Effects of Burn Severity on Stream Buffer Management for Post-Fire Hillslope Erosion in the Inland Northwest Mountain Ranges, USA

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Authorization to Submit Thesis

This thesis of Edwin D. Bone, submitted for the degree of Master of Science with a Major in Water Resources and titled "Effects of Burn Severity on Stream Buffer Management for Post-Fire Hillslope Erosion in the Inland Northwest Mountain Ranges, USA," has been reviewed in final form. Permission, as indicated by the signatures and dates below, is now granted to submit final copies to the College of Graduate Studies for approval.

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Abstract

Forestry practices lack guidance on post wildfire stream buffer widths for harvesting operations. Rills on burned hillslopes were tested for their ability to infiltrate surface runoff and reduce transported sediment. Rills were measured for length and sediment concentration under high and low soil burn severity conditions and at zero, one and two years since fire. Rill length was directly related to soil burn severity at various flow rates. Sediment concentration and transport were related to soil burn severity in year one. Recovery was important and related to the vegetation regrowth and water repellency effects on infiltration. Standard 15 m buffers were sufficient to contain surface runoff at unburned sites. In year 1, low burn severity sites required 2x the standard width, and 4x for high soil burn severity sites. Immediately after fire, high soil burn severity sites required 8x the standard width. In year two, each width was reduced 50%.

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1.1 Background

1.1.1 Post Fire Hillslope Erosion

Due to increasing numbers of large wildland fires in the Western United States in recent decades (Liu et al., 2009; Stephens, 2005), post fire forestry management has become a common situation for land managers. This has raised the issue of how to properly conduct forest management in post fire environments without increasing sediment pollution for downstream water bodies. Inappropriate post fire management activities may increase sediment pollution, violate water quality regulations, and consequently halt operations. Forest managers need more information on how post fire forestry management should be conducted so operational efficiency may increase and environmental objectives may be achieved. Post fire stream buffers is one aspect of forest management that needs to be addressed by the scientific community.

Burned forests are typically more sensitive to management activities than unburned forests (Beschta et al., 2004). Management actions immediately following wildfire events may produce a range of positive to negative consequences lasting years to decades (Sessions et al., 2004). Thus, forest managers should consider both short and long term goals in forest planning (Franklin & Agee, 2003). The dual-disturbance of wildland fire and salvage logging may cause a variety of short term and long term impacts, depending upon the severity and extent of the combined disturbance and the environmental setting (McIver & Starr, 2001).

Salvage logging may offer beneficial opportunities and solutions for forest management. Salvage logging may provide an opportunity to capture the market value of burned timber (Sessions et al., 2004). This option may assist in mitigating potential economic losses caused by wildland fire in forests managed for timber harvesting. Captured revenues may also help offset the expenditures from fire suppression and provide rehabilitation dollars (Barker, 1989). In order to be profitable, available timber resources should be of sufficient value to justify operational expenses (Brown & Kellogg, 1996). Part of this equation includes declining timber quality after mortality (Aho & Cahill, 1984; Lowell & Cahill, 1996). Additional economic factors include harvest methods, transportation costs and market forces. Salvage may also reduce fuel loads of coarse woody debris which may alter future fire behavior (Collins et al., 2012).

The magnitude of wildfire disturbance on forested ecosystems depends upon the severity and extent of the fire (Neary et al., 2005). Generally, fires that cause major disturbance to forests are classified as high fire severity. Moderate and low fire severity classifications are used when alterations are not as severe. Alterations to forest vegetation may not always coincide with physical and biological alterations to forest soils (Safford et al., 2008; Keeley, 2009; Morgan et al., 2014; Moody et al., 2016). However, fires may cause disturbance in mosaic patterns of burn severity.

Wildfire may impact the forest floor in a number of ways. Increased hillslope erosion is a common natural response to wildfire (Moody & Martin, 2001; Robichaud, 2009). With the consumption of vegetation, surface roughness may decrease and cause increased overland flow velocities and erosion rates (Larsen et al., 2009; Lavee, 1995; Moore & Foster, 1990; Scott et al., 2009). Infiltration rates may initially decline and then naturally recover with vegetative regrowth (Cerdà, 1998; Spanos et al., 2005; Wagenbrenner et al., 2015; Robichaud et al., 2016). Fire-induced water repellency may be related to soil burn severity, however its extent and magnitude may vary spatially and dissipate irregularly (Cerdà & Robichaud, 2009; Doerr et al., 2000; Robichaud, 2000).

Post-fire salvage logging may produce a variety of ecological impacts (McIver & Starr, 2001). Potential benefits may include the reduction of fuel loads of coarse woody debris for future wildland fire events (Keyser et al., 2003), and the reduced potential for insect infestations (Brown et al., 2003). Harvesting may also improve surface cover and surface roughness by scattering fine woody debris. Fine woody debris may also provide surface protection for improved seedling recruitment (Castro et al., 2010). Adverse impacts of post-fire salvage logging may include the loss of nutrients and other biological legacies (Lindenmayer & Noss, 2006). Ground disturbance may impair seedling regeneration (Donato et al., 2006). Vegetative regrowth may be delayed, depending upon plant form (Morgan et al., 2015). The removal of snags may also impair habitat for cavity-nesting birds, depending upon the species (Hutto & Gallo, 2006; Saab & Dudley, 1998).

Significant to this study, post-fire salvage logging may also alter soil conditions. This may increase hillslope erosion and adversely impact water quality in adjacent streams (Beschta et al., 2004; Silins et al., 2009). The severity and extent of soil alterations may depend upon the method and extent of harvest operations (Klock, 1975). Some alterations may be helpful. Ground-based machinery may be able to break up fire-induced water repellency, which may improve water infiltration rates of forest soils (Wagenbrenner et al., 2015). However, heavy machinery may also cause soil compaction, which reduces soil pore spaces and may reduce infiltration and supersede the benefits of breaking up water repellent layers (Page-Dumroese et al., 2006; Wagenbrenner et al., 2015). Due to their reduced ground cover, skid trails may also provide preferential flow paths where surface runoff may concentrate and accumulate erosive power and sediment transport capacity (Anderson et al., 1976; Ares et al., 2005; Megahan & Kidd, 1972). The impacts of salvage logging impacts on the surrounding ecosystem may vary from one to several years depending upon the context of burn severity, vegetation, soils, climate and management (McIver & Starr, 2001, Morgan et al., 2015).

Studies have shown that mitigation techniques and prudent operating guidelines may be able to significantly reduce the severity and extent of salvage logging impacts on erosion and vegetative regrowth (Morgan et al., 2015; Spanos et al., 2005). One strategy for erosion mitigation is to exclude logging and machinery from riparian buffers along streams. While riparian buffers may not be spared from fire disturbances, managers may limit or completely preserve them from logging disturbances as a buffer from upland activities. Riparian forests have an essential role in sustaining habitat for aquatic ecosystems and their best management has been the subject of many scientific studies (Gregory et al., 1991; Reeves et al., 2006). Riparian areas function as an interface between aquatic and terrestrial environments (Karr & Schlosser, 1978). They may be managed to reduce the vulnerability of aquatic systems to upland hazards. In general, effective buffer characteristics depend upon the specific hazard being buffered (Castelle et al., 1994). In the case of hillslope erosion, buffers should have the ability to allow transported sediment to deposit within the buffer before reaching the stream channel (Castelle et al., 1994). Sediment deposition in riparian buffers may be caused by a variety of factors including reductions in slope, opportunities for surface detention and infiltration through increased vegetation and surface roughness, and sufficient distance for needed deposition to occur (Shisler et al., 1987).

With a significant loss of vegetative surface cover and forest litter, a burned riparian buffer may lose some effectiveness in mitigating upland erosion. Managers may need to implement wider buffers to compensate for reduced erosion mitigation efficiency. The magnitude of width adjustment may depend upon the severity of fire disturbance within the buffer, and the expected amount of surface runoff from upslope logged and compacted areas. The type of machinery used may be related to the severity of compaction that can be expected in a harvest area (Wagenbrenner et al., 2016). Remote sensing methods with the capability to determine the areal extent of harvest related disturbance to soils are in development (Lewis et al., 2012). Yet, published scientific guidance for managing appropriate buffer widths for post-fire logging erosion seems non-existent.

Stream buffer effectiveness in unburned conditions has been widely studied. In a review of stream buffer size requirements. Castelle et al. (1994) suggested that water resource values and intensities of upland disturbances should be considered when determining buffer sizes. They additionally mention that larger dimensions may be necessary for buffers in poor condition (e.g. high or low soil burn severity). Yet, no specific studies for post fire stream buffers were mentioned in the review.

Lynch et al. (1985) implemented 30 m (98 ft.) wide buffer strips for a green commercial clear-cut sale at an experimental watershed. Along with other best management practices (BMPs), the buffer was able to protect streams from 75% to 80% of excessive sedimentation. In one particular intermittent stream reach where the buffer was not in place, wind-blown trees near the channel were responsible for a large flux of sedimentation during spring runoff.

Wong and McCuen (1982) utilized a modeling approach for sediment control in riparian buffers that included variables such as vegetation, slope, substrate, and flow patterns. They found that buffers of 60 m (197 ft.) were sufficient in most situations. Similarly, a review from the U.S. Army Corp of Engineers (1991) suggests that buffer widths up to 50 m (164 ft.) are typically required to trap sediment from source areas.

These studies do not directly address buffer width requirements for post fire conditions. Bridging this gap of knowledge should provide a better understanding of how to manage and conserve forested watersheds affected by wildfire. Improvements to water quality, and greater efficiency in forestry management are potential benefits from this field of research.

1.1.2 Stream Buffer Management in the Northwest US

1.1.2.1 Introduction

The Clean Water Act (CWA) (33 U.S.C. 1251 et seq.) is the chief water quality law in the United States and is administered by the Environmental Protection Agency (EPA). Section 101 of the CWA describes its purpose to protect the physical, chemical and biological integrity of the Nation’s surface water. This is accomplished by setting water quality standards for each water body through the designation of its intended use. States are required to determine these uses, and then develop EPA approved water quality criteria that will meet the needs of those designations. Water bodies that violate a standard are identified as impaired and require pollution management through a Total Maximum Daily Load [TMDL] (CWA section 303(d)). A TMDL quantifies the maximum allowable loading of a particular constituent to a water body before a beneficial use of the water body is impaired. It then identifies the primary sources of the pollutant within the contributing watershed and sets a maximum allowable loading from each source. Permits are required for point source pollution discharge into surface waters given that State water quality standards are not violated (CWA sections 410, 402 and 404).

The CWA also directs management of disperse sources of pollution. This type of pollution is the primary water pollution associated with silvicultural operations. The CWA does not provide specific regulation or require permitting for nonpoint source (NPS) pollution, but directs states to develop and manage their own NPS programs (CWA sections 208 and 319). If a particular water body exceeds water quality standards, regardless of the pollution source, a TMDL is implemented. NPS allocations must be included. State programs should be capable of identifying NPS sources and developing plans for their control. Therefore, the CWA is a nationally applicable water quality law, but specific NPS goals and programs may vary between states and specific water bodies. Similar to the states, many tribal land management agencies have been permitted to develop and manage water quality programs within their jurisdictions.

States promote or enforce a set of Best Management Practices (BMPs) in an effort to control excessive levels of NPS pollution. They are designed to be preventative measures rather than reactionary treatments of water quality (Ice, 2004). BMPs may be directed toward agriculture, silviculture or other industries and activities. BMPs are not intended to prevent industry activities from occurring, but describe a set of objectives or methods that may be preferred or sometimes prohibited within the jurisdiction. BMPs should provide balance between ecological, social and economic interests, as needed. Once adopted, BMPs should undergo monitoring and periodic evaluation to determine if objectives are being met and if any adjustments should be made (Neary, 2015). This life cycle of BMPs provides opportunities for continuous improvement and directs focus to their ultimate objective of meeting water quality standards.

State BMPs regarding riparian zone management are typically outlined in forest practices rules authorized by respective state law. Northwest states, starting with Oregon, began passing initial forest practices acts in the 1970s, and federal rules were adopted shortly thereafter (Everest & Reeves, 2006). This was in response to mounting scientific research demonstrating that then-existing forestry practices were increasing stream temperatures and sedimentation (Everest & Reeves, 2006). The initial purpose of these guidelines was to reduce stream temperatures and decrease sedimentation. Riparian zone management rules were vital components in the effort to meet these objectives.

Everest and Reeves (2006) explain that early forestry practices BMPs were of the one size fits all variety. Since then, rules have become more adaptive to respond to physiographic, vegetative, and climate variables within a jurisdiction. Management has also evolved into watershed-scale planning, whereas cumulative watershed effects were not considered in initial BMPs (Elliot et al., 2010). Additionally, modern BMPs consider natural disturbance intervals and temporal scales. Forestry BMPs from states typically reflect timber harvest priorities, whereas federal types often prioritize multiple use strategies. BMPs may also target specific conservation projects in specific locations, like with bull trout (*Salvelinus confluentus*) habitat restoration in the Northwestern U.S. (U.S. Fish and Wildlife Service, 2015).

Due to state responsibility for NPS pollution management, other governmental land management agencies must comply with state laws and NPS programs (CWA section 313). The U.S. Department of Agriculture (USDA) Forest Service is a typical example of a governmental land agency that must operate within these rules. The USDA Forest Service (USFS) must comply with applicable State standards at a minimum and forests may simply adopt the established State BMPs where they are located. In other cases, the USFS may follow a different regional or local forest BMP that has been approved by the State as sufficient for meeting their water quality standards (USDA, 2012). Thus, BMPs may legally differ between federal and state forests within the same State.

1.1.2.2 State Management Practices

Riparian management zone BMPs in the northwest region of the United States (i.e. Washington, Oregon, and Idaho) feature some distinct characteristics. General BMP characteristics in the region reflect vibrant forest-based economies, with sometimes competing interests in timber industry, recreation, and conservation.

Northwest jurisdictions have developed riparian guidelines with a high amount of complexity compared with other regions (Lee et al., 2004). Some complexity is derived from classifying waterbodies by size and type. Classifications vary by jurisdiction but may include stream widths, and distinctions between permanent streams, intermittent streams, ephemeral streams, wetlands, seeps, and shorelines. Riparian zone guidelines in the region are also complicated by the presence of fish. Major aquatic conservation projects permeate the region, focusing on threatened and endangered salmonids. These BMPs generally focus on stream shading and sediment filtering functions of riparian reserves. Riparian rules may also be adjusted further by slope gradient, or the presence of unstable soils. Most BMPs in the region allow some degree of selective harvest within riparian buffers (Lee et al., 2004). Selective harvest practices are balanced between minimum conservation targets and economic benefits. In such cases, selective harvest typically decreases with increasing proximity to the waterbody. This strategy intends to increase shading within the riparian zone that may help regulate instream water temperatures. Domestic water supply designations may also modify stream buffer widths in some jurisdictions.

Washington State timber harvest policies are outlined in Washington State Department of Natural Resources (DNR) Title 222 WAC Forest Practices Rules, Chapter 30 (Washington State Department of Natural Resources, 2015). Section 045 of the document specifically covers all salvage logging practices within riparian management zones. It specifies no salvage logging within the bankfull width of any water type. The buffer dimensions are divided into three stages, termed core, inner and outer. The core zone is nearest the stream, the outer zone is the furthest from the stream, and the inner zone sits between them. Salvage logging is prohibited within the buffer's core zone, set at 15 m (50 ft.) in Western Washington and 9 m (30 ft.) in Eastern Washington. Beyond the core zone, the inner and outer zones may also be excluded if salvage would not meet specific leave tree requirements for the corresponding zone. Yet, erosion control qualities of the buffer are not mentioned in determining the suitability of timber harvest within the inner and outer zones. When streams fall within bull trout (*Salvelinus confluentus*) management zones, a special provision of 23 m (75 ft.) buffers with all available shade is prescribed. Section 070 of WAC 222-30 directs all ground-based logging systems to follow riparian zone rules specified throughout the document. Additionally, the number of skid trails allowed within riparian harvest zones is limited. Related best practices for ground-based logging in section 070 includes the following considerations: avoidance of exposed erodible soils or saturated conditions where compaction and excessive erosion may be likely, skid trails shall exist outside the no-harvest zone, and at least 9 m (30 ft.) from the bankfull width of small order unbuffered streams. Trails within 61 horizontal m (200 ft.) of streams are also required to be treated with water bars, slash or grade breaks in a manner that minimizes erosion. Water bar spacing is adaptable to minimize gullies and erosion. Skid trail guidelines applicable to the conditions of burned riparian zones are not explicitly mentioned. See Table 1.1 for a simplified summary.

Oregon Forest Practice Administrative Rules are described in the Oregon Department of Forestry (ODF) Forest Practices Rulebook and authorized by the Oregon Forest Practices Act (ODF, 2014). Division 635-0300 addresses the function of riparian management areas along streams in protecting water quality. Division 635-0200 of the document defines stream classification. Riparian zone widths, as defined in division 635-0310, are based upon the stream classification, in terms of water type and size. Type “F” waters denote fish bearing streams and require wider buffer widths than non-fish bearing streams. At the time of this writing, Oregon is in the process of adopting a new stream type for those bearing sensitive salmonid species, labeled as Salmon, Steelhead and Bull Trout (SSBT) (ODF, 2017). A “large” stream with more than 0.3 cms (10 cfs) of average annual flow require widths of 21 m (70 ft.), or 30 m (100 ft.) for Type F and SSBT. Streams with annual average flow between 0.06 cms (2 cfs) and 0.3 cms (10 cfs) normally require 15 m (50 ft.), 21 m (70 ft.) for Type F streams, and 24 m (80 ft.) for Type SSBT. Requirements for streams below 0.06 cms (2 cfs) average annual flow are 6 m (20 ft.), 15 m (50 ft.) for Type F, and 18 m (60 ft.) for Type SSBT. Measurements start from the high water level of the stream channel. Additional width is required when the slope adjacent to a stream is composed of steep, exposed soil, as in an exposed bank. These riparian zone dimensions primarily define riparian areas for the purposes of general vegetation management and water quality, and do not explicitly define undisturbed harvest exclusion zones. Harvesting rules, outlined in Division 629-630, specify that skid trails on steep or erosion prone hillslopes shall not be located within 30 m (100 ft.) of stream channels. Skid trail runoff must also be diverted onto undisturbed soils, and total disturbed soils must not exceed 10% of the hillslope area. Although practices for recently burned hillslopes are not mentioned, harvest practices for erosion-prone hillslopes may apply in this jurisdiction. See Table 1.1.

Standards for forest practices in the State of Idaho, authorized by Idaho Code Section 38-1304, are documented in the Idaho Administrative Procedures Act (IDAPA) “Rules Pertaining to the Idaho Forest Practices Act,” (IDAPA, 2014). Stream protection zones are divided into two classes, I and II. Class I pertains to any stream used for domestic water supply or function as important fish habitat in terms of spawning, rearing or migration. All fish-bearing streams fall under Idaho’s Shade Rule which applies riparian vegetation retention requirements intended to improve habitat temperatures. Class II streams are minor or headwater streams not included in Class I. Class I Stream Protection Zones extend 23 m (75 ft.) from ordinary high water marks, and Class II zones extend 9 m (30 ft.) from the same mark (Table 1.1). Subsection 30.07 of IDAPA 20.02.01 specifies that ground-based harvesting equipment are not allowed within Stream Protection Zones unless at stream crossings. Landings, skid trails, and fire trails are also prohibited within Stream Protection Zones. Related to soil protection, Subsection 30.03 prohibits skid trails on slopes steeper than 30%, where highly erodible soils are present, as may be in the case after a wildfire event.

1.1.2.3 Federal Management Practices

In addition to its obligation to comply with State water quality programs, the USFS operates under its National BMP Program (USFS, 2012). This program includes a core BMP that acts as an umbrella document for all agency regions and forests. This document ensures compliance with stipulations from the CWA. It also requires cooperation with State water quality programs in an effort to manage NPS pollution on national forest lands. The National BMP Program established an agency-wide priority on safeguarding water quality and improving impaired waters. While the National Core BMPs are applicable throughout the agency, criteria are not standardized to facilitate adaptive forest management strategies and guidance from applicable State, regional or individual forest water quality programs. Yet, monitoring and documentation of compliance with applicable standards is mandated through standardized procedures and data management requirements (USFS, 2012). Such effort is intended to demonstrate compliance with water quality programs and document any improvements of impaired waters. See Table 1.1.

The Northwest Forest Plan (NWFP) is a conservation plan that corresponds with the habitat of the northern spotted owl (*Strix occidentalis caurina*) (USFS & Bureau of Land Management [BLM], 2004). It pertains to both major federal land agencies, the Forest Service and the Bureau of Land Management, within the states of Washington, Oregon and northern California. The Aquatic Conservation Strategy contained within the plan defines standards for management objectives and activities. One of the larger goals of the plan is to protect salmonid habitat in the region. Thus, a major component of the Aquatic Conservation Strategy is to protect riparian areas due to their influence on water quality and instream habitat. Conservation of riparian reserves within this plan is approached with a series of objectives that target the many functions and ecosystem services that riparian areas provide. Included among the objectives is the maintenance and restoration of natural sediment regimes along with appropriate rates of surface erosion. Target conditions may vary temporally and spatially. Recommended riparian reserve widths are classified by waterbody type, as shown in Table 1.1. The strategy prohibits any activities within riparian reserves that may not meet the objectives described within the plan.

The Pacific Anadromous Fish Strategy (PACFISH) is a conservation strategy that applies to federal land agencies (USFS & BLM, 1995). The plan focuses on conservation of anadromous fish populations (i.e. salmon). This region comprises the interior Columbia River Basin, including eastern Washington, eastern Oregon, Idaho and parts of western Montana. PACFISH defines Riparian Habitat Conservation Area (RHCA) widths by waterbody type, similarly to the NWFP (see Table 2.1). Objectives of the BMP include the restoration of natural sediment regimes, and the management of shading for instream temperature regulation. The Inland Native Fish Strategy (INFISH) is a similar federal plan that focuses on bull trout conservation in the same jurisdictions.

1.1.2.4 Post Fire Buffer Management

After a high or low severity wildfire has occurred in a forested ecosystem, some difficulties may arise with the appropriate management of riparian stream buffers with current guidelines. In the situation of high severity wildfire, meeting near stream minimum shade requirements for habitat conservation objectives may no longer be achievable. Therefore, planning salvage logging projects around stream shading targets may no longer be applicable. In the case of both high and low severity wildfire, typical hillslope erosion regimes may not be expected to occur until sufficient time passes for vegetation regrowth and infiltration capacity recovery. Therefore, evaluating erosion risk with typical sediment regime metrics may not be realistic after these events. Additionally, occurrences of exposed and highly-erodible soils may be widespread, making the appropriate adjustment of stream buffer widths based on this metric unclear. Managers may also weigh the potential increase of erosion after salvage logging with the overall erosion impact of the wildfire disturbance (Lewis et al., 2012). In conjunction, total area disturbed by salvage logging operations may need to be evaluated for cumulative watershed impacts (Silins et al., 2009). Moreover, if riparian management zones are in poor condition for sediment filtering functions, normal buffer widths may no longer be sufficient to control stream sedimentation. Finally, managers may evaluate how the timing of recovery and the disturbance of management activities may affect vegetation recovery (Morgan et al., 2015) and cumulative water quality targets within a watershed. Consequently, it seems appropriate to work toward an understanding of how riparian buffer zone BMPs may be applied or modified in post wildfire forested landscapes.

The purpose of this research was to measure concentrated flow or rill travel lengths and their sediment concentrations in high and low soil burn severity classes. This was to determine if buffer widths should be increased under these post fire conditions. My hypothesis was that rill lengths and sediment concentrations would be directly related to burn severity.

1.2 Methods

1.2.1 Site Descriptions

Experiments were conducted at two locations within the Okanogan Highlands of the Inland Northwest Region of the U.S. (Figure 1.2). Both locations had recent wildfire activity. The 2015 North Star Fire in Washington served as the primary location. This mixed severity fire burned over 88,000 ha (218,000 ac.), the majority of which occurred on the Colville Federation Tribal Reservation (InciWeb, 2015). Additional burned areas occurred in the Colville National Forest, to the north of the reservation (Figure 1.3). Experimental sites for high, low and unburned conditions were established one year after the fire in July 2016.

The North Star Fire burned in a cool temperate forest with dry open stands of predominantly Douglas-fir (*Pseudotsuga menziesii*) mixed with ponderosa pine (*Pinus ponderosa*) (Clausnitzer & Zamora, 1987; Williams et al., 1995). Douglas-fir stands may be associated with bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*) or ninebark (*Physocarpus malvaceus*) at lower elevations, or with pinegrass (*Calamagrostis rubescens*) and mountain snowberry (*Symphoricarpos oreophilus*) at higher elevations, dependent upon microclimate (Clausnitzer & Zamora, 1987; Williams et al., 1995). The elevation of the fire ranged from approximately 650 to 2050 m (2100 to 6700 ft.). An analysis of historic fires suggest that occasional large, catastrophic fires have played a role in this region for many centuries, however most fires have been of low intensity (Williams et al., 1995).

Climate in the Okanogan Highlands is generally xeric, due to rain-shadow effects from the North Cascades (Williams et al. 1995). Climate data from Republic, Washington report mean annual precipitation of 430 mm (16.9 in.), average maximum temperatures of 13.4 °C (56.1 °F) and average minimum temperatures of -0.1 °C (31.8 °F), averaged from 1981 to 2010 (Western Regional Climate Center [WRCC], 2009). Precipitation is relatively low from July to September, and most snowfall occurs from November to February (WRCC, 2009). Site conditions are influenced by orographic effects.

Soils at the North Star Fire sites belong to the Nevine Series which is an andisol derived from volcanic ash over glacial till parent material, described as ashy over loamy skeletal, glassy over isotic, frigid *Typic Vitrixerand* (Soil Survey Staff, 2016). This soil is well-drained, and typically 50 to 100 cm (20 to 40 in.) in depth to a dense layer (Soil Survey Staff, 2016). The A horizon is very thin. Clay content is reported at 10%, sand is 21.2%, and hydraulic conductivity at 3.2 cm hr^{-1} (1.3 in hr^{-1}) (Soil Survey Staff, 2017). This soil is naturally resistant to erosion, except when disturbed such as compaction with heavy machinery (Williams et al., 1995).

Additional rill experiments were conducted 2 months after the 2016 Cayuse Mountain Fire (i.e. October 2016) which burned over 7,000 ha (18,000 ac.) on the Spokane Tribal Reservation (InciWeb, 2016). The Cayuse Mountain region is located in a warmer climate than the location of the North Star Fire, with 530 mm (20.9 in.) of average annual precipitation, average maximum temperatures of 14.7 °C (58.5 °F) and average minimum temperatures of 2.1 °C (35.8 °F), averaged from 1981 to 2010 (WRCC, 2009). Experimental sites at the Cayuse Mountain Fire sit about 200 m (656 ft.) lower in elevation than those at the North Star Fire, on average.

Forest structure and composition at the Cayuse Mountain Fire resemble those of lower elevation sites at the North Star Fire, with dry open stands of Douglas-fir mixed with ponderosa pine. This suggests a similar fire history as found at the North Star Fire. Soils at the Cayuse Mountain Fire sites belong to the Dragoon Series, which is a mollisol derived from volcanic ash over granite, gneiss or schist, described as fine loamy, isotic, mesic *Vitrandic Argixerolls* (Soil Survey Staff, 2016).

Clay content is 12.5%, sand is 30.9%, with hydraulic conductivity reported as the same as the Nevine Series at 3.2 cm hr^{-1} (1.3 in hr^{-1}) (Soil Survey Staff, 2017).

1.2.2 Field Methods

Experimental Design

Evaluation of this experiment was based upon the ability for a stream buffer to infiltrate concentrated overland flow and deposit entrained sediment before entry into a stream channel. This was a field-based study where controlled concentrated flow paths, or rills or predetermined sediment concentration, were produced and evaluated for the effects of burn severity, flow rate and recovery time (see Figure 1.1).

The rill experiment was an original experimental design, although some concepts were adapted from previous rill experiments (Robichaud et al., 2010). Both the flow rate and the sediment concentration in the added flow were controlled in the rill experiments. Each rill experiment was 40 minutes in duration with incremental changes in flow every 10 minutes. Each experiment consisted of flow rates of 50, 100, and 150 L min^{-1}. The experiment began with a 50 L min^{-1} flow rate for 10 minutes without any sediment added to acquire background erosion rates. Then 25 g L^{-1} of sediment was added for an additional 10 minutes. The same sediment concentration was maintained while the flow rate was increased to 100 and 150 L min^{-1} during the subsequent 10 minute intervals, respectively. Both flow and sediment concentration were adjusted without any break in the experiment. The controlled sediment concentration was based off of findings from previous experiments (Robichaud et al., 2010). The three flow rates were chosen to simulate the flow from a wide range of upslope contributing areas, and storm sizes. Areas may range from 5 m (16 ft.) wide skid trails with water bars placed every 15 m (50 ft.), to 0.25 ha (0.6 ac.) landings with additional hydrologically connected skid trails and roads. Storms may range from 1 year to 5 year storms, or more if soil recovery is slow. Sediment added to the rills consisted of topsoil collected from the North Star Fire which were sieved to 6 mm and dried.

Sediment concentration samples were taken at three locations along the rill for each steady state flow rate. Samples were taken directly from the top of the rill during the flows to verify the target initial sediment concentrations of 25 g L^{-1}. Samples were taken at approximately halfway between the point where water was added, at the top of the rill, and the point where water was completely infiltrating, at the end of the rill. Additionally, samples were taken at the approximate end of the rill where sufficient flow could be reasonably sampled (Figure 1.1). The exact distance between the two sample locations were determined and adjusted in real-time as runoff progressed down the slope. Both crews took three water samples at all four flow rates at the approximate beginning, middle and end of each 10 minute period. Therefore, both sampling teams required 12 sample bottles per rill. When adding the three samples from the top of the rill, 27 samples were collected for each rill. The flow rate at the location where water samples were taken was estimated based upon the collected water volume and the time required to fill the sample bottle.

Additional measurements taken by the sampling teams were travel distance from the top of the rill to the sample location. This was facilitated by pin flags located along the estimated flow path at 5 m intervals. If rills divided into multiple flow paths or sub-rills, the sub-rill with the most estimated discharge was selected for sampling (Figure 1.1). Immediately after taking the water samples, the sampled sub-rill was measured at the sample location for flow width and average depth. Other sub-rills along the same cross section of the sample location were also measured for flow width and average depth. It was not feasible to collect water samples from each sub-rill along the cross-sections within the allotted time. Flow paths were directed away from large depressions or soil pipes with sheet metal hammered into the ground to prevent any major water losses.

A 25 g L^{-1} sediment concentration was maintained by a sediment feed device with a motor driven dry chamber feeder that mixed sediment and water at the controlled ratio. Gravity fed sediment from the dry chamber was fed into a wet chamber where it was mixed with a controlled water flow rate. The wet chamber was fitted for three separate water supply line inputs. In the experiment, each input was provided with a regulated 50 L min^{-1} supply of water. Water was obtained from a local stream and filtered to 100 microns. The water was delivered with an 11,000 L (3,000 gal) capacity water truck, with separate flow regulators and flow meters for all three supply lines. As the experiment progressed, each of the three inputs were opened to provide the three 50 L min^{-1} increments. With each increment of the water flow, the dry chamber motor speed also increased to maintain the target sediment concentration. The three required motor speeds were tested and calibrated during fabrication to allow quick adjustment during the experiment. Finally, the soil and water mix exited the wet chamber and onto the ground to simulate a concentrated flow path.

<u>Site Selection</u>

Experimental test sites were selected based upon soil burn severity (Parson et al., 2010; Shakesby & Doerr, 2006). At the North Star Fire two high and one low soil burn severity locations were selected in addition to an unburned control site. The two high soil burn severity hillslopes each had two rill experiments making a total of four rills. At the low soil burn severity site all four rill experiments were conducted within the same hillslope. Similarly at the unburned site, located just outside the fire boundary, four rill experiments were conducted on the same hillslope.

All rill experiments were conducted on relatively uniform, steep slopes. It was not required to conduct these experiments adjacent to actual streams. Sites were chosen that had a uniform hillslope gradient for at least 100 m (328 ft.) to eliminate excessive sediment deposition or scour due to topographic variability along the rills. The target slope class for all rills was between 20 and 40 percent. Sites were selected that had not been disturbed by management prior or post fire, or was scheduled for disturbance during the anticipated two year duration of this study. Despite having an 88,000 ha (218,000 ac.) fire to work with, finding sites with all these attributes, plus adequate roadside access, made site selection somewhat challenging.

Rills were tested at the Cayuse Mountain Fire to measure effects immediately after a high severity fire. However, transferring all the site requirements from the

North Star Fire to the smaller 7,000 ha (18,000 ac.) Cayuse Mountain Fire increased the challenge to find suitable sites. As a result, only one high burn severity site was located at the Cayuse Mountain Fire where four replicates were tested along the same hillslope.

The effects of recovery time on rill length and sediment concentration was also assessed. All replicates were tested at the North Star Fire in July 2016 and again in July 2017. All replicates at the Cayuse Mountain Fire were tested in October 2016 and again in July 2017. For analysis, time was evaluated as months since fire was contained. Discrete values included 2 (Cayuse Mountain Fire), 10 (both fires), and 22 (North Star Fire) months since fire.

Other Measurements

During the experiments, additional measurements were taken to characterize the soils within each hillslope. Field saturated hydraulic conductivity was measured with a dual head infiltrometer (Reynolds & Elrick, 1990, Meter Group, Inc., Pullman, WA). The infiltrometer was mounted to a 5 cm deep ring that was pressed into the soil to create the pressurized chamber. The device was performed a 15 minute wetting cycle in order to saturate the soil. Then 5 and 10 cm pressure heads were tested for 10 minutes each. This was repeated three times. The device estimated the infiltration rate based upon the final cycle. This procedure was done for each rill.

Flow velocity was measured during each experiment with a salt and dye solution and electrical conductivity (EC) probes (King & Norton, 1992). Velocity was determined from the time it took the peak salt pulse to travel a known distance from one EC probe to another. The two EC probes were placed 2 m apart near the sampling areas. Velocity measurements were not required if no significant flow was present. If possible, velocity was measured during each of the four flow rates, and at more than one location, depending upon total flow length. An average velocity was calculated for each flow rate for each rill.

Soil water repellency was measured with a water drop penetration test at the soil surface, 1 cm and 3 cm depth below the soil surface, with eight drops each

(DeBano, 1981; Robichaud et al., 2008). Time for each water drop to penetrate the soil was recorded from 0 to 300 seconds and classified into weak (0-60 seconds), moderate (61-180 seconds) and strong (>180 seconds) repellency classes. Undisturbed locations for the test were randomly selected near the top, middle and bottom of each rill. Water repellency of the soil in the hillslope was classified by the percent of occurrence in each penetration class.

One bulk density measurement was taken along each rill. A 4.8 cm diameter soil core was extracted using a slide hammer in randomly selected undisturbed locations next to each rill from the 0 to 5 cm and 5 to 10 cm soil depths. Soil cores were oven dried for 24 hours at 105 °C and then weighed. Bulk density was calculated as the soil weight divided by the core volume of 90.5 cm^3.

Surface soil moisture from the 0 to 3 cm depth was measured by taking one small surface soil sample in a random undisturbed area beside each rill. The samples were sealed in bags after sampling. In the laboratory, they were weighed before and after being oven dried for 24 hours at 105 °C. Soil moisture was calculated as ratio of the water weight to the oven dry soil weight.

Soil surface cover was measured using a 1 m quadrat. Percent cover was estimated with a 100 point sample grid (Chambers & Brown, 1983). Cover was measured at a random location along the top, middle and bottom sections of the rill. Cover categories consisted of bare soil, litter, vegetation and other. Bare soil included mineral soil and gravel observations, litter included woody debris and litter observations, vegetation included vegetation and moss observations, and other included rock and tree observations. Rill surface cover was calculated as the percent of occurrence among all points sampled.

Land slope was measured using a clinometer and two persons of the same height. One person measured slope from the bottom of the slope looking upward toward the other person standing at the top of the slope. One overall slope value was calculated for each rill with the mean value of three separate slope measurements taken at 5 m (16 ft.) spacing across the hillslope.

1.2.3 Statistical Methods

Statistical analysis was performed using R statistical software (R Core Team, 2016). Mixed effects models from *lme4* (Bates et al., 2012) were used to compare means and variances among burn severity and other covariates, with a significance level of 0.05 (Winters, 2013). Random variables were rill plots and fire locations. Fixed variables are shown in Table 1.2 and 1.4. Significance was determined with a likelihood ratio test (Winters, 2013). Rill length was analyzed as the rill's maximum travel distance during each flow rate interval. Soil burn severity was represented nominally as 1) unburned, 2) low soil burn severity, and 3) high soil burn severity. Recovery time was compared in discrete values as the number of months since fire was contained. Fire location was also nominally converted to 1) North Star, and 2) Cayuse. Water repellency was compared as the percent occurrence of strong repellency results. Moderate and weak repellency were not used to simplify the model. Soil cover was compared as fractions of cover. Soil moisture and slope were analyzed as percent values.

Model diagnostics included testing residuals for normal distribution, plotting residuals versus predicted values, plotting residuals versus order, and quantile-quantile plots of residuals versus predicted values (Winters, 2013). The model assumptions were met. Rill length was log transformed to meet fit assumptions. Sediment concentration per discharge was also log transformed to fit the data. Covariates were analyzed for collinearity using a variance inflation factor method using the *car* package (Fox & Weisberg, 2011).

1.3 Results

<u>Rill Travel Length</u>

The influence of soil burn severity on rill travel length was significant at every flow rate, and both years at the North Star Fire. Rill length tended to increase with increased soil burn severity (Table 1.3). The results of each set of experiments are shown in Table 1.3. In 2016, rills traveled on unburned sites a mean of 3.3, 5.5, and 8.8 m (11, 18, and 29 ft.) for the 50, 100 and 150 L min^{-1} flows, respectively. For low burn severity sites, rills traveled a mean of 6.5, 11 and 17 m (21, 36, and 56 ft.) for the respective flows. Mean rill travel length in 2016 was consistently twice the distance on low soil burn severity sites than on unburned sites at each flow rate. In the same year, rills at high soil burn severity sites traveled a mean of 17, 24 and 42 m (56, 79, and 138 ft.) for the respective flows. Rills at high burn severity sites traveled 4.4 to 5.0 times farther than at unburned sites, depending upon the flow rate. Mean travel length was significant at the Cayuse Mountain Fire in 2016 and was 45, 83 and 100 m (148, 272, and 328 ft.) for the 50, 100, and 150 L min^{-1} flow rates, respectively.

Initial flow rate was directly related to rill travel distance and was significant for all experiment sets. Rill travel length increased with the increase in initial flow rate, as expected (Table 1.2). Rill length reached steady state within each of the 10 minute flow intervals and then incrementally increased with higher flow rates. Rill travel length between the 50 L min^{-1} flow without sediment and the 50 L min^{-1} flow with sediment showed a small increase, but was not significant. At the Cayuse Mountain Fire, during the 150 L min^{-1} flow rate in 2016, all four rills reached the 100 m (328 ft.) maximum travel length within the 40 minute time span. Beyond 100 m (328 ft.) the hillslope dropped below the target slope class in a concave shape. Using the ratios of flow rate and rill length from the 50 and 100 L min^{-1} flow rates, final lengths of the Cayuse rills may have reached about 120 m (394 ft.), if more hillslope was available. Comparing the mean travel distances from Table 1.3, each increase of 50 L min^{-1} of flow increased the rill length significantly between 1.5 to 1.8 times in the first year at the North Star Fire. The effect of flow increases was higher at the Cayuse Mountain Fire than the North Star Fire.

Time elapsed since fire was did not significantly influence travel length at the burned sites, unless each fire was analyzed separately. Rill length was slightly longer at the unburned sites in the second year at 5.5, 6.5, and 9.4 m (18, 21, and 31 ft.) for the 50, 100 and 150 L min^{-1} flows, respectively. Recovery time tended to decrease rill length (Table 1.2). However, the mean travel length did not change consistently between the two fires, or the two burn severities at the North Star Fire.

The magnitude of change was greater at the Cayuse Mountain Fire than the North Star Fire and at the high soil burn severity sites within the North Star Fire. Rill lengths at low soil burn severity sites decreased 8 to 19% in the second year to 6, 9 and 14 m (20, 30, and 46 ft.) for the respective flows. High soil burn severity sites at the North Star Fire decreased 39 to 42% to 10, 15, and 24 m (33, 49, and 79 ft.) for the same flows. Rill lengths at the Cayuse Mountain Fire decreased 72 to 76% in the second year to 13, 20 and 35 m (43, 66, and 115 ft.) for the same flows.

Sediment Concentration

Sediment concentration was directly related to soil burn severity. The increase in soil burn severity tended to increase sediment concentration by discharge (Table 1.4). Figure 1.4 shows that in 2016, rills at North Star high soil burn severity sites had significantly higher sediment transport than rills in the other burn conditions. Table 1.5 shows the mean concentration measurements by fire location, year, burn severity and flow rate, and includes both upper and lower rill sample locations in the mean. At the North Star Fire, the mean sediment concentrations at the unburned sites in 2016 were 1.6, 5.6, and 15 g L^{-1} for the 50, 100 and 150 L min^{-1} flow rates, respectively. Mean concentrations were 1.5 times higher at low severity sites than unburned sites, except at 150 L min^{-1}, with 2.5, 8.2 and 7.0 g L^{-1} for the respective flow rates. High severity sites had mean concentrations 1.5 to 2 times higher than low severity sites, with 5.0, 13, and 14 g L^{-1} for the respective flows. Mean concentrations in 2016 at the Cayuse Mountain Fire were 8.5, 15 and 19 g L^{-1} for the 50, 100, and 150 L min^{-1} flow rates respectively. Comparing mean concentration among the clean 50 L min^{-1} flows at the North Star Fire, soil burn severity was also a significant influence (Table 1.5). Table 1.6 shows that sediment concentration decreased more slowly in rills at burned sites. At the North Star Fire in 2016, the mean change in sediment concentration averaged over all flows was 5.4, 3.7, and 1.5 g L^{-1} m^{-1} for unburned, low burn severity and high burn severity, respectively. At the Cayuse Mountain Fire in 2016, the sediment concentration was increasing with distance on average for most flows. Table 1.7 and Figure 1.5 show how sediment load changed with distance in the experiment sets.

Large variability in sediment concentration was observed at unburned sites for the highest flow rate. This was due to small litter dams that often formed and then broke during the experiments, releasing pulses of sediment. Ground cover at unburned rills was primarily pine needles and wood. This ground cover was the litter source for the small dams that were observed in the rills. While the effect of litter dams on sediment pulses was noticeable in 2016, this action was observed much more frequently in 2017. This was likely due to the litter being disturbed during the previous year's experiments. Therefore, spikes in sediment concentration influenced the unburned results in 2016 and especially in 2017, as seen in Figure 1.4.

Initial flow rate was directly related to sediment concentration. Sediment concentration was inversely related to the initial flow rate (Table 1.4). Values in Table 1.6 shows that each incremental increase in flow rate increased mean sediment concentration, but the relationship was not linear. The response in sediment concentration was weaker at higher flow rates. High soil burn severity sites also resulted in an overall weaker response to initial flow rate than the low soil burn severity and unburned sites.

Recovery time tended to decrease sediment concentration for both fires. Mean concentrations in 2017 for the North Star low burn severity sites were 3.3, 4.1 and 5.2 g L^{-1} for the 50, 100, and 150 L min^{-1} flow rates, respectively. North Star high soil burn severity sites were 2.2, 2.4 and 3.0 g L^{-1} for the same respective flow rates. Mean concentrations in 2017 at the Cayuse Mountain Fire were 3.2, 5.5, and 6.9 g L^{-1} for the respective flows. Figure 1.4 shows that sediment transport significantly decreased for high soil burn severity rills at both fires in 2017. Mean sediment concentration decreased in rills more rapidly than the first year at burned sites of both fires (Table 1.6). Sediment concentration decreased at the North Star Fire at a rate of 5.5 and 2.4 g L^{-1} m^{-1} for low and high soil burn severity rills, respectively. Concentration decreased at 1.5 g L^{-1} m^{-1} in 2017 at Cayuse Mountain sites.

Ground Cover

Variability in ground cover was significantly related to soil burn severity and recovery time (Figures 1.6 through 1.9). At the North Star Fire in 2016, high soil burn severity sites had the highest mean percent of bare soil (35%), low sol burn severity sites had the most vegetation cover (71%), and unburned sites were dominated by litter cover (85%). One year later, high soil burn severity sites showed significantly less bare soil, and low soil burn severity sites increased in litter cover and decreased in vegetation cover. Cayuse Mountain Fire sites had significantly higher amounts of bare soil (82.5%) than the North Star Fire high severity sites. Although the bare soil fraction decreased at the Cayuse Mountain Fire the following year, it was still significantly higher than North Star Fire high severity sites at the same time elapsed since fire of 10 months.

Travel length was correlated with ground cover when bare soil was the only cover class analyzed in the mixed model. Travel length tended to increase with bare soil, and decrease with litter and vegetation cover (Table 1.2). Rill length had a stronger correlation to bare soil than any other single cover class. High soil burn severity sites at the Cayuse Mountain Fire had the highest mean bare soil fraction in 2016, and also the longest rill lengths measured in this project.

Sediment concentration per discharge was not significantly correlated with ground cover, except when vegetation cover was the only cover class analyzed. (Table 1.3). Similar to rill length, sediment concentration increased with increasing bare soil. Unlike rill length, litter cover increased sediment concentration, but the results were not significant. This result was likely influenced by the litter dams described earlier. Vegetation cover negatively influenced sediment concentration per discharge, but not significantly.

Water Repellency

Rill travel distances generally increased with water repellency, however this relationship was not significant (Table 1.2). Water repellency was directly related to sediment concentration (Table 1.4). Strong water repellency was common at North Star Fire high burn severity sites in 2016, but dropped off significantly the following year (Figure 1.10). A similar magnitude of change was found at the Cayuse Mountain Fire between both years (Figure 1.11). Water repellency at the North Star

Fire in 2016 was the weakest at low burn severity sites. Water repellency was very weak at both fires in general in 2017.

Velocity

There was a significant indirect relationship between ground cover and rill velocity. Vegetation cover had the strongest significance of all cover variables. Rill velocity increased with percent bare soil, however there was not a significant relationship between litter cover and rill velocity (Figure 1.12). Rill velocity tended to increase with soil burn severity class (Figures 1.13 and 1.14). Velocity tended to increase with increased water repellency (Figure 1.15). However, velocity tended to decrease with time elapsed since fire.

Soil Moisture

Soil moisture tended to increase rill travel length (Table 1.2). However, soil moisture tended to decrease sediment concentration (Table 1.4). Soil moisture had no significant relationship to burn severity. Soil moisture was significantly influenced by ground cover. The relationship was positive with vegetation cover and negative with bare soil. Soil moisture had a significant relationship with fire location. A rainfall event occurred at the Cayuse Mountain Fire location the day before the experiments. Thus, fire location was highly collinear with soil moisture, and had an influence on rill travel length. Figure 1.16 shows soil moisture at the North Star Fire by burn severity and year. All experiments in 2016 at the North Star Fire were conducted within three consecutive days and no precipitation was observed in the days before or during the experiments.

Bulk Density

Bulk density in the top 5 cm of soil tended to decrease both rill travel length and sediment concentration but only significantly for sediment concentration (Tables 1.2 and 1.4). Moreover, bulk density was related to soil burn severity at the North Star Fire in both years (Figure 1.17). The lowest bulk density measurements were found in high soil burn severity sites. Yet, results from the Cayuse Mountain Fire were consistently higher than high soil burn severity bulk density measurements from the North Star Fire (Figure 1.18).

Slope

Slope tended to increase sediment transport (Table 1.4). Slope was also directly correlated with rill length (Table 1.2). This relationship was important despite the lower mean slope gradient of North Star Fire high burn severity sites (Figure 1.19). Regarding the target slope class of 20 to 40%, the mean slope at North Star Fire low severity and unburned sites was 40%. High burn severity sites at the same fire had a mean of 26%. Cayuse Mountain Fire sites had a mean of 18%, which was slightly below the target (Figure 1.19).

Infiltration

The duel head infiltrometer was not able to measure the saturated hydraulic conductivity of the soil as the device was not able to establish ponded water. This was due to the high infiltration rates of the soils. The infiltrometer did successfully measure field saturated hydraulic conductivity at a nearby compacted skid trail, but not at the rill sites. However, using rill travel distance as a rough surrogate, it may be assumed that infiltration was highest at unburned sites, and lowest at high burn severity sites.

1.4 Discussion

These results suggests that effectiveness of a buffer to infiltrate upslope runoff and decrease sediment is greatly reduced following wildfire. Although rill travel length and sediment concentration were both significantly influenced by burn severity, rill length showed more drastic effects. All three soil burn severity conditions had significantly different mean rill lengths, implying managed buffer widths should increase with soil burn severity.

The effect of burn severity on rill length was significant regardless of the initial amount of flow added to the rill. Therefore, burn severity may affect rill length for a wide range of runoff events. This means that if a stream buffer was designed to mitigate the erosion risk for a specific target storm size, it will likely not be as effective after the buffer has burned. This may place downstream water bodies at risk for the target storm size, and possibly smaller ones.

We did observe that these buffers recover with time following wildfire. Rill length decreased for both low and high soil burn severity hillslopes with recovery time. Recovery time had a larger influence on reducing rill length at high burn severity sites. In one year, the Cayuse Mountain Fire decreased rill length by about 75%. Rill length at the North Star Fire high soil burn severity sites reduced by 50% after one year, and sediment transport was no longer higher than unburned and low soil burn severity sites (Figure 1.4). Table 1.6 showed that sediment did not travel as far downslope at all burned sites in the second year, but low soil burn severity sites reduced sediment more effectively than high soil burn severity sites. However, despite the reductions in sediment delivery after one year, all burned sites still had statistically different rill lengths than the unburned control sites at all flow rates. Therefore, standard buffer widths for the given storm design may still place downstream water bodies at an elevated risk for at least 1 to 2 years, or more.

Recovery results indicate the importance of the timing of management activities. Buffer effectiveness was the lowest at the Cayuse Mountain Fire two months after the fire. After 10 months of recovery, runoff distances and sediment delivery were much more manageable at both the Cayuse Mountain Fire and the North Star Fire. Therefore, allowing one growing season to pass before management disturbance may be a very effective strategy to reducing water pollution to downstream water bodies. However, in Morgan et al. (2015), results indicated that disturbance resets the clock on revegetation recovery, thus earlier harvesting was suggested. Management may need to weigh the outcomes of both recommendations.

This study suggests that ground cover can be a useful indicator of buffer effectiveness. The amount of bare soil was strongly correlated with burn severity and recovery time, and was closely related to rill velocity. The extent of ground cover of any type was related to the reduction of rill length. Similarly, Benadives-Solorio and

MacDonald (2005) found that bare soil was a predictor of erosion at the hillslope scale. The reduction in rill length compared across burn treatments may indicate the role of regrowth in improving infiltration after wildfire, similar to Cerdà and Doerr (2005). Litter may also have decreased surface runoff by increasing surface storage, reducing rill velocity, and allowing more time for infiltration (see Lavee et al.1995; Pannkuk & Robichaud, 2003; and Scott et al., 2009).

Mean sediment concentration was higher in rills with more bare soil. These rills also had higher mean rill velocities. This finding agrees with the concept of stream power as a quantification of erosive potential. Stream power increases with water velocity, the depth, density and weight of the water column, and slope (Bagnold, 1966; Nearing et al., 1997). Accordingly, rills with bare soil also had higher velocities, slopes and higher sediment concentrations. Therefore, a rill that completely traverses a stream buffer with low surface cover percentage will likely increase the sedimentation of downstream water bodies.

Infiltration capacity may represent the most important functional asset of buffers managed for erosion mitigation. Results from the North Star Fire clearly indicate significantly different infiltration characteristics between unburned, low burn severity and high burn severity hillslopes, agreeable with Robichaud (2000). At unburned sites, 3,500 L of runoff poured onto 40% gradient hillslopes for 40 minutes traveled a maximum of 11 m (36 ft.). An estimated 96% of the water had infiltrated, using wetted area and average depth measurements from the final sample (Table 1.8). The maximum distance for high burn severity rills was 49 m (161 ft.) Mean estimated infiltration was 70% for these sites (Table 1.8). These high infiltration capacities effectively diminished the risk of the runoff.

Martin and Moody (2001) also found high infiltration results in volcanic ash derived soils in a mixed conifer system in New Mexico. Using a portable rainfall simulator, they produced runoff on only one out of three sites and reported a lower limit of 260 mm hr^{-1} (10.2 in hr^{-1}). They also found that after burning, volcanic soils from the mixed conifer ecosystem had higher infiltration rates than burned granitic soils, and almost as high as unburned granitic soils. However, results showed that volcanic soils from a ponderosa pine only ecosystem had lower infiltration rates than volcanic soils from the mixed conifer system. This indicates that results from the North Star Fire experiments may not be entirely applicable to ecosystems of different soil types and outside mixed conifer forests.

Infiltration at the Cayuse Mountain Fire had different results than the North Star Fire. In 2016, all four rills reached 100 m (328 ft.) distances with estimated 39% mean infiltration (Table 1.8). This may be partly due to the experiments occurring immediately after fire containment. Very little time had elapsed since the wildfire. No vegetation regrowth was observed at the test site, and water repellency was very high. However, distances may also be partly attributed to antecedent moisture conditions caused by a rain event the previous day. Mean soil moisture at the Cayuse Mountain Fire was twice the mean value from high severity sites from the North Star Fire in the same year. Yet, widespread strong water repellency results at the Cayuse Mountain Fire may indicate the soil was not fully saturated (see Doerr et al., 2009). Yet, the soils may have had less storage capacity within the soil available for infiltration which may have contributed to the 100 m (328 ft.) distances. Additionally, the precipitation event may have induced some soil crusting (Morin & Benyamini, 1977) or ash sealing (Cerdà & Robichaud, 2009). Although, crusting may not have been significant due to low clay content in the soils (Ben-Hur et al., 1985).

Results from the Cayuse Mountain Fire sites may not be fully comparable to results from the North Star Fire. Yet, the conditions in 2016 at the Cayuse Mountain Fire may better represent soil conditions during actual rainfall events. Since the rill experiment simulated storm water runoff, it may be assumed that soil water content would be higher during a precipitation event, and pores may be sealed with ash. Therefore, the 2016 results from the Cayuse Mountain Fire may demonstrate a more likely outcome for storm water runoff under those conditions. Despite the lower soil moisture conditions at the North Star Fire, all three burn severities were tested under the same conditions. Therefore, those results may demonstrate a scaling relationship between the different burn severities, pertinent to scaling stream buffer widths.

Water repellency may have reduced infiltration at high burn severity sites. Strong repellency was common at the North Star Fire in 2016 and widespread at the Cayuse Mountain Fire the same year. Tables 1.2 and 1.4 show that repellency was only significant for sediment concentration, but still tended to increase rill length. The widespread repellency in 2016 at the Cayuse Mountain Fire may have contributed to reduced infiltration, as in DeBano (1981). This is in contrast to 2017, where mean rill length at the Cayuse Mountain Fire decreased drastically, as did the occurrence of strong water repellency.

It appears that the Cayuse Mountain Fire recovered more quickly than the North Star Fire high burn severity sites, in terms of rill distance, and sediment concentration. After 10 months of recovery, the Cayuse Mountain Fire had shorter rill lengths for each flow rate and lower sediment concentrations than the North Star Fire. At this point both locations had similar mean ground cover fractions however, the recovery of water repellency was much greater at the Cayuse Mountain Fire than the North Star Fire. That faster recovery time at the Cayuse Mountain Fire may be attributed to the soils having a greater sand content. This may indicate that fire induced water repellency may break up faster in coarser soils. Coarser textured soils also tend to have higher infiltration rates than finer textured soils (Abrahams et al. 1988). These effects of soil texture may have contributed to the higher infiltration at the Cayuse Mountain Fire.

1.5 Conclusion

Stream buffer effectiveness was tested in rills by measuring for travel length and sediment concentration on hillslopes with high and low soil burn severity. Overall, buffer effectiveness declined following wildfire. Rill travel length and sediment transport increased in buffers which experienced both low and high soil burn severity fires. Stream buffers may have very little effectiveness immediately after high soil burn severity fires. Following the wildfire buffer effectiveness improves with time as revegetation occurs. Sediment transport was significantly greater due to the fire than unburned slopes in the first year, but was significantly reduced after two years due to increasing vegetation regrowth and litter. Rill travel length was significantly greater in both soil burn severity classes each year, and declined significantly between years, but more significantly for high soil burn severity. Buffer recovery was correlated somewhat to reductions in water repellency, but more significantly with improved ground cover. The effect of wildfire on buffer effectiveness may vary with soil and ecosystem types. These findings suggest that wildfire impacts and recovery relative to unburned conditions may be a beneficial approach for future stream buffer management recommendations.

Chapter 2: Recommendations for Post Fire Stream Buffer Management

2.1 Discussion

Based upon the results from chapter 1, it seems appropriate to consider burn severity adjustments to stream buffer widths for post wildfire erosion mitigation. Rills originating from upslope areas may be able to travel farther through buffers and carry elevated loads of sediment downhill toward water resources. This is due to reduced efficiency in infiltrating surface runoff into forest soils. Infiltration efficiency appears to correlate well with burn severity, and improves over time. Rills also seem to respond to burn severity at all flow rates.

Stream buffers intended for erosion mitigation may be designed for the surface runoff of a particular storm size. Source areas are typically compacted areas, such as roads, skid trails and landings, and their runoff coefficients would be higher than undisturbed forest soils, but lower than paved surfaces. Rainfall intensities are associated with a storm of predictable frequency of occurrence (e.g. 5 year return interval). A stream buffer may be established according to how much runoff may be expected with a particular storm size. The WEPP model is a modern and efficient runoff prediction tool that has been adapted for forest management applications (Elliot, W.J., 2013). Buffer distance to the stream should allow enough time and space for predicted runoff to infiltrate sufficiently to reduce elevated risk of sedimentation in water resources. Water bars or other runoff control treatments may be used to help mitigation efforts (Robichaud et al., 2014).

Burned stream buffers may require extra width to protect water bodies. Although the purpose of this thesis was not intended to produce a model for adjusting stream buffers, the results may help inform managers of the relative scales of burn severity impacts. Compared with unburned hillslopes, rills traveled 2x the distance on low burn severity sites, and 4.4 to 5x the distance on high burn severity sites (Table 2.1). These results occurred almost one year after the fire was contained. Additionally, rills traveled up to 8 to 10x the distance only two months after the fire was contained. In the second year, rill distances decreased by 10 to 20% for low burn severity, and about 40% for high burn severity.

Vegetation regrowth may provide a visual indicator of stream buffer condition. Regrowth is a variable dependent upon burn severity, soils, vegetation communities and climate (Lloret & Zedler, 2009; Morgan et al., 2015). Major deviations from average annual precipitation may influence regrowth rates, and in turn, stream buffer recovery. Therefore, stream buffer rules may benefit from an adaptive management approach, ensuring buffer dimensions are appropriate to actual conditions.

Post fire management activities should preserve the infiltration capacities of stream buffers. Infiltration capacities of forest soils appear to be critical in the mitigation of upslope erosion. Activities that introduce compaction into stream buffers (i.e. skid trails) may be counterproductive to protecting water resources. If compacted areas already exist within stream buffer zones, additional mitigation techniques may need to be implemented to reduce sedimentation risks (Robichaud et al., 2014; Wagenbrenner et al., 2015).

Downstream water bodies may benefit from implementing post fire best management practices (BMPs) according to the results of this study, which may be most applicable in mixed conifer forests with volcanic ash soils. A typical stream buffer of 15 m (50 ft.) from the stream would have been sufficient to contain all runoff tested at the unburned sites in this study in both years. Therefore, under burned conditions, the scaling factors for each burn severity mentioned previously may be appropriate. Scaling 2x for low burn severity conditions would set widths at 30 m (100 ft.) from the stream, which would have been sufficient for the rills in this study (Table 2.1). Applying a 4x scale for high burn severity would increase widths to 60 m (200 ft.) from the stream, which again would have been sufficient for the rills of this study. Doubling this width to 120 m (400 ft.) from the stream would also have been sufficient for the first year of rill tests at the Cayuse Mountain Fire. According to our results, these dimensions may be reduced 50% after the end of each growing season. Therefore, waiting for vegetation recovery may not only reduce the risk of sediment pollution to downstream water bodies, but also allow management activities to operate in closer proximity. The incremental reduction of buffer widths may be delayed based upon annual evaluation of vegetation regrowth. Based upon the sediment concentration and transport results of this study, extended buffers may no longer be necessary after the third growing season following the event. However, longer term studies may be helpful to verify this assumption. A summary of this post fire BMP recommendation is shown in Table 2.1.

It is uncertain how stream buffers may be adjusted for burn severity in other ecosystems. This may depend primarily upon natural infiltration capacities involved and the degree in which fire alters those initial states. It may also depend upon the local adaptations of vegetation in response to fire regimes and precipitation patterns following the event. Similar experiments to those carried out in this study performed in other ecosystems may lead to a more general understanding of how fire events may impact stream buffer management. However, if the same pattern found in this study is validated in other areas, then the same scaling factors from the unburned buffer distance from the stream may apply (Table 2.1). Table 2.2 demonstrates how the scaling factors from Table 2.1 may be used to adjust existing BMPs for post wildfire buffers.

Post fire stream buffer BMPs may work in concert with existing guidelines. A post fire BMP would be provisional by affected area based upon burn severity, and temporary based upon recovery. Areas unaffected by fire events or at no risk of upslope erosion may continue implementation of pre fire BMPs. Stream buffers that are already sufficiently sized to accommodate predicted post fire erosion may not require a post fire BMP. In some cases, existing BMPs may not need additional width but may need to do more to prohibit activities that significantly reduce infiltration within riparian zones following fire events. Where existing BMPs are determined to be insufficient for post fire hillslope erosion, provisions based upon the findings of this research may be implemented to reduce the sediment delivery risk for downstream water bodies.

2.2 Summary

In summary, many state and federal jurisdictions have adopted stream buffer BMPs to decrease NPS sediment pollution caused by forestry management. These results suggest that buffers need modification for post fire conditions, where elevated risk of sedimentation may be expected, according to the degree of burn severity. Burned soils experience reduced infiltration capacity, therefore stream buffers lose effectiveness at containing runoff and require extra width to compensate. Buffers are very ineffective before the first growing season in high soil burn severity conditions. Width requirements may be scaled for both low and high soil burn severity, and reduced annually depending upon vegetation regrowth. Extra caution within post fire buffers is recommended to avoid additional reductions in soil infiltration capacities. Post fire BMPs should strengthen existing forestry practices by improving mitigation according to impacted areas and temporal requirements. Forest management that uses these recommendations, within the context of this study, should have greater confidence that downstream water bodies are properly buffered from sediment pollution caused by upslope management activities.

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Tables and Figures

Table 1.1. Simplified summary of Northwest US state and federal stream buffer BMPs. Class names, definitions, harvest rules, and distances are included for each.

BMP	Classification	Definition	Harvest	Distance to Stream
Washington (western)	Core zone	Closest to stream	None	15 m (50 ft.)
	Inner zone	Between Core and Outer	Selective, no landings	3 to 30 m (10 to 100 ft.) additive from Core zone, variable by class
	Outer zone	Furthest from stream	Selective	7 to 20 m (22 to 67 ft.) additive from Inner zone, variable by class
	Bull trout zone	Presence of species	Shade dependent	23 m (75 ft.)
Washington (eastern)	Core zone	Closest to stream	None	9 m (30 ft.)
	Inner zone	Between Core and Outer	Selective, no landings	14 to 21 m (45 to 70 ft.) additive from Core zone, variable by class
	Outer zone	Furthest from stream	Selective	0 to 17 m (0 to 55 ft.) additive from Inner zone, variable by class
	Bull trout zone	Presence of species	Shade dependent	23 m (75 ft.)
Oregon	Type F	Fish bearing	Selective, no skid trails	
	1. Large	> 10 cfs		30 m (100 ft.)
	2. Medium	2 to 10 cfs		21 m (70 ft.)
	3. Small	< 2 cfs		15 m (50 ft.)
	Type SSBT	Salmon, steelhead, bull trout	Selective, no skid trails	
	1. Large	> 10 cfs		N/A
	2. Medium	2 to 10 cfs		24 m (80 ft.)
	3. Small	< 2 cfs		18 m (60 ft.)

	Type D	Domestic supply	Selective, no skid trails	
	1. Large	> 10 cfs		21 m (70 ft.)
	2. Medium	2 to 10 cfs		15 m (50 ft.)
	3. Small	< 2 cfs		6 m (20 ft.)
	Type N	Other	Selective, no skid trails	
	1. Large	> 10 cfs		21 m (70 ft.)
	2. Medium	2 to 10 cfs		15 m (50 ft.)
Idaho	Class I	Fish bearing and domestic water supply	No machinery, skid trails or landings	23 m (75 ft.)
	Class II	Minor or headwater streams	No machinery, skid trails or landings	9 m (30 ft.)
Northwest Forest Plan (NWFP)		Fish bearing streams	Selective upon Aquatic Conservation Strategy objectives	91 m (300 ft.)
		Permanent non-fish bearing streams	Selective upon Aquatic Conservation Strategy objectives	46 m (150 ft.)
		Intermittent streams	Selective upon Aquatic Conservation Strategy objectives	30 m (100 ft.)
		Natural lakes, ponds	Selective upon Aquatic Conservation Strategy objectives	91 m (300 ft.)
		Constructed ponds, reservoirs and wetlands	Selective upon Aquatic Conservation Strategy objectives	46 m (150 ft.)
PACFISH/INFISH	Category I	Fish bearing streams	Selective	91 m (300 ft.)
	Category II	Permanent non-fish bearing streams	Selective	46 m (150 ft.)
	Category III	Intermittent streams	Selective	46 m (150 ft.)

	Category IV	Ponds, lakes, reservoirs, wetlands	Selective	15 to 30 m (50 to 100 ft.)

Table 1.2. Summary of mixed linear model for rill travel length. Degrees of freedom shown with Chi squared values. P values in bold are significant (p<0.05).

Covariate	Coefficient	Standard Error	(df) X^2	P value
Burn Severity [1]	0.672	0.150	(1) 21.8	**<0.001**
Flow Rate	0.009	<0.001	(1) 1550	**<0.001**
Time Elapsed (month)	-0.018	0.014	(1) 2.32	0.127
Slope %	0.018	0.008	(1) 4.28	**0.038**
Soil Moisture %	0.024	0.007	(1) 15.1	**<0.001**
Bare Soil %	0.569	1.11	(1) 0.355	0.551
Litter Cover %	-0.094	1.11	(1) 0.0126	0.911
Vegetation Cover %	-0.225	1.15	(1) 0.0465	0.829
Bulk Density Upper	-0.549	0.328	(1) 3.77	0.052
Bulk Density Lower	0.150	0.558	(1) 0.102	0.750
Strong Repellency%	0.001	0.003	(1) 0.182	0.670

[1]Unburned (1), low burn severity (2), high burn severity (3)

Table 1.3. Means and ranges of rill travel distances (m) by flow rate. Different letters across a row indicate significant differences in burn severity treatment at α = 0.05.

Flow Rate	Unburned	Low Soil Burn Severity	High Soil Burn Severity
2016 North Star			
50 L min^{-1}	3.3 (3-4) [a]	6.5 (6-10)[b]	17 (14-19) [c]
100 L min^{-1}	5.5 (5-6) [a]	11 (7-16) [b]	24 (20-36) [c]
150 L min^{-1}	8.8 (7-11) [a]	17 (13-30) [b]	42 (33-49) [c]
2017 North Star			
50 L min^{-1}	5.5 (3-6) [a]	6.0 (6-7) [b]	10 (7-10) [c]
100 L min^{-1}	6.5 (6-7) [a]	9.0 (8-11) [b]	15 (15-20) [c]
150 L min^{-1}	9.4 (8-11) [a]	14 (10-15) [b]	24 (19-33) [c]
2016 Cayuse Mountain			
50 L min^{-1}	--	--	45 (35-45)
100 L min^{-1}	--	--	83 (45-90)
150 L min^{-1}	--	--	100 (100-100)
2017 Cayuse Mountain			
50 L min^{-1}	--	--	13 (10-15)
100 L min^{-1}	--	--	20 (14-30)
150 L min^{-1}	--	--	35 (17-51)

Table 1.4. Summary of mixed linear model for sediment concentration per discharge. Degrees of freedom shown with Chi squared values. P values in bold are significant (p<0.05), with significance codes.

Covariate	Coefficient	Standard Error	(df) X^2	P value
Burn Severity [1]	0.820	0.289	(1) 10.1	**0.001**
Flow Rate	-0.007	<0.001	(1) 52.1	**<0.001**
Time Elapsed (month)	-0.007	0.021	(1) 0.179	0.672
Slope %	0.105	0.024	(1) 22.0	**<0.001**
Soil Moisture %	-0.017	0.010	(1) 4.18	**0.041**
Bare Soil %	2.49	1.66	(1) 3.21	0.073
Litter Cover %	1.05	1.64	(1) 0.653	0.419
Vegetation Cover %	-1.58	1.70	(1) 1.20	0.274
Bulk Density Upper	-1.16	0.491	(1) 7.57	**0.006**
Bulk Density Lower	-0.457	0.827	(1) 0.410	0.522
Strong Repellency %	0.012	0.005	(1) 9.20	**0.002**

[1] Unburned (1), low burn severity (2), high burn severity (3)

Table 1.5. Means and ranges of sediment concentration (g L^{-1}) by flow rate. Different letters across a row indicate significant differences in soil burn severity treatment at α = 0.05.

Flow Rate	Unburned	Low Soil Burn Severity	High Soil Burn Severity
2016 North Star			
50 L min^{-1} *clean*	0.1 (0.0-0.7) [a]	0.3 (0.1-1.2) [b]	2.8 (0.0-13) [c]
50 L min^{-1}	1.6 (0.1-4.2) [a]	2.5 (0.6-5.2) [b]	5.0 (1.1-30) [c]
100 L min^{-1}	5.6 (0.3-26) [a]	8.2 (2.3-30) [a]	13 (0.4-27) [a]
150 L min^{-1}	15 (0.7-27) [b]	7.0 (1.0-21) [a]	14 (4.4-38) [b]
2017 North Star			
50 L min^{-1} *clean*	0.6 (0.0-3.1) [a]	0.2 (0.0-0.7) [a]	0.7 (0.0-3.1) [a]
50 L min^{-1}	5.8 (1.8-19) [a]	3.3 (0.2-4.7) [b]	2.2 (0.5-6.9) [b]
100 L min^{-1}	10.4 (2.5-33) [a]	4.1 (1.2-12) [b]	2.4 (0.7-6.6) [b]
150 L min^{-1}	17.0 (0.7-33) [a]	5.2 (1.2-15) [b]	3.0 (1.2-15) [b]
2016 Cayuse Mountain			
50 L min^{-1} *clean*			2.6 (0.1-14)
50 L min^{-1}	--	--	8.5 (2.4-27)
100 L min^{-1}	--	--	15 (3.0-36)
150 L min^{-1}	--	--	19 (7.4-41)
2017 Cayuse Mountain			
50 L min^{-1} *clean*			1.3 (0.0-3.6)
50 L min^{-1}	--	--	3.2 (1.2-7.5)
100 L min^{-1}	--	--	5.5 (2.9-16)
150 L min^{-1}	--	--	6.9 (3.0-10)

Table 1.6. Mean change in sediment concentration per meter (g L^{-1} m^{-1}) downslope. Results shown for by fire, burn severity and year for each flow rate, and all flows rates combined.

Burn Severity	50 L min^{-1}	100 L min^{-1}	150 L min^{-1}	All flows
2016 North Star				
High	-2.7	-1.2	-0.6	-1.5
Low	-5.1	-3.7	-2.2	-3.7
Unburned	-7.9	-3.9	-4.4	-5.4
2016 Cayuse Mountain				
High	-0.7	0.6	1.1	0.2
2017 North Star				
High	-2.6	-2.8	-2.0	-2.4
Low	-6.3	-5.4	-5.1	-5.5
Unburned	-2.8	-5.8	-0.8	-3.2
2017 Cayuse Mountain				
High	-2.4	-1.2	-0.9	-1.5

Table 1.7. Mean (%) difference in sediment load (g min^{-1} m^{-1}) downslope. Results shown by fire, soil burn severity and year for each flow rate, and all flows rates combined. 2017 North Star unburned sites had missing sample times.

Soil Burn Severity	50 L min^{-1}	100 L min^{-1}	150 L min^{-1}	All flows
Initial Load (g)	12,500	25,000	37,500	
2016 North Star				
High	-13.8%	-9.3%	-7.5%	-10.2%
Low	-41.5%	-26.0%	-20.0%	-29.2%
Unburned	-61.3%	-34.8%	-19.0%	-38.4%
2016 Cayuse Mountain				
High	-0.8%	2.3%	0.3%	0.6%
2017 North Star				
High	-23.3%	-15.3%	-11.5%	-16.7%
Low	-35.8%	-26.5%	-21.3%	-27.8%
2017 Cayuse Mountain				
High	-15.5%	-11.0%	-6.5%	-11.0%

Table 1.8. Estimated infiltration of total added water (3500 L). Rill mean and range results grouped by fire, soil burn severity, and year. Estimated from total wetted area, and average depths of rill and sub-rills measured with the final sample of each rill experiment. Mean infiltration rate is based upon the 40 minute test average.

Year	Fire	Soil Burn Severity	Infiltration (%)	Infiltration Rate (cm/hr)
2016	North Star	High	70 (44 to 88)	14.7
2016	North Star	Low	89 (79 to 95)	49.6
2016	North Star	Unburned	96 (95 to 98)	116
2016	Cayuse Mountain	High	39 (8 to 65)	4.6
2017	North Star	High	85 (75 to 94)	30.6
2017	North Star	Low	94 (91 to 98)	56.6
2017	North Star	Unburned	94 (90 to 97)	100
2017	Cayuse Mountain	High	80 (74 to 87)	16.3

Table 2.1. Summary of post wildfire stream buffer width recommendations. Buffer width is the distance recommended for the locations from this research project. The scaling factor may be used to increase existing buffers at other locations for post fire conditions.

Burn Severity	Year Since Fire[1]	Buffer Width[2]	Factors to Adjust Unburned Buffers for Post Fire Conditions
High Burn Severity	0	120 m (400 ft.)	8x
	1	60 m (200 ft.)	4x
	2	30 m (100 ft.)	2x
Low Burn Severity	0	60 m (200 ft.)	4x
	1	30 m (100 ft.)	2x
	2	15 m (50 ft.)	1x

[1]Each year allowing for one full growing season
[2]Contingent upon typical vegetation regrowth

Table 2.2. Adjustments for post fire dimensions of existing BMPs. The original classification and distance to waterbody is given with adjustments for 0, 1 and 2 years since fire. “H” is given for high soil burn severity, and “L” for low soil burn severity. Adjustments are based off of scaling factors from Table 2.1.

BMP	Classification	Original Distance to Waterbody	Distance for Year 0	Distance for Year 1	Distance for Year 2
Washington (western)	Core zone	15 m (50 ft.)	H=120 m (400 ft.) L=60 m (200 ft.)	H=60 m (200 ft.) L=30 m (100 ft.)	H=30 m (100 ft.) L=15 m (50 ft.)
	Bull trout zone	23 m (75 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)	H=91 m (300 ft.) L=46 m (150 ft.)	H=46 m (150 ft.) L=23 m (75 ft.)
Washington (eastern)	Core zone	9 m (30 ft.)	H=73 m (240 ft.) L=37 m (120 ft.)	H=37 m (120 ft.) L=18 m (60 ft.)	H=18 m (60 ft.) L=9 m (30 ft.)
	Bull trout zone	23 m (75 ft.)	H=183 m (600 ft.)	H=91 m (300 ft.)	H=46 m (150 ft.)

			L=91 m (300 ft.)	L=46 m (150 ft.)	L=23 m (75 ft.)
Oregon	Type F				
	Large	30 m (100 ft.)	H=244 m (800 ft.) L=122 m (400 ft.)	H=122 m (400 ft.) L=61 m (200 ft.)	H=61 m (200 ft.) L=30 m (100 ft.)
	Medium	21 m (70 ft.)	H=171 m (560 ft.) L=85 m (280 ft.)	H=85 m (280 ft.) L=43 m (140 ft.)	H=43 m (140 ft.) L=21 m (70 ft.)
	Small	15 m (50 ft.)	H=120 m (400 ft.) L=60 m (200 ft.)	H=60 m (200 ft.) L=30 m (100 ft.)	H=30 m (100 ft.) L=15 m (50 ft.)
	Type SSBT				
	Large	N/A			
	Medium	24 m (80 ft.)	H=195 m (640 ft.) L=98 m (320 ft.)	H=98 m (320 ft.) L=49 m (160 ft.)	H=49 m (160 ft.) L=24 m (80 ft.)
	Small	18 m (60 ft.)	H=146 m (480 ft.) L=73 m (240 ft.)	H=73 m (240 ft.) L=37 m (120 ft.)	H=37 m (120 ft.) L=18 m (60 ft.)
	Type D				
	Large	21 m (70 ft.)	H=171 m (560 ft.) L=85 m (280 ft.)	H=85 m (280 ft.) L=43 m (140 ft.)	H=43 m (140 ft.) L=21 m (70 ft.)
	Medium	15 m (50 ft.)	H=120 m (400 ft.) L=60 m (200 ft.)	H=60 m (200 ft.) L=30 m (100 ft.)	H=30 m (100 ft.) L=15 m (50 ft.)
	Small	6 m (20 ft.)	H=49 m (160 ft.) L=24 m (80 ft.)	H=24 m (80 ft.) L=12 m (40 ft.)	H=12 m (40 ft.) L=6 m (20 ft.)
	Type N				
	Large	21 m (70 ft.)	H=171 m (560 ft.) L=85 m (280 ft.)	H=85 m (280 ft.) L=43 m (140 ft.)	H=43 m (140 ft.) L=21 m (70 ft.)
	Medium	15 m (50 ft.)	H=120 m (400 ft.) L=60 m (200 ft.)	H=60 m (200 ft.) L=30 m (100 ft.)	H=30 m (100 ft.) L=15 m (50 ft.)
Idaho	Class I	23 m (75 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)	H=91 m (300 ft.) L=46 m (150 ft.)	H=46 m (150 ft.) L=23 m (75 ft.)

	Class II	9 m (30 ft.)	H=73 m (240 ft.) L=37 m (120 ft.)	H=37 m (120 ft.) L=18 m (60 ft.)	H=18 m (60 ft.) L=9 m (30 ft.)
Northwest Forest Plan (NWFP)		91 m (300 ft.)	H=732 m (2400 ft.) L=366 m (1200 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)
		46 m (150 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)	H=91 m (300 ft.) L=46 m (150 ft.)
		30 m (100 ft.)	H=244 m (800 ft.) L=122 m (400 ft.)	H=122 m (400 ft.) L=61 m (200 ft.)	H=61 m (200 ft.) L=30 m (100 ft.)
		91 m (300 ft.)	H=732 m (2400 ft.) L=366 m (1200 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)
		46 m (150 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)	H=91 m (300 ft.) L=46 m (150 ft.)
PACFISH/ INFISH	Category I	91 m (300 ft.)	H=732 m (2400 ft.) L=366 m (1200 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)
	Category II	46 m (150 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)	H=91 m (300 ft.) L=46 m (150 ft.)
	Category III	46 m (150 ft.)	H=366 m (1200 ft.) L=183 m (600 ft.)	H=183 m (600 ft.) L=91 m (300 ft.)	H=91 m (300 ft.) L=46 m (150 ft.)
	Category IV	15 to 30 m (50 to 100 ft.)	H=122 to 244 m (400 to 800 ft.) L=61 to 122 m (200 to 400 ft.)	H=61 to 122 m (200 to 400 ft.) L=30 to 61 m (100 to 200 ft.)	H=30 to 61 m (100 to 200 ft.) L=15 to 30 m (50 to 100 ft.)

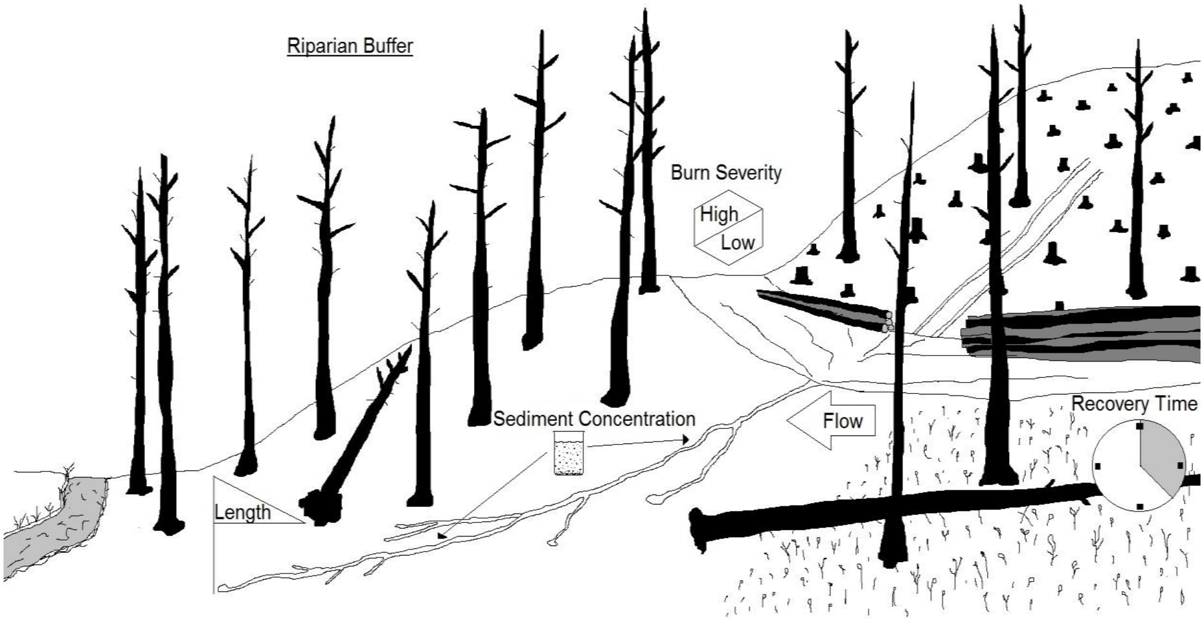

Figure 1.1. Rill experiment sketch. Rills were measured for travel length and sediment concentration under two burn severities, three flow rates, and in two consecutive years.

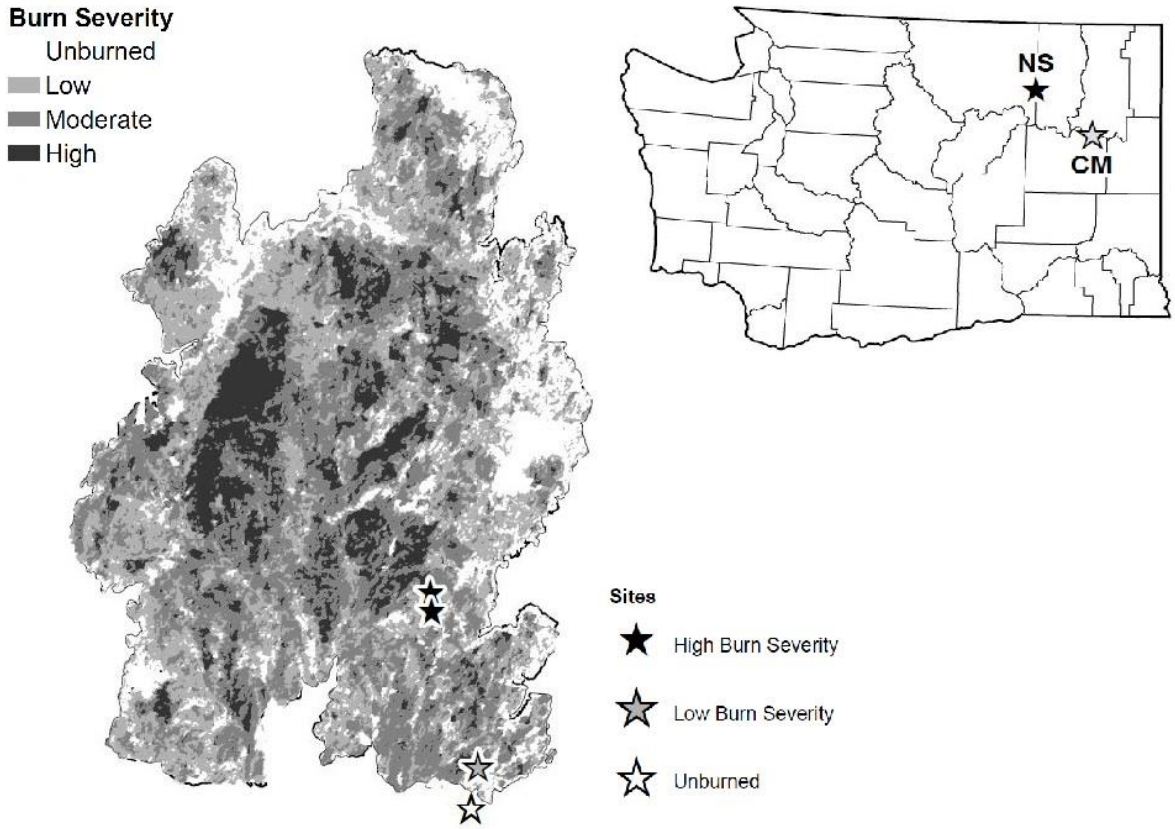

Figure 1.2. Map of field locations. Burn Severity Map of the 2015 North Star Fire, WA. Burned Area Reflectance Classification, (BARC) (Clark & Bobbe, 2006). Fire locations: 2015 North Star Fire (NS) and 2016 Cayuse Mountain Fire (CM).

Figure 1.3. Rill experiment images. a) Overview of a rill site; b) Sediment and water mixer with motor controller; c) Sediment and water mixer in operation; d) a sample being taken with a small piece of sheet metal funneling into a collection bottle.

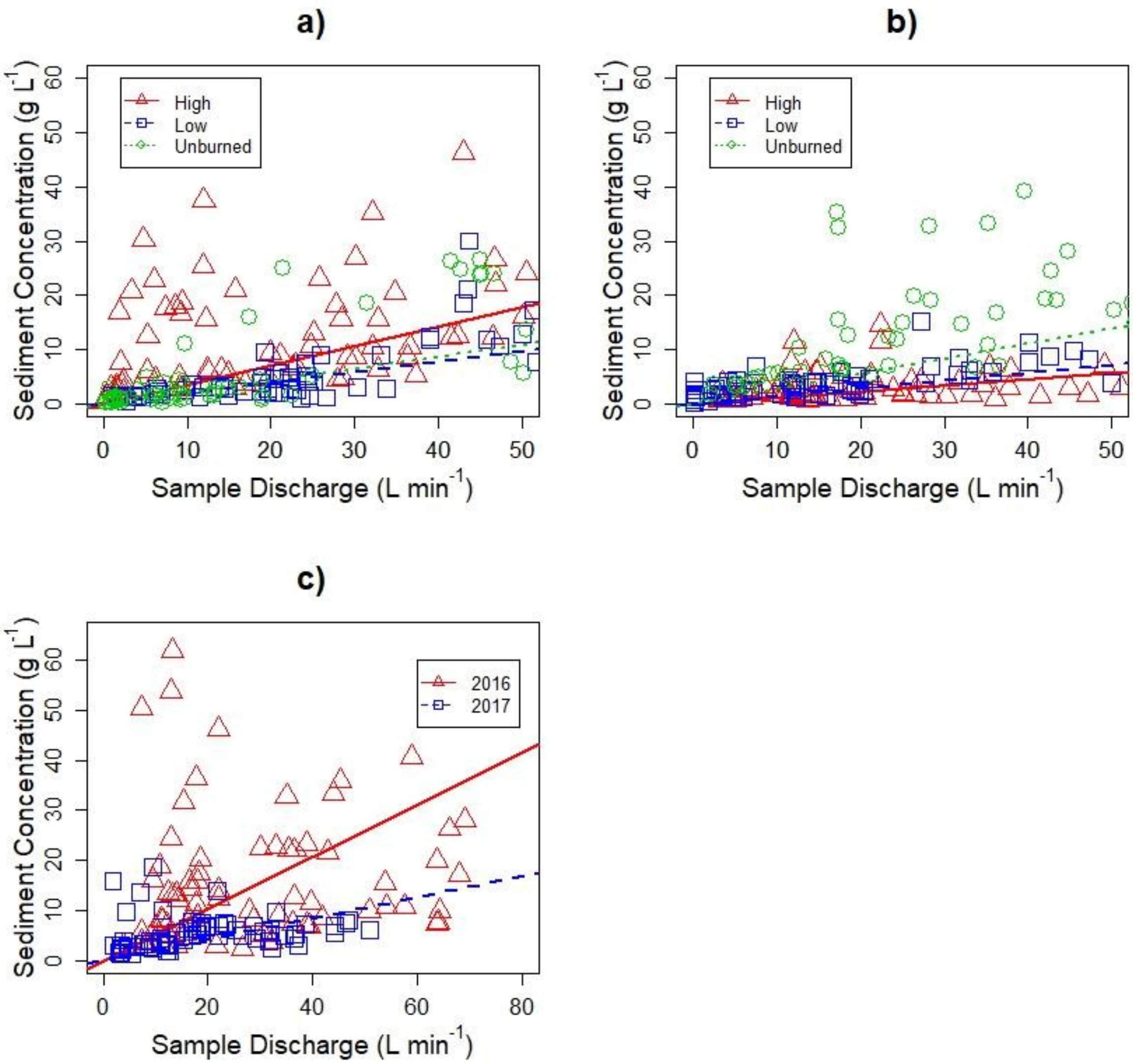

Figure 1.4. Sediment concentration per discharge. a) 2016 North Star Fire; b) 2017 North Star Fire; c) Cayuse Mountain Fire high severity both years. *n*=72 for each plot.

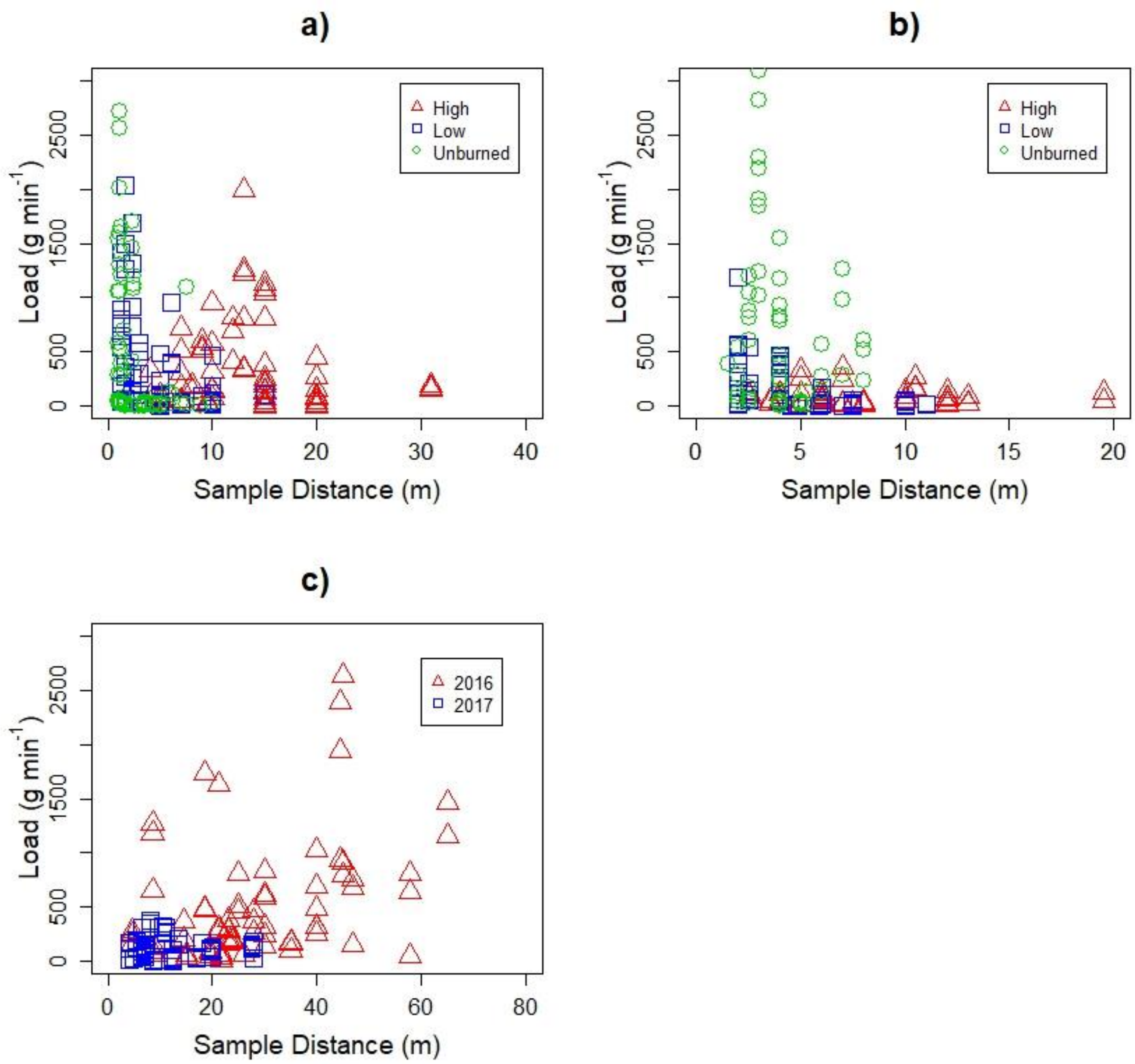

Figure 1.5. Sediment load per distance. a) 2016 North Star Fire; b) 2017 North Star Fire; c) Cayuse Mountain Fire high severity both years. *n*=72 for each plot.

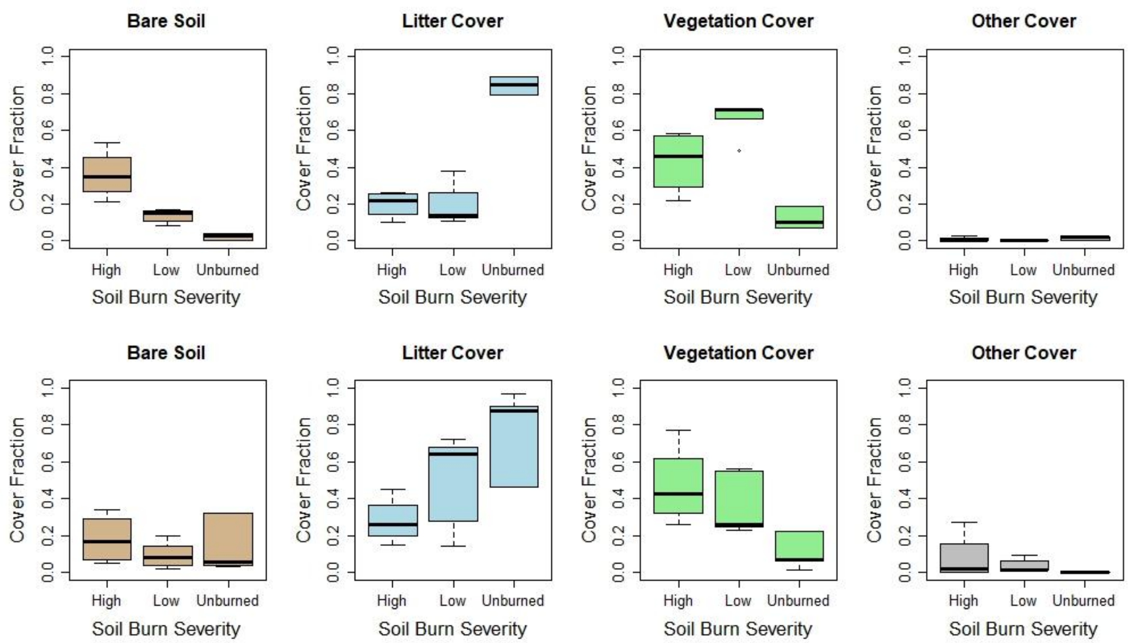

Figure 1.6. Ground cover results at the North Star Fire. 2016 is shown on the top row, and 2017 is shown on the bottom. *n*=4 for each plot.

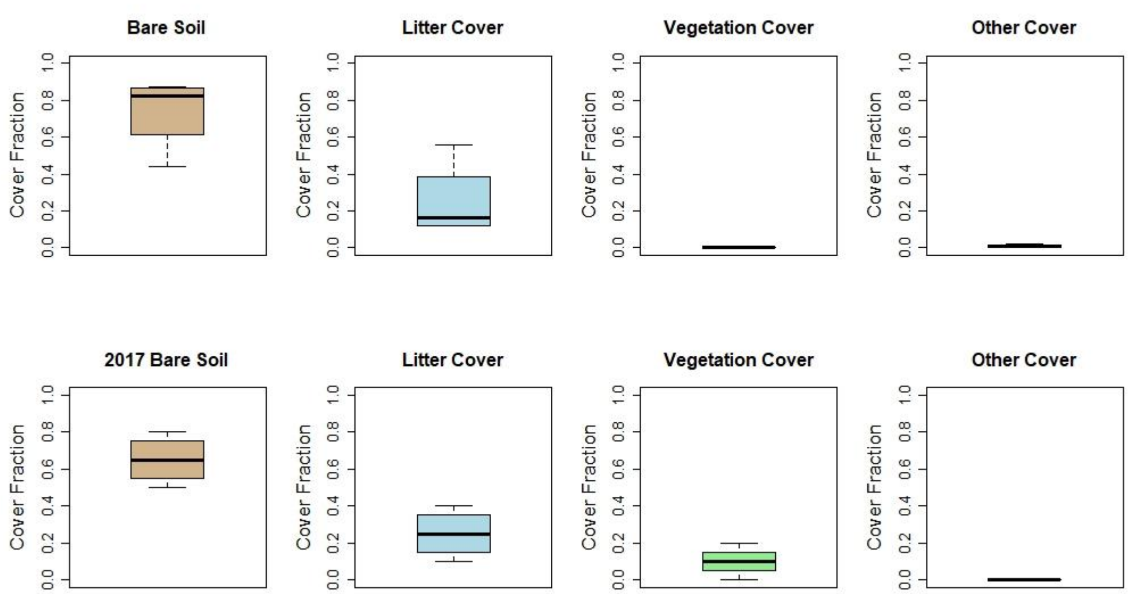

Figure 1.7. Ground cover results at the Cayuse Mountain Fire. 2016 is shown on the top row, and 2017 is shown on the bottom. *n*=4 for each plot.

Figure 1.8. 2016 site images. a) North Star high soil burn severity; b) North Star low soil burn severity; c) North Star unburned; d) Cayuse Mountain high soil burn severity.

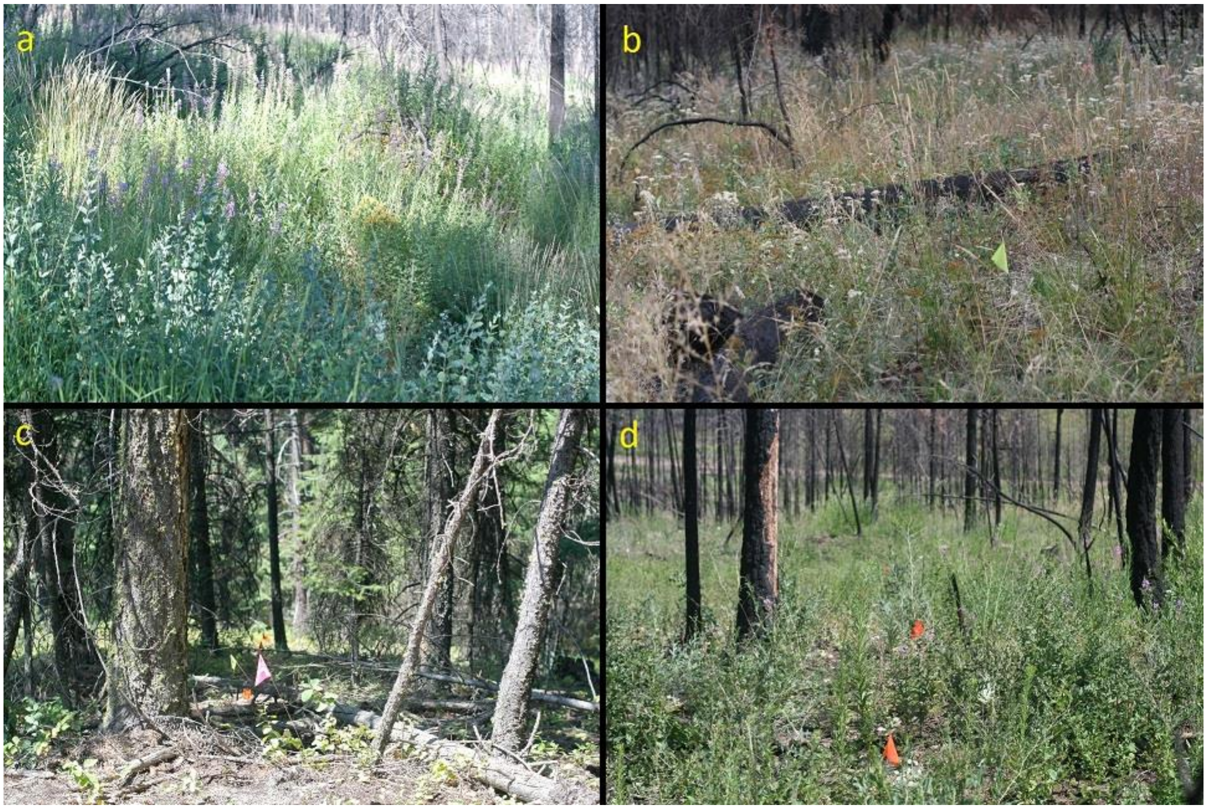

Figure 1.9. 2017 site images. Top-left: North Star high soil burn severity; top-right: North Star low soil burn severity; bottom-left: North Star unburned; bottom-right: Cayuse Mountain high soil burn severity.

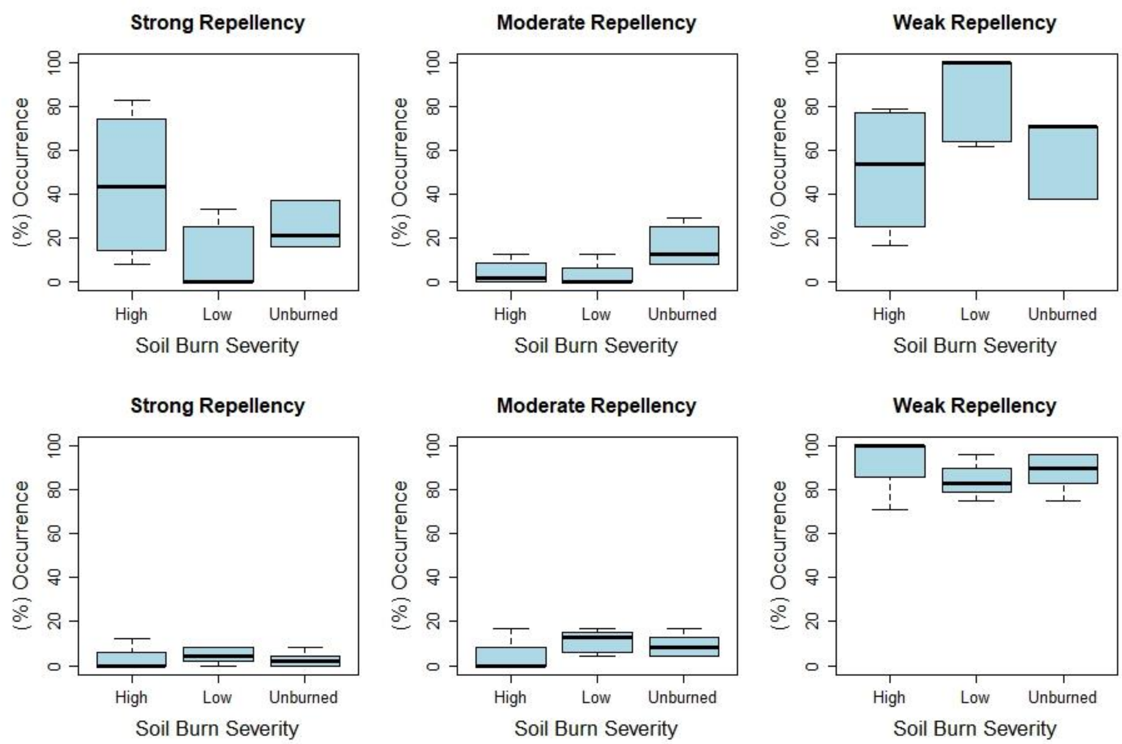

Figure 1.10. Water repellency results at the North Star Fire. 2016 is shown on the top row, and 2017 is shown on the bottom. *n*=4 for each plot.

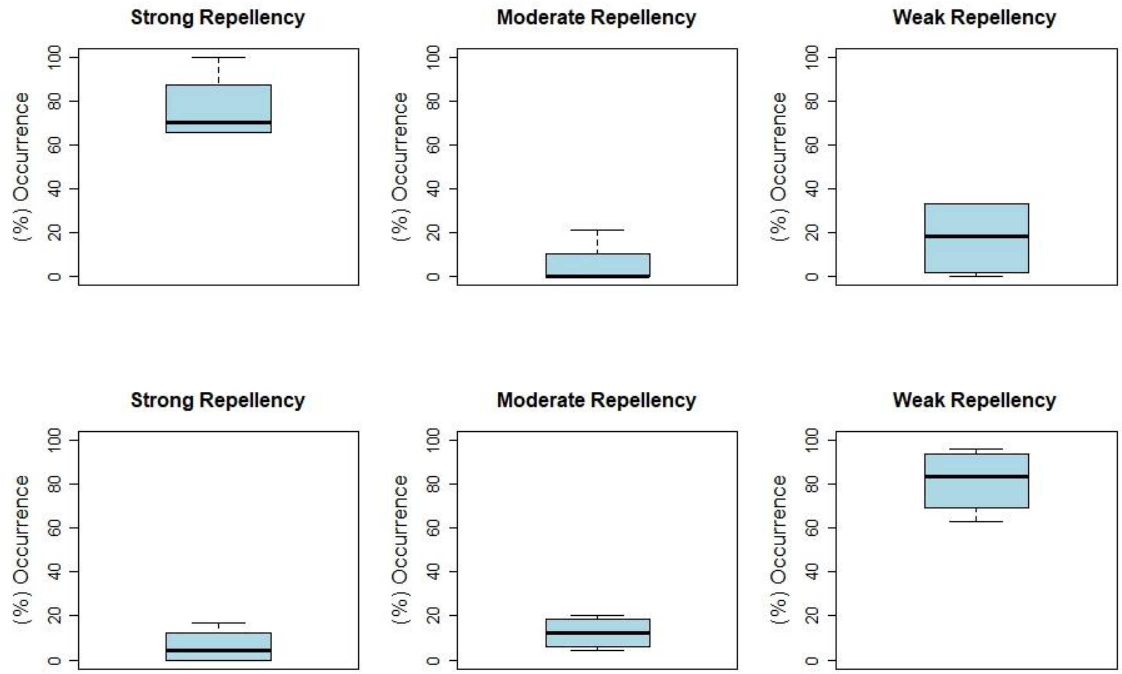

Figure 1.11. Water repellency results at the Cayuse Mountain Fire. 2016 is shown on the top row, and 2017 is shown on the bottom. *n*=4 for each plot.

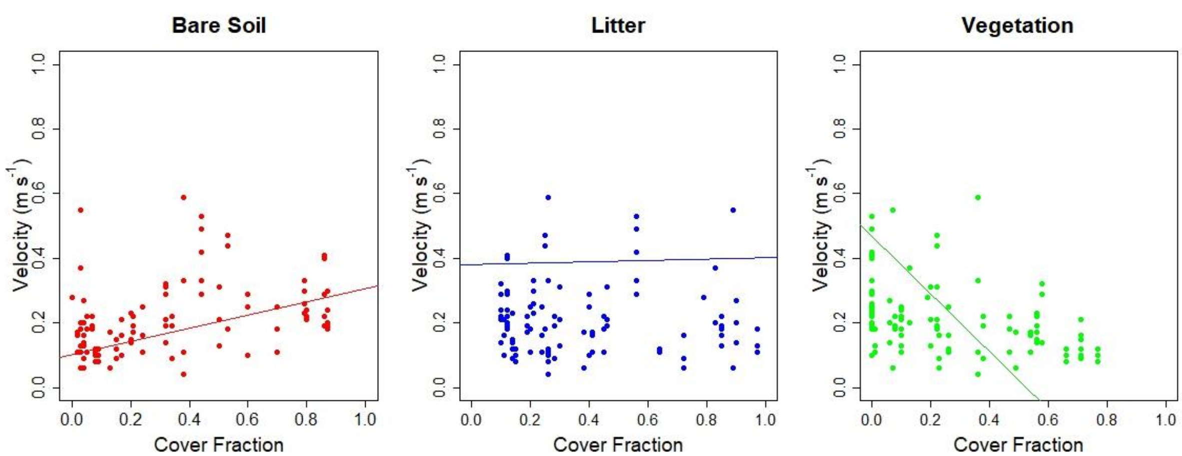

Figure 1.12. Rill velocity measured over all test sites by cover class. *n*=128 for each plot.

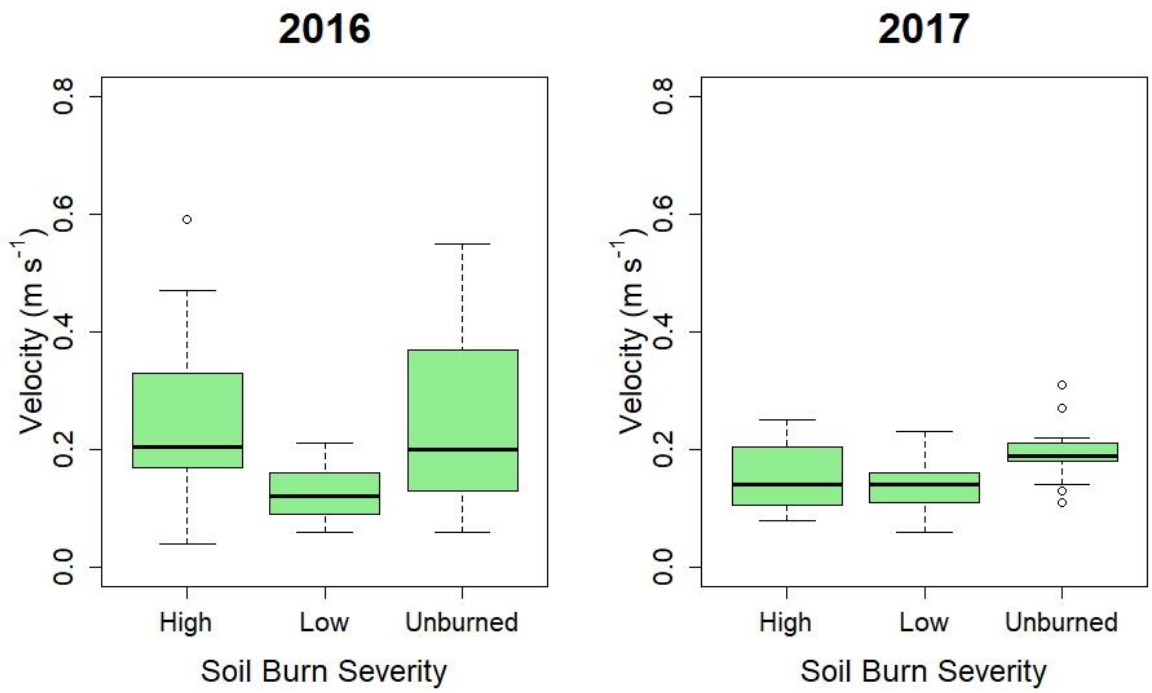

Figure 1.13. North Star Fire rill velocities by soil burn severity and year. *n*=16 for each plot.

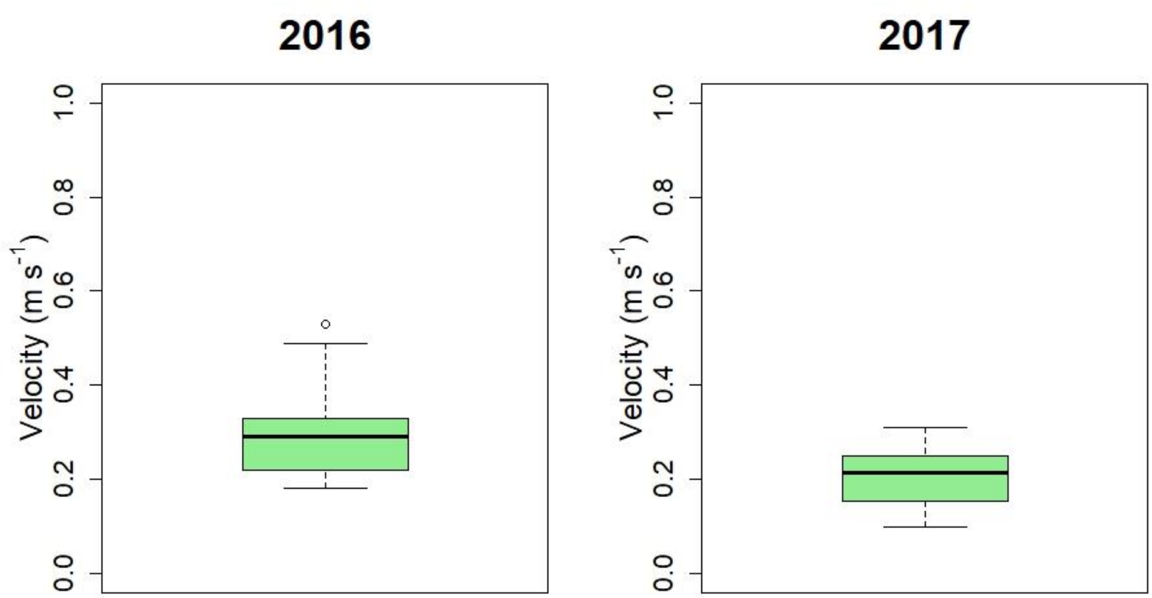

Figure 1.14. Cayuse Mountain Fire rill velocities by year. *n*=16 for each plot.

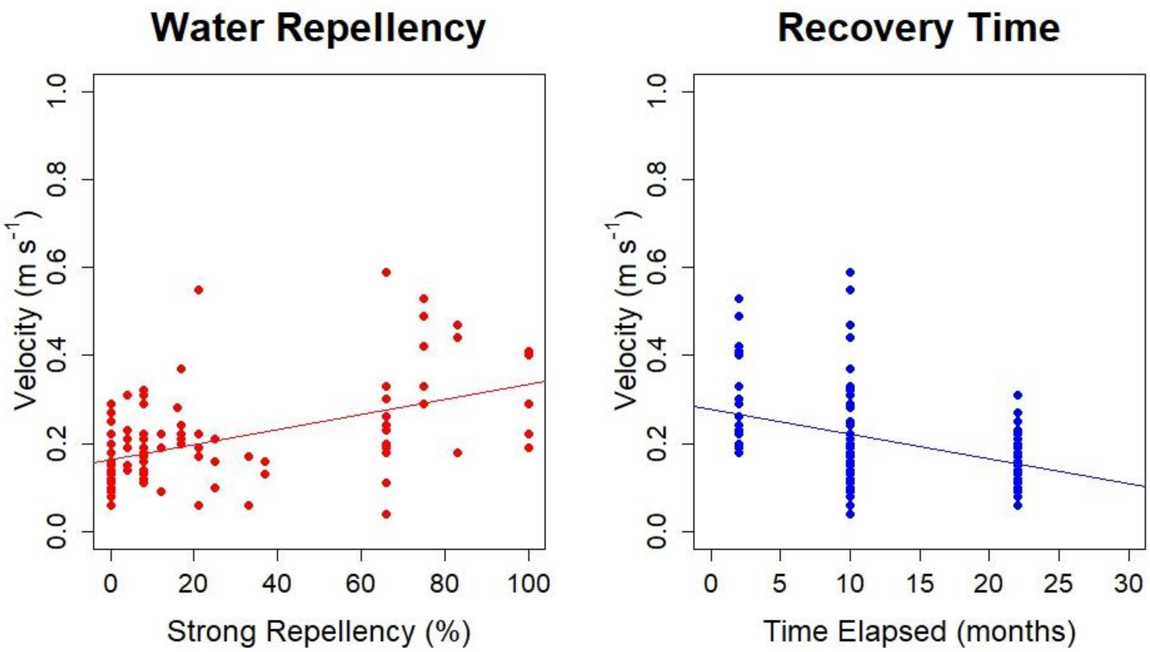

Figure 1.15. Rill velocity measured by water repellency and time. Includes all sites. *n*=128 for both plots.

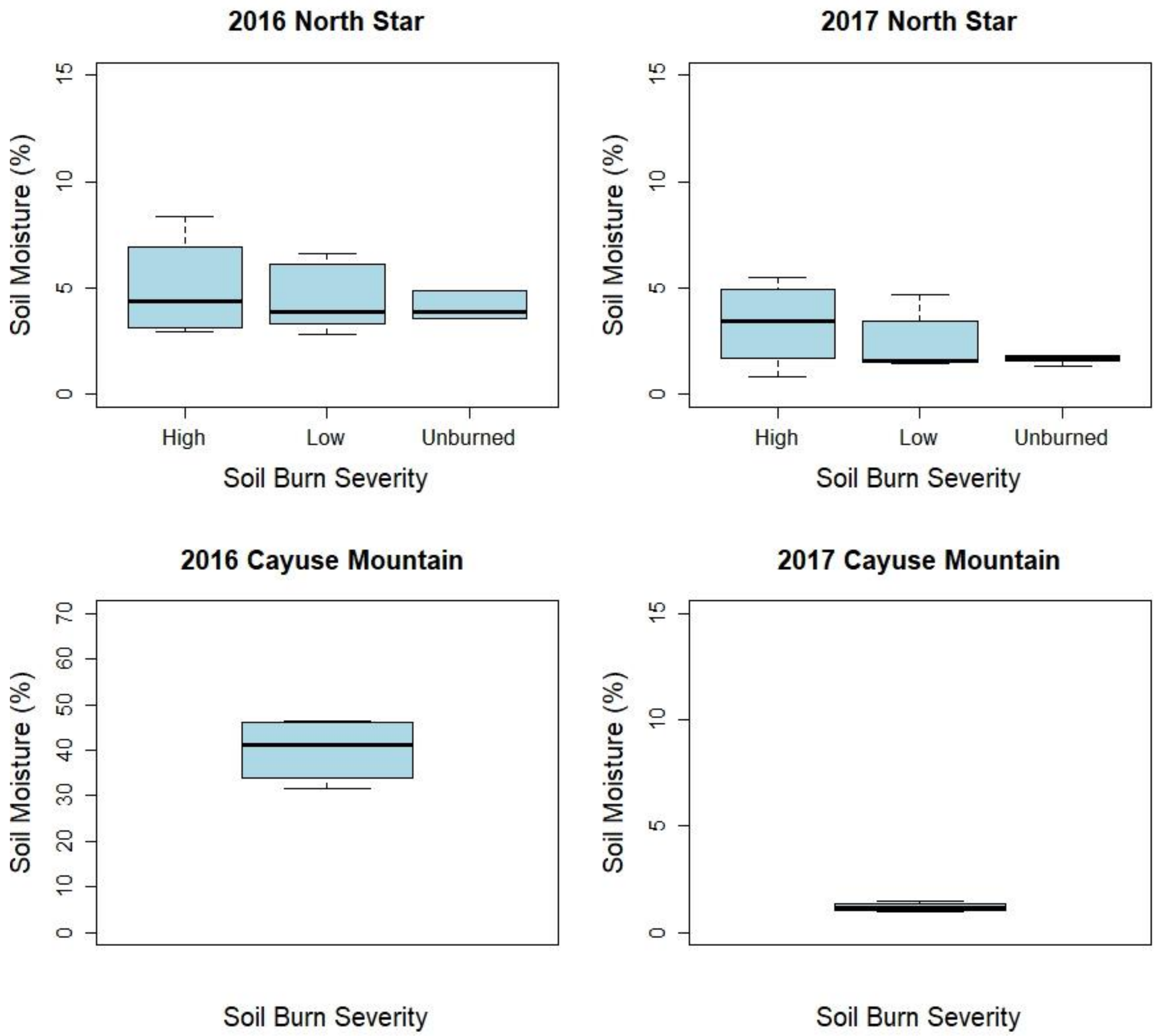

Figure 1.16. Soil moisture measured by fire location and year. Weight (%) of sample. *n*=4 for each plot.

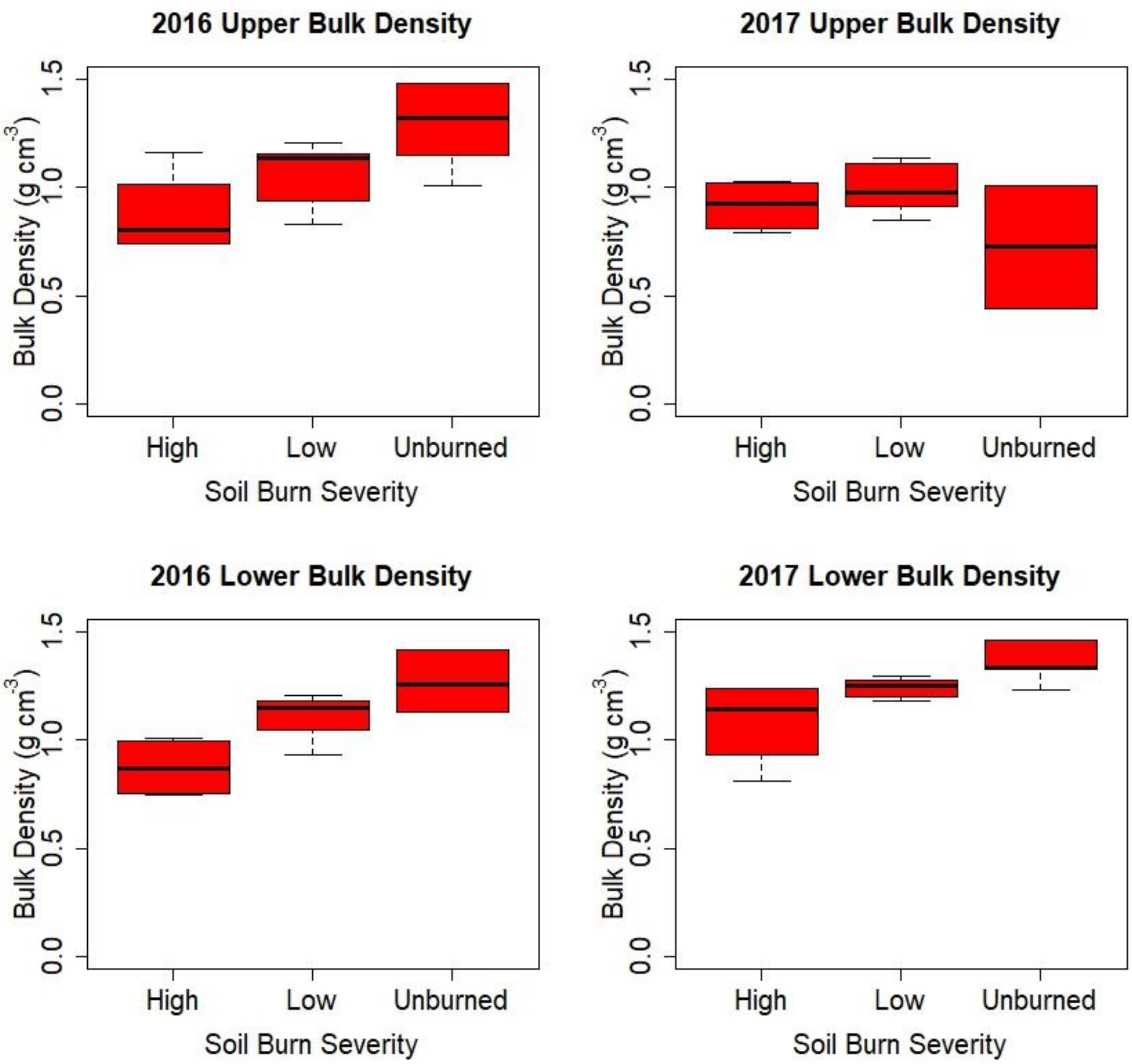

Figure 1.17. North Star Fire bulk density measurements. Separated by soil burn severity, year and depth; upper (0-5 cm) and lower (5-10 cm). *n*=4 for each plot.

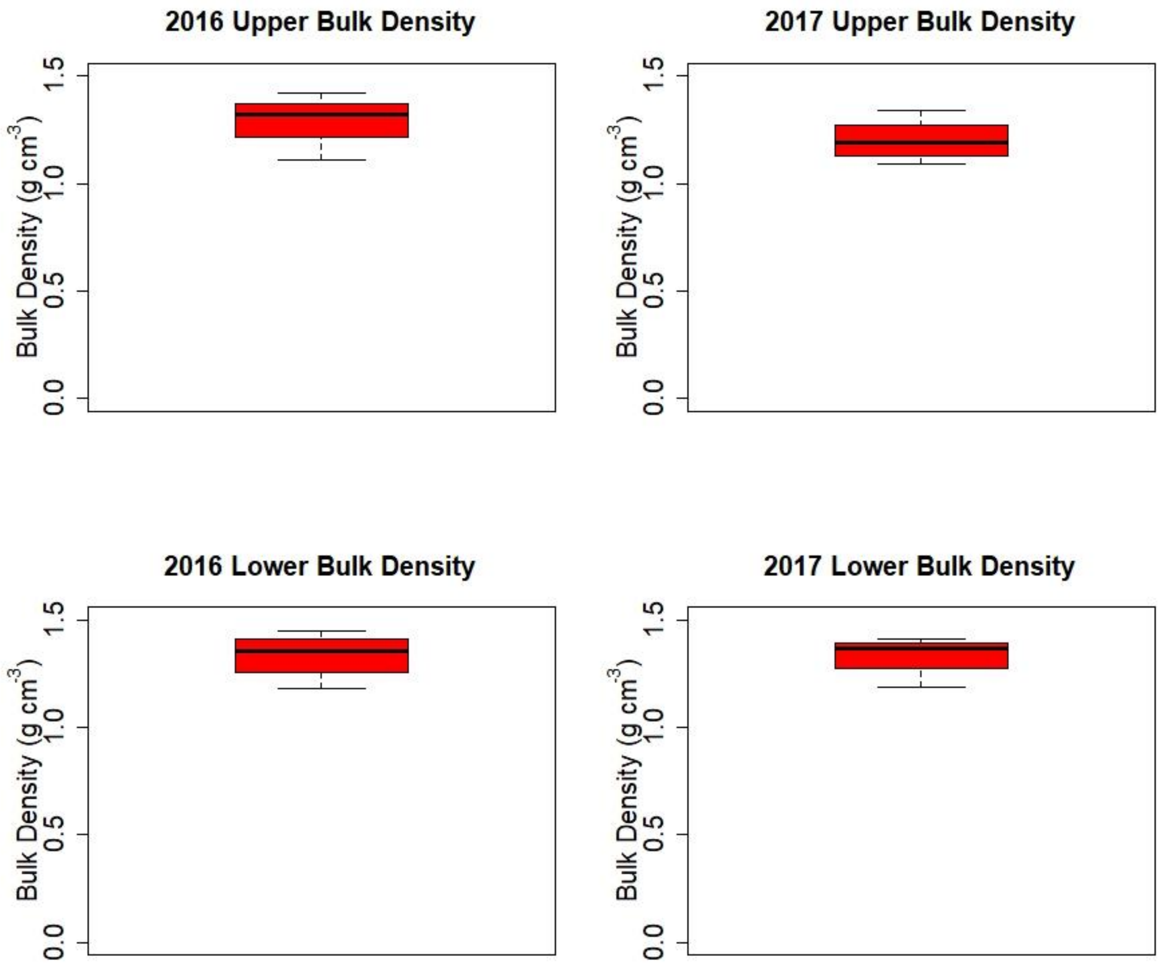

Figure 1.18. Cayuse Mountain Fire bulk density measurements. Separated by year and depth; upper (0-5 cm) and lower (5-10 cm). *n*=4 for each plot.

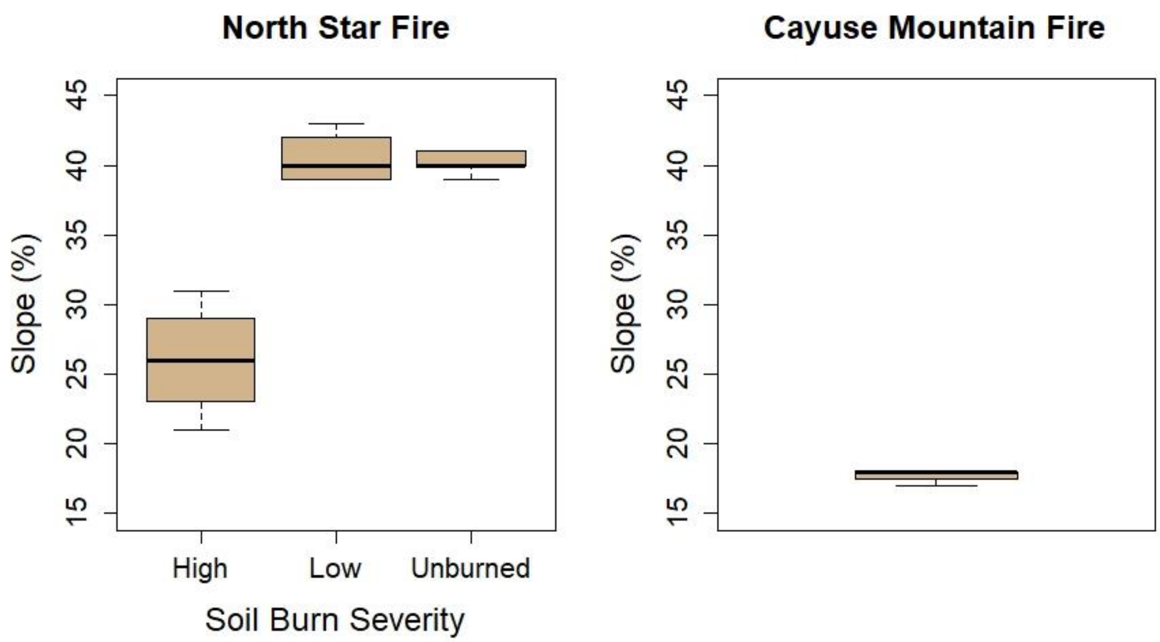

Figure 1.19. Rill slope gradients by fire and soil burn severity. *n*=4 for each plot.