Certification

Evaluation of the Erosion Reduction Potential, Stability, and Resilience of Living Shorelines in Nova Scotia

By Kirsten Ellis

A Thesis Submitted to Saint Mary's University, Halifax, Nova Scotia in Partial Fulfillment of the Requirements for the Degree of Master of Science in Applied Science.

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Abstract

Helping Nature Heal Inc. has developed plant-based methods for coastal erosion management that can be classified as Living Shorelines. These Living Shorelines have been used in Nova Scotia for over a decade and appear to be effective for reducing erosion and increasing slope stability. To quantitatively assess erosion reduction potential of the Living Shoreline techniques, a plot-based experiment tested their effect on runoff and sediment loss from rainfall and experimental watering. Treatments that provided high amounts of soil cover showed significant reductions in sediment loss, showing that these techniques are useful for managing rainfall and overland flow erosion on steep glacial till slopes. Four Living Shorelines at various stages of maturity and representing three different coastal geomorphologies were assessed for indications of stability and resilience. At all but the high-energy cliff site, the Living Shorelines showed general indications of stability and possible resilience in the Nova Scotian coastal environment.

March 27, 2017

Dedication

This work is dedicated to my mom, Angela Busche, who sadly never got to see this project completed but has been a continuous source of love and inspiration.

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Chapter 1: Introduction

Thesis Context and Focus

Coastal erosion management is an important consideration in Nova Scotia,

Canada due to the large amount of coastline and the high percentage of the coastline that
is sensitive to changing coastal conditions such as sea level rise that can exacerbate
erosion issues (Shaw et al. 1995, Himmelstoss et al. 2006). Most of Nova Scotia's land is
privately owned and therefore the general trends in coastal erosion management are
largely reliant on homeowner decisions at the scale of private properties (Manuel et al.
2015). Living Shorelines are one method of coastal erosion management that are gaining
recognition in many places including Nova Scotia (van Proosdij et al. 2016). Helping
Nature Heal Inc., an ecological landscaping company based in Nova Scotia, has
developed their own unique Living Shoreline design and methodology and has been
designing and constructing Living Shorelines for private homeowners in Nova Scotia and
Prince Edward Island for over a decade.

The purpose of this research project was to attempt to quantify the effectiveness of the Living Shoreline techniques developed by Helping Nature Heal (HNH) Inc. A controlled, plot based experiment was designed to test the techniques used by HNH separately and together to determine their effect on sediment loss and runoff due to rainfall, since rainfall and overland flow are some of the key drivers of erosion on Nova Scotia's coastal slopes (personal observation, Brown et al. 2005, Vandamme and Zou 2013). The other component of this research project was a Living Shoreline monitoring program whose purpose was to collect data from existing Living Shorelines built by HNH

that could be used to infer stability and resilience of the Living Shoreline techniques when they are used in a coastal environment.

Soil Erosion

Soil erosion is a natural geoecological process that has been ongoing since the Earth was formed (Boix-Fayos et al. 2006). Erosion is defined by the Oxford Dictionary as, the process of eroding or being eroded by wind, water, or other natural agents (Erosion, 2017). Soil erosion can therefore be defined as the removal of soil from an area by the action of wind, water, or another natural agent. Soil erosion processes are responsible for the formation of many familiar landscape features such as river valleys and coastal bluffs. Depositional landforms are created when sediment eroded from one area is transported by wind or water and deposited in another area forming dunes, beaches, and river deltas for example. Water driven erosion is the primary focus of this study.

Although soil erosion is a natural process it is widely accepted that human activities, especially agriculture and resource extraction, have accelerated the natural rates of erosion resulting in loss of valuable topsoil and nutrients from terrestrial ecosystems (Cerdan et al. 2010). There are many negative impacts of soil erosion; loss of topsoil and associated nutrients reduces crop yields, sediment fills reservoirs increasing maintenance costs, sedimentation in rivers and streams impacts fish habitat, and infilling of bays creates shipping hazards and results in the need for costly dredging (Boardman 2006, Day et al. 2012, Stoffel et al. 2013). Due to the many economic, social, and ecological impacts of accelerating soil erosion it has become an area of concern that has gained more

attention in the scientific, public, and political spheres over the last several decades. This increased awareness has resulted in a large number of studies investigating both the factors controlling soil erosion as well as ways to effectively manage soil erosion.

It is generally recognized that erosion by water and particularly by rainfall and rainfall-generated runoff is a large contributor to global soil erosion (Wei et al. 2009). Water erosion research is based on the theoretical knowledge that interactions between rain drop impacts and runoff facilitate the detachment and transport of soil particles and drive erosion rates (Fox and Bryan 1999). Sloped surfaces emphasize this interaction due to the angle of raindrop impact and the increased velocity of runoff due to slope (Visser et al. 2004). Therefore, in order to effectively manage erosion it is important to reduce the energy of rainfall that reaches the soil surface and the amount and velocity of runoff.

It is generally accepted that vegetation decreases erosion potential through the reduction of rainfall energy by the canopy, reducing runoff velocity through increased surface roughness from stems, roots and litter, and changing soil characteristics (ie. physical, chemical, and biological features) to favour infiltration and decrease runoff amounts (Vásquez-Méndez et al. 2010, Anh et al. 2014, Zhang et al. 2015). It is also thought that the physical structure of the roots helps bind soil together making it more resistant to erosion (Garcia-Estringana et al. 2013). The addition of organic matter to the soil from vegetation has been shown to increase soil cohesion, thereby decreasing its susceptibility to erosion especially in saline soils such as those in coastal areas (Lakhdar et al. 2009).

Coastal Erosion

The erosion of coastal landforms is one aspect of the global erosion issue that has received a lot of attention as sea level rise and increased frequency of intense storms resulting from climate change are causing coastal erosion rates to accelerate in many areas (Himmelstoss et al. 2006, IPCC 2007). Erosion poses a disproportionately large risk in coastal areas because these areas are heavily settled and used by humans (Hsu et al. 2008). In Europe, for example, damage from coastal erosion costs millions of euros every year (Marchand et al. 2011). This high intensity of human pressure in coastal areas can also increase erosion due to inappropriate development and land use. Erosion not only damages human investments and infrastructure but also ecologically important habitats that can result in the reduction of populations of commercially important marine species. Because coastal areas are highly populated, economically valuable, and contain large amounts of infrastructure, the need to protect them from the damage caused by erosion and storm events has been recognized for centuries. The cost of coastal protection in Europe over the last ten years alone is almost 1 billion euros (Marchand et al. 2011).

Historically, hard structures such as seawalls, groins, revetments and breakwaters have been the protection method of choice (Davis et al. 2006). However, recently it has been noted that introducing hard structures into the dynamic coastal system exacerbates erosion issues in many areas (Hsu et al. 2008). The negative on site impacts of hard structures such as steepening or lost beaches, loss of shallow water habitat, decreased water quality, and reduced ecosystem functioning, are becoming more apparent as the amount of shoreline hardening increases (Smith 2006). Hardened shorelines can also have negative off site effects by deflecting water energy and creating erosion issues on

neighbouring properties whose owners then armour their shoreline, which further exacerbates the problem of erosion (Davis et al. 2006). Current practices of shoreline hardening also cut off sediment sources and prevent the natural movement of sediments along the shoreline, both of which are necessary to maintain coastal features like beaches and salt marshes (Preface 2006, van Rijn 2011). Another negative effect of shoreline hardening is the phenomena called 'coastal squeeze', which occurs when human development restricts the natural landward movement of shorelines in response to sea level rise, resulting in the loss of coastal species and critical habitats as well as putting human life and infrastructure at risk (Swann 2008). In addition to emerging information on the negative environmental and social impacts of hard structures, they are also costly to build and maintain and they are unable to adapt to the environmental challenges we are currently facing, particularly sea level rise. This sentiment echoes Borsje et al. [2011] who recognized

"...a strong need for innovative, sustainable and cost-effective coastal protection solutions that deal with threats related to climate change, such as accelerating sea level rise."

Coastal Management and Living Shorelines

A shoreline can be defined as the line along which a large body of water meets the land (Shoreline 2017) and the coast is the part of the land adjoining or near the sea (Coast 2017). Coastal management "involves managing coastal areas to balance environmental, economic, human health and human activities." (NOAA 2015). 'Soft' methods of coastal

management using primarily plants and biological materials have emerged as more environmentally and socially functional, sustainable, and cost effective alternatives to hard structures for preserving coastal assets and reducing erosion risk. One aspect of this type of 'soft' management is the protection and restoration of coastal wetland ecosystems such as salt marshes, mangroves, and sea grass beds, which have been shown to provide cost-effective protection against coastal flooding, storm surge, and the effects of wave action (Gedan et al. 2011). The canopy of submerged aquatic vegetation (SAV) in sea grass beds can also mitigate coastal erosion hazards because it exerts a drag force on the water flowing over it, which results in reduced velocity, turbulence, and increased sediment deposition (Gedan et al. 2011).

Another aspect of 'soft' coastal management is bioengineering of coastal banks and bluffs, which is the use of plants and plant residues (ie. logs, brush, hay) to stabilize coastal slopes. Using plants for coastal bluff bioengineering has been shown to successfully manage bluff erosion because the plant material can attenuate wind and wave energy and bind soil with their roots (Gedan et al. 2011). The concept of using plants and organic materials to protect coastal areas has existed in traditional coastal management for a long time. For example, until recent decades, fishermen in Nova Scotia used to weave brush on the seaward side of wharves and jetties to trap sediment and prevent erosion (personal communication Rosmarie Lohnes 2013). The terms Living Shorelines, soft engineering, and coastal bioengineering have only recently been used to describe techniques that focus on the use of plants and living structures such as oyster reefs to protect coastlines from erosion.

The concept of Living Shorelines was developed about twenty years ago in response to large portions of the United States coastline being armoured to prevent erosion, for example San Francisco Bay was about 75% hardened in 2006 (Davis et al. 2006). As shoreline vegetation is replaced by hard structures, coastal processes such as runoff filtration and sediment trapping that are important to environmental and human health are negatively affected (Davis et al. 2006). The Living Shorelines concept recognizes the importance of natural shoreline processes, appreciates the dynamic and inherent ecological value that natural shorelines provide, and attempts to apply natural principles to coastal management projects in order to enhance habitat and maintain shoreline processes (Smith 2006). Since their conception, Living Shorelines have been developed and adapted to suit a wide range of conditions and their suitability for applications in Nova Scotia is the primary focus of this study.

Existing Erosion Management Knowledge

We know that land use and soil cover have the largest influence on runoff and soil erosion rates of any parameters studied in every environment where they have been investigated (Cerdan et al. 2010, Nunes et al. 2011). As vegetation cover increases, runoff and erosion rates decrease, a phenomenon that has been shown to hold true in most environments (Nunes et al. 2011). Natural vegetation is usually more effective at reducing erosion rates than agricultural or agroforestry plantings (Labrière et al. 2015). Native species forests, shrub lands, and grasslands show high values for the ecosystem service of erosion prevention for both on and off site effects in steeply sloped regions of the Ecuadorean Andes (Balthazar et al. 2015).

High amounts of soil cover can also be accomplished through the use of plant residues or mulch. A variety of mulching techniques have been observed and studied in many environments and have been found to significantly reduce erosion rates (Smets et al. 2008). By providing immediate soil cover mulch reduces raindrop impacts resulting in less dislodgement of soil particles and also provides a rough, absorptive layer that slows down overland flow and absorbs water reducing overall runoff (Smets et al. 2008, Robichaud et al. 2013).

Erosion has been shown to have a negative affect on watercourses and the aquatic biology within them through the transport of nutrients, pollutants, and sediments into water bodies resulting in decreased water quality (Rickson 2014). However, efforts to stop erosion in riparian and coastal environments with hard structures have many negative effects as discussed earlier. Living Shorelines, including coastal wetlands and bioengineered bluffs, are a range of alternatives to shoreline hardening and have been shown to provide the most protection against smaller and more frequent damaging storms but do exhibit protection benefits for larger, catastrophic events (Gedan et al. 2011, Gittman 2014).

Bioengineering is one tool in the Living Shoreline toolkit that is growing in popularity among policy makers, engineers, consultants, and the public as a way to effectively stabilize coastlines and protect coastal assets from erosion while minimizing the negative effects of hard structures. Many coastal plant and animal species can be considered ecosystem engineers because they change the dynamics of the system they live in (Borsje et al. 2011). However, some species such as willow, mangrove, marsh grasses, and oysters have a particularly large effect on biotic and abiotic conditions, for

example wave energy and soil stability, making them ideal for use in bioengineering projects (Borsje et al. 2011).

Erosion Management Knowledge Gaps

There has been much study of the effects of vegetation in erosion prone inland regions including mountains, arid and semi arid climates, and the Mediterranean. There has also been a significant amount of work done in agricultural and agroforestry areas because of their economic importance and disproportionately high erosion rates (Cerdan et al. 2010). However, we are still lacking knowledge such as information on the combined benefits and possible trade-offs of revegetation projects both inland and on the coast, as normally only one benefit is studied at a time (Bryan and Crossman 2008). Models to predict the effectiveness of ecosystem engineering species are not well developed and this results in a lack of certainty about their effectiveness, which can be a barrier to implementation (Borsje et al. 2011).

As far as coastal erosion management is concerned there is a relatively large amount of information on wetlands, salt marshes, and mangroves with less peer reviewed research published on the effects of vegetation in coastal uplands or the effectiveness of coastal bluff bioengineering in regards to their contribution to coastal stability and erosion reduction. The work of David Polster and Polster Environmental Services Ltd. is a notable Canadian example of the successful use of bluff and bank bioengineering techniques (Polster 2006). Living Shoreline design research especially with the consideration of when and where specific techniques are the most effective is also lacking (Bendell et al. 2006). Adaptive management and improvement of Living Shoreline

designs and techniques is impeded because the evaluation of current Living Shoreline projects is insufficient (Currin et al. 2010, Bilkovic et al. 2016).

Objectives

The purpose of this research project is to determine the effectiveness of the coastal bluff bioengineering techniques developed and used by Helping Nature Heal Inc. (HNH) to stabilize sediment and manage erosion in coastal and riparian environments. The three techniques developed by HNH, which are used in combination to produce their proprietary Living Shorelines are; planting native species, brush weaving, and hay mulching. The hypothesis to be tested is that the combination of plants, brush, and hay will reduce runoff and sediment loss when compared to a bare slope. The goal of the primary experiment is quantifying the effect that each of the four treatments (planting native species, brush weaving, hay mulching, and all three techniques in combination) has on erosion rates through comparison with an untreated control. This will provide much needed data on the potential to reduce erosion from rainfall and overland flow on client properties using these techniques. If the techniques show reduced erosion, this will allow HNH to further promote the value of using Living Shorelines in local coastal management and will help strengthen the position of a vegetation-based Living Shoreline approach as a viable alternative to the conventional hard structures currently used to manage erosion in Atlantic Canada.

The broad questions that will be addressed by the primary experiment are:

How do the treatments affect sediment loss?

How do the treatments affect runoff?

Which treatment is most effective at reducing erosion?

Is there a relationship between rainfall amount and sediment loss or runoff?

How does sediment move within the plots?

Monitoring of existing Living Shoreline projects will be used to support the findings of the primary experiment. The Living Shoreline monitoring study will consist of data collection at four different Living Shoreline sites. The information collected from these sites will be used to attempt to answer the question of whether these Living Shorelines are stable and resilient. Stability and resilience will be inferred from measures of erosion, accretion, or no change. Accretion and no change could indicate a stable and/or resilient Living Shoreline in certain environments while erosion could indicate instability depending on the geoecological system.

Another goal of this research project is the creation of a monitoring program that can be used on all current and future HNH Living Shoreline projects to quantify their effectiveness. A robust industry based monitoring program will be especially useful for collecting consistent long-term data on Living Shoreline effectiveness that is badly needed and currently lacking (Bendell et al. 2006). This monitoring program will also help to improve the design, construction, and maintenance of HNH's Living Shoreline projects, ensuring that their clients get the best product possible.

Study Overview

This study employed a field-based experiment to evaluate the effectiveness of four erosion management techniques as well as monitoring existing Living Shorelines that were built using those techniques. Field research measurements were chosen because they take into account in-situ soil and plant composition as well as natural temporal variability in erosion processes (Stroosnijder 2005).

Site Characteristics

The study will include two types of sites, experimental sites and Living Shoreline monitoring sites. All sites are located in Lunenburg County, Nova Scotia, Canada (Figure 1). This region is dominated by a temperate, maritime climate. Average annual rainfall is 1535.7 mm and the mean temperature varies from 19°C in July to -5.3°C in January with an average yearly temperature of 7.5°C (Government of Canada 2015).



Figure 1) Map showing the location of the experimental site (1) and four Living Shoreline monitoring sites (2, 3, 4, 5) in Lunenburg County, Nova Scotia. Map data © 2017 Google.

The experimental site (1) is located in Crousetown, Lunenburg Co. in a recovering forest in the Acadian forest ecozone. The area was clear-cut in 2000, which included the building of a logging road. In some areas the road cut created relatively steep banks, these banks remained free of almost all vegetation as of December 2014 and based on observations, experience active erosion. The primary and secondary experimental plots were set up on these road-cut slopes, which were used as a proxy for coastal bluffs because they share the same soil classification and glacial history (Cann and Hilchey 1958) (Chapter 2 Figure 1).

Controlled Experiment

The primary experiment consisted of fifteen bounded plots with runoff collection devices so that sediment loss under the different treatments could be quantified and evaluated (Figure 2).



Figure 2) Some of the primary experimental plots during construction in July 2015.

The treatments evaluated were; planting native species into the slope, brush weaving, mulching with hay, and a combination of all the three aforementioned techniques that was meant to simulate a Living Shoreline. A list of plant species used can be found in Appendix Table 1. The effectiveness of the treatments was evaluated through comparison with untreated controls. Point scale measurements of soil level within the plots were measured using a profile gauge to determine sediment movement processes within the plots (See Appendix Figure 1). The secondary experiment consisted of twelve open plots

with no runoff collection. These plots were used exclusively in profile gauge analyses and for measuring lag time in the controlled watering experiments (CWEs).

Living Shoreline Monitoring



Figure 3) The four sites used for Living Shoreline monitoring. Two coastal bluff sites (2 and 3), cliff site (4), and dune site (5).

The Living Shoreline monitoring sites were four different coastal sites where Living Shorelines have been installed by Helping Nature Heal and established for more than two years (Figure 3). The sites chosen are in relatively close proximity to each other.

Sites 4 and 5 are located in Kingsburg, Lunenburg Co., Nova Scotia, site 3 is located in Feltzen South, Lunenburg Co., Nova Scotia, and site 2 is in Bayport, Lunenburg Co., Nova Scotia (Figure 1). These sites were used to evaluate the stability and resilience of Living Shorelines in the Nova Scotian coastal environment. The Living Shorelines at the coastal sites were monitored using slope surveys, erosion pins, toe and crest measurements, and vegetation cover estimates. In this context Living Shoreline is the name of the HNH product that is constructed using a specific combination of hay mulching, brush weaving, and planting native species.

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Chapter 2: Experimental Evaluation of Living Shoreline Techniques

Abstract

Helping Nature Heal Inc. uses a combination of hay, brush, and planting native species to reduce erosion on coastal slopes in Nova Scotia. These techniques were tested for their effect on sediment loss and runoff on slopes of similar geology and glacial history as coastal slopes in the area. Sediment loss and runoff data from natural rainfall and controlled watering experiments (CWEs) were collected from closed plots. Sediment loss was significantly reduced in the treatments that provided high amounts of soil cover for both natural rainfall and CWEs. Runoff from natural rainfall was not affected by the treatments but runoff was significantly reduced by the high cover treatments in the CWEs. The combination of treatments used to manage erosion caused by rainfall on coastal slopes reduced sediment loss from natural rainfall by 98% compared to the controls. Reduction in sediment loss was observed for all sediment size classes investigated.

Introduction

It has been well established that soil erosion and its negative effect on soil productivity, terrestrial ecology, and the ecology of receiving water bodies is a major global issue that needs to be addressed, especially in the context of the added pressures of human population growth and climate change (Wei et al. 2009, Cerdan et al. 2010, Bhattarai et al. 2011). Water is the main driver of erosion in most environments with one study estimating that water erosion accounts for as much as 55% of total global soil erosion (Wei et al. 2009). Rainfall amount and intensity are recognized as the most important contributing variables for runoff generation and ultimately erosion and have been studied in a wide variety of environments (Fox and Bryan 1999, Wei et al. 2009).

The effects of rainfall on runoff generation and erosion have primarily been recognized and studied in inland environments. However, a recent study by Nanko et al. [2016] showed that rainfall and runoff also contribute to sediment movement and erosion in coastal environments. Coastal bluffs in Nova Scotia experience erosion caused by a number of rainfall generated phenomena including overland flow, raindrop impact, rill and gully formation, and seepage (personal observation, Brown et al. 2005, Vandamme and Zou 2013). These rainfall-generated phenomena are very similar to those that have been thoroughly studied in inland hillslope environments and are likely subject to similar mitigating factors.

There is abundant evidence that vegetation cover (both living and mulched residues) has a large role to play in mitigating rainfall-generated runoff and soil loss under a wide variety of geologic, land use, slope, and precipitation conditions, including under climate change scenarios (Labrière et al. 2015). It is generally agreed that a large

proportion of bare soil in an area is a strong indicator of high erosion potential (Nunes et al. 2011). The capacity of living plants to protect soil from erosion is primarily related to soil cover from canopy and leaf litter, as well as the soil binding capacity of their roots (Durán-Zuazo et al. 2013). The plant canopy buffers the energy of raindrops that initiate erosion by dislodging soil particles (Durán-Zuazo et al. 2013). Raindrop impacts also cause compaction and surface sealing, which decreases infiltration and increases runoff resulting in increased erosion when no plant canopy is present (Durán-Zuazo et al. 2013).

Plant roots have the ability to change the physical properties of the soil making it less prone to erosion. The presence of plant roots increases soil cohesion by binding sediments and reduces runoff and its ability to transport sediment (Bochet et al. 2000). Plant roots reduce runoff through increased infiltration as a result of decreased soil bulk density and increased soil porosity (Durán-Zuazo et al. 2013). The leaf litter produced by plants reduces erosion by improving soil structure through the addition of organic material, providing soil cover, and improved conditions for microbial/fungal productivity and nutrient cycling (Vásquez-Méndez et al. 2010). Leaf litter also increases the structural variability of the surface thereby reducing runoff velocities and the ability of runoff to transport sediment (Anh et al. 2014).

Species diversity has been identified as one factor that contributes to the effectiveness of plants for mitigating erosion (Durán-Zuazo et al. 2013). Having a diversity of plant species can increase biomass production resulting in increased canopy and litter cover, as well as increased root mass (Durán-Zuazo et al. 2013). Having a diversity of root and shoot forms is likely to increase rainfall interception, reduce runoff velocity, and increase soil stability compared to monocrops (Durán-Zuazo et al. 2013).

This theory is supported by a number of review papers that show land uses that include a diversity of native vegetation and especially mature native forests are consistently proven to have the lowest erosion rates and the greatest ability to store water and stabilize soil structures (Cerdan et al. 2010, Durán-Zuazo et al. 2013, Lin et al. 2014). Where possible it is important to include a diversity of native plant species in erosion management projects. In some locations this may require the addition of topsoil or amendments to improve the soil conditions to support to growth of a greater diversity of species.

When planning for erosion management it is also important to ensure a high proportion of appropriate soil cover in order to effectively reduce erosion rates (Bhattarai et al. 2011). Mulch, which is simply the residue from harvested plant materials is known to be valuable for soil conservation because of its ability to provide surface cover and protection from wind and rain impacts (Smets et al. 2008). Mulch also reduces erosion rates by increasing biological activity, improving soil structure and aggregate stability, decreasing surface sealing, increasing infiltration, and changing runoff patterns such as decreasing flow velocity (Smets et al. 2008). The effectiveness of different types of mulch for reducing erosion has been studied in a variety of environments and it is generally agreed that mulching reduces sediment loss when compared with bare soil in all environments investigated (Smets et al. 2008, Bhattarai et al. 2011). The effectiveness of mulch materials seems to depend on a number of factors including soil properties and slope characteristics (Smets et al. 2008).

The effects of soil properties and slope characteristics on erosion rates have been studied on every continent both in the field and in laboratory experiments (Fox and Bryan 1999, Lal 2003, Cerdan et al. 2010, Fu et al. 2011). Soil organic matter content is an

important indicator of the structural health of the soil; high organic matter content increases aggregate stability and soil cohesion giving it the potential to reduce erosion rates (Zhao et al. 2017). Soil texture has also been investigated in terms of its effect on erosion and it was found that medium and fine soils could be classified as highly erodible while coarse and very fine soils tend to be less erodible in comparison (Cerdan et al. 2010). However, these erodibility classifications were only found to have a significant effect on soil erosion in arable land where soil experienced frequent disturbance (Cerdan et al. 2010). Therefore rainfall, soil cover, and slope characteristics are likely the more influential factors affecting erosion rates in natural environments that do not experience regular mechanical disturbance. Durán-Zuazo et al. [2013] found that bulk density, soil cover percentage, runoff, soil organic carbon, and total rainfall were the variables most strongly correlated to soil erosion of all the parameters they investigated. It is worth noting that Durán-Zuazo et al. [2013] only looked at rainfall and soil parameters and did not consider slope characteristics in their study.

Slope gradient and length are both known to significantly affect erosion rates for bare soil at the plot scale (Cerdan et al. 2010). Runoff tends to increase with slope length on bare land; as a result sediment detachment and transport also increase with increasing slope length (Cerdan et al. 2010). Runoff is also known to increase with slope gradient caused by an increase in flow velocity and a decrease in infiltration as slope gradient increases, this increased flow velocity increases sediment transport and erosion (Fox and Bryan 1999).

Permanent vegetation and mulch can alter the hydrological properties of runoff on slopes through increased surface roughness, reduced flow velocity, increased infiltration

capacity, and runoff absorption by mulch materials, all of which reduce total runoff amounts (Cerdan et al. 2010, Bhattarai et al. 2011). These changes in the hydrological properties of surface runoff result in erosion rates that remain constant or even decrease with slope length and gradient in vegetated and mulched systems (Cerdan et al. 2010, Bhattarai et al. 2011).

Knowing the effect that just one aspect of scale, slope length, can have on the mechanisms driving erosion, it is not surprising that the effect of scale has been thoroughly investigated and debated in the literature. Soil erosion can be studied at a variety of scales depending on the dominant erosion process being investigated (Stroosnijder 2005, Konz et al. 2012). The five spatial scales for measuring erosion defined by Stroosnijder [2005] are: (1) the point (1 m²) scale, (2) the plot (<100 m²) scale, (3) the hillslope (<500 m²), (4) the field (<1 ha), and (5) the small watershed (<50 ha).

Point scale measurements are useful to represent small-scale erosion processes occurring within the immediate vicinity of the measuring device, which is normally a splash cup or erosion pin (Sirvent et al. 1997). Erosion plots are very useful for measuring interrill (unconcentrated flow) and rill (small concentrated flow) erosion but are not large enough to allow gully formation or to capture mass flux events (Boix-Fayos et al. 2006). According to Boix-Fayos et al. [2006], erosion plots are:

"very efficient at representing the hydrological behaviour of different surfaces to allow comparisons between them and to facilitate the study of the influence of soil surface components in the runoff generation and erosion mechanism."

The ability to easily compare between plots is very important for studies like this one whose goal is to investigate different surface covers and their effect on runoff and erosion. The plot scale also has the advantage of being well suited to runoff collection measurement methods, which are fairly simple, accurate, relatively inexpensive, can be employed in many types of land use regimes, and do not require complex technology or technical skill to obtain data (Stroosnijder 2005).

The study of erosion at larger scales (ie. Hillslope, field, and watershed) require more sophisticated measurement methods such as automated flow sampling, LiDAR scanning or radionuclide concentration sampling that require expertise and expensive equipment (Stroosnijder 2005, Konz et al. 2012, Meijer et al. 2013). However, these methods have the advantage of providing data on long-term and large-scale general erosion rates and sediment redistribution processes that are not attainable by other methods.

Due to the many advantages of field plots described above, it is not surprising that the plot scale (1 m² < 100 m²) is the most common scale at which rainfall generated erosion is studied (Smets et al. 2008). The size of the erosion plot is known to have an effect on the estimated erosion rate, which is not surprising given the effect of slope length on runoff and sediment transport (Leys et al. 2010). However, the plot width can also have an affect on erosion estimates due to scouring at the edges (Visser et al. 2004). It has been noted that erosion rates from small plots do not accurately reflect erosion rates at larger scales, particularly when applying rates calculated from short plots to long slopes or applying plot scale data to a landscape or watershed (Leys et al. 2010, Sadeghi et al. 2013). Despite some scaling issues, appropriately designed erosion plots are the most

convenient, cost effective, and standardized way of estimating minimum runoff and soil loss from an area and for these reasons have been chosen for this study. The plots here spanned almost the entire slope, in most cases, so calculated erosion rates are likely an accurate reflection of minimum rainfall generated runoff and erosion on the slope and could be directly applicable to similar sized coastal bluffs with similar slope and geology.

Helping Nature Heal Inc. (HNH) is an ecological landscaping company based in Bridgewater, Nova Scotia. One of the services they offer is coastline erosion management for coastal property owners; this service is called a Living Shoreline. The technique they use on their Living Shoreline projects is their own proprietary combination of mulching and revegetation. HNH uses a combination of hay, brush, and planting native species to stabilize soil and reduce erosion, primarily on coastal slopes such as bluffs and banks.

While these techniques seem very similar to those being studied by many others there are four key differences. (1) Hay is not commonly used as a mulch material in any of the literature reviewed. Likely because the presence of seeds in hay is a concern in agricultural applications, which is what many of the studies are testing soil conservation techniques for. However, it is commonly used to manage erosion during road building and other types of construction projects in Nova Scotia (personal communication, Bob Pett 2017). (2) Brush is also not commonly discussed in the literature, likely because it is too coarse for use in agricultural applications. Calatrava and Franco [2011] investigated the use of pruning residues to reduce erosion in olive orchards but the residues were ground into mulch before application, unlike the whole branch techniques used by HNH. A review by Smets et al. [2008], found only 2 studies out of the 41 reviewed that investigated hay as mulch in plot based erosion experiments and no studies using brush

were reviewed. One paper was found during a review of the literature that investigated brush in relation to sediment and erosion. In that study they found brush to be very effective at trapping sediment in a dune system (Stanley and Watt 1990).

- (3) The steep slope gradient of coastal bluffs that is mirrored at the experimental site is another reason previous studies are not comparable to the techniques investigated here. Most studies on the effectiveness of mulching come from agricultural studies in areas with relatively low slopes (ie. 4, 8 and 16% slopes in Bhattarai et al. [2011]). Smets et al. [2008] reviewed twenty-three field studies that investigated the effects of mulching on erosion rates at the plot scale, 17 of those studies were on slopes with a gradient \leq 10% and only 2 studies were on slopes with a gradient > 25%. There are a few mulching studies that have been done in forest fire affected areas with steeper slopes (ie. 36% in Vega et al. [2015] and 25-50% in Robichaud et al. [2013]). However, the slopes used in this study are considerably steeper than any plot slope in the literature reviewed. At the experimental site the primary experimental plots had a mean slope gradient of 63.5% and the secondary experimental plots had a mean slope gradient of 72.2% as measured through surveying with a Leica Geosystems TCR705 Total Station. Agriculture and Agri-Food Canada [2013] classifies slopes of 31-60% as extremely steep, corresponding to those in the literature, and slopes >60% as excessively steep, consistent with the average gradient of slopes used in this study.
- (4) The technique of planting potted native woody plants into slopes is an unusual treatment that has not been investigated by many authors. Garcia-Estringana et al. [2013] were the only researchers that planted native shrubs to look at their effectiveness for runoff and erosion control. Cuttings from easily rooted species such as willow are more

commonly used in bank stabilization projects (Polster 2006, Bischetti et al. 2010). Most of the studies that looked at the effects of vegetation on soil erosion, simply compared erosion rates for crop plants, agro-forestry plantations, orchards, and different natural vegetation types (ie. grasslands, shrublands, and forests) (Mohammed and Adam 2010, Anh et al. 2014, Labrière et al. 2015).

Natural rainfall events are quite frequent in Nova Scotia and will be the primary source of data for this study. However, natural rainfall can be highly variable in both time and space (Marques et al. 2007). At the site used in this study, differences in overhanging vegetation above some plots could further add to this variability by intercepting and redirecting raindrops (Nanko et al. 2011). Rainfall simulation experiments are a relatively common way control rainfall variability in order to evaluate possible erosion rates under different soil and slope conditions, and have been used to assess the effectiveness of erosion mitigation techniques like mulching (Bhattarai et al. 2011, Fiener et al. 2011, Fu et al. 2011).

Based on the literature there are two key factors to ensure a scientifically robust rainfall simulation experiment. It is important that rainfall simulations adequately model the intensity and duration of the natural rainfall events to be investigated (Marques et al. 2007). The other key factor is ensuring that there is an even distribution of simulated rainfall across the study area (Marques et al. 2007, Bhattarai et al. 2011, Fiener et al. 2011, Fu et al. 2011).

The simulations conducted for this experiment accurately modeled the intensity (mm/sec) of high intensity rainfall events and the total amount of water added fell within the range of natural rainfalls measured at the site (Mueller and Pfister 2011). However,

the lack of availability of appropriate simulation equipment and the constraints of working on an excessively steep slope in a remote location prevented us from accurately modelling natural rainfall duration and ensuring even distribution of water across the plots. Despite these difficulties it is still valuable to assess the effect of the treatments on runoff and sediment loss from a known volume of water and in the absence of spatial variability in rainfall caused by overhanging trees and vegetation above and below the plots. Also supplemental watering of Living Shoreline projects until they become established is part of the maintenance package for projects built by HNH. In order to accurately simulate supplemental watering, we used the same equipment in our watering experiment as HNH uses for their projects. Since the watering simulations did not have the elements needed for a scientifically robust rainfall simulation they were called controlled watering experiments (CWEs).

Overall, this research will help fill some of the knowledge gaps in the coastal erosion management literature. Especially concerning the effectiveness of these Living Shoreline techniques for managing erosion on bluff and bank faces caused by rainfall and overland flow processes. Based on the trends seen in the literature regarding plot scale erosion with different types of erosion management approaches, we expect to see reduced erosion from treatments that cover large portions of the soil surface when compared to treatments with low soil cover values (Garcia-Estringana et al. 2013).

Methods

Site Characteristics

The primary and secondary experimental plots are located at the same site in Crousetown, Lunenburg Co., Nova Scotia (44°14'11.3"N 64°29'59.0"W)(Figure 1). The soil in this area is classified as humo-ferric podzol (Agriculture and Agri-Food Canada 2013 (b), Agriculture and Agri-Food Canada 2015). This area is dominated by slate till of meguma group origin and the regions drumlins, such as the one experimental plots are at the top of, are formed from Hartlen till (Province of Nova Scotia 2013). Hartlen till is a locally derived till consisting mainly of slates and greywackes (Stea 1982). The composition of Hartlen till is typically 40% gravel, 40% sand, and 20% silt and clay (Webster et al. 2014). Data from the Environment Canada weather station in Bridgewater, NS shows the average annual rainfall from 1981-2010 is 1535.7 mm and the mean temperature varies from 19°C in July to -5.3°C in January with an average yearly temperature of 7.5°C (Government of Canada 2015).



Figure 1) Satellite image of primary and secondary experimental plots along a road cut in Crousetown, NS. Imagery © 2017 Cnes/Spot Image, DigitalGlobe, Landsat/Copernicus, Province of Nova Scotia, map data © 2017 Google.

The primary and secondary experimental plots were located on slopes on either side of a road cut (Figure 1 inset). The mean slope gradients for the primary and secondary experimental plots were 63.5% and 72.2% respectively, making them excessively steep (Agriculture and Agri-food Canada 2013). At the time when experimental set up began in May 2015, the slopes were approximately 100% bare soil and appeared to be experiencing active erosion based on a lack of vegetation and prevalence of small rills. According to SEDMODL (a GIS based sediment delivery model that identifies the erosion potential of road segments) outputs, a road cut with a slope gradient of 47.3±7.6 % and a vegetation cover of 28.7±7.8 % falls into the category of

very high sedimentation and erosion risk (Parsakhoo et al. 2014). This supports the observational conclusion that the slopes were actively eroding.

Experimental Treatments

The treatments used in this experiment are based on the techniques developed and used by Helping Nature Heal Inc. (HNH) to manage erosion and stabilize coastal banks in their Living Shoreline projects. HNH constructs their Living Shorelines using a specific combination of hay mulching, brush weaving, and planting native species, therefore this experiment consisted of four experimental treatments; (1) mulching with hay, (2) planting native species into the slope, (3) brush weaving, and (4) all combined, which is a combination of the first three techniques meant to represent the way these techniques are applied in HNH's Living Shoreline projects (Figure 2). Plots of bare slope were left unaltered to function as a control (Figure 2). The control treatment had ~100% bare soil, the only exceptions being a few small plants that had come in through natural regeneration prior to experimental set up. Based on visual estimates the Plant treatment covered an average of 25% of the surface of the plots. This number would have been slightly lower in the fall of 2015 when the plants lost their leaves and may have been slightly higher in the summer of 2016 at the end of their second growing season.



Figure 2) Examples of experimental plots containing the untreated control (A) and the four experimental treatments; native plants (B), hay mulch (C), brush weaving (D), and all combined (E).

Primary Experimental Plots

The primary experimental area consisted of fifteen, closed plots fitted with runoff collection devices. In tables and analyses that include primary and secondary experimental plots, primary plots are labelled 15-1 to 15-15. The four experimental treatments (Hay, Brush, Plants, All) and the Control had three replicates each, which were distributed along the study slope in a random block design according to apparent changes in slope gradient (Figure 3). Plots were bounded at the top and two sides by slab wood,

which was buried at least 10 cm below the slope surface and anchored with wooden stakes to prevent shifting (Figure 4).

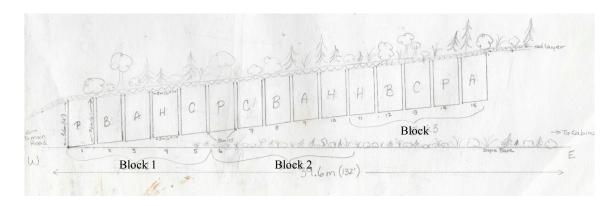


Figure 3) Plot layout for primary experimental plots. Treatment code: Control (C), planting (P), hay mulch (H), brush weaving (B), all combined (A).

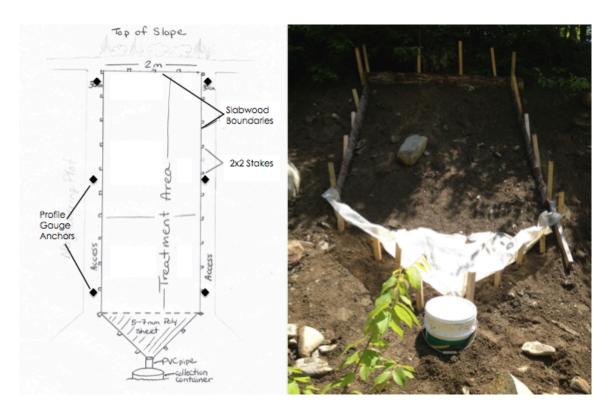


Figure 4) Closed plot design for primary experimental plots (left) and photo of closed plot in the field after installation (right).

Closed plots were chosen for the primary experimental design because the runoff's generation area is known which, facilitates the standardization of runoff and erosion rates by area (Boix-Fayos et al. 2006). The plots were on average 1.78 meters wide and 2.73 meters in length, with a 30-centimeter space between plots to allow access for data collection. This plot width was chosen because it is wide enough to minimize the effect of scouring at the edges of the plots and narrow enough to permit the desired number of replicates along the slope (Visser et al. 2004). Due to variability of slope conditions and the materials used, the plots vary slightly in size and slope. The exact size and slope of each plot was documented through surveying with a Leica Geosystems

TCR705 Total Station, to an accuracy of 2 mm +/- 1ppm, at the beginning and end of the experiment (Table 1).

Runoff collection devices were selected to assess runoff and sediment loss because they are an easy, convenient way to get an accurate estimate of the minimum amount of runoff produced and sediment removed from the bank face during a rainfall event (Gatto 1988). The runoff collection system consisted of a thick plastic sheet (5 mm Duschense house wrap) with one edge attached to a piece of wood as an anchor. The wooden anchor was then buried 15 – 20 cm deep across the bottom of each plot. Each plastic sheet was slightly wider than the plot and the edges were vertically attached to stakes to ensure runoff was contained. Each plastic sheet narrowed into a funnel shape that discharged into a piece of ABS pipe 4 inches in diameter that was inserted into the side of a 20 L bucket. The buckets were inset into the ground to create a downslope angle so that the runoff and sediment could travel easily into the container. Buckets were fitted with lids to prevent evaporation and prevent rain from entering the sample.

Table 1) Area (m²) and slope gradient (%) of each plot at the beginning (June 26, 2015) and end (August 29, 2016) of the study as determined by surveying with a Leica Geosystems TCR705 Total Station. Accurate to 2 mm +/-1ppm.

Plot	2015 Area (m²)	2016 Area (m²)	2015 Slope (%)	2016 Slope (%)
15-1	3.92	3.90	58.4	58.7
15-2	3.65	3.97	64.0	62.6
15-3	3.54	3.41	61.8	60.3
15-4	3.73	3.68	63.0	62.0
15-5	3.60	3.56	64.0	65.6
15-6	3.69	3.54	59.7	63.5
15-7	3.50	3.60	67.0	68.5
15-8	3.93	3.97	64.2	65.3
15-9	3.53	3.77	65.4	66.2
15-10	4.11	3.99	63.3	63.5
15-11	3.51	3.45	66.5	66.0
15-12	3.49	3.23	64.0	64.5
15-13	3.70	3.36	63.2	61.2
15-14	3.54	3.45	64.5	64.4
15-15	3.74	3.78	64.1	60.3
12-1	3.81	4.04	61.4	60.7
12-2	3.64	4.00	67.9	67.6
12-3	4.09	4.06	67.9	67.3
12-4	3.98	4.31	61.5	59.3
12-5	4.17	4.38	65.6	62.1
12-6	4.15	4.18	72.8	73.1
12-7	4.03	4.04	70.4	68.7
12-8	3.88	4.05	72.0	70.1
12-9	3.83	3.89	84.9	81.2
12-10	4.13	4.38	79.7	74.3
12-11	4.07	4.52	80.4	75.2
12-12	3.87	3.86	82.3	75.1

Data Collection

Both the primary and secondary experimental plots were surveyed using a Leica Geosystems TCR705 Total Station when they were first installed (June 26, 2015) and at the end of the monitoring period (August 29, 2016). There was a portable weather station on site with a tipping bucket rain gauge to measure total rainfall amounts and 1-hour rainfall intensity. Unfortunately, due to equipment error only a small portion of the rain gauge data was accurately recorded. There was also a 20L bucket set up next to the weather station to collect rainfall as a back up. The amount of rain in the bucket (mL) was recorded and used to calculate rainfall (mm). After plots were installed and before treatments were put in place, surface soil samples were taken from the middle of each slope (plots 12-6 and 15-8). These samples were sent away for chemical, metal, and nutrient analysis. At the same time, core samples were taken from the top 10 cm of the soil surface in plots 12-1, 12-6, 12-12, 15-1, 15-8, and 15-15. These samples were stored in the fridge at 4°C for bulk density analysis.

Runoff was collected from the collection buckets within 24 hours of the end of each natural rainfall event. The total quantities of runoff and sediment collected were measured because the total sample provides the most accurate data about rates of runoff and sediment loss from an area, in contrast to aliquot sampling (Bagarello and Ferro 1998). Runoff was poured from the collection bucket through a 63 µm sieve into a second 20L bucket with graduations every 250 mL. The amount of runoff (with sediment greater than 63 µm removed) was recorded to the nearest 250 mL. The sediment greater than 63 µm that was collected in the sieve was placed in a labelled Ziploc bag. Even though eroded sediments less than 63 µm are a concern because of their potentially negative

impacts on off-site biological communities, they were not measured in this experiment because of the equipment available (63 μ m was the smallest wet sieve available) and it was not practical to transport, store, and dry the total water and sediment samples because of their significant weight and volume especially after large rainfall events.

The sediment sample bags were taken to the Intertidal Coastal Sediment

Transport (In_CoaST) Research Unit lab at Saint Mary's University, where each sample
was placed in a beaker and allowed to settle for a minimum of three days. The
supernatant was carefully pipetted off and the remaining soil and water was transferred to
aluminum containers of known weight. Purified water was used to rinse remaining
sediment when samples were being transferred from bags to beakers and beakers to
aluminum containers in the lab. The samples were then dried at 75°C in a drying oven for
a minimum of 3 days and placed in a desiccator for a minimum of one hour before being
weighed. Total sample weight in grams was recorded.

The samples were then separated into pebble/granule, sand, and organic components by sieving through a 2 mm sieve. Pebbles and granules were greater than 2 mm and sand was less than 2 mm. Even though Hartlen till typically contains 20% silt and clay, it is assumed that silt and clay components were present in very small amounts in the collected sediment samples since most would have been lost through the 63 μ m sieve used to process samples in the field. Organic material was removed from the samples by hand during the sieving process. Large cobbles (65 mm < 256 mm) were air dried and weighed separately. No boulders (> 256 mm) were eroded during the experiment.

Profile gauge measurements were taken at each plot to measure soil distribution changes within each plot (Appendix Figure 1). The profile gauge is designed to measure point scale soil levels, which were then used to estimate plot scale sediment redistribution over the course of the experiment. The profile gauge was built using a 2x2 inch piece of lumber that was wide enough to span the plots (245 cm this case). Twelve holes were drilled through the lumber at equal spacing starting 30 cm from each end. The holes were just big enough to allow a zinc pin with a 4.8 mm diameter to pass through. The pins were 91.5 cm long.

Three sets of anchor stakes were installed outside the plot area at the top, middle, and bottom of each plot to support the profile gauge and ensure it was in the same position for each subsequent measurement (Figure 3). The profile gauge ends were rested on the anchor stakes with the pins perpendicular to the soil surface. The zinc pins were lowered through the holes to just touch the soil surface and were secured in place with clamps (Appendix Figure 1). The gauge was then moved to a flat surface and the distance from the bottom of each pin to the wooden part of the gauge was measured. This was repeated at each of the three positions for each of the plots. Profile gauge measurements were taken four times during the study period: (1) At the beginning of the experiment after treatments were installed (Fall 2015) (2) Spring 2016, (3) Early Summer 2016, and (4) at the end of monitoring (Late Summer 2016).

Secondary Experimental Plots

The secondary experimental area had twelve open plots constructed the same way and using the same materials as the primary experimental plots, except with no boundary

at the top. Open plots were chosen due to financial and time constraints as well as the type of data to be collected did not require knowledge of the runoff generating area. In tables and analyses that include primary and secondary experimental plots, secondary plots are labelled 12-1 to 12-12. The plots were, on average, 1.75 meters wide and 2.57 meters in length with a 30-centimeter space between plots to allow access for data collection. The plots had a mean slope of 72.2% at the time of installation. Due to variability of slope conditions and the materials used, the plots varied slightly in size and slope. The exact sizes and slopes were documented through surveying with the Leica Geosystems TCR705 Total Station at the beginning and end of the experiment (Table 1). The 12 plots consisted of three replicates of the four treatments; (1) mulching with hay, (2) planting native species into the slope, (3) brush weaving, and (4) all combined. Due to space constraints on the slope there were no control plots. As with the primary experimental plots, the secondary experimental plots were organized in a random block design based on the slope of the site (Appendix Figure 2). The secondary experimental plots were not fitted with runoff collection devices. These plots were used to collect profile gauge data. Runoff lag was also recorded from the secondary plots during the two controlled watering experiments described below.

Controlled Watering Experiment (CWE)

Two controlled watering experiments (CWEs) were conducted over the course of the study. The first CWE was on November 4, 2015 and the second on July 6, 2016. The CWEs were conducted using the same truck, water cube, pump, and hose that are used by Helping Nature Heal (HNH) for supplemental watering at their projects. The watering

time for each CWE was calibrated by setting the hose head to the shower function and timing how long it took to fill a bucket to 20L and that time was then used as the watering duration for each of the plots. The necessary assumption being that the rate of discharge from the hose was constant for each application. Assuming it was, each plot received 20L of water during the CWEs; the average water received by the plots was equivalent to 5.2 mm of rainfall with a range between 4.4 mm and 6.2 mm (mm of rainfall = 20L/plot area (m²).

The CWE calibration for November 4, 2015 was 20 L of water in 186 seconds; the average water received by the plots on this day was equivalent to 5.3 mm of rainfall resulting in an application rate of 0.03 mm/sec. Despite using the same equipment, the second CWE calibration on July 6, 2016 was 20 L of water in 236 seconds; due to the shifting of plot boundaries over time the average water received by the plots on this day was equivalent to 5.2 mm of rainfall resulting in an application rate of 0.02 mm/sec. According to the United States Geological Survey (USGS) this falls in the range of a violent shower (USGS 2016). The highest intensity rainfall found in the literature was 0.7 mm/sec reported by Mueller and Pfister [2011]. Water was applied to each of the primary and secondary experimental plots using the hose set to the shower setting for the time calibrated. During the CWEs an attempt was made to distribute the water as evenly as possible across each plot by spending approximately one third of the allotted time at the top, middle, and bottom of each plot. During the CWEs, the time it took for runoff to begin at the bottom of each plot was recorded as the runoff lag time. After each watering experiment the runoff was recorded and sediment was collected and processed from the primary plots according to the same procedures used for natural rainfall samples.

The dried and weighed sediment samples from each plot and CWE were then sieved into eight size classes based on the Udden-Wentworth sediment grain-size classification chart in preparation for analysis using Gradistat (Blott and Pye 2001). Three very large samples had to be divided into approximately 200 g samples using a riffle splitter before sieving. The size classes after sieving were, silt and clay (< 63 μ m), very fine sand (63 μ m < 125 μ m), fine sand (125 μ m < 250 μ m), medium sand (250 μ m < 500 μ m), coarse sand (CS; 500 μ m < 1 mm), very coarse sand (1 mm < 2 mm), granule (2 mm < 4 mm), and pebble (4 mm< 64 mm). The silt and clay class was likely remnants that were not lost through the 63 μ m sieve when the samples were collected in the field. These size classes were converted to the modified Folk and Ward classification scale adapted for Gradistat, the resulting classifications were; very fine sand (VFS), fine sand (FS), medium sand (MS), coarse sand (CS), very coarse sand (VCS), very fine gravel (VFG), fine gravel (FG), and very coarse gravel (VCG) (See Table 3).

Statistical Analyses

Data analysis was done using R Studio (RStudio 2016). An analysis of covariance with repeated measures was used to determine the effect of treatment, rainfall, and slope on runoff and sediment loss for natural rainfall. Plot was used as the error factor to remove the effect of measuring from the same plots on different days. The same analysis was done in R Studio to determine the effect of treatments on runoff, sediment loss, and runoff lag time for the CWEs. Profile gauge data were also analysed in R studio using analysis of covariance to determine the effect of position and treatment on sediment

movement within the plot. The sieved sediment weights from the CWEs were analysed using Gradistat to determine sample type, textural group, proportions of the different size classes, and the Folk and Ward mean, sorting, skewness, and kurtosis for the eroded sediment from each plot (Folk and Ward 1957, Blott and Pye 2001). The size class proportion data was logit transformed before analysis because proportions are necessarily part of a whole and are therefore not truly independent. The logit transformed size class proportion data was analyzed using analysis of covariance in R Studio.

Results

A basic soil analysis showed that the soil at the experimental site is mineral based, acidic soil with low organic matter and nutrient content (Appendix Table 2). This is consistent with the classification of humo-ferric podzols. The mean bulk densities of samples taken from the primary and secondary experimental plots just before treatments were installed were 1.06 g/cm³ and 1.19 g/cm³, respectively.

Table 2) Bulk density (g/cm³) of each of the surface soil samples taken from primary and secondary experimental plots.

Plot	Bulk Density (g/cm ³)	
15-1	0.98	
15-8	1.02	
15-15	1.19	
12-1	1.29	
12-6	1.15	
12-12	1.11	

Natural Rainfall

Rainfall, runoff, and sediment loss data were collected from the primary experimental plots from September 3, 2015 to December 5, 2015 and from April 28, 2016 to August 29, 2016. Data collection did not occur during the winter months. During the sampling periods there were thirty natural rainfall events that generated runoff. Rainfall data were collected by two methods; a tipping bucket rain gauge attached to a Campbell Scientific CR800 data logger and a 20-litre bucket placed inside of the rain gauge windscreen. The total amount of rainfall was successfully recorded in the 20-litre bucket from twenty-eight of the thirty runoff generating events. Due to equipment error, the data logger only accurately recorded rainfall data for eleven events, all of which were also recorded by the bucket.

The amount of rainfall in the bucket was recorded to the nearest 25 mL and this value was used to calculate rainfall amount in millimeters, therefore this is referred to as calculated rainfall for the rest of the study. Rainfall data obtained from the data logger is called logged rainfall data. It was assumed that the tipping bucket rain gauge would provide more accurate rainfall amounts than those calculated from the bucket. In order to ensure the accuracy of the calculated rainfall data, a linear regression analysis was performed to compare the successfully recorded rainfall amounts (mm) from the logger with the corresponding calculated rainfall amounts (mm) (Figure 5). The results of a Pearson correlation test indicated a very strong correlation ($R^2 = 0.988$) between the logged rainfall amount (mm) and the calculated rainfall amount (mm). It was determined that calculated rainfall amounts could be accurately used to replace logged rainfall in all the following analyses.

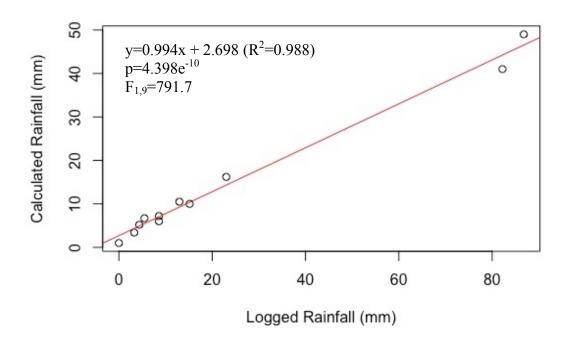


Figure 5) Regression analysis of the relationship between calculated rainfall (mm) and logged rainfall (mm) with Pearson correlation line of best fit. Residual standard error = 1.8 on 9 degrees of freedom.

Slope Gradient

The grade of the slope that the primary experimental plots were installed on appeared to get steeper as you moved from plot 15-1 to plot 15-15. The same trend was apparent for the slope where the secondary experimental plots were installed, with steepness increasing from plot 12-1 to 12-12. This apparent difference was part of the reason for choosing a random block design. Both the primary and secondary experimental plots were organized in 3 blocks (Figure 3; Appendix Figure 2). Primary experimental blocks are labelled 1, 2, and 3 for the following analyses while the secondary blocks are labelled 121, 122, and 123. Block 1 included plots 15-1 to 15-5, block 2 included plots

15-6 to 15-10, and block 3 included plots 15-11 to 15-15. Block 121 included plots 12-1 to 15-4, block 122 included plots 12-5 to 15-8, and block 123 included plots 12-19 to 12-12.

Primary Experimental Block Slope

An analysis of covariance (ANCOVA) was conducted on the mean slope (%) of plots within the three primary experimental blocks (Figure 3) to determine if there was a significant difference between the slope of plots within each block that could affect runoff and sediment loss. There was no significant difference between the slopes of the primary experimental blocks (Appendix Table 3). However, the ANCOVA showed a significant difference within plots for the year-block interaction where year indicates the slope at the initial survey (2015) and the slope at the closing survey (2016) (Appendix Table 3). The year-block interaction occurred because the mean slope of block 2 got steeper over time while blocks 1 and 3 got less steep (Appendix Table 4).

Secondary Experimental Block Slope

The results of an ANCOVA showed that there was a significant difference between the slope gradients (%) of plots within the three secondary experimental blocks (Appendix Figure 2; Appendix Table 5). Block 123 was significantly steeper than blocks 121 and 122; the slopes of plots within blocks 121 and 122 were not significantly different from each other (Appendix Tables 6 and 7). There was also a significant difference within plots for the effect of year and the block-year interaction (Appendix

Table 5). All of the blocks were less steep in 2016 than they were in 2015 and the biggest difference between the 2015 and 2016 slopes was seen in block 123 (Appendix Table 6)

Plot Slope Change

The slope of almost all the primary and secondary plots changed between the initial and final surveys (Table 1). An ANCOVA was used to determine if block or treatment had an effect on the net change in plot slope gradient (%). Block was found to have a significant effect on net change of plot slopes; treatment did not significantly affect the net change of plot slope (Appendix Table 8). A Tukey Highly Significant Differences (HSD) test showed that there was no significant difference between the mean net change in plot slopes for the primary experimental blocks (Appendix Table 9). The same Tukey HSD test showed that mean net slope change of block 123 was significantly different than blocks 121 and 122. The mean net change in slope of block 123 showed that the slope of plots in that block got significantly less steep over the monitoring period when compared to the plots in blocks 121 and 122 (Appendix Table 10).

Runoff from Natural Rainfall

The relationship between runoff (L/m²) and calculated rainfall (mm) was not linear. The effect of calculated rainfall on runoff was analysed using a non-linear least squares regression in R Studio (Figure 6). The Michaelis-Menten equation provided the best fit for the data (Appendix Table 11) (R-Bloggers 2016). An ANCOVA was used to assess the effect of treatment, calculated rainfall amount (mm), block, day, and the treatment-day interaction on the amount of runoff produced from the primary

experimental plots. Based on the ANCOVA results there was no significant effect of any of the analysed factors on runoff (Appendix Table 12). However, calculated rainfall and day had significant within plot effects on runoff (Appendix Table 12).

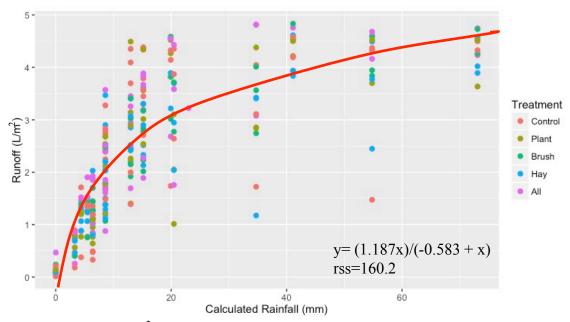


Figure 6) Runoff (L/m^2) from each primary experimental plot produced during natural rainfall events, plotted against the calculated rainfall (mm) for each runoff-generating event. Fitted with a Michaelis-Menten curve. Residual standard error = 0.8 on 271 degrees of freedom. Residual sum-of-squares (rss) = 160.2.

It was hypothesized that the effect of treatment on runoff may differ for small and large rainfall events and that this effect may be masked when all rainfall events were analyzed together. Small events were defined as those having less than 10 mm of calculated rainfall and large events were the five largest rainfall events that runoff was collected for (34.7 mm < 73.1 mm). During the monitoring period there were five rainfall

events with calculated rainfall amounts greater than 73.1 mm. However, the runoff collection buckets were filled to overflowing during those events so accurate runoff data could not be collected. Based on the results of an ANCOVA none of the factors tested, including treatment, had an effect on runoff for small rainfall events (<10 mm), but there was a significant within plot effect of rainfall (mm) and day on runoff (Appendix Table 13). There was also a strong positive linear relationship between calculated rainfall and runoff for small rainfall events (Appendix Figure 3).

For the large rainfall events (34.7 mm < 73.1 mm) there was a significant effect of treatment, calculated rainfall (mm), day, block, and the treatment-day interaction on runoff (L/m²) (Appendix Table 14). There was also a significant within plot effect of calculated rainfall and day on runoff from large rainfall events (Appendix Table 14). There was a weak positive relationship between runoff and rainfall amount for large rainfall events (Figure 7). The All treatment had significantly more runoff than the Control plots and Hay treatment (Appendix Tables 15 and 16). None of the other treatments were significantly different from each other or the Controls (Appendix Table 16). Block 1 had significantly less runoff than blocks 2 and 3 for large rainfall events (Appendix Tables 15 and 16).

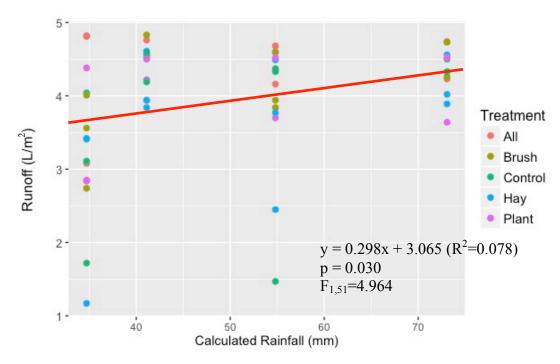


Figure 7) Linear regression of the effect of calculated rainfall (mm) on runoff (L/m^2) for the five largest rainfall events runoff data was collected for (34.7 mm < 73.1 mm). Residual standard error = 0.8 on 51 degrees of freedom.

Sediment Loss from Natural Rainfall

A significant, although weak, positive relationship between calculated rainfall (mm) and sediment loss (g/m²) was found (Figure 8). From Figure 8 there appeared to be trends in the amount of sediment loss from the different treatments. This raised the question of whether the relationship between calculated rainfall and sediment loss was consistent across treatments or if the treatments were affecting the relationship. Linear regressions of the effect of calculated rainfall on sediment loss were done for each of the treatments separately to determine if there was any treatment effect. There was a significant positive relationship between sediment loss and calculated rainfall for the Control plots and Plant treatment (Appendix Figures 4 and 5). There was no significant

relationship between calculated rainfall amount and sediment loss for the Brush, Hay, or All treatments (Appendix Figures 6, 7, and 8). This means that the Brush, Hay, and All treatments negated the effect of rainfall on sediment loss.

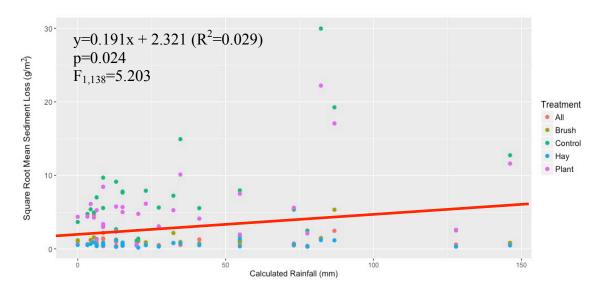


Figure 8) Regression analysis of the effect of calculated rainfall (mm) on sediment loss (g/m^2) . The square root mean sediment loss per treatment per rainfall event was used to improve the readability of the graph. Residual standard error = 4.4 on 138 degrees of freedom.

Factors Affecting Runoff

An ANCOVA was used to determine the effect of various factors on runoff. For natural rainfall events there was no significant effect of calculated rainfall (mm), treatment, day, or block on runoff (L/m²) (Appendix Table 17). However, there was a significant within plot effect of day and calculated rainfall on runoff where larger calculated rainfall amounts resulted in more runoff within plots (Appendix Table 17).

Results of an ANCOVA show that calculated rainfall (mm), treatment, and block had a significant effect on sediment loss (g/m²) from natural rainfall events (Appendix Table 18). We know from previous analyses that there is a significant positive relationship between rainfall amount and sediment loss (Figure 8). Figure 9 and the results of a linear mixed effects (LME) analysis show that the Hay, Brush, and All treatments had significantly less sediment loss than Control plots (Appendix Table 19). Sediment loss from the Plant treatment was not significantly different from the Control plots but was significantly greater than the Hay, Brush, and All treatments (Figure 9). Figure 10 and LME analysis show that block 3 had significantly more sediment loss than blocks 1 and 2, which were not significantly different from each other (Appendix Table 19).

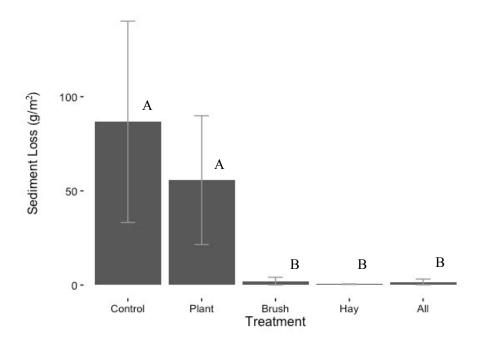


Figure 9) Sediment loss (g/m²) from each of the treatments from natural rainfall during the entire monitoring period. Error bars represent 95% confidence intervals.

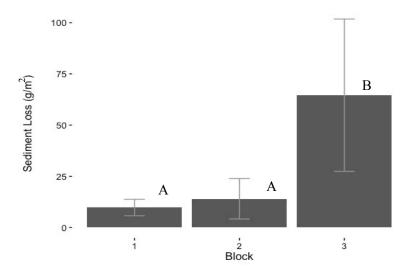


Figure 10) Sediment loss (g/m^2) from each of the primary experimental blocks from natural rainfall over the monitoring period. Error bars represent 95% confidence intervals.

Sediment Loss by Particle Size

Eroded sediment samples from natural rainfall were divided into three sediment size classes, cobbles (65 mm < 256 mm), pebbles/granules (2 mm < 65 mm), and sand (< 2 mm). These size classes were analyzed separately to determine if the treatments were affecting the sediment components differently.

Cobble Component

A linear regression analysis of the effect of calculated rainfall amount (mm) on the largest sediment size class, cobbles (65 mm < 256 mm), shows no significant relationship between the amount of rainfall and the weight (g/m²) of eroded cobbles (Appendix Figure 9). An ANCOVA testing the effect of treatment, calculated rainfall (mm), day, block, and the treatment-day interaction on cobble erosion showed no significant effect of any of the variables tested (Appendix Table 21). It is worth noting that five out of the seven cobbles that eroded during this experiment came from the low cover plots, two from Control plots and three from the Plant treatment. The Brush and All treatments contributed the other two cobbles. Low cover treatments contributed 81% of the weight of eroded cobbles and high cover treatments contributed the remaining 19%.

Pebble/Granule Component

There was a significant, although weak, positive relationship between pebble/granule erosion by weight (g/m²) and calculated rainfall amount (mm) (Appendix Figure 10). The results of an ANCOVA showed no significant effect of treatment, rainfall amount, day, or block on pebble/granule erosion (Appendix Table 22). Although, the

mean weights of pebble/granule erosion from the Control plots and Plant treatment are higher than the Hay, Brush, and All treatments (Appendix Table 23)

Sand Component

There was a significant, although weak, positive relationship between sand erosion (g/m²) and calculated rainfall amount (mm) (Appendix Figure 11). The results of an ANCOVA showed a significant effect of rainfall amount and treatment on sand erosion (Appendix Table 24). Block and day did not have a significant effect on sand erosion for natural rainfall events, however block had a p-value that was very close to significant (Appendix Table 24). The Hay, Brush, and All treatments had significantly less sand erosion than the Control plots and Plant treatment (Appendix Tables 25 and 26). The amount of sand erosion from the Control and Plant treatment were statistically the same. When cobbles and organic matter are excluded sand made up 75% of eroded sediment and the remaining 25% was pebbles.

Profile Gauge

Based on the results of an ANCOVA treatment, position, block, and the treatment-position interaction all had a significant effect on the net soil surface level change (cm) of the plots (Appendix Table 27). Overall, the mean soil surface level net change results were negative indicating a general trend of erosion within the plots (Figure 11). The Hay, Brush, and All treatments had significantly less negative net change than the Plant treatment indicating less erosion (Figure 11). A Tukey HSD test showed that the net change in soil level for the Plant treatment was significantly different from the All, Brush

and Hay treatments but not different from the Control plots (Appendix Table 28). The soil surface net change of the All, Brush, and Hay treatments were not significantly different from the Control plots. However, this may be a reflection of a lack of statistical strength from the Control plot data, because there were only three replicates when all the treatments had six replicates, rather than an indication of true lack of difference between the Control and the All, Brush, and Hay treatments.

Profile gauge measurements were taken at the top, middle, and bottom of each plot; these positions were numbered 1, 2, and 3, respectively in all figures. The top and bottom positions were significantly different from each other with the top position experiencing significantly more erosion than the bottom position (Figure 12). The middle position experienced net surface level change that was intermediate between the top and bottom of the plots, although this was not significantly different from either the top or bottom positions (Figure 12; Appendix Table 28). The primary experimental blocks showed trends in net surface level change. The net change in surface level of blocks 1 and 3 was significantly different based on Tukey HSD test results, with block 3 showing significantly more erosion than block 1 (Appendix Figure 12; Appendix Table 28). There was no significant difference in soil level net change (cm) between the secondary experimental blocks (121, 122, and 123) (Appendix Table 28).

The significant treatment-position interactions are summarized in Appendix Table 28. One important significant interaction is that the bottom position in the All treatment is experiencing net accretion (+ net change) while all the other treatment and position combinations are showing net erosion (- net change) (Appendix Table 29). However, the overall net change at the bottom position is negative as seen in Figure 12 because all the

other treatments are experiencing erosion at that position. A number of significant comparisons occur between position 3 of All treatment and position 1 of the other treatments, not surprising given the overall significant difference between positions 1 and 3 (Figure 12). When you compare the same position between treatments most of the treatments are not significantly different from each other, however the Plant treatment is a notable exception. Position 3 in the Plant treatment experiences significantly more erosion than position 3 in the All and Hay treatments (Appendix Table 29). Position 1 in the Plant treatment also experiences significantly more erosion than position 1 in the Hay treatment. The net surface level changes of the three positions are not significantly different within a treatment except for the All treatment where the net accretion at position 3 is significantly different from the net erosion at positions 1 and 2 (Appendix Table 29). Other factors noticed in the field that could have had an effect on net soil level change were the presence of vegetation or a rock at an individual pin. Vegetation and rock presence were noted during measuring and the results of an ANCOVA showed that there was no significant difference in soil surface net change based on the presence of vegetation or a rock at a pin (Appendix Table 27).

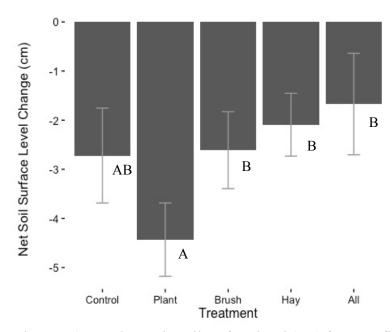


Figure 11) Net change in soil surface level (cm) from profile gauge measurements for each of the treatments between November 2015 and August 2016. Error bars represent 95% confidence interval.

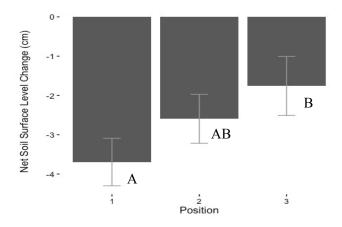


Figure 12) Net change in soil surface level (cm) between November 2015 and August 2016 at the top (1), middle (2), and bottom (3) positions of the plots from profile gauge measurements. Error bars represent 95% confidence interval.

Plant Survival

The All and Plant treatments both had native plants transplanted into them during installation. It was hypothesized that plant survival could be higher in the All treatment compared to the Plant treatment because of the protective effect and moisture retaining capacity of the hay mulch and brush layer present in the All treatment. A two-sample t-test found no significant difference between the Plant and All treatments for plant survival (Appendix Table 30).

Controlled Watering Experiment (CWE)

The first controlled watering experiment (CWE) was conducted in the fall of 2015. The second CWE was conducted in the summer of 2016. For both CWEs, 20 L of water was applied to the primary and secondary experimental plots. Runoff amount (L/m²), runoff lag time (sec), and sediment loss (g/m²) data was collected from the primary experimental plots. Since the secondary experimental plots did not have runoff collection only lag time was recorded.

Runoff

The amount of runoff (L/m²) produced from the two CWEs was significantly affected by treatment and not affected by block or plot slope (Table 3). The treated plots produced significantly less runoff than the Control plots (Figure 13; Appendix Table 31). The watering experiment (1 or 2) had a significant within plot effect, indicating a difference in the amount of runoff generated between the two experiments (Table 3). There was significantly more runoff produced during experiment 1 than there was in

experiment 2 (Appendix Figure 13). The treatment-experiment interaction also had a significant effect on runoff, indicating that the experiment effect significantly impacted the effect of treatment (Table 3). Appendix Figure 14 shows that the decrease in runoff during experiment 2 affected the Control and Plant treatment much more than the Brush, Hay and All treatments. As a result the runoff amounts from the Control plots and Plant treatment are almost the same as those from the Brush, Hay, and All treatments for experiment 2. This reduces the overall significance of the difference between the Control and the Brush, Hay and All treatments and potentially masks any difference between the Plant treatment and the Brush, Hay and All treatments.

Table 3) Results of the analysis of covariance testing the effect of treatment, block, plot slope (%), experiment (1 or 2), and the interaction of treatment and experiment on runoff (L/m²) for the two controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	1.5309	0.3827	42.777	5.25e ⁻⁰⁵
Plot	Block	2	0.0116	0.0058	0.648	0.552
Effects	Plot Slope	1	0.0022	0.0022	0.247	0.634
	Residuals	7	0.0626	0.0089		
Within	Experiment	1	0.6858	0.6858	54.713	4.12e ⁻⁰⁵
Plot	Plot Slope	1	0.0401	0.0401	3.198	0.107369
Effects	Treatment x	4	0.9140	0.2285	18.230	0.000242
	Experiment					
	Residuals	9	0.1128	0.0125		

SS = sum of squares; DF = degrees of freedom; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

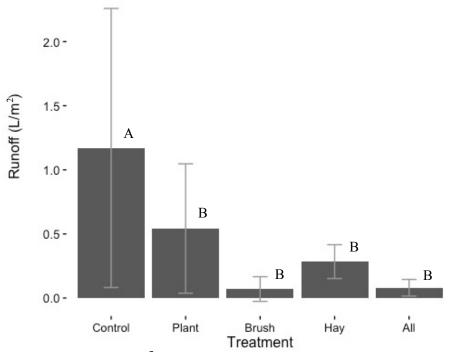


Figure 13) Runoff (L/m²) generated from each of the treatments during the two controlled watering experiments. Error bars represent 95% confidence interval.

Runoff Lag

The runoff lag time (sec) in this study is defined as the amount of time in seconds that it took for runoff to become visible at the bottom of each plot after the CWE began. The runoff lag time was not significantly affected by treatment, block, or plot slope (Appendix Table 32). However, Appendix Figure 15 shows that lag times were longer for the Brush, Hay, and All treatments than they were for the Control plots and Plant treatment, indicating that water took longer to flow over the surface of the Brush, Hay, and All treatments. The experiment and the treatment-experiment interaction had a significant within plot effect on runoff lag time indicating a significant difference between the lag times of the two experiments and that the experiment effect significantly

impacted the effect of treatment (Appendix Table 32). Between the two experiments there was significantly longer runoff lag time from experiment 2 than experiment 1 (Appendix Figure 16). The increase in runoff lag time in experiment 2 was greater for the Control and Plant treatment making their lag times much closer to the lag times of the Brush, Hay, and All treatments and possibly masking any effect of the high cover treatments (Appendix Figure 17).

Sediment Loss

There was a significant positive relationship between runoff (L/m²) and sediment loss (g/m²) for the controlled watering experiments (CWEs) (Appendix Figure 18). However, the relationship appears to be driven by a few points and only explains a small proportion of the variation in the data (Appendix Figure 18). The results of an ANCOVA showed that runoff (L/m²) was the only factor that had a significant effect on sediment loss (g/m²) from the CWEs (Appendix Table 33). However, treatment was shown to significantly affect runoff from the CWEs (Figure 13) and Appendix Figure 18 indicates the possibility of differences in sediment loss between the treatments. When runoff is removed from the ANCOVA, results show that treatment and block had a significant effect on sediment loss (g/m²) from the CWEs (Appendix Table 34). The Brush, Hay, and All treatments had significantly less sediment loss than the Control plots while the Plant treatment was not significantly different from the Control or the Brush, Hay, and All treatments (Figure 14 and Appendix Table 35). Block 3 had significantly higher sediment loss than blocks 1 and 2 for the CWEs (Appendix Table 35 and Appendix Figure 19).

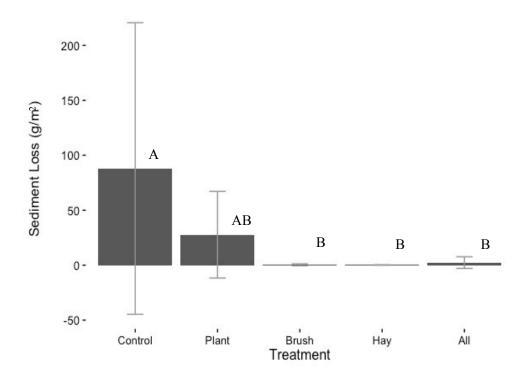


Figure 14) Sediment loss (g/m²) during the controlled watering experiments for each of the treatments. Error bars represent 95% confidence interval.

Sediment Grain Size

Gradistat produced mean proportion data for eight size classes of sediment defined in Table 4 (Adapted from Blott and Pye [2001]). Mean proportion data for all the gravel components together (Gravel) and all the sand components together (Sand) were also included in the Gradistat output. See Appendix Tables 36 and 37 for sediment component weights and proportions.

Overall, the effect of treatment was the same for the proportion of all sediment components except fine sand, which showed the opposite trend and very coarse gravel, which was not significantly affected by treatment. In general the Brush, Hay and All

treatments had low proportions of sediment components and the Control plots and Plant treatment had high proportions of sediment components. Very fine gravel and all the sand components except fine sand had significantly higher proportions from CWE 2, while the fine sand proportion was higher in CWE 1. Block only had an effect on the gravel components with block 3 producing significantly higher proportions of gravel than blocks 1 and 2.

Table 4) Sediment grain size scale used by the Gradistat program compared to Udden-Wentworth grain size scale used to sieve the sediment components. Table adapted from Blott and Pye [2001].

Grain Size		Description		
phi	$mm/\mu m$	Udden-Wentworth	Gradistat	
-6	64 —	Pebbles	Vary Coorse Crayel	
- 5	32	Peobles	Very Coarse Gravel	
- 4	16		Coarse Gravel	
- 3	8		Medium Gravel	
			Fine Gravel	
-2	4 —	Granules	Very Fine Gravel	
-1	2 —	Very Coarse Sand	Very Coarse Sand	
0	1	Coarse Sand	Coarse Sand	
1	500 μm		Coarse Sand	
	•	Medium Sand	Medium Sand	
2	250	Fine Sand	Fine Sand	
3	125	Very Fine Sand	Very Fine Sand	
4	63 —	Silt and Clay	Silt and Clay	

CWEs 1 and 2 were analyzed together and it was found that treatment and block had a significant effect on the logit-transformed proportion of gravel in eroded sediment from the two CWEs (Appendix Table 38). The effect of experiment was significant within plots indicating a difference between the two experiments (Appendix Table 38). The Brush and Hay treatments had significantly less gravel erosion than the Control plots and the Plant treatment, this difference is less pronounced for CWE 2 (Figure 15). Block 3 had a significantly higher proportion of gravel in eroded sediments than blocks 1 and 2 (Appendix Table 39; Appendix Figure 20). There was a greater proportion of gravel in eroded sediments from the treatments in CWE 2 when compared to CWE 1, while the Control plots had a slightly lower proportion of gravel from CWE 2 than from CWE 1 (Appendix Table 39). CWE 2 produced higher proportions of gravel in overall eroded sediment than CWE 1. The logit-transformed proportion of sand eroded from primary experimental plots during the two CWEs was not significantly affected by treatment or block, but there was a significant within plot effect of experiment (Appendix Table 40). There was significantly more sand in eroded sediment from CWE 1 compared to CWE 2.

The gravel component of eroded sediment was divided into very coarse gravel (VCG), fine gravel (FG), and very fine gravel (VFG) by Gradistat. These gravel components were analysed for the effect of treatment, block, and experiment. The proportion of very coarse gravel (VCG) in eroded sediment was significantly higher in block 3 compared to blocks 1 and 2 (Appendix Figure 21). No significant effect of treatment or experiment on proportion of VCG in eroded sediment was observed (Appendix Table 42).

The proportion of fine gravel (FG) in eroded sediment from the two watering experiments was significantly affected by treatment and block (Appendix Table 42). The FG proportion was significantly lower in the Brush, Hay, and All treatments when compared to the Control plots and Plant treatment, while the Controls and Plant treatment were not significantly different from each other (Appendix Figure 22). The effect of the high cover treatments on FG proportion was less pronounced in CWE 2, however this was not significant (Figure 15 and Appendix Table 42). The proportion of FG in eroded sediment was significantly higher in block 3 than blocks 1 and 2 (Appendix Figure 23).

The proportion of very fine gravel (VFG) was significantly affected by treatment and block and there was also a significant difference between CWEs (Appendix Table 43). The Brush, Hay, and All treatments had significantly lower proportions of VFG compared to Controls and Plant treatment, while Control and Plant were not significantly different from each other (Appendix Table 44; Appendix Figure 24). The effect of the high cover treatments on the proportion of VFG was less pronounced in CWE 2 and this effect was significant (Appendix Table 43; Appendix Figure 24). There was a significantly larger proportion of VFG in eroded sediment from CWE 2. There was a significantly greater proportion of VFG in the sediment eroded from block 3 compared to blocks 1 and 2 (Appendix Table 44; Appendix Figure 25).

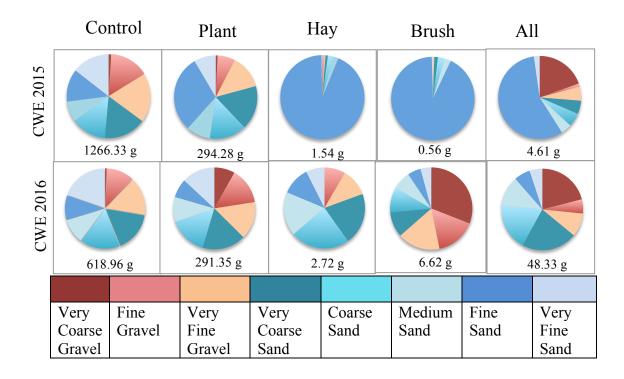


Figure 15) Mean proportion of sediment grain sizes (defined by GRADISTAT) for sediment lost from each treatment from the primary experimental plots during two controlled watering experiments (CWEs). Total weight of sediment sample (g) used for grain size analysis for each of the treatments and CWEs noted under each chart.

The sand component of eroded sediment was divided into very coarse sand (VCS), coarse sand (CS), medium sand (MS), fine sand (FS), and very fine sand (VFS) by Gradistat. These sand components were analysed for the effect of treatment, block, and experiment. Treatment had a significant effect on the proportion of VCS eroded during the CWEs and there was a significant difference in the proportion of VCS between the two CWEs (Appendix Table 45). Eroded sediment from the Hay, Brush, and All treatments had significantly smaller VCS proportions than the Control plots and Plant

treatment did; Control and Plant were not significantly different from each other (Appendix Table 46; Appendix Figure 26). There was a significantly higher proportion of VCS in eroded sediment from CWE 2 (Appendix Table 45; Appendix Figure 27).

The proportion of coarse sand (CS) in eroded sediments was significantly affected by treatment and there was a significant difference between the two CWEs (Appendix Table 47). The Hay, Brush, and All treatments had significantly smaller CS proportions than the Control plots and Plant treatment, while Control and Plant were not significantly different from each other (Appendix Table 48; Appendix Figure 28). There was a significantly higher proportion of CS in eroded sediment from CWE 2 (Appendix Table 47; Appendix Figure 29).

The proportion of medium sand (MS) in eroded sediments was significantly affected by treatment and there was a significant difference between the two CWEs (Appendix Table 49). The Brush treatment had a significantly smaller proportion of MS than the Control plots and Plant treatment (Appendix Table 50; Appendix Figure 30). None of the other treatments were significantly different from each other. There was a significantly higher proportion of MS in eroded sediment from CWE 2 (Appendix Table 49; Appendix Figure 31).

According to an ANCOVA, the proportion of fine sand (FS) in eroded sediments was not significantly affected by treatment or block, but there was a significant difference between the two CWEs (Appendix Table 51). However, a linear mixed effects analysis showed that the proportion of FS in eroded sediment from the Hay treatment was significantly different from the Control plots and the other treatments (Appendix Table 52). Eroded sediment from the Hay treatment had a significantly higher proportion of FS

than any of the other treatments (Figure 15; Appendix Figure 32). Fine sand was the only sediment component to have a higher proportion in a treatment that provided a high amount of soil cover. CWE 1 had a significantly higher proportion of FS in eroded sediment than CWE 2 (Appendix Table 51; Appendix Figure 33). Fine sand was the only sediment component to have a higher proportion in CWE 1. When the proportion of FS is graphed by treatment and CWE we can see that eroded sediment from the Hay treatment had a very high proportion of FS in CWE 1 and a slightly higher FS proportion than the other treatments in CWE 2 (Appendix Figure 34).

Treatment had a significant effect on the proportion of very fine sand (VFS) in eroded sediment and the CWEs were significantly different from each other (Appendix Table 53). The Brush, Hay, and All treatments had significantly smaller proportions of VFS than the Control and Plant treatment while the proportion of VFS in sediment from the Control plots and Plant treatment were not significantly different from each other (Appendix Table 54; Appendix Figure 35). CWE 2 had a significantly higher proportion of VFS in eroded sediment than CWE 1 (Appendix Figure 36).

Discussion

Hypothesis and Expected Results

The Control plots were expected to produce more runoff and sediment loss than the treated plots due to the complete exposure of soil in the Control plots. The Plant treatment also did not provide a high percentage of soil cover so we expected to see high runoff and erosion rates from those plots as well. Erosion and runoff from the Plant treatment may be higher than the Control due to soil disturbance during planting and altered runoff patterns that concentrate flows in between the plants (Abrahams et al. 1995). However, the erosion rate could be lower in the Plant treatment when compared to the Control plots due to factors such as increased infiltration close to the plants and runoff energy deflection by stems and leaf litter (Bochet et al. 2000, Vásquez-Méndez et al. 2010, Durán-Zuazo et al. 2013). Either way we expected to see higher erosion in the Plant treatment compared to the Hay, Brush, and All treatments due to the higher proportion of exposed soil. We expected the high cover treatments (Hay, Brush, and All) to function similarly to each other, especially in the first year. It is possible that decomposition of the Hay treatment and the loss of the needles from the Brush treatment may cause these to lose their soil protection function over time, resulting in higher runoff and sediment loss the second year when compared to the All treatment. Given the relatively short monitoring period of this study, it is not likely that the All treatment will outperform the other high cover treatments for runoff and soil loss. However, this could be expected if monitoring was conducted over a longer time period as the plants in the All treatment mature and the materials added in the Hay and Brush treatments break down. It is also possible that the poor soil at this location would not support the long-term survival of the

plants in the All treatment and that primary colonizers would move into the other treatments resulting in similar soil cover and erosion reduction between the Hay, Brush and All treatments over time.

Calculated Rainfall

Since rainfall is the main driver of runoff and sediment loss at the experimental site, it was important to obtain accurate data for the runoff generating rainfall events during the monitoring periods. As a result of issues with the power supply for the data logger most of the rainfall data from the tipping bucket rain gauge were not reliable. Luckily, there was a rainfall collection bucket set up inside of the rain gauge windscreen as a back up. This allowed the total rainfall to be measured to the nearest 25 mL, which was far from the accuracy of rainfall amount and hourly rainfall intensity data provided by the rain gauge. Fortunately, eleven rainfall events were accurately recorded by the tipping bucket rain gauge so the two rainfall measurement methods could be compared. There was a very strong correlation between calculated and logged rainfall that explained 98.8% of the variation in the data (Figure 5). Based on this strong correlation, it was determined that calculated rainfall amounts could be accurately used in place of logged rainfall for all analyses in this study.

Slope Gradient

The length and gradient of a slope is known to have a positive effect on erosion rates in bare soil and land uses with high proportions of bare soil, due to increasing runoff flow velocities and increasing down slope splash loss with increasing slope gradient (Fox

and Bryan 1999, Cerdan et al. 2010, Fu et al. 2011, Meshesha et al. 2016). When block effects were seen in the initial analyses, differences in slope were thought to be the cause since block 3 appeared to be steeper than the other blocks. However, when the slope gradients of the plots within the three primary experimental blocks were analyzed, there was only a few degrees difference between the blocks and this was not significant. Therefore, the significant differences in sediment loss and runoff between the primary experimental blocks is probably not due to slope gradient and may be due to other factors such as overhanging vegetation or soil bulk density (Nanko et al. 2011, Durán-Zuazo et al. 2013). When the slope gradients of the plots within the three secondary experimental blocks were analyzed block 123 was significantly steeper than blocks 121 and 122, thus any block effects in the secondary experimental data could be due to slope gradient.

The slope gradients of the individual plots and mean slope of the blocks changed over the course of the experiment, presumably due to erosion and deposition. Overall, slope gradient decreased within the plots and blocks. A decrease in slope angle due to erosion at the crest and deposition at the base has been identified as part of the evolution of certain types of hillslopes (Davis 1932 in Young 1972, Hugget 2007). However, this phenomenon is very dependent on the geology, dominant erosion processes, and conditions at the base of the hillslope (Young 1972, Hugget 2007). The slopes studied here are exhibiting this pattern of slope angle decline over the duration of the experiment. Out of the six experimental blocks, block 2 was the only one that got steeper between the initial and final surveys (Appendix Table 10). The change in the slopes of the plots in the primary blocks were not significantly different from each other, so the small increase in

steepness of block 2 is likely due to variability in the data possibly caused by characteristics of the location or measurement error.

Secondary experimental block 123 had the steepest initial slope gradient and was also the only block to have a significant reduction in slope between the initial and final surveys. The most likely explanation is that the initial steepness of block 123 caused more erosion to occur from the top of the plots, through increased splash loss and runoff velocities (Fox and Bryan 1999). Eroded sediment was then deposited downslope within the Hay, Brush, and All (high cover) treatments resulting in an overall decrease in the slope gradients of plots within the block. The majority of the plots (3/4) in each secondary experimental block contained high cover treatments, which profile gauge data shows capture significant amounts of sediment moving downslope. This sediment capturing effect combined with the large percentage of high cover treatment plots within the block, resulted in a significant reduction of the slope of the entire block.

Runoff from Natural Rainfall

Runoff is known to transport sediment, nutrients, metals, and other pollutants thereby contributing to soil erosion as well as sedimentation and contamination of receiving water bodies (Durán-Zuazo et al. 2011, Zhang et al. 2016). Therefore controlling runoff is important not only to reduce erosion but also to protect the health and quality of water resources for the people and organisms that depend on them. The relationship between rainfall and runoff is known to be non-linear due to the fact that the process of runoff generation is affected by a number of other factors including rainfall intensity, antecedent soil moisture conditions, and slope gradient just to name a few

(Zhao et al. 2014). According to a non-linear least squares regression, the equation that best fits the relationship between rainfall and runoff in this study is the Michaelis-Menten saturation equation (R-Bloggers 2016). This relationship is characterized by runoff amounts that increase at a rapid rate with increasing rainfall until a saturation point is reached, after which runoff increases very little with increasing rainfall (Figure 6). This relationship is not seen for runoff amount from multiple rainfalls in any other studies. However, it is seen for runoff rate (ml/s) during a single rainfall event in Ran et al. [2012] and runoff amount (kg/m²) over the duration of a single simulated rainfall event in Vilayvong et al. [2016]. Zokaib and Naser [2012] found a weak exponential relationship between natural rainfall and runoff when 7 years of data from four different land uses were analysed. It is very likely that the relationship between rainfall amount and runoff does not truly reach a saturation point but because the volume of runoff collected was limited by the size of the runoff collection containers, we see a levelling off at the collection limit instead of a continuing increase with rainfall.

It is expected that the vegetation within the treatments may affect runoff differently depending on the size of the rainfall event because the interaction between rainfall parameters, including amount, and vegetation are known to affect runoff generation (Zhao et al. 2014). Small (<10 mm) and large (34.7 mm < 73.1 mm) subsets of the calculated rainfall data were used to assess differences in runoff associated with rainfall event size. It is worth noting that the large subset of rainfall data does not include the five largest calculated rainfall events that occurred during monitoring. This is because corresponding runoff data were not available due to overflow of the collection containers during those rainfall events. The largest calculated rainfall during the monitoring period

was 146.2 mm, almost exactly double the largest rainfall (73.1 mm) in the large rainfall subset. This means that the 'large' rainfall events used in the following analyses were technically intermediate relative to all rainfall events that occurred during the study.

The significant positive linear relationship between rainfall and runoff for small rainfall events is expected because this corresponds to the lower portion of the exponential curve that most likely represents the true relationship between rainfall amount and runoff from multiple events (Zokaib and Naser 2012). There was no effect of treatment for small rainfall events. This is not entirely surprising because soil infiltration capacity and soil suction would not be exceeded by the small amount of water, thereby negating any effect of treatment on runoff due to reduction in flow velocity and increased infiltration (Zhao et al. 2014, Vilayvong et al. 2016). The significant within plot effect of rainfall is not surprising since different amounts of rainfall should produce different amounts of runoff within the same plot. The significant within plot effect of day indicates that differences between days, other than rainfall amount, such as rainfall intensity, wind, temperature, and antecedent soil moisture are likely affecting runoff amounts even for small rainfall events. Antecedent moisture and rainfall intensity, which vary greatly by rainfall event (day), are known to have a large effect on runoff (Zhao et al. 2014).

Runoff from large rainfall events was significantly affected by all of the factors investigated. There was a significant positive relationship between rainfall and runoff for the large rainfall events, however this relationship was very weak and only explained about 8% of the variation in the data. This weak correlation is likely due to the significant effects of the other factors on runoff and agrees with the results of other studies that found that rainfall amount does not explain runoff (Zokaib and Naser 2012) (Figure 7).

Treatment had a significant effect on runoff from large rainfall events, however the type of effect was very surprising with the All treatment having significantly more runoff than the Control plots and Hay treatment. Upon further investigation this effect is not completely unheard of, with one study finding that cover types that have a large proportion of direct contact with the soil surface, like straw and hay, have decreased infiltration and increased runoff compared soil cover types that protected the soil without being in direct contact with it (Leys et al. 2010). This may not be the correct explanation in this case because the All treatment (which contains hay) was significantly different from the Hay treatment and not the Brush treatment which protects the soil without large amounts of direct contact. The important thing to take note of is that this increase in runoff in the All treatment did not translate into increased sediment loss.

The significant effect of the treatment-day interaction on runoff suggests that the effect of the treatments can be significantly impacted by the conditions of the day for large rainfall events. This effect is likely because something about the conditions on Day 18 (May 23, 2016) caused all of the buckets to overflow except the Control plot and the Hay treatment in block 1, compared to the exact same calculated rainfall on Day 6 (October 15, 2015) that did not cause any of the buckets to overflow. The soil moisture conditions on Day 18 were likely higher than on Day 6 based on the amount of time since the previous runoff-generating event, which was 2 days and 9 days, respectively. The higher soil moisture conditions on Day 18 would have favoured higher runoff from the same rainfall amount because the infiltration capacity of the soil and soil suction would have been reduced (Zhao et al. 2014, Vilayvong et al. 2016). Thus the effect of

antecedent soil moisture is stronger than the effect of treatment on runoff amount generated from a rainfall event.

Block 1 had significantly less runoff than blocks 2 and 3 (see Figure 3) and this block effect was only seen for large events. Since blocks do not differ significantly in slope gradient it is not likely a slope effect. The effect of block is fairly consistent across all analyses, less runoff and sediment loss in block 1, intermediate runoff and sediment loss in block 2, and higher runoff and sediment loss in block 3. Initially, the block effect was thought to be due to the density of overhanging vegetation at the top and bottom of blocks 1 and 2 that could intercept raindrop impacts and deflect water away from the plots resulting in the observed block pattern. The results of the controlled watering experiments, discussed later, showed the same block effect except overhanging trees could not intercept the water because it was applied with a hose below the canopy.

Upon further consideration, it is possible that the block effect could be due to a gradient in soil bulk density across the slope (Table 2). The first plot of block 1 had lower bulk density than the middle plot of block 2 and the last plot in block 3 had the highest bulk density of all three. Bulk density measurements were not taken from every plot but it does appear that there may be a trend of increasing bulk density across the slope. If this were the case then the lower bulk density would increase the pore space available for water infiltration in block 1 (USDA 2017). This did not have an effect during small rainfalls but became significant for large rainfalls when the infiltration capacity of the soil with higher bulk density in block 3 was exceeded and more runoff was produced.

There was a significant effect of rainfall amount and day on runoff from large rainfall events. The effect of rainfall is expected since different amounts of rainfall should

produce different amounts of runoff. However, this indicates that the rainfall-runoff relationship did not completely level out at larger rainfalls since larger rainfalls are producing larger amounts of runoff. This supports the theory that the saturation relationship was related to the collection limit of the containers. The significant effect of day again indicates that differences between days, other than rainfall amount, such as rainfall intensity, wind, temperature, and antecedent soil moisture are affecting runoff amounts for large rainfalls as well. Antecedent soil moisture and rainfall intensity are most likely producing the day effect.

Factors Affecting Runoff

None of the factors tested including, calculated rainfall, treatment, block, or day had a significant effect on runoff amounts from natural rainfall. It is understood that rainfall amount alone does not adequately explain runoff so the lack of effect of calculated rainfall on runoff is not surprising (Zokaib and Naser 2012). It is surprising that the treatments did not significantly affect runoff from natural rainfall since it is well established that vegetation and soil cover normally reduce both the rate and amount of runoff (Cerdan et al. 2010, Nunes et al. 2011, Labrière et al. 2015). We may have observed an effect of treatment on runoff from natural rainfall had the buckets not overflowed resulting in a loss of runoff data from the largest rainfalls. Another consideration is that Hartlen till is 80% gravel and sand, which are very porous and would allow rapid infiltration and therefore limit any effects on runoff (Webster et al. 2014).

The expected effect of the treatments on runoff was seen in the two controlled watering experiments (CWEs) with the treatments producing significantly less runoff than

the Control plots. The volume of runoff produced from the Plant treatment in the CWEs was intermediate between the Control and high cover treatments (Figure 13). Even though it was not significant this result supports the hypothesis, discussed in the sediment loss section, that the Plant treatment is reducing sediment loss through a reduction in runoff by increasing infiltration (Leys et al. 2010).

The act of planting the plants also created depressions for water to pool around their base allowing more time for infiltration and further reducing runoff; this was especially effective due to the relatively small amount of water added per unit area (5.24 mm) and short duration of the watering experiments. Larger volumes of water would eventually fill the depressions creating runoff sources that increase runoff and complicate the relationship between runoff and depressional storage in the Plant treatment (Gómez and Nearing 2005). These opposing factors may have contributed to the lack of significant effect of treatment for runoff from natural rainfall.

Other competing factors within the treatments that could have resulted in a lack of significant effect of treatment on runoff for natural rainfall were soil moisture, surface roughness, and infiltration. The high cover treatments likely had higher soil moisture from reduced evaporation that favoured runoff generation while at the same time reducing runoff through increased surface roughness and infiltration. The low cover treatments would experience the opposite of these two opposing effects on runoff generation with the net result being no significant difference in runoff amount between the treatments during natural rainfall. Rainfall intensity and intra-event rainfall patterns are other factors that may have impacted runoff generation from natural rainfall events (Parsons and Stone

2006). Unfortunately, no data was collected on these properties of the natural rainfall events that occurred during this study.

Even though it was not significant, it is worth noting that Hay had the highest runoff of the high cover treatments in the CWEs (Figure 13), which is consistent with the findings of Leys et al. [2010] who found that soil covers that had high levels of soil contact produced more runoff than soil covers that did not have as much contact. The All treatment includes hay mulch and still has less runoff than the Hay treatment, contradicting the results from large natural rainfall events where the All treatment produced significantly more runoff than the Hay treatment. Perhaps the relatively small amount of water added and short duration of the CWEs allowed the depressional storage around the plants in the All treatment to reduce runoff below the level the Hay treatment. By contrast, during the large natural rainfall events the soil becomes saturated and the depressions around the plants become runoff sources creating excess runoff and resulting in the contradictory findings between natural rainfall and the CWEs.

The fact that none of the factors that were thought to control runoff had significant effects on runoff from natural rainfall events indicates that another factor, or combination of factors that were not taken into account must have a strong effect on the amount of runoff generated during a natural rainfall event. Based on the significant effect of day on runoff from natural rainfall and differences seen when analysing runoff from the two CWEs, it is likely that antecedent soil moisture content is the factor having the largest effect on runoff at the experimental site (Zokaib and Naser 2012, Zhao et al. 2014). CWE 1 was conducted on November 4, 2015 and produced significantly more runoff than CWE 2, which occurred on July 6, 2016. In the week prior to CWE 1 about 79 mm of

calculated rain fell, this is in contrast to CWE 2 that was conducted during a severe drought and only had 5.5 mm of calculated rainfall during the preceding week. Therefore, it can be assumed that the antecedent soil moisture was higher for CWE 1 than it was for CWE 2.

High antecedent soil moisture content decreases the amount of rainfall needed to saturate the soil surface, this causes the transition from infiltration excess overland flow to saturation excess overland flow to occur more quickly and allows runoff to occur from rainfall intensities that do not exceed the infiltration capacity of the soil (Cornell Soil and Water 2005, Jones et al. 2017). Soil suction is the negative pressure created by the air in pore spaces within the soil that actually sucks water into the soil. When the soil has high moisture content, soil suction is reduced and less water is pulled into the soil resulting in more runoff (Vilayvong et al. 2016).

The high antecedent soil moisture content presumed for CWE 1 coupled with the intense rate of application (0.03 mm/sec) would result in the infiltration capacity of the soil being exceeded and a higher potential for a transition to saturation excess overland flow, both of which contribute to runoff generation (Cornell Soil and Water 2005). The low antecedent soil moisture content during CWE 2 would also contribute to decreased runoff through increased soil suction (Vilayvong et al. 2016).

There was a significant effect of the interaction between experiment and treatment on runoff from the CWEs. The significant decrease in runoff from CWE 2 was mainly due to a large decrease in runoff from the Control plots and the Plant treatment. Runoff amounts from the high cover treatments only showed a very slight decrease for CWE 2 (Appendix Figure 14). As a result there does not appear to be a difference in runoff

between the high cover treatments and the low cover treatments for CWE 2. So really it was not that the experiment changed the effect of the high cover treatments because they functioned almost the same in both CWEs. It was that the dry soil conditions associated with CWE 2 disproportionately absorbed runoff in the low cover treatments and did not allow a difference in runoff generation to be observed. This finding agrees with Zhao et al. [2014] who described antecedent moisture content as having a positive relationship with runoff generation, with the rate of runoff increase as soil moisture increases being much lower when vegetation is present and much higher for bare soil. In other words bare soil is more sensitive to antecedent moisture content and its effect on runoff generation.

Runoff Lag from Controlled Watering Experiments

Data on the time it took for runoff to be generated after water began to be applied to the experimental plots (runoff lag) was only collected during the CWEs since collecting this type of data for natural rainfall requires specialized equipment. Runoff lag times showed the expected inverse relationship with runoff amount from the two CWEs. Even though there was no significant effect of treatment on runoff lag, the lag times from the high cover treatments were longer than the Control plots and Plant treatments (Appendix Figure 15). This is also consistent with Zhao et al. [2014] who found that runoff lag was shorter on bare soil compared to vegetation. Longer lag times mean that the runoff is being slowed down by the roughness created by the vegetation and mulch in the treatments, permitting more water to be absorbed by the soil (Leys et al. 2010, Zhao et al. 2014). The net result of slowing the runoff flow and increasing infiltration is less runoff, thus producing the inverse relationship between runoff lag and runoff amount

observed for the treatments in the CWEs. A longer lag time also likely means that the runoff is flowing with less velocity and therefore cannot transport sediment as easily, contributing to the observed reduction in sediment loss from the high cover treatments in the CWEs.

Watering experiment 2 had significantly longer lag times regardless of treatment. The low antecedent soil moisture in CWE 2 meant the soil had full suction and absorptive capacity, allowing a longer time until infiltration excess overland flow begins and ultimately resulting in longer runoff lag times (Cornell Soil and Water 2005, Vilayvong et al. 2016). In CWE 1 the soil had high antecedent soil moisture so soil suction and absorptive capacity would be reduced, allowing infiltration and saturation excess overland flow to be generated more quickly, resulting in shorter lag times. Decreases in soil suction at the surface have been shown to occur less than 2 minutes after water is added (Vilayvong et al. 2016). CWE 1 also had a higher intensity application rate and one study showed that lag times tend to get shorter with increasing simulated rainfall intensity (Zhao et al. 2014).

There was a significant treatment-experiment interaction indicating that the treatment effect on lag time was being impacted by the experiment effect. Similar to the treatment-experiment interaction effect for runoff, the increase in lag time for CWE 2 was greater in the low cover treatments than it was for the high cover treatments (Appendix Figure 17). Again the experiment effect did not change the treatment effect it just made the treatment effect less apparent because the lag time of the low cover treatments were increased to be very close to the high cover treatments. Lag time is known to be affected

by antecedent soil moisture content with higher soil moisture resulting in shorter lag times and bare soil is more sensitive to this effect than vegetated plots (Zhao et al. 2014).

Sediment Loss from Natural Rainfall

The weak positive relationship between sediment loss and calculated rainfall, while significant, only explains 3% of the variation in the data suggesting that rainfall is not the driving factor for sediment loss (Figure 8). This agrees with the results of other studies that found that rainfall amount and soil loss are not strongly correlated (Zokaib and Naser 2012, Vega et al. 2015). Zokaib and Naser [2012] also found a weak positive relationship between rainfall and sediment loss while Vega et al. [2015] found that maximum 10-minute intensity was the only rainfall variable significantly correlated to soil erosion. However, previous experiments by Vega et al. [2015] showed a significant effect of total rainfall on soil erosion and no effect of intensity at higher total rainfall amounts, so there may be a threshold after which total rainfall becomes significant.

The stratification of the relationship between mean sediment loss and rainfall by treatment suggests that there are differences in sediment loss between the treatments and that the relationship between sediment loss and rainfall is affected by treatment (Figure 8). The weak positive relationship between sediment loss and rainfall seen for sediment loss in general is also present in the Control plots and Plant treatment, explaining only 5 and 6 % of the variation in the data respectively. However, there is no relationship between sediment loss and rainfall for the Brush, Hay, and All treatments because the high soil cover provided by those treatments reduced sediment loss to a low level for all rainfall events and thus eliminating the small effect of rainfall amount on sediment loss.

Since there was no significant effect of treatment on runoff, the mechanism by which sediment loss is reduced by the Hay, Brush, and All treatments is likely by the interception of raindrop impact. Splash loss from raindrop impacts was found to be the dominant factor in sediment mobilization on short, steep (9-100%) slopes (Fu et al. 2011). While the plots used in this study are longer than those used by Fu et al. [2011], their mean slope gradient (63.5%) is close to the gradient (58%) at which Fu et al. [2011] found peak splash loss to occur. Reducing the raindrop impacts on overland flow would also reduce sediment loss by decreasing the turbidity and transport capacity of the runoff flow (Fox and Bryan 1999). The Hay, Brush, and All treatments were also more effective than the Control and Plant treatment at capturing sediment before it left the plots, as seen in the profile gauge data (Figure 11).

The hypothesis that the effect of the treatments would be mostly related to the amount of soil cover they provide appears to be supported by the experimental results, as can be seen in the above analyses. The Control plots and the Plant treatment, which both have large amounts of bare soil exposed tend to have similar effects on the variables being investigated. The Hay, Brush, and All treatments provide close to 100% soil cover and as a group, tend to exhibit similar effects the variables investigated. In general the effects of the Hay, Brush, and All treatments tend to be significantly different than the effects of the Control and Plant treatments. These high cover and low cover treatment groupings and their effects on sediment loss and runoff variables are found to be relatively consistently throughout the data, further supporting the hypothesis that the effect of the treatments is related to the amount of soil cover they provide.

An interesting phenomenon that is apparent in the sediment loss data is that sediment loss peaks around 80 mm of calculated rainfall in the low cover treatments and then drops again at higher rainfall amounts. It is possible that there is a critical rainfall threshold after which sediment loss drops due to soil saturation or the exhaustion of easily eroded sediments (Boix-Fayos et al. 2006, Cerdan et al. 2010). This pattern of increasing sediment loss to a peak and then declining as rainfall amount increases has been observed for individual rainfall events (Liu et al. 2015).

The two rainfall events that correspond with peak sediment loss occurred on July 8, 2016 (82.2 mm) and August 18, 2016 (86.8 mm). These dates are in the middle of drought conditions when the soil appeared very dry and loose. The two rainfall events that were larger than those associated with the peak, occurred on October 6, 2015 (127.9 mm) and November 24, 2015 (146.2 mm) when soil conditions were more saturated. The peak seen in sediment loss for the low cover treatments is likely caused by the dry soil conditions at the time favouring sediment loss rather than representing a true relationship between rainfall and sediment loss for bare soil (Nunes et al. 2011, Turnbull et al. 2013). This hypothesis is further supported by a similar size rainfall event (77.6 mm) that occurred on May 7, 2016 (before the drought began), that produced much lower sediment loss. Rainfall intensities during the two summer rainfalls with peak sediment loss are unknown but if they were high, as often happens in summer storms, then intense rainfall coupled with dry conditions would have also contributed to the very high sediment loss observed (Nunes et al. 2011).

Factors Affecting Sediment Loss

Sediment loss from natural rainfall was significantly affected by treatment, block, and rainfall amount. The effect of rainfall amount is the weak positive relationship between sediment loss and rainfall. However, this relationship only exists for the low cover treatments because the high cover treatments keep sediment loss consistently low at all rainfall amounts. The significant positive relationship between sediment loss and runoff for the CWEs also appears to be driven by the low cover treatments (Appendix Figure 18). This indicates that bare soil is more susceptible to the erosive effects of rainfall and runoff when compared to soil that is protected by mulch cover.

The results of an ANCOVA that included runoff (L/m²), treatment, block, plot slope (%), experiment, and treatment-experiment interaction as factors, found that runoff was the only factor to have a significant effect on sediment loss from the CWEs (Appendix Table 32). It is expected that runoff and sediment loss would be correlated since the runoff is one of the key transport mechanisms for sediment (Fox and Bryan 1999). However, it was not expected that treatment would not have a significant effect on sediment loss in the CWEs considering the significant reduction in sediment loss from the high cover treatments seen for natural rainfall (Figure 9).

The results of a linear mixed effects analysis produced p-values that indicated a significant difference in sediment loss from some treatments and blocks for the CWEs (Appendix Table 35). When runoff was excluded from the ANCOVA, treatment and block were shown to have a significant effect on sediment loss (Appendix Table 35). Given that runoff is significantly affected by treatment for the CWEs, it makes sense that this nested significance of runoff and treatment on sediment loss was observed. The

results from natural rainfall and CWEs indicate that the low cover treatments had significantly more sediment loss than the high cover treatments, which supports the hypothesis that the high cover treatments are mitigating erosion.

The high cover treatments are likely reducing sediment loss by intercepting raindrops and reducing their ability to mobilize sediment through splash loss (Fu et al. 2011). This should be especially important at the experimental site because the mean slope gradient of the primary plots (63.5%) is very close to the slope gradient at which sediment loss by splash and interrill erosion peaks (58%) (Fu et al. 2011). The interception of raindrops not only reduces erosion by reducing sediment mobilization but also by reducing raindrop impacts on overland flow. Raindrop impacts on overland flow causes turbulence and increases the sediment transport capacity of the flow, ultimately resulting in increased erosion (Fox and Bryan 1999). The result of decreased raindrop energy reaching the soil surface in the high cover treatments is decreased sediment detachment and transport and ultimately the observed reduction in sediment loss.

Sediment loss from the Plant treatment was intermediate between the Control and high cover treatments for both natural rainfall and the CWEs (Figure 9 and Figure 14). While this was not significant it does suggest that there is a small effect of the Plant treatment on sediment loss. Especially since more sediment loss could be expected from the Plant treatment compared to the Control plots due to soil disturbance during planting.

Plant patches, such as those created in the Plant treatment, are known to change the soil beneath them by increasing organic matter content, infiltration rates and surface roughness, and decreasing bulk density, all of which reduce runoff and sediment loss (Bochet et al. 2000, Vásquez-Méndez et al. 2010). Plant patches are also known to alter

overland flow patterns creating excess runoff and increased erosion in the spaces between the plants (Abrahams et al. 1995). These opposing effects of plant patches may have resulted in the lack of significant effect of the Plant treatment in this study.

The plants in the Plant treatment were also likely neither mature nor dense enough to have a significant effect on runoff and sediment loss. As the plants mature their ability to reduce runoff and sediment loss increases (Garcia-Estringana et al. 2013). In the present study, data collection began only 1-2 months after plants were planted and continued until about 12-13 months post-planting. In a study by Garcia-Estringana et al. [2013], plantings were only considered established after 20-30 months and began to show significant reductions in runoff and soil loss at that time. The shrubs in that study did not have a significant effect on runoff or erosion during the establishment period (0-20(30) months post planting) (Garcia-Estringana et al. 2013). This supports the hypothesis that the Plant treatment is reducing runoff and erosion but the plants were not established long enough and were not dense enough for that effect to be significant.

Another possible reason the Plant treatment did not show a significant reduction in runoff and sediment loss is because the canopies of the plants in the Plant treatment covered no more than 25% of the plots on average (Figure 16). Vegetation begins to show significant effects on runoff and sediment loss when 30-40% cover is reached and optimal results occur with 70% vegetation cover (Garcia-Estringana et al. 2013). As the plants mature their canopy cover would increase providing further evidence that the establishment time provided during this experiment was not sufficient to observe significant runoff and erosion reduction by the Plant treatment.



Figure 16) Images of two Plant treatment plots showing lowest (A) and highest (B) amount of soil cover provided by the Plant treatment.

Block 3 had significantly more sediment loss than blocks 1 and 2 for natural rainfall and the CWEs. It is known that this is not due to a higher slope gradient causing increased runoff velocity and sediment transport because the slope of block 3 is not significantly different than blocks 1 and 2. This block effect was first thought to be due to overhanging vegetation intercepting and deflecting raindrops before they entered the plot. However, there can be no effect of the overhanging vegetation in the CWEs because the water was applied directly to the plot and never came in contact with vegetation outside the plot. This led to the investigation of other intrinsic factors within block 3 that could cause increased sediment loss.

Increased sediment loss in block 3 could be due to a higher soil bulk density, assuming that the bulk density from the plots measured are representative of the other plots in the blocks (Table 2). If bulk density is higher in block 3 then that would reduce the pore space available for water infiltration and soil suction, resulting in more runoff and higher runoff velocities that have the ability to transport higher volumes of sediment resulting in higher values for sediment loss in block 3 (Fox and Bryan 1999, Vilayvong et al. 2016, USDA 2017).

Although, no study was found that looked directly at the relationship between bulk density and erosion there is some indication that a small amount of increase in bulk density may result in a disproportionate decrease in infiltration capacity (Halvorson et al. 2003). The implications of this being a disproportionate increase in runoff amount and velocity, which increases sediment transport capacity of the runoff and could ultimately increase sediment loss at higher bulk densities. However, there is a similar variation seen in the bulk densities of the secondary experimental plots with no corresponding block effect suggesting that bulk density may not be the cause of the block effect either.

It is possible that the soil was more disturbed in block 3 when the plots were installed because the block's higher position on the slope relative to the road made it more difficult to work on that part of the slope during plot construction. The disturbance during construction could have resulted in looser soil that was more easily eroded resulting in higher levels of sediment loss, although Ollesch and Vacca [2002] did not find a significant effect of plot installation on sediment loss in their study. However, the plots in that study were much longer (10 m) and had lower slope gradients (12.2% - 46.6%) than those used in this study (Ollesch and Vacca 2002). If the increased erosion in

block 3 were due to plot installation it would be expected that the increased erosion would have been seen at the beginning of monitoring and then decreased with time, which was not the case. It is not possible to say for certain from the data obtained during this experiment what the true cause of the block effect was.

Sediment Loss by Particle Size

Size Classes in Eroded Sediment from Natural Rainfall

Soil at the experimental site is made up of a mixture of different sized sediment particles. Detachment and transport of soil particles tends to decrease as grain size increases (Poesen and Savat 1981). In order to determine if there was differential erosion for different sized soil particles in this study, the eroded sediment from natural rainfall events was divided into three size classes. The largest size class was cobbles. These were not included in any other analyses because they only occurred very occasionally so were not representative of erosion in general and their disproportionately large weights would skew the overall erosion results. There was no significant effect of factors tested on cobble erosion, which is not surprising due to their low sample size and sporadic nature of erosion. However, the fact that the majority of cobbles, by number and weight, came from the low cover treatments does suggest that it is possible that the high cover treatments are reducing cobble erosion.

Pebbles and granules made up the next largest sediment size class. Even though there was no significant effect of any of the factors investigated on pebble/granule erosion, the mean weights of pebbles and granules in eroded sediments from the low cover treatments are much higher than from the high cover treatments (Appendix Table

23). This indicates that the high cover treatments are having some effect on the erosion of pebbles and granules. Since it is unlikely that raindrop impacts, except maybe in the most extreme cases, are able move pebbles and granules it is likely that a combination of overland flow, scouring, and gravity are the agents facilitating pebble and granule erosion. Reduced overland flow velocity due to increased surface roughness in the high cover treatments probably contributes to the reduction in pebble and granule erosion (Bhattarai et al. 2011). The Plant treatment showed pebble and granule loss that was intermediate between the Control plots and the high cover treatments. Even though this was not significant it supports the conclusion that the Plant treatment reduced sediment loss but was not sufficiently mature to produce significant results.

Another factor that likely plays a role in increasing pebble and granule erosion in the low cover plots is frost heave and ice crystal formation (Gatto 2000, Personal observation 2015, Barnes et al. 2016). During data collection in late fall 2015 it was noted that ice crystals had formed on the surface of the bare soil in the low cover treatments and on the top of the ice crystals were relatively large sediment particles (Figure 17). Some movement of sediment would occur when those crystals melted, representing the heave mechanism of soil creep, but the biggest effect would be the loosening of the larger sediment particles at the surface making them more susceptible to movement by overland flow and gravity during subsequent rainfalls (Young 1972, Gatto 2000, Barnes et al. 2016). Ice crystal formation was not seen in the high cover treatments during data collection.



Figure 17) Example of sediment on top of ice crystal formation on the surface of exposed soil in Plant treatment taken November 30, 2015.

Rainfall amount and day had significant within plot effects on pebble/granule erosion. This agrees with research summarized in Parsons and Stone [2006] where researchers found that coarse material (>2mm) is eroded at higher proportions during the highest intensity rain events and that eroded sediment tends to become coarser as rainfall duration increases. The pebble/granule component in the present study corresponds to sediments greater than 2 mm, which increase with increasing rainfall duration and intensity both of which would contribute to large calculated rainfall amounts explaining the significance of calculated rainfall. The day effect is likely more pebbles and granules eroding on days that had high intensity and/or long duration rain events.

Sand erosion from natural rainfall experienced basically the same factor effects as overall sediment loss. The effect of calculated rainfall is likely the increase in sand erosion (g/m^2) as calculated rainfall (mm) increased. The high cover treatments

significantly reduced sand erosion compared to the low cover treatments due to the raindrop interception, runoff alteration, and frost prevention effects described above. The sand component appears to be determining the overall relationship between sediment loss and the factors investigated. Further investigation shows that when cobbles and organic matter are excluded, sand made up the majority of eroded sediment (75%) by weight with pebbles and granules contributing the other 25%, thus it is not surprising that sand erosion is representative of the factors affecting overall sediment loss.

Grain Size Proportions in Eroded Sediment from Controlled Watering Experiments

The two broad classes of eroded sediment defined by Gradistat were Gravel and Sand. Overall, the proportion of gravel in eroded sediment was significantly higher in the low cover treatments, block 3, and CWE 2 while the proportion of Sand in eroded sediment was higher from CWE 1 and was not significantly affected by any other factors. A higher proportion of gravel is seen in sediment eroded from the low cover treatments because the soil surface in the low cover treatments is directly exposed to the water energy compared to high cover treatments where some of that energy is intercepted and dissipated by the mulch cover. More water energy at the surface increases the amount, velocity, and turbulence of the runoff contributing to an increased ability to detach and transport the larger gravel particles, resulting in higher proportions of gravel erosion (Hogarth et al. 2004, Parsons and Stone 2006, Leys et al. 2010). This is consistent with the trend seen for pebble and granule component weights from natural rainfall.

Whatever factor is causing the significant difference in block 3 is acting on the large gravel particles and not the smaller sand particles. This is more evidence against the

hypothesis of soil bulk density because bulk density should affect all the soil particles equally. Interestingly, the significant effect of block on the gravel proportion and no significant effect of block on the sand proportion from the CWEs is the opposite of what was found for pebble/granule and sand erosion by weight from natural rainfall. There is no clear explanation for these contradictory results.

A significantly higher proportion of gravel eroded during CWE 2 when the soil moisture was low. This is consistent with results of a simulation experiment by Vilayvong et al. [2016] who found that high antecedent soil moisture content resulted in the preferential erosion finer sediments while low antecedent soil moisture content favoured the erosion of coarser sediment particles. The significantly higher proportion of Sand in eroded sediment from CWE 1, when antecedent soil moisture was high, also supports the antecedent soil moisture explanation.

The Gravel sediment class was further divided into three subclasses during Gradistat analysis. The largest particles were classified as very coarse gravel (VCG), the next size class was fine gravel (FG), and the smallest class of gravel was very fine gravel (VFG). The proportions of all of the gravel subclasses were significantly higher in eroded sediment from block 3, showing that the factor that is causing the block effect is acting on all sizes of gravel and interestingly not on any of the subclasses of sand. The proportion of VCG was not significantly affected by treatment or experiment. This is likely due to the fact that the proportion of VCG in eroded sediment was very low overall suggesting that neither the application intensity nor runoff velocity needed to detach and transport VCG were reached in the CWEs.

The proportion of FG and VFG were both significantly lower in sediment eroded from the high cover treatments compared to the low cover treatments indicating that application intensity and runoff velocity were sufficient to detach and transport those size particles and that the high cover treatments were able to reduce the water energy enough to significantly reduce FG and VFG erosion. VFG was the only size class of gravel whose proportion was significantly different between the CWEs with a higher proportion of VFG in eroded sediment from CWE 2. The proportions of the other two subclasses of gravel were also higher in eroded sediment from CWE 2 but this was not significant (Appendix Table 37). For the largest gravel particles there is likely a trade off between the low antecedent soil moisture in CWE 2 favouring larger particle erosion and the higher intensity in CWE 1 favouring the detachment of the larger particles (Parsons and Stone 2006, Vilayvong et al. 2016).

The Sand sediment class was divided into five subclasses during Gradistat analysis. In order from largest to smallest sized particles those subclasses were; very coarse sand (VCS), coarse sand (CS), medium sand (MS), fine sand (FS), and very fine sand (VFS). Overall, the sand subclasses' proportions did not behave the same way as the Sand class proportion. The FS subclass was very different from the other subclasses and the only subclass that reflected the same significant effects as the Sand class as a whole. It is unclear from the data collected why this is the case. The only effect that was consistent for the Sand class and all its subclasses was a notable lack of a significant block effect, which was significant in most of the sediment analyses including sand erosion by weight.

Proportions of VCS, CS, MS, and VFS were all significantly higher in eroded sediments from CWE 2. The dry soil conditions at the time of CWE 2 likely favoured the

erosion of the three coarser sand subclasses (Vilayvong et al. 2016). The higher proportion of VFS in CWE 2 is contrary to the results of Vilayvong et al. [2016] who suggest that fine sediments should be preferentially eroded when antecedent soil moisture is high. However, soils with very fine textures have been classified as less erodible than intermediate soil textures due to their cohesiveness and perhaps the high antecedent soil moisture during CWE 1 increased the cohesion of the VFS sufficiently to produce a significantly smaller proportion of VFS (Cerdan et al. 2010). Unlike the other sand subclasses, the proportion of FS was significantly higher in CWE 1 that was conducted when soil moisture was high, which agrees with the results of Vilayvong et al. [2016] who found that high antecedent moisture content resulted in the preferential erosion of fine sediment.

The proportions of VCS, CS, and VFS were significantly smaller in sediment eroded from the high cover treatments compared to the low cover treatments. This indicates that the high cover treatments were able to significantly reduce detachment and transport of these sediment classes. Treatment also had a significant effect on the proportion of MS in eroded sediment however only the Brush treatment produced significantly lower proportions of MS compared to the low cover treatments. The proportions of MS in eroded sediment from the other treatments were statistically the same. An ANCOVA showed no significant effect of treatment on the proportion of FS, however based on the results of a linear mixed effects analysis it was shown that eroded sediment from the Hay treatment had a significantly higher proportion of FS compared to the other treatments.

Even though it was not significant, eroded sediments from all of the high cover treatments, except Brush for MS, had higher proportions of medium and fine sand than the low cover treatments (Appendix Table 37). The medium and fine sand components must be affected by factors such as soil moisture and treatment cover differently than the other grain sizes. Medium and fine sand could be considered intermediate sized particles because they are smaller than gravels and coarse sands and larger than silts and clays, this is important because intermediate sediments are more easily eroded than coarse and very fine sediments (Cerdan et al. 2010). It is possible that the proportion of medium and fine sand in eroded sediment is higher in the high cover treatments because the water droplet energy that was able to penetrate and the flow velocity of runoff reached under the high cover treatments was only sufficient to detach and transport the easily eroded medium and fine sands but not the larger and very fine sediment particles. It is important to keep in mind that only very small amounts of those medium and fine sands were actually removed from the high cover plots they were just removed preferentially to the larger sediments and VFS within those treatments.

Aside from medium and fine sand, the high cover treatments consistently produced significantly smaller proportions of sediment from the CWEs. Even though the proportions of medium and fine sand may have been larger in eroded sediment from the high cover treatments, the overall amount of sediment was greatly reduced. This is important because it means that the high cover treatments will be useful to reduce erosion in all soil textures.

Profile Gauge

The profile gauge was used to take point scale measurements of the soil surface level (cm) at 33 points within each plot (11 points each top, middle and bottom). The difference between the initial surface level and the surface level at the last measurement is the net change in soil surface level (cm) for the experiment. Treatment, position, block, and the treatment-position interaction all had significant effects on the net soil surface level change (cm) of the plots.

The mean net soil surface level change was negative for all of the treatments and Control indicating that they are all experiencing some erosion. The profile gauge data supports the findings of the sediment loss data, indicating that the high cover treatments experienced significantly less erosion than the Plant treatment but surprisingly no difference was found between the high cover treatments and the Control plots. It is very likely that the lack of significance seen for the Control plots is due to lack of statistical strength from only having three Control replicates compared to six replicates of each of the treatments and not a true lack of difference between the Control and the high cover treatments.

The profile gauge measurements showed that erosion was significantly higher at the top (position 1) of the plots when compared to the bottom (position 3) and while not significant, the middle position (2) showed erosion that was intermediate between the top and bottom of the plots (Figure 12). This indicates that some of the sediment eroded from the top of the plots is deposited as it is carried down the slope by overland flow or splashed downslope by raindrop impacts (Meshesha et al. 2016). This result also suggests that more sediment is deposited as it travels further down the plot, which is consistent

with the results of Leys et al. [2010], that show mulching is more effective for reducing sediment loss at longer plot lengths.

There may be some effect caused by the bounded plots in the primary experiment, which do not allow any sediment to enter the top of the plot and thereby exacerbate erosion at the top position (Boix-Fayos et al. 2006). However, the secondary experimental plots were open at the top allowing sediment to enter from above and reducing some of this boundary effect. When the open secondary plots were analysed alone there was no significant effect of position but the Brush and All treatments still show the trend of the most erosion occurring at the top and the least amount of erosion occurring at the bottom. The Hay treatment on the secondary experimental slope also experienced the least erosion at the bottom but the most erosion in the middle. This a further indication that the high cover treatments are capturing or increasing the deposition of sediment as it moves down the slope resulting in the least erosion at the bottom of the plots. The Plant treatments on the primary and secondary experimental slopes exhibited an erosion pattern that was different than the other treatments and the Control, with a large amount of erosion occurring at the bottom of the plots and the least in the middle. Plant communities with large spaces between individuals, which was the case for the Plant treatment, tend to show altered overland flow and erosion patterns and high erosion rates in the areas between plants (Abrahams et al. 1995). This could explain the marked difference in the erosion pattern of the Plant treatment.

Primary experimental block 3 had more negative net change than blocks 1 and 2, which is consistent with block 3 experiencing more sediment loss throughout the

sediment loss data. The profile gauge data does not provide any additional information that would enable a determination of the cause of the block effect.

There was a significant effect of the interaction between treatment and position on soil surface net change. The most striking result is that position 3 in the All treatment is the only location within the plots that is experiencing net accretion. Given the overall significant difference between the Plant treatment and the high cover treatments, it is expected that comparing soil level net change at the same position between the treatments would show some significant differences. As anticipated, the Plant treatment experienced significantly more erosion than the Hay and All treatments at the bottom position. This is likely due to reduced overland flow velocities in the Hay and All treatments resulting in increased sediment deposition at the bottom of the plots containing those treatments (Leys et al. 2010). There could also be an effect of decreased splash loss from raindrop impact at the bottom of the Hay and All treatments. The top position in the Plant treatment also experiences significantly more erosion than the top of the Hay treatment. This is likely due to the protective cover of the Hay treatment reducing splash loss and sediment mobilization at the top of the plots (Fu et al. 2011). This is also an indication that the reduction in sediment loss by the Hay treatment is occurring at all positions within the plot and is not only due to sediment getting trapped or deposited as it moves down slope.

The All treatment is the only treatment that shows a significant difference between positions within the same treatment. The net accretion at the bottom position is significantly different than the net erosion seen at top and middle positions within the All treatment. This suggests that erosion is consistent across the plots within the other

treatments and that the All treatment has an enhanced ability to capture sediment and/or increase sediment deposition at the bottom of the plot.

Plant Survival

Plant survival was higher on the southeast-facing secondary experimental slope (74%) than it was on the northwest-facing primary experimental slope (40%). This may be due to more sun exposure on the southeast-facing slope promoting the growth of the sun-loving species used. Based on the results of other similar planting projects completed by Helping Nature Heal, the survival rates of the native species transplants planted into the All and Plant treatments were lower than expected, 55% and 59% respectively. The low rate of plant survival at the experimental site was probably due to planting in the heat of the summer, under dry conditions, and with minimal supplemental watering because of the remote location of the site. The poor soil quality at the site also likely played a role in lowering survival rates. A higher rate of survival was expected in the All treatment due to the protective and water retaining function of the hay mulch and woven brush, however there was no significant difference between the survival rates. It is possible that there was not enough time to see the true value of the protective capacity of the hay mulch and brush in the All treatment. The plants in the Plant treatment did not appear as healthy as the plants in the All treatment in the second year (Figure 18). There was also a lot of scour erosion observed around the plants in the Plant treatment that left roots exposed, this was not seen in the All treatment (Figure 18). It is possible that the length of the study period was not long enough to observe a difference in plant survival and that over a

longer time period root exposure and water stress would cause higher mortality in the Plant treatment.



Figure 18) Comparison of plant health between the All (A) and Plant (B) treatments on June 30, 2016. C) Exposed roots of plants in the Plant treatment on April 18, 2016 after winter erosion.

Conclusion

The high cover techniques developed and used by Helping Nature Heal to manage soil erosion significantly reduce sediment loss from excessively steep slopes in the immediate and short-term. This reduction of sediment loss is consistent across all sediment size classes and across the length of the slope. None of the treatments completely stop erosion and this is good because in coastal environments where they are primarily used, there needs to be some sediment transfer in order to maintain important coastal processes and healthy coastal environments (Labrière et al. 2015, Brakenridge et al. 2017). Based on the results of this experiment, the hay mulch, woven brush, and all

combined treatments are recommended for immediate soil protection and erosion reduction on slopes. Based on the probability of decay of the Hay and Brush treatments and potential for high amounts of soil cover to be obtained as plants mature, the All treatment, a combination of native plants, hay mulch, and woven brush could be recommended for medium- to long-term stabilization, erosion reduction, and revegetation of eroding slopes.

A variety of factors including rainfall variables and soil conditions contribute to runoff and sediment loss in sloped environments like the one studied here (Cerdan et al. 2010). The only aspect of rainfall that was accurately measured in this study was rainfall amount. Rainfall amount had a small effect on both runoff and sediment loss in this study with both increasing as rainfall amount increased. However, collection limitations resulted in a saturation relationship between rainfall and runoff when an exponential relationship was expected. Runoff from natural rainfall was not strongly correlated with any of the factors investigated. Antecedent soil moisture content, while not quantitatively measured in this study appears to have a large effect of runoff generation. Sediment loss increased with increasing rainfall only in the low cover treatments, the high cover treatments reduced sediment loss to a low level at all rainfall amounts recorded. Overall, runoff was largely determined by factors other than rainfall amount and soil cover was a better predictor sediment loss than rainfall amount.

Living vegetation and mulch have the ability to alter the energy of rainfall hitting the soil surface and change soil conditions in ways that reduce runoff and sediment loss thereby mitigating their negative effects on terrestrial and aquatic ecosystems (Vásquez-Méndez et al. 2010, Anh et al. 2014, Zhang et al. 2015). Runoff from natural rainfall did

not respond as expected to the treatments used in this experiment and it may be that the high and low cover treatments have opposing effects on factors affecting runoff generation such as soil moisture, resulting in a lack of significant difference in runoff between the treatments. When the variability associated with natural rainfall was eliminated in the controlled watering experiments (CWEs) the high cover treatments did significantly reduce runoff as expected. As expected, sediment loss was greatly reduced by the high cover treatments for natural rainfall and the CWEs. This effect of treatment was consistent for all the components of sediment even though the proportions of the most easily eroded size classes were higher from the high cover treatments. This broad effect of the high cover treatments across all sediment size classes is important because it supports the idea that these treatments can work in most environments and on most types of soil.

Part of erosion reduction effect of vegetation and mulch is its ability to increase deposition by slowing down runoff and capture soil particles as they move down slope (Smets et al. 2008, Robichaud et al. 2013). The high cover treatments were more effective at capturing sediment than the low cover treatments. The All treatment was the most effective at capturing down slope moving sediment and was the only treatment that was able to accrete sediment at the bottom of the plot.

The question of which treatment was the most effective at reducing erosion is more difficult to answer than the other questions asked at the beginning of this experiment. This is because the definition of effectiveness is contingent upon the time scale at which it is being evaluated. The immediate and short-term effectiveness of the treatments were evaluated in this experiment. Results show that the high cover treatments

are all equally effective for immediate and short-term erosion reduction. The Plant treatment was the least effective of all the treatments at reducing erosion immediately and in the short-term.

In the Living Shoreline projects where these techniques are used, effectiveness is defined by the immediate and short-term reduction in erosion rates as well as medium- to long-term erosion reduction and stability. The one-year duration of this experiment was too short to definitively say which of the high cover treatments would be the most effective in the medium- and long-term. However, the All treatment would likely be the most effective over the long term since the Brush and Hay treatments were already starting to decompose and lose their soil coverage after only one year, while the All treatment still provided high amounts of coverage. After one year the Hay treatment was also showing signs that it was being washed down the slope leaving areas at the top of the plots bare, this was not observed in the All treatment. By the time the hay and brush used in the All treatment were largely degraded, the plants should be large and mature and able to provide high percentages of soil coverage thereby maintaining high levels of soil protection over the long term. This is what has been observed in the Living Shoreline projects where these treatments have been used and had the opportunity to mature.

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Chapter 3: Monitoring Living Shorelines to Evaluate Stability and Resilience

Abstract

Erosion is a natural part of the evolution of coastlines but also poses a risk to people and infrastructure. Unconsolidated shorelines are particularly susceptible to the negative environmental impact of hard structures used for erosion management and are also the most suitable for ecosystem based erosion management such as Living Shorelines. Living Shorelines of varying ages, built on unconsolidated coasts in Nova Scotia were monitored to assess their stability and resilience. The suitability of the monitoring methods for an industry based monitoring program was also assessed. Areas of the Living Shorelines that had high amounts of vegetation cover experienced relatively little change indicating stability. Beach levels lowered and dune toe vegetation receded during winter and showed indications of recovery the following summer demonstrating possible resilience. The monitoring methods were relatively simple, low cost, and provided reasonably accurate results making them suitable for an industry based monitoring program.

Introduction

Diversity of Coastal Environments

Coastal environments include a large diversity of landforms and habitats based on geology, geomorphology, hydrology, exposure, and the interaction between these factors. In very broad terms coastal landforms can be formed by unconsolidated or overconsolidated (eg. glacial till) sediment or rock. Shorelines formed from sediments include beaches, dunes, and unconsolidated cliffs and bluffs (Davidson-Arnott 2010). Rocky or hard shorelines include rocky headlands, areas of exposed bedrock, and stone cliffs (Davidson-Arnott 2010). The amount of water energy a coastline receives depends largely on its exposure to the ocean. Open coasts are exposed to the open ocean and subjected to high-energy waves and currents. Sheltered coasts experience lower energy waves and currents because they are protected from the open ocean by features like headlands, sand bars, reefs or islands. Sheltered, low energy coastlines tend to host a higher diversity of ecosystems and organisms than open coasts likely due to less environmental stress from currents and wave activity (National Academy of Sciences 2006). Less is known about how processes like erosion and sedimentation function on sheltered coastlines because the interaction between the hydrology, geomorphology, and biology is more complex than on open coasts (National Academy of Sciences 2006). For example Greenwood and Orford [2008] were surprised to discover that the sheltered coasts in their study experienced higher water levels than the open coasts during storm events, they hypothesized that this was due to wind and waves compressing large volumes of water into a small space.

On both open and sheltered coastlines, geomorphology plays a large role in determining the resistance and resilience of coastal features to erosion processes. Shorelines formed from unconsolidated sediments are generally less resistant to erosion processes while rocky shorelines are more resistant. Some coastal types are more resilient than others, for example beaches and dunes can naturally restore themselves after an erosive event given sufficient space and sediment sources while cliffs and bluffs cannot (Marchand et al. 2011). Rocky coastlines are resistant to erosion but not resilient because fine particles scoured from the rock surface cannot be returned. Regardless of their capacity to resist or recover from erosive events, all shorelines are dynamic evolving systems and permanent stability is not a reasonable expectation for any shoreline (Subramanian et al. 2006). The human ideal of a stable shoreline position is a misperception caused by the long time scales of shoreline evolution.

Coastal Erosion

Shoreline evolution is primarily based on the opposing processes of erosion and accretion. While accretion is an important coastal process for building and maintaining many coastal features it does not get as much scientific, political or public attention as its counterpart, erosion. Part of the reason for this is that erosion is much more prevalent with 70% of sandy shorelines experiencing erosion and only 10% accreting (Bird 1987). The other part of the reason is that the risk of damage to coastal properties and development is increasing as climate change and sea level rise accelerate erosion rates.

Coastal erosion is the permanent loss of sand and sediment from the coastal system (van Rijn 2011). Erosion and resilience of unconsolidated shorelines is highly

dependent on the balance between sediment entering the coastal system from sediment sources such as rivers and sediment being lost to the coastal environment when it is transported to sediment sinks such as offshore canyons (Bird 1987, Marchand et al. 2011, van Rijn 2011). The flow of sediment along the shoreline as it is transported from source to sink is also important for the maintenance of coastal features. An area of coastline that contains a full sediment cycle (source, transport, sink) is called a coastal sediment cell and can be used to define the sediment budget for that section of coastline (Marchand et al. 2011). Coastal sediment cells are a useful tool for coastal managers to determine their erosion risk, potential causes of erosion, and how best to manage the shoreline to reduce risk to people and disturbance to the environment.

Coastal erosion is a natural process that is essential to coastal evolution however it is often accelerated by human activities (Roberts 2006). These activities include; damming rivers and constructing hard structures that cut off sediment sources and exacerbate coastal erosion by creating a deficit in the sediment budget; destabilizing coastal sediment through inappropriate construction and recreation activities; interrupting sediment transport processes by building groynes, breakwaters, and artificial headlands that cut off the sediment supply for downdrift beaches and shorelines; and undertaking dredging and sand mining activities that decrease sediment supply and remove sediment from the coast thereby functioning as sediment sinks (Bird 1987). Coastal erosion is a complex process that is dependent on many factors; Bird [1987] described fourteen factors and mechanisms, anthropogenic and natural, for sandy shore erosion alone.

Erosion Processes

Coastal erosion processes and mechanisms depend largely on the geomorphology, hydrology, and exposure of the coast being studied (van Rijn 2011). This section will focus on erosion processes on soft coasts and the mechanisms of dune and cliff erosion specifically because those are the environments where coastal monitoring was conducted in this study. Monitoring was also conducted on coastal bluffs, which are cliffs less than 8 meters (van Proosdij et al. 2016). Coastal bluffs are exposed to the same erosive forces as cliffs and tend to exhibit similar erosion patterns (Davidson-Arnott and Langham 2000, Brown et al. 2005, Himmelstoss et al. 2006).

One factor that controls erosion rates for both dunes and cliffs is beach height and width. A high beach level reduces wave activity at the cliff or dune toe while when the beach level is low the frequency and height of waves reaching the cliff or dune toe is higher and therefore more erosion occurs (Lee 2008). Wider beaches reduce the wave height by exerting drag forces on the wave therefore a wider beach results in fewer waves with enough height and energy to reach the dune or cliff and cause erosion (Himmelstoss et al. 2006, Lee 2008, Gedan et al. 2011).

Dune Processes

Dunes and beaches are known to fluctuate seasonally in temperate regions as intense fall and winter storms transport large amounts of sediment out into the nearshore, which is then washed back in by gentler spring and summer wind and waves (Bird 1987, Lee 2008, Borsje et al., 2011). This process reflects the dynamic equilibrium of a resilient dune and beach that naturally recovers after erosion events, ideally resulting in very little

net change over time because erosion and accretion are balanced. Dune erosion is primarily driven by extreme high water events and the frequency and intensity of smaller events where water reaches above the toe and runs up the dune face removing sediment, which is then deposited onto the beach, in the nearshore or into a sediment sink (Suanez et al. 2011). Undercutting at the toe of dune systems by gentler wave activity that just reaches the toe causes destabilization of the dune face and erosion (Suanez et al. 2011). Erosion can also occur when extreme storm events push dune sediment or an entire section of dune into the backshore (Bird 1987). This type of erosion can reduce the resilience of the dune system because the sediment cannot be acted upon by the constructive summer waves and wind that rebuild the dune.

The mobility of dune sediment has been related to wind and wave energy, precipitation, soil moisture, vegetation coverage, and exposure to storms and storm surges (Levin et al. 2006). Wind is the main force moving sediment in a dune system, however in regions where high winds are also most often associated with high rainfall, as in Nova Scotia, the stabilizing effect of soil moisture can reduce the effect of wind transport (Arens et al. 2004). Dune recovery after an erosion event is largely due to loose dry sand from the upper beach being transported by wind up the dune face and deposited on the dune (Suanez et al. 2011). Vegetation is the last phase of dune recovery after an erosion event, providing stability and reducing sediment movement (Suanez et al. 2011).

Cliff Processes

Coastal cliffs are subject to erosive forces from both marine and terrestrial sources. The relative importance of these erosive forces on cliff recession depends largely

on the exposure of the cliff. Ocean waves and geology are the main factors determining cliff recession on open coasts while slope processes similar to those in terrestrial systems often control cliff recession in sheltered bays (Lee 2008). Generally speaking, the stability of coastal cliffs depends on surface water (rain and runoff), ground water (seepage), and ocean water (wave impacts) (Maryland DNR 2017). High water content from precipitation reduces negative pore water pressure reducing the cohesion of cliff sediments, seepage destabilizes the structure of the cliff, and wave erosion at the cliff toe creates unstable slope angles (Vandamme and Zou 2013). Any of these factors alone or in combination can result in cliff failure and crest recession.

Many cliffs are subject to a recession model of progressive change and episodic failure while others fail immediately as a result of high wave energy at the base (Lee 2008). The model of recession exhibited by a cliff is directly related to the cohesiveness of the sediments that make up the cliff. Collins and Sitar [2008] found that cliffs formed from weakly cemented sands (low cohesiveness) are very susceptible to wave activity at the base and as the slope at the base approaches vertical, failure of a small portion of the toe is initiated. These small failures at the base continue until the entire cliff reaches a near vertical slope then a mass failure causes crest recession and grading to a stable slope and the process begins again (Collins and Sitar 2008). Moderately cemented sands (moderate cohesiveness) tend to form near vertical cliffs that are more susceptible to the effects of precipitation infiltration and seepage than wave action (Collins and Sitar 2008). The high soil moisture content produced by precipitation and seepage destabilizes the outer layer of these cliffs and crest recession occurs when the entire cliff face shears off creating a new near vertical cliff face of lower water content sediment that is now able to

become saturated thereby restarting the cycle (Davidson-Arnott and Langham 2000, Collins and Sitar 2008).

Large cohesive glacial till cliffs experience episodic crest recession with the time between crest recession events dependant on the height of the cliff and the amount of material destabilized during the slump (Manson 2002, Brown et al. 2005, Himmelstoss et al. 2006). The recession cycle begins when a steep unstable cliff experiences a large slump (<20 m crest recession), the slump material protects the base of the cliff and wave erosion at the toe begins to steepen the base of the slump causing failure of small sections of the slump (Figure 1) (Sunamura 1977, Davidson-Arnott and Askin 1980, Brown et al. 2005). This pattern of slump steepening and small failures continues until the slump is completely eroded and the slope becomes a steep unstable cliff again, ready for the next major slump (Brown et al. 2005). On coastal drumlin cliffs greater than 10 meters high in Boston, similar to Site 3 in this study, as the slope steepens and the slump becomes very reduced, a phase of gully erosion caused by rain, runoff, and seepage becomes the dominant form of sediment loss and slope steepening (Figure 1) (Himmelstoss et al. 2006).

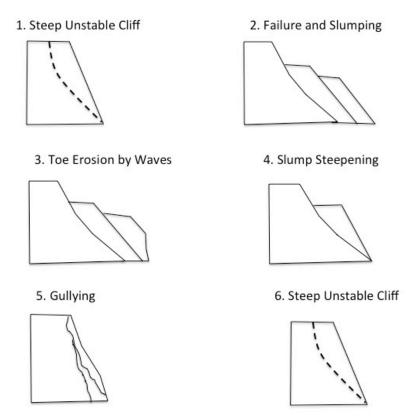


Figure 1) Cycle of erosion and episodic crest recession of a large (>10 meters) cohesive glacial till cliff. Adapted from Brown et al. [2005] and Himmelstoss et al. [2006].

In addition to sediment cohesion, cliff height also plays a role in determining erosion processes. Brown et al. [2005] found that the lag time between toe erosion and crest recession was much shorter for small cliffs (9-13 m) and they did not experience the same degree of slumping as large cliffs. Himmelstoss et al. [2006] noted that the gully erosion phase was not seen in cliffs less than 10 meters in height. Weathering by precipitation, runoff, and gravity were found to be the driving factors for small cohesive cliff erosion compared to erosion of large cliffs that was more driven by wave action (Greenwood and Orford 2008). The small bluffs in this study were on sheltered coasts

very similar to the small cliffs studied by Greenwood and Orford [2008]. In that study they found that seasonal precipitation was strongly correlated with erosion of small cliffs in low energy environments with most crest recession occurring in winter under saturated conditions (Greenwood and Orford 2008). Greenwood and Orford [2008] saw the most extreme erosion in their study during the late fall after a very dry summer followed by a large amount of early fall rain. The next most extreme erosion occurred after a long period of cold temperatures producing a large amount of frost activity. They determined that freeze-thaw cycles loosen the cohesive sediments that are then easily removed from the cliff face by precipitation and runoff (Greenwood and Orford 2008). Even for small cohesive cliffs where weathering is the dominant erosive force, the action of waves removing sediment from the bottom of the cliff is an important contributor to instability, leading to the next failure (Greenwood and Orford 2008). Recession rates of dunes and cliffs are expected to accelerate rapidly with sea level rise due to the increase in frequency, height, and intensity of waves reaching their toes (Himmelstoss et al. 2006).

Coastal Erosion in the Future

Coastal erosion and sediment budgets are strongly influenced by sea level rise (van Rijn 2011). As the sea level rises a larger coastal area is exposed to wave activity and therefore more sediment is needed to maintain coastal features. Increased sediment inputs from sediment sources are necessary if marshes and beaches are going to be able to keep up with sea level rise (SLR) (Marchand et al. 2011). As SLR exposes more of the coast to erosive forces accompanied by an increase in storm frequency and intensity due to climate change (IPCC 2007), sediment inputs to the coastal system should increase

giving beaches and marshes some of the sediment they need to adapt (Perkins et al. 2015). However, the typical human response to the increasing risk posed by erosion to the ever-increasing coastal population and associated development has been hard protection structures that restrict sediment sources and exacerbate erosion in other areas. This response can result in a positive feedback loop toward increasingly hardened shorelines and increasingly degraded coastal systems that can no longer provide critical ecosystem services such as coastal protection.

It is important to remember that coastal erosion is not strictly an environmental issue so much as a social issue that comes from an awareness of risk to property and/or infrastructure and unrealistic ideas about where the coast is 'supposed to be' (Ricketts 1986). Coastal erosion due to SLR is a "chronic and progressive" process that requires solutions that can adapt to changing conditions, unlike the hard structures currently being used (National Academy of Sciences 2006). Mangroves, salt marshes, and dunes have all been scientifically and politically recognized for their natural ability to protect the coast and keep up with sea level rise (Perkins et al. 2015). This represents an important shift in the collective perception of the value of natural coastlines and a shift in coastal management thinking.

Coastal Erosion Management

The new generation of coastal management strives to meet human needs, reduce costs, and lessen ecological impacts, ideally while improving habitat value for native species (Chapman and Underwood 2011). There is also an effort in many places to make coastal erosion management less localized and reactive by implementing regional

regulations and programs to guide coastal managers (Marchand et al. 2011). There is no question that as sea level rise continues so will the need and desire to protect coastal property, assets, and infrastructure (Chapman and Underwood 2011). The public, decision makers, and planners often perceive engineered hard structures, as stable, permanent, and technically advanced and therefore the best solution to coastal erosion issues (Donner and Webber 2014, Betzold and Mohamed 2016). However, recently there has been a realization that hard structures are negatively impacting the health and resilience of our coastline and have a disproportionately large negative impact on soft shorelines (Chapman and Underwood 2011).

Soft sheltered coastlines, which are the most ecologically diverse and provide critical habitat for many commercially important species, also happen to be particularly suitable for natural, ecosystem based forms of coastal management and erosion control (Roberts 2006). The Living Shoreline concept for ecosystem based coastal management and erosion control is to maintain coastal processes, provide habitat, adapt to changing conditions, and maximize functionality from both a human and ecological perspective (Smith 2006).

Projects that use living plants and plant residues to manage coastal erosion are often called Living Shorelines and this is the term used by Helping Nature Heal (HNH) to describe their signature style of coastal erosion management. A basic description of the techniques used by HNH to construct their Living Shoreline projects can be found in Chapter 2. When plants and plant materials are used to stabilize coastal cliffs or bluffs the term coastal bluff bioengineering is often applied. Many of the Living Shoreline projects constructed by HNH are designed with the aim of stabilizing and reducing erosion on

small coastal bluffs and can therefore be considered coastal bluff bioengineering. The All treatment in Chapter 2 was a very basic approximation of the techniques used in actual bluff stabilization. In coastal bluff stabilization projects there is more focus on toe protection using bulkier plant based materials (ie. logs and hay bales) to address wave activity. These techniques were not used in the experimental plots because they would have interfered with the sediment collection system.

In general, healthy dense vegetation cover is recognized as a critical factor for stable shorelines (Jones and Hanna 2004). Vegetation cover is commonly used as an indicator of the stability of coastal features, with higher percentages of vegetation cover indicating greater stability and less erosion potential (Randall 2004, Levin et al. 2008). The use of vegetation as a way to increase stability and control erosion has been investigated for both coastal dunes and bluffs. High vegetation cover is a strong indicator of dune stability and dunes can be successfully stabilized using appropriate plant materials (Duhring 2006, Levin et al. 2008). The opposite is also true with one study looking at remobilization of a stabilized dune through the removal of vegetation cover (Arens et al. 2004). One study generalized that once 50% vegetative cover is reached in a dune system sand movement becomes very low, mainly due to reduced wind energy at the sand surface (Levin et al. 2008).

Native vegetation and more specifically deeply rooted vegetation is recognized as important for bank stabilization and has been used to manage runoff-generated erosion (Duhring 2006). The deeply rooted native vegetation investigated by the Plant treatment in Chapter 2 (See Appendix Table 1) showed reduced erosion when compared to the Controls but this was not statistically significant likely due to insufficient time for the

vegetation to mature and low density of transplants resulting in low amounts of canopy cover. Indicators of slope stability were not assessed for any of the treatments in Chapter 2. Live willow cuttings (used in typical HNH Living Shoreline projects) in particular are recognized for their ability to stabilize slopes and dampen wave energy (Bischetti et al. 2010, Borsje et al. 2011). It is important to note that the erosion reduction potential of vegetation increases as the plants mature (Smets et al. 2008). Increased awareness of the immediate and long-term benefits of living plants has resulted in their increasing incorporation into coastal management plans and an increase in the number and size of Living Shoreline projects. Recent studies have begun to focus on quantifying the effectiveness of Living Shorelines for coastal protection but scientific data is still lacking (Gittman et al. 2014, Bilkovic et al. 2016).

Living Shoreline Effectiveness and Monitoring

Based on a review of the Proceedings of the Living Shoreline Summit [2006] it appears that up until that time the effectiveness of Living Shorelines was largely evaluated by photographic, anecdotal, and qualitative evidence. While there have been a few attempts at quantifying their effectiveness for shoreline protection (see Gittman et al. [2014] and Jones and Hanna [2004]), most of the scientific focus has been on evaluating their habitat value and effect on species diversity and abundance, especially comparing between Living Shorelines and hard structures (Bilkovic et al. 2016). No study in the literature reviewed here has attempted to quantify erosion reduction by Living Shorelines. This is likely because measuring and estimating erosion is very difficult in coastal systems because they are highly dynamic and have many different types of sediment

present (Jones and Hanna 2004). Another issue, which we encountered in this study, is a lack of baseline studies and measurements that are needed to accurately determine effectiveness (van Rijn 2011, Perkins et al. 2015). Aerial photos can and have been used to study long term coastal change where they exist but they are generally not suited to evaluating the relatively short time scale that Living Shorelines have been used (Manson 1999, del Pozo and Anfuso 2008).

From the literature it is clear that there is no standard for monitoring and evaluating the effectiveness of Living Shorelines. Indirect measurements of erosion potential and stability such as shear strength and soil moisture have been used to evaluate the effectiveness of Living Shoreline plantings on banks (Jones and Hanna 2004). Sediment level measurements by erosion pins have been used to estimate the effectiveness of plants and brush for reducing erosion in dune and beach environments (Stanley and Watt 1990, Levin et al. 2008). Estimates of vegetation density and percent cover have also been used to estimate sediment stability in coastal environments (Levin et al. 2008). LiDAR has emerged recently as another way to monitor and quantify cliff and bluff erosion (Day et al. 2012). LiDAR has the advantage of being able to map large areas with a higher accuracy than aerial photography. The technology used by Day et al. [2012] was accurate to 7 mm at 100 m, based on the manufacturer's calculated error. However, the LiDAR technology has difficulty accurately mapping when vegetation is present so is probably not useful for Living Shoreline research (Day et al. 2012).

Unfortunately, due to the issues encountered by many others including lack of baseline data, difficulties accurately measuring sediment loss in the coastal environment, and the episodic nature of cliff erosion, this study was not able to quantify erosion

reduction by HNH's Living Shoreline projects. Although a good quantitative approximation of erosion reduction potential of HNH's Living Shoreline techniques can be found in Chapter 2.

Background and Purpose

HNH's Living Shoreline techniques were first developed and installed on a coastal property about thirteen years ago. Since then the techniques have been adapted and refined as they have been installed on numerous properties around Nova Scotia and Prince Edward Island. The use of hay mulch, woven brush, and native plants to manage erosion is not unique to HNH but the way they are used in combination and how they are arranged and secured on the shoreline is very unique. The majority of HNH's Living Shoreline projects have been installed on small coastal bluffs but HNH has also installed Living Shorelines on one large cliff and one sand dune in Lunenburg County, Nova Scotia. Based on observations, photographs, and homeowner anecdotes, the Living Shorelines installed by HNH have been successful at stabilizing coastal slopes and appear to reduce erosion.

The purpose of this chapter is to empirically evaluate the effectiveness of a representative sample of HNH's Living Shorelines. In accord with other coastal monitoring programs vegetation density estimates, crest measurements, photographs, erosion pins, and Emery Method slope surveys were used to estimate the stability of four Living Shoreline projects (Jungerius and van der Meulen 1989, WHOI Sea Grant 2001, Levin et al. 2006, Taylor 2014). It was not possible to gather baseline measurements or have a control for these monitoring sites because the Living Shorelines were constructed

years prior to this study, they span the entire length of client's shorelines, and the neighbouring properties are often not representative of that shoreline. However, by comparing the observations made at these four Living Shoreline sites with typical models of beach fluctuation and coastal bluff erosion, it should be possible to make some inferences about the stabilizing effects and erosion reduction potential of Living Shorelines constructed by Helping Nature Heal.

An additional benefit of monitoring these Living Shorelines was the opportunity to test different methods for gathering erosion data on Living Shorelines. HNH is planning to implement a monitoring program for their Living Shoreline projects that would include gathering baseline erosion data for each site and then following the projects for three to five years after construction. One of the goals of this monitoring program is to quantify the reduction in sediment loss following the installation of a Living Shoreline. This type of monitoring is badly needed to fill a major gap in the scientific literature on Living Shorelines and will help HNH promote Living Shorelines to prospective clients. Information from this monitoring program will also help refine the design and construction processes and hopefully improve the performance of future Living Shoreline projects.

Methods

Living Shoreline Monitoring Sites

Four Living Shoreline projects designed and constructed by Helping Nature Heal (HNH) were chosen for monitoring based on the desire to represent a diversity of shorelines types and Living Shorelines at different stages of maturity. Year round

accessibility was also a major consideration for site selection. Sites 1 and 2 in Figure 7 are small coastal bluffs (1-3 m high) at the forest edge on sheltered, mixed sediment beaches. The Living Shorelines at Sites 1 (Figures 2 and 3) and 2 (Figure 4) experience low energy conditions and at the beginning of this study were 2 and 12 years old, respectively.

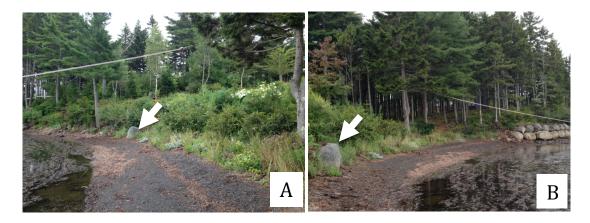


Figure 2) Images showing the eastern (A) and western (B) ends of Site 1. The same rock is indicated in both photos by an arrow.

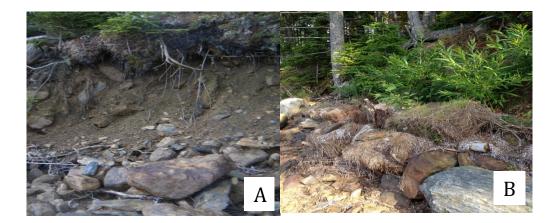


Figure 3) Images showing a section of the bluff at the eastern end of Site 1 before (A) and after (B) the Living Shoreline was installed.

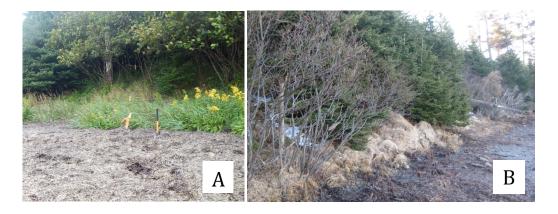


Figure 4) Images of the northern end (A) and central section (B) of Site 2.

Site 3 is a coastal cliff ranging in height from 13-29 meters; the cliff is cut into a glacial drumlin composed of Hartlen till (Taylor 2014) (Figure 5). This is a cohesive, clayey till similar to the cliffs studied by Brown et al. [2005] and Manson [2002]. Site 3 is on an open coast completely exposed to very high-energy ocean waves and currents (Figure 7). The Living Shorelines techniques began to be used at Site 3 eight years previous to the beginning of this study. The Living Shoreline was installed on top of the slump block (Figure 5C) in order to slow crest recession by revegetating and stabilizing the upper cliff while the slump material protects the toe. Seasonal maintenance and addition of material to the Living Shoreline at Site 3 continued from the time of installation up until two years before this study began.



Figure 5) Images of Site 3 taken at the base of the cliff at the western end looking toward the eastern end (A), from the top of the cliff at the gullied eastern end of the monitoring site (B), and at the eastern end of the slump block showing Living Shoreline treatments being installed (C). The arrow in image A shows the eastern end of the slump block.

Site 4 is a sand dune above the sandy beach in Kings Bay (Figures 6 and 7). Kings Bay faces the open ocean although it does receive some protection from Rose Point to the north and Enrage Point to the south. The southern corner of Kings Bay, where Site 4 is located, typically experiences lower intensity wave activity than the northern corner of the bay therefore Site 4 can be considered a moderate energy site, although it is very likely that it does experience high energy conditions on occasion (Personal observation). Site 4 did not receive the same combination of hay mulch, woven brush, and native plantings that the other sites received. Native beach pea (*Lathyrus japonicus* var. *maritimus*) and dune grass (*Ammophila brevigulata*) plantings were used to revegetate and stabilize the dune nine years prior to the beginning of this study. These plants were fed using kelp concentrate inoculated with mycorrhizal fungi seasonally during the first two growing seasons.



Figure 6) Image of the center section of Site 4.



Figure 7) Map showing approximate locations of the four sites used for Living Shoreline monitoring in Lunenburg County, Nova Scotia. Map data © 2017 Google.

Due to the large differences in geomorphology and erosion processes between the four sites, different measurement methods were used to evaluate stability and erosion at each of these sites.

Sites 1 and 2: Small Coastal Bluffs

Sites 1 and 2 were very similar in form and exposure so the same methods were used at both sites. Transects were installed at evenly spaced intervals along the bluffs depending on the length of the Living Shoreline. Site 1 had four transects and Site 2 had six. The transects were composed of three wooden survey stakes installed perpendicular to the waterline, which were used as benchmarks (Figure 8). For each transect benchmark (BM) stake 1 represented the crest of the bluff, BM stake 2 represented the toe of the bluff, and BM stake 3 was installed at the seaward edge of the beach vegetation, which approximately corresponded to the high water mark.

A modified version of the Emery Survey Method was used to determine the profile of each transect (Emery 1961, WHOI Sea Grant 2001). Each transect was divided into segments based on an observable change in the grade of the slope. The height and horizontal distance was measured for each segment as follows. Starting from BM stake 1, a meter stick was held vertically at the first observed change in the slope (See Figure 8). A measuring tape was used to measure the distance from the base of BM stake 1 to the meter stick; this measurement was recorded as horizontal distance (*d*). With the measuring tape held as horizontal as possible, the height on the meter stick where the tape intersected it was recorded as height (*h*). For the next measurement the base of the meter stick became the new starting point for measuring horizontal distance and the meter stick

was moved to the next location where the slope of the bluff changed. This continued for each segment until the waterline was reached.

Surveys were conducted seasonally from Fall 2015 until Fall 2016 at approximately low tide. The time of low tide and the time when the last segment of each transect was measured were recorded. The winter survey could not be conducted at Site 1 due to thick ice cover from the base of the bluff out into the cove. Photos and notes were taken at each transect during each survey to record vegetation cover and any other changes that may have occurred.

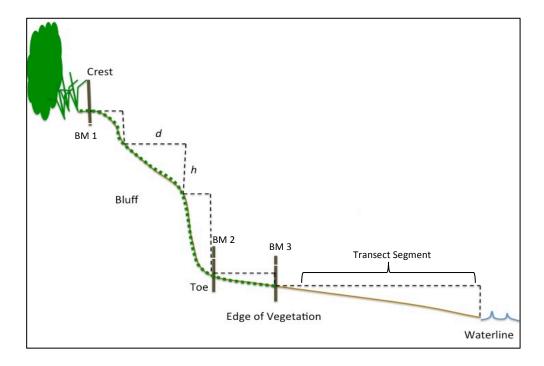


Figure 8) Idealized schematic of small coastal bluff showing the parts of the bluff and the locations of the benchmark (BM) stakes. Dashed lines represent how transect segments were surveyed using the Emery Survey Method to measure horizontal distance (d) and height (h). Not representative of scale or vegetation density.

Site 3: Coastal Cliff

Due to the large size and steep slope of the cliff at Site 3 and the dangers associated with working at height, crest and toe measurements were determined to be the safest, most practical way to measure crest recession and toe erosion at Site 3. Five transects were installed at equal distances across the section of the cliff crest that we had homeowner permission to work on, this happened to be the tallest section of the cliff. Transects 1 and 2 were above the steep gullied part of the cliff face while transects 3, 4, and 5 were above the slump block. The slope above the slump was not as steep as the gullied section of the cliff and no large gully formations were present.



Figure 9) Measuring crest transects at Site 3, February 2016.

The transects consisted of two wooden stakes installed perpendicular to the cliff crest and crest measurements were taken by measuring the distance from the first stake, across the second stake, and out to the cliff edge (Figure 9). Five transects perpendicular to the cliff toe that roughly corresponded to the locations of the crest measurements were

installed in the same way as crest transects. The initial crest and toe measurements were taken in the summer of 2015 and subsequent crest measurements were taken seasonally in 2016. The toe transects were lost between the initial survey and the winter 2016 survey due to the high wave energy experienced at the site. It was determined that due to the exposure of the site, this method of measuring toe erosion would not work so no attempt was made to replace the transects. Photographic evidence was used sparingly for this site because the scale of the site was too large to adequately capture and the crest vegetation was undercut quite drastically in many locations so in the interest of safety, the time spent close to the crest edge was minimized as much as possible.

Site 4: Sand Dune

At Site 4 erosion pin measurements, toe vegetation extent, and vegetation density estimates by quadrat sampling were used to quantify erosion and deposition on the dune. Four transects labelled A, B, C, and D were installed perpendicular to the water at equal distances across the section of dune that had been revegetated (Figure 10). Transect A was located at the southern extent of the study site and transect D was at the northern end of the site. The transects consisted of three wooden stakes placed at the back of the dune, the dune crest, and the edge of the dune toe vegetation. Each transect included three erosion pins one at the back dune stake, one at the dune toe stake, and one on the dune face. The erosion pin was installed on the dune face instead of next to the dune crest stake because the dune face is generally more dynamic (Levin et al. 2008). The erosion pins were 1-meter long zinc rods that were inserted into the dune by approximately three quarters of their length. Each pin was labelled with a letter and a number, the letter

corresponds to the transect the pin was located on and the number corresponds to its position on the dune. Pins in the back dune were given the number 1, the dune face pins were given the number 2, and the pins at the dune toe were given the number 3.

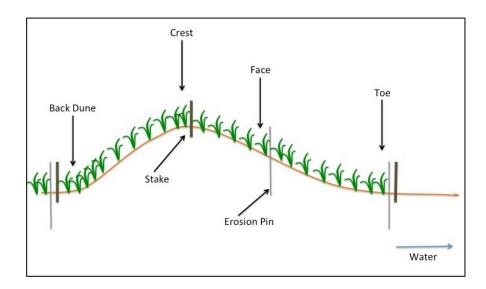


Figure 10) Idealized schematic of dune transect highlighting the parts of the dune and locations of stakes and erosion pins. Not representative of scale or actual vegetation density.

Erosion pin measurements were taken seasonally from Fall 2015 until Fall 2016; no measurements were taken in Winter 2016 due to snow cover and the loss of all dune toe stakes and pins. In Spring 2016 the dune toe stakes were reset using the distance recorded from the crest stake when they were originally installed. Erosion pins were also reinstalled at the same location and the Spring 2016 measurement was used as the new baseline. Erosion pin measurements were taken by placing a washer over the pin,

lowering the washer to the dune surface, and measuring the distance from the washer to the top of the pin (Takei et al. 1981).

The position of dune toe vegetation was measured as the distance from the vegetation to the toe stake. When toe stakes were initially installed in Fall 2015 they were placed at the very edge of the toe vegetation, this was considered the baseline and given a value of 0 meters. Any seaward movement of the vegetation line was given a positive value and any landward movement of the vegetation line was given a negative value relative to the baseline. Dune toe vegetation measurements were taken at the same times as the erosion pin measurements with one additional measurement in the Late Spring 2016 since it was thought that most vegetation growth and spreading would occur early in the growing season.

Visual vegetation cover (%) estimates were made within a square 1 m² quadrat around each erosion pin (Figure 13). The erosion pin was used as the centre point of the quadrat because it was hypothesized that sediment movement measured by the erosion pin might be correlated with the vegetation cover around the pin. Photographs were taken for each vegetation cover measurement so that field estimates could be reconfirmed if necessary. Vegetation cover estimates were made at the same times as erosion pin measurements. The number of plant species within each quadrat was also noted during the vegetation cover estimates.

Results

Sites 1 and 2: Small Coastal Bluffs

Overall, the transect profiles at Sites 1 and 2 showed very little change in the densely vegetated bluff faces and notable seasonal fluctuations in beach level below the mean high water mark (Appendix Figures 37-48). The seasonal fluctuation of beach level was also seen, although to a lesser extent, in the area between the mean high water and the bluff toe. The magnitude of the fluctuations appears, in some cases, to be exaggerated in the survey data when compared to transect photos (See Appendix Figures 37 and 38). The bluff profiles of each individual transect are generally very similar though time with two notable exceptions; the May 2016 profile of Site 1 transect 2 and the October 2016 profile of Site 2 transect 5 (Appendix Figures 37 and 46). The May 2016 profile of transect 3 at Site 2 also appears to drop more than it should in the lower sections. A review of the transect photos indicates that beach lowering was overestimated due to measurement error at the transects noted above during the surveys in question (Appendix Figures 38, 43, and 47).

At Site 1 the magnitude of beach fluctuation was lowest at transect 1 and increased as you move across the beach to transect 4 (Figure 11 and Appendix Figures 37, 39, and 40). Excluding the May 2016 survey at transect 2; the beach profile at Site 1 was lowest in November 2015 and highest in July 2016 except at transect 4 where July and October 2016 profiles showed the same beach level. Excluding the May 2016 survey at transect 2, no change occurred at the crest and upper face of the bluff at transects 1, 2 or 4. A small dip occurred near the bluff crest in the July 2016 profile of transect 3

(Appendix Figure 39). The vegetated transect segments at the bluff toe all showed an increase in elevation between November 2015 and October 2016 at Site 1.

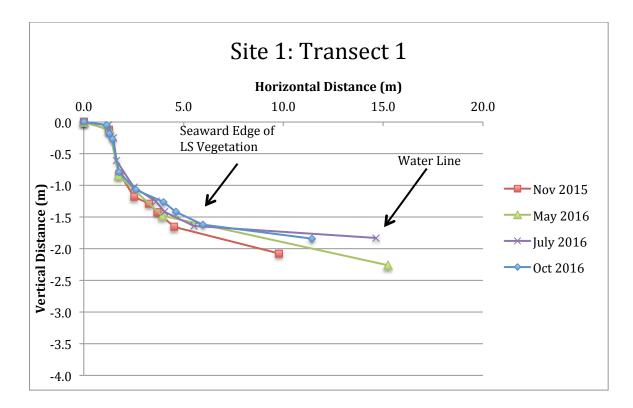


Figure 11) Site 1 seasonal bluff and beach profiles at transect 1 generated from survey data. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest. The seaward edge of Living Shoreline vegetation corresponds approximately to mean high water.

At Site 2 beach fluctuations were of a lower magnitude than at Site 1. Beach height fluctuated the least at transects 1 and 6 and otherwise fluctuation magnitude increased from transect 2 to transect 5 (Figure 12 and Appendix Figures 41, 42, 44, 46, and 48). Excluding the October 2016 profile of transect 5; the beach height at Site 2 was lowest in February and May 2016 except at transect 6, which had the lowest beach level

in October 2016. In general the beach height at Site 2 was highest in July 2016 although at transect 1 the beach level remained at the same height through to the October 2016 survey. Transect 4 had the highest beach level at the beginning of monitoring in November 2015 (Appendix Figure 44).

The profile of the vegetated bluff face at transects 1, 3, 5, and 6 remained consistent for the duration of monitoring at Site 2. The seasonal profiles of transect 2 show a decrease in the elevation of the mid to lower bluff face and the bluff toe between November 2015 and February 2016 (Figure 12). The profiles of transect 4 show a decrease in elevation of the lower bluff, bluff toe, and the area between the bluff toe and mean high water line over the same time frame (Appendix Figures 44 and 45).

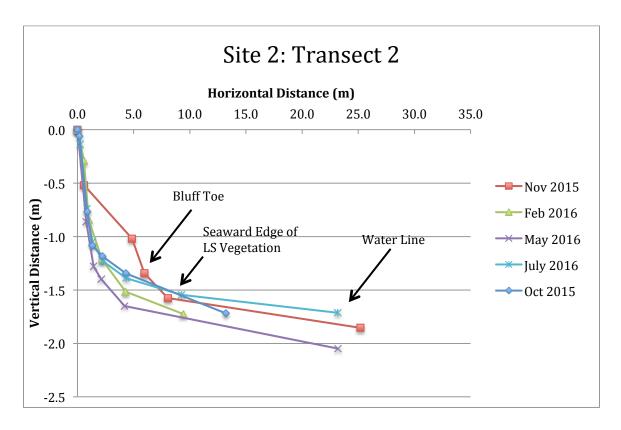


Figure 12) Site 2 seasonal bluff and beach profiles at transect 2 generated from survey data. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest. The seaward edge of Living Shoreline vegetation corresponds approximately to mean high water.

Site 3: Coastal Cliff

The average crest recession of the cliff at Site 3 was 0.13 meters over the 15 months measurements were taken. Transects 1 and 2 had the greatest recession at 0.20 and 0.27 meters over 15 months respectively (Table 1). Transect 3 showed very little recession and transect 5 showed no recession in 15 months of monitoring (Table 1). Since cliff recession is normally reported in the literature as an annual rate, the annual rate of cliff recession is reported here. The average annual (July 2015 – July 2016) rate of cliff recession measured by the transects was 0.08 m/a. There was a lot of variation in the

annual erosion rate between the different transects; transects 3 and 5 had no erosion between July 2015 and July 2016 while transects 1, 2, and 4 showed rates of 0.09 m/a, 0.25 m/a, and 0.08 m/a, respectively. There was no apparent seasonal trend in the total (15 month) or annual crest recession rates at Site 3 (Table 1). The February 2016 crest measurements were not used because they all showed that the crest had grown, which is not likely, although there could be soil expansion due to the frozen conditions.

Table 1) Seasonal and total cliff crest recession in meters measured at five transects at Site 3.

Transect	July 2015 – May 2016 (m)	May – July 2016 (m)	July – Oct 2016 (m)	Total (m)
1	0.09	0	0.11	0.20
2	0	0.25	0.02	0.27
3	0	0	0.05	0.05
4	0.04	0.04	0.04	0.12
5	0	0	0	0

Site 4: Sand Dune

Erosion pin data from twelve locations across the dune was used to calculate the seasonal, net, and absolute change of dune level. All pins except B:2 had a positive net change meaning all experienced sediment accretion (Table 2). The erosion pins at the dune toe were all lost over the winter so no data was gathered for accretion or erosion at that position between September 2015 and May 2016. All remaining erosion pins, except B:2, recorded sediment accretion over the winter (Table 2). Pin B:1 showed almost 10 cm of accretion between September 2015 and May 2016. However, given its position in the

back dune, dense vegetation around the pin, and a lack of apparent newly deposited sediment this was likely caused by something other than accretion. A few pins recorded very small amounts of net erosion during the summer but the majority showed accretion or no change (Table 2). Most pins also saw accretion or no change in the early fall with a couple of the pins showing small amounts of erosion. The largest accretion of sediment for any pin occurred in the early fall 2016 when the toe pin at transect C showed a 2.9 cm rise in sand level. The seasonal absolute change in dune sediment level for winter was 4.6 cm, spring and early summer was 4.1 cm, and late summer early fall was 6.0 cm. These values are very similar, however it is worth noting that the winter calculation is missing data from 5/12 erosion pins.

Table 2) Seasonal, net, and absolute change of the sand dune level, in centimeters, measured at twelve erosion pins on the revegetated dune at Site 4. *Value left out of seasonal absolute change calculation.

	Change in Dune Sediment Level (cm)					
	Sept 2015 –	May – Aug	Aug – Oct			
Pin	May 2016	2016	2016	Net	Absolute	
A:1	0.2	0.4	0.2	0.8	0.8	
A:2	0.4	0	0.6	1	1	
A:3		0.2	0.5	0.7	0.7	
B:1	9.8*	0.4	-0.4	9.8	10.2	
B:2	-1.4	-0.2	0.2	-1.4	1.8	
B:3		1.1	0.1	1.2	1.2	
C:1	0.2	0.4	0	0.6	0.6	
C:2	1.8	-0.1	-0.3	1.4	2.2	
C:3		0.5	2.9	3.4	3.4	
D:1	0	-0.1	0.5	0.4	0.6	
D:2	0.6	0	0	0.6	0.6	
D:3		0.7	0.3	1	1	
Absolute						
Change	4.6	4.1	6.0			

Estimated vegetation cover around the back dune and dune face pins was 100% for the entire monitoring period. Vegetation cover around the dune toe pins was about 50% when the pins were installed in September 2015 because the pins were placed at the edge of dune toe vegetation. By May 2016 the vegetation cover around the dune toe pins was between 0 and 1% due to recession of the dune toe vegetation. Vegetation cover increased over the 2016 monitoring season from 0 to 10% around pin B:3. The area around pins A:3 and D:3 maintained 0% vegetation cover over the monitoring period while vegetation cover around pin C:3 remained at 1%. An analysis of variance

(ANOVA) and a Tukey HSD test showed that the low percent vegetation cover at the toe was significant compared to the vegetation cover in the back dune and dune face (Appendix Figures 54 and 55).

A linear regression analysis of the effect of the vegetation cover around a pin on the net change in sediment level of that pin showed that there was no relationship between net change and vegetation density (Appendix Figure 49). An ANOVA showed no significant effect of pin position within the dune on net change of sand level (Appendix Table 57). A regression analysis of the effect of surrounding vegetation density on absolute change in sediment level at the corresponding erosion pin also showed no significant effect of vegetation density (Appendix Figure 50). An ANOVA of pin position on absolute change showed that sediment movement was not significantly affected by the pin position within the dune (Appendix Table 58). The above ANOVAs were also run with pin B:1 excluded and still found no significant effect of position on net change or absolute change.

The vegetation line at the dune toe receded by between 0.54 and 2.62 meters from September 2015 to May 2016 (Appendix Table 59). The final survey in October 2016 showed that the dune toe vegetation had advanced at all transects except C, which saw a very small amount of further recession (Table 3). The transects that experienced the greatest recession in dune toe vegetation over the winter also showed the greatest toe vegetation advance during the following growing season. The majority of toe vegetation advance happened between May and June 2016 however, transect D saw a substantial amount of toe vegetation advance between August and October 2016 (Table 3). Only transect B showed the most toe vegetation advance between June and August 2016.

Table 3) Seasonal and total advance of dune toe vegetation, in meters, at four transects along the dune at Site 4.

	Dune Toe Vegetation Advance (m)						
Transect	May - June 2016	June - Aug 2016	Aug – Oct 2016	Total			
Α	1.25	0.03	-0.01	1.27			
В	-0.01	0.115	0.005	0.11			
С	-0.01	-0.01	-0.02	-0.04			
D	0.33	0	0.22	0.55			

The number of plant species (species richness) within the square 1 m² quadrat around each erosion pin was significantly affected by the position of the pin on the dune (Appendix Table 60). Species richness was significantly lower at the dune toe compared to the face and back of the dune with only dune grass being found at the toe (Figure 13). There was no significant difference in species richness between the back and face of the dune (Appendix Table 61).



Figure 13) Estimating vegetation cover and species richness in square 1 m² quadrats around the erosion pins at the back of the dune (A), the dune face (B), and dune toe (C). Photos are of transect C and were taken during October 2016 monitoring visit.

Discussion

Small Coastal Bluffs

The four or five bluff profiles for each individual transect were very similar through time indicating that the surveys must have been fairly accurate. If there were large amounts of error, the profiles taken during the seasonal surveys would look very different for the same transect. However, two profiles were conspicuously out of alignment with the other profiles within their transects. Those were the May 2016 profile of transect 2 at Site 1 and the October 2016 profile of transect 5 at Site 2 (Appendix Figures 37 and 46). The May 2016 profile of transect 3 at Site 2 also appeared to drop, just below the bluff toe, more than expected but this was not as conspicuous as the other two profiles (Appendix Figure 42). Based on comparisons between photos and bluff profiles for the three aforementioned anomalies (See Appendix Figures 37 and 38, 42 and 43, and 46 and 47) it is likely that there was an actual change in the profile and that the direction of change shown in the profile is accurate but the amount of change is exaggerated in the profiles. This is expected based on the accumulative nature of Emery Survey errors (Andrade and Ferreira 2006). The change seen at Site 2 transect 3 could have been due to the displacement of a large stump at the location where the change was seen. However, according to photographic evidence the stump was moved prior to the February 2016 survey so if the change in profile was accurate it should have also been seen in the February profile and it was not (Appendix Figure 43).

Measurement error is the most likely cause of the irregular profiles. The most likely source of measurement error observed while conducting the surveys was that the measuring tape might not have always been perfectly horizontal when measurements

were taken. Two people conducted all surveys except the February 2016 survey when there were three people onsite. When surveying with two people it is very difficult to tell if the measuring tape is horizontal, especially over longer distances when it can sag or get caught by the wind. Having three people on site during surveying allowed one person to stand back and confirm that the tape was horizontal, this likely improved the accuracy of the modified Emery Survey method. Another issue was making sure that measurements were taken on the transect line. When there was vegetation or a steep slope impeding the view of the stakes it was easy to stray slightly off the transect line. Whenever this was noticed the transect was remeasured but it is possible that on occasion it was so slight that it was not noticed but would result in not measuring the same part of the slope which could influence the shape of the profile.

The upper parts of the bluff, where there was dense vegetation and no impact from wave activity, maintained the same profiles thorough time indicating stability of the upper bluff at both Sites 1 and 2. The only profile that indicated erosion near the bluff crest was the July 2016 profile of Site 1 transect 3 (Appendix Figure 39). The October 2016 profile at the same transect shows the surface level to be the same as the May 2016 profile so it is very possible that the 'erosion' seen in the intervening July profile was a measurement error.

The lower parts of the bluff slope and upper beach profile showed different trends in elevation change between Sites 1 and 2. The profiles from Site 1 all showed an elevation in the level of the lower segments of the Living Shoreline between the bluff toe and the edge of the shoreline vegetation. While at Site 2, four of the profiles showed no change in the lower bluff and two profiles showed erosion in the lower segments of

profiles corresponding to the lower bluff, bluff toe, and upper beach. The Living Shoreline age and maintenance activities probably caused most of the differences seen between Sites 1 and 2. Site 1 is a relatively young Living Shoreline that was about three years old at the time of monitoring while Site 2 was a very mature Living Shoreline, about twelve years old. There was very good vegetation establishment at Site 1 however, due to the age of the site the vegetation had not reached the density and overall coverage observed at Site 2. At Site 1 many areas of the bluff face are covered by the hay and brush treatments in between plantings similar to the All treatment in Chapter 2.

As shown in Chapter 2, the soil under the All treatment still experiences some erosion and a portion of that sediment is deposited downslope, possibly resulting in the pattern of elevation at the bluff base seen in the profiles from Site 1. At Site 1 the homeowners are very active in maintaining the Living Shoreline and some of the rise in elevation could be from the addition of seaweed and yard waste to the bottom portion of the Living Shoreline. However, the homeowners confirmed that during the monitoring period they did not add much material to the shoreline and avoided the areas where the transects were. This supports the hypothesis that the accretion seen at the base of the bluff may have been due to the Living Shoreline treatments especially since this area was eroding prior to installation. The exact mechanism by which the Living Shoreline could be facilitating accretion is unknown but the three most likely options are that sediment eroded from up slope is being trapped by the treatments, the decomposition of Living Shoreline materials is building soil or the treatments are capturing seaweed and sediments deposited by wind, waves, and ice.

At Site 2 the treatments have decomposed leaving the bluff face covered by grasses and perennials that prevent rainfall and runoff from transporting the sediment downslope resulting in no change at the base of most of the profiles at Site 2. However, erosion was observed for the lower bluff in the profiles of transect 2 and transect 4 at Site 2. Interestingly, both eroded during the winter when beach levels were lowest consistent with the findings of Lee [2008]. It appears that a different process was acting to cause the erosion at each of the two transects. At transect 2 the erosion seen between November 2015 and February 2016 occurred from about mid slope to just above the toe (Figure 12). This location corresponds to segments of the transect where an active animal trail and patch of bare soil were noted during measuring. The heavy animal use of this location has resulted in trampling of the vegetation causing a small patch of erosion that just happened to be captured by the transect. Runoff during heavy fall and spring rainfall events coupled with destabilization from freeze-thaw cycles are probably what caused the majority of erosion to occur over the late fall and winter, this tendency toward increased winter erosion has been seen in other bluffs (Collins and Sitar 2008, Greenwood and Orford 2008).

At transect 4 the erosion seen between November 2015 and February 2016 occurred at the bluff toe and in the area between the bluff toe and the mean high water (Appendix Figure 44). In Appendix Figure 45 it is apparent that the bluff toe is shaded under dense shrub cover during the growing season and that the area between the toe and the high water mark receives enough wave energy to remove thick mats of seaweed during the winter. The erosion at this transect seems to be due to dense shrub cover shading out the understory grasses and herbaceous plants and leaving relatively

unprotected soil at the toe of the bluff that is then eroded by the wave activity of winter storms. The lowering in elevation seen between the toe and the high water appears to be due to winter storms removing all the seaweed and plant material that was trapped by the upper beach vegetation over the summer. The winter beach lowering observed at this transect would also allow more erosive wave activity to reach the bluff toe during winter storms (Lee 2008).

The beach profile was expected to lower over the winter and then increase through the summer (Bird 1987, Collins and Sitar 2008). The beach profile at Site 1 did increase over the summer as expected with the highest overall profile occurring in July 2016. However, it was not expected that the beach profile in May 2016 would be higher than the beach profile in November 2015 since winter erosion commonly causes beach profiles to be lower in the spring. Since the February 2016 survey could not be taken at Site 1 due to ice cover it is difficult to say if the beach profile was actually lower in the winter and then raised again before the May 2016 survey was taken. However, at Site 2 where the February survey was taken, the beach profile was the lowest in February and had raised somewhat by May but the May 2016 profile was never higher than the November 2015 profile. It seems that the ice that prevented the February 2016 survey probably pushed sediment up the beach and also protected the beach sediment from erosive wave action during the winter resulting in a higher beach profile in May 2016 than there was in November 2015.

The magnitude of the fluctuations in beach level differed between the two sites and also varied spatially within each of the sites. While the actual value of the magnitude of beach level change might be exaggerated in some of the survey profiles, it seems that

differences between the sites and spatial differences within the individual sites are accurate. Overall, Site 1 experienced a greater fluctuation in beach level than Site 2, likely due to differences in exposure to wave energy at the two sites. Even though both sites are sheltered and experience low energy conditions, based on observations while working at the sites, Site 1 seems to experience more wave energy than Site 2 thus causing greater changes in beach level at Site 1. The spatial variation within the two sites in beach level fluctuation is probably due to small-scale differences in current, wave deflection off neighbouring hard structures, and exposure to prevailing winds (van Rijn 2011). However, not enough is known about these factors at these two sites to say for certain what is happening.

Coastal Cliff

The cliff at Site 3 experiences episodic slumping events that result in large episodes of crest recession followed by long periods of stability, similar to the cohesive cliffs studied by Brown et al. [2005]. It also exhibits the gullying erosion described by Himmelstoss et al. [2006] for drumlin cliffs in Boston Harbour. Currently, the western end of the cliff is in the post slumping phase where the slope of the upper cliff has a shallower grade and the slump material is protecting the toe (See Figure 1). Crest recession should be fairly slow and steady at this part of the cliff resulting mainly from runoff and precipitation. Based on the amount of slump material protecting the cliff toe, Taylor [2014] estimates another 60 years or more until the next slump and major crest recession event.

To the east of the area where the slump ends is the highest part of the cliff face, which is very steep and characterized by deep gullies and large ridges (Figure 5B. This area is progressively steepening as rain, runoff, and seepage continually erode the ridges and deepen the gullies (Himmelstoss et al. 2006). This section of the cliff is very susceptible to failure at this point and based on the cycle of cliff failure described by Brown et al. [2005], a large slump is likely to occur sooner than later and most likely during the winter or under extremely saturated conditions (Collins and Sitar 2008, Greenwood and Orford 2008). The slump on the neighbouring section of cliff resulted in crest recession of at least four meters (Taylor 2014). The same or larger magnitude of crest recession could be expected when a slump occurs at the gullied section of the cliff because this section is higher than the area that slumped previously (Brown et al. 2005).

The Geological Survey of Canada (GSC) has been monitoring crest recession at the cliff that Site 3 covers a small section of since 1980 (Taylor 2014). This provides a unique opportunity to compare the short-term data collected in this study to some of the long-term results of Robert Taylor, the main GSC researcher at this cliff site. The section of the cliff monitored in this study overlaps two geomorphically distinct sections of the cliff defined by Taylor [2014]. Transects 1 and 2 in this study correspond to Taylor's cliff Section 4 which is defined by its steep slope and extensive gullying (Talyor 2014).

Transects 3-5 correspond to cliff Section 5 which is less steep and lies above the 1980's slump block (Taylor 2014).

The average annual crest recession rate found by this study for Site 3 was 0.08 m/a. When the monitoring period is extended by only three months that rate almost doubled, to 0.13 m/15 months. The rate of 0.13 m/15 months corresponds very closely

with Taylor's findings that cliff recession at cliff Section 5 had slowed to 0.1 m/a between 2000 and 2010. Interestingly, the erosion rate of bluffs in Boston from rill and gully erosion due to precipitation was 0.13 m/a (Himmelstoss et al. 2006).

The transects in this study were divided into two groups based on the sections defined by Taylor [2014]. Transects 1 and 2, which showed total recession rates of 0.20 and 0.27 m over 15 months, respectively, correspond closely with Taylor's findings that Section 4 of the cliff was retreating by 0.2-0.3 m/a. Taylor [2014] found that recession rates at Section 5 have been slowing since the slump event occurred in the 1980's reaching a low of 0.1 m/a between 2000 and 2010. Calculating average recession for transects 3, 4, and 5 separately suggests that the recession rate for Section 5 may have further slowed to an average of 0.06 m/15 months. However, transect 4 showed a total recession of 0.12 m/15 months that is very close to the annual rate reported by Taylor [2014] for section 5 between 2000 and 2010.

Contrary to the results of Greenwood and Orford [2008] from cliffs in Ireland, this study found no seasonal variation in the rates of cliff recession at Site 3. However, given the episodic nature of crest recession, the measuring period may have been too short to discern any seasonal variations. Also the time period that corresponded to fall and winter was very broad in this study because February 2016 crest measurements could not be used in analyses. Error in the February measurements, likely caused by a small amount of snow on the ground during measuring, made it appear as if the crest had advanced since the initial measurement.

Sand Dune

The dune at Site 4 is experiencing a small amount of sediment movement and most of that sediment movement is in the form of accretion. However, the loss of the dune toe pins during the winter may have skewed the results towards low sediment movement and accretion. This is because the dune toe is generally the most active part of a vegetated coastal dune and the winter is when most beach erosion occurs (Bird 1987, Levin et al. 2008). The large recession in dune toe vegetation observed between September 2015 and May 2016 also indicates that large amounts of sediment movement and likely erosion were not captured in the data (Appendix Table 59). The fact that the toe erosion pins that were buried approximately 0.75 m in the sand were removed, also shows that high energy conditions capable of moving sediment and causing erosion were experienced at the dune toe during the winter.

Based on the results of Arens et al. [2004] a seasonal variation in sediment movement could be expected in a dune system with the majority of sediment movement occurring during the fall and winter. From the data collected in this study, sediment movement at Site 4 appears to be relatively evenly distributed throughout the year (Table 2). However, due to the loss of almost half (5/12) the winter data points, winter sediment movement was more than likely underestimated in this experiment. This could cause the similarities in absolute change for winter, summer, and fall and the appearance of a lack of seasonal trend in sediment movement.

In addition to the lost toe pins, pin B:1 was also excluded from the winter absolute change calculation because the 9.8 cm rise in dune level recorded by that pin must have been caused by something other than sediment movement. Photographs show very dense

vegetation cover and no evidence of fresh sediment accretion during May 2016 measuring. There should be some observable features if that much sediment actually accreted at the pin. This pin was located in the back dune close to an area where a large pole with a birdhouse fell over that winter. The pole was cleaned up by the homeowner before the May 2016 visit and pin B:1 would have been directly in the path that they used to access the area where it fell over. It is likely that in the snow covered conditions the pin was stepped on or otherwise pushed down into the dune resulting in a measurement that looks like sediment accretion.

The vegetation cover across the dune did not show a broad range of values. Vegetation cover was consistently 100% in the back dune and on the dune face and initially about 50% at the dune toe but after receding during the winter ranged from 0 to 10% during May to October 2016 monitoring. The significantly lower vegetation cover around the erosion pins at the dune toe was partly a result of the placement of the pins at the edge of the vegetation and was then compounded by the large recession in toe vegetation over the winter that pushed to dune vegetation out of the quadrat range (Appendix Table 59).

In accordance with Levin et al. [2006] the erosion pin data was used to calculate net change to determine erosion or accretion and absolute change as an estimate of overall sediment movement at the site. Surprisingly, the significant differences in vegetation cover around the erosion pins did not have a significant effect on the net change in dune level or the overall sediment movement around the pins. In dune systems it is expected that increased plant cover would result in decreased sediment movement (Levin et al. 2008). The position of the erosion pins on the dune also did not significantly affect net

change or absolute change when we could expect to see more sediment movement at the dune toe (Levin et al. 2006). It is possible that the sample size was too small and the loss of the toe pins in the winter, which would have represented both low vegetation cover and the location and time of year with the most active sediment movement, prevented any significant differences from being observed. It could also be that the area around the pin where vegetation cover was measured was too small. Levin et al. [2006] showed that sand movement around an erosion pin is not only related to the vegetation immediately around the pin but also vegetation 100 - 200 m upwind of the erosion pin.

The dune toe vegetation was quite resilient after being eroded over the winter with all except one transect showing vegetation advance during the following growing season. Interestingly, most of the advance occurred at the transects that had experienced the most recession. It is very possible that where large amounts of recession occurred not all of the roots were removed especially in the more landward area of recession. The persistence of roots would allow the dune vegetation to quickly recover resulting in a rapid advance of the dune toe vegetation. This is supported by the observation that the majority of dune toe advance happened in the early spring (Table 3). Very little vegetation advance occurred over the summer, likely due to the severe drought conditions experienced during the summer of 2016. Unexpectedly, twice as much vegetation advance occurred at the end of the growing season as was seen in the summer. It is possible that increased fall precipitation after the dry summer spurred some late season vegetation growth.

Species richness was significantly higher at the back dune and dune face compared to the dune toe. This is not surprising because the are many stressors including burial, exposure, wind, salt, and low nutrient availability all of which increase as you

move seaward across the dune and likely prevent many species from establishing at the dune toe (Levin et al. 2008). In fact the only plant that was noted at the dune toe was dune grass (*Ammophila breviligulata*). This is not surprising since *Ammophila spp*. are known to be adapted to very mobile dune environments and are often found on the active foredune (Levin et al. 2008).

The higher number of species in the back dune was probably influenced by the fact that about a meter or two behind the back dune erosion pins, where species richness was estimated, there is a long garden bordering the almost the entire length of the dune study site. This garden was built using manure and high quality soil, is regularly mulched, and has been fed with organic fertilizer seasonally for many years. All of these activities would increase the organic matter and nutrient content of the back dune soil and aid plant establishment and growth in the back dune area. There was also some encroachment of rugosa rose from the cultivated garden into the back dune, which was included in species counts. The dune face showed similar species richness to the back dune but was further removed from the enriched garden environment. No analyses were done on species composition but it is worth noting that a few species that were seen in the back dune including rugosa rose, raspberry, and new york aster were not noted on the dune face and species seen on the dune face such as mustard, burdock, and thistles were not recorded in the back dune. So while the species richness was similar the species composition is likely different.

The monitoring methods used in this study were all relatively simple, low cost, did not require specialized equipment or technical skill to use, and provided reasonably accurate results. The modified Emery Survey Method is very useful for shorelines where the entire slope of the bank, bluff, or cliff is accessible and safe to work on. The accuracy of this method may not have been as good as I would have liked, since an error of 5 cm is the least that can be expected (Andrade and Ferreira 2006). However a few small adjustments would likely significantly improve the profiles generated. Installing small pins to mark the location where initial measurements are taken would ensure that the same locations were measured each time. Also having a third person on site to ensure that the tape is horizontal would also improve accuracy. Unfortunately, the cost of this may be prohibitive in many cases so this tactic should be reserved for situations where a high level of accuracy is absolutely required.

The biggest advantage of this method compared to the other methods is the visually appealing bluff profiles that are produced and the ability to layer profiles over each other so that Helping Nature Heal and their clients can visualize what is happening through time and can easily see the effect of the Living Shoreline if surveys are done pre and post construction. Also it is my understanding that if you are confident in the accuracy of your profiles they can be used to estimate the volume of sediment lost from the bluff face and presumably also accretion (personal communication D. van Proosdij 2016). The only real downside to the Emery Survey Method is that some homeowners may find the transect stakes unsightly. If safety of small children or pets is not a concern this could be remedied by using less conspicuous materials such as rebar or zinc rods to

define the transects. Another solution would be to install fewer transects and only in areas that are not immediately visible to the homeowners.

Crest and toe measurements are most applicable on bluff or cliff sites that are too steep or too large to safely do the Emery Surveys or in locations where working frequently on the slope face would cause too much disturbance to soil, plant communities or habitat. Crest recession measurements have the added advantage of being a fairly standardized method of monitoring cliff recession and can easily be cross referenced with other studies or compared to other studies on the same cliff if they exist (Taylor 2014).

In the future, except perhaps in the most protected sites, I would not recommend the toe transects that were tried at Site 3. They are too easily removed by winter conditions and even on sheltered shorelines like Site 1 for example, all but one stake was broken by ice. If toe transects are determined to be necessary at a site, there has to be another stable benchmark that can be used to measure from to replace the stakes if they are lost over the winter. A potentially better way of monitoring erosion at the toe of a large bluff or cliff is erosion pins inserted into the lower face of the bluff or cliff (Manson 2002). However, at very high energy locations or locations experiencing a lot of erosion it is still likely that the pins will be lost or buried by eroding sediment. In these locations photographic documentation of the toe from a marked location so it is the same every time may be the best way to monitor erosion processes at the toe.

In the dynamic dune environment, erosion pins were a very simple and relatively quick way to gather a lot of information about erosion, accretion, and overall sediment movement. Depending on the frequency of measurements, duration of monitoring, and other correlated factors measured at the same time, erosion pins can also provide data on

seasonal processes, long term evolution of the dune, and how things like soil moisture or wind are related to sediment transport within the dune environment (Levin et al. 2006). Measurements of the extent of toe vegetation were very useful in determining conditions at the site and the resilience of the dune. In the future I would probably not install stakes at the edge of the vegetation because they get lost, beach goers may pull them out, and some clients may find them unsightly. Instead measurements taken from a landward stake to the edge of the dune toe vegetation would provide the same information without the issues associated with installing monitoring equipment in the active foredune and areas people use for recreation. Vegetation cover estimates would be most useful as a simple approximation of success in the early stages of a dune revegetation project. In the context of this study working in a stable dune with well-established vegetation, vegetation cover estimates did not provide a lot of useful information. Part of the problem is that the functional area of vegetation related to dune erosion processes is too large to monitor on a private property.

The one monitoring technique that proved very useful at all of the coastal sites was photographs. It is impossible to remember all the details of a single transect let alone multiple transects at multiple sites. The photographs taken during monitoring for this study proved invaluable for interpreting the data collected, especially the bluff profiles. It was surprising how large changes at the sites, specifically the large stump being picked up and moved to a nearby location, went completely unnoticed in the field and was only noticed when comparing the photographic timeline of the transect (Appendix Figure 43).

Overall, the simplicity and accuracy of the monitoring methods make them very suitable for an industry based monitoring program. With minimal set up and instruction

anyone can monitor coastal change using these methods. The low cost of materials and small amount of time investment that it takes to do the measuring also makes it a more palatable undertaking for clients and companies. These methods are ideal for Living Shoreline monitoring because they are easily adapted to densely vegetated conditions as well as baseline data gathering on bare slopes. Incorporating a provision for baseline monitoring before a project is installed and potentially a small untreated section to act as a control where possible would improve the strength of the results from the monitoring program.

Conclusion

Overall, this study showed promising results indicating that Living Shorelines designed and built by Helping Nature Heal are stable and resilient on coastal bluffs in low energy environments and possibly on dunes in moderate-high energy environments.

Longer monitoring is needed to determine if the Living Shoreline treatments are slowing annual cliff recession rates at the high-energy cliff site. A high percentage of vegetation cover is a strong indicator of coastal stability and vegetation cover was high at the bluff and dune sites indicating that those shorelines are relatively stable (Duhring 2006).

Accretion and vegetation recovery at the dune site and beach level recovery at the small bluff sites shows that they are healthy and resilient to the normal seasonal erosion that is occurring. Even though there was some recession of the dune toe vegetation, the area of the dune that was obviously subjected to wave activity over the winter was much larger than the area that saw recession indicating that the vegetation is stabilizing the dune. The vegetation cover appears to be increasing the dune's resistance to disturbance, which is

very beneficial for the stability of the dune because stable dunes tend to remain stable unless a disturbance remobilizes them (Arens et al. 2004).

In addition to stability and resilience, the young Living Shoreline at Site 1 is showing accretion at the bluff toe, which is an indication that the treatments are capturing sediment and protecting the toe from erosion. The mature Living Shoreline at Site 2 is generally stable but it appears that after twelve years the toe is beginning to be eroded in some locations. It is possible that this is part of the natural erosion cycle of small coastal bluffs or a cycle related specifically to Living Shoreline projects on coastal bluffs.

The proposed cycle for Living Shorelines is that after a period of erosion the slope is protected and stabilized using the Living Shoreline treatments including the planting of small herbaceous perennials, grasses, and shrubs. Once plants are established the slope remains stable for a while, in the case of Site 2 at least twelve years. Once the shrubs are fully mature they begin to shade out the perennials and grasses underneath which reduces the canopy cover and root density that was protecting the soil around the shrub base. In the fall and winter the less protected soil is exposed to the erosive forces of rain, runoff, waves, storms, and freeze-thaw cycles. Erosion then begins to destabilize the shrubs, which fall over taking large amounts of soil with them. This exposes the bluff face to further erosion and recession.

It seems that the phase of the cycle where grasses and perennials are lost is just beginning at Site 2 about twelve or thirteen years after installation. Typical maintenance on a Living Shoreline occurs for the first 3-5 years until plants are established. Perhaps a formal ten-year Living Shoreline check-in by Helping Nature Heal staff should be implemented for projects with similar conditions to those seen at Site 2. This check in

would determine if maintenance is needed before any erosion occurs. Maintenance activities should focus on preventing the loss of the ground covers through pruning and cutting back shrubs and trees to reduce shading so that the vegetation can continue to stabilize the bank. That said coastal property owners should be monitoring their shorelines on a consistent basis so that any soil exposure or damage to Living Shorelines can be caught and repaired immediately thereby reducing the slope's exposure to erosive forces. In the case of Site 2 revegetation of the animal trail, the addition of a little more hay and brush to protect some areas of the bluff toe, and some trimming of shrubs to allow more light in to improve the coverage of herbaceous perennials and grasses would go a long way to ensure future stability of the bluff.

Consistent, long-term maintenance will be very important for the potential success of Living Shoreline treatments at the cliff site because of the large amounts of erosion material that bury the Living Shoreline treatments (Personal observation). Potentially trying plants that can accommodate frequent burial may be an option to explore at this site. So far alders seem to be able to tolerate the conditions at the site well. It is possible that the Living Shoreline techniques studied here with the right maintenance when used above the slump at Site 3, could slow crest recession of the area directly above the slump in the medium to long term (~60 years). There is some indication from this study that the annual rate may have slowed to 0.06 m/ 15 months since 2010, which corresponds to the installation of the Living Shoreline treatments. However, the measuring time frame was too short in this study to draw any strong conclusions.

If the treatments can help grade the slope to a lower angle as they were shown to do at the small scale in Chapter 2 and can provide a high percentage of vegetation cover then crest recession could be stabilized over time. At least until the slump material is completely eroded by the ocean and the cliff face becomes too steep to maintain vegetation cover thereby allowing the gullying phase to begin and continuing the cycle of bluff failure, slumping, and crest recession (Brown et al. 2005, Himmelstoss et al. 2006). This cycle may happen faster than the 60+ years predicted by Talyor [2014] as sea levels continue to rise and toe erosion rates increase (Himmelstoss et al. 2006). However, it is important to remember that this sediment is needed to maintain other coastal features like beaches that are so highly valued for tourism and recreation in the area.

Coastal change is very dependant on time scale; seasonal fluctuations, occasional storms and erosive events, long natural cycles of cliff recession, tidal cycles, and of course the slow progressive nature of sea level rise, all influence this temporal variability. Short-term monitoring can over or underestimate erosion depending on the occurrence of erosive events while long term monitoring provides a better overall picture of actual erosion rates it is not always feasible. The issue becomes how to balance the need for long-term erosion monitoring with financial constraints and the relatively short duration of most research projects. Leading to the perennial research question: How long is long enough?

One would think that once the Living Shoreline is mature and stabilized that monitoring would no longer be needed. However, we know that shorelines are always evolving and permanent stability is not a reasonable expectation for any shoreline stabilization project (Subramanian et al. 2006). The results from Site 2 in the present study are further proof of this fact. So while HNH's Living Shoreline techniques have been shown to reduce sediment loss and based on short-term data also appear to stabilize

and provide resilience to coastal bluffs and dunes, consistent maintenance and monitoring is the key to their persistence into the future.

A lot has been learned from this study that will improve Living Shoreline maintenance plans and will help implement an effective Living Shoreline monitoring program at Helping Nature Heal. A long-term monitoring program like this is badly needed to further Living Shorelines knowledge and almost necessarily has to come at least partly from industry because researchers usually do not have the resources to undertake the long-term (10+ year) studies that are needed adequately inform management decisions.

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Chapter 4: Conclusion and Future Directions for Living Shoreline Research in Nova Scotia

Need for Coastal Erosion Management and Living Shorelines

Water driven erosion was the primary focus of this study because the action of water in the form of rain, runoff, seepage, and waves is the main cause of coastal erosion above the mean high water line in Nova Scotia. Erosion is driven by detachment and transport processes, in terrestrial systems raindrop impacts facilitate detachment and runoff transports the detached sediment (Fox and Bryan 1999). In coastal systems waves and ocean currents accomplish detachment and transport of sediment (Bird 1987).

Seepage is a bit more complex acting on the internal cohesion and stability of coastal cliff sediments (Vandamme and Zou 2013).

The coast is particularly vulnerable to erosion because its sediments are subjected to constant detachment and transport by water from terrestrial sources and the ocean. Erosion is part of the natural evolution of coastlines, especially during periods of higher sea levels like the planet is currently experiencing (van Rijn 2011). The coast is also under increasingly heavy pressure from humans for development and recreation as the global population grows (Hsu et al. 2008). These human activities can exacerbate the natural cycle of coastal erosion, as does sea level rise from human induced climate change (Roberts 2006). The inevitable result of increasing coastal development while at the same time accelerating coastal erosion rates is a conflict between humans and coastal processes. Coastal management is needed to resolve this conflict and balance the

protection of human investments with the maintenance of coastal processes that support human life and livelihoods.

It appears that until recent decades most of the coastal erosion management methods have been developed in an attempt to protect assets and infrastructure with no regard for coastal processes. The prominent methods of coastal erosion management to date have been hard, engineered structures that are designed to 'permanently' hold the coastline at the desired location (Davis et al. 2006). As erosion rates, frequency of severe storms, and awareness of the importance of coastal processes continue to increase, it is becoming very apparent that hard coastal defenses are incapable of adapting to the future challenges facing coastal populations (Borsje et al. 2011). On the grand scale of coastal evolution we will inevitably have to surrender large areas of land to the ocean. However, on the scale of human lifetimes coastal managers need to find ways to protect people and their investments from erosion while maintaining the natural functions of coastal ecosystems and their ability to adapt to future conditions.

Living Shorelines are one available option that uses plants and natural coastal ecosystems to stabilize coastal features, reduce erosion rates, and protect coastal assets while maintaining coastal processes (Smith 2006). The health of coastal ecosystems and the human livelihoods that rely on them are progressively being deteriorated by the negative impacts of developed and hardened coastlines (Davis et al. 2006). As a result, the adoption of softer methods of erosion management and coastal protection, like Living Shorelines, is essential if we wish to maintain the ecological integrity of the coastline and protect the people who rely on it in the long term (Davis et al. 2006). One of the barriers to widespread adoption and implementation of Living Shorelines is continued uncertainty

about their effectiveness for reducing erosion rates (Gittman et al. 2014, Bilkovic et al. 2016). A considerable amount of research on Living Shorelines shows strong benefits for ecosystem services, habitat value, and their ability to restore and support populations of aquatic species (ex. Gittman et al. 2016). However, the reality is that most people, communities, and developers are primarily concerned with erosion protection and coastal stability. While ecological value often comes in further down the list of priorities because the impact of a degraded coastal ecosystem on their safety, health, and livelihood is not immediately apparent.

In order to get more widespread incorporation of Living Shorelines into coastal management by governments and the public, the Living Shoreline research community needs to better understand the efficacy, strengths, and limitations of available techniques. The shoreline is a dynamic interface and further advances in coastal management require consideration of both the terrestrial and marine environments. Researchers in terrestrial environments have adequately shown the value of plants and plant residues for reducing erosion rates in many different environments and have also demonstrated that most natural, native plant communities experience less erosion than any other land use (Cerdan et al. 2010). The research summarized below is a step toward gaining knowledge that will lead to continued improvements in coastal management in the Maritime Provinces of Canada

Filling Gaps in Living Shoreline Knowledge

The research presented in Chapters 2 and 3 attempts to provide answers to two important questions about Living Shorelines:

- 1) Are Living Shorelines effective for erosion reduction?
- 2) Can Living Shorelines remain stable and resilient in the coastal environment of Nova Scotia?

The answer to the first question was overwhelmingly yes, the three Living Shoreline techniques (Hay, Brush, and Planting) when combined to simulate a Living Shoreline (All treatment) were very effective for reducing erosion. The Control plots lost 51.4 times more sediment than the All treatments in controlled experiments on extremely steep slopes of glacial till, similar to those found in many locations along the coastline of the South Shore of Nova Scotia (Cann and Hilchey 1958). The Control plots lost a total of 7371 g/m² of sediment while the All treatments only lost 141 g/m² of sediment over the course of the experiment. Importantly, the Living Shoreline techniques did not just reduce erosion but reduced the erosion of all sediment particle sizes larger than 63 µm. So while these techniques were studied on glacial till in Nova Scotia the results should be applicable to all soils with components greater than silt and clay. These techniques are more than likely very effective for reducing the erosion of silt and clays as well since very little fine sediment was observed in the collection containers below the simulated Living Shorelines compared to the Control plots (Figure 1). However, those sediment size classes were simply not measured during the experiment.

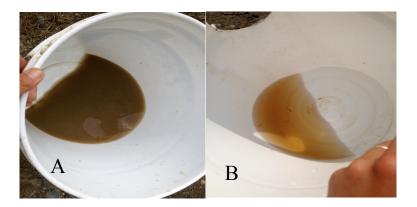


Figure 1) Runoff from a Control plot (A) and an All plot (B) from the same natural rainfall event. Photos taken after seiving through a 63 µm sieve.

Even though it was not specifically investigated in this study there are indications that the simulated Living Shorelines provide the added benefit of habitat value, specifically for birds. Out of the twenty-seven experimental plots in this study, two of the plots with simulated Living Shorelines contained bird's nests the first spring after they were installed (Figure 2). It is interesting that one third of the simulated Living Shorelines provided habitat for bird species within the first year of being installed while none of the other treatments were used by birds in this way. Longer and more in depth monitoring is needed to assess the true habitat value of these treatments for invertebrates, reptiles, mammals, and birds.



Figure 2) Robins nest in one of the simulated Living Shoreline (All treatment) plots on June 6, 2016.

Answering the question about the stability and resilience of Living Shoreline projects was slightly harder because of a lack of baseline monitoring, absence of control sites, and the necessarily short-term nature of this study. However, the data obtained from Living Shoreline monitoring yielded promising initial results indicating the general stability of the Living Shorelines installed on coastal bluffs and dunes in Nova Scotia. Stability was inferred from a lack of change over time, low amounts of sediment movement, and high amounts of vegetation cover at the shoreline sites. Resilience was seen in the recovery of beach level, recovery of dune vegetation, and sediment accretion after natural seasonal erosion.

At the small coastal bluff sites, areas with high amounts of vegetation cover showed no change over time based on the bluff profiles. Indications of erosion were only observed in the profiles when vegetation cover was compromised and the soil was not protected from detachment and transport processes. At the dune site sediment movement was relatively low where vegetation cover was high, strongly indicating that vegetation cover was stabilizing the dune sediment. Plant cover is known to be a strong indicator of

soil stability not only because it protects soil from the erosive forces of rain and wind but also due to the stabilizing effect of the plant roots and stimulation of soil biota that in turn enhance and stimulate further plant growth, nutrient cycling, and the production of biofilms that bind soil particles.

The ability of plant roots to stabilize sediment was especially apparent at the dune site in the spring of 2016 when very little change in dune sediment was measured on the dune face in areas that, based on observations made in February 2016, had obviously been impacted by winter storm waves. Those waves were strong enough to destabilize over two meters of vegetation at some locations on the dune toe but due to the strong root systems of the dune grasses, were not able to destabilize sediment farther landward on the dune face. Two studies looked specifically at the effect of plants on the stability of revegetated coastal banks and cliffs (Jones and Hanna 2004, Bischetti et al. 2010). Jones and Hanna [2004] found that planting into the bank face increased the shear strength of coastal banks by decreasing the soil moisture content. Bischetti et al. [2010] found that live cuttings inserted into the slope face significantly increased slope stability by increasing internal friction and reducing shear stress. This increase in stability was observed to a soil depth of 1.75 m and increases with increasing slope steepness (Bischetti et al. 2010). These studies support the conclusion that HNH's Living Shorelines are increasing the stability of coastal slopes through the use of live plants and cuttings.

The data from the coastal bluff beach level and dune toe vegetation monitoring indicates that these systems may be resilient to erosion caused by winter storms and waves. Seasonal fluctuations are expected in beach and dune environments because they

are naturally dynamic (Bird 1987). The important thing is that the dune and beach systems are healthy enough to recover after these events. The beaches at the small bluff sites experienced erosion during the first winter of monitoring. By the end of monitoring the following fall, almost all transect profiles showed that the beach level had recovered and was higher than it was prior to winter erosion. While at the dune site, toe vegetation showed significant advances over the growing season after being eroded the previous winter. Sediment accretion over the summer was also observed at the eroded dune toe. These results show strong evidence of recovery and resilience at these sites.

However, the dune toe vegetation did not reach the same seaward extent as it had before the winter erosion. Conditions over the 2016 growing season, when recovery was monitored, were exceptionally dry and not conducive to plant growth possibly resulting in the grasses not reaching their pre-winter position. This is an example of why long term monitoring is needed because a determination of whether the toe vegetation is fluctuating around an average position or if it is steadily eroding cannot be made from a one or two year study.

It seems that the scale of the erosion processes and the energy that the cliff site is exposed to are too large and extreme to be completely stabilized in the long term (100+ years). It is generally recognized that open ocean coasts are more difficult to protect due to the high-energy conditions (National Academy of Sciences, 2006). However, if revegetation efforts are focused on the area above the slump block there is definitely potential for a Living Shoreline to reduce crest recession at the cliff site. Care should be taken in what plants are chosen so as not to increase slumping due to increased weight

from plant material, enhancing seepage, and promoting internal destabilization via root growth and freeze-thaw action.

Future Directions

With consistent monitoring, maintenance, and property owner commitment, the medium-term (30-40 years) seems to be a reasonable expected life span for most of these Living Shoreline projects based on current environmental conditions and climate predictions. It is important to recognize that as the sea level continues to rise over the next 100+ years, some coastal erosion is inevitable and unstoppable even with hard structures. The largest and most heavily engineered structures may be able to 'hold the line' in some coastal locations over the next 100 years but probably not without significant fortification. At the scale of private properties and small community developments it is becoming very apparent that in many locations rock walls and bulkheads may not be more effective than soft management methods over the lifetime of the structures. Hard structures are definitely not more resilient than Living Shorelines when storm events happen and they are much more expensive to repair when a failure does occur (Gittman et al. 2014). Collaboration between coastal marine specialists, who have the ability to address erosion forces in the nearshore marine environment using sills and artificial reefs, and bioengineering specialists working to increase stability above the mean high water line is critical for creating holistic solutions that can adequately address coastal issues in the medium term.

In the short term and at a small scale the negative environmental impacts of shoreline hardening on unconsolidated coastlines may not be perceived as a great

concern. However, the widespread loss of healthy coastal ecosystems and the services they provide through accelerated erosion and 'coastal squeeze' resulting from hard structures, among other factors, is already having an impact on livelihoods, food security, and safety in many places (Swann 2008). These impacts will likely continue to worsen and affect more areas in the future. It is recognized that environmental restoration and the ecosystem services provided by healthy environments have become increasingly important as human-induced environmental degradation continues to accelerate and this trend is likely to continue (Trabucchi et al. 2012).

The most effective and economical form of coastal restoration in the intertidal zone is to maintain natural coastal ecosystems and their functions (Chapman and Underwood 2011). I believe that this also applies to areas above the mean high water line where Helping Nature Heal works. The natural function of some coastal features such as bluffs is to erode and contribute sediment to the coastal system. However, allowing erosion is not always feasible when property and other investments are at risk. Helping Nature Heal's Living Shoreline product provides the opportunity to protect coastal assets while maintaining coastal ecosystems in a more natural state than traditional hard structures and possibly even enhance coastal function and resilience. By reducing the negative impacts of introducing additional hard structures into the coastal ecosystem we can preserve the health and integrity of Nova Scotia's coastline well into the future (Airoldi et al. 2005).

Overall, this research has validated the photographic evidence and homeowner anecdotes that indicated that HNH's Living Shorelines were successfully stabilizing coastal landforms and reducing erosion rates. This research has also revealed information

about how Living Shorelines mature in the short term in Nova Scotia and the monitoring and maintenance that is needed to ensure they remain stable and functional. This study also shows what can reasonably be expected from a Living Shoreline in terms of stability and resilience in a variety of coastal environments and how these aspects of Living Shorelines might be improved in the future.

This exemplifies how important monitoring these projects is for broadening Living Shoreline knowledge and improving Living Shoreline techniques in order to enhance the effectiveness of these projects and promote their use. The long term monitoring that will stem from this research and the work of Helping Nature Heal will be invaluable for providing more evidence to encourage the adoption of Living Shorelines as a viable option for coastal management in Nova Scotia. This would align us with the practices of several states along the Atlantic seaboard of the United States (Watkinson and Moon 2006, Bilkovic et al. 2016, Maryland DNR 2017(b)). My greatest aspiration for this research is that it can help influence a transition to a future where Living Shorelines are a familiar concept and their implementation is widespread in coastal management on the East Coast of Canada.

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Appendix

Chapter 1:

Table 1) Plant species used in the primary experiment Plant and All treatments.

Scientific Name	Common Name
unknown	Grass
Morella pensylvanica	Northern Bayberry
Rubus ideaus	Raspberry
Cornus stolonifera	Red Osier Dogwood
Alnus rugosa	Speckled Alder
Spirea alba	Spirea
Rosa virginiana	Wild Rose
Salix spp.	Willow

Chapter 2:



Figure 1) Measuring soil surface position in a Plant treatment plot using profile gauge.

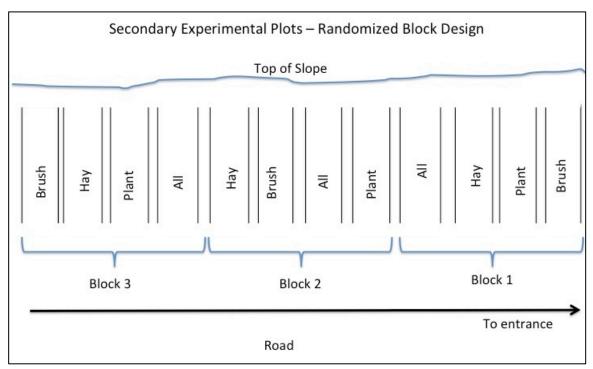


Figure 2) Open plot design and treatment layout of secondary experimental plots.

Table 2) Results of soil analysis of surface samples from primary plot 15-8 and secondary plot 12-6 prepared by Nova Scotia Department of Agriculture Laboratory Services.

Sample ID	12-6		15-8		
Parameter	Analysis	Rating	Analysis	Rating	
Nitrogen (%)	0.05		0.07		
pH (pH Units)	4.95		4.94		
Buffer pH (pH Units)	7.63		7.51		
Organic Matter (%)	1.6		3.0		
P2O5 (kg/ha)	86		65		
K2O (kg/ha)	85		127		
Calcium (kg/ha)	12		64		
Magnesium (kg/ha)	4		26		
Sodium (kg/ha)	16		21		
Sulfur (kg/ha)	28		28		
Aluminum (ppm)	2182		2083		
Boron (ppm)	< 0.50		< 0.50		
Copper (ppm)	0.23		0.29		
Iron (ppm)	37		124		
Manganese (ppm)	2		9		
Zinc (ppm)	< 0.20		0.47		
CEC (meq/100 g)	3.1		4.4		
Base sat. K (%)	2.9		3.1		
Base sat. Ca (%)	1.0		3.7		
Base sat. Mg (%)	0.6		2.5		
Base sat. Na (%)	1.1		1.0		
Base sat. H (%)	94.5		89.7		

Slope Results

Primary Experimental Blocks

Table 3) Results of the analysis of covariance testing the effect of treatment, block, survey year and the survey year-block interaction on mean slope (%) of plots within the primary experimental plots blocks.

j - p						
	Factor	DF	SS	MS	F	р
Between	Treatment	2	36.12	18.062	2.018	0.195
Plot Effects	Block	4	40.61	10.152	1.134	0.406
	Residuals	8	71.61	8.951		
Within Plot	Year	1	0.008	0.008	0.007	0.9324
Effects	Year x Block	2	9.349	4.674	4.205	0.0413
	Residuals	12	13.338	1.111		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

Table 4) Mean slope gradient (%) of plots within primary experimental blocks by survey year.

	Block					
Year	1	2	3			
2015	62.2	63.9	64.5			
2016	61.8	65.4	63.3			

Secondary Experimental Blocks

Table 5) Results of the analysis of covariance testing the effect of treatment, block, survey year and the survey year-block interaction on mean slope (%) of plots within the secondary experimental blocks.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	106.2	26.6	1.047	0.467
Plot Effects	Block	2	921.2	460.6	18.161	0.005
	Residuals	5	126.8	25.4		
Within Plot	Year	1	42.93	42.93	49.48	6.09e ⁻⁰⁵
Effects	Year x Block	2	22.43	11.22	12.93	0.002
	Residuals	9	7.81	0.87		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

Table 6) Mean slope gradient (%) of plots within secondary experimental blocks by survey year.

	Block				
Year	121	122	123		
2015	64.7	70.2	81.8		
2016	63.7	68.5	76.5		

Table 7) Results of a linear mixed effects analysis comparing the mean slope gradient (%) of the plots within the secondary experimental blocks.

Comp	arison	Value	SE	DF	t-value	р
Block 121	Block 122	6.46458	2.853521	5	2.265476	0.073
	Block 123	18.08958	2.853521	5	6.339390	0.001
Block 122	Block 123	11.625	2.5865706	9	4.49437	0.0015

Plot Slope Change

Table 8) Results of the analysis of covariance testing the effect of treatment and block on net slope change (%) of primary and secondary experimental plots.

Factor	DF	SS	MS	F	р
Treatment	4	17.14	4.284	2.198	0.113
Block	5	100.95	20.189	10.358	0.0001
Residuals	17	33.14	1.949		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

Table 9) Results of a Tukey multiple comparison of means test comparing the mean net change in slope (%) between plots within primary and secondary experimental blocks.

Comparison	Difference	LCI	UCI	p adj
Blocks 1-2	1.84	-0.984	4.664	0.340
Blocks 1-3	0.86	-3.684	1.964	0. 0.92
Blocks 2-3	-2.70	-5.524	0.124	0.066
Blocks 121-122	-0.80	-3.958	2.358	0.962
Blocks 121-123	-4.38	-7.533	-1.217	0.004
Blocks 122-123	-3.58	-6.733	-0.417	0.022

Difference = difference between the means; LCI = lower bound of 95% confidence interval; UCI = upper bound of 95% confidence interval; p adj = Tukey adjusted probability level

Table 10) Mean net change in slope gradient (%) of plots within primary and secondary experimental blocks.

	Block					
	1	2	3	121	122	123
Mean Net Change	-0.4	1.5	-1.2	-1.0	-1.8	-5.3

Runoff from Natural Rainfall

Table 11) Parameters of the Michaelis-Menten fit generated by a non-linear least squares regression in R Studio for the relationship between runoff (L/m²) and calculated rainfall (mm).

Parameter	Estimate	SE	t-value	n		
Parameter	Latimate	JL	t-value	р		
Intercept (a)	1.187	0.048	24.607	<2e ⁻¹⁶		
Slope (b)	-0.583	0.130	-4.487	$1.07e^{-0.5}$		
RSE = 0.769						
DF = 271						
Iterations to cor	nvergence = 1					
Achieved convergence tolerance = 7.336e ⁻⁰⁹						
Residual sum of squares = 160.2						

Standard Error = SE; p = probability level; Residual Standard Error = RSE; Degrees of Freedom = DF

Table 12) Results of the analysis of covariance testing the effect of treatment, calculated rainfall (mm), block, day and treatment-day interaction on runoff (L/m²) for natural rainfall events.

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	1.170	1.170	1.900	0.400
Plot Effects	Treatment	4	3.148	0.787	1.279	0.574
	Block	1	6.334	6.334	10.289	0.192
	Day	7	24.213	3.459	5.619	0.314
	Residuals	1	0.616	0.616		
Within Plot	Rainfall (mm)	1	260.7	260.70	1202.088	<2e ⁻¹⁶
Effects	Day	19	166.94	8.79	40.515	$<2e^{-16}$
	Treatment x	73	19.21	0.26	1.214	0.156
	Day					
	Residuals	165	35.78	0.22		

Table 13) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, day, block and treatment-day interaction on runoff (L/m^2) for small rainfall events (< 10 mm).

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	0.0254	0.0254	0.219	0.6594
Plot	Treatment	4	0.2318	0.0579	0.500	0.7391
Effect	Day	1	0.0500	0.0500	0.431	0.5404
	Block	2	1.0149	0.5075	4.380	0.0796
	Treatment x Day	1	0.0075	0.0075	0.065	0.8093
	Residuals	5	0.5793	0.1159		
Within	Rainfall (mm)	1	14.022	14.022	1010.305	$< 2e^{-16}$
Plot	Day	7	2.077	0.297	21.380	1.21e
Effect						15
	Treatment x Day	32	0.614	0.019	1.383	0.126
	Residuals	76	1.055	0.014		

DF = degrees of freedom; SS = sum of squares; MS = mean square; $F = (MS_{factor})/(MS_{error})$; p = probability level

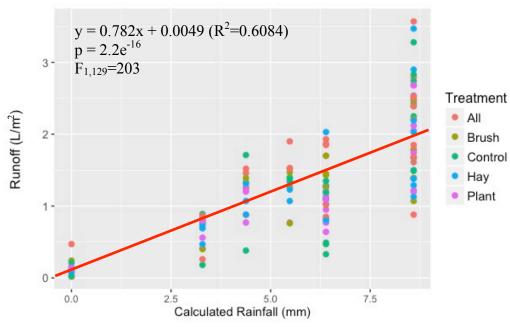


Figure 3) Linear regression of the effect of calculated rainfall (mm) on runoff (L/m^2) for small rainfall events (< 10 mm). Residual standard error = 0.5 on 129 degrees of freedom.

Table 14) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, day, block, and treatment-day interaction on runoff (L/m^2) for large rainfall events (34.71 mm < 73.07 mm).

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	0.185	0.185	23.18	0.041
Plot Effect	Treatment	4	3.940	0.985	123.70	0.008
	Day	2	4.099	2.050	257.40	0.004
	Block	2	5.851	2.926	367.42	0.003
	Treatment x	3	0.718	0.239	30.05	0.032
	Day					
	Residuals	2	0.016	0.008		
Within	Rainfall (mm)	1	3.149	3.149	14.139	0.00115
Plot Effect	Day	3	10.372	3.457	15.526	1.5e ⁻⁰⁵
	Treatment x	13	4.535	0.349	1.567	0.17419
	Day					
	Residuals	21	4.676	0.223		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

Table 15) Runoff (L/m^2) for each treatment type within the three primary experimental blocks from large rainfall events (34.7 mm < 73.1 mm).

		Block						
Treatment	1	2	3					
Control	3.4	3.7	4.1					
Plant	3.4	3.8	4.5					
Brush	3.3	4.1	4.5					
Hay	2.9	3.8	4.3					
All	3.9	4.7	4.8					

Table 16) Results of a linear mixed effects analysis comparing the difference in runoff (L/m^2) from large rainfall events (34.71 mm < 73.07 mm) between the treatments and blocks.

Compa	rison	Value	SE	DF	t-value	р
Control	Plant	0.1958493	0.2948507	8	0.664232	0.5252
	Brush	0.2801980	0.2949599	8	0.949953	0.3699
	Hay	-0.0951579	0.2819893	8	-0.337452	0.7445
	All	0.7779284	0.2966396	8	2.622470	0.0305
All	Plant	-0.5820791	0.3040754	8	-1.914259	0.0919
	Brush	-0.4977304	0.3051180	8	-1.631272	0.1415
	Hay	-0.8730862	0.2899473	8	-3.011189	0.0168
Block 1	Block 2	0.6665132	0.2299206	8	2.898884	0.0199
	Block 3	1.0886521	0.2348799	8	4.634931	0.0017
Block 2	Block 3	0.4221389	0.2232584	8	1.890809	0.0953

SE = standard error; DF = degrees of freedom; p = probability level

Sediment Loss from Natural Rainfall

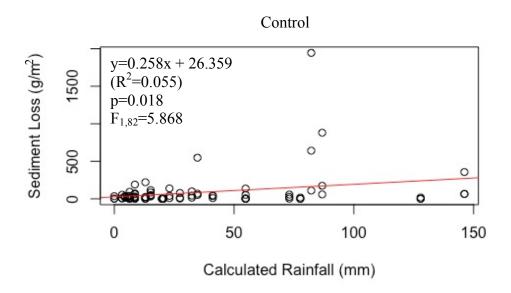


Figure 4) Regression analysis of the effect of calculated rainfall (mm) on sediment loss (g/m^2) for the control plots. Residual standard error = 239.5 on 82 degrees of freedom.

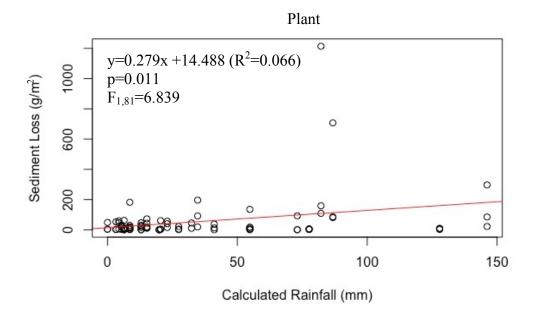


Figure 5) Regression analysis of the effect of calculated rainfall (mm) on sediment loss (g/m^2) for the plant treatment. Residual standard error = 151.4 on 81 degrees of freedom.

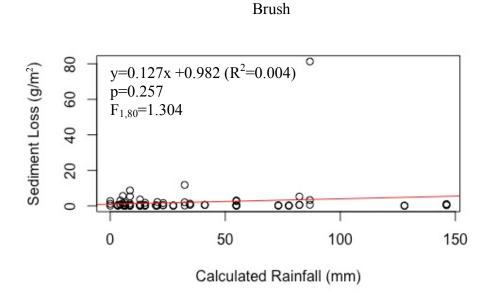


Figure 6) Regression analysis of the effect of calculated rainfall (mm) on sediment loss (g/m^2) for the brush treatment. Residual standard error = 9.1 on 80 degrees of freedom.



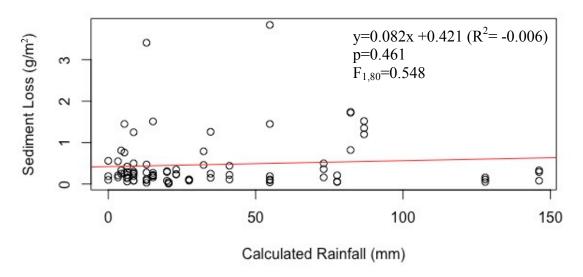


Figure 7) Regression analysis of the effect of calculated rainfall (mm) on sediment loss (g/m^2) for the hay treatment. Residual standard error = 0.7 on 80 degrees of freedom.

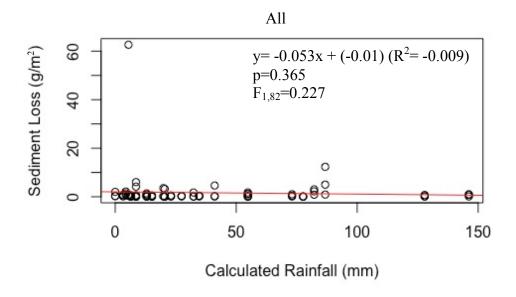


Figure 8) Regression analysis of the effect of calculated rainfall (mm) on sediment loss (g/m^2) for the all combined treatment. Residual standard error = 7.0 on 82 degrees of freedom.

Factors Affecting Runoff

Table 17) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, block, day, and the treatment-day interaction on runoff (L/m²) for natural rainfall events.

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	1.170	1.170	1.900	0.400
Plot Effects	Treatment	4	3.148	0.787	1.279	0.574
	Block	1	6.334	6.334	10.289	0.192
	Day	7	24.213	3.459	5.619	0.314
	Residuals	1	0.616	0.616		
Within Plot	Rainfall (mm)	1	260.70	260.70	1202.088	2e ⁻¹⁶
Effects	Day	19	166.94	8.79	40.515	2e ⁻¹⁶
	Treatment x Day	73	19.21	0.26	1.214	0.165
	Residuals	165	35.78	0.22		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

Factors Affecting Sediment Loss

Table 18) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, day, block, and the treatment-day interaction on sediment loss (g/m^2) for natural rainfall events.

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	122.9	122.92	29.376	0.006
Plot Effects	Treatment	4	1246.2	311.55	74.453	0.0005
	Day	3	27.2	9.06	2.166	0.235
	Block	2	159.0	79.52	19.004	0.009
	Residuals	4	16.7	4.18		
Within Plot	Rainfall (mm)	1	13.9	13.922	13.574	0.0003
Effects	Day	26	468.2	18.007	17.557	2e ⁻¹⁶
	Treatment x	108	191.0	1.768	1.724	0.0002
	Day					
	Residuals	264	270.8	1.026		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

Table 19) Results of a linear mixed effects analysis, comparing sediment loss (g/m²) from natural rainfall events between the treatments and blocks.

Compa	rison	Value	SE	DF	t-value	р
Control	Plant	-0.265630	0.274072	8	-0.969196	0.3608
	Brush	-3.466991	0.2753288	8	-12.59219	0.0000
	Hay	-4.124878	0.2747069	8	-15.01556	0.0000
	All	-3.738373	0.2734652	8	-13.67038	0.0000
Plant	Brush	-3.201361	0.2759384	8	-11.60172	0.0000
	Hay	-3.859248	0.2753024	8	-14.01821	0.0000
	All	-3.472743	0.2740724	8	-12.67089	0.0000
Block 1	Block 2	0.262499	0.2132692	8	1.230833	0.2533
	Block 3	1.577451	0.2129738	8	7.406785	0.0001
Block 2	Block 3	1.314952	0.2126777	8	6.182841	0.0003

SE = standard error; DF = degrees of freedom; p = probability level

Table 20) Sediment loss (g/m²) for each treatment type within the three primary experimental blocks from natural rainfall.

	Block						
Treatment	1	2	3				
Control	23.56	49.08	187.43				
Plant	20.98	18.65	126.29				
Brush	3.32	0.76	2.13				
Hay	0.31	0.38	0.72				
All	0.41	0.43	4.16				

Components of Sediment

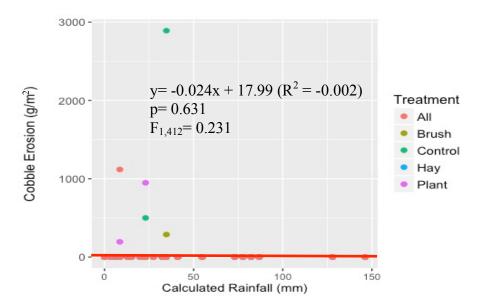


Figure 9) Regression analysis of the effect of calculated rainfall amount (mm) on weight (g/m^2) of eroded cobbles (65 mm < 256 mm) from natural rainfall. Residual standard error = 161.8 on 412 degrees of freedom.

Table 21) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, block, day, and the treatment-day interaction on weight (g/m^2) of eroded cobbles (65 mm < 256 mm) from natural rainfall events.

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	0.2743	0.2743	0.444	0.542
Plot Effects	Treatment	4	1.2525	0.3131	0.507	0.737
	Block	2	0.0002	0.0001	0.000	1.000
	Day	3	1.5562	0.5187	0.840	0.539
	Residuals	4	2.4709	0.6177		
Within Plot	Rainfall (mm)	1	0.46	0.4641	0.729	0.3941
Effects	Day	26	29.85	1.1480	1.803	0.0116
	Treatment x Day	108	50.77	0.4701	0.738	0.9650
	Residuals	264	168.12	0.6368		

DF = degrees of freedom; SS = sum of squares; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level

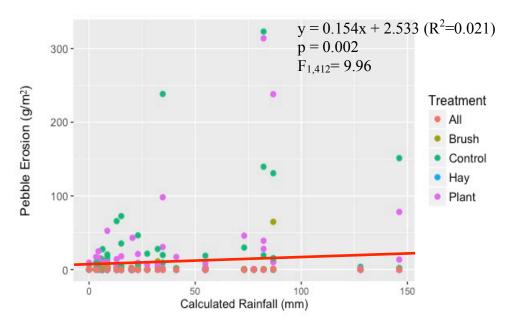


Figure 10) Linear regression of the effect of calculated rainfall (mm) on weight (g/m^2) of eroded pebbles/granules (2 mm > 65 mm) from natural rainfall. Residual standard error = 31.1 on 412 degrees of freedom.

Table 22) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, block, day, and the treatment-day interaction on weight (g/m²) of eroded pebbles/granules (2 mm > 65 mm) from natural rainfall events.

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	10.79	10.79	1.207	0.334
Plot	Treatment	4	167.21	41.80	4.676	0.082
Effects	Block	2	50.64	25.32	2.833	0.171
	Day	3	17.99	6.00	0.671	0.613
	Residuals	4	35.76	8.94		
Within	Rainfall (mm)	1	2.84	2.8354	6.374	0.0122
Plot	Day	26	78.31	3.0118	6.770	2e ⁻¹⁶
Effects	Treatment x Day	108	117.13	1.0845	2.438	3.22e ⁻⁰⁹
	Residuals	264	117.44	0.4449		

Table 23) Mean pebble/granule erosion (g/m²) for each of the treatments from natural rainfall.

Treatment	Pebble/Granule Erosion (g/m²)
Control	18.64
Plant	15.31
Brush	1.17
Hay	0.10
All	0.16

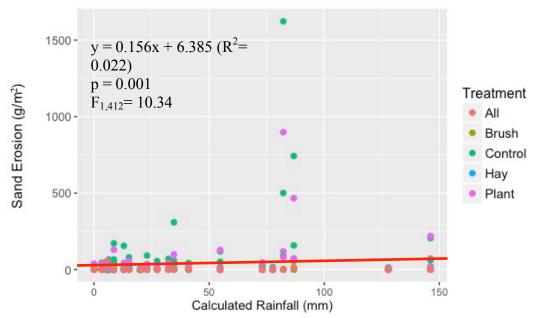


Figure 11) Linear regression of the effect of calculated rainfall (mm) on weight (g/m^2) of eroded sand (< 2 mm) from natural rainfall. Residual standard error = 31.1 on 412 degrees of freedom.

Table 24) Results of the analysis of covariance testing the effect of calculated rainfall amount (mm), treatment, block, day, and the treatment-day interaction on weight (g/m²) of eroded sand (< 2 mm) from natural rainfall events.

	Factor	DF	SS	MS	F	р
Between	Rainfall (mm)	1	35.8	35.84	7. 892	0.048
Plot Effects	Treatment	4	492.5	123.12	27.111	0.004
	Block	2	60.9	30.47	6.709	0.053
	Day	3	10.0	3.33	0.733	0.584
	Residuals	4	18.2	4.54		
Within Plot	Rainfall (mm)	1	11.66	11.655	34.538	1.26e ⁻⁰⁸
Effects	Day	26	145.48	5.595	16.580	$2e^{-16}$
	Treatment x Day	108	134.82	1.248	3.699	2e ⁻¹⁶
	Residuals	264	89.09	0.337		

Table 25) Results of a linear mixed effects analysis, comparing sand (< 2 mm) erosion (g/m²) from natural rainfall events between the treatments and blocks.

Comp	arison	Value	SE	DF t-value p		р
Control	Plant	-0.3273690	0.2852101	8	-1.147817	0.2842
	Brush	-2.4076474	0.2857023	8	-8.427119	0.0000
	Hay	-2.5500438	0.285463	8	-8.933011	0.0000
	All	-2.3876541	0.2849728	8	-8.378535	0.0000
Plant	Brush	-2.0802784	0.2859399	8	-7.275230	0.0001
	Hay	-2.2226748	0.2856984	8	-7.779793	0.0001
	All	-2.0602852	0.2852100	8	-7.223747	0.0001
Block 1	Block 2	0.0807290	0.2213058	8	0.364785	0.7247
	Block 3	0.8695085	0.2211886	8	3.931073	0.0044
Block 2	Block 3	0.7887795	0.2210764	8	3.567905	0.0073

SE = standard error; DF = degrees of freedom; p = probability level

Table 26) Weights (g/m^2) of sand (< 2 mm) eroded from each treatment type within the three primary experimental blocks from natural rainfall.

	Block							
Treatment	1	2	3					
Control	18.30	41.10	144.20					
Plant	16.85	14.25	88.86					
Brush	0.79	0.37	1.21					
Hay	0.21	0.31	0.52					
All	0.20	0.29	3.56					

Profile Gauge Results

Table 27) Results of the analysis of covariance testing the effect of treatment, position within plot (top, middle and bottom), block, presence of vegetation, and the treatment-position within plot interaction on net soil level change (cm) of primary and secondary experimental plots determined by profile gauge measurements.

Factor	DF	SS	MS	F	р
Treatment	4	671	167.7	6.921	1.82e-05
Position	2	438	219.0	9.041	0.000133
Block	5	1933	386.5	15.954	6.64e-15
Vegetation	2	5	2.4	0.097	0.907426
Treatment x	8	614	76.7	3.166	0.001562
Position					
Residuals	700	16959	24.2		

 $SS = sum of squares; DF = degrees of freedom; MS = mean square; F = (MS_{factor})/(MS_{error}); p = probability level$

Table 28) Results of a Tukey multiple comparison of means test comparing the mean net soil level change (cm) of primary and secondary experimental plots between treatments*, positions within the plot (top, middle and bottom), blocks, and the treatment-position within the plot*interaction.

*only significant comparisons shown

Comparison	Difference	LCI	UCI	p adj
Plant-Hay	-2.34	-3.884	-0.790	0.0004
Plant-Brush	-1.82	-3.373	-0.265	0.012
Plant-All	-2.76	-4.270	-1.247	0.000008
Position 1-2	0.99	-0.089	2.067	0.080
Position 1-3	1.92	0.857	2.992	0.00008
Position 2-3	0.93	-0.092	1.963	0.083
Blocks 1-2	-1.53	-3.114	0.043	0.062
Blocks 1-3	-2.16	-3.914	-0.408	0.006
Blocks 2-3	-0.63	-2.383	1.131	0.912
Blocks 121-122	0.01	-2.201	2.223	1.000
Blocks 121-123	0.05	-1.979	2.086	0.999
Blocks 122-123	0.04	-1.891	1.977	0.999
All:3-All:1	3.71	0.587	6.839	0.005
All:3-All:2	3.45	0.525	6.384	0.006
All:3-Control:1	5.69	1.565	9.819	0.0003
All:3-Plant:1	6.30	3.038	9.563	0.000
All:3-Brush:1	4.07	0.993	7.145	0.0008
All:3-Hay:2	3.26	0.334	6.194	0.013
All:3-Plant:3	-5.48	-8.562	-2.409	0.0000002
Plant:1-Hay:1	-3.74	-7.317	-0.168	0.030
Plant:1-Control:3	4.36	0.432	8.294	0.014
Plant:1-Brush:3	3.69	0.275	7.113	0.020
Plant:3-Hay:3	-3.35	-6.563	-0.140	0.031

Difference = difference between the means; LCI = lower bound of 95% confidence interval; UCI = upper bound of 95% confidence interval; p adj = Tukey adjusted probability level

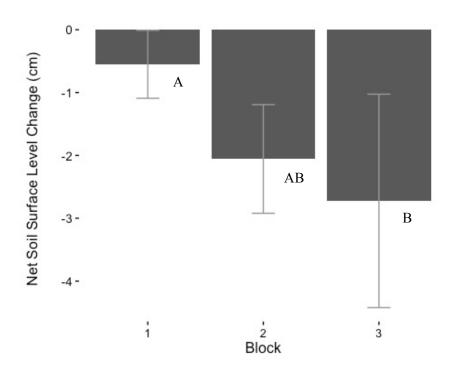


Figure 12) Net change in soil surface level (cm) for plots within primary experimental blocks. Error bars represent 95% confidence intervals.

Table 29) Mean soil surface net change (cm) of primary experimental plots for each

treatment in the three primary experimental blocks.

•	Position							
Treatment	1	2	3					
Control	-5.1	-2.0	-1.5					
Plant	-5.7	-2.3	-5.1					
Brush	-3.5	-2.6	-1.7					
Hay	-1.9	-2.7	-1.6					
All	-3.1	-2.9	0.7					

Plant Survival

Table 30) Results of a two-sample t-test on the percentage of native plant transplants surviving after one year (July/August 2015 - June 10, 2016) in the Plant and All treatment plots.

Treatment	Mean Survival (%)	DF	LCI	UCI	t-value	р
Plant	59.3	10	-32.29	23.68	-0.3426	0.739
All	55.0					

DF = degrees of freedom; LCI = lower bound of 95% confidence interval; UCI = upper bound of 95% confidence interval; p = probability level

Controlled Watering Experiment (CWE) Results

Table 31) Results of a linear mixed effects analysis, comparing runoff (L/m²) between the treatments and blocks for the controlled watering experiments.

Comp	Comparison		SE	DF	t-value	р
Control	Plant	-1.0733718	0.1813090	8	-5.920123	0.0004
	Brush	-1.9847193	0.1624242	8	-12.219357	0.0000
	Hay	-1.7539702	0.1621117	8	-10.819518	0.0000
	All	-1.9708429	0.1630978	8	-12.083813	0.0000
Experiment	Experiment	-1.8147377	0.1619966	9	-11.202319	0.0000
1	2					

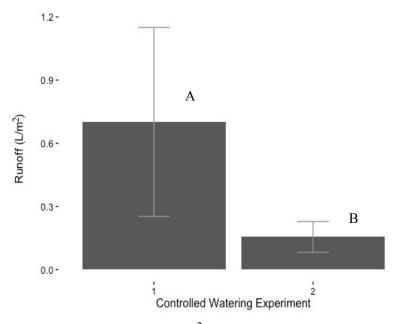


Figure 13) Mean runoff (L/m^2) for the controlled watering experiments. Error bars represent 95% confidence intervals.

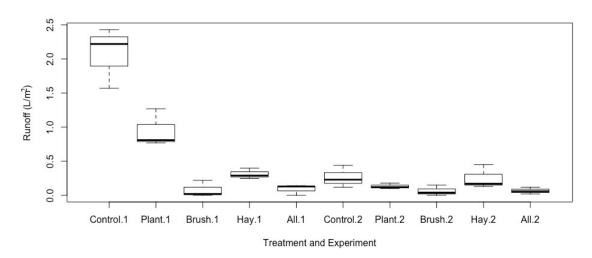


Figure 14) Runoff (L/m^2) from each treatment for controlled watering experiments 1 and 2. Bars represent data range.

Runoff Lag

Table 32) Results of the analysis of covariance testing the effect of treatment, block, plot slope (%), experiment, and treatment-experiment interaction on runoff lag time (sec) for the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	6891	1723	0.478	0.752
Plot Effects	Block	5	19647	3929	1.090	0.403
	Plot Slope	1	555	555	0.154	0.700
	Residuals	16	57683	3605		
Within Plot	Experiment	1	74973	74973	48.467	7.09e ⁻⁰⁷
Effects	Plot Slope	1	2095	2095	1.355	0.2575
	Treatment x	4	18536	4634	2.996	0.0421
	Experiment					
	Residuals	21	32485	1547		

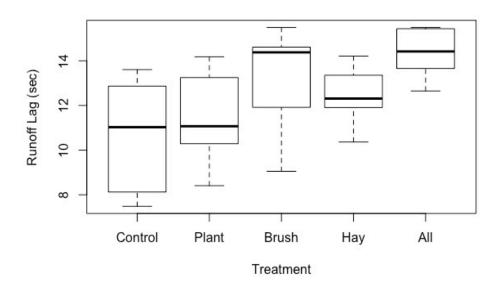


Figure 15) Runoff lag time (sec) for each of the treatments. Bars represent data range.

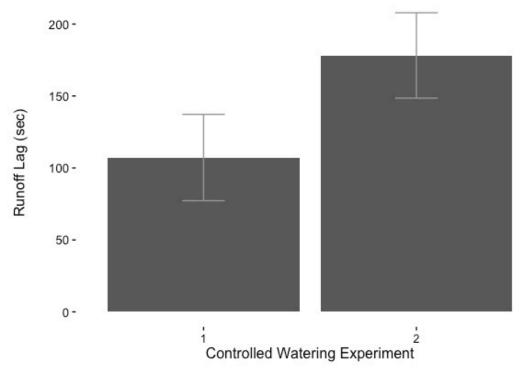


Figure 16) Mean runoff lag time (sec) for the controlled watering experiments. Error bars represent 95% confidence intervals.

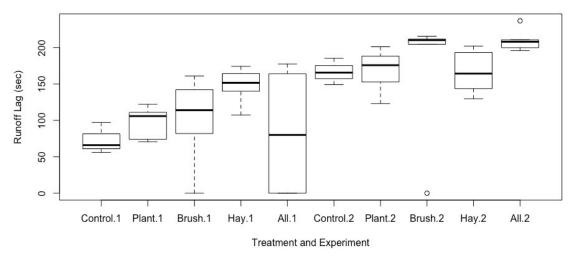


Figure 17) Runoff lag time (sec) for each treatment from controlled watering experiments 1 and 2. Bars represent data range, dots represent outliers.

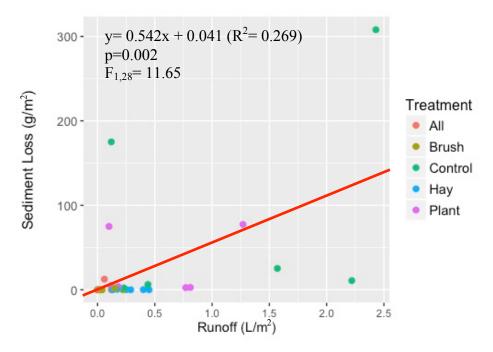


Figure 18) Linear regression analysis of the effect of runoff (L/m^2) on sediment loss (g/m^2) for controlled watering experiments. Residual standard error = 55.3 on 28 degrees of freedom.

Table 33) Results of the analysis of covariance testing the effect of runoff (L/m^2) , treatment, block, plot slope (%), experiment, and treatment-experiment interaction on sediment loss (g/m^2) for the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Runoff	1	48.93	48.93	38.234	0.0008
Plot Effects	Treatment	4	10.73	2.68	2.095	0.199
	Block	2	10.99	5.49	4.293	0.069
	Plot Slope	1	0.85	0.85	0.667	0.445
	Residuals	6	7.68	1.28		
Within Plot	Runoff	1	0.8407	0.8407	5.509	0.047
Effects	Experiment	1	0.7022	0.7022	4.601	0.064
	Plot Slope	1	0.4752	0.4752	3.114	0.116
	Treatment x	4	0.9022	0.2256	1.478	0.295
	Experiment					
	Residuals	8	1.2208	0.1526		

Table 34) Results of the analysis of covariance testing the effect of treatment, block, plot slope (%), experiment, and treatment-experiment interaction on sediment loss (g/m²) for the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	49.58	12.395	9.704	0.006
Plot Effects	Block	2	19.76	3929	7.734	0.017
	Plot Slope	1	0.90	0.901	0.705	0.429
	Residuals	7	8.94	1.277		
Within Plot	Experiment	1	0.0187	0.0187	0.138	0.719
Effects	Plot Slope	1	0.7507	0.7507	5.533	0.043
	Treatment x	4	2.1506	0.5377	3.963	0.040
	Experiment					
	Residuals	9	1.2210	0.1357		

Table 35) Results of a linear mixed effects analysis, comparing sediment loss (g/m²) between treatments and blocks for the controlled watering experiments.

Compa	arison	Value	SE	DF	t-value	р
Control	Plant	-1.960092	0.892366	8	-2.196512	0.0593
	Brush	-3.684587	1.269667	8	-2.902009	0.0198
	Hay	-3.603535	1.166073	8	-3.090317	0.0149
	All	-3.501272	1.259375	8	-2.780165	0.0239
Plant	Brush	-1.724495	0.905722	8	-1.903999	0.0934
	Hay	-1.643443	0.839191	8	-1.958366	0.0859
	All	-1.541180	0.891445	8	-1.728855	0.1221
Block 1	Block 2	0.580046	0.532431	8	1.089430	0.3077
	Block 3	2.070235	0.505965	8	4.091658	0.0035
Block 2	Block 3	1.490188	0.496470	8	3.0015699	0.0170

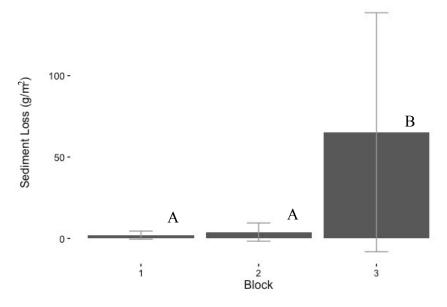


Figure 19) Sediment loss (g/m^2) from primary experimental blocks for the controlled watering experiments. Error bars represent 95% confidence intervals.

Sediment Grain Size

Table 36) Mean weight (g) of sediment components based on Udden-Wentworth grain size classification eroded from each of the treatments during two controlled watering experiments (CWEs). Sediment loss data from primary experimental plots.

		Treatment									
		Con	trol	Pla	ant	Bru	ush	Hay		All	
Size Class	CWE	1	2	1	2	1	2	1	2	1	2
Pebbles (g)		0.58	0.28	0.58	3.05	0.00	0.52	0.00	0.00	0.27	2.62
Granules (g)		7.59	3.09	4.15	3.13	0.00	0.26	0.01	0.01	0.02	0.70
V.C.S (g)		7.94	3.00	5.46	1.98	0.01	0.28	0.01	0.04	0.08	0.68
C.S (g)		6.08	2.94	5.30	1.92	0.02	0.23	0.01	0.06	0.07	0.65
M.S (g)		4.94	2.41	4.25	1.48	0.04	0.17	0.03	0.05	0.07	0.58
F.S (g)		2.61	1.25	3.02	1.00	0.04	0.12	0.03	0.04	0.06	0.32
V.F.S (g)		3.51	1.64	2.77	0.81	1.52	0.09	1.41	0.02	0.86	0.22
Silt/Clay (g)		4.66	2.66	2.48	1.85	0.01	0.07	0.01	0.02	0.03	0.51

Table 37) Mean proportion of each sediment size class, as defined by Gradistat, in samples eroded from each of the treatments during the two controlled watering experiments (CWE). Sediment loss data from primary experimental plots.

		Treatment									
		Con	trol	Pla	ant	Bru	ush	Н	ay		All
Size Class	CWE	1	2	1	2	1	2	1	2	1	2
Gravel		0.35	0.27	0.21	0.37	0.01	0.61	0.01	0.19	0.26	0.36
Sand		0.65	0.73	0.79	0.63	0.99	0.39	0.99	0.81	0.74	0.64
V.C.G		0.01	0.01	0.01	0.08	0.00	0.30	0.00	0.00	0.20	0.21
F.G		0.15	0.11	0.07	0.14	0.00	0.15	0.01	0.08	0.01	0.06
V.F.G		0.19	0.15	0.13	0.15	0.01	0.16	0.00	0.11	0.05	0.09
V.C.S		0.16	0.16	0.17	0.17	0.01	0.13	0.01	0.21	0.05	0.22
C.S		0.14	0.16	0.14	0.15	0.03	0.10	0.02	0.24	0.05	0.19
M.S		0.08	0.10	0.09	0.10	0.02	0.07	0.02	0.17	0.04	0.22
F.S		0.13	0.10	0.30	0.07	0.93	0.05	0.93	0.11	0.57	0.06
V.F.S		0.14	0.19	0.08	0.13	0.00	0.04	0.01	0.07	0.02	0.05

Table 38) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of gravel in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	20.20	5.049	3.905	0.048
Plot Effects	Block	2	17.25	8.623	6.669	0.020
	Residuals	8	10.34	1.293		
Within Plot	Experiment	1	5.383	5.383	8.635	0.015
Effects	Treatment x	4	4.530	1.133	1.817	0.202
	Experiment					
	Residuals	10	6.234	0.623		

Table 39) Results of a linear mixed effects analysis, comparing proportion of gravel in eroded sediment between treatments and blocks for the controlled watering experiments.

Compa	rison	Value	SE	DF	t-value	р
Control	Plant	-0.7640569	0.7993353	8	-0.955865	0.367
	Brush	-2.9794277	0.7993353	8	-3.727381	0.006
	Hay	-2.7008936	0.7993353	8	-3.378924	0.010
	All	-1.6724249	0.7993353	8	-2.092269	0.070
Plant	Brush	-2.2152000	0.7992270	8	-2.771678	0.024
	Hay	-1.9364667	0.7992270	8	-2.422924	0.042
	All	-0.9080000	0.7992270	8	-1.136098	0.289
All	Brush	-1.3072000	0.7992270	8	-1.635580	0.141
	Hay	-1.0284667	0.7992270	8	-1.286827	0.234
Block 1	Block 2	0.580046	0.532431	8	1.089430	0.308
	Block 3	2.070235	0.505965	8	4.091658	0.004
Block 2	Block 3	1.490188	0.496470	8	3.0015699	0.017

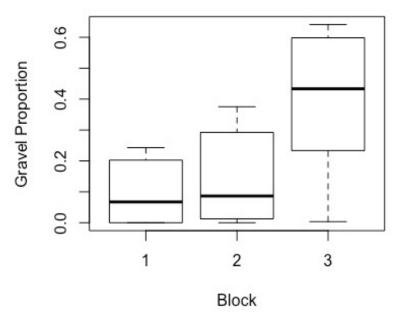


Figure 20) Proportion of gravel in eroded sediments from each of the primary experimental blocks for the controlled watering experiments. Bars represent data range.

Table 40) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of sand in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	62.23	15.556	1.724	0.237
Plot Effects	Block	2	8.87	4.435	0.491	0.629
	Residuals	8	72.18	9.023		
Within Plot	Experiment	1	5.383	5.383	8.635	0.015
Effects	Treatment x	4	4.530	1.133	1.817	0.202
	Experiment					
	Residuals	10	6.234	0.623		

Table 41) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of very coarse gravel in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	4.036	1.009	0.944	0.486
Plot Effects	Block	2	12.596	6.298	5.891	0.027
	Residuals	8	8.552	1.069		
Within Plot	Experiment	1	0.695	0.6951	1.729	0.218
Effects	Treatment x	4	1.323	0.3307	0.823	0.540
	Experiment					
	Residuals	10	4.019	0.4019		

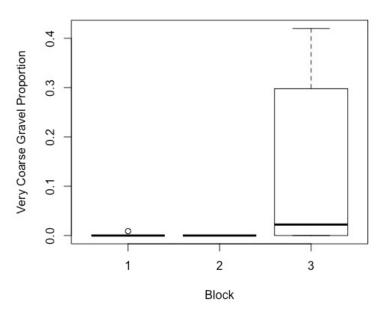


Figure 21) Proportion of very coarse gravel in eroded sediments from each of the primary experimental blocks from the controlled watering experiments. Bars represent data range, dots represent outliers.

Table 42) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of fine gravel in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	10.950	2.7376	8.046	0.007
Plot Effects	Block	2	4.987	2.4933	7.328	0.016
	Residuals	8	2.722	0.3402		
Within Plot	Experiment	1	1.846	1.8461	4.093	0.071
Effects	Treatment x Experiment	4	1.673	0.4183	0.927	0.486
	Residuals	10	4.511	0.4511		

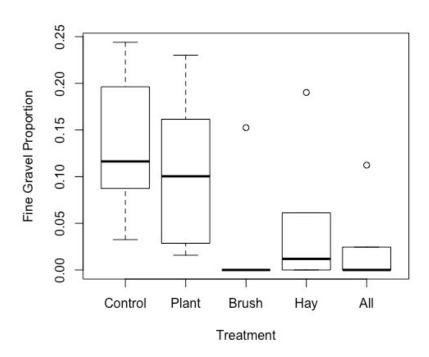


Figure 22) Proportion of fine gravel in eroded sediment from the Control and treated plots for the controlled watering experiments. Bars represent data range, dots represent outliers.

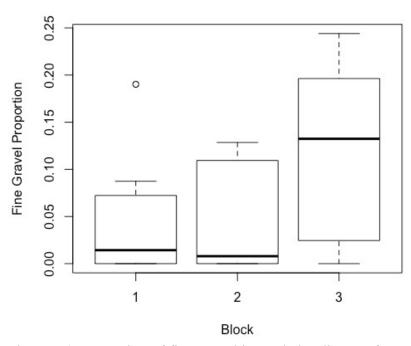


Figure 23) Proportion of fine gravel in eroded sediments from the primary experimental blocks for the controlled watering experiments. Bars represent data range, dots represent outliers.

Table 43) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of very fine gravel in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	13.904	3.476	14.9	0.000527
Plot Effects	Block	2	3.639	3.639	15.6	0.003358
	Residuals	8	2.100	0.233		
Within Plot	Experiment	1	1.930	1.9301	8.26	0.0166
Effects	Treatment x	4	2.617	0.6543	2.80	0.0850
	Experiment					
	Residuals	10	2.337	0.2337		

Table 44) Results of a linear mixed effects analysis, comparing the proportion of very fine gravel in eroded sediment between treatments and blocks from the controlled watering experiments.

Comparison		Value	SE	DF	t-value	р
Control	Plant	-0.3893667	0.4062174	8	-0.958518	0.3659
	Brush	-2.1981000	0.4062174	8	-5.411142	0.0006
	Hay	-2.1166333	0.4062174	8	-5.210593	0.0008
	All	-1.5794333	0.4062174	8	-3.888148	0.0046
Plant	Brush	-1.8087333	0.4062174	8	-4.452624	0.0021
	Hay	-1.7272667	0.4062174	8	-4.252075	0.0028
	All	-1.1900667	0.4062174	8	-2.929630	0.0190
Block 1	Block 2	0.393490	0.2287259	8	1.720356	0.1237
	Block 3	0.853080	0.2287259	8	3.729704	0.0058
Block 2	Block 3	0.4595900	0.2287259	8	2.009348	0.0794

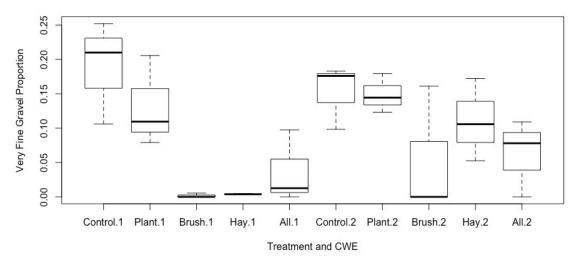


Figure 24) Proportion of very fine gravel in eroded sediment from the Control and treated plots for each of the two controlled watering experiments (CWE). Bars represent data range.

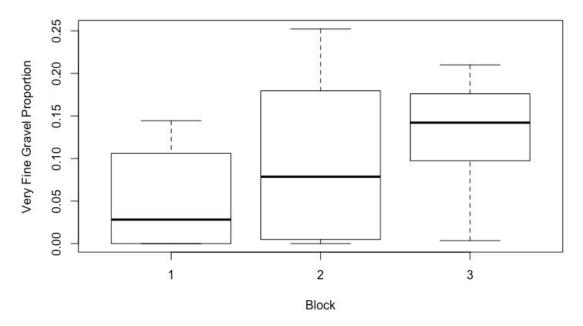


Figure 25) Proportion of very fine gravel in eroded sediments from the primary experimental blocks for the controlled watering experiments. Bars represent data range.

Table 45) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of very coarse sand in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	14.007	3.502	7.178	0.009
Plot Effects	Block	2	1.827	0.914	1.873	0.215
	Residuals	8	3.903	0.488		
Within Plot	Experiment	1	3.956	3.956	10.565	0.009
Effects	Treatment x Experiment	4	4.397	1.099	2.936	0.076
	Residuals	10	3.744	0.374		

Table 46) Results of a linear mixed effects analysis, comparing the proportion of very coarse sand in eroded sediment between treatments for the controlled watering experiments.

Compa	rison	Value	SE	DF	t-value	р
Control	Plant	0.0553667	0.5361153	8	0.103274	0.9203
	Brush	-1.9935000	0.5361153	8	-3.718417	0.0059
	Hay	-1.8361000	0.5361153	8	-3.424823	0.0090
	All	-1.5264667	0.5361153	8	-2.847273	0.0216

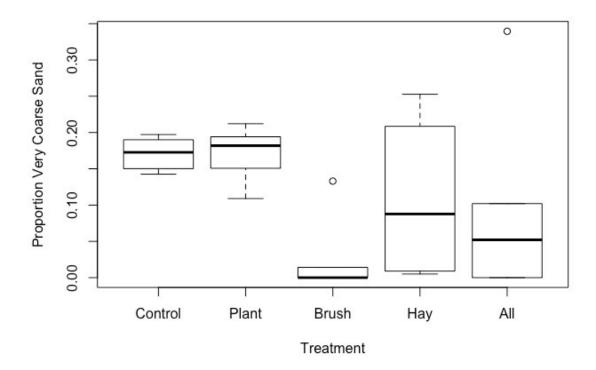


Figure 26) Proportion of very coarse sand in eroded sediment from the Control and treated plots for the controlled watering experiments. Bars represent data range, dots represent outliers.

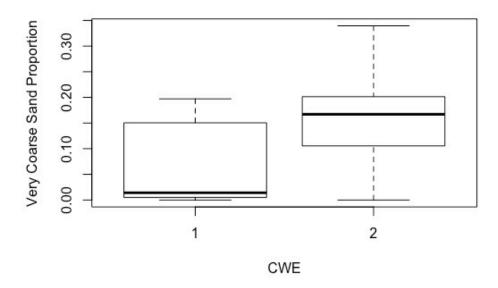


Figure 27) Proportion of very coarse sand in eroded sediment from the two controlled watering experiments (CWE). Bars represent data range.

Table 47) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of coarse sand in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	11.704	2.9260	5.741	0.014
Plot Effects	Block	2	1.095	1.0952	2.149	0.177
	Residuals	8	4.587	0.5096		
Within Plot	Experiment	1	3.208	3.208	11.383	0.007
Effects	Treatment x	4	3.854	0.963	3.419	0.052
	Experiment					
	Residuals	10	2.818	0.282		

Table 48) Results of a linear mixed effects analysis, comparing the proportion of coarse sand in eroded sediment between treatments from the controlled watering experiments.

Comparison		Value	SE	DF	t-value	р
Control	Plant	0.0516000	0.5324306	8	0.096914	0.9252
	Brush	-1.7181000	0.5324306	8	-3.226899	0.0121
Hay		-1.3842000	0.5324306	8	-2.599775	0.0316
	All	-1.3153667	0.5324306	8	-2.470494	0.0387
Plant	Brush	-1.7697000	0.5324306	8	-3.323813	0.0105
	Hay	-1.4358000	0.5324306	8	-2.696689	0.0272
	All	-1.3669667	0.5324306	8	-2.567408	0.0333

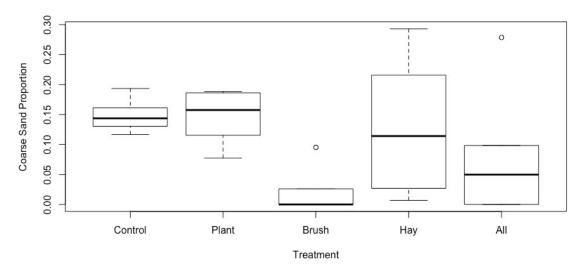


Figure 28) Proportion of coarse sand in eroded sediment from the Control and treated plots for the controlled watering experiments. Bars represent data range, dots represent outliers.

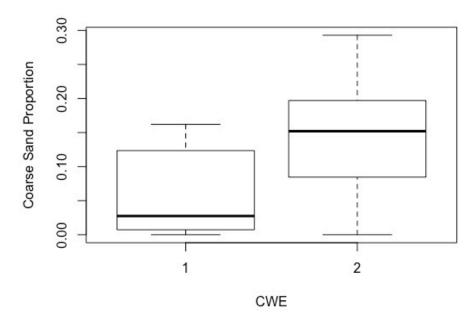


Figure 29) Proportion of coarse sand in eroded sediment from the two controlled watering experiments (CWE). Bars represent data range.

Table 49) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of medium sand in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	6.987	1.7468	4.081	0.0431
Plot Effects	Block	2	0.631	0.3157	0.738	0.5081
	Residuals	8	3.424	0.4280		
Within Plot	Experiment	1	2.073	2.0727	9.025	0.0132
Effects	Treatment x	4	2.614	0.6536	2.846	0.0819
	Experiment					
	Residuals	10	2.296	0.2297		

Table 50) Results of a linear mixed effects analysis, comparing proportion of medium sand in eroded sediment between treatments from the controlled watering experiments.

Comparison		Value	SE	DF	t-value	р
Control	Plant	0.1530333	0.4681961	8	0.326857	0.7522
	Brush	-1.2239667	0.4681961	8	-2.614218	0.0309
	Hay	-0.8860667	0.4681961	8	-1.892512	0.0951
	All	-0.8899333	0.4681961	8	-1.900771	0.0939
Plant	Brush	-1.3770000	0.4681961	8	-2.941075	0.0187
	Hay	-1.0391000	0.4681961	8	-2.219369	0.0572
	All	-1.0429667	0.4681961	8	-2.227628	0.0565
Brush	Hay	0.337900	0.4681961	8	0.721706	0.4910
	All	0.334033	0.4681961	8	0.713448	0.4959

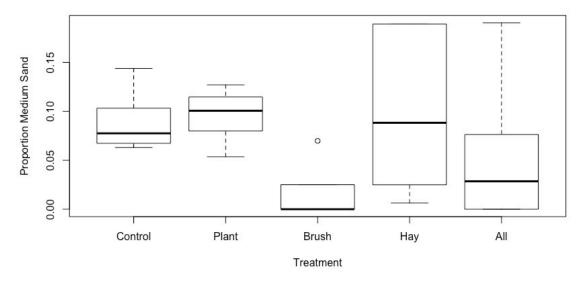


Figure 30) Proportion of medium sand in eroded sediment from the Control and treated plots for the controlled watering experiments. Bars represent data range, dots represent outliers.

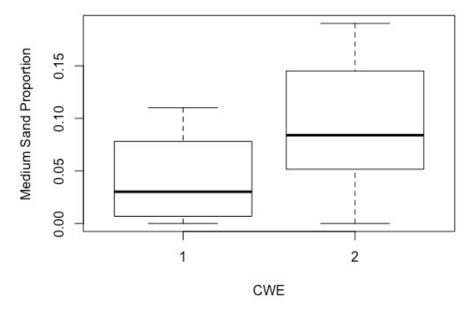


Figure 31) Proportion of medium sand in eroded sediment from the two controlled watering experiments (CWE). Bars represent data range.

Table 51) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of fine sand in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	26.10	6.526	1.539	0.280
Plot Effects	Block	2	2.08	1.040	0.245	0.788
	Residuals	8	33.93	4.241		
Within Plot	Experiment	1	26.59	26.595	17.664	0.00182
Effects	Treatment x	4	13.77	3.443	2.286	0.13171
	Experiment					
	Residuals	10	15.06	1.506		

Table 52) Results of a linear mixed effects analysis, comparing the proportion of fine sand in eroded sediment between treatments for the controlled watering experiments.

Comparison		Value	SE	DF	t-value	р
Control	Plant	0.830933	1.3840508	8	0.6003633	0.5649
	Brush	0.133633	1.3840508	8	0.0965523	0.9255
	Hay	4.230967	1.3840508	8	3.0569446	0.0157
	All	1.046133	1.3840508	8	0.7558489	0.4714
Hay	Plant	-3.400033	1.3840508	8	-2.456581	0.0395
	Brush	-4.097333	1.3840508	8	-2.960392	0.0181
	All	-3.184833	1.3840508	8	-2.301096	0.0504

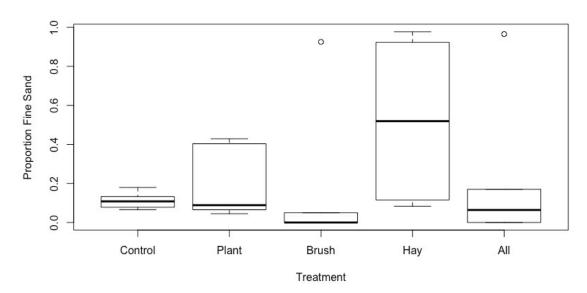


Figure 32) Proportion of fine sand in eroded sediment from the Control and treated plots for the controlled watering experiments. Bars represent data range, dots represent outliers.

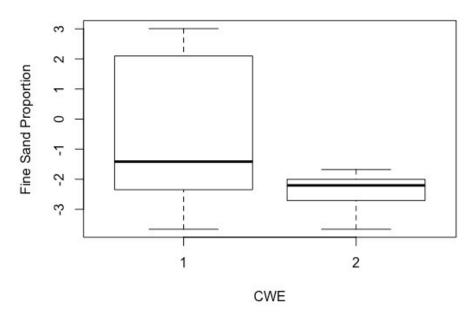


Figure 33) Proportion of fine sand in eroded sediment from the two controlled watering experiments (CWE). Bars represent data range. Bars represent data range.

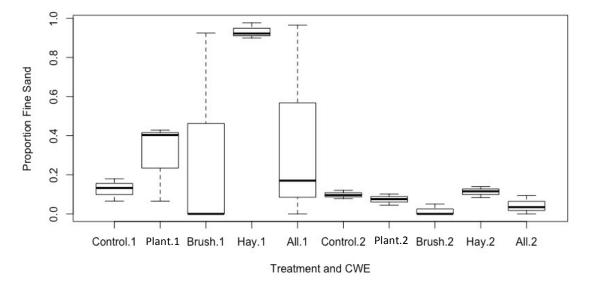


Figure 34) Proportion of fine sand in eroded sediment from the Control and treated plots for each of the two controlled watering experiments (CWE). Bars represent data range.

Table 53) Results of the analysis of covariance testing the effect of treatment, block, experiment, and treatment-experiment interaction on proportion of very fine sand in eroded sediments from the controlled watering experiments.

	Factor	DF	SS	MS	F	р
Between	Treatment	4	16.208	4.052	15.789	0.000727
Plot Effects	Block	2	0.9889	0.2472	2.488	0.11048
	Residuals	8	2.053	0.257		
Within Plot	Experiment	1	1.9778	1.9778	19.907	0.00121
Effects	Treatment x	4	0.9889	0.2472	2.488	0.11048
	Experiment					
	Residuals	10	0.9935	0.0994		

Table 54) Results of a linear mixed effects analysis, comparing the proportion of very fine sand in eroded sediment between treatments for the controlled watering experiments.

Compa	arison	Value	SE	DF	t-value	р
Control	Plant	-0.5049000	0.3444727	8	-1.465718	0.1809
	Brush	-1.9366667	0.3444727	8	-5.622119	0.0005
	Hay	-1.8121333	0.3444727	8	-5.260600	0.0008
	All	-1.6264000	0.3444727	8	-4.721419	0.0015
Plant	Brush	-1.4317667	0.3444727	8	-4.156401	0.0032
	Hay	-1.3072333	0.3444727	8	-3.794882	0.0053
	All	-1.1215000	0.3444727	8	-3.255700	0.0116

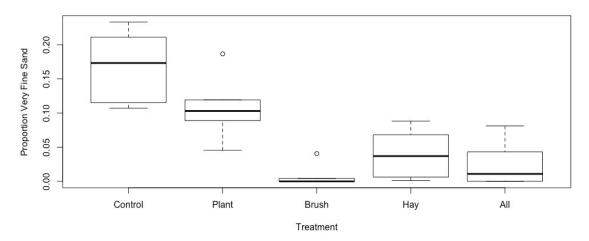


Figure 35) Proportion of very fine sand in eroded sediment from the Control and treated plots for the controlled watering experiments. Bars represent data range, dots represent outliers.

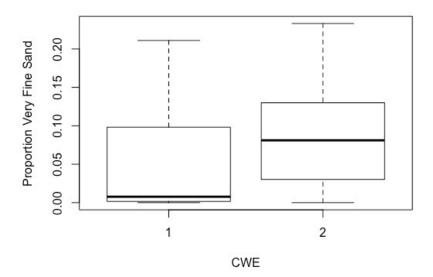


Figure 36) Proportion of very fine sand in eroded sediment from the two controlled watering experiments (CWE). Bars represent data range.

Chapter 3:

Small Coastal Bluff Results

Site 1

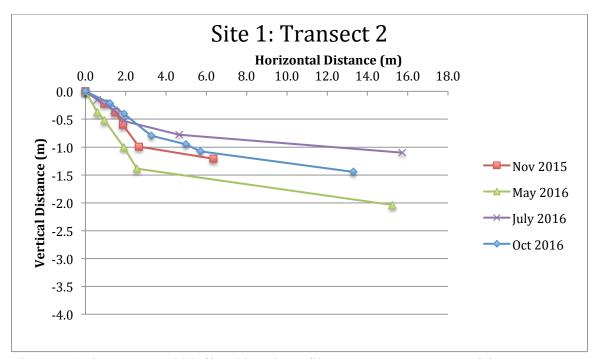


Figure 37) Site 1 seasonal bluff and beach profiles at transect 2 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

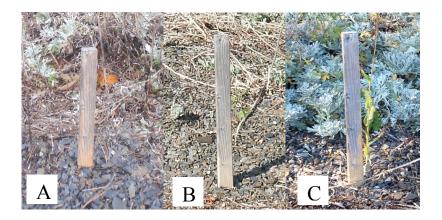


Figure 38) Site 1 transect 2: Close up of stake 3 at the edge of the Living Shoreline vegetation showing actual fluctuation in beach level at the position between (A) November 2016, (B) May 2016, and (C) October 2016.

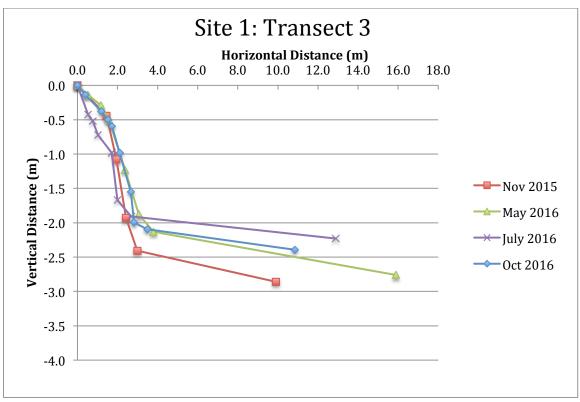


Figure 39) Site 1 seasonal bluff and beach profiles at transect 3 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

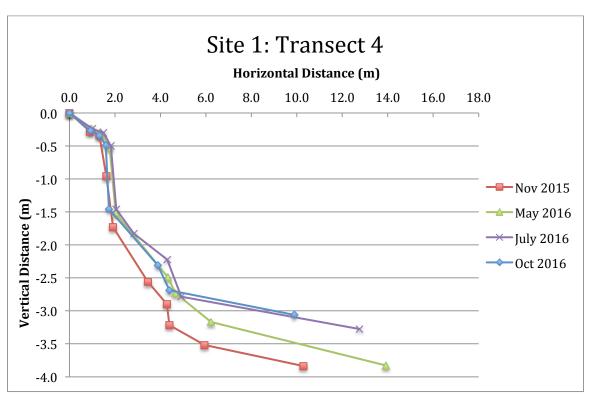


Figure 40) Site 1 seasonal bluff and beach profiles at transect 4 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

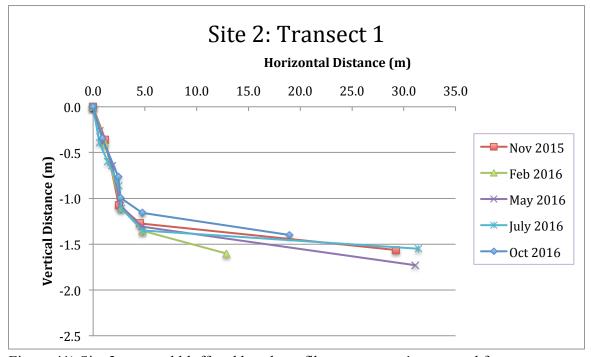


Figure 41) Site 2 seasonal bluff and beach profiles at transect 1 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

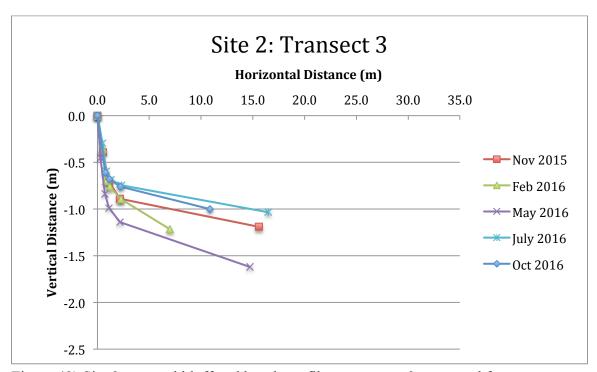


Figure 42) Site 2 seasonal bluff and beach profiles at transect 3 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.



Figure 43) Site 2 transect 3 showing stump displacement between November 2015 and February 2016. Vegetation cover in October 2016 also shown.

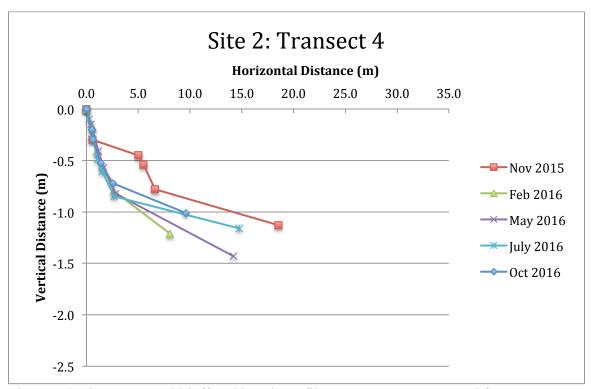


Figure 44) Site 2 seasonal bluff and beach profiles at transect 4 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

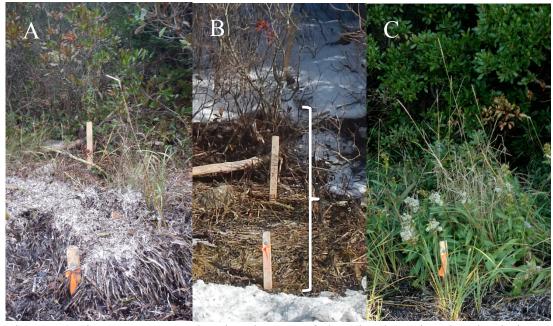


Figure 45) Site 2 transect 4: Showing the area of elevation lowering (bracket in image B) from the bluff toe to stake 3 indicated in Appendix Figure 43 between (A) November 2015 and (B) February 2016. Image C shows same area with vegetation in October 2016.

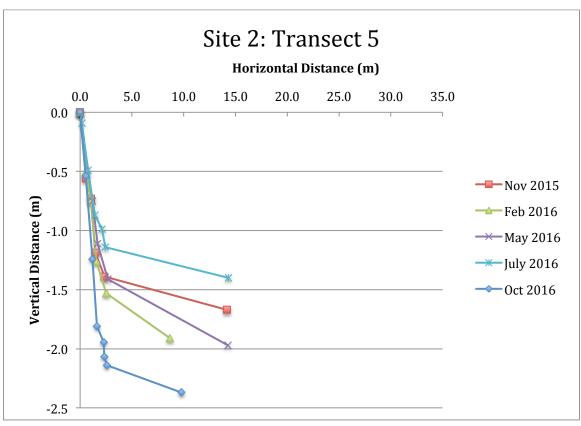


Figure 46) Site 2 seasonal bluff and beach profiles at transect 5 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

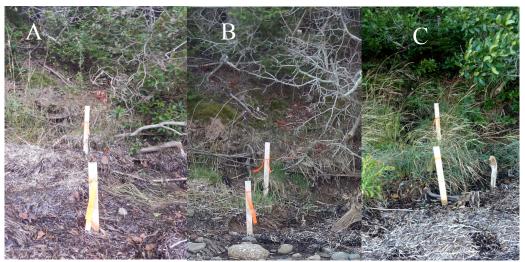


Figure 47) Site 2 transect 5: Showing scouring along the bluff toe apparent in May 2016 (B) and location of the transect directly across a few protective tufts of vegetation. Images A and C show the transect in November 2015 and October 2016, respectively.

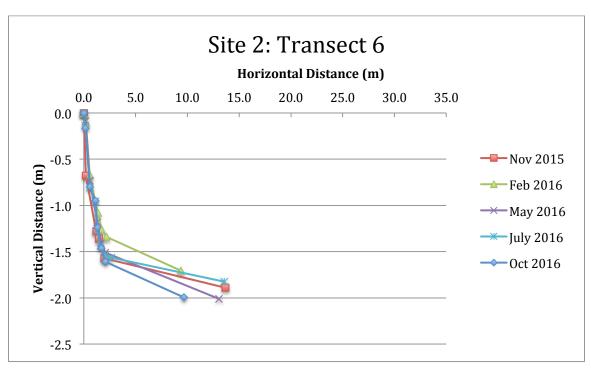


Figure 48) Site 2 seasonal bluff and beach profiles at transect 6 generated from survey data. Axis values are in meters. Point 0.0, 0.0 corresponds with benchmark stake 1 at the bluff crest.

Site 4: Sand Dune

Table 55) Results of an analysis of variance (ANOVA) testing the effect of the position of erosion pins within the dune on the percent vegetation cover within a 100 cm² square quadrat around the pins.

Factor	DF	SS	MS	F	р
Position	2	24074	12037	1361	7.37e ⁻¹¹
Residuals	8	71	9		

Table 56) Results of a Tukey multiple comparison of means, comparing differences in the mean percent vegetation cover within a 100 cm² square quadrat around the erosion pins between the positions of erosion pins within the dune.

Comparison	Difference	LCI	UCI	p adj
Toe – Back	-9.725000e ⁰¹	-103.740145	-90.759855	0
Toe – Face	-9.725000e ⁰¹	-103.258707	-91.24129	0
Back – Face	-1.421085e ⁻¹⁴	-6.490145	6.490145	1

Difference = difference between the means; LCI = lower bound of 95% confidence interval; UCI = upper bound of 95% confidence interval; p adj = Tukey adjusted probability level

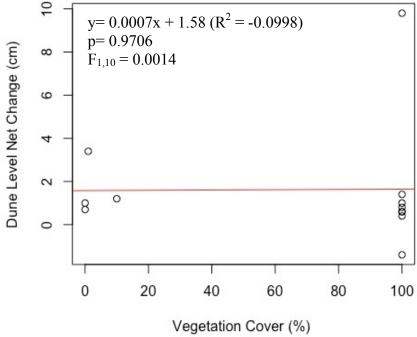


Figure 49) Linear regression analysis of the effect of vegetation cover (%) on the net change in dune sediment level between September 2015 and October 2016 measured by erosion pins at 12 locations at Site 4. Residual standard error = 2.9 on 10 degrees of freedom.

Table 57) Results of an analysis of variance (ANOVA) testing the effect of the position of the pin within the dune on the net change is dune sediment level.

Factor	DF	SS	MS	F	р
Position	2	12.52	6.258	0.774	0.49
Residuals	9	72.77	8.085		

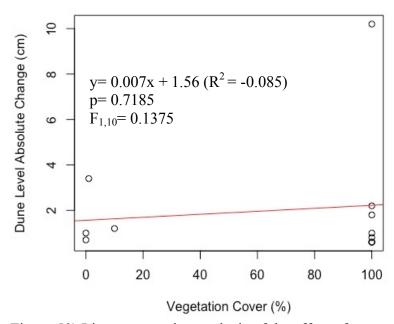


Figure 50) Linear regression analysis of the effect of vegetation cover (%) on the absolute change in dune sediment level measured by erosion pins at 12 locations, four times between September 2015 and October 2016. Residual standard error = 2.8 on 10 degrees of freedom.

Table 58) Results of an analysis of variance (ANOVA) testing the effect of the position of the pin within the dune on the absolute change is dune sediment level.

Factor	DF	SS	MS	F	р
Position	2	0.007	0.0033	0.004	0.996
Residuals	9	8.098	0.8998		

Table 59) Position of dune toe vegetation during seasonal measurements relative to base line of 0 established in September 2015.

	Dune Toe Vegetation (m)							
Transect	May 2016	June 2016	Aug 2016	Oct 2016				
Α	-1.95	-0.70	-0.67	-0.68				
В	-0.82	-0.83	-0.715	-0.71				
С	-0.54	-0.55	-0.56	-0.58				
D	-2.62	-2.29	-2.29	-2.07				

Table 60) Results of an analysis of variance (ANOVA) testing the effect of the position of erosion pins within the dune on the number of plant species within a 100 cm² square quadrat around the pins.

Factor	DF	SS	MS	F	р
Position	2	36.50	18.250	21.66	0.0004
Residuals	9	7.58	0.843		

Table 61) Results of a linear mixed effects analysis, comparing the number of plant species within a 100 cm² square quadrat around the erosion pins between the positions of erosion pins within the dune.

Comparison		Value	SE	DF	t-value	р
Dune Toe	Back Dune	2.000	0.3747427	9	5.336995	0.0005
	Dune Face	2.250	0.3747427	9	6.004119	0.0002
Dune Face	Back Dune	2.250	0.3747427	9	0.667124	0.5214