GEOMORPHIC AND ECOLOGIC PATTERNS AFTER FIRE

WITHIN THE ALPINE TREELINE ECOTONE,

GLACIER NATIONAL PARK, MONTANA

DISSERTATION

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San Marcos, Texas
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DEDICATION

I dedicate this dissertation to my parents, Donald and Sherry Stine.

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First off, I acknowledge and thank my committee. Without their guidance and patience, this dissertation would not have been possible. Thanks to Dr. Suckling for stepping in and being willing to serve on my committee. Thanks to Dr. Dixon for answering my many questions and for his patient guidance. Thanks to Dr. Malanson for his input, guidance, and visiting my field sites. Special thanks to my advisor, Dr. Butler, for his seemingly endless patience, invaluable guidance throughout this process, and for introducing me to the beautiful Glacier National Park. His knowledge, understanding, and support were instrumental to the completion of this degree and dissertation.

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ABSTRACT

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The overall purpose of this study was to evaluate geomorphic and ecologic conditions after recent fires (within the past 10 years) within the alpine treeline ecotone of Glacier National Park, Montana. Specific objectives were focused on characterizing post-fire conditions at three sites, comparing burned to adjacent unburned areas, evaluating results in regard to microtopographic variables, and assessing potential geomorphic-ecologic relationships. Results revealed that 1) several soil variables differed significantly between burned and unburned areas; 2) most of the seedling micro-site variables were significantly different between burned and unburned areas, and seedlings were found to be strongly associated with several fine-scale factors; 3) fine-scale topographic variability corresponded to several soil conditions assessed; and 4) relationships between geomorphic and ecologic factors were significant. The results of this study contribute to a better understanding of alpine treeline disturbance and the geomorphic effects of fire in this high elevation area. As expected, burned areas were significantly different in regard to several geomorphic and ecologic conditions relative to adjacent unburned areas. However, results were not uniform within burned areas. Fine-scale factors were found to be more important to seedling establishment patterns and several soil conditions than more coarse-scale variables within the context of the treeline ecotone. These results contribute to advancing knowledge of biogeomorphic disturbance and ecotone dynamic theories and provide applied information for alpine treeline dynamics and Park Service management.

CHAPTER 1: INTRODUCTION

The alpine treeline ecotone, the zone of transition between the subalpine forest and alpine tundra, is used as a proxy of global climate change (Hofgaard 1997; Camarero and Gutiérrez 2007; Batllori and Gutiérrez 2008), however, disturbance will alter its response to climate conditions (Butler et al. 2007; Tomback and Resler 2007). Very little is known regarding the effects of fire disturbance within the alpine treeline ecotone (treeline) because it was uncommon in the past for fire to reach into this area. Recently, however, climate change and past management practices have resulted in increased occurrences of fires reaching into treeline (Fig.1.1). The goal of this study was to evaluate the effects of fire on geomorphic conditions and associated vegetation, and the potential relationships between these two components, within the alpine treeline ecotone of Glacier National Park, Montana (GNP). Studies have indicated that treeline is affected by geomorphic processes and features within the Northern Rockies (Walsh et al. 2003; Butler et al. 2004; Butler et al. 2007), and fire is well known to influence geomorphic conditions (*e.g.* Swanson 1978; Shakesby and Doerr 2006). The effects of fire on geomorphic factors may subsequently influence vegetation establishment. The combined factors of the effect of fire on vegetation and geomorphology, the harsh environmental conditions found at treeline, and the interactions between geomorphology and vegetation establishment accentuate the need for research that incorporates these multiple, interacting components. This study was innovative in assessing the processes and feedbacks between geomorphic and ecologic factors after disturbance (Viles et al. 2008;

Figure 1.1. The Red Eagle Fire that burned in Glacier National Park, MT in 2006 reached into the treeline ecotone on Divide Mountain, pictured above. Photograph taken in 2010.

Rice et al. 2012). This focus is imperative for holistic understanding of processes, and as such, will contribute to improved management and conservation information and advance emerging theories and methods of coupled geomorphic–ecologic studies.

This research topic is becoming increasingly relevant because upslope movement of treeline is viewed as an indication of a warming climate (Rochefort et al. 1994; Pauli et al. 1996; Grace et al. 2002; Körner and Paulsen 2004), but not enough is known regarding the many confounding variables, including geomorphic components and disturbance, that affect treeline processes to be able to fully understand the movement and position of this ecotone (Butler and Walsh 1990; Daniels and Veblen 2003; Resler et al. 2005; Butler et al. 2007; Tomback and Resler 2007; Resler and Tomback 2008; Holtmeier 2009). There is increasing acknowledgement of the need in understanding fine–scale influences on treeline position before it can be successfully used as a climate change indicator (Danby and Hik 2007; Elliot 2011). Fire at treeline may suppress tree establishment episodically, resulting in a treeline location lower in elevation than it would be if only climate controlled (Bollinger 1973; Noble 1993). Fire may alter factors such as erosion, rock weathering, soil conditions, and vegetation cover – aspects that contribute to the various ecological and hydrological functions of the treeline ecotone. These factors may, alternatively, also facilitate the establishment of seedlings, possibly resulting in an increase in tree establishment and growth and upward tree establishment in the long term.

Climate change is also projected to increase the frequency and intensity of fires (Running 2006; Westerling et al. 2006; Marlon et al. 2009), and therefore, increasing the probability of fire reaching into the treeline ecotone. This research took advantage of recent fires (within the past 10 years) that have occurred in the treeline ecotone in protected areas that were allowed to burn within GNP. The results of this study will advance our understanding of the role of fire as a geomorphic agent, coupled geomorphic–ecologic theory and response to disturbance, and alpine treeline dynamics and position. My specific research questions are the following:

Research Question 1*:* Are soil conditions significantly different in burned areas compared to unburned areas?

Research Question 2**:** Do burn and erosion severity significantly relate to vegetation and soil penetrability?

Research Question 3: Has fire contributed to boulder spalling?

Research Question 4: Are geomorphic variables of seedling micro–sites affected by fire, and what variables are associated with seedling sites compared to random micro–sites?

Research Question 5**:** Do geomorphic variables change in relation to distance from the burn/unburned edge?

Research Question 6**:** What influence does topography have on conditions after fire?

Research Question 7: How does quadrat size influence results?

Research objectives

The study will be conducted within the framework of ecologic and geomorphic disturbance response, and the results will contribute to a better understanding of the impact of fire and climate change in mountain environments and assist in management decisions and practices. Fire can affect both geomorphic and ecologic constituents of a system, and responses of various factors to fire may vary spatially and temporally. The complex topography within mountains, including the treeline ecotone, may result in a range of responses to fire. Slight variations in elevation, aspect, and site features may result in different conditions found after fire. This research will evaluate several ecologic and geomorphic factors to characterize the effects of fire within the treeline ecotone and contribute to information on an aspect of the potential effects that climate change may impart on treeline patterns and processes. My overall research purpose is to determine the coupled geomorphic/ecologic effects of fire at alpine treeline, and my specific objectives are as follows:

Objective 1: Compare soil conditions in burned and unburned areas.

Objective 2**:** Assess potential relationships between paired comparisons of erosion severity, burn severity, vegetation, and soil penetrability.

Objective 3: Evaluate the influence that fire has on boulder weathering.

Objective 4**:** Characterize seedling microsite (0.25 x 0.25 m) conditions and compare them between burned and unburned sites and between plots with seedlings and random plots.

Objective 5: Determine if geomorphic variables change in relation to distance from the burn edge.

Objective 6**:** Characterize soil conditions and vegetation of different sites in relation their topographic features.

Objective 7: Assess various variables within increasing plot size.

Significance

My research objectives address both theoretical and applied components. Geomorphic conditions were assessed in relation to ecological theories of ecotone dynamics, including vegetation succession and facilitation factors. Theoretical ecological mechanisms of ecotones are well studied, but rarely incorporate geomorphic variables (*e.g.* Risser 1993; Malanson 1997). The results of this study are evaluated within the theoretical context of feedback loops between geomorphic and ecologic processes, edge effects and ecotone dynamics, and biogeomorphic disturbance response. The applied contributions include improved understanding of climate change effects on treeline and fire management in a National Park. The treeline ecotone is biologically rich and serves several hydrologic functions (Holtmeier 2009), however, the effects of fire on this system have only been scantily studied. The treeline ecotone is an ideal system in which to conduct this study because the harsh conditions heighten the dependencies between geomorphic and ecologic factors and a great need exists for data on the effects of fire within this system.

The interactions between geomorphic and ecologic processes are often challenging to determine because of the different scales at which geomorphic and ecologic processes operate (Phillips 1995; Post et al. 2007; Renschler et al. 2007). I addressed this issue by 1) collecting field data on both geomorphic and ecologic variables after fire, 2) selecting sites that have burned within a timeframe that encompasses both ecologic and geomorphic processes, and 3) synthesizing the overall results from a coupled geomorphic–ecologic approach. I selected sites that experienced fire between 5 and 10 years ago, a timeframe in which geomorphic conditions and processes are still evident and occurring because the fires were relatively recent, but also allowing enough time for seedlings to have begun establishing. Various statistical tests were employed to determine significant correlations or relationships between geomorphic and ecologic variables and modes of analysis included field data, remote sensing analysis, and statistical tests. The focus of this study on interactions among krummholz densities and burn extent, geomorphic factors affected by fire, and subsequent seedling establishment in relation to geomorphology presents a unique perspective on the two–way interactions between ecologic and geomorphic factors. The results are evaluated in terms of directional movement from both ecology to geomorphology and geomorphology to ecology. This duel directional approach has only recently been recognized but is important for understanding geomorphic–ecologic interactions within a system (Marston 2010). This coupled geomorphic-ecologic approach in conjunction with empirical data will contribute new understanding to the processes and influence of fire on treeline dynamics.

The applied significance of this research includes aiding in fire and resource management plans, contributing to Park visitor interpretation efforts, and increasing knowledge on the potential effects of climate change to the alpine treeline ecosystem. The results of this study contribute information on the influence of fire on species composition at treeline and how fires may affect soil conditions within the ecotone. Fires alter species compositions and densities in many ecosystems, however, the response of vegetation communities is little understood at treeline. The alpine treeline ecotone hosts a high diversity of plants because it contains both subalpine and tundra species (Holtmeier 2009). The plants are often at the limits of their growing conditions, and together the subalpine and tundra species often form a mosaic of vegetation patches within that zone of transition between the two ecosystems. The harsh environment present at treeline results in the occurrence of plants that are adapted to their conditions and/or utilize resources that help the plants survive. However, these conditions also contribute to the high sensitivity of systems within the treeline ecotone, and therefore, disturbance may be especially severe or pronounced, and recovery may proceed at a slower rate compared to lower elevations. Fire has resulted in changes to plant densities and compositions in sub-alpine areas of the Colorado Rocky Mountains (Coop and

Schoettle 2009), and may also affect treeline ecotone vegetation compositions. Shifts in community structure at treeline may subsequently influence further disturbance regimes such as insect invasions and avalanche risks (Butler et al. 1990; Veblen et al. 1994). The harsh establishment and growing conditions within the treeline ecotone provide unique environmental circumstances that result in variations in response to fire relative to lower elevations systems. Species re-establishment will influence future disturbance to treeline. Loss of trees and geomorphic destabilization will also allow for a greater risk of snow avalanches and landslides (*e.g.* Beals 1910; Butler et al. 1990). Trees can serve to mitigate snow movement downslope, stopping or slowing avalanches that may begin above treeline. If fire precludes tree growth or affects its establishment, herbaceous vegetation and other low–lying alpine plants may predominate, precluding tree establishment (Malanson and Butler 1994).

The treeline ecotone is also of particular importance because of its hydrological functions. Mountains are the primary source of water for many locations around the world, and contribute 50% of the water used by the world (Messerli and Ives 1997). The headwaters of many mountain streams are located near treeline, and the processes that occur within treeline will therefore subsequently affect stream processes.

Fire is a natural part of many western ecosystems, and fire management is and has been a significant factor in Park management. The return interval of fire at treeline and higher elevations of the sub-alpine zone within the Northern Rocky Mountains ranges from about 130 years to several centuries (Hawkes, 1980; Romme and Knight 1981). The harsh environmental conditions, including low temperatures, high solar radiation, complex surface topography, and dry soils, and the dynamic interplay between alpine tundra vegetation and the subalpine forest provide an environment that is different from lower elevation forests. These factors may influence the effects of fire to geomorphology, soils, and vegetation, and it is, therefore, important to understand the influence of fire on treeline, and not merely assume that results of fire on lower elevation forests are applicable to treeline systems.

Fires were suppressed for decades before the current policy initiatives allowed fires to burn in certain cases. Glacier National Park's current fire plan objectives include allowing fire to return as part of the ecosystem while protecting property and life from fire. Understanding the role of fire in the alpine treeline ecotone system will assist managers in better understanding the function of fire in the high elevation ecosystems within the Park.

Numerous studies have been performed on the influence of disturbances on the alpine treeline within Glacier National Park (Butler 1986; Butler et al. 1992; Butler and Walsh 1994; Walsh et al. 1994; Butler 1995; Butler et al. 2007; Butler et al. 2009). Few studies have been performed on the effects of fire within the treeline ecotone, and therefore this study will provide baseline information on the effects of fire to both vegetation and geomorphic processes, which may subsequently be incorporated into Park fire management plants. Increased knowledge on the effects of fire on treeline vegetation and geomorphology will lend to a better understanding of treeline movement and response to disturbance, and the coupled or confounding effects of fire and climate change.

The treeline ecotone is also of applied interest because of the high visual characteristic of this ecotone within GNP. The ecotone between the subalpine forest and the alpine tundra is one of the most dramatic and iconic ecotones in the world. Factors that influence this treeline ecotone will likely be of interest to Park visitors because of its great visual appeal.

CHAPTER 2: LITERATURE REVIEW

Global Climate Change and Fire

Global climate change, in conjunction with past fire management practices, is projected to result in conditions well suited for fires of greater intensities and longer duration compared to fires of recent history (McKenzie et al. 2004; Running 2006; Westerling et al. 2006; Marlon et al. 2009). Changes in moisture regimes, warmer temperatures, and an earlier melting snowpack will result in drier forest conditions and a longer fire season (Mote et al. 2005; Westerling et al. 2006). Within the United States and Canada, these conditions will be particularly apparent in the mountains of the west, where vast tracts of forest exist. A more comprehensive knowledge of fire and its effects on both geomorphology and ecology is needed, especially in mountain environments (Swanson 1978; Westerling et al. 2006). Mountains contain unique geomorphic and ecologic conditions, and interactions with climate are often different in these high elevations systems compared to low elevation areas of low relief (Barry 1992; Beniston 2003; Fagre et al. 2003). Climate conditions exert strong influence on geomorphic and ecologic features and processes, and climate within mountains can be harsh and highly variable, forming complex relationships with vegetation and geomorphology (*e.g.* Pauli et al. 1996; Körner 1998; Hiemstra et al. 2002; Trivedi et al. 2008). The combination of steep topographic gradients, compressed climate zones, and distinct vegetation distributions found in mountains create an environment in which interactions and dependencies among various factors are accentuated (Körner 2003; Marston 2010).

The interactions between shifts brought on by climate change in conjunction with disturbance regimes may result in numerous outcomes in the interplay between vegetation distributions and the external forcings of disturbance (Malanson and Butler 1984). Mountain vegetation, which is physiologically and morphologically adapted to harsh climate conditions (Körner 2003), is especially sensitive to global climate change and its subsequent influence on disturbance (Malanson and Butler 1984; Barry 1992; Pauli et al. 1996; Bartlein et al. 1997; Thuiller et al. 2005; Trivedi et al. 2008). Compressed climate zones found in mountains result in high biological diversity, and the isolating conditions of high alpine peaks may form biogeographic islands (Fagre 2009). The relatively narrow climate zones however, may be detrimental for species that cannot migrate at a rate fast enough to remain in a climate zone that is advancing upslope. Many mountain plants are also unable to migrate great distance because of limitations placed by reaching the summit of a mountain (Pauli et al. 1996; Bartlein et al. 1997). Alpine tundra may be particularly vulnerable since it may not be able to migrate to higher elevations, as other species move into its elevational zone (Grabherr et al. 1995). Tundra vegetation is adapted to grow in abiotically stressful conditions, but often cannot compete against more productive species, such as woody vegetation. Climatically stunted trees, or krummholz, and upright trees may migrate upslope and/or become denser in their current locations in some regions (Kittel et al. 2000; Theurillat and Guisan 2001; Klasner and Fagre 2002; Danby and Hik 2007). Conversely, however, Malanson and Butler (1994) presented the perspective of tundra vegetation preventing krummholz from migrating upslope, especially in the case of disturbance events. Disturbances, such as avalanches, can shift the competitive abilities of vegetation, and override the reliance on competitions based primarily on resource acquisition. Geomorphic disturbances are especially influential in mountain environments (Butler and Walsh 1990; Butler and Walsh 1994; Walsh et al. 2003; Butler et al. 2007; Butler et al. 2009).

As woody vegetation becomes denser at higher elevations, simultaneously, warming trends are attributed to increasing tree mortality (van Mantgem et al. 2009). Van Mantgem et al. (2009) concluded that a widespread trend of tree mortality of all age classes in the western United States was linked with climate warming. These conditions of increasing woody vegetation growth in higher elevations in addition to increasing tree mortality will lead to greater amounts of fuel available for fires. A compounding factor to this situation is the build–up of fuel that has occurred after decades of suppressed fires (Weaver 1943; Brown and Arno 1990; Agee 1997). For much of the twentieth century, it was a policy of the federal government to extinguish all forest fires. This action resulted in the accumulation of fuel that would have otherwise likely been burned in low-intensity fires.

Climate regimes, in addition to the accumulation of fuel, will further enhance the possibilities of fires. Warmer temperatures and decreased precipitation and snowpack duration are resulting in forests that contain less moisture and are drier for longer periods during the year. Westerling et al. (2006) found the fire season to have increased by 78 days since 1986 compared to the time between 1970 and 1986, based on the number of days fires were actually burning. Years with early snowmelt resulted in an increase in fires five times that in years of late snowmelt. Snow packs are now melting 1 to 4 weeks earlier than in the past, as spring and summer temperatures have increased about 0.9° C (Mote et al. 2005; Stewart et al. 2005). Westerling et al. (2006) attributed these two conditions to the increase of the fire season, the longer duration of fires, and the increasing area burned by fires. They found that fire activity particularly increased in the higher elevations during their period of study. It was uncommon in the past for high elevation forests to burn because of the long duration of the snowpack. However, with the earlier melting of the snowpack, these forests are now vulnerable to fire. This additional region that is now burning is contributing to the overall increase in fires and burned area.

Understanding the effects of fire in high elevation areas is particularly important because hillslopes and colder environments may respond differently to fire compared to other regions. Loss of vegetation on steep slopes may result in an increased risk of debris flows and snow avalanches (Butler et al. 1990). Fire may exert a significant impact on cryogenic processes, such as needle–ice formation and freeze–thaw patterns within colder, high elevation environments (Swanson 1978). Loss of vegetation leads to a decrease in snow accumulation, and less snow results in decreased soil moisture, an important variable for vegetation, especially on the thinner soils in alpine areas.

The alpine treeline ecotone is a prominent feature in many mountain regions, and may be especially variable and sensitive to the effects of fire. Vegetation communities of both the alpine tundra and subalpine forest within this ecotone are often at their physiological limits, and disturbance will likely compound the dynamic balance and ecological response found within this system. A better understanding of treeline processes in needed to determine the relations between treeline position and climate changes and the potential effects that fire may have on this ecotone.

Alpine Treeline Ecotone

Definitions and Factors

Simply stated, the alpine treeline ecotone is the transition zone between the subalpine forest and the alpine tundra (Holtmeier 2009; Malanson et al. 2012). An ecotone in general is the boundary between two adjacent ecosystems or distinct vegetation communities. As such, ecotones contain components of both systems, and are often characterized as areas of high biodiversity. Complex interactions among biotic and abiotic factors often exist within ecotones as the overlapping features of both systems form a mosaic of vegetation types, soil conditions, and habitat features. The alpine treeline ecotone is especially distinct because it consists of an ecosystem of herbaceous vegetation and one of woody plants.

However, although the alpine treeline ecotone may be one of the most visible and studied ecological boundaries in the world, consensus on the definition of "treeline" is lacking (Holtmeier 2009). Some define the location of this ecotone by tree heights, with ranges from a maximum tree height of 2 m up to 8 m (Holtmeier 2009), whereas others define it by the distance between trees or the percent of closed forest, and others use a combination of these factors. Physiognomic features of tree species further complicate the issue. Four general alpine treeline ecotone types are described by Holtmeier (2009) in relation to tree and krummholz establishment and its abruptness with the alpine tundra. Treeline may form a distinct and abrupt boundary line between forest and alpine tundra vegetation, or trees may gradually decrease in height between the forest and alpine vegetation. The ecotone may alternatively be comprised of patches of trees and/or krummholz that gradually diminish in size and extent as tundra vegetation becomes dominant. A fourth type is typified as containing mostly krummholz mats between the subalpine forest and the tundra. This paper will follow the nomenclature used by Holtmeier (2009) in defining treeline as the belt between closed forest and the upper-most establishment location of trees.

Treeline is currently of particular interest because its location has been linked to temperature regimes, and therefore, it may serve as a response indicator for a changing climate. However, inconclusive results have been found regarding treeline movement in response to climate change, and local scale factors need to be considered when addressing treeline position (Lavoie and Payette 1994; Hessl and Baker 1997; Kullman 2001, 2002; Holtmeier and Broll 2005). Factors such as disturbance, species distributions, and land use history need to be considered when analyzing treeline position. Dullinger et al. (2004) used a temporally and spatially explicit model to predict the movement of treeline in the Alps under a warming climate. They emphasize the importance of species–specific dispersal mechanisms in understanding the potential movement of treeline. In areas that have experienced grazing in the past, land use change may be the greatest influence on treeline position movement (Hofgaard 1997). Likewise, fire disturbance may influence treeline position and movement, and local scale factors are important to the mechanisms of response (Stueve et al. 2009; Coop et al. 2010).

Effects of Fire in the Alpine Treeline Ecotone

Tree establishment is slow in the undisturbed alpine treeline environment, and fire places additional stresses on tree regeneration and growth (Coop and Schoettle 2009; Sass et al. 2012b). Fire may impact geomorphic features and processes, which may subsequently affect tree regeneration. The harsh environment of the alpine treeline ecotone, the vegetation dynamics between the alpine tundra and the sub–alpine forest, and high elevation conditions form unique conditions for tree establishment (Holtmeier 2009). Fire in this area may result in different conditions compared to lower elevation environments, and it is therefore important to understand the dynamics and conditions of treelines impacted by fire.

Several studies have been performed on the effects of fire at alpine treeline, particularly in the Colorado Front Range, the Olympic Mountains, and Cascade Range (Bollinger 1973; Agee and Smith 1984; Shankman and Daly 1988; Coop and Schoettle 2009; Stueve et al. 2009; Coop et al. 2010). Most of the studies were focused on tree regeneration and species compositions following fire. In regard to regeneration rates at or near treeline, various results were found. Stueve et al. (2009) determined that tree regeneration following a fire in the Cascade Mountains was quick and prolific, likely a result of favorable weather conditions during the post–fire years. Coop et al. (2010) and Shankman and Daly (1988) indicated that tree establishment and forest regeneration would take much longer near treeline compared to lower elevations, and that it may be centuries before the areas returned to pre–fire conditions. Each study looked at sites that had experienced fire about 30 years ago, but methods of analysis and study locations were distinctly different between Stueve et al. (2009), who primarily used GIS and remote sensing in the Cascade Mountains, and Coop et al. (2010) and Shankman and Daly (1988) who collected much of their data from field analyses in the Colorado Front Range. Agee and Smith (1984) assessed tree establishment within the Olympic Mountains and found that tree regeneration rates were largely attributable to site–specific conditions. They analyzed reestablishment at one site that had experienced a fire three years prior to the study, one 55 years prior to the study, and one 88 years before the study was performed, as well as corresponding unburned sites. Overall, regeneration rates were higher in the burned areas, findings that were similar to other studies (Coop and Schoettle 2009), compared to unburned areas, but overall slow. They found much variability among the sites, and still few trees within the 55 year old burn site.

Tree species is an important factor in regeneration rates. Although Coop et al. (2010), Coop and Schoettle (2009), and Shankman and Daly (1988), who all performed their research in the Colorado Rocky Mountains, found regeneration rates to be slow, Rebertus et al. (1991) also looked at reestablishment rates in the Colorado Rockies and found regeneration rates to be rather rapid. The quick colonization determined by Rebertus et al. (1991) was *Pinus flexilis*, which were not present in Agee and Smith's (1984) sites. Coop and Schoettle (2009) also found *P. flexilis* to be the quickest tree species to establish following fire. This result may be attributed to the physiologic conditions and preferences for *P. flexilis*, particularly their ability to regenerate in full sun and their seed dispersal mechanisms. *P. flexilis* seeds are carried and buried by Clark's nutcrackers, enabling the species to establish without reliance on close proximity to seed sources and wind speeds and direction. Coop and Schoettle (2009) found the *P. flexilis* was located more in the interior of burn sites and further from burn edges than *Pinus* *aristata*, whose seeds are dispersed by wind and are therefore more spatially associated with their seed sources. Other tree species, such as *Pinus contorta* and *Picea engelmannii*, were also found in decreasing densities with increasing distance from the burn edge towards the burn interior (Agee and Smith 1984; Coop et al. 2010). Additionally, fire intensity contributed to variations in species regeneration rates. *P. flexilis* regenerated faster in complete burn sites relative to partial burns (Coop and Schoettle 2009), and *P. aristata* increased significantly only in the partially burned sites.

Although biotic influences were found to be the most important determinant of tree establishment patterns (Coop and Shoettle 2009), numerous abiotic variables influenced tree regeneration patterns following fire at alpine treeline and in sub–alpine forests (Romme and Knight 1981; Shankman and Daly 1988; Donnegan and Rebertus 1999; Coop et al. 2010). Regeneration establishment was often influenced significantly by topographic and landform variables (Veblen 1986; Donnegan and Rebertus 1999). Coop and Schoettle (2009) and Coop et al. (2010) focused on vegetation patterns in regard to topography and landscapes on their study in Colorado 30 years after high intensity fire within sub–alpine forest near treeline. They found elevation to be the strongest influencing abiotic factor, with regeneration rates and densities greater with decreasing elevation. Stueve et al. (2009) also found elevation to be the most influencing abiotic factor. Slope angle and slope aspect were also determined to be important factors to tree regeneration densities and rates (Shankman and Daly 1988; Stueve et al. 2009). These variables primarily related to moisture and temperature conditions associated with various sites.

Facilitative features were deemed important to tree regeneration by several studies (Bollinger 1973; Coop and Schoettle 2009; Coop et al. 2010). Protective features can be either abiotic or biotic structures (nurse objects), and may include boulders, cobbles, microtopographic structures, logs, and existing vegetation. Coop and Schoettle (2009) found a positive correlation between tree establishment and distance to objects. The nurse objects may provide protection from wind, predation, and solifluction, and improve the micro-conditions for seedlings. Objects may capture precipitation, which would lead to increased soil moisture, and this action may be particularly important when overland flow is increased following fire. Also, objects may cause increased snow accumulation, which may be reduced within the overall area because of a decrease in vegetation cover. Snow capture would also result in increased soil moisture. Bollinger (1973) attributes the inability of treeline advancement after fire to a lack of protective vegetation, causing a decrease in micro–climate conditions for seedling establishment and growth.

In summary, with regard to tree regeneration following fire at or near alpine treeline, elevation, species composition, and seed source comprise the most significant determinants of tree reestablishment according to the literature. Coop and Schoettle (2009) concluded that fire can depress treeline below its hypothetical climatic limit, but that it will return to its pre-fire position. However, regeneration will be slow and can take decades or centuries (Agee and Smith 1984; Coop and Schoettle 2009). Bollinger (1973) hypothesized, however, that fire causes a long-term lowering of the treeline. He found that the loss of vegetation and its facilitative mechanisms resulted in conditions too harsh for trees to re-establish. Fire management practices may also result in confounding treeline response and movement. Some treeline in Glacier National Park moved upslope since 1935, but fire suppression policies may have contributed to this advance rather than its response to climate change (Fagre et al. 2003). Understanding the influence of fire on treeline processes is needed for both projecting the effects of fire on the future of treeline as well as understanding its current position and past movement dynamics.

Tree re–establishment has been the primary focus of the effects of fire on treeline, but geomorphic factors may also be affected by fire and can significantly influence tree establishment (Resler et al. 2005). However, no research was found that assessed the direct geomorphic impacts on tree succession at alpine treeline following fire events. The static landscape features of elevation, slope angle, and aspect were common environmental variables assessed (Agee and Smith 1984; Shankman and Daly 1988; Coop and Schoettle 2009; Stueve et al. 2009; Coop et al. 2010), however many studies have determined that fire can greatly influence geomorphic processes (*e.g*. Zimmerman et al. 1994; Moody and Martin 2001a, b; Dorn 2003; Shakesby and Doerr 2006; Woods et al. 2006; Cannon et al. 2010; Stine 2013). Numerous studies document the importance of geomorphic processes on tree succession following fire at lower elevation locations, as well as the significant impact that site conditions have on geomorphic processes and tree growth. Conditions at treeline, including high winds, shallow soils, higher density of low growing vegetation, and climate limits, provide a different environment compared to lower elevation forests.

Geomorphic Effects of Fire

Introduction

Fire exerts a significant influence on geomorphic processes, at numerous temporal and spatial scales and in varying intensities. The effects of fire, directly from heat and indirectly through ecological changes, can result in geomorphic disturbances, including increased erosion and overland flow rates and changes to soil conditions (*e.g.* Christensen et al. 1989; Brown 1990; Moody and Martin 2001b; Shakesby and Doerr 2006; Cannon et al. 2010). The interactions among fire, geomorphology, and organisms are extensive, and the effects that fire has on organisms can lead to significant changes to geomorphic features and processes. Likewise, the geomorphic processes affected by fire can influence post–fire plant re–establishment and growth. Similar to the varying effects of fire on vegetation, the influence of fire on geomorphology varies widely as a result of differences in fire severity and size, and the location of the fire event. Fire can contribute significantly to landscape denudation (Jackson and Roering 2009; Sass et al. 2010), and is one of the leading causes of erosion in some areas (Shakesby and Doerr 2006).

Fire severity and frequency are significant factors in the resulting extent of geomorphic disturbance. Wildfire severity may be categorized as ground (low), surface (moderate), and crown (high, very high, or extreme), a classification that is generally based on the amount of vegetation consumed and the behavior of the fire (Byram 1959; Neary et al. 1999; Shakeby et al. 2003; Shakesby and Doerr 2006). Ground or low intensity fires burn ground cover and small (<2 m) shrubs. Surface or moderate fires consume ground fuel, shrubs, and possibly small woody vegetation. Crown or high intensity fires burn all ground fuel, shrubs, and at least part of tree crowns (Fig. 2.1).

Extreme intensities burn much of the tree crowns. Fire frequencies vary from every 2–5 years to more than several thousand years, depending on location, environmental factors, and human influences (Baisan and Swetnam 1990; Grissino-Mayer 1995; Wheelan 1995; Fulé et al. 1997; Shakesby and Doerr 2006). In most fire prone areas, low intensity fires were historically a common ecosystem function. These fires prevented a build-up of fuel, and in the process, hampered the occurrence of high intensity fires.

Currently, however, wildfires are increasing in intensity and frequency, and are projected to continue to become larger in size in the future (Moreno et al. 1998; Pausas and Vallejo 1999; Fagre et al. 2003; Westerling et al. 2006; Marlon et al. 2009). The occurrence of high severity fires is often attributed to management decisions and climate change (Lenihan et al. 1998; Price and Rind 1998; McKenzie et al. 2004; Pausas 2004; Running 2006; van Mantgem et al. 2009; Marlon et al. 2009). Marlon et al. (2009) and Westerling et al. (2006) concluded that fire events are increasing because of changes in moisture, temperatures, and storm tract regimes associated with shifts in climate. In the western United States, snow is important to the hydrologic cycle and provides moisture in otherwise often arid regions (Bales et al. 2006). However, decreases in snowfall and earlier melting times associated with climate fluctuations may result in drier conditions and longer fire seasons between snow cover. In addition to climate changes, an increase in fuel is also responsible for greater fire intensities (Schoennagel et al. 2004). Fire is a

Figure 2.1. A crown fire immediately south of Glacier National Park, MT, USA in 2007. Photograph by David R. Butler.

natural disturbance in many ecosystems, however, for decades it was practice and a policy of the United States to suppress wildfires (Busenberg 2004). This action led to a build-up of fuel that would have otherwise been consumed in more numerous but lower intensity fire events.

The extensive influence that fire has on geomorphology; the numerous interactions among fire, geomorphology, and organisms; and the increase in high intensity fires emphasize the importance of understanding the effects of fire on geomorphology. The interactions among fire, vegetation, and geomorphology are many, and geomorphic disturbances as a result of fire may be found around the world in locations where fires occur. Fire affects soil conditions, erosion rates, and hydrologic processes, both directly from intense heat and indirectly from the burning of vegetation. Location and intensity of fire significantly influence the resulting extent of geomorphic disturbance. Much of the literature on fire and geomorphology focuses on fire-prone areas, such as the western United States, Mediterranean Spain, and *Eucalyptus* forests of Australia (*e.g.* Christensen et al. 1989; Doerr et al. 1994; DeBano 2000; Cerdà and Doerr 2005; Shakesby and Doerr 2006).

Soil

Many of the effects that fire has on geomorphology relate to the changes that fire produces in soil. Fire directly influences the soil by subjecting it to high temperatures. After a fire event, distinct temperature gradients are often found in soils. Temperatures on the ground surface can reach 500-700 °C, and possibly even 850° C (DeBano 2000). Temperatures up to 150 °C may be found 5 cm below the surface of mineral soil, but rarely does heat penetrate to 20-30 cm below the surface. Fire duration is often the strongest overall factor in heating depth. Fire-related heat can remain in the soil from a few minutes up to several days following a fire. Fire also influences soil indirectly by damaging vegetation, which can result in a decrease in soil stabilization, nutrient alteration, and loss of an insulating layer. Soil hydrophobicity (water-repellency), porosity, nutrients, infiltration, organic matter, microbial activity, erosion, pH, moisture, leaching, particle size, and color may be affected by fire, and the extent of soil damage is often related to the duration of the fire, temperature of the fire, and location characteristics (Certini 2005). Changes to soil may be short-term, long-term, or permanent (Certini 2005).

Hydrophobicity

Intense heat from fire can cause soil to become hydrophobic (water-repellant), which has numerous ramifications to geomorphic processes (*e.g.* Munns 1920; Rice et al. 1969; Jackson and Roering 2009). Hydrophobic soils may lead to increased erosion and overland flow, enhanced streamflow, uneven wetting patterns, and decreased infiltration. The extent, severity, and spatial pattern of the hydrophobic soils determine the effects that it has on geomorphology (Woods et al. 2006). The degree of hydrophobicity varies greatly, with some soils being hydrophobic for years, whereas others are unaffected by fire (Leitch et al. 1983; Doerr et al. 2004; Sheridan et al. 2007). Water-repellent soils are formed from heat concentrating hydrophobic compounds near the soils' surface (Imeson et al. 1992; Woods et al. 2006), and heat may also bind these compounds to soil particles, further enhancing soils' water repellency. Some soils are naturally hydrophobic, and the hydrophobic compounds may be unaffected by fire. However, the hydrophobicity of the soil may be destroyed if the temperature reaches the threshold in which hydrophobic compounds are volatized. Other soils contain hydrophobic compounds but do not display any water-repellent characteristics until triggered by fire (Doerr et al. 2000; Varela et al. 2005). Sheridan et al. (2007) also found that unburned soil hydrophobicity fluctuated throughout the year, with high water-repellency in the summer and non-repellency in the winter, in their study on *Eucalyptus* forests in Australia. Marston and Haire (1990) found water-repellent soils in both burned and unburned locations following the 1988

Yellowstone fires, but the hydrophobic soils extended to a greater depth in the burned sites. Various organisms and decaying plant matter, including bacteria, fungus, algae, and trees, particularly evergreen trees which often contain waxes and resins, have been associated with hydrophobicity (Doerr et al. 2000). Hydrophobicity is most often associated with coarser-grained soils, such as sands; however, soils with high clay content may have high water repellency as well (Crockford et al. 1991).

Fire intensity and the amount of time after a fire affect the degree of hydrophobicity (Huffman et al. 2001; Shakesby et al. 2003). Moderate intensity fires tend to produce more water-repellant soils than low or high severity fires (Lewis et al. 2006; Woods et al. 2006; Glenn and Finley 2010). Laboratory studies found that hydrophobicity was enhanced between 175 °C and 270 °C but obliterated above 270-400 °C (Doerr et al. 2004). In agreement with the laboratory studies on heat and hydrophobicity, Woods et al. (2006) found that hydrophobic patches were greatest in severity and spatial extent with intermediate intensity fire. Lewis et al. (2006) also found a greater extent of hydrophobic soils after moderate intensity fires, however, they determined that high severity fires resulted in hydrophobic soils at greater depths. Low intensity fires resulted in the least amount of hydrophobicity because they did not generate enough heat to form large areas of hydrophobic soils. Hydrophobic compounds often occur in undisturbed soils, and hydrophobicity will return to background levels usually within months to several years after the fire event, once the soil reaches a certain moisture threshold (Doerr et al. 2003; MacDonald and Huffman 2004).

The duration of hydrophobicity is highly variable and has been found to exist from a few seconds to eight years (DeBano et al. 1976; Dyrness 1976; King 1981; Doerr and Thomas 2000). However, Shakesby and Doerr (2006) noted that little is known concerning the length of time water-repellency exists because of a lack of long-term studies on post-fire conditions, and comparisons between studies are difficult because of differing methods and environmental conditions among sites and studies. The degree of hydrophobicity also varies over time and generally decreases in repellency with increasing time after fire. Huffman et al. (2001) found evidence of repellent soil 19 months after fire, though they had recorded a significant decrease in hydrophobicity 3 months after the fire. Hydrophobic soils are often found in very patchy mosaics and vary extensively in spatial pattern and site location (Cannon and Reneau 2000; Woods et al. 2006).

Soil hydrophobicity influences several geomorphic and hydrologic processes, most of which are discussed in greater detail in following sections of this chapter. When soils are hydrophobic, overland flow and infiltration are reduced (Burch et al. 1989; Letey 2001). Water flow will be increased in the hydrophobic patches, but generally will infiltrate into the soil when it reaches hydrophilic soils or burrows (Shakesby et al. 2000; Woods et al. 2006). Therefore the spatial extent and position of the hydrophobic patches will exert a great influence on the amount of runoff that will occur as a result of soil hydrophobicity. The patchy nature of hydrophobic soils results in differential soil moisture patterns. Hydrophobicity also varies in depth. Spigel and Robichaud (2007) found hydrophobic soils from 9 to 13 mm within their study sites on hillslopes in Montana. Johansen et al. (2001) reported hydrophobicity in a semiarid forest in New Mexico 1-2 cm from the ground surface. Hydrophobicity, in summary, results in numerous geomorphic effects, however, the spatial pattern, presence, and degree of hydrophobicity varies extensively within burned areas.

Infiltration

Decreased infiltration rates may be caused by water-repellent soils, decreased interception from plants, loss of litter, increased bulk density, and a cover of ash as a result of fire (Cerdà 1998a; Robichaud 2000; Martin and Moody 2001). Infiltration after a fire is usually lower than pre-fire rates, which causes an increase in erosion and overland flow (Cerdà 1998a; Robichaud 2000). Various factors brought on by fire affect infiltration rates, including bulk density, frozen ground, loss of vegetation, and decreased animal activity. Bulk density may increase following a fire as a result of changes to soil aggregates and the action of voids being filled with ash and clay (Certini 2005). These changes to fine-scale soil properties result in a decrease in porosity and permeability, and therefore a decrease in infiltration. Infiltration may also be impeded by frozen ground, which could form as a result of a loss of an insulating vegetation layer (Campbell et al. 1977). A decrease of sub-surface faunal activity in the burned area may also decrease infiltration rates (Dragovich and Morris 2002). Intense heat and changes to soil properties may deter animal activity in the soil, decreasing their bioturbating effects, such as burrowing, actions that often increase the porosity and permeability of the soil.

Infiltration rates are highly dependent upon location and environmental factors, as well as time since the fire event. Cerdà (1998a) studied the infiltration rates over the course of several years within a Mediterranean scrubland following fire. He used a rainfall simulator and measured infiltration rates beginning six months after a rangeland fire and continued to record measurements up to 5.5 years following the fire. The lowest infiltration rates were found soon after the fire and increased throughout his study. The timing of runoff was 3 minutes, 47 seconds (StdDev = 1.46 minutes) within the year following the fire, and increased throughout the following 5 years to an average time of 25 minutes, 30 seconds (StdDev = 7.29 minutes). Overland flow dropped from 45% to <6% over the course of the 5.5 years after the fire. Ash provided a continuous ground cover of 3 to 5 cm following the fire. The increase in infiltration rates corresponded with vegetation growth. Other indirect effects of fire may influence infiltration as it relates to vegetation growth. Erosion and other changes to soil may impede vegetation regrowth, which may subsequently decrease infiltration rates.

Nutrients

The effects of fire on nutrients may indirectly influence geomorphic process because of the importance of nutrients to plant establishment. Vital nutrients for vegetation growth, such as nitrogen and phosphorus, may be altered in fires (DeBano and Conrad 1978; Cook 1994; Knicker 2007; Turner et al. 2007). A loss, gain, or redistribution of nutrients may influence vegetation succession patterns, which subsequently affects several geomorphic processes, such as erosion and runoff rates. Nitrogen receives particular attention because it is especially important for vegetation establishment and growth, is often a limiting nutrient for vegetation, and may be used as an indicator for ecosystem function. Fire, however, may burn nitrogen sources, cause mineralization of the nitrogen that does remain, and increase ammonium concentrations. Turner et al. (2007), in their study on fire in the Greater Yellowstone Ecosystem, found that ammonium was higher following fire, however the concentration decreased with time, whereas nitrate increased in the first several years following fire. With regard to fire severity, ammonium resulted in no significant differences resulting from crown and severe surface fires, although nitrate was significantly ($P < 0.05$) higher after crown fires. Net nitrogen mineralization (ammonification and nitrification) varied with time after the fire, with a negative net rate one year following fire, but with highest amounts found two to three years after stand-replacing fire, and decreasing in subsequent years. Their results also revealed drastic changes to nitrogen cycling following fire. Vegetation succession is recognized to strongly influence nitrogen amounts. Herbaceous vegetation in the Northern Rocky Mountains generally increased dramatically in the one to four years following fire. These plants used mineralized nitrogen and added nitrogen-rich litter to the soil. About ten years after the fire, trees, which do not add nitrogen to the soil as herbaceous vegetation does, begin overtaking herbaceous activity, resulting in a decrease in nitrogen input compared to the several years after fire when herbaceous vegetation dominated (Chapman et al. 2006; Turner et al. 2007). In a chaparral ecosystem, DeBano and Conrad (1978) found less nitrogen on the ground surface after a prescribed burn compared to pre-fire amounts. About 110 kg/ha of nitrogen was lost from plants and litter, 21.7 kg/ha from 0-1 cm below the soil surface, and 14.7 from 1-2 cm below the surface during the fire. They attributed this result to the likelihood that most of the nitrogen, which was located in vegetation before the fire, was volatized in the high temperatures generated by the fire. In a different system, Lewis (1974) found an increase in soluble nitrate after fire in a South Carolina pine forest, but no notable difference in biologically available nitrogen. He attributed the increase in soluble nitrate to microbial activity, and postulated that some nitrogen may have been lost through volatization.

Phosphorus is not volatized by fire as is nitrogen (Wells 1971; Debano and Conrad 1978). DeBano and Conrad (1978) found low pre-fire phosphorus amounts in their chaparral study, and similar amounts following fire. However, before fire, phosphorus was located primarily in vegetation, and after the fire significant amounts were found in the ash layer on the ground surface. Lewis (1974) had similar results, with an increase in soluble phosphorus after fire but no significant change to biological phosphorus. Several studies reported greater phosphorus concentrations in runoff and streamflow following fire, likely from increased runoff and available phosphorus (Tiedemann et al. 1978; Schindler et al. 1980; Leitch et al. 1983; Belillas and Roda 1993).

Organic Matter and Litter

Vegetation litter and organic matter (OM), which are factors in the amount of nitrogen present after a burn and influence erosion rates and soil temperature and moisture, may be reduced or volatized during a fire (Debano and Conrad 1978; Marston and Haire 1990; Turner et al. 2007). A loss of surface litter may result in increased soil temperature, greater erosion rates, and decreased soil moisture because of the loss of an insulating layer. In the Greater Yellowstone fires that occurred in 1988, Marston and Haire (1990) found the amount of litter cover to be the greatest factor in lessening sediment loss in burned sites. They found that the amount of ground surface litter dropped from 72.5% in the unburned sites to17.1% in burned areas. Similarly, percent

OM decreased from 14.8% in unburned, unlogged areas to 5.3% in burned, unlogged sites (Marston and Haire 1990). DeBano and Conrad (1978) found an average of 46% loss of OM on the ground surface after a fire compared to pre-fire levels in the same sites with a range of 0 to 70% in their 6 plots. Their study was performed before and after a prescribed burn in a chaparral ecosystem. The extent of OM loss depends heavily on the intensity of the fire and has numerous ramifications regarding vegetation succession and soil erosion. A study in a *Pinus pinaster* forest determined that organic carbon was unaffected by temperatures of 150 °C, but completely oxidized at 490 °C (Fernandez et al. 1999). At 220 °C, 37% of the OM was lost. Combustion of OM in higher severity fires may result in a loss of nitrogen storage but increases nitrogen mineralization, and therefore availability (Turner et al. 2007). Nutrients previously stored within vegetation matter and roots may be released during and after fire as litter is burned and roots decay, resulting in an initial spike in nutrients several years following fire. Though OM may be reduced following fire, it actually may be increased beyond pre-fire levels in the long-term. This result may be attributed to the type of plants that re-establish or a decrease in mineralization following fire (Fernández et al. 1999; Johnson and Curtis 2001).

Microbial and Faunal Activity

A loss of microbial activity in the top layer of soil influences nutrient cycling and geomorphic processes (Klopatek 1987; Shakesby and Doerr 2006). Microbial, fungal, and faunal activity directly affect water infiltration and erodibility through their burrowing activities and secretion of soil cohesion compounds, and indirectly by their influence on vegetation succession. Microbes within the top few centimeters of soil are often destroyed in a fire, but some may return in greater densities than pre-fire populations. Faunal organisms may also move deeper into the soil during a fire event and return near the ground surface after the fire. These organisms may increase infiltration and reduce hydrophobicity by burrowing and bioturbating the soil (Dragovich and Morris 2002; Shakesby and Doerr 2006). However, micro-organisms may vacate an area if the fire has destroyed their food source or habitat.

Soil Temperature and Moisture

Direct heat from fire and a loss of vegetation, ground cover, and soil litter leads to an increase in soil temperature and a decrease in soil moisture (Tiedemann et al. 1978; Loáiciga et al. 2001; Shakesby and Doerr 2006). High temperatures generated during fire can change the structure of soil and decrease soil stability and increase erodibility (DeBano et al. 1998; Neary et al. 1999). Odion and Davis (2000) found spatial variability in re-growth because of variations in soil heating. Patches that burned more intensely from falling canopy fuel resulted in primarily bare ground, and areas that had been spared from the hotter soil heating contained an abundance of re-growth, largely a result of less damage to the seedbank in the soil. Increases in soil temperature over longer time scales (*e.g*. 1-2 years) following a fire may result from a loss in vegetation cover. The resulting warmer soils can affect vegetation establishment, decrease soil moisture, and increase thawing depth in areas that contain permafrost or seasonally frozen ground. In high mountain areas, snow melt may be an important source of soil moisture, however, snow accumulation may be decreased with a loss of vegetation and ground cover, and evaporation rates may be higher without litter and vegetation (e.g. Hiemstra et al. 2002).

Other soil characteristics, such as color and pH, may also be altered. Color change following fire is often a result of the increased temperatures to which the soil was exposed. Low and moderate severity fires often lead to a covering of black or gray ash. High severity fires may result in a redder colored soil if the organic matter is highly combusted and iron oxide forms (Ulery and Graham 1993; Certini 2005). Ground color may affect albedo, which can result in changes to the soil temperature and therefore, possibly soil moisture and plant growth. The burning of vegetation and organic matter may increase pH (Ulery et al. 1993; Pyne 2001; Arocena and Opio 2003). Changes to pH levels may enhance vegetation re-growth (Baath and Arnebrant 1994; Chambers and Attiwill 1994), however, soil type, vegetation, and heat intensity will influence pH values, and some soil pH will not be affected by fire.

Weathering

Relative to the extensive literature on some of the other aspects of fire, less attention has been devoted to fire and weathering (Dorn 2003; Shakesby and Doerr 2006). Several studies have assessed boulder and bedrock weathering rates and found that fire exerts notable influence on the breakdown and erosion of rock, in both field studies and lab experiments (Zimmerman et al. 1994; Allison and Bristow 1999; Dorn 2003). Rock weathering rates vary greatly depending on fire intensities, rock composition, air temperature, and moisture content within the rock (Goudie et al. 1992; Allison and Bristow 1999; Dorn 2003).

Spalling, the breaking of lensoid-shape rock fragments off of boulders, is the most common rock weathering processes that occurs during a fire (Humphreys et al. 2003; Shakesby and Doerr 2006) (Fig. 2.2). Dorn (2003) collected quantitative field data on boulder weathering after a fire in Arizona. He found that spalling often occurred in a distinctly bimodal pattern, in which the spalling depth was usually either zero/very low or caused erosion over 76 cm in depth.

Figure 2.2. Rock spalling on a boulder after a fire. Photograph taken in 2010.

Different rock types also resulted in different erosion amounts. About 60% of sandstone boulders had a spalling depth of 0 cm, whereas 20% contained a spalling depth that was greater than 76 cm in depth. The remaining 20% experienced a spalling depth

>0, but <76 cm. Diorite however, displayed about 60% spalling >76 cm and about 30% with no erosion. Similar to Dorn's findings, other studies have documented the importance of rock type to the amount of weathering (Adamson et al. 1983; Ballis and Bosc 1994). Not only does rock type influence the amount of weathering, but also the pattern of rock breakdown. Dorn (2003) noted that diorite boulders tended to break off in blocks, and stated that this feature may be because of laminar calcrete found in the rock fractures, forming the breaking points for the boulders during intense heating during a fire.

Season and climate also influence rock weathering. Dorn (2003) determined that sandstone eroded more in the winter than the summer, which may be a result of fire-induced fractures that captured moisture and subsequently froze during the winter. Though fire has been attributed to significant rock weathering in a range of environments, some environments are more conducive to rock weathering processes than others. Dragovich (1993) attributed fire as a significant agent in rock weathering in semi-arid regions where precipitation is not an important factor in weathering. In warm climates, such as those in the southwestern United States, fire may be an important agent of boulder weathering, analogous to cryogenic weathering processes in colder regions. Van der Beek et al. (2001), however, noted that fire contributed to rock erosion in humid temperate regions as well. Even in areas where frost is a significant weathering agent, fire may exert an important influence on rock weathering rates. Ballais and Bosc (1994) found that fire may result in 10-100 times more erosion than frost in mountain environments of southern France. Both bedrock fracture and boulder weathering lead to increased overall erosion and additional sediment and rock particles to the ground surface

(Dorn 2003). The break-down of boulders and bedrock results in particles that are more easily transported by erosional processes (Abrahams et al. 1988; Allison and Higgitt 1988).

Figure 2.3. Rock fragments being transported downslope. Photograph taken in 2010.

Erosion

Erosion is one of the most common and obvious geomorphic effects after a fire (Fig. 2.3), and is extensively addressed in the literature (*e.g.* Morris and Moses 1987;

Brown 1990; Kutiel and Inbar 1993; Cerdà 1998b; Shakesby et al. 2000; Moody and Martin 2001b; Cerdà and Lasanta 2005; Sheridan et al. 2007; Spigel and Robichaud 2007; Smith and Dragovich 2008; Cannon et al. 2010). Cerdà (1998b) noted that fire is one of the primary factors of erosion in mountains of the Mediterranean, comparable to erosion rates caused by animal grazing. Loss of vegetation, ground cover, and litter from fire results in increased overland flow and rain splash (Spigel and Robichaud 2007). In most cases, hillslopes experience increased erosion rates after a fire compared to unburned areas, and the amount of sediment erosion usually corresponds to fire intensity, climate, and environmental factors (Inbar et al. 1998; Moody and Martin 2001b; Spigel and Robichaud 2007; Smith and Dragovich 2008). Erosion rates vary extensively, however, and depend on numerous variables, including soil type, slope, and post-fire rain events, and rates generally decrease as vegetation cover begins growing and acts to stabilize soil. Rates ranging from two fold to 2240 fold have been recorded at the plot scale (Ronan 1986; Sheridan et al. 2007).

Surface Erosion

Numerous studies have found a significant increase in surface erosion rates following fire in burned areas compared to unburned locations (*e.g.* Morris and Moses 1987; Inbar et al. 1998; Mayor et al. 2007; Smith and Dragovich 2008) (Fig.2.4). Surface erosion may result from a decreased infiltration capacity of the soil, saturation of the soil to the ground surface, or a decrease in surface roughness (Lavee et al.1995; Shakesby and Doerr 2006). Smith and Dragovich (2008) determined a significant difference between unburned slopes and burned slopes in regard to erosion on their study of hillslope erosion following a moderate severity fire in a sub-alpine environment in the Snowy Mountains of Australia. Unburned areas experienced a net gain of 2.6 mm of soil and a 4.9 mm

Figure 2.4. Soil erosion on a slope that experienced a fire in 2006. Photograph taken in 2010.

change in surface level, whereas the burned areas witnessed a net loss of 3.8 mm and a 6.7 mm surface level change. Morris and Moses (1987) also found significantly increased erosion rates in burned versus in unburned sites in their study in the Colorado Front Range. Similarly, Meyer et al. (2007) found erosion rates of 4563 kg ha^{-1} for burned areas compared to 0.12 kg ha^{-1} in unburned areas in the Mediterranean area of Alicante, Spain. Lavee et al. (1995) determined significant variations in erosion rates in their study on Mt. Carmel, Israel. They concluded that surface roughness accounted for the spatial mosaic in erosion differences that was found in their study location. Areas that had experienced high intensity fires, where only an ash layer covered the ground, resulted in higher erosion rates. Lower intensity burn areas contained greater surface roughness from vegetation, including partially burned plants and an uneven ground surface from fallen trees. Cerdà and Lasanta (2005) found that erosion rates were not significantly different in one burned study plot compared to the control plot, but their other burned plot experienced about 10 times more erosion than the control during the second years after the fire. Average rates for the 8-year study were 89.11 kg ha^{-1} $year^{-1}$ for the control plot, 143.18 kg ha^{-1} $year^{-1}$ for one of the burned plots and 566.44 kg ha^{-1} $year^{-1}$ for the other burned plot. The two burned plots were chosen to be replicates of each other on their study on post-fire erosion and runoff in an abandoned field in the Central Spanish Pyrenees.

The rate of erosion generally decreases with increasing time since the fire event (e.g. Diaz-Fierros et al. 1987; Morris and Moses 1987; Marques and Mora 1992; Inbar et al. 1998; Moody and Martin 2001b; Cerdà and Lasanta 2005: Smith and Dragovich 2008), with the greatest rates usually occurring within a year after fire (*e.g.* Helvey 1980; Inbar et al. 1998). However different environments, even different sites within a study in one general location (Cerdà and Lasanta 2005), exhibit varying relaxation times (Meyer et al. 2007). Smith and Dragovich (2008) found that the highest amount of erosion occurred within the first 59 days following fire, but continued throughout the study that spanned several years. Morris and Moses (1987) found that rates declined rapidly following fire but were still significant four years later. The decrease in erosion over time may be attributed to a loss of water-repellant soils, an increase in vegetation re-growth, and detachment-limited soil (Morris and Moses 1987; Inbar et al. 1998). Sheridan et al. (2007) attributed decreased soil erodibility found two years after fire primarily to soil conditions, such as ash cover and a lack of dry, loose soil, rather than the re-growth of vegetation cover. Inbar et al. (1998) found that erosion rates were highest in the first year following the fire, lowest in the second year, and then increased again above second year rates during the third year as a result of heavy rainfall. Moody and Martin (2001) witnessed increased erosion rates for the first two years following a fire, but then a return to erosion rates of those witnessed before fire. Sheridan et al. (2007), in their study on post-fire erosion in a *Eucalyptus* forest in Australia, found that highest erosion rates occurred during the first winter following the fire. However, Meyer et al. (2007) found erosion rates to increase during the third year after the burn and then decrease with time.

As the previous examples indicated, erosions rates and timing of erosion following fire vary greatly. These differences may be largely attributed to the various environmental conditions associated with individual site locations (Inbar et al. 1998; Bracken and Kirkby 2005; Smith and Dragovich 2008). Smith and Dragovich (2008) found that erosion rates decreased with increasing elevation in the sub-alpine environment of Australia. They attributed this result to longer snow cover at mid- to upper elevations and more organic matter, such as fibrous roots, in the soils at the higher elevations. Inbar et al. (1998) determined that rainfall intensity exerted the greatest influence on erosion amounts in their study on the Mediterranean forest hillslopes of Mt. Carmel, Israel. Most of the sediment yield from their plot occurred on days of high intensity rainfall. Vegetation cover, fire intensity, and slope exposure followed as factors influencing sediment erosion in their study. Slope aspect has been found to be a strong variable of erosion rates in several studies. Moody and Martin (2001b) found more interrill sediment loss on north-facing slopes compared to south-facing slopes within the first year after fire in the Colorado Front Range. They estimated an erosion rate of at least 0.048 kg m^{-1} d^{-1} from north-facing slopes and 0.0070 kg m^{-1} d^{-1} from south-facing slopes during the summer following the fire. However, south-facing slopes had a greater flux of sediment during the winter (~0.9 kg m^{-1} d^{-1}) as a result of freeze-thaw patterns compared to north-facing slopes (~0.09 kg m^{-1} d^{-1}). Cerdà et al. (1995) however, found greater erosion rates on south-facing slopes on their study on a Mediterranean scrubland in La Costera, Valencia, southeast Spain. They compared erosion rates of north-facing and south-facing slopes 10 years after a fire. They attributed this result to slower vegetation re-growth and greater hydrologic stress on the southern slope.

Vegetation type, re-growth, densities, and the spatial variability of plants can have significant effects on erosion rates (Fig. 2.5), and these variables are well acknowledged in the literature for numerous ecosystems and over many time scales (*e.g.* Swanson 1978; Brown 1990; Moody and Martin 2001a, b; Shakesby et al. 2003; Wondzell and King 2003; Cerdà and Doerr 2005; Roering and Gerber 2005). Johansen et al. (2001) and DeBano et al. (1998) suggested that ground cover is a greater factor in erosion rates than soil conditions. Vegetation and erosion share a negative, exponential association (Thornes 1990; Cerdà 1998b). As vegetation increasingly re-establishes to pre-fire densities or greater, erosion rates correspondingly decrease to pre-fire levels. However, numerous factors influence the time between the fire event and subsequent vegetation re-growth and erosion rates. Fire intensity often directly corresponds to the amount of available fuelwood provided by the vegetation, and subsequent geomorphic impacts often relate to the intensity of the fire (Swanson 1978). In low intensity fires, herbaceous vegetation and shrubs may be incinerated, however, many of the trees and larger vegetation will likely still be living. These plants would provide soil stabilization, and

Figure 2.5. Herbaceous vegetation regrowth 4 years after a fire. Areas that contained vegetation retained several centimeters more soil compared to areas that lacked vegetation. Photograph taken in 2010.

only low amounts of erosion may take place in response to a loss of the ground cover. Recovery time would also likely be minimal. Moderate to high severity fires, however, can result in much loss of vegetation and root biomass. This result would potentially lead to greater amounts of surface erosion, changes to soil, debris flows, and overland flow. However, many compounding factors may be involved in erosion rates (Swanson 1978), such as fallen trees acting as sediment traps.

Not only does the presence or absence of vegetation affect erosion, but also vegetation composition and plant type exert a significant influence on the rate of erosion and changes to erodibility over time (Brown 1990; Cerdà and Doerr 2005). Brown (1990) determined that even in areas of the same vegetation type (Mediterranean type in the case of his study), different plant species lead to various erosion rates. He concluded that the life history, phenology, structure, and litter of a plant would influence the spatial pattern of surface erosion. Cerdà and Doerr (2005) studied erosion rates and corresponding plant cover and type over a span of 11 years following a fire in the eastern Mediterranean. They found that erosion rates in plots dominated by herbs decreased to below negligible (1.1 g m^2 h^{-1}) 2 years after the fire, and similarly, shrub plots fell below negligible erosion rates 2.5 years following the fire. Dwarf-shrub and tree plots, however, only reached these levels, or close to them, by 6 and 11 years, respectively, after the fire. They found, however, complex compounding factors associated with the results between the tree plots and the erosion rates. Hydrophobicity was found in increasing amounts up to 8 years after the fire in the tree plots, which would have contributed to greater erosion.

Underground biomass may also have significant implications to erosion rates (Jackson and Roering 2009). Roots within the soil act as a stabilizing agent (Fig. 2.6), however they may be weakened after a fire from death or stress to the plant (Agee 1993).

Fire can directly damage tree roots by scorching root tissue near the ground surface. Tree death and the loss of root systems would contribute to destabilizing the soil and increasing the chance of debris flows (Wondzell and King 2003). Soil destabilization

Figure 2.6. A root apparently acting to stabilize the soil. Photograph taken in 2010.

from loss of tree roots as a result of tree death would likely not be a significant factor immediately after a fire, but rather once the root systems have had time to begin weakening and decaying. Jackson and Roering (2009), however, found a significant loss of root strength only 1 month after a fire. They attribute this result at least partially to damage the trees may have experienced before the fire (*e.g.* logging, insects).

Regional and seasonal differences in rainfall, frequency and patterns of storm events, and freeze-thaw processes exert considerable influence on erosion rates and their temporal variability (Cerdà and Doerr 2005). Fires that occur soon before the rainy season in wet-dry climates, before vegetation becomes seasonally dormant, or preceding high intensity storm events, often lead to increased erosion and overland flow rates than would otherwise occur (Cannon et al. 1998; Cannon and Reneau 2000; Cerdà and Doerr 2005; Vermeire et al. 2005; Gabet and Bookter 2007). Cerdà and Doerr (2005) found average erosion rates of 79.75 g m^{-2} h^{-1} (StdDev. = 42.47 g m^{-2} h^{-1}) within the first year after a fire in the wet season and 30.05 g m^{-2} h^{-1} (StdDev. = 12.96 g m^{-2} h^{-1}) in the dry season. Cannon et al. (1998) assessed the post-fire debris and concentric flows that resulted from torrential rains after fire in an area near Glenwood Springs, CO, USA. The resulting flows covered Interstate 70 with 70,000 m^3 of dry ravel, soil, and other materials from the hillslopes and ravines where an 800 ha fire burned prior to the storm event. Spigel and Robichaud (2007) found that high intensity, short duration thunderstorms resulted in the most erosion on steep Montana hillslopes. Rainfall at a rate of 75 mm h^{-1} for 15 minutes resulted in an erosion rate greater than 20 t ha^{-1}. Low intensity, long duration rain events produced only negligible amounts (<0.01 t ha^{-1}). Although heavy rainfall after a fire can result in an increase in erosion, drought may also lead to long-term increased erosion rates. Mayor et al. (2007) attributed their finding of increasing erosion to drought that impeded vegetation re-growth following fire.

Gully/rill Formation

Erosion can cut channels into hillslopes in the form of rills and gullies, which can transport significant amounts of sediment and create landscape features that persist for years (Wells 1987; Moody and Martin 2001b). These features may become enlarged over time as water flows down through the channels and frost action works to continue to remove sediment. Moody and Martin (2001b) estimated rill erosion to be 40 m^2 for the north-facing slope and 100 m^2 on the south-facing slope during the summer following fire. The average distance between rills was about 10 m and the average cross-sectional area of rills was 0.02 m^2 ($n = 681$). From 2–4 years following the fire, rills widened from freeze-thaw action and heavy rainfall events. Unlike some locations (Schumm 1956), rills remained in place for the four years of the study in the semi-arid mountains of the Colorado Front Range. They hypothesized that over long time spans, rills in semi-arid areas may form into gullies and possibly larger drainage systems, especially on north-facing hillslopes. Cannon and Reneau (2000) found rill formations in moderate-severity burn sites up to 5 cm wide and 4 cm deep and extensive rill erosion in high severity fire sites in their study location of Capulin Canyon, New Mexico. However, erosion response varied widely among their sites with some experiencing little or no rill erosion.

Mass Movements

Fire can result in increased overland flow and sediment movement, decreased infiltration, the formation of water-repellent soils, and loss of vegetation and litter cover – factors that increase the chance for debris flows (Mersereau and Dryness 1972; Wohl and Pearthree 1991; Meyer and Wells 1997; Cannon 2001; Cannon et al. 2001a, 2001b;

Cannon and Gartner 2005; Sass et al. 2012a). Three primary factors that lead to debris flows following fire are a loss of root structures, which leads to sediment destabilization; the accumulation and release of dry ravel deposits; and the accumulation of sediments in overland flow, resulting in a bulked debris flow (Wells 1987; Reneau et al. 1990; Meyer and Wells 1997; Cannon et al. 2001b; Gabet and Bookter 2008; Jackson and Roering 2009). Topography, soil type, lithology, burn mosaic, past debris flows, root matter, and vegetation cover are often, however, more important indicators for debris flow risk than just the event of a fire (Wohl and Pearthree 1991; Spittler 1995; Cannon and Reneau 2000; Larsen et al. 2006). Loose, fine-grained, cohesionless materials and steep slopes are more prone to debris flows than other materials and topography (Spittler 1995). Cannon and Reneau (2000) found that debris flows occurred on the steep-sloped sites of the North Tributary Basin (up to 60% slope), even though the area had experienced less extensive fire than their Capulin Creek site, which had relatively moderate slope angles (about 25% slope). The Capulin Creek basin was also larger in area (39.9 km^2) than the North Tributary basin (6 km^2). This result supported Meyer and Wells' (1997) finding that smaller basins tend to have more debris flows than larger basins, which are more apt to flood (Cannon and Reneau 2000).

Dry ravel (the sliding and bouncing downslope movement of sediment or organic matter) can occur after fire as a result of a loss in stabilizing vegetation (Jackson and Roering 2009). Jackson and Roering (2009) determined dry ravel to be a significant source of erosion and sediment movement following fire on the steep slopes of the Oregon Coast Range, USA. They estimated that dry ravel after a fire accounts for about 5-20% of the long-term erosion in parts of Oregon. In some areas of their study, they found that ravel was likely responsible for removing large patches of the soil mantle and exposing bedrock. Ravel quickly diminished with time after fire (most of the ravel movements occurred within 2 weeks following fire) because of a loss of available sediment material.

A less commonly studied type of debris flow results from sediment bulking, in which overland flow accumulates sediment until it becomes a debris flow. Gabet and Bookter (2007) found bulked debris flows started as surface runoff accumulating sediment, which subsequently incised a rill, increased in size to a gully, and then formed into a bulked debris flow. Their study sites were located on steep, low-order hillslopes in southwestern Montana, an area that had experienced a fire a year prior to the flows. This type of debris flow is indicated by a lack of landslides scarps and may have landform implications (Cannon 1997, 2001; Cannon et al. 2001b; Gabet and Bookter 2007). Gabet and Bookter (2007) discussed that gullies and debris flows likely formed and filled in throughout time, and suggested that sediment from talus slides, tree-throw, and bank failure supply sediments that fill-in the excavated gullies. They noted the importance of this fire-geomorphic relationship to hillslope processes and how it influences landforms over geologic time scales.

Soil creep and snow avalanches may also form as a result of fire and the loss of vegetation on hillslopes (Swanson 1978; Sass et al. 2010). Soil creep, the slow movement of soil downslope, may occur or increase after fire because of the loss of stabilizing vegetation and roots, and may be a significant long term factor resulting from fire. Relative to a fire event, soil creep usually occurs before roots systems have re-established, but after the soil has become wettable and potentially saturated. Vegetation loss in high elevation areas will alter snow accumulation and melting patterns. Vegetation also acts to stabilize snow, and the loss of vegetation in avalanche initiation areas may result in increased avalanche events (Sass et al. 2010).

Wind Erosion

Though erosion from water is a more common occurrence and well addressed in the literature, wind-induced erosion can also occur after fire, especially in non–forested areas (Zobeck et al. 1989; Cannon et al. 1998; Whicker et al. 2002; Vermeire et al. 2005). Wind is a particularly significant agent of erosion after fire in arid and semi-arid lands (Whicker et al. 2002; Ravi et al. 2006, 2009), although even winds at 20 m s^{-1} may only result in wind-blown sediment 20% of the time (Stout 2001). Atkinson (1984) found 8 cm of sediment and ash built-up on the windward side of objects in their study located south of Sydney, Australia. Soil type, soil moisture, hydrophobicity, ground cover, and time of year are factors that influence the amount of wind-blown erosion (Fryrear 1995; Vermeire et al. 2005; Ravi et al. 2009). Ravi et al. (2009) found that wind erosion was enhanced following fire and determined that hydrophobic soils resulted in increased soil loss from wind erosion following fire in Cimarron National Grasslands in Kansas, USA. Hydrophobic soils are drier than normal, and lack of soil moisture is a significant factor in wind-blown erosion. The implications of wind erosion include the loss of soils as well as nutrients from some areas and accumulations of soils and nutrients elsewhere. Wind-blown sediments result in redistribution of nutrients, often from areas of little to no vegetation to vegetation patches, forming a mosaic of soil nutrient patches and influencing vegetation growth (Ravi et al. 2009).

Hydrology

Fire can result in significant changes to hydrology and stream morphology (Brown 1972; Prosser and Williams 1998; Dwire and Kauffman 2003; Shakesby and Doerr 2006; Bart and Hope 2010). Many of the processes that occur after a fire, such as increased erosion, decreased infiltration, mass movements, and loss of vegetation, affect streams. The effects of fire may include a loss of riparian vegetation; sediment influx from debris flows and erosion; the reduction of log jams during fire and the resulting release of stored sediment (Cannon and Reneau 2000; Benda et al. 2003; Larsen et al. 2006); changes to in-channel erosional and depositional features (Moody and Martin 2001b); increased hillslope runoff, stream flow, and storm flow (Keller et al. 1997; Scott 1997; Fierra et al. 2000; Moody and Martin 2001a); the release of dry ravel into the stream channel (Florsheim et al. 1991; Shakesby and Doerr 2006); turbation (Gresswell 1999); and increased amounts of woody debris within several years following fire (Bendix and Cowell 2010). Hydrologic changes after fire have been studied in small and large catchment basins in numerous ecosystems and results have varied extensively.

Runoff

Hydrophobicity, decreased infiltration, and vegetation and litter loss lead to greater overland flow after a fire, which can result in increased base and storm flows in streams (Lindley et al. 1988; Lavabre et al. 1993; Scott 1993; Hessling 1999; Loàiciga et al. 2001). Johansen et al. (2001) used a rainfall simulator to determine runoff rates after a fire in a semiarid forest in New Mexico. They found that 45% of the 120 mm of applied precipitation resulted in runoff in the severely burned areas compared to the 23% in the unburned sites. This result correlated with greater areas of bare mineral soil. Several studies found that runoff was directly dependent on rainfall intensities and regimes (Rubio et al. 1997; Gonzàlez-Pelayo et al. 2006). High intensity rainfall generated more runoff in burned soils than in unburned soils and lower intensity rainfall events (Gonzàlez-Pelayo et al. 2006). Cerdà and Lasanta's (2005) results on runoff in the Central Spanish Pyrenees indicted that rates were also significantly dependent on vegetation and soil characteristics in addition to rainfall events and intensities. Their plots averaged 48.49 mm of runoff per year for the control plot compared to two burned plot runoff values of 78.69 mm $year^{-1}$ and 94.3 mm $year^{-1}$.

Time of year also factors into runoff values because of the nature of storms and vegetation growth. In areas that experience intense summer thunderstorms or high rainfall events, runoff may be greater during the summer than winter, even though less vegetation growth would be present during the winter (Gonzàlez-Pelayo et al. 2006). If rainfall amounts and intensities are comparable in winter and summer, higher runoff rates will likely occur during the winter because of less vegetation growth. Rainfall regimes also influence soil hydrophobicity, which affects runoff rates (Gonzàlez-Pelayo et al. 2006). Consistent wetting of the soil may reduce soil hydrophobicity, leading to increased infiltration and lower runoff values. Gonzàlez-Pelayo et al. (2006) found an increase in runoff of 77.15% in burn plots compared to control plots in their study in a Mediterranean region of Llíria-Valencia, Spain in the year following fire. They determined that the spring/autumn average maximum-intensity rainfall was 18.49 mm h^{-1} within a 30 minute time span, and was of short duration, compared to 5.51 mm h^{-1} within a rain event of long duration during the winter. This difference in precipitation regimes resulted in 57% less runoff in the winter for the high severity fire and 65% in moderate intensity fire plots. A reduction of 18% was found in the control plots.

Streamflow

Effects of fire on streamflow vary widely, with some systems witnessing significant increases to streamflow, whereas others experience no noticeable difference between pre- and post-fire flows (Chandler et al. 1983; Bosch et al. 1984; Lindley et al. 1988; Britton 1991; Lavabre et al. 1993; Scott 1993; Hessling 1999; Loàiciga et al. 2001; Aronica et al. 2002). Streamflow is often a factor of numerous terrestrial variables, which themselves respond to fire in different ways. Hessling (1999) and Loàiciga et al. (2001) found increased streamflow of as much as 30-50% after fire; however, Aronica et al. (2002) did not find any significant post-fire changes in streamflow in a similar sized watershed. Several studies have found that streamflow increases in small (<5 km^2) and large (>50 km^2) watersheds (Lavabre et al. 1993; Scott 1993; Hessling 1999). Bart and Hope (2010) assessed six paired catchment basins greater than 50 km^2 in area in a Mediterranean–Climate Region of California and found that streamflow varied among their sites. They determined that soil wetness was the most significant factor in post–fire streamflow changes and that catchment and burn area did not exert differences to streamflow, either monthly or annually. Comparatively, Beeson et al. (2001) created a hydrologic model to predict and map overland flow in response to fire and determined that post-fire response would lead to significantly greater overland flow and therefore influence stream morphology and increase flow rates. The great variations in results in post-fire streamflow indicated that streamflow reactions depend greatly on spatial and temporal variability, including site- or region-specific characteristics, weather patterns, vegetation, burn severity, season, groundwater and surface water dynamics, and topography (Townsend and Douglas 2000; Beeson et al. 2001; Jung et al. 2009).

Sediment Loads and Channel Morphology

Higher runoff rates and the resulting increase in streamflow following fire may result in added sediment to streams and influence stream channel morphology with both increased flows and sediment loads, especially in areas with steep slopes (Laird and Harvey 1986; Morris and Moses 1987; DeBano et al. 1998; Inbar et al. 1998; Benda et al. 2003). Morris and Moses (1987) concluded that fire events may contribute to much of the long-term sediment yield in the Colorado Front Range. Cerdà and Lasanta (2006) found an erosion rate of 143 kg ha^{-1} $year^{-1}$ in burnt plots of the Central Spanish Pyrenees compared to 89 kg ha^{-1} $year^{-1}$ for the control plots. Johansen et al. (2001) found sediment yields of 76 kg ha^{-1} mm^{-1} in burned plots compared to 3 kg ha^{-1} mm^{-1} in unburned plots on their study in the semiarid forest of New Mexico, USA. They found that sediment yields were directly proportional to loss of ground cover, however, runoff and sediment loads were not proportional. Sediment yields were about 25 times greater in burned plots compared to unburned plots, whereas runoff rates were only about 2 times greater in burned areas. They attributed this result of significant sediment yields to a loss of ground cover, which would lead to the erosive forces of increased rainsplash, shear erosion, and overland flow. Similar to some other studies (Campbell et al. 1977; Davenport et al. 1998), they found that sediment yield increased significantly when bare soil exceeded 60-70%. Sediments may reach stream channels, and subsequently be transported downstream or be deposited within the channel (Bart and Hope 2010). This result will lead to increased stream turbidity, changes to channel form, and enhance erosion as sediments are transported out of a system.

Large Woody Debris and Riparian Zones

Large woody debris has numerous implications to stream channel geomorphology, including sediment retention, bank stability, increased roughness, decreased velocity, and changes to channel and bank morphology (Keller and Swanson 1979; Gurnell et al. 2002; Faustini and Jones 2003; May and Gresswell 2003; Montgomery and Piégay 2003). Woody debris, or potential future woody debris, may be lost during a fire from the incineration of wood in the channel and trees located within the riparian zone. Conversely, however, tree mortality as a result of fire can lead to an increased influx of woody debris in stream channels following fire (Bragg 2000; Bendix and Cowell 2010). Bendix and Cowell (2010) present an overview of large woody debris (LWD) and riparian vegetation after fire in their recent publication on riparian tree mortality. They found post-fire tree mortality of 94% within the riparian zone of their study, which was located in two small watersheds in the southern California chaparral region. After 2 years, 16.8% of the dead trees had fallen. Their results indicated that tree species was an important factor in determining which trees had fallen within the 2 years since their initial sampling following the fire, and therefore, which species would likely serve as woody debris in stream channels.

Pre-historic Fire

Fire histories have attracted much interest because of the information they can provide on long-term climate-fire relationships and fire frequencies and events over the span of thousands or millions of years (e.g. Meyer et al. 1995; Meyers et al. 2001; Gavin et al. 2003; Roering and Gerber 2005). Fire events in the early Holocene were frequent and likely responsible for significant erosion and hillslope aggradation (Personius et al. 1993; Roering and Gerber 2005). An assessment of past fires may be made by analyzing sediment stratigraphies in alluvial fans (Meyer et al. 1995), tree-ring chronologies (Gavin et al. 2003), and with the use of charcoal studies (Long et al. 1998; Gavin et al. 2003). The presence of charcoal and/or ash within a layer of sediment indicates that the layer was deposited as a result of post-fire erosion. Though documented histories of fire extend back 6,000 years for parts of Europe, those of North America are much shorter, and even 6,000 years does not provide much insight into long-term, climate-fire and fire-erosion relationships.

Meyer et al. (1995) and Meyer et al. (2001) analyzed sediment deposits using radiocarbon dating to determine erosion amounts in pre-historic time. Meyer et al. (1995) found that 30% of an alluvial fan deposit was comprised of fire-related sediment in their study on an alluvial fan in northeastern Yellowstone National Park. They determined fire-induced sediment pulses from 7500, 5500, 4600–4000, and 950-800 years B.P., and found that major sediment deposition activity occurred from fire in conjunction with the Medieval Warm Period. Their results regarding prehistoric fire activity as well as more recent events, indicated that increased fire activity corresponded to warmer climate cycles and conditions.

Roering and Gerber (2005) used measurements of geomorphic response rates after a fire with a high-resolution airborne laser swath mapping (ALSM) topographic dataset and built a physically based model to determine erosion rates of natural landscapes on steep soil-mantled slopes in the Oregon Coast Range over long-time scales. Their results indicated that steep slopes were an important factor in post-fire sediment yield. However, most 10 m or 30 m digital elevation models do not contain the accuracy necessary for determining the distributions of steep hillslopes and their specific gradient, which is crucial for calculating more accurate erosion rates. They found that fire may account for about 50% of long-term sediment erosion on steep slopes and concluded that fire could be responsible for high aggradation rates in the early Holocene.

Geomorphic and Topographic Influences on Fire

Geomorphic and topographic features and processes may also influence fire behavior, especially with low to moderate intensity fires. Debris flows and snow avalanches form linear landscape features that extend perpendicular to the contour of slopes and serve as breaks in vegetation (Butler 2001). The lack of fuel in these paths may act to stop, slow-down, or alter fire movement (Malanson and Butler 1984). Unvegetated stream channels, which lack fuel, serve a similar function (Pettit and Naiman 2007). Riparian zones may also form fire breaks because of their humid, moist conditions; however, in dry conditions, they may also function as fire corridors. The orientation of the breaks in regard to fire movement will strongly determine their effectiveness in stopping a fire. Debris flows, avalanche paths, or stream channels that extend perpendicular to the fire will be more effective compared to linear breaks that run parallel to the fire advancement. These breaks, however, will usually not stop high severity or crown fires, which are sufficiently intense to jump breaks.

Topographic features may also influence fire behavior. Slope aspect and angle, mountains, valleys, leeward and windward aspects, and how these features interact with wind direction, moisture patterns, and vegetation growth may influence fire movement on a landscape. In the Northern Hemisphere north of the Tropic of Cancer, southern and western slopes are often drier and therefore facilitate fire ignitions and more intense burning (Jackson and Roering 2009). North-facing slopes usually contain more vegetation and facilitate vegetation regrowth (Rebertus et al. 1991; Stueve et al. 2009), whereas south facing slopes have drier, less cohesive soils (Marques and Mora 1992; Jackson and Roering 2009). These factors can result in significantly higher erosion rates on south- and west-facing slopes. Winds associated with mountains and valleys may carry fire up- or downslope depending on the time of day, season, and location.

Conclusion

Fire unequivocally serves as a geomorphic agent in many locations, particularly in fire prone areas. Changes to soil properties, increases in erosion and weathering rates, and influences on hydrology may all be affected by fire. The relationship between geomorphology and vegetation acts as a strong factor in the influence that fire exerts on geomorphology. Vegetation serves as fuel for fire, which affects geomorphic processes and features with direct heat and vegetation loss. Disturbances to geomorphology subsequently influence vegetation re-growth, and inversely, vegetation re-establishment affects soil conditions, erosion processes, and hydraulic factors. The interactions among fire, geomorphology, and organisms are extensive, and literature on fire reflects the great influence that fire can exert on a location. With particular regard to fire-geomorphic interactions, some topics are well addressed in the literature, such as erosion (e.g. Morris and Moses 1987; Moody and Martin 2001b; Shakesby et al. 2003; Cerdà and Lasanta 2005; Smith and Dragovich 2008; Cannon et al. 2010), whereas other topics are lacking, including the influence of fire on weathering (Zimmerman et al. 1994; Allison and Bristow 1999; Dorn 2003; Shakesby and Doerr 2006) and large woody debris (Bendix and Cowell 2010). Although literature on fire–prone areas is extensive, fewer studies are available on less fire-frequented areas (Gavin et al. 2003). The literature presented in this chapter represents a broad overview on available fire-geomorphic literature, but is by no means all-inclusive. However, this sampling of the literature reveals that fire-geomorphic interactions are often important to many fire-affected landscapes around the world. It also indicates that many gaps still remain in the understanding of fire-geomorphic-organism interactions. The results of fire vary to such great extents depending on local environmental conditions and fire severity that studies are often required for many different locations under various conditions and fire intensities before beginning to understand fire’s effects on particular sites. Changes to fire regimes and the potential effects that climate change may have on fire emphasize the importance of understanding fire’s effects on the landscape, and how changes to fire may subsequently alter geomorphic dynamics.

Theoretical Framework

Biogeomorphic Disturbance

This research topic fits within the overall framework of biogeomorphic disturbance. The interactions between geomorphology and ecology are extensive (*e.g.* Hack and Goodlett 1960; Viles 1988; Butler 1995; Butler et al. 2007; Hjort and Luoto 2009; Marston 2010; Corenblit et al. 2011; Rice et al. 2012) and an understanding of the interrelated processes is necessary for greater basic and applied knowledge of natural processes, especially after a disturbance event (Viles et al. 2008; Rice et al. 2012). Disturbance has been defined as a (relatively) discrete event that disrupts an ecosystem, community, substrate material, or physical environment (Picket and White 1985), and such examples include fire, avalanches, floods, and insect invasions. Biogeomorphology, with its focus on ecologic and geomorphic interaction, is situated to lend a better understanding on how systems respond after a disturbance. Disturbances such as human activities, storm events, fire, and climate change may exert significant changes to both ecologic and geomorphic systems. However, studies often just focus on either the ecologic components or the geomorphic factors, and rarely integrate the two (Stallins 2006; Viles et al. 2008). Although geomorphic and ecologic variables are often obviously affected by disturbance, the response of the geomorphic and ecologic components often occurs at different spatial and temporal scales.

Geomorphology can produce either a negative or positive feedback in biogeographic systems, and likewise, ecology can act to induce a negative or positive feedback in geomorphic systems. Viles et al. (2008) discussed how geomorphic systems respond in a non–linear manner to disturbance events and that biologic factors can influence lag times, response rates, and sediment yields. One of the influencing factors to geomorphic response is vegetation, which can act as a stabilizing or destabilizing factor to geomorphic processes (Viles et al. 2008). Vegetation growth, for example, can stabilize sediment, and, conversely, the loss of vegetation can result in increased soil erosion. Similarly, geomorphic impacts can act as stabilizing or destabilizing agents to ecological systems. The geomorphic actions of weathering and sediment deposition produce sediment for vegetation growth, therefore serving as stabilizing agents for ecological systems. However, sediment deposition and mass movements can bury vegetation, which is a geomorphic factor that acts as a destabilizing agent to biologic features.

Viles et al. (2008) also stated that studies focused on disturbances from external forces, such as fire, and the impacts to coupled biotic and abiotic processes are lacking in the literature. However, fire causes significant impacts to both ecologic and geomorphic systems. As discussed previously, fire impacts geomorphic processes such as soil erosion, soil properties, micro-topography, and cryogenic features (Swanson 1978; Christensen et al. 1989; Woods et al. 2006; Coop and Shoettle 2009).

Complexity Theory

Biogeomorphic disturbance may be evaluated within the concept of complexity theory (Malanson 1999; Stallins 2006; Corenblit et al. 2007). Complex systems have been defined as those that respond or behave in a nonlinear manner, respond with complicated outcomes, and/or lie between order and chaos (Schumm 1973; Schumm

1991; Gell–mann 1995; Malanson 1999). Fire disturbance and the resulting response are often good examples of complex outcomes. Fire can lead to a variety of system responses, which are often related to fire behavior (intensity, duration), environmental conditions (soil and vegetation types, fuel load, topography), and weather patterns. Variability among these, as well as other factors, will result in a suite of outcomes. Understanding the variables of the system may aid in better knowledge and projections of the response of the system. Numerous modes to understanding and evaluating complex systems exist, and Stallins (2006) presents several approaches within the framework of complexity theory in addressing coupled ecology and geomorphology systems. Multiple causality and feedback are two approaches within complexity theory that will aid in evaluating this dissertation topic.

Multiple Causality

Multiple causality will contribute to better evaluation of complex systems. The acknowledgement of more than one factor contributing to an outcome or result (Schwartz 1971), the interlinkages between form and process (Smith et al. 2002), and the relationships among geomorphic variables and ecologic variables (Stallins and Parker 2003) are examples of multiple causality and its potential role in understanding systems. Multiple causality may be especially helpful in understanding and evaluating coupled geomorphic–ecologic disturbance response. Ecologists and geomorphologists apply the concept of multiple causality to topics in their respective fields, but few have used it to address the interactions between geomorphology and ecology (Stallins 2006). Biogeomorphic response to disturbance is often complex and various factors may be dependent on both external forcings, as well as interactions within the system itself (Viles et al. 2008). Geomorphic processes, landforms, vegetation distributions, and successional patterns may interact with each other, forming complex relationships and outcomes that are dependent on the response of each of the individual components.

Both biogeographers and geomorphologists deal with the issue of pattern and process, and the intersection of these two fields provides a rich area for exploration of form and process as it relates to both ecologic and geomorphic factors (Turner 1989; Resler 2006; Stallins 2006). Stallins (2006) presented an example of fire and the process–form interactions between geomorphology and ecology. He described the influence of topography on vegetation distributions, which can be disturbed by fire, and the resulting effect of the loss of vegetation on erosion. If significant erosion occurs, fire breaks can form, which will then influence future disturbance. Therefore, the topography of the landscape – the form, subsequently influences erosion – the process. Form and process are recursive in such an example. The causality of patterns however, is often difficult to determine among vegetation distributions, geomorphic process, and landform (Parker and Bendix 1996). Feedback among factors of process and form may aid in addressing potential causes of resulting patterns (Marr 1977).

Feedback Loops

Causes, response, and interactions within complex biogeomorphic systems will be linked under the theory of feedback (Marston 2010). Feedback loops aid in explaining and exploring multiple causality in form and process, disturbance response between ecologic and geomorphic variables, and patterns in vegetation and geomorphic factors and associated processes (Wilson and Agnew 1972; Malanson 1997; Callaway et al. 2002; Alftine and Malanson 2004; Bekker 2005; Callaway 2007). Interdependencies among various factors, processes, and scales will be recursive and subsequently influence resulting patterns that may then, in turn, affect the involved variables and processes. Viles et al. (2008) explained that biological factors influence the reaction and relaxation times in geomorphic processes following a disturbance. Ecological factors (such as loss of vegetation) lead to geomorphic destabilization, and in this model, increased sedimentation and a greater response time. Vegetation growth, however, leads to geomorphic stabilization and a decreased sedimentation and response time. This model is applicable to geomorphic disturbance following fire. However, treeline conditions would likely add compounding factors. Response times may be increased because of a greater time between the disturbance event and vegetation establishment because vegetation regeneration is often slower at treeline compared to lower elevations. Geomorphology is a factor in vegetation growth at treeline (Butler et al. 2004; Resler et al. 2005; Butler et al. 2007), as well as impacted by fire events (e.g. Moody and Martin 2001b; Woods et al. 2006). The result of fire on geomorphology and vegetation regeneration at treeline was assessed in the context of complex response and non-linear biogeomorphic models (Viles et al. 2008).

The stabilizing and destabilizing interactions between geomorphology and vegetation factors produce both negative and positive feedbacks to tree establishment at treeline after fire. Damage to vegetation from fire will force treeline downslope, and allow for alpine tundra vegetation to dominate where trees had been prior to fire (Bollinger 1973). A loss of vegetation will increase the radiation exposure and evaporation, and remove sources of seed. Treeline advancement would then be repressed and below its otherwise theoretical climatic elevation and any potential advancement hampered, a negative feedback. However, fire could also increase the importance of positive feedbacks. Facilitation mechanisms occur at treeline (Resler 2006), allowing for the growth of vegetation in areas that otherwise would not be able to support woody plants. Coop and Schoettle (2009) found that facilitation objects were significantly associated with seedling establishment after fire within the subalpine in Colorado. Geomorphic features and established vegetation lead to treeline advancement upslope, which will then further facilitate tree establishment and growth, allowing trees to grow above their theoretical limit. Fire could also improve soil conditions, such as the addition of available nutrients from burned plant matter, and decrease competition, a positive feedback. Positive feedbacks have been documented at treeline without fire, however, fire will possibly influence the role of feedbacks and impact the rates or time differences between feedback processes. The role of feedback can change over time and in response to variations in environmental conditions (Bekker 2005). The various results of fire on treeline position emphasize the need for further exploring the influence of environmental site conditions and the potential feedbacks between abiotic and biotic factors.

Spatial Pattern

<u>Ecotone Dynamics</u>

Ecotones, as zones of transition from one ecosystem or vegetation community to another (Holland 1988), are viewed as important indicators of climate change (*e.g.* Gosz 1993; Noble 1993; Risser 1995; Allen and Breshears 1998; Parmesan and Yohe 2003;

Hinzman et al. 2005; Cannone et al. 2007; Beckage et al. 2008; Loarie et al. 2009). The plant species within an ecotone are often at their physiological limits, and changes to the position of the ecotone are often related to changes on the limits of plant establishment and growth. Semiarid, arid, and alpine ecotones are some of the most well studied, and are thought to be the most sensitive to changes in climate (Grover and Musick 1990; IPCC 1996; Camarero and Gutiérrez 2007). Ecotones vary in abruptness, with some displaying distinct lines between two ecosystems or vegetation communities, whereas others extend great distances as one system gradually fades into another (Gosz 1993; Malanson 1997; Walker et al. 2003). The combination of two ecosystems or vegetation communities forms areas of high biological diversity. External and internal controlling factors of ecotones determine the spatial extent and abruptness of ecotones.

The spatial scale at which an ecotone is analyzed is an important factor in understanding the patterns and processes taking place within that ecotone (Gosz 1993; Risser 1993; Malanson 1997; Phillips 1999). Traditionally, ecotones were viewed at three spatial scales – boundaries of wildlife habitat, treeline, and continental scale biomes (Gosz 1993). However, over the past several decades, the study of ecotones has gained significant attention in the literature as an ecological system, an indication of climate change, and as important landscape features (Gosz 1993; Risser 1995; Kupfer and Cairns 1996; Malanson 1997; Parmesan and Yohe 2003). Ecotone studies over the past several decades have expanded to include levels from an individual plant up to continental biome scales (Hinzman et al. 2005). The processes that occur at the fine-scale will subsequently influence patterns of the coarser scale ecotone.

Boundary Dynamics and Edge Effects

The subject of ecological edges is well studied in ecology, especially in regard to forest fragmentation and wildlife management (Leopold 1933; Chen et al. 1995; Rodríguez–Loinaz et al. 2012). Vegetation, animals, and microclimates may be influenced by the presence of edges. Vegetation diversity may increase or decrease in relation to edge effects and species and growth form (Rogríguez-Loinaz et al. 2012). Significance of edge influence (SEI) and depth of edge influence (DEI) were two indices assessed by Chen et al. (1995). Chen et al. (1995) found that soil and air temperatures, humidity, solar radiation, and wind were influenced 30 m to over 240 m into the forest from the edge. Edge orientation and local weather conditions were strong influences on the differences in variables among their sites.

Tree Succession and Seedling Establishment

Abiotic and biotic interactions determine the spatial patterning of trees (Gates 1942; Beals 1965; Diehl 1981; Collins et al. 1982; Germino et al. 2002; Bekker 2005; Resler et al. 2005). Factors such as soil conditions, available moisture, seed rain, micro-topography, microclimate, niche limitations, and competition are important determinants of the micro–scale effects of seedling establishment (Gates 1942; Zedler and Zedler 1969; Schwintzer 1978; Reiners and Lang 1979; Boatman et al. 1981; Hacker and Bertness 1996; Kikvidze et al. 2005). Micro–scale conditions are the primary influence on changes in vegetation conditions and shifts in distributions within macro–scale limitations (*e.g*. climate, geology) (Gosz 1993). Seedling establishment and survival is integral to processes of succession of trees after a disturbance, and resulting distribution patterns (Bazzaz 1979).

Snow is important for seedling establishment and survival in high elevations of the middle and high latitudes. Snow cover provides protection for seeds and seedlings during the winter season and contributes to soil moisture as the snow melts. Hattenschwiler and Smith (1999) found seedlings establishment in association with snow depths between 0.5 m and 1.5 m in ribbon forests at treeline in the central Rockies. Too much snow they hypothesized would result in too short of a growing season, whereas too little snow may provide inadequate moisture in the soil for seedlings during the summer.

Low temperatures, particularly below freezing temperatures, and the duration of the cold temperatures are some of the primary effects on the distribution of vegetation in mountains (Germino et al. 2002; Körner 2003). Plants in various elevations are adapted to the climate of their location because temperature ranges and variations are relatively stable over long time spans (at least centuries). However, early freezes during the fall and late freezes in the spring may result in plant mortality or damage. Snow accumulation patterns and depth, and the length of snow cover, also greatly affect vegetation (Körner 2003). Snow depths influence soil moisture and temperature and solar radiation — factors that affect vegetation establishment and growth (Billings and Bliss 1959; Liston 1999). Snow acts as an insulator during the winter months and provides moisture to growing plants in the spring. Changes to snowfall amounts and snowpack duration will likely influence vegetation distributions. Vegetation also influences snow depth patterns by capturing wind–blown snow (Hiemstra et al. 2002).

Wind has been found to strongly influence the spatial distribution of seedlings and establishment patterns within treeline (Hattenschwiler and Smith 1999; Alftine and Malanson 2004). The effects of wind include desiccation, leaf cuticle abrasion, and flagging (Hadley and Smith 1986). Wind can also affect seedling establishment by influencing snow depth patterns. Bekker (2005) found tree fingers growing in an east–west orientation as a result of the prevailing westerly winds within the treeline ecotone.

Microtopography may also influence the establishment patterns of seedlings (Butler et al. 2004; Bekker 2005; Resler et al. 2005). Features such as relict solifluction terraces can provide protection for seedlings and create conditions more favorable for seedling germination and establishment (Butler et al. 2004). Bekker (2005) and Resler et al. (2005) found seedlings and tree establishment patterns in strong association with microtopographic features within alpine treeline at Glacier National Park, MT. Microtopographic features can shelter seedlings from wind and solar radiation and serve to capture seeds, as well provide areas of soil conducive to seedling establishment, including less compact soil (Butler et al. 2004).

In a similar manner, boulders and surface rocks facilitate seedling establishment by capturing fine sediments and wind–blown seeds, providing shade, reducing evaporation, and offering protection from wind (del Moral and Bliss 1993; Tejedor et al. 2003; Resler 2006; Pérez 2009a, 2009b). The dust-trap effect refers to the process in which wind-blown fine particles that are in suspension are blocked by the rocks and fall to the ground (Díaz et al. 2004). This process not only adds fine sediments, and potentially nutrients, to the ground surface but also increases the moisture retention of the soil (Pérez 2009b). Surface rocks have a greater heat capacity than dry soil and therefore, reduce temperature variations (Mehuys et al. 1975). Diurnal soil temperatures in association with boulders and surface rocks will display the pattern of remaining cooler during the day and warmer during the night relative to nearby soils (Mehuys et al. 1975; Pérez 2009a).

Rock lithology may affect temperatures, rock fragment cover, and fragment sizes, and therefore, plant establishment (Kruckeburg 2002; Malanson et al. 2007). Darker rocks will be associated with warmer temperatures compared to lighter colored rocks. Some rocks will fracture into coarser particles, whereas others will tend to weather into finer particles. The particle sizes will subsequently influence the rocks' ability to capture sediments and seeds and influence temperatures. Extent of rocks on the ground surface and the distribution of fragments and fragment sizes can create a mosaic of micro–scale conditions (Veblen 1992; Pérez 2009a). The literature on surface rocks is well covered in low elevation environments, but is lacking in high elevation areas and in relation to geomorphic and biogeomorphic factors (Pérez 2009a).

CHAPTER 3: HYPOTHESES

Research Question 1*:* Are soil conditions significantly different in burned areas compared to unburned areas?

Objective 1: Compare soil conditions in burned and unburned areas.

Hypothesis 1**:** Fire significantly affects soil conditions.

Sub-hypothesis 1: Duff depth will be significantly lower in burned areas.

Sub-hypothesis 2: Soil penetrability will be significantly higher in burned areas

Sub-hypothesis 3: Effective soil depth (ESD) will be significantly lower in burned areas.

Sub-hypothesis 4: Particle sizes will be significantly higher in burned areas.

Sub-hypothesis 5: Soil loss will be evident in burned areas and not in unburned areas.

Sub-hypothesis 6: Soil pH and other nutrients will be significantly different between burned and unburned areas.

Research Question 2**:** Do burn and erosion severity significantly relate to vegetation and soil variability?

Objective 2**:** Assess potential relationships between paired comparisons of erosion severity, burn severity, vegetation, and soil penetrability.

Hypothesis 2**:** Significant associations will be found among erosion severity, burn severity, and vegetation and soil conditions.

Sub-hypothesis 1: High erosion severity and high burn severity will be significantly associated.

Sub-hypothesis 2: Low erosion severity and high herbaceous vegetation cover will be significantly associated.

Sub-hypothesis 3: Low burn severity and high herbaceous vegetation cover will be significantly associated.

Sub-hypothesis 4: High krummholz density will be associated with lower soil penetrability values.

Research Question 3: Has fire contributed to boulder spalling?

Objective 3: Evaluate the influence that fire has on boulder weathering.

Hypothesis 3: Extent of boulder spalling is greater in burned areas.

Research Question 4: Are geomorphic variables of seedling micro–sites affected by fire, and what variables are associated with seedling sites compared to random micro–sites?

Objective 4**:** Characterize seedling microsite (0.25 x 0.25 m) conditions and compare them between burned and unburned sites and between plots with seedlings and random plots.

Hypothesis 4**:** Variables at the seedling micro–site scale (0.25 x 0.25 m) are significantly affected by fire.

Sub-hypothesis 1: Seedlings will be significantly associated with higher ESD compared to random micro-plots.

Sub-hypothesis 2: Seedlings will be significantly associated with lower soil penetrability values compared to random micro-plots.

Sub-hypothesis 3: Seedlings will be significantly associated with larger particle sizes compared to random micro-plots.

Hypothesis 5: Geomorphic and facilitative factors conducive to seedling establishment will be significantly associated with micro–sites containing seedlings compared to random sites.

Sub-hypothesis 1: Seedlings will be significantly closer to the closest objects than random micro-plots.

Sub-hypothesis 2: Seedlings will be significantly closer to second and third objects compared to random micro-plots.

Sub-hypothesis 3: Distance of objects will vary significantly among species.

Research Question 5**:** Do geomorphic variables change in relation to distance from the burn/unburned edge?

Objective 5: Determine if geomorphic variables change in relation to distance from the burn edge.

Hypothesis 6: Geomorphic variables will change in correlation with distance from the burn/unburn boundary.

Research Question 6**:** What influence does topography have on conditions after fire?

Objective 6**:** Characterize soil conditions and vegetation of different sites in relation to their topographic features.

Hypothesis 7: Differences in soil conditions and vegetation patterns will be found at different burned sites in relation to their topographic characteristics.

Sub-hypothesis 1: Sites at lower elevations will have greater pre-fire krummholz densities and more seedling re-establishment.

Sub-hypothesis 2: Sites of lower degree slope will have lower soil penetrability values, higher ESD, and larger particle sizes.

Research Question 7: How does size of quadrat influence results?

Objective 7: Assess various variables within increasing plot size.

Hypothesis 8: Significant differences in means will be found for various variables among plots of different sizes.

Sub-hypothesis 1: Averages between plots smaller than 25 m^2 and equal to and larger than 25 m^2 will differ significantly.

CHAPTER 4: METHODS

Study Area

Three field sites were positioned within Glacier National Park, Montana (48° W and 113° N). Glacier National Park is located within northwestern Montana, adjacent to the Canada–United States border, and forms part of the Waterton–Glacier International Peace Park — a designated World Heritage Site. The American portion of this Park contains 0.4 million ha. Data were collected from a National Park to mitigate the potential disturbances produced directly by human activity, including logging, roads, grazing, and construction. Such activities could influence alpine treeline dynamics, and therefore, affect its response to fire and introduce a confounding variable to the results of this study. Glacier National Park in particular was selected because it contains alpine treeline and complex micro-topography (Butler and Malanson 1989; Butler et al. 1992; Butler and Walsh 1994; Butler et al. 2004), the treeline ecotone is documented to be strongly influenced by geomorphic activity (Butler et al. 1992; Butler 2001; Butler et al. 2007), and several areas within the treeline ecotone have experienced fire within the previous 10 years. These conditions were necessary to address the questions guiding this research.

Glacier National Park

Glacier National Park was established in 1910. The dramatic, glacially sculpted landscape found within the Park forms the northern Western Cordillera (Fenneman 1931). Elevations range from approximately 960 to 3050 m (Butler et al. 1992). The Lewis and Livingston Ranges are the two main mountain ranges in the Park and they run generally parallel with the long north-south axis of the Park (Butler 1986). The Continental Divide, which extends roughly northwest – southeast diagonally across the Park, lies within the Livingston Range in the northern area of the Park before shifting to the Lewis Range. The Lewis Range is located east of the Livingstone Range and within the orographic shadow produced by the Livingston Range. Therefore, the climate on either side of the Continental Divide is distinctly different.

Climate

The Continental Divide serves as a boundary between the wet western and dry eastern halves of Park. Precipitation amounts on the western side can reach 2500 mm annually, whereas the eastern side averages 585 mm per year (Finklin 1986; Butler et al. 1992). The eastern portion is dry and windy, and is considered to have a continental interior climate regime. The western side, however, experiences intense storms and is designated as a modified Pacific-maritime climate. Winds are generally from the west. Temperatures within the Park reflect its northern location, high elevations, and climate changes since the end of the Pleistocene. Within these broad classifications, however, micro-climate variability is found throughout the Park (Butler and Malanson 1989). Some areas receive great accumulations of snow, whereas exposed, wind-swept slopes may have no snow accumulation. Slope and aspect variations combine to create a mosaic of moisture and temperature regimes found throughout the Park.

Vegetation

Climate variations within the Park are reflected in the vegetation patterns found. Common tree species in the eastern half of the Park are *Pinus flexilis* (limber pine), *Pinus contorta* (lodgepole pine), *Abies lasiocarpa* (subalpine fir), *Picea engelmannii* (Engelmann spruce), and *Populus tremuloides* (quaking aspen) (Butler 1979). On the western side of the Park, *Pinus ponderosa* (ponderosa pine), *Pinus monticola* (western white pine), *Abies lasiocarpa* (subalpine fir), *Pseudotsuga menziesii* (*glauca*) (Douglas fir), *Abies grandis* (grand fir), *Tsuga heterophylla* (western hemlock) *Larix occidentalis* (western larch), *Picea engelmannii* (Engelmann spruce), *Thuja plicata* (western red cedar), *Taxus brevifolia* (Pacific yew), *Betula papyrifera* (subcordata) (northwestern paper birch), and *Alnus* spp. (slide alder) are the most common species. The alpine treeline is located approximately 1,900 – 2,000 m a.s.l. (Butler 1979). The treeline ecotone is a patchy mosaic of woody and alpine vegetation produced by competition among the species and disturbance events (Butler and Walsh 1994; Malanson and Butler 1994; Walsh et al. 1994). Above the treeline ecotone, low-lying herbaceous vegetation and dwarf shrubs are commonly found above about 2,000 m (Butler and Malanson 1999).

Topography and Lithology

The area where Glacier National Park is located experienced extensive glaciation during the Pleistocene and Holocene, resulting in prominent peaks and U-shaped valleys.

Theses steep-sloped valleys are generally positioned in a northeast – southwest orientation and finger lakes fill many of them.

Soils are the product of sedimentary parent material, periglacial environment, and colluvium material. The treeline and alpine soils are characteristically slow to develop in the dry, scantily vegetated areas. A soil fertility gradient was identified by Malanson and Butler (1994), and they found that the areas of greatest fertility were located in the lower elevations.

Fire History

The fire regime of Glacier National Park is strongly influenced by complex topography, fuel availability, and the variations in moisture between north- and south-facing slopes and between areas on either side of the Continental Divide (Keane et al. 1999). Stand replacing, passive crown, and non-lethal underburn fire regimes have been described for GNP (Barrett et al. 1991). Stand-replacing fires have a return interval of 120-150 years and are more common in moist areas where fuel loads are greater. Passive crown, defined as complete burn of patches of trees, and low intensity, ground–burning fires are common regimes in the higher elevation areas of patchy fuel loads. Keane et al. (1999) projections indicated that future fires in Glacier National Park will be more crown fires and cover larger areas than under the climate conditions when the study was conducted.

Study Sites

My three sites are located east of the Continental Divide. Two sites are located on Divide Mountain. Upper Divide (Figures 4.2 and 4.3) is located on a saddle (48° 39.5’

N, 113° 23.9' W) at an elevation of 2,200 m. The saddle is positioned with one slope facing east and the other facing west. The second site on Divide Mountain, Lower Divide (48° 40.4' N, 113° 23.6' W) (Fig. 4.4), is located at a slightly lower elevation (2,100 m) and is positioned on a north–facing slope. Divide Mountain experienced a fire, the Red Eagle Fire, at treeline in 2006. The parent material is Altyn limestone. One site is located on Swiftcurrent Mountain (Fig. 4.5), where a fire (the Trapper Fire) occurred in 2003. This site (48° 46.7' N, 113° 46.1 W) is on a southward facing slope at elevations ranging from 2,260 to 2,340 m. The parent material is Grinnelle argillite.

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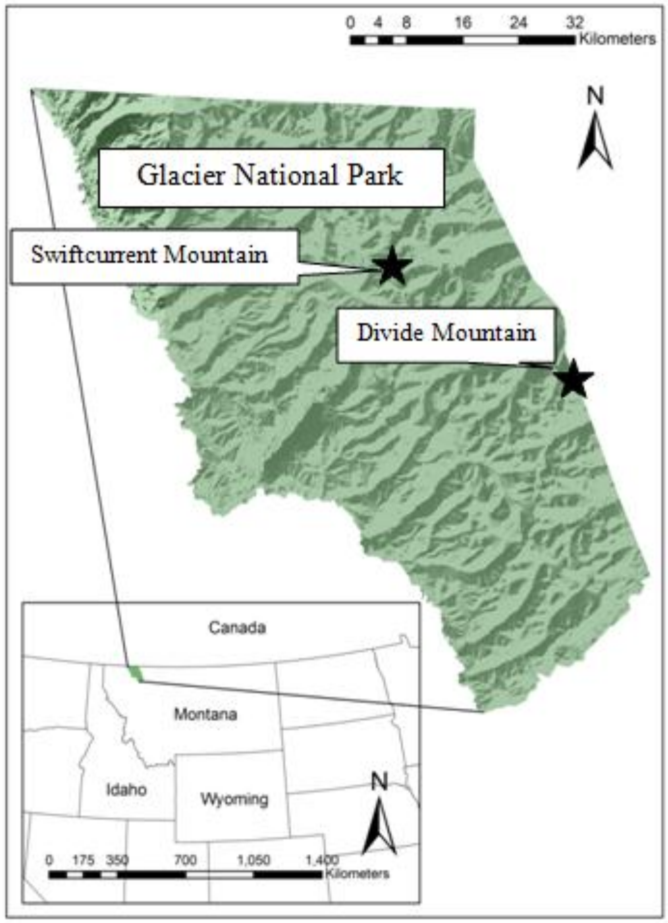

Figure 4.1. Locations of study sites. One site was located on Swiftcurrent Mountain and two were located on Divide Mountain.

Figure 4.2. Upper Divide site. Photograph taken in 2010.

Figure 4.3. Upper Divide site. Photograph taken in 2010.

Figure 4.4. Lower Divide site. Photograph taken in 2011.

Figure 4.5. Swiftcurrent Mountain site. Photograph taken in 2011.

Methods

Quadrat Layout

Field and statistical methods were employed to address the effects of fire on geomorphic and associated vegetation conditions. Quadrats measuring either 5 x 20 m or 5 x 5 m were placed in burned areas and adjacent unburned areas (as control quadrats) at the three sites within the treeline ecotone. The quadrat size used was dependent upon site conditions. If the burned area was too small for a 5 x 20 m quadrat, or if the site changed aspect significantly within a 5 x 20 m area, then the 5 x 5 m quadrat was used. The quadrat size of 5 x 20 m is the same as used by Coop and Schoettle (2009) and the dimensions often fit within the irregularly shaped burned areas. Emphasis was placed on positioning quadrats in unburned areas that most closely resembled the sampled burned area (proximity, degree slope, slope aspect). The number of quadrats per site varied depending on the burn extent and site conditions. The first quadrat was positioned in the uppermost location of the burned area, and the subsequent quadrats placed a random number (as determined by the roll of four dice) of meters from the previous quadrat. Each 5 x 20 m quadrat was partitioned into 5 by 5 m squares. Degree slope (measured with a Brunton clinometer) was recorded for each quadrat by recording the slope of the quadrat sides that run perpendicular to the contours of the slope.

Objective 1

Within each 5 x 20 m quadrat, I took 80 triplicate soil penetrability measurements (20 per 5 x 5 m subplot) and 20 triplicate penetrability measurements within each 5 x 5 m quadrat with the use of a pocket penetrometer. Soil compaction may be influenced by the loss of surface soil and organic material, exposure of the surface to wind and rain resulting from a lack of vegetation, and changes to particle size distributions following fire. I measured the depth of the organic/duff layer at 10 random locations within each quadrat because fire may incinerate this material. I noted evidence of surface erosion and used soil pedestal heights and root exposure as a proxy for measuring the depth of soil loss (Harden 1988). Measurements were collected opportunistically where soil pedestals and exposed roots were present. Soil erosion often occurs after fires because of the loss of stabilizing vegetation in conjunction with runoff and wind. To determine the extent of soil loss over the burned area, I performed a classification on 2005 (pre-fire) and 2009 (post-fire) National Agricultural Imagery Program (NAIP) images. Several different techniques were employed, but only the principle components analysis (PCA) change detection appeared to most accurately determine the best classifications. Unsupervised and supervised classifications were attempted, but they did not result in correct classifications when compared to a natural color image.

I collected soil samples from each site and had them analyzed for pH, conductivity, nitrate-nitrogen (N), phosphorus (P), potassium (K), magnesium (Mg), sodium (Na), sulfur (S), and calcium (Ca). Soil was collected from the top 10 cm of the mineral layer in a stratified manner across the long axis of the quadrats. Soil was collected at five points from within each quadrat and combined for one sample per quadrat. Four samples from the burned area were collected from Lower Divide, three from Swiftcurrent, and seven from Upper Divide. A total of four samples were collected from the unburned areas (one from Swiftcurrent, one from Lower Divide, and two from Upper Divide – one on Upper Divide West and one on Upper Divide East. Samples were analyzed by the Texas A&M Soil Testing Laboratory. Rock clast sizes on the ground surface were measured by recording the long axis of 80 random particles within each 5 x 20 m quadrat and 20 in each 5 x 5 m quadrat. Twenty effective soil depth (ESD) measurements were collected from each 5 x 20 m quadrat and five from each 5 x 5 m quadrat. Effective soil depth was determined by pushing a steel rod into the ground five times at a sampling point and recording the greatest depth.

Data were subjected to statistical analyses. All statistical analyses for this study were performed using SPSS. Data were evaluated for normality using the Kolmogorov-Smirnov test. None of the data were normally distributed. A log transformation was applied but the data remained not normal. Therefore, nonparametric tests were used to address objectives 1, 2, 4, and 6. The Mann Whitney U test was used when comparing two variables, and the Kruskal Wallis test when more than two variables were being compared. A significance level of 0.05 was used. When results of a Kruskal Wallis test were significant, a *post hoc* test was applied by comparing a pair of variables with the Mann Whitney *U* test, and a Bonferroni correction was applied.

Objective 2

Comparisons were made between soil penetrability, percent vegetation cover, krummholz density, erosion severity, and burn severity. Soil penetrability was classified based on the distribution of the data. Percent vegetation was visually estimated. Diameter at ground level (DGL) was recorded for all krummholz within the quadrats. Diameter at ground level was used instead of the more commonly recorded diameter at breast height (DBH) because krummholz often were too short to determine DBH. I measured DGL at the base of each krummholz, immediately above the ground surface.

Erosion severity was ranked from 1 through 5, with 5 being complete and extensive erosion and 1 minimal or none. A 2 indicated about 25% of the area was eroded, a 3 about 50%, and a 4 about 75%. Burn severity was based on the extent of charring and incineration of the krummholz and ranked 1 through 5 (least to most severely burned). A rank of 5 was applied to krummholz burned almost to the ground (Fig. 4.6). A krummholz stem lacking primary branches was categorized as a 4 (Fig. 4.7). If the primary stem plus main branches were present, it was deemed a 3 (Fig. 4.8). Krummholz with small branches and/or bark was a 2 (Fig. 4.9), and if needles were still in place, the burn severity was a 1 (Fig. 4.10). An inherent issue with this classification however, was that I could not determine the health of the krummholz before fire. Those that were dead prior to fire may have been more easily burned and not necessarily reflect the intensity of the fire.

Figure 4.6. Burn severity of 5.

Figure 4.7. Burn severity of 4.

Figure 4.8. Burn severity of 3.

Figure 4.9. Burn severity of 2.

Figure 4.10. Burn severity of 1.

Objective 3

All spalled boulders within the quadrats were recorded. The extent of boulder spalling was evaluated by acquiring the measurement of the long axis of freshly exposed rock on boulders and of rock fragments that spalled off of boulders. I also measured the distance between the spalled fragments and boulder. Only fragments that fit back onto the boulder were measured in association with spalling. Characterization of the spalling and density of spalled boulders were made with the use of descriptive statistics. Overall percent of boulders was determined by visually estimating the percentage of ground covered with boulders for each quadrat.

Objective 4

Geomorphic factors of seedling micro–site conditions were analyzed by randomly placing four 0.25 m x 0.25 m plots (Fig. 4.11) within each 5 x 20 m quadrat or one per 5 x 5 m quadrat, as well as around every seedling found within the quadrats (Coop and Schoettle 2009). A pin toss within each 5 x 5 m subplot determined the location of the random micro-plots. Seedling micro-plots were centered on each seedling. Soil penetrability was measured with 10 penetrometer recordings. Five clast sizes were measured by randomly picking up and measuring the long axis of rocks within the plot. Average effective soil depth was determined by pushing a metal rod into the soil in five random points within the plot and measuring the distance that the rod extended below ground before it would not go any further with a moderate amount of pressure. Percent herbaceous vegetation cover and percent rock cover were visually estimated. Estimates from two persons were acquired to reduce subjectivity. I measured degree slope by placing a Brunton clinometer within the center of the plot. Distance to the nearest three objects (measuring at least 10 cm x 10 cm x 10 cm) was recorded. The distance was measured from the center of the plots to the closest edge of the object. The dimensions of the object and the type of object (boulder, burned krummholz, krummholz stump) were recorded. Seedling species and heights were also recorded. Seedlings were considered individual woody vegetation that had a diameter at ground level equal to or less than 5 mm. Relative sunlight was recorded as either full sun, mostly sun, mostly shade, or full shade in relation to the noon sun. Characterization of the micro–sites was assessed with

Figure 4.11. A random 0.25 x 0.25 m micro-plot in a burned site.

the use of descriptive statistics. Comparisons of the various variables were made between random plots and plots with seedlings, between plots in burned and unburned areas, and plots among the sites using statistical tests.

Objective 5

I assessed the geomorphic components associated with the burned/unburned boundary by placing three transects across a burned patch at Lower Divide. The first transect was placed at the uppermost burn location and successive transects placed a random number of meters downslope from the previous transect. The transect lengths were double the length of the width of the burned patch, and centered on the patch, extending on both side into the unburned krummholz. The first transect was 50 m long, with 10 m in the unburned krummholz, 25 m across the burned patch, and 15 m on the other side of the patch in the unburned krummholz. The second and third transects were each 100 m long and roughly extending 25 on each side of the burned patch and 50 m across the burned patch. I collected data on soil penetrability by measuring 30 triplicate soil penetrations, 30 triplicate effective soil depths, and 30 random clast sizes every 5 m along the length of the transects. Data were analyzed to test for differences in the variables along the transect in relation to distance from the boundary with the use of ANOVA and the Tukey–Kramer *post hoc* test (Moore and Huffman 2004).

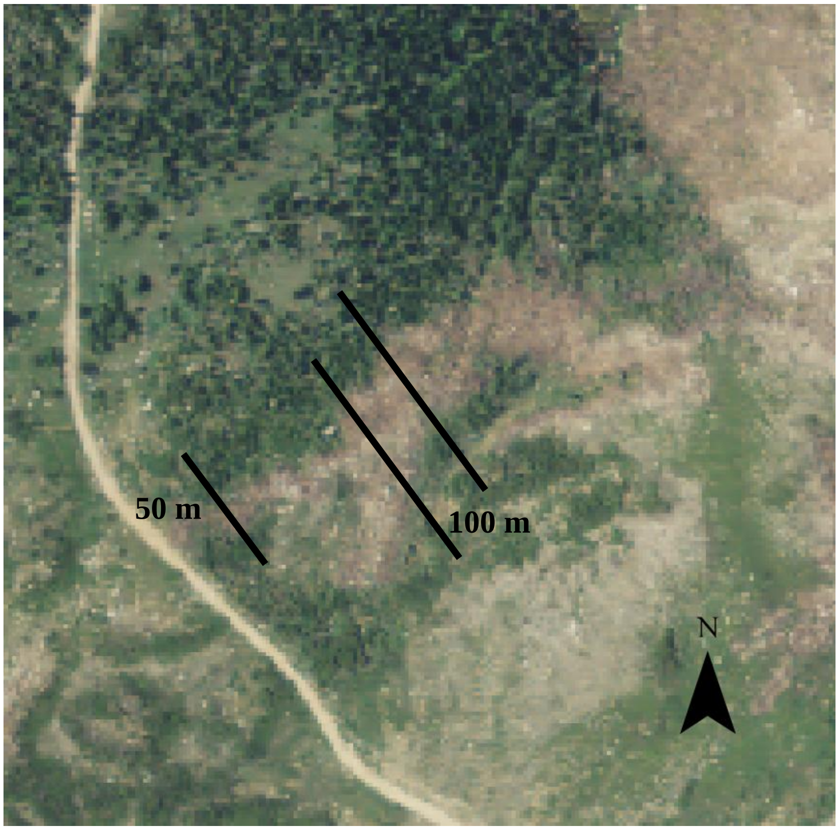

Figure 4.12. Placement of transects across burned patch.

Objective 6

Variables of soil penetrability, ESD, relative sun exposure, particle sizes, and krummholz densities were compared among the three primary sites – Upper Divide (Fig. 4.13), Lower Divide (Fig. 4.14), and Swiftcurrent. Comparisons were also made among locations of different topographic features on Divide Mountain. Divide Mountain contained seven different sites of topographic variation that were identified as sub-sites – Upper Divide West (UDW), Upper Divide Ridge (UDR), Upper Divide East (UDE), Upper Divide Far East (UDF), Upper Divide Islands (UDI), Lower Divide Gully (LDG), and Lower Divide North (LDN). Swiftcurrent Mountain did not provide the topographic complexity observed on Divide Mountain, but quadrats were placed at different elevations on the slope.

Figure 4.13. Upper Divide site. Photograph taken July 2010.

Figure 4.14. Lower Divide site. Photograph taken July 2010.

Objective 7

Twelve of the 5 x 20 m quadrats were sub-divided into smaller plots, with the first 5 x 5 m square divided into 1 x 1 m, 2 x 2 m, 3 x 3 m, 4 x 4 m, and 5 x 5 m subplots (Fig. 4.15). Data were recorded at successively larger subplots to determine if the results are influenced by scale. Data on clast size, soil penetrability, krummholz density, and total basal area were collected from 1 x 1 m plots up through the 5 x 20 m extent of the plot and the data from each analyzed with the within-subject one-way ANOVA.

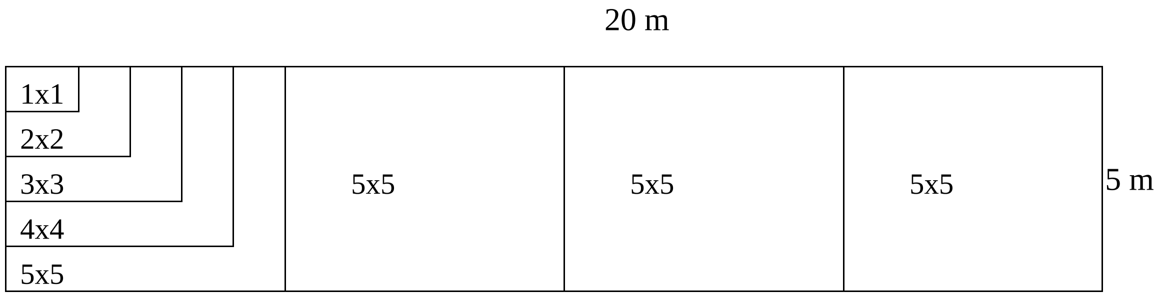

Figure 4.15. Quadrat and subplot layout in meters.

CHAPTER 5: RESULTS

The results chapter is organized into four primary sections – Section 1) site characterization and comparisons among sites, Section 2) comparisons between burned and unburned data, 3) seedling micro-site conditions, and 4) biogeomorphic interactions and methods. The first section is a characterization of each of the three sites. Soil and vegetation conditions are provided for Upper Divide, Lower Divide, and Swiftcurrent. For each site, data are presented on topographic variations and the data collected (such as ESD, soil penetrability, clast size, krummholz density, seedling species) in regard to each sub-site identified. Subsequently, comparisons are made among the three sites, and data collected from each of the topographic variations within each site are contrasted.

Section 2 compares variables in burned areas and unburned areas to answer the question regarding whether or not conditions are different after fire within the alpine treeline ecotone. This section is an aggregate of all the data for burned areas and all the data for unburned areas from the three sites. The soil conditions of duff depth, ESD, soil penetrability, clast size, soil chemistry, and soil loss are presented and compared for both burned and unburned areas. Extent of rock spalling between burned and unburned sites is compared, as are characterizations of boulder spalling, including freshly exposed rock area and distance between the boulder and rock fragments.

In Section 3, seedling micro-site conditions are evaluated. Comparisons are made between seedling and random micro-sites within burned and unburned areas, and between burned and unburned micro-site conditions. Soil conditions, such as ESD, soil penetrability, and clast size, are evaluated along with percent vegetation cover and relative sunlight for each micro-site. Particular focus is placed on the characterization and role of objects in regard to seedlings. Data are presented on the distances to the nearest three objects, seedling species in relation to distance from objects, and comparisons between burned and unburned sites, and between random and seedling sites.

The final three topics are presented in Section 4 and address interactions of vegetation and soil conditions, especially in the context of biogeomorphic methods. Vegetation and soil variables within the burned areas are evaluated to determine if significant correlations are found after fire between vegetation and soil. These results also provide indications to the soil and vegetation variables that may influence each other, and therefore, should be taken into consideration. The second topic is on soil and vegetation conditions in regard to edge effects, and how the variables change from the unburned krummholz to across a burned patch and then back into the unburned krummholz. These data will provide information on how soil conditions relate to changes in vegetation cover, as well as provide indications to the importance of where quadrats should be placed within a burned area. The last topic addresses the influence of quadrat size on the data results. Data were collected from quadrats of different sizes to evaluate whether or not quadrat size influences variable averages. Select soil and vegetation variables are contrasted to each other to determine the importance of quadrat size.

Section 1: Site Characterization

This section focuses on burned site conditions and provides data on soil variables, krummholz characteristics, and seedling micro-plots conditions. Section 1 is organized by site and data are presented for each site. Each site is further sub-divided into sub-sites, based primarily on topographic variability found within each site. Each site section begins with a topographic description and an introduction to the sub-sites, followed by data on soil variables evaluated, rock spalling, krummholz characteristics, and seedling micro-site data.

This section concludes by comparing conditions among the three sites. Clast size, soil penetrability, effective soil depth, soil chemistry, spalling, krummholz density, and krummholz DGL and basal area are statistically contrasted to determine if significant differences among the sites exist, and if so, which sites are either similar or different from each other. The number of quadrats varied among the sites because of differences in site conditions and burned pattern. Table 5.1 provides a summary of the number of quadrats per site for burned and unburned areas, the respective quadrat sizes, and the total area. The number of sampled points for each variable was dependent upon the number of quadrats per site. Table 5.2 lists the total number of sampled point for each variable at the three sites and the total for all sites combined for burned and unburned areas. Those counts do not include the micro-site counts. The number of micro-sites sampled depended upon the number of quadrats as well as the number of seedlings within the quadrats. A total of 226 micro-sites were sampled. Table 5.3 lists the total number of sampled points for each variable within the micro-sites.

Table 5.1. Summary of the size, number, and area of quadrats for burned and unburned areas for each site.

Site	**Quadrat Size**	**Number of Quadrats**	**Total Area (m^2)**
Burned			
Upper Divide			
	5 x 20 m	9	900
	5 x 5 m	11	275
Lower Divide			
	5 x 20 m	5	500
	5 x 5 m	0	0
Swiftcurrent			
	5 x 20 m	3	300
	5 x 5 m	0	0
Unburned			
Upper Divide	5 x 20 m	6	600
Lower Divide	5 x 20 m	3	300
Swiftcurrent	5 x 20 m	3	300

Table 5.2. The number of sampled points for each variable.

Burned

Variable	Upper Divide	Lower Divide	Swiftcurrent	Total
Duff Depth	470	200	120	790
Soil Penetrability	2350	400	240	2990
Clast Size	2350	400	240	2990
ESD	235	100	60	395

Unburned

Variable	Upper Divide	Lower Divide	Swiftcurrent	Total
Duff Depth	240	120	120	480
Soil Penetrability	480	240	240	960
Clast Size	480	240	240	960
ESD	120	60	60	240

Table 5.3. The total number of sampled points within the micro-sites.

Micro-site Variables	Total Count
Soil Penetrability	2,260
Clast Size	1,130
ESD	1,130
Sunlight	226
Degree Slope	226
Object Distances	678
Percent Vegetation Cover	226
Percent Rock Cover	226

Upper Divide

Topographic Description

The Upper Divide site contained the greatest topographic viability of the three sites. Upper Divide was sub-divided into five sub-sites based primarily on slope and aspect. The sub-sites, presented by their relative west to east location, were Upper Divide West (UDW) (Fig. 5.1), Upper Divide Ridge (UDR) (Fig. 5.2), Upper Divide East (UDE) (Fig. 5.3), Upper Divide Islands (UDI) (Fig. 5.4), and Upper Divide Far East (UDF) (Fig. 5.5). Each sub-site displayed distinct and different characteristics, degree slope, and/or slope aspect. UDW was located on a west-facing slope in a broad gully slightly downslope from the crest of the saddle that extends between Divide Mountain and Whitecalf Mountain. Krummholz had established within the gully, extending up from the sub-alpine forest. UDR was located on the opposite side of the saddle, immediately downslope on the eastern side of the saddle. A narrow (about 5 to 10 m in width) strip of krummholz extended along this ridge. UDE was positioned east of the ridge within a slight depression. The UDI quadrats were located on burned patches that had been tree islands. The patches were located leeward of boulders on the eastern side of the saddle, east of UDE and west of UDF. UDI was similar in degree slope and aspect to UDE, but the burned patches were distinctly different because they were smaller patches (about 5 x 5 m) of krummholz that had burned and were located leeward of boulders. UDF was the eastern most site on Upper Divide and was located on an east-aspect slope with an average degree slope of 29° (Table 5.4). The islands on Upper Divide had the lowest degree slope with an average of 4°, followed by UDE with an average of 5°.

Table 5.4. Topographic features of each sub-site at Upper Divide. Sub-sites are listed from west to east and will be presented throughout this dissertation in that manner.

Site	Aspect	Feature	°Slope
UDW	West	Slope	17
UDR	East	Ridge	14
UDE	East	Slope	5
UDI	East	Island	4
UDF	East	Slope	29

Figure 5.1. Upper Divide West (UDW) site. Photograph taken July 2012.

Figure 5.2. Upper Divide Ridge (UDR) site. Photograph taken August 2012.

Figure 5.3. Upper Divide East (UDE) site. Photograph taken August 2012.

Figure 5.4. One of the burned islands at Upper Divide (UDI). Photograph taken August 2012.

Figure 5.5. The Upper Divide Far East (UDF) site. Photograph taken August 2012.

Duff Depth

None of the burned areas on Upper Divide contained a duff layer. Some sites did have detritus scattered about, usually pieces of bark, dead needles, and/or cones. Fire apparently incinerated any organic material on the ground surface. Mineral soil was present on all burned sub-site surfaces.

Effective Soil Depth

Effective soil depth for Upper Divide averaged 7.9 cm with a standard deviation of 7.1 cm (Table 5.5). Averages for each sub-site did not vary significantly when compared to each other with the Kruskal Wallis test (p=0.352). UDW had the greatest average depth at 9.7 cm, and UDI had the lowest with an average of 6.2 cm. The greatest range of depths was found at UDR, which also contained the single greatest depth of 38.1 cm.

Table 5.5. Average effective soil depths in cm at Upper Divide sub-sites.

Site	Average	1SE	1SD	Range	Min.	Max.	n
UDW	9.74	0.976	7.621	27.6	1.4	29	60
UDR	8.56	1.198	8.964	38.1	0.8	38.9	60
UDE	6.69	0.86	5.442	17.5	1	18.5	40
UDI	6.18	1.039	5.194	21	1	22	35
UDF	6.67	0.777	4.916	19.5	1.5	21	40

Soil Penetrability

Overall soil penetrability at Upper Divide averaged 0.81±0.71 kg/cm^2. Soil penetrability values varied significantly among the sub-sites of Upper Divide (Kruskal Wallis, $p<0.001$) (Table 5.6). The Mann Whitney *U* test with a Bonferroni correction was used as a *post hoc* test to determine which sites were significantly different from each other. Significant differences were found between all pairs except UDF and UDR ($p=0.01$) (Table 5.7).

Upper Divide Far East (UDF) and UDR were both on a relatively steeper degree slope compared to most other sites and were located on an east aspect. In general, the highest penetrability values were found on lower-sloped sites and the lowest penetrability values were found at the site with the steepest average degree slope (UDF) (Fig. 5.6). I would have expected the opposite, in which steeper slopes experienced more erosion and therefore would display greater soil penetrability values. However, this trend may allude to the idea that wind erosion is more significant in the area, and the more gently sloped sites were also those that were more exposed to wind.

Table 5.6. Average soil penetrability values (kg/cm^2) for each sub-site at Upper Divide.

Site	Average	1SE	1SD	Range	Min	Max	n
UDW	0.65	0.031	0.478	2.5	0	2.5	240
UDR	0.492	0.83	0.744	4.25	0	4.25	240
UDE	1.24	0.06	0.761	3.5	0	3.5	160
UDI	1.08	0.063	0.754	3.08	0	3.08	140
UDF	0.304	0.037	0.329	1.92	0	1.92	160

Table 5.7. Comparison of soil penetrability differences of the sub-sites at Upper Divide. Averages for each sub-site were compared with the Mann Whitney *U* test and a Bonferroni correction applied. * indicates significant differences at the significance level of 0.072.

Comparison	P-value
UDW and UDE	0.001*
UDF and UDR	0.68
UDW and UDI	0.001*
UDW and UDF	0.001*
UDW and UDR	0.001*
UDE and UDF	0.001*
UDE and UDR	0.001*
UDE and UDI	0.072*
UDF and UDI	0.001*
UDR and UDI	0.001*

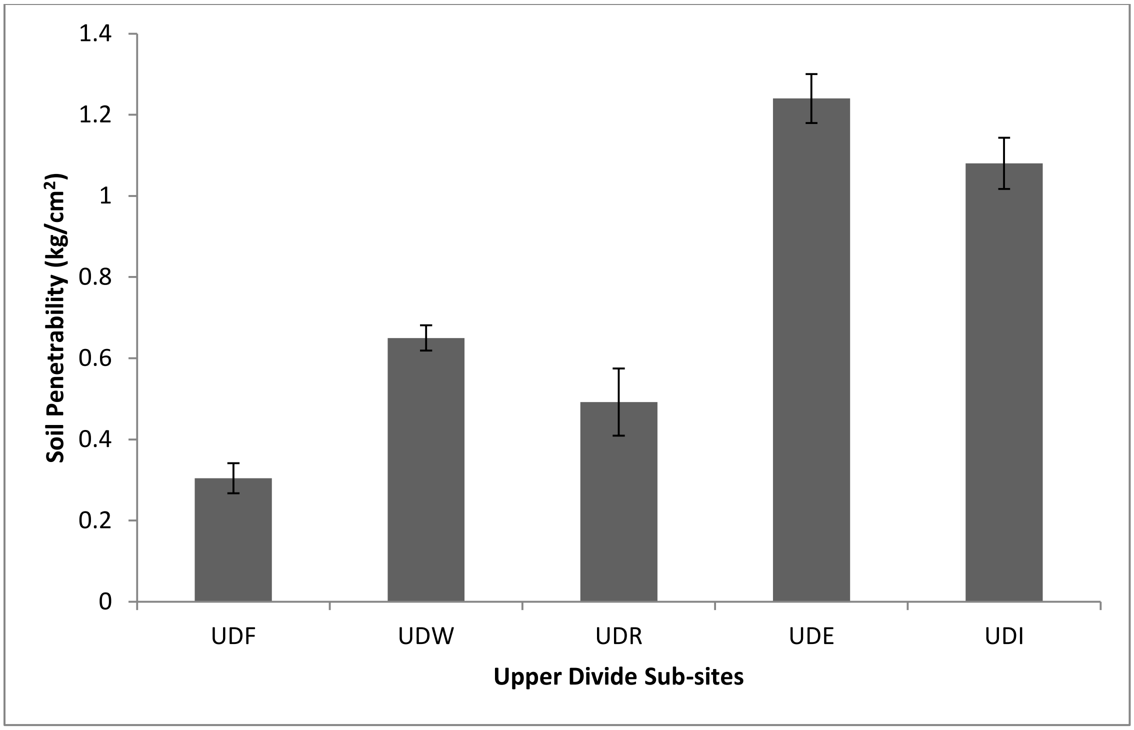

Figure 5.6. Average soil penetrability values for each sub-site at Upper Divide aligned from steepest degree slope on the left (UDF) to the lowest degree slope on the right (UDI). Error bars show ±1SE.

Clast Size

Overall clast size for Upper Divide averaged 5.0±4.4 cm (Table 5.8). Clast size average varied significantly among the five sub-sites at Upper Divide (Kruskal Wallis, p<0.001). Each sub-site was subsequently compared to each other with the Mann Whitney U test and a Bonferroni correction applied (α=0.072). All paired sites were significantly different from each other except UDW and UDI, UDW and UDF, and UDF and UDI (Table 5.9). These sites contained mid-range clast size values among the sub-site averages. Upper Divide Ridge displayed the highest clast size average at 7.6 cm and UDE the lowest with 3.4 cm. Upper Divide Ridge also displayed the greatest range of values. Differences and similarities among the sub-sites did not appear to correspond to topographic variations.

Table 5.8. Average clast sizes (cm) for each sub-site at Upper Divide.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
UDW	4.56	0.14	2.17	11.5	1.0	12.5	240
UDR	7.64	0.47	7.36	43.8	0.2	44.0	240
UDE	3.36	0.13	1.64	8.7	0.8	9.5	160
UDI	4.09	0.17	1.97	11.8	0.2	12.0	140
UDF	4.33	0.13	1.70	8.1	1.0	9.1	160

Table 5.9 Comparisons of clast size between sub-sites at Upper Divide. Analyzed with Mann Whitney *U* test. Bonferroni correction applied with a significance level of 0.072. *indicates significant differences in averages clast sizes values.

Sub-site Comparison	P-value
UDW and UDR	0.001*
UDW and UDE	0.001*
UDW and UDI	0.048
UDW and UDF	0.829
UDR and UDE	0.001*
UDR and UDI	0.001*
UDR and UDF	0.001*
UDE and UDI	0.001*
UDE and UDF	0.001*
UDI and UDF	0.085

Soil Chemistry

Average soil properties for Upper Divide are displayed in Table 5.10, along with averages for each sub-site from which soil was collected. Soil was only collected and analyzed from UDW, UDR, and UDF. Soil pH and nutrients did not vary significantly among burned Upper Divide sub-sites as assessed with the Kruskal Wallis test (α=0.05)

(Table 5.11). Variations among the three sub-sites are, however, evident. Nitrate-nitrogen and phosphorus were notably higher at UDW compared to UDR and UDE. Upper Divide West contained a covering of ash (Fig. 5.7), whereas the other two sub-sites were more of a mixture of mineral soil and, to a lesser extent, ash. Conductivity was also distinctly higher at UDW, especially compared to UDF. Upper Divide West also contained the highest values of potassium, calcium, sulfur, and sodium. Soil pH was lowest at UDR and highest at UDW. Magnesium was highest at UDF and lowest at UDR. The sample sizes are small, however, and are not intended to be conclusive.

Table 5.10. Upper Divide soil samples from burned areas.

Soil Property	pH	Conductivity (umho/cm)	NO_4 (ppm)	P (ppm)	K (ppm)	Ca (ppm)	Mg (ppm)	S (ppm)	Na (ppm)
UDW (n=3)	7.1	143	6	183	177	3757	509	15	11
UDR (n=3)	6.3	78	2	45	94	1972	322	9	6
UDF (n=1)	6.8	117	3	63	150	2590	558	9	7
Overall	7	138.1	4.4	125.7	148.6	3404.7	503.9	12.3	9.6

Table 5.11. Comparisons among UDW, UDR, and UDF (only sub-sites where soil samples were collected) at Upper Divide.

Soil Property	P-value
pH	0.565
Conductivity	0.296
N	0.443
P	0.135
K	0.208
Ca	0.208
Mg	0.721
S	0.319
Na	0.160

Figure 5.7. A layer of ash covered much of the ground at the UDW sub-site.

Spalling

Of the total 258 spalled boulders found within the burned quadrats (Fig. 5.8), 226 or about 88% were located within the Upper Divide site. Upper Divide accounted for about 59% of the total sampled area and about 70% of the sampled area on Divide Mountain. A total of 433 spalled rock fragments were found at Upper Divide, and the average distance they traveled was 14.5 cm. The amount of newly exposed rock surface area averaged 175.7 cm^2 per rock and the amount of rock material fractured off of the spalled boulders averaged 352.5 cm^3 per rock (n=226). Within the Upper Divide site, most of the spalled rocks were found within the UDW subsite, which contained 32% of the 226 spalled rocks, followed closely by UDE with 30%, UDF with 16%, UDR with 15%, and UDI with 6%. UDW and UDE were both located closer to Divide Mountain Peak, from which the boulders were originating, than UDI and UDF. UDE was also close to the Peak, but angled upward from the base of the Peak, and was therefore, not conducive to the accumulation of fallen rock.

Figure 5.8. Spalled boulder at UDE. Photograph taken in August 2012.

Seedlings

Upper Divide West was predominantly *Abies lasiocarpa* (subalpine fir, 29) (subsequently referred to as “fir”), followed by *Pinus albicaulis* (Whitebark pine, 5) (subsequently referred to as “pine”), and *Picea engelmannii* (Engelmann spruce, 3) (subsequently referred to as “spruce”). Subalpine fir seeds are wind dispersed and usually travel within about 30 m from a seed source. Spruce seeds are also wind dispersed but can travel about 91 m from a source. Spruce prefer mineral soils to duff/organic material because the mineral soil will better retain moisture whereas the organic matter will often dry out too fast for slow growing spruce roots. Upper Divide Far East only had pine (13), and UDR had spruce (5) and pine (3) but no fir. One pine was found in UDI, and UDE did not contain any seedlings.

Conditions associated with seedlings at Upper Divide are displayed in Table 5.12. Percent herbaceous vegetation cover was overall rather low with an average of 5.9% for all seedlings found at Upper Divide. The sub-site UDR had the highest average vegetation cover with 16.8%. Average rock cover was much greater at 40.1% overall. The one seedling at UDI was found with a 60% rock cover. The lowest average percent rock cover was 24% with the UDF seedlings. The comparatively low percent herbaceous vegetation cover and high percent rock cover may be because the vegetation can compete with the seedlings for nutrients and moisture, whereas rocks can facilitate conditions helpful to seedling establishment and growth, such as moderating soil temperature and increasing soil moisture.

The overall distance to the average of the three closest objects was 34.3 cm. The averages per sub-site ranged from 28.2 cm at UDW to 60.3 cm at UDR. The differences among the sub-sites are likely related to the amount of objects available at each sub-site. The closest object distance averaged 15.5 cm for Upper Divide overall. The distance to the closest object at UDW averaged 10.9 cm and the seedling at UDI was located 9.0 cm from the closest object. The greatest average distance between the closest object and seedlings was 35.5 cm at UDR.

Degree slope averaged 8.8° for Upper Divide seedling sites and ranged from 6.8° at UDW to 16° for UDF. In general, seedlings were found in association with lower degree slope than the overall sub-site average slope. The average sub-site slope for UDF was 29° compared to the seedling average of 16°. This trend was not the case for the UDI seedling, however, which had a micro-site slope of 10° compared to the average sub-site slope of 4°.

Average clast size for seedling micro-sites at Upper Divide was 1.7 cm. This value was lower than the average clast size of 5.0 cm for Upper Divide overall. Rock clasts have been found to facilitate seedling establishment by acting to capture seedlings, retain soil moisture, and regulate soil temperatures (Pérez 2009a), however, seedlings at Upper Divide are associated with areas of smaller average clast sizes. This finding indicates that rock clasts are not important to seedling establishment. Boulders, other objects, and rock ground cover provide similar services to seedlings and were found to be strongly associated with seedlings, possibly abating the facilitative role that clasts can offer. In support of this idea, the smallest average clasts were found at UDW, which averaged the closest distance between an object and a seedling, and the largest at UDR, at which the greatest average distance was found between seedlings and the closest objects.

Average soil penetrability for seedling micro-sites was 0.26 kg/cm^2, which was much lower than the overall average for Upper Divide of 0.81 kg/cm^2. Sub-sites ranged from the average of 0.17 kg/cm^2 for UDF to 0.55 kg/cm^2 for the seedling at UDI. This trend of the lowest soil penetrability found on the steepest average slope and the highest penetrability value found on the lowest average slope for seedling micro-site values is the same as that of the overall findings for Upper Divide.

Effective soil depth for all seedling micro-sites at Upper Divide averaged 9.3 cm, and ranged from 5.8 at UDI to an average of 10.4 cm at UDR. Upper Divide Islands also had the highest soil penetrability value, indicating the micro-site conditions at this sub-site have more compact, shallower soils (based upon the one seedling micro-site, however).

Table 5.12. Seedling micro-site conditions for each sub-site and combined data (overall) for Upper Divide. *Soil penetrability is abbreviated as Pen.

UDW	Average	1SE	1SD	Range	Min.	Max.	Micro-site n
% Vegetation cover	4.4	1.3	7.9	40	0	40	37
% Rock	45.5	4.9	29.9	95	0	95	37
Avg. distance (cm)	28.2	3.5	21.5	101.3	3	104.3	37
Closest dist. (cm)	10.9	2.6	15.7	80	0	80	37
°Slope	6.8	1.3	7.6	30	0	30	37
Clast size (cm)	1.2	0.24	1.4	5.32	0	5.32	37
Pen.* (kg/cm^2)	0.26	0.05	0.28	1.08	0	1.98	37
ESD (cm)	9.7	0.8	4.6	19.6	4.3	23.9	37
UDR							
% Vegetation cover	16.8	8.8	24.8	72	3	75	8
% Rock	38.1	8.8	24.8	70	0	70	8
Avg. distance (cm)	60.3	13.0	36.9	124	15.7	139.7	8
Closest dist. (cm)	35.5	11.5	32.6	103	6	109	8
°Slope	8.4	3.2	8.5	20	0	8.4	8
Clast size (cm)	3.2	0.67	1.9	5.9	0.8	6.7	8
Pen. (kg/cm^2)	0.38	0.10	0.28	0.87	0.18	1.05	8
ESD (cm)	10.4	2.2	6.3	19.2	6.12	25.3	8
UDI							
% Vegetation cover	3						1
% Rock	60						1
Avg. distance (cm)	43						1
Closest dist. (cm)	9						1
°Slope	10						1
Clast size (cm)	1.72						1
Pen.* (kg/cm^2)	0.55						1
ESD (cm)	5.8						1
UDF							
% Vegetation cover	3.5	1.4	4.4	15	0	15	13
% Rock	24	9.0	28.5	75	0	75	13
Avg. distance (cm)	34.9	5.0	15.9	43.3	20	63.3	13
Closest dist. (cm)	17.2	5.2	16.5	45	1	46	13
°Slope	16	2.8	8.8	20	5	25	13
Clast size (cm)	2.14	0.3	0.94	2.74	1.06	3.8	13

Table 5.12-Continued. Seedling micro-site conditions for each sub-site and combined data (overall) for Upper Divide. *Soil penetrability is abbreviated as Pen.

Pen.* (kg/cm^2)	0.168	0.074	0.02	0.23	0.05	0.28	13
ESD (cm)	7.3	1	3.2	9.6	3.4	13	13
Overall							
% Vegetation cover	5.9	1.6	11.9	75	0	75	59
% Rock	40.1	3.9	29.5	95	0	95	59
Avg. distance (cm)	34.3	3.4	25.3	136.7	3	139.7	59
Closest dist. (cm)	15.5	2.7	20.4	109	0	109	59
°Slope	8.8	1.1	8.5	30	0	30	59
Clast size (cm)	1.67	0.21	1.58	6.69	0	6.69	59
Pen.* (kg/cm^2)	0.26	0.03	0.26	1.08	0	1.08	59
ESD (cm)	9.3	0.63	4.7	21.9	3.4	25.3	59

Krummholz

A total of 533 krummholz was found at Upper Divide. Diameter at ground level (DGL) averaged 15.8 cm and basal area averaged 414.5 cm^2 per krummholz measured (Table 5.13).

Table 5.13 Krummholz diameter at ground level (DGL) (cm) and basal area (cm^2) at Upper Divide (n=533).

	Average	1SE	1SD	Range
DGL	15.8	0.69	16	125.7
Basal Area	414.5	49	1131.2	12719.1

Lower Divide

Topographic Description

The Lower Divide site was located on the north aspect of Divide Mountain, about 100 m lower in elevation compared to Upper Divide. The Lower Divide site was more homogeneous regarding topographic variability compared to Upper Divide. Only two sub-sites were identified for Lower Divide, and these were similar in both degree slope and slope aspect. The difference was their respective settings within vegetation – Lower Divide North (LDN) (Fig. 5.9) was a burned patch of krummholz that was situated within a larger patch of krummholz that had not burned. The burned patch extended downslope, into the subalpine area. The sub-site contained little topographic variation. The other identified sub-site – Lower Divide Gully (LDG) (Fig. 5.10) – was comprised of two gullies that were lined with burned krummholz within an area of low-lying tundra vegetation. The gullies had apparently served as a protective depression for the establishment of krummholz and then fire extended up from the subalpine forest into the treeline ecotone via krummholz within the gullies.

Figure 5.9. Lower Divide North site. Photograph taken August 2012.

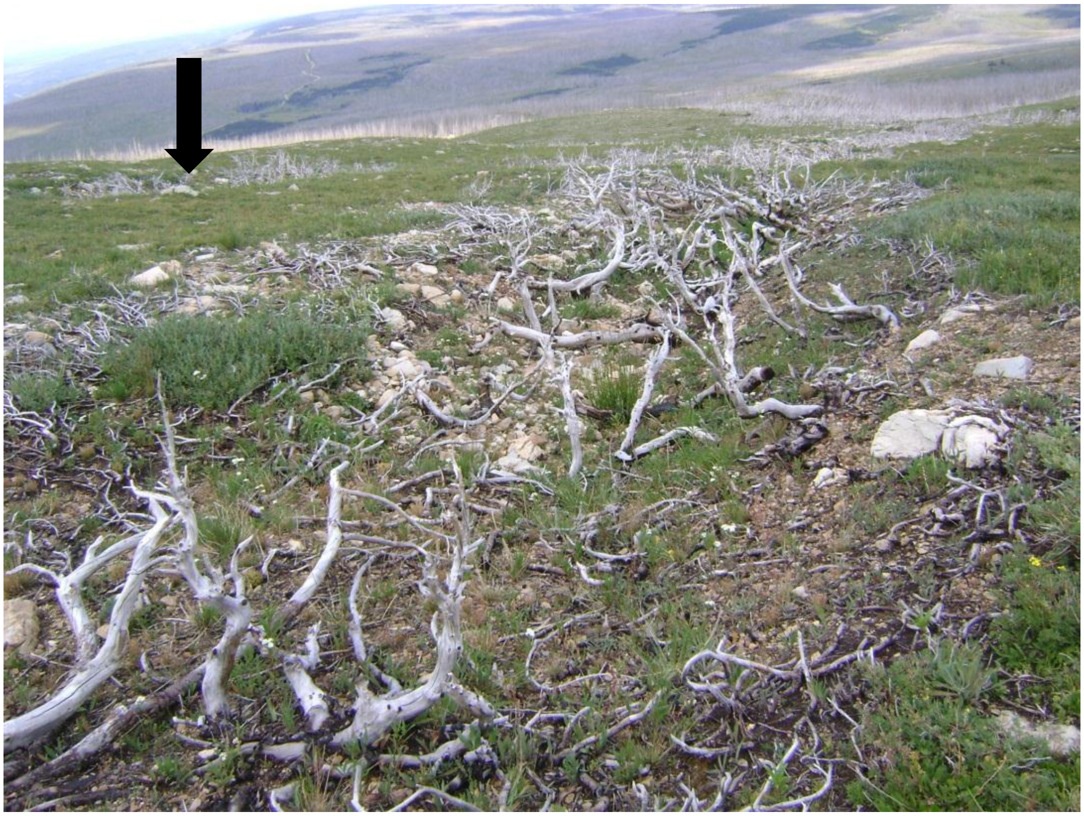

Figure 5.10. One of the gullies in the foreground for the LDG sub-site. The second gully is farther away and slightly downslope in the background of the photograph (indicated by the arrow). Photograph taken August 2012.

Duff Depth

As with Upper Divide, no duff layer was found within the burned area at Lower Divide. A few locations within the quadrats contained some dead needles, and these locations were often near the burned edge where dead needles were sometime still found on the charred upstanding krummholz.

<u>Effective Soil Depth</u>

Effective soil depth averaged 6.0±3.8 cm (n=100) for Lower Divide. The two sub-sites did not vary significantly (p=0.518) (Table 5.14) when analyzed with the Mann Whitney U test. Lower Divide North had a larger range of values, but the sample size was also larger than that of LDG.

Table 5.14. Average ESD (cm) for Lower Divide and Lower Divide Gully.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
LDN	6.1	0.5	4.2	21.3	0.8	22.1	120
LDG	5.8	0.4	2.7	12.5	1.5	14	80

<u>Soil Penetrability</u>

Average soil penetrability values of LDN and LDG did not vary significantly (Mann Whiney U test, p=0.313). However, LDG averaged a lower value of 0.63 kg/cm^2 compared to 0.71 kg/cm^2 of LDN and the maximum value was found at LDN (Table 5.15). Overall average for Lower Divide was 0.67 kg/cm^2.

Table 5.15 Average soil penetrability (kg/cm^2) for LDN and LDG.

Site	Average (kg/cm^2)	1SE	1SD	Range	Min.	Max.	n
LDN	0.71	0.07	0.59	3.24	0.01	3.25	240
LDG	0.63	0.04	0.50	2.17	0.00	2.17	160

Clast Size

Average clast sizes of LDN and LDG were the same at 3.0 cm (Table 5.16). The range of values for LDN was much greater than that for LDG, but LDN had a larger sample size.

Table 5.16. Average clast size (cm) for LDN and LDG.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
LDN	3.0	0.18	2.8	29.9	0.3	30.2	240
LDG	3.0	0.13	1.6	9.3	0.7	10.0	160

Soil Chemistry

No significant differences were found in soil properties between LDN and LDG (Tables 5.17 and 5.18). The sample sizes are small, and only one soil sample was collected and analyzed from LDG. Soil pH averaged 6.6, and was slightly higher for LDG. Conductivity averaged 105 umho/cm for LDN and was 127 umho/cm for LDG. Nitrogen and P were higher in LDN, and K, Ca, Mg, and Na were lower in LDN compared to LDG. Sulfur levels were the same in both sub-sites.

Table 5.17. Soil properties for LDN and LDG.

Soil Property	pH	Cond. (umho/ cm).	N (ppm)	P (ppm)	K (ppm)	Ca (ppm)	Mg (ppm)	S (ppm)	Na (ppm)
LDN (n=3)	6.6	105	2	78	89	2330	351	9	6
LDG (n=1)	7.2	127	1	45	95	2936	528	9	7
Overall	6.6	105	2	78	89	2330	351	9	6

Table 5.18. Significance levels between LDN (n=3) and LDG (n=1).

Soil Property	P-value
ph	0.18
Conductivity	0.655
N	0.346
P	0.655
K	0.655
Ca	0.18
Mg	0.18
S	1.0
Na	0.346

Spalling

Of the total 258 boulders found to have spalled within the sampled area, 31, or 12%, were found at the Lower Divide sites. Lower Divide accounted for 30% of the Divide Mountain sampled area or 25% of the total sampled area. Forty-two percent of the total found within the Lower Divide site were at the LDN sub-site, and 58% were found within the LDG sub-site. LDG may have contained a greater percentage of spalled rock because the heat from the fire may have burned more intensely within the gullies. The amount of newly exposed rock surface area averaged 176.3 cm^2 per rock and the amount of rock material fractured off of the spalled boulders averaged 352.6 cm^3 per rock within Lower Divide.

Seedlings

A total of five seedlings were found in the burned area at Lower Divide. One was located within krummholz roots at LDN, and the other four were in LDG. Those in LDG were all pine, and the one in LDN was a fir.

Seedling site conditions were grouped together for LDN and LDG because few seedlings were found at each sub-site (Table 5.19). Percent vegetation cover was greater than percent rock cover, which was quite low at 0.4%. The average distance to the three closest objects was 98.2 cm and the distance between the closest object and a seedling was 34 cm. Slope averaged 25°.

Clast size averaged 2.3 cm for the seedling micro-sites, and this value was slightly lower than the overall site average of 3.0 cm. Similarly, soil penetrability was lower for the seedling micro-sites compared to overall soil penetrability for Lower Divide with an average value 0.47 kg/cm^2. Average seedling micro-site ESD was slightly higher than the overall Lower Divide site average of 6.0 cm.

Table 5.19 Seedling site conditions at Lower Divide.

Variable	Average	1SE	1SD	Range	Min.	Max.	Micro-site n
% Vegetation cover	9	1	2.2	5	5	10	5
% Rock	0.4	0.4	0.9	2	0	2	5
Avg. distance (cm)	98.2	2.1	4.8	10.7	89.67	100.3	5
Closest dist. (cm)	34	4	8.9	20	30	50	5
°Slope	25	5	11.2	25	5	30	5
Clast size (cm)	2.32	0.12	0.268	0.6	1.84	2.44	5
Pen. (kg/cm^2)	0.47	0.05	0.1	0.23	0.43	0.65	5
ESD (cm)	7.1	0	0	0	7.1	7.1	5

Krummholz

A total of 351 krummholz was measured at Lower Divide (Fig. 5.11). Average DGL was 9.3 cm and average basal area was 102.7 cm^2 per krummholz (Table 5.20).

Figure 5.11. Burned krummholz at Lower Divide. Photograph taken July 2011.

Table 5.20. Krummholz DGL (cm) and basal area (cm^2) for Lower Divide (n=351).

	Average	1SE	1SD	Range
DGL	9.3	0.4	6.7	72.3
Basal Area	102.7	15.6	292.2	4126.1

Swiftcurrent

Topographic Description

The Swiftcurrent site was located on a south-aspect slope on Swiftcurrent Mountain. Average degree slope for this site was 29.3° and ranged from 28 to 30°. Swiftcurrent did not have the topographic variability found on Divide Mountain, but was rather a relatively uniform, steep slope (Fig. 5.12). Three quadrats were positioned, however, over a greater elevation range then those within the Upper Divide or Lower Divide sites, and the results of Swiftcurrent are compared among the three elevational locations of the three quadrats.

Duff Depth

Swiftcurrent did not contain any duff layer in the sampled burned areas. Much of the ground surface was covered in gravel- and pebble-sized rock fragments and loose sediments and ash.

Effective Soil Depth

Effective soil depth averaged 9.3±4.7 cm (n=60) for Swiftcurrent. The three sub-sites varied significantly ($p<0.045$) when analyzed with the Kruskal Wallis test and a significance level 0.05. However, assessing the three sub-sites in comparison to each other with the Mann Whiney *U* test and applying a Bonferroni correction ($\alpha=0.0167$), none of the sub-sites were significantly different from the others (Upper and Middle, $p<0.023$; Upper and Lower, $p<0.045$; Lower and Middle, $p<0.808$). The Upper sub-site averaged a higher ESD value of 11.4 cm compared to the Middle and Lower sub-sites

Figure 5.12. Swiftcurrent site. Photograph taken August 2012.

(Table 5.21), identified by the Kruskal Wallis test. The reduction in statistical power, however, resulting from the Bonferroni adjustment precluded identification of group differences without adjustment, the Upper site ESD is statically thicker than either the Lower or Middle sites.

Table 5.21. Effective soil depth (cm) for each sub-site at Swiftcurrent.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
Upper	11.4	1.1	5.1	16.5	4.0	20.5	20
Middle	8.1	0.9	4.2	12.4	3.6	16	20
Lower	8.5	1.0	4.3	16.6	3.0	19.6	20

Soil Penetrability

Soil penetrability averages ranged from 1.06 kg/cm^2 for Upper sub-site to 0.11 kg/cm^2 for Middle, and Lower was between those two extremes at 0.74 kg/cm^2 (Table 5.22) and averaged 0.61±0.55 kg/cm^2. These differences were significant among the three sub-sites (Kruskal Wallis, p=0.001). The *post hoc* test using the Mann Whitney U test with a Bonferroni correction (α=0.0167) was applied to determine which sub-sites were different or similar among the three. The results indicated that all sub-sites in comparison to each other were significantly different (Table 5.23) at the 0.0167 level.

Table 5.22. Soil penetrability (kg/cm^2) of Swiftcurrent sites.

Site	Average (kg/cm^2)	1SE	1SD	Range	Min	Max	n
Upper	1.06	0.05	0.43	1.92	0.25	2.17	80
Middle	0.11	0.02	0.20	0.92	0.00	0.92	80
Lower	0.74	0.07	0.41	1.58	0.00	1.58	80

Table 5.23. *Post hoc* comparisons of soil penetrability among the three sub-sites.

Comparison	P-value
Upper and Middle	0.001
Upper and Lower	0.001
Lower and Middle	0.001

Clast Size

Average clast size ranged from 3.07 cm at Upper (Fig. 5.13) and Middle (Fig. 5.14) to 2.46 cm at Lower (Fig. 5.15) (Table 5.24). Clast size averages for the three sub-sites varied significantly (Kruskal Wallis, $p<0.001$). A *post hoc* using the Mann Whitney *U* test with a Bonferroni correction ($\alpha=0.0167$) was applied to determine if pair-wise differences among the three sites existed and if so, which sub-site(s) differed from which other(s) (Table 5.25). The results revealed that the Upper and Middle sub-sites differed significantly from the Lower sub-site, which had the lowest average clast size (Table 5).

Table 5.24. Clast size averages (cm) for each sub-site at Swiftcurrent.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
Upper	3.07	0.18	1.65	7.8	1.3	9.1	80
Middle	3.06	0.14	1.21	6.8	1.2	8.0	80
Lower	2.46	0.16	1.42	7.8	0.4	8.2	80

Table 5.25. *Post hoc* significance levels between paired sub-sites determined with the Mann Whiney *U* test and a Bonferroni correction applied. * indicates significant differences in averages at the significance level of 0.0167.

Comparison	P-value
Upper and Middle	0.329
Upper and Lower	0.013*
Middle and Lower	0.001*

Figure 5.13. Clasts on the ground surface at the Upper sub-site.

Figure 5.14. Clasts on the ground surface at the Middle sub-site.

Figure 5.15. Smaller clasts were found at the Lower sub-site in comparison to the Middle and Upper sub-sites.

Soil Chemistry

Variability was not found in soil properties among the three sub-sites at Swiftcurrent when analyzed with the Kruskal Wallis test (p<0.368). The overall average pH was 5.3. The lowest value was found at Upper and the highest at Middle (Table 5.26). Conductivity, phosphorus, calcium, and magnesium also showed the same trend with the lowest value found at the Upper sub-site and highest at the Middle sub-site. Nitrogen, sulfur, and sodium however, were slightly higher at the Upper sub-site. Potassium was highest at Lower, and lowest at Middle.

Table 5.26. Soil properties for the sub-sites at Swiftcurrent and the overall average for the Swiftcurrent site.

Soil Property	pH	Cond-uctivity (umho/cm)	N (ppm)	P (ppm)	K (ppm)	Ca (ppm)	Mg (ppm)	S (ppm)	Na (ppm)
Upper	4.8	57	7	68	92	623	98	11	9
Middle	5.8	65	5	110	88	2628	286	7	7
Lower	5.4	59	5	77	97	1539	151	6	5
Average	5.3	60	6	85	92	1597	178	8	7

Spalling

No rock spalling was found at Swiftcurrent. Swiftcurrent was covered mostly in a layer of gravel and pebbles, and no boulders were located within the sampled quadrats. A few boulders were observed on the slope, but they showed no indications of spalling.

Seedlings

A total of four seedlings were found at Swiftcurrent, and they were all firs located in the Lower sub-site. Their site conditions are displayed in Table 5.27. Average percent vegetation cover was higher than percent rock cover. Swiftcurrent has had more time, compared to Divide Mountain, for herbaceous vegetation to become re-established, and this was evident with the greater percentage of ground covered in vegetation. Similar to the lack of boulders present at Swiftcurrent, very little ground was covered in boulders or had exposed bedrock within the seedling micro-sites.

Distance between the average of the three closest objects and seedlings was 93.6 cm and averaged 15.2 cm for the distance between the closest object and a seedling. Most of the objects at Swiftcurrent were the base of burned, standing krummholz.

Average slope of micro-sites was 23°, and all were between 20 and 25°. Clast size averaged 1.5 cm, which was lower than the overall averages for each sub-site, and ranged from 0.96 to 2.42 cm. Soil penetrability averaged 0.47 kg/cm2, which fell within the range of values found for the sub-sites. Average micro-site ESD was slightly higher at 12.6 cm when compared to the overall ESD value 9.3 cm for Swiftcurrent.

Table 5.27. Seedling micro-site conditions at Swiftcurrent.

Variable	Average	1SE	1SD	Range	Min.	Max.	Micro-site n
% Vegetation cover	23.3	10.1	17.6	35	5	40	4
% Rock	1.7	1.7	2.9	5	0	5	4
Avg. distance (cm)	93.6	15.2	26.3	52.5	66.7	119.2	4
Closest object (cm)	15.2	5.1	8.8	17.5	6.5	24	4
°Slope	23	1.7	2.9	5	20	25	4
Clast size (cm)	1.5	0.45	0.79	1.46	0.96	2.42	4
Penetrability (kg/cm^2)	0.47	0.15	0.26	0.53	0.2	0.73	4
ESD (cm)	12.6	3.3	5.7	11.3	6.7	18	4

Krummholz

A total of 244 krummholz was recorded at Swiftcurrent. Diameter at ground level averaged 8.1 cm and basal area averaged 65.9 cm^2 for krummholz within the three sub-sites combined (Table 5.28). Many of the krummholz at Swiftcurrent were still upright, but some had fallen over (Fig 5.16).

Table 5.28. Krummholz DGL (cm) and basal area (cm^2) at Swiftcurrent (n=244).

	Average	1SE	1SD	Range
DGL	8.1	0.28	4.3	28.1
Basal Area	65.9	5.2	81.8	703.7

Figure 5.16. Burned krummholz at Swiftcurrent. Photograph taken August 2012.

Comparisons among Sites

The following sections present data previously described for each site, but compares the findings of each site. Comparisons are made among the sites with statistical analyses to determine if sites are significantly different from each other in regard to the measured variables.

Effective Soil Depth

Average ESD values were slightly higher at SC (9.3±4.7 cm, n=60) compared to UD (7.9±6.1 cm, n=235) and LD (6.0±3.8 cm, n=100) (Table 5.29), and the averages were significantly different (Kruskal Wallis test, $p<0.001$). A *post hoc* test was used to determine which sites were significantly different from each other (Mann Whitney U test, Bonferroni correction, $\alpha=0.0167$). Swiftcurrent differed significantly from UD and LD ($p<0.001$, $p<0.001$, respectively), but UD and LD were not significantly different ($p<0.288$).

Table 5.29. Average ESD (cm) for each site.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
UD	7.9	0.48	7.08	38.1	0.8	38.9	235
LD	6.0	0.33	3.81	21.3	0.8	22.1	100
SC	9.3	0.61	4.70	17.5	3	20.5	60

Soil Penetrability

Soil penetrability values among the sites were significantly different (Kruskal Wallis, $p<0.006$). A *post hoc* test (Mann Whitney with Bonferroni correction, $\alpha=0.0167$) revealed that UD and SC differed significantly ($p<0.002$), but not UD and LD ($p<0.103$) and SC and LD ($p<0.224$). UD averaged the highest at 0.81 kg/cm^2, followed by LD at 0.66 kg/cm^2 and SC at 0.61 kg/cm^2 (Table 5.30).

Table 5.30. Average soil penetrability values (kg/cm^2) for each site.

Site	Average	1SE	1SD	Range	Min.	Max	n
UD	0.81	0.03	0.71	4.25	0	4.25	2350
LD	0.66	0.03	0.53	3.25	0	3.25	400
SC	0.61	0.04	0.55	2.17	0	2.17	240

Clast Sizes

Clast sizes differed significantly among the sites (Kruskal Wallis, $p<0.001$). A *post hoc* test (Mann Whitney with Bonferroni correction, $\alpha=0.0167$) revealed that UD differed from both LD ($p<0.001$) and SC ($p<0.001$), but LD and SC did not differ significantly from each other ($p<0.107$). Upper Divide had a higher average clast size with 5.03±4.4 cm, compared to LD and SC, which both had an average of 3.0 cm (3.0±2.4, 3.0±1.5 cm, respectively) (Table 5.31). This result may be attributed to rockfall from the Lewis Overthrust on Divide Mountain.

Table 5.31. Average clast size (cm) for each site.

Site	Average (cm)	1SE	1SD	Range	Min.	Max.	n
UD	5.03	0.14	4.36	43.8	0.2	44	2350
LD	2.97	0.11	2.43	29.9	0.3	30.2	400
SC	2.86	0.09	1.46	8.7	0.4	9.1	240

Soil Chemistry

Table 5.32 presents average soil property values for each sites. Soil properties did not vary significantly among the sites as analyzed with the Kruskal Wallis test (Table 5.33), although slight variations are present. Average soil pH, conductivity, calcium, magnesium, and sulfur were highest at UD and lowest at SC. Nitrogen was the highest at SC, followed by UD and then LD. Upper Divide averaged the highest phosphorus level, and LD had the lowest. Potassium was similar between LD and SC, and higher at UD. Sodium was also greatest at UD, followed by SC and LD.

Table 5.32. Average soil properties for each of the three sites.

Soil Property	UD (n=7)	LD (n=4)	SC (n=3)
pH	7.0	6.8	5.3
Conductivity (umho)	138	110	60
Nitrate-N (ppm)	4	2	6
Phosphorus (ppm)s	126	70	85
Potassium (ppm)	149	91	92
Calcium (ppm)	3405	2482	1597
Magnesium (ppm)	504	396	178
Sulfur (ppm)	12	9	8
Sodium (ppm)	10	6	7

Table 5.33. Significance values as a result of the soil properties compared among the sites with the Kruskal Wallis test (α=0.05).

Soil Property	P-value
pH	0.119
Conductivity	0.053
N	0.194
P	0.198
K	0.074
Ca	0.07
Mg	0.133
S	0.376
Na	0.112

Spalling

Density of spalling roughly corresponded to the percentage of ground covered in boulders (Table 5.34). The highest density of spalled rock was found at UDE, followed by UDW. These two sub-sites also contained the highest percentages of rock cover. Likewise, LD and LDG contained the least amount of boulder cover and the least density of spalled rocks. Swiftcurrent did not contain any spalled rock. No boulders were found within the sampled area at Swiftcurrent, however, some of the spalled rocks at Divide Mountain were small (<5 x 5 cm) and Swiftcurrent did contain smaller rocks.

Table 5.34. Spalling density (rock/m^2) per sub-site at Upper Divide and Lower Divide.

Site	Percent rock	Number of spalled rocks	Area (m^2)	Density
UDR	30	35	300	0.117
UDE	65	68	200	0.34
UDI	15	13	175	0.074
UDF	20	37	200	0.185
LDN	8	13	300	0.043
LDG	5	18	200	0.09

Krummholz

Krummholz density was greatest at SC and lowest at UD (Table 5.35). A total of 496 krummholz were measured at UD, followed by 366 at LD and 244 at SC. Krummholz DGL and basal area varied significantly among the three sites (Kruskal Wallis, $p<0.001$ for both DGL and basal area). A *post hoc* test using the Mann Whitney *U* test with a Bonferroni correction was then applied to determine which pairs differed significantly from each other at the significance level of 0.0167. Each site when compared to another was significantly different in both DGL and basal area (Table 5.36).

Table 5.35. Krummholz counts and density per site.

Site	Krummholz Count	Total area (m^2)	Density (krummholz/m^2)
UD	496	1175	0.42
LD	366	500	0.73
SC	244	300	0.81

Table 5.36. Significance levels when comparing krummholz DGL (cm) and basal area (BA) (cm^2). * indicates significance at the 0.0167 level.

Sites	Variable	P-value
LD and SC	DGL	0.006*
	BA	0.006*
UD and SC	DGL	0.001*
	BA	0.001*
UD and LD	DGL	0.001*
	BA	0.001*

Burn severity did not vary notably between LD and UD – both averaged a severity rank of 2.1, and SC was similar at 2.2 (1=severely burned, 5=slightly burned, as indicated by krummholz charring). Relative sunlight in all sites was mostly Full Sun, as determined from random micro-site conditions (Fig 5.17). Sixty-seven percent of random seedling sites were in Full Sun at SC (n=12) compared to 71% at UD (n= 47) and 80% at LD (n=20).

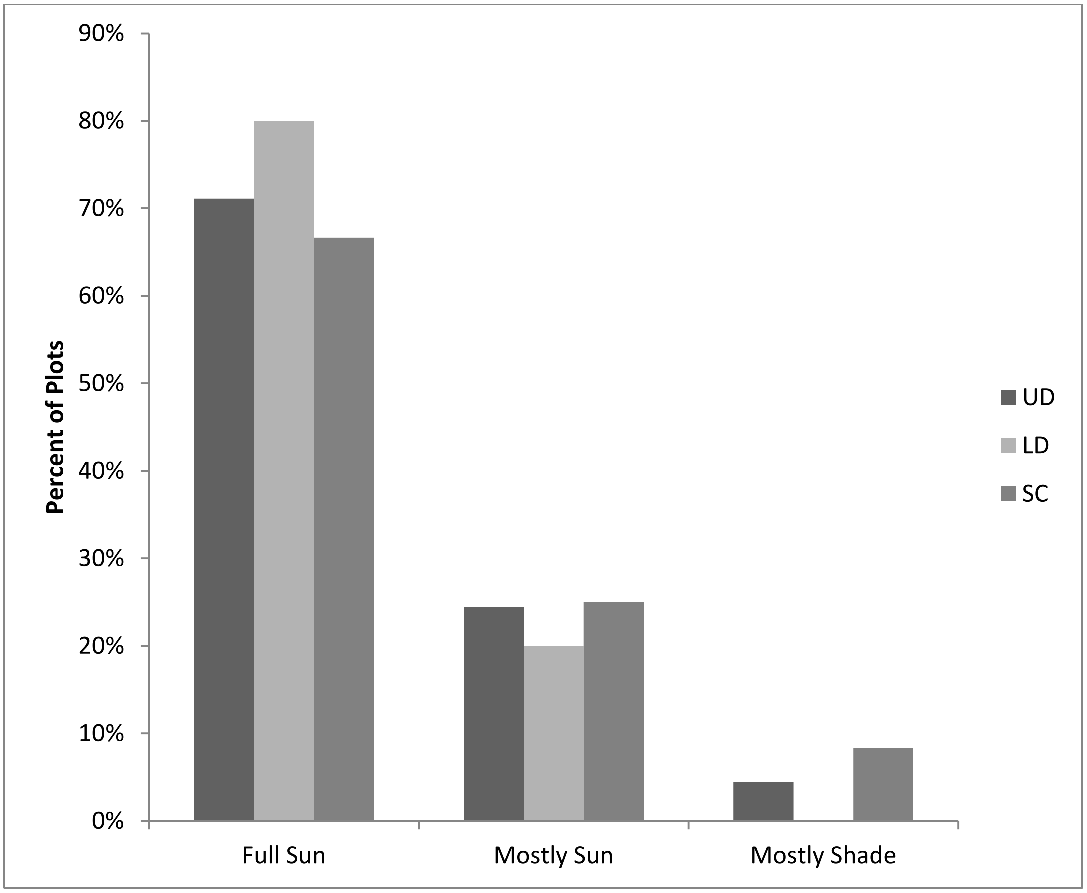

Figure 5.17. Percentage of UD, LD, and SC random micro-plots (burned) in either Full Sun, Mostly Sun, or Mostly Shade. No micro-plots in burned areas were Full Shade.

Section 2: Comparisons between Burned and Unburned Conditions

This section focuses on comparing burned areas to unburned areas. Variables assessed include duff depth, clast size, soil penetrability, effective soil depth, soil loss, and boulder spalling. Data are aggregated from the sites to make overall, general comparisons in attempt to answer the question – what influence does fire have on fine-scale factors within the alpine treeline ecotone? To address this question, data are required on unburned site conditions, and because data were not acquired pre-fire, adjacent unburned areas are used as proxies for conditions before fire. Great emphasis was placed on selecting unburned sampling areas that were most similar, especially in regard to proximity to the burned sampled area, slope aspect, degree slope, and vegetation cover type (krummholz rather than tundra vegetation) to the burned sampling area. This section, therefore, provides data on unburned site conditions, in addition to burned conditions and evaluates their potential similarities and differences.

All the data were evaluated for normality with the Kolmogorov-Smirnov test. High kurtosis and skewness values indicated that the data were skewed. A transformation (natural log) was applied to the data, but they remained not normally distributed. The data did not meet the assumptions for the student's *t* test, and therefore, the nonparametric equivalent - the Mann Whitney *U* test - was used in comparing two independent groups. In cases of more than two groups, the non-parametric equivalent of one-way ANOVA, the Kruskal Wallis test, was used. *Post hoc* tests were performed, when necessary, by comparing pairs of variables with the Mann Whitney *U* test with a Bonferroni correction.

Duff Depth

One of the most obvious differences between the ground surface of burned and unburned areas was the lack of krummholz needles in burned areas (Figures 5.18 and 5.19). My sub-hypothesis that burned areas would contain significantly less duff/organic matter was supported by the data. A layer of needles in various stages of decay was found up to 23 cm thick within the unburned areas compared to 0 cm in burned areas. In unburned areas, the duff layer ranged from 2 cm to 23 cm with an average depth of 9.4 cm (n=480) (Table 5.37). The data were normally distributed (Kolmogorov-Smirnov, 0.022, Skewness, 0.727, Kurtosis, 0.257), but the unburned duff depths were all 0 (n=790). The Mann Whitney *U* test was used to statistically determine differences in means between burned and unburned areas and was found to vary significantly (p<0.001).

Figure 5.18. Ground surface in burned area on Divide Mountain. Photograph taken July 2010.

Figure 5.19. Ground surface in burned area on Divide Mountain. Photograph taken August 2012.

Generally, a layer of needles was found on the ground surface, overlaying a layer of partially decaying needles that ranged from broken needles to a dense mat of organic material. Underlying the mat, a mixture of fine organic matter and sediments were often found. This duff layer was not present in any of the burned sites. A few pieces of detritus, primarily charred bark and dead needles, were found in a few burned areas, but these pieces were few and scattered. Ash was, however, found in many of the burned quadrats. Some areas contained an almost complete covering of ash. Most other sites contained at least some ash mixed in with sediments on the ground surface.

Effective Soil Depth

Contrary to what I had hypothesized regarding ESD, it did not vary significantly between burned and unburned areas when compared using the Mann Whitney *U* test (Table 5.37). Burned ESD values were not normally distributed (Kolmogorov-Smirnov 0.001, skewness, 1.725; kurtosis, 4.596) (n=350). Unburned ESD values were also not normally distributed (Kolmogorov-Smirnov 0.001; skewness, 1.306; kurtosis, 1.256) (n=240). With a loss of duff in burned areas, I would have expected ESD to be significantly lower. However, the average ESD was slightly higher in unburned areas compared to burned areas.

Table 5.37. Average soil condition values with standard deviation. Comparisons between burned and unburned averages were made with Mann Whitney *U* test and *p* values are reported.

	Effective soil depth (cm)	Penetrability (kg/cm^2)	Duff depth (cm)	Avg. clast size (cm)
Burned	7.5±5.6	0.71±0.62	0	3.5±2.3
Unburned	13.1±5.1	0.74±0.25	9.8±4.4	1±1.2
P-value	0.396	0.001	0.001	0.001

Soil Penetrability

Overall soil penetrability averages between burned and unburned sites were compared with the Mann Whitney *U* test and found to be significantly different (p<0.001) with penetrability values lower on average in burned areas (n=2,990) (Table 5.37) compared to unburned areas (n=960). Data were tested for normality and found to significantly deviate from a normal distribution (Kolmogorov-Smirnov, p<0.001) and were positively skewed (skewness, 1.183, kurtosis, 1.609). Higher penetrability values indicate a ground surface that has low compaction, and likewise, lower values indicate a surface of higher penetrability. Burned areas displayed much more variations in penetrability compared to unburned areas. Whereas most unburned ground surfaces were covered in a layer of needles (Fig. 5.20), the burned areas ranged from an accumulation of loose sediments and ash (resulting in low penetrability values) (Fig. 5.21) to exposed compact soil (high penetrability values) (Fig. 5.22).

Figure 5.20. The ground surface of unburned areas usually contained a layer of needles. Photograph taken at Upper Divide, August 2012.

Figure 5.21. Loose sediments cover the ground surface in a burned area, providing an area of high penetrability (low values). Upper Divide in July 2011.

Figure 5.22. An area lacking loose sediments, resulting in low penetrability (high values). Photograph taken at Upper Divide in August 2012.

Clast Size

Average clast size differed significantly between burned and unburned areas as evaluated with the Mann Whitney *U* tests. The data were not normally distributed (Kolmogorov-Smirnov, 0.001; skewness, 1.555; kurtosis, 3.148). Overall, burned sites averaged 3.5 cm (n=2,990) compared to unburned 1.1 cm (n=960) (Table 5.37). Upper Divide varied significantly from LD and SC. Unburned areas among the three sites varied significantly ($p<0.001$, UD and LD, UD and SC, SC and LD) (Table 5.38). Comparisons between unburned and burned areas for each site varied significantly (Fig. 5.23). Clast sizes were significantly larger in burned areas ($p<0.001$ for each site) (Fig. 5.24).

Larger particles were observed beneath the surface in unburned areas, but the loss of soil and duff in burned areas from incineration and erosion exposed the particles to the ground surface. Also, boulders spalled within burned areas and contributed ground surface rock particles (Fig. 5.25).

Table 5.38. Clast size (cm) for unburned sites.

Site	mean	SE	SD	range	min	max	n
UD	1.0205	0.061	1.214	6.4	0	6.4	480
LD	0.3782	0.045	0.565	2.4	0	2.4	240
SC	1.5122	0.064	1.052	6.1	0.4	6.5	240

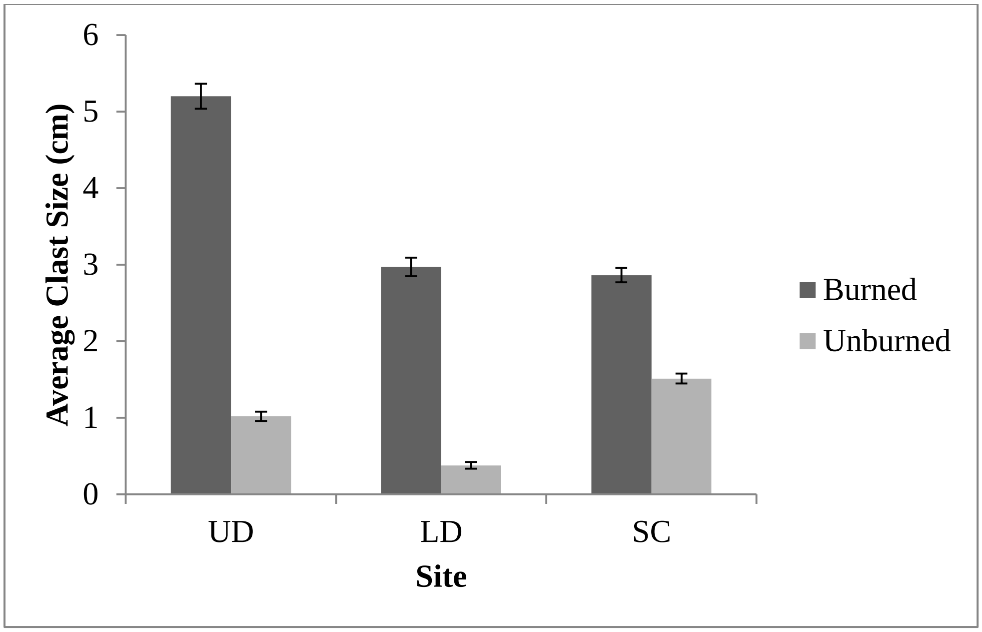

Figure 5.23. Average clast size (cm) for burned and unburned areas at each site. Error bars show ±1SE.

Figure 5.24. Surface differences between unburned ground surface (left side of photograph) and area that burned (right side of photograph).

Figure 5.25. Boulder spalling contributed to rock particles on the ground surface. Photograph taken at Upper Divide, July 2012.

Soil Chemistry

Soil pH, conductivity, and nutrients were compared between burned and unburned sites with the Mann Whitney *U* test because the sample size was small (n=18) (Figures 5.26-5.28). Average nitrogen and phosphorus were significantly (N, $p<0.040$; P, $p<0.006$) higher in burned samples compared to unburned samples. Conductivity, pH, S, Na, Ca, K, and Mg did not vary significantly between burned and unburned sites ($\alpha=0.05$). Phosphorus, which is often a limiting nutrient in the alpine treeline ecotone, was about four times greater in burned areas compared to unburned areas.

Soil pH was slightly lower in unburned areas on Divide Mountain compared to burned soil. Soil pH, however, was higher in unburned areas on Swiftcurrent compared to the burned soil. Calcium was also notably higher in burned sites, but this difference was not statistically significantly.

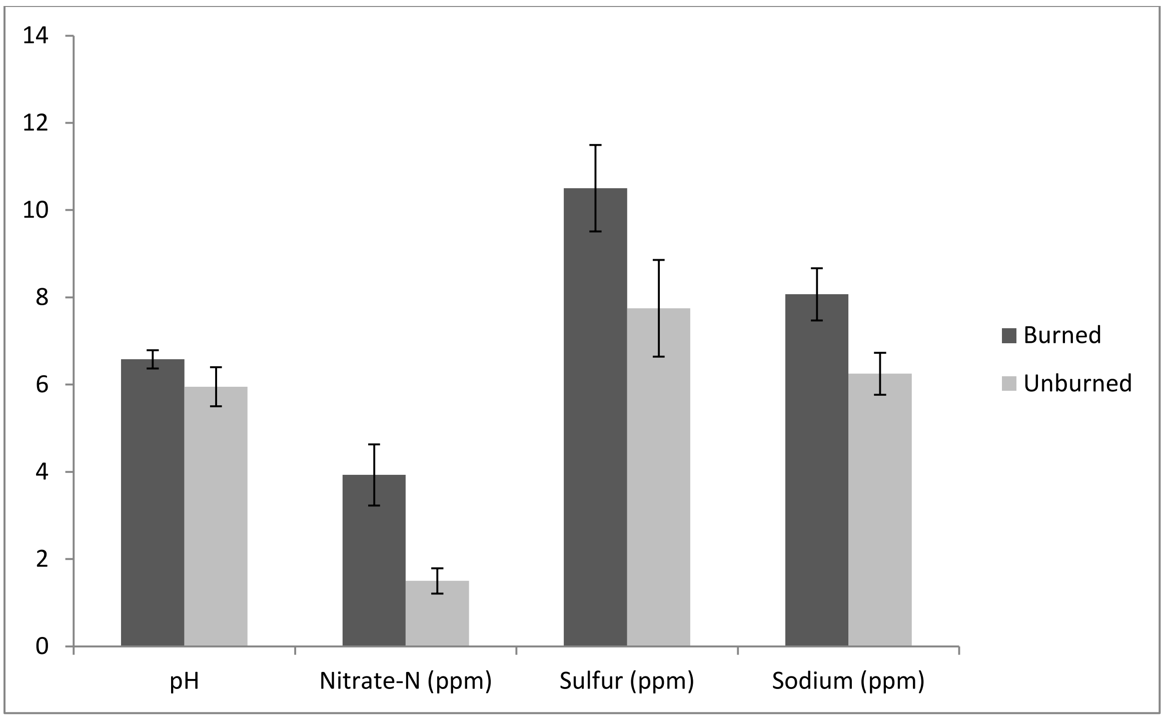

Figure 5.26. Comparison of soil pH, N, S, and Na between burned and unburned sites. Bars show average values for each property for all combined burned sites (n=12) and all combined unburned sites (n=6). Error bars show ±1SE.

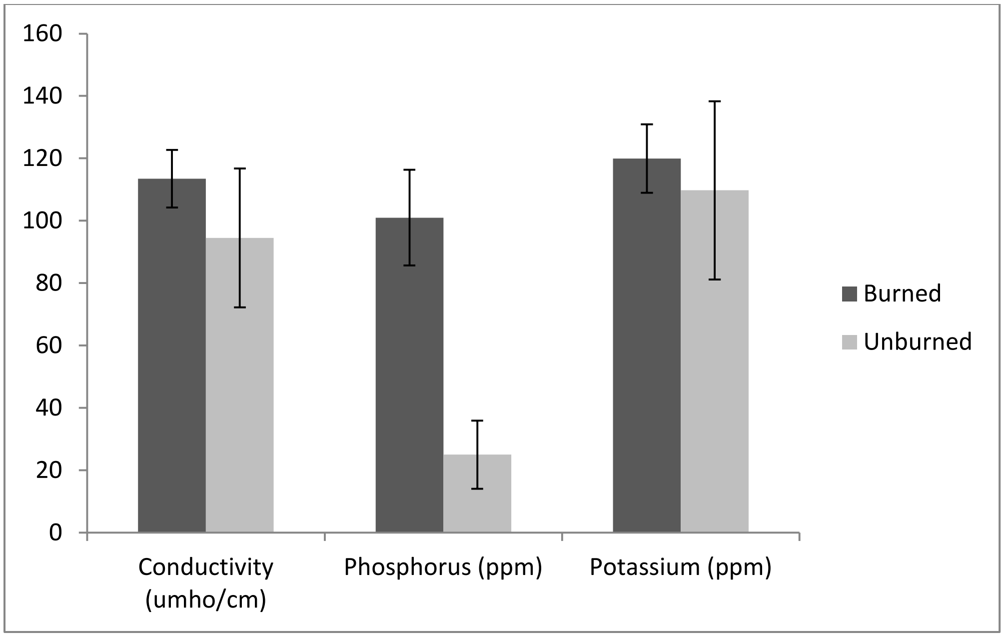

Figure 5.27. Comparison of soil conductivity, P, and K between burned and unburned sites. Bars show average values for each property for all combined burned sites (n=12) and all combined unburned sites (n=6). Error bars show ±1SE.

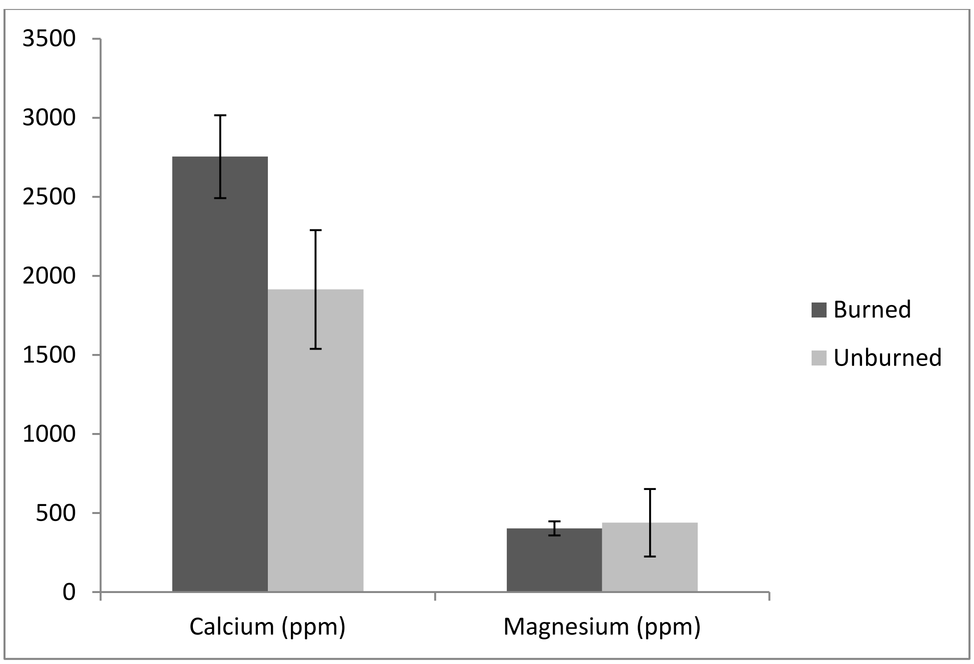

Figure 5.28. Comparison of soil Ca and Mg between burned and unburned sites. Bars show average values for each property for all combined burned sites (n=12) and all combined unburned sites (n=6). Error bars show ±1SE.

Soil Loss

Evidence of erosion was found in every sampled burned area on Divide Mountain. Indications of erosion were noted by exposed krummholz roots, exposed gravel, the presence of soil pedestals, and differences in the ground surface upslope and downslope of roots (Fig. 5.29). Soil pedestals and differences in soil surface elevations were used as proxies to estimate the depth of soil loss (Fig. 5.30). These proxies were measured whenever available within a quadrat. The average soil loss depth was 5.4 cm (n=150). The amount of area that may have experienced this extent of soil loss was determined from a classification of area that experienced high or complete vegetation loss as evident on a NAIP image. Area was measured by applying a PCA change detection (Fig. 5.31). Supervised and unsupervised classifications were attempted with between 5 and 50 classes designated, but did not work. Results of supervised and unsupervised classifications were compared to a natural color NAIP image, and the classifications were incorrectly classifying burned and unburned krummholz. PCA was able to identify areas that had been covered in vegetation before the fire, but were bare or contained minimal vegetation after the fire. An estimated area of complete mortality of the vegetation was 284 ha (2,840,000 m^2) within the designated area (Fig. 5.32).

Figure 5.29. Exposed krummholz roots. Divide Mountain, August 2012.

Figure 5.30. Photograph of ground surface in burned area that experienced soil loss. Divide Mountain, July 2011.

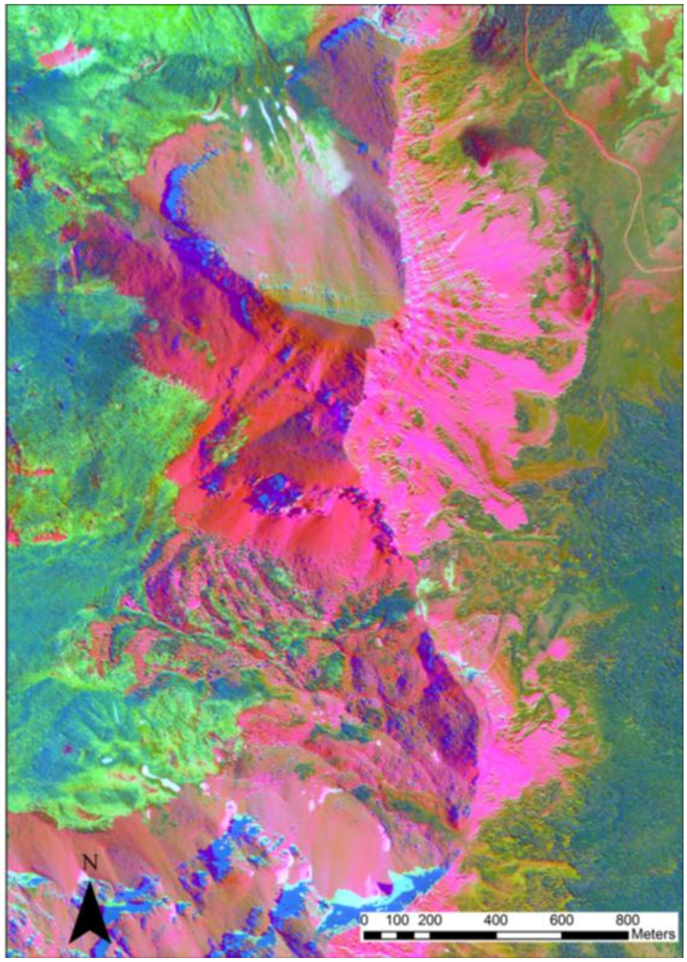

Figure 5.31. Results of the PCA. Bright green areas show features of greatest change and pink shows areas of least change.

Figure 5.32. A 2009 NAIP, natural color image of Divide Mountain.

Boulder Spalling

A total of 258 boulders were found to have spalled within the sampled 1675 m^2 burned area on Divide Mountain (Figures 5.33, 5.34, and 5.35). I found a total of 483 particles that had spalled, and clasts were found from in-place to 84 cm away from where it fractured off the rock. Many particles were not found, as evident from the amount of newly exposed area on the boulders. Newly exposed area of rock was estimated to total 16031.7 cm^2 (1.603 m^2), and averaged 13.8 cm^2 (0.00138 m^2) per boulder. This amount varied greatly (ranged from 1.6 cm^2 to 2,398.3 cm^2) as some rocks just contained one spalled area whereas others were heavily spalled. The estimated total fracturing of rock material was 72402.8 cm^2 (0.0724 m^3), with an average of 280.6 cm^3 (0.000280 m^3). Most of the rocks fragments were not shaped as a square, and therefore these numbers are rough estimates of actual spalling.

Newly exposed area of rock was evident on the limestone boulders by its grey coloring, compared to the exposed white surface of the rock. A general observation was made that four and five years after fire (2010, 2011), the grey color was visually distinct. Six years after fire (2012) however, the grey color was beginning to fade and become whiter. The distinction between spalled area and the unspalled area was, however, still easily apparent.

Rocks were also found to have spalled or fractured within the subalpine forest (Figures 5.36 and 5.37). This finding was just a casual observation of a few rocks in an area that had also burned in the Red Eagle Fire in 2006.

Figure 5.33. Spalled rock at Upper Divide. Photograph taken July 2012.

Figure 5.34. Spalled boulder at Upper Divide. Photograph taken July 2012.

Figure 5.35. A burned krummholz branch adjacent to a spalled boulder.

Figure 5.36. A rock located out of the study area within the burned subalpine forest. A fracture was found that split the rock in two major pieces.
Photograph taken in July 2011.

Figure 5.37. A spalled rock within the burned subalpine forest. Photograph taken July 2011.

Section 3: Seedling Micro-site Conditions

Section 3 presents data on micro-site (0.25 x 0.25 m) conditions. This section is organized in the following manner: conditions are compared between burned and unburned micro-sites, characterization and comparisons are made between seedling and random micro-sites in burned areas, and the role of facilitative objects is evaluated. Data on soil conditions, relative sunlight, seedling species and counts, and object characterizations are provided.

Comparison between Burned and Unburned Micro-sites

Soil

Comparisons between random micro-sites in burned areas and unburned areas were performed to evaluate the influence that fire has on site conditions at the scale of individual seedlings. The Mann Whitney *U* test was used to compare conditions between burned and unburned sites because none of the variables were normally distributed. Particle size and ESD were significantly different between unburned and burned sites for random micro-plots. Average particle size within burned areas was significantly higher and ESD significantly lower than those in unburned areas when comparing data collected from random micro-plots (Table 5.39). Soil penetrability, however, was not significantly different between burned and unburned random plots, although penetrability values averaged slightly lower in burned areas.

Table 5.39. Burned versus unburned variables of random micro-plots.

		Burned	Unburned
Particle Size (cm)	average	2.5	0.45
	1SE	0.27	0.12
	1SD	2.4	0.86
	n	77	48
	P-value	0.001	
Penetrability (kg/cm^2)	average	0.79	0.82
	1SE	0.07	0.04
	1SD	0.63	0.3
	n	77	48
	P-value	0.128	
ESD (cm)	average	7.5	9.2
	1SE	0.54	0.59
	1SD	4.7	4.1
	n	77	48
	P-value	0.001	

Sunlight

Solar radiation is a factor in seedling establishment and growth. Intense solar radiation is often found in the alpine treeline ecotone because of the high elevation. Bollinger (1973) found solar radiation to be a significant cause of seedling mortality in areas that had burned at treeline. As would be expected in areas that experienced complete mortality of woody vegetation, the percentage of micro-sites in sun in burned areas was much greater than those in unburned areas (Figures 5.38 – 5.41). Only 8% of unburned, random micro-plots were in full sun compared to 72% of random micro-plots in burned areas. Full shade went from 42% in unburned plots to 0% in burned areas. Combined, full shade and mostly shade comprised 75% of the unburned random micro-plots, compared to 4% in burned random plots.

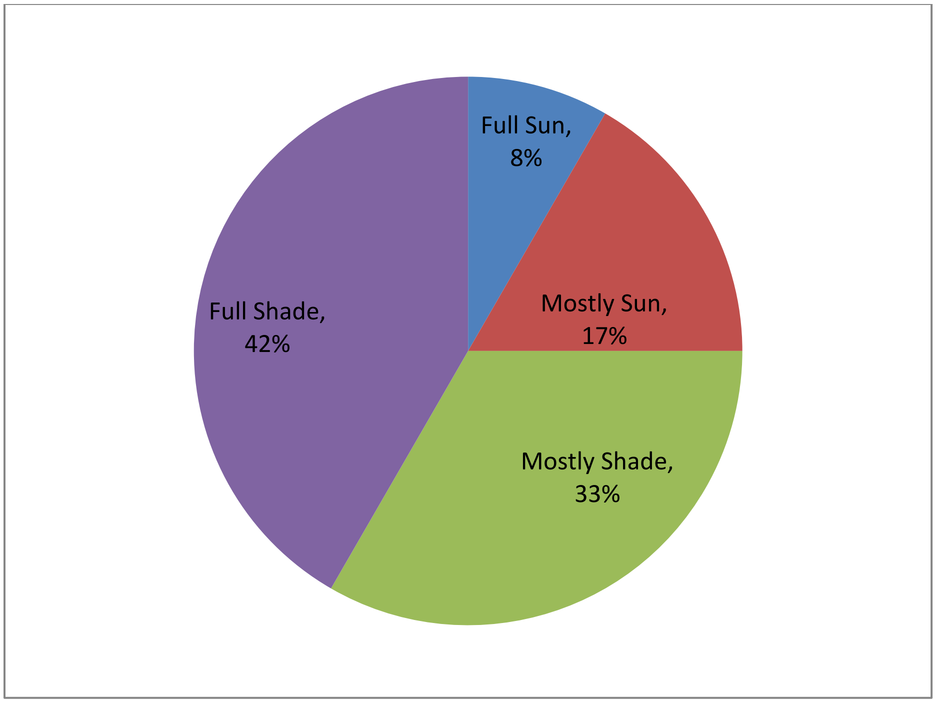

Figure 5.38. Average relative noon sunlight in unburned random micro-plots (n=48).

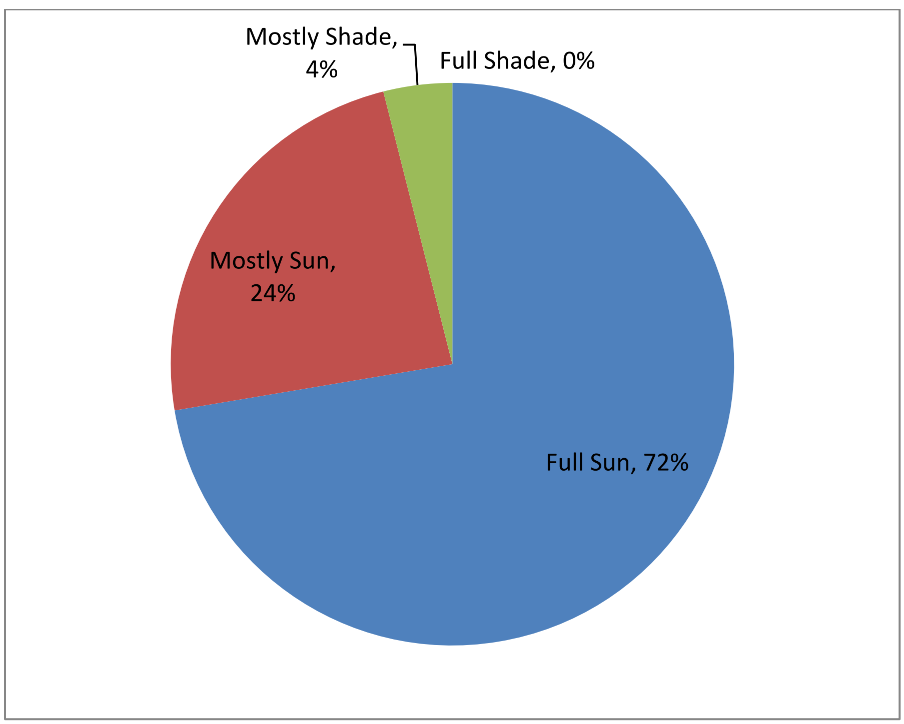

Figure 5.39. Average relative noon sunlight in burned random micro-plots (n=77).

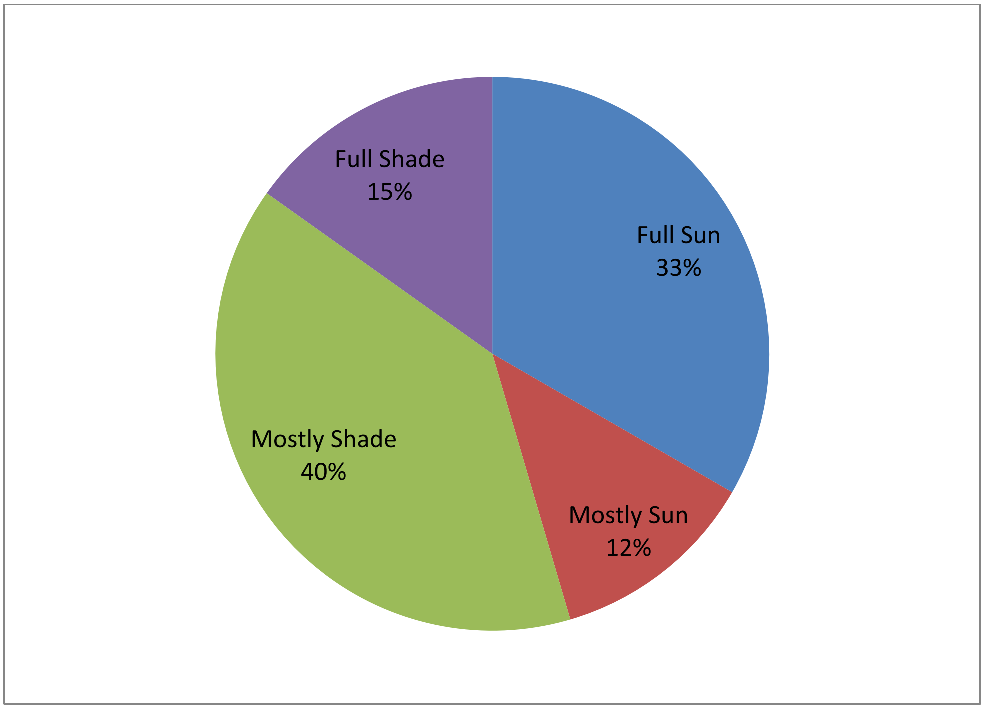

Figure 5.40. Average relative noon sunlight in unburned seedling micro-plots (n=33).

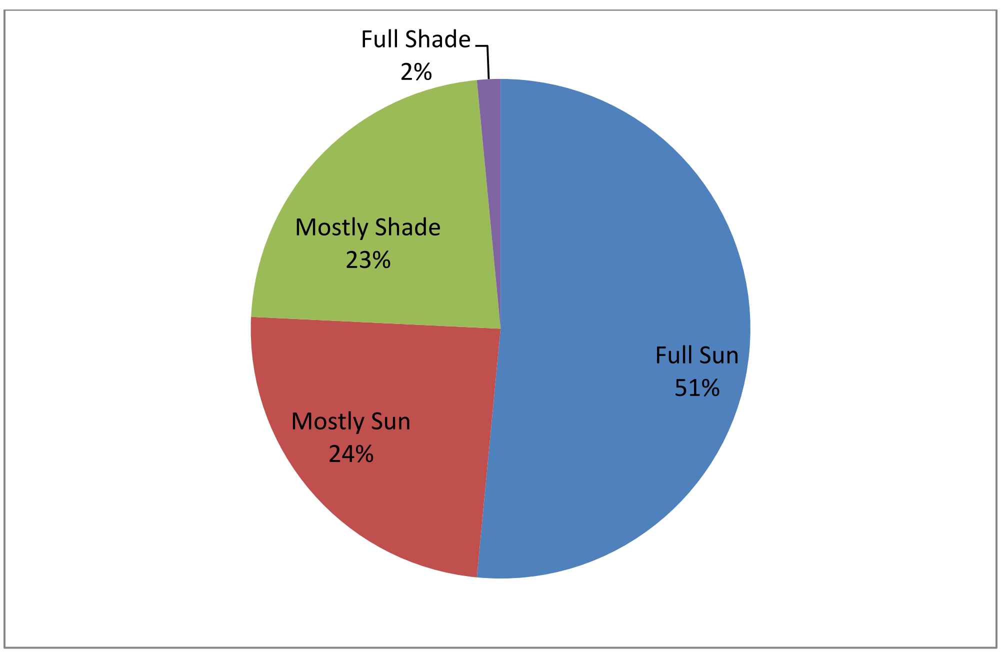

Figure 5.41. Average relative noon sunlight in burned seedling micro-plots (n=68).

<u>Ground Cover</u>

Percent of ground cover in herbaceous vegetation and rock changed between burned and unburned random micro-plots. The average percent of vegetation cover in micro-plots of unburned sites was 24% compared to 8% in burned plots (Fig. 5.42). Seedling plots showed similar percentages with unburned at 27% and burned at 7% (Fig. 5.43). Conversely, percent rock cover averaged 10% in burned random plots compared to 1% in unburned random plots. However, unburned seedling micro-plots averaged 11% rock cover, and 36% in burned seedling plots. The pattern of the burned sites containing more rock and less vegetation than unburned plots is found in both random and seedling plots. A notable difference is that for both unburned and burned seedling plots, percent rock cover is greater than in random plots.

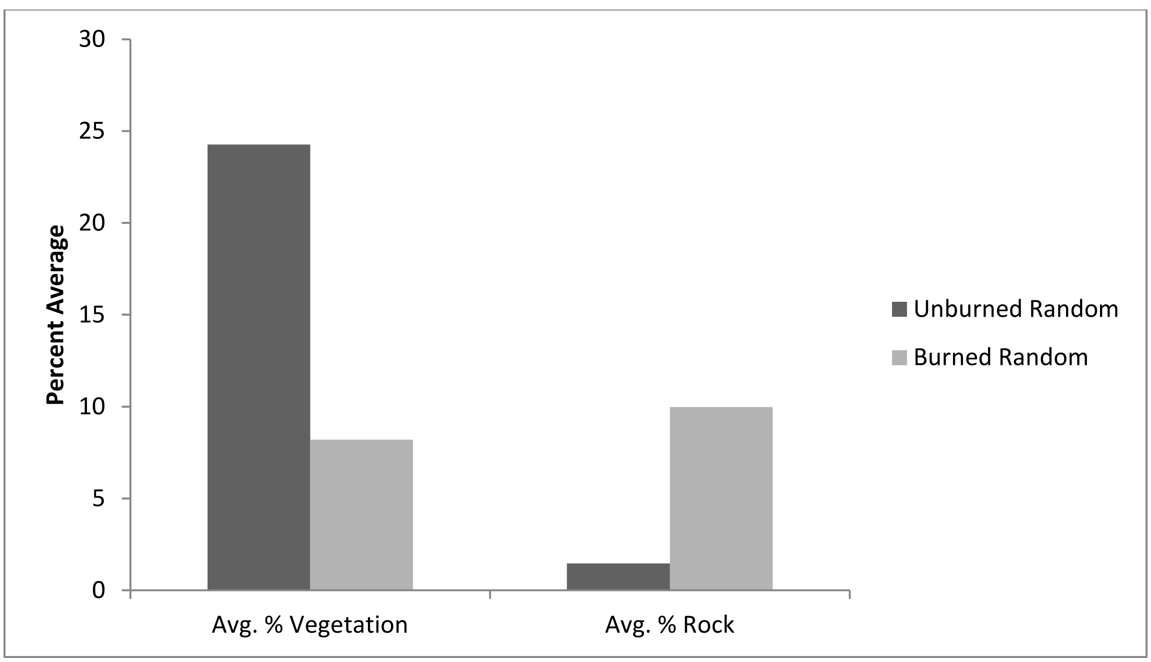

Figure 5.42. Average percent vegetation cover and rock cover in burned and unburned random micro-plots.

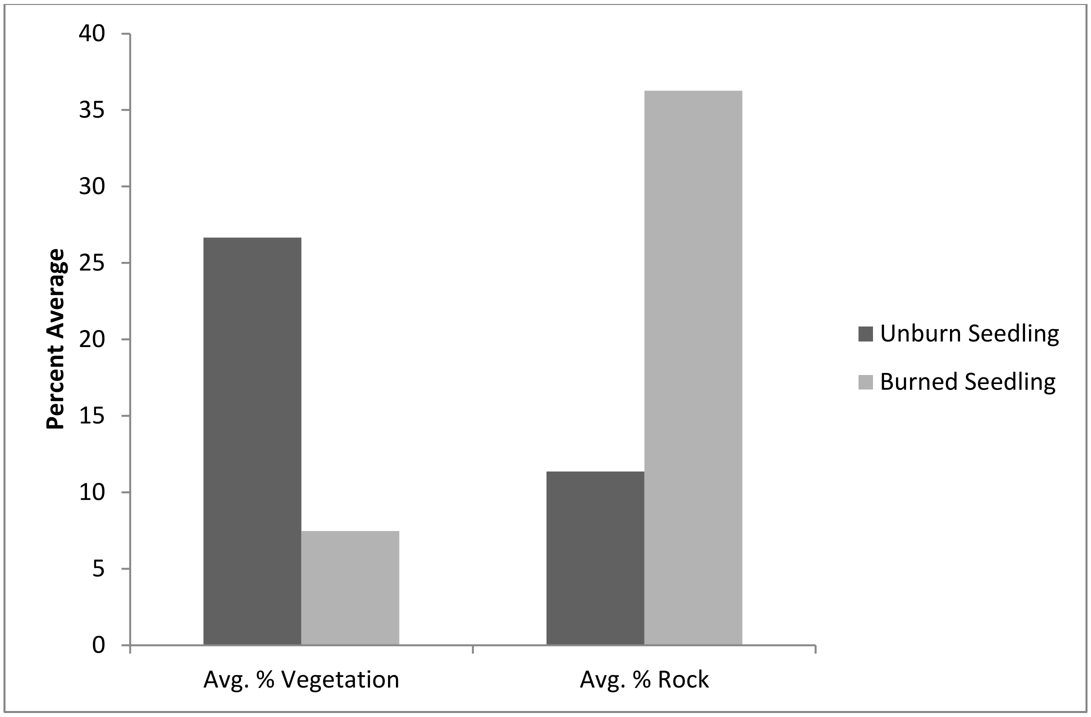

Figure 5.43. Average percent vegetation cover and rock cover in burned and unburned seedling micro-plots.

Comparisons between Random and Seedling Micro-sites

<u>Soil</u>

Soil penetrability was significantly lower in seedling plots compared to random plots, and average ESD was greater for seedling plots (Table 5.40). Soil penetrability and ESD were not significantly correlated for random plots ($p<0.163$) or seedling plots ($p<0.243$), as indicated by two-tailed Kendall's tau b. Average particle size was smaller for seedling plots, but the difference was not significant at the 0.05 level. Among the eight sites, neither random plots nor seedling plots differed significantly in regard to particle size, compaction, or ESD (comparing random to random plots and seedling to seedling plots).

Table 5.40. Soil variables averages and significance of seedling and random micro-plots.

Variable	Seedling (n=68)	1SE	1SD	Random (n=77)	1SE	1SD	P-value
Closest Distance (cm)	18.0	2.9	21.1	47.4	4.6	51.9	0.001
Avg. Distance (cm)	41.8	4.3	30.9	79.5	5.8	65.9	0.001
Particle Size (cm)	1.83	0.21	1.53	2.5	0.27	2.36	0.087
Penetrability (kg/cm^2)	0.31	0.04	0.27	0.79	0.07	0.63	0.001
ESD (cm)	9.6	0.7	4.8	7.6	0.5	4.7	0.002

Sunlight

Comparing relative sun exposure between random micro-plots and seedling micro-plots, seedlings were found in Mostly Shade 23% of the time compared to 4% of random plots in Mostly Shade. Seedlings were found in Full Sun only 52% of time compared to 72% of random micro-plots. One seedling was even found in Full Shade because it had established beneath an overhang of a boulder.

Percent vegetation cover was similar for random and seedling plots in burned areas, with averages of 8% and 7% respectively. Percent rock cover, however was much greater for seedling plots compared to random plots, with values of 36% and 10%, respectively (Fig. 5.44). This finding probably relates to the previously stated result that seedlings were significantly closer to objects such as boulders, and these boulders were sometimes close enough to be within the plot. Also, rocks were sometimes mostly submerged below the ground surface but were exposed and therefore, contributed to part of the ground cover.

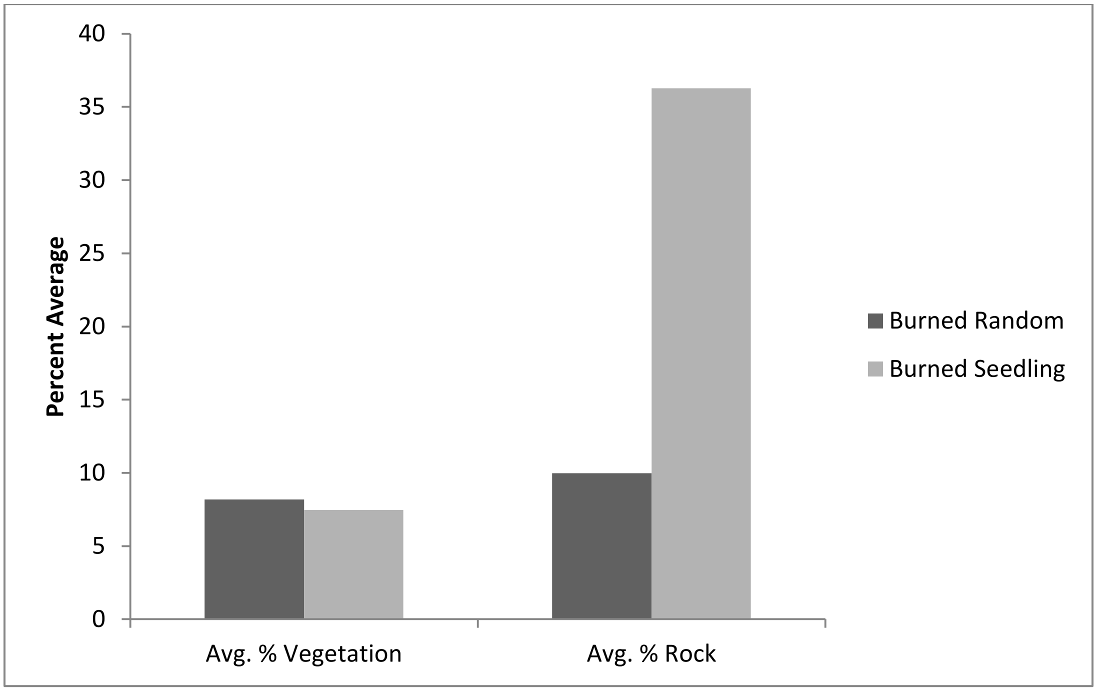

Figure 5.44. Percent average vegetation cover and rock cover in burned areas for random micro-plots compared to seedling micro-plots.

Seedling Species and Counts

LD and SC only had fir (1 and 4, respectively) and LDG and UDI only had pine (4 and 1, respectively). Seedlings were found at all sites but UDE, and most were found at UDW. Subalpine fir was the most common in both burned (Fig. 5.45) and unburned areas (Fig. 5.46) (Fig. 5.47). Seedling heights in the burned areas ranged from 1.5 to 12 cm and averaged 4.2 cm (n=68).

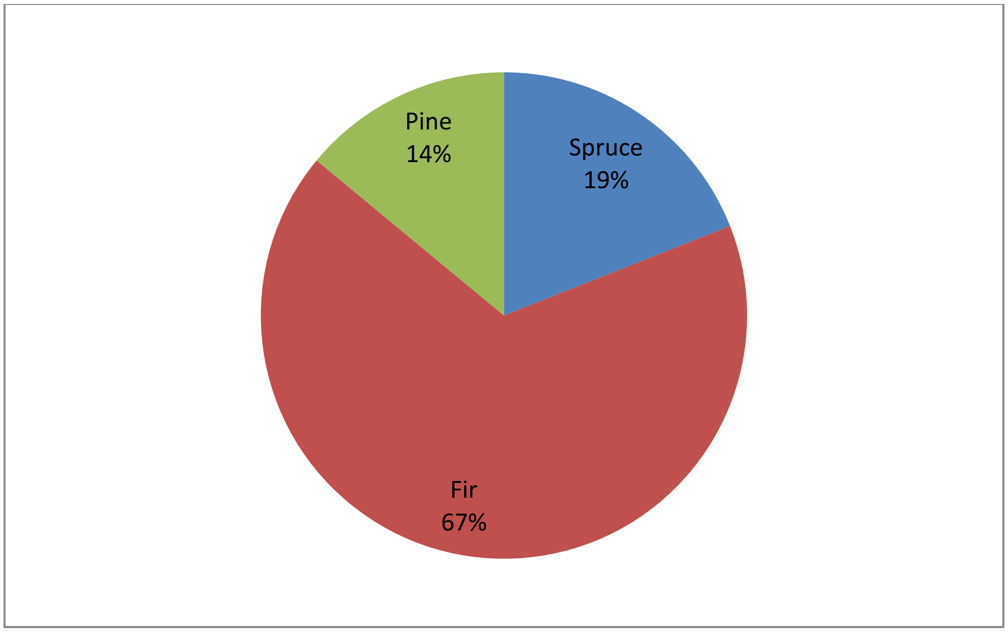

Figure 5.45. Seedling species distribution in burned areas (n=68).

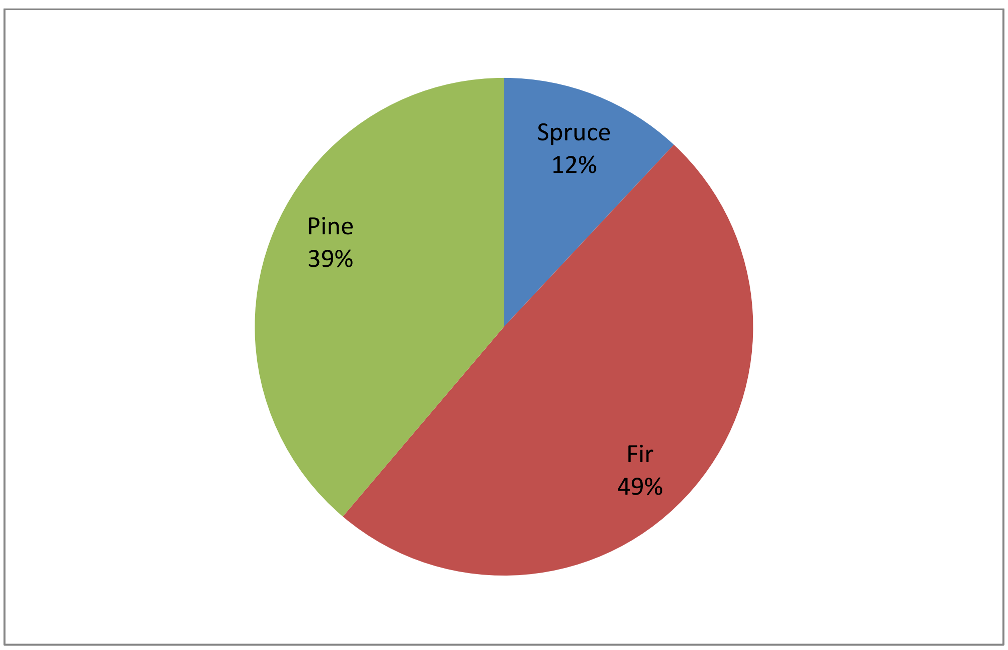

Figure 5.46. Seedling species distributions in unburned areas (n=33).

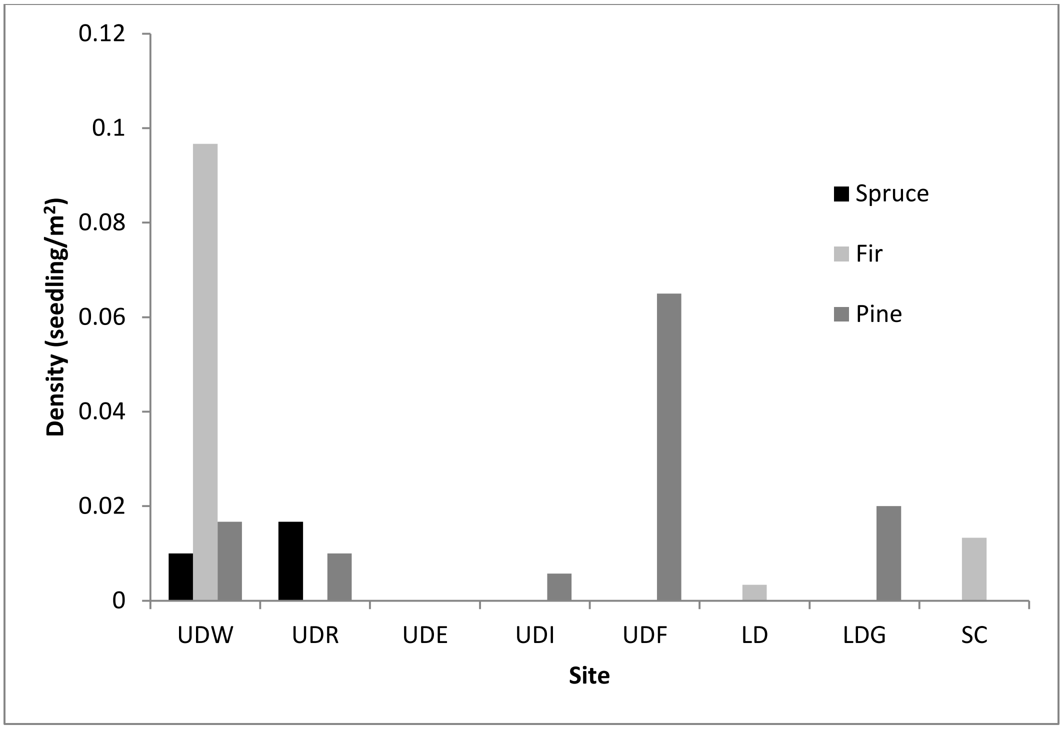

Figure 5.47. Density (seedling/m^2) of seedling species per site (n=68).

Facilitative Objects

Seedling re-establishment has been found in association with nurse objects in harsh environments (Parker 1989; Resler et al. 2005) as well as after fire within the subalpine forest (Coop and Schoettle 2009). Such facilitative objects may include boulders, fallen or dead trees, or live vegetation. Boulders and burned krummholz served as objects within the burned area at treeline, and live krummholz was the primary object in unburned areas. Comparisons were made with the Mann Whitney *U* test and averages are reported with standard deviation. Seedlings in burned areas were significantly closer to an object (18.0±21.1 cm) than the distance between an object and seedlings in unburned areas (79.8±50.4 cm) (α=0.05). In unburned areas, seedlings were not found to be significantly closer to objects compared to random plots (66.71±57.1 cm). Seedlings were significantly closer to objects compared to random plots within burned areas (α=0.05). Distance to the closest object from seedlings was less than half of that found with random plots (Table 5.8). Similarly, the average distance to three objects was much less for seedling plots compared to random plots. Rocks were the most common object (61%) (Figures 5.48 and 5.49), followed by burned krummholz (34%), and burned krummholz stumps (4%). The chart (Fig. 5.50) shows the object type that was closest to the seedling in burned areas. Krummholz was the more common object in unburned areas (Fig. 5.51) at 58%, compared to rocks at 42%. In burned areas, some of the krummholz were burned to ground level or down to a size smaller than 10 cm x 10 cm x 10 cm. Also, boulders may provide more thorough protection for a seedling than krummholz, which would be more important in the exposed burned areas. By definition, only objects that were at least 10 cm x 10 cm x 10 cm were considered, but I noted that several seedlings were closer to a smaller (<10 cm x 10 cm x 10 cm) object (Fig. 5.52).

Figure 5.48. Rocks served as the most common facilitative object for seedlings in burned areas. Photograph taken at Upper Divide, July 2012.

Figure 5.49. Seedling established adjacent to a boulder at Upper Divide. Photograph taken August 2012.

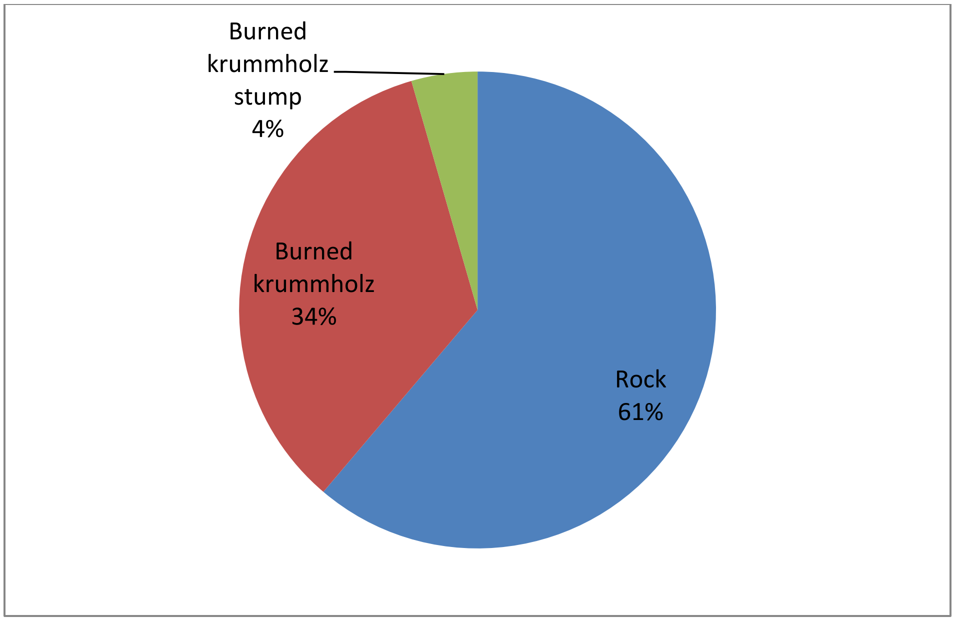

Figure 5.50. Percent of object types that were closest to seedlings in burned areas (n=68).

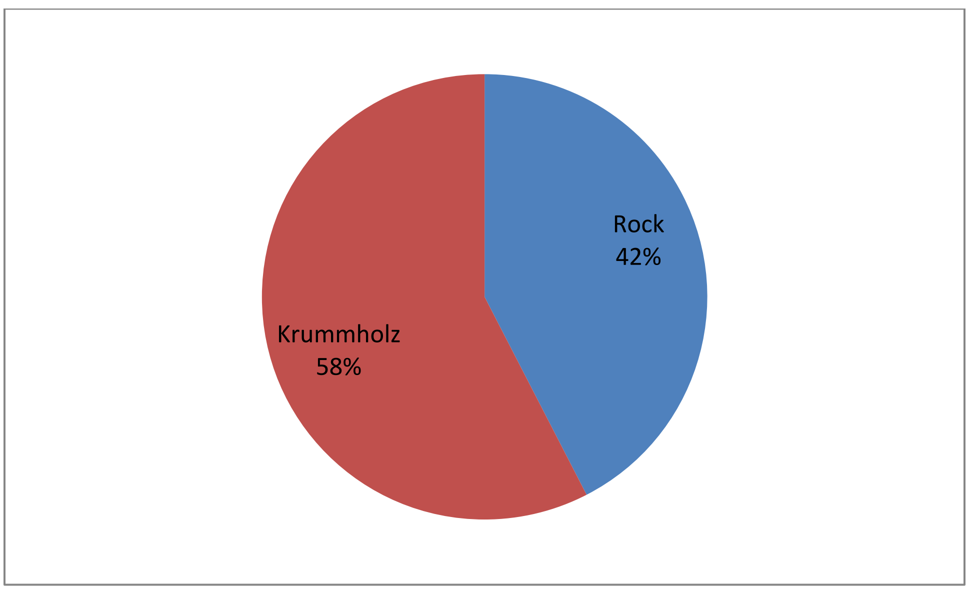

Figure 5.51. Percent of object types that were closest to seedlings in unburned areas (n=33).

Figure 5.52. Some seedlings were located in close proximity to objects less than 10 x 10 x 10 cm.

Table 5.41. Comparison of average distances among species and between each species and random plots. A Kruskal Wallis test indicated that significant differences in distance to the closest object were found among species, and therefore the Mann Whitney U test was used to compare each species to each other with a Bonferroni correction. ^ indicates that the level of significance was 0.0167 because of the Bonferroni correction. All other comparisons were made with a significance level of 0.05. * indicates significant differences were found between the average distances.

Closest object	P-value
fir vs. spruce	0.017^
spruce vs. pine	0.934
fir vs. pine	0.006^*
spruce vs. random	0.005*
fir vs. random	0.001*
pine vs. random	0.001*
Second object	
fir vs. spruce	0.348
spruce vs. pine	0.427
fir vs. pine	0.055
spruce vs. random	0.001*
fir vs. random	0.001*
pine vs. random	0.001*
Third object	
fir vs. spruce	0.438
spruce vs. pine	0.515
fir vs. pine	0.054
spruce vs. random	0.002*
fir vs. random	0.001*
pine vs. random	0.001*

Distance to the Closest Object

Significant differences in means were found among fir, spruce, and pine seedlings in regard to distance to the closest object, as determined with the Kruskal Wallis test (p<0.006) (Table 5.41), and each species was significantly closer to an object than the center of a random plot (α=0.05) (Figures 5.53-5.56). *Post hoc* tests were performed on the data to determine which species were significantly different from the others. The Mann Whitney *U* test with a Bonferroni correction ($\alpha=0.0125$) applied was used as a *post hoc* test. Fir and pine distances were found to be significantly different ($p<0.006$). Fir and spruce were close to being significantly different, but $p<0.017$, which is not less than the significance level of 0.0125. Spruce and pine were also not significantly different ($p<0.934$). Each species also differed significantly from the average random distance (spruce, $p<0.005$; fir, $p<0.001$; pine, $p<0.001$).

Distance to the Closest Second and Third Objects

The Kruskal Wallis test was also used to compare the average distance to the second and third closest object among the species (Figures 5.57 and 5.58). No significant differences among the species were found ($\alpha=0.05$, second, $p<0.132$; third, $p<0.145$). Average distances for each species and random plots were compared for second and third objects with the use of the Mann Whitney *U* test and a significance level of 0.05, and each species was found to differ significantly from the average random distance for both second and third objects (Table 5.41).

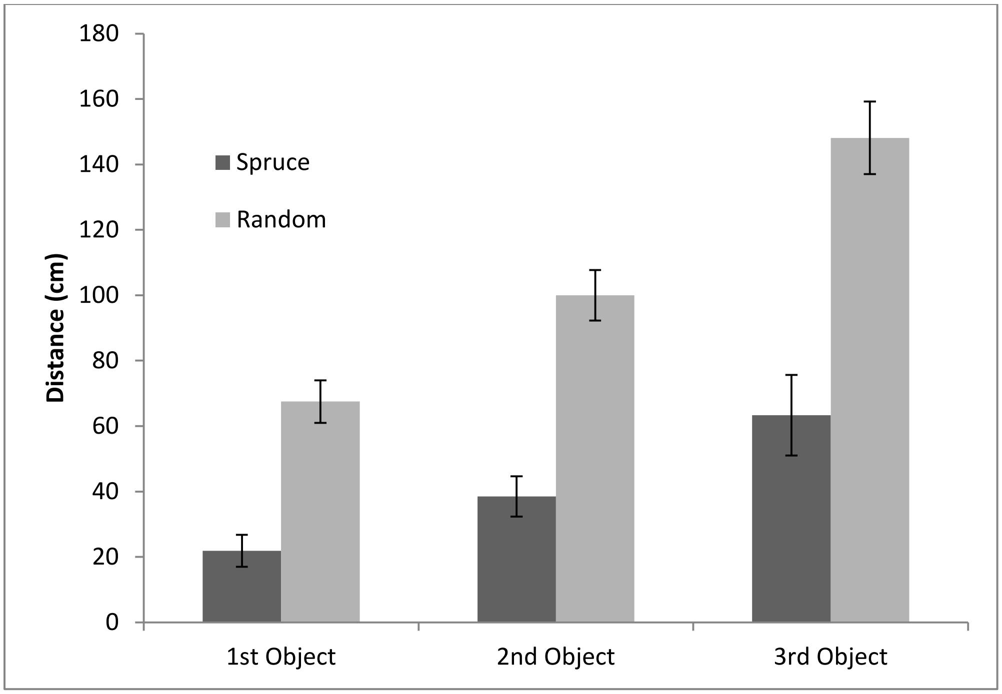

Figure 5.53. Average distances (cm) to objects for spruce and random plots. Error bars show ±1SE.

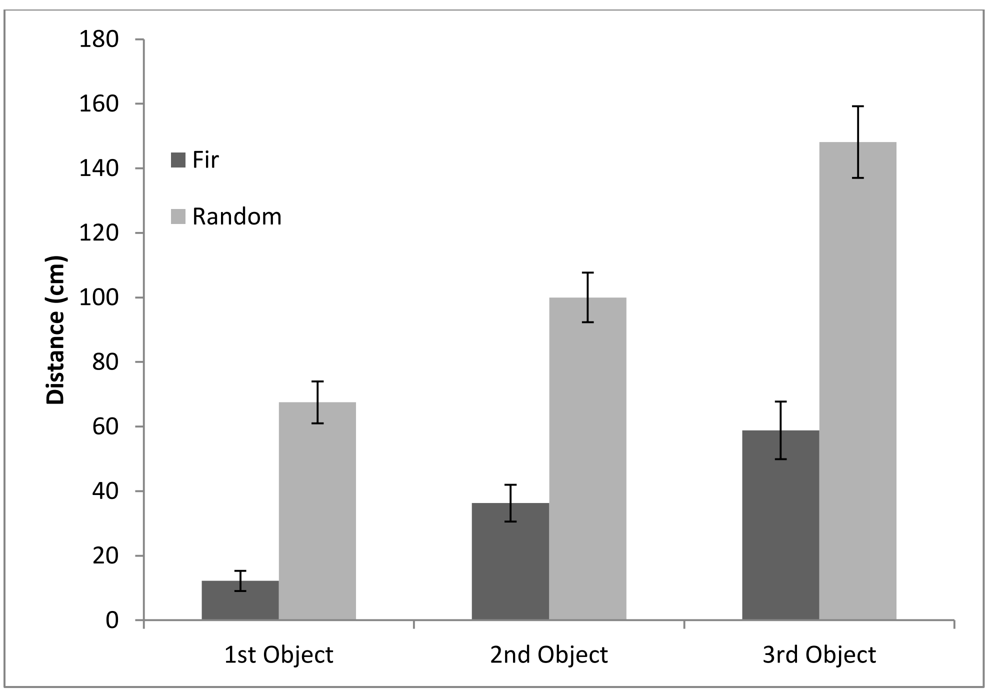

Figure 5.54. Average distances (cm) to objects for fir and random plots. Error bars show ±1SE.

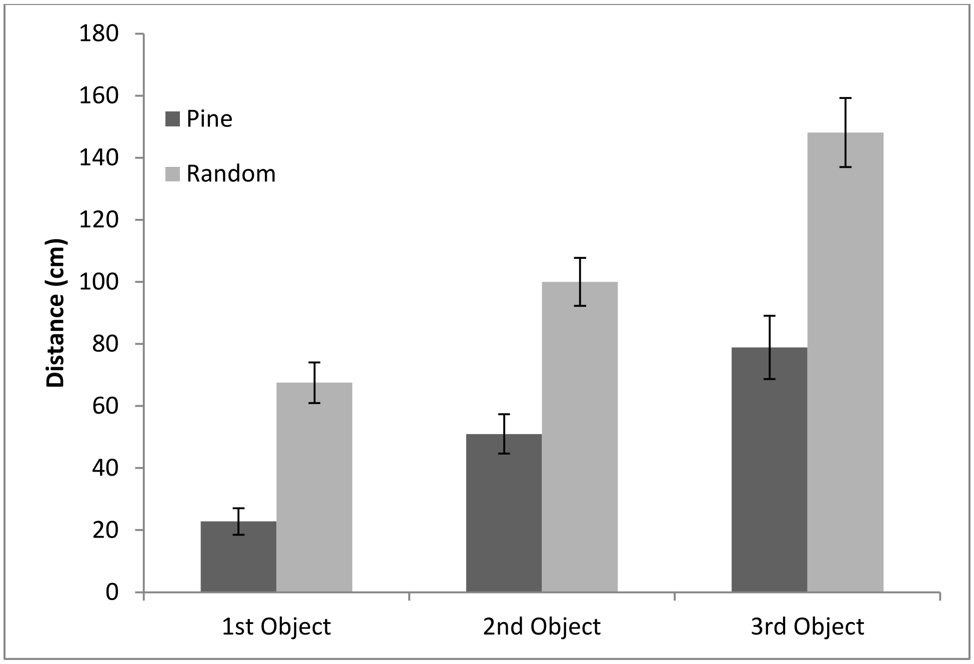

Figure 5.55. Average distances (cm) to objects for pine and random plots. Error bars show ±1SE.

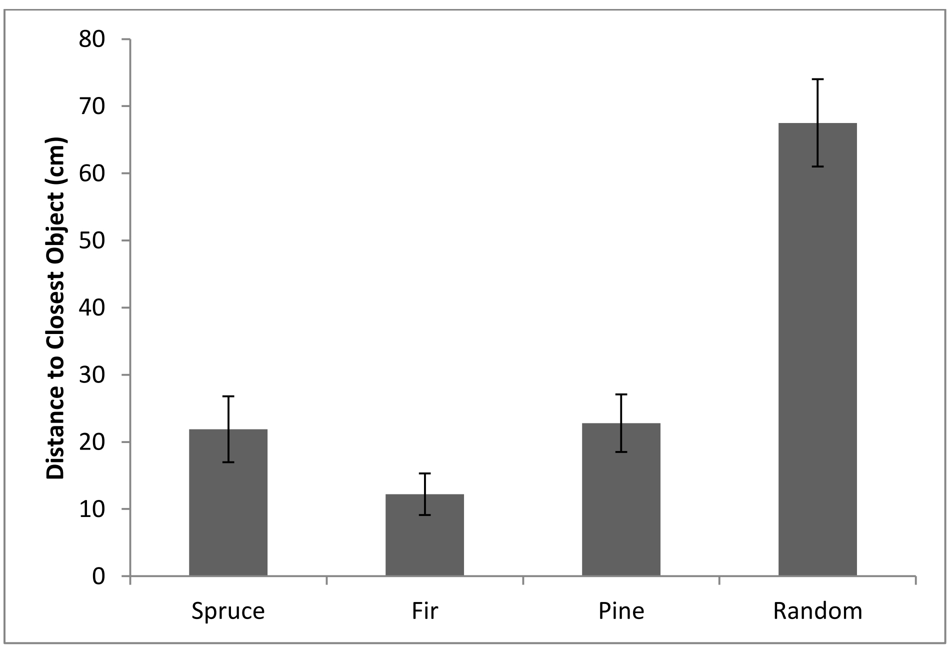

Figure 5.56. Average distance (cm) to the closest objects for each species and the random plots. Error bars show ±1SE.

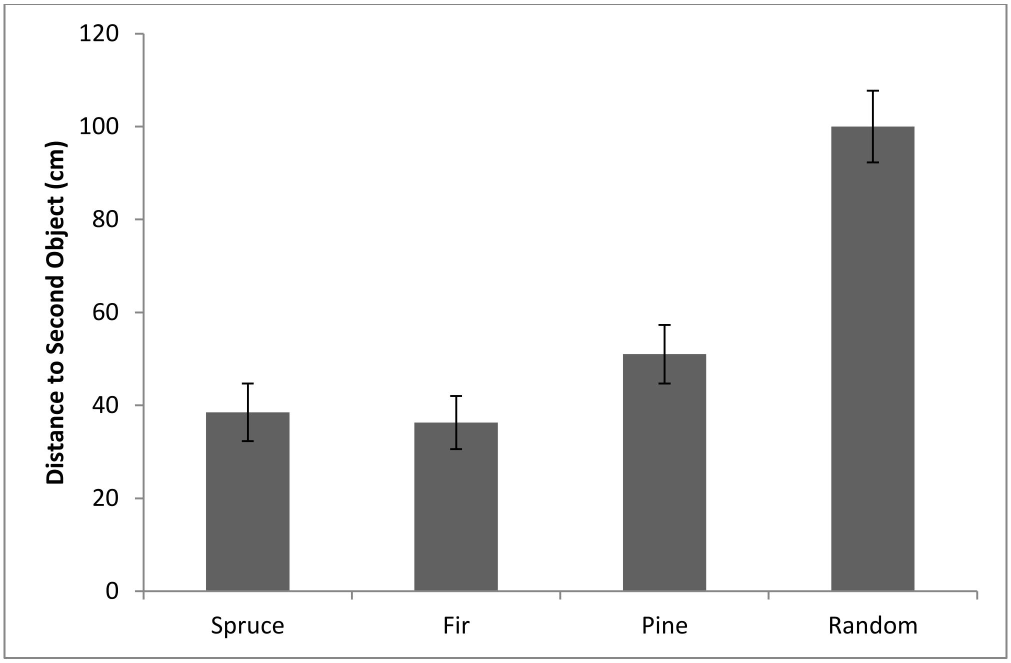

Figure 5.57. Differences (cm) were not significant among species, but each species differed from the average random distance (α=0.05). Error bars show ±1SE.

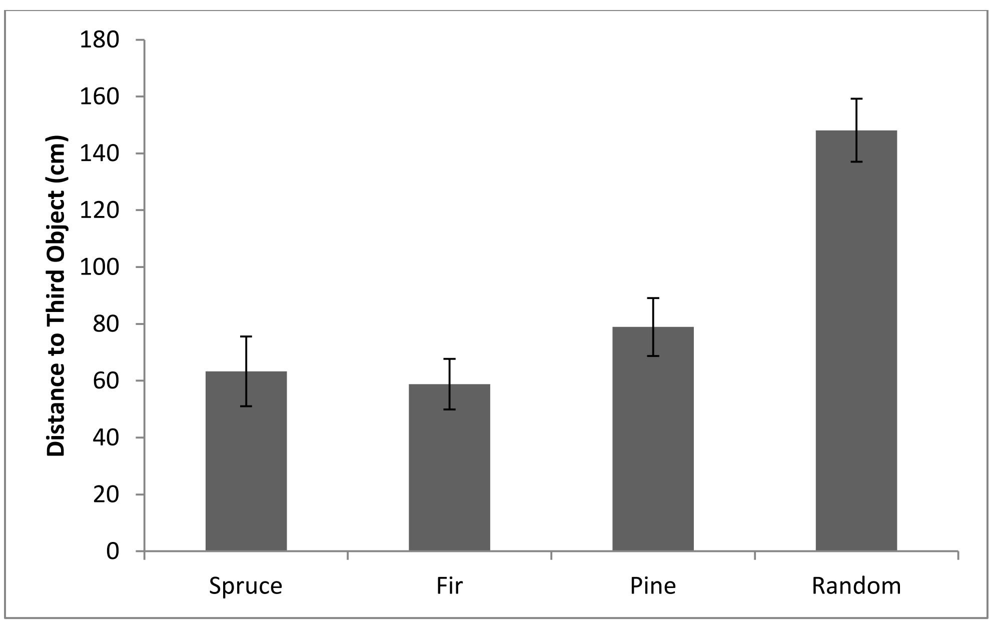

Figure 5.58. Average distance (cm) to third object for spruce, fir, and pine seedlings and the center of random micro-plots. Error bars show ±1SE.

Section 4: Biogeomorphic Interactions and Methods

This final section presents data on combined soil and vegetation variables, especially focusing on methodological factors when collecting data on both soil and vegetation conditions, that may potentially influence results. The first topic is on vegetation conditions, including herbaceous vegetation and krummholz burn severities, and they are subsequently compared to select soil conditions. The second topic is on the burned/unburned edge and how soil conditions change in relation to the ecological edge. Data are presented on ESD, soil penetrability, and clast size along three transects that extended across a burned krummholz patch. Transects were twice as long as the width of the burned patch to evaluate the edge influence on both the burn interior as well as on the unburned krummholz on either sides of the edge. Trends on data across the transects are presented and comparisons are made to determine which areas differ significantly from others in regard to edges. Differences within a burned patch in relation to distance from the edge, may potentially influence results based on quadrat placement. The third topic evaluates the influence of plot size on results. Averages for variables of clast size, soil penetrability, krummholz density, and total basal area are compared among the eight plot sizes. Trends regarding the distribution of the data are discussed in relation to each variable as well as among all four variables.

Vegetation Conditions

Herbaceous Vegetation

Herbaceous vegetation was found to be established at all sites. Average percent cover was 30.6% with a standard deviation of 2.6% and a range of 5 to 80%. Within an area, vegetation was noted to often be patchy, with thick clumps of vegetation interspersed with bare soil, or predominantly bare soil containing patches of vegetation clumps (Figures 5.59 and 5.60). Although no quantitative data were collected on vegetation change throughout the three years of study, observations were made that indicated that herbaceous vegetation cover increased between 2010 and 2012 (Figures 5.61 - 5.64).

Figure 5.59. Herbaceous vegetation patches in the burned area at Lower Divide Mountain. Photograph taken July 2010.

Figure 5.60. Herbaceous vegetation in the burned area at Upper Divide Mountain, on the east facing slope of the saddle. Photograph taken July 2011.

Figure 5.61. Upper Divide West site in late July 2010.

Figure 5.62. Upper Divide West site (slightly different angle from previous picture) in early August 2012.

Figure 5.63. Photograph taken in late July 2010 of Upper Divide West. Very little herbaceous vegetation was present.

Figure 5.64. Photograph taken in late July 2012 of Upper Divide West. Note the increased coverage of herbaceous vegetation.

<u>Krummholz</u>

Recent fires provided the opportunity to collect data on krummholz density and diameters much more efficiently than krummholz with full foliage. A total of 1,138 krummholz were recorded in the burned quadrats. Average DGL was 11.8 cm with a standard deviation of 11.6 cm, and values ranged from 0.2 to 127.3 cm. Density for all burned quadrats averaged 0.58 krummholz per square meter. Average stem count was 1.44 stems per krummholz with a standard deviation of 2.61, and ranged from 0 up to 35 stems. Basal area averaged 224.46 cm^2 with a standard deviation of 757.18 cm^2 and values ranged from 0.03 to 1,272.15 cm^2. Total basal area for all measured krummholz equaled 255,659.1 cm^2.

The mode for burn severity was 2 for all krummholz. Krummholz burned conditions ranged from burned almost flush with the ground, to those that still retained needles. Those that did contain needles were often located near the edge of the burned area. I noted that severely burned krummholz (1) was sometimes located in close proximity (within a meter between the bases) to a krummholz that had burned less (2 or 3), but rarely did I notice a 1 or 2 near a 4 or 5.

Although I did not collect measurements on unburned krummholz, I did note that most were subalpine fir or Engelmann spruce. Several five-needle pines were also observed in the unburned area.

Relationships between Soil and Vegetation Conditions

Potential relationships among the variables of percent vegetation cover, soil compaction, krummholz density, erosion severity, and burn severity were assessed with Kendall’s tau. Soil penetrability values were categorized into 10 classes (Table 5.42) based on the range of the data. Only three of the combinations of those variables were significantly related – soil penetrability and krummholz density, percent vegetation cover and erosion severity, and erosion severity and burn severity (Table 5.43). Soil penetrability values decreased with increased krummholz density (*p*<0.001). As may be expected, percent vegetation cover and erosion severity share an inverse relationship (*p*<0.001). Also, as expected, erosion severity and burn severity have a direct relationship (0.025).

Table 5.42. Classes of penetrability values (kg/cm^2).

Penetrability Range (kg/cm^2)	Class
0-.25	1
.251-.5	2
.501-.75	3
.751-1.0	4
1.01-1.25	5
1.251-1.5	6
1.501-1.75	7
1.751-2.0	8
2.01-2.25	9
2.251-2.5	10

Table 5.43. Significance results between variables tested with Kendall's tau (significance at the 0.05 level indicated with an *).

Variable Comparison		Significance
Penetrability	% Vegetation	0.607
Penetrability	Krummholz Density	0.001*
Penetrability	Erosion Severity	0.079
Erosion Severity	% Vegetation	0.001*
Erosion Severity	Krummholz Density	0.102
Erosion Severity	Burn Severity	0.025*
Burn Severity	Penetrability	0.305
Burn Severity	Krummholz Density	0.292
Burn Severity	% Vegetation	0.69

Burned/Unburned Edge

Edge effects were evaluated in relation to the burned/unburned edge of a burned patch at the Lower Divide site (Fig. 5.65). The topography was overall homogenous, with only slight differences in elevation for the three transects. The burned patch varied in width from 25 m near the upper end of the patch and then expanded to 50 m in width about 20 m downslope of the uppermost krummholz-alpine edge. The transects extended into thick krummholz on both sides of the burned patch (Fig. 5.66)

Figure 5.65. The burned/unburned edge at Lower Divide, facing eastward. Photograph taken in August 2012.

Figure 5.66. Unburned krummholz at Lower Divide.

Soil Penetrability

Soil penetrability varied along a gradient in relation to the burned/unburned edge (Figures 5.67-5.69). These data were analyzed with one-way ANOVA (p<0.001 for each transect). Tables 5.44 - 5.46 show the homogenous sub-set results from a Tukey *post hoc* test. Average penetrability values are shown in regard to their location and what values they differed from at the significance level of 0.05. As expected, the edges did in general have some of the lowest penetrability values for all three transects. The overall trend is that the burned areas contained both the higher and lower penetrability values and the unburned area primarily fell within the middle of the range of values. The highest penetrability values for Transects 2 and 3, at 40 and 55 m respectively, did roughly fall within the middle of the burned patch, as expected. The highest penetrability values for the burned area also were found in the center of the burned patch for Transect 1. I hypothesized that the interior of the burned patch would have the highest penetrability values because the center of the patch is most exposed and may, therefore, experience high amounts of loose soil loss. However, at 55 m in transect 2, one of the lowest penetrability values within the burned area was found. On the western side of the burned patch of both transects 2 and 3, at points 30 and 35 m, the penetrability values are towards the high end of the range. These points along the transect lie just to the east of the western burned/unburned boundary. On the eastern side of the patch, just west of the eastern burned/unburned boundary, the opposite trend is present in Transects 2 and 3. On Transect 2, the penetrability at 70 m is the lowest found across the entire transect. On Transect 3, 65 m has the third lowest value. Transect 1 also displays this trend, in which the lowest penetrability value was found just to the west of the eastern burned/unburned boundary. Transect 1 differs from Transects 2 and 3 primarily in that the unburned values are, for the most part, higher than the burned values, and the highest burned values are lower than those found in Transects 2 and 3. (These differences may be attributed to the smaller patch width of Transect 1, which is smaller and more protected from wind by the unburned vegetation, and therefore probably experiences less soil loss.)

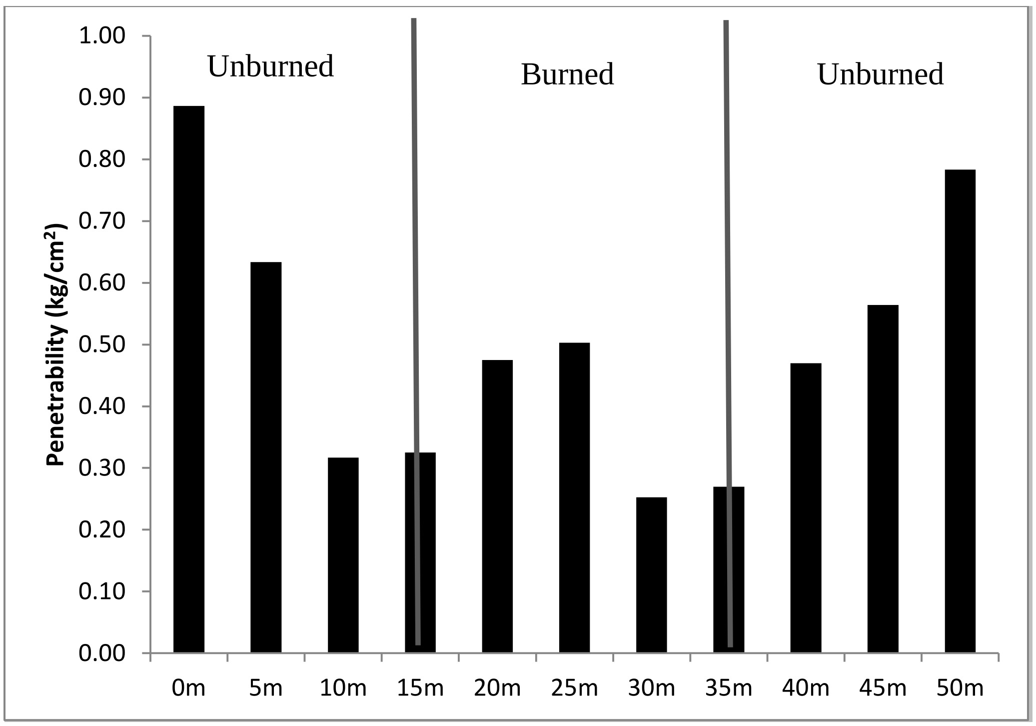

Figure 5.67. Average penetrability values (kg/cm^2) across Transect 1 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

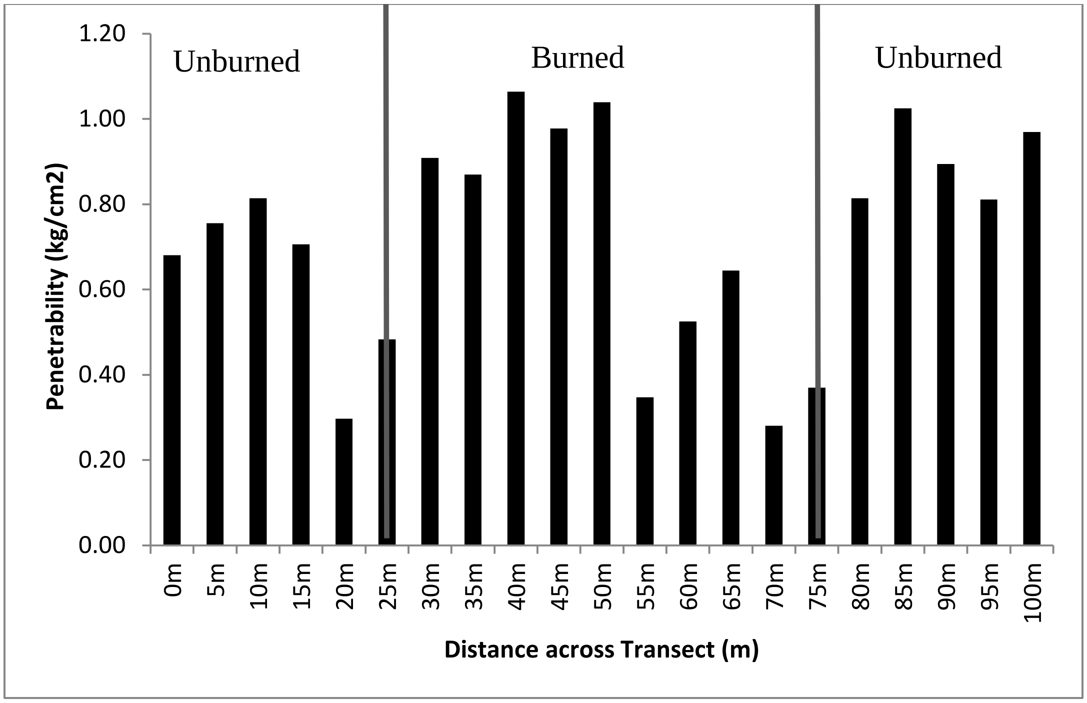

Figure 5.68. Average penetrability values (kg/cm^2) across Transect 2 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

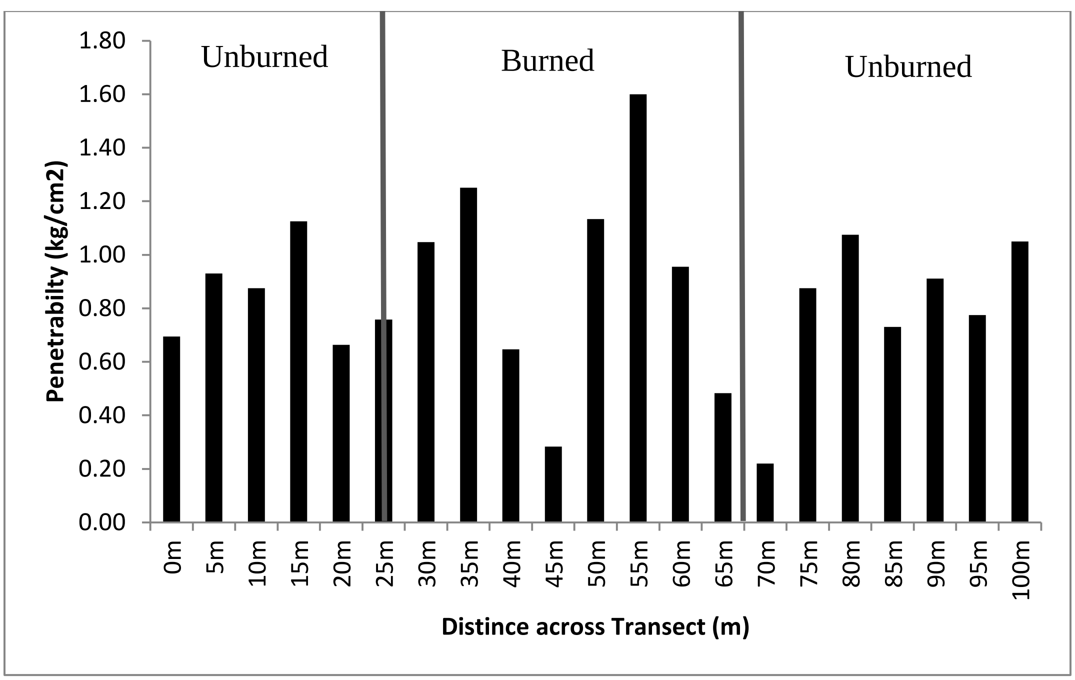

Figure 5.69. Average penetrability values (kg/cm^2) across Transect 3 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

Table 5.44. Homogenous sub-sets revealed through Tukey *post hoc* tests for penetrability across Transect 1. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Location	N	Subset for alpha = 0.05			
		1	2	3	4
30.00	30	.2528			
35.00	30	.2694			
10.00	30	.316			
15.00	30	.3250			
40.00	30		.4694		
20.00	30		.4750		
25.00	30		.5028	.5028	
45.00	30		.5639	.5639	
5.00	30			.6333	
50.00	30				.7833
.00	30				.8861
Sig.		.870	.559	.115	.427

Table 5.45. Homogenous sub-sets revealed through Tukey *post hoc* tests for penetrability across Transect 2. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

	N	Subset for alpha = 0.05								
		1	2	3	4	5	6	7	8	9
70.00	30	.2806								
20.00	30	.2972								
55.00	30	.3472	.3472							
75.00	30	.3694	.3694							
25.00	30	.4833	.4833	.4833						
60.00	30		.5250	.5250	.5250					
65.00	30			.6444	.6444	.6444				
.00	30			.6806	.6806	.6806	.6806			
15.00	30				.7056	.7056	.7056	.7056		
5.00	30					.7556	.7556	.7556		
95.00	30					.8111	.8111	.8111	.8111	
10.00	30					.8139	.8139	.8139	.8139	
80.00	30					.8139	.8139	.8139	.8139	
35.00	30						.8694	.8694	.8694	.8694
90.00	30							.8944	.8944	.8944
30.00	30							.9083	.9083	.9083
100.00	30								.9694	.9694
45.00	30								.9778	.9778
85.00	30									1.0250
50.00	30									1.0389
40.00	30									1.0639
Sig.		.055	.191	.075	.169	.269	.115	.055	.298	.087

Table 5.46. Homogenous sub-sets revealed through Tukey *post hoc* tests for penetrability across Transect 3. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

		Subset for alpha = 0.05									
	N	1	2	3	4	5	6	7	8	9	10
70.00	30	.2194									
45.00	30	.2833	.2833								
65.00	30		.4833	.4833							
40.00	29			.6523	.6523						
20.00	30			.6639	.6639						
.00	30			.6944	.6944	.6944					
85.00	30				.7306	.7306	.7306				
25.00	30				.7583	.7583	.7583				
95.00	30				.7750	.7750	.7750				
10.00	30				.8750	.8750	.8750	.8750			
75.00	30				.8750	.8750	.8750	.8750			
90.00	30					.9111	.9111	.9111	.9111		
5.00	30					.9306	.9306	.9306	.9306		
60.00	30						.9556	.9556	.9556		
30.00	30							1.0472	1.0472	1.0472	
100.00	30							1.0500	1.0500	1.0500	
80.00	30							1.0750	1.0750	1.0750	
15.00	30								1.1250	1.1250	
50.00	30								1.1333	1.1333	
35.00	30									1.2500	
55.00	30										1.600
Sig.		1.000	.241	.160	.099	.054	.090	.241	.101	.218	1.000

Effective Soil Depth

Effective soil depth varied along a gradient in relation to the burned/unburned edge (Figures 5.70-5.72). These data were analyzed with one-way ANOVA (p<0.001 for each transect). Tables 5.47-5.49 show the homogenous sub-set results from a Tukey *post hoc* test. Average penetrability values are shown in regard to their location and what values they differed from at the significance level of 0.05. Effective soil depth was lowest near burned/unburned edges in Transects 2 and 3, and towards the center in Transect 1. On Transect 2, the lowest values were towards the burned interior at both burned/unburned boundaries, but only lower on the easternmost boundary (again, towards the burned interior) of Transect 3. Transect 1 and 2 reveal a strong pattern of lower ESD values in the burned area and higher overall values in the unburned portion of the transect. However, Transect 3 does not present such a clear trend, but rather highest and lowest ESD values are found in the burned position, and the unburned values fall mostly between those two extremes. Moderate depths were also found in burned areas.

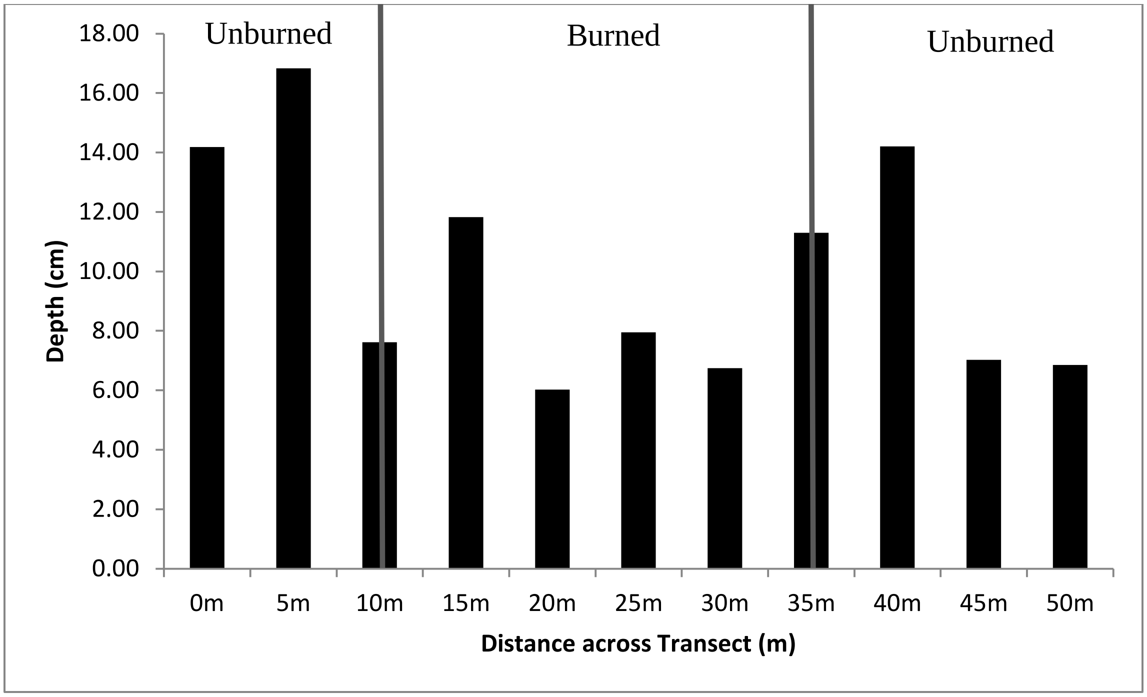

Figure 5.70. Average ESD (cm) across Transect 1 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

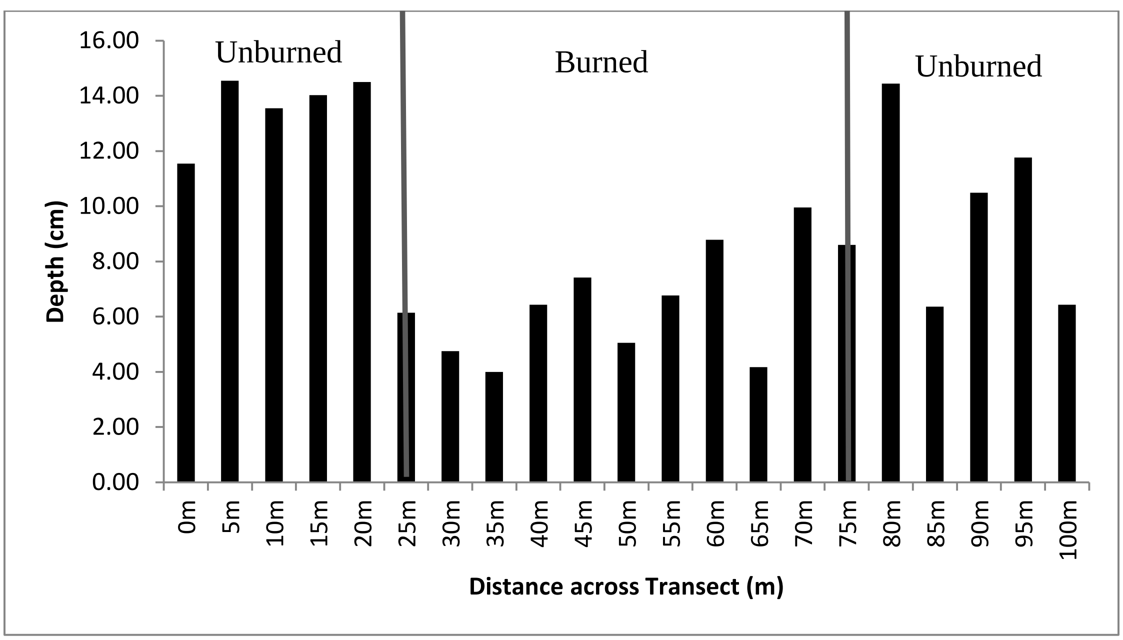

Figure 5.71. Average ESD (cm) across Transect 2 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

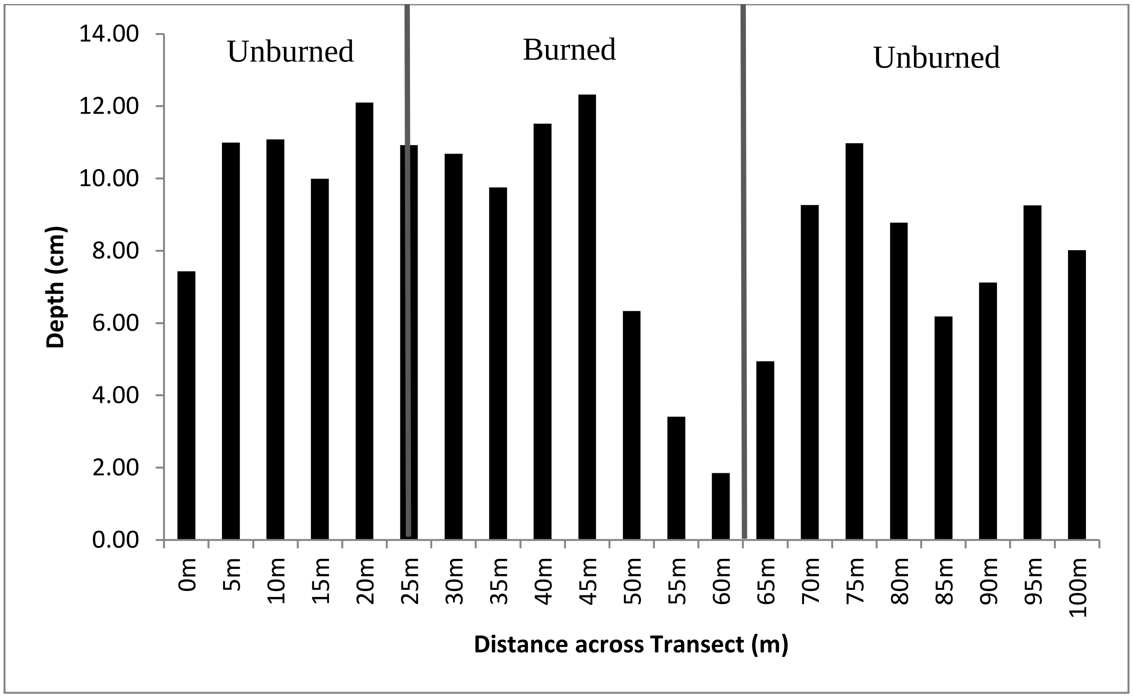

Figure 5.72. Average ESD (cm) across Transect 3 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

Table 5.47. Transect 1 Homogenous sub-sets revealed through Tukey *post hoc* tests for ESD across Transect 1. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Tukey HSD[a]

T1	N	Subset for alpha = 0.05				
		1	2	3	4	5
20.00	30	6.0233				
30.00	30	6.7500				
50.00	30	6.8500				
45.00	30	7.0300				
10.00	30	7.6133	7.6133			
25.00	30	7.9500	7.9500	7.9500		
35.00	30		11.3000	11.3000	11.3000	
15.00	30			11.8233	11.8233	
.00	30				14.1867	14.1867
40.00	30				14.2033	14.2033
5.00	30					16.8300
Sig.		.882	.085	.055	.365	.512

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 30.000.

Table 5.48. Homogenous sub-sets revealed through Tukey *post hoc* tests for ESD across Transect 2. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Tukey HSD[a]

T2	N	Subset for alpha = 0.05							
		1	2	3	4	5	6	7	8
35.00	30	4.0033							
65.00	30	4.1700							
30.00	30	4.7533	4.7533						
50.00	30	5.0567	5.0567						
25.00	30	6.1433	6.1433	6.1433					
85.00	30	6.3600	6.3600	6.3600					
100.00	30	6.4267	6.4267	6.4267					
40.00	30	6.4300	6.4300	6.4300					
55.00	30	6.7700	6.7700	6.7700	6.7700				
45.00	30	7.4200	7.4200	7.4200	7.4200				
75.00	30		8.6000	8.6000	8.6000	8.6000			
60.00	30		8.7800	8.7800	8.7800	8.7800			
70.00	30			9.9533	9.9533	9.9533	9.9533		
90.00	30				10.4867	10.4867	10.4867	10.4867	
.00	30					11.5400	11.5400	11.5400	11.5400
95.00	30					11.7586	11.7586	11.7586	11.7586
10.00	30						13.5500	13.5500	13.5500
15.00	30							14.0267	14.0267
80.00	30							14.4367	14.4367
20.00	30							14.5033	14.5033
5.00	30								14.5433
Sig.		.231	.051	.093	.117	.375	.156	.053	.478

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 30.000.

Table 5.49. Transect 3 Homogenous sub-sets revealed through Tukey *post hoc* tests for ESD across Transect 3. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Tukey HSD[a,b]

T2	N	Subset for alpha = 0.05							
		1	2	3	4	5	6	7	8
60.00	30	1.8533							
55.00	30	3.4100	3.4100						
65.00	30	4.9400	4.9400	4.9400					
85.00	30		6.1867	6.1867	6.1867				
50.00	30		6.3300	6.3300	6.3300				
90.00	30		7.1233	7.1233	7.1233	7.1233			
.00	30			7.4300	7.4300	7.4300	7.4300		
100.00	29			8.1724	8.1724	8.1724	8.1724	8.1724	
80.00	30			8.7767	8.7767	8.7767	8.7767	8.7767	8.7767
95.00	30				9.2567	9.2567	9.2567	9.2567	9.2567
70.00	30				9.2667	9.2667	9.2667	9.2667	9.2667
35.00	30				9.7533	9.7533	9.7533	9.7533	9.7533
15.00	30				9.9900	9.9900	9.9900	9.9900	9.9900
30.00	30					10.6800	10.6800	10.6800	10.6800
25.00	30					10.9233	10.9233	10.9233	10.9233
75.00	30					10.9733	10.9733	10.9733	10.9733
5.00	30					10.9967	10.9967	10.9967	10.9967
10.00	30						11.0833	11.0833	11.0833
40.00	30							11.5167	11.5167
20.00	30							12.1033	12.1033
45.00	30								12.3233
Sig.		.369	.092	.066	.072	.059	.108	.050	.141

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 29.951.

b. The group sizes are unequal. The harmonic mean of the group sizes is used. Type I error levels are not guaranteed.

<u>Clast Size</u>

Clast size varied along a gradient in relation to the burned/unburned edge (Figures 5.73-5.75). The average values among the distances along the transects lengths varied significantly when analyzed with a one-way ANOVA (*p*<0.001 for each transect). Tables 5.50-5.52 show the homogenous sub-set results from a Tukey *post hoc* test. Average clast size values are shown in regard to their location and what values they differed from at the significance level of 0.05. All three transects displayed a roughly bell-shaped curve in regard to clast size, with largest values found towards the center of the burned patches and smallest clast sizes found in the unburned areas.

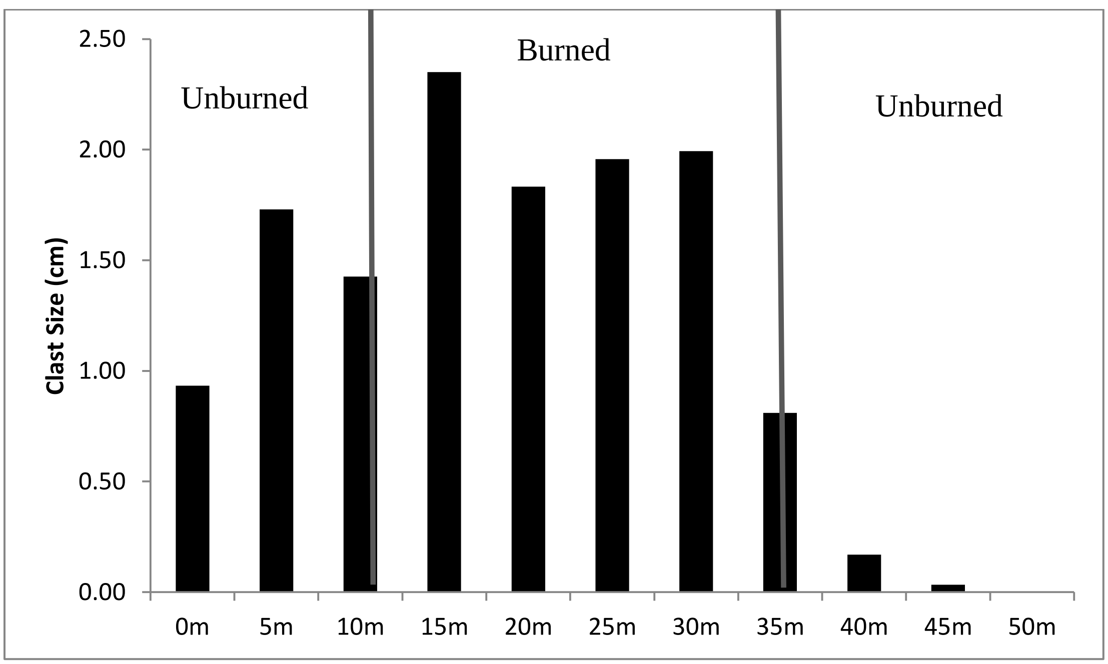

Figure 5.73. Average clast (cm) size across Transect 1 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

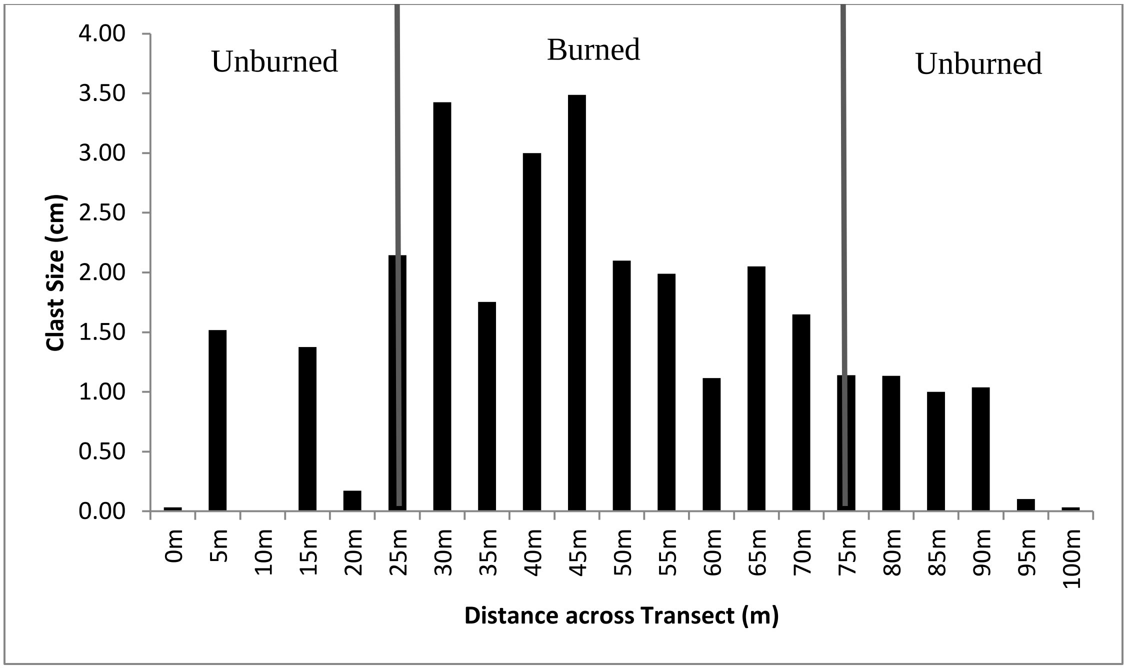

Figure 5.74. Average clast size (cm) across Transect 2 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

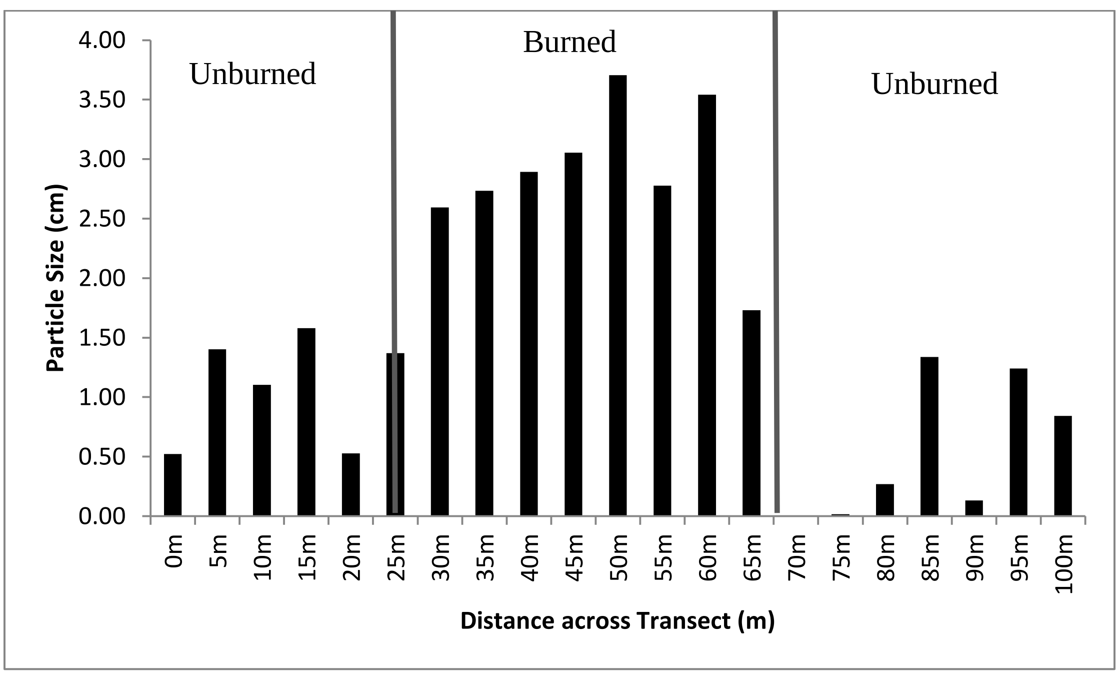

Figure 5.75. Average particle size (cm) across Transect 3 (n=30 for each bar). Grey lines indicate burned/unburned boundary location.

Table 5.50. Homogenous sub-sets revealed through Tukey *post hoc* tests for clast size across Transect 1. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Tukey HSD[a]

T1	N	Subset for alpha = 0.05					
		1	2	3	4	5	6
50.00	30	.0000					
45.00	30	.0333					
40.00	30	.1700	.1700				
35.00	30	.8100	.8100	.8100			
.00	30		.9333	.9333	.9333		
10.00	30			1.4267	1.4267	1.4267	
5.00	30				1.7300	1.7300	1.7300
20.00	30					1.8333	1.8333
25.00	30					1.9567	1.9567
30.00	30					1.9933	1.9933
15.00	30						2.3500
Sig.		.088	.139	.426	.101	.558	.417

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 30.000.

Table 5.51. Homogenous sub-sets revealed through Tukey *post hoc* tests for clast size across Transect 2. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Tukey HSD[a]

	N	Subset for alpha = 0.05				
		1	2	3	4	5
10.00	30	.0000				
.00	30	.0333				
100.00	30	.0333				
95.00	30	.1034				
20.00	30	.1733	.1733			
85.00	30		1.0000	1.0000		
90.00	30			1.0367		
60.00	30			1.1167		
80.00	30			1.1333		
75.00	30			1.1400		
15.00	30			1.3767	1.3767	
5.00	30			1.5167	1.5167	
70.00	30			1.6500	1.6500	
35.00	30			1.7533	1.7533	
55.00	30				1.9900	
65.00	30				2.0500	
50.00	30				2.1000	
25.00	30				2.1433	
40.00	30					3.0000
30.00	30					3.4267
45.00	30					3.4867
Sig.		1.000	.056	.140	.120	.877

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 30.000.

Table 5.52. Homogenous sub-sets revealed through Tukey *post hoc* tests for clast size across Transect 3. Averages that do not differ significantly from each other are shown in each column. Values that also overlap across the row do not vary significantly. Yellow indicates the burned/unburned boundary, red represents areas in the burned patch, and green was located in the unburned krummholz.

Tukey HSD[a,b]

T2	N	Subset for alpha = 0.05							
		1	2	3	4	5	6	7	8
70.00	30	.0000							
75.00	30	.0167							
90.00	30	.1333							
80.00	30	.2700	.2700						
.00	30	.5233	.5233	.5233					
20.00	30	.5267	.5267	.5267					
100.00	29	.8379	.8379	.8379	.8379				
10.00	30		1.1033	1.1033	1.1033				
95.00	30			1.2400	1.2400				
85.00	30			1.3367	1.3367				
25.00	30			1.3700	1.3700				
5.00	30			1.4033	1.4033				
15.00	30				1.5800				
65.00	30				1.7300	1.7300			
30.00	30					2.5933	2.5933		
35.00	30						2.7333	2.7333	
55.00	30						2.7767	2.7767	
40.00	30						2.8933	2.8933	2.8933
45.00	30						3.0533	3.0533	3.0533
60.00	30							3.5400	3.5400
50.00	30								3.7067
Sig.		.115	.121	.071	.061	.086	.966	.160	.149

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 29.951.

b. The group sizes are unequal. The harmonic mean of the group sizes is used. Type I error levels are not guaranteed.

Comparison among Quadrat Sizes

An issue that this research attempted to address was to collect data on both geomorphic and vegetation conditions as they relate to each other. Scale is often a concern because geomorphic/soil conditions do not necessarily reflect changes and patterns in vegetation density and cover. I analyzed this question by collecting soil penetrability, particle size, krummholz density and basal area within varying quadrat sizes, from 1 x 1 m up to 5 x 20 m. Quadrats measuring 5 x 20 m contained nested plots, as described in the methods section, to measure the influence of quadrat scale on the data results and how that may relate to both soil conditions and vegetation.

One-way within-subjects ANOVA was performed on the krummholz density, soil penetrability, and particle size variables. Mauchly's Test of Sphericity implied that the data variables' variance-covariance matrices were not homogenous. The ANOVAs were evaluated for significance using the Greenhouse-Geisser correction to the mean squares. Results of the analysis show no significant differences among the means of krummholz density, soil penetrability, and particle size across quad size (α=0.05).

Clast Size

Average clast size for all quadrats combined was lowest at 3.9 cm for the smallest plot size (1 x 1 m) and greatest at 4.9 cm for the 5 x 5 m plot size (Figures 5.76 and 5.77, Table 5.53). Then the average values dip down again, particularly with the 75 m^2 plot, before increasing back to 4.5 in the 100 m^2 plot. The differences among the plot sizes were not significant at the 0.05 level (Greenhouse-Geisser, p<0.380). The values of the 1 x 1 plot varied from 1.8 (LD1) to 6.7 cm (UDR2); 2 x 2 m 3.2 (UDE5 and SC3) to 7.1

(UDR2); 3 x 3 m, 3.0 (UDE5) to7.8 (UDR1); 4x4 m 2.9 (UDE5) to 9.3 (UDR1); 5 x 5 m 2.8 (UDE5) to 11.6 (UDR1); 5 x 10 m, 2.6 (SC3) to 11.3 (UDR1); 5 x 15 m, 2.6 (SC3) to 7.3 (UDR1); 5 x 20 m, 2.5 (SC3) to 12.5 (UDR1). The greatest scatter of the twelve quadrats is seen in the smaller plot sizes (1 m^2 and 4 m^2), and then the values are more clustered together, except for the UDR1 averages.

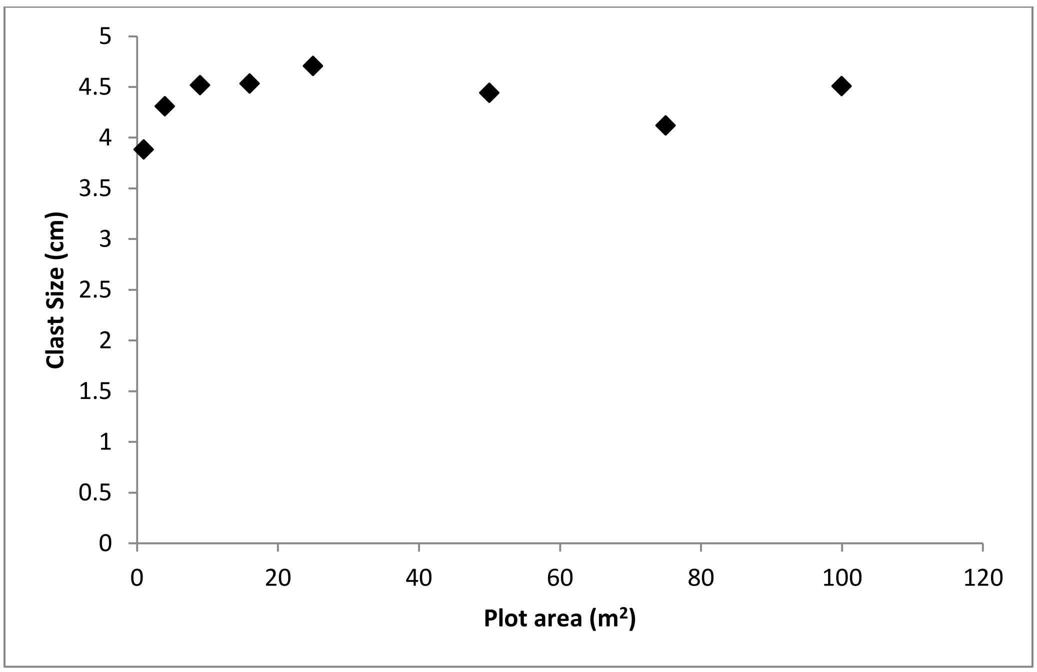

Figure 5.76. Average clast sizes (cm) in relation to plot size (n=12 per quadrat size).

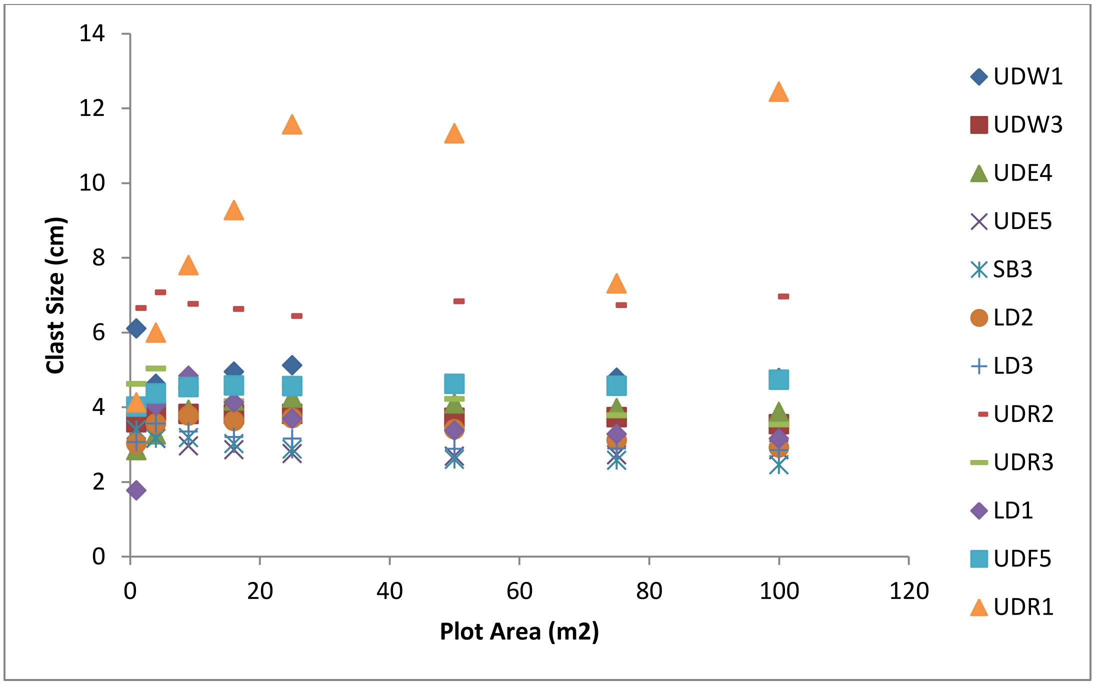

Figure 5.77. Clast size (cm) for each plot in relation to plot size (n=12 per quadrat size).

Table 5.53. Average clast size (cm) for each sub-site assessed in relation to plot size.

Plot Area (m^2)	1	4	9	16	25	50	75	100
UDW1	6.1	4.6	4.8	4.9	5.1	3.6	4.8	4.8
UDW3	3.6	3.9	3.8	3.8	3.8	3.7	3.7	3.5
UDE4	2.9	3.3	3.9	4.2	4.3	4.1	4.0	3.9
UDE5	3.4	3.2	3.0	2.9	2.8	2.7	2.7	2.9
SC3	3.4	3.2	3.2	3.0	2.9	2.6	2.6	2.5
LD2	3.0	3.6	3.8	3.6	3.7	3.4	3.1	2.9
LD3	3.1	3.6	3.3	3.2	3.2	2.9	2.9	2.8
UDR2	6.7	7.1	6.8	6.6	6.4	6.8	6.7	7.0
UDR3	4.6	5.0	4.5	4.2	4.6	4.2	3.8	3.5
LD1	1.8	4.0	4.8	4.1	3.7	3.4	3.3	3.2
UDF5	4.0	4.4	4.5	4.6	4.6	4.6	4.6	4.7
UDR1	4.1	6.0	7.8	9.3	11.6	11.3	7.3	12.4
Average	**3.9**	**4.3**	**4.5**	**4.5**	**4.7**	**4.4**	**4.1**	**4.5**

Soil Penetrability

The lowest average penetrability value of the combined quadrats was 0.86 kg/cm^2 for the 1 m^2 plot and the largest value was 0.95 kg/cm^2 for the 50 m^2 plot (Figures 5.78 and 5.79, Table 5.54). The standard deviation was greatest for the 100 m^2 plot (0.90±0.59 kg/cm^2). The differences among the plot sizes were not significant at the 0.05 level (Greenhouse-Geisser, p<0.213). Most of the values were clustered rather close together for each plot size, however the UDE5 quadrat displayed greater penetrability values than the other quadrats.

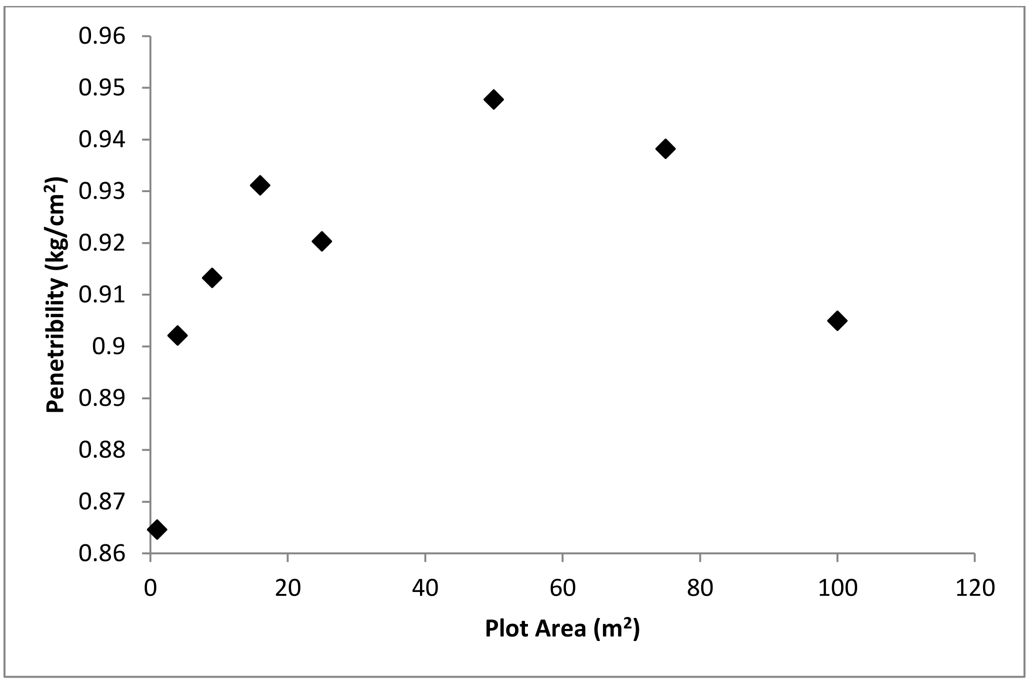

Figure 5.78. Average penetrability (kg/cm^2) in relation to plot size (n=4 per quadrat size).

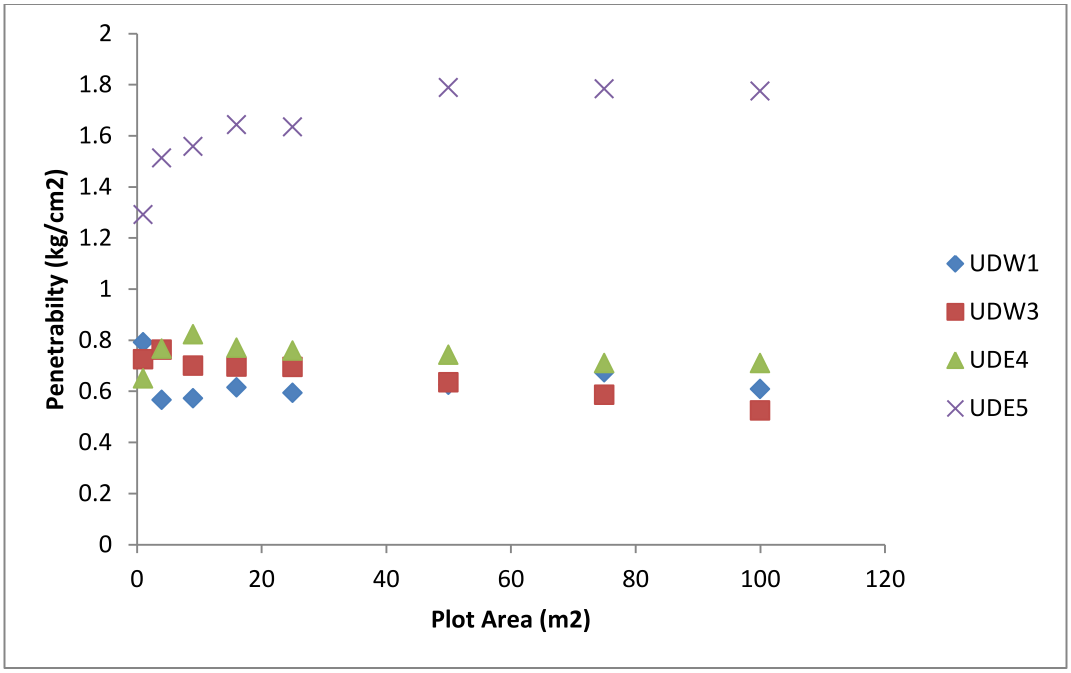

Figure 5.79. Average penetrability values (kg/cm^2) for each sub-site in relation to plot area.

Table 5.54. Average soil penetrability values (kg/cm^2) for each quadrat assessed in relation to plot area.

Plot Area (m^2)	1	4	9	16	25	50	75	100
UDW1	0.79	0.57	0.57	0.61	0.59	0.63	0.67	0.61
UDW3	0.73	0.76	0.70	0.70	0.70	0.63	0.59	0.53
UDE4	0.65	0.77	0.82	0.77	0.76	0.74	0.71	0.71
UDE5	1.29	1.51	1.56	1.64	1.63	1.79	1.78	1.78
Average	**0.86**	**0.90**	**0.91**	**0.93**	**0.92**	**0.95**	**0.94**	**0.90**

Krummholz Density

Average krummholz density for all quadrats combined was lowest for the 1 m^2 plot, which averaged 0.42±0.51 (krummholz/m^2) (Figures 5.80 – 5.81, Table 5.55). The highest krummholz density was found in both the 16 m^2 and 25 m^2 plots, with values of 0.62±0.48 and 0.62±0.41, respectively. The greatest standard deviation was found for the

4 m^2 plot which averaged 0.56 with a standard deviation of 0.56 krummholz per m^2. The lowest standard deviation was found with the 100 m^2 plot, which averaged 0.61 with a standard deviation of 0.29 krummholz per m^2. The differences among the plot sizes were not significant at the 0.05 level (Greenhouse-Geisser, $p<0.255$).

More scatter is seen in the data with plot sizes equal to or less than 25m^2 when comparing averages per quadrat for each plot size. Data in plot sizes 50 m^2 and larger tend to be clustered into two groups. LD1 and LD2 averaged higher krummholz density and were grouped together, whereas the other quadrats were clustered together with slightly lower krummholz densities. However, with plot sizes of 75 m^2 and 100 m^2, UDE5 and SC3 began grouping together, between the higher densities of LD1 and LD2 and lower densities of the remainder of the quadrats.

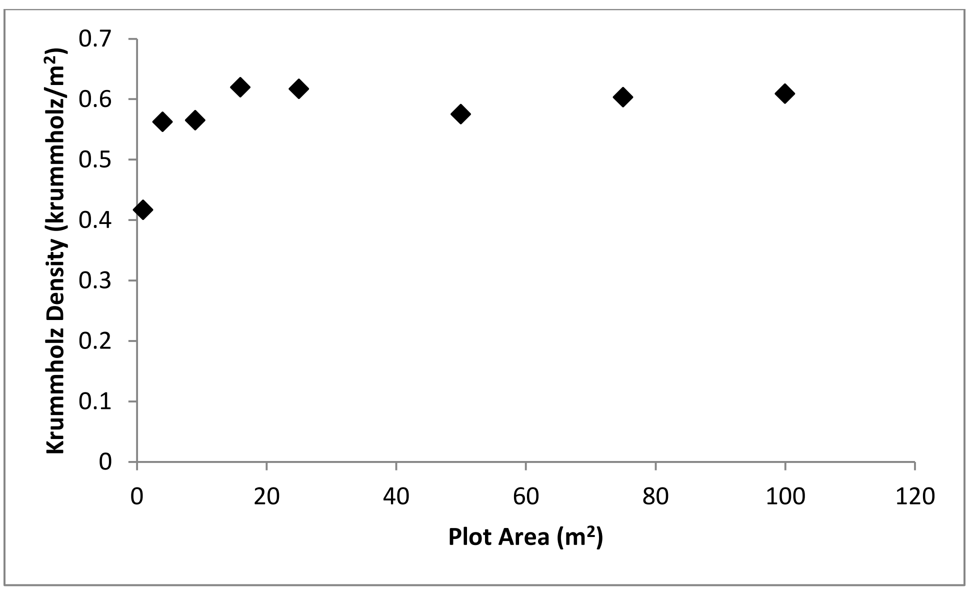

Figure 5.80. Average krummholz density (krummholz/m^2) for all quadrats combined in relation to plot area.

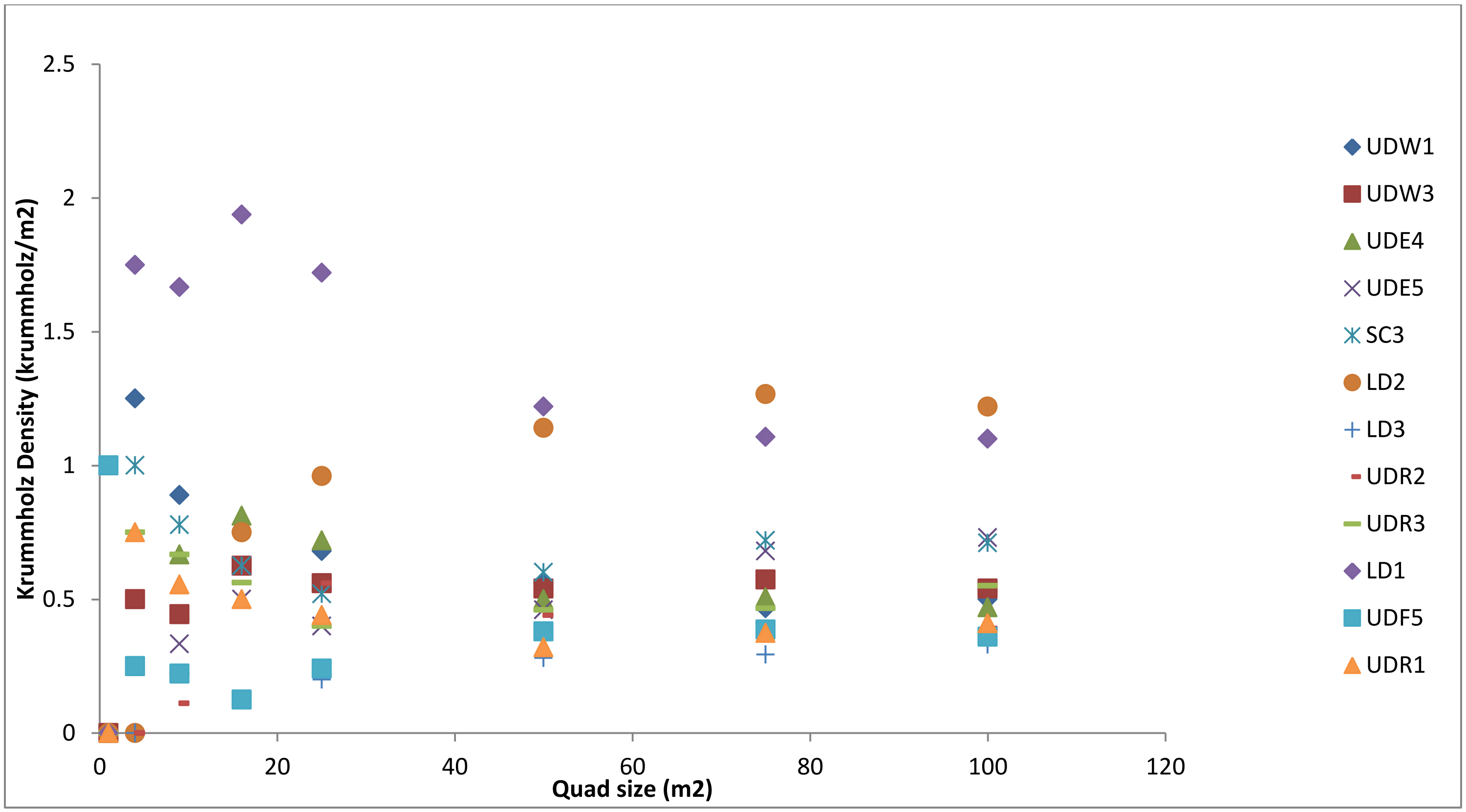

Figure 5.81. Krummholz density (krummholz/m^2) for each site in relation to plot size (n=12 for each plot size).

Table 5.55. Average krummholz density (krummholz/m^2) for each quadrat assessed in relation to plot area.

Plot Area (m^2)	1	4	9	16	25	50	75	100
UDW1	0.00	1.25	0.89	0.75	0.68	0.56	0.47	0.50
UDW3	0.00	0.50	0.44	0.63	0.56	0.54	0.57	0.54
UDE4	1.00	0.25	0.67	0.81	0.72	0.50	0.51	0.47
UDE5	1.00	0.25	0.33	0.50	0.40	0.46	0.68	0.73
SC3	1.00	1.00	0.78	0.63	0.52	0.60	0.72	0.71
LD2	0.00	0.00	0.22	0.75	0.96	1.14	1.27	1.22
LD3	0.00	0.00	0.22	0.13	0.20	0.28	0.29	0.33
UDR2	0.00	0.00	0.11	0.13	0.56	0.44	0.40	0.39
UDR3	0.00	0.75	0.67	0.56	0.40	0.46	0.47	0.55
LD1	0.00	1.75	1.67	1.94	1.72	1.22	1.11	1.10
UDF5	1.00	0.25	0.22	0.13	0.24	0.38	0.39	0.36
UDR1	0.00	0.75	0.56	0.50	0.44	0.32	0.37	0.41
Average	**0.33**	**0.56**	**0.56**	**0.62**	**0.62**	**0.58**	**0.60**	**0.61**

Basal Area

Averages of total basal area for quadrats combined and plotted in relation to plot size reveal a linear direct relationship with a high R^2 value of 0.996 (p<0.001) (Fig. 5.82). Average total basal area was 12.5 cm^2 for the 1 m^2 plot size and increased up to an average of 1297.2 cm^2 for the 100 m^2 plot (Figure 5.83, Table 5.57). UDR1 increased quickly starting at 50 m^2, whereas LD3 and UDR2 remain comparatively low. Basal area for UDE4 increases rapidly beginning with the smaller plot sizes. As would be expected, standard deviations increase with increase in averages and plot sizes (Table 5.58).

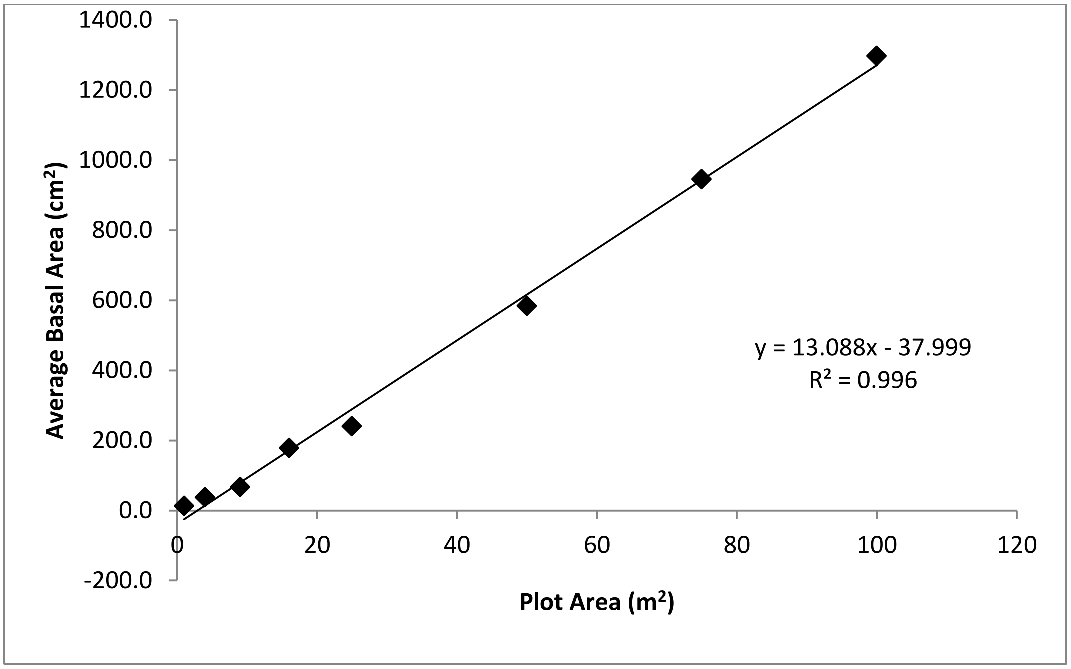

Figure 5.82. Average krummholz basal area in relation to plot area.

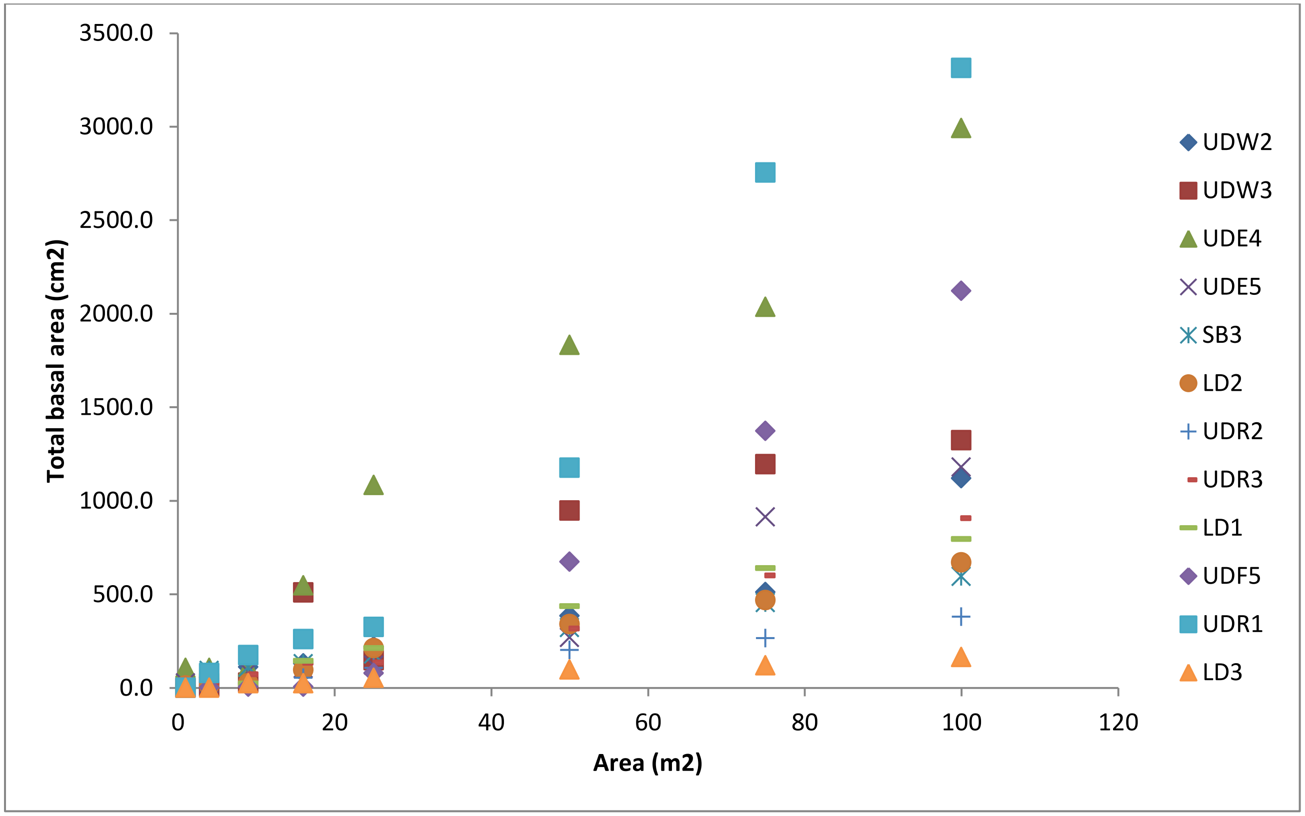

Figure 5.83. Total basal area of krummholz for each site in relation to plot size (n=12 for each plot size).

Table 5.56. Average krummholz basal area (cm^2) for each quadrat assessed in relation to plot area.

Plot Area (m^2)	1	4	9	16	25	50	75	100
UDW1	0.0	30.6	112.6	130.6	220.9	385.8	511.6	1119.7
UDW3	0.0	6.5	27.4	510.2	147.4	947.5	1196.2	1324.5
UDE4	107.5	107.5	112.3	547.9	1083.8	1833.6	2036.5	2992.3
UDE5	26.9	26.9	88.4	102.8	120.3	268.9	914.2	1179.8
SC3	0.0	92.2	110.4	128.9	180.2	322.5	455.5	596.0
LD2	12.3	12.3	25.4	97.1	212.1	341.4	469.3	671.3
LD3	0.0	0.0	23.9	23.9	53.1	97.6	120.7	164.8
UDR2	0.0	0.0	20.6	53.6	107.3	201.7	265.9	381.0
UDR3	0.0	27.8	73.1	133.3	135.9	317.2	601.1	906.4
LD1	0.0	63.3	23.4	143.9	212.5	437.0	640.1	795.5
UDF5	3.1	3.1	6.3	6.3	79.8	673.5	1373.9	2121.6
UDR1	0.0	79.9	175.8	260.2	325.8	1176.8	2755.5	3313.8
Average	**12.5**	**37.5**	**66.6**	**178.2**	**239.9**	**583.6**	**945.0**	**1297.2**

Table 5.57. Averages and standard deviations of total basal area per plot size.

Plot Area (m^2)	Average	1SD
1	12.5	31.0
4	37.5	38.4
9	66.6	53.3
16	178.2	176.5
25	239.9	275.8
50	583.6	504.0
75	945.0	781.8
100	1297.2	1003.4

CHAPTER 6: DISCUSSION AND CONCLUSION

The overall goal of this study was to assess conditions at treeline after recent fires, with particular focus on geomorphic response, associated seedling re-establishment conditions, and biogeomorphic interactions. The results indicated that 1) of the soil variables assessed, several varied significantly within or among sites or between burned and unburned areas; 2) seedling micro-sites were significantly different between burned and unburned areas and seedlings were found to be strongly associated with several variables; 3) significant relationships were found between several soil and vegetation conditions; 4) soil conditions were found to vary in relation to the burned/unburned edge; and 5) quadrat size did not influence the results of several soil and vegetation conditions.

This chapter begins with discussion on the influence of fire within the alpine treeline ecotone in general. Results of soil and vegetation conditions and rock spalling are contextualized in regard to relevant literature and topographic conditions. Subsequently, the concepts of facilitation, edge effects, biogeomorphic disturbance, and multiple causality are applied to the results and the contributions of this study to those ideas are discussed. The discussion section concludes by providing the applied contributions of the results, specifically regarding sampling methods and management implications. Future research and a conclusion section subsequently complete the chapter.

The Influence of Fire on the Alpine Treeline Ecotone

Several studies have addressed the influence of fire on the alpine treeline ecotone or subalpine forest, but these studies were primarily focused on vegetation re-establishment several decades after fire (Coop et al. 2009; Stueve et al. 2009). Likewise, the geomorphic influences of fire are well addressed in the literature (*e.g*. Inbar et al. 1998; Shakesby and Doerr 2006). However, very little research has been performed on the geomorphic influence of fire within the alpine treeline ecotone. The alpine treeline ecotone is an area of distinct topographic, ecologic, and climate interactions compared to many other systems, and response to fire from other systems cannot be assumed to be applicable here. Additionally, a re-occurring theme within the literature regarding fire and geomorphology is the great variability experienced in different environments. Research on the influences of fire on geomorphic conditions is lacking for high elevations within the Northern Rocky Mountains and this study provides data collected with a multi-faceted approach to assess alpine treeline conditions after fire.

My results, overall, supported some of my hypotheses that fire significantly alters soil conditions, but other soil variables were not significantly affected by fire. Table 6.1 summarizes the variables that were significantly different between the burned and unburned areas. Soil conditions of soil loss, duff depth, penetrability, clast size, N, and P were found to be significantly different in burned areas compared to adjacent unburned areas.   Within burned areas, significant differences in soil penetrability, clast size, ESD, and krummholz DGL, density, and basal area averages were found among the three sites (Table 6.2). Few variables, however, differed significantly within each site (Table 6.3). At the micro-site scale, ESD and clast size varied significantly between burned and unburned areas, but soil penetrability did not (Table 6.4).

Soil loss was distinctly evident within all burned sites and was one of the major differences between burned and unburned areas. The alpine treeline ecotone is often very exposed and contained lower-lying vegetation, factors that enhance soil erosion. Soil penetrability values were used as a proxy to gauge locations that contained less fine sediments, presumably because of erosion. Lower penetrability values appeared to correlate with a greater presence of fine sediments, and likewise, areas that did not contain much fines, produced higher penetrability values. Some of the highest penetrability values were found on the lowest sloped areas within a site, whereas the higher slopes tended to have lower penetrability values. The lower sloped sites were also those that were most exposed, whereas the steeper slopes often were protected to some extent by the slope itself. This finding counters that of many studies performed after fire that concluded steeper sloped sites experienced the most soil erosion. These studies, however, often assessed much larger areas, and I evaluated fine-scale variations. My results indicating that fine-scale variations in topography are influencing soil conditions are similar to the findings of Butler et al. (2004), who also assessed fine-scale soil patterns within the alpine treeline ecotone. They found that turf-exfoliated sites contained significantly lower penetrability values compared to other micro-topographic features, and concluded that such sites would be more amenable to seedling establishment. Similarly, this study indicated that erosion was mitigated by the protective action of the slopes itself, leading to lower penetrability values in relation to micro-topographic features.

Soil loss is of particular interest within the alpine treeline ecotone, and the burned areas experienced extensive soil loss compared to the unburned areas, which provided no indications of soil loss. The loss of vegetation cover and a duff layer resulted in the soil being exposed to wind and water erosion. Soils are often very slow in developing within the treeline ecotone, and a loss of even several centimeters could take centuries to be replaced. Headwaters of streams often form within or near the treeline ecotone and erosion from treeline could provide an influx of sediments into streams that are fed by areas that burned. Also the loss of soil will result in a redistribution of sediments and nutrients that may influence plant re-establishment patterns.

Table 6.1. Comparison of soil conditions between burned and unburned areas.

Variable	**Significant?**
Soil loss	Yes
Duff depth	Yes
ESD	No
Penetrability	Yes
Clast Size	Yes
pH	No
Conductivity	No
N	Yes
P	Yes
K	No
Ca	No
Mg	No
S	No
Na	No

Duff depth averages were also found to significantly vary between burned and unburned areas. Duff within the alpine treeline is of interest because of its role in seedling establishment and carbon storage (Malanson et al. 2009). The duff layer in unburned sites was often a thick mat of needles covering the ground surface. This layer can serve as a protective layer for seedling establishment, providing a surface away from frost-heaving action (Swanson 1978). Duff also serves as a reservoir of biomass and has been found to be significantly higher in tree islands within the treeline ecotone compared to alpine tundra (Sanscrainte et al. 2003). Duff helps protect the soil from erosion and reduces its exposure to the desiccating effects of wind and solar radiation. Burned areas, however, lacked a duff layer, which was likely incinerated during the fire.

Table 6.2. Variables from burned areas compared among three sites.

Variable	**Significant?**
Duff Depth	No
ESD	Yes
Penetrability	Yes
Clast Size	Yes
pH	No
Conductivity	No
N	No
P	No
K	No
Ca	No
Mg	No
S	No
Na	No
Krummholz density	Yes
Krummholz DGL	Yes
Krummholz basal area	Yes

Table 6.3. Comparison of variables within each site, among or between sub-sites.

Variable	Upper Divide	Lower Divide	Swiftcurrent
	Significant?		
ESD	No	No	No
Penetrability	Yes	No	No
Clast Size	Yes	No	Yes
pH	No	No	No
Conductivity	No	No	No
N	No	No	No
P	No	No	No
K	No	No	No
Ca	No	No	No
Mg	No	No	No
S	No	No	No
Na	No	No	No

Table 6.4. Comparisons of variables between micro-sites.

Random	Burned vs. Unburned
Variable	**Significant?**
Clast Size	Yes
Penetrability	No
ESD	Yes

Rock clasts can influence soil conditions and vegetation establishment patterns, and burned areas contained significantly larger particle sizes. A loss of organic matter and duff exposed the often underlying gravel, and rock spalling produced additional rock particles to the ground surface. Chambers et al. (1991) found the diaspore entrapment increased as soil particle size increased. Seed entrapment by soil and rock particles in windy environments on exposed soil can have a significant influence on vegetation patterns (Reichman 1984; Chambers et al. 1991). Cerdà (2001) focused a study on coarse particles (>2 mm) and found that surface particles had significant influence on soil erosion. Rock particles delayed runoff, reduced erosion, and increased infiltration.

The role of ESD in influencing seedling establishment at treeline appears to change after fire. Malanson et al. (2002) did not find any spatial association of ESD within the alpine treeline ecotone, and concluded that woody vegetation would not be advancing in relation to patterns of ESD. The results of this study, similarly, did not find significant differences between burned and unburned ESD values. However, seedlings re-establishing in burned areas were found to be significantly associated with higher ESD. Stoniness of the soil, which is indicated with ESD, can potentially influence soil factors related to vegetation establishment (Poesen and Lavee 1994).

Soil chemistry is of particular interest within the alpine treeline ecotone because of the changes in vegetation cover from the subalpine forest up into the alpine tundra and potential responses to changes in climate (Malanson and Butler 1994; Cairns 1999; Malanson 2001; Liptzin and Seastedt 2009, 2010). Soil nitrogen and phosphorus increased significantly after fire. This trend commonly occurs after fires because the incineration of vegetation results in the addition of nutrients to the soil. The addition of nitrogen is of particular interest because the alpine treeline serves as a nitrogen deposition area (Liptzin and Seastedt 2010). Fire may further raise nitrogen amounts in this often nitrogen-saturated zone.

McKenzie and Tinker (2012) found that fires greatly altered the tree composition in Glacier National Park. They found that tree re-establishment densities maintained those of pre-fire conditions but that shifts in species occurred with quaking aspen and lodgepole pine becoming much more common. Seedling species densities change somewhat within the ecotone as well. Firs remained the most common species after fire as before fire, but the proportion of pine seedlings was greater in burned areas compared to adjacent unburned areas. Pine is intolerant to moderately intolerant of shade, and therefore may benefit from the loss of krummholz canopy. Spruce seedlings were slightly greater in burned areas compared to unburned. Soil moisture is often an important factor of tree re-establishment after fire because conditions after fire tend to increase soil desiccation. Therefore, moisture preferences of seedlings can strongly dictate the species re-establishment patterns and densities (Denslow and Battaglia 2002). Fir is relatively quick growing and can tolerate soils that do not retain moisture better than spruce seedlings, which are slow growing and prefer more soil moisture.

Topographic Variations

Seedling density often lessens with higher elevation within sub-alpine and treeline zones (Agee and Smith 1984; Coop et al. 2010). Although based on a small sample area, this pattern was found at Swiftcurrent, in which the only seedlings present were those within the lowest elevation quadrat. The topography at Swiftcurrent was generally uniform, other than change in elevation, and was a great contrast to the topographic complexity present at Divide Mountain. Interestingly, more seedlings were found at the higher elevations than the lower elevations at Divide Mountain. This finding may indicate several factors contributing to this pattern – 1) topographic variations at the sub-site scale (gullies, ridges) are exerting stronger influence than coarser scale factors (elevation), 2) burn patch dynamics and seed source distances may be different between the two areas, 3) fine-scale features (soil conditions, boulders) are overriding coarse scale topography, and/or 4) the elevation gradient that I assessed was not extensive enough to compare to other studies on seedling re-establishment and elevation.

The pattern of seedling establishment sites on Divide Mountain indicates that topographic variations at a fine-scale (100 m^2) may override the influences of topography at coarser scales (>1 km^2) for seedling establishment after fire. General declines in seedling establishment with increasing elevation have been attributed to lower reproductive outputs at higher elevations. Cold temperatures, frost damage, winter desiccation, and dry soil conditions can also impede seedling establishment within higher elevations (Germino et al. 2002). All sites within treeline on Divide Mountain probably experience these harsh conditions, however, fine-scale differences in topography may enhance or mitigate the severity and extent. For example, UDW, which is a gully and contained the most seedlings, may facilitate the retention of soil moisture and lessen the desiccating effects of wind. Numerous studies have found that fine-scale factors overcome coarse-scale controls in regard to seedling establishment within treeline areas that have not experienced fire (Walsh et al. 1992; Cairns and Malanson 1998; Butler et al. 2004; Resler et al. 2005; Resler 2006; Malanson et al. 2007; Butler et al. 2009; Malanson et al. 2009). Fine-scale topographic features can produce micro-sites more favorable to seedling establishment, such as providing areas of protection from wind or facilitating the retention of soil moisture.

It must be kept in mind, however, that this comparison is all within the treeline ecotone. From personal observation, lower elevation seedling establishment and growth is much more advanced than that within treeline. Other studies that have found a lower density of trees with gains in elevation generally cover a greater elevation gradient

(Stueve et al. 2009) that may extend from treeline down through the sub-alpine forest. Therefore, micro-site influence can only override coarse-scale controls within the context of a given environment, in this case, that of the alpine treeline ecotone (Resler 2006; Butler et al. 2009).

Distance to seed sources and burn patch patterns may also be a controlling factor (Agee and Smith 1984; Malanson 1997; Stueve et al. 2009). At the higher elevations on Divide (Upper Divide), the burned areas were smaller than those at slightly lower elevation (Lower Divide). Smaller patches have greater edge density and more burned area is in closer proximity to unburned vegetation, which can enhance seedling re-establishment. Coop et al. (2010) found that most tree species declined in density with increasing distance from the burn edge. For seed-generating species, the burn interior is often farther from seed sources and provides a more inhospitable location for germination. The larger burn patches at Lower Divide may result in more burned area farther from seed sources than the smaller patches at Upper Divide.

Fine-scale variations between Lower Divide and Upper Divide may also influence seedling establishment patterns. Several conditions associated with seedlings (Table 6.5) were found to be more common at Upper Divide. Facilitative objects, which were primarily boulders, were found to be significantly closer to seedlings, and Upper Divide displayed a more extensive covering of boulders. Lower soil penetrability values and higher ESD were also found to be in strong association with seedling establishment. Average soil penetrability was lower in the Upper Divide sub-sites of UDW, UDF, and UDR (which contained the most seedlings) compared to Lower Divide, and ESD averages were greater at Upper Divide compared to Lower Divide. These more favorable micro-site conditions at Upper Divide may create fine-scale areas amenable to preliminary seedling establishment after fire.

Table 6.5. Comparison of variables between seedling to random micro-sites in burned areas.

Burn	Random vs. Seedling
Variable	**Significant?**
Closest Object	Yes
Avg. Object Distance	Yes
Clast Size	No
Penetrability	Yes
ESD	Yes

The argument that fine-scale factors are exerting significant influence on seedling establishment patterns, to the extent of over-coming topographic characteristics of slope and aspect, is further supported by the fact that Lower Divide actually was positioned on what is often considered a more favorable aspect. Slope aspect has been found to influence vegetation re-establishment after fire (Agee and Smith 1984), and north-facing slopes generally are more conducive to vegetation re-establishment because they retain more soil moisture and experience less erosion after fire (Rebertus et al. 1991; Stueve et al. 2009). However, the result that fewer seedlings were found on the north-aspect of the Lower Divide sites counters this trend. Upper Divide sub-sites were positioned on east and west aspects, and west aspects particularly, are generally drier than east and north aspects. North aspects, however, may experience a shorter growing season as the snowpack persists longer, but conversely, snow cover would protect seedling from exposure during winter (Körner 2003).

Fine-scale topographic variations, however, may also capture snow and provide areas of protective snow cover (Körner 2003; Walsh et al. 2009), and therefore, mitigate the perceived benefits of a north aspect. Features, such as gullies and leeward areas of a ridge or slope, can capture and retain more snow than areas of little topographic variability. In regard to seedling establishment within the Upper Divide sites, the areas that contained the most seedlings were also those that were leeward of a slope or ridge, or within a gully. Snow serves as a protective blanket to seedlings during the winter months, and exposure to wind, desiccation, and ice may be fatal to seedlings (as well as established krummholz) (Holtmeier 2009). Therefore, even though the north aspect of Lower Divide may provide a greater protective layer of snow in winter than the overall Upper Divide area, the topographic variations at Upper Divide can serve a similar purpose in providing fine-scale site conditions that allow for the accumulation of snow.

Fire as an Erosional and Weathering Agent

Erosion rates and total soil loss vary greatly after fires and may correspond to vegetation type, topography, climate, or weather conditions (Robichaud and Brown 2000; Benavides-Solorio and MacDonald 2001; Spigel and Robichaud 2007; Sass et al. 2012b). Difference in ground surface depth provides indications that a significant amount of soil was lost from the treeline area after fire. This process is especially important to understand within the treeline ecotone because headwaters often are located near or begin in this area. Nutrients are highly variable across the alpine ecotone, and soil loss can result in a redistribution of soil nutrients.

My method to estimate soil loss (measuring soil pedestals, differences in ground surface, and root exposure) was probably not the most accurate estimation, however, it has been used by others to estimate changes in soil depth (Harden, 1988; Pérez 2007) and was logistically possible, unlike the placement of erosion fences or pins immediately after fire. Also, many studies on soil erosion employ rainfall simulators, but wind exerts a significant influence on the treeline ecotone. Rainfall simulation, therefore, would not have provided an accurate account of erosion rates or processes. Also, because this study was conducted several years after fire, initial sediment load most likely could not be replicated.

Rock spalling at Upper Divide was extensive. Blackwelder (1927) first called attention to the great influence that fire has on rock weathering, but also noted the importance of an environment and conditions conducive to boulder spalling. For example, certain rock types are more susceptible to spalling and the presence of woody vegetation is important. Associations between boulders and krummholz were necessary for boulder spalling. Boulders were witnessed to have spalled near krummholz, whereas boulders within the same rock pile but not in association with krummholz had not spalled. Vegetation provides the fuel to generate the heat that results in the boulder spalling.

Great contrast was witnessed between Divide Mountain and Swiftcurrent Mountain. The process of spalling can include the expansion of moisture within the rock, and Swiftcurrent may be much drier on its south-aspect slope compared to Divide Mountain. The primary difference, however, may be attributed to variations in lithology of the two sites. Grinnell argillite is found at Swiftcurrent, and this rock is characterized by fracturing into platy pebble- and cobble-sized fragments. Altyn limestone, found at Divide Mountain, however, commonly weathers into boulders. Upper Divide experienced more spalling than Lower Divide, and this finding is probably because Upper Divide was closer to the source of rock, and therefore, also had a greater ground cover of boulders.

Facilitation after Fire

The loss of krummholz cover in burned areas creates conditions similar to the alpine tundra, including enhancing solar radiation and wind velocity and lessening snow deposition, all factors that also result in decreased soil moisture (Bollinger 1973). These changes will make it more challenging for seedlings to become established. Bollinger (1973) found that seedlings established in greater densities in areas that had burned when a "nursemaid" object – often a shrub that offered protection from the elements – was present. In Bollinger's study in the Colorado Front Range, willow often served as the nursemaid plant, however, willows are not found the alpine treeline ecotone in the Northern Rockies and rocks, instead, frequently served in this role. Boulders have been found to serve as facilitative objects within the alpine treeline ecotone without fire disturbance (Resler et al. 2005). Boulders offer protection from wind, moderate temperature, capture and retain moisture, and provide shade without using resources as another plant likely would.

This study found that facilitative objects are important to re-establishment of seedlings after fire within the alpine treeline ecotone. Average distance to the closest object averaged 17 cm, which was a similar distance that Coop and Schoettle (2009) found for bristlecone pine seedlings (16.6±28.1 cm). In Coop and Schoettle's (2009) study, the closest random object was located an average of 26.2±35.5 cm, which was closer than the 47.4 cm that I found for the distance between random plot centers and the closest object. Limber pine seedlings averaged only 11.0±12.9 cm from the closest object, but again, the distance between random plot centers to the closest object was only 23.3±32.7 cm. The difference between seedlings and the closest object, and random plot centers and the closest object was 30.4 cm within the alpine treeline, compared to 9.6 cm (bristlecone pine) and 12.3 cm (limber pine) in Coop and Schoettle's (2009) findings.

Several factors may contribute to the differences in distance averages between seedlings and random plots found in the two studies. Coop and Schoettle (2009) defined their study area as being "sub-alpine," whereas this study was situated within treeline. Treeline often experiences harsher climatic conditions, lending to the idea that facilitative objects will be more important to seedling establishment than within the sub-alpine zone. The differences in our results may also be related to the number of years since fire for each study. Their study took place several decades after fire, whereas this study was conducted within 10 years of fires. This difference may indicate that facilitative objects are more important for initial seedling establishment and subsequently decrease in importance over time.

I also used a slightly more conservative definition of "object." I only considered objects that were at least 10 cm in at least three directions, whereas they used objects at least 10 cm in two dimensions. Some objects were 10 cm by 10 cm, but much less in the third dimension, which may have been serving as the "protective side" for a seedling. Therefore, to take into account variations in object shape and retain consistency, I used only objects at least 10 cm in three dimensions. In retrospect, I would still consider objects in three dimension, but for a study conducted soon after fire with seedlings still very small (<10 cm in height), I would consider an object that is at least 5 cm on three sides.

Fir was significantly closer to the first object compared to pine seedlings. Pine seedlings are shade intolerant, and therefore may not benefit from shadows cast by objects as fir seedlings may. No significant differences were found among seedling species in regard to distance between the seedling and the second and third object. The second and third objects were often not close enough to the seedlings to provide direct influence, such as shade. However, each species was found to be significantly closer to second and third objects in comparison to the distance between random plots and second and third objects. This result indicates that near-by objects, not only those that are closest, may also be important for seedling establishment. Second and third objects may provide additional protection to seedlings, including influencing wind dynamics and, potentially, seedling dispersion and deposition. The presence of one object may also indicate a greater likelihood that another object or other objects may be nearby.

Resler et al. (2005) found that seedlings on Divide Mountain, before it burned, were in significant association with nurse objects. My result in not finding any significant differences in comparing the distance to object between seedlings in unburned areas to random plots without seedlings is probably related to several factors. Resler et al. (2005) sampled tree islands on Divide Mountain in which boulders likely facilitated the island’s establishment. However, tree islands burned in the 2006 fire (Fig. 6.1) and, except for one individual, did not contain re-established seedlings. Therefore, seedlings in tree islands were not measured in this study in relation to objects. Unburned patches adjacent to burned areas were not tree islands, but rather often larger patches of krummholz. Seedlings in such a setting would not be as dependent on the protective effects of a boulder within a covering of krummholz as they are within islands. Also, our sampling methods were different and tailored to address different research questions. Resler et al. (2005) placed larger quadrats (30 x 30 m) spread out in five different elevation zones, whereas this study was limited to locations where fire had occurred. They also defined the presence of shelter as within 2 cm of a conifer, and I was interested in seedling distance to nearest objects, a methodology adapted from Coop and Schoettle (2009).

Figure 6.1. A burned tree island at Upper Divide Mountain. Photograph taken August 2012.

Not only were objects apparently aiding in seedling re-establishment, but krummholz roots also facilitated and influenced seedling patterns. Roots collected downslope sediment movement and/or prevented soil loss (Fig. 6.2), and several seedlings were noted to have established immediately upslope of a root. Surface levels were often greatly reduced on the downslope side of a krummholz root and the surface scoured of fine sediments.

Figure 6.2. A charred krummholz roots capturing sediments and/or preventing soil from eroding, and apparently capturing a seed and allowing for its establishment. Photograph taken at the Upper Divide sub-site of UDF in August 2012.

Facilitative objects can offer an important source of shade for seedlings established in the exposed burned areas. Relative sun exposure was higher in burned plots compared to unburned random plots. Solar radiation is especially intense at higher elevations because sunlight does not travel through as much atmosphere as it does to reach lower elevations, and therefore it is not as diffused. Also areal flux and radiation extremes are greater in higher elevations (Körner 2003). Solar radiation can be detrimental to seedlings establishing at treeline after fire (Bollinger 1973). Bollinger (1973) found seedling mortality greatly increased after two summers exposed to the sun in comparison to seedlings in shade. Intense radiation coupled with a loss of vegetation can result in a harsh environment for seedling establishment. Objects can greatly mitigate the intensity of solar radiation (Greenlee and Callaway 1996; Bader et al. 2007), and seedling establishment patterns found in this study support this relationship (Figures 6.3 and 6.4).

Figure 6.3. A seedling established beneath the overhang of a boulder. Photograph taken at Upper Divide in July 2012.

Figure 6.4. A seedling (location indicated by the arrow) under a leaning burned krummholz, which is providing shade to the ground surface beneath it. Photograph taken at Upper Divide in July 2012.

Even though UD has conditions not favored by seedlings (lower ESD and higher soil penetrability values) and contains the least pre-fire krummholz density, the greatest seedling density was found at UD (0.05 seedlings/m^2) compared to SC and LD (both contained a density of 0.01 seedlings/m^2) (Table 6.6). Of all the variables assessed, the presence of facilitative objects (particularly rocks) appears to be an important factor of seedling establishment after fire. Interestingly, Swiftcurrent had a greater density of pre-fire krummholz compared to the sites on Divide Mountain, even though the Swiftcurrent site is located on a south-facing slope. This finding may be attributed to greater geomorphic disturbance on Divide Mountain, especially rock fall and freeze-thaw action.

Table 6.6. Relative site comparison for various variables.

Most Seedlings	UD	Least Seedlings	LD and SC
Highest Krummholz Density	SC	Lowest Krummholz Density	UD
Largest Particle Size	UD	Smallest Particle Size	LD and SC
Least Penetrable Soil	UD	Most Penetrable Soil	SC
Highest ESD	SC	Lowest ESD	LD and UD
Most Sun	LD	Least Sun	SC

Edge Effects

The concept of edge effects is well established in ecology, and studies address such factors as soil temperature and moisture in relation to an ecological edge (*e.g.* Camargo and Kapos 1995; Chen et al. 1999). However, the application of this theory to geomorphologic variables and how they may change in relation to an ecological edge is lacking. Butler (1979) and Malanson and Butler (1984) did address the idea of geomorphic-ecologic interaction in regard to disturbance edges, but did so from the perspective of a geomorphic disturbance and the resulting vegetation patterns in relation to the disturbance edge. Butler (1979) found the longitudinal pattern of vegetation re-establishment in avalanche paths to coincide with the intensity of impact the vegetation would experience within the path. Malanson and Butler (1984) assessed vegetation across avalanche paths from the trimline to the ravine and found three distinct zones, likely as a result of avalanche damage and topography. Their results highlight the potential for interactions between geomorphic and ecologic edges and the resulting patterns produced.

Soil surfaces, especially those exposed by a disturbance, are subjected to the forces of wind and precipitation, and the presence of vegetation will probably influence that relationship as it affects wind and precipitation velocities. Wind is an especially strong influence on dry, exposed soils (Ravi et al. 2009), such as those after fire. My results did indeed indicate that a pattern, with respect to the burned/unburned edge, exists among the soil conditions. Wind erosion and deposition are probably a strong influence on the burned patches because of the exposed soil, dry conditions, harsh climate, and limited vegetation (Cannon et al. 1998; Ravi et al. 2006, 2009).

Soil penetrability values were lower near the edges, as hypothesized, and assumed to be the result of sediments blown from the burned area and then deposited when the wind hits the unburned vegetation. However, I did not expect to see such complex patterns with soil conditions across the burned patch. I expected to find high soil penetrability values, larger clast sizes, and lower ESD in the center of the burned patch as a result of greater soil loss within the more exposed burned interior. Instead, however, soil penetrability and ESD followed a roughly undulating patterning, with penetrability values highest and ESD values lowest, adjacent to the edges and within the center. This condition may be attributed to wind speeds and its interaction with the burned/unburned edge. Wind may dip down towards the ground surface, coming from the unburned vegetation, after it has crossed the edge, resulting in the areas adjacent to the edge region experiencing the strongest surface winds. However, clast sizes did display an expected pattern, in that larger sizes were found in the center of the burned patch and smaller sizes towards both edges.

Biogeomorphic Disturbance

The interactions between vegetation and geomorphic factors are extensive. Seedling site preferences included areas of lower soil penetrability, higher ESD, and the presence of facilitative objects. These conditions were influenced by topographic patterns and pre-fire krummholz density. Post-fire seedling establishment may follow a self-organized pattern in relation to geomorphic response patterns. The harsh conditions within treeline enhance the interactions between vegetation and geomorphic conditions (Butler et al. 2004; Resler et al. 2005; Butler et al. 2007; Zeng et al. 2007), and fire appears to further strengthen this relationship (Sass et al. 2012b)

Biogeomorphic response to a disturbance is often difficult to quantify because of the varying response times of vegetation and geomorphic factors (Viles et al. 2008; Sass et al. 2012b). For example, erosion may be greatest soon after a fire (1-2 years), but then decrease as sediment becomes limited and as vegetation begins to establish (Cerdà and Lasanta 2005). In alpine areas, however, Sass et al. (2012b) found that erosion was still occurring at greater rates in burned areas 60 years after fire compared to unburned slopes. It may be several years or decades for vegetation to become established, and therefore, the initial geomorphic patterns after fire will likely be mitigated by the time vegetation patterns are evident. Also, the change between primarily geomorphic response and primarily vegetation response will probably take place at different times in respect to fire within different environments. Weather patterns, vegetation types, and topography will likely influence this shift. The diagram below (Fig. 6.5) shows a simplified idea regarding the shift from primarily geomorphic instability (black lines) to primarily vegetation instability (green lines). The dashed lines indicate phases of change between geomorphic and vegetative instability as the process shifts from geomorphic to vegetative instability. The length of time for each phase (each line segment) will vary depending on environmental factors present at different locations.

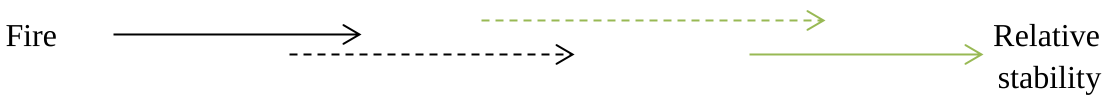

Figure 6.5. A simple diagram indicating a shift from primarily geomorphic response after fire to primarily ecologic response after fire.

Spatial variations may also be difficult to quantify because of the differences in spatial extent and patterns of vegetation and geomorphic factors (Renschler et al. 2007; Sass et al. 2012b). I attempted to address these issues by selecting sites that are probably shifting between geomorphic stability and vegetation re-establishment. Vegetation re-establishment is often slower at the higher elevations of alpine treeline compared to lower elevation areas. Seedlings have likely established within the past one to three years, but vegetation cover was not extensive enough to significantly impede geomorphic and soil processes.

Multiple Causality

The resulting patterns and conditions present at the fine-scale after fire within the alpine treeline appear to be the response of abiotic and biotic variables to feedbacks between vegetation and soil conditions, interactions between topography and weather conditions, and influences of pre-fire vegetation patterns. Similar to the findings of other studies on treeline, these results indicated that meso-scale variables (topography, wind) exert significant influence on fine-scale factors (soil conditions), which, therefore, will influence seedling establishment patterns (Resler 2006; Butler et al. 2009). Seedling establishment patterns will, in turn, influence meso-scale vegetation patterns and wind dynamics. The movement of fire into treeline was likely influenced by both the presence of krummholz and the topography, in addition to wind patterns experienced during the burn. The gullies in Lower Divide, for example, served as protection for the establishment of krummholz, but subsequently, likely facilitated the movement of fire upslope within the gully.

Re-establishment rates and patterns after fire appear to be influenced by topographic variations. UDW may become even more densely covered in krummholz with the addition of nutrients into the soil, whereas, krummholz establishment on UDE will likely be retarded for much longer. Lower Divide and Swiftcurrent did not contain as much topographic variations as Upper Divide. Lower Divide does not appear to be as exposed as Upper Divide overall, but it also did not contain micro-scale features that could potentially serve as protective locations for seedling establishment. The one seedling that was found in Lower Divide North was growing up through the tangle of roots of a burned krummholz (Fig. 6.6).

Figure 6.6. Seedling in LDN. Divide Mountain, July 2011.

These results indicated that simplifying krummholz re-establishment patterns to aspect, degree slope, or elevation will not be accurate. Sub-alpine studies have found significant correlations with such variables, but the topographic complexity at the fine-scale within the alpine treeline appears to override meso-scale factors of aspect and elevation (when comparing treeline to treeline, not in the context of comparing treeline to subalpine) (Sass et al. 2012b). This study supports the general finding by many studies on fire disturbance that fire exerts wide variations on a landscape.

Temporal patterns of each site's response will likely change. Lower Divide and SC showed more homogenous krummholz establishment, whereas pre-burn patterns at UD were patchier. Once establishment begins to occur at LD and SC, subsequent establishment may become consistent. Although, Upper Divide had the greatest initial re-establishment density, it may begin to lag in re-establishment rates because of the greater mosaic pattern of burned patches and alpine tundra vegetation. It may also take more time for tree islands to become established again, compared to larger krummholz patches because of the farther distance to seed sources and smaller area of pre-burn krummholz inhabitation.

Site-specific conditions exerted notable influence on the results found in this study. Seedlings were significantly closer to objects compared to random micro-plots, and rocks served as the most common type of object. Objects were closest to seedlings at the Upper Divide site compared to Lower Divide and Swiftcurrent. Objects may be one of the reasons contributing to the greater density of seedlings at the Upper Divide site. Upper Divide contained the greatest covering of boulders because it is situated downslope of a rock face made of sedimentary rock that is breaking off and falling down into (and below) the alpine treeline ecotone (Fig. 6.7). Lower Divide is also on Divide Mountain, but it is located farther downslope and on a more gentle degree slope, conditions not as conducive to the input of boulders. Therefore, not only do variations in topography, and topography's relationship to weather patterns, influence post-fire conditions, but also the pre-existing disturbances, lithology, and relation to topographic features (Walsh et al. 1992).

Figure 6.7. Boulders at Upper Divide. Photograph taken August 2012.

Soil conditions and seedlings appear to exhibit a self-organizing pattern with respect to both abiotic and biotic variables. Strong indications of relationships between vegetation and soil conditions were evident. Seedlings were significantly associated with the near-by boulders and several soil conditions, soil conditions changed in regard to distance from the vegetation burned/unburned edge, and the loss of vegetation from fire significantly altered soil and geomorphic factors. The mere presence of krummholz resulted in the potential of burning, and the vegetation fueled the fire to cause direct changes to the soil, and then the loss of vegetation subsequently can allow for greater soil exposure and loss. Likewise, the soil conditions will subsequently influence seedling establishment patterns. This reciprocal relationship indicates that self-organization may be occurring (Zeng et al. 2007). The treeline ecotone contains such harsh conditions as to enhance the dependence and importance of micro-scale patterns, climates, and feedbacks (Butler et al. 2004; Resler 2006; Zeng et al. 2007).

Greater krummholz density corresponded to lower penetrability values. This result is likely because more krummholz helps prevent soil loss and acts to capture sediments, therefore increasing the fine particles on the ground surface and resulting in lower penetrability values. Lower erosion classifications were found to be associated with greater herbaceous vegetation cover. I cannot determine which may have been in place first and resulted in the other, but less erosion would likely benefit herbaceous vegetation establishment, and in turn, more herbaceous vegetation would likely prevent erosion. Erosion and burn severity were directly related. Areas that experienced more intense fire may contain less herbaceous vegetation re-establishment as well as less krummholz. Lacking these features may result in higher erosion because not as much vegetation is in place to capture and prevent sediment loss. However, contradictory to this explanation, percent vegetation cover and burn severity do not share a significant relationship. Soil hydrophobicity after fire may alternatively have contributed to this relationship between erosion severity and burn severity. Soil that experienced more intense heat may have become hydrophobic and contributed to erosion.

Sampling Methods

The results did not indicate significant differences between plots sizes of 1 m^2 up to 100 m^2 in particle size distribution, penetrability, or krummholz density. Total basal area similarly showed a strong correlation with plot size (R^2=0.996), with higher total basal area as plot size became greater. In terms of these variables, data could have been collected using 1 x 1 m quadrats rather than 5 x 20 m quadrats and the average of the results would not have significantly differed. Soil conditions were apparently uniform within a given 100 m^2 area. Malanson et al. (2002) found a similar patter with ESD in the treeline ecotone. Krummholz was somewhat patchy, with some 1 m^2 and 4 m^2 plots not being large enough to capture any krummholz, but other 1 m^2 and 4 m^2 plots did and therefore, the averages came out to not vary significantly. Although average values did not vary significantly, standard deviations tended to become smaller with larger plot sizes as a result of greater sample sizes.

However, placement of the quadrats may result in different outcomes based on the variations found across the burned patch. Quadrats measuring 1 x 1 m were positioned along a transect and were found to differ significantly in regard to position along the transect and the burned/unburned edge. Average clast size, soil penetrability, and ESD varied significantly in relation to position along the transect, and therefore, placement of a quadrat may potentially influence the results. For example, sampling from near the center of a burned patch would likely result in a higher average clast size than sampling close to the edges. Variations in ESD and soil penetrability were not as simply and directly related to edge position but rather undulated across the burned patch. For both variables, however, placement of a quadrat close to the edge would likely provide values not representative of most of the interior of the burned patch.

Krummholz density probably would not vary in relation to the edge, unless a pre-fire condition influenced it. Seedling re-establishment densities will likely be influenced by the edge because of the affects that the edge has on soil conditions and the finding that indicated that strong relationships exist between soil conditions and seedling establishment.

Management Implications

High intensity fires increase the possibility of fire reaching into the alpine treeline ecotone, and this occurrence may result in several ramifications for the treeline ecosystem. A larger number of high intensity fires is, in part, a response to past fire management practices and well as global climate change. The greater occurrence of fire within treeline may lead to shifts in species distributions, greater sediment in stream headwaters, changes to carbon balance within treeline, higher avalanche potential, and nutrient redistributions. Changes within the treeline ecotone are of special interest because it is viewed as an indication of climate change and is of great interest to visitors of areas that contain treeline.

Fire reoccurrence intervals at treeline may extend hundreds, if not thousands of years (Sass et al. 2012b). A shorter return interval will, therefore, alter the disturbance regime of the treeline system, potentially resulting in a shift of the dominant species present. If a system is adapted to infrequent fires, the species present may not be resilient to fire, allowing other species or one of the species to dominate. For example, the results indicated that fir densities rose and pine dropped in burned areas compared to unburned areas. Fir seedlings are able to withstand dry soils and are able to germinate comparatively quickly. If fires continue to burn into treeline, firs may replace or outcompete other treeline species, such as pine. This factor is especially important for whitebark pine because whitebark serves as a source of food for several species and is suffering from a fungus within the mountains of the western United States and Canada.

Sass et al. (2012b) found that erosion continued to remain high in burned alpine areas many years after fire. They noted that bare soil patches of high erosion severity had expanded from 6% to 19% over the course of four years. They attribute this to the break-down of krummholz roots as well as positive feedback processes, in which high erosion inhibited vegetation establishment, which allowed for more erosion. Erosion within the alpine treeline may either result in a redistribution of sediment on the slope or contribute sediment to stream systems. A redistribution of sediments may influence vegetation re-establishment patterns and nutrient distributions, and a loss of sediment to stream systems will lead to an increase in stream sediment further downslope. Soil development is slow within the alpine treeline ecotone and it may take hundreds, or thousands, of years for that soil to be replaced.

More fire events within the alpine treeline may also result in raising avalanche potential (Butler et al. 1990; Sass et al. 2012a). This indirect effect of fire within the alpine treeline has particular implication to areas that contain structures, trails, and/or roads. Greater avalanche potential may last decades after fire because vegetation re-establishment and growth are slow within treeline and the decay of burned krummholz can occur years after fire and result in even greater slope destabilization than immediately following a fire event. Therefore, it is important that burned slopes be monitored long after fire occurred for indications of slope instability and avalanche potential.

Carbon flux within the alpine treeline ecotone has received relatively little attention in the literature (Grafius and Malanson, in preparation). Grafius and Malanson found that a high proportion of biomass is stored within dead material at treeline, which may subsequently serve as fuel for fire. Fires within treeline would, therefore, result in a quick release of this stored carbon into the atmosphere and influence the distribution of carbon within the treeline ecotone. Interactions among climate change, carbon, and fire within treeline appear to be complex and need further investigation.

Visitors to Glacier National Park are interested in the alpine treeline, which represents an iconic landscape feature of high mountain regions, and fire can leave a very visible influence on that ecotone. A better understanding of how fire affects treeline will aid visitor interpretation efforts. Also, the topic can lend itself to discussion on why low intensity fires within the sub-alpine forest are necessary to mitigate the occurrence of fire reaching into the treeline ecotone.

The overall contribution of this research to management concerns is that it further supports the need for allowing fires to burn and encouraging low intensity fires to occur more frequently. Climate change and fires within the alpine treeline may result in a positive feedback loop, in which drier conditions lead to more high intensity fires, and these fires release more carbon into the atmosphere, which results in further enhancing climate change conditions. A greater number of fire events at treeline also may result in a higher occurrence of avalanches and a lessening of alpine treeline diversity. Fine-scale and biogeomorphic interactions need to be recognized and incorporated into management and treeline dynamic models to better understand the treeline system.

Future Research

I plan on further analyzing the data, including testing for spatial autocorrelation among quadrats and within quadrats and evaluating additional relationships among the variables. I have begun evaluating the data for potential differences in results based on whether the data are considered related or independent, and the results so far are inconclusive. The data collected were analyzed to the extent of answering the research questions, but may be further evaluated to explore additional relationships.

For future research, I plan on expanding this study to include additional study locations within the Rocky Mountains. Ideally, I would select study sites located along a longitudinal gradient extending from the Southern Rockies into the Canadian Rockies. However, fires within treeline are not common and sites would be highly dependent on where recent fires have occurred. I would employ similar methods, depending on specific site situations, to compare burned areas to adjacent unburned areas. Methods would include the same variables addressed in this study, but the sample size for each site would be smaller to more efficiently collect data from more sites within a field season. Additionally, if the situation were to present itself, I would like to sample a site within weeks after it burned to measure soil hydrophobicity and initial erosion.

Along similar lines of this study, I would also like to develop a project to investigate the potential relationships between vegetation life history traits and the geomorphic effects of fire. This study would be a comparative assessment of various sites located throughout the mountains of the western United States that provide a diverse array of vegetation types that have recently experienced fire. Field methods and remote sensing techniques would be employed to evaluate geomorphic response, including erosion and soil conditions, following fire in relation to vegetation life history traits. Vegetation life history traits that may influence geomorphic response include a species or community's density before fire, its root structure (tap root systems, for example, may not be able to retard erosion as efficiently as roots located near the ground surface), and the species' ability to regenerate in post-fire conditions.

I also plan on further investigating interactions between geomorphic and ecologic variables in regard to disturbance edges. This study would not be limited to fire disturbance, but would consider different types of disturbance and the resulting relationships between geomorphic processes and vegetation patterns in mountain environments

The effects of fire on snow avalanches and debris flows may last years after the fire occurred. Sass et al. (2012a) found debris flows were still occurring in response to fire over 50 years later. Butler et al. (1990) delineated areas particularly susceptible to avalanching after fire within central and east Glacier National Park, and noted that an avalanche in many of these areas would impact structures, roads, and or railroads. It is well acknowledged that fire on slopes increase avalanche potential, but I am interested in investigating what affects fire within treeline has in comparison to fires within the subalpine area and how long the influence of fire may last on avalanche potential and occurrence in the Rocky Mountains.

Conclusion

The overall goal of this study was to evaluate the fine-scale influences of fire within the alpine treeline ecotone at three sites within Glacier National Park. Three primary findings were that fire contributed to alterations in soil, influenced vegetation cover and seedling establishment conditions, and highlighted the importance of biogeomorphic interactions. The results have several implications for treeline dynamics and management.

The results of this study indicate that several soil variables were significantly altered after fire. Soil loss was much greater in burned areas compared to adjacent unburned areas. Also, nitrogen and phosphorus were significantly higher after fire. Ground surface conditions became less compact overall, lost all duff material, and contained larger clasts. However, within burned areas, clast size and soil penetrability values varied spatially and in relation to topographic features. Boulder spalling was extensive in burned areas that contained boulders.

Vegetation was also significantly altered. Krummholz within the sites experienced complete mortality in the burned areas. Seedling micro-sites were altered after fire and seedling showed a distinct preference for select site conditions. Seedlings in burned areas were strongly associated with deeper effective depth and lower soil compaction (as indicated by soil penetrability). The loss of vegetation resulted in a great increase in the relative amount of sunlight received by the ground surface, and seedlings also preferred areas of more shade when compared to random micro-plots.

Biogeomorphic interactions were found in relation to several variables and in regard to methods of analysis. Facilitative objects were primarily rocks, and were found to be more important after fire than in unburned areas, for seedling establishment. Soil conditions also responded to the distinct burned/unburned edge around a path of burned krummholz. This result indicates that not only do soil conditions correspond to an ecological edge, but that placement of quadrats within a burned patch may also generate different results depending on the location of the quadrat relative to the edge. Results, however, of four variables evaluated, were independent of quadrat size. No significant differences in the averages of soil penetrability, clast size, krummholz density, and krummholz basal area were found among quadrat sizes ranging from 1 m^2 to 100 m^2. Standard deviations were generally larger for smaller quadrats.

The overall results of this research imply that the incorporation of fine-scale factors and biogeomorphic interactions are needed to fully understand treeline dynamics, and that low intensity fires should be allowed to burn, in theory, to decrease the chance of high intensity fires occurring and reaching into treeline. Treeline positions are monitored around the world for their response and movements relative to climate and other variables, but disturbance events, such as fire, may alter treeline dynamics. These systems of high species diversity may also alter compositionally if fire begins to reach into treeline more frequently. The results of this research further support the idea that fires are important to ecosystem functions and low intensity fires should be allowed to burn.

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