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EFFECT OF *SALIX PHYLICIFOLIA* PATCHES ON MICROSITE AVAILABILITY ON DEGRADED SITES IN ICELAND

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ABSTRACT

Within any degraded sites are "islands of fertility" that differ in environmental characteristics on the spatial scale of the individual seed or seedling. By studying conditions at these microsites we can learn about the barriers to restoration and also use the microsites to kick-start the process of restoration. This study assesses the presence of microsites created by "tea leaved" willow a native shrub in Iceland. This study seeks to answer the questions whether "tea leaved" willow (*Salix phylicifoli)* improves microsites for colonization and succession in degraded sites, if the effect of the patch extends beyond the patch canopy and whether the microsite is a result of biological processes or a physical process due to capture of organic debris, aeolian material and snow by the willow canopy. The study also assesses if the presence of willow patches leads to variation in plant composition and reduced susceptibility to erosion. Soil samples were taken in and around willow patches of differing canopy diameter and analysed for total carbon, phosphorus, nitrogen, pH and water holding capacity. These samples were compared to a control sample taken in an area without any willows. The willow site was also compared to another site without willows to assess resistance to wind and water erosion which are major drivers of degradation in Iceland. The results show that the presence of willows improved soil quality, as seen in higher values of total C, N, and P. The presence of willows also ameliorates soil pH which could benefit plants that thrive in low pH soils. Willows ultimately increased foliar cover and improved site stability to soil erosion by both water and wind. It is concluded that willows can be used for land restoration as a low cost alternative to seeding and fertilizer application.

Key words: land restoration, microsites, *Salix phylicifoli.*

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1. INTRODUCTION

1.1 Background to the study

Land degradation is a gradual process in which the land loses its ecosystem functions and productivity due to several disturbances from which it may not recover if unaided (Bai et al. 2008). There is general consensus now that land degradation is not just a collection of global problems but an issue of global concern (Gisladottir & Stocking 2005). However, the extent of land degradation varies from one place to another, with some regions being worse off than others (Oldeman et al. 1991). In Uganda, land degradation is a serious problem (Banadda 2010). According to Olson & Berry (2003), land degradation is estimated to cost Uganda about 4-12% of gross net product. Most of the land degradation is in the form of soil erosion (Sserunkuuma et al. 2001), especially in the highland areas (National Environmental Management Authority 2010), due to a combination of natural factors (topography and climate) and anthropogenic factors (high population and poor farming methods) that work in synergy to exacerbate the problem.

The process of land degradation is not spontaneous, but a step-wise process in which the ecosystem regresses from a state of full functionality (Whisenant 1999). This creates a continuum of degradation from the most degraded areas in which all ecosystem functions are compromised (Rapport et al. 1998) to ones where some functionality still exists, only needing management interventions to achieve restoration goals (Aradottir 2007). In a healthy ecosystem there is an in-built mechanism to transition or self-repair (Whisenant 1999), but if the degrading agents exceed the transition threshold, the ecosystem will revert to a lower state. Transitions are controlled by either biotic or abiotic interactions. The kind of interaction controlling a transition will determine the kind of intervention required for restoration and subsequently the cost of restoration (Holl et al. 2007; Arnalds & Thorsson 2012).

In order to regain the use of degraded land, there is a need to initiate and accelerate natural processes and functions (SER 2004). This can be achieved by undertaking ecological restoration (Holl et al. 2007; Hobbs et al. 2011). The Society for Ecological Restoration International Science & Policy Working Group (SER 2004) defines ecological restoration as *"*the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed." (p. 3). Restoration is dependent on identifying factors that prevent an ecosystem from reverting to a less degraded state (Whisenant 1999) in order to prescribe interventions that would enable such factors to be overcome. If the limits of the ecosystem to naturally follow its ecological trajectory have not been exceeded, use of external inputs may be uneconomical (Maestre & Cortina 2004) since recovery can occur naturally.

The ultimate challenge for restoration programs is to repair systems and regain the primary processes of energy capture, hydrology and nutrient recycling (Whisenant 1999). In Iceland the task was further complicated by wind erosion, and therefore land restoration in Iceland initially involved the use of physical barriers to stem drifting sand (Halldorsson et al. 2012). Lyme grass (*Leymus arenarius)* and fertilizer were used to stop drifting sand wherever the sowing technology and seed availability permitted (Crofts 2011). Later, the Soil Conservation Service of Iceland, established in 1907, started systematic restoration involving a number of approaches like fencing off degraded land (Ágústsdóttir 2004), reduction of sheep numbers since the early 1970s to reduce the pressure on grazing land (Arnalds & Barkarson 2003), use of organic and inorganic fertilizers, seeding with grass and a combination of seeding and fertilizer applications (Crofts 2011). Nootka lupine (*Lupinus nootkatensis)*, an exotic nitrogen fixing legume, has also been used although it's highly invasive (Magnússon 2010).

Restoration methods have evolved over time as the field of ecological restoration has developed but the objective remains to increase plant cover and accumulate carbon in the soil (Aradóttir et al. 2000). While re-vegetation using agronomic means like fertilization and seeding with desired species of grass and herbs can help restore degraded land (Gretarsdottir et al. 2004), they can be prohibitively expensive over large areas. Moreover, there is still an ongoing debate on who should pay for the cost of restoration (Rees et al. 2007). According to Mcghee et al. (2007) there is a lot of promise in the fact that restoration can occur naturally in spite of human interventions and investment.

The use of exotic species shows that proper ecological functioning is seldom the only motivation for restoration programs (Hagen et al. 2013). It rather illustrates the prominence of the "return of structure" mindset (e.g. species composition and nutrient availability) (Whisenant 1999) in restoration programs. The time-bound nature of restoration programs and need to show results for funds spent could explain the preference for "return of structure" over "return of function". However, concentrating on structural repair can create more problems than it solves, especially with regard to exotic species becoming invasive (Aradottir et al. 2004). As an alternative, reclamation could be undertaken by enhancing colonization of degraded sites by native species (Elmarsdottir et al. 2003).

Colonisation and succession in degraded areas may be hampered by nutrient limitations, lack of propagules and limited water retention among other factors (Tsuyuzaki et al. 1997; Aradottir 2007). In order to kick-start germination and colonisation, a conducive environment has to be available in the immediate surrounds of the seed. This is an important prerequisite, especially for non-colonisers that cannot survive under adverse conditions of severe nutrient deprivation. However, early colonizers may be able to become established under harsh conditions and create patches of vegetation on otherwise degraded poor sites. Under and around these patches colonization blossoms (Titus & del Moral 1998), probably due to the ability to trap propagules and/or to improve nutrient availability. The patches may create differences in temperature, soil nutrients and soil organic matter which make optimum conditions for growth and survival of other plants by breaking the abiotic thresholds of the system.

According to Kondo et al. (2012), these patches create "islands of fertility" in which the limiting factors to germination and colonization have been broken. These "islands of fertility" are called microsites, i.e. "areas that differ in environmental characteristics on the spatial scale of an individual seed or seedling" (Titus & del Moral 1998, p. 13). Other landscape features like rocks (Titus & del Moral 1998) and biological soil crust (Elmarsdottir et al. 2003) can create microsites important for seedling recruitment (Jones & Moral 2005). The study of microsites is important because the success of colonization and emergence of seedlings is affected by the immediate surroundings of the seedling (Grubb 1977; Elmarsdottir et al. 2003). In a study in semi-arid Australia, Maestre and Cortina (2004) discovered that restoration of degraded land could be initiated if efforts were focused on maximizing the patches by increasing their number and reducing the downward slope distance between them. Vegetation patches have been used for restoration with success (see Tongway & Ludwig 1996). Restoration is not only important to bring land into production either as pasture land or agricultural land, but can also be used to halt further degradation. For instance, increasing plant cover can significantly reduce susceptibility of land to the effects of water and wind erosion.

Tea-leaved willow *(Salix phylicifolia*) is a native early colonizer species of Iceland (Aradottir 2007) which has been highly recommended for use in restoration (Svavarsdóttir, as cited in Aradottir 2007), probably due to its ability to grow in adverse conditions. Willow colonization has been seen to occur naturally in the Hekluskogar area (Aradottir 2007) and willow patches show close association with other plants implying a different microclimate in relation to the presence of propagules, nutrient availability or improved water retention, thereby breaking the limiting conditions for plant colonization and succession.

Over 40% of land in Iceland is degraded (Arnalds et al. 2001a). Degradation in Iceland is either a result of natural phenomena like climatic change and volcanic activity (Arnalds 1987; Gísladóttir et al. 2010; Gísladóttir et al. 2011) or due to anthropogenic drivers, mainly unsustainable sheep grazing (Arnalds & Barkarson 2003; Gísladóttir et al. 2010) and deforestation (Ritter 2009). Subsequent to the centuries of land degradation, almost half of Iceland has sparse to no vegetation cover (Arnalds et al. 2001b) especially in the central highland areas (Greipsson 2012). Moreover, Icelandic Andosols have low bulk density (Arnalds 2004) which makes them prone to wind erosion due to saltation, more than in any other part of the world (Arnalds et al. 2001b; Arnalds 2004). As a result, large areas have been left bare, with limited ecological function, like carbon sequestration (Óskarsson et al. 2004), water and nutrient cycling (Arnalds & Kimble 2001) needed to support healthy ecosystems.

Most of the degraded land in Iceland requires physical manipulation (cf. Whisenant 1999) to increase infiltration, reduce soil erosion and increase capture of organic matter. This was the focus of earlier restoration efforts in Iceland which concentrated on reducing soil loss by wind erosion (Runólfsson 1987) and later to initiating and accelerating vegetation colonization and succession. The latter was mainly done by seeding with native species but also with exotic fast growing species like Nootka lupine (*Lupinus nootkatensis).*

Without good knowledge of ecology, restoration programs may prescribe unnecessary and expensive interventions (Bradshaw 1987). For instance, it is important to understand the mechanism of natural succession and what is required to trigger and accelerate it. Whereas most restoration programs involve use of seed, seeding is not always necessary (Elmarsdottir et al. 2003), but may be needed if the limiting factor is the absence of propagules; likewise, some land may require seeding if the propagules are a limiting factor, in which case application of fertilizer may be unnecessary (Aradottir et al. 2001).

Seedling establishment is associated with certain types of vegetation patches or microsites (Elmarsdottir et al. 2003). However, the mechanism by which microsites enhance seedling establishment is not well known. Moreover, understanding the characteristics of microsites would be very useful in understanding patterns of colonization of new areas (Sohlberg & Bliss 1984) and in ensuring that restoration interventions are based on understanding of natural ecosystems. It is envisaged that understanding the characteristics of microsites created by willows will improve the ability to reclaim successfully degraded sites.

This study seeks to answer the questions whether willows improve microsites for colonization and succession in degraded sites, whether the effect of the patch extends beyond the patch canopy and whether the microsite is a result of biological processes by the willow or a physical process due to capture of organic debris, aeolian material and snow by the willow canopy. The study also assesses if the presence of willow patches leads to differences in plant composition and reduced susceptibility to erosion when compared with a site without willows.

1.2 Aim and objectives of the research

1.2.1 Aim of the research

The goal of this study was to assess the effect of willows on formation of microsites for plant establishment and survival and to assess the effect of willows on ecosystem plant composition.

1.2.2 Objectives of the research

1. To assess microsite characteristics by measuring selected soil attributes and nutrient availability in and near willow patches.
2. To assess changes in microsite characteristics with distance from willow patches and along the prevailing wind direction.
3. To compare plant composition of willow-covered and willow-free sites.

2. METHODS

2.1 Study area

The study was carried out at Geitasandur in South Iceland near the Soil Conservation Service of Iceland (63°50.561'N, 20°09.740'W, ca. 77 m above sea level). The area receives 920-1510 mm of annual precipitation with mean annual temperatures of 4 °C (Icelandic Met Office 2014) and mainly North-Easterly winds (Fig. 1). Due to its proximity to the Hekla volcano, the study area has experienced land degradation due to tephra deposition (Arnalds 2013). As a result, the vegetation of the area is scanty apart from areas that have received restoration treatments over the years. The study site has been fenced off from grazing for over 40 years and has not received any fertilization treatments for at least 20 years. The soils are black gravelly sandy that is low in nitrogen (N), total carbon (C) (Hunziker 2011) and water holding capacity (Arnalds & Kimble 2001).

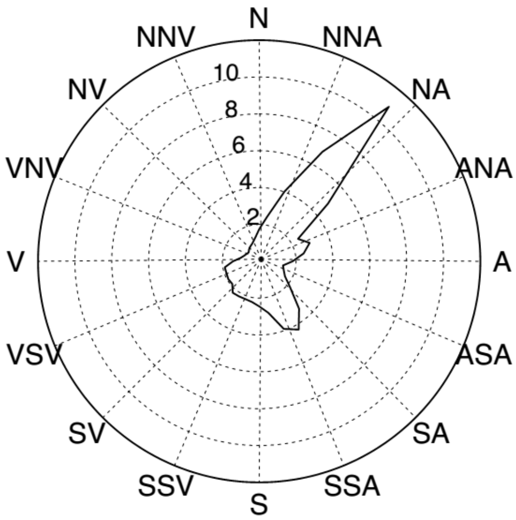

Figure 1. Wind rose of frequency of wind direction for nearest meteorological station at Hella for the period 27 April 2006 to 31 December 2013 showing the direction of the predominant winds (Icelandic Met Office 2014). North (N), East (A), South (S), West (V), North-east (NA), South-east (SA), South-west (SV), North-west (NV), North-north-east (NNA), East-north-east (ANA), East-south-east (ASA), South-south-east (SSA), South-south-west (SSV), West-south-west (VSV) West-north-west (VNV) and North-north-west (NNV).

2.2 Research design

2.2.1 Sampling design

Eight patches were selected for the study. Patches were purposively selected based on their size as either large (average canopy diameter 6.25 m) or small (average canopy diameter 1.97 m) (Fig. 2). Four small and four large patches were used in the study. All patches were on gently sloping ground and within 1 to 16 m of another patch. The control samples were taken from spots at the same site without any willow patches within a distance of 50 m in all directions.

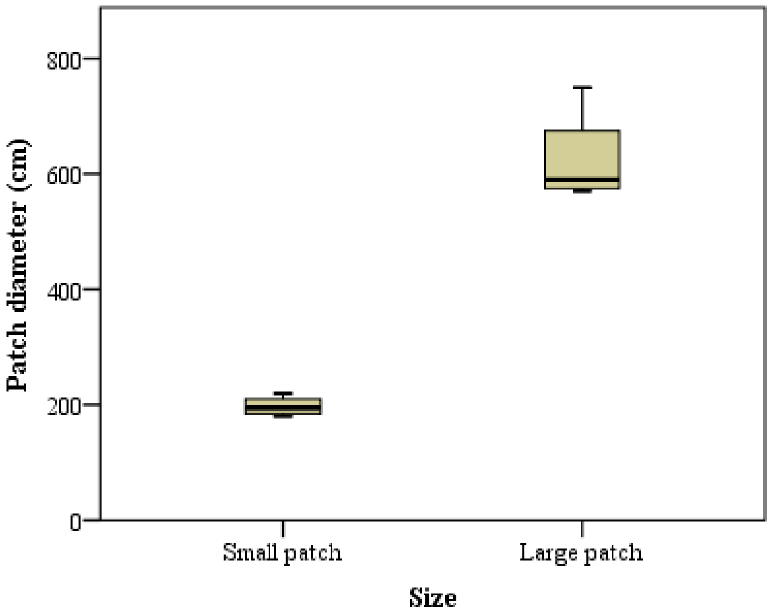

Figure 2. Box plot of patch canopy diameter for small and large patches. The top and bottom whiskers show the maximum and minimum values without outliers, respectively. The line shows the mean of the distribution and the box shows the boundaries of the upper and lower quartile.

In order to evaluate the effect of patches on microclimate availability, a transect running north-east was laid across each of the eight patches. Two soil samples were taken equidistant (0.5 m) from each canopy edge at opposite sides of the prevailing wind direction. This was done to examine whether the prevailing wind direction had an effect on the measured variables in the study since it is possible that accumulation of organic debris, aeolian material or snow in winter might influence characteristics of the microsite. In addition, two more samples were taken at the epicentre of each willow patch and the other at 3 m on the leeward side of the patch canopy (Fig. 3). This design was intended to reveal if patch size has an effect on microsite characteristics, the extent of the microsite and whether the microsite effect is towards the direction of the prevailing wind, i.e. determined by capture of materials by the willow canopy. The direction of the wind was used to locate the area of snow, organic debris and aeolian material accumulation based on the observation that willows trap materials on one side of the patch that is shielded from wind by the canopy.

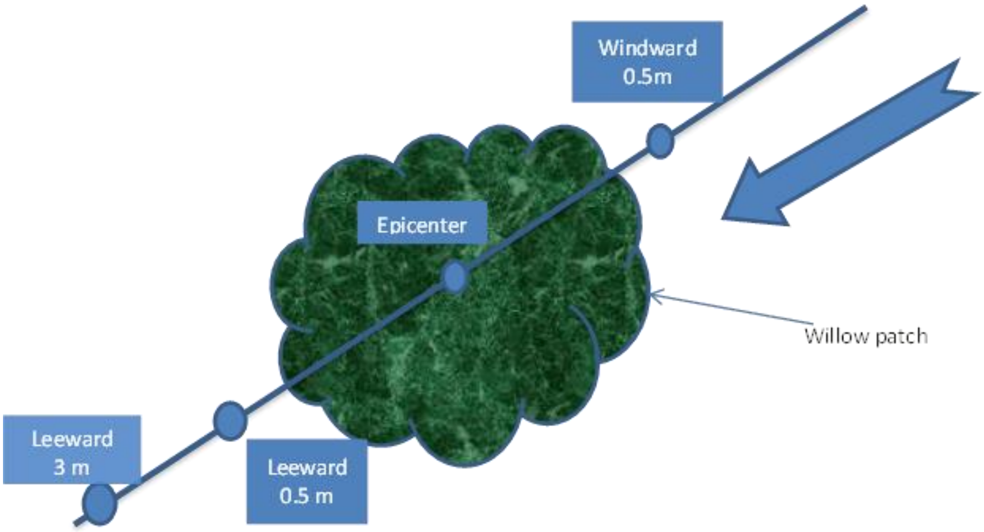

Figure 3. Willow patch canopy showing location of sampling points.

2.2.2 Soil sampling procedure

Samples were taken by a soil auger (1.5” diameter) to a depth of 10 cm (Fig. 4). This was considered the most active part of the soil profile necessary for seedling recruitment. In addition, a control sample was collected from within the same site but in an area without willows. A total of 35 samples were collected.

Figure 4. Soil sample taken 0.5 m from willow patch (left). Soil taken at 10 m depth by a soil auger (right). (Photos: K. Balikoowa, 29 June 2014).

2.3 Soil analysis

Each sample was divided into two parts; one part for dry analysis of phosphorus (P), nitrogen (N), pH and total carbon (C) and the other part for water holding capacity measurement. The portion for dry analysis was dried at 35° C and sieved through a 2 mm sieve. A ball milled sample was used for total C and N analysis.

Total C and N were measured by a dry combustion method in a Vario MAX C/N–Macro Elementary Analyzer. The C, N and P were adjusted for dry matter (Blakemore et al. 1987) for comparison across different samples.

Soil pH was measured with an Oakton pH meter in a 1:2.5 soil to water ratio after shaking for 2 hours. The pH was measured in duplicates of each sample.

Water holding capacity was determined on wet samples using the pressure-plate extraction method (Klute 1986). Three pressures of 0.33 bar, 1 bar and 15 bars were used. Ceramic plates were first saturated in water overnight. For each of the pressures, retainer rings were placed onto the appropriate saturated plate and filled with 10 to 15 g of saturated soil. The apparatus was closed airtight and the specified pressures applied. The apparatus was monitored to check if water had ceased being emitted from the outflow tube. When water had stopped being emitted, the samples were weighed, then placed in an oven to dry at 105°C for 24 hours and reweighed. Calculations for moisture content followed Blakemore et al. (1987). The difference between water holding capacity at field capacity (0.33 bar) and water holding capacity at permanent wilting point (15 bars) was used to calculate the plant available water.

2.4 Vegetation assessment

At the same site used for soil sampling, the effect of willows on plant composition was assessed using a methodology adapted from rangeland ecosystem assessment by Herrick et al. (2005). Two areas were purposively chosen, one with willow patches and a control (Fig. 5). At each of these areas ten transects were laid for gap intercept data collection. The percentage covered by gaps between plant canopies was estimated.

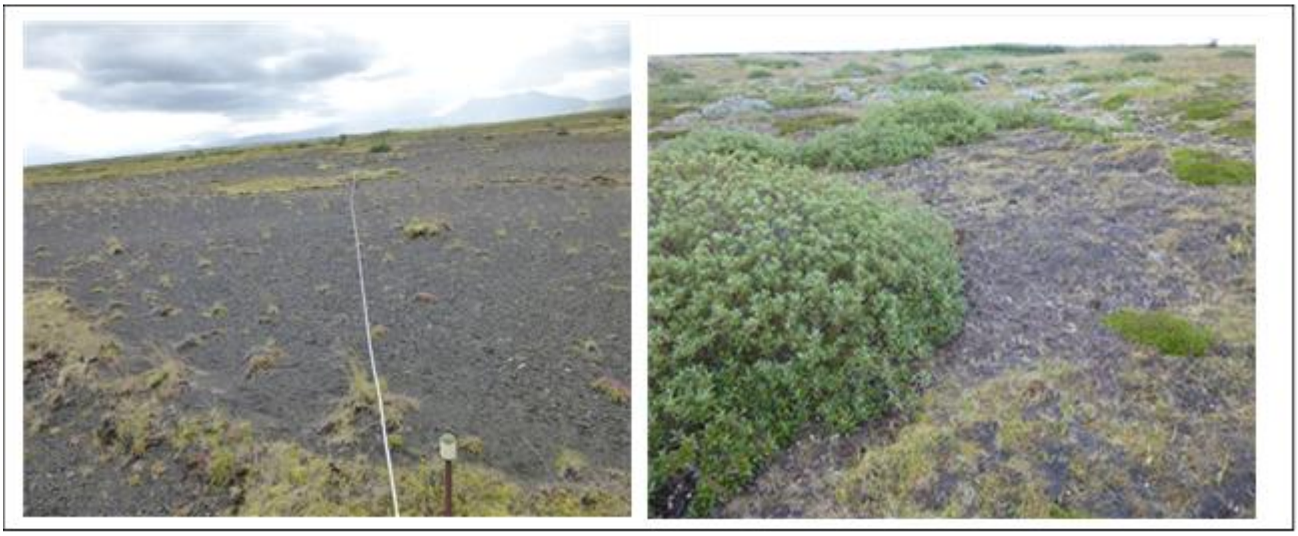

Figure 5. Transect set for gap intercept at the control site (left). Site covered with willow (right). (Photos: K. Balikoowa, 23 July 2014).

Gap intercept was used to assess the risk of the sites to wind erosion based on the size distribution of canopy gaps at the site (Herrick et al. 2005). Basal gap analysis was used to assess the risk of the two sites to frost action in winter.

2.5 Data analysis

Pearson's correlation was used to analyse the relationship between the patch size and measured attributes. Differences in soil attributes between soil samples were determined by ANOVA and Tukey's test for mean comparison. To show levels of significance between large and small patches, a t-test was performed by the Pooled method or Satterthwaite, depending on whether the test for equality of variances was significant. For phosphorus analysis, the Kruskal-Wallis test was used because of the non-homogeneity of variances. When p-values were lower than 0.05, differences were considered significant.

Gap intercept was used to calculate the proportion of the sites covered by the various surface types. All data analyses were done using the SAS Enterprise Guide 6.1 (SAS Institute 2012).

3. RESULTS

3.1 Effect of patch size on soil attributes

3.1.1 Total C, N, CN ratio and soil pH

The results for total C, N, CN ratio, pH and available soil moisture are shown in Table 1. Pearson correlation showed that there was a significantly positive relationship between patch diameter and total C ($r = 0.8321$; $p = 0.0028$) and N ($r = 0.8765$ $p = 0.0009$). The relationship between patch diameter was significantly negative for the CN ratio ($r = -0.6701$; $p = 0.0340$) and pH ($r = -0.9103$, $p < 0.0003$).

Table 1. Mean values and standard error (mean ± standard error) of measured soil attributes for soil collected from willow patches of different size categories and the control. Comparisons were conducted by analysis of variance (ANOVA), and were significant when $p < 0.05$.

Variable	Unit	Control (n = 2)	Small willow (n = 4)	Large willow (n = 4)	F	p value
Nitrogen	%	0.03 ± 0.00	0.05 ± 0.00	0.06 ± 0.00	23.91	0.0007
Carbon	%	0.34 ± 0.02	0.62 ± 0.05	0.75 ± 0.02	25.32	0.0006
CN ratio		13.02 ± 0.88	13.01 ± 0.27	12.10 ± 0.11	2.74	0.1325
pH (H_2O)		7.02 ± 0.00	6.98 ± 0.04	6.72 ± 0.04	17.30	0.0020
Plant available water*	%	7.95 ± 0.55	10.25 ±0.76	11.10 ± 0.56	4.16	0.0646

*Plant available water was calculated from the difference between water content at 15 bars and water content at 0.33 bar.

Analysis of variance showed that there is a significant difference among the soil samples with respect to N and total C. Levels were highest for large canopy patches and smallest for the control. CN ratio was highest in the control followed by the small patch samples and lowest in the large patch samples. The t-test showed that there was a significant difference between soils at large and small patches in level of C ($t = 2.53$, $p = 0.0447$) and N ($t = 3.03$, $p =$

0.0231). Also, Tukey's test showed that total C levels for large and small patches of soil were significantly different from the control at α = 0.05. The level of N for large patches was significantly different from the control but did not vary from the small patches. The CN ratio was only significantly different between soil from large and small patches (t = -3.13, p = 0.0205), there was no significant difference between the control and large or control and small patches.

Tukey's test showed a significant difference in pH between the large and control and the large and small patches (t = -4.72, p = 0.0033). However, there was no significant difference between the small patches and the control.

3.1.2 Phosphorus

The amount of P significantly increased with patch diameter (r = 0.6914; p = 0.0268). The Kruskal-Wallis test showed that there were significant differences in levels of phosphorus (X^2 = 5.7273, p = 0.0343) among the samples. Large patches had the highest levels of P (85.37 $mgkg^{-1}$) followed by the control (17.74 $mgkg^{-1}$) and lastly small patches (3.92 $mgkg^{-1}$) (Fig. 6).

However, there was no significant difference between the levels of P in the small and large patches due to large variation.

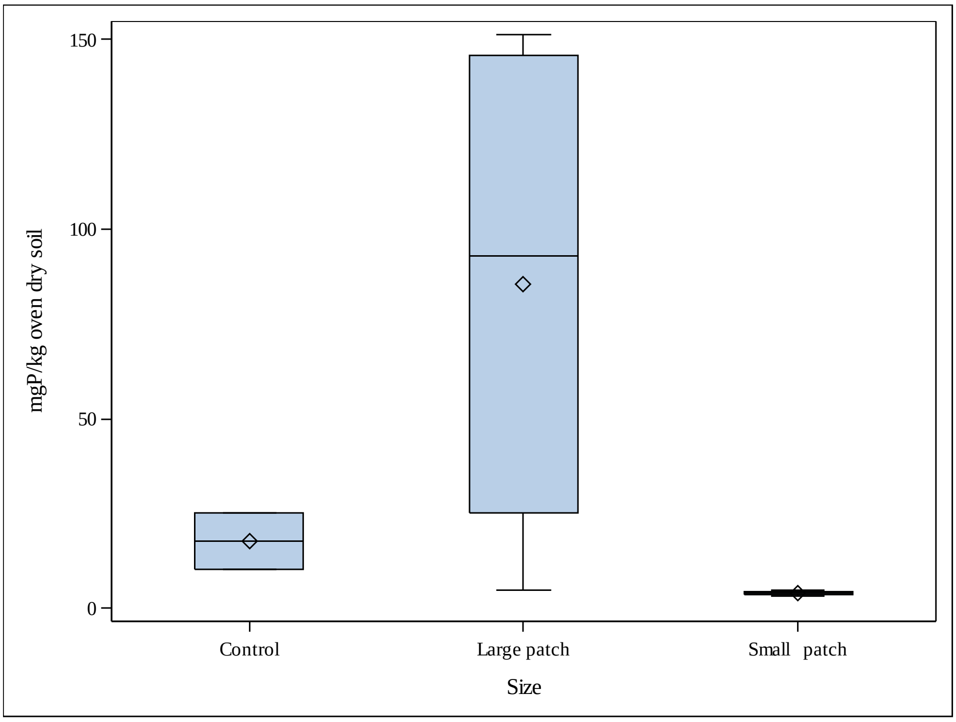

Figure 6. Box plot of P for large (n = 4) and small patches (n = 4) and the control. Kruskal-Wallis test showed significant (X^2 =5.7273, p = 0.0343) difference in P levels at α = 0.05. The top and bottom whiskers show the maximum and minimum values without outliers, respectively. The line shows the mean of the distribution, the diamond shows the median and the box shows the boundaries of the upper and lower quartile.

Levels of phosphorus at 0.5 m from the patch canopy were the same as those at the epicentre (t = -0.05, p = 0.0959).

3.1.3 Water holding capacity and plant available water

Pearson's correlation showed that water holding capacity at 0.33 bar, 15 bars and the available soil moisture increased with size of the patch although the increase was not statistically significant. Analysis of variance showed that water holding capacity at 0.33 bar varied among samples (F = 5.99, p = 0.0372) with the large patches registering the highest followed by the small patches and lastly the control. Tukey's test showed that at 0.33 bar the big patch samples were significantly higher than the control, but did not differ from the small patch sample. No significant difference was shown at 15 bars (F = 4.71, p = 0.0590) among the samples, but there was a general trend of increase with patch diameter (Fig. 7) with respect to water content at different pressures. The t-test did not reveal any differences in water holding capacity at the different pressures between big and small patch soils. The plant available water was not significant among the soil samples (Table 1).

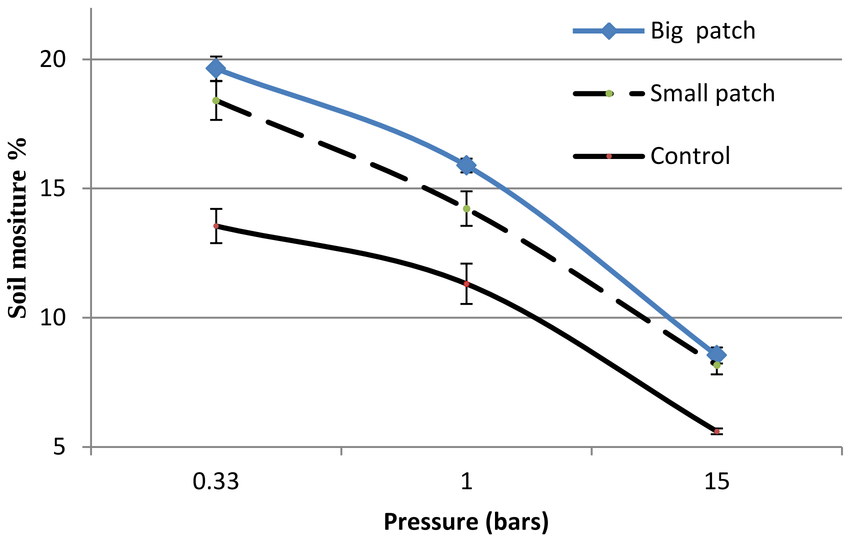

Figure 7**.** Water holding capacity at different soil pressures in patches of different sizes and average of the control samples. Error bars show standard error of measured moisture content.

3.1.4 Changes in measured attributed along the direction of the wind

The levels of total C, N, CN, pH, available water and water holding capacity at 0.33 bars did not vary at different positions under and near the patches.

3.2 Vegetation analysis

Vegetation analysis showed differences in the proportion of different types of surface cover measured by the basal gap intercept over a transect of total length 250 m for each site (Fig. 8). Heath was the most common vegetation type (32.3%) followed by tree/shrub (21.55%), then moss (20.39%) and lastly rocks covering less than 0.1% of the transect length at the willow site. The control site was mostly covered by rocks (58.78%), followed by heath (7.42%). Trees/shrubs were missing at the control site while grass/herbs covered only 0.27% of the transect.

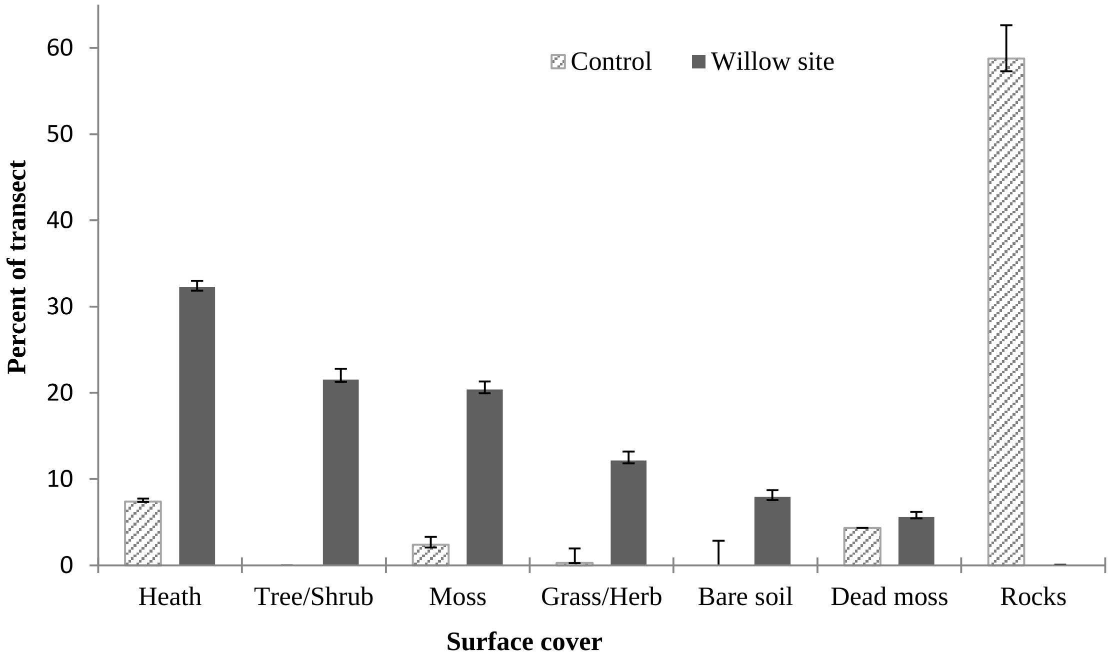

Figure 8. Percentage of different surface cover types for transects laid at willow site and control. Error bars show standard error of percentage cover for each surface cover type.

Vegetation data showed that there were differences in the size and distribution of canopy gaps (Fig. 9). The willow covered site had only 5% of the line covered by canopy gaps larger than 1 m while the control had 45% of the line covered by large canopy gaps. Also the willow covered site had a much higher portion of the transect covered by canopy and/small canopy gaps.

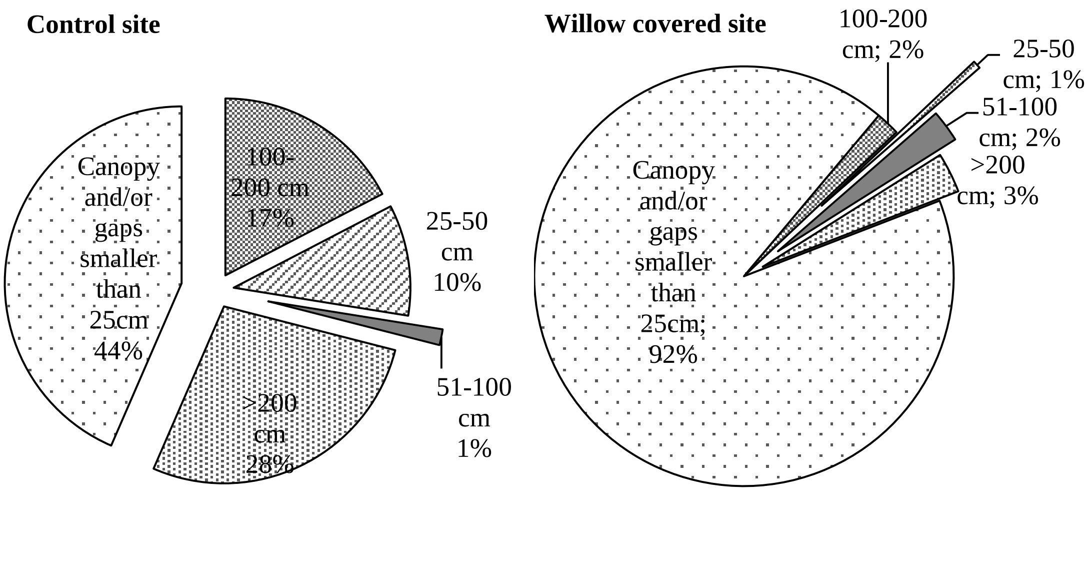

Figure 9. Size distribution of canopy and gaps at willow covered area and the control.

4. DISCUSSION

4.1 Effect of willow patches on nutrient availability and selected soil attributes

The levels of soil C for all samples were low (cf. Arnalds 2004) but even much lower than would be expected from well weathered Andosols (Óskarsson et al. 2004; Arnalds 2008); probably the soils at the study site are still young. Low total C and N were probably due to the high aeolian input and tephra deposition (Arnalds 2004). High pH values indicated soils dominated by allophane (Nanzyo et al. 1993; Arnalds 2004). The high pH could also be attributed to the soils being young and the presence of fresh parent material causing recharge of basic cations during weathering (Arnalds 2004). The water holding capacity was much lower than for typical well weathered Andosols because the soils were still young and not fully weathered.

Soils around willow patches had more improved soil properties than the control and the results show that the effect extended as far as 3 m from the canopy edge even for the relatively small basal canopies used in this study. This was also shown by Jumpponen et al. (1998). Total carbon and nitrogen were higher for large than for small patches due to increased primary production (cf. Ludwig et al. 2004). This suggests that in the larger patches, nutrient enrichment was more intensified (Kondo et al. 2012) in and around the canopy. Patches are expected to grow in canopy size with time, and if smaller patches are assumed to be younger than larger patches, it indicates that the effect of willows on a site will increase with time. The high total C and N content may be the result of organic matter deposition from litter fall (Hook et al. 1991; Scholes & Archer 1997), since 10 cm depth top soil was used (Williams 1980). It is also presumed that the large patches are older than the small patches; if this is so, then the difference in C and N could be an age effect due to accumulation of carbon over a longer time.

The soil acidity differed between large and small patches, probably due to differences in the soil organic matter in the top soil. Several studies have shown that soil organic matter affects soil reactions (Pocknee & Sumner 1997; Tang & Yu 1999; Xu et al. 2006) and this study showed that willow litter significantly lowered soil pH. The lack of variation between small patches and the control could have been due to the low level of organic matter added into the soil under the small patches because amelioration of pH depends on the amount of plant residues (Wang et al. 2013) added, in this case on the amount of plant litter. Under small patches, there may not have been enough soil organic matter to significantly change the soil pH.

The study indicated that the presence of willows creates beneficial microsites. This characteristic of willows can be used in restoration programs in Iceland since the success of colonization and emergence of seedlings is affected by the immediate surroundings of the seedling (Grubb 1977; Elmarsdottir et al. 2003), usually to the nearest centimetre. Multiplication of willow-created microsites can be used to create favourable conditions for other plants to germinate and survive (cf. Tongway & Ludwig 1996).

While few chemical properties were included in this study, the results point to willows contributing to nutrient availability due to biological processes created by roots or litter supply (Garner & Steinberger 1989) as opposed to physical processes due to capture of organic materials and snow. Indeed the leeward side and the windward side were not significantly different from each other for the same patch for most of the measured variables.

Particle and water capture are expected to be important drivers for microsite creation and properties, especially in water stressed areas.

Since patch samples were collected within 3 m of the patch basal area, the high water holding capacity was likely due to higher total C content near the canopy. Areas with dense cover of willows would be expected to have similar water holding properties. The available water content did not vary significantly among samples because all samples were collected within a small area with similar soil properties. Additionally, the difference in total C content between samples was not large enough to significantly affect the water holding capacity of the soil though it is known that soil C can influence the water holding properties of the soil (cf. Hudson 1994). Patch size did not significantly increase water holding capacity at different pressures and plant available water. The sample size could have been too small but also all the soils were sandy and expected to have the same water holding properties.

With regard to the spatial arrangement of the sampling points, within and around the patches, the water holding capacity at 15 bars was significantly higher on the leeward side than at any other positions. This could have been because this area captures organic debris which could slightly change the soil properties. Also on the leeward side there was more dieback of moss, thus increasing the amount of organic matter which could have improved the water holding capacity.

Phosphorus was higher under the larger patches probably due to increased biomass production. Willows take up phosphorus and translocate it to the leaves and later release it as litter (Ens et al. 2013). Larger patches are expected to have a denser root network and because willows have been found to be associated with mycorrhiza, this could have contributed to the high levels of P in the larger patches, as the mycorrhiza acquire P from the soil. However, the study could not explain the big variation in P levels under the large patches.

4.2 Effect of willows on plant composition

The high proportion of rocks on the control site was probably a result of frost heaving which moves coarse rock particles to the surface in areas without vegetation (Brady & Weil 1999). Higher cover by plants in the willow site could have been due to the improved nutrient availability caused by the presence of willows. Availability of nutrients can be a big limiting factor for plant populations (Crawley & Ross 1990; Eriksson & Ehrlén 1992), but since willows cause "pockets of fertility" the better conditions could benefit other plants. The presence of microsites may create better conditions for seedling growth and survival (Fowler 1988) and was probably the reason for better plant cover in the willow site.

In addition, higher plant cover possibly protected the site from further frost heaving, creating conditions conducive for seedling emergence and survival. Willows could also be acting as "nurse plants" to other vegetation (Dona & Galen 2007) or even a trophic mutualist (Galatowitsch 2012) because willows have been found to be associated with ectomycorrhiza and abscular mycorrhiza (Dhillion 1994; Parádi & Baar 2006).

The willow covered site was probably more resistant to wind erosion due to the size distribution of canopy and canopy gaps because according to Herrick et al. (2005) in the Western United States, disturbed soil in gaps 1-2 m in diameter is as susceptible to wind erosion as in gaps with no vegetation. Since wind erosion is a concern in Iceland due to the low density soils, having more willows could improve site resistance to wind erosion.

5. CONCLUSION

In conclusion, the study showed that the presence of willows improves soil quality, as seen in the higher values of total C, N, and P; the effect has been greater with time because it is a biological process due to biomass production and the action of mycorrhiza. The presence of willows also ameliorated soil pH which could benefit certain types of plants that thrive in low pH soils.

Improved soil quality under and around willow patches can be used to kick-start land restoration by multiplying the number of willow patches in degraded sites.

The presence of willows ultimately improves foliar cover by nurturing other plants. The increased plant cover leads to improved site stability to soil erosion by both water and wind. Increased foliar cover is not only important for soil erosion control but could also directly bring degraded land into production if palatable species germinate and establish.

It is recommended that more research be done to determine the spatial extent of the individual microsites in order to determine the optimum willow configurations for land restoration.

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REFERENCES

Ágústsdóttir, A. M. 2004. Revegetation of eroded land and possibilities of carbon sequestration in Iceland. Nutrient Cycling in Agroecosystems **70:**241-247.

Aradottir, A. and O. Arnalds. 2001. Ecosystem degradation and restoration of birch woodlands in Iceland. In: F.E. Wielgolaski (ed.) Nordic Mountain Birch Ecosystems, pages 293-308. Man and Biosphere Series Vol. 27. UNESCO, Paris and Parthenon Publishing, Carnforth.

Aradottir, A., E. v. Santen, and G. Hill. 2004. Does the nootka lupin facilitate or impede colonization and growth of native birch in Iceland? Pages 184-190. Wild and cultivated lupins from the Tropics to the Poles. Proceedings of the 10th International Lupin Conference, Laugarvatn, Iceland, 19-24 June 2002. International Lupin Association.

Aradóttir, A., K. Svavarsdóttir, T. H. Jonsson, and G. Gudbergsson. 2000. Carbon accumulation in vegetation and soils by reclamation of degraded areas. Icelandic Agricultural Sciences **13:**99-113.

Aradottir, A. L. 2007. Restoration of birch and willow woodland on eroded areas. Pages 67. Effects of afforestation on ecosystems, landscape and rural development.

Arnalds, A. 1987. Ecosystem disturbance in Iceland. Arctic and Alpine Research **19:**508-513.

Arnalds, A., and J. Thorsson. 2012. Soil erosion and landcare in Iceland. Newsletter. European Society for Soil Conservation.

Arnalds, O. 2004. Volcanic soils of Iceland. Catena **56:**3-20.

Arnalds, O. 2008. Adosols. Pages 39-46 in W. Chesworth, editor. Encyclopedia of Soil Science. Springer, Netherlands.

Arnalds, O. 2013. The Influence of Volcanic Tephra (Ash) on Ecosystems. Pages 331 in D. L. Sparks, editor. Advances in Agronomy. Academic Press, San Diego, CA, USA.

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

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

Arnalds, Ó., E. F. Thorarinsdottir, S. Metusalemsson, Á. Jónsson, E. Gretarsson, and A. Arnason. 2001b. Soil erosion in Iceland. Agricultural Research Institute, Soil Conservation Service.

Bai, Z. G., D. L. Dent, L. Olsson, and M. E. Schaepman. 2008. Proxy global assessment of land degradation. Soil Use and Management **24:**223-234.

Banadda, N. 2010. Gaps, barriers and bottlenecks to sustainable land management (SLM) adoption in Uganda. African Journal of Agricultural Research **5:**3571-3580.

Blakemore, L. C., P. L. Searle, and B. K. Daly. 1987. Methods for chemical analysis of soils. NZ Soil Bureau, Department of Scientific and Industrial Research, Lower Hutt.

Bradshaw, A. 1987. The reclamation of derelict land and the ecology of ecosystems. Pages 54-74 in R. W. Jordan, E. M. Gilpin and D. J. Aber, editors. Restoration ecology: a synthetic approach to ecological research.

Brady, N., and R. Weil. 1999. The nature and properties of soil, 12th ed. Prentice-Hall Inc. Upper Saddle River, New Jersey.

Crawley, M. J., and G. Ross. 1990. The population dynamics of plants [and discussion]. Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences **330:**125-140.

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

Dhillion, S. S. 1994. Ectomycorrhizae, arbuscular mycorrhizae, and Rhizoctonia sp. of alpine and boreal Salix spp. in Norway. Arctic and Alpine Research **26:**304-307.

Dona, A., and C. Galen. 2007. Nurse Effects of Alpine Willows (Salix) Enhance Over-winter Survival at the Upper Range Limit of Fireweed, Chamerion angustifolium. Arctic, Antarctic, and Alpine Research **39:**57-64.

Elmarsdottir, A., A. L. Aradottir, and M. Trlica. 2003. Microsite availability and establishment of native species on degraded and reclaimed sites. Journal of Applied Ecology **40:**815-823.

Ens, J., R. E. Farrell, and N. Bélanger. 2013. Early effects of afforestation with willow (Salix purpurea,“Hotel”) on soil carbon and nutrient availability. Forests **4:**137-154.

Eriksson, O., and J. Ehrlén. 1992. Seed and microsite limitation of recruitment in plant populations. Oecologia **91:**360-364.

Fowler, N. L. 1988. What is a safe site?: neighbor, litter, germination date, and patch effects. Ecology 69**:**947-961.

Galatowitsch, S. M. 2012. Ecological restoration. Sinauer Associates, Sunderland. Massachusetts.

Garner, W., and Y. Steinberger. 1989. A proposed mechanism for the formation of fertile islands in the desert ecosystem. Journal of Arid Environments **16:**257-262.

Gísladóttir, G., E. Erlendsson, and R. Lal. 2011. Soil evidence for historical human-induced land degradation in West Iceland. Applied Geochemistry **26:**S28-S31.

Gísladóttir, G., E. Erlendsson, R. Lal, and J. Bigham. 2010. Erosional effects on terrestrial resources over the last millennium in Reykjanes, southwest Iceland. Quaternary Research **73:**20-32.

Gisladottir, G., and M. Stocking. 2005. Land degradation control and its global environmental benefits. Land Degradation & Development **16:**99-112.

Greipsson, S. 2012. Catastrophic soil erosion in Iceland: Impact of long-term climate change, compounded natural disturbances and human driven land-use changes. Catena **98:**41-54.

Gretarsdottir, J., A. L. Aradottir, V. Vandvik, E. Heegaard, and H. J. B. Birks. 2004. Long-Term Effects of Reclamation Treatments on Plant Succession in Iceland. Restoration Ecology **12:**268-278.

Grubb, P. J. 1977. The maintenance of species-richness in plant communities: the importance of the regeneration niche. Biological Reviews **52:**107-145.

Hagen, D., K. Svavarsdottir, C. Nilsson, A. K. Tolvanen, K. Raulund-Rasmussen, À. L. Aradòttir, A. M. Fosaa, and G. Halldorsson. 2013. Ecological and social dimensions of ecosystem restoration in the Nordic countries. Ecology and Society **18:**34.

Halldorsson, G., G. Halldórsson, Á. L. Aradóttir, A. M. Fosaa, D. Hagen, C. Nilsson, K. Raulund-Rasmussen, A. B. Skrindo, K. Svavarsdóttir, and A. Tolvanen. 2012. ReNo: Restoration of Damaged Ecosystems in the Nordic Countries. Nordic Council of Ministers.

Herrick, J. E., J. W. Van Zee, K. M. Havstad, L. M. Burkett, and W. G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. Volume I: Quick Start. Volume II: Design, supplementary methods and interpretation. USDA-ARS Jornada Experimental Range.

Hobbs, R. J., L. M. Hallett, P. R. Ehrlich, and H. A. Mooney. 2011. Intervention ecology: applying ecological science in the twenty-first century. BioScience **61:**442-450.

Holl, D. K., L. Pejchar, and G. S. Whisenant. 2007. Overcoming Physical and Biological Obstacles to Restore Natural Capital. Pages 249-255 in J. Aronson, J. S. Milton and N. J. Blignaut, editors. Restoring natural capital. Science, business, and practice. Island Press, Washington, DC, USA.

Hook, P. B., I. C. Burke, and W. K. Lauenroth. 1991. Heterogeneity of soil and plant N and C associated with individual plants and openings in North American shortgrass steppe. Plant and Soil **138:**247-256.

Hudson, B. D. 1994. Soil organic matter and available water capacity. Journal of Soil and Water Conservation **49:**189-194.

Hunziker, M. 2011. A study on above-and belowground biomass and carbon stocks as well as sequestration of mountain birch (Betula pubescens Ehrh.) along a chronosequence in southern Iceland. Institut für Geographie.

Icelandic Met Office. 2014. Meðalvindhraði hverrar vindáttar, Hella, www.vedur.is.

Jones, C. C., and R. Moral. 2005. Effects of microsite conditions on seedling establishment on the foreland of Coleman Glacier, Washington. Journal of Vegetation Science **16:**293-300.

Jumpponen, A., K. Mattson, J. Trappe, and R. Ohtonen. 1998. Effects of established willows on primary succession on Lyman Glacier forefront, North Cascade Range, Washington, USA: evidence for simultaneous canopy inhibition and soil facilitation. Arctic and Alpine Research **30:**31-39.

Klute, A. 1986. Water Retention: Laboratory Methods. Pages 635-662 in A. Klute, editor. Methods of Soil Analysis: Part 1—Physical and Mineralogical Methods. Soil Science Society of America, American Society of Agronomy.

Kondo, J., M. Hirobe, Y. Yamada, J. Undarmaa, K. Sakamoto, and K. Yoshikawa. 2012. Effects of Caragana microphylla patch and its canopy size on “islands of fertility” in a Mongolian grassland ecosystem. Landscape and Ecological Engineering **8:**1-8.

Ludwig, F., H. de Kroon, F. Berendse, and H. H. Prins. 2004. The influence of savanna trees on nutrient, water and light availability and the understorey vegetation. Plant Ecology **170:**93-105.

Maestre, F. T., and J. Cortina. 2004. Insights into Ecosystem Composition and Function in a Sequence of Degraded Semiarid Steppes. Restoration Ecology **12:**494-502.

Magnússon, B. 2010. NOBANIS–Invasive Alien Species Fact Sheet–Lupinus nootkatensis. Pages 11. North European and Baltic Network on Invasive Alien Species NOBANIS.

Mcghee, W., J. Craig, R. De Groot, J. S. Miller, and K. Bowers. 2007. Making restoration work: non-monetary mechanisms. Pages 294-302 in J. Aronson, J. S. Milton and N. J. Blignaut, editors. Restoring natural capital. Science, business, and practice. Island Press, Washington, DC, USA.

Nanzyo, M., R. Dahlgren, and S. Shoji. 1993. Chemical Characteristics of Volcanic Ash Soils. Pages 145-187 in M. N. Sadao Shoji and D. Randy, editors. Developments in Soil Science. Elsevier.

National Environmental Management Authority. 2010. State of the Environment Report for Uganda 2010. National Environment Management Authority (editor,) Kampala.

Oldeman, L., R. Hakkeling, W. Sombroek, and A. Ayoub. 1991. World Map of the Status of Human-Induced Soil Degradation: A Brief Explanatory Note. International Soil Reference and Information Centre [and] United Nations Environment Programme.

Olson, J., and L. Berry. 2003. Land degradation in Uganda: its extent and impact. available at lada. virtualcentre. org/eims/download. asp.

Óskarsson, H., Ó. Arnalds, J. Gudmundsson, and G. Gudbergsson. 2004. Organic carbon in Icelandic Andosols: geographical variation and impact of erosion. Catena **56:**225-238.

Parádi, I., and J. Baar. 2006. Mycorrhizal fungal diversity in willow forests of different age along the river Waal, The Netherlands. Forest Ecology and Management **237:**366-372.

Pocknee, S., and M. E. Sumner. 1997. Cation and nitrogen contents of organic matter determine its soil liming potential. Soil Science Society of America Journal **61:**86-92.

Rapport, D., R. Costanza, and A. McMichael. 1998. Assessing ecosystem health. Trends in Ecology & Evolution **13:**397-402.

Rees, E. W., J. Farley, É.-T. Vesely, and R. De Groot. 2007. Valuing natural capital and the costs and benefits of restoration. Pages 227-236 in J. Aronson, J. S. Milton and N. J. Blignaut, editors. Restoring natural capital. Science, Business, and Practice. Island Press, Washington, DC, USA.

Ritter, E. 2009. The Role of Afforestation in Soil Restoration: effects on soil nutrients in degraded volcanic soils on Iceland. 6th International Symposium on Ecosystem Behaviour BIOGEOMON 2009.

Runólfsson, S. 1987. Land reclamation in Iceland. Arctic and Alpine Research **19:**514-517.

SAS Institute. 2012. SAS Visual Analytics 6. 1: User's Guide. Sas Institute.

Scholes, R., and S. Archer. 1997. Tree-grass interactions in savannas. Annual review of Ecology and Systematics **28:**517-544.

SER. 2004. The SER International primer on ecological restoration in A. Clewell, J. Aronson and K. Winterhalder, editors.

Sohlberg, E. H., and L. C. Bliss. 1984. Microscale pattern of vascular plant distribution in two high arctic plant communities. Canadian Journal of Botany **62:**2033-2042.

Sserunkuuma, D., J. Pender, and E. Nkonya. 2001. Land management in Uganda: Characterization of problems and hypotheses about causes and strategies for improvement. Project on Improved Land Management in Uganda, International Food Policy Research Institute, Washington, DC.

Tang, C., and Q. Yu. 1999. Impact of chemical composition of legume residues and initial soil pH on pH change of a soil after residue incorporation. Plant and Soil **215:**29-38.

Titus, J. H., and R. del Moral. 1998. Seedling establishment in different microsites on Mount St. Helens, Washington, USA. Plant ecology **134:**13-26.

Tongway, D. J., and J. A. Ludwig. 1996. Rehabilitation of semiarid landscapes in Australia. I. Restoring productive soil patches. Restoration Ecology **4:**388-397.

Tsuyuzaki, S., J. H. Titus, and R. Moral. 1997. Seedling establishment patterns on the pumice plain, Mount St. Helens, Washington. Journal of Vegetation Science **8:**727-734.

Wang, Y., C. Tang, J. Wu, X. Liu, and J. Xu. 2013. Impact of organic matter addition on pH change of paddy soils. Journal of Soils and Sediments **13:**12-23.

Whisenant, S. G. 1999. Repairing damaged wildlands: a process-orientated, landscape-scale approach. Cambridge University Press, Cambridge, Kingdom.

Williams, C. 1980. Soil acidification under clover pasture. Animal Production Science **20:**561-567.

Xu, J., C. Tang, and Z. Chen. 2006. The role of plant residues in pH change of acid soils differing in initial pH. Soil Biology and Biochemistry **38:**709-719.