EVALUATING THE EFFECT OF ROAD DECOMMISSIONING PRACTICES ON SOILS, SUB-SURFACE WATER, INSECTS AND PLANTS

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Section 1: General Introduction to Forest Road Decommissioning

Since its creation in 1905, the USDA Forest Service has been involved in debates over how to best manage public lands. In 2003, the former Forest Service chief listed the four main threats to America's grasslands and forests: invasive species, fuels and fires, disappearance of open space, and unmonitored recreation. The common link among these threats was Forest Service roads (Grace and Clinton, 2007).

Roads have been important to generations of humans for maintaining economic activities, accessing recreation sites, and general transportation (Lugo and Gucinski, 2000). Many forest roads were originally constructed for the purpose of timber harvesting or to create fire breaks. Consequently, the quality of construction of many of these "temporary" roads was related to the time it would take to complete a timber harvest. When these roads are no longer used for their original purposes, the cost of maintaining them may be outweighed by negative impacts including loss and fragmentation of habitat and changes in hydrology and geomorphology (Switalski *et al.*, 2003). The use of forest roads for recreational purposes has increased almost 20% since the 1940s, and the excess traffic has caused accelerated erosion and the need for increased road maintenance (Grace and Clinton, 2007). Decisions about whether to maintain or decommission roads are complex and controversial. Often, debates associated with decisions regarding forest road decommissioning have centered on competing land use interests such as recreation, timber harvesting, and biodiversity (Lugo and Gucinski, 2000).

Human uses of forest roads including natural resource extraction, recreation, and improper use of roads can lead to decreased quality of aquatic and terrestrial habitat (Switalski *et al.*, 2003). Damage to soil resulting from forest roads includes removal of the organic layer and topsoil and compaction and erosion of the exposed soil, which can lead to nutrient loss from the ecosystem (Kolka and Smith, 2004). Soil compaction resulting from the creation and use of unpaved forest roads can impair soil infiltration, which can lead to high levels of erosion. Increased erosion can result in increased sedimentation of streams (Brady and Weil, 2007). The decommissioning of forest roads is one way to restore habitat and ecosystem function (Thompson, 2008) and is one aspect of the Gallatin National Forest's Travel Management Plan. Decommissioning of forest roads is defined by the US Forest Service as "activities that result in the stabilization and restoration of unneeded roads to a more natural state" (Foltz *et al.*, 2007).

Section 1.1: Forest Road Decommissioning: From Policy to Practice

Since the 1980's, "increasing demand, new information on the potential effects to resources, and diverse personal value sets have raised more attention and concern as to how the public uses the Forest" (Anon, 2010). Historically, forest management plans have not accounted for changes in road management aside from mandated requirements of extermal agreements such as those imposed by the Endangered Species Act or the Memorandum of Understanding and Conservation Agreement with the Bureau of Land Management for westslope cutthroat trout (Anon, 2006a).

The Gallatin National Forest (GNF) was established in 1899 (Erikson, 2008). In the last 30 years, the amount of land managed by the GNF and the number and variety of land uses has increased greatly (Anon, 2006b). The legacy left by the timber industry includes a complex and extensive system of roads throughout the forest used for automobiles, all terrain vehicles (ATV), and snowmobiles. Thus, GNF land historically used for timber harvest and grazing has recently seen a drastic increase in the amount of recreational traffic. In 2005 the Forest Service began the process of making a comprehensive travel management plan for the entire GNF. The intent of this plan was to "identify and establish opportunities for public recreation use and access using the Forest's road and trail system." The plan established goals and objectives for managing these uses (Anon, 2006c):

> "The overall goal is [to] improve the hydrologic function of the watersheds and reduce sediment delivery into streams…[with the objectives to] Eliminate detrimental uses on those roads identified for decommissioning; Obliterate user-built trails not part of the Gallatin National Forest transportation system…; improve infiltration of water into the road surface to help restore surface and subsurface water flows [and to]reestablish natural stream flows where roads cross streams."

The National Environmental Policy Act required an Environmental Impact Statement (EIS) for a portion of the GNF in December 2006. The EIS involved analysis of seven different alternatives, including no new action, for a forest travel plan. The Forest Service decided to use a modified version of an alternative that would leave the total amount of publically accessible roads generally unchanged (~740 miles), would increase the amount of passenger vehicle accessible roads to 400 miles, reduce ATV and motorcycle use trails to 145 and 279 miles, respectively, and decrease the amount of area open to snowmobile use to 53% of the total area of the GNF. The selected alternative included directives to "close or restore non-system and user-built roads" (Anon, 2006c). This particular directive to "close or restore," or decommission, certain roads was created in an effort to reduce redundant and unnecessary roads (Story, 2010). The Forest Service determined that some roads within the GNF were not needed and were having a negative contribution to surrounding ecosystems (Anon, 2006d).

As part of the EIS, the Forest Service completed an Environmental Assessment (EA) of road decommissioning in 2006 for the Bangtail Mountains portion of the GNF. The EA stated an overarching concern about "reduc[ing] the level of sediment entering streams that is attributable to roads and trails on National Forest System Lands." According to the EA, levels of sediment were found to have been washing off roads and road culverts at levels harmful to fish habitat and riparian areas. By taking roads out of public use (through various methods) the Forest Service hopes to eventually reduce sediment levels in streams and also help restore natural infiltration of surface water into groundwater. (Anon, 2006d)

Because the Forest Service recognized the potential for short term increases in sediment loading due to disturbances created by decommissioning techniques. Consequently, the Bangtail EA included many mitigation measures to help decrease this impact and enacted extensive monitoring protocols. Monitoring measures as listed in the Bangtail EA included annual assessment of sites, observation to determine weed abundance, and inspections to "determine success in stopping vehicle use, reestablishment of hydrologic function, and growth of seeded grasses" (Anon, 2006d). The completed Bangtail EA concluded no finding of significant negative environmental impact; therefore, a more comprehensive and expensive EIS was not necessary (Anon, 2006). The Bangtail EA established monitoring criteria relevant to road decommissioning efforts in other parts of the GNF.

Section 1.2: Road Decommissioning Practices

The Forest Service has adopted several techniques that are used in a variety of road decommissioning situations. The most commonly used methods of road removal are gating the road to restrict access, stream crossing restoration, ripping, and recontouring hillslopes (Switalski *et al.*, 2004; Switalski *et al.*, 2003; Coghlan and Sowa, 1998). The ripping technique involves dragging a specialized plow over the roadbed surface to a depth of 30-90 cm (Switalski *et al.*, 2004). Stream crossings are restored by the removal of culverts and recontouring the slopes on either side of the stream (Switalski *et al.*, 2003). A full slope recontour is the most costly and complete form of road removal and involves removing the fillslope that was moved during road construction and placing it in the cutslope (Figure 1.1). After ripping or recontouring, the area is usually reseeded and strewn with native organic debris (Switalski *et al.*, 2003). For instance, fertilizers, bio-solids (wastewater leftovers), and mulches are often added to assist in nutrient cycling and to enhance revegetation (Switalski *et al.*, 2004). The aspects of road removal are multifaceted and include hydrologic, geomorphic, and ecological concerns. As part of our senior capstone project, we evaluated how ripping and reseeding, recontouring and reseeding, and no treatment affected soil compaction, water infiltration, insect diversity, and plant and weed richness and abundance at various study sites within the Hyalite Canyon of the GNF (see Section 1.3).

Research has shown that forest roads can cause increased erosion and negative impacts on water quality (Grace and Clinton, 2007). Water flows are concentrated by roads from higher hillslopes which can lead to a smaller first order drainage basin and an overall longer channel network as well as altering subsurface flows (Alexander and Forman, 1998). Exporting sediments via the network of forest roads is a large concern for forest management (Grace and Clinton, 2007). Recontouring of road cut hillslopes has been shown to significantly decrease sediment yield and erosion from those hillslopes (Madej, 2001) and thus is a common treatment used on steep, road-cut hillslopes.

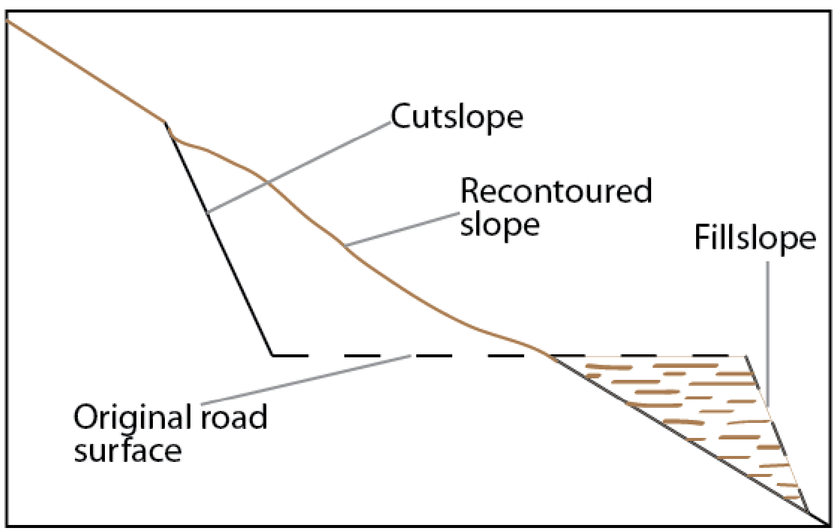

Figure 1.1: Diagram of a recontoured road.

Another common issue associated with forest roads is soil compaction and erosion. Compaction of a soil system can result in decreased porosity, with porosity being dependent on pore size distribution, soil texture and structure, organic matter percentage, and soil-moisture tension (Gebhardt *et al.*, 2009). It is hypothesized that ripping decommissioned roads should increase infiltration and decrease erosion, with the extent of improvements depending upon slope stability, soil texture and revegetation success (Switalski *et al.* 2003).

A collective goal of road decommissioning is improving overall ecological function and one way to determine this is by species richness of plants as well as insects. Insects are considered to be effective bioindicators, which could be indicative of ecological health (Da Mata *et al.*, 2007). Insects are useful for two reasons: low resilience and short life-cycles making them sensitive to small environmental changes (Brown, 1997; Michaels, 2007). Using indicators to their best advantage will require understanding the responses of target species to environmental disturbance in both spatial and temporal scales (Michaels, 2007).

One of the most obvious visual aspects of road decommissioning and subsequent ecological restoration is the presence of vegetation (Switalski *et al.*, 2004). Roadsides generally have high species richness with a distinct presence of disturbance tolerant plant species (Alexander and Forman, 1998). When roads are removed, establishing vegetation is essential, and the process of ripping the road bed and adding amendments such as organic matter can greatly speed up the rate at which vegetation is reestablished (Switalski *et al.*, 2004). Evaluating species richness off the road and within different decommissioning treatments would be beneficial to understanding how well these different approaches work.

Another aspect of vegetation reestablishment in association with road sites is the presence or absence of introduced species. The disturbance associated with roadways generally harbors more weedy species (Alexander and Forman, 1998), and the disturbance provided by decommissioning may increase this presence and lead to invasion further into the forest (Parks *et al.*, 2005). However, many non-native species are often incapable of infiltrating deep into a forested system (Pauchard and Alaback, 2006). Although it has been documented that road removal restores geomorphic and hydrologic functions and reduces erosion, additional research will be necessary to get a better picture of ecological impacts of road removal (Switalski *et al.*, 2003).

Section 1.3: General Study Site Description

The study site was located in the Gallatin National Forest, accessed by the Forest Service road in Hyalite Canyon, roughly 25 km south of Bozeman, MT. The area covers several square kilometers from Langhor Campground south to History Rock. The average elevation is approximately 2000 m, average air temperature is 1-3 ºC, average precipitation is 65 to 90 cm, with a frost free period of 50 to 70 days. The study sites are mostly on steeper slopes from 10 to 45 percent.

The majority of the soils within the research area were of the Alfisol and Inceptisol soil orders with the dominating classification being Mollic Cryoboralfs (NRCS, 2010). According to the US Department of Agriculture, Alfisols are defined by having an argillic, kandic or a natric horizon with a base saturation greater than 35% and are typically found under forests or mixed vegetative cover. Inceptisols have minimal horizon development and have been found under a wide range of ecological settings. They are generally found in mountainous areas on steep slopes, young geomorphic surfaces and, on resistant parent materials. The geologic parent material for the area was diverse and contributed to the variation in soil type and vegetative cover that was observed.

Vegetation on these gravelly loam soils typically consisted of spruce (*Picea spp.)* and sweetscented bedstraw as well as subalpine fir (*Abies lasiocarpa*), with huckleberry shrubs (*Vaccinium deliciosum*) (NRCS, 2010). Higher up along the mountain slopes, the soil parent material changed to a loamy colluvium derived from volcanic breccia with a silt loam to sandy, clay loam soil profile while other slopes were derived from limestone, sandstone or shale (NRCS, 2010). Vegetation on these soils were mainly subalpine fir (*Abies lasiocarpa*), with twinberry (*Linnaea borealis*), and/or huckleberry (*Vaccinium deliciosum*) (NRCS, 2010). The mountaintops and ridges, above our sites, were comprised of volcanic substratum with a very gravelly loam soil profile (NRCS, 2010). The vegetation at these higher elevations included subalpine firs (*Abies lasiocarpa*), and white bark pines (*Pinus albicaulis*), with grouse whortleberry (*Vaccinium scoparium*) understory (NRCS, 2010).

The capstone class separated into five groups concerned broadly with, soils, water, insects, native and non-native plants. While the exact sampling protocol differed between groups, all groups sampled one- three times from late June to early August before decommissioning treatments were imposed, and one to three times from late August to late September after treatments had been imposed. Four roads were identified that were ripped and reseeded (RS), another four that were recontoured and reseeded (RC), four that remained untreated (UN) and three that were ripped and reseeded 14-15 years ago (H). Locations of these roads are shown in Figure 1.2.

In the following sections, each research group has introduced a specific project, objectives, methods, results and discussion. In the final section the findings of all the groups were brought together and general observations discussed.

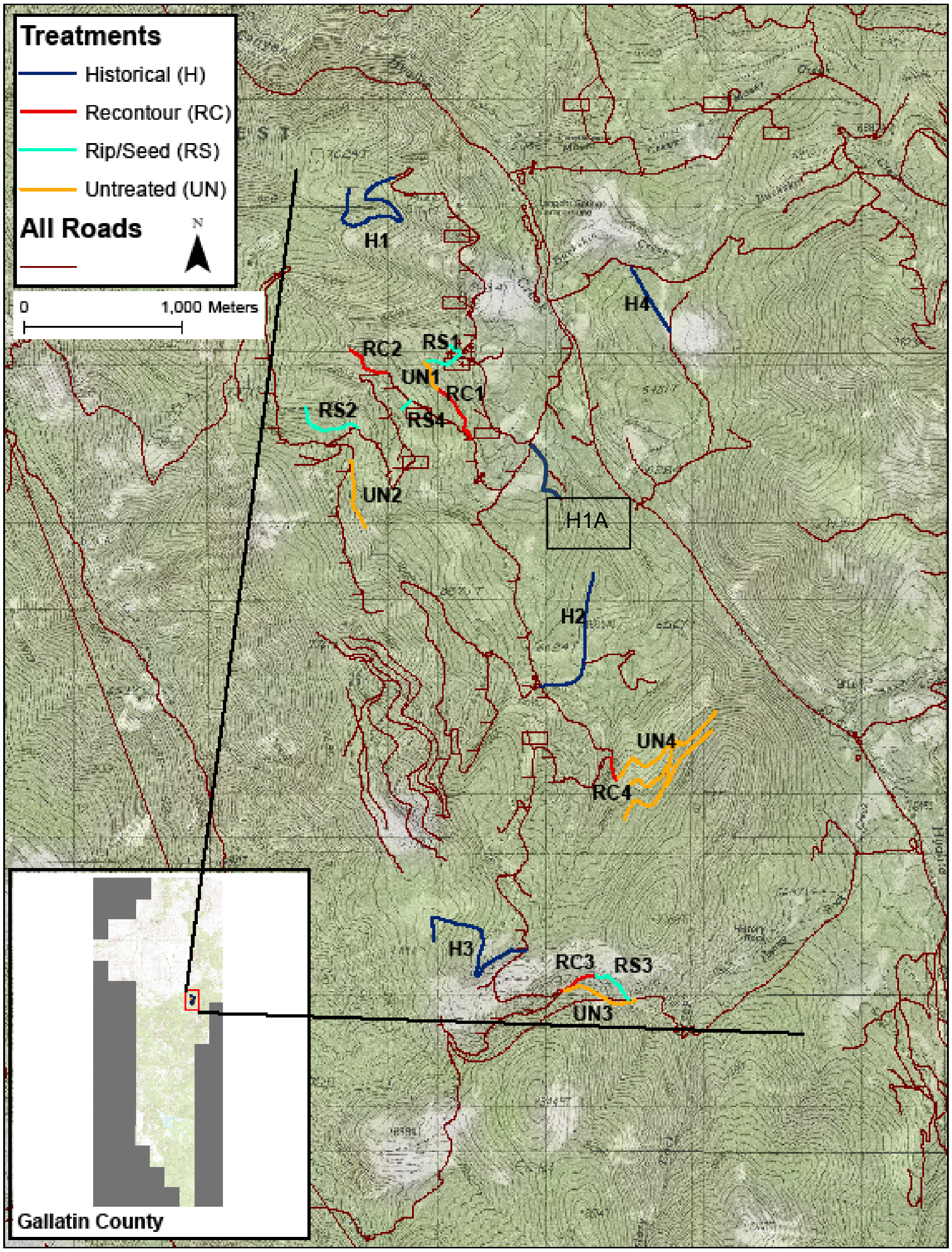

Figure 1.2: Locations and treatment types of roads in Hyalite Canyon study area.
Note that H1 as labeled here was also decommissioned for a second time in 2010 due to undesired public access. The road labeled H1A was therefore used instead as it was also decommissioned in 1994/5.

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Section 2: Soil Compaction

Researchers: Anna Bergstrom, Kendra E. Kaiser, Galen H. Laird, Christine R. Miller.

Section 2.0: Abstract

Increased soil compaction on forest roads decreases infiltration and can increase the runoff of these roads. Bulk density and cumulative infiltration measurements were taken in order to assess the effectiveness of the different road commissioning treatments used (recontoured and reseeded, ripped and reseeded, and untreated) in the Hyalite Canyon area, on soil decompaction. Cumulative infiltration was found to be higher off road than on road, and the cumulative infiltration of a historically ripped site was found to be significantly different between on and off road sites. Although there was no significant difference between on and off road sites prior to the decommissioning, the bulk density values from the recontoured sites indicate that they became more similar following treatment. It was found that the most intensive the decommissioning treatment, recontouring, had the greater the reduction in soil bulk density. The historic road was found to have statistically different bulk densities when comparing on road and off road. It was also found to have a median very similar to that of a site that was ripped recently (Summer 2010), suggesting that it has not returned to a pre-road condition.

Section 2.1: Introduction

Soil compaction can be described as the reduction in soil volume resulting from a loss of natural porosity (Cakir *et al*., 2010). This causes a reduction in the infiltration to percolation ratio and increases overland flow and erosion, changing the hydrologic and ecological properties and processes (Foltz *et al*., 2007). Improving soil compaction is an important factor of restoration in order to re-establish the hydrologic and ecological properties and processes (Foltz *et al*., 2007).

There are several methods for describing soil compaction. Bulk density (or porosity) is the parameter usually used to describe soil compactness. If soil is compacted, bulk density increases and porosity decreases correspondingly (Keller and Hakansson, 2010). Infiltration is the rate at which moisture enters the soil profile. Luce (1997) measured infiltration rates with two different soils on roads directly after ripping, roads with no treatment, and ripped roads covered by straw mulch. He concluded that both ripping treatments resulted in increased hydraulic conductivity, but after a few minutes of simulated rainfall, soil particles begin to clog macropores and infiltration rates dropped. Little research exists on the durability of the increased infiltration rates over time, but it appears that ripping only produces short-term gains (Luce, 1997). Anecdotal observations by Luce (1997) revealed that after one winter, ripped road surfaces were nearly as solid and dense as the original road surfaces. Incorporating organic matter into the ripped surface may be important to increase the longevity of these gains (Bergeron, 2003).

It was suggested by Gifford (1975) in his review that the effectiveness of ripping treatments on compacted rangelands decreased over a period of years. However, in western Montana, Bradley (1997) found that ripping improved infiltration rates three months after treatment. Kolka and Smith (2004) suggested that the use of recontouring techniques on forest roads lead to lower bulk densities, less surface runoff and sediment production, and greater seedling growth in comparison to subsoiling or no treatment. All of the roads chosen in the Kolka and Smith (2004) study were located in Kentucky and had a similar grade, aspect, and soil profile. The authors used an undisturbed hillslope as their control and found soil moisture increased downslope from recontoured sites, and moisture tended to be higher on the road for subsoiling sites. They measured surface bulk density, not subsurface, and total bulk density without removing the coarse fragments.

Generally, ripping increased vegetation reestablishment and infiltration rate, as well as discouraged weed invasion and reduced erosion potential (McNabb, 1994). It was concluded in the Switalski *et al.* (2003) review that the degree of effectiveness of road ripping was directly related to the stability of the slope (Bloom, 1998), soil texture (Luce, 1997), and the use of soil amendments (Hektner and Reed, 1989, Cotts *et al.*, 1991). These studies quantified changes directly after treatment, but did not evaluate them over a number of years. Consequently, we have determined a benefit from quantifying the relative change in bulk density and infiltration rate between the decommissioning types; as well as studying a single treatment type and its effectiveness over a longer period of time. Switalski *et al.* (2003) proposed that more research be focused on how road removal techniques affect soil aggregation. Bulk density is a measurable parameter related to soil aggregation. Future research questions were suggested including: "How does road removal influence soil aggregation and bulk density?"

It was the aim of this study to quantify soil compaction and determine which treatment type (ripped and reseeded (RS), recontoured and reseeded (RC), or untreated (UN)) achieved soil compaction levels most comparable to an adjacent off road condition. Our specific objectives are to (1) measure bulk density across the three treatment types on three sample roads/sites per treatment, two historically treated sites, and (2) measure cumulative infiltration rate on one of each treatment type.

Section 2.2: Materials and Methods

Section 2.2.1: Site Sampling

A total of 11 road sites were visited. Sampling from the same sites took place two times during the summer: once in June (before treatment) and once in August (after treatment). Three sites each were sampled from four treatment types: ripped and reseeded "RS"; recontoured and reseeded "RC"; untreated "UN"; and historically ripped and reseeded fifteen years ago "H". A number was assigned to each of the sites of each treatment type in order to delineate the road segment (Table 2.1). Each of the 11 site locations were logged using a GPS unit, and marked with yellow flags in June.

Treatment Type	Site	June	August
Recontoured	RC1	BD	BD
	RC2	BD, Inf	BD, Inf
	RC3	BD	BD
Ripped	RS1	BD	BD
	RS2	BD, Inf	BD
	RS3	BD	BD
Historic	H2		BD
	H3		BD, Inf
Untreated	UN1	BD	BD
	UN2	BD, Inf	BD, Inf
	UN3	BD	BD

Table 2.1: RC=Recontoured and reseeded, RS=Ripped and reseeded, UN=Untreated road site. The number associated with each of the road sites delineates the particular road segment at a known location. BD=Bulk density sampling, Inf= Infiltration sampling.

Section 2.2.2: Bulk Density Sampling

Within a site we randomly chose a flat area of the road, and marked transects as specified in section 2.2.1. Eight samples were taken along each of these transects, two off road samples on each side of the road, and four on the road (Figure 2.1). The pre-treatment samples were dug with a hand trowel (holes 5-10 cm deep and 5-10 cm diameter) volumes were found by lining the hole with a plastic bag and determining the volume of water using a graduated cylinder. The post-treatment samples were taken by using one of two soil corers of known volume (either 177.157 or 442.25 ml) at the RS, RC and UN, as well as two H sites.

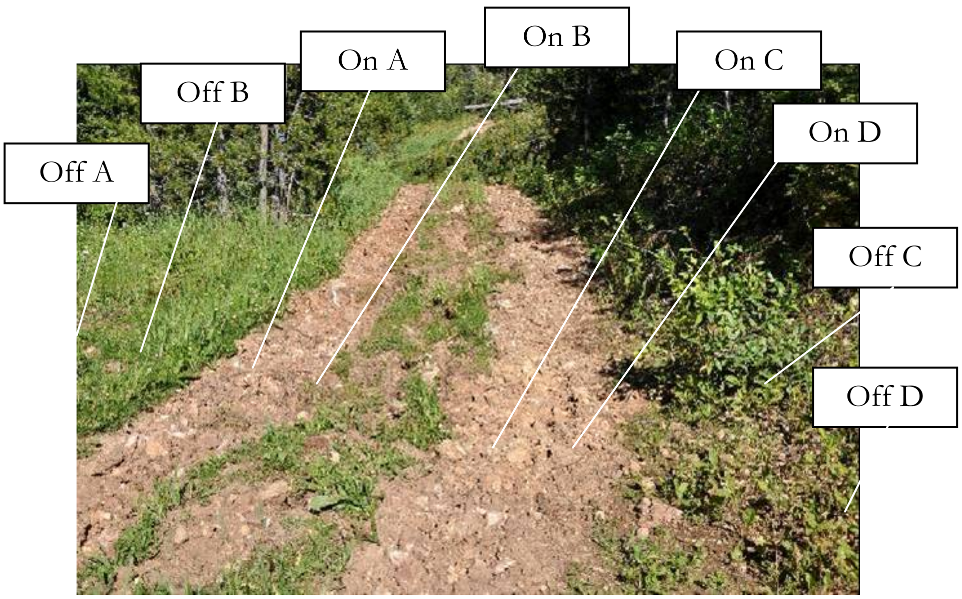

Figure 2.1: Bulk density sampling locations at example ripped and reseeded (RS) site.

Section 2.2.3: Bulk Density Data Analysis

In the laboratory each soil sample was dried at 110 °C for 24 hours. An adjusted bulk density without coarse fragments was determined by grinding and sieving the soil with a 2 mm sieve. The sieved soil was reweighed, and the volume of coarse fragments was determined by assuming a bulk density of 2.65 g/cm^3. By finding the difference in weight by the removal of coarse fragments, we could divide by the rock bulk density (2.65 g/cm^3) to determine a volume change. This procedure provided an adjusted volume, from which we could determine an adjusted bulk density of only the soil by using this volume and the dried weight.

Section 2.2.4: Infiltration Sampling

Infiltration data was collected in June and August (Table 2.1). Double-ring infiltrometers were used. The inner ring of the infiltrometer was 28 cm and the outer ring was 53 cm and the height of both was 30 cm. Infiltrometers were pounded into the ground at least 2 cm with the goal of 5 cm. This proved difficult because of the rocky substrate. If the infiltrometer was not pounded in to a 5 cm depth, mud and soil were packed around the outside of the infiltrometer to keep the water from escaping.

Once the infiltrometers were in place, both the outer and inner rings were filled simultaneously with approximately 6 cm of water and the first measurement was immediately taken. Two infiltration readings were taken on-road, and two were taken off road. The water height was recorded into the field notebook every three minutes, until a constant rate was achieved, or after 30 minutes, whichever came first. Water levels in the rings were kept at approximately the same level and the rings were not allowed to go dry. If the water level dropped, the infiltrometers were refilled, and the new levels were recorded. Water height was measured with a standard ruler at the same location on the inside of the infiltrometer, marked with two vertical lines, to eliminate error due to uneven ground surfaces.

Section 2.2.5: Data Analysis

Exploratory data analysis was performed on the bulk density data to identify specific questions to be answered statistically. First, we inquired whether there was a difference between on and off road bulk density at all pretreated sites across all treatments. Next, we compared on road bulk densities before and after treatment, by treatment type. Additionally, we asked whether there was a difference between on and off road bulk density in following treatments grouped by treatment type to show how each treatment compared to its adjacent off road conditions. We then compared post-treatment on road bulk density by treatment type to compare the effects of treatments with each other. This last question included the historic treatments to allow us to compare previously treated roads to those just treated. Analysis of Variance (ANOVA) models were used to analyze the bulk density data and pairwise comparisons were made using Tukey's Honestly Significant Differences (HSD). The level of significance selected for our pairwise comparisons was set at 0.01 and was chosen due to the variability and heterogeneity of forest soils, the small sample sizes, and accuracy of our sampling methods.

Section 2.3: Results

Section 2.3.1: Bulk Density Results

Comparing the untreated site data between pre and post-treatment samples, no significant difference was found between the on and off road samples. However, the medians appeared to be higher on road than off road pre-treatment (Figure 2.2). At the later sampling on the untreated sites, the off road bulk density was statistically lower than on road ($p = 0.051$; Figure 2.2).

For all of the ripped and reseeded sites no significant difference was found between the on and off road samples before or after treatment ($p= 0.27$ and 0.20, respectively). However, the medians tended to be higher on road than off road both before and after treatment (Figure 2.3).

The recontoured and reseeded sites also showed no significant difference between on and off road samples before or after treatment ($p= 0.35$ and 0.60, respectively). However, these p-values show us that the median bulk density values became more similar following treatment (Figure 2.4).

Bulk density samples on the historic sites were only collected in August, and a significant difference was found between on and off road sites ($p= 0.004$). The difference in median bulk density between the on and off road sites was 0.41 g/cm^3 (Figure 2.5). When comparing treatment effects based upon their respective bulk densities in August (Figure 2.6), the only significant difference found was between untreated sites and recontoured sites, with a p-value of 0.01. However, the medians appeared to differ between all treatment types with the untreated sites having the highest bulk density and the recontoured sites having the lowest bulk density. The historic sites had similar median bulk density to ripped sites and those values fell between the untreated and recontoured bulk density values.

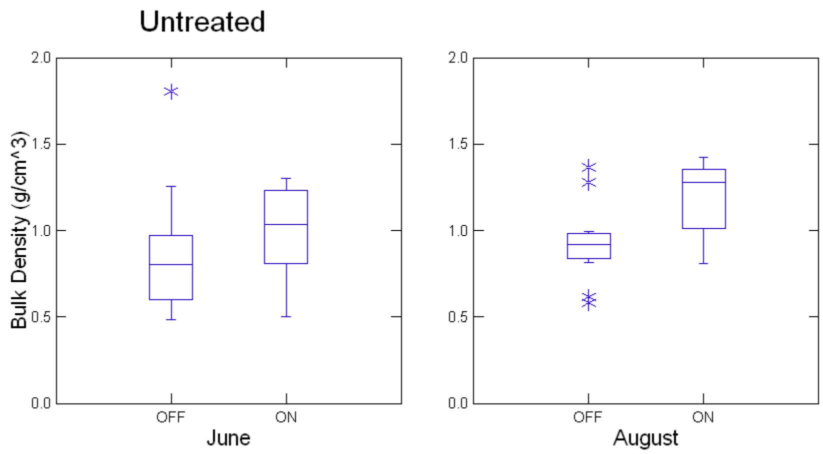

Figure 2.2: Box plots of all untreated site data for June (left plot) and August (right plot). No significant difference was observed between on and off road samples in July but was in August (p = 0.051).

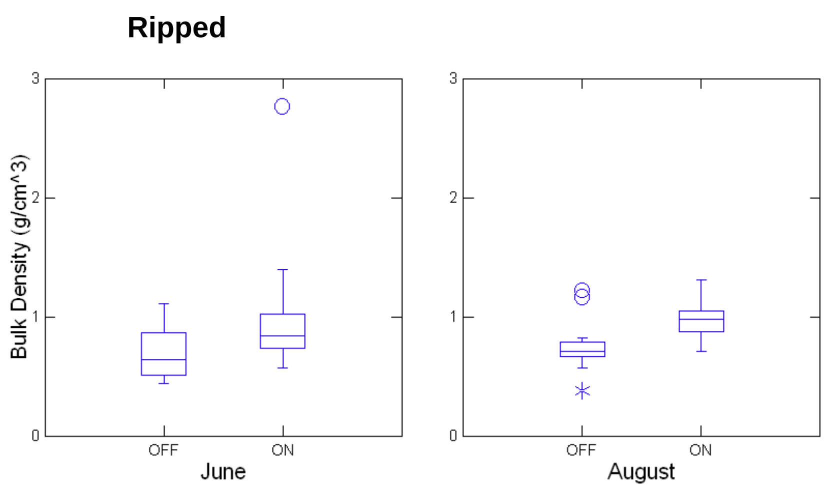

Figure 2.3: Box plots of all reseeded site data for June and August. No significant difference was observed between on and off roads in June or in August (p = 0.27, 0.20, respectively).

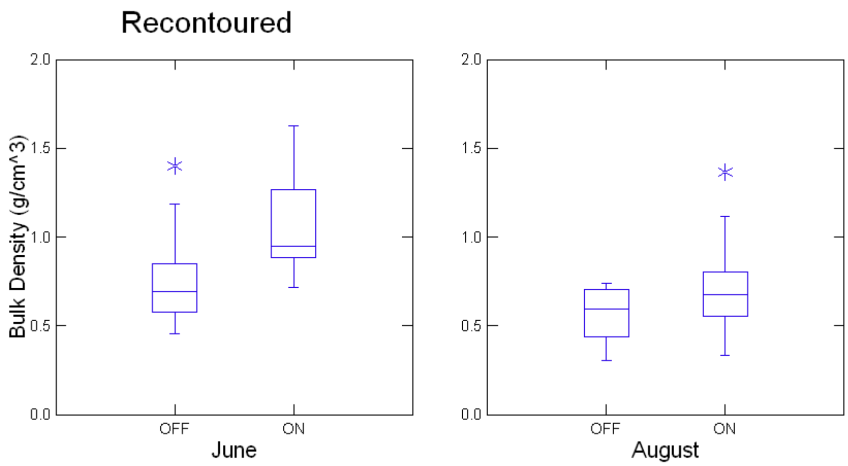

Figure 2.4: Box plots of all recontoured site data for June and August. No significant difference was observed between on and off roads in June or in August ($p = 0.35$ and 0.60, respectively).

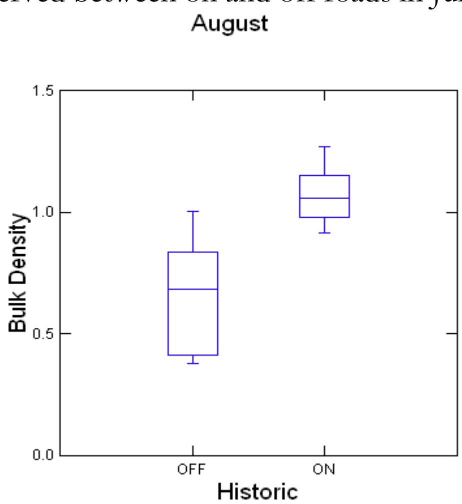

Figure 2.5: Box plots of all historic site data for August. A significant difference was observed ($p = 0.004$). The difference in mean bulk density between the on and off road sites was 0.41.

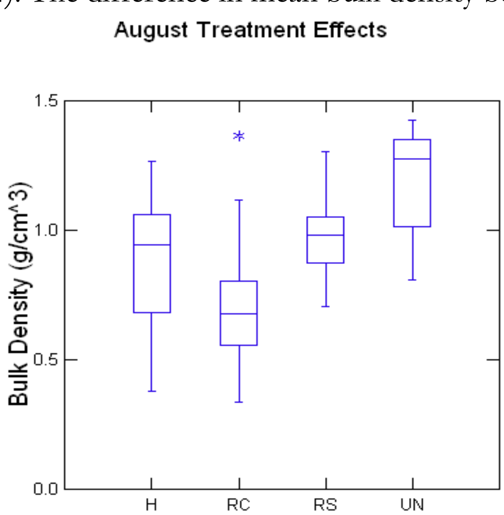

Figure 2.6: Box plots of treatments effects based upon their respective bulk densities in August. The only significant difference was between untreated and recontoured, with a p-value of 0.01.

Section 2.3.2: Infiltration Results

Infiltration data was collected on only five roads (one pre-treatment recontoured, one pre-treatment ripped, one pre-treatment untreated, one post-treatment untreated, and one historic site) due to the limited scope of the project. As a consequence of only collecting data on one site within each treatment type, the data was unable to be statistically analyzed.

Treatment Type	On Road Average Cumulative Infiltration (mm)	Off Road Average Cumulative Infiltration (mm)
Recontour/Reseed	5	117
Ripped/Reseed 2	18	119
Untreated 2	Pre-treatment: 41 Post: 16	Pre-treatment: 120 Post: 74
Historic 3	15	243

Table 2.2: Cumulative infiltration on and off road for each treatment type. Untreated 2 (UN2) was the only site sampled in June (pre-treatment) and August (post-treatment).

Infiltration rates for RS2 in June pre-treatment were higher off (labeled A) road than on road (labeled B) (Figure 2.7). The infiltrometers placed at each "Off A" and "On B" sites (Figure 2.1) were both stopped at 15 minutes because water was in short supply. Average cumulative infiltration was higher off road than on road (Table 2.2).

The cumulative infiltration at untreated site two (UN2, Figure 2.8a) in June showed that the off road sites had a higher infiltration rate and were capable of infiltrating more total water than the on road sites. The Off A and On B sites were stopped short of 30 minutes due to a lack of water available to the experiment. In August, UN2 rates decreased compared with June (Figure 2.8b). However, infiltration rates over 30 minutes off road are still higher than on road rates (Figure 2.8a, b). Both On B and Off A had side leakage from the infiltrometer that could have added considerable experimental error.

Infiltration rates for RC2 in June were higher off road than on road (Figure 2.9a). Average cumulative infiltration was 18 mm and 119 mm for on and off road, respectively. A paired post-treatment sampling was not completed; the reasons are in the discussion section.

Infiltration rates for H3 were much higher off road than on road (Figure 2.9b). H3 off road values were higher than all other off road values for every site. Off A data were not included due to the hydrologic influence of the adjacent road cut. Average cumulative infiltration was again higher off road than on road. On road cumulative infiltration values were similar to those measured on the untreated site following treatment.

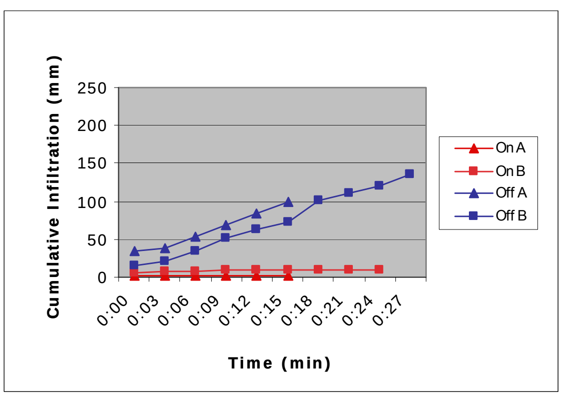

Figure 2.7: Cumulative infiltration over time RS2 in June.

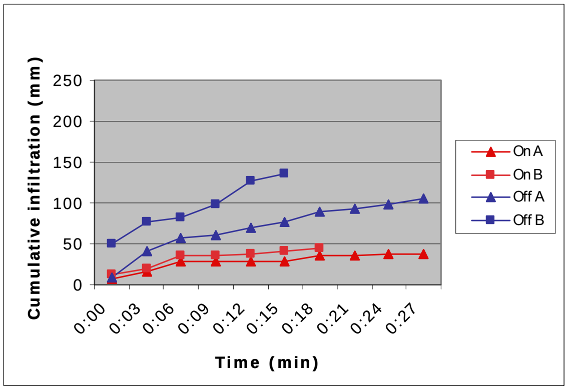

Figure 2.8a: Cumulative infiltration over time UN2 in June.

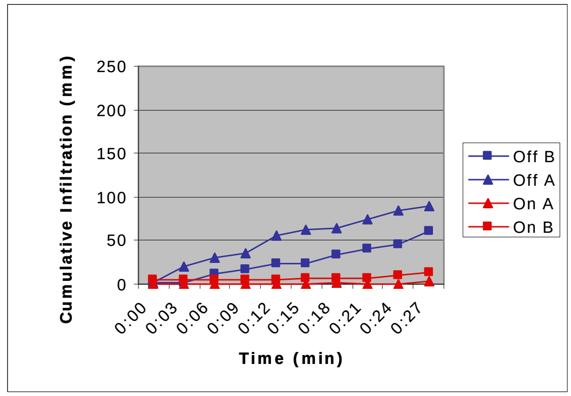

Figure 2.8b: Cumulative infiltration over time UN2 in August.

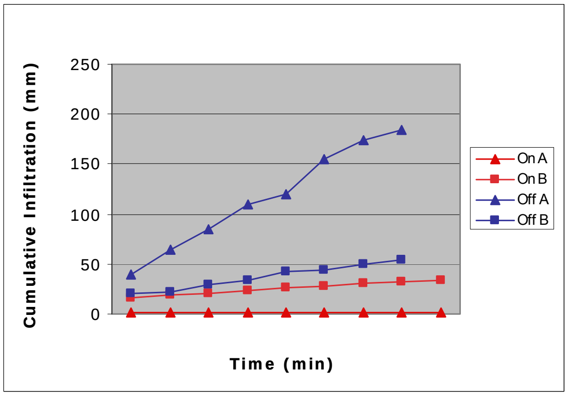

Figure 2.9a: Cumulative infiltration over time RC2 in June.

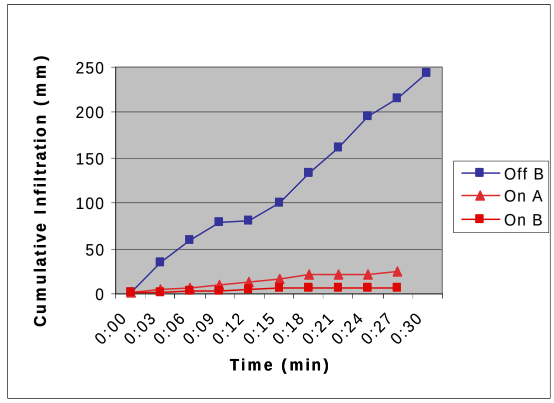

Figure 2.9b: Cumulative infiltration over time H3 in August.

Section 2.4: Discussion

Section 2.4.1: Bulk Density

The bulk density data provided insight on the original state of the roads, the effectiveness of each treatment, and how the treatments differed. The first question was how the bulk density differed on these roads and adjacent non-road areas. There was no statistically significant difference between on road and off road for any of the sites in June, prior to treatments being imposed. The outliers in both the untreated and the recontoured showed too much variation in order to observe statistical patterns within the data. However, the median bulk densities for all treatment types are higher on road than off road suggesting compaction on road. (Figures 2.2-2.6). The third quartiles of the off road recontoured and untreated were below their on road medians. Even though no statistical significance was found it appeared that the on road areas were more compact than off road. The fact that areas on road have increased compaction has been supported by previous studies (Kolka and Smith, 2004; Brady and Weil, 2007).With increased sample size this pattern may be more evident.

At untreated sites in August, the on road bulk density was higher than off road. This was expected, nothing was done to these roads. Compaction levels were still high on these untreated roads. Longer-term observations must be made to show how road closure without treatment affects compaction levels.

The ripped and reseeded road sites were not different between on and off road plots. However, if the box plots are taken into consideration, the median of the on road was much higher than off road. The ripping treatment on the road involved passing over the roadbed with tines to break up the soil. In some places, the tines did not rip up the whole road leaving pieces intact. Due to the variability in the treatment, it may return parts of the road to an off road condition but our data suggest that it was not completely effective.

The recontoured roads were also not statistically different between on and off road in August. It cannot be said that the treatment was effective because on and off road bulk densities were not different pre-treatment. On road bulk densities decreased from June to August and looked much closer to off road in August. Tukey HSD tests were performed for on road bulk densities for each treatment between June and August. The p-value for recontoured roads in June vs. August was 0.161. With an alpha level of 0.01 this was not significant but low enough to show a slight difference. This suggested that bulk density did change due to recontouring roads between June and August and in August the recontoured roads were similar to the off road references.

When treatments were compared by looking at on road bulk densities in August, differences in immediate effectiveness of reducing compaction became apparent. The bulk density values of the untreated roads were significantly higher than those of recontoured roads. The reseeded roads were not significantly different from the untreated or recontoured roads suggesting that the compaction levels were somewhere in the middle. This was expected because of the nature of the treatments. Untreated roads will obviously still have higher bulk densities because nothing was done to attempt to change them. Recontouring involves moving earth to remove the road cuts and returning the hillside to the natural slope. It should have much lower bulk density. Reseeded roads were treated but not as intensely as recontoured roads, implying that their bulk densities would be lower than untreated roads but higher than recontoured. These are only the immediate effects of treatment, after several years, long-term differences in treatment will become apparent.

Samples from roads ripped and reseeded 15 years ago should provide some idea of the long-term effects of treatment. First, on road bulk density was higher on road than off with a mean difference of 0.743 g/cm^3. Furthermore, bulk density on historic roads was similar to untreated and reseeded roads. These data suggest that historically treated roads are still more compact than adjacent off road areas and have not changed much since treatment albeit that we did not observe any statistical significance, or test these roads before treatment 15 years ago. Based on trends apparent in the bulk density data, it appeared that the recontouring treatment was most effective at reducing compaction on road and returning it to an off road condition. These findings are similar to a Kolka and Smith (2004) study which suggested that recontouring roads lead to lower bulk densities, less surface runoff and sediment production, and greater seedling growth in comparison to ripping or no treatment. The reseeding treatment was somewhat effective but compaction still remained higher than off road areas after 15 years. Not treating the road does not change compaction levels, but with more time these findings may change. Without treatment decompaction may occur slowly through frost action, biotic activity and wetting and drying (Foltz *et al.,* 2007).

Section 2.4.2: Infiltration

It was found that the instantaneous infiltration rate under non-pre-wetted conditions was substantially lower on the road than off road for all pretreated sites. This shows that the on road infiltration rate was much lower than the off road references. The historic road was also found to have substantially greater infiltration off road. The average on road cumulative infiltration for the historic site was 15 mm. Cumulative infiltration on untreated roads were 5mm and 16mm in June, 18 mm and 41 mm in August, suggesting that the historic site has not been returned to a state similar to the off road reference sites. The inconsistency between the June and August sampling of UN2 is likely due to the fact that the soils were not pre-wetted. This allowed the heterogeneity of the soil, such as large macro-pores, to increase the variability in the cumulative infiltrations.

Taking accurate infiltration measurements proved to be difficult. The high amount of rocks greatly reduced the ability to pound the infiltrometer 5 mm into the ground, in some locations barely making it to 2mm. This caused an additional problem of water leaking out from the sides of the infiltrometer to the surrounding soils rather than saturating the soils within the infiltrometer. The amount of water leaking from the sides was reduced by packing soil around the outside of the infiltrometer. Measuring infiltration on the recontoured road was challenging because within the first minute all of the water initially poured into the infiltrometer had already infiltrated. This was likely due to large macro pore development following the recontouring and that the roads were decommissioned within two weeks prior to sampling. We noted the extremely fast infiltration rate but were unable to continue measuring due to not having enough water to continually fill it for 30 minutes. For this reason, no measurements were taken on the ripped site.

For future studies a different method for quantifying infiltration should be employed. Luce (1997) had success with a rainfall simulator. This would give the additional observation of rain splash action on the soil surface. The size of the simulator should be considered. Ripped roads are heterogeneous after treatment. Areas of the road where the tines ripped through the soil surface will have a very different infiltration rate than areas the tines missed. Ideally, the simulator would be as wide as the road, but this would require equipment too large to transport into forests. Alternatively, real rainfall could be observed. This would eliminate the need for excessive amounts of water and equipment. A simple rainfall gauge and sediment trap could quantify the amount of sediment runoff generated given the magnitude of the rainfall observed. Though, this method would be dependent on weather conditions, which can be highly unpredictable.

Section 2.4.3: Conclusions

Infiltration and bulk density measurements result from the combination of key soil parameters including mineralogy, structure, texture, and landscape position. Generally, they are related: the lower the bulk density, the higher the infiltration. The importance of water infiltrating into the soil profile cannot be ignored. In Ward and Trimble's 2003 text its importance is summed eloquently, "(Infiltration) is a necessity for vegetative growth, contributes to underground water supplies that sustain dry-weather stream flow, and decreases surface runoff, soil erosion, and the movement of sediment and pollutants into surface water systems. Infiltration directly affects deep percolation, groundwater flow, and surface runoff contributions to the hydrologic balance in a watershed." Clearly, accounting for infiltration is fundamental to understanding and evaluating ecosystem processes. With these soil results, the logical next step is to evaluate the below ground hydrology of forest roads, and accordingly, it is the next section of this report.

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Section 3: Hillslope Hydrology

Researchers: Kadie Huse, Amanda Hyman, Emily Lewis, and Zach Twombly

Section 3.0: Abstract

Forest roads have been shown to alter hillslope hydrologic connectivity by intercepting groundwater and converting it to surface flow. Many studies have observed the detrimental impacts of the forest roads on local hydrologic regimes, yet few have proven if these effects can be reversed by decommissioning practices. This study investigated the impacts of recontouring treatments on hillslope hydrology along forest roads. From June to September 2010, peak water table measurements were recorded across four transects of recontoured forest roads in the Gallatin National Forest, Montana. Wells downslope of the recontoured road responded to precipitation similarly to wells upslope of the road prior to recontouring. Following treatment however, the downslope wells responded significantly differently than their respective upslope wells. These results suggest that recontouring forest roads does indeed alter hillslope hydrology, although further studies are required to discern causes and long-term responses.

Section 3.1: Introduction

Forest roads have been known to cause a variety of changes to their surrounding ecosystems. Such changes include altered hillslope hydrology, conversion of subsurface flow to surface flow and concentration of overland flow that can result in significant geomorphic changes (Switalski *et al.*, 2004). When these detrimental environmental impacts are deemed to outweigh the economic benefits of such roads, decommissioning treatments are applied in an attempt to reduce environmental degradation. The primary reasons to undertake road-decommissioning treatments include restriction of public travel allowing for vegetation regrowth, reducing landslide potential, reducing erosion, restoring natural drainage patterns, and restoring aquatic and terrestrial habitat (US Forest Service, 2005). Depending on which of these issues are of concern there are a variety of decommissioning treatments that can be applied to a given road or road segment. Common decommissioning treatments include road closure, "ripping" of the roadbed, removal of stream crossings, and fill recontouring of the hillslope (Switalski *et al.*, 2004).

Since water resources are particularly important in the Western United States, much of the research surrounding the benefits of road decommissioning has focused on its effects on water quality and aquatic ecosystems. In many forest ecosystems sedimentation from roads is a serious concern and is often a major reason to undertake a road-decommissioning project (Wemple *et al.*, 2003). Roads constructed across steep slopes are highly susceptible to large landslides and extensive gullying that transports sediment straight into stream channels. Road cuts have also been known to intercept the flow of groundwater and interflow downslope converting subsurface flow to surface flow, and thus increasing runoff and erosion in channels from an increase in discharge (Madej, 2001). In most cases, road-decommissioning treatments are highly successful in reducing overall sediment production from these roads. A study from Redwood National Park in California that looked at over 500 km of treated forest roads found that roads treated by various methods (ripped and drained and various degrees of recontouring) produced only one-quarter of the sediment produced from untreated roads and almost 80 % of the decommissioned road reaches had no erosion at all following a 12 year recurrence interval storm (Madej, 2001). This research suggested that some road decommissioning techniques can successfully reduce sediment transport and sedimentation of stream channels.

Many forest roads that have been heavily used suffer from severe compaction that causes increased runoff from decreased infiltration. These roads intercept and channelize water during storm events causing an increase in sediment yield and making the road susceptible to erosion. One common decommissioning treatment for reducing compaction and improving infiltration is ripping, where a bulldozer is used to break up the compacted layers of soil in the road. This method has been shown to significantly increase the hydraulic conductivity of a road surface and decrease runoff production from the roadway (Luce, 1997). While the amount of subsurface flow that is intercepted by a road is a function of the area of the hillslope above the road and the interaction of the road cut with subsurface flow paths, any water that is intercepted is transported directly to stream channels by unnatural flow paths. This can alter peak discharge magnitudes as well as alter the catchment scale hydrologic response (Wemple *et al*., 2003).

A 1999 study of five forest roads in Alaska attempted to measure the effects of roads on groundwater flows in various soils and hydrologic conditions. This study found that most roads had a depressed water table below the road in relation to the water table above the road. These effects were most significant from 5 to 10 m below the road surface with the severity of the depressed water table diminishing with distance below the road indicating the roadway as the likely cause of the depressed water table (Kahklen *et al*., 1999). While it has been established that roads have the potential to intercept subsurface flows and alter hillslope hydrologic connectivity, little has been studied regarding the effectiveness of methods to reduce or stop this process from occurring. Road decommissioning techniques such as hillslope recontouring and ripping have been shown to decrease runoff and increase hydrologic conductivity (Luce, 1997; Madej, 2001) however, nothing in the literature has shown whether this is effective in restoring a natural hydrologic connectivity to the hillslope. Economical and ecological management of forest roads require management agencies to have a complete understanding of the benefits and impacts of various road-decommissioning techniques. This means a full understanding of the impact of roads on water movement as well as the hydrologic impact of various road-decommissioning treatments. To date, an evaluation of the effectiveness of these restoration efforts on restoring natural hydrologic processes has not yet been made (Switalski *et al*., 2004). Our study, focused on forest road-decommissioning in Gallatin National Forest in southwest Montana, attempted to determine if current hillslope recontouring techniques are able to restore the disrupted hydrologic connectivity of road-cut hillslopes. The objectives of this study were to (1) understand subsurface water flows with respect to decommissioned roads, and (2) determine if hillslope recontouring was effective in reestablishing a natural hillslope hydrologic connection.

Section 3.2: Methods

Section 3.2.1: Site Description

Hyalite Canyon is located in the Gallatin National Forest just south of Bozeman, Montana. Recontouring is one road decommissioning technique used in this area. We focused on recontoured and undisturbed (off road) sites. Four sites that were recontoured in mid August 2010 were chosen, three of which were east facing, while one was north facing. Soils were well-drained, loams with cobbles and pebbles. Dominant parent materials of the soils at the site were alluvium over glacial outwash (NCSS, 1999). There are many small creeks that feed into Hyalite Creek, which eventually feed into the Gallatin River.

Section 3.2.2: Determining Depth to Water Table Pre-and Post- Restoration

At each sampling road (site), saturated areas at water convergence points were located in July 2010 by looking for evidence of gullies and low topographic points. Water convergence points were found approximately 25 m and 5 m upslope of the road and 15 m downslope of the road (Figure 3.1). At each of the convergence points a PVC well was pounded into the ground. Each PVC well was 1.5 m in length with a 1.3 cm diameter and had holes drilled every 5 cm into the bottom third of it. The holes allowed water to infiltrate into the well so that groundwater heights could be assessed. A thin, plastic rod covered in blue chalk was then placed in the well, and the well was covered with a waterproofed seal. Three wells were placed at each of the four sites.

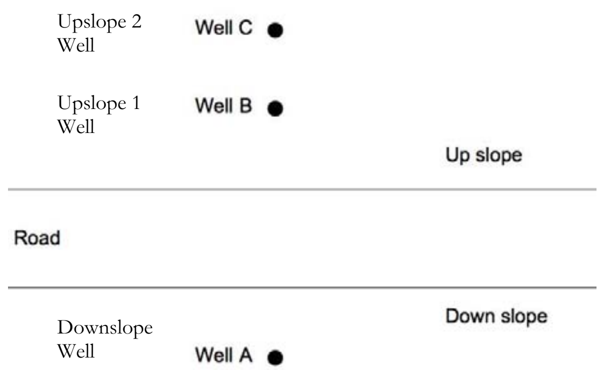

Figure 3.1: Depiction of the well placement in relation to the road sites.

After the installation of each well in July, the depth of the groundwater was recorded two times each month, until early September 2010, by measuring the height at which the chalk remained on the dowel. After each observation of groundwater depth, the chalk was reapplied to the dowel and was replaced in the well and re-sealed. As sites were recontoured in August this provided 2-3 samples pre and 3-4 samples post-treatment on the different road sites.

Daily precipitation amounts were obtained from the Lick Creek Snotel site (NRCS) for the duration of the sampling period. The maximum total precipitation in a 72 hours period in between each sampling date was calculated and used in analysis.

Section 3.2.3: Data Analysis

The depth measurements for each well were averaged. For each well the average was subtracted from each individual measurement to normalize the data. The normalized depths were then plotted against the maximum total precipitation in a 72-hour period in between each sampling date. The slope of the regression represented the responsiveness of the wells to precipitation. The responsiveness of all upslope 1 and upslope 2 wells were contrasted against the responsiveness of all downslope wells both pre- and post-recontouring to infer the effects of recontouring on hydrologic connectivity. An ANCOVA (analysis of covariance) was also performed to determine if any statistically significant relationship between each well, measurement date, and 72-hour maximum precipitation existed.

Section 3.3: Results

The range of normalized depth to ground water data for each well that was used in the analysis can be interpreted in the following way: negative values indicate depth ground water values that were below the average for that well and positive values indicate depth to ground water values that were above the average (Figure 3.2). The wells which were furthest upslope of the road (C) had the greatest variability in the normalized depth to ground water, whereas the wells below the roads (A) had the least variability (Figure 3.2).

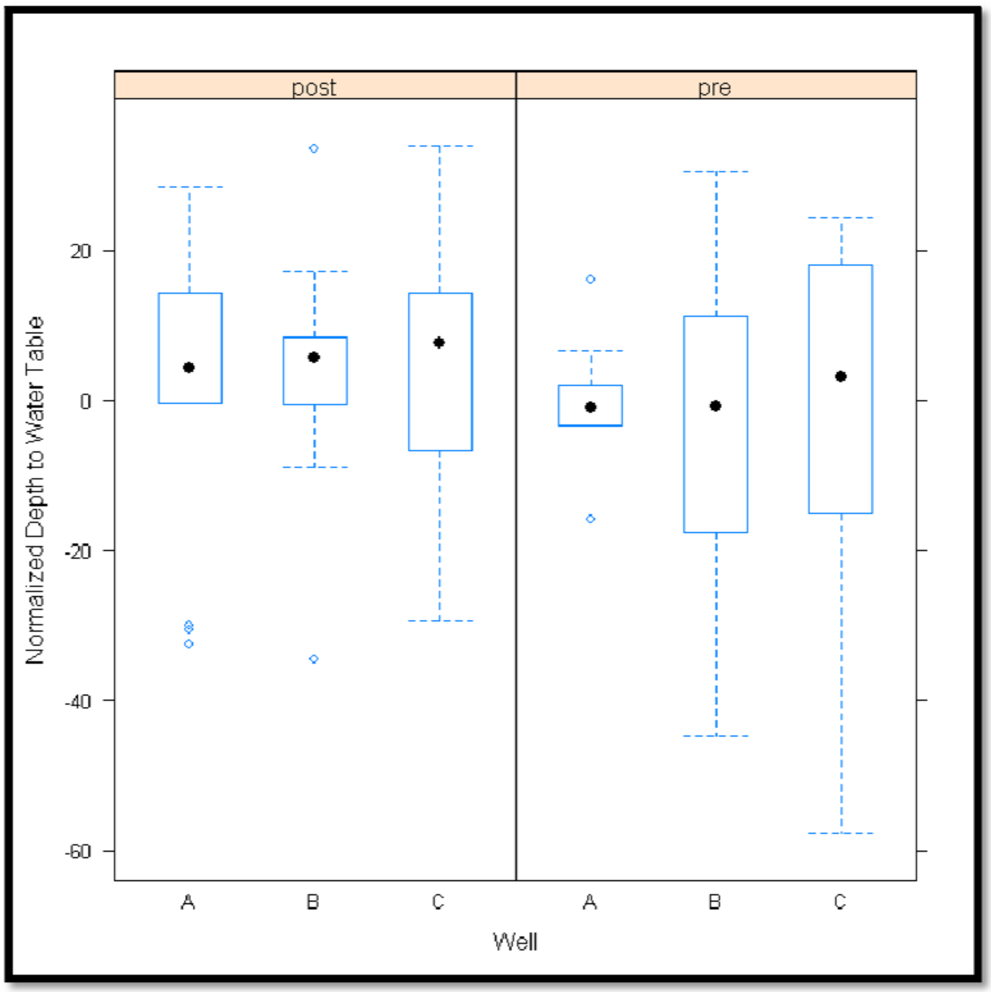

Figure 3.2: Normalized depth to ground water data for each well both post- and pre-treatment.

The response of the two upslope wells was compared to the downslope well both pre and post-treatment to determine if the road was influencing subsurface water flow. In Figures 3.3-3.6 the slopes of each line represent the response of that well to precipitation. Similar slopes between two wells indicate that those wells respond similarly to precipitation. Graphs displaying a negative slope mean that the more precipitation the well received the lower the depth to ground water was in that well. Graphs displaying a positive slope can be interpreted as the more precipitation that well received the greater the depth to ground water was in that well.

Before treatment there was no significant difference between the responses of the upslope wells and the response of the downslope well (Figures 3.3 and 3.4).

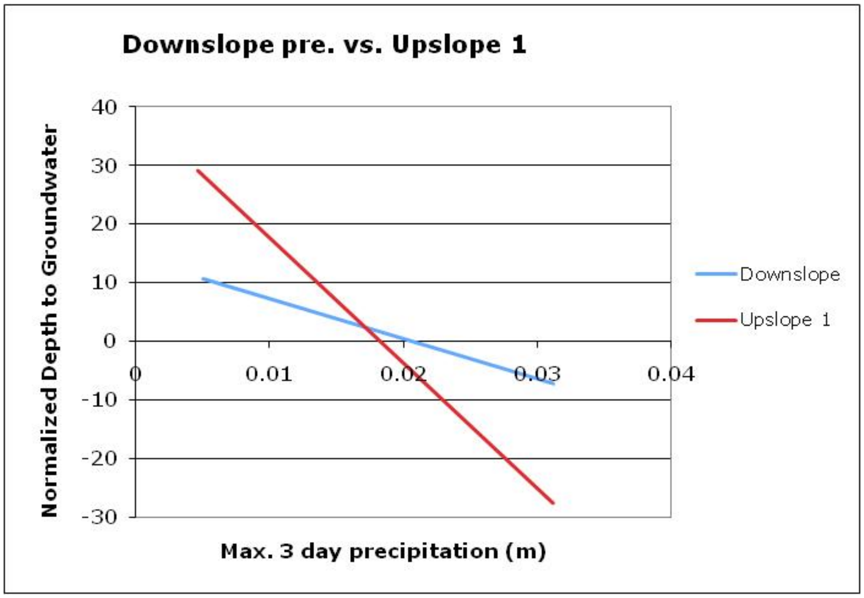

Figure 3.3: Response of all downslope wells pre (A) compared to response of upslope well B. ($p = 0.49$)

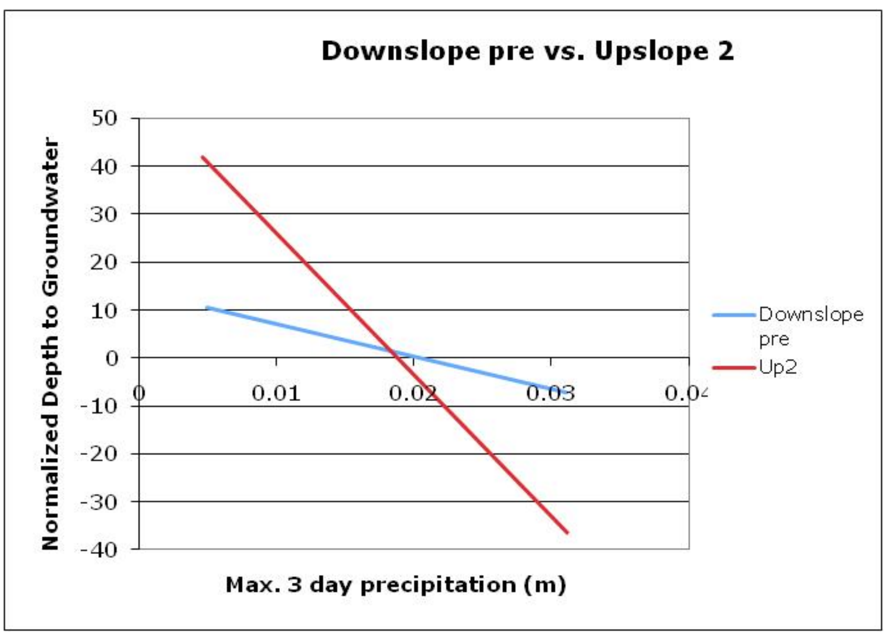

Figure 3.4: Response of all downslope wells pre (A) compared to response of upslope well C ($p = 0.61$)

After treatment there was a significant difference between the response of the downslope well and the response of the upslope wells (Figure 3.5 and 3.6).

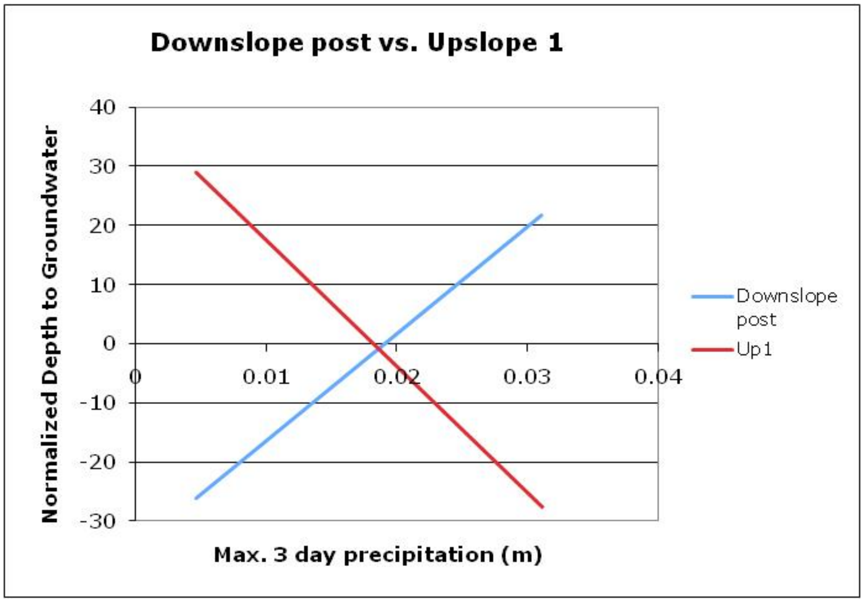

Figure 3.5: Response of all downslope wells post (A) compared to the response of upslope well B. ($p = 0.02$)

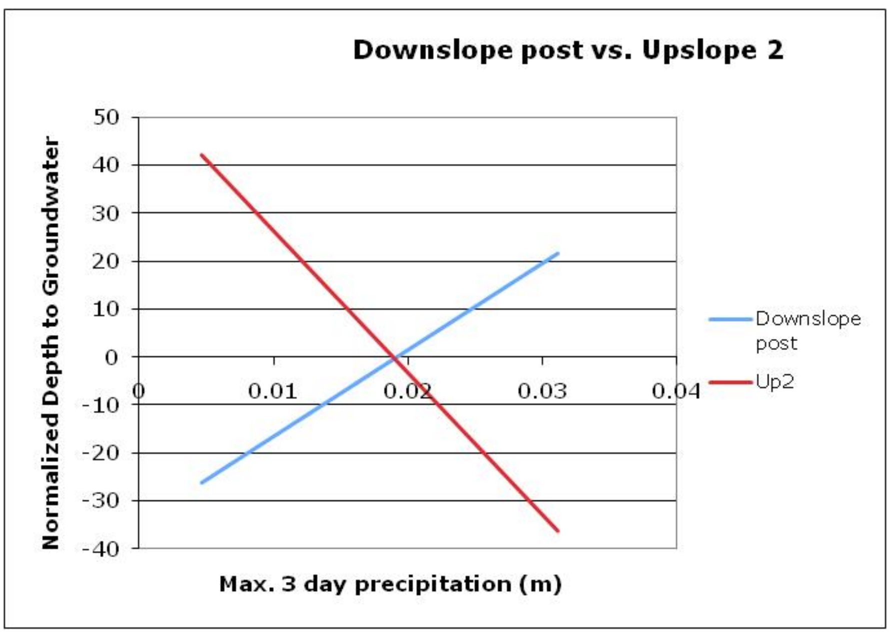

Figure 3.6: Response of all downslope wells post (A) compared to the response of upslope well C ($p < 0.001$)

Section 3.4: Discussion

While the recontouring of the forest roads in this study did not create the hydrologic response that was hypothesized it is evident that there was a significant change in the hydrology of these hillslopes caused by recontouring of the road. Before treatment there was no significant difference between the responses of each well to precipitation. This suggests that the three wells were all influenced by the same hydrologic conditions and that the road cut did not significantly influence the hydrology of the hillslope. After re-contouring however, there was a significant change in the response to precipitation between the downslope well and the two upslope wells. This suggests that the re-contouring treatment altered some hydrologic component of the hillslope that affected the downslope well but not the two upslope wells. While further study is necessary to determine exactly what this altered hydrologic component could be there is evidence to suggest that an increase in infiltration observed in this study was caused by the recontouring treatment.

Forest road decommissioning treatments have been shown to greatly increase the hydrologic conductivity of soils (Luce, 1997). On the same recontoured forest roads used in our study, the Capstone Soils Group infiltration study found that the recontoured road surfaces had a significantly increased infiltration capacity compared to the untreated road surface. In our study, since the wells used to measure subsurface flow were located in areas of water convergence, there was likely a significant amount of surface or near-surface flow occurring during large rain events. With the greatly increased infiltration capacity of the roadway after recontouring the water moving from the upslope wells likely infiltrated rapidly and to a greater depth and therefore never reached the downslope well. Due to the short time scale of this study (2- 6 weeks post-treatment) it was not evident whether or not this flow pattern would change with time. A previous study of infiltration rates on forest roads after ripping treatments found that hydrologic conductivities of ripped roads decreased by 50% after significant rainfall had reached the road. This was caused by settlement of the soils and large clods corresponding with a significant increase in bulk density seen after the first rainfall (Luce, 1997). Since the scope of this study did not cover a significant amount of rainfall or time after treatment it cannot be concluded how these roads will respond to water as the disturbed roadway settles, compacts and revegetates.

Section 3.4.1: Study Errors and Extensions

Throughout the data collection process various sources of error were observed that could have skewed the ground water measurements. First, it was the intention to select road sites that were similar in topography, soil type etc. However, due to the number of possible recontoured sites within our study area there were variations in slope, aspect (we had three sites east facing and one north facing), soil type, and vegetation features that could have created differences in how ground water moved in the subsurface. While the assumption that all of these sites were similar it must be noted that this caused considerable variance within the groundwater heights that were observed.

Other sources of error involved the design of the wells and well placement. Since holes were only drilled in the bottom third of the wells during times of high water table height the well could have acted like a piezometer and water could have risen in the well higher than the height of the actual water table. Additionally, in some cases, the wells were not perfectly flush with the ground around them leaving space for surface flow to enter the hole and fill the well, thus skewing the reading for the height of ground water.

Another source of error that is suspected to have had a significant influence on some of the downslope wells was pooling of water on the road surface that ran off and reached the downslope well as surface runoff. Since the wells were placed in areas of water convergence, the portion of the road between the upslope and downslope wells became an area where water pooled. This water would then runoff downslope when the pool became full or it was forced off the road by vehicle traffic pre-treatment. With this happening the water was still reaching the downslope well however, it was occurring as surface flow rather than subsurface flow. This is believed to be one potential reason why the downslope well appeared to act similarly to the two upslope wells in the pre-treatment sample period. Further study is necessary to determine if this effect significantly influenced downslope wells and what implications this could have for the water quality downslope.

In addition, the different hydrologic response obtained after the roads were recontoured may have been due to the recontoured roads infiltrating water readily and due to the depth of the treatment the water may have flowed below the wells. It would have been difficult to make the wells deeper but having the treatment and wells at similar depths was a limitation of the study.

As a first time study, there are many potential ways to improve upon our experimental methods that could lead to a greater understanding of the flow of water associated with road-cut hillslopes. In this study the measurements of groundwater that were taken represented only the peak ground water table which does not tell much about the flux of groundwater in the subsurface that could controlled by a myriad of factors including topography, vegetation and soil type. Measuring the base height of the ground water in each well along with the maximum height would indicate the flux response of each well without the effects of the many confounding variables that would likely influence the peak ground water height from well to well. A tracer test would be a useful addition to this study to prove the connectivity of the three wells. At this point it has simply been assumed that much of the water influencing the upslope wells is the same water that is influencing the downslope well. A dyed tracer test would prove whether or not the three wells are hydrologically connected or not and could help to identify some of the flow patterns that are occurring in the subsurface. (Flury and Wai, 2003)

Another significant improvement on this study would be to increase both the number of wells on each transect and the number of transects on each road but this was logistically unfeasible for us. This would help to capture the large variation in hydraulic conditions that is likely occurring in these forest road systems. In addition, precipitation gauges could be positioned at each transect to capture precipitation patterns. This would assist in understanding the large variation in hydraulic conditions that is likely occurring on these forest road systems over time as the road settles and begins to revegetate.

Section 3.4.2: Conclusion

Directly contrary to the original hypothesis the downslope well responded more similarly to the upslope wells before treatment than post-treatment. This suggests that some aspect of the road recontouring treatment disrupted or in some way altered the hydrologic connectivity of these hillslopes. Without further study to determine what caused the disruption in connectivity observed on these roads after recontouring it would be inappropriate to make recommendations to the Forest Service regarding the impacts of recontouring treatments on hillslope hydrology. The factors causing the observed relationships must first be identified and a longer period of observation must be used to determine how these relationships will evolve over time.

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Section 4: Utilization of Insects as Bioindicators to Evaluate Road Decommissioning Practices.

Researchers: Alex M. Gaffke, Carmel E. Johnston, and Elizabeth M. Usher

Section 4.0: Abstract

The practice of road decommissioning and the use of insects as bioindicator species are two concepts that have been researched thoroughly and have many years of data behind them; however the connection between the two has little to no known research. Combining the practice of road decommissioning with the use of insect bioindicator species has the possibility to create a realm of monitoring that has not been used in the past. Insect richness was analyzed on three road treatment types: undisturbed, untreated, and ripped and seeded historic roads. Samples were collected in June, July, and August on transects using sweep nets, pit fall traps, and soil samples. Only the July and August sweep net samples could be analyzed and 880 insects in 54 families were identified. A Tukey HSD test showed no significant differences in the raw abundance of insect number between road treatment types. Narrowing in on bioindicator insect orders Hymenoptera and Lepidoptera showed no significant differences between undisturbed (control) sites to historic sites, as well as no significant differences between untreated and historic sites. Control and untreated were significantly different. These data indicate that while the ripped and seeded historic sites still have characteristics of the untreated roadway sites, they are showing characteristics of the control sites, and may be still be in a state of recovery.

Section 4.1: Introduction

Long-term environmental impacts can be lessened when a road is restored to its natural state through ripping, re-contouring, and/or reseeding the surface area (Thompson, 2008). Road decommissioning and revegetation treatments could increase forest connectivity and enhance ecosystem health by supporting higher diversities of forest flora and fauna (Switalski *et al.*, 2004). Therefore, ways to assess the health of ecosystems are needed. Species richness is a basic marker of ecosystem health; a healthier ecosystem tends to have more organisms, as it can support more life. Ecosystem health and stability is commonly assessed with organism diversity (Brown, 1997). Studies with butterflies in Neotropical forests have demonstrated an increase in diversity with disturbance levels near or below natural, non-anthropogenic levels (Brown, 1997). Vegetation diversity also illustrated a positive relationship with species richness of insects (Jonsson *et al.*, 2009). Grasshoppers, common to many ecosystems, can have high diversity, high functional importance, and high sensitivity to disturbance, making them excellent indicators of the effects of land management (Andersen *et al.*, 2001). In fact, insects could be used as biological indicators of the state of a system.

Insects are a key factor in identification of ecosystem function and health and constitute a sizeable percentage of terrestrial biomass (McGeoch, 1998). Insect populations are important for energy cycling through the food web, nutrient cycling, and population regulation by acting as vectors for diseases. Pollinating insects significantly influence the dynamics of plant populations, specifically angiosperms. More than 90% of angiosperms rely on insects to complete their life cycles by spreading pollen to aid in reproduction (Wilcock and Neiland, 2002). When environmental conditions and anthropogenic activities negatively impact these pollinators, angiosperms are also affected (Wilcock and Neiland, 2002; Quintero, 2010).

Destruction of habitat affects pollinating insects in three ways: (1) destruction of food sources, (2) destruction of nesting or oviposition sites, and (3) destruction of resting or mating sites (Kevan, 1999). Because of their sensitivity to ecosystem changes, specific insect groups have been used as bioindicators of ecosystem health and have been shown to work well in such a role (Akutsu *et al.*, 2006). Bioindicators are a group of species which reflect the state of an environment, represent the impact of environmental change on an environment, or typify the diversity in an area (McGeoch, 1998). An advantage for using insects as bioindicators is their quick life cycle, which can be completed several times over in one season, as well as their relatively high abundance. Moreover, their high sensitivity to habitat changes, resulting in a quick response to any changes in the environmental conditions (Brown, 1997), and their presence in numerous environments (Akutsu *et al.*, 2006) makes them potentially well-suited to act as bioindicators. Insects have high functional importance, high species diversity, and are influenced by 'bottom-up' effect making them sensitive to disturbances to the grass layer (Andersen *et al.*, 2001). The term 'bottom-up' effects is used to describe a system where initial impacts are small but have an increasingly large effect on the system overall. In respect to grasshoppers, this means that they are controlled by the grasses that they use for feeding and habitat which in turn controls other populations dependent on them.

Certain bioindicator species can help focus the study on what exactly is desired. The relative abundance of Lepidoptera family can represent the diversity for the whole ecosystem where these insects are distributed (Kahn, 2004). Tscharntke *et al.* (1998) found that bees can be promising bioindicators for ecological change or habitat quality. They also found that species richness of bees was closely related to plant species richness. For these reasons, we chose to utilize Lepidoptera and Hymenoptera families as the bioindicators.

No studies using insects as bioindicators with respect to road decommissioning have been located in the literature. However, there is abundant published research using insects as bioindicators in many parts of the world. This experiment will be the leading edge for connecting the gaps between research on insects as bioindicators and research solely on road decommissioning. Insects are an integral part of every environment, and to fully evaluate road decommissioning, the effects on insects are a valuable, mandatory part of the research.

The objectives of our study were to (1) compare the abundance of gathered insect samples resulting from roads that were decommissioned 15 years ago by ripping and reseeding, those that were designated to be untreated but access prevented in 2010 and control sites off road, and (2) evaluate if insect abundance could be used as an indicator for the road decommissioning process.

Section 4.2: Methods

Section 4.2.1 Site Description

Six different sites were sampled which included two roads targeted to be closed but untreated in 2010, two roads previously ripped and reseeded, and two undisturbed control sites off road. The previously ripped and reseeded sites were decommissioned in 1994/1995 and will be referred to as historic sites. The third kind of site sampled was a control site. This was an area that was approximately 50 meters away from the untreated site in the surrounding forested area. All of the sites were in Hyalite Canyon west of Langhor Campground and south of History Rock in the southwestern part of Montana as described in the general site description in Section 1. Hyalite Canyon is a mixed conifer forest with some open meadows dominated by grasses and forbs. The samples from Hyalite Canyon were taken on June 15, 2010 on a partly cloudy and windy day. Samples were also taken on July 24th and 25th, 2010 and August 25th and 26th, 2010 with sunny, cloudless days and little to no wind. The temperature was approximately 15 °C when the first samples were taken. The next four sample dates were taken at temperatures between 21-32 ºC.

Section 4.2.2: Sampling sites

- Untreated: These two roads were selected as untreated sites. The focus of our study was not on the effectiveness of the treatment types, but rather the recovery over time. Samples were taken on these sites to analyze insect populations on a currently non-decommissioned road for comparison with roads that had been decommissioned, and with control sites. These roads were identified as UN1 and UN2.
- Historic: These sites were ripped and reseeded in 1994/1995 as part of a previous decommissioning project. Samples were taken to analyze insect populations after road decommissioning had taken place.
- Off road Control: These two sites were in the forest, Control 1 and Control 2 were approximately 50 meters from UN1 and UN2, respectively (Figure 4.1). They had similar topographical, climatic, and geological features to the other treatment sites. Samples were taken with the assumption that these sites would show insect populations without any influence from treated or untreated roads.

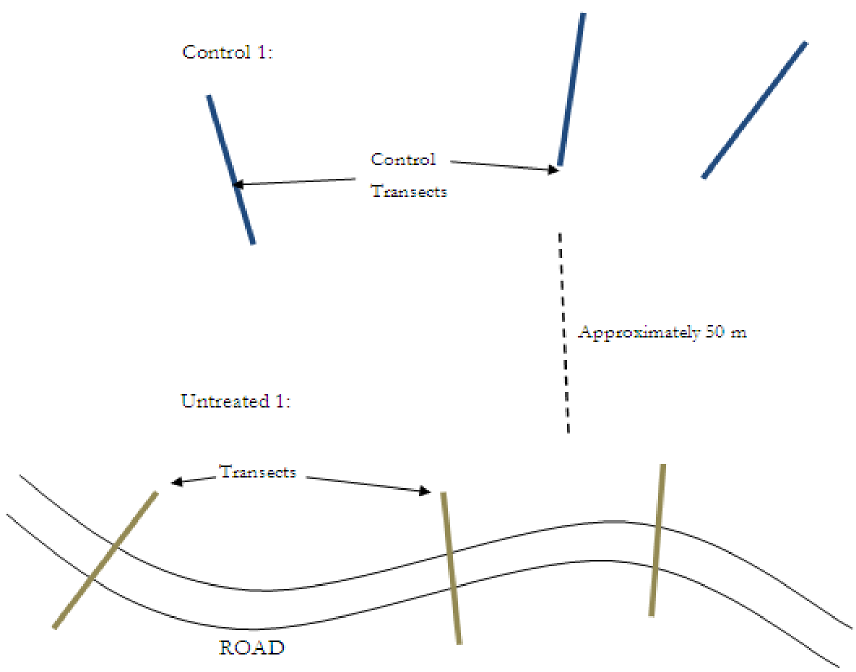

Figure 4.1: Schematic of transect placement. Road (black lines) with three untreated site transects (brown lines), and three control transects (blue lines).

Section 4.2.3: Sweep Net Sampling

Sweep net sampling was used to assess abundance and diversity of flying and plant inhabiting insect species. Three transects were established at random intervals along the selected portion of each of the six sites. The transects were approximately 10 m long so as there would be approximately 3 m of sampling beyond each side of the roads. Samples consisted of 35-45 sweeps at approximately 0.2 m off of the ground of each transect at a constant rate of sweep speed and arc to minimize error. Once each transect had been sampled over the entire length, the net was quickly whipped in one direction to concentrate insects into a ball. The net was then twisted once to keep insects caught in one ball, and then quickly untwisted and then thrust into gallon Ziploc bags, and shaken to dislodge any insects gripping or stuck to the net. Each bag was labeled for site, transect, date, and time. Samples were brought back to lab and put into freezers as soon as possible so they could be sorted by hand at a later time.

Section 4.2.4: Soil Samples

To obtain the soil samples, approximately 0.3 m x 0.3 m x 0.13 m volume prisms of soil were collected at three random locations along each of the transects, but avoiding any boundary areas where the vegetation and the road areas met, and ensuring one collection from the middle of the road, and one from either side of the road. The samples were combined from the three transects of one road into one conglomerate sample to be tested. This was performed once for each and every site on each sampling date given in Section 4.2.1. A Berlese funnel (Figure 4.2) was used to separate the live insects from the soil sample. The Berlese funnel separated the insects out of the soil by the use of light and heat. Soil samples were placed on a mesh screen that allowed for the soil invertebrates to migrate through it. Insects residing in the soil sample should move away from the light and heat produced by the lightbulb and will ultimately migrate out of the sample and into a collection bin below. This allowed for the collection and sorting of the soil biota that we would previously had to pick out by hand. Insects in these samples were put in Ziploc bags and then into the freezer to preserve them and allow for sorting. Unfortunately, time was short, and the samples were unable to be evaluated and analyzed.

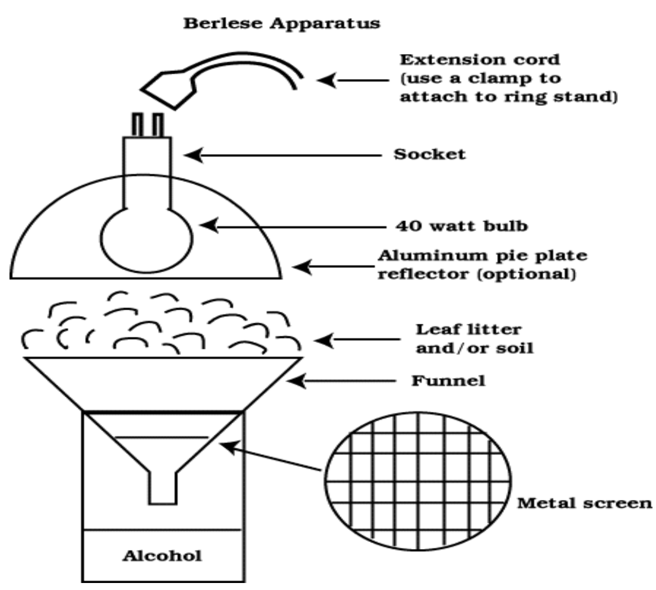

Figure 4.2: Berlese Funnel

Section 4.2.5: Pitfall Samples

Pitfall samples were used to assess the diversity of terrestrial invertebrates that would not be present in either the sweep net or soil sample methods. The pitfall trap consisted of a red

473 ml plastic Solo cup placed into the ground with its top flush with the soil level (Figure 4.3). This allowed for insects to fall into the pitfall. A liquid mix of water and soap was used for sampling dates June 15th, July 24th, and July 25th, and dishwasher detergent and water was used on August 25th and 26th. A 2.5 cm layer of water was poured in the bottom of the traps and a film of soap, or detergent in the August samples, was placed in the water in order to prevent the insect from escaping and to help counteract evaporation. Pitfall traps were located at three random points along each transect with two cups on each side of the road, and one in the middle. Pitfall traps were set up on sampling dates given above and collected after five days. The liquid was poured out and the cups were then frozen. Unfortunately, time was short, and the samples were unable to be evaluated and analyzed.

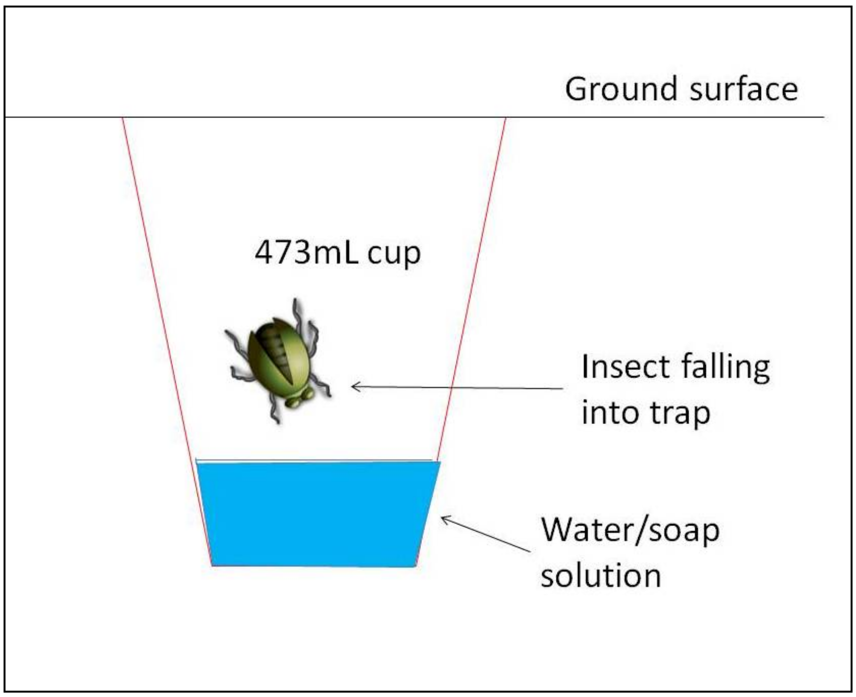

Figure 4.3: Example of a pitfall trap

Section 4.2.6: Sample sorting

Sweep samples were organized to determine the different orders and families of insects at each site. Sorted insects were put into centrifuge tubes organized by family type and stored in a freezer. Once the samples were sorted into the differing orders and families, an insect from each species was pinned and labeled to represent the species and the abundance of the particular species recorded in a separate file. This was done to prevent the waste of time and space of pinning duplicates of insects. If the insects were large enough, they were directly pinned with a pinning needle. However, if the insects were too small or delicate to handle being pinned directly they were glued to a small piece of paper, a process called pointing, and the paper was then pinned to mounting board.

Section 4.2.7: Data Analysis

When all the data had been collected, Excel 2007(Microsoft®) was used to organize the data for the comparisons of interest. Because of time constraints, only the sweep net data were used in the analyses. Graphs were created using box plot functions in the R statistical software. An analysis of variance (ANOVA) was completed with number of families, total number of individuals, number of bioindicator families and number of bioindicator individuals as response variables, and site, treatment, time and site: treatment interaction as indicator variables. A Tukey Honestly Significant Differences (HSD) multiple comparison of means was then calculated for significant main effects.

Section 4.3: Results

A total of 880 individual insects were counted from 54 families, as analyzed from the sweep net samples in July and August. Many insects were in early life stages, especially in July.

Section 4.3.1: Graphical Results and Statistical Analysis

The number of families observed at each time, treatment, and site were plotted (Figure 4.4). From July to August for each treatment per site, there was not a significant difference between the means. However there was considerable variation between some sites and treatments, particularly between historic site one and historic site two, as confirmed using Tukey HSD. Because of this significant site and treatment effect and the site to treatment interaction, no further analyses were performed.

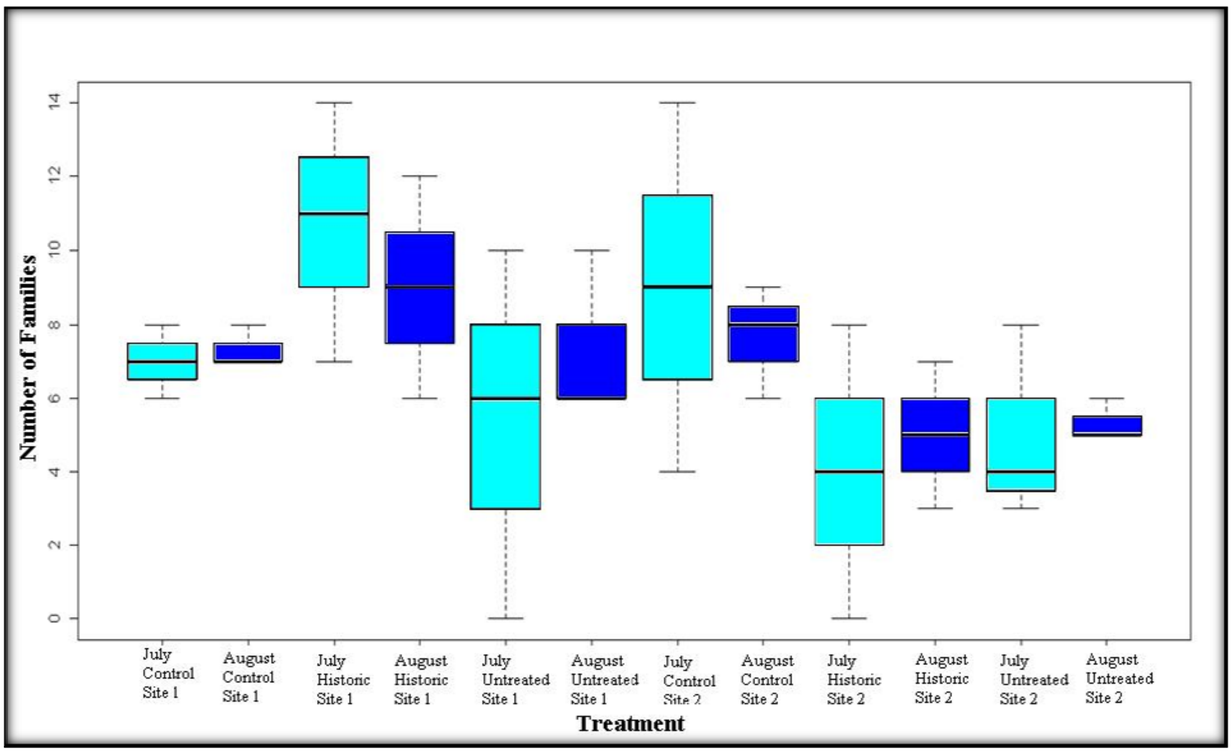

Figure 4.4: Number of individual families collected at each treatment and site for July and August sweep net sampling.

The number of individuals for each time, treatment, and site were also plotted (Figure 4.5). There is a trend showing more insects in July than in August, excluding historic site one and untreated site two. While there did appear to be a difference between the sites in so far as which month samples were taken, comparing the sites by different treatments showed no significant differences. For example, July control one was not significantly different from July untreated one, and August control one was not significantly different from August untreated one. Though, July historic one showed an extreme difference from all the others, due to the fact that at this site a lot of ants were recorded. Indeed when compared to the number of families in the treatment types (Figure 4.4) this site did not show as dramatic of a difference as it did in the number of individuals at the site. There were sites that were significantly different, and July historic 1 dominated those values (Table 4.1).

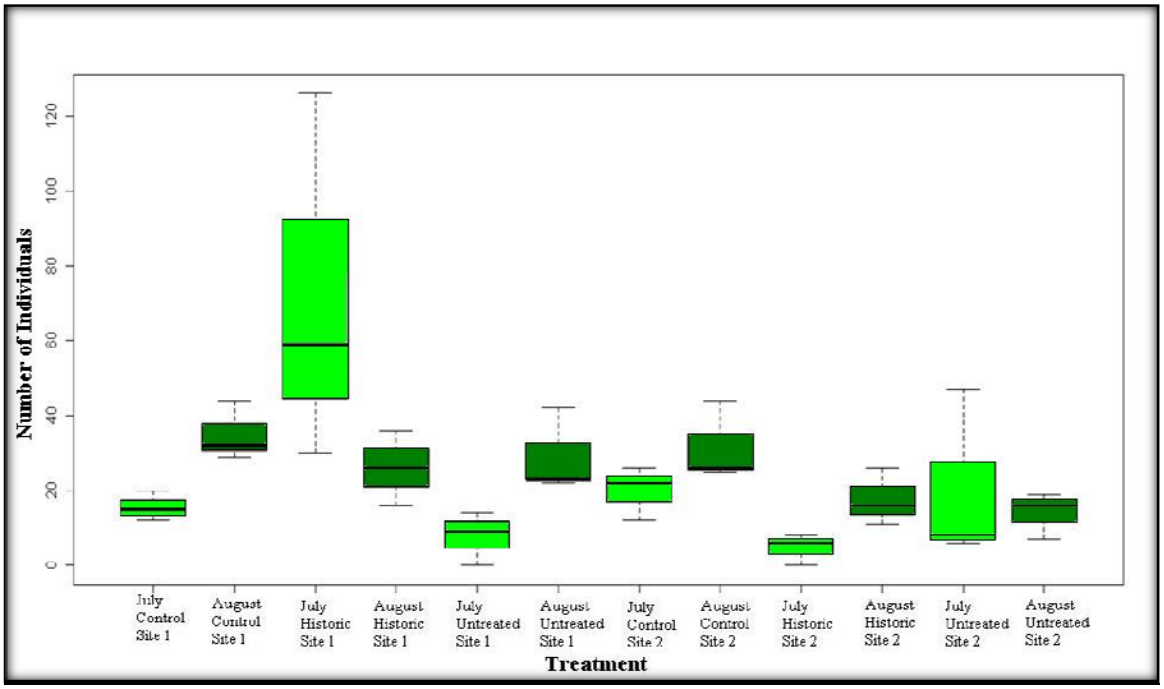

Figure 4.5: Number of individual insects collected at each grouping of time, treatment and site in July and August sweet net samples.

Time, treatment, site	P-value
Untreated site 1and historic site 1	0.0545
Historic site 2 and historic site 1	0.0108
Untreated site 2 and historic site 1	0.0424
July historic site 1 and July control site 1	0.0227
July untreated site 1 and July historic site 1	0.006
July control site 2 and July historic site 1	0.0449
July historic site 2 and July historic site 1	0.0037
August historic site 2 and July historic site 1	0.0312
July untreated site 2 and July historic site 1	0.0473
August untreated site 2 and July historic site 1	0.0174

Table 4.1: P-values for comparison of significant difference between number of individuals at the different sites, treatments, and sampling times.

Hymenoptera and Lepidoptera were chosen as bioindicator families. Analysis of bioindicator families showed higher numbers at the control sites, lower numbers at the untreated sites, with the historic sites showing values between those two (Figure 4.6). There were no significant differences between the number of bioindicator families at the historic and control sites, or between the historic and the untreated sites. However, the control and the untreated were significantly different (P = 0.02, Table 4.2).

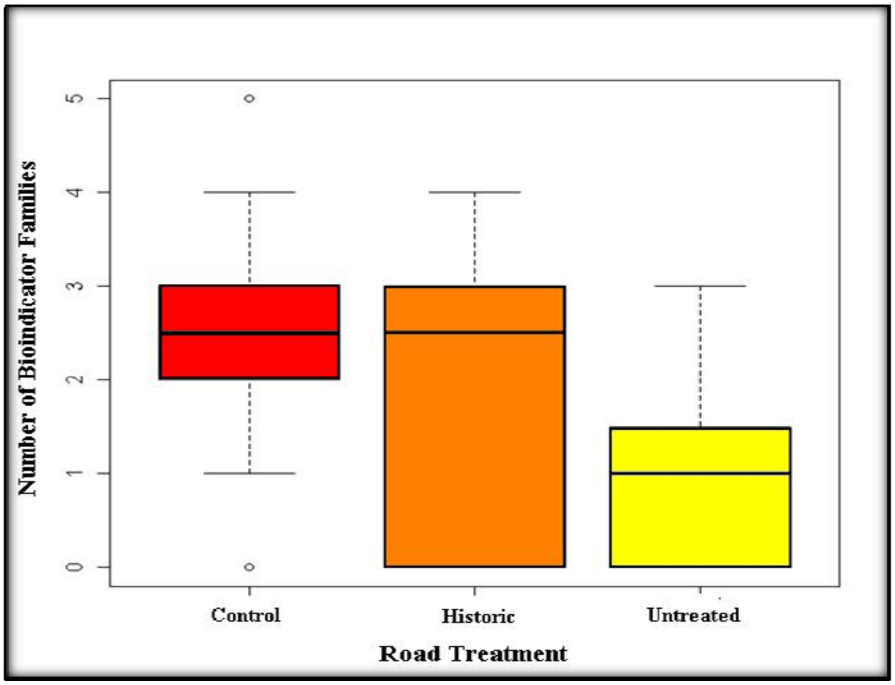

Figure 4.6: Number of bioindicator families for Control, Historic, and Untreated site types.

Treatment comparison	P-value
H-C	0.4225
UN-C	0.0197
UN-H	0.2658

Table 4.2: Tukey multiple means comparison P-values between three site types. H=Historic sites C= Control sites UN = Untreated sites

The number of Hymenoptera and Lepidoptera individuals in the untreated sites showed lower numbers at untreated sites, with higher numbers in the control and historic sites (Table 4.3). The number of individuals were scattered between more families in the control and historic sites.

		Number of Individuals		
Order	**Family**	**Control**	**Untreated**	**Historic**
Hymenoptera	Formicidae	31	11	30
Hymenoptera	Pompilidae	1	0	0
Hymenoptera	Braconidae	4	0	2
Hymenoptera	Sphecidae	3	0	1
Hymenoptera	Chrysididae	3	0	1
Hymenoptera	Vespidae	1	0	3
Hymenoptera	Ichneumonidae	1	1	0
Hymenoptera	Apidae	1	0	1
Hymenoptera	Pelecinidae	1	0	0
Hymenoptera	Megachilidae	0	1	0
Lepidoptera	Unknown	12	1	11
Lepidoptera	Noctuidae	2	1	1
Lepidoptera	Tortricidae	0	2	2
Lepidoptera	Hesperiidae	0	1	2
Total Families		**11**	**7**	**10**
Total Individuals		**60**	**18**	**54**

Table 4.3: Number of individual bioindicator insects in the Hymenoptera and Lepidoptera families and totals per treatment type.

Section 4.4 Discussion

When number of families, and total number of individuals were analyzed, there was no significant difference between control, untreated, and historic sites. Individual sites were significant players in the difference between samples when categorized by time, treatment, and site (Figure 4.4, 4.5). For the sake of repeatability, this is not ideal. We hypothesize these sites differences were influenced by differences in site topography where we researched. Time and treatment were also significant factors, but only when associated with individual site (Figure 4.5).

We analyzed the numbers of families within two orders, Lepidoptera and Hymenoptera. These orders contain species including butterflies, moths, ants, wasps, sawflies, and bees. The relative abundance of Lepidoptera families can represent the diversity for an entire ecosystem (Roasrio and Hellmann, 2006). They have been shown to be sensitive to environmental changes, such as habitat fragmentation (Roasrio and Hellmann, 2006). Habitat fragmentation has been defined at either a patch scale or a landscape scale, depending on the researcher (Fahrig, 2003). In this way, the roads we looked at can be considered as causing habitat fragmentation. Our data showed that the areas with untreated road disturbances did have less Lepidoptera individuals. The diversity of this family is determined by vegetation, food availability, temperatures, and exposures (Rosario and Hellmann, 2006).

Hymenoptera have also been used to indicate changes in population vulnerability, community structure, and ecosystem functioning (Tscharntke *et al.,* 1998). Habitat fragmentation has also been shown to have negative effects on population numbers, and species richness has been shown to correlate with plant diversity (Tscharntke *et al.,* 1998). Our untreated transects showed lower diversity in Hymenoptera populations, and the vegetation group also collected data that indicated low plant richness on untreated road plots.

The results of characterizing the number of these bioindicator families at the sites observed a significant difference between the control and untreated sites but no significant difference between the control and historic sites or historic and untreated sites. The control sites had one to four bioindicator families with outliers at zero and five and the untreated sites had zero to three families with no outliers. The historic sites had between zero and four families, which went as low as the untreated values and as high as the majority of the control values. This suggests that the historic site was still affected by the disturbance of being a road, but it has the potential to reach recovery because its diversity reached the same maximum as the control sites. Our experiment successfully showed that bioindicator insect families can be a useful technique for determining the recovery of roads after decommissioning.

In order to better determine the recovery of a road, a more thorough analysis of all insects in the area with several methods of collection would be needed. We collected pitfall and soil samples but did not have time to analyze them, leaving an entire functional type of insects, that is, those within the soil and surface dwelling fraction, completely overlooked. In addition, sites should be selected with better care and in a greater number as well. Plus, analyzing roads that have been decommissioned for more time would be valuable.

The historic sites were decommissioned in 1994/5 using ripping and seeding techniques. At these sites, there was an attempt to return the environment back to a pre-disturbed status. Long timescales are associated with ecosystem recovery (Hurtt *et al.*, 2002), far less time than time since decommissioning.

Section 4.5 Conclusion

The data collected and analyzed through this process supported our initial hypothesis that the untreated, control, and historically ripped/seeded sites would have different insect diversity. The further analysis with bioindicator families showed that there is currently a transition or succession occurring from the untreated sites to the historic sites. The analysis suggested that with additional time, the historically ripped/seeded sites should be able to achieve the diversity observed in the control sites. This supports the Forest Service's goal of decommissioning roads to return them to a pre-disturbance state. However, our data suggest that the time that it will take to achieve the original state is longer than the 15/16 year timeframe post rip and seed treatment that we studied. Further monitoring of insect populations at historically treated sites may provide more insight to the time it will take for a road to recover.

Section 4.6: References

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Section 5: Evaluating Plant Species Presence in Forested Road Environments

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Section 5.0: Abstract

Section 5.0.1: Native Vegetation

Forest road decommissioning seeks to restore the site to a natural state, but the effects of decommissioning on native vegetation are uncharacterized at this time for the Northern Rockies. Indigenous plant establishment on restored sites contributes to many other site functions including soil properties and hydrologic processes. To determine how the decommissioning techniques affected native plant species, sites in Hyalite Canyon, Gallatin National Forest, Montana were sampled during the summer of 2010. Three roads for each of four treatment types, including recontouring and seeding, ripping and seeding, and elimination of vehicle traffic (untreated) all of which took place in 2010, as well roads ripped and seeded in 1994 and 1995 (historical), were studied. Three transects were located on each road, bisecting the road perpendicularly, with a 0.25 m^2 plot at the center of the road, one 15 m upslope, and one 15 m downslope. Each plot was sampled before and immediately following treatment, species richness and canopy cover measurements were taken, and these data were evaluated using ANOVA and Tukey HSD comparison of means. Untreated roads were found to have a significant difference (α=0.05) for species richness and canopy cover between road and upslope plots, while no significant difference was found between untreated and historically treated road plots.

Section 5.0.2: Seed Germination

The US Forest Service applies seed mixes of mainly native plant species to decommissioned roads in Gallatin National Forest, Montana, but the germination and emergence of these species in this environment is unknown. The four species comprising the greatest proportions of the dry seed mix were selected and tested for germination rates and viability for scarified and unscarified seeds at a range of temperatures possible at the restoration sites. Petri dishes with two layers of blotting paper saturated with water were loaded with ten seeds of a species, and nests of one plate for each of the four selected species were positioned at five evenly spaced points on a thermogradient table with a surface temperature range of 0˚C to 32˚C. For 15 days seeds were observed for germination and emergence, and on the 15th day remaining seeds were tested for viability (indicated by seed firmness). Germination rates were high at the four warmest positions on the thermogradient table, but seeds rarely germinated at the coldest position. Scarification had no significant impact on seed germination but did reduce seed viability in all species. No germination was observed in *Bromus marginatus*.

Section 5.1: Introduction

Roads are a critical component of civilization. They provide access for people to study, enjoy, or contemplate natural ecosystems (Lugo and Gucinski, 1999). In forested areas, road effects and uses may be somewhat arbitrarily divided into beneficial and detrimental. The largest group of beneficial variables relates to access and includes timber harvesting, grazing, mining, recreation, fire control, land management, research and monitoring, access to private holdings, restoration, local community critical needs, subsistence, and the cultural value of the roads themselves. Non access-related benefits include edge habitat, firebreaks, the absence of economic alternatives for land management, and the jobs associated with building and maintaining the roads (Gucinski *et al.,* 2000). Undesirable consequences include adverse effects on hydrology and geomorphic features (such as debris slides, sedimentation), habitat fragmentation, predation, road kill, invasion by non-native species, dispersal of pathogens, degraded water quality and chemical contamination, degraded aquatic habitat, use conflicts, destructive human actions (Gucinski *et al.,* 2000).

Increased national transportation and subsequent road construction within the United States has resulted in significant impacts to plant communities and soil stability within road ecosystems (Petersen *et al*., 2004). It is widely recognized that roads have numerous ecological impacts, including increasing erosion, altering wildlife distribution patterns, habitat fragmentation, and introduction of non-native vegetation (Frenkel, 1970; Rich *et al.* 1994; Klein *et al.* 1995). As a result, opposition to road building and pressure to decommission roads in rural landscapes and public lands will continue to increase as roadless areas decrease (Lugo and Gucinski, 1999).

According to the National Forest Service, it is estimated that out of the 435,000 miles of forest service roads, 52,000 miles are not maintained for vehicle use (Northern Arizona University, 2010). Roads can have negative effects on a forest ecosystem. Soils can become compacted, which can reduce infiltration, aeration, pore space, disrupt water movements, increase erosion, and impact biological activities (Northern Arizona University, 2010). Native plant species are unable to thrive in such an environment and opportunistic species may become established.

The Northern Arizona University Ecological Restoration Institute defines the goals of decommissioned road restoration should include reduction in soil erosion, reestablishment of vegetation, restoration of terrestrial and aquatic habitats, and promotion of hillside stability (Northern Arizona University, 2010). An underlying assumption of an ecosystem-level restoration approach is that management activities promoting succession and increasing vegetation complexity will have indirect benefits for other, non-manipulated components of the ecosystem.

Vegetation plays a key role in the way an ecosystem functions; it reduces erosion, creates habitat, and adds to soil organic matter. Opportunistic species or poor vegetative cover can greatly affect such processes and the overall ecosystem.

Techniques commonly employed for forest road decommissioning and subsequent restoration are ripping, recontouring, slashing, and revegetation by means of broadcast seeding (M. Story, personal communication, February 2010). Ripping loosens the soil, increasing infiltration and creating microclimates that favor germination. Recontouring involves a bulldozer to bring in topsoil from adjacent downslope areas in order to recreate the previous slope. Revegetation is commonly paired with ripping and recontouring to reduce weedy plants species establishment with a mixture of native grasses and forbes, and also some desired non-native species. In addition, a common practice includes slashing the area of the decommissioned road with pine, fir, or willow branches to deter public use (M. Story, personal communication, February 2010).

Multiple studies have looked at the impacts of logging practices and skid roads on soil properties and herbaceous cover. Soil compaction resulting from forest road use can severely impair plant growth by restricting root growth, and subsequently decreasing photosynthesis rates (Barzegar *et al.,* 2006). A study of a sessile oak forest in Istanbul showed that the most disturbance resistant herbaceous species yielded <5% cover on compacted skid roads. Sparse cover provides little protection from further soil structure degradation, habitat loss, erosion, or competition to prevent the invasion of noxious weeds (Gungor *et al.,* 2008).

Natural recolonization of disturbed areas is the goal of most restoration practices. However, this goal is difficult to achieve and is typically accompanied by additional inputs to speed up the process and prevent invasion by undesirable weedy species. Such inputs include seeding and transplanting indigenous species, mulching, fertilizing, adding topsoil with a native seed bank as well as seeding with a native nurse grass (Densmore *et al.*, 1990; Cotts *et al.*, 1991; Dawson and van Bergen, 1991; Shearer *et al.*, 1996).

The US Forest Service currently utilizes dry broadcast seeding: the most affordable and time efficient means of seeding restoration sites. However, this method can be problematic due to exposure to predation and erosion. Soil compaction is generally acknowledged as a limitation to germination and emergence of a seeded species. Seedbed preparation is therefore a key variable to revegetation success on disturbed sites (Montalvo *et al.*, 2002).

According to a two-year southern California agricultural field study, soil ripping was found to have unclear and relatively insignificant impacts on seeded native species establishment on sites where topsoil was removed and soils compacted (Montalvo *et al.,* 2002). However, fewer weedy species established on ripped treatments compared to untreated compacted soils at these study sites (Montalvo *et al.,* 2002). This may indirectly augment native species cover due to less interspecific competition in the long term. The study showed that the seeding technique had the most significant impact on seed emergence and species density.

A study in Glacier National Park (GNP) addressed the effectiveness of several treatments to establish native prairie vegetation and reduce the occurrence of alien species following road construction (Tyser *et al.,* 1998). Although non-native species have commonly been used in roadside revegetation, these species may be susceptible to environmental stress and may also hinder the establishment of indigenous species (Wilson, 1989; Jefferson *et al.*, 1991). The relevance of this study involved five seeding treatments; three of which included a seed mix of native prairie species. Additionally, they evaluated the incorporation of a short-lived sterile wheat hybrid into certain seed mixes (Tyser *et al.,* 1998). The purpose of the nurse grass was to provide soil stabilization on steeper slopes, thus permitting establishment of the slower growing species (Densmore *et al.*, 1990). The study sites were topographically similar with similar southeastern exposures and elevation (1380 m) along with comparable native and non-native vegetation (Tyser *et al.,* 1998). The native seeding mixes provided the best native graminoid establishment (Tyser *et al.,* 1998). However, none of the seeding treatments significantly established native forb cover (Tyser *et al.,* 1998). This could be due to the fact that the native forbs were seeded at a much lower rate compared to the native graminoids. The study suggested that over time, the amount of native forbs may increase because of their seed bank longevity and the high amount 1000 seeds per m^2 in the seed bank prior to seeding activities (Tyser *et al.*, 1998).

Previous research has examined different seeding methods or soil treatments separately. How these factors might together affect the success of native plant establishment, weed suppression, and development of a diverse and healthy Rocky Mountain conifer forest community is currently unknown. Furthermore, previous studies have not compared ripping and recontouring as road treatments; these variable treatments may have significant impacts on the results of restoration practices on decommissioned forest roads.

At this moment, there are ten seed species in the dry seed mix which the Gallatin National Forest Service spreads onto treated roads; this includes nine grass species and one forb species. In previous studies, it has been shown that certain plant species do well over a wide range of temperatures, while others need a more specific, constant range (Lu *et al.*, 2008). The germination success of these species in the Hyalite area is unknown.

The objectives of our study were to i) measure and compare percent canopy cover of the seeded restoration species on untreated, ripped, recontoured and historically ripped roads before and after treatment; ii) compare richness and abundance of species within each treatment, and; iii) compare road revegetation in treated areas to off road vegetation; and iv) measure and compare dry seed mix germination rates at different temperatures and treatments.

Section 5.2: Methods

Section 5.2.1: Field sampling

Road sites consisted of three roads of each of treatment: untreated (UN), historically decommissioned (1994/5) (H), and roads that were recontoured and seeded (RC), and ripped and seeded (RS) in August 2010. The road sites were selected prior to treatment based upon proximity to each other and where topography, vegetation, and aspect were somewhat analogous. Once the sites were selected, three random baseline points were chosen along the road within the length of each treatment boundary. These random points became the road sample location type where disturbance was to occur and the slope was approximately zero. The non-road samples were located ~15 m upslope and ~15 m upslope of the road samples where disturbance from the treatments was unlikely (Figure 1). The only exception to this design was the rip/seed at road 1 (RS1). For this site, the upslope (undisturbed) sample locations were placed approximately 40 m from the road due to anthropogenic disturbances (i.e. camping).

On July 8th and 12th, pre-treatment samples were collected for the three RS and RC sites as well as the UN1 site. There were a total of nine pre-treatment samples collected within each treatment per road section for a total of 63 samples. The sampling included using 0.25 m^2 circular quadrat frames to collect vegetative data that included percent canopy cover and species richness. Any species that could not be identified in the field were collected and placed in bags labeled with the date, site, transect location and a description of the vegetation type, for example: “unknown grass A”. These samples were taken back to the laboratory to be identified. In order to return to the sample locations for the post-treatment measures, each plot was marked with road-hairs and flags in addition to collecting geographic coordinates (latitude/longitude) using a Trimble© XT GPS unit with sub-meter accuracy.

On August 25th and 26th, post-treatment samples for all RS, RC and UN sites were collected. In addition, historically ripped/seeded (H) sites (1994/95) were sampled using the same experimental design. A similar process was implemented in selecting sample locations for the historical treatments. A baseline transect was positioned where the road once existed and three random points were selected along this transect. These randomly selected points became the historic road samples where the rip/seed process occurred. Furthermore, the non-road samples were located ~15 m upslope and ~15 m upslope of the road samples where disturbance from the previous rip/seed activity was unlikely. Within each of the individual sites were three randomly located transects that ran perpendicular to the road. Three half-meter squared quadrats were assessed for all plant species: upslope, downslope and in the roadway. The locations of the 108 quadrats were acquired with Trimble GeoXT GPS receivers.

Section 5.2.2: Greenhouse Experiment

Our objectives with the greenhouse experiment were to determine the range and optimal germination temperature for the four species (*E. glaucus, P. spicata, B. marginatus, E. trachycaulus*) comprising the largest proportion of the seeds from the US Forest Service (USFS) dry zone seed mix (Table 5.1). Our second objective was to evaluate if seed scarification affected germination rates as well as viability over the same temperature range. Our experimental procedure was as follows. Five centimeter diameter petri dishes were prepared with two layers of blotting paper covering the bottom of each dish and saturated with approximately 10 ml volume of distilled water. Ten seeds of a given species were arranged evenly in five dishes for each of four species. The thermogradient equipment consisted of the six replicated aluminum plates, two PolyScience Programmable Temperature Controller (one hot and one cold) that pumped distilled water through the plates to create the temperature gradient from 0˚C to 30˚C and an Omega OMG-DAQ-56 USB Data Acquisition System recorded the temperature at 10 cm intervals within the plate every 10 minutes. At each of five points on a temperature gradient plates four petri dishes were positioned at random. Three gradient replicates were used for both unscarified and scarified seeds and were randomly assigned lanes on the thermogradient plate. Seeds were scarified using a sand paper lined tray and a sand paper block with ten passes in alternating perpendicular directions over the top of the seeds.

The plates received fluorescent light for 12 hours each day and more distilled water was added to plates if they dried out. The seeds were examined for germination daily. At the end of 15 days, a pressure test was conducted to assess seed viability. Percent germination was then calculated for each dish and compared across the temperature gradient using Microsoft Excel 2003.

Section 5.2.3.1: Data Analysis

Analysis of variance (ANOVA) tests were used to compare the means of the treatments and plots for all data collected. This allowed all factors to be simultaneously evaluated against each other, with the assumption that the data were independent, normal, and had an equality of variance. Tukey Honestly Significant Difference (HSD) multiple comparison of means was then implemented to assess all possible pairs of means and determine which pairs were significantly related at $\alpha=0.05$. Pairwise comparisons were made between relevant plots, broken down by treatment type and plot location (i.e. untreated upslope vs. untreated road or untreated road vs. historical road), within each month of collection. Each of these tests were performed using R© statistical software. All graphs were plotted and means calculated using SPSS statistical software.

Section 5.2.3.2: Seed Experiment

Data analysis for the seed experiment was conducted in Excel 2003. Pivot tables and pivot charts were used to organize data and create figures. The average of percent germination and percent viability was calculated along with standard deviation.

Common name	Scientific name	Variety	Origin	Dry mix	% in mix	Wet mix	% in mix
Blue wildrye	*Elymus glaucus*	Elkton	WA	x	20.81	x	6.37
Mountain bromegrass	*Bromus marginatus*	Garnet	MT	x	20.15	x	15.42
Bluebunch wheatgrass	*Pseudoroegneria spicata*	Goldar	"-"	x	14.17	x	9.76
Slender wheatgrass	*Elymus trachycaulus*	Pryor	ND	x	12.08	x	15.39
Idaho fescue	*Festuca idahoensis*	Joseph	WA	x	7.07	x	6.44
Lewis blue flax	*Linum perenne*	Appar	OR	x	6.74	x	9.35
Thickspike wheatgrass	*Elymus lanceolatus*	Critana	MT	x	6.71	x	12.34
Canada bluegrass	*Poa compressa*	Reubens	ID	x	3.58		
Quickguard	*Triticale*	Quickguard	WA	x	3.32	x	3.05
Inert Matter				x	2.45		2.9
Other Crop				x	1.45		2.57
Western Yarrow	*Achillea millefolium*	Eagle Mountain	WA	x	0.66		
Weed Seed				x	0.07		0.1
Big Bluegrass	*Poa secunda*	Sherman	WA			x	3.53
Tufted Hairgrass	*Deschampsia cespitosa*	Nortron	CAN			x	2.44
Violet prairie clover	*Dalea purpurea*	Kaneb	TX			x	10.34
Quickguard	Sterile tricale hybrid						
Seed from:	http://www.graniteseed.com/						

Table 5.1: Seed mix species used for reseeding in 2010 by Gallatin National Forest. Highlighted rows indicate species used in the germination experiment. Seed was sourced from the Granite Seed Company, UT.

Section 5.3: Results

Section 5.3.1: Native Vegetation

The data collected to compare species richness and percent canopy cover between treatments included all species present within the 0.25 m^2 plots, including both natives and non-natives. Due to time constraints, there was only one pre-treatment untreated (UN) site surveyed for richness and cover compared to three sites for each the recontoured (RC) and ripped/seeded (RS) pre-treatment conditions. No statistical test could therefore compare the untreated site with the RC and RS sites between pre-treatment and post-treatment.

Comparison of the RS and RC in June, before road decommissioning treatments were imposed, determined that the UN road had lower species richness and cover than all the UN off road plots, and both RC and RS plots (Figure 5.1), although there was high variability at all sites.

Pre-treatment total species richness on road was 1 for the untreated compared with approximately 6 for RS and RC. The average species cover for UN road plots was 43% compared with approximately 90% for UN off road plots. Cover on the RS and RC was similar (40-50% respectively) compared with 60-70% at off road sites. Off road species richness was approximately 3 for UN plots compare with approximately 8 for RS and RC plots. Note that only one of three UN roads was sampled pre-treatment compared with three roads each for RS and RC.

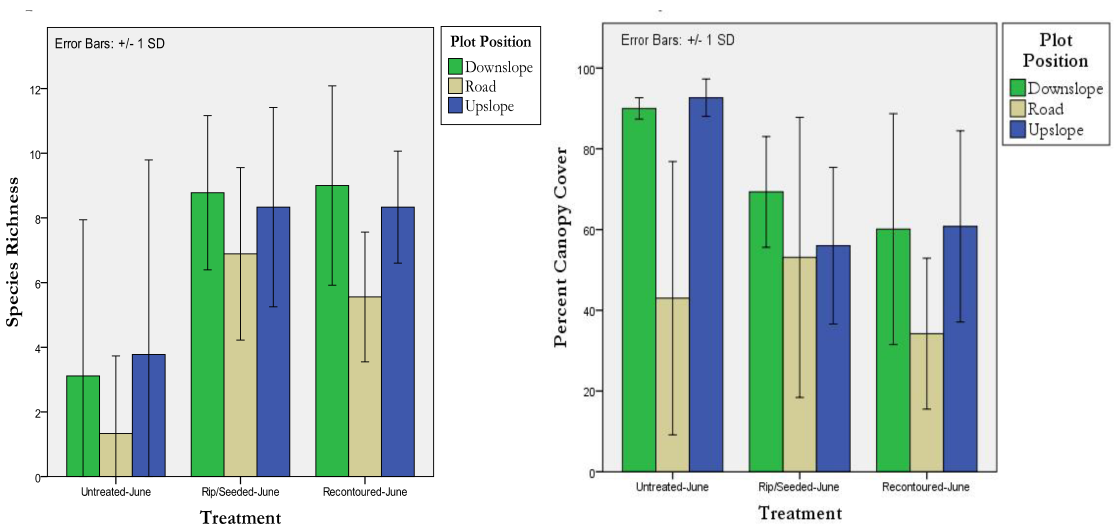

Figure 5.1: Species richness and percent canopy cover on each road position pre-treatment.

The post-treatment species richness and cover field collection did not take into account any effect decommissioning had on re-establishment of vegetative communities due to the fact that collection was done before any seeded vegetation was able to germinate (roughly 45 days after decommissioning). Figure 5.2 shows the lack of recovery on decommissioned roads in August. There was residual vegetative cover on RS roads (canopy cover μ = 11%, richness μ = 3), but RC roads exhibited no vegetation post-treatment.

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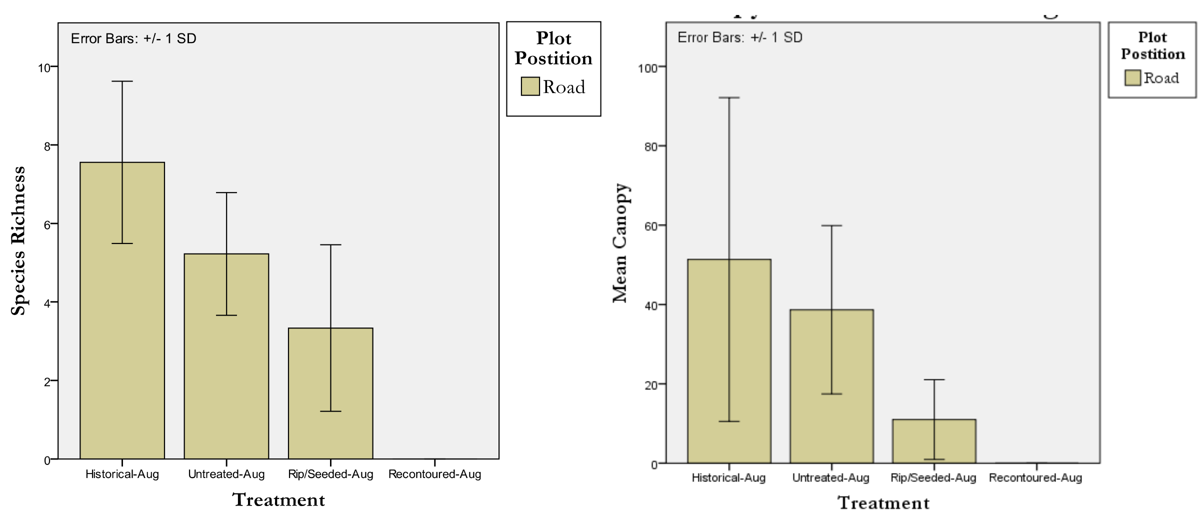

Figure 5.2: Species richness and percent canopy cover on roads post-treatment.

In order to meet the goal of comparing how road-decommissioning techniques affect species richness and cover compared to untreated roads, a comparison was made between historically ripped/seeded roads in 1994/95 and roads left untreated in 2010.

ANOVA tests determined that untreated road plots and historical road plots did not differ significantly in richness. The historical road plots did not significantly differ in either richness or canopy cover compared with the adjacent undisturbed up and downslope plots. The UN road plots, however, had significantly less richness than the adjacent upslope plots (Figure 5.3).

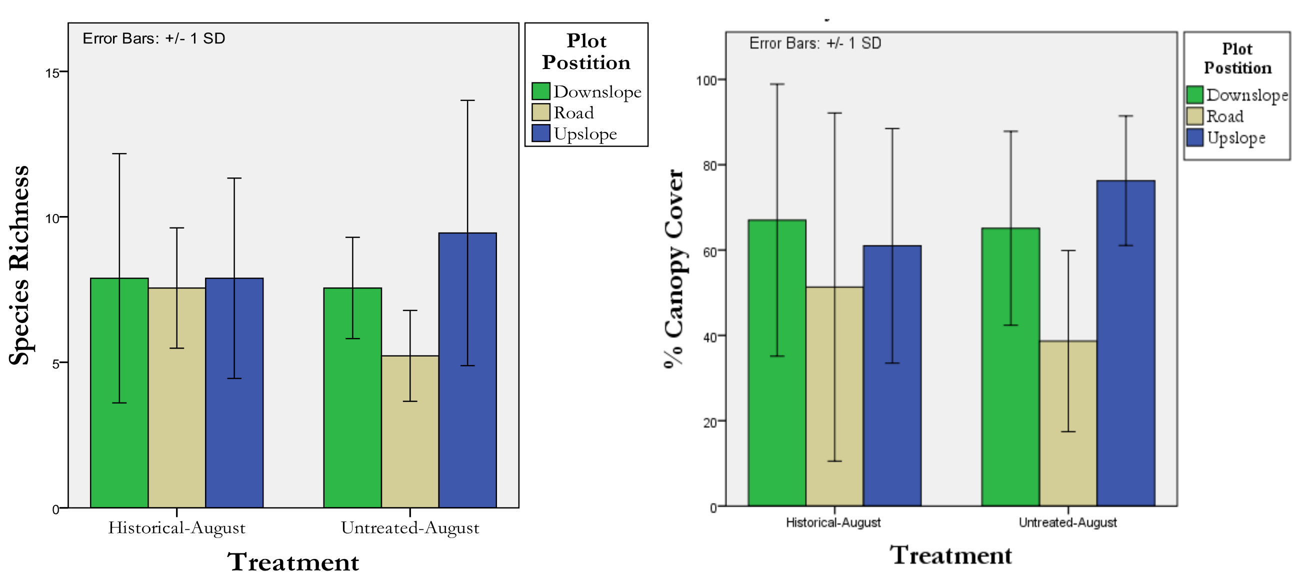

Figure 5.3: Percent cover on all plot positions for historically treated and untreated sites.

To summarize the species most commonly observed on roads and adjacent reference sites, Tables 5.2 and 5.3 were drafted. Species observed on four or more of the nine plots measured for each position, on pre and post-treatment road, were included in the lists. The species highlighted in blue font are species classified as non-native by the weed group (Section 6.0).

Pre Treatment Roads					
Ripped and Seeded			**Recontoured**		
Upslope	**Road**	**Downslope**	**Upslope**	**Road**	**Downslope**
Arnica cordifolia	*Achillea millefolium*	*Achillea millefolium*	*Arnica cordifolia*	*Bromus inermis*	*Arnica cordifolia*
Aster conspicuus	*Epilobium*	*Arnica cordifolia*	*Aster conspicuus*	*Taraxacum officinale*	*Epilobium*
Carex spp	*Fragaria virginiana*	*Carex spp*	*Erythronium grandiflorum*	*Trifolium pratense*	*Fragaria virginiana*
Spiraea betulifolia	*Phleum pratense*	*Fragaria virginiana*	*Fragaria virginiana*	*Trifolium repens*	*Linnaea borealis*
	Taraxacum officinale	*Taraxacum officinale*			*Lupinus sericeus*
	Trifolium pratense				*Osmorhiza depauperata*
	Trifolium repens				*Spiraea betulifolia*

Table 5.3.1. Summary of species present on plot positions prior to treatment, June, 2010.

Post Treatment and Untreated Roads					
Historically Treated			**Untreated**		
Upslope	**Road**	**Downslope**	**Upslope**	**Road**	**Downslope**
Arnica cordifolia	*Achillea millefolium*	*Arnica cordifolia*	*Achillea millefolium*	*Achillea millefolium*	*Arnica cordifolia*
Fragaria virginiana	*Epilobium*	*Carex spp*	*Arnica cordifolia*	*Phleum pratense*	*Aster conspicuus*
Calamagrostis rubescens	*Fragaria virginiana*	*Fragaria virginiana*	*Aster conspicuous*	*Taraxacum officinale*	*Fragaria virginiana*
Spiraea betulifolia	*Lupinus sericeus*	*Calamagrostis rubescens*	*Fragaria virginiana*	*Trifolium pratense*	*Spiraea betulifolia*
	Phleum pratense	*Spiraea betulifolia*	*Lupinus sericeus*	*Trifolium repens*	*Taraxacum officinale*
	Taraxacum officinale	*Vaccinium scoparium*	*Calamagrostis rubescens*		
	Trifolium pratense		*Spiraea betulifolia*		
	Trifolium repens		*Trifolium repens*		

Table 5.3.2. Summary of species present on plot positions of historically treated and untreated roads in August, 2010.

Section 5.3.2: Seed Experiment

Section 5.3.2.1: Overall Species Germination versus Temperature:

Germination of unscarified seeds varied by species and over the temperature gradient. *E. glaucus* had very low germination success at the lowest temperature (0.2°C) but had similar germination at the other four temperatures with an average of ~96 % germination (Figure 5.4). This species (*E. glaucus*) lacked standard deviation bars at 9.5°C and 24.3°C due to equal germination over all replications. Both *P. spicata* and *E. trachycaulus* had more normal bell shaped germination over the temperature range; however the standard deviations overlap for *E. glaucus* suggesting it can germinate well over the full temperature range. *E. trachycaulus* had no growth at 0.2°C, but had a bell curve peaking at 17.4°C where all seeds germinated in all three replicates, hence the lack of standard error bar for this temperature. Germination was less variable for this species with 24.3°C providing the second highest germination, and 31.7°C and 9.5°C providing less germination but with no difference between them. *B. marginatus* had no germination at any of the temperatures.

Seeds that were scarified had varying germination rates for all species across the temperature gradient (Figure 5.4). *E. glaucus* had the highest percent germination (96%) between the temperatures of 9.5°C through 24.3°C. Results from replications for *E. glaucus* on temperature 0.2°C ranged greatly, giving it a high standard deviation. Overall, standard deviations overlapped between all temperatures. *P. spicata* resulted in normal bell shape germination, similar to unscarified, with temperature 24.3°C being the highest average germination (90%). *E. trachycaulus* had considerably lower germination at temperature 0.2°C. All other temperatures overlapped with standard deviation. Similar to the unscarified treatment, scarified *B. marginatus* did not germinate at any of the five temperatures.

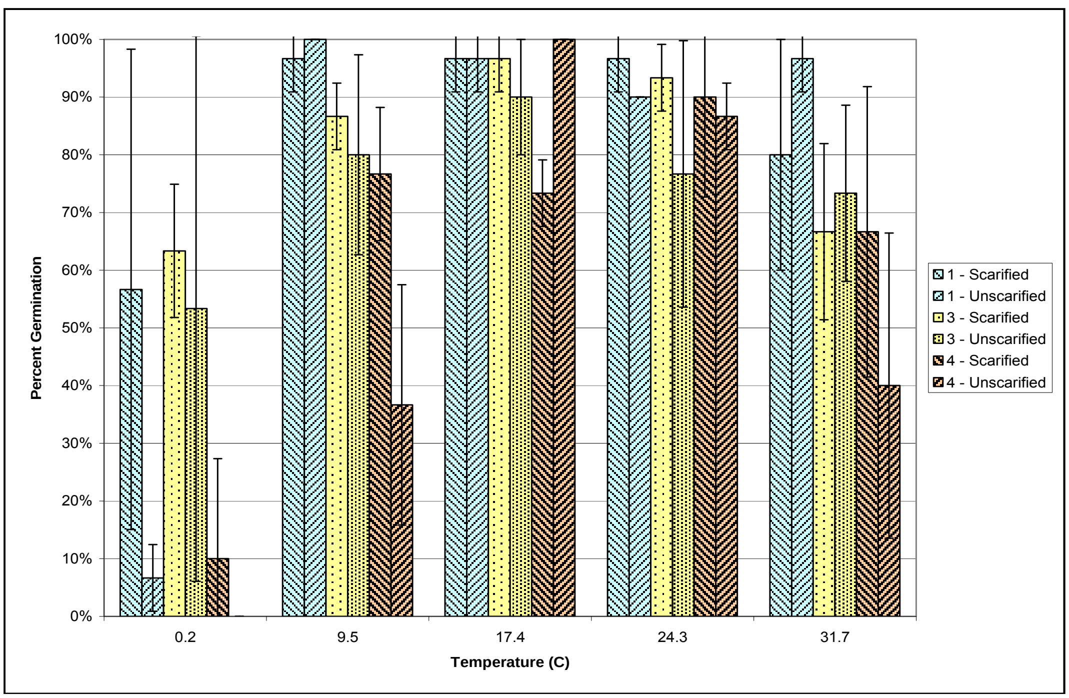

Figure 5.4: Percent germination for unscarified and scarified seeds versus temperatures between 0.2°C and 31.7°C. 1= *Elymus glaucus*; 3= *Pseudoroegneria spicata*; 4= *Elymus trachycaulus*.

Section 5.3.2.2: Cumulative Germination Rates

When comparing between the scarified and unscarified treatments, there is only slight variation between the two; most standard deviations overlapped one another. The few which did not were *E. glaucus* at 0.2°C, *E. trachycaulus* at 9.5°C, and *E. trachycaulus* at 17.4°C.

E. glaucus consistently presented the highest percent germination, albeit, with significant variation (Figure 5.4-5.6). Unscarified and scarified treatments showed similar rates of germination. Temperatures of 24.3°C and 17.4°C resulted in the highest rate of germination for this species, taking three to five days to plateau. However, 31.7°C and 9.5°C presented a more gradual germination rate, increasing until close to the end of the study. At 0.2°C, *E. glaucus* took twelve days to germinate. Scarified *E. glaucus* presented a higher rate of germination than unscarified treatments.

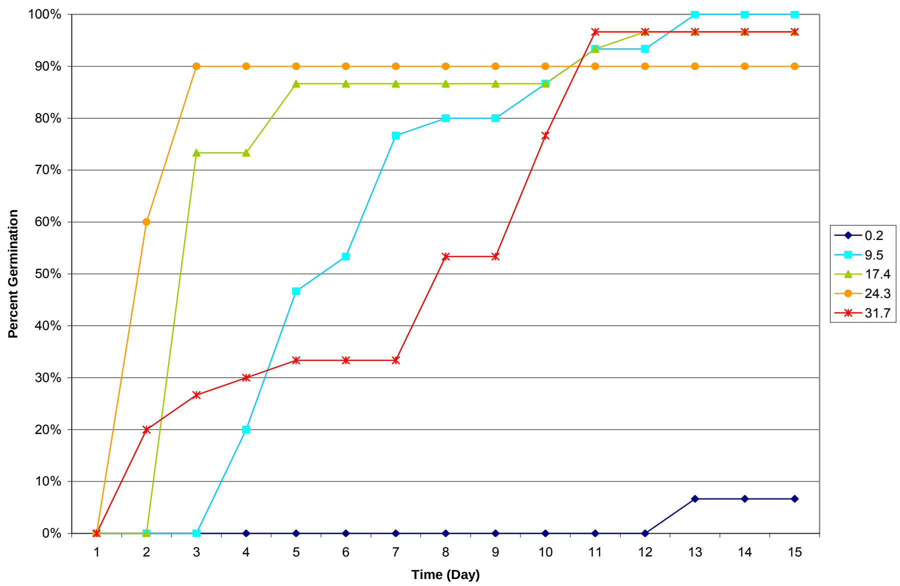

Figure 5.4: Unscarified *Elymus glaucus* germination rate for temperatures (°C).

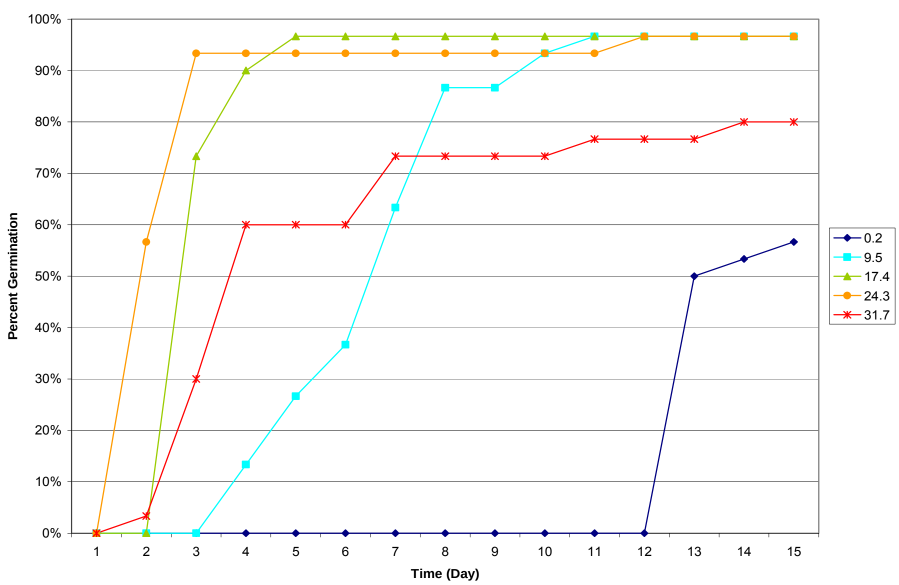

Figure 5.5: Scarified *Elymus glaucus* germination rate for temperatures (°C).

Section 5.3.2.3: Species Viability

At day 15, seed that had germinated was totaled, the remaining seeds were assessed for viability and divided into two groups, viable but not germinated or non-viable (Figure 5.6). The results encompass all temperatures separated by seed species and by scarified versus unscarified. The results show there was high seed germination in species *E. glaucus*, *P. spicata*, and *E. trachycaulus* in both scarified and unscarified. All unscarified *E. trachycaulus* replications showed 100% viability. *B. marginatus* shows a reduced average viability compared to the others, and was lowest viability when scarified, though there was high variation (not shown). The variability in total germination was less for unscarified seeds (data not shown).

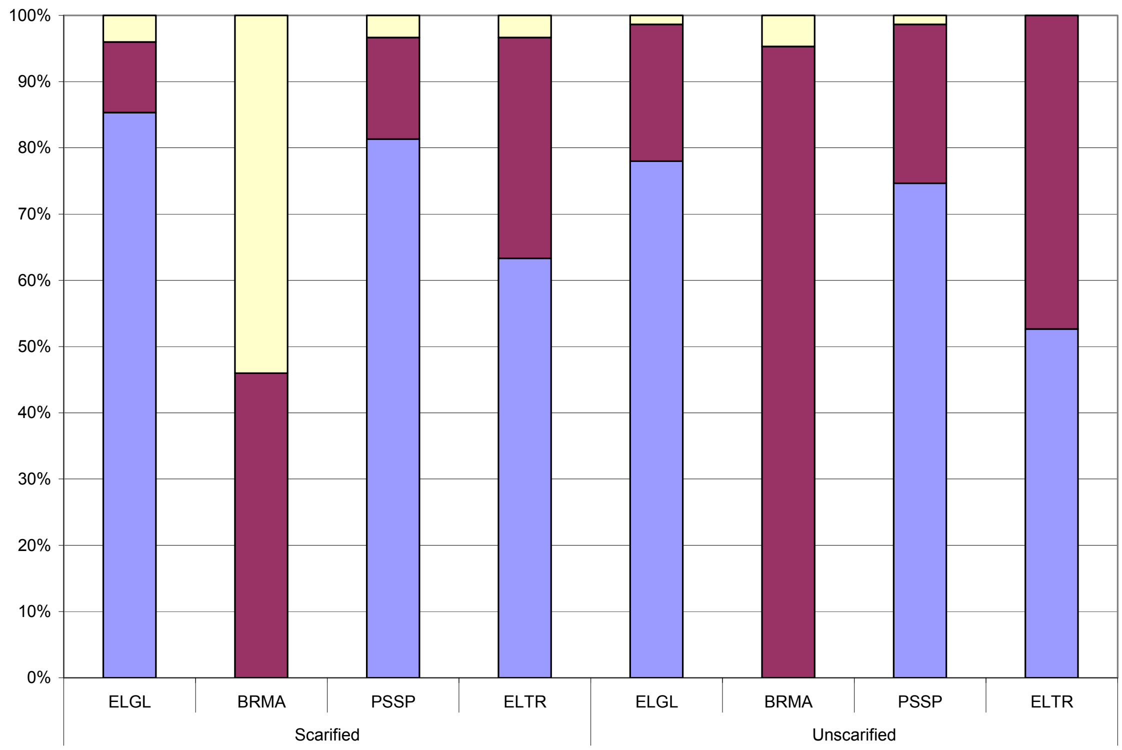

Figure 5.6: Percent germination (blue), viable but not germinated (burgundy), and unviable seed (yellow) for each species, for both the scarified and unscarified treatments. ELGL= *Elymus glaucus*; BRMA= *Bromus marginatus;* PSSP= *Pseudoroegneria spicata*; ELTR= *Elymus trachycaulus*.

Section 5.4: Discussion

Section 5.4.1: Native Vegetation

New roads established in forested landscapes often lead to economic development as well as deforestation and habitat fragmentation (Forman and Alexander, 1998). This leads us to the ongoing debate of whether roads lead to development or if development leads to roads? This has greater ramifications as environmental quality becomes more important in the transportation-land use interaction (Forman and Alexander, 1998). It is widely recognized that roads have significant ecological impacts ranging from compacted soils to disrupting wildlife distribution patterns. However, roads are a necessary component of society and will continue to be built and maintained even as roadless areas continue to decrease. On many occasions, decommissioning of roads in rural settings has been implemented in order to remove the road's influence and restore a sustainable ecosystem much like the ecosystem prior to disturbance. Such decommissioning techniques used by the United States Forest Service include rip and seeding, re-contouring and seeding and no treatment save blocking anthropogenic access, particularly vehicular. We have used untreated areas to represent controls from which we can evaluate the affect of the treatments. For this research, the vegetation measurements were limited to pre-treatment conditions, historically rip/seeded site measurements and post-treatment measurements (post-treatment measurements occurred within a short time period (~45 days), we expected to observe little (if any) vegetation. In the pre-treatment measure, the untreated roads and adjacent positions were only measured in Rep 1, due to time constraints. Since we could not measure what will happen along these treatments in subsequent years, we compiled and compared the canopy cover and species richness within each pre-treatment to historically rip/seeded roads. The goal was to provide baseline plant species richness and canopy cover data prior to any treatment for all roads including roads left untreated and compare the date to the historically ripped/seeded roads treated in 1994/95.

The pre-treatment measures were collected on and adjacent (upslope and downslope) to the roads to be ripped/seeded, recontoured/seeded, and those to remain untreated (control) roads. From these results, we observed that the untreated roads had much lower species richness values along all three sampling positions (up/downslope and road) when compared to the pre-treatment conditions of the rip/seeded and recontoured/seeded roads (Figure 5.1). The differences in variability occurring on the untreated road may have been a product of sampling a single road for the control compared to three roads for each of the pre-treatment rip/seed and recontoured/seeded conditions. If sampling error was not the cause, the data suggest that the Forest Service does not consider the canopy cover or species richness of a road when determining which treatment to administer. Instead, it is likely a question of accessibility to the public that is used to determine which roads receive specific treatment (*i.e.* an untreated road is still accessible whereas a recontoured/seeded road will not be).

Following the treatments in August there was, unsurprisingly, very limited species richness or canopy cover in the road plots. Therefore we compared total plant species richness of the road plots between untreated and historically treated (rip/seed) roads (Figure 5.2) measured in August. The results show that the species richness for untreated roads and the historically rip/seeded road did not differ significantly ($p = 0.81$) with mean values of 7.5 and 5. This suggests that even though the historically rip/seeded road has been decommissioned for 15/16 years, there is still very little difference in regards to species richness to roads that have been used more recently. Comparison of canopy cover between the historically rip/seeded and untreated (measured in August), told a similar account (Figure 5.2). Canopy cover for the historically rip/seeded site was not significantly different between any of the positions (up/down/road). This suggests that over time, treated roads will resemble the up/downslope areas. As for the untreated roads, the only positions that were significantly different were the upslope versus road ($p = 0.05$; Figure 5.3). However, the downslope versus road positions were not significantly different. A comparison of canopy cover on road positions between the untreated and historically rip/seeded roads did not yield a significant difference as well ($p = 0.98$; Figure 5.3), which is a similar result to the species richness comparison between these two positions and treatments.

Frequency of species occurrence in our up- and upslope and road plots and treatments was assessed (Tables 5.2 and 5.3). Four species commonly occurred on road plots and these were non-native species, three of which were introduced to improve cattle forage including: *Phleum pratense,* timothy, *Taraxacum officinale,* common dandelion, *Trifolium pratense,* red clover, and *Trifolium repens*, white clover. *Phleum pratense* is a grass while the others are forbs. An invasive grass species observed on four pre-recontour road plots was *Bromus inermis*, smooth brome. Two native species were also commonly observed both on road plots and either the adjacent upslope or downslope positions:

Achillea millefolium, common yarrow, and *Fragaria virginiana*, Virginia strawberry, were observed in. This was likely due to either remnant viable seed in the seed bank, or volunteers from upslope and downslope positions that have spread from native (undisturbed) conditions to disturbed conditions.

Reseeding of formerly bare and compacted ground has been conducted with success on campsites in subalpine sites in Oregon (Cole, 2007). In the Oregon study the ground was thoroughly scarified to a depth of 15 cm, followed by seeding achieved a density of 55 plants/m^2 on restored. These results were significantly increased when soil amendments were included in the treatment (Cole, 2007).

This project has established a baseline inventory that could be utilized in future endeavors when road decommissioning is considered along U.S. Forest Service roads in Hyalite Canyon, and other parts of the Gallatin National Forest. Continuing to research and study the effects of these treatments on species richness and canopy cover will be essential to understanding which treatments should be implemented in the future. Future studies should include determining species diversity changes over time in order to have a better understanding of the ecological impact of these decommissioning techniques.

Section 5.4.2: Seed Experiment

Section 5.4.2.1: Germination versus Temperature

For the unscarified treatment, *E. trachycaulus* presented the most variable germination differences between temperatures; with an optimal germination at 17.4°C. *E. glaucus* and *P. spicata* germination overlapped between temperatures 9.5°C and 31.7°C with no optimal temperature due to standard deviation bar overlap. *E.glaucus, P. spicata*, and *E. trachycaulus* had the highest percent growth between 9.5°C and 24.3°C for scarified seeds. At the high (31.7°C) and low (0.2°C) temperature extremes, the average percent germination for the three species was much lower. *E. glaucus* showed a more consistent high percent germination throughout the temperature range with no obvious optimal germination temperature. *B. marginatus* did not germinate at any temperature regardless of treatment.

Over most temperatures in this experiment, germination for each species was not significant between treatments. *E. glaucus* at 0.2°C, *E. trachycaulus* at 9.5°C showed a higher percent germination when scarified, whereas, *E. trachycaulus* at 17.4°C had a significantly lower percent germination. Overall, the results showed it would be unnecessary to scarify seeds prior to dispersal since the difference between the two was negligible.

Section 5.4.2.2: Germination Rate

E. glaucus, as well as *P. spicata*, and *E. trachycaulus* rate of germination varied between temperatures. Temperatures of 24.3°C and 17.4°C resulted in the highest rate of germination, taking only three to five days to plateau; whereas temperatures at 31.7°C and 9.5°C presented a more gradual germination rate, and 0.2°C resulted in minimal growth. *E. glaucus*, as well as *P. spicata*, and *E. trachycaulus* germination preferred temperatures between 17.4°C and 24.3°C.

Section 5.4.2.3: Species Viability

E. glaucus, *P. spicata*, and *E. trachycaulus* had high germination and viability showing low microbe and fungus growth both for scarified and unscarified seeds in the laboratory. *B. marginatus*, however, showed high amounts of fungal growth such as penicillium, and an unknown black fungus, especially when scarified. *B. marginatus* was more susceptible when scarified than unscarified. Upon research, NRCS stated that *B. marginatus* must be treated with a fungicide before planting.

Section 5.4.2.4: Overall/Conclusion

There was no significant difference in percent germination between *E. glaucus*, *P. spicata*, and *E. trachycaulus* within the Forest Service seed mix between scarified and unscarified. However, *B. marginatus* were viable though they did not germinate within the 15 day time span. With this, it can be seen that it is unnecessary, and could be detrimental to scarify seeds before dispersal.

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Section 6: Non-Native Species in Relation to Forest Roads and Decommissioning

Researchers: Dan Campbell, Amanda Lipe and Bobby Peters

Section 6.0: Abstract

This study focused on the ratio of non-native to native plant species observed on road, upslope and downslope plots along untreated, ripped and seeded in 2010 and in 1994/5, and recontoured and seeded in 2010 roads. The non-native to native ratio was less than one, *i.e.* more natives, for all off road plots at all sites. On the road sites there were more non-natives to natives at both the June and August sampling, with the exception of the recontoured and historic plots in August. The recontoured plots had bareground due to the recently imposed treatment. The historically ripped and reseeded sites had a median ratio below one, suggesting that this treatment has potential for recovery to a more native vegetation over time. *Taraxacum officinale, Trifolium pratense* and *Trifolium repens* were observed on all road treatment plots, and *Phleum pratense* on three of the road treatments. A greenhouse study of the timing of herbicide application on seeds and seedlings using picloram and glyphosate showed variable responses but nine of the ten species were negatively affected by the treatment. The tenth species failed to germinate.

Section 6.1: Introduction

There has been controversy over the management of forested public lands since the creation of the USDA Forest Service in 1905 (Grace and Clinton, 2007). In some cases, roads have been described as ecosystems in the sense that they are structured, support biota, change over time, exchange energy and matter with adjacent systems and occupy ecological space (Lugo and Gucinski, 1999). Road cuts generally have high species richness as more light and moisture reaches plants (Forman and Alexander, 1998). Reasons for removing roads include: restricting access, reducing erosion, protecting endangered species, increasing hillslope stability and restoring habitat (Switalski *et al.,* 2004).

Certain forest service roads are being decommissioned because of degraded conditions such as erosion, trash accumulation, and soil compaction. The process of decommissioning as well as prior road use provides a potentially suitable habitat for non-indigenous plants. Some non-native plant species can negatively affect natural ecosystems due to competition for resources such as light, nutrients, or water (Flory and Clay, 2009).

The treatments commonly used in road decommissioning are ripping and re-contouring both combined with seeding, and natural re-vegetation. These methods are often used in combination with slash. Slash is a technique where vegetation cuttings are used to block roads from use. Non-native species management should not only consider the effectiveness of removal but how removal methods influence native plant responses (Flory and Clay, 2009). A previous study showed that native species establishment was effective when prior vegetation was disturbed; likewise, native plant establishment was enhanced with herbicide and tillage (Skousen and Venable, 2008).

When roads are decommissioned, it is important to reestablish desirable vegetation as quickly as possible to inhibit the growth of less desirable and invasive species. In addition, plants provide a visual sign of ecosystem recovery, create animal habitats, and prevent erosion (Switalski *et al.*, 2004). While any vegetation will slow erosion, it is important that the introduction of non-native species is avoided, particularly on state and federal land which are under mandates to control and manage noxious weed species. "The invasion of natural communities by non-indigenous plant species is a threat to native biodiversity and is currently rated as one of the most important global scale environmental problems" (Vitousek *et al.*, 1996).

The problem with native species restoration is slow establishment. Two factors affect road restoration (1) non-native species interference and competition, and (2) limited knowledge of growth requirements of slow-establishing native species (Skousen and Venable, 2008).

The first objective of this study was to measure native and non-native species cover and richness to determine how the ratios differed between road treatment types and off and on road locations. The second objective was to determine if timing of application of the herbicides picloram and glyphosate affected germination rates of seeds or injured seedlings of species used in the Forest Service revegetation seed mix.

Section 6.2: Materials and Methods

Section 6.2.1: Site description

For details on the site please see the general site description in Section 1.

Section 6.2.2: Sampling

Road sites consisted of three roads of each of treatment: untreated (UN), historically decommissioned (1994/5) (H), recontoured and seeded (RC), and ripped and seeded (RS) in August 2010. The road sites were selected prior to treatment based upon proximity to each other and where topography, vegetation, and aspect were somewhat analogous. Once the sites were selected, three random baseline points were chosen along the road within the length of each treatment boundary. These random points became the road sample location type where disturbance was to occur and the slope was approximately zero. The off road samples were located ~15 m upslope and ~15 m upslope of the road samples where disturbance from the treatments was unlikely (Figure 6.1). The only exception to this design was the rip/seed on road 1 (RS1). For this site, the upslope (undisturbed) sample locations were placed approximately 40 m from the road due to anthropogenic disturbances (i.e. camping).

On July 8^{th} and 12^{th}, pre-treatment samples were collected for the three RS and RC sites as well as the UN1 site. There were a total of nine pre-treatment samples collected within each treatment per road section for a total of 63 samples. The sampling included using 0.25 m^2 circular quadrat frames to collect vegetative data that included percent canopy cover and species richness. Any species that could not be identified in the field were collected and placed in bags labeled with the date, site, transect location, and a description of the vegetation type, for example: "unknown grass A". These samples were taken back to the laboratory to be identified. In order to return to the sample locations for the post-treatment measures, each plot was marked with road-hairs and flags in addition to collecting geographic coordinates (latitude/longitude) using a Trimble© GeoXT GPS receiver with sub-meter accuracy.

On August 25^{th} and 26^{th}, post-treatment samples for all RS, RC and UN sites were collected. In addition, historically ripped/seeded (H) sites (1994/95) were sampled using the same experimental design. A similar process was implemented in selecting sample locations for the historical treatments. A baseline transect was positioned where the road once existed and three random points were selected along this transect. These randomly selected points became the historic road samples where the rip/seed process occurred. Furthermore, the non-road samples were located ~15 m upslope and ~15 m upslope of the road samples where disturbance from the previous rip/seed activity was unlikely. Within each of the individual sites were three randomly located transects that ran perpendicular to the road. Three half-meter squared quadrats were assessed for all plant species: upslope, downslope and in the roadway. The locations of the 108 quadrats were acquired with Trimble GeoXT GPS receivers.

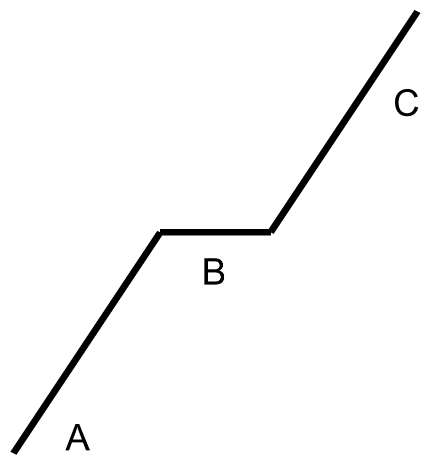

Figure 6.1: Sampling profile cross section with 0.25 m^2 quadrats placed at A) downslope from the road B) road C) upslope from the road.

Section 6.2.3: Greenhouse Experiment

After recontouring or ripping of a site is completed, revegetating the site is a priority to prevent erosion, maintain water quality, and discourage recreational use. The Forest Service uses seed mixes consisting of grasses and forbs native to the local area. This experiment was broken down into two parts: pre-emergence and post-emergence herbicide treatments.

The first step was to fill 240 10 cm diameter square pots with the MSU soil mix. Half of the pots were planted with ten seeds each of the species listed (Table 6.1) and were divided into three groups (with four replications each) denoted by colored label sticks: white for control, yellow for picloram, blue for glyphosate, and allowed 11 days for germination prior to herbicide application. Yellow marked pots were sprayed with picloram, and blue marked pots were sprayed with glyphosate, and controls were unsprayed. Rates similar to those used by the Forest Service were used.

On the 11th day, the 120 remaining pots were also divided into three groups (again with four replications each) marked with colored markers: yellow marked pots were sprayed with picloram, and blue marked pots were sprayed with glyphosate. Three days later, the pots were planted with one of the ten selected species – resulting in four reps of each species per treatment. Gloves were worn to eliminate exposure to the herbicides. Sixteen days were allowed for the seeds to germinate and emerge prior to assessment.

Throughout the greenhouse experiment, the pots were sprayed every twelve hours with a mist of water, and due to uneven spraying patterns, the pots were randomly moved every two or three days. At the end of the sixteen days, the levels of injury to the seedlings and the germination/emergence rates and seedling injury were noted visually and quantified graphically.

Each species was evaluated after both pre- and post-emergence herbicide applications on a (0-5) scale. Zero showing no herbicide injury and five showing seedling death or failure to germinate due to herbicide application. Identification of herbicide application was unknown by evaluators to produce unbiased results.

Common name	Scientific name	Abbrev -iation	Variety	Origin
Blue wildrye	*Elymus glaucus*	ELGA	Elkton	WA
Mountain bromegrass	*Bromus marginatus*	BRMA	Garnet	MT
Bluebunch wheatgrass	*Pseudoroegneria spicata*	PSSP	Goldar	MT
Slender wheatgrass	*Elymus trachycaulus*	ELTR	Pryor	ND
Idaho fescue	*Festuca idahoensis*	FEID	Joseph	WA
Lewis blue flax	*Linum lewisii*	LILE	Appar	OR
Thickspike wheatgrass	*Elymus lanceolatus*	ELLA	Critana	MT
Canada bluegrass	*Poa compressa*	POCO	Reubens	ID
Quickguard	*Triticale sp.*	QUIK	Quickguard	WA
Violet prairie clover	*Dalea purpurea*	DAPU	Kaneb	TX

Table 6.1: Seed mix species used for reseeding in 2010 by Gallatin National Forest. Seed was sourced from the Granite Seed Company, UT.

Section 6.2.4: Data Analysis

From the data collected at our field sites, a new response variable, non-native: native species ratio was derived. The non-native: native species ratio measures the relative contribution of non-native (exotic) and native species to the total species richness in a single value. Non-native:native ratio = number of non-native species/number of native species, meanings of the values are listed as follows:

Ratio > 1.0: more non-native species relative to native species
Ratio < 1.0: more native species relative to non-native species
Ratio = 1.0: equal number of non-native and native species
Ratio = 5.0: composed entirely of non-native species
Ratio = 0.0: composed entirely of native species
Ratio = -1.0: bare ground (no plant species present)

The non-native: native species ratio for every road treatment and plot position combination was evaluated graphically using box and whisker plots. Likewise, results of our Greenhouse Experiment were analyzed graphically.

Section 6.3: Results

Results of our graphical analyses using the non-native: native ratios show that there was little variation in the data for the upslope and downslope sites (Figure 6.2). At the majority of these sites, the non-native to native ratios were less than one which indicates there was a greater proportion of native species than non-native species in these areas. The only exceptions were three of the post-treatment downslope sites that had ratios greater than one. Although this result was likely not statistically significant, there was a slight trend of a greater proportion of non-native species inhabiting the areas downslope from the road. The road plots all had an non-native:native ratio of greater than one showing that the roads had more non-natives than natives. The exception was the recontoured plots post-treatment that was bareground due to the recently imposed treatment that turned over all the soil. In the post-treatment plots both the ripped and reseeded, and untreated plots had similar medians and wide variability. Some of the variability in the ripped plots was due to the tine used in ripping leaving bareground.

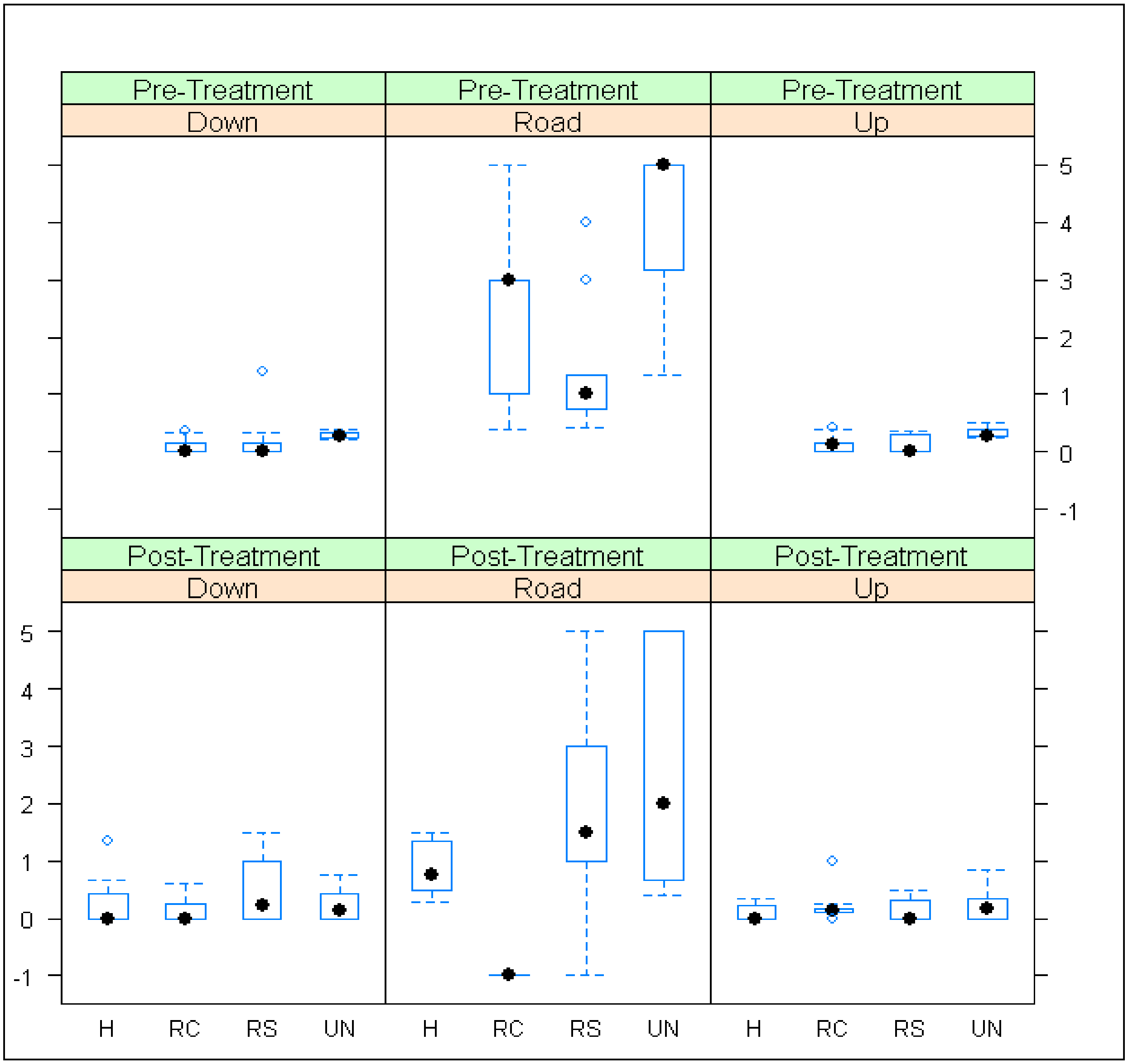

Figure 6.2: Non-native to native ratios for all decommissioning treatments at both observation times.

Taraxacum officinale, Trifolium pratense and *Trifolium repens* were observed on all road treatment plots, *Phleum pratense* on the RS, H and UN road treatments, and *Bromus inermis* was frequent on the RC plots (Section 5, Figure 5.3.2). None of these species were frequent off the road.

Section 6.3.1: Greenhouse Experiment Results

Emergence of the seeds planted into unsprayed pots ranged from 55 to 95% of the different species, with the exception of the *B. marginatus* which did not emerge in the control plot (data not shown). The germination of species from pots sprayed with picloram or glyphosate was compared relative to the control. Emergence in the picloram pots was lower than the control for all species except *P. spicata*. Emergence from pots sprayed with glyphosate was generally higher than the picloram treatment, and equaled or exceeded the control for six of the ten species (Figure 6.3).

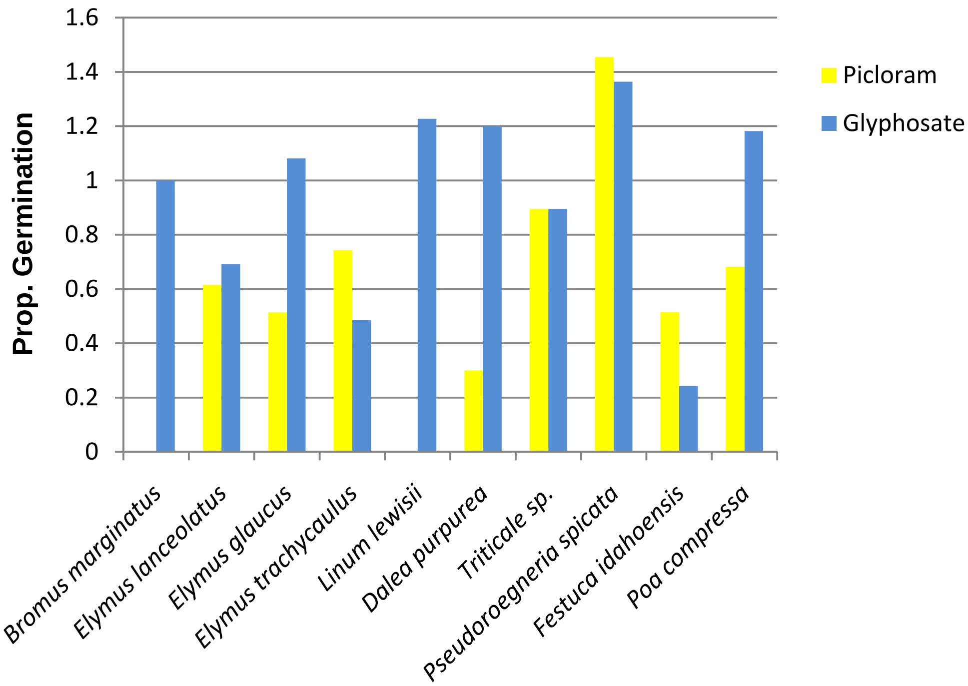

Figure 6.3: Relative proportional emergence of ten species to the untreated control after spraying the soil with picloram or glyphosate, or no herbicide three days prior to planting the seeds.

Injury to the seedlings when herbicide was applied eleven days after emergence was low (injury scale of 1) for most species and herbicides (Figure 6.4 and 6.5). Picloram application showed injury level 2 to the three *Elymus* species and *P. spicata*. The forb *L. lewisii* showed the greatest injury (rating 3) but the other forb, *D. purpurea*, only had an injury rating of 1 (Figure 6.4). Glyphosate treatment produced an injury rating of 1 for five species, a rating of 2 for *P.compressa*, rating of 3 for *E. glaucus* and *P. spicata,* and a rating of 4 for *E. tracycaulus* (Figure 6.5).

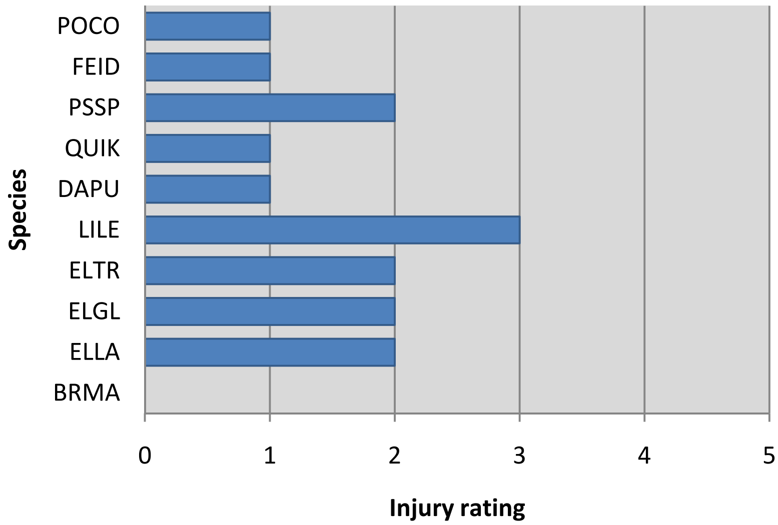

Figure 6.4: Injury to seedlings treated with picloram 11days after planting.

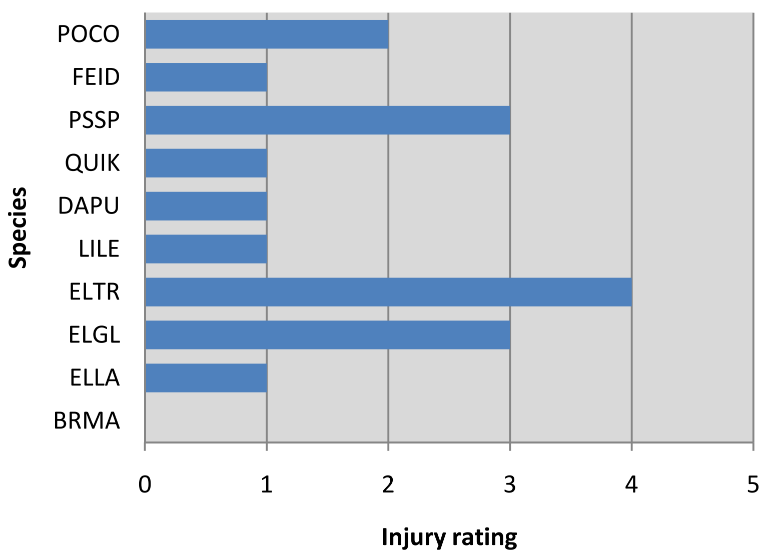

Figure 6.5: Injury to seedlings treated with glyphosate 11days after planting.

Section 6.4: Discussion

The non-native to native ratio on the roads were mostly between one and five indicating the high proportion of weed species to native species established on roads. The value of negative one at the RC post-treatment site was due to sampling soon after treatment while no vegetation was present. At the historic road sites, the ratio was just less than one suggesting that this treatment may provide a desirable habitat for natives over the 15-16 years since it was ripped and reseeded.

The non-native to native species ratios for off road plots were typically less than one both in July and August suggesting that these positions are not favorable to non-native species invasion.

The results of the greenhouse experiment were not clear other than all species were negatively impacted by the herbicide treatments. In the pre-treatment experiment, where seeds were planted into soil that was sprayed with herbicide three days prior, germination and emergence was inhibited relative to the control for most species as would be expected, with the picloram treatment having a more negative impact that glyphosate. When glyphosate and picloram were applied to seedlings these were also negatively affected with *P.compressa*, *P. spicata*, *E. trachycaulus* and *E. glaucus* more affected by glyphosate and *L. lewisii* more affected by picloram.

To our knowledge the Forest Service has not applied herbicide or performed any treatments on the historically treated sites since they were decommissioned. Our evaluation of the roads that were set for decommissioning this season with those decommissioned in 1994/5 the data suggest that over this time period decommissioned roads have the potential to have higher native to non-native cover and richness without requiring additional management.

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Section 7: General Discussion of Our Findings

Effective ecosystem management and manipulation requires knowledge about overall ecosystem process and function. Numerous factors and their interactions must be considered if ecosystem managers wish to reach their desired goals. In this study, key ecosystem functions and structures were analyzed: soil structure, sub-surface water connectivity, plant distribution, and insect communities. All groups examined the effect of forest roads and subsequent decommissioning practices on these ecosystem functions. Current practices were evaluated based on the success of delivering previous road areas back to a pre-disturbed state. More specifically, the research conducted asked whether there were noticeable trends or statistically significant differences between the roads that were untreated and roads that were treated by recontouring, and ripping both now and historically.

To address this question, field sampling was conducted in the summer of 2010. All groups sampled sites in June and again in August 2010 with some groups sampling at additional times. This approach ensured that all sites were assessed before treatments (road entrance blockage/untreated, ripped and seeded, and recontoured and seeded) were imposed on the roads in early August. These samples were compared with three roads historically ripped and seeded in 1994 and 1995. Below we discuss our interdisciplinary results regarding soils, water, insects, total plants and weeds. Several overall points concerning the effects of each road decommissioning treatment became apparent.

Untreated

The least expensive of the mentioned road decommissioning techniques is to leave the roads untreated and simply occlude entrance. The untreated roads were closed prior to the August sampling events. Because these roads were not treated over the summer, they were not expected to undergo drastic changes. Therefore, untreated roads acted as a control for sampling methods and provided a baseline to compare against other road treatments. Except for the insect data, there no other data sampled showed statistically significant differences of the untreated roads between June and August. Untreated road soils had the highest median bulk density of any treatment and a mean cumulative infiltration difference between on and off road sites of 66 mm; plant species had a richness of one and a canopy cover difference between on and off sites of 50%. Untreated roads had the highest mean non-native to native ratio of any treatment, and finally, there was a median of only one bioindicator insect family at each on-road sampling site. Clearly, if the goal was to restore the road to a pre-road condition simply closing the entrance to the road was not effective, at least in the timeframe we measured.

Recontoured

Contrary to the untreated roads, recontoured roads represented the most expensive form of decommissioning. Recontouring had the greatest visual impact on the roads, as well as many of the metrics. The soils along the road and up to two meters alongside the shoulder were completely agitated and homogenized to about a meter depth. Essentially, the area became a strip of loose, unconsolidated soil along the hillside. In August after treatment, there were no statistical differences in bulk density on and off road. This suggests that recontouring returned the soil to a pre-road condition, albeit there was no significant difference before treatment either. Attempts to perform infiltrometer tests on these sites demonstrated that these areas would infiltrate copious amounts of water (more than the group could carry to the site). Accurate infiltration tests were impossible due to an insufficient supply of water. However, this rapid infiltration may provide an explanation to the hydrologic results from these areas.

Recontouring was the only treatment evaluated for its hydrologic change. The wells installed in transects along the recontoured roads before treatment showed that, the downslope well responded similarly to the upslope well. After a large rain event, as the depth to water table decreased upslope, it also decreased downslope. After treatment, the upslope and downslope wells responded differently. Because these roads had high surface roughness and had low bulk density, it is possible that during these rain events, when the water ran on to the road, it infiltrated and was trapped within the loosened earth that was formerly the road. As the depth to water table decreased upslope during a rain event, it increased downslope. Thus recontouring reduced soil compaction and increased infiltration by eliminating surface runoff.

Plants were highly impacted by the recontouring treatment. Prior to the treatment, the flora consisted of grasses and forbs with an non-native/native ratio of three and a canopy cover near 35%. After treatment however, all plants were eliminated, but a good environment for plant establishment may have been created. The loose soil and large crevices in the soil surface will probably provide favorable conditions for seedling germination and emergence. Insect populations were not evaluated on recontoured roads, but due to the obliteration of all plants, it is likely that insect population would be severely depressed directly after treatment. Unfortunately, it is impossible to quantify the long-term effects of recontouring without conducting later field sampling to further evaluate the findings and hypotheses put forth from these short-term data.

Ripped & Seeded

The rip and reseed treatment imposed in 2010 presented a compromise between the costs and short term benefits of recontouring with the savings of leaving a road untreated. Accordingly, the road treatment produced mixed results. Visually, the treatment was variable in the areas. Where the tines had broken the soil surface vegetation was destroyed, whereas between the tines or where they could not break through the soil surface vegetation remained. Consequently measuring bulk density reliably was difficult post-treatment. The level of soil compaction, measured as bulk density, was not statistically different between on and off road in June or August, but the boxplots show that the median on road bulk densities were higher before than after treatment. This suggests that bulk density was decreased after treatment.

Ripped roads before treatment had a higher non-native to native ratio on road than off road with a mean value of 1.2. After the rip and reseed treatment in August, there was much higher variability but the median non-native to native ratio remained about the same. Total canopy cover on road plots decreased from a mean of around 50% to around 10%. Species richness also appeared to decrease from a median around 7 pre-treatment to around 3 post-treatment. Predictably, plant communities were negatively impacted directly after treatment. However, plant communities on roads before ripping already had lower species richness and canopy cover compared to their off road references. The question of whether ripping and reseeding can restore vegetative communities to an off road condition could possibly be answered using the data from historically ripped and seeded roads.

Historically Ripped and Seeded

Evaluating the success of road decommissioning is a relatively new phenomenon and to date, no long term monitoring data exists. This study was fortunate to have been conducted in the Gallatin National Forest where decommissioning has been a priority for some time. The historic sites were ripped and reseeded in 1994 and 1995. The data collected from these sites is extremely valuable for land managers because it represents some of the first and only long-term data available.

Soils on these historically treated roads had a significantly higher mean bulk density ($\alpha = .05$) than the off road values with a difference of 0.41 g/cm^3. When comparing the compaction medians between 2010 ripped/reseeded and historic sites, there is little difference. Furthermore, cumulative infiltration was still much higher off road than on historically treated roads with a difference of 219 mm. These results indicate what pedologists already know, specific soil properties may only develop over hundreds to thousands of years. The main visual difference between on and off road sites was the lack of organic matter, which corresponded to the lower species richness on roads. However, grass and forb species from upslope and downslope appeared to have colonized rather readily.

Plant species composition on the road was not perceived to resemble the seed mix used in 1994/5 – albeit information on the exact formula of the mix is sparse. The only species potentially present in the seed mix that was also present in the on road sample plots was yarrow (*Achillea millefolium*). However, species richness and canopy cover were not statistically different from either an upslope or a downslope site. Thus, species from the adjacent hillsides were dispersing onto the road where some were able to germinate and establish. Following the observation that only one potentially sown species was present 15-16 years after treatment a germination and scarification experiment was applied to the most common seeds in the mix. Most of the species germinated at temperatures above 9.5 C. Scarification did not enhance germination of most species and greatly reduced that of *Bromus marginatus*.

Many weedy species prefer disturbed areas and thus could be successful on ripped roads. However, the non-native to native ratio was less than one on historic roads, meaning there were more native plant species. This was lower than all mean values for pretreated roads, but was still higher than all mean values for every off road reference site. Clearly, many factors play into successful revegetation of decommissioned roads. These factors need to be identified and studied if future rip and seed treatments are to be effective.

Before long term ripping and reseeding success is judged simply by the similarity of values between species richness and canopy cover for on road and reference conditions, it must be stated that there was also no significant difference between these parameters with untreated and historically treated roads. If only means are compared, however, historical sites have a higher species richness and canopy cover than untreated sites. In short, the results are mixed; plants have established well on historically ripped roads, but weedy species may make up a greater percent of the cover than on reference conditions, the seed mix appears to be transitory at best, and the plants have not begun to alter the soil structure.

Insects rely on vegetation for food and habitat. Their abundance and distribution may elucidate the state of the vegetative community on the historically treated sites. The median number of bio-indicator families found at the historically treated roads was similar to the off road control. However, the interquartile range included samples where no bioindicator families were found. This finding was more similar to an untreated site. In other words, the historic site may share characteristics of both untreated sites and reference sites (in terms of bioindicators). Further speculation leads one to believe that historically ripped and reseeded sites are still in a state of transition, or succession. This same explanation fits well with the mixed findings presented in the plant section. Possibly with more time, in this context, ripped and reseeded roads will return to natural conditions.

Conclusion

In conclusion, while none of our studies showed significant evidence that the decommissioning treatments have affected the return of our assessed metrics to off road levels, some of our data do show that the treated roads from 1994/5 may be on their way towards ecosystem recovery of the flora and insect fauna, but that improvements in soil compaction and infiltration were minimal relative to the untreated and pre-treatment sites. To solve the debate of whether to rip or recontour a forest road, long-term data needs to be collected on recontoured sites and compared with the historic data collected in this report.

At the very least, this research has provided baseline data for pre-treatment conditions in regards to the soils, hydrology, insect population, as well as native and non-native vegetation. All are considered important fields when contemplating the complexities associated with mountain ecosystems. One is not more important than the other and all need to be examined to gauge the relative success of a decommissioning treatment. In the future, the Forest Service could use the results of this study to determine the relative status of a decommissioned road.