UNITED NATIONS UNIVERSITY

UNU-LRT

Land Restoration Training Programme
Keldnaholt, 112 Reykjavik, Iceland

Final project 2013

MONITORING RESTORATION PROGRESS AROUND HELLISHEIÐI, A LOW ALPINE ENVIRONMENT IN SOUTH-WEST ICELAND

Wellencia Clara Mukaru
University of Namibia
340 Mandume Ndemufayo Ave, Pionierspark, Windhoek, Namibia
cmukaru@unam.na

Supervisor
Kristín Svavarsdóttir
Soil Conservation Service of Iceland
kristins@land.is

ABSTRACT

Monitoring is essential for evaluating achievement of restoration goals and should be an integral part of any restoration project. The aim of the current study was to monitor the progress of restoration activities at two sites previously mined for gravel in south-west Iceland. Two types of inputs had been applied for restoration purposes: moss branches were spread at a crater and turfs transplanted onto a vehicle track. For the moss input, eight transects (four treated and four controls) were sampled. Quadrates of 50 x 50 cm in size were randomly placed along each transect and the frequency of mosses was determined for each quadrate. Moss and vascular plant cover was visually estimated in each quadrate. For the turf input, photographs were compared for visual changes that may have occurred since the transplanting of turfs. The percentage frequencies of all moss species/groups, apart from “other mosses” were significantly higher at the treated than the untreated site, while only the *Racomitrium* species were higher in cover. *Racomitrium lanuginosum* was the targeted species for the moss spreading and therefore expected to be more abundant in the treated area. There was no significant difference in vascular plant cover and species richness between the two sites. This was attributed to three quadrates in the control that had unusually high vascular plant cover and species richness. All mosses apart from *R. lanuginosum* increased in frequency between 2009 and 2013. *R. eriocoides*/*R. canascens* cover increased significantly over that period. Thus, moss abundance had increase since the spreading of moss branches five years earlier. A comparison of photographs taken prior and after turfs had been transplanted, suggested that vegetation cover on the track had increased. This

study highlights the need of monitoring procedures at all stages of ecological restoration activities.

This paper should be cited as:

Mukaru WC (2013) Monitoring restoration progress around Hellisheiði, a low alpine environment in South-West Iceland. United Nations University Land Restoration Training Programme [final project] http://www.unulrt.is/static/fellows/document/mukaru2013.pdf

TABLE OF CONTENTS

1. INTRODUCTION....1
2. LITERATURE REVIEW....3
2.1 Ecological restoration and succession....3
2.2 Plant-plant interactions....4
2.3 Monitoring of restoration progress....5
3. METHODOLOGY....6
3.1 Study site description....6
3.2 Restoration....7
3.2.1 Moss spreading....7
3.2.2 Turf transplanting....8
3.3 Data collection....8
3.3.1 Moss treatment....8
3.3.2 Vegetation turf treatment....9
3.4 Data analysis....9
4. RESULTS....10
4.1 Moss spreading....10
4.1.1 Impact of moss spreading....10
4.1.2 Changes in moss cover and frequency over time (2009-2013)....12
4.2 Vegetation turfs....13
5. DISCUSSION....15
5.1 Moss spreading....15
5.2 Vegetation turfs....17
5.3 Monitoring of restoration progress....17
6. CONCLUSIONS....18
ACKNOWLEDGEMENTS....20
REFERENCES....21
APPENDICES....26

1. INTRODUCTION

Disturbances, both natural and anthropogenic, have in some cases resulted in severe degradation of ecosystems. Thus, ecosystem services such as clean air and water, and fertile soils have been deteriorated or lost as degraded ecosystems have a reduced ability to provide such services (Constanza et al. 1997; Society for Ecological Restoration International Science and Policy Working group 2004; Greipsson 2011; Galatowitsch 2012). Ecosystems that have been severely degraded also lose their capacity to repair themselves and are therefore unable to prevent further degradation (Whisenant 1999). Disturbances like volcanic eruptions, erosion, extensive drought and mining may drive an ecosystem to the point where intervention is required through ecological restoration (Whisenant 1999; Walker & del Moral 2003). Ecological restoration is an intentional process of assisting the recovery of degraded ecosystems (Society for Ecological Restoration International Science and Policy Working group 2004). This may include initiating or accelerating the recovery of ecosystem attributes such as species composition, community structure, ecological function, suitability to support life, and connectivity with the surrounding landscape (Society for Ecological Restoration International Science and Policy Working Group 2004; Clewell & Aronson 2007).

Ecological restoration can be conducted in a number of ways, including seeding of desired vascular plant species (Zamfir 2000; Davy 2002; Krautzer & Wittmann 2006; Poulin et al. 2012), spreading of surface vegetation (Poulin et al. 2013; Rochefort et al. 2013) and the translocation of whole plant communities through turf transplants (Bullock 1998; Conlin & Ebersole 2001; Davy 2002; Krautzer & Wittmann 2006; Klimeš et al. 2010; Aradottir 2012). In order to increase its success, a restoration project must have clear and realistic goals that describe measurable targets to be reached by a specific time (Kondolf 1996; Whisenant 1999; Hobbs & Harris 2001; del Moral et al. 2007). Such goals are useful when putting the project plan in place. Proper planning is therefore a key for a successful restoration project, as it provides the procedures as well as making provisions for maintenance and management to reach project goals (del Moral et al. 2007).

Most countries around the world have degraded ecosystems that need ecological restoration regardless of whether they are developing or developed, or their climatic conditions. Iceland, the country where the study was conducted, and my home country Namibia, have very different climates, the former being generally colder and the latter warmer, but yet they both have land degradation issues that necessitate the need for ecological restoration. Ecological restoration as a practice has a long history in Iceland, which dates back to the beginning of the last century (Crofts 2011). The combined effects of human interventions, mostly through livestock grazing and deforestation (Arnalds & Barkarson 2003), and harsh environmental conditions have led to severe land degradation that has resulted in soil erosion being the biggest environmental problem in Iceland (Arnalds et al. 2001b; Arnalds & Kimble 2001; Crofts 2011). Icelandic soils are derived from volcanic material, and thus they are very light and easily transported once exposed, and as a result the whole soil profile is often lost through erosion (Arnalds et al. 2001b; Arnalds 2008). Advancing sand then impacts new areas through abrasion and burying vegetation (Arnalds et al. 2001a, 2001b; Arnalds 2008; Croft 2011). Other forms of disturbance that necessitate ecological restoration in Iceland include volcanic lava flows (Arnalds & Kimble 2001; Arnalds et al. 2001a) and gravel mining, mainly for the construction of roads (Haney 2010).

Mining is a critical sector of the Namibian economy and mineral assets form a major source of national wealth (Lange 2003). According to the Chamber of Mines of Namibia (2010), there is a long history of mining in Namibia, which dates back many hundreds of years. Many mines closed in the late 20th century, and this has led to over 200 abandoned mines (Chamber of Mines of Namibia 2010). These mines have sometimes left substantial environmental damage behind. This has had a negative impact on Namibia's natural resources and resulted in losses of valuable ecosystem services on which the Namibian people depend for their health and safety (Chamber of Mines of Namibia 2010). As there were no proper regulations for mine closures in place at the time, the Namibian government was left with the responsibility of restoring these degraded areas (Chamber of Mines of Namibia 2010). Other types of land degradation that may drive an ecosystem to lose its resilience and therefore raise a need for ecological restoration in Namibia include overgrazing, bush encroachment, and desertification (de Klerk 2004).

As mentioned above, each restoration project needs to have a detailed project plan which specifies every aspect of the project from the goals, to the budget and its implementation and subsequent management (Herrick et al. 2006). A carefully planned project saves resources and increases the chance of success (Brainbridge 2007). An important aspect of a restoration project, which should be addressed in the project plan is monitoring, that will amongst other things show if restoration goals are being achieved (Kondolf 1996; Woodward & Hollar 2011). It can be used to identify problems that may come up while they are still manageable (Elzinga et al. 1998). Monitoring results can also provide an opportunity to re-evaluate project objectives if they reveal that the initially set objectives were not realistic or conditions have changed. Monitoring should be conducted at different stages of the projects, and for different reasons. Usually baseline information is collected at the beginning of the project (Elzinga et al. 1998; Lewis et al. 2009). Information can further be collected mid-project and post-project to assess progress and to detect whether the desired changes have been achieved, respectively (Elzinga et al. 1998; Lewis et al. 2009).

This study aimed at monitoring the progress of restoration activities in an area in which two different types of restoration inputs were applied in 2007 and 2008. The inputs were: spreading of moss together with sowing of grass species and transplanting of turf.

The study sought to answer the following research questions:

a) Has the spreading of moss contributed to an increase in moss frequency and cover at the treated site?
b) Is there a difference in vascular plant cover between the treated and control sites?
c) Is there a difference in plant species richness between the treated and control sites?
d) Is there a difference in plant species composition between the treated and control sites?
e) Have moss frequency and cover in the treated area changed since 2009?
f) Is there any visual change in vegetation cover at a site treated with turf transplants five years ago?

2. LITERATURE REVIEW

2.1 Ecological restoration and succession

Functional ecosystems have the ability to repair themselves after a disturbance, but sometimes severe disturbances may drive an ecosystem to a point where it loses its resilience, making it vulnerable to change in the face of a disturbance (Whisenant 1999; van Andel & Grootjans 2006). Under normal circumstances, species will return to an area after a disturbance, following a somewhat predictable sequence of ecological changes called succession (Walker et al. 2007; Galatowitsch 2012). Ecological restoration usually addresses shorter time scales than successional studies, but is dependent on successional processes and patterns for its success (Palmer et al. 1997; Walker et al. 2007). Species used for restoration should be selected with a final state in mind, and this choice of species relies mainly on the understanding of successional patterns as it offers insight into the roles of species dispersal, species interactions and interactions between biotic and abiotic factors (Palmer et al. 1997; Choi 2004; van Andel & Grootjans 2006; Walker & del Moral 2008; Greipsson 2011). The aim of restoration most often is therefore to influence direction and accelerate the rate of succession (Palmer et al. 1997; Greipsson 2011).

Ecological restoration mostly involves a single event that initiates or accelerates succession. This may include excluding grazing, seeding, or planting of nurse species (Greipsson 2011). Nurse species have been used for a long time to assist the establishment and growth of other species (Davy 2002; Galatowitsch 2012). Such plants may be specifically selected early successional species due to their ability to improve the physical environment of a degraded site, thus enabling the subsequent establishment of the target community (Davy 2002). When restoration activity introduces early successional species in an area, connectivity to surrounding areas is very important to provide a source of propagules to allow for changes in species composition towards a desired community (Palmer et al. 1997; Walker & del Moral 2003; Walker et al. 2007).

Assuring that the area under restoration is connected to the surrounding ecosystems is however only one of the hurdles in restoration ecology because only those species with the best dispersal characteristics will reach the area (Walker & del Moral 2003, 2008). While the dispersal characteristics and isolation of species will determine the number of species that will arrive in an area, eventually site conditions will determine which of those species can establish (Palmer et al. 1997; Walker and del Moral 2003, 2008). This is because not all dispersed seeds fall on suitable sites for germination (Whisenant 1999; Walker & del Moral 2003). Seeds may fall on a site on which they are exposed to harsh environmental conditions, limited resources, seed predation or competition from established plants (Whisenant 1999; Walker & del Moral 2003).

Only very few of the propagules that reach an area do germinate and grow to maturity (Picket at al. 1987; Jumpponen et al. 1999). Therefore the distribution and abundance of sites that satisfy the regeneration requirements for desired species (safe sites) is important for both germination and seedling establishment (Oesterheld & Sala 1990; Palmer et al. 1997; Jumpponen et al. 1999). Furthermore, abiotic conditions play a crucial role in establishment of species in a primary succession, as most early successional surfaces are stressful (Walker & del Moral 2003), making the choice of pioneer species in ecological restoration very important. The presence of plant species in an area after a disturbance is therefore dependent on survival, successful migration, and establishment of its propagules (Whisenant 1999).

2.2 Plant-plant interactions

In ecological restoration early-successional plants may be used to modify the microclimate or soil in a way that allows later successional plants to enter the community (Young et al 2001), a process called facilitation (Picket at al. 1987; Whisenant 1999; del Moral et al. 2007; Greipsson 2011). The establishment and modification of a habitat by the facilitator is therefore essential for the establishment of later species (Walker & del Moral 2003). Another form of plant-plant interaction is inhibition which, as the name suggests, is the negative effect of one species on another (Whisenant 1999; Walker and del Moral 2003; del Moral et al. 2007). The inhibitor may slow or arrest successional change by preventing establishment of species (Walker and del Moral 2003; Greipsson 2011). It is important for restoration practitioners to recognize that both these types of interactions among plants occur simultaneously in many ecosystems and therefore it might not always be advisable to consider them in isolation (Picket at al. 1987; Whisenant 1999; Palmer et al. 1997).

Mosses have been shown to both facilitate and inhibit the establishment of vascular plants, depending on the moss species (Zamfir 2000; Sedia & Ehrenfeld 2003; Kirkpatrick et al. 2006). The mechanisms behind the moss effects include light reduction, moderation of temperature and soil moisture and allelopathic effects (Kirkpatrick et al. 2006), depending on the type of environment and vascular plant species (Zamfir 2000).

Rayburn et al. (2012) conducted a study on how moss may affect the distribution and performance of a primrose species (*Primula cusickiana* var. maguirei), which is endemic to canyons in northern Utah, USA. They found that moss may facilitate *P. cusickiana* var. maguirei by trapping primrose seeds and through the provision of increased soil resources (Rayburn et al. 2012). Although Rayburn et al. (2012) revealed a facilitative effect of moss on the primrose species, they also pointed out that there might be alternative explanations and that both moss and primrose might be responding to other factors than those measured in their study, such as soil depth or microtopography, which might further explain their results.

An experimental study revealed a facilitative effect of perennial grass (*Ichaemun aristatum* var. glacum) tussocks on the regeneration of native vascular plants in Ukishima Marsh, eastern Japan (Wang et al. 2012). The identified mechanisms for facilitation were provision of safe sites for seed germination and seedling establishment against high inundation and water disturbance (Wang et al. 2012). They further found that the occurrence of moss in the tussocks showed significantly positive effects on the seedling survival of some species, including endangered species (Wang et al. 2012). In contrast, the presence of moss in a greenhouse experiment at the University of Puget Sound reduced the establishment of two species of Monkey flowers (*Mimulus lewisii* and *M. guttatus*) from seed compared to treatments without moss (Kirkpatrick et al. 2006). Seedling establishment of the two species, however, showed conflicting results; the presence of moss seemed to facilitate the establishment of *M. guttatus* seedlings but inhibit that of *M. lewisii* (Kirkpatrick et al. 2006). Kirkpatrick et al. (2006) attributed this difference in seedling establishment in the presence of moss to differences in their early growth characteristics, to how they germinate. This was further supported by the curved and twisted stems of *M. guttatus* seedlings that managed to establish in the presence of moss, indicating that they were able to take advantage of small gaps in the moss canopy to reach the surface which *M. lewisii* was unable to do (Kirkpatrick et al. 2006). Similar results were obtained in another greenhouse experiment Zamfir

(2000), studying the effects of bryophytes and lichens on seedling emergence of four alvar (calcareous) grassland species; the studied plants responded differently to the presence of moss and lichen. An interesting observation from that study is that seed size may have affected the emergence of seedlings from greater depths, with larger seeds being less affected than smaller ones (Zamfir 2000).

Conlin and Ebersole (2001) showed a high overall success of turf transplants in a study in Colorado, USA. They concluded that despite lower overall plant cover in the transplant plots, the use of turf transplants was a successful method of rehabilitating severely eroded trails in the Colorado mountains as different growth forms had a high survival rate and species richness was relatively high (Conlin & Ebersole 2001). Both Aradottir (2012), and Krautzer and Wittmann (2006) reported positive effects of turfs transplants on the colonization and establishment of native vegetation at disturbed sites, but cautioned that the method might not always yield the desired results. Turf expansion and establishment, for example, varies with the conditions of the receptor site, and therefore it is advisable to select a donor community that does not differ greatly from the receptor site in abiotic conditions (Bullock 1998; Klimeš et al. 2010; Aradottir 2012).

A possible disadvantage of using turf transplants is a disturbance of the natural vegetation at the donor site for providing the turf. This can however be minimized by using turf from roadbeds, mine sites and other areas that are being stripped of vegetation (Krautzer & Wittmann 2006; Aradottir 2012). Turf size has also been shown to be important (Bullock 1998; Krautzer and Wittmann 2006; Klimeš et al. 2010; Aradottir 2012). Klimeš et al. (2010) suggested that size should be preferred over the number of transplanted turfs, while Krautzer and Wittmann (2006) recommends a turf size of 0.15-0.5 m^2.

Plant-plant facilitation has been demonstrated in studies with shrubs as nurse plants. Shrubs have been shown to have positive effects on forest tree sapling establishment and survival, mainly through protection from abiotic stress, such as solar radiation, frost and draught (Castro et al. 2004; Gomez-Aparicio et al. 2008). Armas and Pugnaire (2005) also found that a shrub species (*Cistus clusii*) in south-east Spain acted as a nurse species for a perennial grass (*Stipa tenacissima)* through improving its water and nutrient content, carbon dioxide assimilation rate, and growth.

2.3 Monitoring of restoration progress

Ecological restoration success can be viewed as an ongoing process from a successful establishment of initially introduced species, to an establishment of ecosystem attributes that ensure a self-sustaining and functioning system (Reay & Norton 1999). Thus the success of an ecological restoration project can be measured by periodically assessing whether restoration goals have been attained or are in the process of being achieved. According to Elzinga et al. (1998), monitoring is a tool for identifying problems early and before they become unmanageable. It involves collection and analysis of repeated observations and measurements to evaluate changes in conditions of the project site and progress towards meeting pre-defined objectives (Elzinga et al. 1998; NAVFAC [Naval Facilities Engineering Command] Risk Assessment Workgroup and Argonne National Lab 2004). Monitoring should ideally be carried out in permanent plots or at specifically marked points in order to reduce sampling variation and ensure that spatial variations are not confused with temporal change (Walker & del Moral 2003).

Monitoring of ecological restoration projects is divided into four phases; baseline assessment, implementation, effectiveness, and validation (Elzinga et al. 1998; Lewis et al. 2009). Baseline (pre-project) assessment involves documenting the initial state of the project site and implementation monitoring refers to the assessment of whether the restoration actions were conducted as planned (Elzinga et al. 1998; Lewis et al. 2009). Effectiveness monitoring is used to determine whether the activities or restoration inputs are having the desired habitat response, while validation monitoring is used for confirming that the activities in the project have brought about the desired biotic and abiotic responses in the restored ecosystem (Elzinga et al. 1998; Lewis et al. 2009).

The level of intensity for conducting effectiveness monitoring varies, ranging from a routine to an intensive evaluation, depending on the complexity of the restoration project (Machmer & Steeger 2002). Routine evaluation is usually used for projects with relatively straightforward objectives on relatively small or homogeneous areas over a limited period of time (Machmer & Steeger 2002). The method mainly relies on the use of qualitative data, involving rapid data collection at low cost to compare the condition of the project site before and after the completion of restoration work (Machmer & Steeger 2002; Lewis et al. 2009). When more in-depth quantitative monitoring is required to meet project objectives, intensive effectiveness monitoring is the recommended method (Machmer & Steeger 2002). This is usually conducted over a longer time, at a higher cost than routine evaluation (Machmer & Steeger 2002).

Quantitative measurements are not always feasible for most ecological restoration projects (Woodward & Hollar 2011), as they can be time consuming, very costly and often have a narrow focus (Machmer & Steeger 2002; Lewis et al. 2009). Qualitative data may be useful in identifying a broad spectrum of issues that might otherwise be undetectable with the more narrow focus of the quantitative method (Brainbridge 2007; Lewis et al. 2009). An example of qualitative monitoring is photopoints; a series of photographs taken over time starting before project implementation, in order to detect changes over time (Lewis et al. 2009). However, qualitative monitoring is subjective, as it involves observations, and the interpretation may vary from one observer to another (Lewis et al. 2009). Ultimately, the method used for monitoring restoration progress depends on the monitoring objectives (Hobbs & Harris 2001; NAVFAC Risk Assessment Workgroup and Argonne National Lab 2004; Driscoll et al. 2007; Lewis et al. 2009) and using both qualitative and quantitative methods may be advantageous where resources are sufficient, as the methods can complement one another (Machmer & Steeger 2002; Lewis et al. 2009).

3. METHODOLOGY

3.1 Study site description

The study was carried out at and near the crater Gígahnúkur in Hellisheiði, south-west Iceland (64°02 N, 21°22 W) at 400-440 meters above sea level (Aradottir & Grétarsdóttir 2011). The Gígahnúkur crater is the largest crater in the lava field in Hellisheiði (Sæmundsson, as cited in Aradottir & Grétarsdóttir 2011). The study site is located within the construction area of the Hellisheiði geothermal power plant (Fig. 1).

Fig. 1. *The location of the study/research area in Hellisheiði, south-west Iceland. (Source: Fanney Ósk Gisladottir, Agricultural University of Iceland, 2013).*

The mean daily temperature in the Hellisheiði area for the period 2001-2012 was 2.6°C, and the mean annual precipitation for the same period was 2315 mm. During the restoration project period (2007-2012), the average daily temperature was 2.7°C, and the mean annual precipitation for the same period was 2229 mm. The precipitation was unevenly distributed through the year, with the lowest readings recorded in early summer, and the highest in autumn and winter. It should be noted that the weather data obtained lacked precipitation readings for 19-28 January 2012. This may have affected the annual precipitation calculated for that year. There were no rainfall data available for 2013 at the time of submission, and the mean daily temperature for January was 0°C and 11.7°C for July. For the period 2007-2012, the mean daily temperature for January was -1.6°C and for July it was 10.1°C. January was used to give an indication of winter temperatures, while July was used for the summer. The weather data were obtained from the Icelandic Meteorological Office (unpublished data from Hellisskarð, the weather station nearest to the study site).

3.2 Restoration

3.2.1 Moss spreading

The Gígahnúkur crater had been mined for gravel and parts of it left severely disturbed due to the removal of substrate and vegetation (Aradottir & Fridriksdottir 2011). Restoration of the crater started in 2007, with the physical rebuilding of the hills based on old aerial photographs and a three-dimensional model (Aradottir & Fridriksdottir 2011). Lava slag was then spread on the top of the surface of the site under restoration, to make it resemble the surrounding areas and to stimulate vegetation establishment (Aradottir & Fridriksdottir 2011). In an attempt to speed up the establishment of the dominant moss species (*Racomitrium lanuginosum*), its branches were spread on top of the lava slag (Aradottir & Fridriksdottir 2011). The spread moss was collected approximately 5 km west of Gígahnúkur crater at a little less than 250 meters above sea level, and was distributed by hand from 25 August to 5 September in 2008 by a group from the British Trust for Conservation Volunteering (Aradottir & Fridriksdottir 2011).

Moss establishment was assessed in the autumn of 2009, about one year after it was spread. A total of four 30 m long transects were laid down on a south-facing hill of the crater, starting from the top of Gígahnúkur crater (Aradottir & Fridriksdottir 2011). On each transect, 10 randomly placed quadrates were sampled (Aradottir & Fridriksdottir 2011). No control site was sampled in 2009. An attempt was made in 2013 to establish four transects in an untreated area as a control. As most of the area with comparable surface had received some restoration input, either moss spread or turf transplant, a large enough area to place four 30 m transects was not found. Thus the control had to be located in two areas with similar disturbance regimes as the treated sites and the length of the transects varied. An area adjacent to three of the four treated transects contains a restoration experiment, from which five of the experimental control plots were used. In addition, a 21 m transect (the maximum available transect length at that site) was laid down and five quadrates randomly placed along it. Thus, 10 quadrates were sampled at that site and were regarded as one replicate. The other control area was an old vehicle track with similar substrate material located in the same area, east of the other transects. On the track a sample area ranging from N64 01 61.8 and W21 21 95.2 to N64 01 67.2 and W21 21 75.5 (Appendix 1) was used for the study. Three transects were placed within the sampling area. A total of 25 quadrates were sampled there; 10 along 25 and 30 m transects, and five plots along a 10 m transect. All transects were marked with flags on each end and their position recorded using GPS.

3.2.2 Turf transplanting

Near the Gígahnúkur crater a vehicle track was removed in 2007. The surface is lava slag and it is comparable to the Gígahnúkur crater. In an attempt to restore vegetation cover after the construction, small turfs were transplanted onto the track. No detailed recording of the methodology is available and baseline data were not collected in 2007. Photographs were however taken before and after transplanting the turfs onto the track (M. Magnúsdóttir, 16 July 2013, Reykjavík Power Company, personal communication). No monitoring had been done of the restoration treatment since the turf transplanting.

3.3 Data collection

3.3.1 Moss treatment

In 2013, the same 40 treated quadrates and procedures were used as in 2009 (Aradottir & Fridriksdottir 2011). For control, a total of 35 quadrates were sampled, as described earlier.

Quadrates (50 x 50 cm) were randomly placed along each transect and the frequency of moss species occurrence was determined by dividing the quadrate into 100 subquadrates of 5 x 5 cm and counting the number of subquadrates in which each species occurred (Fig. 2). Only green branches of moss were recorded and the data were collected under moist conditions, in order to maximize moss visibility. The frequency of *Racomitrium lanuginosum*, *R. ericoides*/*R. canascens*, *Polytrichum* sp./*Pogonatum* sp. and other mosses was recorded. Only moss frequencies were determined because there were few plants at the study site.

In 2013, moss cover was visually estimated in each quadrate, as had been done in 2009 (Aradottir & Fridriksdottir 2011). The cover was categorized according to the following classes: 1: < 1%, 2: 1-5%, 3: 6-10%, 4: 11-15%, 5: 16-25%, 6: 26-50%, 7: 51-75% and 8: 76-100% cover. In 2013, all vascular plant species were identified and their cover percentage visually estimated using the same cover classes as for mosses.

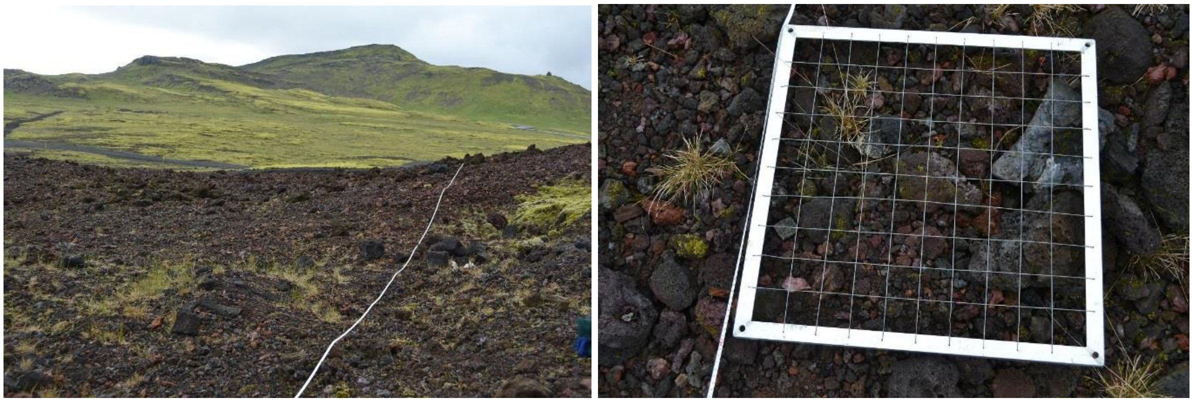

Fig. 2. *The photo on the left shows an example of a transect laid down at the Gígahnúkur crater. A quadrate as seen on the right was randomly placed along the transect for sampling frequency and cover of species. (Photos taken by K. Svavarsdóttir on 16 July 2013.)*

3.3.2 Vegetation turf treatment

As there was insufficient information on the initial procedure on the track revegetated with turf (i.e. the original turf sizes, their plant species composition, the design of transplanting, etc.), it was difficult to assess progress with measurements as no comparable data were available. As a result, photographs were compared to provide a visual demonstration of the changes that had occurred from the time the turfs were transplanted in 2007 to 2013. I would however like to point out that this is not a typical photopoint monitoring, as no monitoring was planned at the beginning of the project. Hence the photographs taken in 2007 and 2008 were not carefully located as photo points for future monitoring. The photos from different years may therefore not be from exactly the same points. An attempt was made to find comparable photos from different times to allow for a general overview of the restoration progress.

3.4 Data analysis

Cover classes were converted into percentage cover by using the median of each class. As there were few vascular plants at the study site, a sum of the cover of all vascular plant species was calculated to determine total cover of vascular plants for each quadrate. For data analysis transects were regarded as replicates.

Both the cover (including the data from 2009) and frequency data were tested for normality using the Kolmogorov-Smirnov test. The frequency data for *Racomitrium lanuginosum* and cover data for other mosses in 2009 were not normally distributed and therefore transformed using log-transformation. The t-test for independent variables was used to compare moss frequencies, cover of the mosses and vascular plant cover and species richness between the treated and untreated sites.

The mean cover data for *Racomitrium ericoides* in 2009 was reported as < 0.1% for all four transects (Aradottir & Fridriksdottir, 2011). In order to allow for a comparison between 2009 and 2013 for that species, < 0.1% was replaced with 0.09%. The highest number without adding too many decimal points was chosen to be conservative as the cover of *Racomitriium ericoides*/*R. canascens* had increased in 2013. A paired t-test was used to test for differences in moss frequency and cover between 2009 and 2013.

All data analyses other than ordination were done using the Minitab statistical software, version 14 (Minitab Inc.). Ordination, detrended correspondence analysis (DCA), was performed on a plots-species matrix consisting of 75 quadrates (plots) and 24 plant species, using presence and absence data. This was done to reveal relationships in species composition amongst the different quadrates. Canoco 5 was used for ordination analyses (ter Braak & Šmilauer 2012).

4. RESULTS

4.1 Moss spreading

4.1.1 Impact of moss spreading

The percentage frequency of *Racomitrium lanuginosum*, *Racomitrium eriocoides*/*R. canascens* and *Polytrichum* sp./*Pogonatum* sp. was significantly higher at the treated than the control (untreated) sites (Table 1). There was no significant difference in the occurrence of other mosses between the two sites ($p > 0.05$).

Table 1. *Mean percentage frequency (± standard error) of mosses for the untreated and treated sites near and at the Gígahnúkur crater in south-west Iceland. The superscript letters show the outcome of the comparison between mean frequencies of species (or group) with different letters indicating a significant difference.*

Moss species / groups	Untreated	Treated
Racomitrium lanuginosum	9.5 (± 4.52)^{a}	29.8 (± 2.57)^{b}
Racomitrium eriocoides/*R. canascens*	7.6 (± 4.10)^{a}	73.9 (± 5.66)^{b}
Polytrichum sp./*Pogonatum* sp.	1.6 (± 1.37)^{a}	6.6 (± 0.90)^{b}
Other mosses	0.7 (± 0.66)^{a}	2.6 (± 0.46)^{a}

Racomitrium lanuginosum and *R. eriocoides*/*R. canascens* were the most frequent mosses at the untreated sites, both had just under 10% frequency (Table 1). Both species greatly increased their occurrence in the treated area ($p < 0.01$ and $p < 0.001$, respectively), although *R. eriocoides*/*R. canescans* increased much more and was found in about three of every four quadrats on the restored hill. *Polytrichum* sp./*Pogonatum* sp. also had a significantly higher frequency in the treated area than the control ($p < 0.05$), but was much less frequent than the *Racomitrium* species at both sites.

Vegetation cover at both sites was generally low, with total vegetation covering less than 5% at the untreated site and about 13% at the treated site (Table 2). There was a significant difference in percentage cover of *Racomitrium lanuginosum* ($p < 0.01$) and *Racomitrium eriocoides*/*R. canascens* ($p < 0.001$) between the treated and untreated sites, both having higher cover at the treated site. *Polytrichum* sp./*Pogonatum* sp., other mosses and vascular plants did not differ significantly between the two sites (Table 2). Three quadrates (5, 6 & 7) along transect 2 of the untreated site had very high vascular plant cover and species richness compared to the other quadrates from the same site (Appendix 2). Removing the three quadrates from the dataset revealed a significantly higher vascular plant cover at the treated site ($p < 0.05$), while there was no significant difference in species richness ($p > 0.05$) between the two sites (Table 2 and 3).

Table 2. *Mean percentage cover (± standard error) for the untreated and treated sites near and at the Gígahnúkur crater in south-west Iceland. The superscript letters show the outcome of the comparison of mean cover of species (or group) with different letters indicating a significant difference.*

Species/groups	**Untreated**	**Treated**
Racomitrium lanuginosum	0.4 (± 0.05) [a]	3.9 (± 0.39) [b]
Racomitrium eriocoides/R. canascens	0.4 (± 0.05)[a]	4.2 (± 0.77) [b]
Polytrichum sp./*Pogonatum* sp.	0.3 (± 0.22) [a]	0.6 (± 0.10) [a]
Other mosses	0.2 (± 0.11) [a]	0.3 (± 0.05) [a]
Vascular plants	1.7 (± 1.46) [a]	3.5 (± 0.55) [a]
Vascular plants*	0.3 (± 0.10) [a]	3.5 (± 0.55) [b]
Total vegetation cover	**3%**	**12.5%**

*Vascular plant cover without the three quadrates from transect 2 at the untreated site.

Vascular plant species richness did not differ significantly between the untreated and treated sites ($p > 0.05$). Transect 2 at the untreated site had 10 species, which was the highest number of species per transect, and differed greatly from the other controls (Table 3). Five or seven species were recorded along transects at the treated site, with only one exception where three species were recorded (Table 3).

Table 3. *Species richness of vascular plants on each transect at the untreated and treated sites near and at the Gígahnúkur crater in south-west Iceland. The means and standard errors (SE) for each site are shown at the bottom.*

Transect	**Untreated**	**Treated**
1	4	3
2	10	7
2*	4	7
3	2	5
4	2	7
Mean	**4.5**	**5.5**
SE	**1.89**	**0.96**

*Species richness without the three outliers on transect 2 at the untreated site. When the three quadrates were excluded, the mean of the untreated site was 3, and the standard error 0.58.

Detrended correspondence analysis of the 75 plots and 24 species did not show a clear separation between the treated and untreated plots (Fig. 3). It is evident from the ordination diagram, however, that less variation was found amongst the treated plots than the control plots (Fig. 3). The untreated plots varied greatly in their species composition. The eigenvalue of DCA axis 1 was 0.44 and its gradient length 3.40. The second axis had 2.79 in gradient length and an eigenvalue of 0.34. Total variation in the data was 3.42, of which the two first axes combined explained 23.3%.

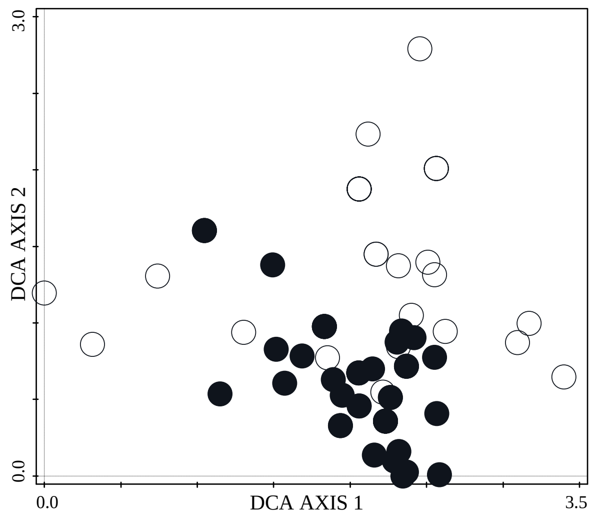

Fig. 3. *DCA ordination diagram showing species composition of quadrates in the study area near and at Gígahnúkur crater in south-west Iceland. The closer quadrates are located in the graph the more similar was their species composition. The empty circles represent untreated quadrates, while the filled circles represent treated ones.*

4.1.2 Changes in moss cover and frequency over time (2009-2013)

In 2009 *Racomitrium lanuginosum* was the most frequent moss species in the study area, and its mean frequency had not changed significantly in 2013 ($p > 0.05$) (Fig. 4). *R. eriocoides*/*R. canascens* had become the most wide spread species in 2013 (Fig. 4), with a mean percentage frequency significantly higher than in 2009 ($p < 0.01$). Although they were seldom encountered (Fig. 4), both *Polytrichum* sp./*Pogonatum* sp. and other mosses had significantly increased their occurrence since 2009 ($p < 0.01$).

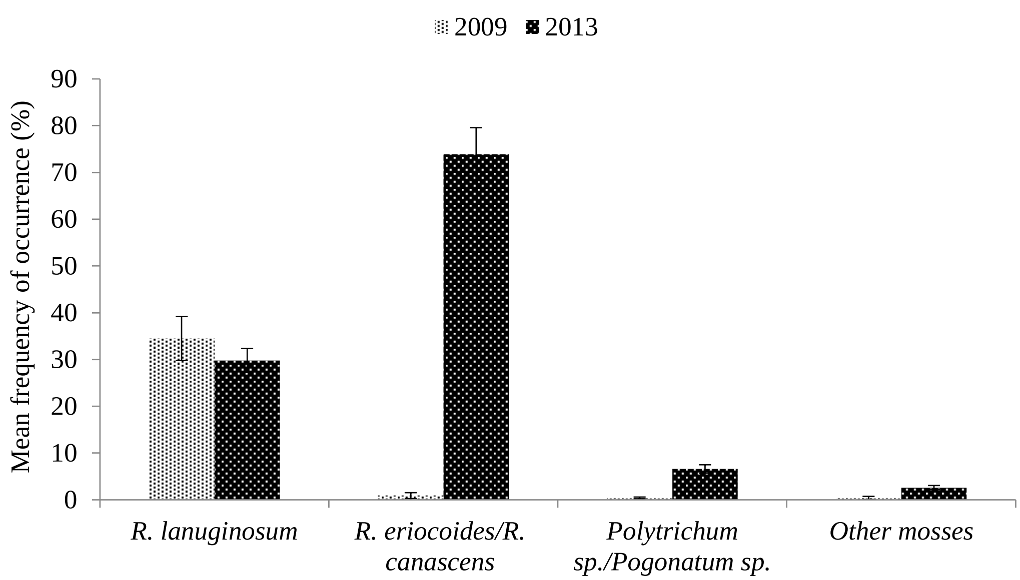

Fig. 4. *Mean percentage frequencies (±standard errors) of mosses at the treated site in the Gígahnúkur crater area in 2009 and 2013.*

The cover of *Racomitrium eriocoides*/*R. canascens* had increased significantly since 2009 ($p < 0.05$), from a mean cover of below 1% in 2009 to just over 4% in 2013 (Fig. 5). There was no significant difference in the cover of *Racomitrium lanuginosum* between 2009 and 2013 ($p > 0.05$).

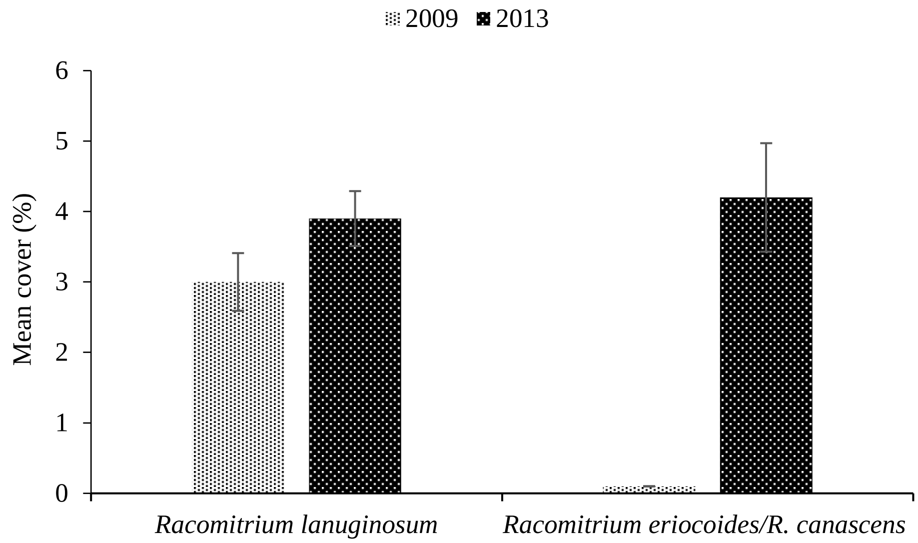

Fig. 5. *Mean percentage cover (±standard errors) of* Racomitrium lanuginosum *and* Racomitrium eriocoides/ R. canascens *in the Gígahnúkur crater area in 2009 and 2013.*

4.2 Vegetation turfs

There appeared to be more vegetation cover in 2013 than in 2007 after the turfs were transplanted onto the track in 2007 (Fig. 6). Figure 6A shows how the track looked before the turfs were transplanted and there clearly was no vegetation on the track at the time. The transplanted turf pieces can be seen in Figure 6B, while in Figure 6C a spread of vegetation from the transplants of turfs can be observed.

A

B

C

Fig. 6. *A sequence of photos of the vehicle track near the Gígahnúkur crater in south-west Iceland that had turfs transplanted in 2007. Photo A was taken before turfs were transplanted, photo B was taken a year after the turfs were transplanted, and photo C was taken six years after the transplants. Changes over time can be detected from the photos. (Photo A: Reykjavik Energy Company, 2007; Photo B: Kristín Svavarsdóttir, 1 September 2008; Photo C: Wellencia Clara Mukaru, 17 July 2013).*

In Figure 7, turf pieces that were transplanted in 2007 can be seen on the left and on the right the establishment of vegetation in the transplanted turf. Another observation that is evident is the establishment of vegetation between the originally transplanted turfs (Fig. 6 and 7).

Fig. 7. *Examples of close up transplanted turfs on a vehicle track near the Gígahnúkur crater in south-west Iceland. The turfs were transplanted in 2007 and were distinct turfs a year later (left) but five years later some vegetation establishment was observed between the transplanted turfs (right). (Photos: Kristín Svavarsdóttir, 1 September 2008; and Anna Sigríður Valdimarsdóttir, 17 July 2013).*

5. DISCUSSION

5.1 Moss spreading

Spreading of moss branches as an ecological restoration method assisted moss colonization at the Gígahnúkur crater site, as shown in significantly higher percentage frequencies of all moss species/groups recorded apart from "other mosses" at the treated site. Although mosses were generally more frequent in the treated area than at the untreated site, only the *Racomitrium* species had higher cover (Table 1 and 2). This was not surprising, especially for *Racomitrium lanuginosum*, because it was the targeted species for the moss spreading and is the most dominant moss species in the area (Aradottir & Fridriksdottir 2011). Other mosses may have been accidentally collected with the *Racomitrium lanuginosum* branches but they are though likely to have been limited in the spreading.

Five years after the initial input only very few vascular plants occurred in the treated area. Campeau and Blanchard (2010) conducted a study in which local moss fragments and vascular plant seeds were sown at the same time and their establishment monitored over time. They found that when moss cover was low (between 5-6%), vascular plant cover was below 1% but became higher with increasing moss cover (Campeau & Blanchard 2010). The low moss cover in the study area may explain at least partly the low vascular plant cover and that it was not significantly greater in the treated sites than at the untreated sites (Table 2). Mosses are thus at least not yet facilitating the establishment of vascular plants. The possible mechanisms of potential facilitation in this case may include seed trapping, moderation of temperature (Kirkpatrick et al. 2006), and soil stabilization, allowing the vascular plants to anchor themselves into the more stable soils (Brink 2005).

An alternative explanation for the lack of difference in vascular plant cover between treated and control sites was the presence of three quadrates along the control transect 2, as they had exceptionally high vascular plant cover compared to the other control quadrates (Appendix 2 and 4). This might have masked the difference in vascular plant cover between the two sites. This argument is further supported by a significant difference in vascular plant cover between treated and untreated sites being revealed after removing these three quadrates from the dataset (Table 2). The three quadrates were located in a depression with a small rock outcrop close to the adjacent vegetation on the edge of the track. This area might have served as a safe site for seedling establishment, through the trapping of seeds and water, and the outcrop further acted as a barrier protecting the site against wind and preventing seeds from being blown away (Whisenant 1999).

The same three quadrates also contributed to a high number of species along transect 2 at the untreated site, the highest number recorded on any of the studied transects (Table 3 and Appendix 2). Removing them from the data changed the transect species richness from 10 to 4, but there was still no significant difference in species richness between the two sites although there was a general trend of a higher number of species at the treated site (Table 3). The lack of a significant difference, even after removing the three outliers, may be explained by the high variation in the species richness data at the treated site reflected in its high standard error of the mean.

Although the DCA ordination did not reveal a clear separation between treated and untreated quadrates, it is evident that there was more similarity in species composition among the treated quadrates (Fig. 3). This indicates that the treated area has become more compositionally uniform due to the spreading of moss and sowing of *Festuca.* The results further indicate that species composition at the control site was more stochastic than in the treated plots, as was evident from the spread of untreated quadrates along DCA axis 1 and 2 (Fig. 3). It should however be noted that the multivariate analysis was mostly done for training purposes. The data were limited due to very few plant species at the study site (Table 3 and Appendix 2, 3, 5 and 6) and this restricts ordination analyses. Eight of the 24 species occurred only once, and because of the limited data, even rare species were used in the analyses.

Racomitrium lanuginosum, the targeted species for the moss spreading and the most dominant moss in the adjacent area (Aradottir & Fridriksdottir 2011), did not seem to spread much as both its cover and frequency of occurrence had not increased significantly since 2009. On the other hand, *Racomitrium eriocoides*/*Racomitrium canascens* increased greatly in cover and frequency throughout the study area. Since *R. lanuginosum* had both much higher cover and frequency in 2009 than *R. eriocoides*/*R. canascens* (Aradottir & Fridriksdottir 2011) the results suggest that *R. eriocoides*/*R. canascens* is a better colonizer than *R. lanuginosum*. *R. eriocoides*/*R. canascens* may therefore be the dominant moss species in the early stages of succession. Five years are most likely too short a time for predicting future trends so further monitoring is needed to determine the development of the site. Herrick et al. (2006) suggested that short term monitoring of species composition was insufficient as a predictor for long term restoration success. Herrick's argument is valid because without local knowledge of the area from former studies and observation from surrounding undisturbed areas, this study could have concluded that *R. eriocoides*/*R. canascens* would eventually become the dominant species. However, as *R. lanuginosum* is the dominant species in the surrounding undisturbed areas, the results can be attributed to transient dynamics, where the better colonizing *R. eriocoides*/*R. canascens*, is likely to eventually be replaced by *R. lanuginosum*, which may be a better competitor but poor colonizer (Gleeson & Tilman 1990; Walker & del Moral 2003).

5.2 Vegetation turfs

Although no measurements were carried out due to lack of baseline information for comparison, the photographs indicate that the turf treatment had influenced the development of vegetation on the track, resulting in some restoration progress (Figs. 6 and 7). This is based on observations of photographs showing more vegetation cover on the photographs taken in 2013 compared to those from 2008. Therefore, not only had the transplanted turf patches established themselves, but vegetation had been established between the patches as well. This might be an indication that the donor and recipient sites had similar abiotic conditions or optimal turf sizes were used in the restoration, as these were found to be critical contributing factors to the success of turf transplants in previous studies (Bullock 1998; Krautzer & Wittmann 2006; Klimeš et al. 2010; Aradottir 2012).

The lack of baseline information made the collection of quantitative data impractical, as there were no data to compare my findings with in order to statistically determine whether any changes had occurred on the track. Valid observations indicating an increase in vegetation cover could, however, be made from comparisons of the photographs in Figure 6B and 6C, as they were taken at almost the same points. This is evident from the curve of the track and the mountain in the background (Fig. 6B and 6C). On the other hand, the close-up photographs were not taken from the same points (Fig. 7) and therefore do not compare exactly the same parts of the track, thus probably giving false positive results. Thus, although they may possibly give an implication of progress on the track since before the turfs were transplanted, they do not allow any conclusion to be made. An experimental study in the same area has shown that vegetation cover increased significantly only two years after transplanting turfs (Aradottir 2012). Bullock (1998) further cautions that plant species composition may be modified during the process of transplanting, as the conditions may be altered to favour a different community from the native one. Due to the above limitations, this study was not aimed at and could not determine changes in species composition between what was initially transplanted and what is currently on the track.

5.3 Monitoring of restoration progress

The focus for monitoring restoration progress depends directly on the restoration activities and the associated goals and objectives, which should be stipulated in the project plan (Elzinga et al. 1998; NAVFAC Risk Assessment Workgroup and Argonne National Lab 2004; Herrick et al. 2006; Lewis et al. 2009). There was no written project plan with clearly stipulated restoration goals for the Hellisheiði area, nor detailed recordings of the treatment designs, including where and how much of the treatments were applied. This made it generally difficult to determine progress or evaluate the restoration methods in the area. Further limiting the study was the fact that there were no provisions for future monitoring in place, apart from the permanent transects set out for the study by Aradottir and Fridriksdottir (2011) at the treated site. No control sites were identified for that study. The turf treatment as mentioned earlier also lacked baseline data for future comparisons during monitoring. These problems are not unique to these restoration projects, however, as restoration activities are often not set up in a way that allows for future monitoring (Brainbridge

2007). All of the above emphasize the need for clear and realistic goals and objectives in all restoration projects, and an elaborate plan that includes baseline information and provision for future monitoring of restoration progress (Elzinga et al. 1998; Herrick et al. 2006; Lewis et al. 2009).

Although this study highlights limitations described above and thus could only reach limited conclusions, future monitoring of these areas would still be beneficial. The current study has identified control areas for the moss treatment which could also be used for the turf treatment as both treatments were applied in the same area, with similar substrates and environmental conditions. Permanent transects have been placed at the control sites and the GPS coordinates to those transects are included in Appendix 1. I recommend that these transects should be left untreated for future comparisons. In addition, three other quadrates should be randomly selected along transect 2 at the control site, to replace the outliers. To incorporate the new quadrates into the dataset I recommend that they are established next summer and all quadrates from this year together with the new quadrates measured. Future monitoring of all transects in the moss treated area and control should be carried out every 1-5 years (Herrick et al. 2006). Relevant information for monitoring, such as species composition and vegetation cover on the track, soil and site stability, and biotic integrity on both study sites can also be collected at this point, to allow for future comparisons. Furthermore, permanent photo points for monitoring should be put in place as soon as possible, for increased accuracy of comparisons over time. Generally, in order to enable future monitoring, restoration projects should provide sufficient data for the site before the commencement of restoration activities, including site identification and mapping, and complete descriptions of what was done (Brainbridge 2007).

6. CONCLUSIONS

The ecological restoration method of spreading moss branches has triggered an increase in the frequency of mosses recorded in the study with the exception of the group "other mosses". The treatment also appeared to influence vascular plant species richness and cover, although neither was significantly greater at the treated site than the control. This lack of significance may, however, most probably be explained by three outliers at the untreated site with a higher number of species and higher cover compared to any other quadrates in the study, thus possibly masking the difference between the two sites. All mosses apart from *R. lanuginosum* increased their frequency between 2009 and 2013. The cover of two moss species was compared between 2009 and 2013, and only *R. eriocoides*/*R. canascens* increased significantly. Thus I conclude that over the five year period since the moss branches were spread there had been a general increase in frequency of mosses and cover of *R. eriocoides*/*R. canascens*, one of the two compared mosses.

From photographs taken prior and after turfs had been transplanted on the vehicle track I conclude that vegetation cover on the track appears to have increased, but due to lack of baseline information no further conclusions can be drawn from the photographs.

ACKNOWLEDGEMENTS

Working on this project has been a real learning experience, not only for the Icelandic context, but it also provided me with knowledge on the importance of monitoring and setting clear goals and objectives for any restoration project, which I can incorporate into my work and apply in Namibia. I would therefore, first and foremost, like to thank GOD, my creator and source of all wisdom for carrying me through this process. For providing me with the chance of a lifetime, my gratitude goes to the UNU Land Restoration Training Programme and the Icelandic Ministry for Foreign Affairs. To the Agricultural University of Iceland and the Soil Conservation Service of Iceland, I would like to extend a special vote of thanks for hosting us during the programme. Hafdis Hanna, Berglind and Halldora, thank you for driving us around and always wanting to know if we were okay, your concern for our well-being did not go unnoticed and is much appreciated. To my supervisor Kristin, I would like to say thank you for your guidance and patience, it was a real pleasure working with you. I further acknowledge all the instructors we had throughout the programme for laying the foundation for our project ideas. I would also like to convey my appreciation to Anna Sigga and Magnea for assisting me with data collection. Azfar, thank you for tirelessly driving us to and from the bus stop so that we could meet our supervisors, for taking us shopping and generally devoting so much of your time to us. To the other UNU-LRT fellows of 2013, thank you for being my family away from home. Last, but certainly not least, my sincere gratitude and appreciation go to my husband Mufaro Nesongano for enduring over six months of loneliness for me to be able to do this. I could not have done this without your support my love, thank you.

REFERENCES

Aradottir, A.L., and H. Fridriksdottir. 2011. Dreifing gamburmosa á raskað land [Spread of *Racomitrium lanuginosum* on disturbed land]. Pages 89-94 in A. L. Aradottir and J. Grétarsdóttir, editors. Endurheimt staðargróðurs á röskuðum hálendissvæðum [Ecological restoration of native vegetation in disturbed highland areas]. Rit Lbhí **29**. (in Icelandic).

Aradottir, A.L., and J. Grétarsdóttir. 2011. Endurheimt staðargróðurs á röskuðum svæðum [Restoration of adjacent vegetation on eroded land]. Pages 3-13 in A. L. Aradottir and J. Grétarsdóttir, editors. Endurheimt staðargróðurs á röskuðum hálendissvæðum [Ecological restoration of native vegetation in disturbed highland areas]. Rit Lbhí **29**. (in Icelandic).

Aradottir, A. L. 2012. Turf transplants for restoration of alpine vegetation: does size matter? Journal of Applied Ecology **49**:439-446.

Armas, C., and F. I. Pugnaire. 2005. Plant interactions govern population dynamics in a semi-arid plant community. Journal of Ecology **93**:978-989.

Arnalds, O., and J. Kimble. 2001. Andisols of deserts in Iceland. Soil Science Society of America Journal **65**:1778-1786.

Arnalds, O., F. O. Gisladottir, and H. Sigurjonsson. 2001a. Sandy deserts of Iceland: An overview. Journal of Arid Environments **47**:359-371.

Arnalds, O., E. F. Thorarinsdottir, S. Metusalemsson, A. Jonsson, E. Gretarsson, and A. Arnason. 2001b. Soil erosion in Iceland. The Soil Conservation Service and the Agricultural Research Institute, Reykjavik, Iceland.

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.

Arnalds, O. 2008. Soils of Iceland. Jökull **58**:409-421.

Brainbridge, D. A. 2007. A guide for desert and dryland restoration: New hope for arid lands. Island Press, Washington D.C, USA.

Brink, V. 2005. Alpine plants: Adapting to a harsh environment. Davidsonia **16**(3):95-105.

Bullock, J. M. 1998. Community translocation in Britain: Setting objectives and measuring consequences. Biological Conservation **84**(3):199-214.

Campeau, S., and K. Blanchard. 2010. Use of locally sourced moss, lichen and vascular plant propagules for the revegetation of mineral disturbances in a boreal climate. URL https://circle.ubc.ca/bitstream/handle/2429/30349/15%20Campeau.pdf?sequence=1 [accessed on 11 August 2013].

Castro, J., R. Zamora, J. A. Hódar, J. M. Gómez, and L. Gómez-Aparicio. 2004. Benefits of using shrubs as nurse plants for reforestation in Mediterranean mountains: A 4-year study. Restoration Ecology **12**(3):352-358.

Chamber of Mines of Namibia. 2010. Namibian Mine Closure Framework, final report. URL http://www.chamberofmines.org.na/index.php?id=247 [accessed on 25 May 2013].

Choi, Y. C. 2004. Theories for ecological restoration in changing environment: Toward 'futuristic' restoration. Ecological Research **19**:75-81.

Clewell, A. F., and J. Aronson. 2007. Ecological restoration: principles, values, and structure of an emerging profession. Island Press, Washington, D. C, USA.

Conlin, D. B., and J. J. Ebersole. 2001. Restoration of an alpine disturbance: Differential success of species in turf transplants, Colorado, USA. Arctic, Antarctic, and Alpine Research **33**(3):340-347.

Costanza, R., R. D'Arge, R.S. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature **387**:253-260.

Crofts, R. 2011. Healing the land. Soil Conservation Service of Iceland, Gunnarsholt, Iceland.

Davy, A. J. 2002. Establishment and manipulation of plant populations and communities in terrestrial systems. Pages 223-241 in M. R. Perrow and A. J. Davy, editors. Handbook of ecological restoration, volume 1: Principles of restoration. Cambridge University Press, Cambridge, UK.

de Klerk, J. N. 2004. Bush encroachment in Namibia. Ministry of Environment and Tourism, Government of the Republic of Namibia, Windhoek, Namibia.

del Moral, R., L. R. Walker, and J. P. Bakker. 2007. Insights gained from succession for the restoration of landscape structure and function. Pages 19-44 in L. R. Walker, J. Walker and R. J. Hobbs, editors. Linking restoration and ecological succession. Springer, New York, USA.

Driscoll, D. L., A. Appiah-Yeboah, P. Salib, and D. J. Rupert. 2007. Merging qualitative and quantitative data in mixed methods research: How to and why not. Ecological and Environmental Anthropology (University of Georgia). Paper 18.
URL http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1012&context=icwdmeea [accessed on 26 July 2013].

Elzinga, C. L., D. W. Salzer, and J. W. Willoughby. 1998. Measuring and monitoring plant populations. Bureau of Land Management, Technical Reference 1730-1, BLM/RS/ST-98/00+1730. URL http://www.blm.gov/nstc/library/pdf/MeasAndMon.pdf [accessed on 23 July 2013].

Galatowitsch, S. M. 2012. Ecological restoration. Sinauer Associates, Inc. Sunderland, MA, USA.

Gleeson, S. K., and D. Tilman. 1990. Allocation and the transient dynamics of succession on poor soils. Ecology **71**(3):1144-1155.

Gomez-Aparicio, L., R. Zamora, J. Castro, and J. A. J. Hódar. 2008. Facilitation of tree saplings by nurse plants: microhabitat amelioration or protection against herbivores? Journal of Vegetation Science **19**:161-172.

Greipsson, S. 2011. Restoration ecology. Jones and Bartlett Learning, Sudbury, MA, USA.

Haney, G. 2010. Visual impact assessment of small-scale mining in Iceland: A tool for municipal planning and decision making. MSc Thesis, Faculty of Life and Environmental Sciences, University of Iceland, Reykjavik, Iceland.

Herrick, J. E., G. E. Schuman, and A. Rango. 2006. Monitoring ecological processes for restoration projects. Journal for Nature Conservation **14**:161-171.

Hobbs, R. J., and J. A. Harris. 2001. Restoration ecology: Repairing the earth's ecosystems in the new millennium. Restoration Ecology **9**(2):239-246.

Jumpponen, A., H. Vöre, K. G. Mattson, R. Ohtonen, and J. M. Trappe. 1999. Characterization of 'safe sites' for pioneers in primary succession on recently deglaciated terrain. Journal of Ecology **87**:98-105.

Kirkpatrick, H. E., J. W. S. Barnes, and B. A. Ossowski. 2006. Moss interference could explain the microdistributions of two species of Monkey-flowers (*Mimulus*, Scrophulariaceae). Northwest Science **80**:1-8.

Klimeš, L., I. Jongepierová, J. Doležal, and J. Klimešová. 2010. Restoration of a species-rich meadow on arable land by transferring meadow blocks. Applied Vegetation Science **13**:403-411.

Kondolf, G. M. 1996. A cross section of stream channel restoration. Journal of Soil and Water Conservation **51**(2):119-125.

Krautzer, B., and Wittmann. 2006. Restoration on alpine ecosystems. Pages 208-220 in J. van Andel and J. Aronson, editors. Restoration ecology: The new frontier. Blackwell Science Ltd., Oxford, UK.

Lange, G. 2003. The contribution of minerals to sustainable economic development: Mineral resource accounts in Namibia. DEA research discussion paper, Number 54. Directorate of Environmental Affairs (DEA), Ministry of Environment and Tourism, Windhoek, Namibia.

Lewis, D., M. Lennox, and S. Nossaman. 2009. Developing a monitoring program for riparian revegetation projects: University of California Division of Agriculture and Natural Resources. URL http://ucanr.org/freepubs/docs/8363.pdf [accessed on 27 July 2013].

Machmer, M., and C. Steeger. 2002. Effectiveness monitoring guidelines for ecosystem restoration: Final report. Pandion Ecological Research and Ministry of Water, Land and Air

Protection, Victoria, B.C., Canada.
URL http://www.env.gov.bc.ca/wld/documents/fia_docs/rest_effect_mon_guidelines_s.pdf [accessed on 26 June 2013].

NAVFAC (Naval Facilities Engineering Command) Risk Assessment Workgroup and Argonne National Lab. 2004. Guidance for habitat restoration monitoring: Framework for monitoring plan development and implementation. Naval Facilities Engineering Command, Washington, D.C., USA. URL http://www.tidalmarshmonitoring.org/pdf/NAVFAC2004_HabitatRestorationMonitorGuide0704.pdf [accessed on 23 July 2013].

Oesterheld, M., and O. E. Sala. 1990. Effects of grazing on seedling establishment: The role of seed and safe-site availability. Journal of Vegetation Science **1**:353-358.

Palmer, M. A., R. F. Ambrose, and N. L. Poff. 1997. Ecological theory and community restoration ecology. Restoration Ecology **5**(4):291-300.

Pickett, S. T. A., S. L. Collins, and J. J. Armesto. 1987. Models, mechanisms and pathways of succession. Botanical Review **53**(3):335-371.

Poulin, M., T. Landry, V. Laberge, and L. Rochefort. 2012. Establishing vascular plants from seeds around pool margins in restored peatlands. Extended abstract. 14th International Peat Congress, Peatlands in Balance. Stockholm, Sweden. URL http://www.gret-perg.ulaval.ca/uploads/tx_centrerecherche/Poulin_264_03.pdf [accessed on 30 July 2013].

Poulin, M., R. Andersen, and L. Rochefort. 2013. A new approach for tracking vegetation change after restoration: A case study with peatlands. Restoration Ecology **21** (3):363-371.

Rayburn, A. P., J. B. Davidson, and H. M. White. 2012. Possible effects of moss on distribution and performance of a threatened endemic primrose. Western North American Naturalist **72**(1):84-92.

Reay, S. D., and D. A. Norton. 1999. Assessing the success of restoration plantings in a temperate New Zealand forest. Restoration Ecology **7**(3):298-308.

Rochefort, L., F. Isselin-Nondedeu, S. Boudreau, and M. Poulin. 2013. Comparing survey methods for monitoring vegetation change through time in a restored peatland. Wetlands Ecology and Management **21**:71-85.

Sedia, E. G., and J. G. Ehrenfeld. 2003. Lichens and mosses promote alternate stable plant communities in the New Jersey Pinelands. Oikos **100**:447-458.

Society for Ecological Restoration International Science and Policy Working Group. 2004. The SER international primer on ecological restoration. Society for Ecological Restoration International. URL http://www.ser.org/docs/default-document-library/english.pdf [accessed on 26 November 2012].

ter Braak, C. J. F., and P. Šmilauer. 2012. Canoco reference manual and user's guide: Software for ordination (version 5.0). Microcomputer Power, Ithaca, NY, USA.

van Andel, J., and A. P. Grootjans. 2006. Concepts in restoration ecology. Pages 16-28 in J. van Andel and J. Aronson, editors. Restoration ecology: The new frontier. Blackwell Science Ltd., Oxford, UK.

Walker, L. R., and R. del Moral. 2003. Primary *s*uccession and *e*cosystem rehabilitation. Cambridge University Press, Cambridge, UK.

Walker, L. R., J. Walker, and R. del Moral. 2007. Forging a new alliance between succession and restoration. Pages 1-18 in L. R. Walker, J. Walker and R. J. Hobbs, editors. Linking restoration and ecological succession. Springer, New York, USA.

Walker, L. R., and R. del Moral. 2008. Lessons from primary succession for restoration of severely damaged habitats. Applied Vegetation Science **12**:55-67.

Wang, Z., J. Nishihiro, and I. Washitani. 2012. Regeneration of native vascular plants facilitated by *Ischaemum aristatum* var. glaucum tussocks: An experimental demonstration. Ecological Restoration **27**:239-244.

Whisenant, S. G. 1999. Repairing damaged wildlands: A process-orientated, land-scale approach. Cambridge University Press, Cambridge, UK.

Woodward, A., and K. Hollar. 2011. Monitoring habitat restoration projects: U.S. Fish and Wildlife Service Pacific Region Partners for Fish and Wildlife Program and Coastal Program protocol. Techniques and methods 2-A11. U.S. Department of the Interior and U.S. Geological Survey. URL http://pubs.usgs.gov/tm/tm2a11/pdf/tm2a11.pdf [accessed on 23 July 2013].

Young, T. P., J. M. Chase, and R. T. Huddleston. 2001. Community succession and assembly: Comparing, contrasting and combining paradigms in the context of ecological restoration. Ecological Restoration **19**:5-18.

Zamfir, M. 2000. Effects of bryophytes and lichens on seedling emergence of alvar plants: Evidence from greenhouse experiments. Oikos **88**:603-611.

APPENDICES

APPENDIX 1. *GPS coordinates for the sampled transects in a study at the Gígahnúkur crater, south-west Iceland, and the locations of quadrates along these transects.*

Transect	GPS location for transect		Quadrates location
	Start	End	
Treated 1	N64 01 39.8 W21 22 05.6	N64 01 39.7 W21 22 03.3	2, 4, 11, 12*, 13, 21, 22, 24, 25, 29
Treated 2	N64 01 41.4 W21 22 04.5	N64 01 41.3 W21 22 02.3	6, 8, 11, 12, 14, 15, 18, 19, 22, 27
Treated 3	N64 01 42.2 W21 22 01.3	N64 01 42.2 W21 21 59.2	0, 2, 4.5, 10, 14, 16, 22, 27, 28, 29
Treated 4	N64 01 45.4 W21 21 58.4	N64 01 45.8 W21 21 56.3	3, 9, 11, 15, 16, 18, 20, 21, 23, 27
Untreated 1	N64 01 62.6 W21 22 17.1	N64 01 61. 9 W21 22 19. 1	2, 6, 15, 19, 20 (plus 5 control plots from an experiment)
Untreated 2	N64 01 61.8 W21 21 95.2	N64 01 62.5 W21 21 91.6	1, 4, 7, 9, 18, 21, 22, 24, 25, 26
Untreated 3**	N64 01 63.4 W21 21 87.5	N64 01 63. 6 W21 21 86.5	2, 3, 4, 6, 7
Untreated 4	N64 01 67.2 W21 21 75.5	N64 01 68.3 W21 21 73.9	1, 3, 5, 8, 11, 13, 16, 21, 22, 24

*This quadrate was sampled on the left side of the transect line as was done in 2009 (Aradottir and Fridriksdottir 2011).

**This transect was 10 m long.

APPENDIX 2. *Cover classes of vascular plant species along the four untreated transects near the Gígahnúkur crater, south-west Iceland in 2013.* Festuca *sp. (s) refers to sown individuals of the species* Festuca*, while* Festuca *sp. (n) refers to unsown* Festuca.

Transect	Quadrate	Species	Class
1	**1**	*Festuca* sp. (n)	1
	2	None	
	3	*Festuca* sp. (n)	1
	4	*Festuca* sp. (n)	1
	5	*Festuca* sp. (n)	1
		Cerastium fontanum	1
	6	None	
	7	*Koenigia islandica*	1
	8	None	
	9	*Cerastium fontanum*	1
		Poa pratensis	1
	10	*Cerastium fontanum*	1
		Koenigia islandica	1
2	**1**	None	
	2	*Festuca* sp. (n)	1
	3	None	
	4	*Festuca* sp. (n)	1
		Luzula spicata	1
		Poa pratensis	1
	5	*Festuca* sp. (n)	3
		Luzula spicata	2
		Bistorta vivipara	2
		Salix herbacea	4
2	**5**	*Silene acaulis*	2
		Juncus trifidus	2
		Carex bigelousii	2
	6	*Festuca* sp. (n)	2
		Luzula spicata	1
		Bistorta vivipara	1
		Salix herbacea	1
		Juncus trifidus	2
	7	*Festuca* sp. (n)	2
		Luzula spicata	2
		Salix herbacea	2
		Juncus trifidus	2
		Alchemilla alpina	1
	8	*Descampsia* sp.	2
	9	None	
	10	None	

3	**1**	None	
	2	None	
	3	None	
	4	None	
	5	*Festuca* sp. (n)	1
		Luzula spicata	1
4	**1**	*Luzula spicata*	1
	2	None	
	3	None	
	4	*Festuca* sp. (n)	1
		Luzula spicata	1
	5	None	
	6	None	
	7	None	
	8	None	
	9	*Festuca* sp. (n)	1
	10	None	

APPENDIX 3. *Cover classes of vascular plant species along four treated transects at the Gígahnúkur crater, south-west Iceland in 2013.* Festuca *sp. (s) refers to sown individuals of the species* Festuca*, while* Festuca *sp. (n) refers to unsown* Festuca.

Transect	Quadrate	Species	Class
1	**1**	*Festuca* sp. (n)	1
	2	*Festuca* sp. (n)	2
	3	*Festuca* sp. (s)	2
		Festuca sp. (n)	2
1	**3**	*Poa pratensis*	1
	4	*Festuca* sp. (s)	2
		Festuca sp. (n)	2
		Poa pratensis	1
	5	*Festuca* sp. (n)	2
	6	None	
	7	*Festuca* sp. (n)	1
	8	*Festuca* sp. (n)	1
	9	*Festuca* sp. (n)	1
	10	*Festuca* sp. (n)	2
2	**1**	*Festuca* sp. (n)	1
		Arabidiopsis petraea (cart petr)	1
	2	None	
	3	*Festuca* sp. (n)	2
		Arabidiopsis petraea (cart petr)	1
		Poa pratensis	1
		Cerastium fontanum	1
	4	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
	5	*Festuca* sp. (n)	1
		Festuca sp. (s)	3
		Luzula spicata	1
	6	*Festuca* sp. (n)	1
		Festuca sp. (s)	1
		Luzula spicata	1
	7	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
	8	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
	9	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
		Agrostis capilleris	1
	10	*Festuca* sp. (n)	2
		Festuca sp. (s)	2

Transect	Quadrate	Species	Class
3	**1**	*Festuca* sp. (n)	1
	2	*Festuca* sp. (s)	1
	3	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
		Empetrum nigrum	1
	4	*Festuca* sp. (n)	1
		Festuca sp. (s)	2
		Festuca sp. (n)	1
3	**5**	*Festuca* sp. (s)	2
	6	*Festuca* sp. (n)	2
		Festuca sp. (s)	2
	7	*Festuca* sp. (n)	1
		Festuca sp. (s)	2
	8	*Festuca* sp. (n)	1
		Festuca sp. (s)	1
		Luzula spicata	1
	9	*Festuca* sp. (n)	2
		Juncus trifidus	1
	10	*Festuca* sp. (n)	1
		Festuca sp. (s)	1
4	**1**	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
		Agrostis capilleris	2
	2	*Festuca* sp. (n)	2
		Festuca sp. (s)	1
	3	*Festuca* sp. (n)	1
		Festuca sp. (s)	4
		Agrostis capilleris	1
	4	*Festuca* sp. (s)	1
	5	*Festuca* sp. (s)	2
		Agrostis vinealis	2
	6	*Festuca* sp. (n)	1
		Festuca sp. (s)	2
		Agrostis vinealis	1
	7	*Agrostis vinealis*	1
	8	*Festuca* sp. (s)	2
		Galium normanii	1
	9	*Festuca* sp. (s)	2
		Cerastium fontanum	1
	10	*Festuca* sp. (n)	1
		Festuca sp. (s)	2
		Agrostis vinealis	2

Juncus trifidus	1

APPENDIX 4.

Appendix 4. *One of the quadrats (quadrat 5) along control (untreated) transect 2 with very high vascular plant cover and species richness (left) and a typical quadrat at the untreated site (right) near the Gígahnúkur crater, south-west Iceland in 2013.*

APPENDIX 5.

Appendix 5. *Cover and frequency of mosses along four untreated transects near the Gígahnúkur crater, south-west Iceland in 2013. The maximum and minimum values are indicated for* R. lanuginosum *and* R. eriocoides/ R. canascens, *which are the most common and abundant species in the study area.*

Transect	*R. lanuginosum*				*R. eriocoides/R. canascens*				*Polytrichum* sp./*Pogonatum* sp.		Other mosses	
	%Cover	%Frequency			%Cover	%Frequency			%Cover	%Frequency	%Cover	%Frequency
	Mean	Mean	Min	Max	Mean	Mean	Min	Max	Mean	Mean	Mean	Mean
1	0.25	0.8	0	3	0.5	18.6	2	51	0.05	0.2	0.05	0.1
2	0.4	4.8	0	17	0.5	9.1	3	16	0.95	5.7	0.5	2.7
3	0.5	21.6	13	29	0.3	0.6	0	1	0.1	0	0.1	0
4	0.4	10.6	1	27	0.4	2.2	0	7	0.05	0.5	0.05	0.1
Mean	**0.4**	**9.5**			**0.4**	**7.63**			**0.3**	**1.6**	**0.2**	**0.7**
SE	**0.05**	**4.5**			**0.05**	**4.1**			**0.22**	**1.4**	**0.11**	**0.66**

APPENDIX 6.

Appendix 6. *Mean cover and frequency of mosses along four treated transects at the Gígahnúkur crater south-west Iceland in 2013. The maximum and minimum values are shown for* R. lanuginosum *and* R. eriocoides/ R. canascens*, which are the most common and abundant species in the study area.*

Transect	***R. lanuginosum***				***R. eriocoides/R. canascens***				***Polytrichum* sp./*Pogonatum* sp.**		**Other mosses**	
	%Cover	**%Frequency**			**%Cover**	**%Frequency**			**%Cover**	**%Frequency**	**%Cover**	**%Frequency**
	Mean	**Mean**	**Min**	**Max**	**Mean**	**Mean**	**Min**	**Max**	**Mean**	**Mean**	**Mean**	**Mean**
1	5	34.6	19	71	3	67	32	86	0.4	4.2	0.25	2.7
2	3.3	33.5	0	79	3.7	75.5	50	100	0.35	6.1	0.2	2
3	4	27.5	7	46	6.5	89.2	71	100	0.7	7.8	0.4	3.9
4	3.5	23.7	2	53	3.8	63.9	47	88	0.75	8.1	0.35	1.9
Mean	**3.9**	**29.8**			**4.2**	**73.9**			**0.6**	**6.55**	**0.3**	**2.6**
SE	**0.39**	**2.57**			**0.77**	**5.66**			**0.1**	**0.9**	**0.05**	**0.46**