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Title: Effects on Vegetative Restoration of Two Treatments: Erosion Matting and Supplemental Rock Cover, in the Alpine Ecosystem

Running Head: Matting and Rock Cover Alpine Restoration

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Abstract

Unsanctioned travel routes through alpine ecosystems can influence water drainage patterns, cause sedimentation of streams, and erode soils. These disturbed areas can take decades to revegetate. In 2012 a volunteer-driven project restored a 854 m section of unsanctioned road along the Continental Divide in Colorado, USA. The restored area was seeded with three native grass species and then treated by installing erosion matting or adding supplemental rock cover. Four years later, results suggest that the seeding, along with the use of erosion matting or supplemental rock can enhance revegetation. Matting appeared to accumulate litter, and this effect might have contributed to enhanced moisture retention. Treated areas contained 40% of the vegetation cover found on adjacent controls, which averaged 69% vascular plant absolute cover. Recovery on both treatments was markedly higher than published estimates of passive revegetation of disturbed areas measured elsewhere suggesting seeding with added cover or protection led to substantial vegetative cover after four years. Two of the three seeded grass species, *Trisetum spicatum* and *Poa alpina*, dominated the restored plots, composing 81.7% of relative vegetation cover on matting sites and 73.4% of relative cover on rock supplemented areas. Presumably due to its preference for moister sites, *Deschampsia cespitosa*, had low establishment rates. Volunteer species, i.e., species that appeared on their own, contributed 6.3% to the absolute vegetation cover of matting and rock sites, and species such as *Minuartia biflora, Minuartia obtusiloba, Poa glauca* and *Festuca brachyphylla*, should be considered for use in future restorations.

Key words: road reclamation, alpine, seeding, vegetative cover

Implications for Practice:

- Natural recovery of vegetative denuded sites in dry alpine tundra ecosystems can take decades to reach even 30% cover if unaided and is often difficult to access. Thus, any improvements in the restoration process can have a large saving of time and money for land managers.
- Seeded restoration sites responded positively to two treatment types: erosion matting and supplemental rock cover suggesting anthropogenic assistance can make a difference in the timeline and success of natural processes such as vegetative recovery.
- Two of three seeded species represented the majority of the vegetative cover in restored sites supporting the concept of having diversity in seed mixes as some species will likely do better than others under different circumstances.
- Several unseeded native species exhibited recruitment to the sites and should be considered for future seed mixes. These species present a potential tool to future restoration projects as they established on their own in disturbed sites.

INTRODUCTION

Alpine landscapes represent a large economic and ecologically important ecosystem (Loomis et al. 2000, Hesseln et al. 2004, Grêt-Regamey, et al. 2008a). Alone, the recreational visitation of three of Colorado’s peaks above 14,000 feet in elevation has been estimated to bring over $1.94 million dollars in annual revenue to the state and create an estimated 42 annual jobs (Keske & Loomis 2008). Though representing 3% or less of the earth’s surface (Körner 1995) such studies indicate that the alpine ecosystems can be a significant part of economic systems. Additionally, alpine systems are of global ecological importance. Alpine ecosystems provide a supply of fresh water to many regions (Walker et al. 1993), have very high plant diversity (Körner 1995) and are a key indicator for the effects of global environmental change (Benedict 1970, Lapp et al. 2005, Neff et al. 2002, Schmidt et al. 2008, Grêt-Regamey et al. 2008b). At the same time, these important alpine systems across the globe are showing increased degradation from fragmentation and loss of diversity (Cole and Landres 1995, Urbanska and Fattorini 2000, Zabinski et, al. 2000, Hagen et al. 2014) due to a variety of factors including climate change (Theurillat & Guisan 2001) and increasing recreational use (Ebersole et al. 2002, Bay & Ebersole. 2006, Hagen et al. 2014).

The increasing disturbance of alpine habitat is especially of concern. Willard & Marr (1971) and Ebersole (2002) have shown the natural recovery of these systems can take decades or longer. For example, Ebersole (2002) reported that devegetated, 1 m^2 dry meadow plots in a Colorado, USA alpine site had only 20% of the relative vegetation cover of control plots after 13 years of recovery. Further, restoration of alpine vegetation can be challenging because areas are often difficult to access, the zone has a limited number of colonizers, and many areas have a very short growing season (Billings 1973, Chambers 1997, Rydgren et al. 2013).

A meta-analysis by Benayas et al. (2009) synthesized 89 restoration assessments finding that ecological restoration increased biodiversity by 44% and ecosystem services by 25% on average. Previous studies show that restoration, or assisted revegetation can speed up the recovery process (May et al. 1982, Bay & Ebersole 2006, Mallik & Karim 2008, Jorgenson et al. 2010, Güsewell & Klötzli 2011, Hagen et al. 2014). In these studies seeding and transplanting are two common revegetation techniques (Bayfield 1980, Behan 1983, Roach & Marchand 1984, Guillaume et al. 1986, Chambers 1997, Conlin and Ebersole 2001, Hagen et al. 2014, McDougall 2001, Ebersole et al. 2002, Rydgren et al. 2017). Transplanting, while a viable option can be time consuming and costly, especially in the alpine environment where access to sites is often limited and difficult. For this reason, seeding is often a common technique in the alpine (Hagen et al. 2014). While the current body of knowledge on alpine restoration is beginning to grow, it is largely based on research conducted on post resource extraction or mine reclamation (Chambers et al. 1987, Smyth 1997, Rieder et al. 2013, Cohen-Fernandez & Naeth 2013). These studies show the importance of restoration and the challenge of limited resources but in a limited perspective. By continuing to broaden the literature on recovery of alpine ecosystems to recreationally disturbed sites it will allow increased efficiency through optimized treatments for a variety of alpine disturbances and can allow more work to be accomplished.

The graminoid species, *Deschampsia cespitosa, Trisetum spicatum,* and *Poa alpina* were used in previous alpine restoration studies (Chambers et al. 1987, Kershaw & Kershaw 1987, Smyth 1997, Payson et al. 2005, Isselin-Nondedeu & Bédécarrats 2007, Cohen-Fernandez & Naeth 2013). *Deschampsia cespitosa* is a widespread bunch grass prevalent, with observed tillers and often dominant in moist meadows across the alpine but present in a variety of other alpine plant communities (May et al. 1982, Gehring & Linhart 1992, Walker et. al 2001, Suding et al. 2015). The rapid growth and rhizomatous tillering make it an ideal species for many restoration sites (May et al. 1982, Fattorini et al. 2001, Suding et al. 2004, Payson et al. 2005). *Deschampsia cespitosa* is a good indicator of nitrogen deposition (Farrer et al. 2013) and has been shown to create a positive feedback loop for nitrogen deposition (Bowman & Steltzer 1998) adding to its value as a species for active restoration. *Trisetum spicatum* is noted as a rapid colonizer by Harper and Kershaw (1996) and was listed as one of the top species at providing cover in multiple studies by Urbanska & Fattorini (2000) and Ebersole et al. (2002). Finally, *Poa alpina* is a tufted and moderately compact alpine grass (Isselin-Nondedeu & Bédécarrats 2007) that occurs in moist and dry meadows (Ebersole 2002).

While many factors contribute to establishment of vegetation, seed availability and the presence of microsites or microclimates have been shown as two limiting elements (Turnbull et al. 1999, Roach & Marchand 1984, Lindgren et al. 2007). Previous studies by Urbanska (1997) and Chambers (1995) suggest that microsites mitigate difficulty in early plant development by providing shelter from the alpine ecosystem’s harsh conditions and delivering essential, limited resources such as moisture. This process has been described as the “nurse effect,” where surrounding biotic or abiotic structure provides an advantage for newly establishing vegetation (Cavieres et al. 2002, 2014, Padilla & Pugnaire 2006). It is important to note that while vegetation can provide a nurse effect it can also create competition (McDougall 2001, Cavieres et al. 2002, 2014, Padilla & Pugnaire 2006, Dullinger et al. 2007, Hagen et al. 2014).

The manipulation of the abiotic environment is one proven way to accomplish revegetation goals. In a previous study the addition of rock cover surrounding installed plugs of *Deschampsia cespitosa and Trisetum spicatum* suggests that supplemental rock cover helps create microclimates and facilitate survivorship of transplants (Roberts 2012). In addition to using supplemental rock cover in the creation of microclimate environments, Burroughs and King (1989) along with other studies have used matting to aid seedling establishment in the alpine (Lewis 1995, Whitall 1995, Lavendel 2002, Ebersole et al. 2004, Krautzer et al. 2006).These techniques of using matting to aid in alpine seeding dates back at least as far as 1857 according to a review by Gorer & Harvey (1979). Matting serves to alter the microclimate for seedling establishment reducing wind and increasing seedling germination by as much as five to six times compared to seeding without matting (Ebersole et al (2002). Further, seeding with two species, including *Deschampsia cespitosa,* under erosion matting produced 400 seedlings per square meter after 2 years in a trail restoration study done by Ebersole et al. (2002), 20 to 28 times more than untreated plots. Matting has additionally been shown to reduce erosion from the splash of rain which can disrupt the establishment of new vegetation through impacting and eroding soils (Berglund 1978, Bhattacharyya et al. 2010). Erosion matting clearly increases vegetation cover and reduces erosions but is expensive and difficult to transport to remote, high-elevation sites (Carr 1975).

To understand how seeding can affect restoration on recreationally disturbed alpine sites, this study compared the vegetative cover and differences in plant community compositions four years after reclamation of alpine road under two treatments: utilization of erosion matting and use of added rock cover. We characterized differences in vegetative cover between the two treatments and explored the potential for future species of interest by comparing treated plots to the surrounding source populations. Half the treated sites were applied with erosion matting while the other half were applied with supplemental rock cover. Both applications received the same seeding rates and all installation was done over a single weekend. Each treated site was paired with an adjacent native site for comparison. The study then compared the differences between treated and native sites to answer three main questions: First, do the different treatments provide differing overall vegetative cover? Second, will the two treatments result in varied species compositions? Third, utilizing timeframes from other studies as a baseline, will these treatments and seeding increase recovery rates? Furthermore, this study sought to examine potential species for use in seeding. We predicted that matting would result in overall higher vegetative cover compared to the rock cover. However, the addition of either would result in increased revegetation compared to documented rates in previous studies of unaided restoration.

METHODS

Site Description

Within the southern Rocky Mountains of the United States of America, the study site is located within the White River National Forest, along the Continental Divide and between the Colorado towns of Breckenridge and Jefferson near Georgia Pass. The study site runs 853 linear meters from the base of the slope at 3535 m above sea level to the first major ridgeline at 3658 m in elevation (latitude and longitude 39.463506, -105.904778 to 39.468906, -105.901624). Annual precipitation for the specific site was not available, however at a nearby research site Niwot Ridge 80 km away and approximately the same elevation (3743 m) recorded 1322 mm in 2010, 1141 mm in 2011, 1161 mm in 2012, 1277 mm in 2013 and 767mm (January-August only) in 2014, 1250 mm in 2015 and 1179 mm in 2016 suggesting adequate precipitation over the study period (NWT 2019). However, the dry meadows are the study site are often wind scoured of their snow so inputs could be substantially less. The location of the study was largely homogenous in typical vegetative cover, aspect, soil moisture and disturbance. The major dissimilarity between plots was elevation which varied 123 meters from the lowest site to highest.

The study took place on a section of unsanctioned and recently closed road. This section of road ran directly up the fall-line, was heavily eroding, averaged over 2.7 meters in width, and had begun to braid into multiple paths (Figure 1). Observation notes over visits two years apart describe gullying up to 42 cm in depth, rilling and incised areas. The linear distance of the study site runs roughly south to north with an aspect ranging from 120 degrees to 142 degrees and slope grades ranging from 8% to 26%. The area is best catalogued as a dry alpine meadow as described by Komarkova (1976) and later by May et al. (1982).

Restoration Project Implementation

Success of the project and study depended heavily on ensuring future off-road vehicle travel did not continue to occur at the site. As such, it was important that the unsanctioned road which was closed parallels a second road which will remain open and maintained by the Forest Service. Thus, a large ecological gain through improved habitat connectivity and reduced erosion could be made without restricting recreational access to surrounding areas. Additionally, the project and study were performed in conjunction with installation of a fence and signage put in place to keep vehicles from driving on the restoration work and to educate the public about the project goals.

The volunteer driven restoration project took place between August 24 and September 12, 2012. A backhoe and operator prepared the site by removing larger rock cover, smoothing and decompacting the surface with a backhoe. Decompaction was achieved by using the teeth of the backhoe bucket to till the top approximately 15 cm of soil. Modest ditching with straw wattles for water runoff was added, where necessary, at 30 m intervals and some transplanting of native plugs was added to plots where data were not recorded. This process took roughly 40 hours of equipment operation time. Soon after, 66 volunteers from a local Colorado non-profit, Wildlands Restoration Volunteers, worked for two days to finish site preparation, seed the entire decommissioned road, and add either erosion matting or rock cover to each section of the project. Rock cover was added from rock disturbed during site preparation or collected outside the study site. Seeding was done at a rate of 1250 seeds per square meter. The seed mix consisted of 30% *Poa alpina*, 40% *Deschampsia cespitosa* and 30% *Trisetum spicatum* by number of pure live seed. Using a large group of volunteers allowed better control of the temporal variability when installing seed and differing restoration techniques on a large scale, ensuring consistent conditions during installation over a very large project site. No soil amendments or supplemental watering were used partially due to the extreme difficulty to get large trucks or these materials to the site. Additionally, supplemental water or soil amendments were not typically used in high elevation projects by the sponsoring company or in previous studies to be used for comparison.

At the time of this study the raw cost of material for a biodegradable erosion matting comprised of coconut husk, 33.5 m long by 2.4 m wide, was $89 US, not including installation. To save money as well as generate a test of techniques, the study design alternated supplemental rock cover and erosion matting along the elevational gradient. Erosion matting was installed covering 2.4 meters of the road width over 30.5 meter-long sections, fastened in place with metal staples. Minimal rock removed before installing the matting was also replaced to help fasten the matting in place. The alternation of treatments avoided some confounding variables such as elevational gradient, aspect, slope, and location along the project site by ensuring both matting and rock treatments were spread along the entire road closure. Supplemental rock ranging from 20 cm to 60 cm in diameter was added to the rock cover areas so that 60 or more percent of the surface was visually covered by supplemental rocks. This procedure produced 18 sections of 30.4 m lengths of matting and 12, 30.4 m lengths of rock cover (Fig S1). The 30.5 m section lengths were attributed to the matting roll length of 100 linear feet. The size of the rock has a large variance because it was harvested at the project site but typically ranged from approximately 10 square cm to 30 square cm.

Cover Sampling

Point-intercept techniques were used to collect vegetative cover and species composition. Collection occurred from July 20 to July 26, 2016. The timing of collecting data was chosen because it provided the ability to identify many early season species before senescence while still being able to identify late season species still in and early growth phase. Within each 30.5 m section a point was selected roughly two meters from the bottom of the section end. This point became the marker for placing a 1 m^2 plot. In total, 30 treatment plots (18 matting, 12 rock cover) were chosen spanning the study site. At each plot a paired sample of native vegetation was taken 20 meters from the edge of the restored road. The east or west side of the restored road was randomly selected for this paired sample. The meter squared plots of the restored road and paired sample were divided into 100 points. At each point, using point-intercept methods, the species or substrate hit by a vertical pin placed from above was recorded. Species were recorded to the lowest taxonomic level possible in the field and samples were taken of any unknown plants to be identified later in the lab. At each plot the slope, aspect, soil moisture, date, recorder, observer, site number, restoration technique, species at point intercept, and additional species were recorded. Soil moisture was collected using a Rapitest, Moisture Meter™ once during this collection period and did not provide precise measurements but did allow a generalized comparison of the sites soil moisture content. We used first hit (100 records per plot) to quantify absolute vegetation, rock, and litter cover. We used total hits (multiple plant species found beneath a single point) to calculate relative vegetation cover.

Plant cover and plant species responses on reference plots, erosion matting, and rock-supplemented plots were summarized using SAS 9.4 (SAS 2019) programs. Cover characteristics were compared using a one-way ANOVA, with a post-hoc SNK test used when statistically significant differences were found among treatments. While additional explorations of community composition were undertaken, only summary findings are reported here, and a larger suite of analyses and all data are available in Roberts (2018).

RESULTS

The commercially purchased seed mixture was tested by the supplier to provide 1292 pure live seeds per meter, adequate for ample vegetative coverage under ideal circumstances.

In total 55 species were identified using point-estimate methods with an overall mean of 10.3 species per square meter. Five species not scored as ‘hits’ but seen in quadrats were also observed. The mean number of species for the rock cover treatments was 4.6, the mean for erosion matting was 4.2 species and the mean for the native plots was 16.1 species.

Total vascular vegetation was not statistically different between matting and rock cover plots and was lower in both than in native, undisturbed plots (Table 1). Further analysis shows a statistical difference in the combine performance of the three seeded species with the highest cover in the matting plots (Table 1). Despite ample seeding rates, *Deschampsia cespitosa* was less prevalent in restored matting and rock cover plots than in the native surrounding plots. The other two seeded species, *Trisetum spicatum and Poa alpina* however were significantly more prevalent in the matting than in the native plots. *Trisetum spicatum* exhibited a statistically higher presence in the matting plots than rock cover or the native plots making it the only of the three seeded species to statistically different cover in all three plot types (Table 2).

Non-vascular plant cover information was only obtained if vascular plants were not encountered first and therefore do not reflect absolute amounts on the plots. Given this limitation, litter had statistically higher mean counts in matting plots than rock cover or native plots (Table 2). Additionally, results suggest that bare ground was more prevalent in the rock cover plots (46.2%) than either matting (20.8%) or reference plots (14.9% cover; p<.0001).

Within the undisturbed reference plots, there were eight species that had greater than 5% relative cover (Table 1) and 21 species that comprised over 1% relative cover. The matting and rock cover treatments comparatively had three species with over 5% relative cover including two of the seeded species, *Trisetum spicatum* and *Poa alpina.* There were six species volunteer (non-seeded) species with relative cover above 1% in the matting treatment and four species in the rock cover treatments (Table 1). The percent of absolute cover from volunteer species did not differ between matting and rock cover treatments and was 6.3%.

Discussion

This study sought to examine the benefits of using erosion matting and rock cover as restoration treatments of a road obliteration as measured by relative and absolute vegetative cover. Comparing effectiveness with restoration goals such as creating specific plant communities or maximizing cover will better inform future restoration projects on the value of these treatments. Our findings addressed both the relative benefits of proactive restoration and compared a reduced budget (rock addition) to a preferred (erosion matting) technique to reduce erosion and enhance seedling establishment.

Our results suggest that the addition of matting or supplemental rock can lead to substantial vegetative growth in disturbed sites. This trend is consistent though less pronounced than a study by Ebersole et al. (2002) that looked at restoration of smaller disturbed trails and found matting provided 5 to 6 times more seedling establishment 2 years after restoration. Supplementing natural processes with matting appears to correlate with increased vegetative cover over a short period of time in other studies and to considerable vegetative growth in this study. In previous studies by Chambers (1993) and Ebersole (2002) restored sites that received treatments provided higher cover than plots left untreated, leading to the conclusion that the effort required to perform restoration may be worthwhile to reach adequate levels for vegetative cover.

Trisetum spicatum was the most successful seeded species in our restoration. At the well-studied Niwot Ridge alpine site, approximately 80 km north of our study, this species is also common and one of the few to be as abundant in disturbed, unseeded restoration sites as in adjacent controls (Ebersole 2002). Both this species and the combined *Poa* species, *P. alpina* and *P. glauca,* have been shown to be strong responders to increased nitrogen and phosphorus abundance at the Niwot Ridge site (Theodose and Bowman 1998; Seastedt, unpublished results). This response criteria might suggest that these species are likely good candidates for restoration of disturbed alpine areas, but perhaps do not need to be a large percentage of the seed mix to prevent heavy dominance by a single species.

Findings for *Poa alpina* were consistent with previous studies by Ebersole (2002), where the species was noted as only occurring in disturbed plots of dry meadows. In this study *Poa alpina* accounted for only 0.1% of relative cover undisturbed, native plots but was 19.0% of the relative cover in matting plots and 28.8% of cover in rock cover plots. This suggests *Poa alpina* is a good species for increasing vegetative cover on disturbed, dry meadow, alpine plots. However, depending on the goals of the restoration project, *Poa alpina*'s ability to establish may also be a sign of out competing other species which could lower overall diversity.

Deschampsia cespitosa is a species which, along with other grasses, has been shown to dominate the reclamation of disturbed sites in alpine ecosystems (Chambers et al. 1984). Despite being present in native plots and accounting for 40% of the seed mix, *Deschampsia cespitosa* was largely absent in the treated plot types. While present in the native plots at 6.0% relative cover, *Deschampsia cespitosa* is more likely to dominate systems with higher moisture content (Chambers et al. 1984). The high relative cover of *Carex rupestris* at 9.0% and *Kobresia myosuroides* at 5.8% in the native plots, support the conclusion that the study site was characteristic of a dry windblown meadow (Walker et al. 1994) which could help explain the lack of *Deschampsia cespitosa* in restored plots.

This study also identified which of the locally abundant species might be used in future seed mixes. Among the non-seeded species that composed about 6% of the total absolute vegetation cover in treated plots, *Festuca brachyphyla, Poa glauca, Cerastium arvense,* and two species of *Minuarta* emerged as potentially useful species in subsequent restorations making up at least 0.5% of absolute cover in restored plots (Table 1).

One important question that arises from the dominance of seeded species in restoration is what the long-term effect will be on species composition. Previous studies have considered other possibilities to the notion of succession in the alpine (Schmidt 2008, Smyth 1997), noting that many species act as both colonizers and components of established vegetative systems. With seed species making up 71-83% of the restored plots in this study, future studies should consider looking at the lasting effect of using a limited seed mix that includes species capable of potentially out-competing native recruited species. Future studies should also consider comparing the effect of a more diverse seed mix to mitigate for the dominance of a few species, as well as to evaluate its impact on absolute vegetation cover.

Vegetative recovery in the alpine ecosystem was found by Willard & Marr (1971) to be a very slow process across Colorado’s Front Range. The observed absolute vegetative cover in the present study for matting plots at 35% and rock cover plots at 26% after four years were much higher than absolute cover on plots in Ebersole’s (2002) study looking at disturbed sites left untreated at Niwot Ridge. Ebersole’s (2002) study, showed that after 13 years untreated plots had an absolute cover of 14% (±6.8%). The same study (Ebersole 2002) showed 58% cover (±38.9%) after 30 years. While not directly comparable, the recovery of vegetative cover in 4 years at Georgia Pass was similar to natural recovery over 13 years at Niwot Ridge. Similarly, in Chambers’ (1993) study untreated plots on the Beartooth Plateau, Montana had vegetative cover of 25% after 35 years. Again, less than assisted recovery in this study.

The potential increased rate of recovery due to the two treatments could save decades compared to natural recovery rates. Additionally, the restored plots were placed in the middle of a much wider road, removing the effect of edge expansion which can enhance restoration recovery measurements. In Ebersole's (2002) study, edge expansion was thought to be a large contributing factor to increasing vegetative cover. This makes the present recovery rate potentially even more significant in jump-starting the restoration process. These findings are promising for future restoration efforts but is it important to acknowledge the site-specific nature of the current research.

With increasing environmental and recreational pressure on alpine ecosystems and very slow natural recovery management, agencies should prevent unnecessary disturbances that will require restoration efforts. Such efforts are costly to repair and even with the increased recovery rates seen in the present study and others, full recovery will take decades, if not longer (Colin & Ebersole 2001). Leveraging volunteers can reduce costs and build capacity for future restoration projects but most importantly, engaging volunteers can create buy-in from local user groups and further develop a sense of stewardship for public lands. If loss of plant cover does occur on slopes and especially in drier communities where they are more likely to occur (Colin & Ebersole 2001), then restoration efforts such as seeding may be aided by rock cover and matting to expedite recovery.

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Table 1. Raw cover, relative cover and absolute cover of plant species on restoration matting or rock addition areas of an obliterated road in the Colorado alpine.

	Rock Cover			**Erosion Matting**			**Undisturbed**		
Plot Type	Count	Relative Cover (%)	Absolute Cover (%)	Count	Relative Cover (%)	Absolute Cover (%)	Count	Relative Cover (%)	Absolute Cover (%)
Bare Ground	**554**		46.2	**374**		20.8	**446**		15.7
Litter	**74**		6.2	**508**		28.3	**155**		5.4
Moss	**3**	n/a	0.3	**2**	n/a	0.1	**43**	n/a	1.5
Rock	**249**		20.8	**281**		15.6	**114**		4.0
Lichen	**0**		0.0	**0**		0.0	**23**		0.8
Species									
Trisetum spicatum	**143**	44.7	11.9	**397**	62.7	22.1	**60**	2.9	2.1
Poa alpina	**92**	28.8	7.7	**120**	19.0	6.7	**2**	0.1	0.1
Festuca brachyphylla	**19**	5.9	1.6	**37**	5.9	2.1	**167**	8.1	5.9
Poa glauca	**11**	3.4	0.9	**20**	3.2	1.1	**25**	1.2	0.9
Deschampsia cespitosa	**10**	3.1	0.8	**3**	0.5	0.2	**167**	8.1	5.9
Minuartia biflora	**9**	2.8	0.8	**4**	0.6	0.2	**3**	0.2	0.1
Minuartia obtusiloba	**7**	2.2	0.6	**8**	1.3	0.4	**46**	2.2	1.6
Cerastium arvense	**7**	2.2	0.6	**3**	0.5	0.2	**38**	1.8	1.3
Oreoxis alpina	**4**	1.3	0.3	**1**	0.2	0.1	**120**	5.8	4.2
Luzula spicata	**3**	0.9	0.3	**24**	3.8	1.3	**23**	1.1	0.8
Artemisia scopulorum	**3**	0.9	0.3	**0**	0.0	0.0	**149**	7.2	5.2
Arenaria fendleri	**2**	0.6	0.2	**5**	0.8	0.3	**161**	7.8	5.7
Phacelia sericea	**2**	0.6	0.2	**3**	0.5	0.2	**4**	0.2	0.1
Draba aurea	**2**	0.6	0.2	**1**	0.2	0.1	**1**	0.1	0.0
Androsace septentrionalis	**2**	0.6	0.2	**0**	0.0	0.0	**0**	0.0	0.0
Sedum lanceolatum	**1**	0.3	0.1	**0**	0.0	0.0	**31**	1.5	1.1
Polygonum bistortoides	**1**	0.3	0.1	**0**	0.0	0.0	**12**	0.6	0.4
Trifolium nanum	**1**	0.3	0.1	**0**	0.0	0.0	**10**	0.5	0.3
Draba streptocarpa	**1**	0.3	0.1	**0**	0.0	0.0	**0**	0.0	0.0
Lloydia serotina	**0**	0.0	0.0	**3**	0.5	0.2	**41**	2.0	1.4
Geum rossii	**0**	0.0	0.0	**1**	0.1	0.1	**32**	1.6	1.1
Heterotheca pumila	**0**	0.0	0.0	**1**	0.1	0.1	**25**	1.2	0.8
Elymus scribneri	**0**	0.0	0.0	**1**	0.1	0.1	**9**	0.4	0.3
Agoseris glauca	**0**	0.0	0.0	**1**	0.1	0.1	**1**	0.1	0.0
Carex rupestris	**0**	0.0	0.0	**0**	0.0	0.0	**257**	12.4	9.0
Kobresia myosuroides	**0**	0.0	0.0	**0**	0.0	0.0	**164**	7.9	5.8
Trifolium dasyphyllum	**0**	0.0	0.0	**0**	0.0	0.0	**129**	6.2	4.5

Trifolium parryi	**0**	0.0	0.0	**0**	0.0	0.0	**88**	4.3	3.1
Campanula rotundifolia	**0**	0.0	0.0	**0**	0.0	0.0	**70**	3.4	2.5
Artemisia pattersonii	**0**	0.0	0.0	**0**	0.0	0.0	**68**	3.3	2.4
Selaginella densa	**0**	0.0	0.0	**0**	0.0	0.0	**34**	1.6	1.2
Calamagrostis purpurascens	**0**	0.0	0.0	**0**	0.0	0.0	**32**	1.6	1.1
Sibbaldia procumbens	**0**	0.0	0.0	**0**	0.0	0.0	**21**	1.0	0.7
Other Species (<1 Individually)	**0**	0.0	0.0	**0**	0.0	0.0	**78**	3.8	2.7

Table 2. Absolute cover of restored alpine areas found on erosion matting, and rock-supplemented sites versus that of undisturbed alpine tundra. Values are means, ± std errors. Means followed by different letters are significantly different (P<.05) using a post-hoc SNK test.

	Undisturbed Native Plots (n=30)	Erosion Matting Plots (n=18)	Supplemental Rock Cover Plots (n=12)
Total vascular vegetation	68.9 (3.4) A	35.2 (2.8) B	26.7 (2.8) B
Litter	5.2 (1.0) B	28.2 (2.4) A	6.2 (1.2) B
All seeded species	7.6 (2.3) C	28.9 (2.0) A	20.4 (3.1) B
Deschampsia cespitosa	5.6 (2.4) A	0.2 (0.2) B	0.8 (0.5) B
Trisetum spicatum	2.0 (0.5) C	22.1 (2.5) A	11.9 (2.4) B
Poa alpina	0.1 (0.1) B	6.7 (1.2) A	7.7 (1.8) B
Volunteer vascular vegetation	n/a	6.3 (1.6) A	6.3 (2.5) A

Figure Legends:

Figure 1. Road in 2011, before (A) and after restoration (B) in 2014 In addition to the seeding effort, vegetation plugs, seen here as patches of vegetation, were inserted into flatter, less rocky portions of the former road.

Figure 2. Example of erosion matting (A) and supplemental rocks (B) added to the former road area.

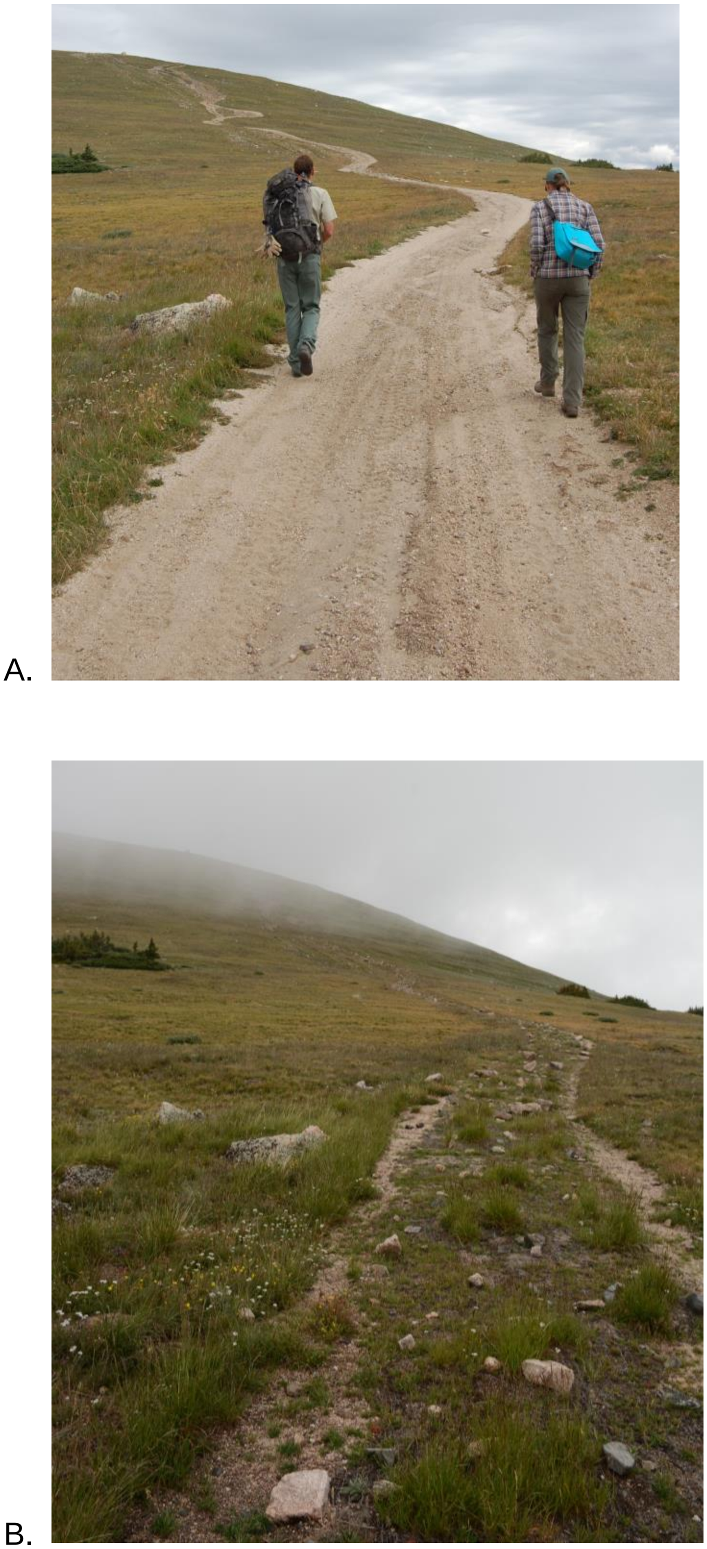

Fig. 1

Fig 2.