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Development of a Post-Fire Monitoring Protocol for Evaluating Treatment Effectiveness and Cheatgrass Abundance Using Quickbird Imagery and Ground Observations

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DEVELOPMENT OF A POST-FIRE MONITORING PROTOCOL FOR EVALUATING TREATMENT EFFECTIVENESS AND CHEATGRASS ABUNDANCE USING QUICKBIRD IMAGERY AND GROUND OBSERVATIONS

by

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ABSTRACT

Development of a Post-Fire Monitoring Protocol for Evaluating Treatment Effectiveness and Cheatgrass Abundance Using Quickbird Imagery and Ground Observations

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The Bureau of Land Management (BLM) manages 9.3 million hectares of land in Utah and has implemented an Emergency Stabilization and Rehabilitation (ESR) Program to protect life and property, combat soil erosion, and reduce the invasion of exotic/noxious weeds following wildland fire. In highly vulnerable sites, seeding treatments may be applied to establish an interim landcover to stabilize the soil and competitively exclude weed invasions. Monitoring treatment effectiveness is mandated through ESR guidelines and necessary for the submission of annual Accomplishment Reports for the first three years following fire containment. Ground monitoring has been the traditional approach to fulfilling this ESR monitoring mandate.

Ground monitoring of vegetation within a large burn can be complicated or rendered infeasible by the logistical constraints presented by size, topography, and remoteness. The inherent weaknesses of ground monitoring in large remote areas provide the impetus for augmenting these approaches with remotely sensed data. The Rattle Fire Complex (RFC) is a 2002 burn that demonstrates a need and an opportunity to develop a remote sensing-based monitoring tool.

This project utilized high spatial resolution Quickbird imagery and ground data to monitor treatment effectiveness and vegetative recovery within the RFC ESR project area and shows that remote sensing and statistical modeling can significantly improve knowledge regarding ESR treatment effectiveness when combined with traditional ground monitoring methods. The image acquisition cost and labor investment may be prohibitive, making this approach feasible only on large, high priority projects. This methodology arguably represents the simplest approach from both a remote sensing and statistical modeling approach and was accomplished using software currently available within the Bureau of Land Management computer network. It is unlikely that current technology can provide a cheaper or simpler alternative. Testing of this methodology on other projects will provide better insight into its utility and transferability.

(98 pages)

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CONTENTS

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION

The Bureau of Land Management (BLM) manages 9.3 million hectares of land in Utah and has implemented an Emergency Stabilization and Rehabilitation (ESR) Program to protect life and property, combat soil erosion, and reduce the invasion of exotic/noxious weeds following wildland fire. In cases of high complexity fires, the Burned Area Emergency Response (BAER) Team is called in to make specific rehabilitation recommendations using the general ESR guidelines, described as follows. In highly vulnerable sites seeding treatments may be applied to establish an interim landcover that may stabilize the soil and competitively exclude weed invasions. Monitoring treatment effectiveness is mandated through ESR guidelines and necessary for the submission of annual Accomplishment Reports for the first three years following fire containment. Cooperative efforts in monitoring and dissemination of the results of ESR projects are encouraged (Interagency BAER Handbook 2002). Ground monitoring has been the traditional approach to fulfilling this ESR monitoring mandate.

Ground monitoring of vegetation within a large burn can be complicated or rendered infeasible by the logistical constraints presented by size, topography, and remoteness. Acquiring ground data over a large area is often impossible, forcing interpolation and extrapolation from small sample sizes. Additionally, ground monitoring is usually restricted to one annual visit which may not best capture the true vegetative recovery. Comparisons of vegetation data collected at different growth stages may provide spurious conclusions regarding vegetative condition. The inherent weaknesses of ground monitoring in large remote areas provide the impetus for augmenting these approaches with remotely sensed data.

The Rattle Fire Complex (RFC) is a 2002 burn that demonstrates a need and an opportunity to develop a remote monitoring tool. The RFC is located in an extremely remote area of the Book Cliffs, making overland access difficult and sometimes dangerous in wet conditions (personal experience, 2003-2006). Elevation ranges from 1,768-2,850 m with slopes reaching 80%. The RFC contains 38,251 ha of burned vegetation with approximately 10,702 treated hectares under the ESR monitoring mandate.

The goal of this project is to develop a method for monitoring seeding treatment effectiveness on large and remote rehabilitation projects by integrating:

- Traditional ground monitoring
- Remote sensing
- GIS

This project explored the applicability of high resolution remotely sensed data to ESR monitoring goals. Emphasis was given to determining treatment effectiveness by mapping vegetative cover, bare soil, and monitoring the post-fire expansion of cheatgrass (*Bromus tectorum*).

BACKGROUND

Cheatgrass & Fire

Wildland fires are widely viewed as natural components of healthy ecosystems in the Intermountain West. However, the influence and affect of wildland fires within semi-arid rangelands has changed in the context of exotic and noxious weed invasion, changes in plant community structure, climatic variability, and consistent land-use. Fires provide a niche for the invasion of exotic and noxious weeds and increase the potential for soil erosion. Fire frequency on Utah rangelands has increased with the invasion of cheatgrass (Pellant 1996). Post-fire conditions typically favor the establishment of annual cheatgrass over native perennials creating a self-perpetuating cycle of fire, erosion, and further weed infestation. The invasion of exotic and noxious weeds, including cheatgrass, poses a major threat to ecosystem structure, function, and biodiversity of these dry ecosystems (Mooney and Cleland 2001). Increases in the rate of soil erosion can cause serious and irreparable ecological damage including decreases in site productivity, downstream sedimentation of streams and rivers (Pritchett and Fisher 1987), permanent habitat loss and desertification. Fire suppression, rehabilitation and post-fire monitoring of these lands are often necessary to avoid serious land degradation.

Cheatgrass, in the semi-arid western United States, is the most prolific and successful annual grass introduced from Eurasia (Hurlbert 1955) and is believed to have reached its current distribution by 1930 (Mack 1981). Cheatgrass invasion is directly and indirectly responsible for the decline of native species in shrublands and perennial grasslands through direct competition for resources and by shortening fire return intervals

(Pellant 1996). Additionally, cheatgrass costs federal and state agencies a significant amount of their annual budget in increased fire suppression and ESR costs due to more frequent fires although exact figures are unknown.

Cheatgrass is a winter annual that can germinate in both the fall or spring given the right climatic conditions (Mack and Pyke 1983). Additionally, cheatgrass exhibits rapid elongation of the roots (Harris 1967) and is a prolific seed producer (Hurlbert 1955). These characteristics in conjunction with fall germination provides a competitive advantage over native perennial species (Harris 1967). Cheatgrass is rapidly growing when native perennials are initiating growth allowing cheatgrass to monopolize soil water and nutrients (Pellant 1996).

Cheatgrass invasion has reduced the fire return interval relative to pre-settlement conditions causing more frequent fires (Billings 1948). As mentioned, cheatgrass greenup phenology occurs earlier than most native perennials but senescence also occurs earlier. Cheatgrass is typically flammable four to six weeks earlier and remains flammable one to two months longer than most perennial perennials (Platt and Jackman 1946). Perennial grass, forb and shrub species decrease with each fire and the recovery time to reach pre-fire conditions is longer (Pellant 1996) although the available recovery window is shorter. In this manner, by altering the fire return interval and increasing perennial mortality cheatgrass indirectly affects the survivorship of perennial species and gains a competitive advantage through reduced inter-species competition for resources.

Emergency Stabilization and Rehabilitation

The ESR program initiated by the Department of Agriculture and the Department of the Interior is differentiated into stabilization and rehabilitation sub-programs. Emergency stabilization is defined as “planned actions to stabilize and prevent unacceptable impacts to natural and cultural resources, to minimize threats to life and property resulting from the effects of a fire, or to repair/replace/construct physical improvements necessary to prevent degradation of land or resources” (Shepard 2007). Rehabilitation is defined as “efforts undertaken within three years of containment of wildland fire to repair or improve fire-damaged lands unlikely to recover naturally to management approved conditions, or to repair or replace minor facilities damaged by fire” (Shepard 2007).

ESR treatments are designed to minimize the effects of wildfire by reducing (Juenger 2005):

- Loss of additional watershed cover (vegetation)
- Loss of soil and on-site productivity
- Loss of water control and deterioration of water quality
- Damage to property on and off site
- Invasion of burned areas by highly flammable plants (annuals)
- Invasion of noxious weeds
- Destruction of wildlife habitats
- Post-fire erosion to cultural remains

The seeding treatments initiated in the RFC are directly related to increasing vegetative cover, minimizing the loss of soil, minimizing the invasion of noxious weeds, and protecting against damage to private property.

Ground Monitoring of Vegetation

Ground monitoring of vegetation has been the traditional method of evaluating ESR treatment effectiveness and provides a quantitative species specific examination of vegetative condition. A variety of methodologies have been developed and utilized over the past century. Common metrics include: (1) cover, (2) nested frequency, and (3) density.

The cover metric quantifies the percentage of the soil surface covered by either aerial or basal vegetative cover from a vertical perspective. Cover is the metric most directly related to the biomass of the plant and provides a good estimation of plant composition (Elzinga et al. 1998). Cover is arguably the metric most useful in developing relationships with remotely sensed data because it is what the sensor detects from above. Also, cover does not require identification of what constitutes an individual, as does density. One disadvantage of cover measures is that the values may change dramatically over the course of a growing season while density and frequency remain more static. This can present problems for large areas where sampling may take several weeks. Additionally, determining whether decreases in cover are related to mortality or decreased vigor due to climatic conditions can be difficult with the cover metric alone. Arguably, the most important disadvantage to cover is that it does not adequately sample species with low cover (Walker 1970, Bonham 1989, Elzinga et al. 1998), an occurrence common following ESR seeding treatments.

Frequency of a plant species is the probability of finding the species when a particular size of quadrat is randomly located within the project area (Bonham 1989) and is therefore influenced by the size and shape of the sample unit (Bonham 1989, BLM 1996). Frequency measurements are the easiest, simplest and fastest vegetative monitoring method (BLM 1996). Rhizomatous species and weed invasions are often measured using frequency because the identification of individuals is not necessary only their presence within the plot (Bonham 1989, Elzinga et al. 1998). The primary advantage to frequency methods is that the observer need only decide the presence or absence of the species within the plot. The primary disadvantage is that changes in frequency can be difficult to interpret as they may result from decreases in density or changes in the plant's distribution (Bonham 1989, Elzinga et al. 1998). In the case of ESR projects, the combination of cover and frequency is a powerful dataset because vegetation with low cover and high frequency often indicates that plants are small but well distributed within the sampling area.

Density is defined as the number of individuals per unit area (Bonham 1989) and is calculated by counting the number of plants occurring within a quadrat of known area. The use of density allows direct comparison with other sites that may be using different quadrat sizes (Elzinga et al. 1998). Density is most sensitive to changes in vegetation relating to the mortality or recruitment of individual plants (Elzinga et al. 1998). However, the ability to recognize, define, and count individuals is central to this metric. This may be difficult with rhizomatous plants (BLM 1996) and even with bunchgrasses that break apart (Bonham 1989) making density a poor choice for these species (Herrick et al. 2005).

Remote Sensing of Vegetation

Remote sensing techniques provide a potential means of detecting, quantifying, mapping and monitoring changes in vegetation on local, regional, and global scales (Leprieur et al. 2000) as well as at monthly, weekly, or daily time intervals. The increased spatial coverage provides monitoring capabilities over a large burn or treatment area. The temporal resolution achievable with remotely sensed data creates the opportunity to monitor vegetation throughout a growing season or during different vegetative conditions. Vegetation condition is often quantified using common spectrally derived vegetation indices.

Vegetation indices use spectral band ratio techniques to enhance the visualization and analysis of vegetation based on the unique spectral characteristics of vegetation. Healthy green vegetation reflects approximately 40-50% of the incident near-infrared (NIR) energy while 80-90% of the incident visible energy is absorbed by chlorophyll for photosynthesis (Jensen 1996). Senescent vegetation reflects less in the NIR and more in the visible wavelengths. Specific spectral measurements are combined to form ratios that are well correlated with green leaf area, standing biomass, fractional cover, photosynthetic activity, and productivity (Baret and Guyot 1991). The Normalized Difference Vegetation Index (NDVI) (Rouse et al. 1973) is one of the most commonly used vegetation indices and is often used in areas of lower canopy cover as it becomes asymptotic at higher Leaf Area Index (LAI) values.

Temporal Aspects of Vegetation & Multi-temporal Data Analysis

Studies that monitor or analyze the biological cycles (e.g. growing season) of plants and their connection to climate are called *phenological* studies (Berube 1982). Phenophases are specific stages within a growing season like budburst or flowering. The phenological foci of this study are the unique green-up and senescence patterns of cheatgrass which was used to differentiate it from other landcover types for evaluating and mapping treatment effectiveness and vegetation recovery.

In this application, remotely sensed imagery was used to monitor cool and warm season vegetative growth patterns and has several advantages over traditional ground monitoring including: (1) the capability to delineate between cool and warm season herbaceous vegetation based upon their spectral signatures at unique phenological stages over the entire project area; and (2) the spatial and temporal variability of emergence and establishment may be mapped over large aerial extents.

Remote phenological monitoring and vegetative delineation has scientific precedent. Phenological characteristics of plant species or landcover types can be differentiated by studying their spectral and temporal signatures (Dall'Olmo and Karnieli 2002). For example, vegetation-cover classes have been separated in multi-temporal space according to their phenological variations using 1 km AVHRR (Advanced Very High Resolution Radiometer) NDVI scenes (Justice et al. 1985, Tucker 1985, 1986, Ehrlich et al. 1994, Hobbs 1995, Schmidt and Karnieli 2002). The differences in coarse resolution phenological patterns of C_3 shrub and C_4 grasses have been used to delineate vegetative growth forms at the community level (Peters et al. 1997). There is promising evidence that an NDVI time-series generated from remotely sensed data is capable of phenologically delineating semi-arid vegetation including biological soil crusts (Schmidt and Karnieli 2002). Therefore, vegetative growth forms and in some cases individual species may be delineated using a time series of fine resolution multispectral imagery.

Mapping Invasive and Noxious Weeds Using Remote Sensing

Few studies, if any, have attempted to utilize remote sensing techniques to monitor treatment effectiveness on relatively small scale rehabilitation projects. A handful of studies have attempted to map cheatgrass on a regional scale using coarse or moderate resolution imagery such as Landsat Enhanced Thematic Mapper (ETM+) and the AVHRR. Continuous cheatgrass cover was modeled using tobit regression and the seasonal change in the Normalized Difference Vegetation (ΔNDVI) derived from 30 m Landsat ETM+ data across 13.3 million has in Nevada (Peterson 2005). Another study used a time series of Landsat ETM+ and AVHRR derived ΔNDVI during regionally wet and dry years to delineate cheatgrass (Bradley and Mustard 2005). They used the high interannual variability of cheatgrass growth as expressed in the ΔNDVI to differentiate it from other landcover types. However, satellite data with 20, 30, and 80 m spatial resolution often cannot detect small weed infestations or infestations occurring within mixed vegetation (Lass et al. 2005).

The Fire and Study Area

The RFC started on June 20, 2002 from multiple lightning strikes and resulted in 38,251 ha of burned landscape (BAER Report Rattle Fire Complex 2002). The RFC includes the Diamond Creek Fire (35,753 ha) and the Black Canyon Fire (2,498 ha). The fire crossed multiple administrative boundaries including:

- Bureau of Land Management - Moab Field Office
- Bureau of Indian Affairs – Uintah and Ouray Agency
- State of Utah – School Institutional Trust Lands Administration
- State of Utah – Division of Wildlife Resources
- Private property

The RFC is located in the Book Cliffs north and northwest of the town of Thompson Springs, Utah and approximately 10 km east of the Uintah and Ouray Indian Reservation (Fig. 1). The approximate center of the study area is located at latitude 39°13’14.84 and longitude 109°32’13.74. The RFC is regionally located within the Utah portion of the Colorado Plateau physiographic province on the Tavaputs Plateau, a Cretaceous and Tertiary period formation which spans much of eastern Utah and western Colorado (McNab and Avers 1994). A geomorphological examination shows that these deposits rise gradually southward and upward from the center of the Uinta Basin. The plateau continues to rise until it reaches elevations between 2,438 and 3,048 m and monolithic erosional cliffs (McNab and Avers 1994). The Book Cliffs, cut primarily from marine Cretaceous sandstone, form the southwestern and southern terminus of the Tavaputs plateau and mark the transition into the valleys of Carbon, Emery, and Grand counties.

The Book Cliffs begin near Helper, Utah located in Carbon County and initially extend eastward making a smooth arc southward to Green River, Utah. The cliffs change direction at Green River extending eastward, paralleling I-70, eventually arcing northeast toward Colorado. The portion of the RFC burn perimeter intersecting the Diamond and Cottonwood watersheds defines the study area for this project (Fig. 1).

The majority of the burned area occurring on BLM land was located within the Diamond and Cottonwood watersheds. These watersheds were designated as Wilderness Study Areas (WSA) and the BLM retired grazing permits in the mid 1990s. However, many years of preferential grazing of grasses and forbs by livestock and wildlife resulted in bottomlands primarily filled with tall decadent Basin Big Sagebrush (*Artemisia tridentata*) and rabbitbrush (*Chrysothamnus spp.*) shrublands with a cheatgrass understory. A series of beaver dams created a marshy riparian zone in some areas which included various willows (*Salix spp.*), Box Elder (*Acer negundo*), Fremont Cottonwood (*Populus fremontii*), and riparian grasses and grasslike species. Uplands consisted of Pinyon-Juniper (*Pinus edulis* & *Juniperus osteosperma*), Gambel Oak (*Quercus gambelii*), Douglas Fir (*Pseudotsuga menziesii*), Aspen (*Populus tremuloides*), and some Ponderosa Pine (*Pinus ponderosa*) communities. Quantitative pre-fire data is generally lacking and this information is compiled from anecdotal information, photographs, and ground reconnaissance.

The upland and bottomland vegetation in the middle and upper reaches of both watersheds were almost completely consumed by the moderate to high burn severities of the fire. Remnant beaver ponds and riparian vegetation were scoured away by the intense

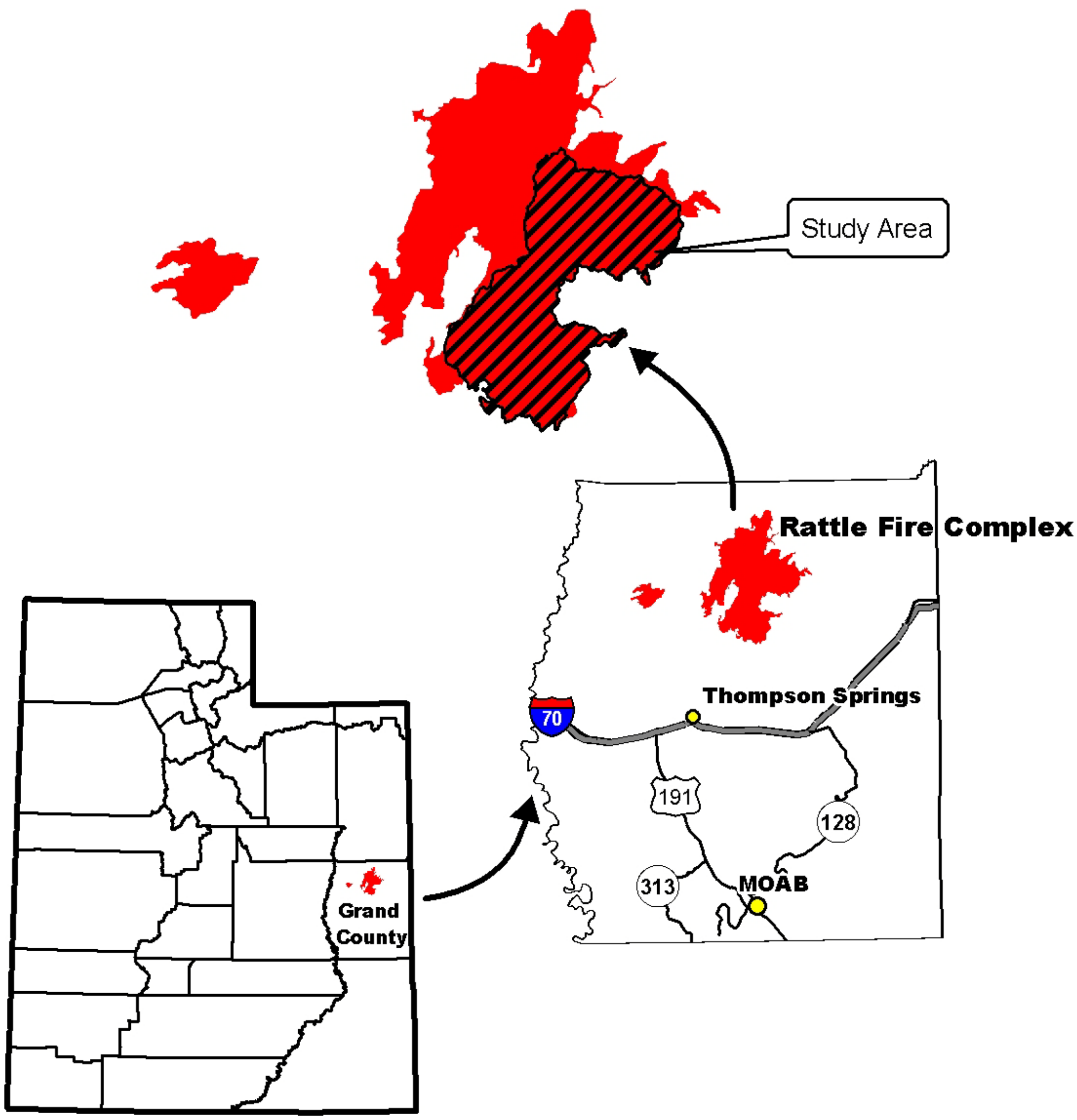

Figure 1. Study area. RFC shown in red and study area with cross-hatching.

Figure 2. Channel incision in Diamond Canyon.

flow resulting from the loss of vegetative cover and litter (A. Aubry, BLM Hydrologist, personal communications, 2005). Stream channels have become deeply incised (Fig. 2) and floodplains have expanded in some areas causing annual scouring of floodplain vegetation. These extreme hydrologic cutting and filling events appear to be the natural processes responsible for carving the Book Cliffs into their present physiographic condition but are problematic with the potential for weed invasion and the risk to private property.

ESR Treatments

In order to stabilize the watersheds and slow the invasion of cheatgrass, the BAER Team made recommendations to treat areas of moderate to high burn severity occurring on slopes less than 60%. An aerial seed treatment was applied to upland and bottomland areas on BLM lands within the Cottonwood and Diamond watersheds in fall 2002 (Fig. 3). The 10,702 ha treatment consisted of 88,904 kg of seed composed of seven species of native grasses, forbs, and shrubs and will be termed Treatment 1 hereafter (Appendix A).

Treatment 2 consisted of a 567 ha follow-up aerial seed treatment that was applied to areas of both the Cottonwood and Diamond watersheds in the fall of 2003 overlapping the bottomland portions of Treatment 1 (Fig. 3). The seed mix included 12 species of native grasses, forbs, and shrubs that were treated with a ballistics coating designed to minimize drift and increase penetration into the soil by increasing seed weight (Appendix A). The treatment consisted of 3,624 kg of seed but this weight increased to 24,434 kg after the ballistics coating was added. An endomycorrhizal seed coating was applied to six selected species (Appendix A) in an attempt to give the seeded species a competitive advantage over cheatgrass and increase the soil stabilization potential by increasing plant establishment.

Mycorrhizae is a term that describes a mutualistic symbiotic relationship that occurs between the roots of some plants and fungi (Allen 1991). In this relationship the fungi obtain a steady supply of carbohydrates from the plant and the plant utilizes the large surface area of the fungal mycelium to absorb water and nutrients from the soil (Allen 1991). The mycorrhizal inoculum contained 150 propagules per gram of arbuscular mycorrhizal fungal spores, colonized root pieces and fungal mycelium of *Glomus* species. *Glomus* species included *Glomus intraradices* (50%), *Glomus aggregatum* (25%), and *Glomus mosseae* (25%).

A third treatment of hydromulch was also applied to 263 ha of the bottomland areas of both the Diamond and Cottonwood watersheds in the fall of 2003 overlapping portions of Treatment 1 and Treatment 2 (Fig. 3). The mulch treatment was applied in an attempt to stabilize both the soil and the seed on site thus minimizing the loss of seed from erosion or predation. The mulch was also intended to decrease the hydrophobicity of the soil by increasing the surface organic layer. The treatment consisted of 530,703 kilograms of thermo-mechanically refined virgin wood fiber mulch with 53,070 kilograms of guar tackifier.

There is essentially one upland treatment and two bottomland treatments that are derived from the overlap of all three treatments:

Treatment 1 (upland seeding):

- Aerial seeding fall 2002 not overlapping Treatment 1 and Treatment 2

Treatment 2 (bottomland mycorrhizal seeding):

- Treatment 1 (Aerial seeding fall 2002)
- Aerial seeding (mycorrhizae) fall 2003

Treatment 3 (bottomland mycorrhizal seeding and mulch):

- Treatment 1 (Aerial seeding fall 2002)
- Treatment 2 (Aerial seeding (mycorrhizae) fall 2003)
- Mulch fall 2003

There are several noteworthy issues associated with the application of the treatments that may either significantly influence or make it difficult to determine treatment effectiveness. These issues are addressed here to provide context for the methodological approach employed in this project.

The proportion of species in the seed mixes applied to the RFC was determined using kilograms of seed as a reference. This approach makes little sense ecologically because the amount of seed per pound varies between species. For example, Sand Dropseed (S*porobolus cryptandrus*) comprised 4% of the total weight in Treatments 2

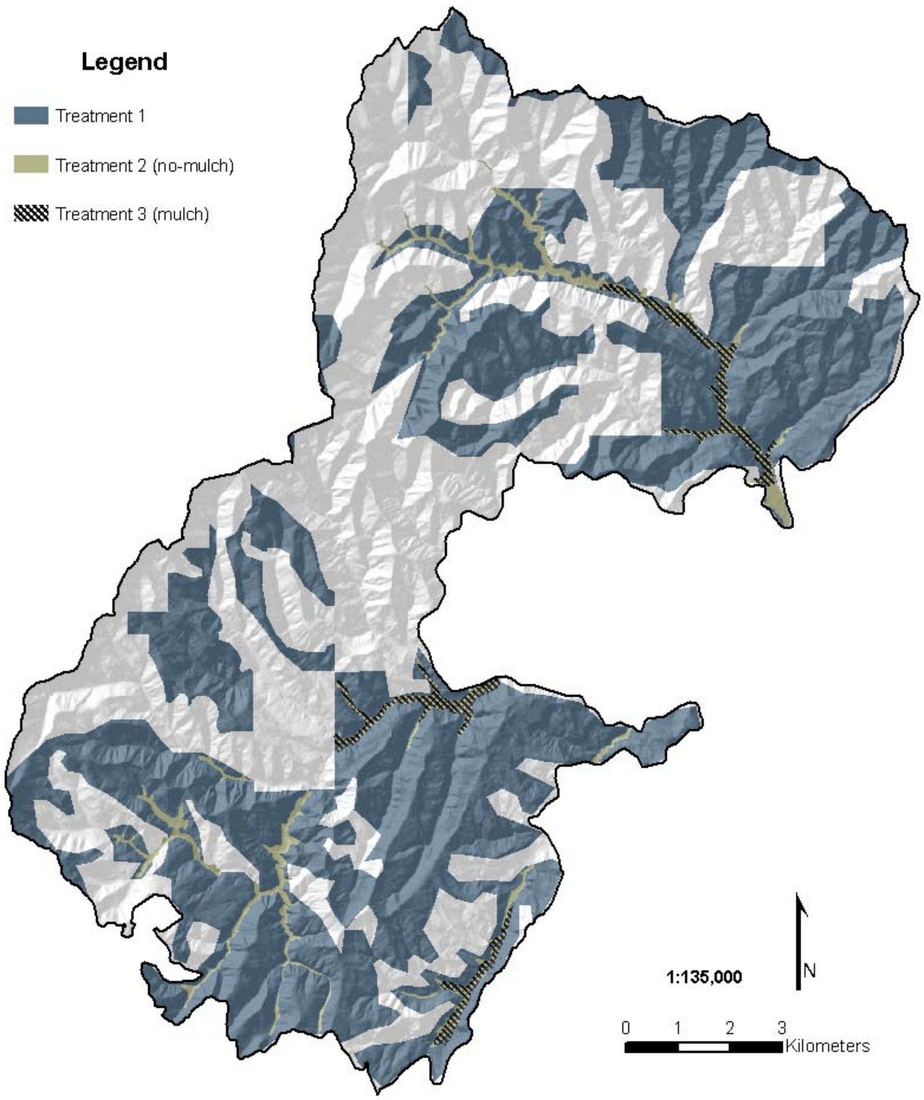

Figure 3. Treatments within study area.

and 3 but was 39% of the seed mix when evaluated by the number of viable seeds per square meter the same seed mix, Indian Ricegrass (*Oryzopsis hymenoides*) comprised 15% of the total weight but only 4% when using viable seeds per square meter as a metric. Therefore, in a seed mix, the weight of various species may be similar but the actual number of viable seeds may be highly disproportionate. In terms of rehabilitation, number of seeds equates more to the number of potential individuals that may be established than does a weight metric. Appendix A shows the proportion of seeds in each treatment based on the number of viable seeds derived using Pure Live Seed (PLS) values.

Secondly, it is not possible to quantify the effectiveness of the mycorrhizal coating because the mycorrhizal treatment was applied to the entire bottomland. In order to address this question a control treatment of the same seed mix without the mycorrhizal coating should have been applied. Adequate controls would provide the opportunity for valid statistical comparisons in addressing this research question.

Thirdly, no representative untreated control plots were present. Control plots were established in a 2004 pilot study, but several of these control plots had to be located in side canyons and were unexpectedly scoured away by overland flooding. The remaining control plots were located on a state tract of land in Cottonwood canyon that was hypothetically left untreated. It showed a relatively high percentage of seeded species which were not native to this area in the Book Cliffs indicating that these were not true controls. It is therefore impossible to quantify the effect of not seeding. The lack of good control sites has dictated the use of defined target/threshold objective in the determination of treatment effectiveness.

METHODS

This project employs a multi-scale approach to evaluate treatment effectiveness by incorporating both a traditional ground monitoring study and a remote sensing study. The methodologies of each are described separately within this section. The original work was performed in English units at the request of the BLM but was converted to metric for this thesis. This explains the unusual nested frequency quadrat sizes and macroplot dimensions.

Ground Monitoring Study

Ground sampling occurred between July 1 and August 25 in 2005 and was confined to the Diamond watershed for logistical and statistical reasons. Access to the Cottonwood and Diamond watersheds is extremely difficult. A rockslide in the 2004-2005 winter blocked road access up Cottonwood canyon. The remaining area is primarily roadless and motorized travel is restricted under WSA status. Existing roads are often inaccessible during the field season, even by ATV, making helicopter access necessary in some cases. Bottomlands are long and narrow and deeply incised by both ephemeral and perennial streams. These access issues are compounded by the large extent of the treatment areas in the Cottonwood and Diamond watersheds and the brief data collection window available before the Arizona Monsoon season arrives in mid-July. It was logistically difficult to obtain adequate sample sizes for rigorous statistical inference or adequate ecological inference when attempting to sample both watersheds.

This issue was addressed by limiting the statistical population of interest to the Diamond watershed. In the ground monitoring study, statistical inference was limited to the Diamond watershed but ecological inference was made to Cottonwood watershed. In other words, the Diamond watershed was both qualitatively and quantitatively monitored with the assumption that the levels of vegetative recovery and treatment effectiveness would be similar in Cottonwood to base future management decisions upon. The remote sensing analysis provides quantitative support and validation for this approach.

This broad ecological inference is also supported by similarities between the Diamond and Cottonwood watersheds as determined in the 2004 pilot study (Appendix B), ground reconnaissance, and background research. The similarities include:

- Similar topographic features
- Similar pre- and post-fire vegetation
- Similar hydrologic characteristics
- Similar soil map unit (Flatnose Loamy Bottomland) (Hansen 1989)
- Same ecological site description with same site potential (Hansen 1989)
- Same treatments

The BAER report recommended monitoring for treatment effectiveness and overall vegetative recovery. Management objectives have been defined post-hoc for both categories as none were explicitly defined prior to the application of the treatments. Quantitative sampling was only undertaken in Treatments 2 and 3 of the bottomland areas. Sampling locations were not located in monotypic stands of Gambel Oak or on the floodplains subjected to seasonal scouring. Monotypic Gambel Oak stands were not sampled because substantial herbaceous growth is not observed in either burned or unburned stands. Seasonally scoured floodplains were not sampled because the vegetative response is more directly related to flooding than fire rehabilitation. Qualitative monitoring methods were utilized in the Treatment 1 due to the challenges of accessing and sampling this acreage on steep and dangerous slopes.

Ecological Model and Treatment Objectives. The ecological model (Fig. 4) used as a baseline to derive the quantitative post-fire ESR objectives for Treatments 2 and 3 is based on SSURGO soils data (i.e. Soil Survey Geographic Data). These treatments are predominately composed of the Flatnose Loamy Bottom ecosite (438 ha). The upper reaches of each drainage, however, transition into Plite Mountain Loam (36 ha). Several other soil types are present in small amounts. Since the SSURGO Loamy Bottom ecosite dominates the bottomland treatment area, it was used in the development of the following ecological model (Fig. 4). The Loamy Bottom ecosite shows a potential absolute vegetative cover of 50% for grasses/grasslikes, 5% for forbs, and 15% for shrubs. The cover potentials of these functional groups were used in the determination of target/threshold objectives presented in the next section. Seeded species are considered surrogate inputs into these functional groups augmenting the potential for natural recovery from existing species (i.e. native species) in order to stabilize the watershed and minimize the invasion of cheatgrass.

Treatment effectiveness and overall vegetative recovery was quantified using cover and frequency data. Aerial line-point intercept cover and nested frequency data provide a powerful combination of measurable vegetation attributes and can be collected in a relatively short period of time. In this instance, cover is a vertical projection of vegetation from the ground as viewed from above (Elzinga et al. 1998). Cover values are the most directly related to biomass and will equalize the contribution of plant species to the overall vegetative cover. The use of points is the oldest method to quantify vegetative cover and is considered to be the most objective (Bonham 1989). This methodology for collecting cover data (i.e. line-point intercept) tends to underestimate rare species comprising less than 15% cover and species with narrow vertical growth habits (Bonham 1989). Based on the 2004 pilot study (Appendix B) the seeded species are being treated as rare species and therefore frequency data was collected concurrently with cover data to provide a more powerful assessment of the overall vegetative recovery and treatment effectiveness.

Since the post-fire establishment responses of the multitude of individual species present in the burn are variable, objectives were set based upon the establishment of functional groups (i.e. grasses, forbs, shrubs). The seeded species have been categorized into seeded grasses, seeded forbs, and seeded shrubs to evaluate treatment effectiveness and preferred grasses, preferred forbs, and preferred shrubs to evaluate overall vegetative recovery. The treatment may still be successful if functional group targets are achieved but establishment of individual species is low. Preferred life forms are non-invasive species that are either native to the area and are establishing naturally or have been seeded by the BLM. They may or may not have significant forage or cover value to wildlife, but do provide some important early seral ecological niche or competition against cheatgrass invasion. A list of preferred species defined for this project can be examined in Appendix C.

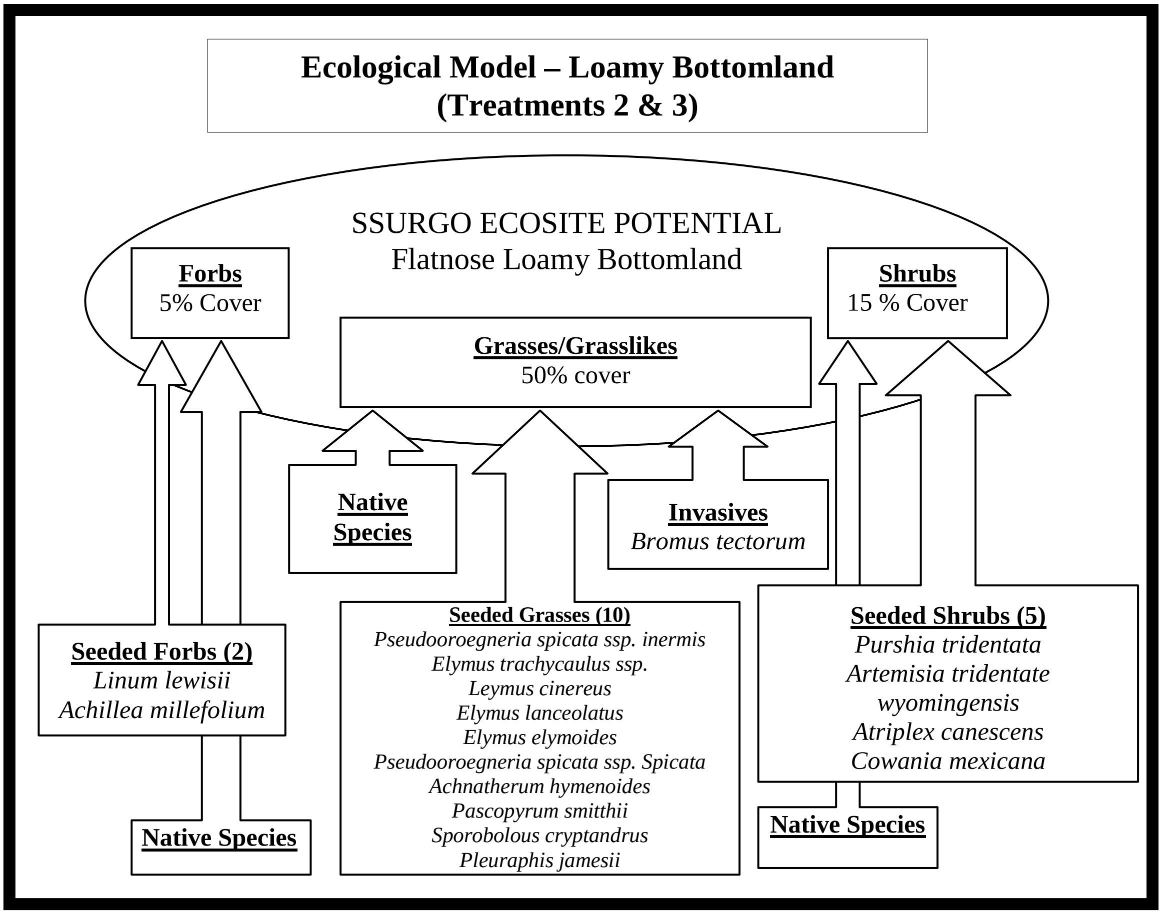

Figure 4. Ecological Model of the Loamy Bottomland Ecological Site (Treatments 2 and 3).

The objectives were defined based on a 2004 pilot study, field reconnaissance, SSURGO soils data, SSURGO ecosite descriptions, and consultations with BLM resource specialists. Relative values have been used to normalize for variations in absolute vegetative cover relating to climate variability or other factors. Target/Threshold objectives are intentionally weighted toward the establishment of forbs and grasses because these life forms dominate the early successional stages of the loamy bottomlands. Cover targets are lower than frequency targets as canopy cover may still be fairly low on early seral plants. The four target/threshold management objectives are:

1. **Cover Objective for Overall Vegetative Recovery**

   Obtain relative vegetative cover values for preferred life forms (native/seeded species) of 20% for grasses, 20% for forbs and 5% shrubs within the study area of the loamy bottomland ecological site within the Diamond/Cottonwood watersheds by 2006.

2. **Frequency Objective for Overall Vegetative Recovery**

   Obtain relative frequency values for preferred life forms (native/seeded species) of 30% for grasses, 30% for forbs, and 10% for shrubs by 2006 in the loamy bottomland of the Diamond/Cottonwood Watersheds.

3. **Cover Objective for Treatment Effectiveness**

   Limit the relative vegetative cover to 50% for cheatgrass (Bromus tectorum) within the study area of the loamy bottomland ecological site within the Diamond/Cottonwood watersheds by 2006.

4. **Frequency Objective for Treatment Effectiveness**

   Obtain relative frequency values for the seeded species of 50% for seeded grasses, 5% for seeded forbs, and 10% for seeded shrubs sp (ArTr, AtCa, CoMe) with the study area of the loamy bottomland ecological site of the Diamond/Cottonwood Watersheds by 2006.

Objective 1 allows for 55% of the relative vegetative cover to be comprised of undesirable species including cheatgrass. While this objective is not the optimal ecological scenario it is a realistic one based upon the ability of cheatgrass to proliferate after fire and its prevalent pre-fire distribution. **Objective 2** essentially states that for every 10 frequency quadrats containing vegetation, three should include native/seeded grasses, three should include native/seeded forbs, and one should include native/seeded shrubs. Cover was chosen to assess the spread of *Bromus tectorum* in **Objective 3**

because frequency values would be extremely high for all but the smallest nested frequency quadrats. High frequency values would neither provide a useful measure of cheatgrass nor provide room to detect change in subsequent years of monitoring. Frequency will be used in **Objective 4** to assess the establishment of seeded species. The seeded species are considered to be rare species based on the 2004 pilot study (Appendix B) and the frequency method helps provide a better assessment of composition and establishment where cover values would generally be underestimated.

Ground Sampling Protocols. Treatments 2 and 3 were sampled using quantitative methods between July 1 and August 25 in 2005. Five 39.6 x 18.3 meter (697 m^2) macroplots (Fig. 5)(BLM 1996, Elzinga et al. 1998) were established in Treatment 2 and Treatment 3. Eight of the macroplots (i.e. 4 each per treatment) were established at the eight randomly located transect locations from the 2004 pilot study (Appendix B) and one additional macroplot was randomly established in each treatment in 2005.

For each macroplot, a 39.6 meter baseline transect was randomly established and monumented using 0.6 meter rebar (Fig. 5). GPS locations were collected using a Trimble GeoXT and were differentially corrected and exported into shapefile format. Ten subtransects were systematically placed perpendicular to the baseline transect within each macroplot every 3.9 m. The first subtransect was located randomly between 0-2.7 m.

Nested frequency quadrats were placed systematically along the each subtransect every 1.8 m for a total of 10 quadrat readings per subtransect. The location of the first quadrat was located randomly between 0-1.5 m on each subtransect. The nested frequency sampling frame contained 7.6 x 7.6 cm, 15.2 x 15.2 cm, 30.4 x 30.4 cm and

60.8 x 60.8 cm quadrats. For nested frequency, plants were considered inside the quadrat if more than 50% of the plant was rooted within that quadrat. Fifty cover points were systematically placed along each subtransect with a random start and only top canopy hits were collected. A sample datasheet is located in Appendix C. Total data collection for each macroplot included 10 subtransects sampled, 100 nested frequency quadrats and 500 total cover points.

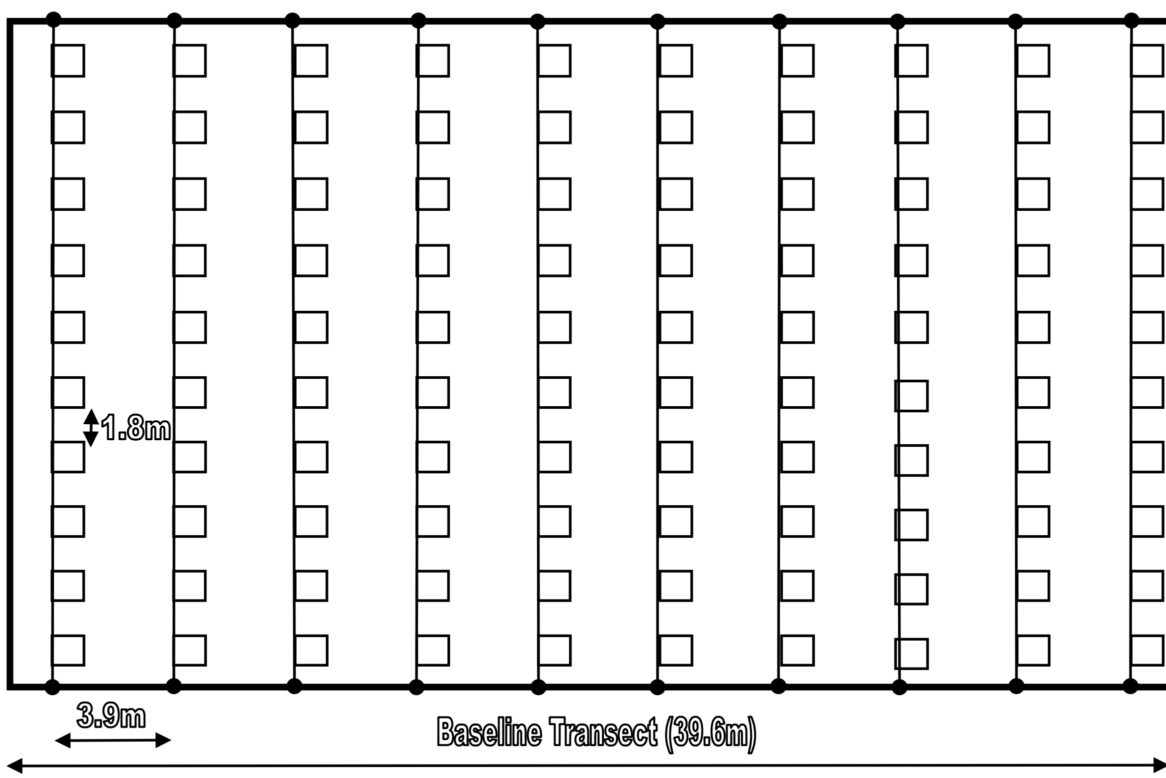

Figure 5. Macroplot Layout. Lines with dots are subtransects and squares are sampling frame locations. Cover points not shown but include 50 pts/subtransect.

Treatment 1 was monitored using qualitative monitoring methods. Four photopoints were randomly established on slopes of 20-40 degrees on varying aspects in

Diamond Canyon. Photopoints are landscape or feature photographs repeated through time from the same location so that changes can be observed over time (Elzinga et al. 1998). Photos were taken in a panorama at each photo point. A permanent 0.9 x 0.9 meter photoplot was established at each photopoint site. A photopoint is a photograph taken of a small area or plot taken from a specified height (Elzinga et al. 1998). A species composition list indicating presence/absence of all species was collected within 11.3 meter radius (0.04 ha) of each photopoint center. Photopoints and photoplots were monumented using 0.6 meter rebar and 0.6 meter angle iron respectively.

Statistical Analysis. Macroplot means were compared both individually against the target/threshold objective to determine any spatial variability in the success/failure of the defined objectives. Individual macroplot means were then aggregated to treatment level means and again compared against the target/threshold objectives. Confidence intervals were calculated using an alpha of 10%. This project is willing to accept a 10% probability of making a false-change error. Unlike laboratory studies better able to control system variability, field studies must encompass the high vegetative variability of post-fire ecosystems making the use of the traditional 5% alpha unreasonable. T-values were used instead of Z-values in these statistical calculations because of the small sample size (Elzinga et al. 1998, Durham 2008).

When reporting results for the management objectives there are some cases of uncertainty where the cover or frequency estimate and the upper bound of the confidence interval crossed the target threshold but the lower bound of the confidence interval did not. In these cases, if the 75% of the confidence interval, including the mean, has crossed the threshold than the objective will be considered to be met (Elzinga et al. 1998, Wirth and Pyke 2007). If the less than 75% of the confidence interval has crossed the threshold then no valid determination can be made and the objective will not be considered met. The 60.8 x 60.8 cm nested frequency quadrat was used for all species and functional groups for the purposes of this analysis. Means and standard deviations were calculated for each functional group and plotted against the target threshold objectives.

A Student's T-test was performed between the Treatment 2 (no-mulch) and Treatment 3 (mulch) means for each functional group and cheatgrass using an alpha of 10% (Tables 3.4 and 3.5). The null hypothesis being tested were:

H_0 = There is no significant difference between the means of the functional groups within the mulch (TRT 3) and no mulch (TRT 2) treatments.

H_1 = There is a significant difference between the mean of the functional groups within the mulch (TRT 3) and no mulch (TRT 2) treatments.

Photoplot data is semi-quantitatively analyzed by summarizing the occurrence of each seeded species collected at each upland photoplot. The occurrence of each species is then summarized by occurrence in all the photoplots.

Remote Sensing Study

Image Acquisition. Quickbird imagery was acquired over 28,300 ha within the RFC on May 26, 2005 and July 19, 2005. Image acquisition windows were centered during both the cool and warm seasons in order to capture the spectral differences resulting from changes in phenological condition. Several unsuccessful acquisition attempts were made before the six day satellite return interval coincided with cloud-free conditions.

The phenological patterns of both seeded and native species within this post-fire plant community are highly variable in time and space and by definition are influenced by the variability of weather and climate. The multispectral May image captured the green-up phenophase of cool season native grasses and forbs including cheatgrass. Cheatgrass was senescent in the July image while other native grasses and forbs remained green which allowed delineation of the cheatgrass landcover type. An analysis of these two images both phenologically and spectrally is the foundation for remote monitoring of treatment effectiveness and vegetative recovery.

The 11-bit bundled product included both multispectral and panchromatic scenes. The multispectral dataset contains four spectral bandwidths (Table 1) and is delivered with a 2.4 m spatial resolution. The multi-spectral data (4-band) includes the visible light bands (i.e. blue, green, and red) and the near infrared (NIR) band. The panchromatic dataset incorporates the four bands of data into a single band at a 60 cm spatial resolution.

Table 1. Quickbird Spectral Resolution.

Spectral Band	Band Pass (nm)
Panchromatic	525-924
Blue	447-512
Green	499-594
Red	620-688
NIR[1]	755-874

[1] NIR = Near Infrared

Pre-Processing. The imagery underwent multiple preprocessing techniques to correct for differences in solar geometry, atmospheric effects, and terrain distortions. The COST without Tau (Dark Object Subtraction) Atmospheric Correction was used to minimize the atmospheric effects and to normalize the two images for variations in brightness resulting from differences in the Earth-Sun Distance and Solar Zenith angle and to ultimately convert the digital numbers to both top-of atmosphere radiance and reflectance. The COST without Tau (Dark Object Subtraction) Atmospheric Correction provides results comparable to the more complex radiative transfer models (Chavez 1996) and performs better in arid rather than in humid environments (Wu et al. 2005). The *Radiometric Use of Quickbird Imagery Technical Note* (Krause 2005) was used to provide sensor-specific constants needed as inputs into the COST without Tau (Dark Object Subtraction) equation. Dark object input values were determined by selecting the values at the base of the slope of the histogram (Lowry 2003). The rectangular images were then clipped to the Diamond and Cottonwood watershed boundaries and the fire perimeter resulting in a 16,659 ha image. Small parts of the treatment areas occurring outside the watershed boundaries were eliminated. The images were georeferenced and georectified, by DigitalGlobe, using 1:4,800 user-supplied ground control points (GCPs) and 1:24,000 DOQQs using the nearest neighbor resampling algorithm and projected into the UTM NAD27 Zone 12 North.

Data Generation and Multi-temporal Stacking. The differential growth patterns represented within these time series are the key to the phenological delineation of vegetative growth forms. The Normalized Difference Vegetation Index (NDVI) is a commonly used vegetation metric that measures "greenness" and is highly correlated with the spectral reflectance characteristics of photosynthetically active vegetation (Asrar et al. 1992). NDVI has been successfully used to predict the potential distribution of cheatgrass and Dyers woad (*Isatis tinctoria*) (Lass et al. 2005, Peterson 2005). The NDVI was calculated for both image dates in order to capture canopy growth patterns (i.e. green-up and senescence) within the cool and warm seasons. The NDVI calculation is shown in Equation 1.

Equation 1. $\frac{NIR - red}{NIR + red}$

The ΔNDVI (i.e. change-NDVI) was calculated by taking the difference between the warm and cool season NDVI (Equation 2) and is a measures of the change in photosynthetic activity within that time period. In this ecosystem, pixels that show an extremely negative ΔNDVI are assumed to represent areas of high cheatgrass cover.

Equation 2. $\Delta NDVI = NDVI_{warm} - NDVI_{cool}$

Fractional cover images were generated using the NDVI* method (Owen et al. 1998). This metric of fractional cover uses NDVI values from bare soil and highly vegetated pixels as inputs to quantify horizontal vegetation structure by estimating the percent of the ground that is covered by photosynthetically active vegetation (Equation 3, Equation 4).

Equation 3. $NDVI^* = \frac{NDVI - NDVI_{bare\ soil}}{NDVI_{max} - NDVI_{bare\ soil}}$

Equation 4. Fractional Cover = $(NDVI^*)^2$

The time series of pre-processed spectral data, NDVI, and ΔNDVI data were combined into a single dataset or spectro-phenological layerstack for analysis. The layerstack is an 11-band image that captures the unique spectral vegetation signatures and unique canopy growth patterns within the project area. The layerstack was rescaled from floating point to integer to save disk space. Fractional cover data was excluded because it did not meet the assumptions of normality required from the classification algorithms.

Supervised Classification. Landcover information was generated using a supervised Maximum-Likelihood (MLH) statistical classification method using ERDAS IMAGINE software available on the Bureau of Land Management computer network. Training sites were identified for each landcover class using reference data derived from ground mapping (see Ground Sampling section), Quickbird panchromatic imagery, and 2006 National Agricultural Imagery Program (NAIP) imagery. Spectral signatures were derived from the spectro-phenological layerstack at each of the training sites. The signatures were analyzed for spectral separability using the statistical models and graphical displays available in IMAGINE. The final signature set represents a statistically separable set of signatures that incorporate spectral and phenological information. The spectro-phenological layerstack was then classified using the final signature set and the MLH algorithm.

The MLH algorithm uses the mean reflectance of each band to determine the spectral pattern for a given landcover type (Jensen 1996, Lass et al. 2005). All pixels are assigned to a class based on the probability that they belong to that class. MLH was used because it is often the most accurate of the classifiers available in ERDAS IMAGINE and all the input bands met the assumption of normality (Smith and Brown 1999).

Landcover classes were selected that were specifically related to ESR monitoring objectives but also sufficiently general as to be statistically discernible to a classification algorithm given the limited spectral resolution of Quickbird. The landcover classes were:

1. Light Bare Soil / Rock
2. Dark Bare Soil / Rock
3. Skeleton Forest
4. Mixed Grass-Forb
5. High Cover Cheatgrass
6. Broadleaf Deciduous Canopy
7. Coniferous Evergreen Canopy

The resultant classification image was post-processed using the CLUMP and ELIMINATE functions in IMAGINE. Groups of similar adjacent pixels were identified using the CLUMP function. The ELIMINATE function was then used to remove clumps smaller than four pixels (23 m^2) and replace these pixels values with the values of their dominant neighbor. This resulted in a final landcover map with minimum clump sizes of four pixels and minimized some of the “salt and pepper” appearance within the image.

In order to better understand treatment effectiveness the distribution of the bare soil class was further refined by cool/wet and warm/dry aspects. Warm dry aspects are likely to have greater amounts of bare soil similar to pre-fire conditions. Cool/wet aspects are defined by aspects ranging 315-135 degrees and warm/dry aspects ranging from 135-315 degrees.

Ground Sampling and Accuracy Assessment. Ground sampling was completed in 2005 between July 1 and August 25. Ground sampling consisted of:

- 10 randomly located macroplots (39.6 m x 18.3 m) quantified using line-point intercept cover and nested frequency (Fig. 5).
- Repeat photography
- Sub-meter GPS mapping of training sites using Trimble GeoXT GPS receiver.

Cover and frequency data collection methodologies can be found in the Ground Sampling Protocols section above. Forty representative training sites were also collected in the field using a Trimble GeoXT and differentially corrected to improve positional accuracy. Training sites consisted of areas of homogenous or mixed vegetation larger than six m^2. Random macroplot locations were generated using a GIS random sampling algorithm.

The accuracy of the classification was evaluated at the pixel-level using an error matrix (Jensen 1996, Congalton and Green 1999). Fifty reference coordinates were generated at random within each of the seven landcover types. These reference pixels were "ground-truthed" using the multispectral, panchromatic, 2006 NAIP imagery, and ground data. Additionally, some of the mapped polygons that were not used as training sites were utilized as reference pixels. The training sites used to generate spectral signatures were not used as reference sites. The error matrix was created by comparing the reference pixels against the classification output.

The overall, user's accuracy, producer's accuracy, and kappa statistic were calculated to better evaluate the classification accuracy. The overall accuracy represents the general accuracy of the classification while the producer's and user's accuracies are specific to each landcover type. The overall accuracy was calculated by dividing the total correct pixels (i.e. sum of the major diagonal) by the total number of pixels in the error matrix (Jensen 1996, Lillesand and Kiefer 2002). The producer's accuracy was calculated for each landcover class by dividing the number of correctly classified pixels within the column by the total number of pixels in the column. The producer's accuracy represents the probability of a reference pixel being correctly classified and indicates how well the landcover class can be classified (Jensen 1996, Lillesand and Kiefer 2002). The user's accuracy was calculated by the number of correctly classified pixels within each row by the total number of pixels in that row. The user's accuracy indicates the probability that a pixel classified on the map actually exists on the ground (Jensen 1996, Lillesand and Kiefer 2002). The kappa statistic relates the classification to one resulting from chance and was calculated using the methodology described in Jensen (1996).

Cheatgrass Cover Linear Regression Model. Simple linear regression techniques were used to create a model that predicts absolute cheatgrass cover within the mixed grass-forb landcover type by relating the macroplot cheatgrass cover data to changes in NDVI. A mathematical relationship between the mean cheatgrass cover value (i.e. ground data) and the mean ΔNDVI (i.e. remote sensing) was calculated using nine of the ten macroplots (725 m^2) and analyzed using simple linear regression. A tenth macroplot was not used because it was not representative of the grass-forb landcover type.

The linear regression equation was applied to the ΔNDVI dataset resulting in a predictive map of continuous cheatgrass cover within the mixed grass-forb landcover class. The dataset was grouped into cover classes 0-24.999%, 25-49.999%, 50-74.999% and 75-100% and summarized by treatment.

In order to evaluate the performance of the model, a mean model was created using the mean cheatgrass cover value of all the macroplots combined. The bias and mean absolute error (MAE) were calculated for both models. The linear regression model was then compared against the mean model to see if the regression model is more informative.

Subsampling/Bootstrap Analysis. The macroplot mean cheatgrass cover is derived from the measurement of 10 subtransects within each macroplot (i.e. 90 subtransects) requiring a significant time investment. A subsampling analysis was undertaken to determine how many subtransects need to be sampled within each macroplot to generate a linear regression model of the same power. The bootstrapping analysis generated a linear regression model and associated R^2 value, at each iteration, by utilizing all nine macroplot means calculated from incrementally fewer subtransects (i.e. 1-9). The model was generated 100 times for each sample size and the results reported using simple summary statistics.

RESULTS

Ground Monitoring Study

Treatment 1-Upland Seeding. Each photoplot contained from three to five seeded species (Table 2). *Pseudoroegneria spicata spp. inermis* and *Elymus trachycaulus* occurred in all of the photoplots. *Leymus cinereus* occurred in one of the four photoplots. *Elymus lanceolatus* occurred in three of the four photoplots. *Purshia tridentata* was not present in any photoplot. *Achillea millefolium* occurred in three out of four photoplots. *Linum lewisii* occurred in one of four photoplots. When consolidated into functional groups, seeded grasses occurred in all of the photoplots; seeded forbs occurred in 3 out of the four photoplots; and the seeded shrubs occurred in zero of the photoplots (Table 2).

Table 2. Occurrence of Seeded Species in Treatment 1 at each Photopoint (0=absence; 1=presence).

Seeded Species	Photopoint 1	Photopoint 2	Photopoint 3	Photopoint 4	Total	%
Pseudoroegneria spicata	1	1	1	1	4	100%
Elymus trachycaulus	1	1	1	1	4	100%
Leymus cinereus	1	0	0	0	1	25%
Elymus lanceolatus	1	1	1	0	3	75%
Purshia tridentata	0	0	0	0	0	0%
Achillea millefolium	1	0	1	1	3	75%
Linum lewisii	0	0	1	0	1	25%
Total # of seeded species present	5	3	5	3		

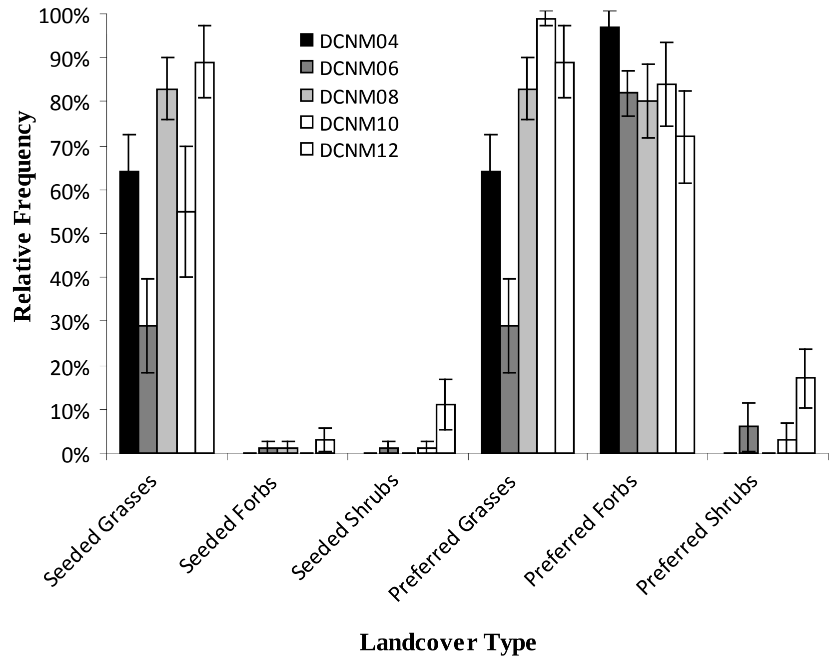

Figure 6. Mean Relative Frequency for Each Macroplot in Treatment 2 (No-Mulch). Error bars represent 90% confidence. Black lines show management objectives. DCNM04 stands for Diamond Canyon No-Mulch Macroplot 4

Treatment 2–Mycorrhizal Bottomland Seeding and Overlap of Treatment 1. The Frequency Objective 4 for treatment effectiveness was not fully achieved by all of the macroplots. The objective was reached by 60% of the macroplots with respect to seeded grasses (Fig. 6). The seeded forb objective and the seeded shrub objective were not reached by any macroplots. However, the Cover Objective 3 for treatment effectiveness was reached 80% of the macroplots since their mean relative cheatgrass cover was below 50% (Fig. 7).

There were no macroplots that fully achieved the Cover Objective 1 for overall vegetative recovery for each functional group (Fig. 7). The preferred grass cover and

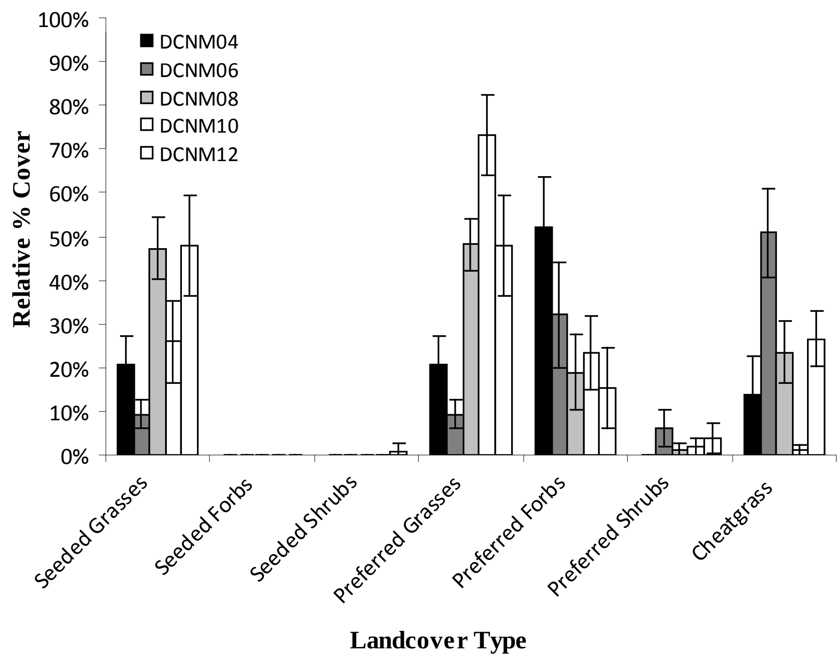

Figure 7. Mean Relative Cover for Each Macroplot in Treatment 2 (No-Mulch). Error bars represent 90% confidence. Black lines show management objectives. DCNM04 stands for Diamond Canyon No-Mulch Macroplot 4.

preferred forb cover objectives were both met by 60% of the macroplots. Only 20% of the macroplots reached the cover objective for preferred shrubs.

Frequency Objective 2 for overall vegetative recovery for all functional groups was achieved by only 20% of the macroplots (Fig. 6). The preferred grass objective was reached by 80% of the macroplots while 100% of the macroplots reached the objective for preferred forbs. Only 20% of the macroplots reached the frequency objective for preferred shrubs.

Treatment 3-Hydromulch Applied to Areas of Treatment 2. The Frequency Objective 4 for treatment effectiveness was met by 80% of the macroplots with respect to seeded grasses (Fig. 8). No macroplots reached this objective for seeded forbs or seeded

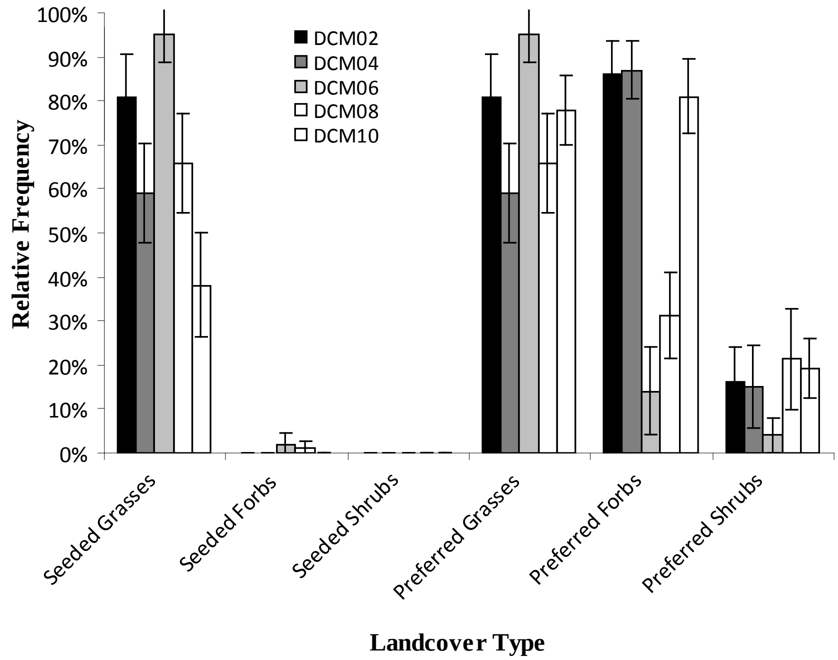

Figure 8. Mean Relative Frequency for Each Macroplot in Treatment 3 (Mulch). Error bars represent 90% confidence. Black lines show management objectives. DCM02 stands for Diamond Canyon Mulch Macroplot 2.

shrubs. The Cover Objective 3 for treatment effectiveness was reached by only 20% of the macroplots while 80% of macroplots had relative cheatgrass cover values greater than 50%. The overall mean cheatgrass cover for the Treatment 2 was 49% (±15.1%; Fig. 10).

There were no macroplots that fully achieved the Cover Objective 1 for overall vegetative recovery for each functional group (Fig. 9). Cover Objective 1 for preferred grasses and preferred forbs was met by 40% of the macroplots while 80% of the macroplots reached the cover objective for preferred shrubs.

There were four out of five (i.e. 80%) macroplots that reached the Frequency Objective 2 for overall vegetative recovery for all functional groups. All of the

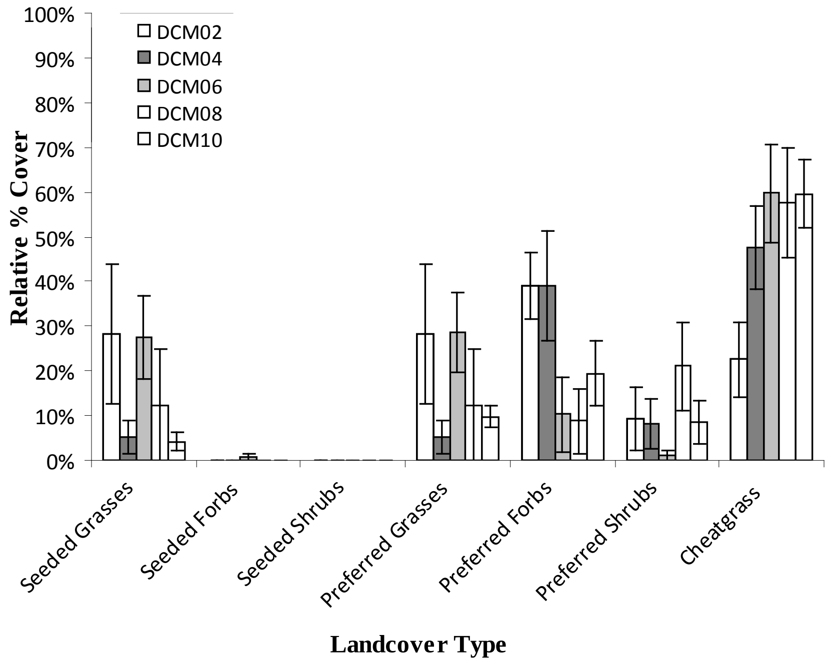

Figure 9. Mean Relative Cover for Each Macroplot in Treatment 3 (Mulch). Error bars represent 90% confidence. Black lines show management objectives. DCM02 stands for Diamond Canyon Mulch Macroplot 2.

macroplots reached the frequency objective for preferred grasses. The objectives for preferred forbs and preferred shrubs were met by %80.

Mulch vs. No Mulch. The T-test showed that Treatment 3 (mulch) had a significantly higher cover and frequency value for preferred shrubs and higher cheatgrass cover (Table 3). Conversely, Treatment 2 (no mulch) had a significantly higher cover value for preferred grasses (Table 3). There were no statistical differences between any other categories. The assumptions of normality and homogeneity of variances were met according to the guidelines outlined in Measuring and Monitoring Plant Populations (Elzinga et al.,1998), which allows for differences in variances of a factor of 2 to 3. Only the frequency of preferred forbs was questionable on the assumption of homogeneity of variances. The mean cover and frequency values for each functional group and 90%

confidence intervals are shown below (Figs. 9-10). The width of the confidence intervals are notably increased as the variation between macroplots is incorporated into a single statistic. The general distribution of ground cover (Fig.12) resulted in absolute vegetative cover of approximately 50% for both treatments. The frequency metric is used to evaluate the distribution and effectiveness of individual species (Fig. 13). The first ten species were seeded and the last seven occurred naturally.

Table 3. Results of t-test for Relative Frequency and Relative Cover. Significant relationships are shown in bold italics (alpha=10%).

	Relative Frequency		Relative Cover	
	Mulch Mean	No Mulch Mean	Mulch Mean	No Mulch Mean
Seeded Grasses	67.8	64.0	15.5	30.2
Seeded Forbs	0.6	1.0	0.1	0.0
Seeded Shrubs	0.0	2.6	0.0	0.2
Preferred Grasses	75.8	72.8	***16.8***	***39.8***
Preferred Forbs	59.9	83.0	23.4	28.4
Preferred Shrubs	***15.1***	***5.2***	***9.6***	***2.6***
Cheatgrass	89.4	63.2	***49.4***	***23.2***

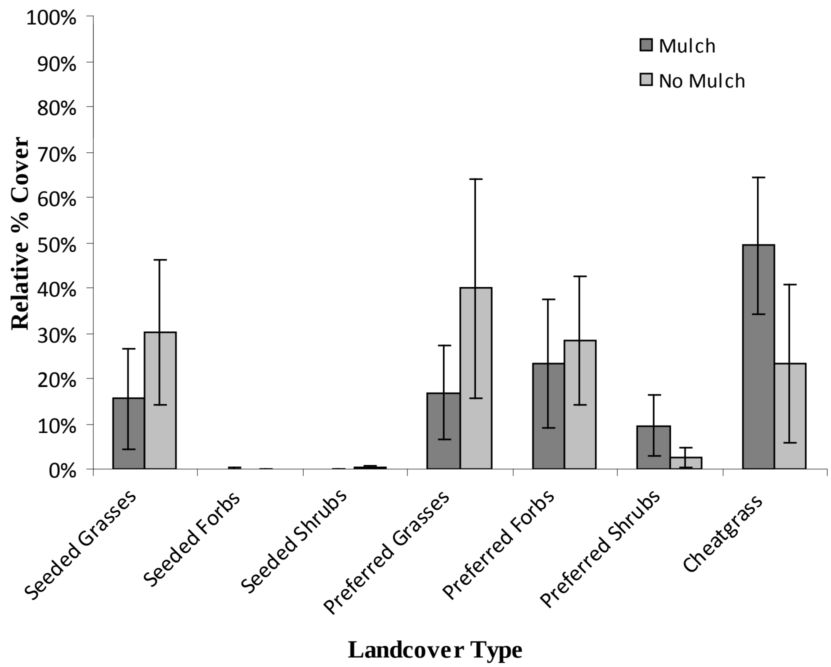

Figure 10. Mulch (Trt 3) vs. No Mulch (Trt 2) Comparison of Relative Cover. Error bars represent 90% confidence.

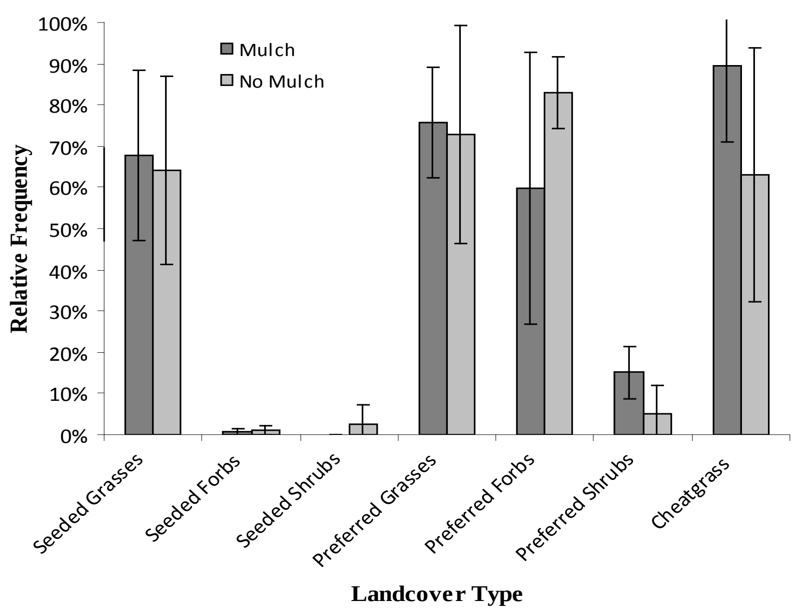

Figure 11. Mulch (Trt 3) vs. No Mulch (Trt 2) Comparison of Relative Frequency. Error bars represent 90% confidence.

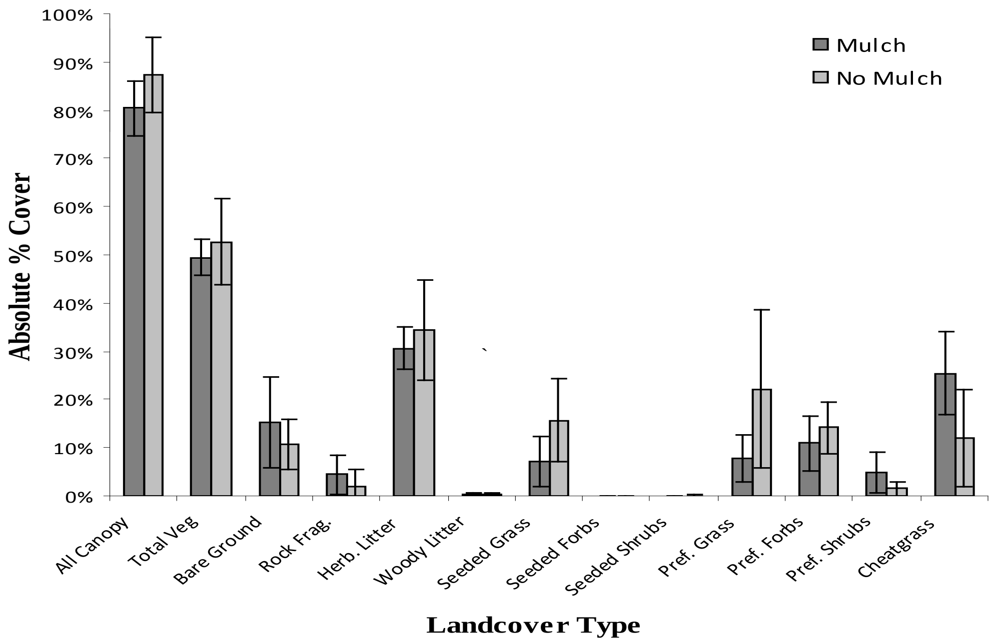

Figure 12. Mulch (Trt 3) vs. No Mulch (Trt 2) Comparison of Absolute Cover of All Landcover Types. Error bars represent 90% confidence.

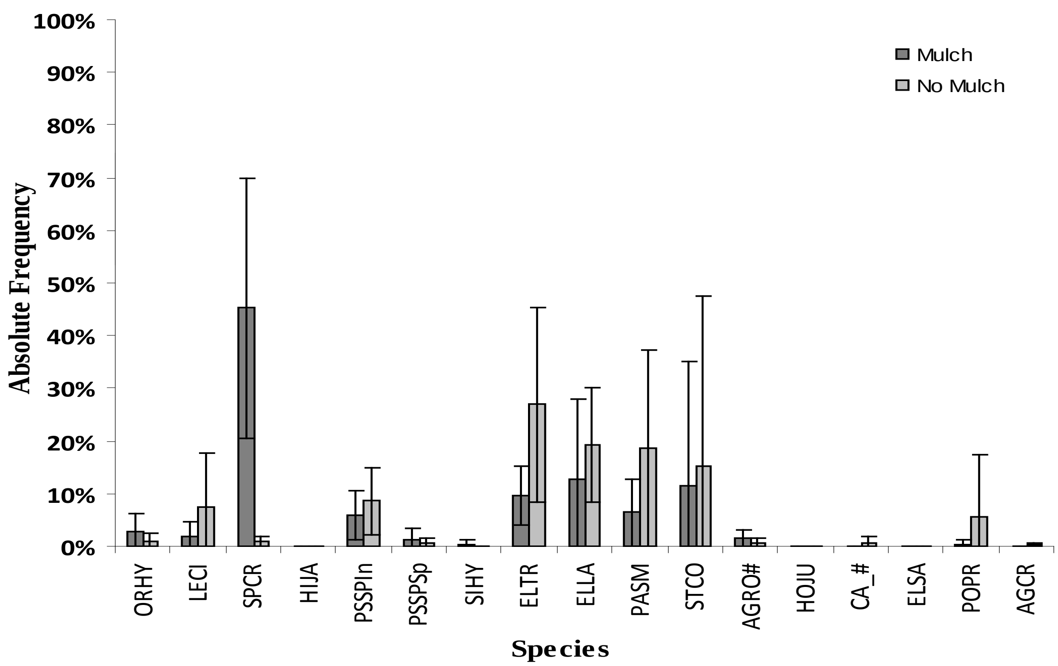

Figure 13. Absolute Frequency of Seeded and Native Grass Species. Error bars show the 95% confidence interval. The first 10 species are seeded the last 7 are natural regrowth. Species code abbreviations are described in Appendix A.

Remote Sensing Study

Accuracy Assessment. The results of the accuracy assessment including the user's, producer's and overall accuracy are shown in Table 4. The associated Kappa statistic is 0.80. The Kappa statistic can be interpreted to mean that this classification is 80% better than one resulting from chance (Lillesand and Kiefer 2002, Viera and Garrett 2005). The overall accuracy of the classification was 83%. The evergreen class was difficult to classify as indicated by a low producer's accuracy of 51%. The broadleaf class was also had a relatively low producer's accuracy of 75%.

Table 4. Error Matrix for Supervised Classification

	Reference Data								
Classification Data	L. Bare Soil/Rock[1]	D. Bare Soil/Rock[2]	Skeleton Forest	Mixed Grass/Forb	Cheatgrass Mono[3]	Broadleaf	Evergreen	Row Total	User's Accuracy
L. Bare Soil/Rock[1]	84	0	1	13	0	0	2	100	84%
D. Bare Soil/Rock[2]	4	55	0	1	0	0	0	60	92%
Skeleton Forest	0	0	57	0	0	4	0	61	93%
Mixed Grass/Forb	0	2	0	50	4	1	20	77	65%
Cheatgrass Mono[3]	0	0	0	2	47	0	9	58	81%
Broadleaf	0	0	0	0	0	55	3	58	95%
Evergreen	0	0	1	0	0	13	35	49	71%
Column Total	88	57	59	66	51	73	69	463	
Producer's Accuracy	96%	97%	97%	76%	92%	75%	51%		83%

[1] Light Bare Soil & Rock
[2] Dark Bare Soil & Rock
[3] Cheatgrass Monoculture

Supervised Landcover Classification. The distribution of landcover classes for the overall project area and within each treatment area is reported in Table 5. Cheatgrass monocultures contributed to only 111 ha (0.7%) of the entire study area and 49 ha (1%) of Treatment 1. Treatment 2 contained 13 ha (4.2%) of cheatgrass monoculture while Treatment 3 had 19 ha (7.2%). Bare soil was prevalent on 4,129 ha (50%) of Treatment 1 and 46 ha (15%) of Treatment 2. Treatment 3 contained 70 ha (26%) of bare soil. The distribution of bare soil by aspect is shown in Table 6. Bare soil within Treatment 1 was proportionally more prevalent (Table 6; 56.7 %) on warmer/drier aspects.

Table 5. Landcover Distribution by Treatment. Values shown are hectares and % of area in parentheses. Due to rounding, values shown do not total to 100%.

Landcover	Treatment 1	Treatment 2	Treatment 3	Study Area
Light Bare Soil/Rock	3,261 (40)	23 (7)	53 (20)	6,001 (36)
Dark Bare Soil/Rock	867 (11)	24 (8)	17 (6)	2,127 (13)
Skeleton Forest	811 (10)	9 (3)	2 (1)	1,885 (11)
Mixed Grass-Forb	1,278 (16)	113 (37)	126 (48)	2,572 (15)
Cheatgrass Monocultures	49 (1)	13 (4)	19 (7)	111 (1)
Broadleaf Tree	892 (11)	51 (17)	23 (9)	1,754 (11)
Coniferous Tree	1069 (13)	75 (24)	24 (9)	2,209 (13)
Total Hectares	8,227 (100)	307 (100)	263 (100)	16,659 (100)

Table 6. Bare Soil Distribution by Aspect & Treatment. Values shown are hectares and % of area in parentheses. Due to rounding, values shown do not total to 100%.

All Bare Soil/Rock	Treatment 1	Treatment 2	Treatment 3	Study Area
Cooler/Wetter Aspects	1,785 (43)	24 (51)	32 (46)	3,418 (42)
Hotter/Drier Aspects	2,336 (57)	23 (49)	38 (54)	4,703 (58)
Total Hectares	4,121 (100)	46 (100)	70 (100)	8,121 (100)

Cheatgrass Cover Linear Regression Model. The linear regression model had an R^2 value of 0.75 (Fig. 14) between the ΔNDVI and the absolute cheatgrass cover measured within the sampled macroplots. The model was statistically significant ($P<0.01$; F-test) with a standard error of 6.1. The bias and mean absolute error of the

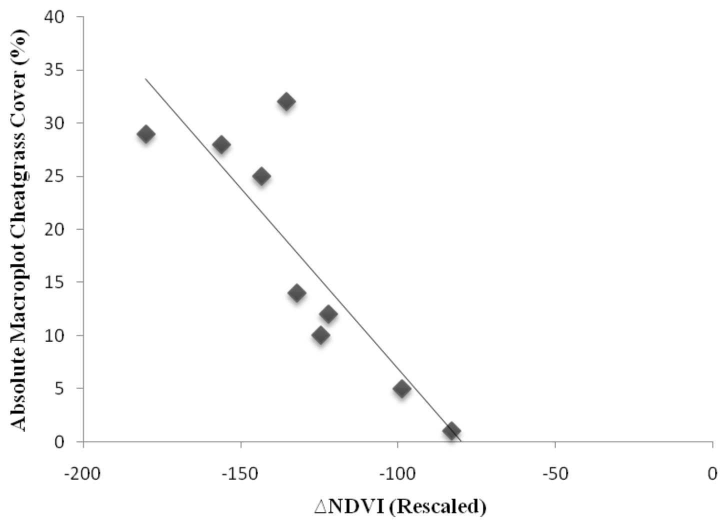

Figure 14. Macroplot Cheatgrass Cover and ΔNDVI Regression Model. Y= -0.34x – 27.17; Intercept SE = 9.97; X Variable SE = 0.075; Model SE = 6.1; R^2=0.75.

Table 7. Macroplot Regression Model

Macroplot	Predicted	Observed	Predicted-Observed	Abs (Predicted-Observed)
DCM02	15.9	10	5.9	5.9
DCM04	21.2	25	-3.9	3.9
DCM06	17.3	32	-14.7	14.7
DCM10	38.9	29	9.9	9.9
DCNM04	7.0	5	2.0	2.0
DCNM06	25.5	28	-2.5	2.5
DCNM08	14.8	12	2.8	2.8
DCNM10	1.3	1	0.3	0.3
DCNM12	18.3	14	4.3	4.3
Bias =	0.5			
Mean Absolute Error (MAE) =		5.1		

Table 8. Mean Regression Model

Macroplot	CgAvg[1]	Observed	Predicted-Observed	Abs (Predicted-Observed)
DCM02	17.8	10	7.8	7.8
DCM04	17.8	25	-7.2	7.2
DCM06	17.8	32	-14.2	14.2
DCM10	17.8	29	-11.2	11.2
DCNM04	17.8	5	12.8	12.8
DCNM06	17.8	28	-10.2	10.2
DCNM08	17.8	12	5.8	5.8
DCNM10	17.8	1	16.8	16.8
DCNM12	17.8	14	3.8	3.8
Bias =	0.5			
Mean Absolute Error (MAE) =		10.0		

[1] Mean Macroplot Cheatgrass Cover (N=9)

linear regression model predictions were 0.5 and 5.1 respectively (Table 7). The bias and mean absolute error calculated for the mean model predictions were 0.5 and 10.0 respectively (Table 8).

The results of the linear regression model are divided into four cover classes 0-24.999%, 25-49.999%, 50-74.999% and 75-100%. The distribution of cover classes by acreage and % of total area are shown in Table 9 and Figure 15. Treatment 3 shows the highest percent of the total area with cheatgrass cover greater than 25%.

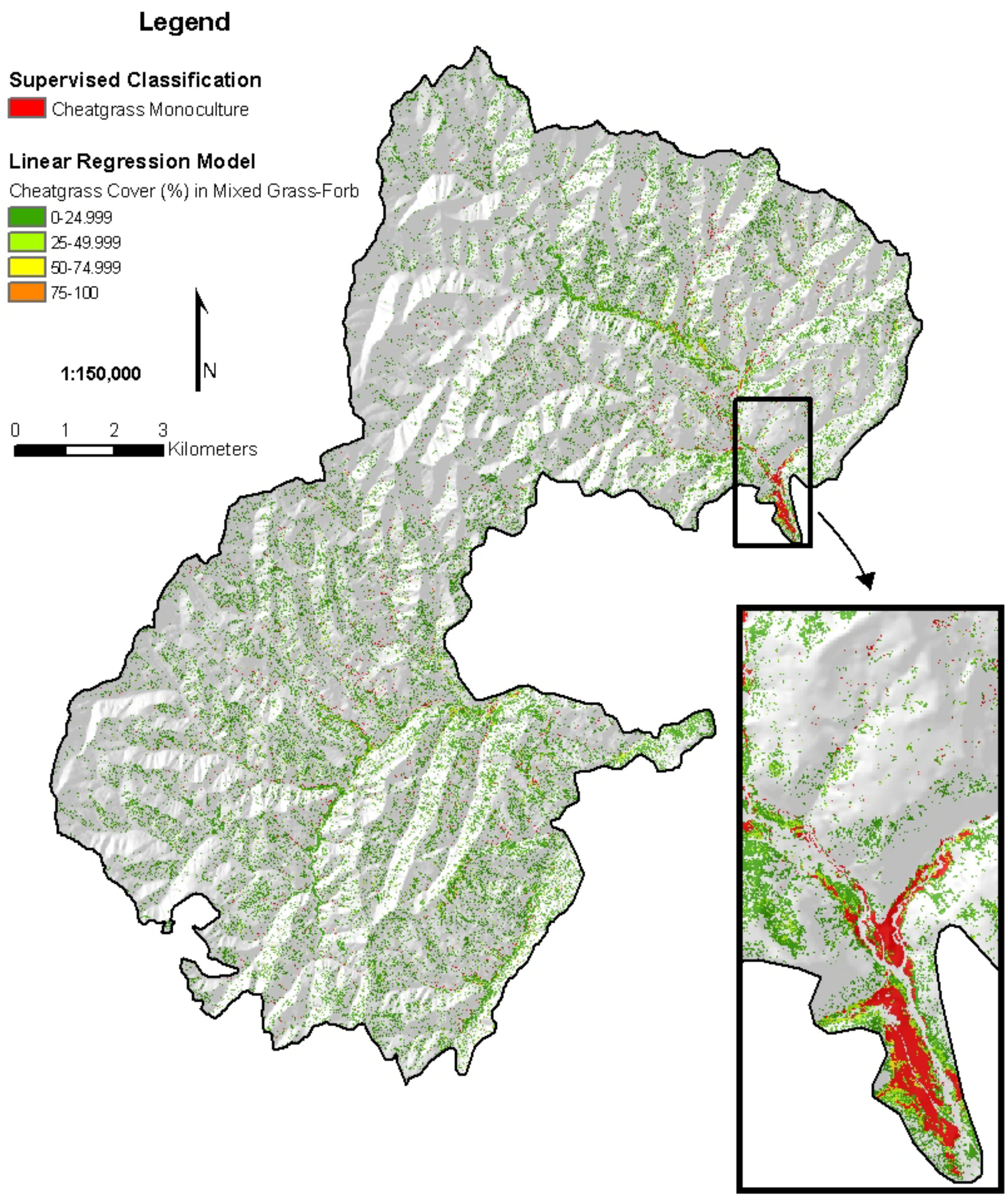

Figure 15. Cheatgrass distribution within study area.

Table 9. Distribution of Cheatgrass Cover Classes in Mixed Grass-Forb Landcover. Values shown are hectares with % of area in parentheses. Due to rounding, values shown do not total to 100%.

% Absolute Cheatgrass Cover	Treatment 1	Treatment 2	Treatment 3	Study Area
0-24.999%	1,150 (90)	95 (84)	85 (68)	2,293 (89)
25-49.999%	109 (9)	16 (14)	33 (26)	237 (9)
50-74.999%	17 (1)	2 (2)	7 6)	39 (2)
75-100%	2 (0)	0 (0)	1 (1)	4 (0)
Total Hectares	1,278 (100)	113 (100)	126 (100)	2,573 (100)

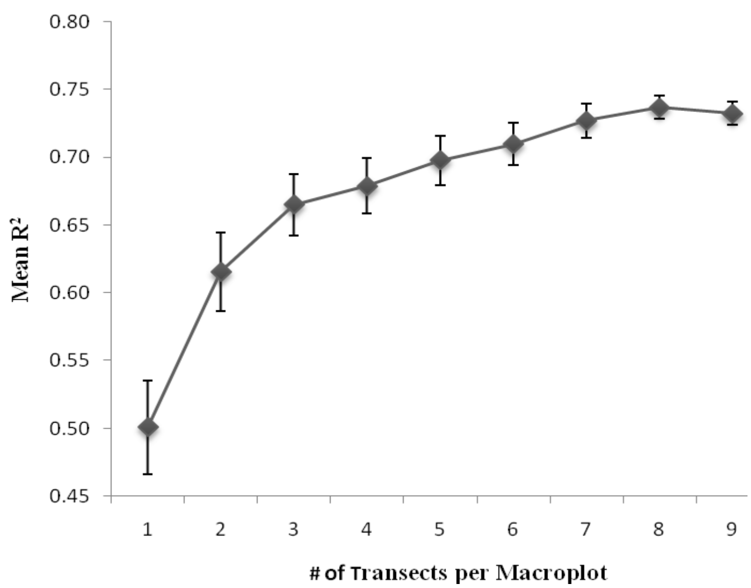

Figure 16. ΔNDVI Regression Model Performance and Sampling Intensity. Error bars show the 95% confidence interval.

Subsampling/ Bootstrap Analysis. The subsampling analysis showed significant improvement of the R^2 model values when increasing the sample size to three subtransects (Fig. 16). Increasing the number of transects beyond three showed incrementally smaller improvements in R^2. The standard deviation exhibits a strong inverse relationship with the number of transects sampled.

DISCUSSION

The primary goals of this ESR seeding project were to establish a ground cover of seeded species to minimize the expansion of cheatgrass and stabilize the soil within the Diamond and Cottonwood watersheds. The post-treatment distribution of cheatgrass and bare soil are therefore central to understanding treatment effectiveness. Significant distribution and abundance of either imply failure of the treatments to establish sufficient seeded ground cover to meet ESR objectives.

This methodology combined a traditional ground analysis with a two pass remote sensing approach in which a first pass supervised classification mapped the distribution of general landcover types (i.e. cheatgrass monocultures, bare soil/rock, etc) while the second pass utilized simple statistical models to gain more detailed insight into the distribution of cheatgrass. This methodology allows managers to commit various levels of time and resources to monitor ESR treatment effectiveness. Managers can either perform a simple ground analysis or can exploit the spectro-phenological characteristics of cheatgrass by integrating a first pass and/or second pass remote sensing analysis.

Supervised Classification

This first pass mapping exercise allows managers to identify the “hot-spots” of cheatgrass invasion where germination of seeded species is inherently low by employing simple remote sensing and ground techniques. The spectro-phenological signature of high cover cheatgrass sites is unique and statistically separable within the study area. Interestingly, the cool season spectral signature of a vigorous cheatgrass monoculture is statistically indistinguishable from broadleaf deciduous canopies (e.g. *Populous tremuloides, Quercus gambelii, Acer negundo*, *Populous fremontii*) with Quickbird data. Likewise, the warm season spectral signature of cheatgrass is typical of any senescent grass. However, the phenological signature generated when the two spectral signatures are combined is distinguishable from other vegetation phenologies present. In other words, the timing of cheatgrass greenup and senescence is unique from all other species within the study area. Areas of high cheatgrass cover can thus be identified using the MLH supervised classification algorithm and simple ground validation. Information on the abundance and distribution of cheatgrass monocultures is important to the adaptive management process and understanding treatment effectiveness.

The distribution and abundance of bare soil was also mapped during the 1st pass, allowing managers to gain a better understanding of treatment effectiveness and the potential for soil erosion and flooding. The relationship between % vegetative cover and erosion potential has been well established; bare canopy interspaces within Pinyon-Juniper woodlands generate, on average, about three times more sediment than patches with herbaceous cover and 24 times more sediment than patches underneath Pinyon-Juniper canopies (Reid et al. 1999). It is mainly these bare intercanopy patches that produce runoff during precipitation events (Reid et al. 1999). Patches of vegetation act as barriers which slow and trap runoff, sediments, and nutrients derived from bare canopy interspaces (Wilcox and Breshears 1995, Ludwig et al. 2005). The cover and distribution of vegetation patches is often reduced by grazing or fire, greatly reducing the ability of the system to trap and retain water and resources (Scanlan et al. 1996).

The vegetation within the Pinyon-Juniper woodlands of the RFC was almost completely consumed, converting the majority of the system into bare canopy interspace which resulted in higher rates of runoff and erosion within the system. ReGAP data show that broadleaf and coniferous landcover types occurred on 8,381 ha of the Treatment 1 area prior to the fire. The Quickbird landcover classification shows that these areas comprised only 1,880 ha after treatment in 2005. While there is a significant difference in the spatial resolution between the ReGAP classification (30m) and the Quickbird classification (2.4m) there does appear to be a decrease of approximately 75% in the amount of broadleaf and coniferous canopy cover. This reduction on tree canopy cover clearly resulted in an increase in the amount of bare soil.

The Quickbird classification shows that bare soil/rock occurred on 4,129 ha (51%) of Treatment 1. These areas are essentially bare intercanopy patches with the highest rate of runoff and erosion. The treatment was not effective in establishing seeded species within these areas. However, the warmer and drier upland slopes within the study area have historically had sparse herbaceous cover and bare soil was more prevalent in these areas prior to the fire than on cooler/wetter aspects. As a result, treatment effectiveness on the warmer/drier aspects was expected to be lower than on cooler/wetter aspects. The results from the analysis follow this trend and show that 57% of the bare soil in Treatment 1 occurred on warmer/drier aspects. While the absolute amount and distribution of pre-fire bare soil is unknown it is clear that there is still more bare soil after the fire even with the application of seeding treatments than existed prior to the fire.

A more detailed analysis of watershed stability could be performed with the output from this analysis but is beyond the scope of this project.

Accuracy Assessment

The largest classification error involves the commission error of evergreen landcover into mixed grass/forb landcover type. Therefore the distribution of the mixed-grass forb landcover type appears to be overestimated in the upland areas where the evergreen landcover was most prevalent. This author believes that the error is primarily due to a scale mismatch between the pinyon and juniper communities and the 2.4 meter resolution of imagery. The distribution of Pinyon-Juniper is characterized by single or small groups of trees surrounded by areas of canopy interspace. Therefore, pixels rarely were composed of pure Pinyon-Juniper canopies but were more often composed partially of Pinyon-Juniper canopies and bare soil resulting in pixel values similar to those of the mixed grass-forb class. Deciduous canopies were primarily composed of Gambel Oak stands which often formed larger more continuous stands resulting in fewer mixed pixels. The classification error between the mixed grass/forb and coniferous landcover classes may be improved by using a third image for further phenological separation, employing a finer spatial resolution, increasing spectral resolution, or using more sophisticated object-oriented remote sensing software.

Cheatgrass Cover Linear Regression Model

The second pass technique allows managers to map continuous cheatgrass cover using line-point intercept cover data and the remotely sensed ΔNDVI as inputs into a simple linear regression model. The regression model is developed for use within heterogeneous grass-forb communities because the spectro-phenological signature of cheatgrass is attenuated by the phenological patterns of other species. Increases in intra-pixel grass-forb diversity decreases the ability of “hard” classification algorithms like MLH to delineate the spectro-phenological patterns of individual species including cheatgrass. A regression model provides a way to predict how much cheatgrass cover is contributing to the overall spectro-phenological signal within a given pixel. The regression model is tuned using line-point intercept cover data gathered locally and is inherently ecosystem specific. In other words, the resulting linear regression model from this study may not be directly transferable to another ecosystem due to variations in soil brightness and vegetation composition. However, this methodology can be tuned to other project areas by coupling ground data and imagery collected locally.

When creating a regression model using both ground data and remotely sensed data the consideration of plot location, plot size (i.e. macroplot size) and subsampling intensity is imperative. The macroplots should be representative of the cheatgrass cover continuum in order to provide an adequate sample of cheatgrass conditions. Sampling macroplots with subtle differences in cheatgrass cover will likely result in poor model performance. However, stratifying a sampling campaign based on cheatgrass cover is not necessarily being advocated. Theoretically, random sampling techniques will capture the cheatgrass cover continuum given enough samples. If random sampling fails to capture the perceived range of variability than “key areas” may be established and used as inputs into the regression model.

The spatial resolution of the imagery and the degree of subsampling intensity should be considered in the determination of the appropriate plot size. The plot should be sufficiently large as to encompass several pixels. It is important to obtain a sample of ΔNDVI pixels because their average value is related to the average cheatgrass cover value collected using the line-point intercept method. Conversely, the macroplot needs to be small enough so that it can be adequately subsampled on the ground. It is important to minimize “within plot” variation so that the mean value input into the model is an accurate representation of the ground condition. The subsampling analysis shows that subsampling intensity within each macroplot can be reduced to between three and seven transects while still maintaining reasonable model performance (i.e. R^2 values). In other words, sampling 10 transects/macroplot appears to be an inefficient use of both time and money. Land managers must evaluate the desired model accuracy against the resources available for ground sampling. In some cases, sampling three transects/macroplot may provide adequate model results.

Treatment Effectiveness and Vegetative Recovery

Treatment effectiveness is considered low during the growing season of 2004 based on the pilot study and photography. Seeded species exhibited low cover and abundance and were difficult to quantify using only the line-point intercept cover method. Cheatgrass was prevalent with high cover values in many areas. However, the following is a brief discussion of the observed role of annual *Chenopodium* species and perennial *Sphaeralcea* species. The discussion is based on observations and more research is needed to better understand the relationships and mechanisms involved

The invasion and dominance of cheatgrass was attenuated in more mesic areas by the presence of native annual forb species like Desert Goosefoot (*Chenopodium pratericola)*, Fremont's Goosefoot (*Chenopodium fremontii*, and Mapleleaf Goosefoot (*Chenopodium simplex).* These forb species were prolific in and around the floodplain areas often growing in mulch (Fig. 17). Cheatgrass cover was very low in areas dominated by *Chenopodium* species suggesting a positive competitive advantage although no research on the topic could be found.

In terms of ESR objectives, author speculation suggests that these naturally occurring *Chenopodium* forb species appear to fill a key primary successional niche in this ecosystem by quickly providing extensive annual groundcover and competition against cheatgrass. Ecological intuition suggests, based on their post-fire prevalence, that Chenopods are able to compete effectively for light and water resources. The maximum

Figure 17. Annual Chenopod species establishing in hydromulch (2004). .

rooting depth of these Chenopods is similar to that of cheatgrass indicating similar access to water and nutrients (Allen and Knight 1984). Additionally, Desert Goosefoot has a phenological pattern similar to Russian Thistle (*Salsola kali*) characterized by warm season flower and seed production (Allen and Knight 1984). This temporal resource partitioning may provide an advantage, similar to Sand Dropseed, allowing these Chenopods to grow within cheatgrass invaded communities. A related species, Sandhill Goosefoot (*Chenopodium cycloides*), is considered an early successional species common in sandy soils adjacent to "blowouts" (Ladyman 2006). "Blowout" is a term for an unvegetated saucer- or trough-shaped depression formed by wind erosion on a sand deposit (Bates and Jackson 1984). Areas of loose unconsolidated sandy deposition denuded by fire and scoured by both wind and water provide habitat similar to "blowouts" within the RFC. Also, where established their rapid and tall broadleaf growth form may significantly limit resource availability to cheatgrass seedlings germinating in the understory. The prevalence of annual Chenopod species diminished by 2005 and field observations show that they were replaced by other preferred species except on more xeric sites where cheatgrass became established. Populations of these species appear to be short-lived early seral cheatgrass competitors that may reserve a niche for successional transitions towards perennial grasses and forbs given the right climatic conditions.

Smallflowered globemallow (*Sphaeralcea parvifolia)* was also very prevalent in both 2004 and 2005 occurring in extensive areas outside the floodplains and co-dominated sites with cheatgrass. Globemallow species are perennial, cool season forbs growing best in open and disturbed sites on sandy- to clay-loam soils in 200-350 mm precipitation zones (Pendery and Rumbaugh 1993). Studies have shown that globemallow species resprout from the root crown or from rhizomes following disturbances (Jaynes and Harper 1978, Pendery and Rumbaugh 1993) including fire (Pendery and Rumbaugh 1993). Data from this study show that cheatgrass cover was lower in areas where smallflowered globemallow was prevalent. Gooseberryleaf globemallow (*Sphaeralcea grossulariifolia*), a related species, has been used to suppress cheatgrass and other annuals (Stevens et al. 1985). The competitive pressure from smallflowered globemallow appears to have limited cheatgrass in some areas but to an extent less than that of the annual Chenopod forbs. Although smallflowered globemallow is also a broad-leaved forb species its canopies provide less cover allowing more light and water resources to reach the ground. Little research exists to support or refute this speculation. Populations of this species appear to be longer lived early seral species providing moderate levels of competition against cheatgrass.

Treatment effectiveness during the growing season of 2005 is considered to be moderate. The above average precipitation resulted in significant germination, establishment and growth of some of the seeded species. A discussion of treatment effectiveness and overall vegetative recovery for the 2005 growing season follows below.

Treatment 1–Upland Seeding. Treatment 1 is considered a partial success based on the remote sensing analysis and the qualitative monitoring which included species lists, site descriptions, photos, and site reconnaissance. The 1st pass analysis indicates that the mixed grass-forb class was distributed over 16% of treatment area. The species composition of the mixed grass-forb community can be interpreted from ground observations, which in upland areas was predominantly seeded species (Fig. 18). Although only four photopoints were established to monitor upland areas, site reconnaissance in other areas indicates that the mixed-grass forb areas in the uplands can be characterized by these photopoints. The ground monitoring showed that seeded grasses and forbs exhibited good distribution and vigor occurring on 100% and 75% of sites respectively. Additionally, the 2nd pass analysis indicates that 90% of the mixed grass-forb landcover type exhibited cheatgrass cover of less than 25% while cheatgrass monocultures occurred on only 1% of the treatment area. While treatment 1 was only effective in establishing seeded grasses and forbs on 16% of

Figure 18. Upland Photopoint 1.

the treatment area the expansion of cheatgrass was not prolific.

The shrub component consisting of Antelope Bitterbrush (*Purshia tridentata)* appears to be completely unsuccessful as it was not detected at any of the photopoints or during field reconnaissance. The absence of seeded shrub species may be due to inadequate site conditions for germination, inability of the seed to reach a safe site by aerial seeding, competition from herbaceous species or the ecological timeline on upland sites may be longer than three years for germination and establishment (Lambert 2005). In California, seeded shrubs emerged no earlier than natural regeneration and seeded grasses appeared to inhibit the growth of native shrubs and forbs (Robichaud et al. 2000). Antelope bitterbrush was observed, albeit rarely, in the bottomland areas indicating that it can establish in the deeper bottomland soils of the Book Cliffs but still may not be the best choice for either short-term stabilization or rehabilitation objectives.

Natural vegetative recovery or fire survival occurred within 24% of the treatment area in coniferous and broadleaf landcover cover types. The coniferous canopies appear to be primarily Pinyon-Juniper woodlands that survived the fire. Broadleaf deciduous canopies typically consist of Gambel Oak stands and isolated Box Elder and Fremont Cottonwood.

The remaining 61% of the treatment area consisted of bare soil and skeleton forest. Approximately 57% of the bare soil occurred on warm dry slopes typically low in herbaceous cover. The ground study suggests that seeded grass and forb species were present on all aspects but drier aspects had decreased cover. Treatment effectiveness and natural vegetative recovery on these aspects should be expected to be lower. Therefore, if these warm dry areas were removed from treatment effectiveness "equation" than the effectiveness of treatment increases. Skeleton forest represents areas where there does not appear to be a detectable treatment or natural recovery effect. The spectral signature of charred timber may have masked the spectral signal of any underlying herbaceous vegetation.

Overall vegetative recovery and the influence of the upland seeding on post-fire succession are largely based upon the existing pre-fire vegetative community. ReGAP vegetation data (USGS 2005) show that Treatment 1 spanned 17 vegetative communities (Table 10). These 17 vegetative communities, among others, were lumped into a smaller more manageable subset of vegetative communities during the development of the Normal Year Fire Rehabilitation Plan (NFRP) for the Moab Fire District. Table 11 shows the NFRP groupings that were treated in 2002. Pinyon and Juniper Woodlands, Mountain Shrub (i.e. Rocky Mountain Gambel Oak-Mixed Shrubland), and Douglas Fir/Mixed Conifer/Aspen were the three dominant vegetative communities. A brief discussion about the levels of natural vegetative recovery and the treatment effect for each of these community types is included below.

Pinyon-Juniper Woodland comprised the largest pre-fire vegetative community treated at 4,678 ha. The 1st pass analysis shows coniferous tree canopies consisted of 1,069 ha (13%) of the treatment area and represents areas where determining treatment effectiveness is difficult but indicates survival or natural vegetative recovery. BLM GIS fire history data and paper records show that Pinyon-Juniper woodlands located in the RFC were frequently burned by both natural and man-made fires. Until 1951, grazing permitees were allowed to burn areas in the Book Cliffs to maintain grass and forb abundance for cattle and sheep grazing (Ed Maloney, personal correspondence). The relatively high fire frequency in Pinyon-Juniper forests of the Book Cliffs resulted in age classes presumed to be approximately 55 and 100 years old in many areas. Old growth

Figure 19. Box elder and oak skeleton forest on 04/18/04.

pinyon-juniper stands do not appear to be common in the Book Cliffs and therefore overall vegetative recovery to pre-fire conditions was based on younger age classes.

A model of succession for Pinyon-Juniper Woodlands in southwestern Colorado progresses from skeleton forest and bare ground, to annual stage, to perennial grass-forb stage, to shrub stage, to shrub-open tree stage, to climax woodland (Brown and Smith 2000). The abundance of seeded grass and forb species from the 2002 seeding indicate that the treatment was successful in establishing a more dominant, vigorous and diverse perennial grass-forb stage after three years than might otherwise occur.

There has been some natural establishment of shrubs but the transition into a well developed shrub stage is expected to occur within next 5-15 years. Tree reestablishment

Figure 20. Box elder and oak skeleton forest on 06/04/04.

will occur slowly (Erdman 1970) through the introduction of juniper berries and pinyon cones (Floyd et al. 2000) by gravity or animal vectors (Bradley et al. 1991). Large burned patches are expected to colonize slowly from the outside in as seed sources are distant from the interior. The progression to well developed climax woodland similar to pre-fire conditions may take from 46-71 years (Barney and Frischknecht 1974) depending on the aspect, proximity to surviving trees, regional climatic patterns in the future.

The Mountain Shrub (i.e. Rocky Mountain Gambel Oak-Mixed Shrubland) was the second largest treated vegetative community occurring on 2,454 ha. Gambel Oak stands resprouted from root crowns (Engle et al. 1983) within days of the fire containment and had shown considerable foliar regrowth by the end of the 2005 growing season. Stands are typically dense with little to no grass-forb understory beneath the overstory canopy although grasses and forbs have established in the small open spaces between oak canopies. Figures 19 and 20 show a two-date photo series of Box Elder (*Acer negundo*) in the foreground and Gambel Oak (*Quercus gambelii*) in the background. Although figure 19 was taken in 2004 it shows what the post-fire skeleton forest looks like. Figure 20 shows the considerable resprouting that occurred by the early part of the 2004 growing season. Gambel Oak (*Quercus gambelii*) communities are extremely resilient to fire due to there extensive rhizomatous root system which stabilize the soil and their ability to resprout quickly. The 1st pass analysis shows broadleaf tree canopies, which are primarily comprised of Gambel Oak stands, occurred on 892 ha (11%) of the treatment area. The broadleaf landcover represents areas that have survived the fire or have recovered naturally but determining treatment effectiveness underneath broadleaf canopies using remote sensing is difficult. These communities should generally be considered a low priority for seeding treatments because of their fire resiliency and their competitive exclusion of other vegetation.

The Douglas Fir/Mixed Conifer/Aspen comprised the third largest treated vegetative community at 1,244 ha. Post-fire germination and establishment of Douglas Fir after severe wildfire will typically rely on wind-dispersed seeds reaching a safe site with bare mineral soil and optimal moisture conditions (Steinberg 2002). Seed bearing cones usually travel only a few hundred yards from the source (Shearer 1981). There are pockets of Douglas Fir that have survived in unburned or low burn severity areas which will provide a seed source for regeneration. However, speculation suggests areas where moderate and high burn severities occurred that are more remote from seed trees may see minimal conifer regeneration for many years. Aspen stands burned in the RFC are expected to resprout quickly from the extensive root system that typically remains after fire (Howard 1996). Aspen may be more prolific in some areas as the post-fire

Table 10. Pre-fire ReGAP Vegetative Communities Located Within Treatment 1 Boundary

Description	Hectares
Colorado Plateau Pinyon-Juniper Woodland	4,381
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	2,459
Rocky Mountain Montane Dry-Mesic Mixed Conifer Forest and Woodland	696
Rocky Mountain Montane Mesic Mixed Conifer Forest and Woodland	338
Colorado Plateau Pinyon-Juniper Shrubland	290
Rocky Mountain Aspen Forest and Woodland	192
Inter-Mountain Basins Big Sagebrush Shrubland	149
Inter-Mountain Basins Montane Sagebrush Steppe	123
Rocky Mountain Lower Montane Riparian Woodland and Shrubland	74
Inter-Mountain Basins Greasewood Flat	60
Rocky Mountain Alpine-Montane Wet Meadow	32
Inter-Mountain West Aspen-Mixed Conifer Forest and Woodland Complex	18
Rocky Mountain Cliff and Canyon	7
Colorado Plateau Mixed Bedrock Canyon and Tableland	7
Southern Rocky Mountain Montane-Subalpine Grassland	3
Inter-Mountain Basins Mixed Salt Desert Scrub	1
Rocky Mountain Subalpine Mesic Spruce-Fir Forest and Woodland	1
Total Acres	8,830

Table 11. Pre-fire NFRP Vegetative Groupings Located Within Treatment 1 Boundary.

Description	Hectares
Pinyon and Juniper Woodland	4,678
Mountain Shrub	2,459
Douglas Fir/Mixed Conifer/Aspen	1,244
Sagebrush	272
Riparian Wetland	106
Salt Desert Scrub/Shrub	61
Insignificant Vegetation Type[1]	11
Total Acres	8,830

[1]Rocky Mountain Cliff & Canyon

competition from coniferous species is reduced (Howard 1996).

Skeleton forests comprised 9.9% of the total treatment area and represent areas where the establishment of seeded species is difficult to discern. Skeleton forests are areas where the dominant spectral signature is derived from charred standing timber snags. These areas appear to be lacking significant natural vegetative recovery although no ground data was collected.

Treatment 2–Mycorrhizal Bottomland Seeding and Overlap of Treatment 1. Treatment 2 was effective in establishing seeded grass species and minimizing cover of *Bromus tectorum* but was ineffective in establishing seeded forbs and shrubs. The 1st pass analysis indicates that the mixed grass-forb communities were distributed over 37% of the treatment area while cheatgrass monocultures covered only 4.2%. The species composition of the mixed grass-forb class can be interpreted from ground observations which indicate that both cheatgrass and seeded species are common. Seeded grasses, preferred grasses, and preferred forbs exhibited a high frequency with variable relative cover. The 2nd pass analysis shows that cheatgrass has been limited to 0-24.999% cover on 83.7% of this treatment area. Moderate cheatgrass cover of 25-49.999% occurred on 14% of the area while high cheatgrass cover greater than 50% occurred on 2.3% of the treatment area. The treatment effectiveness and overall vegetative recovery of the treatment area is acceptable based upon an early seral grass-forb dominated ecological model.

The ground analysis shows that Treatment 2 was successful in meeting the frequency objective of 50% for seeded grasses with a mean of 64% despite the lower bound of the confidence interval dropping below 40%. However, since the majority of the 90% confidence interval, including the mean, lies well above the threshold it is considered a success. The mean relative cover of seeded grasses was 30% ±16.5%. Meeting the relative frequency objective for seeded grasses with a significantly lower relative cover value is noteworthy. There are several likely factors influencing this phenomenon. First, the line-point intercept method tended to underestimate the cover of narrow leaved plants (Bonham 1989) like the seeded bunchgrasses. Secondly, the wetter 2005 growing season resulted in increased germination of seeded grass species (Bissonette et al. 2006). While the frequency of seedlings is high their biomass and aerial cover are still low indicating that given another wet growing season frequency should stay static while cover values would increase. Thirdly, the mean value for the entire treatment incorporates data from sites on a soil moisture continuum. Variability in vegetation is typically high in post-fire ecosystems and can be explained by several factors including differences in microsite water characteristics, life stage characteristics, burn severity or species growth form.

The ground analysis corroborates the results from the remote sensing analysis and shows that the cover objective of 50% for minimizing *Bromus tectorum* cover was met. The mean value for the entire treatment shows that the relative cover of *Bromus tectorum* was limited to 23% + or – 18.3%. The upper end of the confidence interval was well below the target/threshold of 50%. The establishment of seeded grasses in conjunction with vigorous natural revegetation of preferred grasses and forbs was able to minimize the cover of *Bromus tectorum*.

In contrast, the frequency objective of 5% for seeded forbs was not met. The mean value for the entire treatment was 1% with upper confidence limits well below the 5% target/threshold. The relative cover of seeded forbs was less than 1% with very little variance. These seeded forbs were qualitatively observed in the bottomlands during sampling but with extremely low cover and frequency. The lack of germination and establishment cannot be attributed to a lack of available seeds because seeded forbs comprised 10.38% of the seed mix for *Achillea millefolium* and 2.93% for *Linum lewisii* based on the number of viable seeds (Appendix A). *Achillea millefolium* was observed frequently in the uplands indicating that aerial seeding can be successful for seeded forbs. One explanation is that these seeded forbs did not compete well with the abundance of other annual/perennial grasses and forbs present in the bottomlands.

The frequency objective of 10% for seeded shrubs was also not met and Treatment 2 and indicates the treatment was not effective with regard to this functional group. The mean value for the entire treatment was 3% with upper confidence limits well below the 10% target/threshold. One explanation for the lack of establishment of seeded shrubs is that the seeding rate was too low. *Cowania mexicana*, *Atriplex canescens, Purshia tridentata,* and *Cercocarpus ledifolius* comprised only 0.67% of the entire seed mix when evaluated by the number of viable seeds (Appendix A). A second hypothesis may be that the high absolute cover of vegetation (49%) in the bottomland areas competitively excludes the germination of shrubs. One study indicates that in a post-fire seeding treatment 30% cover of seeded ryegrass during the first year caused increased shrub seedling mortality (Beyers 2004). Ryegrass cover values of 55% reduced shrub seedling density to zero by the end of the first summer (Beyers 2004). A third possibility may be that seeded shrubs are operating on a longer ecological timeline for germination or establishment (Lambert 2005).

The frequency objective of 30% and cover objective of 20% for preferred grasses were both met as the mean frequency was 73% (±26.3%) and relative cover of 43% (± 23.9%). This indicates that an abundance of preferred grasses are present on the landscape. Additionally, the frequency objective of 30% and cover objective of 20% for preferred forbs was met with mean frequency values of 83% (± 8.6%) and relative cover values of 27 % (± 14.1%). This indicates an abundance of preferred forb species within the treatment area.

Preferred shrubs did not reach the objective of 10% frequency or 5% relative cover. The mean relative frequency was 5% (± 6.8%) and mean relative cover 3% (± 2.6%). There is some uncertainty as the upper confidence boundary of each metric is slightly above the target/threshold. However, the means and majority of the confidence intervals are below the target/threshold indicating that preferred shrub objectives were not met. Preferred shrubs like *Chrysothamnus nauseosus* and *Chrysothamnus viscidiflorus* have the ability to resprout quickly after fire if the buds located in the root crown are not damaged in the fire (Tirmenstein 1999). The treatment area was exposed to high burn severities which resulted in the almost complete consumption of above ground biomass but may have also increased the mortality of buds in the root crown. A high bud mortality would both minimize shrub regeneration from resprouting and increase the time of shrub recovery as regeneration becomes more reliant on off-site seed sources.

Furthering this line of speculation, this treatment area will likely be in a grass-forb dominated early successional stage for a longer period of time than areas of more moderate burn severities.

The 1st pass analysis also shows that 15.1% of treatment area consisted of bare soil with little difference between cooler and warmer aspects. The treatment area is characteristically a flat bottomland zone with shallow slopes and subtle differences in aspect. The bottomlands are dissected by stream channels and banks which represent a large component of this bare soil acreage. With the effect of extreme topography minimized in these bottomland areas the difference between cool and warm aspects is also minimized. The treatment was, therefore not effective in establishing seeded species on 19.3% of the treatment based on the assumption that bare soil and the presence of cheatgrass monocultures indicate failure. Broadleaf and coniferous canopies existed on 40.9% of the treatment area and represent areas where determining treatment effectiveness is difficult but fire survival and/or natural vegetative recovery was successful. Skeleton forest comprised the remaining 3.1% of the treatment area.

Treatment 3-Hydromulch Applied to Areas of Treatment 2. Treatment 3 was successful in establishing seeded grasses but was unsuccessful in establishing seeded forbs, seeded shrubs, and minimizing the proliferation of cheatgrass. The 1st pass analysis indicates that the mixed grass-forb communities were distributed over 48% of the treatment area while cheatgrass monocultures occur on 7%. The species composition of this class can be interpreted from ground observations which indicate that cheatgrass was dominant but seeded species were present. Seeded grasses, preferred grasses, preferred forbs, and preferred shrubs exhibited a high frequency with variable relative cover. The 2nd pass analysis shows that cheatgrass has been limited to 0-24.999% cover on 67.7% of the treatment area. However, moderate cheatgrass cover of 24-49.999% occurred on 26.3% of the area while high cheatgrass cover greater than 50% occurred on 6% of the treatment area. The treatment effectiveness and overall vegetative recovery of the treatment area is low based upon an early seral grass-forb dominated ecological model.

The frequency objective for seeded grasses of 50% was met with a mean frequency of 68% (± 22%) indicating that the objective was met for seeded grasses. Mean relative cover of seeded grasses was 14% (± 11.2%). The hypotheses explaining this high frequency and low cover phenomenon are the same as discussed above.

The ground study shows that the frequency objective for seeded forbs of 5% and shrubs 10% was not met. The mean frequency of seeded forbs was 1% with upper confidence limits well below the 5% target/threshold. The cover value for seeded forbs was less than 1% with very little variance. These seeded forbs were qualitatively observed in the bottomlands during sampling but with extremely low cover and frequency. The hypothesis for the lack of germination and establishment is similar to the explanation described in the Treatment 2 section although the competitive influence of higher cheatgrass infestation would likely make forb establishment more difficult. The frequency of seeded shrubs was 0% with no variance. No seeded shrubs were recorded in any of the individual macroplots. The explanation for this is described in the Treatment 2 section.

The ground study also corroborates the remote sensing analysis and shows that the cover objective of 50% for minimizing cheatgrass (*Bromus tectorum)* cover was not met. The mean relative cover of *Bromus tectorum* was 49% (± 15.1%). The mean is slightly below the target/threshold objective and confidence interval is evenly distributed on both sides.

The mean frequency of preferred grasses was 76% (± 13.5%) indicating the frequency objective of 30% was met and an abundance of preferred grass species are present on the landscape. Interestingly, the cover objective for preferred grasses of 20% was not met as the mean relative cover was 16% (± 10.4%) for the entire treatment. The mean and majority of the confidence interval fall well below the threshold of 20% indicating the failure to meet this objective. The area defined by treatment 3 has reached a sufficient level of vegetative recovery with respect to the abundance of preferred grass seedlings but biomass and cover are still lower than desired.

The mean value for the entire treatment shows that Frequency Objective 2 of 30% was also met for preferred forbs with statistical certainty. The mean frequency and 90% confidence interval for preferred forbs are 60% (± 33.1%). This indicates an abundance of preferred forb species within the treatment area. The Cover Objective 1 of 20% was not met for preferred forbs with a mean of 22% (± 14.2%). The mean is slightly above the target/threshold with the confidence interval fairly evenly distributed on either side. When examining cover data from individual macroplots the forb target was met by two macroplots and not met by two macroplots. Only one macroplot was statistically borderline. These data indicate the high abundance of preferred forbs with low levels of cover (i.e. small plants).

The preferred shrub target/thresholds of 10% defined in the frequency objective and 5% defined in the cover objective were met. The mean relative frequency was 15% (± 6.48%) and mean relative cover 10% (± 6.9%). There is slight uncertainty as the lower bounds of each metric's confidence interval are slightly below the target/threshold but there is enough certainty to assume the objectives have been met.

The 1st pass analysis shows that 27% of treatment area consisted of bare soil with little difference between cooler and warmer aspects. The treatment area is characteristically a flat bottomland zone with shallow slopes and subtle differences in aspect. The bottomlands are dissected by stream channels and banks which represent a large component of this bare soil acreage. However, treatment 3 has more bare soil areas outside of the stream channel than does treatment 2. With the effect of extreme topography minimized in these bottomland areas the difference between cool and warm aspects is also minimized. Therefore, the treatment was not effective in establishing seeded species on 33.7% of the treatment area assuming bare soil occurring on all aspects and the presence of cheatgrass monocultures indicate failure. Broadleaf and coniferous canopies existed on 17.8% of the treatment area and represent areas where determining treatment effectiveness is difficult but natural vegetative recovery was successful. Skeleton forest comprised the remaining 0.7% of the treatment area.

Mulch vs. No Mulch. Treatment 3 (mulch) did not have a positive influence on treatment effectiveness or success. A T-test did not show statistical differences in the cover or frequency of seeded grasses, forbs or shrubs although Treatment 2 (no-mulch) had 100% higher seeded grass relative cover values indicating potential ecological significance. Treatment 3 had a significantly higher cover and frequency of preferred shrubs and higher cheatgrass cover but also had a significantly lower cover of preferred grasses when compared against the no mulch treatment. The results of the linear regression model also show that Treatment 3 had nearly twice as much cheatgrass in the 25-49.999%, 50-74.999%, and 75-100% categories as Treatment 2. Two factors that may explain these patterns are differences in burn severity and the application of mulch.

The mulch treatment area in Diamond watershed suffered a moderate burn severity while the treatment area without mulch was characterized by high burn severities. While both moderate and high burn severities will top-kill most shrubs by eliminating above ground biomass the increased temperatures associated with higher burn severities often increases the mortality of buds at the root crown. *Chrysothamnus nauseosus* and *Chrysothamnus viscidiflorus* both resprout from the buds located at the root crown (Tirmenstein 1999) and are the most abundant components of the preferred shrub category. The increased shrub cover and frequency in the mulch treatment area are likely due to an increased survival of buds at the root crown resulting from the lower temperatures of the moderate burn severity. Regeneration of these rabbitbrush species in Treatment 2 (i.e. high burn severities) will likely occur more slowly by seed resulting in an extended early seral grass-forb stage.

The difference in burn severity is also a possible factor explaining the difference in cheatgrass cover between these treatments. Cheatgrass seeds are susceptible to heat kill and seed densities are typically higher on sites of lower burn severity (Zhouhar 2003) resulting in a higher potential for cheatgrass proliferation (Humphrey and Schupp 2001). The immediate post-fire densities of cheatgrass seed may have been higher in the mulch treatment area resulting in higher cheatgrass cover and increased competition against seeded/preferred grasses.

Several factors related to the application of the mulch may also have influenced the proliferation of cheatgrass. Germination of cheatgrass seedlings in dry environments requires that the seed must be covered by soil or litter (Evans and Young 1972) and establishment of cheatgrass seedlings is favored under high mulch conditions (Evans and Young 1970) up to two inches in depth (Harris and Goebel 1976). An untested alternate hypothesis suggests that the increased winter/spring soil moisture trapped by the layer of hydromulch may be depleted by early cheatgrass germination and growth. In this scenario, much of the additional soil moisture trapped by the layer of hydromulch is utilized and depleted by cheatgrass before native grass and forb species initiate growth. There is essentially a net increase in water availability for cheatgrass but little water for later growing cool season species. This may provide a competitive advantage to winter annuals that initiate growth early in the growing season. This effect will likely be increased in drought years like 2003 and 2004 and minimized in years of above average precipitation like 2005. While the hydromulch may have been effective in stabilizing seeds from the treatment the benefit may have been offset by creating more desirable conditions for cheatgrass germination and establishment within an area more likely to have higher densities of cheatgrass seed in the seedbank.

It is the opinion of this author that the increased cover of cheatgrass in the mulch treatment is a significant factor contributing to the difference in preferred grass cover. In Treatment 3 (mulch), 59% of the preferred grass frequency consisted of the seeded warm season grass Sand Dropseed (*Sporobolus cryptandrus*) with no other warm season grasses present. In Treatment 2 (no-mulch) only 2% of the preferred grass frequency was consisted of this grass. Ground observations show seedlings and juvenile Sand Dropseed (*Sporobolus cryptandrus*) growing through mats of senescent cheatgrass. The prevalence of this seeded warm season grass in the mulch treatment indicates that it has a competitive advantage in areas of higher cheatgrass cover. The competitive advantage is apparently gained through its phenological difference in growing season as it initiates growth under hotter and drier conditions when cheatgrass is senescent. Sand Dropseed (*Sand Dropseed*) has a growth form that typically has significantly less cover than many of the cool season grass occurring within the no mulch treatment. In this case many occurrences of this grass are seedlings or juvenile plants with lower cover than mature growth forms. The difference in preferred grass cover between treatments is largely due to the difference in species, growth form, and life stage.

Success of Seeded Grasses Species. The success of the seeding treatments is based upon the successful germination and establishment of the seeded grass species (Fig. 21). Germination and establishment varied greatly between species and location. An understanding of which species were the most successful is useful in terms of future seeding treatments in fire prone areas similar to the Book Cliffs.

Figure 21. High cover of seeded cool season grasses.

The species that had the highest frequencies across variable burn severities and treatments were *Elymus trachycaulus*, *Elymus lanceolatus*, *Pascopyrum smithii* and *Pascopyrum spicatum inermis*. These cool season species were marginally successful in the first two post-fire years most likely in response to persistent drought conditions. These species responded dramatically with the above average precipitation in 2005. *Elymus lanceolatus* and *Pascopyrum smithii* were present in large high frequency sod-forming patches by the end of the 2005 growing season. *Leymus cinereus* did not account for a large portion of the absolute vegetative frequency but was well established when the pilot study was initiated in 2004. The early success of this species under drought conditions, monsoonal scouring and strong competition from annual forbs and grasses is worth noting. These perennial grass species should be considered in future seeding treatments in similar ecosystems.

Seedlings of *Orhyzopsis hymenoides* and *Sporobolus cryptandrus* were prolific during the 2005 growing season. High densities of *Orhyzopsis hymenoides* seedlings were noticed on fresh alluvial deposits within the main channel and are not likely to

Figure 22. Vigorously growing western wheatgrass.

persist. Orhyzopsis hymenoides seedlings were also present in lower numbers in areas outside the channels. Persistence of these seedlings will depend on future climatic conditions and frequency of fire disturbance. The prevalence of *Sporobolus cryptandrus* seedlings growing in areas of high cheatgrass cover highlight the potential role of warm season grasses. The potential effectiveness of incorporating warm season grasses into ESR seed treatments should be further examined. *Hilaria jamesii*, *Pascopyrum spicatum spicatum* and *Sitanion hystrix* exhibited low levels of germination and establishment and should not be considered for future seeding treatments in the Book Cliffs.

Mycorrhizae. The mycorrhizal coating was applied to species known to development mycorrhizal relationships. It was believed that germination and establishment would increase by packaging the seed with mycorrhizal inoculum. It is impossible to quantify the effect of the mycorrhizal coating on germination and establishment without the proper control treatments or sites. There were no areas where the same seed mix was applied without the mycorrhizal component. However, some ground observations support a possible effect on the establishment and vigor of *Pascopyrum smithii*. In the DCNM08 macroplot *Pascopyrum smithii* was observed growing in strong sod-forming monoculture with heights approximately 12 inches taller than typically observed in the region (Fig. 22). It is possible that the mycorrhizal treatment has influenced this phenomenon. However, these monitoring studies cannot provide conclusive evidence indicating either success or failure of the mycorrhizal coating. Establishing control treatments and sites prior to treatment application is necessary to evaluate the success or failure of mycorrhizal coatings.

Inferences to Cottonwood Canyon. The levels of treatment effectiveness and overall vegetative recovery of the adjacent Cottonwood watershed are expected to be very similar to that of the Diamond watershed. Extending this ecological inference to Cottonwood is considered reasonable based on the rationale presented earlier.

The seeding treatments on the uplands of the Cottonwood watershed are expected to be successful in establishing the same vigorous perennial grass community present on the cooler upland aspects in Diamond watershed. Seeded forbs are expected to be moderately abundant with *Achillea millefolium* being more abundant than *Linum lewisii*. Native forbs species (i.e. asters, penstemon, etc.) are also expected to be abundant. Seeded shrubs are not expected to have germinated or established to any significant degree. Drier upland aspects are expected to have a similar composition and less abundant distribution of both the seeded and native species. The overall vegetative recovery of the upland areas in Cottonwood watershed are expected to be similar to the recovery in Diamond as described for the Pinyon-Juniper Woodland, Mountain Shrub and Douglas Fir/Mixed Conifer/Aspen.

The effectiveness of the seeding treatments and overall vegetative recovery in the bottomland areas are expected to by largely dependent on the burn severity. Areas of moderate burn severity are likely to have higher cover and frequency of preferred shrubs but little germination and establishment of seeded shrubs or forbs. Cheatgrass will have generally higher cover and seeded grass species are expected to exhibit low cover but relatively high frequencies. The warm season grass, *Sporobolus cryptandrus*, is expected to comprise the majority of seeded grasses present. Areas of higher burn severity are expected to have lower cover and frequency of preferred shrubs. Germination and establishment of seeded forbs and shrubs is expected to be minimal. Cheatgrass cover will likely be lower as seeded and native grass cover is expected to be higher. Seeded grass species will predominately consist of the cool season grasses, *Pascopyrum smithii*, *Elymus lanceolatus*, *Elymus trachycaulus*, *Pascopyrum spicatum inermis* and *Leymus cinereus*. Bottomland areas of high burn severity are expected to exhibit higher cover of preferred grasses.

CONCLUSIONS

The results of this project show that the ESR seeding treatments applied to the RFC were partially successful in establishing seeded species and minimizing the proliferation of cheatgrass. While the establishment of seeded species was not directly quantified using remote sensing, the location of grass-forb communities was mapped and the species composition was derived from the cheatgrass cover linear regression model and ground data. Treatment effectiveness was higher in Treatments 1 and 2 as shown by lower cheatgrass cover values within the mixed grass-forb landcover class. In other words, where grasses and forbs did become established within these treatments, the composition of cheatgrass was low while the composition of seeded species was high. However, Treatment 1 was not able to establish seeded grasses and forbs across the majority of the treated area. Interestingly, the application of hydromulch (i.e. Treatment 3) did not provide a significant benefit and may have provided a competitive advantage for cheatgrass. The results from this remote sensing study are in agreement with the results from the ground study.

In hindsight, the treatment effectiveness target/threshold objectives were reasonable except for the objectives set for seeded shrubs. Research shows that seeded shrubs are slower to germinate and establish then are seeded grasses and forbs. As previously mentioned, seeded grasses and forbs may actually inhibit the growth of seeded shrubs in early post-fire conditions. If used, seeded shrub objectives should be evaluated on a timeframe longer than three years.

This project utilized high spatial resolution Quickbird imagery and ground data to monitor treatment effectiveness and vegetative recovery within the RFC ESR project area and shows that remote sensing and statistical modeling can significantly improve knowledge regarding ESR treatment effectiveness when combined with traditional ground monitoring methods. The image acquisition cost and labor investment may be prohibitive making this approach feasible only on large priority projects. The methodology above arguably represents the simplest approach from both a remote sensing and statistical modeling approach and was accomplished using software currently available within the Bureau of Land Management computer network. It is unlikely that current technology can provide a cheaper or simpler alternative. Testing of this methodology on other projects will provide better insight into its utility and transferability.

REFERENCES

Allen, E. B., and D. H. Knight. 1984. The effects of introduced annuals on secondary succession in sagebrush-grassland, Wyoming. The Southwestern Naturalist **29**:407-421.

Allen, M. F. 1991. The Ecology of Mycorrhizae. Cambridge University Press, New York, NY.

Asrar, G., R. B. Myneni, and B. J. Choudhury. 1992. Spatial heterogeneity in vegetation canopies and remote sensing of absorbed photosynthetically active radiation: a modeling study. Remote Sensing of Environment **41**:85-103.

BAER Report Rattle Fire Complex. 2002. Rattle Fire Complex: burned area emergency stabilization and rehabilitation plan. Bureau of Land Management, Moab, UT.

Baret, F., and G. Guyot. 1991. Potentials and limits of vegetation indices for LAI and APAR assessment. Remote Sensing of Environment **35**:161-173.

Barney, M. A., and N. C. Frischknecht. 1974. Vegetation changes following fire in the pinyon-juniper type of West-central Utah. Journal of Range Management **27**:91-96.

Bates, R. L., and J. A. Jackson. 1984. Dictionary of Geological Terms, 3rd edition. Anchor Books - Doubleday, New York, NY.

Berube, M. S. 1982. The American Heritage Dictionary. Houghton Mifflin, Boston.

Beyers, J. L. 2004. Postfire seeding for erosion control: effectiveness and impacts on native plant communities. Conservation Biology **18**:947-956.

Billings, W. D. 1948. Preliminary notes on fire succession in the sagebrush zone of Western Nevada. Bulletin of the Ecological Society of America **29**.

Bissonette, G. J., A. A. Aubry, and B. K. Keating. 2006. Rattle Fire Complex emergency stabilization and rehabilitation final monitoring report. Bureau of Land Management, Moab, UT.

BLM. 1996. Sampling vegetation attributes. BLM/RS/ST-96/002+1730, U.S. Department of the Interior, Bureau of Land Management, National Applied Sciences Center, Denver, CO.

Bonham, C. D. 1989. Measurements for terrestrial vegetation. Wiley-Interscience, Malden MA.

Bradley, A. F., N. V. Noste, and W. C. Fischer. 1991. Fire ecology of forests and woodlands in Utah. Gen. Tech. Rep. INT-287, U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT.

Bradley, B. A., and J. F. Mustard. 2005. Identifying land cover variability distinct from land cover change: Cheatgrass in the Great Basin. Remote Sensing of Environment 94:204-213.

Brown, J. K., and J. K. Smith. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Ref. RMRS-GTR-42-Vol.2, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT.

Chavez, P. S. 1996. Image-based atmospheric corrections - revisited and improved. Photogrammetric Engineering and Remote Sensing **62**:1025-1036.

Congalton, R. G., and K. Green. 1999. Assessing the accuracy of remotely sensed data: principles and practices. Lewis Publishers, Boca Raton, FL.

Dall'Olmo, G., and A. Karnieli. 2002. Monitoring phenological cycles of desert ecosystems using NDVI and LST data derived from NOAA-AVHRR imagery. International Journal of Remote Sensing **23**:4055-4071.

Ehrlich, D., J. E. Estes, and A. Singh. 1994. Review article: applications of NOAA-AVHRR 1 km data for environmental monitoring. International Journal of Remote Sensing **15**:145-161.

Elzinga, C. L., D. W. Salzer, and J. W. Willoughby. 1998. Measuring and monitoring plant populations. USDI, BLM National Applied Resources Center, Denver, CO..

Engle, D. M., C. D. Bonham, and L. E. Bartel. 1983. Ecological characteristics and control of gambel oak. Journal of Range Management **36**:363-365.

Erdman, J. A. 1970. Pinyon-juniper succession after natural fires on residual soils of Mesa Verde, Colorado. Biological Series **11**:1-26.

Evans, R. A., and J. A. Young. 1970. Plant litter and establishment of alien annual weed species in rangeland communities. Weed Science **18**:697-703.

Evans, R. A., and J. A. Young. 1972. Microsite requirements for establishment of annual rangeland weeds. Weed Science **20**:350-356.

Floyd, M. L., W. H. Romme, and D. D. Hanna. 2000. Fire history and vegetation pattern in Mesa Verde National Park, Colorado, USA. Ecological Applications **10**:1666-1680.

Hansen, D. T. 1989. Soil survey of Grand County, Utah. Natural Resource Conservation Service, United States Department of Agriculture, Moab UT. Survey available online: http://soils.usda.gov/survey/printed_surveys/state.asp?state=Utah&abbr=UT

Harris, G. A. 1967. Some Competetive Relationships Between *Agropyron spicatum* and *Bromus tectorum*. Ecological Monographs **37**:89-111.

Harris, G. A., and C. J. Goebel. 1976. Factors of plant competition in seeing Pacific Northwest bunchgrass ranges. Washington State University, College of Agriculture Research Center, Pullman, WA.

Herrick, J. E., J. W. Van Zee, K. M. Havstad, L. M. Burkett, and W. G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. USDA - ARS Jornada Experimental Range, Las Cruces, NM.

Hobbs, T. J. 1995. The use of NOAA-AVHRR NDVI data to assess herbage production in the arid rangelands of Central Australia. International Journal of Remote Sensing **16**:1289-1302.

Howard, J. L. 1996. *Populus tremuloides*. *In* U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Lab, Missoula MT.

Humphrey, D. L., and E. W. Schupp. 2001. Seed banks of Bromus tectorum - dominated communities in the Great Basin. Western North American Naturalist **61**:85-92.

Hurlbert, L. C. 1955. Ecological studies of Bromus tectorum and other annual bromegrasses. Ecological Monographs **25**:181-213.

Interagency BAER Handbook. 2002. Interagency burned area emergency stabilization and rehabilitation handbook. Available online : www.fws.gov/fire/ifcc/esr/policy/es%20**handbook**%202-7-06.pdf

Jaynes, R. A., and K. T. Harper. 1978. Patterns of natural revegetation in arid southeastern Utah. Journal of Range Management **31**:407-411.

Jensen, J. R. 1996. Introductory Digital Image Processing: A Remote Sensing Perspective. Prentice Hall, Upper Saddle River, NJ.

Juenger, K. 2005. Normal year fire stabilization and rehabilitation plan. EA UT-060-2005-043, Bureau of Land Management, Moab Field Office, Moab, UT.

Justice, C. O., J. R. G. Townshend, B. N. Holben, and C. J. Tucker. 1985. Analysis of phenology of global vegetation using meteorological satellite data. International Journal of Remote Sensing **6**:1271-1318.

Krause, K. 2005. Radiometric use of QuickBird Imagery. DigitalGlobe, Longmont, Colorado.

Ladyman, J. A. R. 2006. *Chenopodium cycloides* A. Nelson (sandhill goosefoot): a technical convservation assessment. USDA Forest Service, Rocky Mountain Region, Centennial, CO.

Lambert, S. 2005. Guidebook to the seeds of native and non-native grasses, forbs and shrubs of the Great Basin. Technical Bulletin 2005-04, United States Department of the Interior, Bureau of Land Management, Idaho State Office, Boise, ID.

Lass, L. W., T. S. Prather, N. F. Glenn, K. T. Weber, J. T. Mundt, and J. Pettingill. 2005. A review of remote sensing of invasive weeds and example of early detection of spotted knapweed (Centaurea maculosa) and babysbreath (Gypsophila paniculata) with a hyperspectral sensor. Weed Science **53**:242-251.

Leprieur, C., Y. H. Kerr, S. Mastorchio, and J. C. Meunier. 2000. Monitoring vegetation cover across semi-arid regions: comparison of remote observations from various scales. International Journal of Remote Sensing **21**:281-300.

Lillesand, T., and R. Kiefer. 2002. Remote Sensing and Image Interpretation. John Wiley & Sons, Malden, MA.

Lowry, J. 2003. Image Standardization (At-Sensor Reflectance and COST Correction). *in*. Utah State University.

Ludwig, J. A., B. P. Wilcox, D. D. Breshears, D. J. Tongway, and A. C. Imeson. 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology **86**:288-297.

Mack, R. N. 1981. Invasion of *Bromus tectorum* L. into Western North America: an ecological chronicle. Agro-Ecosystems **7**:145-165.

Mack, R. N., and D. A. Pyke. 1983. The demography of *Bromus tectorum*: variation in time and space. Journal of Ecology **71**:69-93.

McNab, W. H., and P. E. Avers. 1994. Ecological Subregions of the United States. USDA Forest Service. Available online: www.fs.fed.us/land/pubs/ecoregions/

Mooney, H. A., and E. E. Cleland. 2001. The evolutionary impact of invasive species. Procedures of National Academy of Science **98**:5446-5451.

Owen, T. W., T. N. Carlson, and R. R. Gillies. 1998. An assessment of satellite remotely-sensed land cover parameters in quantitatively describing the climatic effect of urbanization. International Journal of Remote Sensing **19**:1663-1681.

Pellant, M. 1996. Cheatgrass: The Invader that Won the West. Idaho State Office Bureau of Land Management, Boise, ID.

Pendery, B. M., and M. D. Rumbaugh. 1993. Globemallows. Rangelands **15**:127-130.

Peters, A. J., M. D. Eve, E. H. Holt, and W. G. Whitford. 1997. Analysis of desert plant community growth patterns with high temporal resolution satellite spectra. Journal of Applied Ecology **34**:418-432.

Peterson, E. B. 2005. Estimating cover of an invasive grass (Bromus tectorum) using tobit regression and phenology derived from two dates of Landsat ETM+ data. International Journal of Remote Sensing **26**:2491-2507.

Platt, K., and E. R. Jackman. 1946. The Cheatgrass Problem in Oregon. Oregon State College, Corvalis, OR.

Pritchett, W. L., and R. F. Fisher. 1987. Properties and Management of Forest Soils. John Wiley & Sons, NY.

Reid, K. D., B. P. Wilcox, D. D. Breshears, and L. MacDonald. 1999. Runoff and Erosion in a Pinon-Juniper Woodland: Influence of Vegetation Patches. Soil Science Society of America Journal **63**:1869-1879.

Robichaud, P. R., J. L. Beyers, and D. G. Neary. 2000. Evaluating the Effectiveness of Postfire Rehabilitation Treatments. RMRS-GTR-63, U.S. Department of Agricuture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Rouse, J. W., R. H. Haas, J. A. Schell, and D. W. Deering. 1973. Monitoring vegetation systems in the great plains with ERTS: Third ERTS symposium. Pages 309-317 *in* NASA, editor. NASA SP-351.

Scanlan, J. C., A. J. Pressland, and D. J. Myles. 1996. Runoff and soil movement on mid-slopes in North-east Queensland grazed woodlands. Rangeland Journal **18**:33-46.

Schmidt, H., and A. Karnieli. 2002. Analysis of the temporal and spatial vegetation patterns in a semi-arid environment observed by NOAA AVHRR imagery and spectral ground measurements. International Journal of Remote Sensing **23**:3971-3990.

Shearer, R. C. 1981. Clearcutting and Fire in Larch/Dougals-fir forests of Western Montana - A Multifaceted Research Summary. Gen. Tech. Rep. INT-99, U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Shepard, E. 2007. ESR Handbook. H-1742-1, Bureau of Land Management, Washington D.C.

Smith, C., and N. Brown. 1999. Erdas Field Guide, 5th edition. Erdas, Altlanta, GA.

Steinberg, P. D. 2002. Pseudotsuga menziesii var. glauca. Pages *in*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Fire Sciences Lab, Missoula MT.

Stevens, R., N. Shaw, and C. G. Howard. 1985. Important nonleguminous forbs for intermountain ranges. Range Plant Improvement in Western North America: Proceedings, Annual Meeting of the Society for Range Management:102-112.

Tirmenstein, D. 1999. Chrysothamnus nauseosus. Pages *in*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Lab., Missoula MT.

Tucker, C. J. 1985. African land-cover classification using satellite data. Science **227**:369-375.

Tucker, C. J. 1986. Maximum normalized difference vegetation index images for sub-Saharan Africa for 1983-1985. International Journal of Remote Sensing **7**:1383-1384.

USGS. 2005. Southwest regional gap analysis project stewardship digital data. Data housed at. RS/GIS Laboratory, College of Natural Resources, Utah State University, Logan, UT.

Viera, A. J., and J. M. Garrett. 2005. Understanding interobserver agreement: the kappa statistic. Family Medicine **37**:360-363.

Walker, B. H. 1970. An evaluation of eight methods of botanical analysis on grasslands in Rhodesia. Journal of Applied Ecology **7**:403-416.

Wilcox, B. P., and D. D. Breshears. 1995. Hydrology and Ecology of Pinyon-Juniper Woodlands: Conceptual Framework and Field Studies. Pages 109-119 *in* Desired Future Conditions for Pinyon-Juniper Ecosystems. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Flagstaff, AZ.

Wirth, T. A., and D. A. Pyke. 2007. Monitoring post-fire vegetation rehabilitation projects- a common approach for non-forested ecosystems. 2006-5048, U.S. Geological Survey Scientific Investigations Report. Persistent URL: http://pubs.water.usgs.gov/sir20065048

Wu, J., D. Wang, and M. E. Bauer. 2005. Image-based atmospheric correction of Quickbird imagery of Minnesota cropland. Remote Sensing of Environment **99**:315-325.

Zhouhar, K. 2003. Bromus tectorum. *in*. U.S. Department of Agriculture, U.S. Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula Mt..

APPENDICES

APPENDIX A. Seeding Treatment Details. % of area in parentheses. Due to rounding, values shown do not total to 100%.

Common Name	Scientific Name	Abbreviation	2003 Mycorrhizal Coat	2002 (seeds/ft^2)	2002 PLS	Seed 2002 (viable seeds/ft^2)	2003 (seeds/ft^2)	2003 PLS	Seed 2003 (viable seeds/ft^2)	TRT 1 (viable seeds/ft^2)	TRT 2 & 3 (viable seeds/ft^2)
Slender Wheatgrass	*Elymus trachycaulus*	ELTR	N	89.03	0.9389	7.77	0.00	0.0000	0.00	7.77(29)	7.77 (12)
Western Yarrow	*Achillea millefolium*	ACMI	N	7.21	0.9414	6.79	0.00	0.0000	0.00	6.79(25)	6.79 (10)
Thickspike Wheatgrass	*Elymus lanceolatus*	ELLA	N	5.35	0.8467	4.53	0.00	0.0000	0.00	4.53(17)	4.53 (7)
Great Basin Wildrye	*Leymus cinereus*	LECI	Y	4.51	0.8452	3.81	1.28	0.9000	1.15	3.81(14)	4.96 (8)
Beardless Bluebunch Wheatgrass	*Pseudoreoegneria spicata ssp.* Inermis	PSSPIn	N	4.06	0.8130	3.30	0.00	0.0000	0.00	3.30(12)	3.30 (4)
Lewis Flax	*Linum lewisii*	LILE	Y	0.76	0.8945	0.68	1.44	0.8572	1.23	0.68 (3)	1.91 (3)
Antelope Bitterbrush	*Purshia tridentata*	PUTR	N	0.13	0.9229	0.12	0.00	0.0000	0.00	0.12(0)	0.12 (0)
Wyoming Big Sagebrush	*Artemisia tridenta ssp.* wyomingensis	ARTR	N	0.00	0.0000	0.00	26.24	0.1277	3.35	0.00	3.35 (5)
Sand Dropseed	*Sporobolus cryptandrus*	SPCR	N	0.00	0.0000	0.00	26.06	0.8790	22.91	0.00	22.91(35)
Bluebunch Wheatgrass	*Pseudoroegneria spicata* ssp. spicata	PSSPSn	Y	0.00	0.0000	0.00	3.21	0.8620	2.77	0.00	2.77 (4)
Indian Ricegrass	*Achnatherum hymenoides*	ORHY	Y	0.00	0.0000	0.00	2.77	0.9175	2.54	0.00	2.54 (4)
Western Wheatgrass	*Pascopyrum smithii*	PASM	Y	0.00	0.0000	0.00	2.16	0.9018	1.95	0.00	1.95 (3)
Four Wing Saltbush	*Atriplex canescens*	ATCA	N	0.00	0.0000	0.00	0.05	0.3242	0.02	0.00	0.02 (0)
Bottlebrush Squirreltail	*Sitanion hystrix*	SIHY	Y	0.00	0.0000	0.00	0.79	0.8771	0.69	0.00	0.69 (1)
Galleta Grass	*Pleuraphis jamesii*	HIJA	N	0.00	0.0000	0.00	2.31	0.6241	1.44	0.00	1.44 (2)
Curl-leaf Mountain Mahogany	*Cercocarpus ledifolius*	CELE	N	0.00	0.0000	0.00	0.25	0.9074	0.23	0.00	0.23 (0)
Cliffrose	*Cowania mexicana*	COME	N	0.00	0.0000	0.00	0.11	0.7228	0.08	0.00	0.08 (0)
TOTALS				**30.30**		**27.0**	**66.67**		**38.36**	**27.00**	**65.36**

APPENDIX B. 2004 Ground Monitoring Pilot Study

The 2004 pilot study included 22 individual 50-meter line-point intercept transects quantified using line-point intercept cover. Thirteen transects were established in the Diamond watershed and nine were established in the Cottonwood watershed (Table B1). Four control plots were established in both Diamond and Cottonwood canyons (Table B1). Data was collected between May 27 and July 22 in 2004. Measured response variables were plant composition and cover.

Transects were read once using a systematic (1/2 m intervals) line-point intercept method. A portable 10-point angled (15 º) laser point bar was used in place of a pin flag for intercept measurements. Repeat photography was initiated at each transect. Digital photos were taken looking down and back from the origin and end of each transect. Three additional photos were taken in orthogonal directions from origin (Appendix B). Results of the cover data are shown in Figure B1.

Table B1. Distribution of Transects in 2004 Pilot Study.

	# of Transects	
Treatment	Diamond	Cottonwood
Control	4	4
Treatment 1 (Upland)	4	4
Treatment 2 (No-Mulch)	4	4
Treatment 3 (Mulch)	1	1

Collecting FY2004 transect cover data within the RFC was time-consuming and logistically difficult due to monsoon rains, flash flooding, intense lightening storms and blown-out roads. As a result, cover data collection in FY2004 occurred over a 7 week period on the cusp of the cool/warm season which appears to have had a significant effect on the cover values. The mulch transect data (TRT 3) was collected early in the growing season when cover values from seedlings were very low. Conversely, the no-mulch (TRT 2) and control transects were read later in the season when cover values were more static but had been influenced by increased growth. The difference between treatments and controls is an artifact of the timing of the sampling and not the treatments themselves.

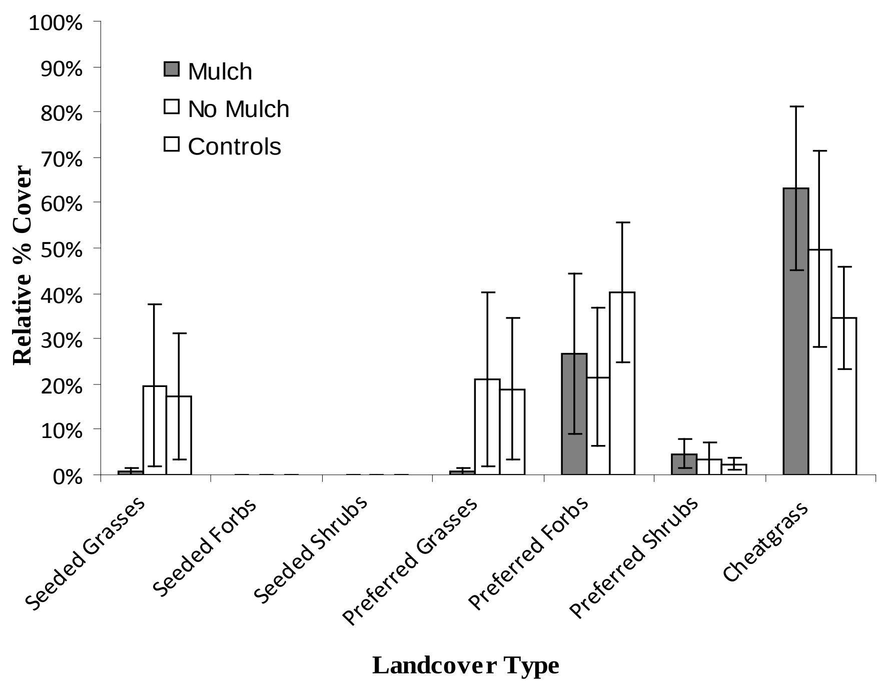

Figure B1. Relative Cover of Functional Groups from 2004 Pilot Study. Error bars show the 90% confidence interval.

The control plots established in the 2004 pilot study are not considered good controls. Several of these control plots were located in side canyons out of necessity but were subsequently blown out by monsoonal overland flooding. The remaining control plots were located on a state tract of land in Cottonwood canyon that was hypothetically left untreated. It showed a relatively high percentage of seeded species, some of which were not native to this area in the Book Cliffs indicating that these control areas were being influenced by the treatments. As a result, the 2004 pilot study was not used to determine treatment effectiveness or overall vegetative recovery. However, these data exhibit the same general trends that are evident in the 2005 data which provides additional validation for the conclusions. In particular, the cover of seeded grasses is higher and the cover of cheatgrass is lower in the no-mulch treatment. The FY2004 pilot study was used as an important exercise to determine the limitations of the sampling method and provide insight used to revise the monitoring plan for FY2005.

Several revisions were made to the 2004 pilot study before sampling began in 2005. During the pilot study it was observed that the line-point intercept method was underestimating the cover of the seeded grasses. The line-point intercept method does not work well when cover values are less than 15% (Bonham 1989). In other words, the seeded species were present in low abundance and cover but were not being adequately quantified by this method. Compounding this problem is the fact that the most accessible data acquisition window occurs during a dynamic part of the growing season. Cover values are highly susceptible to the phenological stage of the plant (Bonham 1989) which, as mentioned above, had a significant effect on the 2004 pilot study data. A method that was able to quantify rare species and was less susceptible to the phenological stage of the plant was needed.

Nested frequency was initiated in FY2005 along with the continued collection of cover data. Nested frequency is less susceptible to phenological stages and better able to quantify the presence of rare species. Cover data was still collected as it is more directly related to biomass and can be correlated with erosion potential. Cover data collection was initiated on July 1 during the warm season prior to the onset of monsoon precipitation when composition and cover were more static.

APPENDIX C. List of Preferred Species.

Preferred Grasses	Preferred Forbs	Preferred Shrubs	Non-Preferred/Weeds
Indian Ricegrass	Western Yarrow	Sagebrush[5]	Cheatgrass
Basin Wildrye	Lewis Flax	Antelope Bitterbrush	Descurainia spp.
Sand Dropseed	Globemallow[3]	Mountain Mahogany	Annual Forbs[7]
Galleta	Scarlet Globemallow	Cliffrose	Perennial Forbs[7]
Bluebunch Wheatgrass[1]	Globemallow[4]	Fourwing Saltbush	Annual Grasses[7]
Bluebunch Wheatgrass[2]	Chenopod spp.	Woods Rose	Perennial Grasses[7]
Bottlebrush Squirreltail	Wavy-leaf Thistle	Gray Rabbitbrush	Common Mullein
Slender Wheatgrass	Aster spp.	Green Rabbitbrush	Kochia (Fireweed)
Thickspike Wheatgrass	Primrose spp.	Skunkbush Sumac	Prickly Lettuce
Western Wheatgrass	Desert 4 O'clock	Elderberry	Opuntia spp.
Needleandthread grass	Stickseed	Utah Serviceberry	
Unknown Agropyron	Western Stoneseed	Snowberry[6]	
Foxtail Barley	Louisiana Wormwood	Chokecherry	
Carex spp.	Rock Clematis	Gardner's Saltbush	
Salina Wildrye	Veiny Dock	Broom Snakeweed	
Kentucky Bluegrass	Showy Milkweed		
Crested Wheatgrass	Coreopsis spp.		
	American Licorice		

[1] Inermis
[2] Spicatum
[3] Small Flowered
[4] Gooseberryleaf
[5] Wyoming Big
[6] Grey Mountain
[7] Unknown