016). Many of these habitat groups are subjective to some minor anthropogenic disturbances and can be defined as natural or semi-natural to some extent.

The habitat classification provides an overview of all the various, unique and rare habitats found, as well as their characteristics and general distribution over the country. Habitats can provide information regarding the conservation value of land areas and be important indicators for appropriate land use, *e.g.,* forestry and soil conservation. Due to the unique Icelandic weather and geological conditions some of the Icelandic habitats are not found in other regions (Ottosson *et al.*, 2016).

1.2. Icelandic flora

The Icelandic flora consists of roughly 600 different species of mosses, more than 700 species of lichens, and around 490 species of vascular plants that grow wild in Icelandic nature (Kristinsson, 2010; Thorhallsdottir & Kristinsson, 2019). Of the 490 species, there are 300 dicotyledons, 145 monocotyledons, and around 40 different pteridophytes with 23 of them being various ferns and nine species of clubmoss. There are also four different gymnosperms with only one being native to the country (Kristinsson, 2010). The identification, distribution, mapping and further data regarding the Icelandic flora has been systematically collected since the late 19th century and onwards (Babington, 1871; Grönlund, 1881; Stefansson, 1901; Kristinsson, 2010). Roughly half of all the vascular plant species are commonly found and evenly distributed over the whole country, whilst other species are less common and more bound to specific land areas, landscapes and environmental factors (Kristinsson, 2010).

The Icelandic flora is recognized for its unique features and how it differs from other neighboring areas, such as Greenland and Scandinavia, by possessing an Atlantic European element more prominent than present in other Arctic and Subarctic areas. Most of the taxa forming the flora found in Iceland originate from Europe (Wasowicz *et al.*, 2014). Most of the species (over 45%) are considered boreal; thereafter, Arctic and Boreal-Arctic species account for up to 40% and temperate species are the least abundant, accounting for around 20% of the flora (Elven *et al.*, 2011).

During the Pleistocene period, Iceland was extensively glaciated, resulting in the entire flora being considered to be of a postglacial origin. Therefore, the flora and its evolutionary timeline is ecologically young and, hence lacking endemic species. However, a lot of the taxa present exhibit some morphological differentiation in comparison to their ascendants from Europe (Elven *et al.*, 2011). Some external environmental factors contribute to the unique flora with presence of specific ecological niches, *e.g.,* geothermal areas, volcanic activity and specific soil types, and complex weather, temperature and precipitation patterns. Also, long range bird migrations can introduce new features to the flora (Wasowicz *et al.*, 2014).

1.2.1. Red listed plant species

The International Union for Conservation of Nature (IUCN) provides scientific and quantitative criteria's for assessing the conservation status of species on a global and regional scale in terms of their exposure to threats and risks for extinction (Butchart *et al.*, 2007). The red lists are based on inventories and general judgement by experts that apply the IUCN criteria based on precise data collection and scientific knowledge of the distribution of the specific species, total number of individuals within the species, and population dynamics, population size and density (IUCN, 2022).

The regional red lists covering the threatened species that are a part of the Icelandic biota are compiled by the Icelandic Institute of Natural History (IINH) (Wasowicz & Heiðmarsson, 2019). The most recently published IINH red list for all the vascular plants was released in 2018, with the initial one being published back in 1996 (IINH, 1996). According to the more recent observations, a total of 56 vascular plant species are present on the current red list. One species has been classified as regionally extinct (RE), eight are critically endangered (CR), seven endangered (EN), and 31 vulnerable (VU). The initial red list the IINH presented in 1996 additionally red-listed 74 species of mosses as well as 67 lichen species (IINH, 1996). However, due to insufficient data, ten species in total are classified as data deficient (DD). The IINH has recommended that all species present on the red list are to be protected in Iceland (Wasowicz & Heiðmarsson, 2019).

1.3. Icelandic soil

Icelandic soil has been categorized into four main groups; Andosols, Histosols, Vitrisols and Leptosols (Arnalds, 2004). The soil type that is most dominant in Iceland is Andosol, which is the type of soil formed throughout the years in volcanic areas (Arnalds, 2004). The Andosols unique properties make the Icelandic soil rather distinctive and different from other soil types. One of the properties distinguishing the Andosol from other soil types is its extreme weathering properties, as well as formation of special chemical compounds (Arnalds, 2004).

The Andosol is based on tephra, which weathers at a significant rate in the presence of moisture (Arnalds & Óskarsson, 2009). Tephra is the term for any airborne pyroclastic material which is ejected during an active volcanic eruption, covering expansive land areas (Bradley, 2015). The weathering causes aluminum (Al) and silicon (Si) cations to precipitate with the oxygen (O) and hydroxide (OH^-), making up the clay minerals (Arnalds & Óskarsson, 2009). The main clay minerals found in the Andosol are allophane, imogolite, and halloysite. These clay minerals, as well as other special chemical compounds, provide the Andosol with its unique properties. Some of these properties are the ability to accumulate and conduct water, high ion capacity, lack of cohesion, high carbon storage, and high fertility (Arnalds & Óskarsson, 2009).

Andosols are the predominant soil types in Iceland, and they are also the most prone to erosion (Arnalds *et al.*, 2001b). The volcanic nature of the Andosols have a drastic effect on the soil's resistance to erosion processes. The soil is characterized by soil grains of low density which enables wind to move and erode soil particles that have a diameter of up to 3 cm (Arnalds *et al.*, 2001b). Moreover, particles exceeding a diameter of 0.8 mm are generally not as prone to movement caused by external environmental factors (Skidmore, 1994). However, due to the low density of the Andosols grains, strong winds are able to carry a much greater amount of soil particles around resulting in erosion caused by wind in Iceland being more excessive than in other areas (Arnalds *et al.*, 2001b).

1.4. Land degradation & soil erosion

Desertification has been defined as the lasting decline and degradation of land fertility in arid and semi-arid areas, often attributed to climate change and extensive land use (Arnalds *et al.*, 2001a). Soil erosion in rangelands is one of the main drivers of desertification (Peri *et al.*, 2021). The Icelandic deserts, spanning over 40% of the land area, are distinguished by their dark surfaces that absorb solar heat, resulting in significant evaporation during summer (Arnalds, 2015). These deserts face a chronic deficit in water retention, leading to dry soil conditions even in humid surroundings.

Desertification in Iceland is primarily anthropogenic but also enforced by natural stressors (Arnalds, 2015). Before the Norse settlement in Iceland in the late 8th century there were no large herbivores on the island (Karlsson, 2000). It has been stated that before the settlement a vast part of the land area was covered by a more or less continuous vegetation, over 60%, thereof, 15-40% was forested, mainly by birch (Bergthorsson, 1996). Since then, Icelandic ecosystems have been influenced and formed by improper management practices resulting in land degradation. Iceland have been under intense grazing pressures since the settlement with grazing patterns and practices all year round, which have detrimental effects on the ecosystems during the short growing season of wild vegetation. However, winter and spring grazing was discontinued in the 1970s (Arnalds & Barkarson, 2003).

At the time of Iceland's settlement, native birch woodlands covered approximately a fourth of the country. However, by the early 20th century, these woodlands were nearly eradicated, now spanning only around 1.5% of Iceland's land area (Snorrason *et al.*, 2016). The protection of remaining birch woodlands and the restoration of woodland ecosystems in degraded areas have become key conservation objectives in Iceland (Aradottir & Eysteinsson, 2005). While the extent of birch woodlands has modestly increased in recent decades due to natural regeneration resulting from changes in land use (Snorrason *et al.*, 2016), active restoration efforts remain essential to achieve current restoration targets (Aradottir & Eysteinsson, 2005).

Today, land degradation poses a significant environmental challenge in the Icelandic highlands, which are often utilized for extensive summer grazing for sheep (Barrio *et al.*, 2018). The Icelandic rangelands are very susceptible to extensive and heavy grazing, due to cold temperatures and short growing seasons (Arnalds *et al.*, 2023), which has shaped the Icelandic landscape by affecting the biodiversity within those areas, above and below ground, as well as biomass quantity and quality and soil stability (Arnalds, 2015).

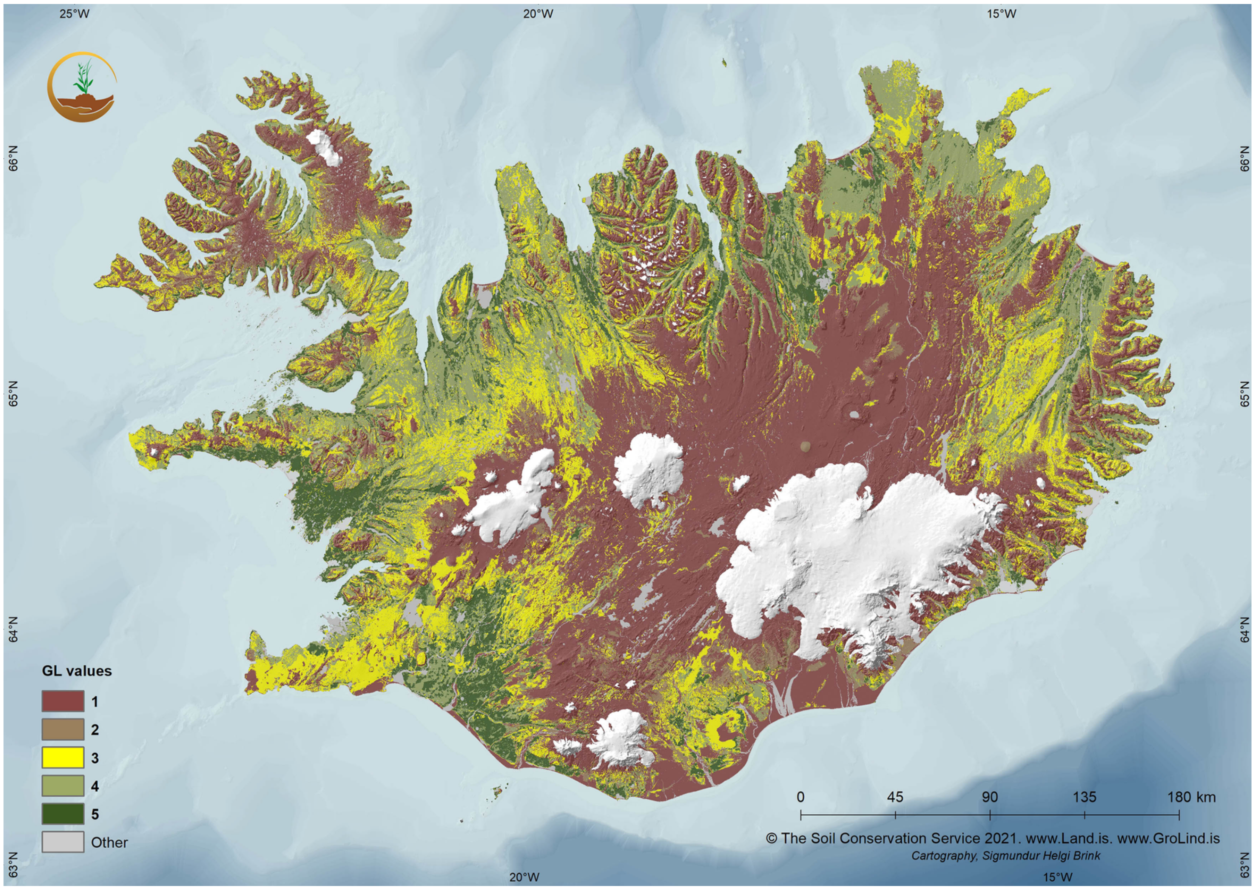

Figure 1 - A map of the condition classification of terrestrial habitats in Iceland. Class 1 represents the areas with limited activity and reduced stability resulting in extensive soil erosion. Class 2 represents the areas with little stability and ecological activity. Class 3 shows those habitat areas where little to considerable activity and erosion is present. Class 4 represents the moderately active habitats with considerably good stability and activity followed by little erosion, and then the final class, 5, showcases those areas where the habitats are very active and stable resulting in no active erosion (Arnalds et al., 2023).

A mapping of the conditions of terrestrial habitats in Iceland in correspondence to the level of activity, water regulation and habitat stability (*Figure 1*), demonstrates the poor conditions of many Icelandic ecosystem. Around 45% of Iceland is classified having little to limited ecological activity and soil stability resulting in soil erosion (Class 1 and 2, *Figure 1*). With only 40 % of the area having little to no erosion, and good stability and ecological activity (Class 4 and 5, *Figure 1*) (Marteinsdottir *et al.*, 2021). Not all the desertified areas of Iceland are due to anthropogenic factors as some have been formed throughout the centuries due to volcanic eruptions, flooding, and varying land elevations (Aradottir *et al.*, 2013). Since the early 1900 Iceland has been battling soil erosion with various restoration efforts (Aradottir & Hagen, 2013).

1.5. Restoration projects & actions

Ecological restoration can be driven by various interacting factors, motivators and mechanisms. One of the main motivators is the possibility of increased ecosystem productivity within an area, considering long term benefits and consequences (Hobbs & Norton, 1996). Another important factor that drives restoration projects are conservation values of a certain area and species, including the conservation of rare, endangered and threatened species, communities, and whole landscapes (Hobbs & Norton, 1996).

Organized restoration projects and soil conservation in Iceland has been at work for more than a century with varying drivers and results over time (Crofts, 2011). The improvement of the management of rangelands and habitat restoration has been a major objective within the Icelandic agricultural and Environmental policies since the early 1990s. Furthermore, soil conservation has been an official policy goal since 1907, with the establishment of the Soil Conservation Service of Iceland (SCSI, now Land and Forest Iceland) (Aradottir & Hagen, 2013). A majority of the restoration actions taken in Iceland have been carried out by public agencies and governmentally funded institutions. However, increasingly more restoration projects are being driven by various stakeholders, *e.g.*, landowners, farmers, non-governmental organizations (NGO's), and the general public (Aradottir & Johannsson, 2006).

1.5.1. Revegetation

The process of restoring plant coverage within areas where vegetation has been lost or damaged, partially or fully, is defined as revegetation (Byrne *et al.*, 2011). Such processes can involve the replanting of native plant species back to a specific area or other appropriate species which can further assist in stabilizing the soil, improve ecosystem functions and services, and enhance biodiversity. Revegetation is most commonly applied in areas undergoing fragmentation, extensive erosion, and degradation (Byrne *et al.*, 2011).

In Iceland in the early 1900 the main focus of restoration activities was on trying to stabilize the major problem of sand drift and preventing catastrophic sand encroachment. For this purpose, a native Lyme grass species, *Leymus arenarius*, was seeded in the affected areas, with the addition of extensively constructed barriers in some cases (Magnusson, 1997). Revegetation and range improvements on a larger scale became more common in the 1950s with the seeding of various grass species and with the application of mineral fertilizers. For such revegetation projects both native and non-native species were used. The most widely introduced grass species are *Deschampsia beringensis* and *Festuca rubra*, and then less extensively the native species *Deschampsia caespitosa* and *Festuca richardsonii* (Magnusson *et al.*, 2004).

During the past decades there has been an increasing interest in using legume species and more native trees and shrubs for the restoration of degraded and eroded land areas (Magnusson *et al.*, 2004). Implementing the use of nitrogen fixing legume species for reclamation services can reduce the additional need for fertilizers, therefore reducing effort and cost that can be involved in restoration projects of degraded habitats (Aradóttir & Jóhannsson, 2006). Reclamation with the introduction of *Lupinus nootkatensis* started in the 1980s, with the species being initially introduced to the country in the 1940s (Magnusson *et al.*, 2004; Magnusson, 2010). With the lupin being a nitrogen fixing species with extensive colonizing, self-distributing and production capabilities on nutrient deficient soils it suited well as a revegetation species for the vastly eroded areas in Iceland. The Nootka lupine has deep and extensive root systems that can bind the soil together, increasing its cohesion, and preventing further erosion due to wind and precipitation (Magnusson *et al.*, 2004). However, due to its invasive nature, it outcompeted most of the native vegetation already present resulting in it being declared as an invasive alien species by the Icelandic Ministry of the Environment in 2017, limiting any potential future use for restoration purposes. Since 2018, government institutions have stopped using Nootka lupine (Vetter *et al.*, 2018; IINH, 2024).

in Iceland was mainly a top down approach with majority of the work and initiative coming from the governmental institutions. But in the 1990 more focus was put on participatory approaches (Petursdottir, 2011). Farmers Heal the Land (FHL) is a large-scale restoration program, organized and launched by the Soil Conservation Service of Iceland (SCSI) back in 1990s, and initially covered about 150 km² of areas under restoration (Petursdottir, 2011). It was the first project carried out by SCSI that systematically involved other public stakeholders with the objective to encourage the restoration of severely degraded lowland rangelands. Involved in the project are around 600 farmers, located all over the country, that take part in sustainable rangeland grazing management and the revegetation of extremely degraded areas located within their lands. About 20% of all Icelandic farms were participating in the project by the year 2011, and by 2012, about 300 km² of degraded lowland rangelands have been treated by the FHL programme (Brynleifsdottir, 2012). Moreover, further studies have looked into the characteristics of soil and vegetation within these areas with the conclusions that these treatments, provided by the programme, have stimulated short-term ecosystem development, as well as overall public understanding and attitude towards the restoration practices (Petursdottir *et al.*, 2013).

1.5.2. Woodland restoration

Restoration projects have relied on the use of various tree and shrub species for erosion control through reforestation and afforestation methods such as planting of seedlings and, on a smaller scale, direct seeding (Aradottir & Johannsson, 2006). European white birch, *Betula pubescens*, and willows, *e.g., Salix phylicifolia* and *Salix lananta*, are species that possess the ability to colonize in the early stages of succession and also thrive in plantations on degraded and eroded sites, making them valuable for land rehabilitation efforts. Restoring birch woodlands not only aids in ecosystem restoration by reestablishing structure and function, but also expands land-use options (Aradottir & Eysteinsson 2005).

A restoration project, Hekla woodlands, with the aim to restore approximately 600 km² of native woodlands and shrublands was initiated in 2006 (Aradottir, 2007). The project's objective was to enhance the resilience of local ecosystems to disturbances caused by tephra fallout from Hekla volcano eruptions, while also mitigating potential damage from the secondary dispersal of tephra by wind. A significant portion of the project area is severely degraded, with sparse vegetation cover and active soil erosion. Restoration efforts in the Hekla woodlands involve both high-density planting and natural regeneration from existing stands. However, the main approach is to establish woodland clusters that act as seed sources for further expansion, stabilizing the soil surface and mitigating soil erosion where necessary (Aradottir, 2007). Another current project being carried out in Iceland in the matter of restoration of birch woodlands is the interdisciplinary project EcoBirch. The main objective of the project is to increase general knowledge of the importance of woodland restoration, and also examine the benefits and consequences of such restoration practices in terms of water management, carbon sequestration, landscape, and biodiversity (Aradottir *et al.*, 2022).

2. Methodology

2.1. GróLind

Despite Iceland's history of land degradation, limited efforts were put into place to monitor the conditions of Icelandic ecosystems, until the year 2017, when GróLind, the first nationwide soil and vegetation monitoring project was launched (Marteinsdottir *et al.*, 2021). This thesis was partly based on this ongoing monitoring project carried out by Land and Forest Iceland. In GróLind, monitoring data in relation to various environmental factors connected to ecosystem function and structure are used to estimate the condition of the rangelands in Iceland as well as detect any possible changes over time. The overall objective is to use the information collected to promote sustainable land management (Marteinsdottir *et al.*, 2021). The project is based on adaptive monitoring, meaning that methods and approaches are always being evaluated and improved, *e.g.,* in light of new data (Lindenmayer & Likens, 2009).

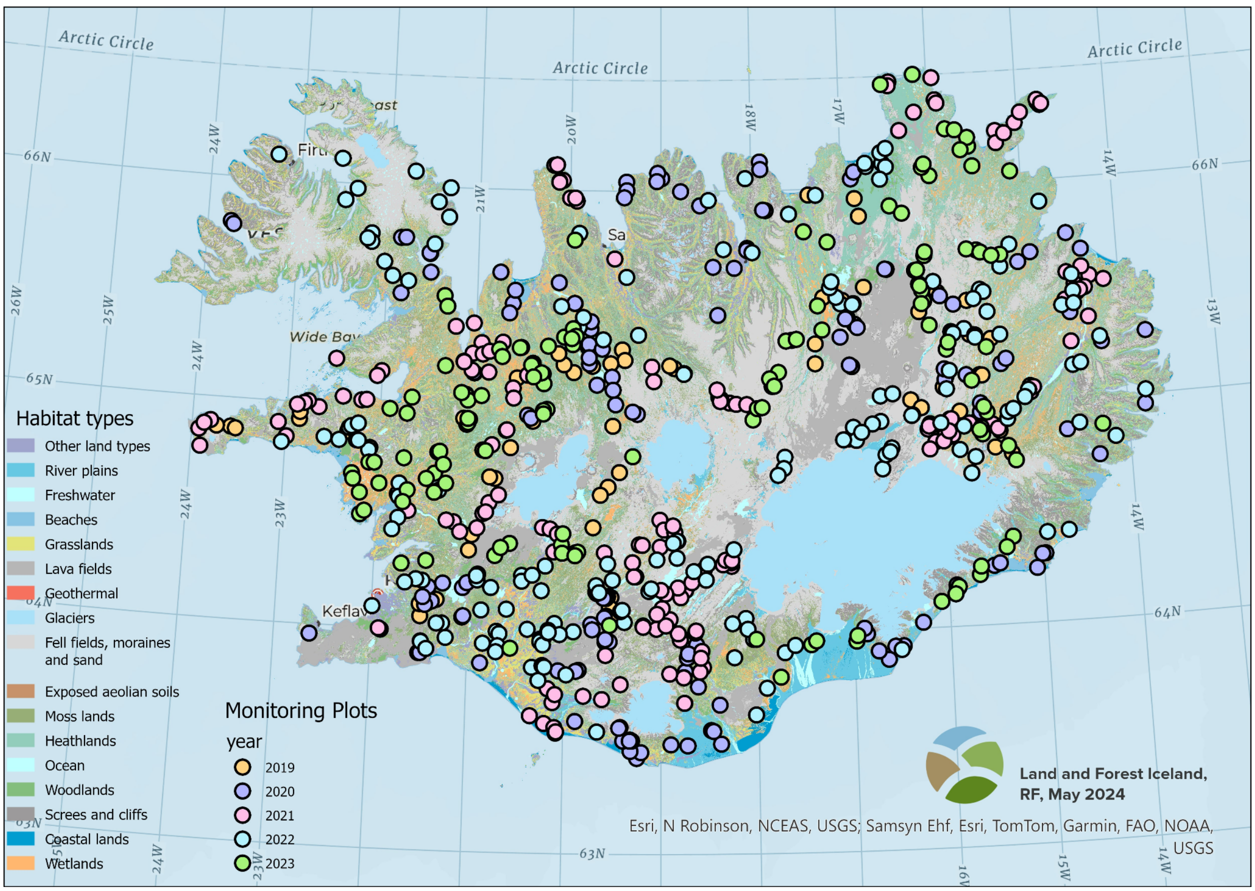

Figure 2 - A map of Iceland showing the distribution and locations of the 701 monitored plots that were measured between the years 2019-2023. All plots were included in this study and I took part in the sampling of the ones from 2023. All main habitat types are also present on the map.

In each sampling point in GróLind more than ten variables, related to ecosystem functioning, are measured or estimated, including habitat classification, soil type, soil erosion, elevation above sea level, list of species and vegetation cover (Marteinsdottir *et al.*, 2021).

The research area of GróLind covers around 75% of Iceland, excluding only the habitats that were irrelevant to the project, *e.g.,* human influenced areas, agricultural lands, forestry, and areas covered by other monitoring projects. Within the research area, 1500 sampling points were initially distributed randomly over the whole study area. Each sampling point will be measured every 5 years, and the first round of sampling finished in 2024 with 918 plots being established in total but the other 600 plots being omitted due to various reasons, mainly due to lack of accessibility.

For this study, data and information from 701 plots measured between the years 2019-2023 were used (*Figure 2*). The project was conducted in collaboration with Land and Forest Iceland, and then carried out in the Department of Biology, Lund University.

2.2. Data collection

This thesis focused on the variables collected in GróLind that were needed in order to answer the research questions and hypotheses. These variables were elevation, Line Point Intercept (LPI), species identification, soil classification, habitat classification, estimations of vegetation cover, and level of erosion for each plot. The methods performed for gathering data for each variable are described in detail below.

2.2.1. Site selection & coordinates

The placement of the monitoring plots for GróLind, was based on a stratified random sampling (Ding *et al*., 1996; Nguyen *et al.*, 2020). The country was stratified based on habitat type and height intervals (Marteinsdottir *et al.*, 2021). The height intervals used were 0-200 m, 200-400 m, 400-600 m, 600-800 m, 800-1000 m, and then >1000 m above sea level. The proportional area of each stratum was used to calculate the proportion of points within those strata. The points were then placed randomly within each stratum no further than 1.5 km from the closest road and no closer than 50 m to the road. Hence, the points were more likely to land within habitats with larger areas, relatively.

2.2.2. Plot criteria

Each plot was either rejected or used to establish a monitoring plot. Certain criteria were used to reject selected plots in the field in order to prevent any external anthropogenic influence on the results from land use practices, infrastructure, man-made land, etc. In some cases, the plots could be rejected beforehand by looking at map data, *e.g.,* if the plot was located in a habitat not being monitored, such as forestry areas and agricultural lands, and if the plot was not accessible in the field by car or at walking distance. Also, in some cases, the plot was rejected due to landowners not consenting to the procedure on their land.

2.2.3. In the field

The data collection in the GróLind plots was carried out by specialists and others that had received adequate training and experience in plant- and soil identification, as well as land literacy. All the measurements gathered from the field were registered in pre-made forms, either into Excel sheets or into well-constructed survey forms in ArcGIS Survey 123 from ESRI (ESRI, 2024).

2.3. Experimental design

Each sampling plot covered an area of 50 m x 50 m with the use of two 50 m line transects which were placed perpendicular over each other forming a cross section, one transect crossing the plot from south to north and the other from west to east (*Figure 3*). The starting points for each transect were the south and west points, and at each point a small wooden post was placed at the starting, end, and middle points in order to mark the plot for future re-measurements. A larger pole was then placed at the south point, which also held the number for the specific plot.

Within the plots a GPS coordinate was taken at each point. Photos were taken, two overviewing the whole plot from the south and east, and then four overviewing photos from the middle point in each direction. All the measurements were done within the whole area of the plot or along the transects.

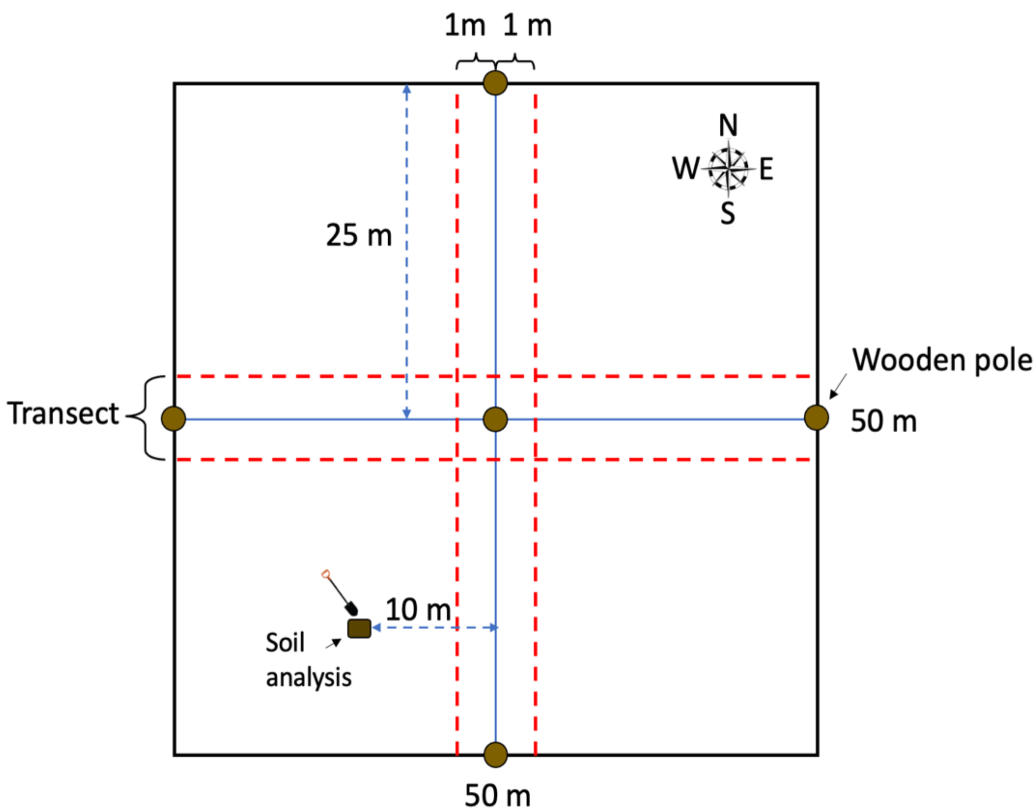

Figure 3 - A schematic illustration of a sampling plot in the GróLind project with the two crossing line transects (50 m x 50 m). The blue lines illustrate the placement of the measuring tapes where along which parts of the measurements took place, e.g. erosion % and other coverage estimations. The red dotted lines illustrate the area at each side of the measurement tapes (1 m in each direction) where a vascular plant species list was noted for the sampling area. At 10 m north and 10 steps west from the south point a hole was dug for soil profiling and classification. For the whole area of the plot, black outlines, measurements such as habitat type, vegetation cover, and erosion level were estimated.

2.4. Habitat classification

Table 1 - List of the habitat types in Iceland used for this research (EEA, 2024; IINH, 2024).

	Habitat type classes		**Habitat types**
L1	Fell fields, moraines & sands	**L1.1**	Sparsely- or un-vegetated habitats on mineral substrates not resulting from recent ice activity
		L1.2	Sparsely- or un-vegetated habitats on mineral substrates not resulting from recent ice activity
		L1.3	Oroboreal *Carex bigelowii-Racomitrium* moss-heaths
		L1.4	Glacial moraines with very sparse or no vegetation
		L1.5	Volcanic ash and lapilli fields
L2	Exposed aeolian soils	**L2.1**	Icelandic exposed andic soils
L3	Screes & cliffs	**L3.1**	Icelandic talus slopes
L4	River plains	**L4.2**	Icelandic braided river plains
L5	Moss lands	**L5.1**	Boreal moss snowbed communities
		L5.2	Icelandic *Racomitrium ericoides* heaths
		L5.3	Moss and lichen fjell fields
L6	Lava fields	**L6.1**	Barren Icelandic lava fields
		L6.2	Icelandic lava field lichen heaths
		L6.3	Icelandic lava field moss heaths
		L6.4	Icelandic lava field shrub heaths
L7	Coastal lands	**L7.6**	Icelandic *Carex lyngbyei* salt meadows
L8	Wetlands	**L8.2**	Icelandic stiff sedge fens
		L8.4	Juncus arcticus meadows
		L8.5	Boreal black sedge-brown moss fens (highlands)
		L8.6	Boreal black sedge-brown moss fens (lowlands)
		L8.9	Icelandic black sedge-brown moss fens
		L8.10	Icelandic *Carex rariflora* alpine fens
		L8.11	Common cotton-grass fens
		L8.13	Basicline bottle sedge quaking mires
		L8.14	Icelandic *Carex lyngbyei* fens
L9	Grasslands	**L9.1**	Icelandic *Carex bigelowii* grasslands
		L9.2	Insular *Nardus-Galium* grasslands
		L9.3	Wavy hair-grass grasslands
		L9.4	Boreal tufted hairgrass meadows
		L9.5	Icelandic *Festuca* grasslands
		L9.6	Boreo-subalpine *Agrostis* grasslands
		L9.7	Northern boreal *Festuca* grasslands
L10	Heathlands	**L10.1**	Icelandic *Racomitrium* grass heaths
		L10.2	Arctic *Dryas* heaths
		L10.3	Icelandic *Carex bigelowii* heaths
		L10.4	Icelandic *Empetrum Thymus* grasslands
		L10.5	Icelandic lichen *Racomitrium* heaths
		L10.6	North Atlantic boreo-alpine heaths
		L10.7	Oroboreal moss-dwarf willow snowbed communities
		L10.8	North Atlantic *Vaccinium-Empetrum-Racomitrium* heaths
		L10.9	Icelandic *Salix lanata/S. Phylicifolia* scrub
		L10.10	Oroboreal willow scrub

A habitat classification was performed for the whole area within each plot based on the habitat classification system from the Natural History Institute of Iceland (IINH) (*Table 1*). Moreover, an identification key was used in the field to further estimate the exact habitat type for the particular environment (Ottosson *et al.*, 2016; Magnusson, 2019).

2.5. Land assessment & erosion levels

The vegetation cover within the whole area of each plot was assessed as relative percentages and put into the corresponding coverage categories, 0-10%, 11-33%, 34-66%, 67-90%, and 91-100%. Then, for each category a corresponding level value was provided from 1 to 5, where 1 corresponded to the least coverage and 5 the most.

Table 2 - Erosion scale used to determine the level of erosion for each area (Arnalds & Aradóttir, 2015).

Erosion level		**Description**
0	Non existent	No traces of erosion
1	Minimal	Some traces of erosion, non-active
2	Modest	Some traces of erosion, slighly active
3	Considerable	Slowly active yet growing erosion, gravel
4	Substantial	Active erosion characterized by loose sand
5	Extensive	Very active and immense erosion, open areas with loose sand

Furthermore, the condition level of the habitat in relation to different erosion stages was estimated for the whole area of each plot. An erosion scale (*Table 2*), following a pre-specified criteria and methodology used by the Agricultural Research Institute and the Soil Conservation Service of Iceland in order to map the occurrence of erosion in Iceland in the late 1900s (Arnalds *et al.*, 2001b), was used when estimating the level of soil erosion within the whole area of each plot. The objective for estimating the erosion was to evaluate the type and level of erosion as the measurement of current loss of vegetation cover, *i.e.* amount of bare soil. For this project, the focus was only on the level of erosion rather than the specific erosion type. The different levels ranged from 0 to 5, where 0 indicated a non-existent and non-active erosion, while on the contrary the highest level, 5, corresponded to a very active and extensive erosion.

2.6. Soil characteristics

The soil properties were estimated by digging a 30 cm hole at approximately the same place in each plot, 10 m north and 10 steps west from the south starting point (*Figure 3*). The soil type was estimated by looking at the grain size and texture. The hole was excavated and backfilled after doing the soil measurements in a concise manner, to leave the area as undisturbed as possible.

One side of the hole was used for soil profiling, recording all the soil layers present. Different soil layers in the profile can influence the hydrology of the system due to differing permeability, and therefore, affect erosion risks (Hartemink *et al.*, 2020). After recording the soil profile, the specific soil type for the area could be estimated by assessing the soil texture.

The soil texture can be used to determine the permeability of the soil. Soil permeability is defined as the property of soil to transmit water and air through the soil profile. Different soil types and horizons have varying physical and chemical properties, resulting in different permeability (Elhakim, 2016). The soil texture was estimated by hand using the Soil Texture by Feel method, also called the Ribbon method, following a soil texture determining flowchart (Thien, 1979; Ritchey *et al*., 2015). This method of analysis was used as it is thought to be best suited for the Icelandic soil (Arnalds, 2015) (*Table 3*). For this procedure, a small handful of soil was taken from the top 5 cm of the soil profile and the sample was then broken apart by hand, moistened as much as needed and kneaded until it withheld a round, ball-like structure. If the sample could not be kneaded into a ball-like structure, then we could straight away characterize the soil as sand. However, if it held structure to some level the following steps could be performed.

Table 3 - A guide for determining soil type from soil texture and other characteristics using the Soil Texture by Feel method (Ribbon method) based on a soil texture estimation flow chart (Thien, 1979; Ritchey et al., 2015).

Structure group	Coherence	Ribbon length (cm)	Texture	Soil type
Sand	None	0	Sandy	Sand
	Slight	0	Sandy	Loamy sand
Loam	Just coherent	< 2.5	Mixed	Loam
	Just coherent	< 2.5	Sandy	Sandy loam
	Just coherent	< 2.5	Smooth	Silt loam
Clay loam	Coherent	2.5 - 5	Mixed	Clay loam
	Coherent	2.5 - 5	Sandy	Sandy clay loam
	Coherent	2.5 - 5	Smooth	Silty clay loam
Clay	Strongly coherent	> 5	Mixed	Clay
	Strongly coherent	> 5	Sandy	Sandy clay
	Strongly coherent	> 5	Smooth	Silty clay

The soil sample was pressed out between the thumb and forefinger, and its ability to hold structure and cohesion without breaking could determine the main soil category. If the sample withheld no structure when pressed, we had loamy sand, if it held slightly and formed a so called ribbon of certain degree (< 2.5 cm) it could be categorized as a type of loam, if it held relatively well (2.5-5 cm) then we had a type of clay loam, and finally if it held its form strongly (> 5 cm) then we had a sort of clay soil.

The next step, after determining the specific category from the structure, was to look at the soil texture. In order to perform this analysis, a small part of the initial handful sample was taken and rewetted more than before until it had more of a paste consistency. Then we used the fingertips to estimate the texture of the soil on our palm. The texture would either be very sandy, very smooth, or neither particularly rough nor smooth (*Table 3*).

If the soil sample had been categorized as a type of loam and the texture was very rough, we could estimate that we had sandy loam. If the texture was very smooth then we had silt loam, and then if it did not fall under either category, it would be characterized as loam.

If the soil sample had been categorized as a type of clay loam, then it could be classified as so if there were no decisive texture differences. However, if the texture was rougher, then we had sandy clay loam, and with a very smooth texture we had silty clay loam.

Then finally, if the soil sample had been categorized as a type of clay, if very rough then the texture indicated that we had sandy clay and with a smooth texture we had silty clay. Then again, we simply had clay if there were no decisive texture characteristics.

2.7. Line Point Intersect

Line point intercept (LPI) is a standardized vegetation monitoring method that provides estimates for plant community structure, vegetation cover and biodiversity within an area, as well as quantifying bare soil cover (Godínez-Alvarez *et al.*, 2009). LPI is a visual assessment method that can be used to accurately estimate the cover of either a certain species or different plant functional groups in natural ecosystems (Thacker *et al.*, 2015).

The LPI method was performed at every 0.5 m along the transects (101 measurements for each line transect in total) with the use of a relatively long pin in order to estimate the different plant functional groups, soil cover and litter within the area (*Figure 3*). The functional groups were based on the growth form of the plants as well as their ecological properties (*Table 4*). The different groups were differentiated due to their various responses to environmental changes and habitat choice. Each group was only registered once per measurement and the groups were identified top-down, *i.e.*, the group with the first upper contact with the pin and then we worked our way down towards the surface. In order to prevent any measurement bias the individual measuring ensured that the pin was placed at each interval randomly without any specific aim or preference.

Table 4 - The plant functional groups used for the Line Point Intersect measurements.

Functional groups	**Descriptions**
Bryophytes	All species of moss. Exclude if they are growing on rocks.
Lichens	All species of lichen. Exclude if they are growing on rocks.
Biocrust	Biological soil crust. Cryptogamic crust.
Pteridophytes	Ferns, horsetails & clubmoss/spikemoss
Graminoids	Grasses & other herbaceous species, *e.g.*, sedge & rush.
Loose litter	Unattached litter and plant residues.
Rooted litter	Litter still attached to live plant bodies.
Herbs	Herbaceous flowering vascular plants.
Alien species	All alien species, *e.g.*, alaskan lupin, hedge parsley & self distributed pine.
Shrubs & trees	*E.g.*, dwarf birch, downy birch, mountain ash, & some willows, excluding dwarf willow.
Evergreen shrubs	*E.g.*, heather, mountain avens, common crowberry shrubs, mountain azalea, purple mountain saxifrage, & mountain thyme.
Deciduous shrubs	*E.g.*, blueberry shrubs, bilberry shrubs, moss plant, & dwarf willow.
Bare soil	All bare soil (sand, loam, clay, etc.) and rocks with a diameter of <5 cm.
Rocks	All rocks with a diameter of >5 cm.

2.8. Species identification

The number of different species, vascular plants, mosses, and lichens, found within a plot can be used as an estimation of species richness, and moreover, a measurement of biodiversity (McGlinn *et al.*, 2018). High biodiversity and high vegetation cover in an ecosystem has been connected to increased carbon flux and storage of nitrogen and phosphorus, resulting in more stable and productive ecosystem processes (Yadav & Mishra, 2013).

A species list was compiled along both transects within an area of 1 m from the measuring tape in each direction (*Figure 3, Table A2 in appendix*). The vascular plant identification was based on the Icelandic plant identification manual by Hörður Kristinsson (2010). All species seen within the transect were recorded and marked down as present. Only certain moss- and lichen groups were registered based on their ecological role in the natural system; *Racomitrium* sp., *Sphagnum* sp., *Peltigera* sp., *Stereocaulon* sp., *Cetraria islandica*, *Cetraria delisei*, *Cladonia arbuscula,* and also Cryptogamic crust. The number of species was then referred to as species richness (SR) at a scale of 100 m^2 for each plot.

In addition, all the registered species were then later categorized into different plant functional groups. These groups were mosses, lichens, graminoids, forbs, evergreen shrubs, deciduous shrubs, ferns, and equisetum.

2.9. Data analysis

Data was collected and gathered from the plots I took part in measuring and monitoring for a few weeks during the summer of 2023, as well as being provided with additional corresponding data and information from all plots measured during the years of 2019 to 2023. The data was partly in the form of two separate excel sheets, where one including the full species list and the other one the Line Point Intersect (LPI) measurements for both transects in each plot. All other information, such as habitat type, elevation, soil type, vegetation cover, erosion levels, etc., was provided on separate excel datasheets that had been constructed from the ESRI survey forms. From the excel sheets, covering information from roughly 700 plots, I constructed my own databases by combining all necessary data and information and transferring it to a clear and applicable format for further statistical analysis in R studio.

Cover of different functional group in each coverage layer for individual plots was calculated from the Line Point Intercept (LPI) data as (*Table 4*):

$$Coverage\ (\%) = \frac{Number\ of\ occurring\ measurements}{Total\ amount\ of\ measurements} \times 100$$

(*Eq. 1)*

Every layer was estimated proportionally from the 101 measurements per transect (South to North and West to East) at every 0.5 m (0 to 50 m). The first layer calculated was the canopy cover which was defined as the upper layer of the vegetation zone. This was only calculated for points with more than one measurement and the first hit being categorized as vegetation. The next layer calculated was the basal cover which is present when the final hit of a point is vegetation. The stone base layer was estimated from all the point measurements where the final hit was a stone. The percentage of soil erosion (Erosion %) for the plot was then estimated from the point measurements where the final hit was bare ground, moreover, the percentage of bare ground.

All statistical analysis was performed with R-studio (R Core Team, 2024), with additional packages *ggplot2* (Wickham, 2016), *dplyr* (Wickham *et al.*, 2023), *MASS* (Venables & Ripley, 2002), *vegan* (Oksanen *et al.*, 2024), and *rstatix* (Kassambara, 2023). To test for differences in median soil erosion (percentage and level) between the different habitats and soil types, non-parametric Kruskal-Wallis tests were used. To further assess differences between groups, post-hoc paired comparisons (Dunn test) were used with a Bonferroni adjusted alpha level to account for the number of groups being compared.

To further test if different estimations of soil erosion were significantly related to different environmental factors and measured variables, linear models were used. In addition, the correlations between all environmental factors were checked using a general linear model (*Table A1 in appendix*). To measure the linear correlations between different data sets the Pearsons correlation coefficient, R, was used (Pearsons, 1895).

All the linear models were built using the following form:

$$Y \sim \alpha + x\beta + \varepsilon$$

(*Eq. 2*)

Where Y represents the response variable, α the model intercept, β is the slope on the explanatory variable, x, and ε is an error term representing model residuals. All models were fitted using *lm* function in base R (R Core Team, 2024). When testing if habitat, soil type and elevation influenced erosion, the erosion (erosion % or erosion level) was the response variable (Y) and the other explanatory variables (X). When testing if soil erosion effected vegetation cover, LPI cover measurements, species richness, and the number of functional groups, the erosion factors were the explanatory variables (X), whilst the other variables were the response (Y). Model selection was done using R^2, as well as adjusted R^2, and Residual Standard Error (RSE) values in order to identify the best fitted model to the data. The model selected for the data was a linear model using a second order (k = 2) polynomial regression forming a quadratic expression, along with a relevant confidence interval, to further represent the correlation and relationship between the variables.

When determining if species composition was related to varying erosion levels for all monitored plots a non-metric multidimensional scaling (NMDS) method was applied to the data. Only plots where one or more species was present were used for the execution of the NMDS analysis, 692 plots in total. The data was prepared by filtering the data and removing data from plots where no species were registered, providing us a new data set representing the presence and absence of all registered species for all relevant plots. The NMDS analysis was performed using the *metaMDS* function from the *vegan* (Oksanen *et al.*, 2024) package in R, along with the Jaccard distance metric for assessing the dissimilarity between species compositions across all plots in relation to erosion level.

3. Results

3.1. Comparing different estimates and measurements of erosion

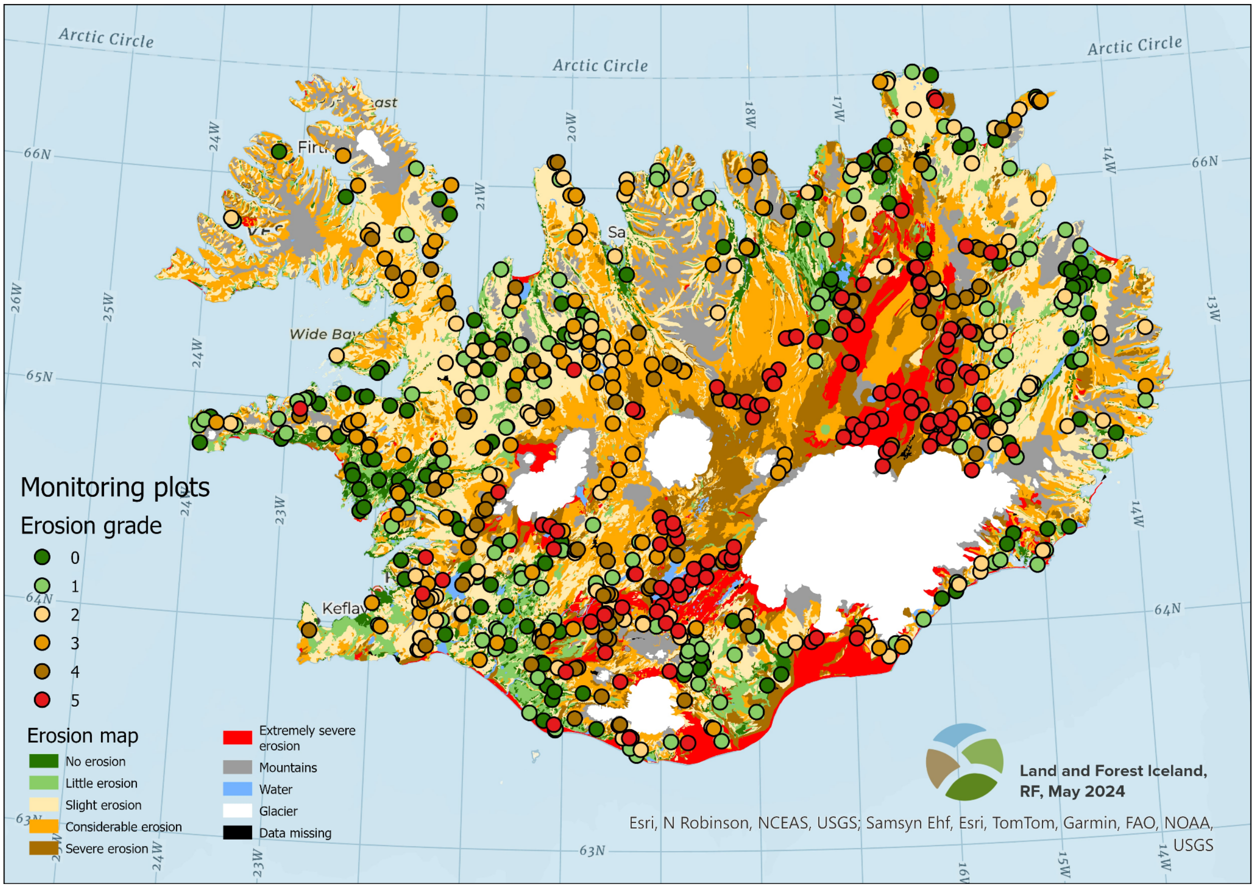

Figure 4 - A map of Iceland representing both a layer of a previously published erosion map and on top of that points for all the measured plots, N = 701, from the years 2019-2023 with their corresponding erosion grade (0 to 5).

Our estimated erosion levels (0 to 5) for each plot in this study seem to correspond well with an erosion map of Iceland published in 1997, that was based on aerial photography and mapping (*Figure 4*) (Arnalds *et al.*, 2001b). There was a positive correlation between the soil erosion estimations of each plot measured in this study, erosion %, calculated as the percentage of bare soil from the LPI measurements along the transects, and erosion level, estimated in increment levels of 0 to 5 for the whole area the plots ($R = 0.81$, *Table A1 in appendix*). Linear regression models were also performed between the two erosion factors resulting in a significant relationship between the two estimates of soil erosion used in this study (estimate = 16.729, SE = 0.439, $p < 0.001$), where increasing erosion levels correspond to increasing erosion percentage ($R^2 = 0.749$) (*Figure 5*). At approximately 85% erosion the erosion levels reach a peak at almost level 5, plateauing as the measured erosion reaches 100%.

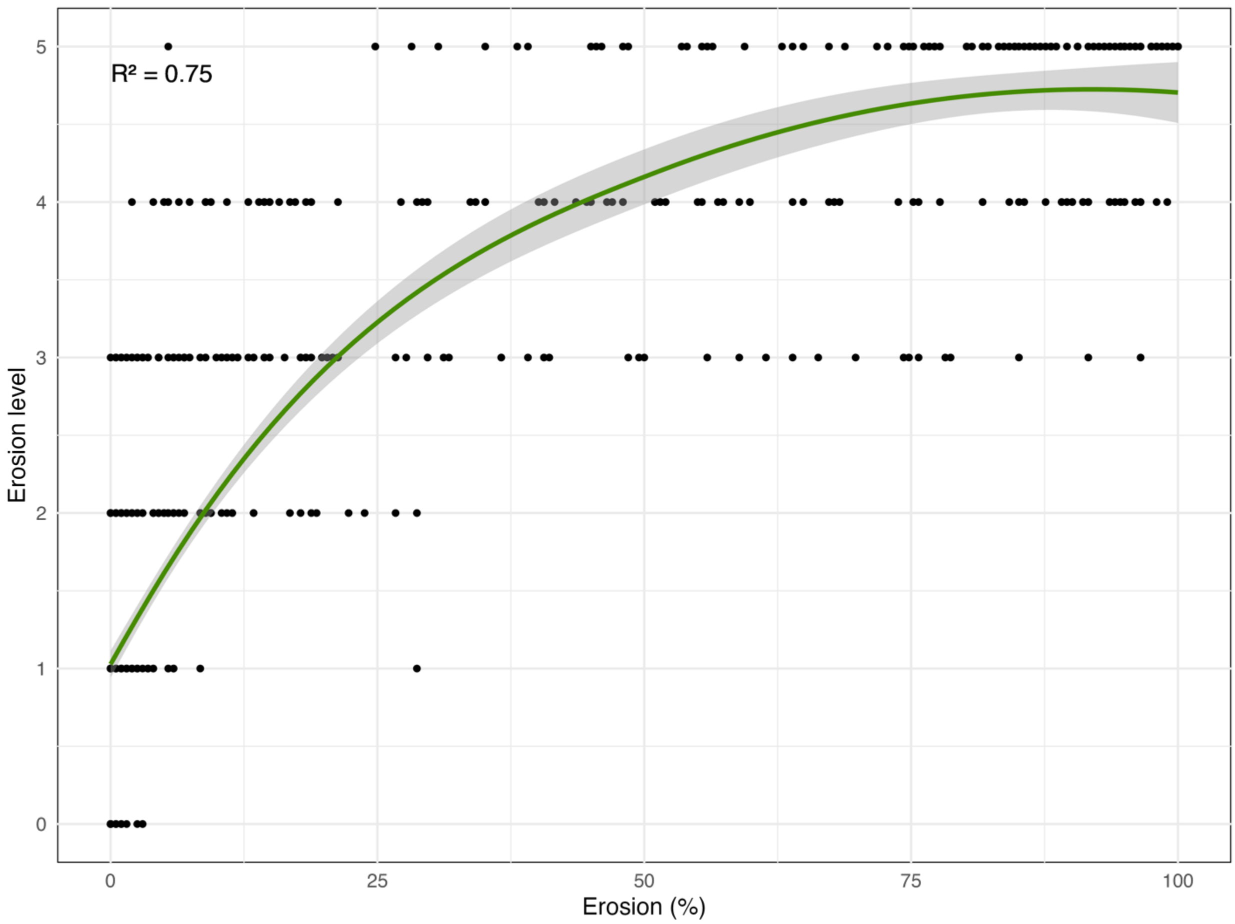

Figure 5 – Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship between the estimated Erosion level (0 to 5) for the whole area of each plot and Erosion (%) measured from Line Point Intersect (LPI) and calculated from bare ground percentages.

3.2. Relationship between environmental parameters and the erosion factors

Habitat types differed significantly in erosion % (Kruskal-Wallis test: x^2 = 449.12, df = 8, p < 0.0001). Fell fields, moraines and sands had the highest overall erosion % (median = 84.7%) followed by river plains (median = 59.9%), exposed aeolian soils (median = 45%) and screes and cliffs (median = 34.2%). The remaining habitat types generally displayed low erosion % with median values ranging between 0-10% (*Table 5*). However, erosion % within habitat types were highly variable for some habitat types (*e.g.,* L1, L4, and L6) which together with low samples sizes for some habitat types (*e.g.*, L2, L3, and L7) made subtle differences scarce. Post-hoc tests generally suggested higher erosion % in fell fields, moraines and sands, river plains, and lava fields compared to vegetated habitats such as wetlands, grasslands, and heathlands for which erosion % was close to zero. All significance levels between group comparisons are listed in *Table A3*.

Table 5 – Summary statistics between erosion % and different habitat types (Habitat), including N which is the sample size for each habitat, mean erosion %, standard deviation and the median for the relationship between erosion % and habitat. Note that habitat type L7 was excluded from the statistical analysis due to low sample size.

	Habitat	N	Mean	SD	Median
L1	Fell fields, moraines & sands	204	70.0	30.5	84.7
L2	Exposed aeolian soils	3	39.6	19.2	45.0
L3	Screes & cliffs	3	44.7	26.9	34.2
L4	River plains	8	59.7	31.3	59.9
L5	Moss lands	59	6.9	12.5	1.49
L6	Lava fields	47	36.6	36.7	11.9
L7	Coastal lands	1	0	-	0
L8	Wetlands	101	0.9	3.1	0
L9	Grasslands	38	1.1	2.5	0
L10	Heathlands	237	5.2	8.7	1.5

Overall, there were significant differences between habitat types in observed erosion level (Kruskal-Wallis test: x^2 = 566.2, df = 5, $p < 0.0001$). However, exchanging erosion % with erosion level revealed a slightly different pattern. High levels of erosion were present in habitat types L1, L3, L4 (median level = 5) and L2 (median level = 4) and intermediate erosion was observed in L6 as well as L5 and L10 (median level = 3 and 2 respectively) (*Table 6*). For habitat types L5 and L10 this is notable given that both habitats displayed low erosion % (~1.5%). As with erosion %, habitat types L8 and L9 both displayed low levels of erosion (*Table 5-6*). Post-hoc tests generally suggested higher erosion level in fell fields, moraines and sands, river plains, and lava fields compared to wetlands and grasslands (*Table A4*. There were also suggested differences for habitats L2 (L2-L8) and L3 (L3-L8 and L3-L9), however due to the low sample sizes in the respective habitats it is difficult to draw any meaningful conclusions. Overall these results suggested that habitats with sparse or no vegetation exhibited more erosion than vegetated habitats.

Table 6 - Summary statistics between erosion level and different habitat types (Habitat), including N which is the sample size for each habitat, mean erosion %, standard deviation and the median for the relationship between erosion % and habitat. Note that habitat type L7 was excluded from the statistical analysis due to low sample size.

	Habitat	N	Mean	SD	Median
L1	Fell fields, moraines & sands	204	4.2	1.0	5
L2	Exposed aeolian soils	3	4.0	1.0	4
L3	Screes & cliffs	3	4.7	0.6	5
L4	River plains	8	4.6	0.7	5
L5	Moss lands	59	1.8	1.2	2
L6	Lava fields	47	3.3	1.5	3
L7	Coastal lands	1	1.0	-	1
L8	Wetlands	101	0.5	0.8	0
L9	Grasslands	38	0.8	0.9	0
L10	Heathlands	237	1.8	1.2	2

Comparisons of erosion % between different soil types revealed significant differences (Kruskal-Wallis test: x^2 = 287.7, df = 10, $p < 0.0001$). The highest overall erosion % was present in soils consisting of sand (median = 88.6 %) followed by loamy sand soils (median = 51%), indicating greater susceptibility to erosion in coarser-textured soils. Erosion % in the remaining finer soil types were generally low (median < 10%) (*Table 7*). Post-hoc tests revealed significantly higher erosion % in sand soils compared to all other soil types (all $p < 0.0001$). Loamy sand soils also showed a significantly higher erosion % than the other soil types except for sand, sandy loam, and clay soils. However, the low sample size ($n = 4$) and the presence of an outlier in clay soils likely generated this effect (*Table 7* and *Table A5*). Moreover, the generally low erosion % in soils consisting of clay and loam suggests an overall effect of sand presence in the soil on erosion %.

Table 7 – Summary statistics between erosion % and different soil types, including N which is the sample size for each soil type, mean erosion %, standard deviation and the median for the relationship between erosion % and soil type.

Soil type	N	Mean	SD	Median
Sand	101	79.7	25.9	88.6
Loamy Sand	95	49.8	36.4	51.0
Loam	72	5.7	14.5	0.5
Sandy Loam	62	30.2	35.7	9.9
Silt Loam	61	4.8	11.5	0
Clay Loam	72	4.2	8.6	0.9
Sandy Clay Loam	40	8.0	8.5	5.9
Silty Clay Loam	50	4.1	7.9	1.2
Clay	4	3.6	6.9	0.2
Sandy Clay	7	2.7	5.7	0
Silty Clay	12	1.8	2.0	1.2

The median erosion levels across the sampled soil types also differed significantly (Kruskal-Wallis test: $x^2 = 245.2$, df = 10, $p < 0.0001$) and showed a similar clear trend related to soil texture. Soils consisting of sand exhibited the highest median erosion level (median = 5), followed by loamy sand (median = 4). Sandy loam and sandy clay loam had intermediate median erosion levels (median = 3 and 2, respectively). In contrast, finer-textured soils such as loam, silt loam, clay loam, silty clay, silty clay loam, and clay all had low median erosion levels (range: 1-1.5). Sandy clay was the only soil type with a zero median erosion level (*Table 8*). As with erosion %, most of the pairwise post-hoc comparisons occurred between soil consisting of sand and loamy sand (*Table A6)*. Overall, this pattern highlights the influence of soil texture on erosion susceptibility, with coarser soils experiencing greater erosion than finer soils.

Table 8 - Summary statistics between erosion level and different soil types, including sample size (N) for each soil type, mean erosion %, standard deviation and the median for the relationship between erosion % and soil type.

Soil type	N	Mean	SD	Median
Sand	101	4.4	1.2	5
Loamy Sand	95	3.5	1.6	4
Loam	72	1.5	1.2	1
Sandy Loam	62	2.6	1.6	3
Silt Loam	61	1.1	1.3	1
Clay Loam	72	1.6	1.3	1.5
Sandy Clay Loam	40	2.1	1.3	2
Silty Clay Loam	50	1.4	1.2	1
Clay	4	1.5	1.9	1
Sandy Clay	7	0.9	1.6	0
Silty Clay	12	1.4	1.1	1.5

Both soil erosion % and erosion level increased significantly with increasing elevation (estimate = 0.079, SE = 0.004, $p < 0.01$; *Figure 6A*, and estimate = 0.004, SE = 0.0002, $p < 0.001$; *Figure 6B*).

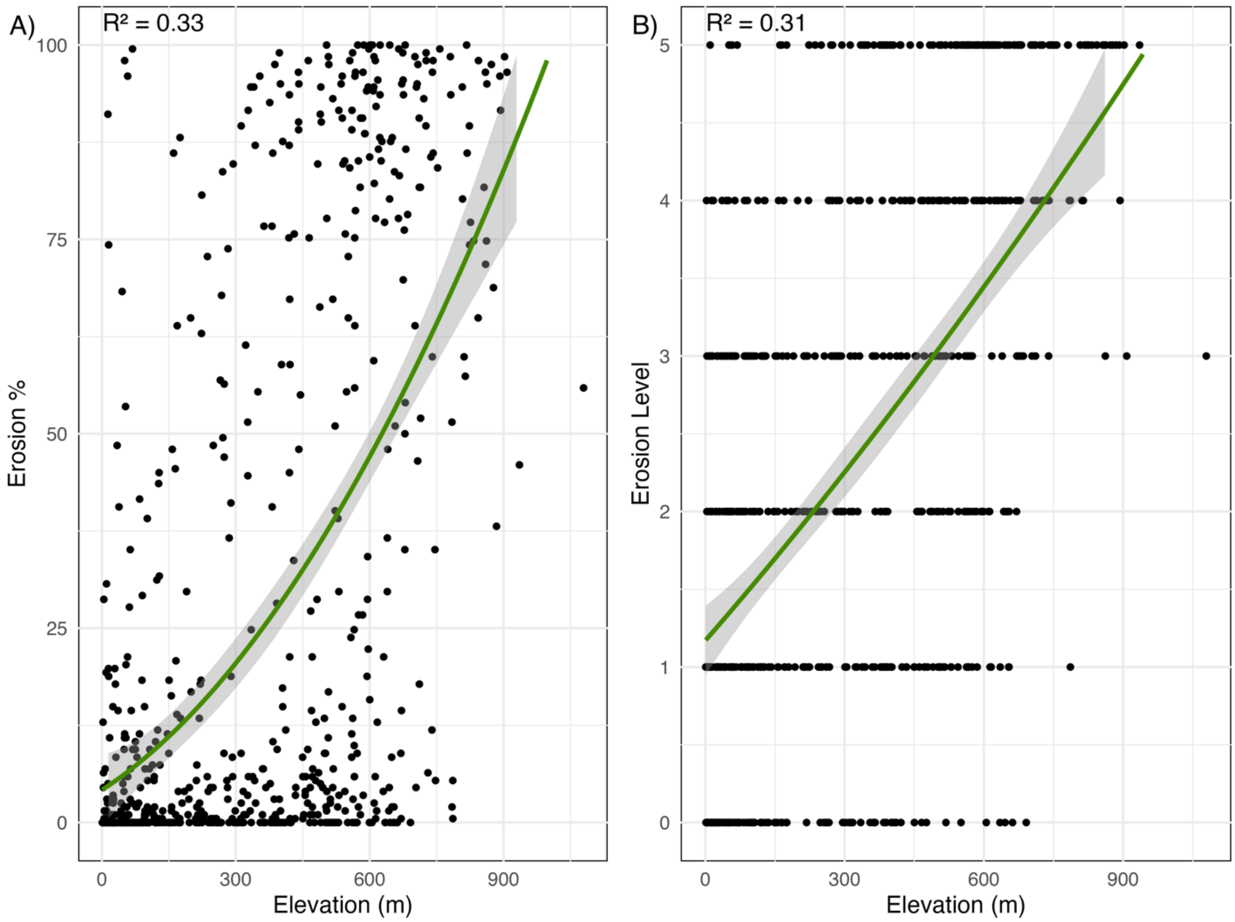

Figure 6 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship between Elevation (m) and the two different erosion factors measured, Erosion (%) from Line Point Intersect (LPI) estimated from bare ground percentages (R^2 = 0.33), and Erosion level estimated visually in the field (R^2 = 0.31).

Elevation explained 33% (R^2 = 0.327) of the variability with erosion % (*Figure 6A*) and 31% (R^2 = 0.311) with erosion level (*Figure 6B*). With increasing elevations, there was more erosion. At approximately 600 m elevation there was around 50% erosion, but at 900 m it increased to 100% erosion. However, 100% erosion could also be detected from a range at very low elevations, 50-100 m, up to the highest elevation, >1000 m. Plots with no erosion were most common at elevations of 0-100 m, ranging up to roughly 700 m. At 0 m elevation, the mean erosion level was 1, then being increased to level 3 at approximately 500 m elevation, before reaching a peak with level 5 erosion at 900 m.

There was a significant negative relationship between both erosion % and vegetation cover (estimate = -0.041, SE = 0.001, $p < 0.001$; *Figure 7A*) and between erosion level and vegetation cover (estimate = -0.799, SE = 0.016, $p < 0.001$; *Figure 7B*).

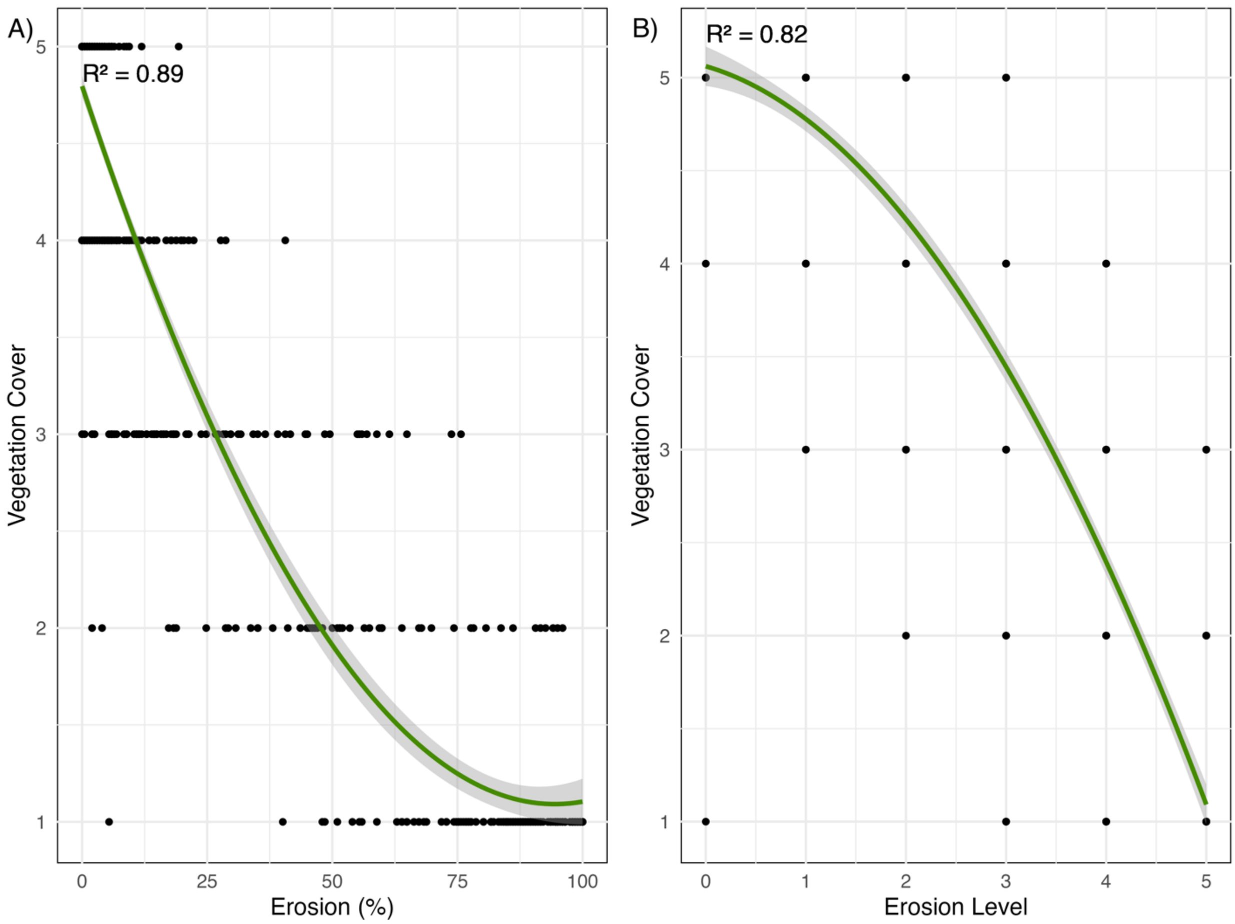

Figure 7 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship between Vegetation cover levels (1 to 5) and the two different erosion factors measured, Erosion (%) from Line Point Intersect (LPI) estimated from bare ground percentages, and Erosion level (0 to 5) estimated visually in the field.

Erosion % explained 89% (R2 = 0.891) and erosion level 82% (R2 = 0.822) of variation vegetation cover among plots, suggesting a very strong relationship. As erosion % increased the vegetation cover rapidly decreased down to 34-66% cover at ~25% erosion, 11-33% coverage at 50% erosion, and eventually no evident vegetation cover at around 90% erosion (*Figure 7A)*. The changes were quite rapid in a similar manner for the relationship with erosion levels, where the vegetation cover levels had a decreasing respond in correspondence to the increasing erosion levels. The mean vegetation coverage was reached at erosion level 3.5 and at level 5 erosion the vegetation cover was fully absent (*Figure 7B*).

Erosion % and erosion level significantly affected the canopy cover (CC), basal cover (BC), and stone cover (SC) (erosion %: CC, estimate = -0.827, SE = 0.021, $p < 0.001$; BC, estimate = -1.080, SE = 0.013, $p < 0.001$; SC, estimate = 0.062, SE = 0.008, $p < 0.001$; *Figure 8*, and erosion level: CC, estimate = -16.206, SE = 0.454, $p < 0.001$; BC, estimate = -19.874, SE = 0.439, $p < 0.001$; SC, estimate = 2.129, SE = 0.159, $p < 0.001$; *Figure 9*).

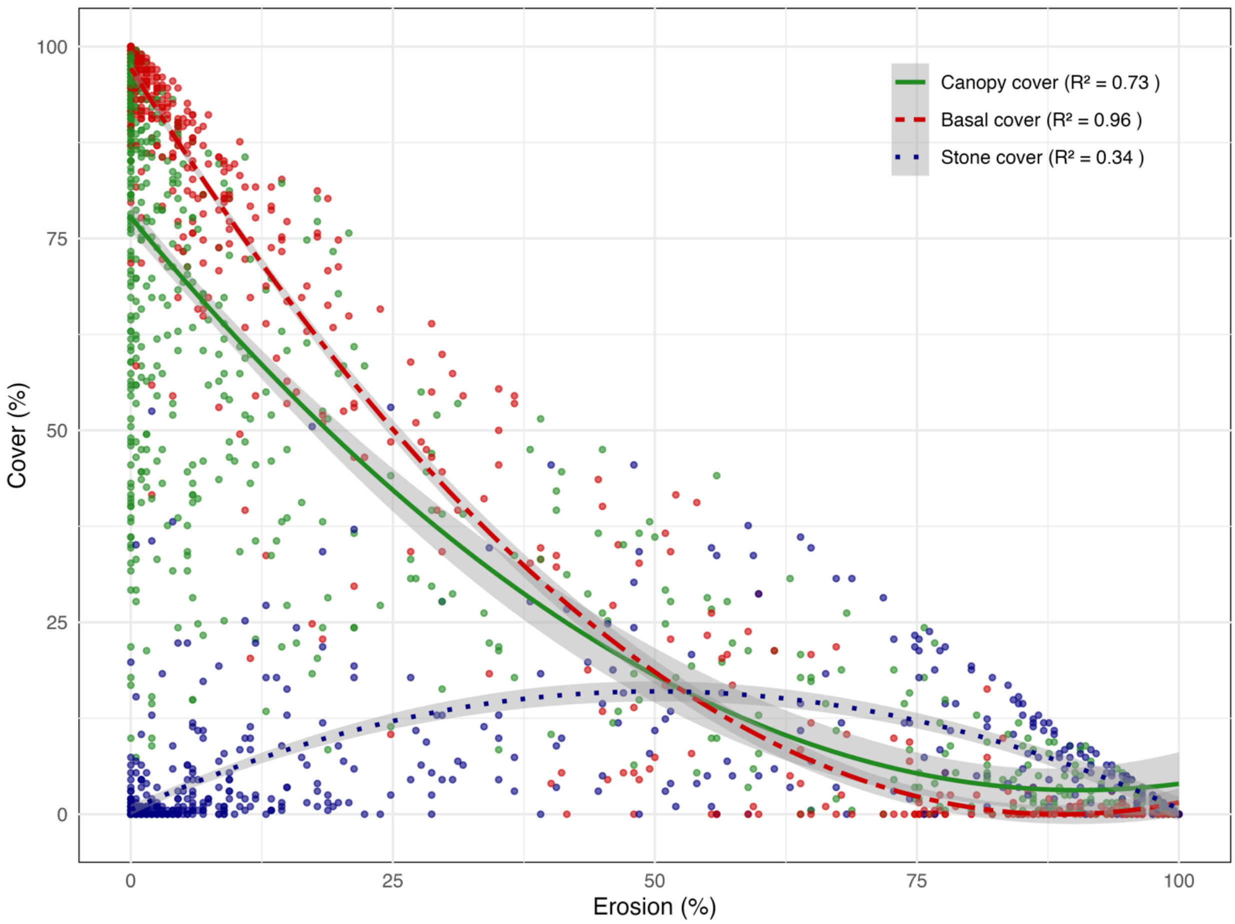

Figure 8 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship of Canopy cover (%), Basal cover (%) and Stone base cover (%) between Erosion (%) measured from Line Point Intersect (LPI) and estimated from bare ground percentages.

The model for the response of canopy cover to erosion % explained 73% (R^2 = 0.729) of the variability, suggesting a strong inverse relationship, 96% (R^2 = 0.959) of the variability in basal cover was explained by erosion %, representing an extremely strong inverse relationship, and finally, 34% (R^2 = 0.339) of the variability in stone cover was explained by erosion %, suggesting a weaker moderate relationship (*Figure 8*).

Basal cover decreased quite rapidly in response to the erosion % reaching a coverage of 50% at only 25% erosion, and at an erosion cover of ~85% the basal cover was fully absent. The canopy cover decreased as well as a response to the erosion % reaching a 50% coverage slightly quicker than the basal cover at approximately 20% erosion. However, the canopy cover decreased at a less rapid rate then the basal cover, and it still persisted to some extent around the highest % of erosion. The stone base cover was initially nonexistent where no erosion was occurring, then as erosion increased the stone cover increased as well. However, at around 50% erosion the stone cover started to steadily decrease again resulting in the absence of any stone cover as erosion reached 100%.

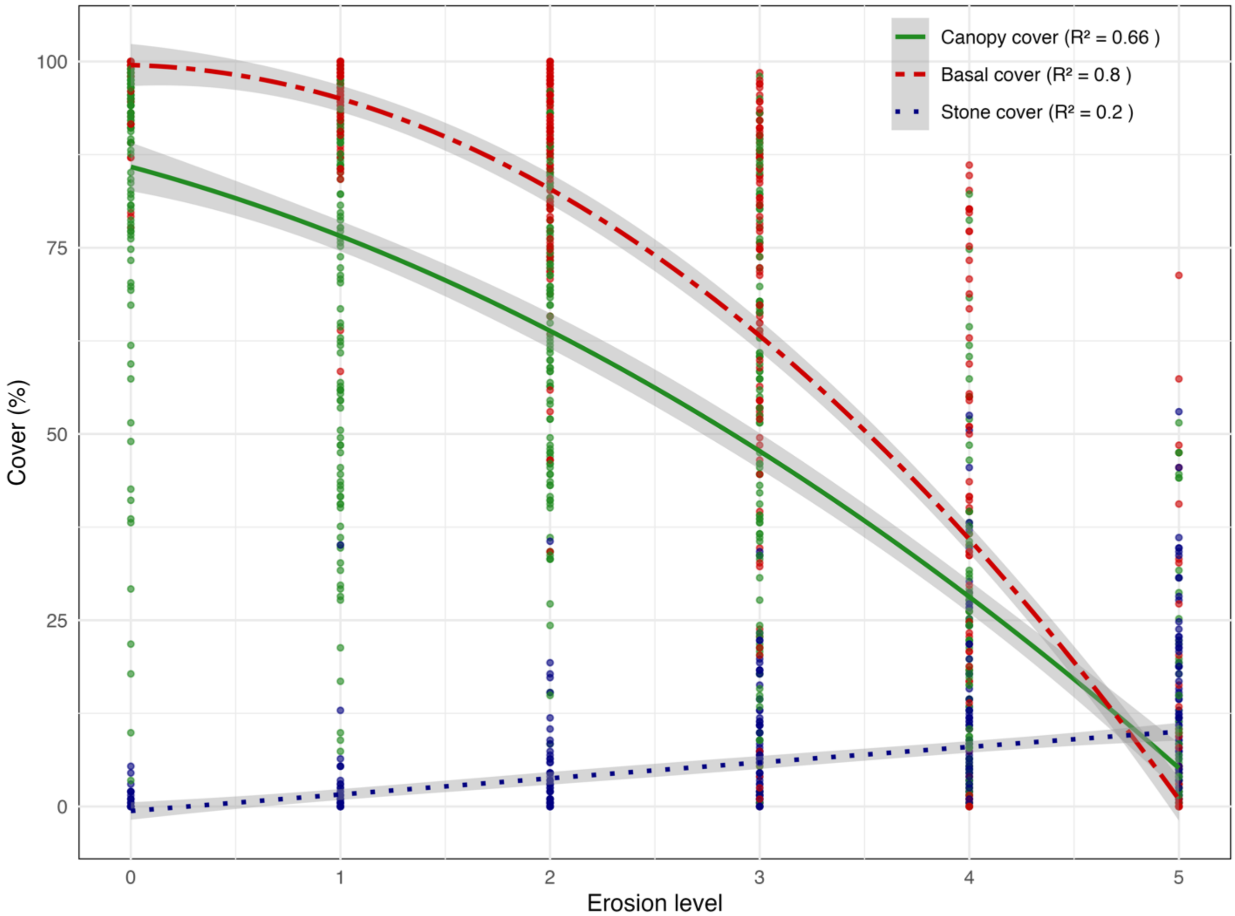

Figure 9 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship of Canopy cover (%), Basal cover (%) and Stone base cover (%) between Erosion level (0 to 5) estimated visually in the field.

The model for the response of canopy cover to erosion level explained 66% (R^2 = 0.661) of the variability, suggesting a strong negative relationship, 80% (R^2 = 0.80) of the variability in basal cover was explained by erosion level, representing a very strong negative relationship, and 20% (R^2 = 0.203) of the variability in stone cover was explained by erosion level, suggesting a weak relationship (*Figure 9*).

The basal cover decreased steadily until reaching 50% coverage at an erosion level of 3.5 and then ultimately becoming fully absent at level 5 erosion. The canopy cover decreased constantly before reaching 50% coverage at the erosion level of approximately 3 and then reaching very low percentages at the highest level of erosion. The stone base cover responded in a different manner as it increased slowly but steadily throughout the increasing erosion levels and reached a peak at level 5 erosion.

3.3. Relationship between erosion and plant community structure and species composition

Table 9 - Linear regression model between the erosion factors and plant community structure, including estimates, standard error and p- values from the linear regression model for the relationship between erosion % and erosion level with species richness and the number of functional groups. Relationships resulting in a significant relationship with the erosion factors (p<0.001) are presented in bold.

Disturbance relationship	Estimate	Standard error	p - value
Species richness ~ Erosion %	-0.233	0.010	**2×10^{-16}**
Species richness ~ Erosion level	-3.176	0.248	**2×10^{-16}**
No. functional groups ~ Erosion %	-0.030	0.001	**2×10^{-16}**
No. functional groups ~ Erosion level	-0.422	0.031	**2×10^{-16}**

There was a significant negative relationship between soil erosion and both species richness and plant community structure (*Table 9*).

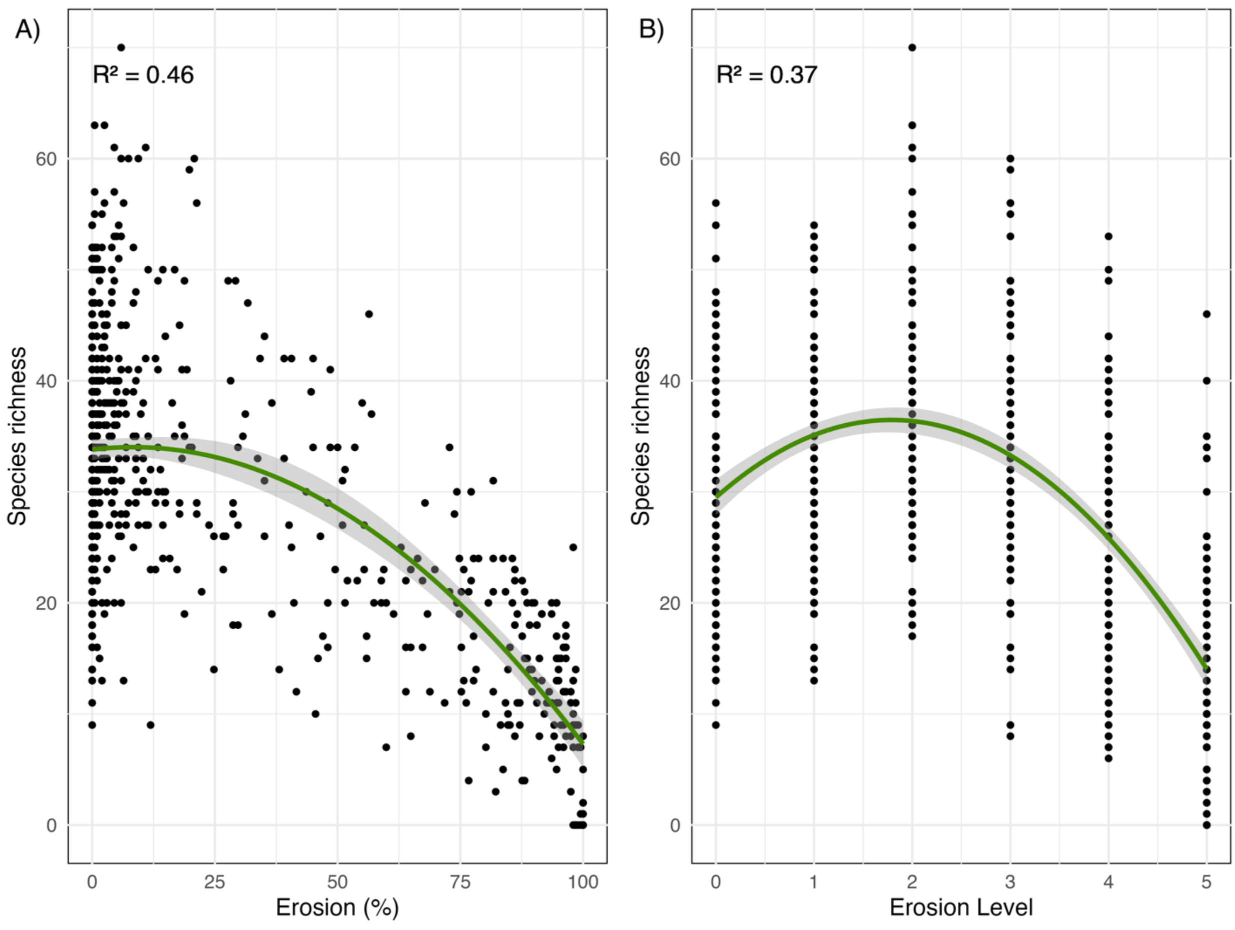

Figure 10 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship between species richness (SR = 100m^2) and the two different erosion factors measured, Erosion (%) from Line Point Intersect (LPI) estimated from bare ground percentages), and Erosion level (0 to 5) estimated visually in the field.

Erosion % explained 46% ($R^2 = 0.455$) of the variability in species richness, suggesting a moderate relationship between the variables, and 37% ($R^2 = 0.373$) of the variability in species richness was explained by erosion level, indicating a slightly weaker relationship (*Figure 10*).

As the erosion % increases the species richness decreases steadily. At 0% erosion the number of species present in a plot ranged from approximately 10 species up to over 70 species. As the erosion increased the number decreased and the range becomes narrower. At 50% erosion the number ranges from ~10 to ~45, and at 100% erosion the range spans from 0 species to ~20 (*Figure 10A*). Species richness responded slightly differently to the erosion level estimations as the number of species initially increases with higher levels of erosion reaching a mean peak at level 2 erosion. After that the number started to decrease, reaching the lowest species richness at level 5 erosion. In comparison, the number of species at level 0 ranged from ~10 to ~50, at level 2 erosion, when the species richness reached a peak, the number ranged from approximately 20 up to 70 species, and lastly at the highest level of erosion, level 5, the range was from 0 species up to ~45 (*Figure 10B*).

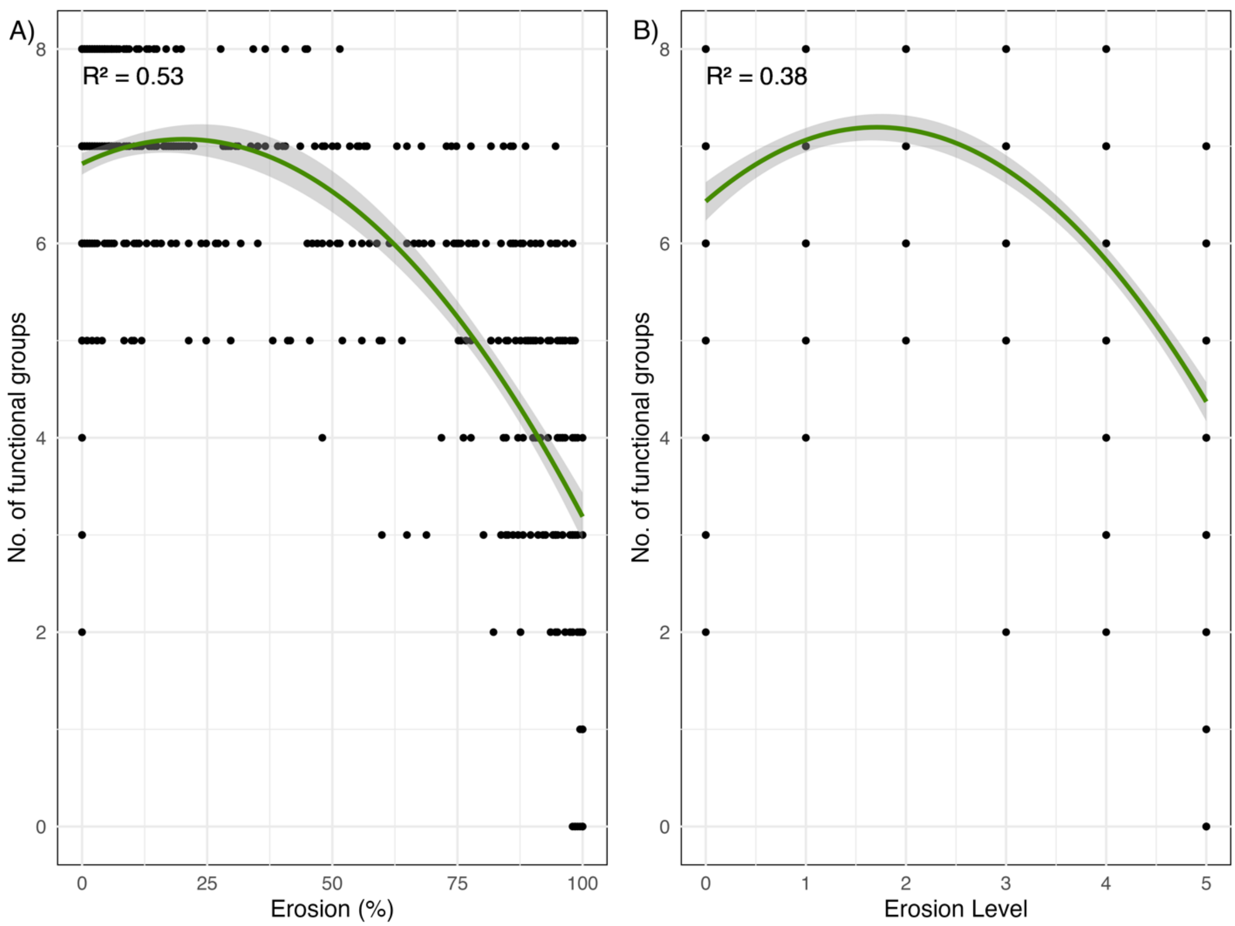

Figure 11 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship between the number of plant functional groups present and the two different erosion factors measured, Erosion (%) from Line Point Intersect (LPI) estimated from bare ground percentages, and Erosion level (0 to 5) estimated visually in the field.

Erosion % explained 53% (R^2 = 0.528) of the variability in the number of functional groups, suggesting a moderate relationship (*Figure 11A*). Erosion level explained 38% (R^2 = 0.384) of the variability, representing a weaker relationship (*Figure 11B*). Both relationships responded in a similar way where initially the number of functional groups increased slightly with increasing erosion, reaching a certain peak before declining towards total erosion. The number of functional groups reached a peak at 25% erosion and the same peak is reached at approximately level 2 erosion. At maximum erosion % the number of functional groups was around 3, and at the same maximum but for the erosion levels, the number of groups was ~4.5.

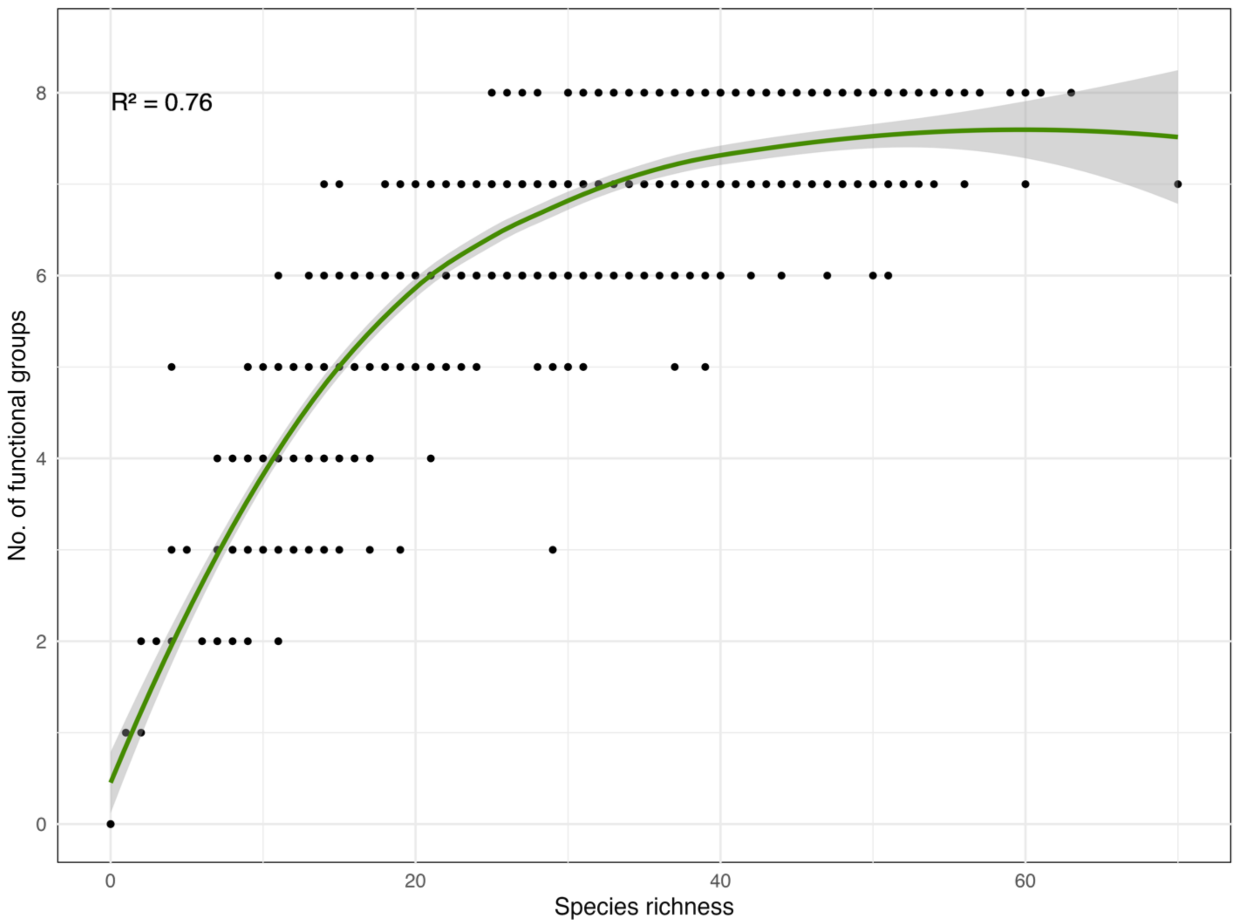

Figure 12 - Linear model made from all measurements gathered from over 700 plots using polynomial regression and a relevant confidence interval showing the relationship between species richness and the number of plant functional groups present.

There was a strong significant positive relationship between species richness and number of functional groups (estimate = 6.156, SD = 0.194, $p < 0.001$). With species richness explaining around 76% (R^2 = 0.757) of the variability in number of functional groups (*Figure 12*).

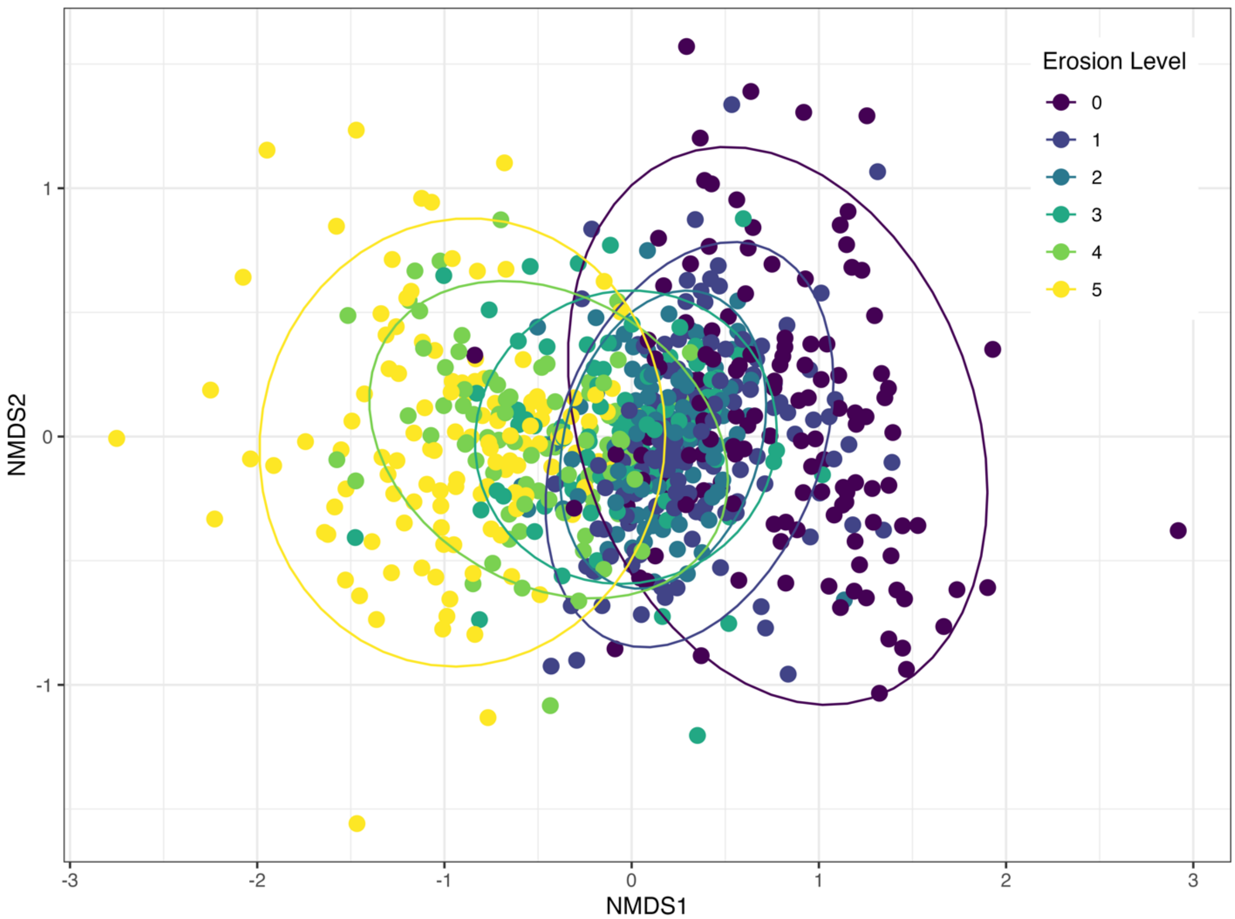

Figure 13 - Non-metric multidimensional scaling (NMDS) plot for the 692 study plots that had one or more species present in relation to erosion level estimated in the field for each corresponding plot. The species composition is grouped by color, based on corresponding erosion level, and corresponding circles, assuming a multivariate normal distribution.

There was a systematic change in species composition in relation to varying erosion levels for each monitoring plot (*Figure 13*). Erosion level influenced plot species composition as plots with the same erosion level were generally closer to each other in the NMDS plot. Species composition seems to change gradually from plots with no erosion (to the right in the plot), to plots with intermediate erosion (middle of the plot) to plots with severe erosion (left in the plot). Also, variations in species composition among plots at the same erosion level was the highest for plots with no (level 0) or severe (level 5) erosion (the largest circles on the plot).

4. Discussions

In this study, I examined how soil erosion in Iceland is influenced by various environmental variables and how erosion affects plant community structure, species richness and composition, in various habitats. Furthermore, I compared two different methods for estimating the amount of erosion and whether they result in different responses to the environmental factors, as well as how the two measurement techniques perform in relation to accuracy for future re-measurements.

The main results suggest that erosion risk is significantly affected by increasing levels of elevation, habitat type, and soil type. Moreover, vegetation cover, species richness, and the number of plant functional groups present within each plot all had a significant negative correlation to erosion, with increased richness at intermediate disturbance levels. In addition, species composition corresponds to the level of erosion present. Furthermore, this research highlights the similarities between the two methods GróLind uses to estimate soil erosion and how they correspond to each other when representing the severity of erosion within each area.

Estimating and evaluating the degree of land degradation and soil erosion within habitats is important to fully understand the current productivity within an area (Arnalds *et al*., 2001b). Furthermore, recurrent monitoring is essential to observe any degradation changes that might occur over time, whether that change may result in positive or negative responses. The responses can then possibly be connected to ongoing environmental impacts, including anthropogenic influence.

The current condition of Icelandic ecosystems seems to be highly dependent on and affected by elevation above sea level (*Figure 6*). Increasing elevations result in a correspond with increase in the degree of erosion for both estimation methods, % and level. At 600 m elevation more than half of the areas were exposed to erosion, with the highest elevations being completely exposed to erosion. However, complete exposure to erosion could also be detected at lower elevations, suggesting that erosion is also being induced by other environmental factors than elevation. A significant relationship was found between elevation and both erosion estimations, resulting in a comparable response between both values and the environmental factor, elevation. These results indicate that land conditions are generally poorer for all areas of Iceland at higher elevations, being more prone to occurring and complete erosion. Increasing elevations reflect as induced environmental constraints, affecting ecosystem resilience towards climatic impact and land use pressures, *e.g.*, grazing. These results are in line with conclusions gathered by other studies pointing out a reduction of plant growth and vegetation cover with increasing elevations (Magnusson & Svavarsdottir, 2007; Alewell *et al.*, 2008; Draebing *et al*., 2022; Arnalds *et al*., 2023).

There were pronounced differences between the various habitat types and the amount of observed erosion. Most importantly, fell fields, moraines, and sand habitats (L1), as well as river plains (L4), displayed the highest extent of erosion. The overall results from the comparisons between the various habitat types and the amount of observed erosion suggested that habitats with scarcer vegetation covers are generally eroded to a larger extent than habitats where a lot of vegetation is present. Habitats such as grasslands, heathlands, and wetlands all showed little to no erosion suggesting a stabilizing effect of vegetation on habitat resistance to erosion (*Table 5-6*). This is further supported when looking at the comparisons of erosion between different soil types. Soil types with high sand contents were generally more eroded than soil types more suitable for vegetation such as clay soils. Here, soils with high contents of clay generally exhibited low erosion which might be due to the clay having higher cohesion capabilities making it more resistant to erosion then the sandy soils (Firoozi *et al.*, 2016). Clay soils also often support denser vegetation covers due to more optimal nutrient and moisture contents. Eroded areas might lack in the presence of clay due to the parent material not weathering into clay particles (Velde & Meunier, 2008). The loamier soils possess higher contents of soil organic matter (SOM) and nutrients due to the presence of more vegetation, with increased value of occurring degradation of organic matter (OM), allowing for the constant formation of newly introduced humus into the system, making it less sensitive to erosion inducing factors (Bogunovic *et al*., 2014).

Erosion had a negative effect on the presence of vegetation. The response of vegetation cover towards both erosion factors is represented in the same manner, with a generally full vegetation cover at low erosion stages, until being fully reduced to no apparent vegetation cover representing an extensively eroded area. Similarly, canopy cover and basal cover decreased with more erosion (*Figure 8-9*), corresponding to the previous responses of estimated vegetation covers towards erosion. This is not surprising as erosion in this study is defined as, either the percentage of bare soil (soil not covered by vegetation) or visually as the amount of bare soil and indications of soil movement. The presence of rocky terrains (stone cover) was also significantly affected by erosion. Stone cover initially increased with erosion % , reaching a certain peak around 50% erosion and then decreased again. This suggest that at certain stages of erosion, the stone cover is consistent with the decreasing vegetation cover, perhaps influencing the erosion itself (Toy *et al.*, 2002), before becoming less eminent in highly erosion prone areas, possibly due to further climatic and environmental factors. However, the relationship between the stone cover and erosion level results in a positive linear relationship, where the stone cover increases with increasing levels of erosion. Stone covers on eroded soils can affect soil erosion processes by increasing infiltration, decreasing runoff, and influencing the overall hydrological processes occurring with soil erosion (Zhang *et al*., 2016). These stone covers can protect the soil, increasing the roughness of the surface, prevent sediment transportation to an extent, and reduce external environmental impact (Omidvar *et al*., 2019), reducing soil erosion to up to approximately 70% in some cases (Lv *et al*., 2019). Other studies have also suggested that stone cover can induce runoff and sediment erosion (Rodrigo-Comino *et al.,* 2017).

Erosion percentage had a negative effect on species richness, but the number of species did not start to decline until around 25% erosion. For erosion level, species richness was the highest at erosion level 2 before declining rapidly (*Figure 10*). There was also a negative relationship between erosion and the number of functional species (*Figure 11*). Number of functional groups showed a similar pattern to the species richness, slightly increasing until erosion reached 25% or level 2 and then rapidly decreasing as erosion increased. Number of functional groups was also positively associated with species richness (*Figure 12*). These results might suggest the presence of a species richness threshold in response to erosion, at around 25% erosion, and that loss of species richness and diversity may reduce soil erosion resistance. These thresholds generally occur in systems that have failed to recover from disturbance (Barrio *et al.*, 2018). Perhaps a suitable amount of disturbance can be present for different species to still thrive (Berendse *et al*., 2015; Bendix *et al*., 2017). Less vegetation cover might not resonate with the diversity still present within the area, resulting in a response where less species richness is less vulnerable (Helm *et al*., 2005). These results might correspond to the intermediate disturbance hypothesis (IDH), which suggests that species diversity reaches a threshold, is maximized, in the presence of intermediate levels of disturbance, *i.e.*, intermediate levels of soil erosion (Wilkinson, 1999). This hypothesis was initially proposed by Grime (1973) and according to the hypothesis, species communities reach a certain maximum in diversity at disturbance that is considered to be at an intermediate stage. This occurs due to a balance between colonization and competition, allowing for a larger number of coexisting species (Moi *et al*., 2020). This hypothesis corresponds to the results for the relationships between species richness and the number of functional groups present in relation to erosion, where a maximum is reached at an intermediate level of disturbance.

The species composition was estimated in relation to varying erosion levels in order to evaluate whether specific species compositions within all the measured plots followed a certain pattern in relation to erosion (*Figure 13*). The results indicate that certain species group together forming specific compositions separating them from other groups in relation to erosion. Some species seem to be more frequent at low levels of erosion, whilst other species thrive at the higher levels. Then there are some overlapping species, as well as varying compositions that are more abundant at intermediate erosion levels. This suggests that erosion has an effect on determining the type of species growing and thriving within an area, possibly where more resistant species occur where erosion is more extreme, while some species might be more vulnerable to the disturbance, occurring more in highly vegetated areas. The different compositions follow a strategic direction with each increasing erosion level. Furthermore, there is increased species variability within the compositions at level 0 and level 5. This might suggest that at high levels of erosion only the most resistant species survive, and at sites with no erosion, the best competitors become dominant and shape the community, which species resist or dominate seams to vary allot probably due to variability in environmental factors. At intermediate erosion, the species composition seems to be more congruent among sites. Soil erosion, and other disturbances, can act as a key factor when determining the species composition within specific areas (Čepelová & Münzbergová, 2012; Sharma *et al*., 2023).

Here, two methods of measuring erosion were used, one direct measurement (erosion %) and one based on visual estimation (erosion level). There was a high correlation between the two methods (*Figure 5*), making them both applicable when evaluating the erosion level of an area. The two estimates of erosion also responded similarly to environmental variables, with few noticeable differences, and had a similar relationship with plant composition. The main differences were more imprecise results from the erosion level in comparison to erosion %, where more broad data did not reach the same significance levels in correlations and relationships with other variables as erosion %, mainly when it comes to the categorical environmental variables, *e.g.*, habitat type and soil type. Moreover, erosion level resulted in more insensitive responses at high levels of erosion as compared to erosion %, which provided higher resolution at high disturbance levels. However, it is important to recognize that erosion % is based on much more detailed measurements with less apparent deviation, whilst erosion level is based on few increment levels which might not fully represent small changes occurring within the area, as well as possibly being more prone to subjectivity. If the goal is to get an estimation of the general state of an area, visual estimation of erosion levels, is a quick, cheap and easy method to apply. Furthermore, estimating erosion levels allow for comparisons with other research, since that method is commonly used for general erosion estimation for areas in Iceland (Arnalds *et al*., 2023). When it comes to monitoring, the crucial factor is to detect change over time. Visually estimating erosion levels will not allow for detection of subtle changes. An erosion estimation method based on more data points provide more precise information regarding the stage of erosion within an area and might be a more reliable method for detecting future changes in erosion over a specific period of time. The erosion % method allows for the detection of changes in the form of slight deviations from the initial calculations. Therefore, putting resources towards measuring erosion in more detail might be needed to detect those changes. Performing LPI measurements at 0.5 m intervals (N = 202), like in the GróLind methodology, allowed for a 95% chance of the calculations being 0-7% from the correct average measurements, hence, providing a very accurate representation for erosion, *i.e.*, the amount of bare soil (Marteinsdottir *et al.*, 2021) and allowing for the detection of subtle changes over time.

This research project was performed and worked in collaboration with Land and Forest Iceland, which is a new joint institute of the Soil Conservation Service of Iceland and the Icelandic Forestry Agency. A large data set was compiled from information collected in the years of 2019 to 2023 from over 700 plots distributed all over Iceland. The plots are a part of an ongoing monitoring programme, GróLind, which is a long-term vegetation and soil monitoring programme that has just completed its first round of data collection and therefore, the first round of monitoring. The next steps of the project will be to revisit and remeasure all the currently established monitoring plots, which will allow for detection of possible changes over time. The data, from GróLind, used in this project has not been extensively analyzed before. Thus, this research project not only aimed at answering the research questions regarding the effects of different environmental factors on erosion, the influence of erosion on the plant community and the differences between various erosion factors, but also to explore how the large data set of GróLind is suited to answer the above questions. With GróLind being a relatively new project based on adaptive monitoring, and the first if its kind in Iceland, it is important to explore the data set and use that in a study like this, to validate the methods and practices being used and further certifying their applicability for the future of the project. The results from this project were in line with ecological theory, indicating that the GróLind project methods, are monitoring relevant parameters. This project also underlines the importance of choosing the right method for each project. If the aim of a project is to get information on the general state of an area, visually estimating soil erosion might be enough, while more detailed and intricate methods are needed for estimating gradual changes over time.

5. References

ACIA. (2004). Impacts of a Warming Arctic: Arctic Climate Assessment. *ACIA Overview Report*. Cambridge University Press, 140.

Alewell, C., Meusburger, K., Brodbeck, M., & Bänninger, D. (2008). Methods to describe and predict soil erosion in mountain regions. *Landscape and Urban Planning, 88*(2-4), 46–53. https://doi.org/10.1016/j.landurbplan.2008.08.007

Aradottir, A. L. & Eysteinsson, T. (2005). Restoration of birch woodlands in Iceland. In: JA Stanturf, P Madsen (eds) *Restoration of boreal and temperate forests*, 195–209. CRC Press, Boca Raton, Florida. https://doi.org/10.1201/9780203497784

Aradottir, A. L. & Johannsson, M. H. (2006). Ecological restoration with native species in Iceland. *Soil-Bioengineering: Ecological Restoration with Native Plant and Seed Material*, 175-179. https://www.cabidigitallibrary.org/doi/pdf/10.5555/20073117064

Aradottir, A. L. (2007). Restoration of birch and willow woodland on eroded areas. In: Halldorsson, G., Oddsdottir, E. S., Eggertsson, O. (eds), *Effects of afforestation on ecosystems, landscape and rural development.* TemaNord 2007, 508, 67–74. Nordic Council of Ministers, Copenhagen, Denmark

Aradottir, A. L., Petursdottir, T., Halldorsson, G., Svavarsdottir, K., & Arnalds, O. (2013). Drivers of Ecological Restoration: Lessons from a Century of Restoration in Iceland. *Ecology and Society*, 18(4). http://www.jstor.org/stable/26269410

Aradottir, A.L. & Hagen, D. (2013). Ecological restoration: approaches and impacts on vegetation, soils and society. *Advances in Agronomy*, 120, 173–222. https://doi.org/10.1016/B978-0-12-407686-0.00003-8

Aradottir, A. L., Svavarsdottir, K., Thorhallsdottir, Þ. E., Behrend, A. M., Snorrason, A., Traustason, B., Oddsdottir, E. S., Johannsdottir, G., Thorsson, J., Thorlaksdottir, J. S., Arnalds, O., Palsson, S., Sanchez, S., & Petursdottir, Þ. (2022). Endurheims birkivistkerfa - áskoranir, leiðir og ávinningur (Birkivist). *Rit Mógilsár*, 47, 37-41.

Arnalds, O., Gisladóttir, F. O., & Sigurjonsson, H. (2001a). Sandy deserts of Iceland: an overview. *Journal of Arid Environments*, 47(3), 359-371. https://doi.org/10.1006/jare.2000.0680

Arnalds, O., Thorarinsdóttir, E. F., Metusalemsson, S., Jonsson, A., Gretarsson, E., & Arnason, A. (2001b). Soil erosion in Iceland. The Soil Conservation Service and the Agricultural Research Institute, Reykjavík, Iceland. [online] URL: http://www.rala.is/desert/Soil%20Erosion.pdf

Arnalds, O., and B. H. Barkarson. (2003). Soil erosion and land use policy in Iceland in relation to sheep grazing and government subsidies. *Environmental Science & Policy*, 6, 105-113. http://dx.doi.org/10.1016/S1462-9011(02)00115-6

Arnalds, O. (2004). Hin Íslenska jarðvegsauðlind. Fræðaþing landbúnaðarins 2004, 94-102.

Arnalds, O. & Oskarsson, H. (2009). Íslenskt Jarðvegskort. Náttúrufræðingurinn. 78(3-4), 107-121.

Arnalds, O. (2015). The Soils of Iceland. In World Soils Book Series. Springer Netherlands. https://doi.org/10.1007/978-94-017-9621-7

Arnalds, O. & Aradottir, A. L. (2015). Að lesa og lækna landið. Landvernd, Landgræðsla ríkisins og Landbúnaðarháskóli Íslands: Landvernd, Landgræðsla ríkisins og Landbúnaðarháskóli Íslands.

Babington, C. C. (1870). A Revision of the Flora of Iceland. *Journal of the Linnean Society of London Botany*, 11(53), 282–348. https://doi.org/10.1111/j.1095-8339.1870.tb00067.x

Barrio, I. C., Hik, D. S., Thorsson, J., Svavarsdottir, K., Marteinsdottir, B., & Jonsdóttir, I. S. (2018). The sheep in wolf's clothing? Recognizing threats for land degradation in Iceland using state-and-transition models. *Land Degradation & Development*, 29(6), 1714–1725. https://doi.org/10.1002/ldr.2978

Bendix, J., Wiley, J. J., & Commons, M. G. (2017). Intermediate disturbance and patterns of species richness. *Physical Geography, 38*(5), 393–403. https://doi.org/10.1080/02723646.2017.1327269

Beraldi-Campesi, H. (2013). Early life on land and the first terrestrial ecosystems. *Ecological processes*, 2(1). https://doi.org/10.1186/2192-1709-2-1

Berendse, F., van Ruijven, J., Jongejans, E., & Keesstra, S. (2015). Loss of Plant Species Diversity Reduces Soil Erosion Resistance. *Ecosystems, 18*(5), 881–888. https://doi.org/10.1007/s10021-015-9869-6

Bergþorsson, P. (1996). Temperature and vegetation. *Icelandic Agricultural Sciences*, 10, 141–164.

Bjornsson, H., Jonsson, T., Gylfadottir, S. S., & Olason, E. Ö. (2007). Mapping the annual cycle of temperature in Iceland. *Meterologische Zeitschrift*, 16(1), 45-56. https://doi.org/10.1127/0941-2948/2007/0175

Bjornsson, H. (2016). The Glaciers of Iceland: A Historical Cultural and Scientific Overview. *Atlantis Advances in Quaternary Science*.

Blanco-Canqui, H., Shapiro, C. A., Wortmann, C. S., Drijber, R. A., Mamo, M., Shaver, T. M., & Ferguson, R. B. (2013). Soil organic carbon: The value to soil properties. *Journal of Soil and Water Conservation*, 68(5), 129A–134A. https://doi:10.2489/jswc.68.5.129a.

Bogunovic, I., Mesic, M., Zgorelec, Z., Jurisic, A., & Bilandzija, D. (2014). Spatial variation of soil nutrients on sandy-loam soil. *Soil and Tillage Research, 144*, 174–183. https://doi.org/10.1016/j.still.2014.07.020

Bradley, R. S. (2015). Paleoclimatology: Reconstructing Climates of the Quaternary, Third Edition. https://doi.org/10.1016/C2009-0-18310-1

Bridges, E.M. (2001). Response to Land Degradation (1st ed.). CRC Press. https://doi.org/10.1201/9780429187957

Brynleifsdottir, S. J. (2012). Bændur græða landið. Yearly report, 2011. LR-2012/13. Soil Conservation Service of Iceland, Gunnarsholt, Iceland. [online] URL: http://www.land.is/images/pdf-documents/arsskyrslur/bgl_skyrslur/bgl-skyrsla.pdf

Butchart, S. H. M., Resit Akçakaya, H., Chanson, J., Baillie, J. E. M., Collen, B., Quader, S., Turner, W. R., Amin, R., Stuart, S. N., & Hilton-Taylor, C. (2007). Improvements to the Red List Index. P*LoS ONE, 2*(1), e140. https://doi.org/10.1371/journal.pone.0000140

Byrne, M., Stone, L., & Millar, M. A. (2011). Assessing genetic risk in revegetation. *Journal of Applied Ecology*, 48(6), 1365-1373. https://doi.org/10.1111/j.1365-2664.2011.02045.x

Callaghan, T. V., Björn, L. O., Chernov, Y. I., Chapin III, F. S., Christensen, T., Huntley, B., Ims, R., Johansson, M., Jolly, D., Matveyeva, N., Panikov, N., Oechel, W., & Shaver, G. (2005). Arctic tundra and Polar Desert Ecosystems. In ACIA. *Arctic Climate Impact Assessment* (Ed.), Arctic Climate Impact Assessment Cambridge University Press.

Carter, M.R., & Stewart, B.A. (1995). Structure and Organic Matter Storage in Agricultural Soils, First edition. CRC Press. https://doi.org/10.1201/9781003075561

Čepelová, B., & Münzbergová, Z. (2012). Factors determining the plant species diversity and species composition in a suburban landscape. *Landscape and Urban Planning, 106*(4), 336–346. https://doi.org/10.1016/j.landurbplan.2012.04.008

Chytrý, M., Tichý, L., Hennekens, S. M., Knollová, I., Janssen, J. A. M., Rodwell, J. S., Peterka, T., Marcenò, C., Landucci, F., Danihelka, J., Hájek, M., Dengler, J., Novák, P., Zukal, D., Jiménez-Alfaro, B., Mucina, L., Abdulhak, S., Aćić, S., Agrillo, E., & Attorre, F. (2020). EUNIS Habitat Classification: Expert system, characteristic species combinations and distribution maps of European habitats. *Applied Vegetation Science, 23*(4), 648–675. https://doi.org/10.1111/avsc.12519

Crofts, R. (2011). Healing the land: the story of land reclamation and soil conservation in Iceland. Soil Conservation Service of Iceland, Gunnarsholt, Iceland. https://api.semanticscholar.org/CorpusID:129983217

Ding, C., Haieh, C., Wu, Q., & Pedram, M. (1996). Stratified random sampling for power estimation. Proceedings of International Conference on Computer Aided Design. https://doi.org/10.1109/iccad.1996.569913

D'Odorico, P., Bhattachan, A., Davis, K. F., Ravi, S., & Runyan, C. W. (2013). Global desertification: Drivers and feedbacks. *Advances in Water Resources, 51*, 326–344. https://doi.org/10.1016/j.advwatres.2012.01.013

Doran, J. W., Jones, A. J., Bezdicek, D. F., Papendick, R. I., & Lal, R. (1996). Introduction: Importance of Soil Quality to Health and Sustainable Land Management. *Methods for Assessing Soil Quality*, 49. https://doi.org/10.2136/sssaspecpub49.introduction

Draebing, D., Mayer, T., Jacobs, B., & McColl, S. T. (2022). Alpine rock wall erosion patterns follow elevation-dependent climate trajectories. *Communications Earth & Environment, 3*(1). https://doi.org/10.1038/s43247-022-00348-2

Dubey, A., Malla, M. A., Khan, F., Chowdhary, K., Yadav, S., Kumar, A., Sharma, S., Khare, P. K., & Khan, M. L. (2019). Soil microbiome: a key player for conservation of soil health under changing climate. *Biodiversity and Conservation*, 28(8-9), 2405–2429. https://doi.org/10.1007/s10531-019-01760-5

Einarsson, M. A. (1976). Veðurfar á Íslandi [Climate in Iceland]. Iðunn, Reykjavík, 150 p.

Elhakim, A. F. (2016). Estimation of soil permeability. *Alexandria Engineering Journal, 55*(3), 2631-2638. https://doi.org/10.1016/j.aej.2016.07.034

Elven, R., Murray, D. F., Razzhivin, V., Yurtsev, B. A. (2011). Checklist of the panarctic flora (PAF). Available: http://www.nhm.uio.no/english/research/infrastructure/paf/introduction. Accessed 2014 April 22.

ESRI. (2024). ArcGIS Survey123 (version 3.21.62) [Software]. Environmental Systems Research Institute, Inc. Available at https://survey123.arcgis.com

European Environment Agency (EEA). (2024). EUNIS habitat classification. Retrieved from https://eunis.eea.europa.eu/habitats.jsp

Fath, B. D. (2019). Encyclopedia of Ecology. Second Edition. https://www.sciencedirect.com/referencework/9780444641304/encyclopedia-ofecology

Finch, H. J. S., Samuel, A. M., & Lane, G. P. F. (2014). Soils and soil management. *Lockhart & Wiseman's Crop Husbandry Including Grassland,* 37–62. https://doi.org/10.1533/9781782423928.1.37

Firoozi, A. A., Firoozi, A. A., & Baghini, M. S. (2016). A review of clayey soils. *Asian Journal of Applied Sciences* (ISSN: 2321–0893), 4(06).

Godínez-Alvarez, H., Herrick, J. E., Mattocks, M., Toledo, D., & Van Zee, J. (2009). Comparison of three vegetation monitoring methods: Their relative utility for ecological assessment and monitoring. *Ecological Indicators, 9*(5), 1001–1008. https://doi.org/10.1016/j.ecolind.2008.11.011

Grime, J. P. (1973). Control of species density in herbaceous vegetation. *Journal of Environmental Management, 1*. 151–167.

Grönlund, C. H. (1881). Islands Flora. Copenhagen: Gyldendalske Boghandels Forlag.

Hartemink, A. E., Zhang, Y., Bockheim, J. G., Curi, N., Silva, S. H. G., Grauer-Gray, J., Lowe, D. J., & Krasilnikov, P. (2020). Chapter Three - Soil horizon variation: A review. *Advances in Agronomy, 160*(1), 125-185. https:/24/doi.org/10.1016/bs.agron.2019.10.003

Helm, A., Hanski, I., & Partel, M. (2005). Slow response of plant species richness to habitat loss and fragmentation. *Ecology Letters,* 051109031307003. https://doi.org/10.1111/j.1461-0248.2005.00841.x

Hobbs, R. J. & Norton, D. A. (1996). Towards a conceptual framework for restoration ecology. *Restoration Ecology*, *4*, 93-110. http://dx.doi.org/10.1111/j.1526-100X.1996.tb00112.x

Icelandic Institute of Natural History (IINH) (1996). Válisti 1 – Plöntur. https://utgafa.ni.is/valistar/valisti_1.pdf

Icelandic Institute of Natural History (IINH). (2024). Terrestrial habitat types. Retrieved from https://www.ni.is/en/flora-funga/habitat-types/terrestrial-habitat-types

Icelandic Institute of Natural History (IINH) (2024). Invasive plant species. https://www.ni.is/en/flora-funga/invasive-plant-species

International Union for Conservation of Nature. (2022). The IUCN Red List of Threatened Species. Version 2022-1. https://www.iucnredlist.org

Jiao, J., Zou, H., Jia, Y., & Wang, N. (2009). Research progress on the effects of soil erosion on vegetation. Acta Ecologica Sinica, 29(2), 85–91. https://doi.org/10.1016/j.chnaes.2009.05.001

Karlsson, G. (2000). Iceland's 1100 Years: History of a Marginal Society. London: Hurst &Company.

Kassambara, A. (2023). rstatix: Pipe-Friendly Framework for Basic Statistical Tests. R package version 0.7.2.

Kristinsson, H. (2010). The Icelandic Plant Handbook. Flowering Plants and Ferns of Iceland. Mál og menning, Reykjavík, 364.

Lal, R., Kimble, J.M., Follett, R.F., & Stewart, B.A. (1997). Soil Processes and the Carbon Cycle, First edition. CRC Press. https://doi.org/10.1201/9780203739273

Leblans, N. I. W., Sigurdsson, B. D., Vicca, S., Fu, Y., Penuelas, J., & Janssens, I. A. (2017). Phenological responses of Icelandic subarctic grasslands to short-term and long-term natural soil warming. *Global Change Biology*, *23*(11), 4932-4945. https://doi.org/10.1111/gcb.13749

Lindenmayer, D. B. & Likens, G. E. (2009). Adaptive monitoring: a new paradigm for long-term research and monitoring. *Trends in Ecology & Evolution*, *24*(9), 482-486. https://doi.org/10.1016/j.tree.2009.03.005

Lovett, G. M., Burns, D. A., Driscoll, C. T., Jenkins, J. C., Mitchell, M. J., Rustad, L., Shanley, J. B., Likens, G. E., & Haeuber, R. (2007). Who needs environmental monitoring? *Frontiers in Ecology and the Environment*, *5*(5), 253-260. https://doi.org/10.1890/1540-9295(2007)5[253:WNEM]2.0.CO;2

Lv, J., Luo, H., & Xie, Y. (2019). Effects of rock fragment content, size and cover on soil erosion dynamics of spoil heaps through multiple rainfall events. *CATENA*, *172*, 179–189. https://doi.org/10.1016/j.catena.2018.08.024

Macheyeki, A. S., Li, X., Kafumu, D. P., & Yuan, F. (2020). Elements of exploration geochemistry. *Applied Geochemistry*, 1–43. https://doi.org/10.1016/b978-0-12-819495-9.00001-3

Magnusson, S. H. (1997). Restoration of eroded areas in Iceland. In: Urbanska, K.M., Webb, N.R., Edwards, P.J.: *Restoration Ecology and Sustainable Development*, Cambridge University Press, Cambridge, 188-211.

Magnusson, B., Magnusson, S. H., & Sigurdsson, B. D. (2004). Plant succession in areas colonized by the introduced Nootka lupin in Iceland. In: Santen E., Hill, G.D.: *Wild and Cultivated Lupins from the Tropics to the Poles*. Proceedings of the 10th International Lupin Conference., Laugarvatn, Iceland, 19-24 June 2002, 170-177.

Magnusson, S.H. & Svavarsdóttir, K. (2007). Áhrif beitarfriðunar á framvindu gróðurs og jarðvegs á lítt grónu landi. Reykjavík: Náttúrufræðistofnun Íslands. *Fjölrit Nátturufrædistofnunar, 49*. Icelandic

Magnusson, B. (2010). NOBANIS - Invasive Alien Species Fact Sheet - Lupinus nootkatensis. Reykjavik: Icelandic Institute of Natural History. online] URL:http://www.nobanis.org/files/factsheets/Lupinus_nootkatensis. pdf, 11

Magnusson, B. (2019). Vistgerðalykill Náttúrufræðistofnunar Íslands - I. Vistgerðir á landi.

Marteinsdottir, B.,Thorarinsdottir, E. F., Halldorsson, G., Stefansson, J. H., Thorsson, J., Svavarsdóttir, K., Finnsdottir, R., & Jonsdóttir, I. S. (2021). "GróLind – Sustainable Land Use Based on Ecological Knowledge". IGC Proceedings (1993-2023). 25. https://uknowledge.uky.edu/igc/24/1/25

McGlinn, D. J., Xiao, X., May, F., Gotelli, N. J., Engel, T., Blowes, S. A., Knight, T. M., Purschke, O., Chase, J. M., & McGill, B. J. (2018). MoB (Measurement of Biodiversity): a method to separate the scale-dependent effects of species abundance distribution, density, and aggregation on diversity change. *Methods in Ecology and Evolution, 10*(2), 258-269. https://doi:10.1111/2041-210x.13102

Meltofte, H., editor. (2013). Arctic Biodiversity Assessment: Status and trends in Arctic Biodiversity. Akureyri: Conservation of Arctic Flora and Fauna. 674 p.

Moi, D. A., García-Ríos, R., Hong, Z., Daquila, B. V., & Mormul, R. P. (2020). Intermediate Disturbance Hypothesis in Ecology: A Literature Review. *Annales Zoologici Fennici, 57*(1-6), 67. https://doi.org/10.5735/086.057.0108

Montanarella, L. (2016). The Importance of Land Restoration for Achieving a Land Degradation–Neutral World. Land Restoration, 249–258. https://doi.org/10.1016/b978-0-12-801231-4.00020-3

Nguyen, T. D., Shih, M.-H., Srivastava, D., Tirthapura, S., & Xu, B. (2020). Stratified random sampling from streaming and stored data. Distributed and Parallel Databases, 39(3), 665–710. https://doi.org/10.1007/s10619-020-07315-w

Oksanen, J., Simpson, G., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., O'Hara, R., Solymos, P., Stevens, M., Szoecs, E., Wagner, H., Barbour, M., Bedward, M., Bolker, B., Borcard, D., Carvalho, G., Chirico, M., De Caceres, M., Durand, S., Evangelista, H., FitzJohn, R., Friendly, M., Furneaux, B., Hannigan, G., Hill, M., Lahti, L., McGlinn, D., Ouellette, M., Ribeiro, C. E., Smith, T., Stier, A., Ter Braak, C., & Weedon, J. (2024). vegan: Community Ecology Package. R package version 2.6-8. https://CRAN.R-project.org/package=vegan.

Olafsson, H., Furger, M., & Brümmer, B. (2007). The weather and climate of Iceland. *Meteorologische Zeitschrift, 16*(1), 5-8. http://doi:10.1127/0941-2948/2007/0185

Omidvar, E., Hajizadeh, Z., & Ghasemieh, H. (2019). Sediment yield, runoff and hydraulic characteristics in straw and rock fragment covers. *Soil and Tillage Research, 194*, 104324. https://doi.org/10.1016/j.still.2019.104324

Ottosson, J. G., Sveinsdottir, A., & Hardardottir, M., editor. (2016). Vistgerðir á Íslandi. Fjölrit Náttúrufræðistofnunar, 54, 1-299.

Oskarsson, H., Arnalds, O., Gudmundsson, J., & Gudbergsson, G. (2004). Organic carbon in Icelandic Andosols: geographical variation and impact of erosion. *CATENA*, *56*(1-3), 225–238. https://doi.org/10.1016/j.catena.2003.10.013

Pearson, K. (1895). Notes on regression and inheritance in the case of two parents. *Proceedings of the Royal Society of London, 58*, 240–242. https://doi.org/10.1098/rspl.1895.0041

Peri, P. L., Lasagno, R. G., Chartier, M., Roig, F., Rosas, Y. M., Pastur, G. M. (2021). Soil Erosion Rates and Nutrient Loss in Rangelands of Southern Patagonia. *Earth Systems and Environmental Sciences*. doi.org/10.1016/B978-0-12-821139-7.00183-5

Petursdottir, T. (2011). Bændur græða landið – samstarf um uppgræðslu og endurheimt vistkerfa [Farmers Heal the Land – a cost share program on revegetation and restoration of ecosystems]. Pages 84-85 in Á. L. Aradottir and G. Halldorsson, editors. Vistheimt á Íslandi [Restoration in Iceland]. Landbúnaðarháskóli Íslands og Landgræðsla ríkisins, Reykjavík, Iceland.

Petursdottir, T., Aradottir, A. L., & Benediktsson, K. (2013). An evaluation on the short-term progress of restoration combining ecological assessment and public perception. *Restoration Ecology, 21*, 75–85. https://doi.org/10.1111/j.1526-100X.2011.00855.x

R Core Team. (2024). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Ritchey, E., McGrath, J., & Gehring, D. (2015). Determining Soil Texture by Feel. *Agriculture and Natural Resources Publications, 139*. https://uknowledge.uky.edu/anr_reports/139

Rodrigo-Comino, J., García-Díaz, A., Brevik, E. C., Keestra, S. D., Pereira, P., Novara, A., Jordán, A., & Cerdà, A. (2017). Role of rock fragment cover on runoff generation and sediment yield in tilled vineyards. *European Journal of Soil Science, 68*(6), 864–872. https://doi.org/10.1111/ejss.12483

Snorrason, A., Traustason, B., Kjartansson, B. Þ., Heidarsson, L., Isleifsson, R., & Eggertsson, Ó. (2016). The natural birch woodland in Iceland—a new assessment of distribution and state. *Icelandic Agricultural Sciences, 86*, 97–111.

Sharma, A., Patel, S. K., & Singh, G. S. (2023). Variation in Species Composition, Structural Diversity, and Regeneration Along Disturbances in Tropical Dry Forest of Northern India. *Journal of Asia-Pacific Biodiversity, 16*(1), 83–95. https://doi.org/10.1016/j.japb.2022.11.004

Skidmore, E. L. (1994). Wind erosion. In: Lal R. (ed.): Soil erosion, research methods. 2nd ed. *Soil and water conservation society*. St. Lucie Press. 265–293.

Stefansson, S. (1901). Flóra Íslands. Copenhagen: Hið íslenzka bókmenntafélag.

Stockmann, U., Padarian, J., McBratney, A., Minasny, B., de Brogniez, D., Montanarella, L., Hong, S. Y., Rawlins, B. G., & Field, D. J. (2015). Global soil organic carbon assessment. *Global Food Security, 6*, 9–16. https://doi.org/10.1016/j.gfs.2015.07.001

Thacker, E., Messmer, T., & Burritt, B. (2015). Sage-Grouse Habitat Monitoring: Daubenmire versus Line-Point Intercept. *Rangelands, 37*(1), 7–13. https://doi:10.1016/j.rala.2014.12.002

Thien, S. J. (1979). A flow diagram for teaching texture-by-feel analysis. *Journal of Agronomic Education, 8*(1), 54–55. https://doi.org/10.2134/jae.1979.0054

Toy, T. J., Foster, G. R., & Renard, K. G. (2002). Soil erosion: processes, prediction, measurement, and control. John Wiley & Sons, New York, 41-42.

Tsujimoto, M., Kajikawa, Y., Tomita, J., & Matsumoto, Y. (2018). A review of the ecosystem concept — Towards coherent ecosystem design. *Technological Forecasting and Social Change, 136*, 49–58. https://doi.org/10.1016/j.techfore.2017.06.032

Tyler, T., Herbertsson, L., Olofsson, J., & Olsson, P. A. (2021). Ecological indicator and traits values for Swedish vascular plants. *Ecological Indicators, 120*(4), 106923. https://doi.org/10.1016/j.ecolind.2020.106923

Tyler, T., Herbertsson, L., Olsson, P. A., Fröberg, L., Olsson, K.-A., Svensson, Å., & Olsson, O. (2018). Climate warming and land-use changes drive broad-scale floristic changes in Southern Sweden. *Global Change Biology, 24*(6), 2607–2621. https://doi.org/10.1111/gcb.14031

Venables, W. N. & Ripley, B. D. (2002). Modern Applied Statistics with S. Fourth Edition. Springer, New York. ISBN 0-387-95457-0

Velde, B., & Meunier, A. (2008). The Origin of Clay Minerals in Soils and Weathered Rocks. Springer. https://doi.org/10.1007/978-3-540-75634-7

Vetter, V. M. S., Tjaden, N. B., Jaeschke, A., Buhk, C., Wahl, V., Wasowicz, P., & Jentsch, A. (2018). Invasion of a Legume Ecosystem Engineer in a Cold Biome Alters Plant Biodiversity. *Frontiers in plant science, 9,* 715. https://doi.org/10.3389/fpls.2018.00715

Walker, D. A., Raynolds, M. K., Daniëls, F. J. A., Einarsson, E., Elvebakk, A., Gould, W. A., Katenin, A. E., Kholod, S. S., Markon, C. J., Melnikov, E. S., Moskalenko, N. G., Talbot, S. S., Yurtsev, B. A. (†), & The other members of the CAVM Team. (2005). The Circumpolar Arctic vegetation map. *Journal of Vegetation Science, 16*(3), 267–282. https://doi.org/10.1111/j.1654-1103.2005.tb02365.x

Walker, D. A., Epstein, H. E., & Welker, J. M. (2008). Introduction to special section on Biocomplexity of Arctic Tundra Ecosystems. *Journal of Geophysical Research, 113*(G3). https://doi.org/10.1029/2008jg000740

Wasowicz, P., Przedpelska-Wasowicz, E. M., & Kristinsson, H. (2013). Alien vascular plants in Iceland: Diversity, spatial patterns, temporal trends, and the impact of climate change. *Flora - Morphology, Distribution, Functional Ecology of Plants, 208*(10-12), 648-673. https://doi.org/10.1016/j.flora.2013.09.009

Wasowicz, P., Pasierbiński, A., Przedpelska-Wasowicz, E. M., Kristinsson, H. (2014). Distribution Patterns in the Native Vascular Flora of Iceland. *PLoS ONE 9*(7): e102916. https://doi.org/10.1371/journal.pone.0102916

Wasowicz, P., & Heiðmarsson, S. (2019). A Vascular Plant Red List for Iceland. *Acta Botanica Islandica, 16*, 31–48. https://doi.org/10.5281/zenodo.2875162

Wickham, H. (2016). ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

Wickham, H., François, R., Henry, L., Müller, K., & Vaughan, D. (2023). dplyr: A Grammar of Data Manipulation. R package version 1.1.4. https://CRAN.R-project.org/package=dplyr.

Wiegmann, S. M., & Waller, D. M. (2006). Fifty years of change in northern upland forest understories: Identity and traits of "winner" and "loser" plant species. *Biological Conservation, 129*(1), 109–123. https://doi.org/10.1016/j.biocon.2005.10.027

Wilkinson, D. M. (1999). The Disturbing History of Intermediate Disturbance. *Oikos, 84*(1), 145. https://doi.org/10.2307/3546874

Yadav, S.K., & Mishra, G.C. (2013). Biodiversity measurement determines stability of ecosystems. *International Journal of Environmental Science, 4*(3), 2231-1289. https://www.ripublication.com/ijesdmspl/ijesdmv4n3_15.pdf

Thorhallsdottir, Þ. E. (1991). The effect of fertilizer and sowing in protected areas in North Iceland. In: Thorsteinsson, I., (ed.): Uppræðsla á Auðkúluheiði og Eyvindarstaðaheiði 1981–89. *Agricultural Research Institute*. Reykjavík. 89–103.

Thorhallsdottir, Þ. E. & Kristinsson, H. (2019). Flora of Iceland - Flowering plants and ferns. Vaka-Helgafell.

Zhang, Y., Zhang, M., Niu, J., Li, H., Xiao, R., Zheng, H., & Bech, J. (2016). Rock fragments and soil hydrological processes: Significance and progress. *CATENA*, *147*, 153–166. https://doi.org/10.1016/j.catena.2016.07.012

Zonn, I.S., Kust, G. S., & Andreeva, O. V. (2017). Desertification paradigm: 40 years of development and global efforts. *Systematic Study of Arid Territories*, 7, 131-141. https://doi.org/10.1134/S2079096117030118

6. Appendix

Table A1 - Correlation tests between all numerical environmental factors measured.

Environmental factor	1.	2.	3.	4.	5.	6.	7.
1. Vegetation cover	-	- 0.88	- 0.59	- 0.93	0.86	0.95	- 0.48
2. Erosion level	- 0.88	-	0.56	0.81	- 0.80	- 0.86	0.45
3. Latitude	- 0.59	0.56	-	0.56	- 0.62	- 0.55	0.27
4. Erosion %	- 0.93	0.81	0.56	-	- 0.84	- 0.96	0.27
5. Canopy cover %	0.86	- 0.80	- 0.62	- 0.84	-	0.85	- 0.45
6. Basal cover %	0.95	- 0.86	- 0.55	- 0.96	0.85	-	- 0.49
7. Stone base cover %	- 0.48	0.45	0.27	0.27	- 0.45	- 0.49	-

Table A2 - The species list used with all the main species identified in Icelandic habitats. Additional species can also be found and are then added to the list.

Mosses & Lichens

Sphagnum spp.
Racomitrium spp.
Cladonia arbuscula
Stereocaulon spp.
Peltigera spp.
Cetraria spp.
Cryptogamic crust

Grasses

Agrostis capillaris
Agrostis stolonifera
Agrostis vinealis
Anthoxanthum odoratum
Calamagrostis neglecta
Deschampsia alpina
Deschampsia caespitosa
Deschampsia flexuosa
Hierochloë odorata
Leymus arenarius
Phelum alpinum
Poa alpina
Poa annua
Poa glauca
Poa pratensis
Trisetum spp.
Festuca richardsonii
Festuca vivipara

Rush

Juncus arcticus
Juncus biglumis
Juncus trifidus
Juncus triglumis
Luzula arcuata
Luzula multiflora
Luzula spicata

Sedge

Carex bigelowii
Carex capillaris
Carex capitata
Carex maritima
Carex nigra
Carex rariflora
Carex rupestris
Carex vaginata
Eriphorum angustifolium
Eriphorum scheuchzeri
Kobresia myosuroides

Ferns

Botrychium lunaria

Horsetails

Equisetum arvense
Equisetum hyemale
Equisetum palustre
Equisetum pratense
Equisetum variegatum

Clubmoss, Spikemoss

Huperzia selago
Selaginella selaginoides

Trees & Shrubs

Betula nana
Betula pubescens
Juniperus communis
Salix arctica
Salix lanata
Salix phylicifolia

Shrublets & Heathers

Empetrum nigrum
Arctostaphylos uva-ursi
Calluna vulgaris
Loiseleuria procumbens
Vaccinium uliginosum
Vaccinium myrtillus
Dryas octopetala
Thymus praecox
Salix herbacea

Flowering plants

Geranium sylvaticum
Campanula rotundifolia
Pinguicula vulgaris
Coeloglossum viride
Dactylorhiza maculata
Platanthera hyperborea
Lupinus nootkatensis
Chamerion latifolium
Epilobium spp.
Viola palustris
Bartsia alpina
Euphrasis frigida
Rhinanthus minor
Plantago maritima
Armeria maritima
Sedum villosum
Arenaria norvegica
Cerastium alpinum
Cerastium cerastoides
Cerastium fontanum
Lychnis alpina
Minuartia spp.
Sagina spp.
Silene acaulis
Silene uniflora
Cardamine nymanii
Cardaminopsis petraea
Draba incana
Draba norvegica
Achillea millefolium
Alchemilla alpina
Alchemilla filicaulis
Erigeron borealis
Hieracium spp.
Leontodon autumnalis
Gentiana nivalis
Parnassia palustris
Galium boreale
Galium normanii
Galium verum
Comarum palustre
Geum rivale
Potentilla crantzii
Ranunculus acris
Thalictrum alpinum
Saxifraga cespitosa
Saxifraga hirculus
Saxifraga hypnoides
Saxifraga oppositifolia
Saxifraga stellaris
Bistorta vivipara
Koenigia islandica
Oxyria digyna
Rumex acetosa
Rumex acetosella
Tofieldia pusilla
Pyrola minor

Other species found

Table A3 – Dunn's multiple-comparison test results for a Kruskal-Wallis analysis of erosion % between the different habitat types. Asterisks indicates significance level for pairwise comparisons after adjusting the p-values according to the Bonferrioni method. p < 0.0005 (***); p < 0.005 (**); p < 0.05 (*); not significant (ns).

Habitat	L1	L2	L3	L4	L5	L6	L8	L9	L10
L1	-								
L2	ns	-							
L3	ns	ns	-						
L4	ns	ns	ns	-					
L5	***	ns	ns	**	-				
L6	***	ns	ns	ns	**	-			
L8	***	ns	ns	***	*	***	-		
L9	***	ns	ns	***	ns	***	ns	-	
L10	***	ns	ns	**	ns	**	***	ns	-

* *L1 = Fell fields, moraines & sands; L2 = Exposed aeolian soils; L3 = Screes & cliffs; L4 = River plains; L5 = Moss lands; L6 = Lava fields; L8 =Wetlands; L9 = Grasslands; L10 = Heathlands.*

Table A4 – Dunn's multiple-comparison test results for a Kruskal-Wallis analysis of erosion level between the different habitat types. Asterisks indicates significance level for pairwise comparisons after adjusting the p-values according to the Bonferrioni method. p < 0.0005 (***); p < 0.005 (**); p < 0.05 (*); not significant (ns).

Habitat	L1	L2	L3	L4	L5	L6	L8	L9	L10
L1	-								
L2	ns	-							
L3	ns	ns	-						
L4	ns	ns	ns	-					
L5	***	ns	ns	**	-				
L6	ns	ns	ns	ns	**	-			
L8	***	*	**	***	***	***	-		
L9	***	ns	**	***	ns	***	ns	-	
L10	***	ns	ns	**	ns	***	***	*	-

* *L1 = Fell fields, moraines & sands; L2 = Exposed aeolian soils; L3 = Screes & cliffs; L4 = River plains; L5 = Moss lands; L6 = Lava fields; L8 =Wetlands; L9 = Grasslands; L10 = Heathlands.*

Table A5 – Dunn's multiple-comparison test results for a Kruskal-Wallis analysis of erosion % between the different soil types. Asterisks indicates significance level for pairwise comparisons after adjusting the p-values according to the Bonferrioni method. $p < 0.0005$ (***); $p < 0.005$ (**); $p < 0.05$ (*); not significant (ns).

Soil type	1.	2.	3.	4.	5.	6.	7.	8.	9.	10.	11.
1.	-										
2.	**	-									
3.	***	***	-								
4.	***	ns	***	-							
5.	***	***	ns	***	-						
6.	***	***	ns	**	ns	-					
7.	***	**	ns	ns	ns	ns	-				
8.	***	***	ns	*	ns	ns	ns	-			
9.	*	ns	ns	ns	ns	ns	ns	ns	-		
10.	***	*	ns	ns	ns	ns	ns	ns	ns	-	
11.	***	**	ns	ns	ns	ns	ns	ns	ns	ns	-

** 1 = Sand; 2 = Loamy Sand; 3 = Loam; 4 = Sandy Loam; 5 = Silt Loam; 6 = Clay Loam; 7 = Sandy Clay Loam; 8 = Silty Clay Loam; 9 = Clay; 10 = Sandy Clay; 11 = Silty Clay.*

Table A6 – Dunn's multiple-comparison test results for a Kruskal-Wallis analysis of erosion level between the different soil types. Asterisks indicates significance level for pairwise comparisons after adjusting the p-values according to the Bonferrioni method. $p < 0.0005$ (***); $p < 0.005$ (**); $p < 0.05$ (*); not significant (ns).

Soil type	1.	2.	3.	4.	5.	6.	7.	8.	9.	10.	11.
1.	-										
2.	ns	-									
3.	***	***	-								
4.	***	ns	*	-							
5.	***	***	ns	***	-						
6.	***	***	ns	ns	ns	-					
7.	***	**	ns	ns	ns	ns	-				
8.	***	***	ns	*	ns	ns	ns	-			
9.	ns	ns	ns	ns	ns	ns	ns	ns	-		
10.	***	**	ns	ns	ns	ns	ns	ns	ns	-	
11.	***	*	ns	ns	ns	ns	ns	ns	ns	ns	-

** 1 = Sand; 2 = Loamy Sand; 3 = Loam; 4 = Sandy Loam; 5 = Silt Loam; 6 = Clay Loam; 7 = Sandy Clay Loam; 8 = Silty Clay Loam; 9 = Clay; 10 = Sandy Clay; 11 = Silty Clay.*