AVIAN PROGRAM

SEPTEMBER 2012

WILDLIFE RESEARCH REPORT

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COLORADO DIVISION OF PARKS AND WILDLIFE
Research Center, 317 W. Prospect, Fort Collins, CO 80526

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Colorado Division of Parks and Wildlife
September 2011 – September 2012

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ABSTRACT

In 2005, the Colorado Division of Wildlife, now Colorado Parks and Wildlife (CPW), initiated a research project to evaluate the demography and movement patterns in two Gunnison sage-grouse (GUSG) populations. The objective is to develop population and landscape models that will be used in the development and refinement of management plans for GUSG and to update the population viability analysis used in the Gunnison Sage-grouse Rangewide Conservation Plan (2005). Acquiring more precise estimates of demographic rates (and their variances) will enable us to evaluate the relative importance of various environmental and demographic factors that potentially influence population abundance, dynamics and persistence of GUSG.

There are seven GUSG populations distributed across southwestern Colorado and southeastern Utah. Six of the populations are relatively small (with < 100 males counted on leks and < 100,000 acres of sagebrush habitat) compared to the Gunnison Basin (with 750-1,000 males counted on leks and > 500,000 acres of sagebrush habitat). We chose to contrast the demography and movement patterns of GUSG in one of the small populations in San Miguel County with the large population in Gunnison County. The population in San Miguel County is one of the more complex of the smaller populations with approximately six isolated population segments. The Miramonte population segment contains the largest population of GUSG outside of the Gunnison Basin. We currently lack information on the role of landscape features (e.g., the composition and fragmentation of sagebrush habitat) on the movement patterns and demography of GUSG. We completed the field work for this project in 2011. This report represents the progress on the data analysis for this project. We provide the Abstracts from the completed Ph.D. dissertation completed by Amy Davis in June 2012, along with commentaries containing additional information.

WILDLIFE RESEARCH REPORT

DEMOGRAPHY AND DISPERSAL OF GUNNISON SAGE-GROUSE (*Centrocercus minimus*)

MICHAEL PHILLIPS and AMY DAVIS

PROJECT OBJECTIVES

1. Acquire current estimates of nest success, survival of chicks, juveniles, yearlings, and adults) and estimates of variation (temporal and spatial) for two populations of Gunnison sage-grouse (GUSG).
2. Model nest survival using nest-site vegetation structure and various spatial and temporal factors (e.g., time of nest initiation, nest age, and population) as covariates.
3. Model chick, juvenile, yearling, and adult survival using life history and various spatial and temporal factors as covariates (e.g, age, sex, time of year, population, weather).
4. Record movement patterns of GUSG and use the information to develop landscape use, movement models and spatial models of demographic parameters.
5. Use the above estimates to update and refine a population viability analysis (PVA) developed for the GUSG Rangewide Conservation Plan (Gunnison Sage-grouse Rangewide Steering Committee, 2005) and to develop a landscape model and a spatially-explicit population model (SEPM) specific to GUSG.
6. Use estimates of demographic parameters and model output to develop and evaluate the projected consequences of alternative management plans.

SEGMENT OBJECTIVES

1. Compile and error check telemetry data in preparation for development of demography and landscape models.
2. Compile and error check nest and nest-site vegetation data in preparation for analysis of nest survival.
3. Compile an error check of chick, juvenile, yearling, and adult survival in preparation of survival analyses; update a PVA model; and development a SEPM for GUSG.
4. Complete analyses of the demographic data. Amy Davis will defend her doctoral dissertation at Colorado State University.
5. Collect and georeference GIS databases, and create appropriate location (i.e., telemetry locations and nest locations) databases to be used in the development of a landscape model and a spatially-explicit population model.

INTRODUCTION

Our ability to conserve Gunnison sage-grouse (GUSG) will depend on our ability to restore and manage a biologically relevant mosaic of habitats. Population viability analysis (PVA) and spatially explicit population models (SEPM) are two analytical tools increasingly used by conservation biologists to evaluate the relative effects of demographic rates and changing landscape structure on the viability of populations. Before such models can be constructed, there is a need for reliable estimates of demographic and behavioral data for GUSG.

A PVA was developed for the Gunnison Sage-grouse Rangewide Conservation Plan (GUSG RCP, 2005). The model required information that was either not available (e.g., chick survival) or not current (e.g., nest success). Acquiring more precise estimates of demographic rates (and their variances) will enable us to develop more reliable models of population viability, as well as evaluate the relative importance of various landscape and demographic factors that potentially influence population abundance, dynamics and persistence of GUSG. Development of relevant management strategies for GUSG will depend on modeling efforts derived from reliable estimates of demographic rates and behavioral patterns (e.g., movement and dispersal). Demographic models developed from more precise estimates will allow more realistic projections of persistence time and a more rigorous evaluation of the relative demographic and environmental factors influencing the viability of GUSG. Many standard models of population viability indicate that adult survival has the greatest impact on viability. However, the PVA developed for the GUSG RCP indicated that juvenile mortality and female productivity have significant impacts on population growth. Our intent is to evaluate the relative impacts of different demographic rates on viability. Depending on the outcome of these models, management actions should be developed that will directly influence these parameters. Furthermore, the population targets reported in the GUSG RCP may need to be revised if there are significant differences in species-specific demographic and behavioral data.

A SEPM will be a valuable tool that will allow researchers and managers to develop and evaluate alternative management plans specifically for GUSG. A SEPM will allow land managers to evaluate the relative merits of proposals for land acquisition and easements based on spatially explicit demographic and behavioral data (e.g., what is the potential for sage-grouse to use a specific land parcel and what effect may it have on the local population persistence). We will use the information collected in this project to evaluate how productivity, recruitment and movement patterns of GUSG may be a function of landscape features (e.g., the composition and configuration of sagebrush habitat, and other anthropogenic features).

The development of a PVA and a SEPM are valuable tools for evaluating the relative threats to GUSG. However, the models do not automatically indicate which management strategies will have the greatest impact on minimizing the threats. Therefore, species-specific data and modeling results acquired in this study will ultimately be used to develop a decision analysis approach to evaluating alternative management programs. Decision analysis is an analytical approach to evaluate the relative outcomes of alternative management actions. Using this approach, managers can assess alternative strategies by incorporating the probability of an event occurring, given a particular strategy, and the probabilities of several potential outcomes as a result of that event. In this manner, the consequences of management strategies can be evaluated more quantitatively than in a probabilistic framework.

The effect of the landscape on the demography and movement of GUSG may depend on the relative size of the population. The potential variation in demographic rates and movement patterns may have a greater impact on the viability of small populations in contrast to larger population by significantly reducing persistence time. We collected data from two GUSG populations of contrasting size. The Gunnison Basin has the largest GUSG population (750-1,000 males counted on leks in the last eight years) and the largest amount of relatively homogeneous sagebrush habitat (> 500,000 acres) of the seven GUSG populations. In contrast, the sagebrush communities in the other six populations are smaller (< 100,000 acres of sagebrush habitat), potentially more fragmented and contain fewer GUSG (< 100 males counted on leks). We selected the Miramonte population segment of the San Miguel Basin population. Among the small populations, it contains the largest population of GUSG outside of the Gunnison Basin population and therefore provided us the best opportunity of making a meaningful comparison with the Gunnison Basin population.

STUDY AREAS

There are seven GUSG populations distributed across southwestern Colorado and southeastern Utah. Six of the populations are relatively small (with < 100 males counted on leks and < 100,000 acres of sagebrush habitat) compared to the Gunnison Basin (with 750-1,000 males counted on leks and >500,000 acres of sagebrush). The Gunnison Basin is an intermontane basin that includes parts of

Gunnison and Saguache Counties in Colorado. Elevation ranges from 7,500-9,000 feet. Steep sloped mesas are scattered throughout the basin. Uplands are divided by permanent or intermittent drainages. Big sagebrush (*Artemisia tridentata* spp.) dominates upland vegetation. Habitat along major stream drainages has been converted to hay and pastureland.

The San Miguel population is located in Montrose and San Miguel Counties. Sagebrush habitat in the San Miguel population is one of the more complex with approximately six isolated communities (GUSG RCP). Elevation ranges from 6,300-9,000 feet. Habitat varies from a patchy big sagebrush (*Artemisia tridentata* spp.) distribution with sparse grass and forb understory in the Dry Creek community to more diverse sagebrush stands of big sagebrush (*Artemisia tridentata* spp.), low sagebrush (*Artemisia arbuscula*) and black sagebrush (*Artemisia nova*) and a more abundant grass and forb understory in the other San Miguel communities.

METHODS

Trapping and Radiomarking

Adult and yearlings: We focus our radiomarking efforts on females since two key demographic parameters of the current project are nest success and juvenile survival. GUSG were captured using spotlighting techniques (Giesen et al. 1982, Wakkinen et al. 1994). Spring trapping began mid-March and ended in early May. Fall trapping began in late August and continued into September, but avoided spotlighting during the fall hunting season. Spring spotlighting efforts were centered initially in areas near leks with increasing effort further away from leks. We did not search or trap on leks. The search effort during both periods was opportunistic. Areas where grouse are more likely to be located (e. g., open areas near sagebrush and along adjacent ridge tops) were searched more thoroughly.

Each captured individual was radiomarked. Using radiomarked individuals provides a valid, although intensive, method for estimating demographic parameters (White and Garrott 1990, White et al. 2002). We used necklace-mounted radio transmitters (Advanced Telemetry Systems or Holohil Systems, Inc.). The transmitters (17 g) were equipped with a four-hour mortality circuit and have a nominal battery life of 18 months. The transmitter weight is < 2% of the body weight of an adult female, and < 1% body weight of an adult male GUSG. An aluminum band (size 14 female; size 16 male, National Brand and Tag Company) was attached to each individual. We recorded body weight, age and sex of each individual (Crunden 1963, Dalke et al. 1963, Beck et al. 1967). We located females with broods six days each week, females without broods three to four times each week and males two to three times each week. All trapping and handling procedure followed the CPW Sage-grouse Trapping and Handling Protocol previously approved by the Animal Care and Use Committee (project # 02-2005).

Juveniles: We used radiotelemetry to estimate juvenile survival and movement patterns. The status of nests was monitored daily. Immediately after hatching, we located the brooding hen either early in the morning or at dusk (i.e., during periods when the juveniles are most likely to be closely brooded by the female), visually located juveniles, estimated brood size, then flushed the female off her brood and captured the brood by hand or with a lightweight hand-held mesh net. Trapping juveniles was not attempted in inclement weather (e.g., rain or snow) or during extreme cold temperatures (< 20° F). We randomly selected approximately half of the juveniles from the brood and fitted them with a 1.0 g transmitter with a nominal battery life of 18 days (Advanced Telemetry Systems). GUSG juveniles weighed 25-30 g at birth (Phillips, preliminary CPW data). The 1.0 g transmitters are < 5.0% body weight of a juvenile GUSG. Procedures for attaching light-weight transmitters to juveniles have been developed for juvenile greater sage-grouse (Aldridge 2000, Burkpile et al. 2002) and juvenile ruffed grouse (Larsen et al. 2001). The transmitter is attached to a juvenile by suturing it to the interscapular region of the juvenile. There were no reported mortalities or overt signs of stress for juveniles during attachment of transmitters to greater sage-grouse or ruffed grouse. Juveniles were handled as quickly as possible to minimize stress. The brood was released where they were originally captured. We estimated a Universal Transverse Mercator (UTM) location using triangulation for each radio-marked juvenile every two days.

We recaptured juveniles after 18 days to replace the 1.0 g transmitter with a 4.0 g transmitter with a nominal battery life of six months (Advanced Telemetry Systems) by suturing it to the interscapular region (as described above). A 4.0 g transmitter is < 3.0% body weight of an 18-day old GUSG (Phillips, preliminary CPW data). In the fall (August-September) we recaptured juveniles to attach an adult 17g transmitter if the juvenile weighs > 800 g and is in good condition. If the juvenile is < 800 g and in good condition then the 4.0 g transmitter was replaced with another 4.0 g transmitter. These individuals will be recaptured the following spring and refitted with a 17 g transmitter (only if they weigh > 800 g when recaptured). The sex of each juvenile was estimated using primary feather and molting sequence (Dalke et al. 1963, Beck et al. 1967), but will not be confirmed until following spring using behavioral (mating behavior at leks) and plumage characteristics.

Radiotelemetry

Following release, radiotelemetry locations of radiomarked individuals were estimated on the ground using hand-held Yagi antennas once every one to three days (from date of capture through September) to monitor status (dead or alive) and movement patterns. UTM locations and appropriate measurement error were estimated by triangulation using the program LOCATE II (Nams 1990) using ≥ 3 bearings. During fall and winter (October-March), all radiomarked individuals were located one to two times per month using either ground or aerial telemetry to document movement patterns and seasonal habitat use.

Vegetation sampling

Vegetation characteristics were measured at all nest locations using established techniques (Connelly et al. 2003). Microhabitat data has the potential of being an important covariate in estimating nest survival. After a hen left a nest (whether successful or unsuccessful), a 30 m transect was placed along a north-south direction bisected by the nest. The shrub height and canopy cover were determined using line-intercept (Canfield 1941). Shrub species was separated into three categories: *Artemisia tridentata* spp., other sagebrush species and non-sagebrush shrubs (e.g., antelope bitterbrush). Total shrub cover included all three categories. Height of a sagebrush shrub within 1 m of the transect line was measured every 5 m along the transect. The percent of grass cover, forb cover, bare ground, and litter was estimated using 20 x 10 cm Daubenmire frame at 5 m intervals along the line transect (Daubenmire 1959). Grass height, number of grass species, forb height and number of forb species were measured within the Daubenmire frame at 5 m intervals along the line transect (Daubenmire 1959).

Animal Care and Use Committee approval (project # 02-2005) was granted February 2005 and updated August 2008 with an addendum to include collaboration with Colorado State University.

Analyses

Nest initiation, estimates of nest (and renesting) success and nest survival were determined by locating and monitoring nests of radiomarked females We used nest survival models (Dinsmore et al. 2002) in Program MARK (White and Burhnam 1999) to estimate rates of daily nest survival and examine the relationship between nest success and vegetation and temporal covariates. We calculated adult and yearling monthly and annual survival using known fate models in Program MARK (White and Burhnam 1999). We calculated daily chick survival for chicks up to 30 days old also using known fate models in Program MARK (White and Burhnam 1999). Variation in survival estimates were generated using bootstrapped estimates from different populations over time.

Estimates of demographic rates and their variance were used to develop a population viability model specifically for GUSG. The models were constructed as matrix models incorporating information on age- and/or sex-specific parameters.

Movement and dispersal metrics will include information on distance, rate and spatial orientation. These metrics will be evaluated using a GIS to assess the effect of landscape features (e.g., habitat composition, configuration of landscape features, barriers, etc.) on movement and dispersal patterns (McGarigal and Marks 1995, White and Garrott 1990, Turchin 1998). These metrics will be critical in the development of landscape model and a SEPM. These analyses will require an accurate GIS database.

RESULTS AND DISCUSSION

The following are Abstracts from a doctoral dissertation by Amy Davis. I added additional comments at the end to expand on the Abstract or to include any information not in the Abstract. Davis successfully defended her doctoral dissertation in June 2012 and is now employed as a statistician by the Missouri Department of Conservation. Further details of these results can be found in her dissertation.

Factors Affecting Nest Success of Gunnison Sage-Grouse in Two Populations in Southwest Colorado

I investigated factors affecting nest success at two dynamically different populations of Gunnison sage-grouse (*Centrocercus minimus*) to determine what selection pressures may be acting on these different populations. The Gunnison Basin population is believed to comprise 85-90% of all Gunnison sage-grouse and is relatively stable. The San Miguel population is one of six relatively small populations, which contains 3-5% of Gunnison sage-grouse and is on the decline. My objective was to compare the demography of these two populations by evaluating the relationship between nest success and 1) vegetation characteristics (e.g., sagebrush height, shrub cover, grass cover, and forb cover), 2) temporal factors (e.g., year, timing of nest initiation, and nest age), and 3) age of the nesting female (yearling or adult). Although expecting nest success to be related to vegetation characteristics is logical, my results did not suggest a strong connection. These results may be due, in part, to measuring characteristics at a different scale than they are acting on the system. My results indicate that temporal factors were strongly related to nest success in both populations. Nest success varied considerably between years (21.4%-60.1%); the average was 38.8%. Within years, I found nests that initiated earlier in the season had higher success than those that were initiated later in the season. Nests were also at greater risk of failure the longer they had been incubated. I found no evidence for a difference in nest success with relation to hen age or between populations.

Davis examined histograms of the vegetation covariates and the related nest fates to determine if there was a pattern. The plots show that the hatch/fail rates are similar across all levels of vegetation cover. This suggests that differences in vegetation cover are not explaining why some nests fail and others are successful. The span of this study includes a recovery from severe drought conditions at our study sites which may have introduced additional variation in vegetation that overwhelmed any effect on nest success. The role that vegetation plays in nest success should not be dismissed. Future work should examine the relationship between vegetation structure and nest success at different scales. Davis examined the potential of scale on nest success by comparing the 15 m radius transect vegetation data to data within 1m from the nest. She found the original model still performed better. Therefore, narrowing the scale of the vegetation sampling did not appear to improve the detection of a nest site level relationship. However, we need to further investigate the relationship between vegetation and nest success at larger scales.

The importance of temporal covariates on nest success may be due to factors that we did not explicitly examine, such as: variation in habitat quality, weather conditions or predator levels. One example of this possibility is the success rates from 2008. The 2007-2008 winter in Colorado was extremely harsh with heavy snowfall that persisted late into spring. However, the highest nest success recorded in our study was in the spring of 2008. The high success rate might be a result of better habitat quality due to the increased moisture, and/or less predation pressure on nests because of the abundance of winter killed deer and elk, and/or the harsh winter may have led to fewer predators overall.

Estimation and Evaluation of Gunnison Sage-Grouse Juvenile Recruitment in Southwest Colorado

Juvenile recruitment is one of the most important vital rates influencing the population growth of many bird species and is fundamental to understanding trends in population size. Gunnison sage-grouse (*Centrocercus minimus*) have declined substantially from their historic range and are currently a candidate species under the U.S. Endangered Species Act. In order to assess the status of this species, my research focused on establishing baseline juvenile recruitment rates and testing population-level, individual (i.e., hatch date) and temporal hypotheses (i.e., month) associated with juvenile recruitment for the Gunnison sage-grouse. I tested these hypotheses on data from two populations of Gunnison sage-grouse in the southwest of Colorado: the Gunnison Basin population was monitored from 2005-2010 and the San Miguel County population that was monitored from 2007-2010. I evaluated juvenile recruitment by combining both chick survival (hatch-30 days of age) and juvenile survival (31 days of age) to the start of the first breeding season. I found strong support for a difference between the two populations in the chick survival analysis - no chicks survived to 30 days of age in San Miguel (n=8). Chick survival was 0.44 in Gunnison Basin (n=282). Thus, there was no recruitment in San Miguel. I found a slight negative trend in chick survival and a stronger negative trend in juvenile survival from 2005-2010 in the Gunnison Basin. Juvenile survival ranged from 0.60 in 2005 to 0.11 in 2010 (n=87). The overall juvenile recruitment rate in the Gunnison Basin declined from 0.26 in 2005 to 0.05 in 2010. These declines mimic declines observed in population index data which might suggest juvenile recruitment declines are contributing to population declines.

We accounted for the potential lack of independence between chicks in the same brood by using methods developed by Bishop et al. (2008). In situations where there is evidence of dependence between siblings, the method uses an inflation factor for the variance in order to compute an appropriate estimate of the variance. We found a significant level of sibling dependence in chicks, but not in juveniles. Chicks during the first 30-days of life stay close to their mother and their siblings and therefore sibling proximity and behavior are inherently linked and their risk of predation is not independent. However, juvenile grouse tend to remain in flocks with their hen and siblings into the fall, but then separate into larger flocks in the winter.

Even though we did not detect differences in nest success and adult survival between these the two populations, we did detect a dramatic difference between the two populations in chick survival. No chicks survived beyond eight days of age in the San Miguel population. Heavy predation pressure prevented juvenile recruitment into this population. Based on this information, CPW initiated a predator control project in order to help this declining population recover.

We found juvenile survival rates to be more variable and lower during the summer months (June-September) and consistently higher during the fall and winter (October-March). This ?single year pattern of survival is similar to that of adult and yearling GUSG where survival rates of yearling and adults are high and constant during the non-breeding season (fall-winter). This survival pattern suggests that juveniles that survive until October will likely survive to the breeding season.

Adult and Yearling Survival of Gunnison Sage-Grouse in Southwest Colorado

Gunnison sage-grouse (*Centrocercus minimus*) populations have declined from their historic numbers and range and recent monitoring has suggested that some populations are continuing to decline. The evaluation of long-term, population-specific survival estimates is important to evaluate population stability, which is necessary for conservation of this species of concern. I evaluated adult and yearling survival in two populations of Gunnison sage-grouse. The Gunnison Basin population is believed to comprise 85-90% of all Gunnison sage-grouse and is relatively stable. The San Miguel population is one of six relatively small populations and contains 3-5% of the species and is on the decline. I examined the relationship between survival and population and tested hypotheses with regards to temporal effects (across years and within year) and individual effects (sex, age, breeding status). I also examined the effect of harsh winters on survival using average monthly snow depth as an indicator of winter harshness. I evaluated monthly survival using known-fate models in Program MARK on 217 radiomarked birds in the Gunnison Basin from 2004-2010 and 25 birds in San Miguel from 2007-2010. I compared the relative support for each covariate using cumulative AICc model weights. The within? (other term?) year pattern of survival is different for males compared to females (cumulative AICc weight 0.878). Males had the lowest survival during the lekking season (March –April), females had lower survival during the nesting and chick rearing season (May – August). Survival also varied among years: between 0.52 and 0.89 for females and between 0.30 and 0.71 for males. My data suggest that harsh winters have little effect on sage-grouse survival. I found no evidence for a difference in survival between yearlings and adults or between the Gunnison Basin and San Miguel population.

Males have the lowest survival during the months of March and April (i.e., the lekking season). Fighting between males for breeding opportunities and an increased risk of predation are likely to add to the higher mortality at this time of year. Although cause of mortality was often unable to be determined, predation was the cause in 98% of the cases where it was able to be determined.

Females have the lowest survival during the months of May to August. This is the nesting and chick rearing time of the year. Hens are limited by their broods as to where they forage, how fast they travel, and how visible/detectable they are to potential predators. It has been suggested that survival during the nesting season was higher for females that initiated a nest than for those that did not initiate a nest. We examined this relationship and found a similar pattern. However, this effect was not as influential as compared to that of sex, season, and year. Perhaps a hen sitting on a nest for 28 days limits her probability of being detected by a predator during that time compared to her counterparts that are probably more active.

Survival rates of both sexes remain fairly constant during the non-breeding season.

Evaluating Viability and Translocation Strategies of Gunnison Sage-Grouse Using a Population Projection Model

Gunnison sage-grouse (*Centrocercus minimus*) are a candidate species under the U.S. Endangered Species Act. Species-specific vital rate information and analyses are important for implementing effective conservation and management actions and have been unavailable until now. I created a female-based, Leslie-type, post-birth pulse population model with three age classes (chicks, yearlings, and adults) to assess the viability of and sensitivity of growth rates to vital rates in two populations of Gunnison sage-grouse; Gunnison Basin which comprises ~90% of the individuals in the species, and San Miguel which comprises ~3% of the species. I also evaluated translocation strategies from the larger Gunnison Basin population to the smaller San Miguel population. I found adult survival to be the most influential vital rate (based on sensitivity metrics) when the population is declining. Juvenile survival and nest success have the largest sensitivity proportional to their variation suggesting these rates might be ideal targets for management actions. Translocation strategies that move birds every five, or fewer, years result in an increase in population persistence. Moving more birds (e.g., > 400 over the course of 30 years) improves the expected population size, but does not improve the persistence probability as much as frequent translocation (e.g., moving birds every year or every other year).

Viability

The growth rates experienced by GUSG during this study indicate declines occurred and that if these rates are indicative of the future, GUSG will likely continue to decline. The population indices of lek counts for the Gunnison Basin population indicate there was a population decline during the time frame of this study. Even though lek counts are at an all time high for the Gunnison Basin, our study included a period where lek counts peaked at over 1,000 males and declined to around 750 males, where it has remained for the last six years. This decline is a defining characteristic of the Gunnison Basin population during our study. Population projections of the Gunnison Basin assume this characteristic will remain true into the future. However, given the highly variable population cycles of GUSG this is unlikely to be true. The time span of this study may not be long enough to encompass broader time trends that might be at play for such a population.

If this study had been conducted just a few years earlier, or later, we might have found a different trend across time. Davis examined the effects of adding one more year to the study where the year had an increasing growth rate. If the next year was a good year (using the best year from the simulated data with a lambda of 1.27), the population projections change from obviously declining to widely variable (i.e., some simulations projected an increasing population, some stable population, some a decreasing population).

It is reasonable to assume that each component of a grouse's life history plays a part in the current trajectory of the population. Fluctuations in adult survival will likely have the largest effect on population growth. However, adult survival is typically a very stable vital rate in nature and therefore not typically the cause of population declines. Chick and juvenile survival were consistently high in importance in the sensitivity analyses on GUSG. Juvenile survival is typically more influential than chick survival on GUSG population growth rates and juvenile survival appears to be the best management target when the population growth rate is slightly decreasing to increasing.

Translocations

The overall results of the translocation analysis suggest that moving birds into a small population (such as San Miguel) will improve the persistence of a declining population. Both the extinction time and resulting population size were improved for the destination population with some form of translocation compared to no translocation. The simulations suggest the optimal translocation strategy is that more frequent translocations had a greater impact on improving population persistence of the small population. We recommend moving birds at a frequency of every year or two to have the greatest impact on the destination population, but if the fate of the source population is in question, then moving birds every five or six years will mitigate some of the effect of bird removals. Based on our analysis, we suggest moving a total of 300-500 birds over 30 years as it balances the impact on the source population and yet will still have considerable improvement on the destination population

Translocations alone may not be sufficient for maintaining a population. Underlying causes of decline (e.g., not enough habitat, large predator populations) must be addressed. Although translocations improve population persistence, the quality of the destination population is more important than either number or timing of translocation in determining population persistence. However, keeping a population viable until management actions can take place is a better strategy for small populations since establishing a population may be more difficult than augmenting an existing population.

Another result from the translocation simulation is that translocating yearling birds appears to result in larger population sizes on the destination population than translocating adults. Adult female sage-grouse are known to have high site fidelity both in breeding and wintering locations. Adult birds that are moved often disappear from the destination population (CPW unpublished data). The yearling females may not have established a movement pattern as strongly as adults and this may relate to their higher success rates in the destination population.

We also examined the effect of removing birds from the source population (Gunnison Basin) based on the same translocation strategies. Removing birds all at once had the least effect on the source population in terms of resulting population size and mean and minimum extinction time. The more frequent the removal, the more substantial the effect on the source population. This result was more pronounced when more than 400 birds were removed from the population. Removing birds more frequently than every 10 years substantially reduced the mean extinction time for the source population - at least a 10 year reduction in extinction time. The minimum extinction time was also more sensitive to frequent translocations, removing birds every 15 years reduced the minimum extinction by seven years as compared to removing birds only once. These results may be due to an accumulative effect of removing birds from the source population. After removing a large number of birds all at once, demographic rates are able to eventually compensate the loss of birds. Otherwise, the repetitive removal of birds results in a longer recovery time since the demographic rates are working on population size that is constantly affected by the translocations.

An Integrated Modeling Approach to Estimating Gunnison Sage-Grouse Population Dynamics: Combining Index and Demographic Data

Evaluation of population dynamics for rare and declining species is often limited to data that are sparse and/or of poor quality. Frequently the best data available for rare bird species are based on large-scale, population count data (e.g., Breeding Bird Survey, Christmas Bird Count, etc.). These data are commonly based on sampling methods that lack consistent sampling effort, do not account for detectability, and are complicated by observer bias. For some species short-term studies of demographic rates have been conducted as well, but the data from such studies are typically analyzed separately. To utilize the strengths and minimize the weaknesses of these two data types, I developed a Bayesian integrated model that innovatively links population count data and population demographic data through population growth rate (λ) for Gunnison sage-grouse (*Centrocercus minimus*). The long-term population index data available for Gunnison sage-grouse are annual (1953-2011) male lek counts. An intensive demographic study was also conducted from 2005-2010. I was able to reduce the variability in expected population growth rates across time, while correcting for potential small sample size bias in the demographic data. I found the population of Gunnison sage-grouse to be slightly declining over the past 16 years ($\lambda = 0.94$, 95% CI 0.90,1.00). However, it is important to keep in mind that these results are preliminary as this methodology is novel and has not been fully vetted.

The dominant eigen value from Leslie matrices that use demographic data is generally considered to be an estimate of population growth (λ). Additionally the rate of population change from one time step to another ($\lambda = Mt+1/Mt$) is another method that estimates population growth; this method is applicable to lek count data. This study combines the population growth estimates (λ) from these two sources of information.

We recognize that the growth rates may not be directly related, especially given that the demographic data are based on a female driven model and the population count data is only of males. Therefore, we estimated a linear relationship between the log of the growth rates. This evaluation, based on the more reliable 1996-2011 time series, suggests that lek count estimates of population growth are typically biased high and exhibit extreme high values that are not realistic based on demographic analysis. The population growth rates exhibited by the lek count data varied wildly (max λ near 2), and the range was much greater than is typically seen in growth rate estimations from Leslie matrix calculations for sage-grouse species

A key aspect to this analysis was to help eliminate potential bias in the shorter time series. These results suggest that the population, on average, is relatively stable over the past 16 years, but the end of the time series shows a slight decline.

Additional Analyses in Progress

Landscape model: this analysis will examine the impact of landscape features (e.g., the quantity and configuration of GUSG habitat and anthropogenic features) on movement patterns of GUSG. This will help us evaluate the potential of these features to act as barriers. The information from this model will provide parameter estimates that will be used to evaluate a spatially-explicit population model, as well as evaluate whether the sagebrush habitat is fragmented in relation to GUSG.

Spatially-explicit population model (SEPM): This is a complex, data intensive model that is potentially very useful. It will examine the demographic information we have collected, but in a spatially-explicit framework. This analysis will examine how productivity (survival, nest success, etc.), recruitment and movement patterns of GUSG may be a function of temporal and spatial variation. The models of demography described above treat the populations (San Miguel and Gunnison Basin) as a homogeneous process. Information from the landscape model will provide information to be used in a SEPM (i.e., how far and under what conditions GUSG move). A SEPM will examine how landscape features influence demographic parameters (e.g., nest success, and survival). Furthermore, a SEPM will be a valuable tool that will allow researchers and managers to develop and evaluate alternative management plans specifically for GUSG. A SEPM will allow managers to evaluate the relative merits of proposals for land acquisition and easements based on spatially explicit demographic and behavioral data (e.g., what is the potential for sage-grouse to use a specific land parcel and what effect may it have on the local population persistence).

Fecal sampling and lek counts: We have no estimates of the accuracy of lek counts. This project will involve using the DNA from fecal samples collected on leks and mark-recapture methods to estimate lek size. This allows an estimate of precision of lek counts. Traditional lek counts were recorded on the day that fecal samples were collected. A comparison will give us insight to the potential error of lek counts. This project is a collaboration with Dr. Sara Oyler-McCance of the U.S. Geological Survey, who conducted the genetics analysis and Dr. Paul Luckas of the University of Montana, who is experienced in the use of noninvasive sampling for population estimates.

Metapopulation model: Another potentially useful analysis will evaluate the movement of GUSG in a metapopulation framework. This analysis could potentially expand on the SEPM by modeling the Gunnison Basin as a metapopulation. A SEPM assumes a static parameter of movement based on the movement data. A metapopulation model could be used to examine varying rates of movement through simulations. Genetic data can also be more easily incorporated into a metapopulation model than a SEPM. Typical applications often draw more directly on ecological theory (e.g., island biogeography). This analysis could be used to evaluate the potential of applying a metapopulation across all GUSG populations (i.e., by examining the extent extinction-colonization dynamics within the Gunnison Basin we may have a better understanding of GUSG as a rangewide metapopulation). Spatially realistic metapopulation models may be used to generate more refined species-specific and landscape-specific predictions of population persistence, which could potentially replace the debated rules of reserve design based on the dynamic theory of island biogeography.

Decision Analysis: The development of a population viability model and a SEPM are valuable tools for evaluating the relative threats to GUSG. However, the models do not automatically indicate which management strategies will have the greatest impact on minimizing the threats. Therefore, species-specific data and modeling results acquired in this study will ultimately be used to develop a decision analysis approach to evaluating alternative management programs. Decision analysis is an analytical approach to evaluate the relative outcomes of alternative management actions. Using the approach, managers can assess alternative strategies by incorporating the probability of an event occurring, given a particular strategy, and the probabilities of several potential outcomes as a result of that event. In this manner the consequences of management strategies can be evaluated more quantitatively in a probabilistic framework.

These analyses will continue in 2013, with manuscripts prepared for submission to peer-reviewed journals.

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Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3420	:	Avian Research
Work Package:	0659	:	Gunnison Sage-grouse Conservation
Task No.:	N/A	:	Gunnison sage-grouse captive-rearing
Federal Aid Project No.	N/A		

Period Covered: September 1, 2011 – August 31, 2012

Author: A. D. Apa, M. Phillips, and L. Wiechman

Personnel: K. Fox, Colorado State University; A. B. Franklin, U.S.D.A. National Wildlife Research Center; J. V. Azua, Jr., Denver Zoo; P. Alden, C. Binschus, C. Davis, M. Downey, K. LeDoux, C. Mix, S. Ogden, R. Sadowski, B. Sedinger, L. Stoorza, S. Vincent, C. Wilson and L. Wolfe, CPW

ABSTRACT

Gunnison sage-grouse (*Centrocercus minimus*, hereafter GUSG) is a species of concern in Colorado. Two conservation issues addressed in the Gunnison Sage-grouse Rangewide Plan (RCP) are the population persistence of GUSG (especially the small populations) and the relatively low genetic diversity among GUSG. Augmenting the small GUSG populations is a potentially useful management tool to address these conservation concerns. Five alternative techniques for transplanting yearling or adult individuals are discussed in the RCP, including use of captive-reared GUSG. Researchers at the U.S.D.A. National Wildlife Research Center (NWRC) in Fort Collins, Colo. were able to maintain 18 yearling greater sage-grouse (*C. urophasianus*, hereafter GRSG) in captivity for eight months. Recent Colorado Parks and Wildlife (CPW) research on GRSG has evaluated different aspects of captive-rearing techniques. The objectives for this project segment are to: 1) collect 70 GUSG eggs; 2) artificially incubate and hatch eggs; 3) develop captive breeding techniques for GUSG; 4) determine if captive GUSG can initiate incubation and rear a brood in captivity; and 5) augment wild surrogate broods with domestically-reared chicks at one, three, five, and seven weeks of age.

Female GUSG were captured using spot-lighting techniques. Females were radio-marked and monitored to assist in locating nesting females. Eggs were collected from laying and incubating females. Eggs were transported from the Gunnison Basin to the CPW Foothills Wildlife Research Facility (FWRF) in Fort Collins and placed in an incubator in a newly constructed building until an external pip was observed (25-26 days), and then they were moved to a hatcher.

Fourteen females were captured in the Gunnison Basin. They were added to 36 previously marked females and were available for egg collection and to serve as potential surrogate for domestically-reared chicks. Forty-seven of 50 females (94.0%) initiated a nest. The three females who did not initiate a nest died before a nest was detected (April 20, April 27, and May 5). Eggs were collected from 14 females. Five of the eight females that had eggs collected during laying successfully hatched (62.5%).

All of the six females who were forced to abandon their nests initiated a renest, and three of the six hatched successfully (50.0%). Excluding females forced to abandon nests for egg collection purposes (n=6), 62.5% (25/40) initial (?inital what?) and 50.0% (5/10) of the renests were successful.

Thirty-seven eggs were collected from five incubating females, and 38 eggs were collected from nine laying females. Thirty-two eggs were collected from five captive-reared females. All eggs laid were collected and no females were allowed to incubate. Seventy-one of 107 eggs artificially incubated hatched (66.4%). Of the 36 eggs that did not hatch, 14 were infertile, and four were malpositioned (head down, away from air cell). If those eggs are censored from the analysis, our hatch success of viable eggs is 79.8% (71/89).

Chick survival appeared to be higher in 2011. Fifty-two of 71 chicks that hatched successfully survived to introduction (73.2%). Fifteen wild broods were augmented with 51 captive-reared chicks over 19 separate introductions. Overall adoption success (defined as successful if the chick is with the surrogate brood 24-36 hours post-introduction) was 35.3% (n = 18/51). Within Treatment I (seven days), our adoption success was 60% (15/25), although one chick was lost due to exposure, and two surrogate broods, including seven domestic chicks were depredated within 24 hours of release, accounting for most of our failed adoptions. Apparent survival of the domestically reared chicks was 0% (0/39). Four of the 51 chicks were censored from the analysis after their transmitters fell off. Eight of the remaining 47 were missing and their fate is unknown.

Additional detail is provided in the 2011 progress report. From September through November 2011, actions were initiated to transition from a research project to a long-term captive-flock care and maintenance at the CPW FWRF. During this time, the flight pens at the National Wildlife Research Center (NWRC) were restored to their previous condition and eight GUSG (6 F: 2 M) were fed daily. On 23 November 2011, the captive GUSG were captured, weighed, and blood samples were obtained for DNA archival storage. Grouse were transported to the CPW FWRF and released into a newly constructed aviary with no injuries. Husbandry care is being conducted by CPW FWRF staff. This transition technically ended the captive-rearing portion of the research project.

Currently the project personnel are in the process of proofing and editing data and conducting analyses. Manuscripts are currently being prepared and are expected to be submitted for publication in early 2013.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	0659	: Gunnison Sage-grouse Conservation
Task No.:	N/A	: Baseline habitat monitoring for Gunnison sage-grouse in the Gunnison Basin, Colorado
Federal Aid Project No.	N/A	

Period Covered: September 1, 2011 – August 31, 2012

Author: J. H. Gammonley

Personnel: A. Apa, M. Phillips, CPW; A. Hild, M. Williams, University of Wyoming

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ABSTRACT

Analyses were completed and a final report was prepared and distributed: Williams, M. I., and A. L. Hild. 2012. Charateristics of Gunnison sage-grouse habitat in Dry Mountain Loam and Mountain Loam ecological sites of the Gunnison Basin. A manuscript was prepared and submitted for publication: Williams, M. I., A. L. Hild, J. H. Gammonley, A. D. Apa, and M. L. Phillips. In review. Evaluating Gunnison sage-grouse breeding habitat in the Gunnison Basin. Journal of Wildlife Management. Below is the abstract from the draft publication:

Gunnison sage-grouse (*Centrocercus minimus*) is a sagebrush-obligate species proposed for listing as endangered by the US Fish and Wildlife Service under the Endangered Species Act. Seven populations exist on the Colorado Plateau in western Colorado and eastern Utah. Since habitat loss and degradation are primary threats to the species, there is great interest in the assessment of populations and their habitat over time and space. We measured habitat characteristics on 392 30-m transects in 6 study areas within the Gunnison Basin, CO during 2010 and 2011. Transects were stratified between two, shrub-steppe ecological sites in each study area. We evaluated cover and height of sagebrush (*Artemisia tridentata*), grasses, and forbs for meeting minimum breeding habitat guidelines and assessed for spatial and temporal variation by ecological site, study area, and year. Few (<4%) individual monitoring transects met all the guidelines, but many transects in all study areas, ecological sites, and years met or exceeded the minimum guidelines for one or more of the habitat characteristics. Accounting for spatial and temporal variation, model-averages of sagebrush foliar cover and height, grass canopy cover and height, and forb height fell within the breeding habitat guidelines. Forb canopy cover did not meet the minimum breeding guideline in some study area, ecological site, and year combinations. Use and interpretation of habitat guidelines with reference to study area, ecological site, and year can account for spatial and temporal variation in the potential for the Basin to support target habitat characteristics.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3420	:	Avian Research
Work Package:	0660	:	Greater Sage-grouse Conservation
Task No.:	N/A	:	Greater sage-grouse natal dispersal and brood augmentation with captive-reared chicks

Federal Aid
Project No. N/A

Period Covered: September 1, 2011 – August 31, 2012

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ABSTRACT

In response to population declines, recent research on greater sage-grouse (*Centrocercus urophasianus*) has focused on the population ecology, habitat relationships, and response to management practices by this species. However, the mechanisms, patterns, and consequences of movements between seasonal habitats, especially by juveniles during natal dispersal, and the effects of this movement on survival, recruitment, the redistribution of individuals, as well as the population dynamics within and between populations remains largely unknown. Quantifiable data and information on juvenile dispersal and survival in the greater sage-grouse is one of the least understood aspects of this species' life history. Dispersal patterns and recruitment processes of juvenile sage-grouse are lacking, as well the landscape characteristics that influence and contribute to these movements. Knowledge of the dispersal ecology (timing, distances moved, frequency and rate of movement, immigration and emigration rates within and between populations, and juvenile survivorship) will provide better information on how to manage this species at the landscape level, as well as within and between populations. This information will be useful in attempting to improve and plan for the conservation of this species as its habitat becomes more fragmented and altered. The objectives of our study are to: 1) determine the sex-specific movement patterns of juvenile sage-grouse during natal dispersal including timing, duration, rate of movement, distances moved and recruitment rate, 2) determine the effects of these dispersal patterns on survival rates and causes of mortality, 3) determine how landscape structure influences both the movement patterns and survival of juveniles during this period, 4) verify and evaluate the mechanisms and conditions of adoption in wild broods through the introduction of domestically-hatched chicks and observation of natural adoption rates, 5) assess the movement patterns and survivorship of successfully adopted domestically-hatched 2- and 7-day-old chicks from the natal area of the surrogate brood to chick independence and brood break-up at approximately 10 weeks of age, and 6) compare the movement patterns and survivorship of domestically-hatched chicks with the movement patterns and survivorship of wild-hatched chicks in mixed and unmixed broods from the natal area of the surrogate brood to chick independence and brood break-up. The study areas were located in the Axial Basin and Cold Springs Mountain in northwestern Colorado from 2005 – 2007. The project field research and final report is

complete with the delivery of a Ph.D. dissertation. Manuscripts based on chapters of the dissertation are currently being prepared and submitted for publication in peer-reviewed journals. The following summaries are excerpts from Dr. Thomas R. Thompson's Dissertation (Thompson 2012).

[1]With the exception of the project abstract, all excerpts are from Dr. Thompson's Dissertation.

DISPERSAL ECOLOGY OF GREATER SAGE-GROUSE IN NORTHWESTERN COLORADO: EVIDENCE FROM DEMOGRAPHIC AND GENETIC METHODS

ABSTRACT

The greater sage-grouse (*Centrocercus urophasianus*; here after sage-grouse) has undergone dramatic population declines over the last 25 years as a result of loss, fragmentation and degradation of sagebrush (*Artemisia tridentata* spp.) habitats on which it depends. Because of these declines and the subsequent loss of habitat, knowledge concerning the juvenile ecology of sage-grouse, including natal dispersal patterns and abilities and its influences on population persistence, colonization, and connectivity are critical for the conservation planning and management of this species

The focus of this dissertation was two-fold: first, to assess the feasibility of actively collecting and hatching sage-grouse eggs from wild radiomarked sage-grouse and rearing subsequent domestically-hatched (DH) chicks from 1-10 days of age before augmenting wild sage-grouse broods (Chapter 2), and second to investigate natal dispersal in greater sage-grouse through both demographic (radiotelemetry) and genetic methods. In Chapter 3, I monitored survival and causes of mortality in wild-hatched chicks (n = 431) in wild broods (n = 115) from hatch to 16 weeks of age in the Axial Basin and Cold Springs Mountain study areas in northwestern Colorado, 2005-2007 and evaluated potentially important predictors of brood and chick survival. In addition, I monitored survival from hatch to 16 weeks of age for a cohort of DH chicks raised to 1-10 days of age in captivity (n = 116) and introduced into a subset of wild broods during this same time period. Model averaged estimates of brood and chick survival indicated that survival varied both temporally and spatially. In Chapter 4, I captured, radiomarked and monitored survival and recruitment of 183 transmitter-equipped juveniles (from Chapter 3) from Sept. 1 – March 31. Survival from September through March was similar for all juveniles, but varied by month, study area, and gender. Median dispersal distance was greater for juvenile males compared to females (M: 3.84 + 1.26 km; F: 2.68 + 0.30 km), as well as the proportion dispersing > 5 km (M: 31.6%; F: 15.5%). In Chapter 5, I examined the patterns of dispersal, gene flow, and genetic structure at 15 leks in six population management zones (PMZs). Genetic analyses were largely congruent and suggested that gene flow followed an isolation-by-distance pattern, and supported male-biased dispersal findings based on demographic data (Chapter 4). Finally, in Chapter 6, I investigated how coarse-grained landscape characteristics influenced dispersal and settlement patterns. Landscape metrics primarily differed between study areas rather than genders, and among pre-dispersal, winter, and post-dispersal landscapes. Effect of extent upon analyses depended upon the specific metric and landscape.

CHAPTER 1 - This is a review chapter and an abstract is not supplied.

CHAPTER 2 - CAPTIVE-REARING SAGE-GROUSE FOR AUGMENTATION OF SURROGATE WILD BROODS: EVIDENCE FOR SUCCESS

ABSTRACT

Both species of North American sage-grouse (*Centrocercus spp.*) have shown substantial declines in distribution and abundance. Translocation of adult birds from a stable population to a small or declining population has been one of the management tools used by wildlife agencies to support

population persistence in these areas. Captive-rearing chicks with subsequent release into wild surrogate broods is an untested alternative to augment declining populations of sage-grouse. Captive-rearing involves collecting eggs from wild females in a stable population, hatching and rearing them to a specified age in captivity, then releasing them into surrogate wild broods. We developed techniques to successfully captive-rear sage-grouse chicks, evaluated predictors of hatching and captive-rearing success, and estimated chick survival to 30 days in surrogate broods.

We collected 304 eggs from radiomarked female greater sage-grouse (*C. urophasianus*) during 2004-2007 in three study areas in northwestern Colorado. Hatching success of collected eggs was 71.5% and was significantly higher for eggs collected from incubating females (88.9%) than eggs collected from laying females (59.2%) or salvaged eggs collected after nest abandonment or female mortality (22.7%). The top model (lowest Δ AIC*c* score) suggested hatching success was influenced by refrigeration of eggs, and percent egg weight loss during incubation. These parameters occurred in the top two models and accounted for 88% of the weight in the candidate model set. Predicted hatchability of collected eggs based on the top model decreased over time with < 50% of eggs hatching if refrigerated longer than 9 days or if percent of egg weight loss during storage and incubation was > 15%. We raised 176 chicks in captivity for 1-10 days. The top two models to predict captive-rearing success accounted for 91% of the weight in the candidate model set. The top models suggested captive-rearing success was influenced by daily weight gain, weight of chick at hatch, refrigeration of the egg, and nesting status of the female when the egg was collected. Based on the top models, predicted captive-rearing success was > 80% for chicks weighing at least 32 g at hatch and for chicks that gained more than 1g per day during captivity. We radiomarked 133 captive-reared chicks and introduced them to surrogate wild females in 53 separate release events. Adoption success of domestically-hatched chicks into surrogate broods at 24 hours post release and > 50 m from adoption site was 88.7%.

Overall survival of domestically-hatched chicks to 30 days of age was 0.42 (95% CI = 0.33 – 0.52). Survival did not differ between treatment groups; however, it differed between study areas. Depredations accounted for 26.3% of fates and exposure accounted for 25.6%. Survival modeling suggested model selection uncertainty in the model set. However, the top three models all included hatching date, surrogate brood size after adoption, surrogate female age, and age difference between wild- and domestically-hatched chicks at adoption, and accounted for 93% of the total model weights. We discuss recommendations for the use of captive-rearing as a management tool to demographically or genetically reinforce small populations of sage-grouse.

CHAPTER 3 – SURVIVAL OF GREATER SAGE-GROUSE BROODS AND CHICKS FROM HATCH TO BROOD INDEPENDENCE IN NORTHWESTERN COLORADO

ABSTRACT

Survival of chicks from hatch to brood independence and recruitment into fall populations is an important, but poorly understood life history trait that can have important consequences on the dynamics and viability of greater sage-grouse (*Centrocercus urophasianus*) populations. Little is known about how the factors of gender, hatch date, hatch weight, distance traveled from nest, and brood size contribute both individually and ecologically to survival of chicks. We monitored survival and causes of mortality in wild-hatched (WH) chicks (n = 431) in wild broods (n = 115) from hatch to 16 weeks of age in the AB (Axial Basin) and CSM (Cold Springs Mountain) study areas in northwestern Colorado between 2005 and 2007 and evaluated potentially important predictors of brood and chick survival. In addition, we monitored survival from hatch to 16 weeks of age for a cohort of domestically-hatched (DH) chicks raised to 1-10 days of age in captivity (n = 116) and introduced into a subset of wild broods during this same time period.

Overall brood survival from 2005-2007 (both wild and wild broods augmented with DH chicks) to 16 weeks of age was 0.381 (95% CI: 0.264 – 0.514) at CSM compared to 0.533 (0.405 – 0.657) in the AB. Within the AB, we observed higher model-averaged survival rates among broods with DH chicks (0.631, SE = 0.088) compared to broods without (0.430, SE = 0.104), while at CSM the pattern was reversed (0.205, SE = 0.102 and 0.573, SE = 0.080). When we included broods that were depredated 1-3 days post-hatch and before radiomarking of chicks our overall apparent brood survival decreased from 47.8% (55/115) to 43.7% (55/126).

The main cause of chick death was from predation, although exposure accounted for 27% of mortalities among DH chicks. Model averaged estimates of brood and chick survival indicated that survival varied both temporally and spatially. Brood and chick survival were higher in the AB compared to CSM, and WH chicks had higher survival in both areas compared to DH chicks. Similarly, DH and WH chicks at CSM in both augmented and wild broods had lower survival than DH and WH chicks in the AB with the largest differences occurring during weeks 1 – 3. Among the WH chicks survival rates among years in the AB ranged from 0.158 to 0.446 and at CSM from 0.088 to 0.339. Between study areas and among years survival was lowest during the first 3 – 4 weeks. We found evidence that chick survival increased with age and decreased with advancing hatch date, but found limited support for the influence of gender or distance traveled.

We recommend that managers develop better understanding and knowledge of the relationship between nesting cover and brood habitat, as well as movement patterns between these areas within a landscape for each population. Managers need to consider prioritizing the protection and restoration of both early- and late brood-rearing habitat within specific landscapes, as our study demonstrates two bottlenecks through which chick survival significantly decrease at < 21-day post-hatch and during brood independence at > 10 weeks of age. We suggest that > 3 areas of each seasonal brood habitat type be dispersed within a breeding population to maintain traditional use patterns and to facilitate the use of new areas (i.e., restorations or plantings such as CRP), so as to help reduce predation risks and exposures due to concentration of broods in poor quality or limited critical habitat.

CHAPTER 4 – SURVIVAL, NATAL DISPERSAL AND RECRUITMENT OF JUVENILE GREATER SAGE-GROUSE IN NORTHWEST COLORADO

ABSTRACT

Natal dispersal can play a key role in the demographic, genetic, and evolutionary processes at both the population and species level. Recognition of dispersal‘s importance in the persistence, distribution, and regulation of populations within a species is well supported at least theoretically; however for most species, including the greater sage-grouse (*Centrocercus urophasianus*) this life history trait is largely unknown, and its implications on specific populations and population processes unexplored. We captured, radiomarked, and monitored survival and recruitment of 183 transmitter-equipped juveniles from Sept. 1 – March 31 at two study areas in northwest Colorado (AB: Axial Basin, CMS: Cold Springs Mountain). We also documented movement patterns and characteristics between pre-dispersal (late brood-rearing), winter, and post-dispersal (breeding) ranges for juveniles surviving through these periods. Juveniles monitored included three main types: known wild-hatched (WH; location of natal nest is known), random (natal nest unknown and captured > 40 days post-hatch), and domestically-hatched (DH; chicks hatched in captivity and released into same-age wild surrogate broods at < 10 days of age).

Survival from September through March was similar for all juveniles (WH, DH, and random), but varied by month, study area, and gender. Juvenile females (AB: 0.754, SE = 0.052; CSM: 0.549, SE = 0.060) had higher survival than juvenile males (AB: 0.621, SE = 0.070; CSM: 0.410, SE, 0.081), and

survival for each was greater in the AB compared to CSM. We observed that juvenile survival was lowest during September and October and coincided with brood independence and integration into winter flocks, but before initiation of dispersal. We did not find any evidence for the effect of body mass, hatch date (age), or year on juvenile survival. We documented higher survival for a subset of radiomarked adult females (AB: 0.831, SE = 0.061; CSM: 0.835, SE = 0.057) compared to juveniles (AB: 0.723, SE = 0.076; CSM: 0.426, SE = 0.098), and that differences occurred primarily during September and October. We observed brood independence to be a distinct behavioral event separate from initiation of dispersal. The initial movements related to dispersal occurred approximately four to six weeks after the initiation of brood independence. Both studied populations showed a high degree of variation in movement and distances traveled from late brood or pre-dispersal areas to wintering ranges during November and December (fall phase of dispersal) and then again during the spring phase of dispersal (March) from wintering ranges to post-dispersal or breeding areas.

We documented primarily intra-population dispersal (within a breeding population) with only one occurrence (a juvenile male) of inter-population dispersal (between breeding populations) despite often extensive movements (> 10 km) to wintering ranges. Median dispersal distance was greater for juvenile males compared to females (M: 3.84 + 1.26 km; F: 2.68 + 0.30 km), as well as proportion dispersing > 5 km (M: 31.6%; F: 15.5%). Additionally, juvenile females had smaller home ranges and overlapped their maternal home range more than juvenile males, even though home ranges were larger for males. Outside of gender, we did not find any evidence for the influence of other measured explanatory variables on dispersal distance or occurrence. Recruitment of surviving juveniles was primarily local in both study areas (into natal breeding population; AB: 98.2%, CSM: 100%), however average survival from hatch to recruitment into the natal breeding population (March) varied between each (AB: $\bar{x}$ = 0.287, SE = 0.039; CSM: $\bar{x}$ = 0.122, SE = 0.054). This information on survival, dispersal, and recruitment of juvenile sage-grouse has important implications for the management of this species at local, landscape, and regional levels.

CHAPTER 5 - DISPERSAL, GENE FLOW, AND POPULATION GENETIC STRUCTURE IN THE GREATER SAGE-GROUSE: IMPLICATIONS FOR CONNECTIVITY AND NATURAL RECOLONIZATION

ABSTRACT

Dispersal and its influence on gene flow can be a life-history trait that has important consequences on the population dynamics and genetic structure of populations. However, for most species the degree to which dispersal and gene flow maintain population connectivity both demographically and genetically remains unknown. Here, we examine the patterns of dispersal, gene flow, and genetic structure in greater sage-grouse (*C. urophasianus*) at 15 leks in six population management zones (PMZs) in a stable population in northwest Colorado by genotyping 275 individuals at 17 microsatellite loci.

All leks showed high levels of genetic diversity in terms of average number of alleles, allelic richness, and heterozygosity. We inferred moderate to high levels of gene flow between neighboring PMZs. Global Mantel tests of genetic distance vs. geographic distance, principal coordinates analysis, and Bayesian clustering analyses revealed that leks and PMZs had an isolation by distance pattern in which gene flow followed a directional or two-dimensional stepping-stone pattern that was primarily local and between neighboring leks, but also between neighboring PMZs.

Contrary to the traditional view of female-biased dispersal in avian and grouse species, we observed evidence of male-biased dispersal in both direct (radiotelemetry) and indirect (genetic) methods. Median natal dispersal distance of radio-marked males was slightly greater than females (3.8 + 1.3 km

and 2.7 + 0.3 km, respectively; z = 468.0, P = 0.0206). We also detected higher levels of genetic structure in females for the sex-specific F_{ST} (0.040 and 0.025, respectively; P = 0.032) and relatedness (r) (0.076 and 0.048, respectively; P = 0.034) indices indicating higher dispersal rates and less genetic structure in males. Spatial autocorrelation analyses indicated significant fine scale structuring for both males and females at distances < 15 km, and both males and females showed positive autocorrelation out to 30 km. First-generation migrant tests between PMZs were inconclusive but suggest that dispersal was rare at distances > 40 – 60 km for both sexes.

Our data indicate that genetic and demographic connectivity (observed movements of radiomarked individuals) occurs at different scales and thus different thresholds exist beyond which populations can become functionally isolated due to loss or reduction in the number of dispersers and amount of gene flow. Results suggest that within-population processes (i.e., internal population dynamics including production, immigration and emigration of individuals) occur at smaller scales than would be expected for a large, highly-mobile species, and that these process drive both demographic and genetic connectivity. This can have dramatic effects on a population‘s ability to persist or re-colonize under current habitat and population threats. Our study demonstrates the importance of using both demographic and genetic methods to define and understand population characteristics that can provide insight into conserving and managing populations at appropriate scales.

CHAPTER 6 - RELATIONSHIP OF LANDSCAPE CHARACTERISTICS TO MOVEMENT BEHAVIORS AND SETTLEMENT PATTERNS OF GREATER SAGE-GROUSE IN NORTHWEST COLORADO

ABSTRACT

Range-wide declines in greater sage-grouse (*Centrocercus urophasiansus*) populations have largely been attributed to loss, degradation, and fragmentation of sagebrush habitats and landscapes that are believed to negatively impact population vital rates, movements and distribution patterns. Current understanding of these processes in sage-grouse is primarily limited to adult age individuals with little understanding of their influences on juvenile movement behaviors and settlement patterns. In this study we assessed how landscape composition (percent land cover) and edge density (m/ ha) within the dispersal range (winter and dispersal locations) and dispersal period landscapes (pre-dispersal, winter, and post-dispersal locations) differed between male and female juvenile sage-grouse in two study areas (Axial Basin and Cold Springs Mountain) in northwestern Colorado.

In 2005 – 2008 we monitored 95 juveniles (74 female and 31 males) from September through April. Before running landscape analyses we performed an accuracy assessment on three potential Landsat satellite imagery sources (Colorado Vegetation Classification Project, LANDFIRE, and Southwest Regional GAP) and used overall accuracy, and kappa coefficients to determine which data source would have the highest quality and less uncertainty in derived land cover maps. Using the LANDFIRE (2006) Existing Vegetation Map, we compared ?the? proportion of land cover types and edge densities in four dominant land cover types (sagebrush dominated community (*Artemisia tridentata* spp), salt desert shrub dominated community (shadscale saltbush (*Atriplex confertifolia*); greasewood (*Sarcobatus vermiculatus*)), grassland/ rangeland/ perennial grass and forb, and deciduous shrub/ mountain-shrub dominated community (bitterbrush (*Purshia tridentate*); Gambel oak (*Quercus gambelii*); serviceberry (*Amelanchier spp*); snowberry (*Symphoricarpos spp.*), and tested for effect on genders, areas, dispersal ranges, and among dispersal period landscapes at two spatial extents (500- and 2,000-m).

Dispersal ranges and dispersal period landscape metrics were not significantly different between genders at either buffer extent. Within dispersal ranges, percent cover in sagebrush did not significantly differ between study areas at the 500-m buffer extent; however at the 2,000-m buffer extent proportion of

land cover in sagebrush was higher in the Axial Basin. Among dispersal period landscapes, measured metrics significantly differed between areas and among periods. At the 500-m buffer extent winter and post-dispersal landscapes in the Axial Basin had higher land cover in sagebrush, lower edge density in sagebrush, and lower cover in salt desert shrub compared to Cold Springs Mountain. At the 2,000-m buffer extent a similar pattern was observed, as well as higher land cover in sagebrush and shrub, as well as shrub edge density in the Axial Basin. The grassland cover type did not significantly differ at either buffer extent for dispersal range or dispersal period landscapes.

We believe this suggests natal dispersal movement behaviors and settlement patterns within our study areas, where percent land cover in sagebrush are > 60%, are not directly influenced by landscape structure or composition in the dispersal range or period, but by individual and population pressures and demands (e.g., access to resources, inbreeding avoidance, traditional use) related to the breeding and production (brood-rearing) areas.

LITERATURE CITED

Thompson, T. R. 2012. Dispersal ecology of greater sage-grouse in northwestern Colorado: evidence from demographic and genetic methods. Ph.D. Dissertation, University of Idaho, Moscow, Idaho, USA.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3420	:	Avian Research
Work Package:	0663	:	Terrestrial Species Conservation
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Personnel: T. Apa, K. Eichhoff, J. Gammonley, B. Petch, M. Phillips, B. Walker

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the author. Manipulation of these data beyond that contained in this report is discouraged.

ABSTRACT

The identification of core habitat areas and the resulting prediction maps are vital tools for land managers. Often, agencies have large datasets from multiple studies over time that could be combined for a more informed and complete picture of a species. Colorado Parks and Wildlife has a large database for greater sage-grouse (*Centrocercus urophasianus*), including 11 radio-telemetry studies completed over 12 years (1997-2008) across northwestern Colorado. We divided the 49,470 km^2 study area into 1 km^2 grids with the number of sage-grouse locations in each grid cell counted as the response variable. We used a generalized linear mixed model (GLMM) that used land cover variables as fixed effects and individual birds and populations as random effects to predict greater sage-grouse location counts during the breeding, summer, and winter seasons. The mixed effects model enabled us to model correlations that may exist in grouped data, e.g., correlations among individuals and populations. We found only individual groupings accounted for variation in the summer and breeding seasons, but not the winter season. The breeding and summer seasonal models predicted sage-grouse presence in the currently delineated populations for Colorado, but there was little evidence of selection for the winter season. About 50% of the study area in Colorado is considered highly or moderately suitable habitat in both the breeding and summer seasons.

As oil and gas development and other landscape changes occur in this portion of Colorado, it will become more critical to know where management actions can be accomplished or possible restoration can occur. These seasonal models provide data-driven, defensible distribution maps that managers and biologists can use for identification and exploration when investigating greater sage-grouse issues across the Colorado range. Using historic data for future decisions on species management, while accounting for issues found from combining datasets, allows land managers the flexibility to use all information available. A manuscript was prepared based on these analyses and accepted for publication:

Rice, M. B., A. D. Apa, M. L. Phillips, J. H. Gammonley, B. Petch, and K. Eichhoff. In Press. Analysis of regional species distribution models based on combined radio-telemetry datasets from multiple small-scale studies. Journal of Wildlife Management.

WILDLIFE RESEARCH REPORT

DEVELOPMENT OF DISTRIBUTION MODELS FOR MANAGEMENT OF GREATER SAGE-GROUSE IN COLORADO

MINDY B. RICE

PROJECT OBJECTIVES

The objective of this study is to provide managers and biologists with a comprehensive map of greater sage grouse habitat in northwestern Colorado using a consistent and repeatable methodology that can delineate the entire range in more detail.

SEGMENT OBJECTIVES

1. Run various biologically relevant models in the telemetry database of sage-grouse locations and use GIS to predict the probability of sage grouse locations in Colorado based on vegetation.
2. Provide initial prediction model for evaluation and validation before adding North Park data into model.

INTRODUCTION

Predicting species distribution across a broad geographic range can be challenging because of variability within and among regions and populations. Some studies indicate high levels of predictability among regions (Vanreusel et al. 2006) while others argue against a uniform management program for species with a large geographic range (McAlpine et al. 2008). When conserving for multiple populations across a large range, we need to develop predictive species distribution maps with details specific to each population, but applicability across the entire range (McAlpine et al. 2008). The identification of core or priority habitat areas for a species provides a vital tool for communication to private landowners and partner wildlife professionals.

Species distribution models have become an important tool for wildlife management and there are many methods to accomplish this goal. Often, the data available across a landscape scale are the presence locations of a species. Commonly used methods for this type of data are resource selection functions (RSF) using a logistic regression framework with presence and absence locations where absences are often represented by random locations or pseudo-absences (Ciarniello et al. 2006, Greaves et al. 2006, Johnson and Gillingham, 2008). Problems associated with these resource selection methods include pseudo-absences whose absolute presence or absence is not known with certainty (Johnson et al. 2006), contamination and overlap in used and available locations (Johnson et al. 2006), and unequal sampling of the species range (Latimer et al. 2006). In addition, many methods for evaluating logistic regression model predictions are inappropriate for presence/available data because used sites are drawn directly from the distribution of available sites (Boyce et al. 2002). This overlap causes the variance estimates from the logistic regression to be biased (Johnson et al. 2004). Another option is utilizing the frequency of animal locations as a count variable for modeling habitat associations when absence is unknown across a species range such as resource utilization functions (RUF) and resource selection probability functions (RSPF) (Marzluff et al. 2004, Sawyer et al. 2009). These analyses focus on an individual animal's habitat selection which is then combined to yield population level inference and may require a large number of locations per individual which may not be available for all datasets. Another method for presence only modeling is based on maximum entropy using the software Maxent (Phillips et al. 2006). Maxent produces estimates of a suitability index rather than occurrence probability and is not suitable for making explicit predictions (Royle et al. 2012). Maxent can also be sensitive to modeling choices and input localities (Rodda et al. 2011).

Most of these methods have been shown to provide adequate predictive species distribution maps, but may become more problematic for data not originally collected for habitat analyses, data collected from multiple populations, or data from multiple studies. These data are common for state agencies and if analyzed correctly, may provide a better picture of the overall species' range.

To demonstrate an alternative method for producing predictive species maps with more complicated datasets, we used the greater sage-grouse (GRSG) in Colorado for which multiple telemetry studies have been completed over the last 13 years in multiple populations across their range in the state. The GRSG is a species of conservation concern because of historical population declines and range contraction (Schroeder et al. 2004) and thus requires high quality seasonal distribution maps in the state of Colorado. Many anthropogenic developments (Knick and Connelly 2011) are stressors on GRSG habitat and populations. More recently, energy development in some of the largest energy reserves in western North America are located in regions characterized by sagebrush habitat (Copeland et al. 2009, Harju et al. 2010, Naugle et al. 2011). Identifying priority GRSG habitat that overlaps with potential development may be essential to the future management of the species (Doherty et al. 2010*a*). Currently, GRSG monitoring and modeling across its range has focused on breeding season specific locations. including nests and leks (spring breeding grounds), which are often a key source of information for these models (Walker et al. 2007, Copeland et al. 2009, Doherty et al. 2010*b*, Doherty et al. 2011, Aldridge et al. 2012). In addition, most of the current modeling methods for GRSG use some form of RSF, RSPF, or program Maxent (Homer et al. 1993, Aldridge and Boyce 2007, Yost et al. 2008, Doherty et al. 2008, Doherty et al. 2011).

Multiple populations of GRSG in Colorado are separated geographically and may exhibit seasonal and regional differences in habitat preferences. Regional variations, such as in these separate populations, and their habitat relationships across a broad geographic range are an emerging issue in ecology (McAlpine et al. 2008). Currently, Colorado utilizes an occupied range map based on field personnel expert opinion and observations. This map provides a generalized spatial extent of GRSG activity in Colorado, but there is no specific association between the habitat encompassed by the occupied range and GRSG habitat use. However, more detailed data exist in the form of radio-telemetry locations that provide information for six populations across the Colorado range. These data were collected during multiple smaller-scale studies on multiple individuals and contain information throughout the GRSG lifecycle including the breeding, winter and summer seasons.

Incorporating multiple datasets into habitat mapping along with information collected on individuals to predict distributions at a broader scale can be critical for large-scale population management (Klar et al. 2008). Generalized linear mixed models provide a method of analysis that accounts for variation and covariation in and among groups which may be more difficult with traditional RSF methods. By incorporating random effects in the model, variation that may exist between genotypes, species, regions or time periods can be investigated (Bolker et al. 2008). By accounting for differences among the six Colorado GRSG populations as well as differences in individual birds, we can provide a more accurate and valid assessment for the probability of presence of GRSG across multiple populations at a regional scale.

Modeling the frequency of GRSG locations as a continuous variable while accounting for data from multiple datasets and different populations provides a unique method for managing GRSG on a large scale where the knowledge of species absence may be unconfirmed or in areas where data has not been collected. It is the goal of this project to test this method of modeling GRSG distribution using seasonal land cover associations in northwest Colorado at a 1 km^2 scale. The ultimate goal is to provide refined seasonal GRSG distribution maps for managers and biologists to target prime areas of suitable habitat for future management and restoration projects. We want to accomplish this by accounting for as many assumptions and variability in these combined datasets for GRSG and extend habitat occupancy prediction across the entire range in Colorado.

STUDY AREA

The study area included Eagle, Garfield, Jackson, Moffat, Rio Blanco, and Routt counties in northwestern Colorado. This area is occupied by six populations of GRSG (Fig. 1): North Park (NP), Middle Park (MP), North Eagle/South Routt (NESR), Meeker (M), Parachute/Piceance/Roan (PPR), and Northwest (NW). This 49,470 km^2 study area encompasses the current range of GRSG in Colorado.

METHODS

Greater Sage-Grouse Location Database Compilation

Telemetry locations of GRSG were compiled from 1997 to 2010 in all populations except North Park (North Park data were collected after the initial project started and will be used as validation data). These data were not originally obtained with the goal of a distribution analysis, but instead mostly for demographic studies. This analysis used all GRSG telemetry studies in the state opportunistically with the idea of combining and summarizing overall GRSG habitat use patterns. Only individual GRSG that had ≥3 telemetry locations using VHF radio transmitters were included in the dataset used to develop the seasonal distribution models. All locations were projected to NAD83 UTM 13 and cross checked with the original data as well as with the other datasets to exclude possible duplicates. For each location, we included the population, individual, sex, age, date of location, year, season, and UTM coordinates. Some studies did not include sex or age in their datasets which resulted in a restricted set of information that included these variables. Therefore, age and sex were not used in this modeling effort at the overall range scale. For those datasets that did contain age and sex information, 90.8% (12909/14213) of telemetry locations were from females, 9.2% (1304/14213) were from males, 65.3% (9151/14005) were from adults, and 34.7% (4854/14005) were from juveniles.

Seasons were classified as follows: the breeding period was March through July, the summer period was July through September, and the winter period was December through February (modified from Connelly et al. 2000). Location data from October and November were not included because these months were considered transition periods (8.8% data removed). Our interest (? Intent, method?) was to analyze important seasons relevant to the GRSG population rather than transition periods where habitat choices may be indistinct. Populations were separated based on management boundaries used in Colorado (Fig. 1).

GIS Layers and Variables

We used GIS data layers to assess land cover associated with GRSG use locations. The land cover data were obtained from the former Colorado Division of Wildlife, now the Colorado Parks and Wildlife, “basinwide” land cover layer. This land cover layer was constructed with 25 m resolution landsat imagery as part of the Colorado Vegetation Classification Project (CVCP) administered by the Colorado Division of Wildlife in collaboration with the Bureau of Land Management and the U.S. Forest Service, which was completed in 2005. Initially, we used expert opinions about GRSG habitat use to combine 122 vegetation types in the original basinwide data into 13 land cover categories. As a result, across all counties, we included proportions of sagebrush (*Artemisia spp*), pinyon (*Pinus spp*)-juniper (*Juniperus spp*), agriculture, shrubland, urban, alpine, riparian, grassland, forest, bare, forest shrubland, mountain shrubland, and salt desert shrubland as variables in the model. Urban areas were removed as a variable because no GRSG locations occurred in this category. Subsequently, we overlaid the six county study area with 1km^2 grid cells which provided the sampling scale for summarizing the data. We used this scale to account for location error associated with multiple radio-telemetry methodologies used over numerous years. This grid size was also most appropriate for the large area in the analysis. Only cells that contained GRSG locations were included in model building as we had no information about GRSG presence or absence in other cells. Within each grid cell, the number of GRSG locations were summed and used as the response variable for the models.

Land cover was used to model GRSG habitat because GRSG are sagebrush obligates (Connelly et al. 2004, Connelly et al. 2011) and require sagebrush for food and cover, which may indicate dependence on vegetation types (Homer et al. 1993). GRSG require sagebrush through all phases of their life history to varying degrees, however the amount and composition of sagebrush in the landscape may differ among breeding, summer and winter seasons (Connelly et al. 2011). In addition, numerous other studies have demonstrated that vegetation type is an important influence on GRSG distribution (Aldridge and Boyce 2007, Doherty et al. 2008, Yost et al. 2008, Carpenter et al. 2010). Land cover also provided a consistent layer to evaluate GRSG habitat at the regional scale and generalizations based on basic resources such as land cover can provide a conservative accuracy for predicting species occurrence (Shanahan and Possingham 2009). More localized studies have indicated that topographic variables may be important predictors of GRSG distribution, such as elevation, slope, aspect, and terrain ruggedness (Doherty et al. 2008, Doherty et al. 2010*b*), but these variables likely operate at finer scales (<1 km^2). For example, in a 1 km^2 grid, there are roughly 1,100 grid cells in our study area and for continuous variables such as elevation, information related to actual GRSG locations can be lost.

Land cover variables were extracted using the Geospatial Modeling Environment (version 0.5.3 beta, Hawthorne L. Beyer, 2009-2011) tools and converted to proportion of land cover type within each grid cell where GRSG locations were recorded. The database was categorized into seasonal datasets based on breeding, summer, and winter locations. The proportions of each of the land cover types were the explanatory variables and were standardized (mean=0, SD=1) to address possible convergence issues (Bolker et al. 2008, Zuur et al. 2009). We calculated Pearson Correlation Coefficients (PCC) for the land cover variables for each seasonal dataset separately and removed those variables with a PCC > 0.60. No variables were removed from the breeding season, alpine and forest were removed from the winter model, and alpine, agriculture, and riparian were removed from the summer model. Each seasonal model was fit separately with the count of GRSG locations in each season as the response variable.

Data Analysis

We used a generalized linear mixed model (GLMM) to investigate if the variance could be explained by similarity in grouping variables. GLMM is a possibly non-Gaussian regression that incorporates random effects to separate potential effects on the models by co-varying factors (Martinez-Abrain et al. 2003, Bolker et al. 2008). By associating common 'random' effects to observations sharing the same level of a classification factor, mixed effects models more appropriately represent the covariance structure induced by grouping data (McAlpine et al. 2008). We were also more interested in the predictive versus explanatory power for predicting GRSG possible habitat, which lends itself to the GLMM approach (Goetz et al. 2012). We recognize that estimating habitat preferences by pooling telemetry data from all individuals is likely to bias toward data rich individuals (Aarts et al. 2008), but random effects allows us to recognize this unbalanced sampling effort such as the number of observations among individuals or groups (Gillies et al. 2006).

The response variable used in the models was a count based on the number of GRSG locations in a grid cell, with the proportion of land cover classes as fixed effects and individual and population as random effects. In our sample, we only used those grid cells that had known presence locations, so there were no zero counts in the dataset, which may bias our estimates (Creel and Loomis 1990). Therefore, we utilized a zero truncated Poisson distribution with a log link to model GRSG counts within each grid cell (Plackett 1953, Zuur et al. 2009). This type of censoring the data so that inference is conditional on observed presence is an adapted hurdle model. Hurdle models consist of two components. The first component models zero versus counts equal to or larger than 1 (Seifert et al. 2010). The second component is left truncated to exclude zero counts and determines the number of prediction events. In our model, we already had knowledge of presence with the GRSG locations, so we utilized the second part of a hurdle model which excludes zero counts.

Let y_{ijk} be the value of the grid cell count of telemetry locations for individual i, population j and grid cell k. The expected number of counts is log $u_{ijk} = \eta_{ijk}$, with linear predictor $\eta_{ijk} = x_{ijk*}\beta_j + z_{ijk}* \nu_i$ where x are fixed variables, β are the fixed variable coefficients, ν are the parameters of the random effect, and z are the levels of the random effect. Assuming a truncated Poisson process for count y_{ij}, the probability that $y_{ij} \geq y$ conditional on random effects ν, is

$$P(y_{ij} = y|\nu_i, x_{ij}, z_{ij}) = \frac{\mu_{ij}{}^{y} \exp^{(-\mu_{ij})}}{(1 - exp^{-\mu_{ij}})y!}$$
$$y = 0, 1, 2, \ldots$$

We investigated two grouping variables that may influence the GRSG dataset. The first random variable was radio-marked individuals, which account for multiple locations from individual GRSG. By including marked individuals as a grouping variable, we account for the lack of independence associated with multiple observations made on the same individual over time (Wagner et al. 2011). Including an individual random effect is also useful for accommodating overdispersion often observed with Poisson or count distributions (Breslow and Clayton 1993). A random intercept influences overall prevalence, which often arises because of unbalanced samples, in our case multiple studies from multiple populations or multiple locations per individual (Gillies et al. 2006). The second grouping variable of interest was the influence of population. We assumed relationships between GRSG and land cover variables in one population would be more similar than those in another population because of the large scale of this analysis. In addition to the random effects, the variance attributed to the possible autocorrelation in all three seasonal models was evaluated because ignoring dependence can lead to inaccurate and invalid conclusions (Olea 2009). We did this by using a semivariogram which defines the spatial scales over which patterns are dependent (Rossi et al. 1992). We used the residuals for each of the full seasonal models without random effects to construct each semivariogram.

We first ran each seasonal model with only an individual random effect, a population random effect, and a combination of both random effects without explanatory variables to determine if the random effects provided a better fit for the data based on a lower AIC value. In both cases, only the individual affect made an impact on the null model, so we therefore only used the individual random effect in the following models. We then fitted a set of all possible linear models from our set of 12 explanatory variables and the appropriate random effect(s) for each season and evaluated them with AICc (Burnham and Anderson 2002; Boccadori et al. 2008; McAlpine et al. 2008). We included sagebrush in all models as GRSG are sagebrush obligates and we were interested in its explanatory power (Aldridge and Boyce, 2007). We fit all combinations of variables along with sagebrush in every model because we did not have a clear reason to include or exclude the other land cover variables as subsets of all possible models. This resulted in 2,049 models in the breeding season model, 513 in the winter season model, and 257 summer season models. Doherty et al. (2010*c*) recommended using all model combinations with model averaging and Lukacs et al. (2010) determined that model averaging can protect against spurious results and is appropriate for developing a prediction tool (Burnham and Anderson 1998). Therefore, rather than selecting the one 'best' model, we also averaged across all reasonably well fitted models (Burnham and Anderson 2002, Bolker et al. 2008). Tables 2, 4, and 5 indicate that in each season, the top model has no higher than 0.14 (?)weight and no clear top model. Model averaging allows inference to be based on information from a number of models (Greaves et al. 2006). We calculated the relative importance of variables by summing the Akaike weights across all competing models in which each variable appears (Murray and Connor 2009). All GLMM models were fit with the *glmmADMB* library (Skaug et al. 2012) in the R statistical computing environment (R core development team 2012).

The averaged model for each season was then applied to the proportion of land cover raster layers in ESRI ArcMap (ArcGIS 10, Environmental Research Systems Institute, Redlands, Calif.). This resulted in a relative probability of GRSG occurrence layer for each seasonal model that ranged from 0 to 1. To standardize the probabilities for comparison between seasons, probability values were combined based on quartile breaks in the data for each seasonal model resulting in four categories of high, moderate, low and rare (Johnson et al. 2004, Sawyer et al. 2009). This means that a quarter of all values across the predictive surface in each season fall within each of the four bins. In addition, proportions of each prediction category were calculated for each of the current GRSG population boundaries in Colorado (Fig. 1).

The gold standard for any modeling exercise is the testing of predictions against an independently collected dataset (Wiens et al. 2008). In addition, it was important to test predictions in areas where data had not been collected to verify that the model results could be extrapolated across the GRSG range. We used two different independent datasets for comparison. The first validation dataset consisted of active leks collected across the GRSG range in our study area (*n* = 263) similar to methods used by Aldridge et al. (2012). These data are collected on a yearly basis and provide an especially good validation for the breeding model, but less so for the summer and winter models. Therefore, we used a second dataset that was collected specifically in the North Park population across all three seasons. This population was not included in the original analysis as data did not exist at that time, but has since been collected in a separate telemetry study. Initially, 95 females were radio-marked in April 2010 in the North Park population, with 51 still being monitored in March 2011. An additional 22 females were radio-marked in March 2011, providing a total of 117 GRSG and 3,132 locations to use in validating the rangewide seasonal distribution models. We separated the North Park validation dataset into seasons based on the original model for the best validation in each season. For both validation datasets, in each season we calculated the proportion of locations or leks that occurred in each quartile (Sawyer et al. 2007). A Spearman rank correlation was then used to assess the relationship between the predicted probability of occurrence for the original locations for each season and the test dataset locations in each quartile (Johnson et al. 2004). We would expect the presence locations from the validation datasets to be more prevalent in the high and moderate categories than the low and rare categories.

RESULTS

We found no patterns of latent spatial autocorrelation in any of the seasonal datasets (Appendix 1). We used 16,796 GRSG radio-telemetry locations from 959 individual GRSG in northwest Colorado from 1997-2010. Of the total locations, 74.1% were during the breeding season, 8% during the winter and 17.9% in the summer. GRSG location counts for each individual per population per grid cells ranged from 1-57 during the breeding, 1-12 during the winter, and 1-37 during the summer seasons. Only the individual random effect contributed to the variance in the breeding and summer models. The variances associated with these variables will be discussed in detail with each seasonal model.

In each season, sagebrush comprised the largest proportion of land cover in the 1 km^2 cells used by sage grouse (Table 1). The highest proportion of sagebrush was found during the winter and breeding seasons followed by the summer season. The second highest proportion of land cover type found in notable quantities during the winter period was salt desert shrubland. The second highest land cover type during the summer and breeding period was mountain shrubland.

For the breeding season model, the averaged model included all land cover variables (Fig. 2a, Table 2a). GRSG are negatively associated with alpine, bare, forest, and forested shrub, but had a positive association with all other variables (Table 2b). The relative variable importance indicated that bare, forest, mountain shrub and sagebrush were the more important variables found in the models that were averaged (Table 2b). The quartile cutoff values for the breeding season predictive surface were 0.04, 0.53, 0.76, and 1.00. North Park (NP) and Meeker had >90% of their population area categorized as high or moderate habitat (Fig. 2a; Table 3). The NW and PPR populations had >80% of their population area as high or moderate. The MP population had the least amount of highly or moderately suitable habitat with 65.6%. The individual random variable accounted for 13.2% of the variance in the model.

The averaged winter model included all variables except alpine and forest and indicated positive relationships with forested shrubland, grassland, mountain shrubland, pinyon-juniper, sagebrush, and salt desert shrubland (Fig. 2b; Table 4a,b). There were negative relationships with the other variables. Most of the variables in the winter model had a low contribution to the overall averaged model as variable importance was negligible (Table 4b). The top models for the winter season had very little support which indicated the lack of data in the winter model or perhaps the inability to distinguish land cover effects during the winter period. For this reason, we did not extend this model to a prediction surface as no variables significantly contributed to the averaged model.

The averaged summer model included all variables except riparian, alpine and agriculture (Fig 2b; Table 5a). GRSG were positively associated with grassland, mountain shrubland and shrubland, but negatively associated with the other variables (Table 5b). Bare, grassland, sagebrush, and shrubland were the most important variables in the averaged model (Table 5b). The quartile cutoff values for the summer model predictive surface were 0.23, 0.46, 0.55, and 1.00. The resulting probability map indicated 90.6% high and moderate habitat in Meeker and > 80% in PPR and NESR (Table 3). MP had the lowest probability of high and moderate habitat with 55.1%. The variance associated with the individual random effect was 37%.

Validation showed that both the breeding and summer seasonal models were good predictors for NP and the breeding model was also a good predictor for active lek datasets (Table 6). The original locations for the breeding model had a Spearman correlation of 0.9940 with NP locations and 0.9982 with the active lek locations. The original locations for the summer model had a spearman rank correlation of 0.7971 with the NP locations. The lower R^2 value for the summer model and NP locations was mostly driven by the high percentage of NP locations found in the high category and no locations in the rare and low categories.

DISCUSSION

Our analysis combined Colorado GRSG radio-telemetry data and modeled the distribution of GRSG presence in six counties. The seasonal habitat maps represent the basic land cover associations for GRSG in this region while accounting for possible covariation within individuals and populations. The resulting seasonal maps provide managers and biologists with a useful method for identifying large-scale high use areas throughout the annual lifecycle of the GRSG in Colorado.

The breeding and summer seasons predict suitable habitat within the current estimated GRSG range in Colorado. The summer habitat model is mostly driven by negative associations with forest, forested shrubland and bare ground, which follows patterns described in other habitat studies (Connelly et al. 1988, Connelly et al. 2011). There was a very slight negative relationship with sagebrush in the summer model which may suggest that GRSG use a much more diverse suite of land cover types in the summer period that include mesic non-sagebrush dominated cover types (Connelly et al. 2011).

The breeding season model was driven by negative relationships with alpine, bare, forest, and forested shrub. The absence of tall shrubs and trees appears to be critical for lekking (Colorado Greater

Sage-Grouse Steering Committee 2008). There was a positive association with grass in the understory which has been shown to be an important component of GRSG nesting and early brood-rearing sites (Connelly et al. 2000). The winter land cover may also be a limiting factor for GRSG based on this model, but more work needs to be done to confirm this.

The winter model had poor model performance due to a lack of support for any model which resulted in no significant variables in the model. Overall, there were only 1,348 locations from 446 individuals in the winter model compared to 3,000 in the summer and 12,444 in the breeding model. In order for a winter model to be feasible while accounting for variation in the dataset, more winter locations would need to be collected.

The effect of individual random effect was relatively large for the breeding and summer seasons where 13% or more of the variation was explained. Including this as a random effect in the models allows a more robust way to identify the explanatory variables that actually affect seasonal distribution (Vall-llosera and Sol 2009). It remains important to account for covariation in individual locations. Failure to account for this repeated measurement can lead to incorrect conclusions regarding effects of ecological variables and how those variables affect model selection and scientific conclusions (McAlpine et al. 2008).

The NP independent dataset and the active lek locations validated well for the breeding and summer season models. This indicates that overall, these models are robust in predicting areas of use that were not part of the original analysis or that were outside the scope of the objectives of the original telemetry locations. Most importantly, it is possible to build accurate and robust predictive distribution models based on functional resources such as land cover and extrapolate across a larger geographic range (Vanreusel et al. 2007). This can be important for management agencies when decisions need to be made for areas where sampling has been limited or non-existent. In the case of Colorado, it provides a refined occupied map of GRSG throughout the year.

At the broad scale of this study, it is unlikely that small differences between individual uses and population uses will be detected. For example, the breeding season habitat model includes lekking habitat for males and both nesting and early-brood rearing for females that can be found in a broad spectrum of sagebrush cover values (heavy cover to open areas). Our seasonal habitat map does not discern between the habitat occupied during these differing behaviors. Another issue with these probability maps is the hierarchical nature of habitat selection from the local scale immediately adjacent to the GRSG to the larger landscape (Johnson 1980). For example, taking measurements in a 1 m vegetation plot may indicate that a bird chooses pinyon-juniper for cover. Expand the scale to a 1 km grid cell and the vegetation in that grid cell might be 90% sagebrush and 10% pinyon juniper indicating that the individual strongly prefers sagebrush over pinyon-juniper. At the broad scale of these models, detecting specifics for individual birds or individual locations are not possible. In addition, terrain variables such as slope or elevation are masked when using a larger scale analysis such as 1 km^2 grids.

Each of the seasonal models indicates a high probability of GRSG in areas currently populated (Appendix B). This in itself provides a good validation for the models overall, but additional validation has been done with bootstrapping. The bootstrapping confidence intervals for all three seasonal models indicated that variable estimates were robust with 95% confidence. In addition, the all three seasonal models predicted habitat in the North Park population where no locations were included in this analysis. This provides further evidence that these models are robust and useful for management purposes.

At this broad scale, small differences between individual choices and population choices will not be detected. For example, the breeding season map includes lek attendance by males and nesting and early brood rearing by females that can be found in either heavy cover or open areas. This map will not decipher between these various behaviors. Another issue with these probability maps is that what a GRSG chooses directly around itself may be different from its location in a landscape. For example, taking measurements in a 1 m vegetation plot indicates a bird chooses pinyon juniper for cover. Expand the scale to a 1 km grid cell and the vegetation in that grid cell might be 90% sagebrush and 10% pinyon juniper indicating that individual prefers sagebrush. At the broad scale of these models, detecting specifics for individual birds or individual locations are not possible.

All three seasonal models provide managers and biologists on the ground a useful tool for interpreting general GRSG habitat characteristics for management actions at a large scale. It also provides maps that have been tested and validated so they can have confidence in broad scale decisions that may be made based on the map. Future analyses will investigate multiple scale issues and how it effects GRSG habitat predictions. This will include mapping habitat at the individual population scale and comparing it to the overall northwest prediction map.

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Table 1: Average values within each seasonal dataset for each land cover class (all values in proportion of land cover type)

Variable	Breeding	Summer	Winter
Agriculture	0.0196	0.0258	0.0131
Grassland	0.0428	0.0595	0.0231
Shrubland	0.0371	0.0334	0.0361
Sagebrush	0.7016	0.6402	0.7025
Salt desert shrub	0.0260	0.0121	0.1247
Pinyon juniper	0.0400	0.0385	0.0191
Mountain shrub	0.0863	0.1271	0.0355
Bare	0.0148	0.0093	0.0226
Forest shrub	0.0144	0.0283	0.0143
Forest	0.0036	0.0063	0.0008
Alpine	0.0006	0.0015	0.0003
Riparian	0.0124	0.0171	0.0071

Table 2: a) The top breeding season model from averaging all models based on all combinations of variables tested with sagebrush as a constant in all models (variables are abbreviated as follows: sb=sagebrush, ag=agriculture, gl=grassland, sd=salt desert shrub, sh=shrubland, pj=pinyon juniper, f= forest, fs=forested shrub, ms=mountain shrub, b= bare, a=alpine) b) coefficients for the top model-averaged estimates for the breeding model

a).

Model	AICc	Delta	Weight
a+ag+b+f+gl+ms+pj+r+sb+sd+sh	23796.27	0.00	0.13
a+ag+b+f+gl+ms+pj+r+sb+sh+fs	23796.87	0.60	0.09
a+b+f+fs+gl+ms+r+sb+sd+sh	23797.46	1.19	0.07
a+ag+b+f+fs+ms+pj+sb+sd	23797.65	1.38	0.06
ag+b+f+gl+ms+pj+r+sb+sd+sh	23797.86	1.59	0.06
a+b+f+fs+gl+ms+sb+sd+sh	23798.05	1.78	0.05
a+b+f+fs+gl+ms+pj+r+sb+sd+sh	23798.07	1.80	0.05
a+ag+b+f+fs+gl+ms+pj+r+sb+sd+sh	23798.07	1.80	0.05
a+ag+b+f+fs+gl+ms+r+sb+sd+sh	23798.67	2.40	0.04

b).

Variable	Coefficient	SE	Confidence interval Lower	Upper	Relative Variable Importance
Int	0.8626	0.0105	0.8421	0.8832	
a	- 5.4455	2.6390	- 10.6189	- 0.2721	0.82
ag	0.9949	1.5841	- 2.1100	4.0997	0.77
b	- 3.7752	1.2803	- 6.2847	- 1.2657	0.93
f	- 3.6238	1.3896	- 6.3477	- 0.9000	0.90
fs	- 1.2002	1.5373	- 4.2133	1.8130	0.80
gl	1.9087	1.4281	- 0.8904	4.7078	0.86
ms	2.1420	1.4596	- 0.7187	5.0027	1.00
pj	1.2312	1.5382	- 1.7837	4.2461	0.77
r	1.8682	1.4582	- 0.9900	4.7263	0.74
sb	1.1703	1.4617	- 1.6946	4.0352	1.00
sd	0.3097	1.5820	- 2.7909	3.4103	0.87
sh	1.8470	1.4206	- 0.9374	4.6315	0.86

Table 3: Proportion of high, moderate, low, and rare habitat in each of the six populations in the northwestern region of Colorado based on breeding, winter and summer seasonal models (populations are abbreviated as follows: PPR=Piceance/Parachute/Roan, NW=Northwest, NP=North Park, NESR= North Eagle/South Routt, MP=Middle Park, and M=Meeker)

		Breeding				Summer			
Population	Acreage	High	moderate	low	rare	High	moderate	low	rare
PPR	544993.9	0.4431	0.3993	0.1573	0.0001	0.4162	0.4302	0.1483	0.0053
NW	2650113.3	0.4784	0.3389	0.1810	0.0019	0.2909	0.4490	0.2454	0.0148
NP	414094.6	0.5928	0.3135	0.0936	0.0002	0.3980	0.9487	0.2519	0.0014
NESR	233416.9	0.3761	0.3905	0.2241	0.0092	0.2395	0.5752	0.1706	0.0147
MP	270747.1	0.2269	0.4286	0.3388	0.0057	0.2038	0.3472	0.3973	0.0517
M	164558.2	0.6316	0.2710	0.0931	0.0043	0.4792	0.4270	0.0870	0.0058

Table 4: a) The top winter season model from averaging all models based on all combinations of variables tested and sagebrush in all model (variables are abbreviated as follows: sb=sagebrush, ag=agriculture, gl=grassland, sd=salt desert shrub, sh=shrubland, pj=pinyon juniper, f= forest, fs=forested shrub, ms=mountain shrub, b= bare, a=alpine) b) coefficients for the top model-averaged estimates for the winter model

a).

Model	AICc	Delta	Weight
ag+b+gl+sb	1080.20	0.00	0.03
ag+b+gl+ms+sb	1080.33	0.13	0.03
ag+b+gl+pj+sb	1081.24	1.04	0.02
ag+b+gl+ms+pj+sb	1081.25	1.05	0.02
ag+b+gl+r+sb	1081.52	1.32	0.02
fs+gl+ms+pj+sb+sd+sh	1081.60	1.40	0.01
ag+b+gl+sb+sd	1081.61	1.40	0.01
ag+b+gl+sb+sh	1081.61	1.41	0.01
ag+b+gl+ms+sb+sh	1081.64	1.44	0.01
ag+b+sb+sd+sh	1081.65	1.44	0.01

b).

Variable	Coefficient	SE	Confidence interval Lower	Upper	Relative Variable Importance
Int	- 1.2697	0.0797	- 1.4260	- 1.1134	
ag	- 3.8513	3.6224	- 10.9553	3.2528	0.69
b	- 7.4574	3.7410	- 14.7955	- 0.1193	0.89
gl	4.3436	3.2761	- 2.0805	10.7978	0.71
sb	0.7003	2.4583	- 4.1192	5.5198	1.00
ms	2.4508	3.1183	- 3.6631	8.5647	0.57
pj	3.0205	3.7676	- 4.3671	10.4081	0.44
r	- 4.7878	6.1931	- 16.9354	7.3599	0.41
fs	1.8953	4.0669	- 6.0789	9.8696	0.34
sd	0.8478	3.5097	- 6.0330	7.7286	0.47
sh	- 0.1139	3.8633	- 7.6886	7.4609	0.39

Table 5: a) The top summer season model from averaging all models based on all combinations of variables tested with sagebrush in all models (variables are abbreviated as follows: sb=sagebrush, ag=agriculture, gl=grassland, sd=salt desert shrub, sh=shrubland, pj=pinyon juniper, f= forest, fs=forested shrub, ms=mountain shrub, b= bare, a=alpine) b) coefficients for the top model-averaged estimates for the summer model

a).

Variable	AICc	Delta	Weight
b+f+fs+gl+sb+sh	4941.75	0.00	0.14
b+f+gl+sb+sh	4942.55	0.80	0.09
b+f+fs+gl+pj+sb+sh	4943.13	1.38	0.07
b+f+fs+gl+sb+sd+sh	4943.15	1.40	0.07
b+f+fs+gl+ms+sb+sh	4943.69	1.94	0.05
b+f+gl+ms+sb+sh	4944.27	2.52	0.04
b+f+gl+sb+sd+sh	4944.33	2.58	0.04
b+f+gl+pj+sb+sh	4944.55	2.80	0.03
b+f+fs+gl+pj+sb+sd+sh	4944.59	2.84	0.03
b+fs+gl+sb+sh	4944.97	3.22	0.03

b).

Variable	Coefficient	SE	Confidence interval Lower	Upper	Relative Variable Importance
Int	0.3398	0.0294	0.2909	0.3887	
b	- 3.4166	1.3059	- 5.9780	- 0.8551	0.95
f	- 2.5111	1.1862	- 4.8377	- 0.1845	0.87
fs	- 0.8445	0.4832	- 1.7921	0.1031	0.66
gl	1.0848	0.4211	0.2589	1.9108	0.91
sh	0.9577	0.3762	0.2198	1.6955	0.89
sb	- 0.5511	0.2164	- 0.9754	- 0.1268	1.00
pj	- 0.2662	0.4382	- 1.1257	0.5933	0.32
sd	- 0.5163	0.7499	- 1.9870	0.9545	0.33
ms	0.0172	0.2909	- 0.5533	0.5877	0.31

Table 6: Percentage of the original locations, the lek locations, and the North Park locations as predicated by each season's model in each of 4 bins based on the quartile cutoff values along with the spearman rank correlation coefficient between the lek locations and the original locations as well as the North Park locations and the original locations.
a) Breeding

	Original locations	Lek locations	North Park Locations
Rare	0.000	0.000	0.000
Low	0.041	0.053	0.012
Moderate	0.278	0.251	0.227
High	0.681	0.696	0.761
		R^2=0.9982	R^2=0.9940

b) summer

	Original locations	North Park Locations
Rare	0.000	0.000
Low	0.066	0.092
Moderate	0.539	0.325
High	0.395	0.584
		R^2=0.7971

Figure 1. Populations (n = 6) of Greater sage-grouse in northwestern Colorado. North Park is not included in the initial model as no data was collected from this population at the time this model was generated.

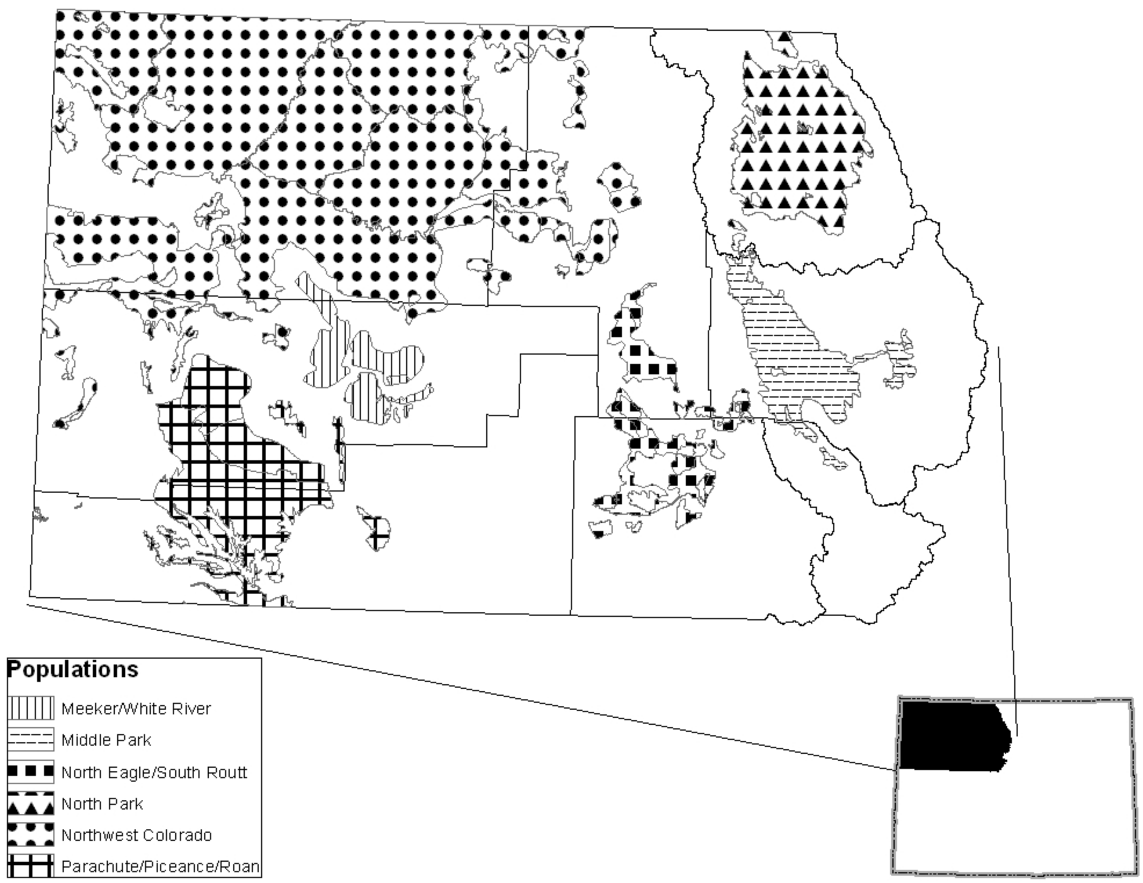
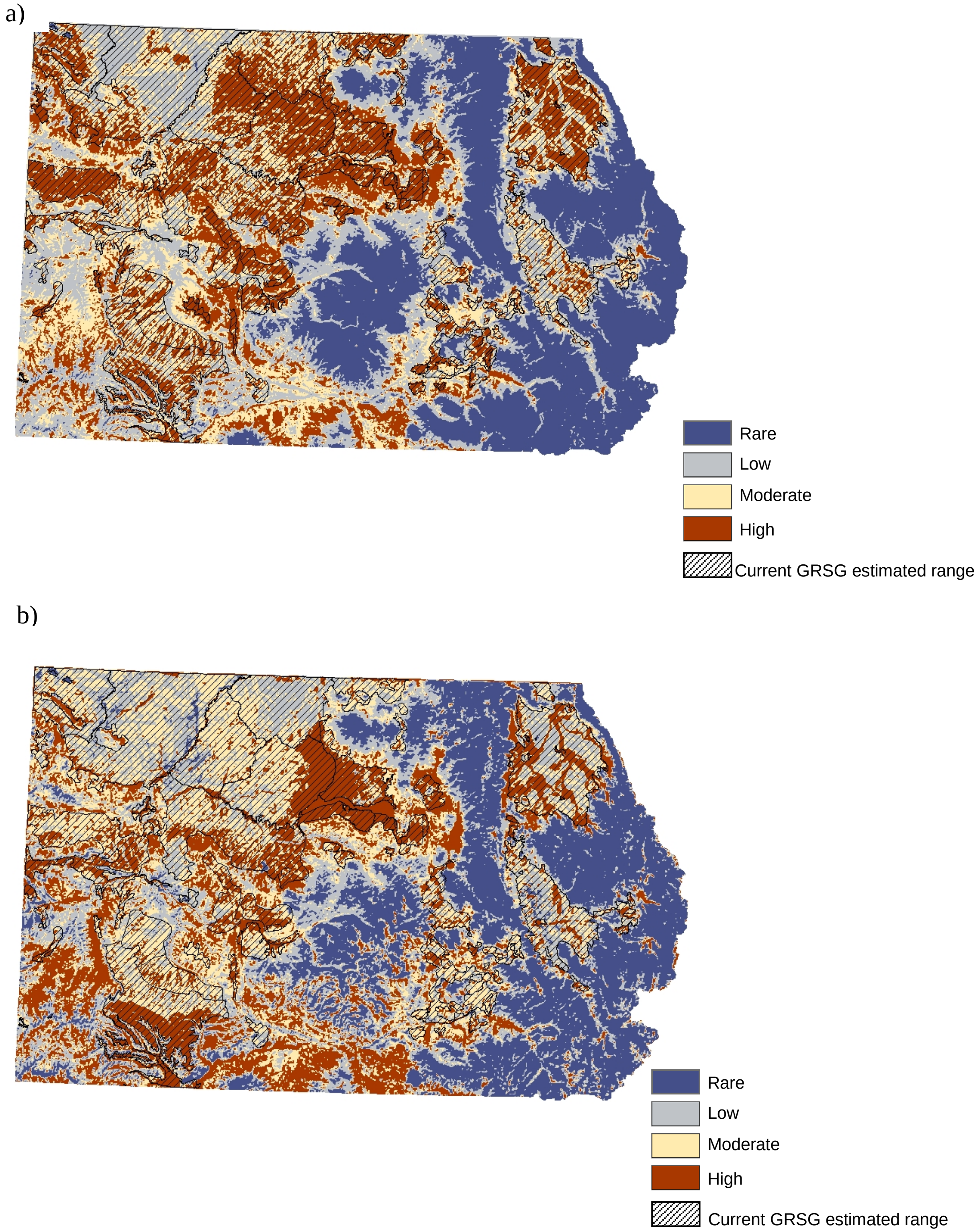
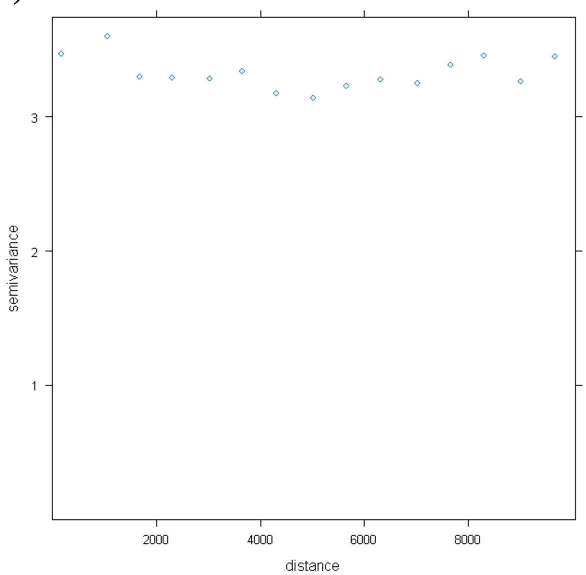


Figure 2. Probability of sage-grouse presence in northwest Colorado during the (a) breeding season and (b) summer season based on radio-telemetry locations from 1997-2010 with the current estimated range of GRSS.

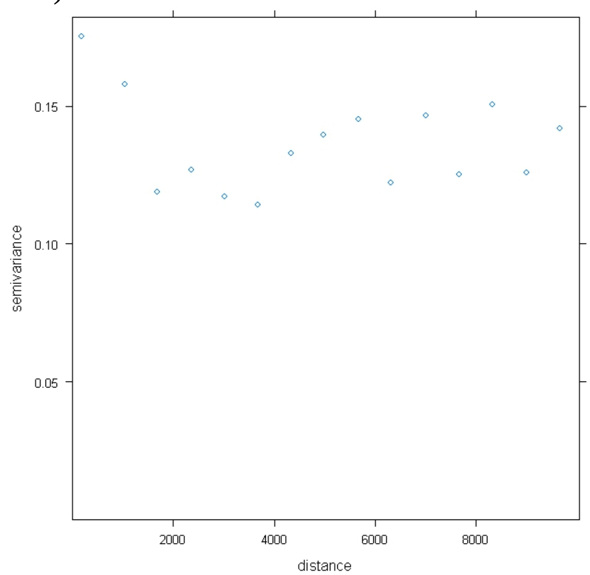


Appendix 1: variograms for each seasonal model based on all variables included without random effects to detect for autocorrelation within the data. a) breeding, b) winter, c) summer.

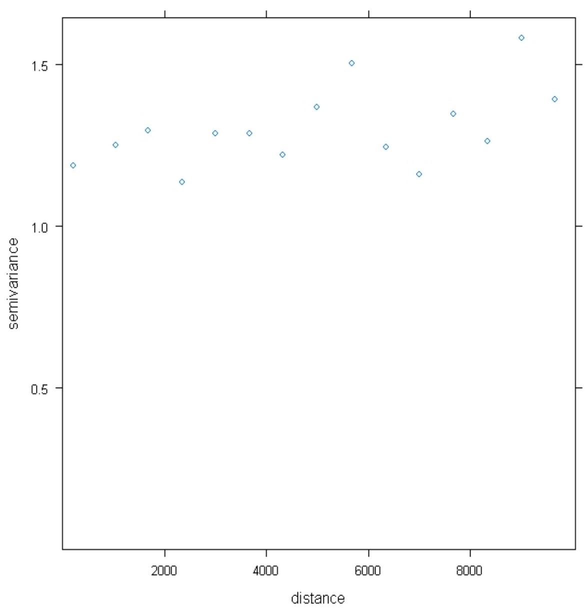
a)



b)



c)



Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3420	:	Avian Research
Work Package:	0660	:	Greater Sage-grouse Conservation
Task No.:	N/A	:	Greater sage-grouse seasonal habitat use and demographics in North Park
Federal Aid Project No.	N/A		

Period Covered: September 1, 2011 – August 31, 2012

Author: A. D. Apa, L. Rossi, and M. B. Rice

Personnel: J. Haskins, B. Petch, CPW; A.Timberman, U.S. Fish and Wildlife Service; North Park Greater Sage-Grouse Local Working Group

ABSTRACT

Rangewide declines and recent energy development within sagebrush habitat has led to concerns about conserving greater sage-grouse (*Centrocercus urophasianus*) (GRSG) populations across Colorado, including in North Park, which supports approximately 20% of the state's GRSG. Breeding, summer/fall, and winter habitat has been described at the local scale across the GRSG range and in Colorado. GRSG habitat use is known to be influenced by both landscape-scale factors, such as extent of sagebrush habitat and topography, and by local factors. However, the relative importance of local vs. landscape scale variables in habitat selection remains unknown. Also unknown is information on survival and reproductive effort that was obtained both before and during energy development, and comparisons with similar areas where development is not occurring. That information is needed to evaluate population-level responses of GRSG to energy development.

Spatially explicit, high-resolution maps depicting seasonal habitat areas within larger landscapes across the range of GRSG in Colorado would be useful for agencies and industry to make informed decisions for conservation and mitigation. On-the-ground efforts to map GRSG habitat within proposed oil and gas fields are expensive, time-consuming, and by necessity, limited in geographic scope. However, uniformly-applied mitigation buffers may include areas with non-critical habitat in which spatial and temporal restrictions on development could be relaxed. Conversely, uniform buffers may not adequately protect all required seasonal habitats because GRSG use a variety of seasonal habitats. The proposed EOG Resources Energy Development (EOG RED) is within occupied range of GRSG and includes seven active GRSG leks as well as two active leks adjacent to the EOG RED project area. The objectives of this research project are to:

1. Generate high-resolution digital maps showing seasonal GRSG seasonal habitat throughout North Park, the proposed EOG RED oil field and across the distribution of GRSG in Colorado.
2. Evaluate a hierarchical modeling approach to mapping GRSG seasonal habitat in North Park and across the distribution of GRSG in Colorado.
3. Provide current estimates of key demographic parameters (nest initiation rates and success rates, and juvenile, yearling, and adult survival) inside and outside the EOG RED areas as well as seasonal movement patterns inside and outside the EOG RED area.
4. Provide managers with estimates of local habitat variables in relation to established guidelines.
5. Provide managers with estimates of local habitat variables in relation to established guidelines.

To accomplish these objectives we radio-marked 117 female GRSG in April 2010 and 2011.We will continued to radio-track these GRSG through the winter 2011/12 and February 2012 using financial support provided by the U.S. Fish and Wildlife Service. We collected habitat measurements at used and unused locations for all seasonal habitats. Analysis will be completed and a manuscript prepared for publication in 2013.

WILDLIFE RESEARCH REPORT

GREATER SAGE-GROUSE SEASONAL HABITAT USE AND DEMOGRAPHICS IN NORTH PARK

ANTHONY D. APA, LIZA ROSSI, and MINDY B. RICE

PROJECT OBJECTIVES

The goal of this study is to obtain detailed, current information on GRSG habitat use and demography in North Park. Specific objectives include:

1. Generate high-resolution digital maps showing GRSG seasonal habitat throughout North Park and across the distribution of GRSG in Colorado.
2. Evaluate a hierarchical modeling approach to mapping GRSG seasonal habitat in North Park and across the distribution of GRSG in Colorado using physical and vegetation variables.
3. Provide current estimates of critical demographic parameters inside and outside the EOG RED areas as well as seasonal movement patterns inside and outside the EOG RED area.
4. Provide managers with estimates of local habitat variables in relation to established guidelines (Colorado Division of Wildlife (CDOW) 2009).

SEGMENT OBJECTIVES

1. Radio-mark female GRSG in North Park in spring 2010 and monitor movements, survival, nest and brood success.
2. Monitor used and unused sites for seasonal habitat model development.
3. Mark an additional sample of females and obtain a second season of demographic and movement data.
4. Continue to monitor previously marked females through February 2012 to obtain an addition winter habitat locations and habitat measurements.

INTRODUCTION

The greater sage-grouse (*Centrocercus urophasianus*) (GRSG) is a species of conservation concern due to historical population declines and range contraction (Schroeder *et al*. 2004), and there have been repeated attempts to list the species under the Endangered Species Act of 1973 (DOI 2005). Rapid, widespread energy development within sagebrush habitats of the western U.S. has raised additional concerns, as several recent studies have documented demographic impacts to GRSG in areas with active gas development (Lyon and Anderson 2003, Holloran 2005, Kaiser 2006, Aldridge and Boyce 2007, Walker et al. 2007). Extensive efforts have been made by industry and federal and state agencies to avoid, minimize and mitigate impacts of energy development on GRSG (CDOW 2008). Such efforts include wildlife surveys, environmental planning, alternative siting, and adherence to spatial and timing restrictions designed to minimize impacts to GRSG. However, the effectiveness of these efforts in reducing impacts on GRSG populations needs to be evaluated, and industry and agencies need better information to use in planning energy development activities.

North Park (Jackson County) is an important area for GRSG in Colorado, supporting approximately 20% of the statewide population (CDOW 2008). The proposed EOG Resources Energy Development (EOG RED) project encompasses most of the southwestern portion of North Park. The project area is also within occupied range of GRSG and includes seven active GRSG leks as well as two active leks adjacent to the project area. The Colorado Division of Wildlife (CDOW) is interested in developing information that will assist in avoiding impacts to GRSG through development planning for the EOG RED, and in better understanding GRSG response to energy development in North Park.

GRSG require sagebrush throughout the year. However, specific habitat requirements may differ among breeding, summer brood-rearing, fall, and winter seasons, and the juxtaposition of suitable areas of these different habitats determine the seasonal movements and distribution of GRSG throughout the year. (Connelly *et al.* 2000). Current patterns of seasonal habitat use by GRSG across the landscape in North Park are not well-documented. Sage-grouse habitat requirements at the local scale are generally well known, but no study to date has simultaneously addressed the influence of both landscape- and local-scale factors on GRSG habitat use. For example, soil type has never been included in habitat analyses. In other sagebrush-obligate species, specific soil types are key predictors of occupancy and abundance because of the direct influence of soil on the structure and composition of sagebrush (Vander Haegen *et al.* 2000). More research is needed to understand the full range of biotic and abiotic (i.e., current and historic energy development) factors influencing GRSG habitat selection.

Addressing wildlife requirements can be costly because it can result in delays in permitting, disruption of drilling and construction activities, seasonal lay-offs, and repeated revisions to maps and planning documents. On-the-ground efforts to identify important seasonal GRSG habitats within proposed oil and natural gas fields are expensive, time-consuming, and due to logistical constraints, limited in area. Moreover, fixed mitigation or avoidance buffers around critical seasonal habitats may include areas of non-critical habitat in which restrictions could be relaxed with little impact to GRSG populations, thereby reducing costs of planning and mitigation. Conversely, fixed buffers also may not adequately protect all seasonal habitats or impacts to habitats, and areas suitable for off-site mitigation need to be identified over a larger landscape. Thus, there is a need to identify and delineate important seasonal habitats for GRSG on a landscape scale prior to energy development.

Recent advances in modeling using high-resolution satellite imagery now allow researchers to more effectively classify and map seasonal habitat over large scales. These techniques provide spatially explicit information at a resolution sufficient to undertake detailed planning, mitigation and conservation efforts. This approach also allows for the delineation of seasonal habitats at local and landscape levels using a hierarchical (individual-population-statewide range) modeling approach. The approach allows external validation of selected models against independent datasets to ensure that findings are robust (Boyce et al. 2002, Johnson et al. 2006). There are six major populations of sage grouse in northwestern Colorado including the North Park, Middle Park, Meeker/White River, North Eagle/South Routt, Northwest, and Parachute/Piceance/Roan populations. Currently, there are just under 20,000 sage grouse telemetry locations in northwestern Colorado, but none of the locations are in the North Park population. The absence of data in North Park, which accounts for 20% of the state population, would bias a statewide habitat model as the variability within North Park would not be captured or included.

A preponderance of recent research on oil and gas development impacts on GRSG has been conducted in actively developing fields (Lyon and Anderson 2003, Holloran 2005, Kaiser 2006, Aldridge and Boyce 2007, Walker *et al.* 2007, Walker 2008). Assuming that oil and gas development will occur in the near future, North Park presents the unique opportunity for CDOW to collect baseline demographic data prior to substantial development. There has been little or no information collected, through either long-term research or retrospective analyses, on the long-term response of GRSG to historic development, where fields are in the production phase and sites have largely been reclaimed. North Park provides a unique opportunity to retrospectively assess the response of GRSG to historic development that occurred over 30 years ago.

In addition to quantifying accurate and precise estimates of habitat use at the local and landscape level, quantifying precise and accurate estimates of demographic parameters (survival rates, recruitment rates, etc.) is critical to successful conservation and management (Skalski et al. 2005) of GRSG. In a recent GRSG population viability analysis (CDOW 2008), juvenile and adult female survival was found to be crucial to population viability. There are no recent estimates of these demographic parameters for North Park and some demographic estimates were reported over 30 years ago. Current estimates of survival and reproduction of GRSG in North Park will provide a baseline for future monitoring and management, and allow comparison of demographic rates between the EOG RED project area and areas in North Park where development is not occurring, as well as comparison to historic estimates.

STUDY AREA

The study area includes the currently mapped occupied range of GRSG in North Park, Jackson County, Colorado. This study area includes the EOG RED project boundary, as well as adjacent areas in North Park that have proposed leases to facilitate modeling and increase our ability to make inferences to the entire North Park population of GRSG (Figs. 1 and 2).

METHODS

GRSG were captured and radio-marked during April 2010 and 2011 using spot-lighting techniques (Giesen et al.1982, Wakkinen et al.1994), and a four-wheel drive ATV. All GRSG captured were weighed (±1 g) using an electronic scale and marked with uniquely numbered aluminum leg bands. The age and gender of each GRSG was determined using wing (Dalke et al.1963) and other plumage or morphological characteristics. VHF transmitters were 17 g necklace-mounted radio transmitters with a 30 cm antenna lying between the wings and down the back of the grouse. Transmitters have a minimum battery life of 18 months and a four-hour mortality circuit. The radio transmitter package was 1.0% or 1.2% of the body weight for adult and yearling females, respectively.

Following release, the movements and survival of all radio-marked GRSG were monitored one to two times per week. GRSG general locations were determined by triangulation and radio-marked birds were not flushed. Hand-held Yagi antennas, attached to a receiver/scanner, were used to locate radio-marked grouse. The loudest-signal method was used to locate grouse/transmitters (Springer 1979). Monitoring efforts were distributed equally among three diurnal periods; morning (< 4 hours following sunrise), midday (> 4 hours after sunrise) and evening (< 4 hours before sunset). All grouse were circled at a 50 – 100 m radius (Apa 1998) to determine habitat type use. A precise Universal Transverse Mercator (UTM) location was not possible at the time of location (the bird will not be flushed out). To obtain more precise use locations, the observer selected a location approximately 50 m in one of the four cardinal directions from the estimated location of the bird. The observer took a Global Positioning System (GPS) location, and then manually corrected the UTM location. VHF collars allowed field crews to collect real-time local-scale data while in the field (i.e., snow depth, flock size and composition, etc.). A fixed-wing aircraft assisted to locate any grouse not located by ground monitoring or lost during ground monitoring efforts. General locations were identified aerially and ground locations will be identified within 48 hours.

When a female is incubating, the nest location was determined using binoculars as described by Apa (1998). Once a female is identified as incubating, she was not disturbed. A diagram of the nest location was drawn to assist with nest location after the cessation of nesting, when the precise UTM location was collected. A nest was considered successful if one egg hatches (Rearden 1951). At all nest sites, four 10 m transects were placed in the cardinal directions intersecting at the nest bowl. Nest shrub species and height were measured. The height of the lowest live and dead nest bush branch above the nest bowl was measured from the edge of the nest bowl. Canopy cover (foliar intercept) of the shrub species over-story was determined using line intercept (Canfield 1941). The intercept by the lowest possible taxa was measured. Height of the of the nearest nest bush type shrub within 1 m of the transect line was measured at 2.5 m, 5 m, and 10 m. Grass and forb height was measured for the nearest, tallest grass/forb part at the point where the edge of the nest bowl and the transect intersected, and at the 2.5 m, 5 m, and 10 m point on each transect.

The percent of forbs and grass cover, bare ground, and litter horizontal understory cover was estimated using 50 x 50 cm micro-plots (Daubenmire 1959). Eleven cover classes were used and delineated as follows: Trace: 0-2%, 1) 3-9%; 2)10-19%; 3) 20-29%; 4) 30-39%; 5) 40-49%; 6) 50-59%; 7) 60-69%; 8) 70-79%; 9) 80-89%; 10) 90-100%. A single micro-plot was located at the nest bowl. Subsequent plots were placed systematically along the transects at 2.5, 5 and 10 m. In addition, the distance to nearest visible roadways, telephone poles, powerlines, and fence posts were determined. The same vegetation data collection techniques were applied to at least one random location for each nest. Random locations were obtained by using randomly selected UTM coordinates located in cells considered "unused" in the study area based on a spatially balanced random design.

Females with broods and unsuccessful females were located one to two times per week. At each location, date, time, UTM coordinates, slope and aspect were recorded. Unsuccessful females were located in the same manner as females with broods. Microhabitat variables were measured at brood and unsuccessful female sites. We estimated survival probabilities using the Kaplan-Meier product limit estimator (Kaplan and Meier 1958) with staggered entry design (Pollock et al. 1989a, Pollock et al. 1989b).

RESULTS AND DISCUSSION

A total of 95 female GRSG (Yearling = 22, Adult = 73) were radio-marked throughout the study area in April 2010. There were 51 remaining females and being tracked in March 2011, and an additional 22 females (Yearling = 18, Adult = 3, Unknown = 1) were radio-marked in 2011. In 2010 and 2011, 86 and 62 nests were documented, respectively. In 2010, apparent nest success was 47% (n = 40/86) and female success was 44% (n = 40/90). In 2011, apparent nest success was 64% (n = 40/62) and female success was 61% (40/65). Additionally, in 2010 and 2011 nest initiation rates were 88% (n = 80/91) and 85% (n = 55/65), respectively.

To date more than 4,500 use locations have been obtained. Use locations suggest that North Park is extensively used during the breeding (Fig. 3), summer (Fig. 4) and winter (Fig. 5) seasons. Vegetation sampling was conducted on 132 nest, 115 non-use, 28 brood sites and 13 non-brood female, and 30 winter sites and these data are in the analyses phase. The vegetation data has been summarized and include the variables of total shrub cover (TOTSHRUBCC), grass height (GRASSHT), big sagebrush height (SAGEHT), total sagebrush cover (TOTSAGECC), forb cover (FORBCOV), perennial grass cover (PERGRASSCOV), bare ground (BAREGRD), litter cover (LITTERCOV), non-sagebrush cover (NONSAGECC), forb height (FORBHT), sagebrush nest bush height (SAGEHTNESTBUSH) (this variable represents sagebrush height at the 0 m point (?) for the four intersecting transects), forb species richness (FORBRICHNESS) and grass species richness (GRASSRICHNESS) for nest (Table 1), brood (Table 2), non-use (Table 3), and non-brood female (Table 4) sites.

Twenty-two and 73 yearling and adult females were used to estimate annual survival from April 2010 through March 2011, respectively (Fig. 6). Annual survival for yearling females was 0.82 ± 0.02 (95% CI = 0.66 – 0.97) and 0.49 ± 0.01 (95% CI = 0.37 – 0.60) for adult females. These estimates are generally consistent with a previous survival estimate for yearling (0.78 ± 0.03; 95% CI 0.72 – 0.75) and adult (0.59 ± 0.01; 95% CI 0.57 – 0.61) females for a 17-year banding study in North Park (Zablan *et al.* 2003). Yearling female survival was relatively consistent and steady through January followed by a small decline in February and March (Fig. 6). Adult female survival (females two years of age or older) declined consistently from capture through March 2011 (Fig. 6). The 2010-11 winter was considered severe and the annual survival rate for yearling females was higher than previously reported and was slightly lower than previously reported for adult females in North Park (Zablan et al. 2003). Additional data analyses are on-going with a final report planned for mid-2012. Since March 1, 2012, no additional data has been collected and project authors are in the process of proofing, editing and analyzing data to accomplish and fulfill the objectives of this study.

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Table 1. Vegetation structure variable mean, SE, minimum and maximum values at greater sage-grouse female nest sites in North Park, Colorado, 2010.

	SITE			
	NEST		RANGE	
VARIABLE	***n***	**Mean ± SE**	**MIN**	**MAX**
TOTSHRUBCC	82	49.0 ± 1.4	23.4	81.8
GRASSHT	82	13.7 ± 0.5	7.0	32.2
SAGEHT	82	39.9 ± 2.2	12.3	177.4
TOTSAGECC	82	42.2.± 1.5	0.0	72.8
FORBCOV	82	10.0 ± 1.1	0.1	46.5
PERGRASSCOV	82	22.4 ± 1.1	4.9	51.7
BAREGRD	82	28.4 ± 1.9	0.4	70.0
LITTERCOV	82	60.0 ± 1.9	20.8	89.2
NONSAGECC	82	6.7 ± 1.1	0.0	63.4
FORBHT	81	7.7 ± 0.6	2.2	35.0
SAGEHTNESTBUSH	82	57.1 ± 1.6	32.0	124.0
FORBRICHNESS	81	5.4 ± 0.3	1.0	13.0
GRASSRICHNESS	81	4.5 ± 0.2	1.0	8.0

Table 2. Vegetation structure variable mean, SE, minimum and maximum values at greater sage-grouse female brood sites in North Park, Colorado, 2010.

	SITE			
	BROOD		RANGE	
VARIABLE	***n***	**Mean ± SE**	**MIN**	**MAX**
TOTSHRUBCC	28	19.0 ± 3.6	0.0	55.3
GRASSHT	28	13.5 ± 1.1	6.1	37.5
SAGEHT	28	33.1 ± 3.9	14.3	63.9
TOTSAGECC	28	13.5.± 3.1	0.0	47.0
FORBCOV	27	15.5 ± 2.3	1.2	44.2
PERGRASSCOV	28	43.3 ± 4.9	5.8	85.8
BAREGRD	28	19.3 ± 3.5	0.1	55.8
LITTERCOV	28	53.0 ± 4.0	21.1	91.1
NONSAGECC	28	5.5 ± 1.6	0.0	33.9
FORBHT	81	7.7 ± 0.6	2.2	35.0
SAGEHTNESTBUSH[1]	14	39.4 ± 5.8	10.0	89.0
FORBRICHNESS	28	5.5 ± 0.5	1.0	11.0
GRASSRICHNESS	28	4.9 ± 0.3	2.0	7.0

[1]A nest is not located at this site. This variable is representative of a sagebrush bush measured at the transect intersection for comparison to the same variable in Table 1.

Table 3. Vegetation structure variable mean, SE, minimum and maximum values at unsuccessful greater sage-grouse female use sites in North Park, Colorado, 2010.

	SITE			
	FEMALE SUMMER		RANGE	
VARIABLE	***n***	**Mean ± SE**	**MIN**	**MAX**
TOTSHRUBCC	13	17.8 ± 4.2	0.0	42.2
GRASSHT	13	13.2. ± 3.3	5.8	51.4
SAGEHT	10	23.3 ± 2.0	12.5	30.8
TOTSAGECC	13	15.2.± 3.7	0.0	40.2
FORBCOV	13	16.1 ± 5.7	0.8	76.5
PERGRASSCOV	13	32.6 ± 6.6	8.9	87.3
BAREGRD	13	23.8 ± 5.3	0.0	56.9
LITTERCOV	13	54.6 ± 5.8	26.5	91.1
NONSAGECC	13	2.6 ± 0.8	0.0	9.1
FORBHT	13	7.2 ± 1.9	2.3	29.5
SAGEHTNESTBUSH[1]	9	24.8 ± 4.5	3.0	46.0
FORBRICHNESS	13	4.8 ± 0.6	2.0	9.0
GRASSRICHNESS	13	4.4 ± 0.3	3.0	7.0

[1]A nest is not located at this site. This variable is representative of a sagebrush bush measured at the transect intersection for comparison to the same variable in Table 1.

Table 4. Vegetation structure variable mean, SE, minimum and maximum values at non-use sites in North Park, Colorado, 2010.

	SITE			
	NONUSE		RANGE	
VARIABLE	***n***	**Mean ± SE**	**MIN**	**MAX**
TOTSHRUBCC	115	24.5 ± 1.8	0.0	84.4
GRASSHT	113	15.4 ± 1.0	4.7	87.3
SAGEHT	78	30.6 ± 1.9	6.3	97.0
TOTSAGECC	115	15.5 ± 1.6	0.0	79.9
FORBCOV	109	14.7 ± 1.4	0.0	77.3
PERGRASSCOV	109	30.8 ± 2.3	0.4	94.2
BAREGRD	114	27.1 ± 2.0	0.0	85.8
LITTERCOV	114	54.6 ± 2.4	6.5	100.0
NONSAGECC	82	6.7 ± 1.1	0.0	63.4
FORBHT	108	8.5 ± 0.7	0.7	41.7
SAGEHTNESTBUSH[1]	71	28.2 ± 2.4	4.0	80.0
FORBRICHNESS	112	5.3 ± 0.3	0.0	14.0
GRASSRICHNESS	113	4.3 ± 0.2	1.0	11.0

[1]A nest is not located at this site. This variable is representative of a sagebrush bush measured at the transect intersection for comparison to the same variable in Table 1.

Figure 1. Study area depicting land ownership, core area designation, and lek and oil and gas well density in North Park, Jackson County, Colorado, 2010.

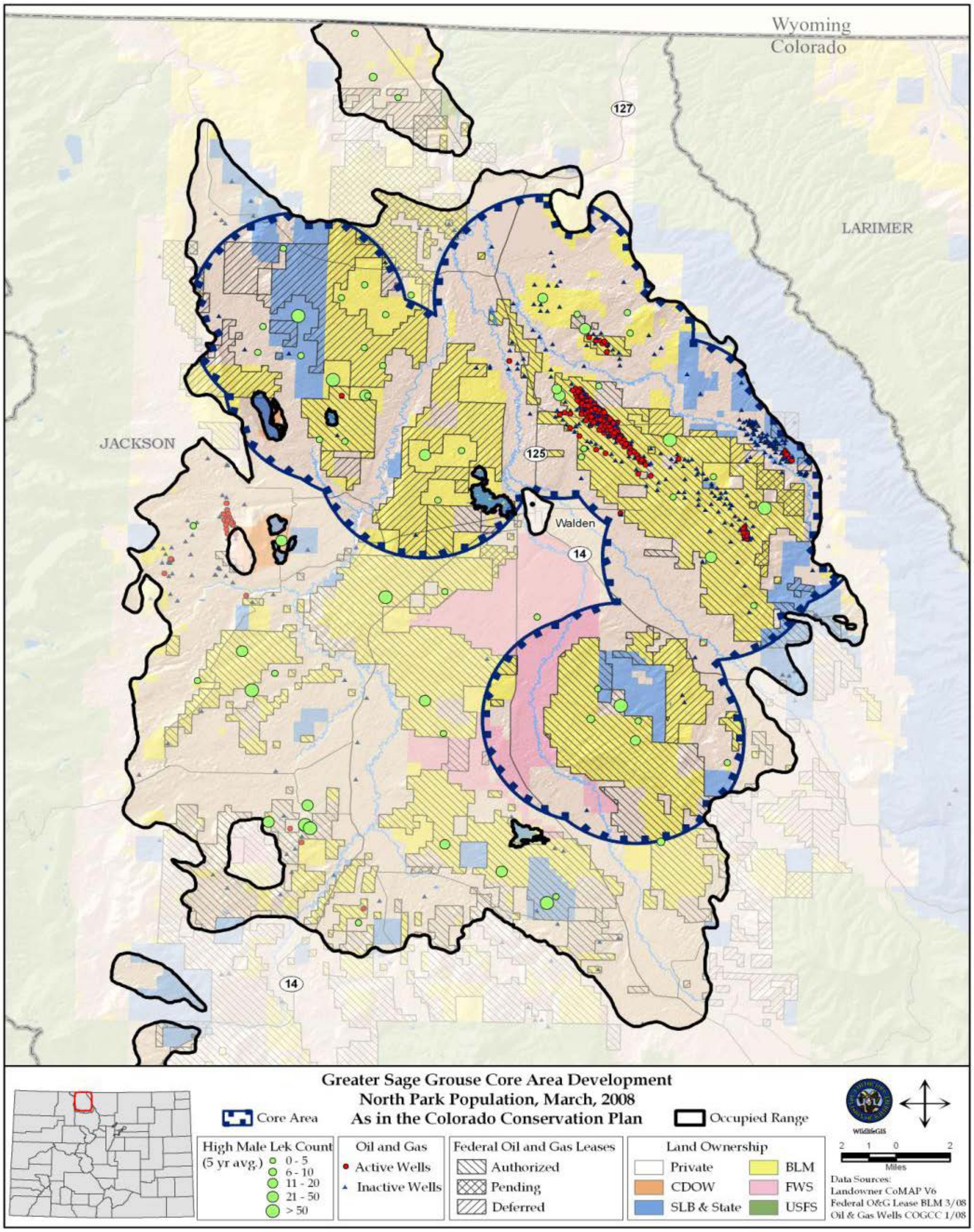


Figure 2. Study area depicting lek locations, core areas and varying status of EOG exploration activities in North Park, Jackson County, Colorado, 2010.

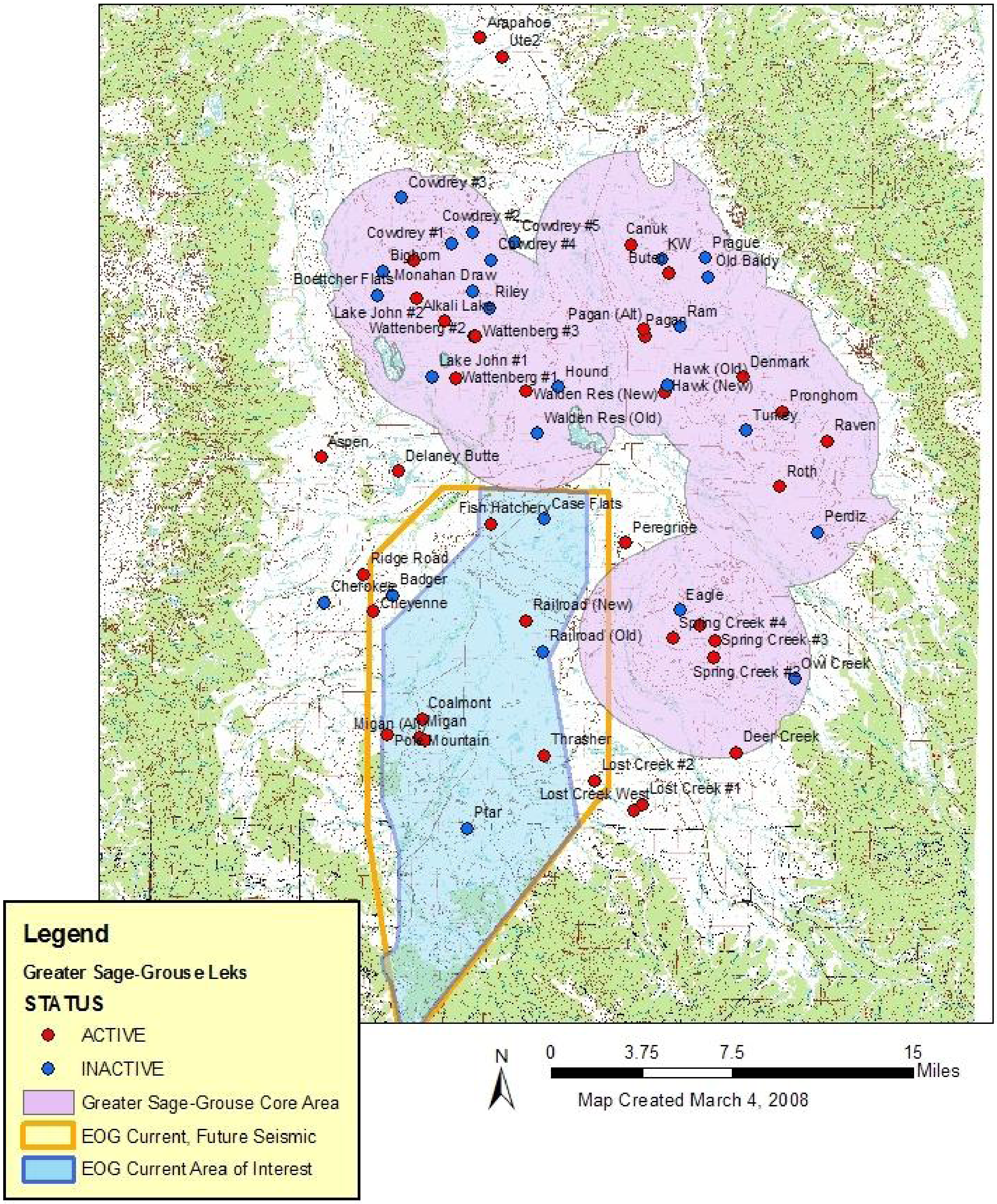


Figure 3. Female greater sage-grouse locations in April, May, and June in North Park, Colorado, 2010.

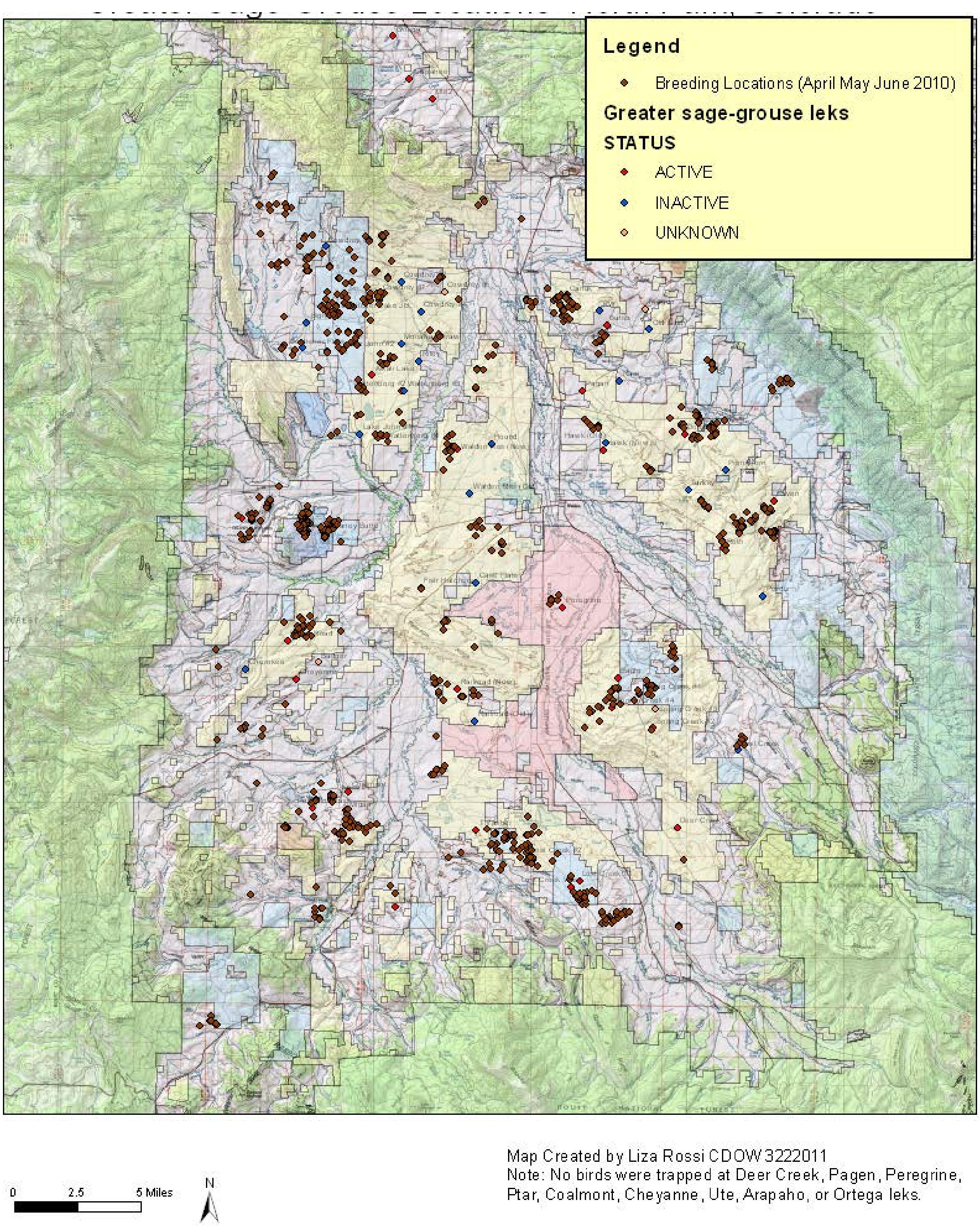


Figure 4. Female greater sage-grouse locations in July and August in North Park, Colorado, 2010.

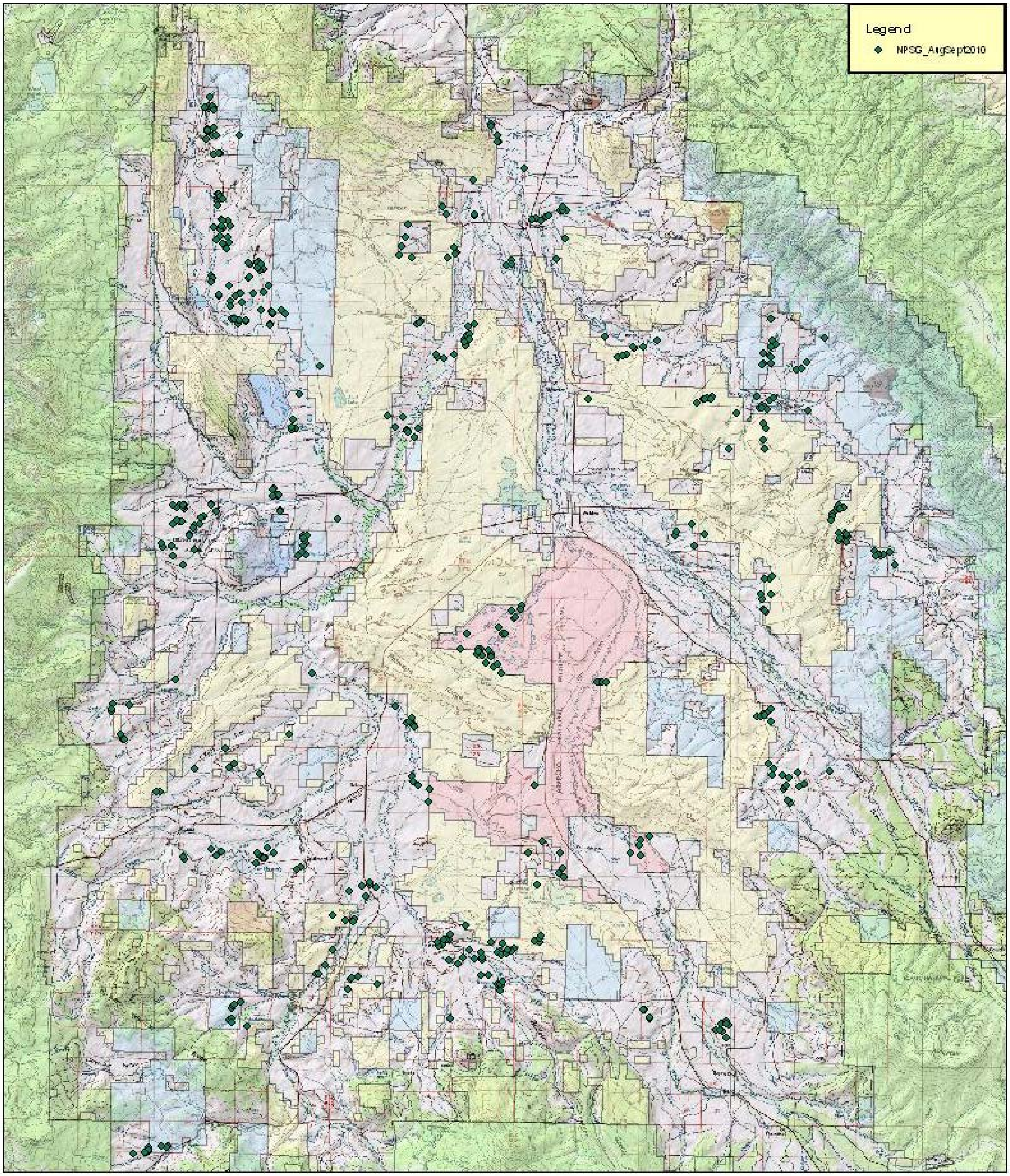


Figure 5. Female greater sage-grouse locations in January and February in relation to previously mapped sage-grouse severe winter range and non-severe winter range in North Park, Colorado, 2011.

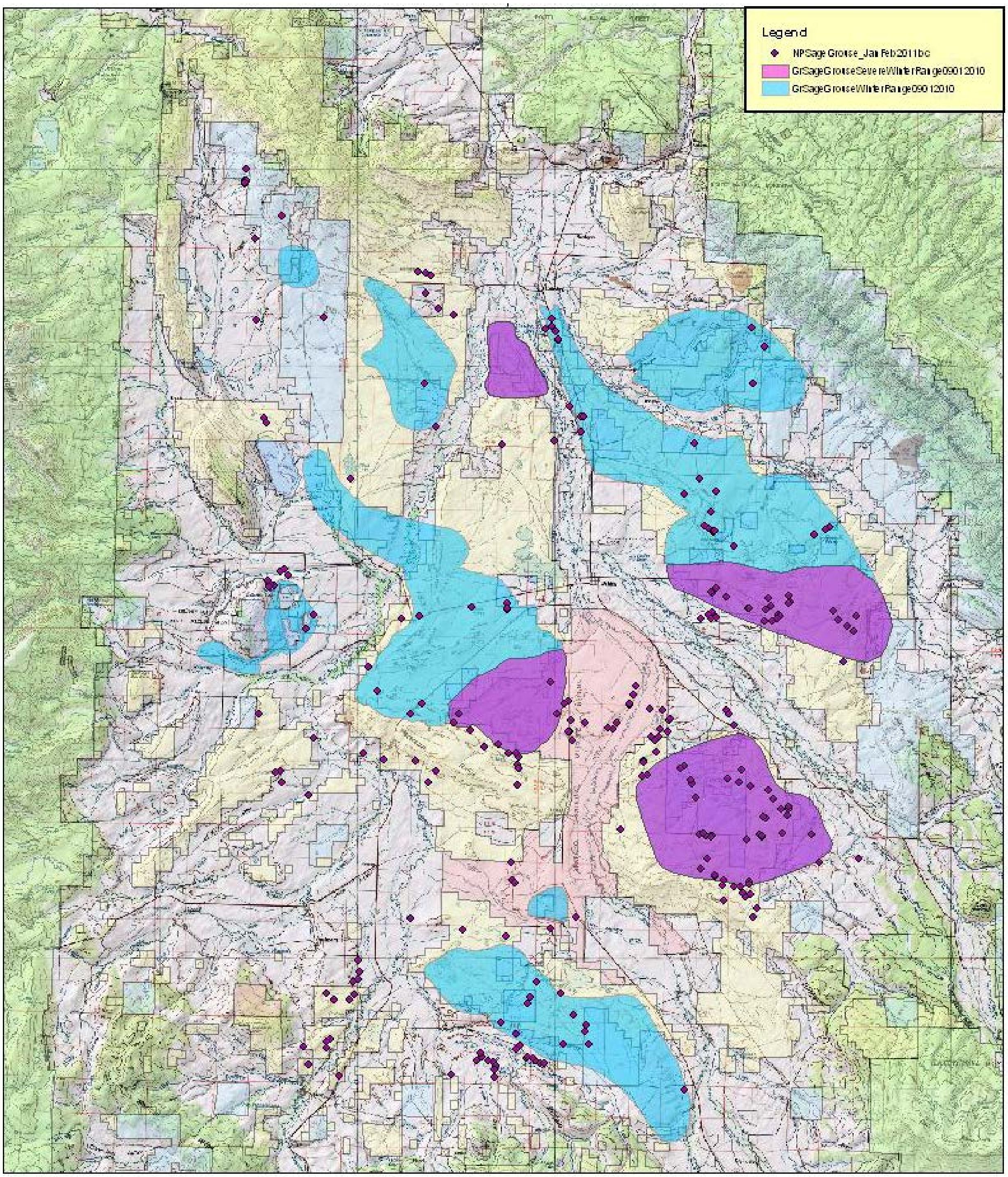
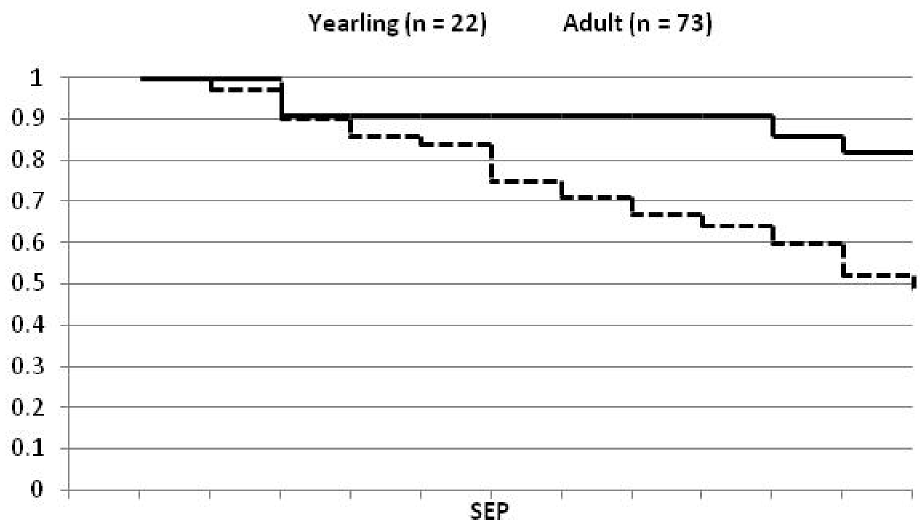


Figure 6. Annual product-limit survival curve for radio-marked female greater sage-grouse by age class in North Park, Colorado, April 2010 – March 2011.



Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	0660	: Greater Sage-grouse Conservation
Task No.:	N/A	: Using GPS satellite transmitters to estimate survival, detectability on leks, lek attendance, inter-lek movements, and breeding season habitat use of male greater sage-grouse in northwestern Colorado

Federal Aid
Project No. N/A

Period Covered: September 1, 2011 – August 31, 2012

Author: B. L. Walker

Personnel: B. Holmes, B. Petch, B. deVergie

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ABSTRACT

Implementing effective monitoring and mitigation strategies is crucial for conserving populations of sensitive wildlife species. Concern over the status of greater sage-grouse (*Centrocercus urophasianus*) populations has increased both range-wide and in Colorado due to historical population declines, range contraction, continued loss and degradation of sagebrush habitat, and recent federal listing of the species as warranted but precluded under the Endangered Species Act in 2010. Despite untested assumptions, lek-count data continue to be widely used as an index of abundance by state and federal agencies to monitor sage-grouse populations. Lek locations are also commonly used to identify and protect important sage-grouse habitat. However, the use of lek counts and lek locations to monitor and manage sage-grouse populations remains controversial because it is unknown how closely lek-count data track actual changes in male abundance from year to year, or if lek buffers are effective at reducing disturbance to male sage-grouse and their habitat during the breeding season. Colorado Parks and Wildlife is color-banding and deploying solar-powered GPS transmitters on male greater sage-grouse and conducting double-observer counts and resighting at leks to obtain data on male survival, lek attendance, inter-lek movements, detectability, and diurnal and nocturnal habitat use around leks during the breeding season in and near the Hiawatha Regional Energy Development project area in northwestern Colorado. These data will allow us to evaluate whether GPS transmitters have an impact on males, whether current lek-based monitoring methods provide reliable information about sage-grouse population trends, and whether current lek buffers are effective for conserving greater sage-grouse. We captured 42 non-juvenile (yearling or adult) males in fall-winter 2011-2012 and 25 juvenile males in late winter 2012. Of these 67 males, we deployed GPS transmitters on 47 (23 non-juveniles and 24 juveniles). Field crews discovered two new potential leks in 2012 by checking clusters of previous early-morning locations of GPS males, and they confirmed strutting again at three leks newly discovered in 2011. Field crews conducted 90 standard lek

counts at 26 different leks, 131 mornings of resighting on 15 different leks, and 58 unreconciled double-observer counts at 16 leks. Male survival, lek attendance, and inter-lek movement data are still being analyzed. We documented round-trip seasonal movements of males of 34-58 km and one-way movements of 25-59 km. Problems with color-band retention observed in 2012 may preclude mark-resight analyses and comparison of color-banded vs. GPS male survival. Field work will be completed in 2013.

COLORADO PARKS AND WILDLIFE RESEARCH REPORT

USING GS SATELLITE TRANSMITTERS TO ESTIMATE SURVIVAL, DETECTABILITY ON LEKS, LEK ATTENDANCE, INTER-LEK MOVEMENTS, AND BREEDING SEASON HABITAT USE OF MALE GREATER SAGE-GROUSE IN NORTHWESTERN COLORADO

BRETT L. WALKER

PROJECT OBJECTIVES

1. Test the effect of GPS transmitters on male greater sage-grouse:
    a. Estimate and compare seasonal and annual survival rates of yearling and adult male greater sage-grouse with GPS transmitters to published and empirical estimates for leg-banded males.
    b. Compare fitted leg-loop size for adult vs. yearling males to assess whether yearling males will outgrow harnesses; if needed, recapture yearling males and refit harnesses in the field.
    c. Compare strutting display rates between GPS males and color-banded or unmarked males.
    d. Compare raw lek attendance rates between GPS males and color-banded males.
2. Use locations of GPS males to locate, verify, and count new leks in and around the study area.
3. Estimate the number of known and unknown leks in the study area
4. Use unreconciled double-observer lek counts and time-to-detection models with lek-count and resighting data to estimate detectability of males attending leks.
5. Develop a modified sightability model approach to estimate daily, seasonal, and annual variation in male lek attendance.
6. Use movements of GPS males to determine presence near leks in the study area and to estimate the frequency, timing, and distance of breeding-season movement among leks.
7. Estimate daily and breeding-season survival rates of GPS males.
8. Use simulations to quantify how variation in age-specific male survival, presence, detectability, lek attendance, movement, and count frequency affect lek count indices and trend estimation.
9. If possible, use mark-resight data and counts of marked and unmarked males and females at leks to generate annual estimates of age- and sex-specific population size.
10. Quantify male habitat use and movement around leks to test the effectiveness of current oil and gas lease stipulations for lek buffers.

SEGMENT OBJECTIVES

1. Capture and color-band enough non-juvenile (adult or yearling) males in fall to maintain an approximate sample size of 60, and deploy GPS transmitters on half the males captured.
2. Capture and deploy GPS transmitters on 30 juvenile male greater sage-grouse in Feb-Mar 2012.
3. Locate, verify, and count new leks confirmed during the 2012 breeding season.
4. Resight color-banded males on leks attended by GPS males.
5. Conduct standard lek-counts and unreconciled double-observer at leks attended by GPS males
6. Enter and proof field data.

INTRODUCTION

Greater sage-grouse (*Centrocercus urophasianus*) are a major conservation concern due to historical population declines, range contraction, and recent federal listing of the species as warranted but precluded under the Endangered Species Act (Connelly et al. 2004, Schroeder et al. 2004, USFWS 2010). The species continues to be threatened by ongoing anthropogenic and ecological changes to their habitat, including residential housing development, wildfire, invasive plants, pinyon-juniper encroachment, West Nile virus, agricultural conversion, and energy development (Connelly et al. 2004, CGSSC 2008, USFWS 2010). Accurately monitoring changes in sage-grouse abundance is crucial for assessing the current conservation status of populations, for quantifying responses of populations to potential stressors, and for documenting success or failure of conservation and mitigation efforts. Management strategies to protect sage-grouse habitat must also be validated to ensure they are effective at preventing unwanted impacts to populations.

Greater sage-grouse populations are typically monitored and managed using data collected at leks. Each spring, male sage-grouse congregate on traditional mating grounds, or leks, to display and mate with females (Schroeder et al. 1999). Males attending leks are then counted by observers on the ground or from aircraft following standardized protocols (Jenni and Hartzler 1978, Beck and Braun 1980, Autenrieth et al. 1982, Connelly et al. 2000). Lek counts are the primary index used by all state wildlife agencies in the western U.S., including the Colorado Division of Wildlife, to monitor sage-grouse population trends (Connelly et al. 2004, CGSSC 2008, WAFWA 2008). Changes in lek size and lek persistence derived from lek count data are also used to investigate how regional and range-wide populations respond to changes in habitat and to anthropogenic stressors such as oil and gas development (e.g., Braun et al. 2002, Walker et al. 2007, Aldridge et al. 2008, Doherty at el. 2010*b*, Harju et al. 2010, Tack 2010). Lek locations are also used to help identify and protect important habitat in land-use planning efforts because leks are typically centrally located within breeding areas (Gibson 1996, Doherty et al. 2010*c*). For example, federal oil and gas lease stipulations include timing and surface occupancy restrictions on oil and gas development within specific distance buffers around sage-grouse leks to minimize disturbance to males and their habitat during the breeding season. Many state and regional “core areas” have been delineated by placing buffers around known leks that meet male count and lek density criteria (e.g., CGSSC 2008, Doherty et al. 2010*a*, Hagen 2010, State of Wyoming 2010).

The importance of accurate and effective monitoring and management strategies is heightened in areas slated for energy development. A major threat factor in the listing decision was expanding energy development in the eastern portion of the range (USFWS 2010). Accumulated evidence suggests that sage-grouse populations show substantial declines following oil and gas development, even when standard mitigation measures are implemented (e.g., Holloran 2005, Walker et al. 2007, Doherty et al. 2008, Harju et al. 2010, Holloran et al. 2010). However, measured population responses to oil and gas development, while consistently negative, are not always of the same magnitude due to variation in: (a) the intensity of development; (b) the type of infrastructure required to develop the resource, which in turn affects the ecological processes by which impacts occur; (c) lag times between development and detection of impacts; (d) inherent differences in habitat quantity and configuration among populations subject to development; and (e) extent of overlap between development and important seasonal habitats (Harju et al. 2010). These same factors have also lead to the suggestion that it may not be appropriate to apply a one-size-fits-all protective buffer around leks based on range-wide data to local populations (Harju et al. 2010). Uncertainty about how quickly and how much sage-grouse populations will decline in response to development, and about the size of lek buffers required to minimize impacts on populations, creates potential for conflict among agencies, industry, and other stakeholders and underscores the need to test, validate, and implement scientifically defensible strategies for monitoring and managing populations in portions of greater sage-grouse range that overlap with planned energy development.

Lek-based Monitoring

Lek-based monitoring and management strategies are also subject to empirical criticisms and require additional research to understand their uses and limitations (Applegate 2000). Using lek-count data as an index of population size has been called into question because the quantitative relationship between lek counts and actual population size has never been established (Beck and Braun 1980; Applegate 2000; Walsh 2002; Walsh et al. 2004, 2010). The probability of detecting an individual male during a lek count (*p*) is the product of: (1) the probability that a male is alive (*survival*, p_{alive}); (2) the probability of the male being present in the survey area, given that it is alive (*presence*, $p_{present}$); (3) the probability of the male attending the lek, given that it is alive and present (*availability*, p_{avail}); (4) the probability of detecting the male, given that it is alive, present, and attended the lek (*detectability*, p_{detect}); and (5) the probability that the lek is counted (*count probability*, p_{count}), such that: $p = p_{alive} * p_{present} * p_{avail} * p_{detect} * p_{count}$ (Walsh et al. 2004, Alldredge et al. 2007, Riddle et al. 2010). To understand the quantitative relationship between lek counts and male population size and to quantify how that relationship changes on an annual basis, we need daily and annual estimates of the proportion of males alive over the course of the breeding season, the proportion of those males present in the study area, the proportion of males attending leks, the proportion of males detected on lek counts, and the probability that the lek is counted, which depends on count effort.

At present, too few quantitative data are available to estimate survival, presence, lek attendance, and detectability for male greater sage-grouse during the breeding season. No published studies have quantified how much annual variation occurs in the proportion of males detected or how much detectability varies among observers or with male age, weather, the observer's distance from lek, equipment used (binoculars vs. spotting scopes), or count method (e.g., ground vs. aerial counts) (Connelly et al. 2003, Walsh et al. 2004). Male lek attendance is known to vary with age, time of day relative to sunrise, date, weather, snow depth, renesting by females, predator activity, and human disturbance, but standardization of lek-count protocols only minimizes variation associated with some of these variables (Patterson 1952, Dalke et al. 1963, Rogers 1964, Hartzler 1972, Jenni and Hartzler 1978, Beck and Braun 1980, Autenrieth et al. 1982, Emmons and Braun 1984, Ellis 1984, Dunn and Braun 1985, Connelly et al. 2000, Connelly et al. 2003, Boyko et al. 2004, Walsh et al. 2004). Past studies that have addressed male lek attendance also did not collect or report data in a consistent fashion, making generalization across studies difficult (Walsh 2002, Walsh et al. 2004). In the most rigorous studies on lek attendance for greater sage-grouse to date, Walsh et al. (2004, 2010) emphasized the importance of individual heterogeneity, age, sex, time of day, and date, but because their data on lek attendance of greater sage-grouse came from only one year in one population, they concluded that additional research was needed to quantify annual variation in lek attendance. Age-specific inter-lek movements by males have been reported in several studies, with 4-50% of males known to have attended more than one known lek during a single breeding season (Dalke et al. 1963, Gill 1965, Wallestad and Schladweiler 1974, Emmons and Braun 1984, Dunn and Braun 1985, Bradbury et al. 1989, Walsh et al. 2004), but the effect of inter-lek movements on lek count data has not been quantified.

Several other factors that influence lek-count data have also never been addressed quantitatively including: disturbance by observers, predator activity (Ellis 1984), disturbance from activities associated with energy development (Braun et al. 2002), and annual variation in female attendance associated with renesting effort (Dalke et al. 1963, Eng 1963 *in* Walsh 2010). Methodological considerations may also affect counts. Non-random access to leks due to logistical constraints (e.g., road conditions, landowner permission) could bias population estimates derived from count data if access is correlated positively or negatively with attendance or abundance (e.g., if attendance is lower near roads). The number of counts conducted per breeding season can also influence lek-count data. Some states only record the maximum count of males at each lek in state-wide databases, and the maximum count is likely to be higher with more counts because any given count is more likely to coincide with peak attendance (Walsh et al. 2004).

Despite these shortfalls, lek-count data continue to be widely used. Large decreases in lek counts or disappearance of leks over large areas over time are thought to reliably indicate population decline or range contraction (Walker et al. 2007, Aldridge et al. 2008, Doherty et al. 2010*b*, Harju et al. 2010). The fact that core areas have been established based largely on counts of males on leks and lek density also suggest that state and federal agencies still consider higher lek counts, on average, to represent larger populations (CGSSC 2008, Doherty et al. 2010*a*, Hagen 2010). This raises an important, but unresolved question. How big of a change or difference in lek counts is required to confidently and reliably infer an actual change or difference in male population size? Investigating these questions and assessing the reliability of lek-count data collected using current, standard protocols for measuring changes in actual population size over time has been identified as a range-wide research priority (Naugle and Walker 2007).

There are two main options for resolving these issues. First, mark-recapture or mark-resight models using either marked birds or genetic data could be used to generate annual population estimates (Lukacs and Burnham 2005, Walsh et al. 2010), and over time, these estimates could be compared to maximum counts of males on leks to better understand the relationship between the two metrics. Using mark-resight approaches is probably the most rigorous way for generating defensible point population estimates (Clifton and Krementz 2006, Walsh et al. 2010), but they are generally too costly, and too time and resource-intensive to implement over large areas or on an annual basis (Walsh 2002, Walsh et al. 2004, Clifton and Krementz 2006, Walsh et al. 2010). A cheaper, easier method would be preferable for long-term monitoring.

Another alternative would be to estimate survival, presence, lek attendance, and detectability from field data in relation to measured ecological and methodological variables, then correct lek count data to obtain annual population estimates and measures of precision. Double-observer approaches, originally developed for use with songbird point counts (Nichols et al. 2000), have recently been modified to use raw count data from independent observers to estimate detectability of males on lek counts (Riddle et al. 2010). Time-to-detection models can also be used to estimate the effects of individual-, group-, survey- and time-specific covariates on detectability (Alldredge et al. 2007). In addition, sightability models have been widely used with other species to estimate the effects of covariates on detection probability and to generate corrected population estimates from annual count data (e.g., Samuel et al. 1987, Rice et al. 2008, Walsh et al. 2009). Such models can be modified for use with lekking species to estimate the probability that individual males attend leks as a function of ecological and methodological covariates that can be measured or recorded in the field (Walsh et al. 2004). Intensive monitoring of individuals with transmitters in the field can be used to calculate daily probability of survival, presence, and lek attendance during the breeding season.

Simulations would also be valuable for exploring the consequences of variation in survival, presence, lek attendance, detectability, and count probability on lek-count data. Most lek-count data are currently collected according to standardized protocols, but it may be that directing biologists to collect one or two more key covariates (e.g., distance from lek, type of optics used) would increase precision of population estimates without increasing cost. Even after following standard count protocols, it may still be beneficial, in terms of the precision of population and trend estimates, to collect and correct count data for weather, time of day, and count method using a modified sightability model. Moreover, not all variables known to influence detectability and lek attendance can be measured when collecting annual lek count data (e.g., inter-lek movement). It would be informative to use simulations based on empirical field data to quantify and illustrate how much lek-count data are likely to vary when I either do not correct for measurable covariates, cannot correct for unmeasured covariates, or both, even in the absence of actual population change. Simulations have been successfully used with other species to assess the effects of unmeasured sources of variation on count data, estimated abundance, and estimated population trends (e.g., Rice et al. 2008).

Lek-based Management

Lek-based management strategies are also subject to criticism. First, such strategies incorrectly assume that all lek locations are known. Several states, including Colorado, have used a combination of known lek locations, male counts at those leks, and vegetation layers to delineate priority areas for conservation of sage-grouse (e.g., “core areas”; CGSSC 2008, NRCS 2009, Doherty et al. 2010*a*, Hagen 2010). Each analysis used slightly different criteria and methodologies, but each assumed that all lek locations were known. This assumption is clearly violated. New leks are discovered annually, particularly in more remote portions of the species' range where surveying is more difficult. The number of known leks monitored range-wide increased 10-fold between 1965 and 2007 due to the discovery of new leks, and the majority of leks currently monitored were discovered after 1994 (WAFWA 2008). For that reason, current core areas are geographically biased toward areas with greater survey effort (which are typically areas closer to population centers and with easier access) and the extent of core areas is most likely underestimated. Moreover, lek-based oil and gas lease stipulations can only be applied to leks whose locations are known. In combination, the presence of unknown leks and underestimation of core areas could lead to inadequate levels of protection in oil and gas fields. Although monitoring data can be adjusted to account for unknown leks using area-based sampling designs (e.g., dual-frame sampling; WAFWA 2008) or estimators that incorporate correction factors (e.g., Huggins estimators; Walsh et al. 2004), lek-based management strategies. For this reason, one of the keys to appropriately managing sage-grouse in oil and gas fields is to locate all leks within and adjacent to the field prior to leasing and development. One way to do this would be to intensively track a representative sample of males to see where they go to display in the early morning hours during the breeding season.

Second, lek-based approaches for managing populations in areas with energy development have not been empirically validated. Oil and gas leases typically stipulate either no surface occupancy (NSO) or restricted surface occupancy (RSO) within certain buffer distances around leks. Historically, the Bureau of Land Management implemented a 0.25-mi. NSO or RSO buffer around leks to minimize disturbance to lekking males and to prevent degradation of the habitat males use during the breeding season, with the overall intention of minimizing long-term population declines and preventing extirpation in areas with development. However, the 0.25-mi. stipulation has no scientific basis (p. B-5, Appendix B, CGSSC 2008). More recently, a review of range-wide studies of male diurnal habitat use and movements during the lekking season suggested that a 0.6-mile buffer around leks may be more appropriate (p. B-6, Appendix B, CGSSC 2008), and this criterion is now recommended by state agencies in Colorado and Wyoming (CGSSC 2008, State of Wyoming 2010). However, the distribution of suitable habitat around leks often is not homogenous and no studies to date have empirically tested how large buffers need to be to protect habitat for males during the lekking season, so it is unclear whether a 0.6 mi. buffer is too large, adequate, or too small. Research is needed to quantify the buffer size needed by intensively tracking both day-time and night-time habitat use of individual males around leks during the breeding season without disturbing the males.

Testing GPS Transmitters

Recent technological advances have led to production of 22-30 g, solar-powered, global positioning system (GPS) satellite transmitters that may be well-suited for generating the data needed to resolve lek-based monitoring and management issues. Most research studies use very high frequency (VHF) transmitters attached to a neck collar to radio-track individual sage-grouse. VHF necklace collars are widely accepted as the current standard method for radio-marking (Connelly et al. 2003), and necklace collars have been widely used on males (Ellis et al. 1987, Walsh et al. 2004, Knerr 2007, Robinson 2007, Wisinski 2007, Holloran et al. 2010, Walsh et al. 2010). However, no studies to date have tested the impact of VHF collars on male (or female) survival, and field observations have generated concern whether males can safely be fitted with necklace-style VHF collars. Necklace collars may interfere with male displays by bouncing up and striking the male's beak during strutting; they may restrict breathing or foraging when neck and breast tissue swells during the breeding season; they may prevent yearling males from swallowing as their necks grow over time, leading to suffocation or starvation; and males may become distressed and repeatedly attempt to remove the collar, thereby increasing their detectability to predators (B. Walker, pers. obs.). Lek attendance of females with necklace-mounted VHF collars did not appear to be affected (Walsh et al. 2004), but females do not display, so whether necklace collars reduce male lek attendance remains unclear.

GPS transmitters have several advantages over VHF necklace collars. GPS transmitters record multiple locations per day at specific, pre-programmed times; logistical problems that prevent crews from locating birds on the ground are eliminated (e.g., weather, road conditions, truck breakdowns, technicians oversleeping, denied access, etc.); data are gathered without disturbing the bird or its flock mates; and they provide high-resolution data on survival, movement, lek attendance, and diurnal and nocturnal habitat use around leks. Collecting data of comparable resolution and accuracy using VHF collars would result in excessive disturbance to birds and be logistically impossible. However, solar cells require that transmitters be mounted dorsally so they are exposed to the sun. Because of their similarity to backpack-style transmitters (Brander 1968, Amstrup 1980), there is concern that rump-mounted transmitters may directly or indirectly reduce survival of sage-grouse. Moreover, as with any new technology, data are also needed to assess the proportion and accuracy of GPS locations acquired, transmitter durability and longevity under field conditions, and cost effectiveness in comparison with VHF collars.

Current studies with female greater sage-grouse indicate that rump-mounted leg-loop harnesses may be a viable option for attaching GPS transmitters to males as well. Satellite GPS transmitters cannot be used with necklace collars because solar cells under the neck receive insufficient sunlight to charge the battery (B. Henke, Northstar Science and Technology; C. Bykowsky, Microwave Telemetry, pers. comm.). Five separate studies are now using GPS transmitters with a rump-mounted leg-loop harness design to track female sage-grouse. Survival of females with VHF transmitters ($n = 42$) and GPS transmitters ($n = 50$) was tracked in northwestern Colorado from spring 2009 to spring 2010. VHF and GPS females had similar survival rates through October 2009, but survival of GPS females was lower from October 2009 - March 2010, resulting in lower estimates of annual survival (0.556 ± 0.073 SE for VHF vs. 0.406 ± 0.068 SE for GPS). Despite limited sample sizes, these results suggest that the ratio of transmitter size to body size reduced, the harness design should be made more flexible, transmitters should be fit less snugly, or all three. Because males are larger than females, 30-g transmitters (38 g including harness and crimps) would be proportionally less of male body mass, approximately 1.1-1.9%, depending on male age (~2000-2400 g for yearlings, ~2800-3300 g for adults; Beck and Braun 1978). Leg-loop harnesses may still cause skin irritation under the legs, particularly during males' vigorous strutting displays. Moreover, if harnesses are fitted too tightly around the legs, or if swelling occurs around the legs prior to the breeding season (as it does around the neck and breast), this may also restrict the ability of males to display in spring. Having a GPS transmitter with a highly reflective solar cell attached dorsally may also increase detectability of males to predators or alter their distribution of body weight such that it impedes flight and makes them more susceptible to predation or targeted by visual predators. If yearling males grow during the course of the study, they may also outgrow a less flexible or snugly-fitting harness. If leg-loop harnesses impact survival of males, I would predict lower survival rates for GPS males than those published for leg-banded males. Band-recapture data suggest that survival rates of male sage-grouse vary annually and by age (0.635 ± 0.034 SE for yearlings vs. 0.368 ± 0.007 SE for adults; Zablan et al. 2003). Males with VHF collars in southwestern Montana averaged 0.34 ± 0.067 SE annual survival, but the author did not distinguish between yearling and adult males (Wisinski 2007). If harnesses hinder movement or displays of males, I would also predict reduced display rates for GPS males compared to either color-banded or unmarked males of the same age under the same conditions.

This study is intended to be a three-year investigation of greater sage-grouse lek monitoring and management strategies using males deployed with GPS transmitters in the Hiawatha Regional Energy Development Project area in NW Colorado and SW Wyoming.

STUDY AREA

The study area covers the Hiawatha Regional Energy Development project boundary in northwestern Colorado and southwestern Wyoming and includes birds from both Colorado (Zone 1; "Cold Spring Mountain/Hiawatha") and Wyoming ("Salt Wells") core breeding populations (Fig. 1). This area holds a large, robust population that is contiguous with the rest of greater sage-grouse range in northwestern Colorado and south-central Wyoming. The maximum count of males on known leks in Colorado's Zone 1 varies annually (in part due to variation in effort), but it is considered a stable population (CGSSC 2008, p. 259). Previous data from VHF- and GPS-marked females in this region indicate that sage-grouse typically winter in or near the Hiawatha project area and attend leks both within the project area and at higher elevations surrounding the project area. There are currently nine known leks within the study area plus six more immediately adjacent to the study area. Of these 15 leks, seven were discovered in 2007 or later. It is unclear what proportion of males in the population our sample will represent because not all leks are known and it is unclear how the number of males counted on known leks relates to actual population size. Research is being conducted with the support of the Wyoming Game and Fish Department and the Rock Springs (WY) and Little Snake (CO) field offices of the Bureau of Land Management.

METHODS

Capture and Handling

Most males are captured in fall and winter habitat prior to the onset of the breeding season to prevent biasing data on lek attendance the following spring (Walsh et al. 2004). A small number males trapped early in the breeding season are used to estimate inter-lek movements and habitat use around leks. Trapping effort and GPS unit deployment follow a probability-based sampling scheme based on winter habitat identified in seasonal habitat models (Walker 2010) to ensure that males from all potential wintering areas and therefore all leks in the project area are represented in the sample. I plan to capture and attach 30-g, rump-mounted solar-powered GPS PTT satellite transmitters (Northstar Science and Technology, King George, VA) on 30 adult male sage-grouse in November-December and on 30 yearling male sage-grouse in February each year. I selected 30-g transmitters because they have larger battery capacity than 22-g models, which decreases risk of transmitter failure (or temporary failure to transmit data) under low-light conditions. GPS males will also receive individually numbered aluminum leg bands (size 16) and distinctive combinations of colored leg bands. I also plan to capture and individually color-band 30 adult males in November-December and 30 yearling male sage-grouse in February each year. I will alternate marking methods during captures to maintain equal proportions of GPS males versus color-banded males in each portion of the study area. Trapping yearling males in February rather than in the fall will allow them to reach larger body mass prior to deploying the transmitter, thereby reducing the chance that they will outgrow the harness during the breeding season. Transmitters from birds that die may be recovered, cleaned, refurbished, and redeployed to maintain or increase sample sizes for survival analyses and or collecting mark-resight data.

Capture and handling methods will follow standard operating procedures established for sage-grouse (Appendix A), with the exception that the decision whether injured birds will either be released or euthanized will be made by the PI in the field rather than transporting birds back to Fort Collins. This is because no known rehabilitators in Colorado currently have the facilities to care for wild sage-grouse. I will use night-time spotlighting and hoop-netting for all captures (Wakkinen 1992). I selected a sample size of 30 individuals per age class (yearling vs. adult). It is crucial to estimate parameters for each age class separately because they have different survival rates (Zablan et al. 2003) and different rates of lek attendance (Walsh et al. 2004). Sample size must also be balanced with the potential for impacts on the population should GPS transmitters have highly detrimental effects on male survival. With a sample size of 30 males in each age class, statistical power will be > 0.80 if survival of adult males is < 0.14 or > 0.62 or if survival of yearling males is < 0.39 or > 0.86. This sample size will only allow detection of relatively large differences in survival with statistical power > 0.80. However, deploying more GPS transmitters would be unethical without data regarding whether the transmitters have catastrophic effects on survival. The loss of > 30 males in any given age class in any given year in this population would likely pose an unacceptable risk to stakeholders and cooperators. A sample size of 30 should inform us whether GPS transmitters have catastrophic effects on survival.

GPS Transmitter Attachment

I will use a rump-mount attachment for GPS transmitters based on the method B design described in Bedrosian and Craighead (2007) modified for sage-grouse (Figs. 2-3). Transmitters will be manufactured with a medium-brown, sand-textured finish to reduce reflected light. A thin layer of neoprene is glued to the bottom side of the transmitter to ensure that contact between the transmitter material and the bird's lower back is padded and insulated. The harness material is 0.55-cm (0.25-inch) wide, brown Teflon ribbon (Bally Ribbon Mills, Bally, PA). A 12 cm length of 0.55-cm wide elastic cord is sewn into the center of a 75-cm (36-inch) length of Teflon ribbon such that 4-6 cm of stretchy Teflon ribbon extends out from the attachment points on either side (Fig. 2). The elastic gives the harness flexibility when the bird extends its legs during take-off and when males are displaying. Yearling harnesses will be sewn with more elastic (16 cm) to accommodate possible increases in body size over time. The transmitter side, front, and back are painted camouflage to decrease visibility to predators (Fig. 3). Harnesses are fit with the bird held in a standing position. The transmitter is mounted on the bird's lower back centered between the legs (as seen from behind and as seen from the side of the bird) with the antenna extending toward the rear above the tail (Fig. 3). The Teflon ribbon is fitted down, around, and underneath the legs and attached to the loops using a small section of 0.55-cm (0.25-inch) diameter copper tubing as a crimp (Fig. 2). Copper crimps typically quickly become tarnished with exposure to the elements, but as a precaution, crimps are marked with black ink before release to be sure they don't reflect light. I trim the Teflon ribbon at an angle and leave just enough excess ribbon on each side (~3 cm) to allow us to refit or enlarge the harness should it become necessary during fitting. The end of the excess ribbon is dabbed with Superglue® (Super Glue Corporation, Rancho Cucamonga, CA) to prevent fraying. The life span of the exposed Teflon ribbon has not been tested, but it has been used successfully with rump-mount transmitters on female sage-grouse for >16 months already without showing signs of fraying, wear, or deterioration. The life of the elastic cord is unknown. Transmitters will be fitted just snugly enough to prevent birds from dropping transmitters.

The units are solar-powered and may last for 3-5 years, which is longer than the life span of almost all male sage-grouse (Zablan et al. 2003). All GPS units will be pre-programmed to collect 8 locations per day from March-May so as to get data on early morning lek attendance (6 am, 7 am, 8 am), mid-day feeding/loafing areas (12 pm), evening feeding areas (6 pm), and night roost locations (12 am). Units will be programmed to collect two locations per day at 12 pm and 12 am from June-Feb to capture basic patterns of seasonal habitat use and movements while reducing demand on the battery during low-light conditions encountered in fall and winter. I anticipate studying males for multiple seasons (if possible), so I do not plan to remove GPS transmitters unless leg-loop data suggest refitting of yearlings will be required. If preliminary survival data indicate little or no impact of GPS-transmitters on male survival, I will clean and re-deploy GPS transmitters recovered from mortalities on additional males to maximize sample sizes for survival analyses. I was trained in the attachment technique in the field in March 2009 by Bryan Bedrosian, who has used GPS transmitters with raptors, corvids, and sage-grouse (Craighead and Bedrosian 2009).

The ARGOS system sends GPS transmitter data as a text file by email every three days. I then parse the raw text files using the "DSDCODE" software provided by Northstar Science and Technology. This software automatically parses the data and amends new locations from GPS birds to an ArcGIS shapefile for each individual. I amend the parsed data (in .dbf format) to an existing Microsoft Excel® spreadsheet of GPS bird locations and manipulate the spreadsheet to remove duplicate records, flag date and location errors, and to identify records signifying important events (e.g., mortality).

Lek and Lek Attendance Definitions

I define a lek as any area within which ≥ 2 males have displayed in ≥ 2 years, which is consistent with previous state-wide and range-wide definitions (Connelly et al. 2000, CGSSC 2008). I use this definition to ensure that small leks and "satellite" leks are included, but that locations where males do not consistently display are excluded (i.e., one-time use locations). The status of a lek may be active or inactive in any given year. Leks used by displaying males at least once within the past 5 years are considered active (CGSSC 2008). Newly-discovered leks > 500 m from all other known leks will be designated as potential leks. If those locations have displaying males in ≥ 2 years, they will be classified as new leks and assigned a name based on local geography. I will delineate a "count boundary" for each known lek prior to the first count and for each new lek immediately following its discovery. The count boundary represents the specific perimeter within which males would be visible and available for counting by observers during any given count. The purpose of establishing a count boundary is to ensure that the geographic area of observation for each lek is consistent over time. This prevents the characteristics of specific leks (e.g., their area, location, topography, etc.) from changing over time. This count boundary will necessarily be larger than the outer perimeter around displaying males on any given date because: (a) observers can typically see and count males over an area larger than just the area where displaying males are found, (b) males may shift the location where they strut slightly from day to day (WAFWA 2008), and (c) observers typically adjust the location from which they count males from day to day to maximize their ability to obtain complete counts of males.

It is also important to unambiguously define lek "attendance" because some males use habitat near leks, but they may or may not be within the area that can be counted by observers. I define lek "attendance" for each male as a binomial variable. Lek attendance is classified as 1 if the male is inside the count boundary (i.e., visible and available for counting by observers) at any time during the standard count period (0.5 hrs before sunrise to 1.5 hrs after sunrise) and as 0 if the male is either: (a) outside the count boundary (i.e., not visible and unavailable for counting) during the standard count period, or (b) inside the count boundary at a time other than during the standard count period. Lek attendance of GPS males should be straightforward to assess when resighters are present, but there may be some ambiguity about lek attendance for GPS males that are not directly observed (those that attend leks at which no observers are present). The accuracy of high-quality locations derived from GPS transmitters is typically ≤ 26 m. Only GPS males with early morning locations within 26 m of the count boundary will be considered to have attended a lek.

Lek Counts and Resighting

CDOW lek-count protocol instructs observers to obtain a maximum count of males by conducting repeated counts 5-10 minutes apart over a 30-minute period between 0.5 hr before and 1.0 hr after sunrise (Appendix B). Although no specific guidelines are given for exactly how far away to be, biologists and wildlife managers typically count leks from 50-400 m away, depending on topography, access, and how far away they need to stay to keep from disturbing birds at the lek. They use whichever optics are required to obtain a reliable count (binoculars or spotting scope) and whichever mode of transportation (truck, ATV, on foot) gets them close enough to the lek to count it. A truck is preferred because it reduces disturbance to birds and is logistically easier and more comfortable for conducting repeated scans.

Field crews will focus on collecting count data and resighting data at only those leks attended by GPS males, most of which are likely to be within or adjacent to the study area. Observers will visit each of these leks once a week. The field crew will be divided into three groups: resighters, counters, and surveyors. Surveyors will check locations of potential new leks as needed, and if males are present, will conduct a standard 30-minute lek count. Resighters will each go to a different lek and collect resighting data on GPS and color-banded males during each 30-minute interval from 0.5 hr before local sunrise to either 1.0 hrs after sunrise or to when all birds depart the lek, whichever is later. Resighters will use a spotting scope from a portable blind placed ~50 m from the lek (Walsh et al. 2004). The goal of each resighter is to collect accurate data on the identity of all GPS and color-banded males present on the lek during each 30-minute interval. Portable blinds will be placed near leks either the night before or >1 hr before to sunrise to prevent disturbance to birds on the lek (Walsh et al. 2004). Blinds will have raptor perch deterrents installed on top to prevent aerial predators from using blinds as perches. Counters will work in pairs, and each pair will conduct a 30-minute lek count during the standard count period at two leks per day (the same leks being observed by resighters). For counters, each 30-minute visit to a lek will be divided into six 5-minute scan intervals. Counters will follow CDOW count protocols and record the maximum number of yearling and adult males and females counted during each 5-minute interval. The goal of each counter is to get an accurate count of yearling and adult males and females during each scan interval and to determine the number (and eventually, the identity) of all GPS males present on the lek. Counters will also record any birds that arrive or leave the lek during each interval. Counters will alternate between using a spotting scope and binoculars during each scan interval. Each observer will be allowed to scan the lek multiple times within each 5-minute interval because that is typically how lek counts are conducted by CDOW biologists and wildlife managers. At the end of each count, the counters will consult with the resighter by two-way radio to reconcile and confirm the identity of any GPS males observed on the lek.

Observers will be systematically rotated such that each observer conducts an equal number of lek counts and resighting days with each other observer. I will only hire observers with experience conducting lek counts. All observers will be trained in standard lek-count protocols, will practice resighting prior to collecting field data, and will collect data on standardized forms. All counts will be conducted from within a realistic distance from leks, depending on topography and optics (50-400 m), and all counters will record the distance (m) to the approximate lek center using a laser rangefinder. All observers will conduct counts using the same standard make and model of 10x binoculars and 20-60x zoom spotting scopes.

Aerial lek counts will be conducted using two observers flying in fixed-wing aircraft (the pilot and an independent observer). Aerial observers will record the number of males observed, but cannot distinguish between marked and unmarked males or between adult and yearling males.

Objective 1a: Survival comparison. –I will use location and mortality data from males with GPS transmitters to estimate seasonal and annual survival rates of yearlings and adults. The null hypothesis is that male greater sage-grouse with GPS transmitters in each age class have survival rates indistinguishable from those reported for leg-banded males in the published literature. If location data from a GPS male indicate a stationary transmitter, field crews will visit all subsequent locations to determine whether it was mortality or a dropped transmitter and to recover the transmitter using a metal detector. Transmitters deployed so far have typically been recovered within 20 m of their last set of stationary location(s) (B. Walker, unpub. data). I do not anticipate estimating cause-specific mortality rates because the delay between when birds are killed, the acquisition and processing of satellite data, and when locations can be checked by field crews is typically 4-7 days.

I will use an information-theoretic approach (Burnham and Anderson 2002) to evaluate sets of *a priori* candidate models describing variation in daily and seasonal survival rates of males during breeding, summer, fall, and winter. Survival analyses of GPS male data will use a continuous-time approach such as a Cox proportional hazards model (Murray 2006). Age will be a fixed effect in all seasons (adult vs. yearling), and landscape-scale habitat variables known to influence habitat selection in each season (e.g., terrain ruggedness, proportion sagebrush habitat within 1 km; Walker 2010) will be included as additional explanatory variables. During the breeding-season, daily lek attendance status will be included as an explanatory variable to quantify risk due to lek attendance.

As a separate test of the effects of GPS transmitters on breeding-season and annual survival, I will use a Barker model to jointly estimate survival of GPS and color-banded only males (Barker 1997) in MARK. I will use GPS location data, mark-resight data from each spring's resighting effort, and recoveries from harvested birds to generate capture histories. Age and mark (GPS vs. color-banded) will be fixed effects.

Objective 1b: Leg-loop size comparison. – Leg loops will be marked at various distances from the front attachment point using colored iridescent, permanent markers. The exact length of the leg loop from the front to the rear attachment point will be recorded in the field on each leg on each bird after fitting. Means and variances of harness lengths will be compared between yearlings and adults using a standard one-sided, two-sample t-test because of the *a priori* expectation that yearlings will have smaller leg-loop lengths than adults. If needed, yearlings may have to be recaptured after the breeding season to refit them with adult-sized harnesses. Recapture of yearlings may be difficult because the transmitters cannot be tracked in real time. If needed, I will use location data to identify recent night roost locations of yearling males and attempt to find and capture those males by trapping in those areas.

Objective 1c: Comparison of GPS and color-banded male display rates. – During lek counts at which marked males are present, the resighter will record the display rate (no. struts/minute) of the GPS male nearest the observer and of the color-banded male in the same age class that is nearest the observed GPS male. The resighter will conduct three 1-minute observations per individual spaced 1 minute apart. Data from the three 1-minute observation periods will be summed. Observation periods will alternate between GPS and color-banded males, and the first bird to be observed will be randomly determined. If no color-banded males are present on the lek, the resighter will observe the nearest color-banded only or unmarked male in the same age class. The observation period will occur during at some time during the first 1.0 hr after local sunrise to ensure that light is sufficient to record behavioral data, but after resighting data have been collected. When more than one GPS male and more than one color-banded male are present, the resighter will collect on the next pair of marked males at the next earliest opportunity. Time spent fighting with other males or copulating with females will be excluded when calculating display rates. The null statistical hypothesis is that GPS males and color-banded males will show no difference in mean display rate. Comparisons will be made using a paired-sample, repeated-measures design because the dataset will include repeated observations from the same individuals over time.

Objective 1d: Comparison of GPS and color-banded male lek attendance rates. – The null statistical hypothesis is that GPS males and color-banded males will show no difference raw rates of season-long lek attendance. Raw lek attendance for each individual will be calculated as the proportion of the total number of 30-minute intervals during the breeding season during which each marked bird was resighted on a lek. I will then compare raw lek attendance among GPS and color-banded males separately for each age class because the two age classes will be marked at different times of year.

Objective 2: Using GPS males to find new leks. – Early morning locations of GPS males will be compared against locations of known leks every three days as satellite data arrive and are processed to identify potential new lek locations in and near the study area. Males that make ≥ 2 early morning visits to the same location on consecutive mornings during the breeding season will be considered to have visited a potential lek location. The surveyor will then visit those locations or they will be checked from the air at least once during the next 7 days to document whether displaying males or their sign (e.g., pellets, tracks, feathers) are present or absent, and if so, how many. If displaying males or sign are present at a newly discovered lek, then that lek will be added to the list of regularly counted leks following standard protocols, and the count boundary determined prior to the next visit. A GPS male that uses a location within the count boundary during the count period that is subsequently discovered to be a lek will be considered to have attended that lek on that date.

Objective 3: Estimate no. of leks in the study area. – Data from GPS males will be used in a mark-recapture framework to estimate the number of leks in the study area. Visits by marked GPS males can be used to "capture" leks and subsequent visits by marked birds to that lek constitute "recaptures" of that lek. Recapture histories for individual leks can then be derived and analyzed using an appropriate mark-resight model (Bartmann et al. 1987, Bowden and Kufeld 1995, McClintock et al. 2008).

Objective 4: Estimating detectability of males on leks. – I will compare three methods for estimating detectability of males on leks. Two of the methods have only recently been published and require validation for use with lekking species (Alldredge et al. 2007, Riddle et al. 2010). The third method is included as a way to double-check an assumption of the first two methods.

First, I will use an unreconciled, independent, double-observer approach to estimate detectability from lek-count data (Riddle et al. 2010). Standard double-observer and removal models require that observers match or reconcile specific individual animals that were or were not detected by each observer (Nichols et al. 2000). Because there may be as many as 80 or more males on any given lek and most of these males will be unmarked, this would be impossible to do on most lek counts. Unreconciled double-observer models use raw maximum counts of the number of individuals detected (in each age class) from each of two independent observers to generate a site history for each observer on each count (e.g., 13, 15) (Riddle et al. 2010). Site histories are then analyzed using the repeated-counts hierarchical model of Royle (2004) in program PRESENCE, with the difference being that, in the unreconciled double-observer model, each observer is considered an independent "visit" (Riddle et al. 2010). One of the benefits of this approach is that leks do not actually have to be visited twice, and the closed population assumption is met (Riddle et al. 2010). The method may require using a negative binomial or zero-inflated Poisson distribution in place of a Poisson distribution if data are overdispersed (Riddle et al. 2010). This estimator may become unstable when detectability is low (P. Lukacs, pers. comm.). However, I anticipate relatively high detectability because observers typically position themselves to maximize their ability to detect males attending the lek.

The counting protocol outlined above (under *Lek counts and resighting*) results in dataset with six repeated counts from the same lek on each date for each counter for each age class of males and for females, with three of the six counts by each counter done with a spotting scope and three with binoculars. Counters will record distance to approximate lek center and presence or absence of snow cover on the count as well as predator activity and weather (temperature, wind speed, precipitation, visibility, illumination) at the end of each 5-minute interval. Predator activity will be broken into three classes (no predator detected, predator visible near lek, predator on, over, or attacking males) based on observations of potential predators of adults (eagles, hawks, falcons, owls, coyote, red fox, bobcat, mountain lion, feral dog) that, in the opinion of the counter or resighter, should have been visible to males attending the lek. Covariates in the analysis of site histories will include a random effect of lek and fixed effects of lek size (i.e., max no. of males counted), distance from lek, optics used (binoculars vs. spotting scope), predator activity, weather, and an interaction between optics and distance from the lek. Because the data consist of repeated counts from the same lek within and among days, this dependence will have to be addressed using a repeated-measures approach.

Second, I will use a time-to-detection approach with resighting data from GPS males collected by counters to estimate detectability. Time-to-detection approaches use resighting data to generate capture histories for individual males detected during the count, and at least four intervals are required for modeling (Alldredge et al. 2007). In the field, counters will record the number of GPS males they detect on the lek during each of the six 5-minute scan intervals. GPS transmitters should be visible at distances at which counts are typically conducted using binoculars. Counters will then double-check with resighters by two-way radio to confirm the identity of GPS males observed on the lek. Resightings will also be checked against early morning locations of GPS males to ensure correct identification of males. I use data from counters instead of from resighters to ensure that detectability measured is representative of how counts are typically conducted. Detections by resighters are not used in detectability calculations because lek counts generally are conducted at distances > 50 m from leks. Resighting data from counters will result in a dataset of capture histories for each marked individual observed during each scan interval for each count period on each date on each lek (e.g., 101011). Capture histories will then be linked with individual-, group-, count-, and interval-specific covariates. This method assumes that males do not arrive, leave, or leave then return to the lek between intervals within each 30-minute count period (i.e., it assumes a closed population). The method has fewer assumptions and more flexibility for modeling than either traditional double-observer (Nichols et al. 2000) or removal methods (Farnsworth et al. 2002). Covariates will include a random effect of either lek or observer (but not both at the same time) and fixed effects of distance from lek, optics used (binoculars vs. spotting scope), predator activity, weather, and an interaction between optics and distance from lek. Time-to-detection models for estimating detectability will be run in program MARK, version 6.0 (Alldredge et al. 2007, White 2010).

I will also estimate detectability by calculating the proportion of GPS males known to have attended a lek that were also detected by either resighting or counting observers during lek counts on that same date. This is to test the implicit assumption that all males that attend a lek are available for counting. It is possible that not all males attending a lek are necessarily visible to both observers (e.g., some may be hidden by topography). Although time-to-detection and unreconciled double-observer approaches should both theoretically account for males attending that are hidden from view, it would be good to directly test this assumption. To do this, I will compare early morning locations of males with GPS transmitters against records of individual marked GPS males observed by resighters during lek counts at approximately the same times that GPS transmitters are scheduled to record early morning locations (6 am, 7 am, and 8 am). The resighting observer will estimate individual marked bird locations by correcting observer UTM locations for direction (θ, in degrees) and distance (m) using the formulas: $\text{northing}_{\text{male}} = \text{northing}_{\text{observer}} + \cos(\theta) * \text{distance}$ and $\text{easting}_{\text{male}} = \text{easting}_{\text{observer}} + \sin(\theta) * \text{distance}$. Resighters will record their locations in Universal Transverse Mercator (UTM) coordinates in the North American Datum 1983 using a high-sensitivity GPS unit (Garmin eTrex Vista HCx), they will estimate direction to males from true north with a declinated compass (Silva Ranger CL), and they will estimate distance to those males using a laser rangefinder (Nikon Prostaff 550).

To estimate the effect of counting males from the air on detectability, I use the maximum raw count of all males combined from counters on the ground versus the maximum raw count from a counter in a fixed-wing aircraft (either the pilot or an observer) using the same unreconciled double-observer approach as above. In this case, the difference in detection probability among observers represents the difference in detectability of counting on the ground versus from the air. The comparison will be made between data recorded on the flight and data recorded over the entire 30-minute count period on the ground. This comparison is appropriate because ground counts based on data from a 30-minute count period and flight counts based on data from 3-5 minute count periods are recorded with equal weight in statewide count databases. I will attempt to conduct 40 paired lek counts per year on the ground and from fixed-wing aircraft on the same dates and at the same times. Detectability from the air may be lower because data are derived from only 2-3 passes during a brief window of time (3-5 minutes) rather than counted for an extended period of time during the morning (30 minutes) as is typical for ground counts. However, it is possible that ground-based counts could result in lower counts if topography prevents observers on the ground from detecting all males.

Objective 5: Estimate age-specific lek attendance of males. – I will analyze lek attendance data in two ways. First, as recommended by Walsh et al. (2004), I will develop a modified “sightability” approach to estimate lek attendance for adult and yearling males using data from GPS males. I will use early morning locations of GPS males to determine which leks (or potential leks) GPS males are attending or likely to attend. Field crews will make every effort to count and resight GPS and color-banded males on each of those leks at least once in random order during each week-long resighting occasion throughout the season. Resighting observations will be lumped into 30-minute resighting periods starting at 0.5 hr before local sunrise for lek attendance analyses. Resighters will also collect data on covariates likely to influence lek attendance for each 30-minute interval. Covariates collected by observers at the lek will also be applied to non-attending GPS males because the focus of this analysis is on testing factors that influence presence on the lek rather than factors influencing presence at locations away from leks. Because GPS males sometimes move between leks, they may not always be present on leks they previously attended that get counted. For this reason, data for the modified "sightability" model will necessarily come from a subset of our sample of GPS males. Data from males that attend non-counted leks will be excluded from this analysis. The dependent variable is lek attendance (1 = attended lek, 0 = did not attend lek). Covariates will include a random effect of lek, fixed effects of time of day, date, snow depth, the previous day's weather, presence or absence of females on the lek, probability of female attendance, lek size (i.e., maximum count of males), and marking type (GPS vs. color-banded), as well as fixed effects of weather variables, predator activity, and frequency of anthropogenic disturbance during the previous 30-minute interval. Logistic regression will be conducted in program R (version 2.11.0, R Development Core Team 2010). Although misidentification of color-bands combinations is a concern for resighting, comparison of color-band combinations recorded against early-morning locations should allow us to correct any misidentification of GPS males by resighters. If misidentification is a problem, new mark-recapture approaches may be available to address that issue (e.g., Link et al. 2010).

Second, I will use the entire GPS male dataset to estimate lek attendance as a function of variables that can be measured without observing attending males directly. I will compare early morning locations of males with GPS transmitters against the count boundary for all known active lek locations to determine whether or not GPS males attended leks (see definition of "attending a lek" in *Objective 4,* above). I can then estimate daily rates of lek attendance for each male using logistic regression. Field crews will document all major weather events that could influence male attendance throughout the field season (e.g., storms, high winds). Daily lek attendance will be modeled as a function of date, current weather (temperature, wind speed, precipitation), the previous day's weather, resighter presence, counter presence, average lek size, previous lek attendance (as a measure of reproductive effort), and probability of female attendance (estimated from counts of females at leks over the course of the season). I include observer presence because having observers count leks may cause males or females to move to another lek or to forgo lek attendance that day, yet this has never been tested. Overall lek attendance for each individual over the season will be calculated by summing the total number of days that each bird attended a lek and dividing that value by the total number of days for which each individual was alive and its early morning location was known.

Detection probability (the joint probability of detectability and lek attendance), may also be estimated as part of estimating male population size (see *Objective 9,* below) and can be compared against the product of detectability and lek attendance estimated separately.

Objective 6: Estimating probability of age-specific presence using movements of males. – As outlined above, probability of presence is one of five key components of detection probability for sage-grouse males on leks that need to be estimated for running simulations. I will use location data to estimate the daily probability of presence for each GPS male for each lek within the survey area on each day of the breeding season. Data will be stored as an N x L x D matrix, where N = the number of GPS males in the sample, L = the number of leks attended by GPS males, and D = the number of days during the breeding season. Each cell in the matrix is assigned a 1 or a 0 based on the presence (1) or absence (0) of each GPS male within a certain distance of each lek on each day. A value of 1 does not denote lek attendance because males have the option of either attending or not attending the nearest lek to them on each day. From this matrix, I can calculate the raw proportion of the males in our population that were present at or near each lek we're studying on each day. This is the daily probability of presence that will be used in simulations. Because I will be tracking the location and movement of each male and identifying all leks used by males, where GPS males move and where field crews can determine lek status will define the survey area. However, if a male moves so far outside of the study area that field crews cannot survey any leks he might be attending and it is impossible to determine whether or not he attended a lek, then he will be excluded from the dataset (all values for that male for those days will remain blank).

I will also use location data from GPS males overlaid with locations of all known active leks to document the frequency, timing, and distance of inter-lek movements by yearling and adult males.

Objective 7: Estimating age-specific survival during the breeding season. – The purpose of estimating daily survival is to determine the proportion of males in each age class that remain alive on each date over the course of the breeding season in each year for simulations. Age will be used as the only predictor variable in this analysis (adult vs. yearling). Survival analysis will use a continuous-time Cox proportional hazards model (Murray 2006).

Objective 8: Simulate lek-count data. – I will use empirical data on variation in male survival, presence, lek attendance, detectability, and lek-count effort in conjunction with important covariates (e.g., time of day, date, weather, etc.) to simulate how much lek counts are likely to vary in the absence of population change when conducted according to standardized protocols. I will simulate data for the same sample of leks for which I have data on the number of males counted, as well as data on survival, presence, lek attendance, and detectability. The number of males in the simulated population will be set at a value equal to the maximum count of yearling or adult males at each lek during the period of peak attendance for each age group divided by age-specific detectability estimated during that period. I can use these data to simulate what proportion of the simulated population of adult and yearling males would actually be alive, present, and attending each lek during each time period of the morning on each day of the breeding season in each year. I would then run scenarios using this simulated dataset with realistic combinations of measured and unmeasured variables that influence detectability (e.g., time of day, optics, distance from lek, weather, and number of counts per season). Scenarios would include counts conducted: (a) under more restrictive (0.5 hrs before to 0.5 hrs after sunrise) or less restrictive (0.5 hrs before to 1.0 hrs after sunrise) time of day requirements; (b) with binoculars versus spotting scope; (c) close to leks, farther away from leks, or at various distances from leks; (d) in good versus marginal weather conditions; (e) using a varying number of counts per season from 1 to 6 on randomly selected dates at least a week apart (to mimic data contained in state databases); (f) with varying proportions of leks counted to mimic access problems encountered in the field. Simulations will be set up in program R (version 2.11.0, R Development Core Team 2010).

Objective 9: Estimate age-specific population size. – If sufficient data from repeated counts are available at leks within the study area, I will use mark-resight and lek count data to estimate detection probabilities and population size for yearling males, adult males, and females. Because this is an open population (many leks surround the study area), I will analyze the data using an immigration-emigration mixed logit-normal mark-resight model (Bartmann et al. 1987, Neal et al. 1993, Clifton and Krementz 2006) in program MARK (version 6.0, White 2010). Population estimates will be generated in each year of the study. Although previous authors concluded that the joint hypergeometric estimator was unsuitable for greater sage-grouse because it does not allow for individual heterogeneity in lek attendance, violation of the closed-population assumption could lead to even greater bias in population estimates.

Objective 10: Test 0.6-mile lek buffer. – Portions of the study area have had oil and gas development since the 1920's (Walker 2010). However, most leks within the study area are far enough away from areas with oil and gas development that I should have sufficient data to measure how male sage-grouse use habitat around leks in the absence of disturbance related to oil and gas development. If the hypothesis that males avoid disturbance is true, I would predict a pattern of constrained habitat use around leks within or near development compared to those outside development after accounting for habitat features. This can be tested by comparing buffer distances required to protect the same proportion of the male population at leks inside and outside development after controlling for habitat and topography.

I will measure distances of three off-lek locations per day (at noon, 6 pm, and midnight) for each male to the center of the lek attended that day, the lek most recently attended, the nearest active lek (as recorded in CDOW databases or by field crews), and the lek attended on the next visit. I will then calculate the proportion of off-lek locations (for each portion of the day) that fall within specific distances of dissolved buffers around the centers of known active leks to test the effectiveness of the current 0.6-mi. NSO/RSO stipulation for lek buffers and to make recommendations on the most efficient buffer size to use to protect specific proportions of the population. It may also be possible to use a kernel or bivariate normal mixture model to estimate the probability of males using the area around leks (D. Walsh, pers. comm.). I will also compare the effectiveness of conserving areas that fall within different circular buffer sizes to areas of high priority habitat of similar size already identified using VHF locations of females (Walker 2010).

RESULTS AND DISCUSSION

In fall 2011, I modified our handling technique such that birds are held on their sides while attaching the transmitters rather than in a standing position to improve our ability to properly fit the harness. Field crews captured and color-banded a total of 40 non-juvenile (i.e., adult or yearling) male greater sage-grouse from Oct 2011-Feb 2012 and 25 juvenile males from Jan-Mar 2012 within or immediately adjacent to the Hiawatha Regional Energy Development boundary. Two additional adult males were captured after the breeding season started, for a total of 67 males captured and marked. Of these 67 males, we deployed GPS transmitters on 47 (23 non-juveniles and 24 juveniles). Two males with GPS transmitters (two juveniles) slipped their transmitters.

I monitored the locations of all GPS males every three days throughout the spring breeding season in 2012 (10 March – May 20) to examine: (a) locations of potential new leks for crews to check, (b) their live-dead status, (c) which males remained within the study area, (d) which males were attending leks, (e) the leks they attended, (f) any inter-lek movements, and (g) nocturnal and diurnal habitat use around leks.

Field crews checked numerous potential lek locations and located, verified, and counted two previously unknown leks in 2012 (North Scrivner, Cow Creek Reservoir) (Fig. 1). Crews also confirmed strutting at three leks first discovered in 2011 (North Kinney Rim, Owl Bench 2, and Central Sand Wash).

A lack of snow in winter 2011-2012 led to a shorter breeding season and earlier end to male strutting than in 2011 (likely due to less renesting by females). Field crews conducted standard lek counts, unreconciled double-observer counts, and resighting at leks within the study area from 15 March - 14 May 2012. This effort resulted in 32 standard lek counts at 10 different lek locations (6 active) in Colorado, 58 standard lek counts at 16 lek locations (of which 15 were active) in Wyoming, 131 lek-days of resighting at 15 different leks, and 58 unreconciled double-observer counts. Crews did not conduct paired display rate observations in 2012. Field crews entered and proofed all field data by 1 June 2012.

Data on GPS male survival, lek attendance, and inter-lek movement are still being analyzed. A cursory review of location data suggests that yearling males started attending leks later in the season, moved significantly further, and visited more leks during the breeding season than adult males, but further analysis is needed to quantify these differences. Several males with GPS transmitters disappeared without any evidence of mortality. It is unclear whether those transmitters failed, whether the transmitters were destroyed or lost power following mortality, or whether the transmitters slipped off and failed to transmit their last location. Because random censoring is an assumption of survival analysis, we will be attaching miniature VHF mortality transmitters underneath the GPS transmitters in fall 2012 to test whether GPS males whose transmitters stop transmitting data are still alive or dead.

Three males made round-trips of 34-58 km between fall 2011 and summer 2012 (Fig. 4). Yearling male 105064 captured near the Hell's Canyon lek in March 2011 (sc. Sweetwater Co., WY) took a route south of Powder Rim and traveled east ~67 km to summer near the Hangout Ridge and Little Robber leks (w. Carbon Co., WY). The same bird returned ~58 km west in late fall 2011, traveling a route north of Powder Rim this time (crossing Sand and Skull creeks), to winter near the Stateline lek (sc. Sweetwater Co., WY). In 2012, the bird moved back to the east ~58 km via a slightly different route north of Powder Rim in February, attended the Hangout Ridge lek from March through early May as an adult, then summered again near the Little Robber lek. Yearling male 105075 was captured in March 2011 near Sugarloaf South lek, moved 34 km SE to spend the spring and summer in Sand Wash, returned ~34-44 km to winter near the Sugarloaf, G Flat, and Whiskey Draw leks within the Hiawatha field boundary, then moved ~44 km back to Sand Wash in early March 2012 and attended Central Sand Wash lek as an adult through early May. Adult male 105058B was captured near the Chicken Creek lek in Oct 2011. He then moved NE in Dec 2012 to winter 16-22 km away on the east side of Kinney Rim near Sand Creek lek. In March 2012, he moved 21 km further NE to attend the Ft. LaClede lek. He then returned 36 km SW back over Kinney Rim in late June 2012 and summered near the Alkali Creek lek.

Five males made one-way trips of 25-59 km between fall 2011 and summer 2012 (Fig. 4). Yearling male 107564 was captured near G Flat lek in Feb 2012, then traveled SW over Cold Springs Mountain, down to Brown's Park, then into Utah and over the Uinta Mountains in late March. The male then attended the Davenport Junction and Chicken Springs leks in spring 2012 (n. Uintah Co., ~27-31 km north of Vernal, Utah). Yearling male 107690 was captured near the G Flat lek in Feb 2012 and moved 38 km SE to Sand Wash (via Dry Ridge) in April 2012. Adult male 93629R was captured on Rifes Rim in Nov 2011 and moved ~22 km north in early March 2012 to attend the Dripping Rock lek. Yearling male 107692 was captured near the Hell's Canyon lek in Feb 2012, then traveled NE ~9 km to spend the spring near Muir Reservoir lek east of Kinney Rim. The male then moved 20 km NE up near Skull Creek Rim for May-June, then another 33 km east to summer near the Courthouse Butte lek just west of Hwy 13 north of Baggs, WY. Yearling male 107694 was captured in Feb 2012 on the east side of the Hiawatha project area and moved 13 km east over Kinney Rim to spend the spring near Muir Reservoir, then another 20 km north to the upper end of Shell Creek (near the Eversole Ranch) to summer. Adult male 105079B was captured on the Rifes Rim East lek in early April 2012 then moved 30 km west in late May and summered near the Little Basin and Buffalo Springs 2 leks.

In winter and spring 2012, we discovered that color-bands older than one year were deteriorating and either expanding and sliding down over the metal band or breaking and falling off. We were only able to recapture two males in February 2012 to replace and correct their color-band combinations. Field crews conducting resighting at leks reported numerous cases of incomplete or incorrect band combinations caused by color-bands breaking and falling off or expanding and slipping down over metal bands. Because of these problems, I anticipate even more males with incorrect band combinations being recorded showing up on leks in spring 2013. It is logistically impossible to recapture all previously marked males, and we will be unable to capture sufficient numbers of new males in fall 2012 to estimate differences in return rates or survival. For that reason, we decided not to color-band a separate sample of non-juvenile (adult or yearling) males in fall 2012 and will focus instead only on marking GPS males. Field work will continue through summer 2013.

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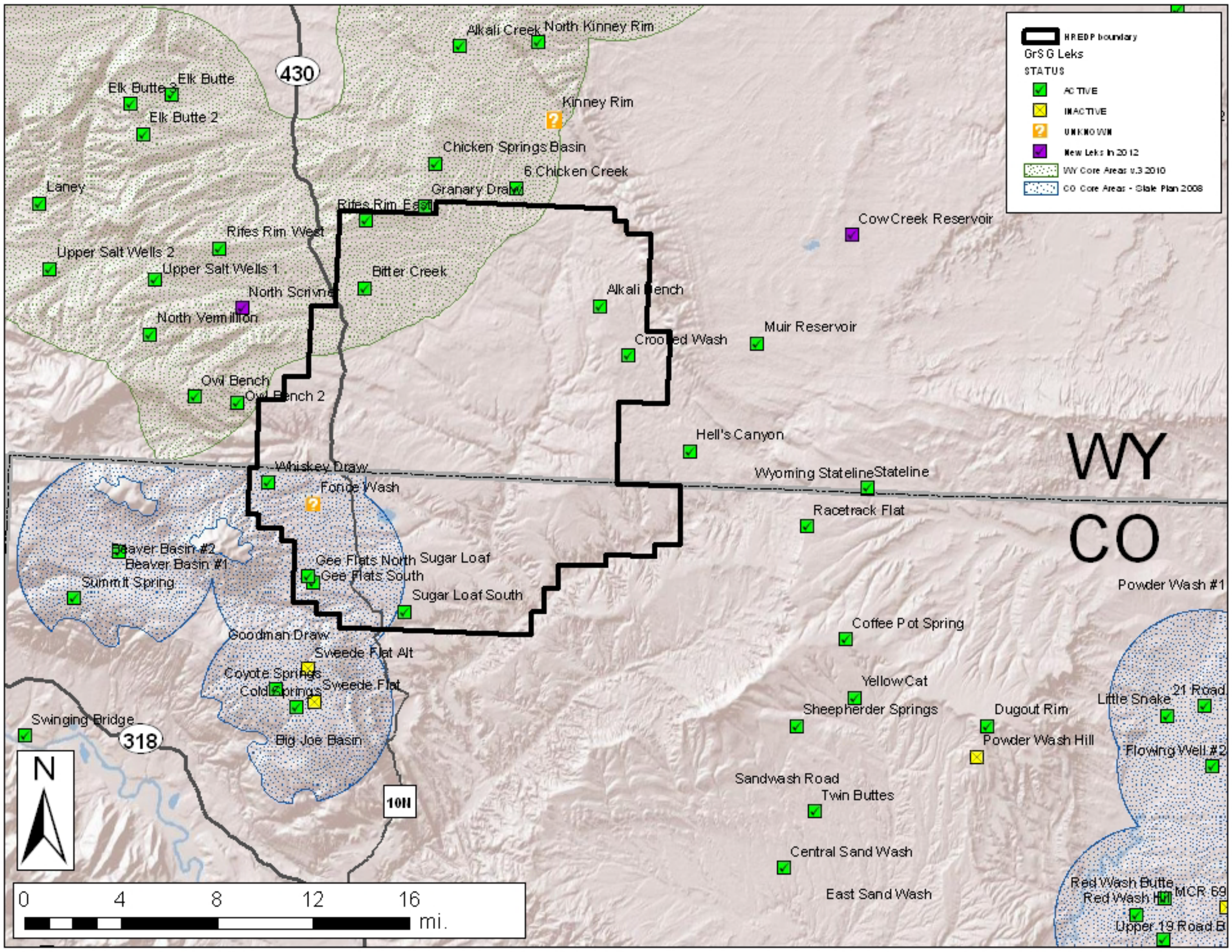

Figure 1. Hiawatha Regional Energy Development project area, Colorado and Wyoming greater sage-grouse core areas, and surrounding region showing known active, inactive, and unknown status greater sage-grouse leks as of 2012, plus two new potential leks that were discovered by monitoring and checking early-morning locations of GPS males in 2012.

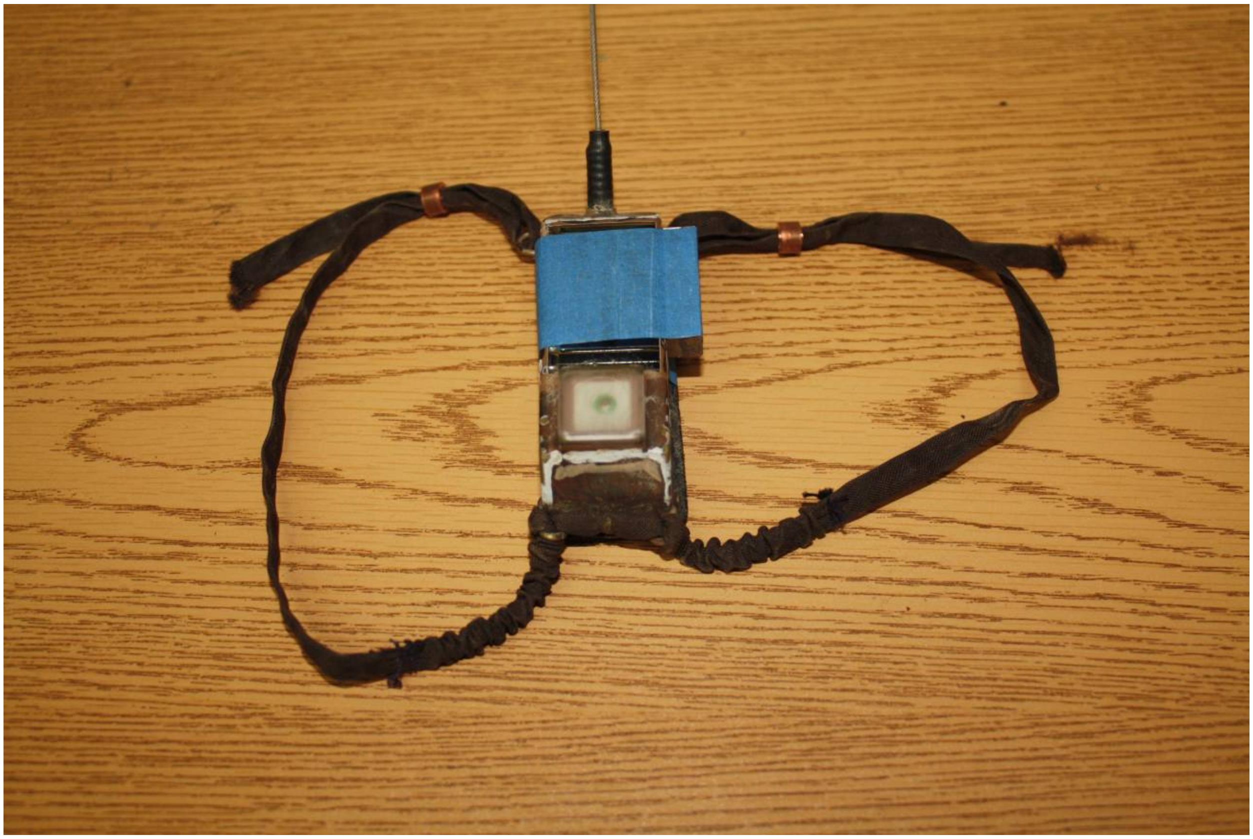

Figure 2. Harness design for rump-mounted leg-loop attachment of solar-powered GPS satellite PTT transmitters to male greater sage-grouse.

Figure 3. Attachment, placement, and camouflage of rump-mounted, solar-powered, GPS satellite PTT transmitters for male greater sage-grouse.

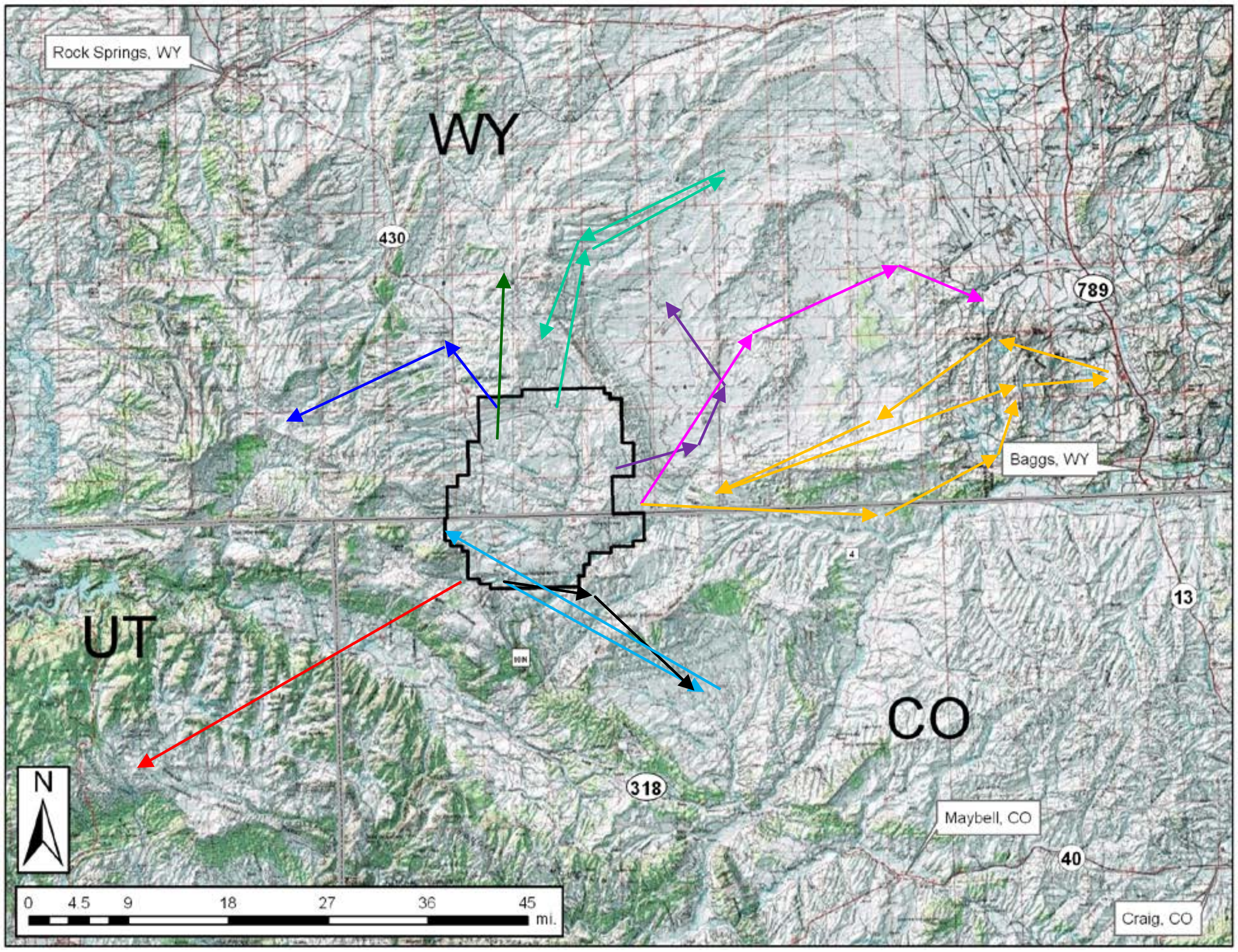

Figure 4. Long-distance round-trip and one-way movements by selected GPS males from fall 2011 through summer 2012 in relation to the Hiawatha Regional Energy Development project area and known lek locations in Sweetwater and Carbon counties, WY and Moffat Co., CO.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	0660	: Greater Sage-grouse Conservation
Task No.:	N/A	: Assessment of greater sage-grouse response to pinyon-juniper removal in the Parachute-Piceance-Roan population of northwestern Colorado
Federal Aid Project No.	N/A	

Period Covered: September 1, 2011 – August 31, 2012

Author: B. L. Walker

Personnel: B. Holmes, B. Petch, T. Knowles, B. deVergie

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the author. Manipulation of these data beyond that contained in this report is discouraged.

ABSTRACT

Greater sage-grouse (*Centrocercus urophasianus*) in the Parachute-Piceance-Roan (PPR) region of western Colorado face at least two major potential stressors: projected habitat loss from energy development and a long-term decline in habitat suitability associated with pinyon-juniper (PJ) encroachment. PJ removal may be a useful mitigation tool to offset potential habitat losses associated with energy development. Although PJ removal is commonly used to improve habitat for greater sage-grouse, no studies to date have quantified the timing or magnitude of how birds respond to treatments. Since 2008, Colorado Parks and Wildlife (CPW) has cooperated with industry and landowner partners to investigate the effectiveness of PJ removal in restoring sage-grouse habitat in the PPR.

In fall 2008, I established nine “survey” study plots, arranged in three groups of three, with each group consisting of a sagebrush control plot, an untreated PJ control plot, and a PJ treatment plot. Treatments were completed in 2010 and 2011. Track surveys were incomplete in winter 2011-2012 due to a lack of snow. Surveying for pellets over four summers (2009-2012) indicated that mean pellet occupancy was higher on sagebrush control plots (range 0.197-0.449) than on plots with encroaching PJ across all years (range 0.023-0.076). The proportion of sample units containing pellets increased within two years after PJ removal on one plot treated in 2010 (Upper Galloway) and within one year on a plot treated in 2011 (Ryan Gulch). There was no meaningful increase in pellets on the third survey plot treated in 2010 (Black Sulphur). Twelve additional transect plots were added in fall 2010 and two more transect plots were added in summer 2011. Winter transects were incomplete in winter 2011-2012 due to a lack of snow. All 14 transect plots were surveyed for pellets in summer 2011 and 2012. Preliminary transect data indicated low mean pellet densities on three PJ-Control plots over two years (range 0.00-0.58 pellet piles/km), and low densities on PJ-Treatment plots prior to treatment (mean = 0.03 pellet piles/km). There was no detectable increase in mean pellet density on treated transect plots within 10-12 months after PJ removal (mean = 1.04 pellet piles/km). Pellet density was substantially higher on four Sagebrush-Control transect plots over two years (range 11.10 - 27.14

piles/km) and on one transect plot within five years after treatment (Lower Barnes, 13.78 - 25.71 piles/km) than on PJ-Control plots (range) or PJ-Treatment plots prior to treatment (range). Detectability of pellets varies substantially among observers and also by pellet age/condition class.

WILDLIFE RESEARCH REPORT

ASSESSMENT OF GREATER SAGE-GROUSE RESPONSE TO PINYON-JUNIPER REMOVAL IN THE PARACHUTE-PICEANCE-ROAN POPULATION OF NORTHWESTERN COLORADO

BRETT L. WALKER

PROJECT OBJECTIVES

The objective of this study is to measure short-term (two to five years) responses of greater sage-grouse to experimental PJ removal using changes in winter track and pellet occupancy in a before-after control-treatment framework.

SEGMENT OBJECTIVES

Objectives of this study from fall 2011 through summer 2012 were to:

1. Complete PJ removal treatments.
2. Conduct winter track surveys and pellet surveys on 9 survey plots and 14 transect plots.
3. Summarize results of winter track surveys and pellet surveys from 2008-2012.

INTRODUCTION

Large-scale changes to sagebrush ecosystems and historical population declines (Schroeder et al. 2004) have raised concern about the status and conservation of greater sage-grouse (*Centrocercus urophasianus*) and contributed to the recent listing of the species as warranted but precluded under the Endangered Species Act (DOI 2010). The Parachute-Piceance-Roan (PPR) region holds one of seven distinct geographic populations of greater sage-grouse in northwestern Colorado. Greater sage-grouse in the PPR are of conservation concern due to a long-term reduction in habitat suitability caused by encroachment of pinyon pine (*Pinus monophylla*) and juniper (*Juniperus scopularum* and *Juniperus utahensis*) into sagebrush and potential impacts from rapidly increasing energy development.

Removal of pinyon-juniper (PJ) from areas with an existing sagebrush understory may help restore sage-grouse habitat and offset future potential habitat losses from energy development. Pinyon-juniper encroachment into sagebrush over the last 150 years has been identified as a threat to the species' habitat in the PPR, in Colorado and range-wide (CGSSC 2008; Chapter IV; Fig. 30). Encroachment is thought to be caused by fire suppression, reduced fire frequency due to removal of residual grass via livestock grazing, and a window of climatic conditions suitable for PJ establishment during the late 1800s and early 1990s (Miller and Rose 1999). Pinyon-juniper removal has been widely implemented in Colorado and range-wide (CGSSC 2008). However, sage-grouse responses to PJ removal remain poorly studied (Commons et al. 1999), and the timing and magnitude of greater sage-grouse responses following treatment is unknown. For this reason, it is difficult to judge whether PJ removal can effectively increase available habitat and offset impacts from energy development.

Since 2008, the Colorado Division of Wildlife, now Colorado Parks and Wildlife (CPW), and industry and landowner partners have been cooperating on research to assess the value of removing encroaching PJ as a mitigation strategy in the PPR. The main objective of this study is to measure short-term (<5 years) responses of greater sage-grouse to experimental PJ removal using changes in winter track and pellet occupancy in a before-after control-treatment framework. This progress report summarizes preliminary results from winter track and summer pellet surveys for the period December 2008 - August 2012.

STUDY AREA

Study plots are within or immediately adjacent to the current occupied range of greater sage-grouse in the PPR (Fig. 1). Birds in the PPR population inhabit the tops of ridges and plateaus dominated by mountain big sagebrush (*Artemisia tridentata vaseyana*) and a mixture of sagebrush and "mountain shrubs" (e.g., serviceberry, *Amelanchier* spp.; Gambel oak, *Quercus gambelii*, snowberry, *Symphoricarpus* sp.; antelope bitterbrush, *Purshia tridentata*, mountain mahogany, *Cercocarpus* sp., etc.). These areas are typically interspersed with patches of aspen (*Populus tremuloides*) and Douglas-fir (*Pseudotsuga menziesii*). Sagebrush and mixed sagebrush-mountain shrub habitats at higher elevation give way to PJ woodland on lower-elevation ridges that largely precludes use by sage-grouse. Our study plots are situated along the ecotone where PJ is encroaching upslope into sagebrush and sagebrush-mountain shrub habitat.

METHODS

Plot Selection

I used vegetation, topography, marked bird locations, aerial photography, and on-site visits to identify nine study plots in 2008 (Fig. 2) and 14 additional study plots in 2010 and 2011 (Figs. 2-4). Two of the plots added in 2010 (Upper and Lower Bar D) were treated by the Bureau of Land Management and were included opportunistically in our study (Fig. 2). Study plots were selected based on: (1) density of PJ; (2) shrub composition, density, and height; (3) topography; (4) proximity to areas of known use by greater sage-grouse; and (5) proximity to, and likelihood of, energy development within five years. All plots had a sagebrush-dominated shrub layer, typically intermixed with mountain shrubs, and topography similar to habitats used by sage-grouse in the PPR from 2006-2010 (Apa 2010, Walker 2010). The southeast portion of the PPR population is experiencing intensive energy development, but there is currently no development within the study plots and limited development nearby.

Assessing Response to Pinyon-Juniper Removal

I am using a before-after, control-treatment design to compare changes in sage-grouse winter track occupancy and pellet occupancy among control and removal plots before and after encroaching PJ is removed. Caution must be exercised in interpreting results because estimates of track and pellet occupancy only give an index of frequency of use during a defined survey period, rather than a measure of abundance, density, habitat quality, or habitat selection (*contra* Dahlgren et al. 2006).

I have three levels of treatment: 1) "PJ-Treatment" plots where encroaching PJ is removed, 2) "PJ-Control" plots where encroaching PJ is not removed, and 3) "Sagebrush-Control" plots with suitable sagebrush habitat and no PJ. Data from PJ-Treatment plots are used to measure changes in track and pellet occupancy before and after PJ removal. PJ-Control plots allow us to measure background changes and variation in track and pellet occupancy in areas with encroaching PJ in the absence of treatment. Sagebrush-Control plots allow us to estimate background changes and variation in track and pellet occupancy in habitats already suitable for sage-grouse. Most plots were surveyed for one to three years prior to PJ removal and will be surveyed for two to five years following removal. I established three study plots per treatment in fall 2008 for a total of nine original "survey" plots (Fig. 2). I established 12 additional transect-based study plots in fall 2010 and two more transect-based plots in summer 2011 (Figs. 2-4), for a total of 23 study plots (nine survey, 14 transect).

Winter Track Surveys and Transects.

I originally planned to estimate frequency of winter use using occupancy and density of tracks measured on snowshoe surveys (McKenzie et al. 2006) using two different sampling methods based on square sample units (Becker et al. 1994, 1998) or with a transect-intercept probability estimator (Becker 1991, Becker et al. 1994). However, winter track surveys were discontinued in 2012 due to lack of snow and ongoing logistical problems associated with using a time-sensitive survey methodology (see Results and Discussion, below). See previous annual reports for details of winter track survey methodology.

Summer Pellet Surveys and Transects

I obtained an index of frequency of sage-grouse use of plots by surveying for pellets during the summer and estimating pellet occupancy. I used two different sampling techniques for pellet surveys. The first technique involves surveying a systematic random sample of 30 x 30 m sample units on each "survey" plot. The second technique involves surveying for pellets within 1.5 m on either side of linear transects spaced 50 m apart that ran the length of each "transect" study plot.

With both sampling methods, field crews searched for, counted, and removed pellets from each sample unit (or transect line) within the plot once per year in July-August. For all pellets, field crews recorded their condition and appearance (crumbly-bleached vs. hard-dry, vs. fresh-wet) to estimate age and used composition to estimate when during the year they were deposited. Field crews identified those containing intact insect parts and flower heads as "summer" pellets (April-October) and those containing only digested sagebrush leaves as "winter" pellets (November-March) (Wallestad et al. 1975). Field crews recorded single pellets, pellet piles (i.e., day or night roost piles), and cecal droppings separately. Observers recorded pellets or groups of pellets within 10 cm of each other as one pile, with the constraint that pellets or groups of pellets > 10 cm apart could not be counted as the same pile.

I initially planned to sample each study plot twice per summer – at the start and end of each three month summer period. However, surveys took too long to complete, so I opted to sample each plot only once per year in July-August and to instead change the "survey period" (i.e., the period during which birds can deposit pellets) from three months to 12 months. Field crews marked all sample unit centers and transect lines with aluminum tags, high-visibility stakes and high-visibility flagging to ensure consistency in sampling locations across years. On survey plots, each observer carefully and thoroughly surveyed sample units by slowly walking 10 parallel 30-m lines spaced 3 m apart. On transect plots, observers survey for pellets by walking flagged transect lines and searching within 1.5 m on either side of the line. Observers recorded anecdotal evidence of occupancy while surveying (e.g., clockers, nests and eggs, feathers, birds, etc.).

These two survey methods for pellets have the following assumptions: 1) all pellets can be correctly identified to species; 2) all pellets can be correctly distinguished as either a chick or adult pellet by size; 3) all pellets deposited during the survey period (during the previous year) can be correctly distinguished from pellets deposited prior to the survey period by condition and appearance; 4) all pellets can be correctly distinguished by season ("winter" vs. "summer") by pellet composition; 5) surveying does not influence whether or not pellets are present in sampled units (or along transects). To address assumption 1, I trained observers to distinguish dusky grouse from sage-grouse pellets prior to surveys. Adult pellets of the two species can be distinguished by composition and smell in any season. Adult-sized sage-grouse typically consume 13-39% sagebrush throughout the spring and summer and >99% sagebrush in winter (Wallestad et al. 1975, Schroeder et al. 1999). For this reason, pellets of adult sage-grouse contain sagebrush year-round, and unlike dusky grouse, consistently smell like sage, even after a year in the field. To address assumption 2, I trained crews to distinguish adult from chick pellets by length and diameter Quantitative analyses will only include data on adult-size pellets because it may not be possible to distinguish pellets of dusky grouse vs. sage-grouse chicks due to overlap in chick diets (both species consume primarily insects and forbs as chicks; Zwickel 1992, Drut et al. 1994). To address anssumption 3, crews differentiated pellets in the field based on condition (bleached and crumbly vs. hard and dry vs. fresh and green/moist). I am also testing how pellet condition changes with age by placing piles of fresh test pellets within representative sagebrush habitat in the field and photographing changes in condition over time. I will test anssumption 4 by testing how often observers correctly assess composition and season for pellets collected from marked birds in different seasons. Assumption 5 may be violated if surveyors flushed birds that then landed within another sample unit or along a transect line later surveyed. However, violation of this assumption is unlikely to meaningfully influence analyses because the number of pellets birds could deposit before the unit or transect gets surveyed (on the order of minutes or hours) is miniscule compared to the 12-month survey period.

Pellet Detectability

Detectability of sage-grouse pellets is typically low and may vary among observers (Dahlgren et al. 2006) and with pellet condition and appearance. In summer 2009, I determined that crews were unable to sample each plot more than once per summer due to logistical constraints, so I instead estimated variation in detectability of pellets among observers by having each observer survey eight test sample units in which I placed piles of fresh pellets of various sizes at random directions and distances within the sample unit from the sample unit center. Pellet piles of different sizes and different condition and appearance classes (bleached-crumbly vs. hard-dry and fresh-green/moist) were also placed at random distances within 1.5 m along two 400-m long test transects in 2011. Test piles were placed at the nearest point on the ground (i.e., not on top of vegetation) to the randomly selected location. Sample units and transects used for testing detectability and observer bias were exhaustively grid-searched for pellets and pellets were removed prior to testing to ensure that no other pellets were present during the test. Testing of observers was blind; observers did not know which sample units or transects contained pellets, how many, or their condition or appearance.

Vegetation Sampling

Field crews sampled vegetation at locations used by marked sage-grouse in winter and within a systematic-random subset of sample units and transect points in July-August 2011 to determine whether local-scale habitat on study plots would be suitable for wintering sage-grouse once the PJ overstory was removed. At each sampling point, field crews laid out two 30-m perpendicular tapes running N-S and E-W and measured: shrub canopy cover by species using the line-intercept method (Canfield et al. 1941); and species and height of the nearest shrub (excluding inflorescences) within 2.5 m every 5 m along each tape.

Pinyon-Juniper Removal

All treatments were done by contractors using either a Bobcat with a Fecon head or a Hydro-axe. Contractors were instructed to remove only pinyon-juniper and to avoid removing sagebrush or mountain shrubs. A partial treatment was done on the Black Sulphur plot in August 2009 and completed in July 2010, the Upper Galloway plot was treated in November 2010, and the Ryan Gulch plot was treated in August 2011. Among transect plots, Upper and Lower Bar D plots were treated by BLM in January 2011, and three plots (Cottonwood, Magnolia South, and Lower Wagonroad) were treated in August-November 2011.

RESULTS AND DISCUSSION

Winter Track Surveys and Transects

Field crews completed one to three rounds of winter track surveys per winter on the original nine survey plots in each winter 2008-2009, 2009-2010, and 2010-2011 and one to three rounds of surveys on 12 transect plots in winter 2010-2011 (Tables 1-2). The number of rounds of surveys completed each winter depended on the number of storms that deposited fresh snow and on weather conditions following each storm.

Track surveys in winter 2011-2012 could not be completed due to lack of snow. Those data are not presented. We plan to discontinue future winter track survey work due to the logistical difficulties of implementing time-sensitive survey protocols in winter and because of unpredictable nature of snow conditions from year to year. The first assumption of winter track surveys is that all sage-grouse leave tracks in the snow during the survey period (i.e., between when snowfall stops and when the survey is conducted). This assumption was regularly violated in winter 2011-2012. Winter 2011-2012 was extremely dry, so many of our study plots lacked snow cover for most of the winter. Those plots that did get snow often had it melt within 48 hrs before field crews could conduct surveys because of unusually warm and sunny weather. Field crews collected pellet samples for genetic analyses in winter 2008-2009, 2009-2010, and 2010-2011, but no pellets were collected in winter 2011-2012. However, genetic analyses will also be discontinued because we will not be able to compare them with winter track survey data.

Summer Pellet Surveys and Transects

Field crews conducted one round of pellet surveys on each of the nine original plots in August 2009, July-August 2010, July-August 2011, and July-August 2012 and one round of pellet surveys on each of 14 transect plots in July-August 2011 and July-August 2012 (Tables 3-6). Pellet survey data from 2009-2012 indicated substantially higher mean seasonal and year-round pellet occupancy on Sagebrush-Control survey plots than on PJ-Control (no treatment) plots or on PJ-Treatment plots prior to treatment. There was no observed increase in year-round pellet occupancy on the Black Sulphur plot within two years following treatment. Year-round pellet occupancy increased on the Upper Galloway plot (to 0.245) and to a lesser degree on the Ryan Gulch plot (to 0.188) in 2012 (Fig. 5, Tables 3-6). The mean proportion of sample units with pellets detected on Sagebrush-Control survey plots in 2012 rebounded to 2009-2010 levels (Fig. 5). Higher pellet densities were also observed on Sagebrush-Control plots in 2012 than in 2011. No transect plots showed a detectable increase in pellet occupancy within one year following treatment. Pellet density on the Lower Barnes plot (treated by BLM in 2007-2008) was again comparable to Sagebrush-Control plots in 2012.

Pellet Detectability

Observers conducted pellet surveys on test sample units on the Upper Galloway plot in 2009, on test sample units on the Stake Springs plot in 2010, on sample units on both plots in 2011, and on plots on Upper Galloway in 2012. Each test sample unit contained sage-grouse pellets distributed randomly in piles of different sizes. Test units in 2009 only contained fresh green pellets. To more realistically mimic pellets encountered during field surveys, test units in 2010-2011 contained both fresh and aged sage-grouse pellets, and test units in 2012 contained pellets of varying age, but no fresh pellets. I used test units on Stake Springs in 2010 because the Upper Galloway plot was inaccessible when testing was conducted. Mean detectability of fresh test pellets on Upper Galloway in 2009 was low overall (0.125) and variable among the five observers (range 0.06-0.22). Mean detectability of mixed fresh and aged pellets on Stake Springs was higher in 2010 (0.375) and less variable among the three observers (range 0.344-0.406). Detectability of mixed fresh and aged pellets on Upper Galloway was higher in 2011 (0.323) than in 2009 (0.125) but lower on Stake Springs in 2011 (0.297) than in 2010 (0.375). Detectability among the six observers in 2011 was less variable than in 2009 but similar to values in 2010 (range 0.281-0.375 on Upper Galloway, range 0.281-0.375 on Stake Springs). There was no indication of any substantive difference in average pellet detectability between plots in 2011 (0.323 ± 0.027 SE on Upper Galloway; 0.297 ± 0.021 SE on Stake Springs). Detectability of pellets of varying age (but none fresh) among four observers on the Upper Galloway plot in 2012 was higher than in 2009 and 2011 (range 0.375-0.625). I suspect that part of the reason for low detectability in 2009 was that only fresh green pellets were used that year. Higher detectability in 2011 and 2012 compared with 2009 may be due to inclusion of a higher percentage of older, more bleached (and therefore more visible) pellets.

Observers conducted test pellet surveys along two 400 m long transects on Bar D ridge in August 2011. The first test transect had shorter shrubs and less shrub cover but more grass cover than the second test transect. Mean detectability of pellets on test transects was similar to that on test survey plots (Transect 1: 0.220 ± 0.040 SE; Transect 2: 0.390 ± 0.020 SE), but detectability was higher on the plot with taller, denser shrubs and less grass cover. Variation in detectability among observers ranged from 0.100-0.350 on the first transect and from 0.350-0.450 on the second transect.

Vegetation Sampling

Vegetation sampling data were collected on the nine original survey plots in August 2010. Vegetation sampling data were collected on the 14 transect plots in August-September 2011. A summary of vegetation sampling data will be presented in a subsequent progress report.

Overall, preliminary data from winter track surveys and summer pellet surveys during the pre-treatment phase of the project were largely as expected, with substantially greater use of Sagebrush-Control plots than either PJ-Control plots or PJ-Treatment plots prior to treatment. There was a detectable increase in the proportion of sample units containing pellets on two of three treatment plots within one to two years following treatment. There was no detectable increase on in pellets found on transect plots < 1 year following treatment.

The amount of time it took to conduct pellet surveys prevented us from repeating surveys within years. This in turn, prevented us from estimating detectability in an occupancy framework and forced us to estimate detectability experimentally using test plots instead. Variation in pellet detectability among observers may be an issue for interpretation of data from pellet surveys if the effect size of PJ treatment is small compared to variation in detectability of pellets among observers. Additional testing will be conducted to quantify the effects of pellet age and condition on detectability.

Annual variation in snow conditions and weather continued to make winter track surveys logistically challenging, and in many cases, impossible for field crews. For this reason, we discontinued winter track surveys (and associated pellet sampling for genetics work) in 2012 and will focus future survey effort on pellet surveys conducted during the summer. Discontinuing winter track surveys will allow funding to be redirected exclusively to summer pellet surveys and extend the number of years we can document sage-grouse responses to PJ treatment using summer pellet surveys (Table 7).

Current post-treatment data are insufficient to draw final conclusions about responses of greater sage-grouse to PJ removal. Use on two of three survey plots appears to be increasing within one to two years following treatment (Upper Galloway and Ryan Gulch), but we observed little obvious response within one year post-treatment on five other treated plots. Response to PJ treatments may take longer than one to two years if few yearlings are available to colonize newly created habitat. For that reason, response to PJ treatments may not be detected until there is a good year for sage-grouse reproduction.

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Table 1. Preliminary winter track occupancy estimates ($\Psi \pm$ SE) and number of tracks detected for greater sage-grouse in the Parachute-Piceance-Roan region of western Colorado, USA, during the winters 2008-2009, 2009-2010, and 2010-2011. Crews attempted to survey all 30 x 30 m sample units within each plot on each sampling visit, but it was not always possible, so sample size of units per plot (n) varies. The Upper Galloway, Ryan Gulch, Eureka, and Stake Springs plots were reduced in size between winter 2008-2009 and winter 2009-2010 to better match treatment areas (Upper Galloway and Ryan Gulch) or to ensure that each plot within each group could be sampled within a single day (Eureka and Stake Springs). Detectability of sage-grouse winter tracks in snow within 30 m x 30 m sample units is assumed to be 1.0.

	PJ - Treatment Plots			**PJ - Control (No Treatment) Plots**			**Sagebrush - Control Plots**		
	1-Upper Galloway	2-Black Sulphur	3-Ryan Gulch	1-Dry Ryan	2-Eureka	3-Stake Springs	1-Dry Gulch	2-Canyon Creek	3-Black Cabin
					Winter 2008-2009				
Ψ[a]	0.000[b] ± 0.000	0.000[b] ± 0.000	0.000[b] ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.076 ± 0.006	0.012 ± 0.012	0.036 ± 0.020
T[c]	0, 0	0	0	0, 0	0	0	45, 21	17	18
N[d]	2	1	1	2	1	1	2	1	1
n[e]	109, 109	74	76[d]	65, 65	93[d]	76 [d]	84[d], 87	82	83[d]
					Winter 2009-2010				
Ψ[a]	0.000[b] ± 0.000	0.000[f] ± 0.000	0.000[b] ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.015 ± 0.010	0.012 ± 0.012	0.006 ± 0.006
T[c]	0, 0, 0	0, 0, 0	0, 0	0, 0, 0	0, 0, 0	0, 0	10, 12, 0	10, 0, 0	2, 0
N[d]	3	3	2	3	3	2	3	3	2
n[e]	82, 84, 84	74	71[d], 72	65, 65, 65	86, 86, 81	63, 63	87, 87, 87	82, 81, 82	85, 84
					Winter 2010-2011				
Ψ[a]	0.000[g] ± 0.000	0.000[h] ± 0.000	0.000[b] ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.017 ± 0.006	0.006 ± 0.006	0.006 ± 0.006
T[c]	0, 0	0, 0	0, 0	0, 0	0, 0	0, 0	7, 2	0, 5	0, 2
N[d]	2	2	2	2	2	2	2	2	2
n[e]	84, 84	74, 74	72, 72	65, 65	86, 86	63, 63	87, 87	84, 84	84, 84

[a] Ψ = estimated winter track occupancy. Values for plots surveyed on multiple visits are means averaged across visits.
[b] Pre-treatment data
[c] T = total no. of individual tracks detected per sampling visit.
[d] N = no. of sampling visits per winter.
[e] n = no. of 30 x 30 m sample units surveyed per plot per sampling visit.
[f] The Black Sulphur plot was partially treated in fall 2009. Values represent data collected 3-5 mo. following partial PJ removal.
[g] The Upper Galloway plot was completed in summer 2010. Values represent data collected 6-8 mo. following complete PJ removal.
[h] Treatment on the Black Sulphur plot was completed in summer 2010. Values represent data collected 6-8 mo. following complete PJ removal.

Table 2. Preliminary estimates of track density for greater sage-grouse on transect plots in the Parachute-Piceance-Roan region of western Colorado, USA, in winter 2010-2011. Detectability of sage-grouse winter tracks in snow that cross transect lines is assumed to be 1.0.

	PJ – Treatment Transect Plots						**PJ – Control (No Treatment) Transect Plots**				**Sagebrush – Control Transect Plots**			
	Pre-treatment					**Post-treatment**								
	Lower Bar D	Upper Bar D	Magnolia South	Cotton-wood	Lower Wagon-road	Lower Barnes[a]	Bar D Split	Magnolia North	Sprague	Upper Wagon road	Bar D Control	Magnolia Control	Upper Barnes Control	Wagon-road Control
x[b]	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00	0.00	0.00[a]	-	0.00 ± 0.00	0.00	0.00	0.00 ± 0.00	7.21 ± 0.29	- -	0.00
T[c]	0, 0, 0	0, 0, 0	0, 0	0	0	0, 0[a]	-	0, 0	0	0	0, 0	51, 47	-	0
I[d]	0, 0, 0	0, 0, 0	0, 0	0	0	0, 0[a]	-	0, 0	0	0	0, 0	51, 47	-	0
N[e]	3	3	2	1	1	1	-	2	1	1	2	2	-	1
km[f]	9.58	6.36	9.80	7.70	7.64	5.88	-	6.44	13.40	9.80	5.88	6.80	-	5.88

[a] Data on Lower Barnes were collected opportunistically and are presented for comparison only. Data represent 3 years post-treatment. Treatments on Lower Barnes were started in summer 2007 and completed in summer 2008 by the Bureau of Land Management. No pre-treatment data are available.
[b] x = no. of individual tracks encountered per km of transect. Estimates for plots surveyed on multiple sampling visits are means averaged across visits ± SE.
[c] T = total no. of individual tracks detected per sampling visit.
[d] I = total no. of tracks detected intersecting transects per sampling visit.
[e] N = no. of sampling visits per winter.
[f] Total kilometers of transect line surveyed per plot per sampling visit.

Table 3. Preliminary estimates of proportion of sample units (p ± SE) containing greater sage-grouse pellets on survey plots in the Parachute-Piceance-Roan population of western Colorado, USA in 2009. Values presented do not account for variation in pellet detectability.

	PJ - Treatment Plots				PJ - Control (No Treatment) Plots				Sagebrush - Control Plots			
	Pre-treatment											
	1-Upper Galloway[a]	2-Black Sulphur[a]	3-Ryan Gulch[a]	Mean ± SE	1-Dry Ryan	2-Eureka	3-Stake Springs	Mean ± SE	1-Dry Gulch	2-Canyon Creek	3-Black Cabin	Mean ± SE
	n = 49[b]	n = 38[b]	n = 48[b]	N = 3[c]	n = 35	n = 50	n = 38	N = 3	n = 54	n = 41	n = 42	N = 3
p_S[d]	0.061	0.000	0.042	0.034	0.000	0.040	0.026	0.022	0.111	0.049	0.119	0.093
	± 0.034	± 0.000	± 0.029	± 0.018	± 0.000	± 0.033	± 0.026	± 0.012	± 0.043	± 0.034	± 0.05	± 0.022
p_W[d]	0.061	0.000	0.104	0.055	0.000	0.000	0.079	0.026	0.389	0.268	0.381	0.346
	± 0.034	± 0.000	± 0.044	± 0.03	± 0.000	± 0.000	± 0.044	± 0.026	± 0.066	± 0.069	± 0.075	± 0.039
p_{YR}[d]	0.082	0.000	0.146	0.076	0.000	0.040	0.105	0.048	0.444	0.293	0.452	0.397
	± 0.039	± 0.000	± 0.051	± 0.042	± 0.000	± 0.033	± 0.05	± 0.031	± 0.068	± 0.071	± 0.077	± 0.052

[a] Data represent pre-treatment values. Numbers preceding plot names refer to which set the plot is in (e.g., Set 1 = Upper Galloway, Dry Ryan, Dry Gulch; Fig. 2).

[b] n refers to no. of 30 x 30 m units sampled within the study plot.

[c] N refers to no. of study plots.

[d] p = proportion of sample units per plots in which greater sage-grouse pellets were detected. Subscripts refer to season (S = summer; W = winter; YR = year-round).

Table 4. Preliminary estimates of proportion of sample units (p ± SE) containing greater sage-grouse pellets on survey plots in the Parachute-Piceance-Roan population of western Colorado, USA in 2010. Values presented do not account for variation in pellet detectability.

	PJ - Treatment Plots				PJ - Control (No Treatment) Plots				Sagebrush - Control Plots			
	Pre-treatment			Post-treatment								
	1-Upper Galloway[a]	3-Ryan Gulch[a]	Mean ± SE	2-Black Sulphur[a,b]	1-Dry Ryan	2-Eureka	3-Stake Springs	Mean ± SE	1-Dry Gulch	2-Canyon Creek	3-Black Cabin	Mean ± SE
	n = 49[c]	n = 48[c]	N = 2[d]	n = 38[c]	n = 35	n = 50	n = 38	N = 3	n = 43	n = 41	n = 42	N = 3
p_S[e]	0.000	0.042	0.021	0.000	0.000	0.020	0.000	0.007	0.326	0.024	0.095	0.148
	± 0.000	± 0.029	0.021	± 0.000	± 0.000	± 0.02	± 0.000	± 0.007	± 0.071	± 0.024	± 0.045	± 0.091
p_W[e]	0.000	0.063	0.031	0.026	0.029	0.020	0.026	0.025	0.558	0.293	0.333	0.395
	± 0.000	± 0.035	0.031	± 0.026	± 0.028	± 0.02	± 0.026	± 0.003	± 0.076	± 0.071	± 0.073	± 0.083
p_{YR}[e]	0.000	0.104	0.052	0.026	0.029	0.040	0.026	0.032	0.721	0.293	0.333	0.449
	± 0.000	± 0.044	± 0.052	± 0.026	± 0.028	± 0.028	± 0.026	± 0.004	± 0.068	± 0.071	± 0.073	± 0.136

[a] Data represent pre-treatment values. Numbers preceding plot names refer to set (e.g., Set 1 = Upper Galloway, Dry Ryan, Dry Gulch, etc.; Fig. 2).

[b] The Black Sulphur plot was partially treated in fall 2009. PJ removal was completed in June 2010 prior to pellet surveys in July-August 2010 .

[c] n refers to the no. of 30 x 30 m units sampled within the study plot.

[d] N refers to the no. of study plots.

[e] p = proportion of sample units per plots in which greater sage-grouse pellets were detected. Subscripts refer to season (S = summer; W = winter; YR = year-round).

Table 5. Preliminary estimates of proportion of sample units (p ± SE) containing greater sage-grouse pellets on survey plots in the Parachute-Piceance-Roan population of western Colorado, USA in 2011. Values presented do not account for variation in pellet detectability.

	PJ - Treatment Plots				**PJ - Control (No Treatment) Plots**				**Sagebrush - Control Plots**			
	Pre-treatment	**Post-treatment**										
	3-Ryan Gulch[a]	1-Upper Galloway[b]	2-Black Sulphur[b]	Mean ± SE	1-Dry Ryan	2-Eureka	3-Stake Springs	Mean ± SE	1-Dry Gulch	2-Canyon Creek	3-Black Cabin	Mean ± SE
	n = 48[c]	n = 49[c]	n = 38[c]	N = 2[d]	n = 35	n = 50	n = 38	N = 3	n = 43	n = 41	n = 42	N = 3
p_S[e]	0.000[a] ± 0.000	0.020[b] ± 0.020	0.000[b] ± 0.000	0.010 ± 0.010	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.000 ± 0.000	0.093 ± 0.044	0.000 ± 0.000	0.000 ± 0.000	0.031 ± 0.031
p_W[e]	0.063[a] ± 0.035	0.061[b] ± 0.034	0.000[b] ± 0.000	0.031 ± 0.031	0.000 ± 0.000	0.020 ± 0.020	0.000 ± 0.000	0.007 ± 0.007	0.349 ± 0.073	0.171 ± 0.059	0.071 ± 0.040	0.197 ± 0.081
p_{YR}[e]	0.063[a] ± 0.035	0.082[b] ± 0.039	0.000[b] ± 0.000	0.041 ± 0.041	0.000 ± 0.000	0.020 ± 0.020	0.000 ± 0.000	0.007 ± 0.007	0.349 ± 0.073	0.171 ± 0.059	0.071 ± 0.040	0.197 ± 0.081

[a] Data represent pre-treatment values.

[b] Data represent 1 yr post-treatment. Treatments on Black Sulphur and Upper Galloway were completed in summer 2010. Numbers preceding plots names refer to which set the plot is in (e.g., Set 1 = Upper Galloway, Dry Ryan, Dry Gulch, etc.; Fig. 2).

[c] n refers to the no. of 30 x 30 m units sampled within the study plot.

[d] N refers to the no. of study plots.

[e] p = proportion of sample units per plots in which greater sage-grouse pellets were detected. Subscripts refer to season (S = summer; W = winter; YR = year-round).

Table 6. Preliminary estimates of proportion of sample units (p ± SE) containing greater sage-grouse pellets on survey plots in the Parachute-Piceance-Roan population of western Colorado, USA in 2012. Values presented do not account for variation in pellet detectability.

	PJ - Treatment Plots (Post-treatment)				**PJ - Control (No Treatment) Plots**				**Sagebrush - Control Plots**			
	1-Upper Galloway[a]	2-Black Sulphur[a]	3-Ryan Gulch[b]	Mean ± SE	1-Dry Ryan	2-Eureka	3-Stake Springs	Mean ± SE	1-Dry Gulch	2-Canyon Creek	3-Black Cabin	Mean ± SE
	n = 49[a,b]	n = 38[a]	n = 48[c]	N = 3[d]	n = 35	n = 50	n = 38	N = 3	n = 43	n = 41	n = 42	N = 3
p_S[e]	0.061	0.000	0.000	0.020	0.000	0.040	0.000	0.013	0.279	0.000	0.071	0.117
	± 0.034	± 0.000	± 0.000	± 0.02	± 0.000	± 0.028	± 0.000	± 0.013	± 0.068	± 0	± 0.04	± 0.084
p_W[e]	0.204	0.026	0.188	0.139	0.029	0.020	0.000	0.016	0.581	0.341	0.214	0.379
	± 0.058	± 0.026	± 0.056	± 0.057	± 0.028	± 0.020	± 0.000	± 0.008	± 0.075	± 0.074	± 0.063	± 0.108
p_{YR}[e]	0.245	0.026	0.188	0.153	0.029	0.040	0.000	0.023	0.605	0.341	0.262	0.403
	± 0.061	± 0.026	± 0.056	± 0.065	± 0.028	± 0.028	± 0.000	± 0.012	± 0.075	± 0.074	± 0.068	± 0.104

[b] Data represent 2 years post-treatment. Treatments on Black Sulphur and Upper Galloway were completed in summer 2010. Numbers preceding plots names refer to which set the plot is in (e.g., Set 1 = Upper Galloway, Dry Ryan, Dry Gulch, etc.; Fig. 2).

[b] n refers to the number of sample units per study plot.

[c] Data represent 1 year post-treatment. Treatment on Ryan Gulch was completed in summer 2011.

[d] N refers to the total no. of study plots.

[e] p = proportion of sample units per plots in which greater sage-grouse pellets were detected. Subscripts refer to season (S = summer; W = winter; YR = year-round).

Table 7. Preliminary estimates of the density of greater sage-grouse pellets encountered on transect plots in the Parachute-Piceance-Roan population of western Colorado, USA in 2011. Values presented do not account for variation in pellet detectability.

	PJ - Treatment Plots						**PJ - Control (No Treatment) Plots**				**Sagebrush - Control Plots**			
	Pre-treatment			**Post-treatment**										
	Magnolia South	Cottonwood	Lower Wagon-road	Lower Bar D[a]	Upper Bar D[a]	Lower Barnes[b]	Bar D Split	Magnolia North	Sprague	Upper Wagon-road	Bar D Control	Magnolia Control	Upper Barnes Control	Wagon-road Control
x_S[c]	0.00	0.00	0.00	0.00	0.00	2.38[b]	0.00	0.00	0.00	0.00	2.38	0.59	0.00	1.70
X_S[d]		0.00 ± 0.00		0.00 ± 0.00[e]				0.00 ± 0.00				1.17 ± 0.72		
x_W[c]	0.10	0.00	0.00	0.00	0.00	11.39[b]	0.00	0.00	0.00	0.00	3.23	18.24	13.51	4.76
X_W[d]		0.03 ± 0.03		0.00 ± 0.00[e]				0.00 ± 0.00				9.94 ± 4.43		
x_{YR}[c]	0.10	0.00	0.00	0.00	0.00	13.78[b]	0.00	0.00	0.00	0.00	5.61	18.82	13.51	6.46
X_{YR}[d]		0.03 ± 0.03		0.00 ± 0.00[e]				0.00 ± 0.00				11.10 ± 3.84		
km[f]	9.80	7.70	7.64	9.58	6.36	5.88	4.68	6.44	13.40	9.80	5.88	6.80	5.92	5.88

[a] Treatments on Lower Bar D and Upper Bar D were completed in January 2011. Data represent 6 mo. post-treatment.

[b] Data on Lower Barnes were collected opportunistically and are presented for comparison only. Data represent 3 years post-treatment. Treatments on Lower Barnes were started in summer 2007 and completed in summer 2008 by the Bureau of Land Management. No pre-treatment data were available.

[c] x = no. pellet piles detected per km of transect. Subscripts refer to season (S = summer; W = winter; YR = year-round).

[d] X = mean no. pellet piles detected per km of transect across study plots in each treatment. Subscripts refer to season (S = summer; W = winter; YR = year-round).

[e] Post-treatment mean values for PJ – Treatment plots include data from the Lower and Upper Bar D plots and exclude data from Lower Barnes.

[f] Total kilometers of transect line surveyed per plot.

Table 8. Preliminary estimates of the density of greater sage-grouse pellets encountered on transect plots in the Parachute-Piceance-Roan population of western Colorado, USA in 2012. Values presented do not account for variation in pellet detectability.

	PJ - Treatment Plots (Post-treatment)						**PJ - Control (No Treatment) Plots**				**Sagebrush - Control Plots**			
	Magnolia South	Cotton-wood	Lower Wagon-road	Lower Bar D[a]	Upper Bar D[a]	Lower Barnes[b]	Bar D Split	Magnolia North	Sprague	Upper Wagon-road	Bar D Control	Magnolia Control	Upper Barnes Control	Wagon-road Control
x_S[c]	0.20	0.00	0.00	0.00	0.00	7.31[b]	0.00	0.00	0.00	0.10	9.52	11.91	0.00	2.72
X_S[d]		0.07 ± 0.07		0.00 ± 0.00[e]				0.03 ± 0.02				6.04 ± 2.80		
x_W[c]	2.65	0.27	0.00	0.00	0.63	18.20[b]	0.00	2.02	0.00	0.20	13.44	34.12	7.94	28.91
X_W[d]		0.97 ± 0.84		0.31 ± 0.31[e]				0.56 ± 0.42				21.10 ± 6.21		
x_{YR}[c]	2.86	0.27	0.00	0.00	0.63	25.51[b]	0.00	2.02	0.00	0.31	22.96	46.03	7.94	31.63
X_{YR}[d]		1.04 ± 0.91		0.31 ± 0.31[e]				0.58 ± 0.42				27.14 ± 7.97		
km[f]	9.80	7.52	7.64	9.58	6.36	5.88	4.68	6.44	12.52	9.80	5.88	6.80	5.92	5.88

[a] Treatments on Lower Bar D and Upper Bar D were completed in January 2011. Data represent ~18 mo. post-treatment.

[b] Data on Lower Barnes were collected opportunistically and are presented for comparison only. Data represent 4-5 years post-treatment. Treatments on Lower Barnes were started in summer 2007 and completed in summer 2008 by the Bureau of Land Management. No pre-treatment data were available.

[c] x = no. pellet piles detected per km of transect. Subscripts refer to season (x_S = summer; x_W = winter; x_{YR} = year-round).

[d] X = mean no. pellet piles detected per km of transect across study plots in each treatment. Subscripts refer to season.

[e] Post-treatment mean values are calculated separately for the Upper and Lower Bar D plots and for the Lower Barnes plot.

[f] Total kilometers of transect line surveyed per plot. Portions of transects on Cottonwood and Sprague conducted in 2011 were eliminated in 2012 because they had shrub understory or topography unsuitable for greater sage-grouse.

Table 9. Timeline for research on greater sage-grouse response to pinyon-juniper removal in the Parachute-Piceance-Roan population, western Colorado, 2008-2015.

Task	Initiation	Completion
Identification of plots for PJ removal	COMPLETE	COMPLETE
Remove encroaching PJ on survey treatment plots (2009-2011)	COMPLETE	COMPLETE
Remove encroaching PJ on transect treatment plots (2011)	COMPLETE	COMPLETE
Pellet surveys (annually)	25 June	31 Aug
Prepare cumulative report (annually)	1 Sep	1 Oct
Prepare cumulative final report	1 Aug 2015	1 Oct 2015

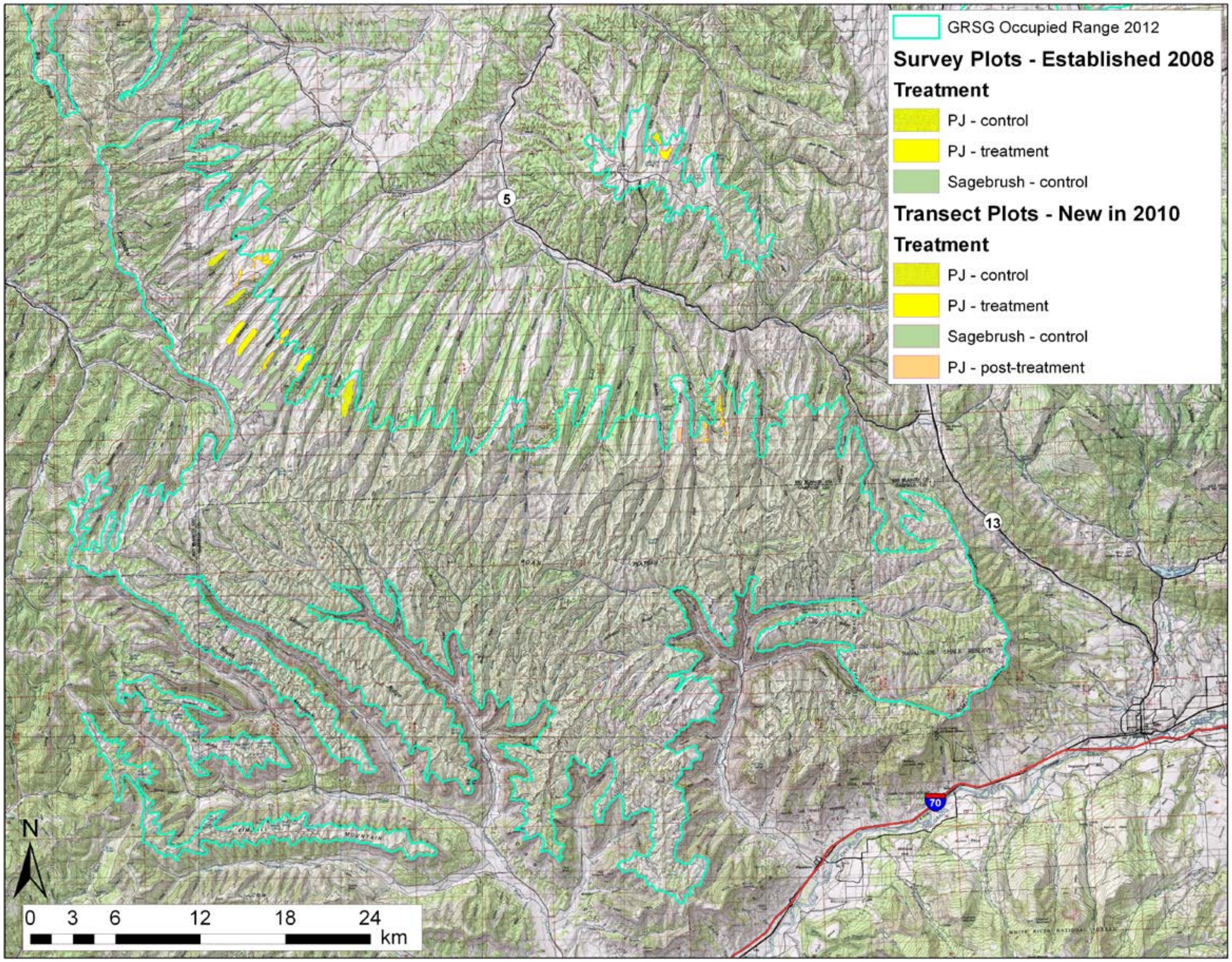

Fig. 1. Map of greater sage-grouse occupied range (as of 2012) showing study plot locations for the pinyon-juniper removal experiment in the Parachute-Piceance-Roan population of western Colorado, USA.

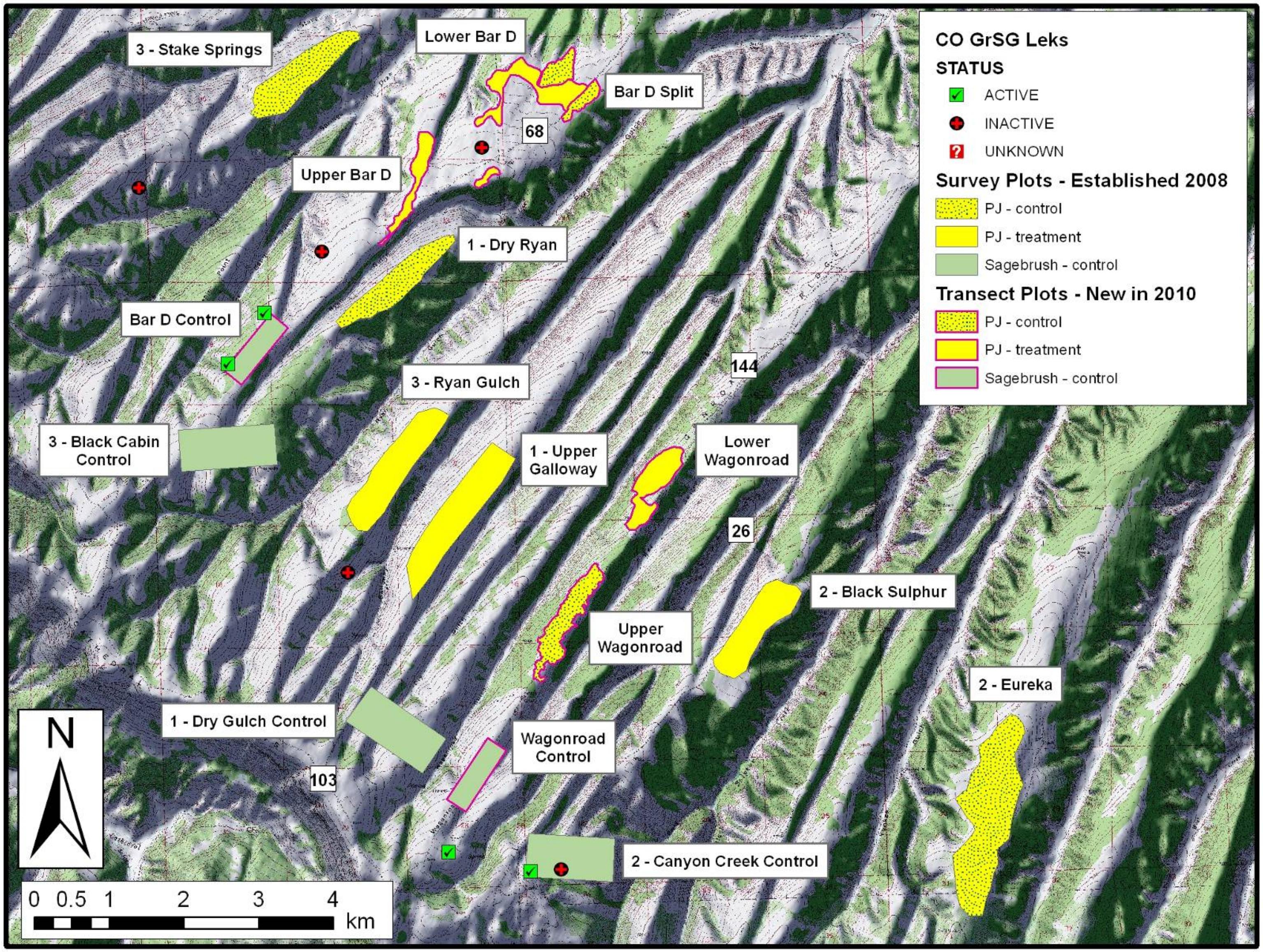

Fig. 2. Map of the west side plots, showing the original nine survey plots established in 2008 and seven new transect plots established either in fall 2010 or summer 2011 (Bar D Split) for the pinyon-juniper removal experiment in the Parachute-Piceance-Roan greater sage-grouse population.

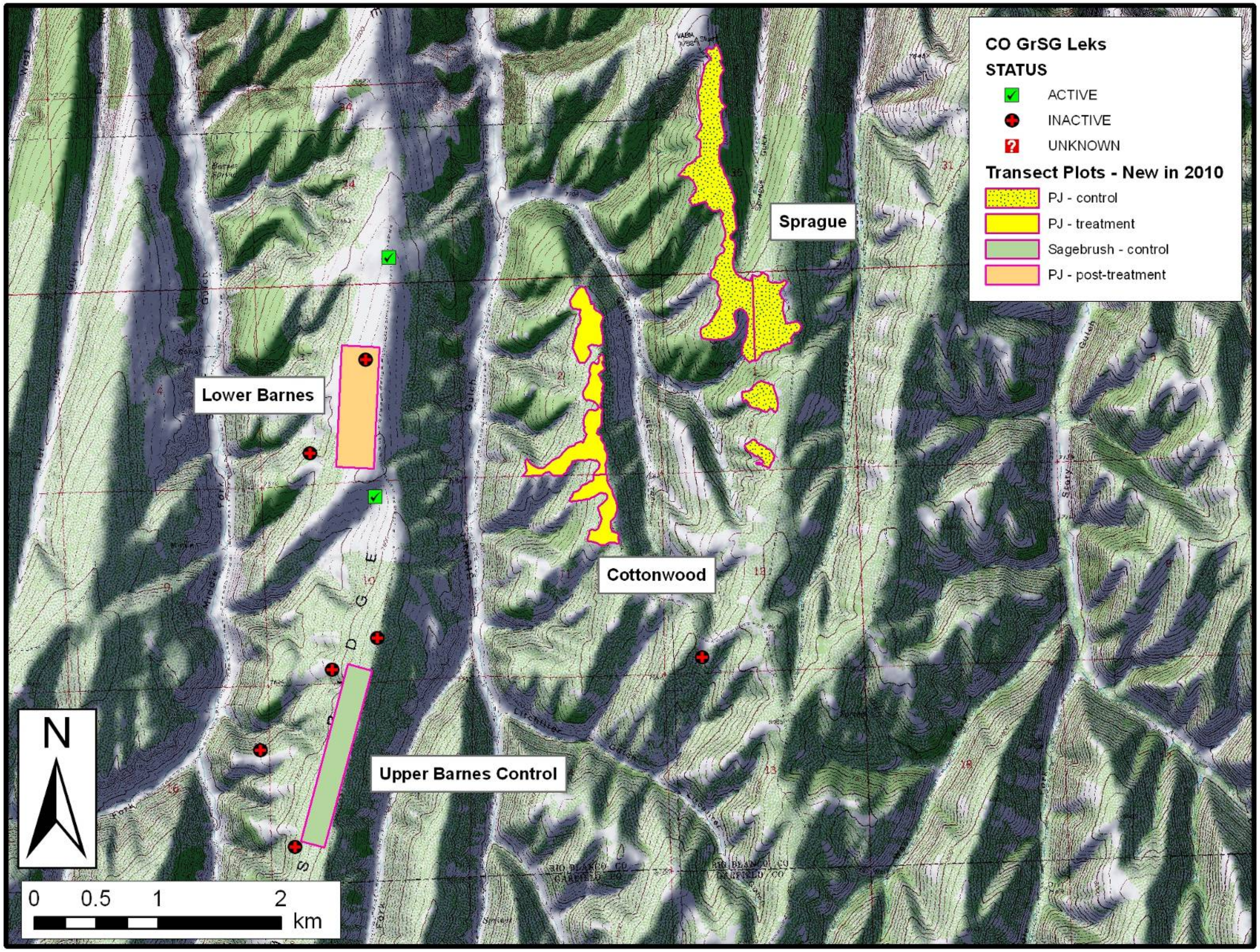

Fig. 3. Map of the east side plots, showing the four new transect study plots for the pinyon-juniper removal experiment in the Parachute-Piceance-Roan greater sage-grouse population. All plots shown were established in fall 2010 except Upper Barnes, which was established in summer 2011.

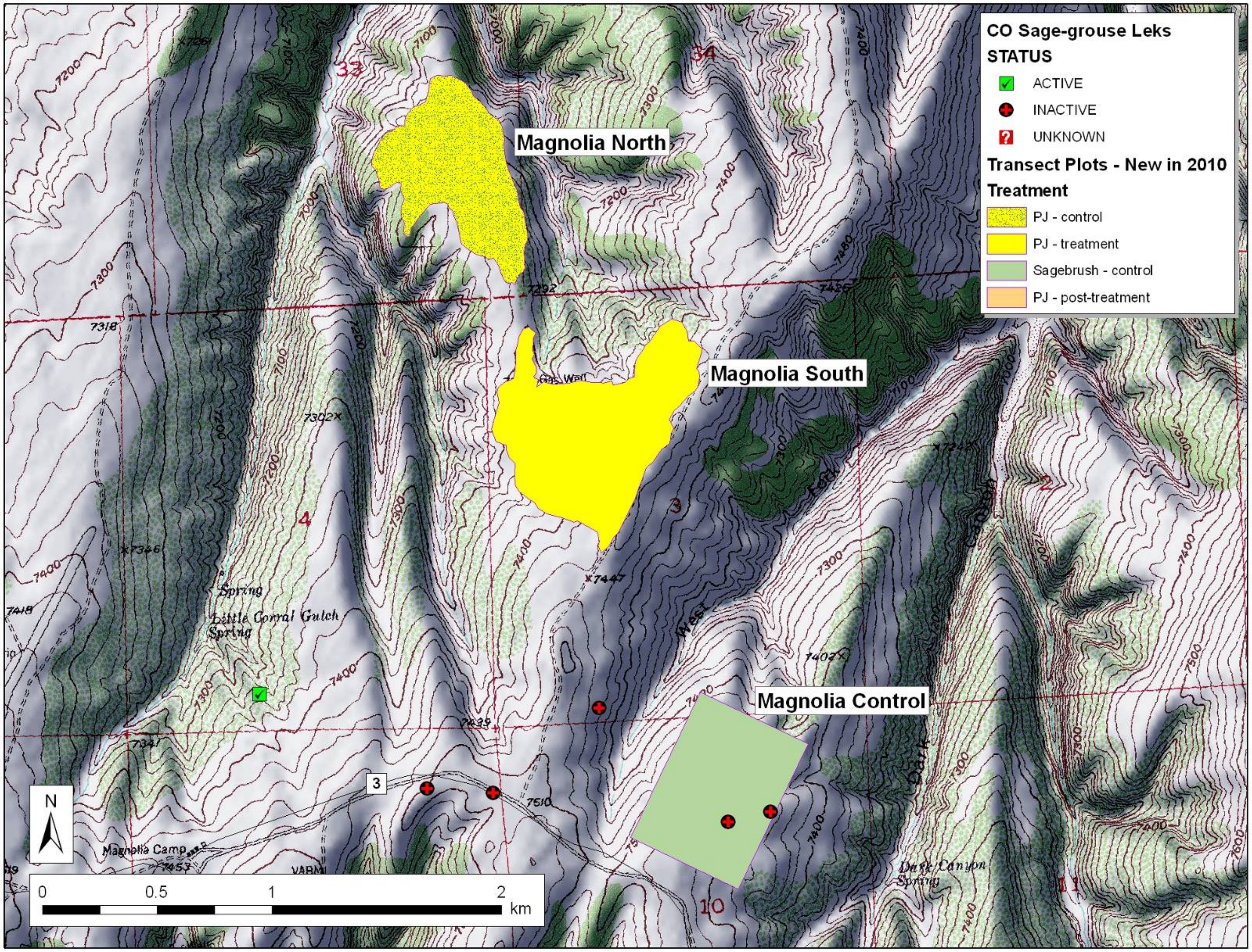

Fig. 4. Map of the Magnolia plots, showing the three new transect study plots established in fall 2010 for the pinyon-juniper removal experiment in the Parachute-Piceance-Roan greater sage-grouse population.

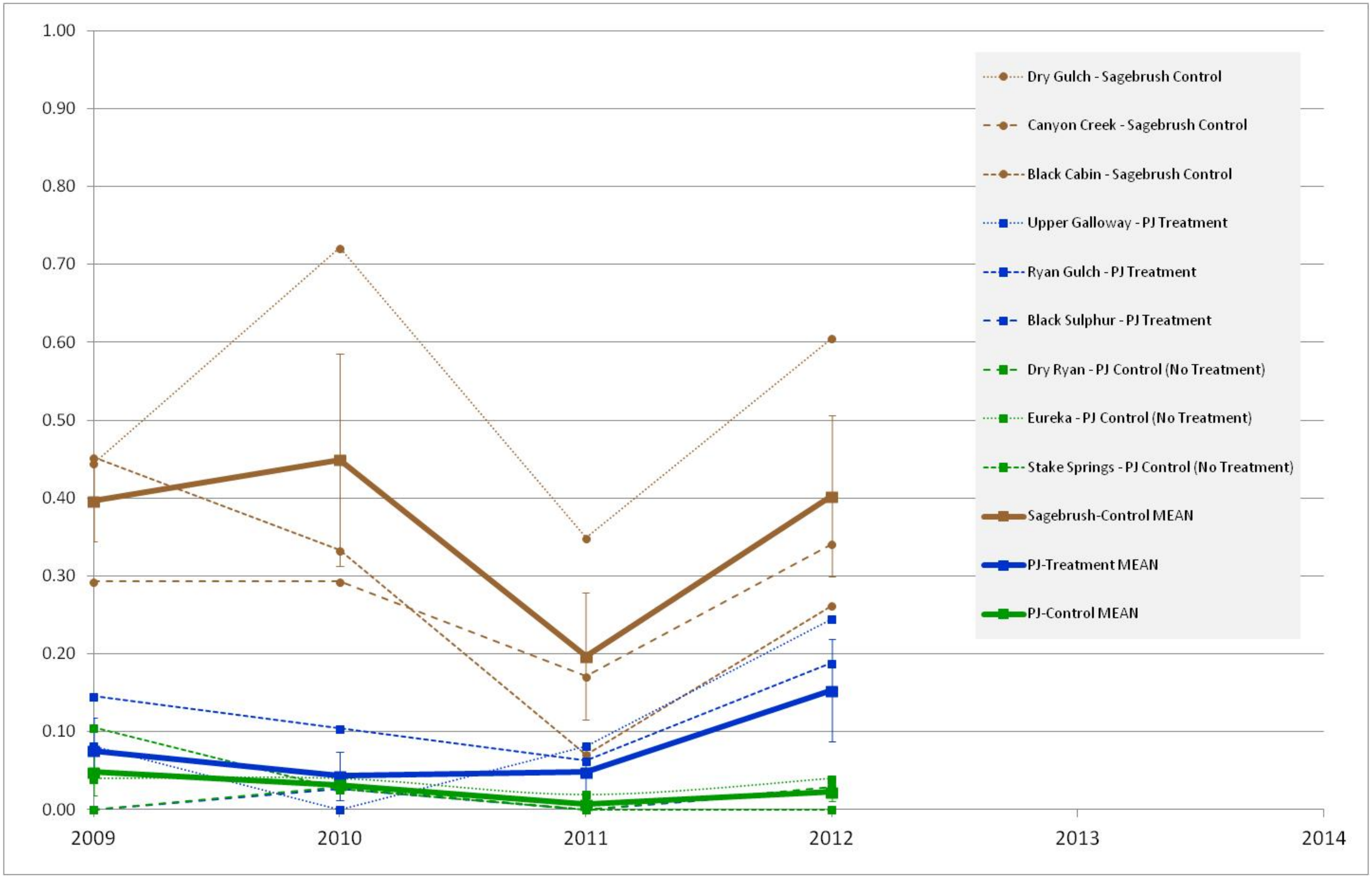

Fig. 5. Trends in the proportion of sample units detected with greater sage-grouse pellets by study plot (and treatment) in the Parachute-Piceance-Roan population of western Colorado, USA, 2009-2012. Values presented do not account for variation in pellet detectability.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	0660	: Greater Sage-grouse Conservation
Task No.:	N/A	: Evaluation of alternative population monitoring strategies for greater sage-grouse (Centrocercus urophasianus) in the Parachute-Piceance-Roan population of northwestern Colorado

Federal Aid
Project No. N/A

Period Covered: September 1, 2011 – August 31, 2012

Author: B. L. Walker

Personnel: B. Holmes, B. Petch, B. deVergie

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the author. Manipulation of these data beyond that contained in this report is discouraged.

ABSTRACT

Robust estimates of population size and population trends provide the scientific basis for managers to make appropriate and defensible recommendations regarding land-use decisions, harvest regulations and mitigation efforts to conserve wildlife. When linked with environmental variables, robust monitoring programs also allow managers to examine wildlife responses to disease, land-use patterns, habitat treatments, weather, ecological succession, and disturbance. Significant progress has been made over the past three decades in sampling methodology, statistical analysis, and tracking technology to estimate wildlife abundance. However, many wildlife monitoring programs continue to use uncorrected population indices, even though they may or may not provide reliable information on population status or trends. For this reason, it is essential to evaluate alternative approaches to population monitoring in terms of estimator precision, cost, practicality, and level of disturbance to wildlife. Lek counts are the primary index used by all state wildlife agencies in the western U.S. to monitor changes in greater sage-grouse (*Centrocercus urophasianus*) abundance, but lek counts rely on untested assumptions about male lek attendance, detectability, inter-lek movement, sex ratio and the proportion of leks in the population that are counted. Colorado Parks and Wildlife (CPW) currently uses maximum counts of males from multiple uncorrected counts at each lek as the basis for estimating 3-year running averages for each lek and population zone. Given the availability of new methodological and statistical approaches for estimating wildlife populations, it is worth comparing the use of uncorrected lek counts with other potential monitoring methods. Both dual-frame sampling of leks and non-invasive genetic mark-recapture methods appear suitable and promising for monitoring trends in greater sage-grouse populations. The purpose of this study is to evaluate and compare the reliability and efficiency of dual-frame sampling, genetic mark-recapture and standard lek counts for estimating population size and trend and to estimate sex ratios in the Parachute-Piceance-Roan (PPR) population of greater sage-grouse in northwest Colorado. The first of three years of dual-frame sampling by helicopter was successfully completed in April-May 2012, with

each of 137 list- and area-frame cells surveyed three times. We discovered four new active lek locations in sampling cells (three list-frame, one area-frame) and three other leks while in transit between cells. Capture and marking of females began July 15, 2012. Fall-winter pellet sampling for the genetic mark-recapture portion of the study is scheduled to begin in November 2012.

WILDLIFE RESEARCH REPORT

EVALUATION OF ALTERNATIVE POPULATION MONITORING STRATEGIES FOR GREATER SAGE-GROUSE (*Centrocercus urophasianus*) IN THE PARACHUTE-PICEANCE-ROAN POPULATION OF NORTHWESTERN COLORADO

BRETT L. WALKER

PROJECT OBJECTIVES

1. Estimate proportion of known leks, the average number of males attending known leks and the total number of male greater sage-grouse attending leks in the population during three consecutive lekking seasons using dual-frame sampling from helicopter.

2. Estimate population size using genetic mark-recapture during two consecutive fall/winter seasons.
3. Estimate sex ratio using genetic sampling during two consecutive fall/winter seasons.
4. Compare and contrast methods for estimating population size and evaluate the application of auxiliary data for improving estimations based on standard lek-count data.

SEGMENT OBJECTIVES

1. Conduct the first year of dual-frame sampling of leks from helicopter in April - May 2012.
2. Capture, mark, and monitor VHF females starting July 15, 2012 for genetic mark-recapture.

INTRODUCTION

Population monitoring programs are essential for the proper management of wildlife species. Well-designed monitoring strategies allow researchers to determine the status of species of interest; these are often keystone, umbrella, threatened or endangered, candidate, game, or invasive species. For candidate species under the Endangered Species Act, effective monitoring plays a critical role in determining their appropriate conservation status. Additionally, data from monitoring programs informs managers and allows for adjustment of land use strategies, federal or state legal status, hunting regulations, and mitigation plans in response to current population status and trend. Monitoring programs also allow investigators to identify key factors such as disease, human land use, or natural disturbances that influence populations. To provide the information needed to evaluate the status of a population and inform management decisions, researchers need to provide accurate and defensible estimates of population size and trend.

Significant progress has been made in wildlife population estimation since the 1970s, driven by the practical need to estimate abundance and monitor populations over time (Burnham 2004). Advancements in methodology, statistical analysis and technology have been paramount in improving population size estimation and monitoring. Methods have expanded to include stratified and cluster sampling, mark-recapture, occupancy, dual-frame sampling, line intercept, adaptive cluster sampling, distance sampling, and indices from point or lek count data, among others. The development of technologies such as radio telemetry, satellite telemetry, global positioning systems (GPS), global information systems (GIS), genetic analysis and computer programs also represent major advances that contribute to improved monitoring strategies for wildlife species. Progress has also been made in the development of methods that reduce or eliminate disturbance to wildlife species, such as genetic mark-recapture and track surveys. Additionally, innovations in the size and design of radio and satellite transmitters have reduced the impact on study animals, allowed researchers to evaluate habitat use and management in a greater variety of species, and reduced the cost of monitoring per individual.

Despite the recent progress, generating accurate and defensible estimates of population size and trends remains a key challenge for wildlife biologists and managers. This is particularly true for monitoring "rare" species with a small number of individuals and for populations occurring at low densities. Researchers investigating rare species face numerous challenges including: a lack of appropriate methods, greater susceptibility to estimation bias, issues with imperfect detectability, clustering of animals, and insufficient funding. As a result of these logistic challenges, investigators often turn to population indices to estimate abundance or monitor populations. While indices may be easier to obtain, they are often based on assumptions or unknown variables. Therefore, their relation to the true population may be unclear (Witmer 2005) or inaccurate. As a result, indices may be inadequate when accurate estimates of abundance and trend are required to determine proper management of a wildlife population.

Recent declines in greater sage-grouse populations and substantial restriction of pre-settlement distribution of the species have been observed nationwide (Connelly and Braun 1997, Schroeder et al. 2004). These declines, in combination with habitat loss and human land use conflicts, have prompted repeated petitions for federal listing. In 2010, the species status was designated as warranted, but precluded under the Endangered Species Act (Leonard et al. 2000, Connelly et al. 2004, Schroeder et al. 2004, Braun et al. 2005, Aldridge et al. 2008, USFWS 2010). This decision by the U.S. Fish and Wildlife Service (USFWS) has created a critical need for accurate, defensible population estimations based on sound monitoring techniques. The development of innovative, yet practical methods for estimating populations of rare or cryptic animals, such as the sage-grouse, is essential to accurately determining population size, monitoring trends in abundance and instituting proper management practices.

During the spring breeding season, male greater sage-grouse gather to display on traditional strutting grounds (Patterson 1952), known as leks. These leks offer a unique opportunity to observe and count individuals, particularly males. Historically, lek counts have been considered to be the best, if not the only, means for monitoring populations of lekking species and are currently used by state wildlife agencies throughout the western United States (Connelly et al. 2004). These counts are based on standard protocols (Patterson 1952) and are assumed to provide information on population trends (Fedy and Aldridge 2011). However, lek counts are subject to numerous sources of sampling bias and do not generate rigorous and defensible population estimates required for protection and management of species (Walsh et al. 2010). Standard state lek count indices, like those used by Colorado Parks and Wildlife (CPW), are based on seasonal high counts of males attending leks to estimate population size. The use of lek-count indices to estimate population size and trend rely on untested assumptions and do not account for spatial and temporal variation in detectability. Implicit in lek count indices are assumptions about the proportion of leks that are known and counted, the proportion of males that attend leks, the proportion of males on leks that are detected by observers, the frequency of inter-lek movements by males, and the sex ratio of the population. Each of these factors affect the accuracy of greater sage-grouse abundance estimates to an unknown degree. Therefore, there is a great need to either quantify these variables and adjust lek count index estimates accordingly or develop new methods for estimating population size and trends over time.

Despite legitimate criticism, lek-count indices continue to be the primary means for monitoring changes in sage-grouse population size. Investigations into the reliability of lek-count data for monitoring changes in population size should be a research priority for greater sage-grouse (Naugle and Walker 2007) and the development of alternative population monitoring methods is essential for greater sage-grouse population monitoring and management, both in the Piceance and range-wide.

In recent years, attempts have been made to evaluate or improve lek count protocols for greater sage-grouse in order to generate more robust estimates of population size and trends. In addition to projects which evaluate the reliability of standard lek counts, alternative population estimation methods are being developed. These include survey methods that reduce disturbance to the species by employing non-invasive techniques. Recent advances in genetic mark-recapture using sources of DNA such as scat, feathers and hair have created new, innovative opportunities using mark-recapture theory (Lukacs and Burnham 2005a). Sampling of fecal DNA was first attempted in coyotes (Kohn et al. 1999) and has since been used in a variety of other species, including grizzly and brown bears (Mowat and Strobeck 2000, Boulanger et al. 2004, Bellemain et al. 2005), black bears (Coster et al. 2011), northern pike (Miller et al. 2001), northern goshawks (Bayard de Volo et al. 2005), Gunnison sage-grouse (Oyler-McCance, unpublished data) and humpback whales (Palsboll 1997). Genetic mark-recapture is a promising strategy for estimating and monitoring greater sage-grouse populations.

This study will allow us to evaluate the efficacy of using novel techniques for estimating population size and observing trends in the small, isolated population of greater sage-grouse in northwest Colorado. These techniques will be conducted in the field, assessed for reliability of estimates and evaluated for feasibility in long-term population monitoring. These monitoring strategies will also be compared to traditional methods for monitoring sage-grouse populations (i.e. lek counts), focusing on the benefits and disadvantages of each.

STUDY AREA

The Parachute-Piceance-Roan (PPR) population of greater sage-grouse is located northwest of Rifle and southwest of Meeker in western Colorado (Fig. 1). It is recognized as one of six distinct populations in the state (CGSSC 2008). Occupied range in the PPR is characterized by high-elevation ridges and plateaus broken up by steep canyons and drainages. Vegetation is dominated by mountain big sagebrush (*Artemisia tridentata vaseyana*), mixed sagebrush-mountain shrub habitat and pinyon-juniper (*Pinus edulis, Juniperus spp.*) woodlands with occasional patches of aspen (*Populus tremuloides*). Mixed sagebrush-mountain shrub is primarily comprised of mountain big sagebrush and serviceberry (*Amelanchier* spp.) with Gambel oak (*Quercus gambelii*), snowberry (*Symphoricarpus* sp.), antelope bitterbrush (*Purshia tridentata*), mountain mahogany (*Cercocarpus* sp.), and wild rose (*Rosa* sp.). Greater sage-grouse are largely restricted to elevations from 6,500-9,000-ft. (PPR-GSGWG 2008). Approximately 35% of the occupied range is managed by state or federal agencies with the remaining 65% privately owned, primarily by energy companies and ranchers. Only 4-5% of the total number of male greater sage-grouse counted on known leks in Colorado are located in the PPR (CGSSC 2008, PPR-GSGWG 2008).

The PPR population of greater sage-grouse was estimated to have approximately 340 total males in 2007 (PPR-GSGWG 2008). This small population is experiencing substantial landscape changes, including energy development and pinyon-juniper encroachment into areas of formerly suitable sagebrush habitat (PPR-GSGWG 2008). The PPR population may be especially vulnerable due to its small size and imminent reductions in suitable habitat resulting from the ongoing changes in land use. However, reliable information on population size or trends is not currently available to accurately assess the level of risk to the population. The limited data available to estimate long-term trends in the PPR population include estimates of male population size based on state lek counts conducted by helicopter dating back to 2005 (CGSSC 2008). Unfortunately, the utility of lek count data for reliably estimating population size or monitoring trends in lekking species has been heavily criticized (Beck and Braun 1980, Applegate 2000, Walsh et al. 2004, Walsh et al. 2010) and there is currently no scientifically defensible population estimate available for the PPR (PPR-GSGWG 2008).

METHODS

Dual-Frame Sampling

Dual-frame sampling of leks estimates the proportion of 1-km^2 cells covering occupied range that contain one or more greater sage-grouse leks as well as the average number of males attending those leks.

This information can be used to estimate: (1) the proportion of known leks, (2) the average number of males attending known leks, and (3) the population size of male greater sage-grouse attending leks. We will conduct dual-frame sampling of 1-km^2 cells from helicopter during three consecutive spring lekking seasons in 2012, 2013, and 2014.

We will survey for leks and count any leks found within two distinct sampling frames, the list frame and the area frame. The *list* frame consists of all 1-km^2 cells known to contain an active lek. The *area* frame consists of a spatially balanced random selection of 1-km^2 cells generated using a Reversed Randomized Quadrant-Recursive Raster (RRQRR; Theobald et al. 2007). Active leks are defined by Colorado Parks and Wildlife (CPW) as a location on the ground where one to two strutting males have been observed in two or more of the past five years. Leks of unknown status are those that have been recently identified and require a second year or observation before they can be defined as active. For the purposes of dual-frame sampling, unknown leks will be sampled as active leks. We will sample 39 1-km^2 list-frame cells in the PPR population (excluding the "Magnolia" portion) known to contain 49 known active leks and approximately 100 area-frame cells (Fig. 2). Any leks newly discovered in 2012 within either list-frame or area-frame cells will be added to the list frame for sampling in 2013 and 2014.

We investigated the statistical power to detect a 5%, 7.5% and 10% change in occupancy (the proportion of sample units containing one or more leks) based on lek activity observed during dual-frame sampling using simulations in Program Mark (White and Burnham 1999). We used input parameter values expected to represent the true population occupancy and anticipated sampling effort for the list and area frames. Capture histories were simulated based on input data (expected occupancy rate, detection probability and number of sample units surveyed) and analyzed in Program Mark to obtain standard errors (Runge et al. 2007). Power calculations were generated using Program R. Results indicate that with expected sampling effort (125-140 total 1-km^2 cells per season), power to detect a minimum of 7.5% annual rate of decline in occupancy will be approximately 0.95 with 15 years of surveillance.

All sampling will be conducted by helicopter. Helicopter sampling protocols are based on methods developed by Dr. Paul Lukacs (formerly with the Colorado Division of Wildlife, now at the University of Montana) with slight modifications to manage logistical problems unique to the PPR. Surveys will be conducted from mid-April to early May, the primary lekking period for sage-grouse in the PPR, and from 30 minutes before local sunrise to 2 hours after sunrise in accordance with standard CPW lek-count protocols. Observers will count leks by circling the lek, scanning with binoculars, and recording the sex of all birds present. A minimum of three rounds of surveys and counts will be conducted in each cell sampled. This allows estimation of detectability of leks and adheres to standard state lek-count protocols that stipulate at least three counts per year. If leks are discovered incidentally while flying to or from area or list-frame cells, those lek locations will be recorded and used to improve the following year's list frame.

For area-frame cells, observers will survey the entire cell and count any newly discovered leks. A waypoint will be marked at the center of the lek and the location will be added to the list frame the following year. New leks located in area-frame cells will be sampled on any subsequent survey of those cells. Observers will survey list-frame cells the same way as area-frame cells, but observers will also check and, if birds are present, count all previously known lek locations. If new leks are located within list-frame cells, those leks will also be counted and a waypoint marked at the center of the lek.

Dual-frame sampling data will be analyzed in an occupancy framework. Equations modified from Haines and Pollock (1998) will also be used to estimate population size using lek counts and the proportion of known vs. unknown leks.

Genetic Mark-Recapture

This project will use genetic mark-recapture methods to estimate population size. Birds will be captured, marked and have feathers sampled from July to October of each year within occupied range. Fecal pellets obtained from sampling in fall-winter will be genotyped to identify individual sage-grouse. While fecal pellets will predominantly consist of sagebrush DNA, traces of DNA from the intestinal walls of sage-grouse are transferred to the pellets and can be used to amplify microsatellites for genotyping.

Sage-grouse pellets are expected to be at low density due to the apparently small population size of birds in the Piceance and the relatively large size of the study area (1,473 km^2). In addition, unpredictable winter weather conditions, including snowfall and blowing snow, may obscure or bury tracks and pellets. In order to increase the number of samples collected, pellet samples will be obtained using several sampling strategies, including pellet collections from: (1) a random sampling scheme; (2) incidental locations of roost sites or pellets; and (3) back-tracking of incidentally located greater sage-grouse tracks in order to locate roost sites and pellets.

Roosting behavior of greater sage-grouse offers a unique opportunity for collecting pellet samples. As sage-grouse roost at night, they typically remain at a single location on the ground, regularly dropping fecal pellets. This results in condensed piles of pellets referred to as “roost piles” (Patterson 1952) (Fig. 3). Greater sage-grouse are a gregarious species with both males and hens forming flocks, particularly in the non-breeding or winter months (Patterson 1952). The average flock size of the PPR population is estimated at five to six birds with flocks as large as 24 birds observed (CPW, unpublished data). In the winter months when snow is abundant and temperatures often remain below freezing, fecal DNA is expected to remain viable for several days, particularly for those pellets concealed in roost piles and protected from sun exposure and desiccation.

In addition to the collection and analysis of fecal pellets, feather samples from captured birds will also be collected and used as a source of DNA for individual identification. A related project currently involves the capture of male greater sage-grouse in the PPR and the attachment of 30 g rump-mounted, solar-powered GPS PTT satellite transmitters (Northstar Science and Technology, King George, VA). Up to 35 GPS transmitters per year will be attached to males during 2012 and 2013. We will capture and attach 22 g battery-powered VHF necklace collars equipped with mortality sensors (Advanced Telemetry Systems, Isanti, MN) to an equal number of hens (up to 35 per year) in 2012 and 2013.

Capture and marking of greater sage-grouse in the PPR serves two purposes. First, feathers collected during capture will be genotyped to identify (or “mark”) individual birds and the resulting capture data will constitute the first mark-recapture occasion. The addition of this initial capture occasion will increase precision of population abundance estimates generated from mark-recapture data based on the sampling of fecal pellets. Second, systematic monitoring of marked birds with radio-collars will allow us to assess the assumption of demographic and geographic closure of the population (i.e. no death or emigration) during sampling periods, a crucial assumption for the use of closed mark-recapture models.

Random transects will be generated using a spatially balanced (GRTS) sample design (e.g. Reversed Randomized Quadrant-Recursive Raster (RRQRR) (Theobald et al. 2007)). Spatially balanced samples allow for more complete coverage of the study area while increasing the probability of sampling clustered individuals, such as winter flocks of sage-grouse. Approximately 65% of the study area is privately owned (mostly by energy companies or private ranches). Spatially balanced sampling is likely to be advantageous as it will avoid clustering of random points in locations where it may be difficult to gain access for sampling. Spatially balanced sampling will also allow greater coverage of the study area and should aid in reducing heterogeneity in detection probability of individual sage-grouse. Stratification of random samples may be achieved using RRQRR with the incorporation of relative probability of use maps developed for greater sage-grouse in the PPR (Walker 2010). The number of random plots to be sampled will ultimately be constrained by funding and logistics.

Sampling plots will be surveyed for roost piles or other evidence of grouse, particularly tracks in the snow. Fecal piles identified at roost sites or along tracks will be sampled with roost sites being the preferred source for pellet sampling whenever they are available. Roost sites or pellets encountered outside of the random sampling scheme will also be included as incidental sampling locations. When tracks from flocks or individual birds are encountered, they will be followed in an attempt to locate a roost site for sampling. If a roost piles are unavailable, pellets will be sampled along the tracks.

The location of each roost pile will be clearly marked by a staked flag to facilitate sampling. Following an initial search, a 30-meter buffer surrounding roost piles will be searched for additional piles and the buffer reset around the new location until no additional piles are identified within the buffer. This search strategy was developed during the 2011 pilot work conducted at roost site locations near Hiawatha, CO and was designed to maximize pellet detection. At each roost pile, a total of four to five pellets will be collected with a focus on pellets in the best condition (i.e. least exposed or least desiccated). Caecal piles will not be sampled and pellets having contact with caecum will be avoided. When sampling pellets from tracks, the number of birds present in the flock will be counted and an attempt made to collect several pellets from each individual.

Pellet samples will be placed in sterile Whirlpak® bags with a single FTA® desiccant pouch, sealed, labeled for individual identification and stored on ice or snow until they can be transferred to a -20°F non-frost-free freezer at the Little Hills SWA bunkhouse). Pellets will be later transported on dry ice to the USGS Fort Collins Science Center Molecular Ecology Laboratory for DNA extraction.

Capture, Handling, Transmitter Attachment, and Feather Sampling

We plan to capture adult and yearling female greater sage-grouse in July-November in each year. All captured females will be sexed, aged, weighed, and fitted with individually-numbered, aluminum leg bands (size 20) and will have a 22-g, necklace-style, battery-powered VHF transmitter attached (Model A4060, Advanced Telemetry Systems, Isanti, MN). VHF transmitters have a 4-hour mortality switch, a guaranteed life of 15 months, and a range of several miles both from the ground and from the air (depending on terrain and radio age). Transmitters from birds that die may be recovered, cleaned, refurbished and redeployed as necessary to maintain sample sizes. Crews will capture females using CODA net launchers (Giesen et al. 1982), night-time spotlighting and hoop-netting (Wakkinen et al. 1992), walk-in traps (Schroeder and Braun 1991) modified for sage-grouse, or Super Talon net guns (Advanced Weapons Technology, La Quinta, CA), all of which have been approved for capture in this population. The trapping effort will be distributed across the population so that it is proportional to and representative of the amount of local breeding habitat present as identified in seasonal habitat models (Walker 2010). Otherwise, capture and handling methods will follow standard CPW operating procedures established for sage-grouse. The decision whether injured birds will either be released or euthanized will be made by the PI rather than transporting birds back to Fort Collins. No known rehabilitators in western Colorado currently have the facilities to care for wild, injured sage-grouse.

Feather samples will be collected from each captured bird following modified protocols based on those previously used by the USGS Fort Collins Science Center Molecular Ecology Laboratory (MEL) for collection of Gunnison sage-grouse feathers for DNA analysis.

Sample Analysis

DNA extraction and microsatellite analysis will be conducted using protocols developed by the MEL and demonstrated to be reliable for genotyping DNA from fecal pellets of the Gunnison sage-grouse (*Centrocercus minimus*) (Oyler-McCance and St. John, unpublished report). Protocols used for genotyping Gunnison sage-grouse from fecal pellets will be equivalent to those used for greater sage-grouse. DNA extraction will be performed using the QIAmp DNA stool mini kit (Qiagen, Germantown, MD) following protocols for "Isolation of DNA from stool for human DNA analysis" with a slight modification that decreases the final elution volume to 60 ul. Polymerase Chain Reaction (PCR) for amplification of DNA will be performed using a 2-step, pre-amplification method (Piggot et al. 2004) based on primer recipes and thermal profiles currently used by the MEL for genotyping from fecal samples of sage-grouse (pers. comm. Sara Oyler-McCance and Jennifer Fike, USGS). Microsatellite analysis will focus on loci previously identified by the USGS laboratory as reliable for use in identifying individual sage-grouse. Genetic analysis of feather samples will be performed using similar methods developed by the MEL for use in genotyping individual sage-grouse.

A major challenge for researchers conducting genetic mark-recapture studies using non-invasive samples such as feces is the potential for genotyping error. Non-invasively collected samples are often characterized by low quantity or quality of DNA (Broquet et al. 2007), which may be highly variable among samples (Miquel et al. 2006). Problems facing analysis of these samples include amplification failure, allelic dropout and mutation during amplification (Lukacs and Burnham 2005a), each of which may result in genotyping error and violation of a critical assumption of closed models that "marks" are correctly identified and recorded. Lukacs and Burnham (2005b) showed that genotyping error may result in biased abundance estimates from closed mark-recapture models.

To address these concerns, actions will be taken in the field and laboratory to reduce genotyping error and estimate the rate at which it occurs. Throughout this project, special care in the collection and storage of samples will be taken to prevent contamination and maintain sample integrity. In the laboratory, genotyping error rates can be greatly decreased or eliminated with proper training of personnel, careful protocols, the systemization and automation of methods and the use of a reliable set of microsatellite loci (Paetkau 2003). In addition to these measures, each pellet sample will be analyzed twice to monitor for and estimate rates of genotyping error. Sample pairs that fail to match (indicating that potential genotyping error has occurred) will be resampled. Additionally, occasional inclusion of blind duplicate samples will be employed to validate the accuracy of laboratory methods.

Data Analysis

All greater sage-grouse in the study population possess a unique genetic fingerprint and are therefore inherently marked. DNA from fecal pellets will be genotyped, referenced to unique individuals in the population and used to generate encounter histories for those individuals. Encounter history data will be analyzed using closed mark-recapture models in program MARK (White and Burnham 1999). Analysis using closed capture models requires that fecal pellets be sampled across several unique temporal occasions and that closed model assumptions (i.e. demographic and geographic closure, no mark loss or misidentification) are satisfied.

Power Analysis

Simulations were performed in program MARK, using the *Closed Captures* and *Full Closed Captures with Heterogeneity* data types to estimate the sampling effort required to achieve acceptable levels of precision and bias in abundance estimates. Simulations of 500 repetitions were run using a range of probable values for true population size (N), detection probability (p), heterogeneity mixtures (p_i), and the number of sampling occasions. Results from these simulations indicate that, in the absence of individual capture heterogeneity, at least 10% of individuals (p=0.1) in the population should be encountered during each sampling occasion for a minimum of four to five occasions to obtain abundance estimates with acceptable accuracy (i.e. CV < 0.15 and 95% coverage of N). These results were used, in combination with current expectations for population size and considerations for sampling with replacement, to compute cost estimates for the project reported in the budget section of this proposal.

Simulations also indicated that heterogeneity in detection probabilities may make it difficult to obtain unbiased estimates of abundance when using closed mark-recapture models. As a result, emphasis will be placed on sampling strategies that reduce heterogeneity in encountering pellets of individual sage grouse. These strategies include the use of spatially balanced random sampling (RRQRR) to improve sampling coverage of the study area and generation of a unique set of random sampling transects for each occasion to reduce the chance of repeatedly encountering individual birds with fidelity to certain locations.

Sex Ratio

Sex ratios of greater sage-grouse populations have been estimated in several states. However, the majority of data used for these estimates were obtained from hunter harvest efforts such as wing-barrel programs (Connelly et al. 2011) which may have bias due to hunter behavior or preference. Sex ratio of the PPR greater sage-grouse population will be estimated using genetic samples from the genetic mark-recapture component of this project that allows us to determine sex of individual birds. Sex ratio will be estimated using data from two sources of DNA (fecal pellets and/or feathers) collected during sampling efforts for Objective 2. The genetic data obtained from this project will also provide an opportunity to investigate variation in sex composition of greater sage-grouse winter flocks.

Method Comparison

Following the conclusion of the sampling periods and data analysis, a comparison of key population estimation methods investigated in this study will be conducted. Population size estimates from dual-frame sampling, genetic mark-recapture and standard state lek count techniques will be compared. Factors, including variance of population size estimates, cost, practicality of methods, and disturbance to birds associated with each method will be evaluated and recommendations made regarding continued monitoring of the species, both in the PPR and range-wide.

Additionally, we will discuss the efficacy and potential consequences of employing these methods to estimate greater sage-grouse population size and/or determine trends in population size. We will also discuss opportunities for the improvement of lek count-based population estimations through the use of supplemental population information. Sex ratio, inter-lek movements of male sage-grouse and the proportion of known versus unknown leks will be estimated by this project. Related research being conducted in the Piceance by Dr. Brett Walker will additionally provide estimates of male lek attendance rates and detectability. Combined efforts from the two studies will generate estimated values the five unknown variables which are lacking, or assumed, in traditional state lek count estimations.

RESULTS AND DISCUSSION

Dual-frame Sampling

We surveyed 39 list-frame and 98 area-frame cells for greater sage-grouse leks from helicopters (Bell 47 Soloy and Bell 206-B3) from one-half hour before sunrise to two hours after sunrise three times each from April 22 to May 2, 2012 on three rounds of five flights (15 flights total). We confirmed active leks in 19 of 39 list-frame cells (including three new lek locations) and we found a new lek in one area-frame cell. Three leks were also found incidentally in transit between cells for a total of seven new leks identified during dual-frame sampling flights.

Genetic Mark-recapture

Field crews captured 25 male greater sage-grouse (25 yearlings or adults and one juvenile) in the study area from February through August 2012 as part of Dr. Brett Walker's GPS male project. Field crews also captured 30 VHF collared adult/yearling females and 10 juvenile females (banded and released) from July through September 2012. Fall-winter pellet transects are scheduled to begin in November 2012.

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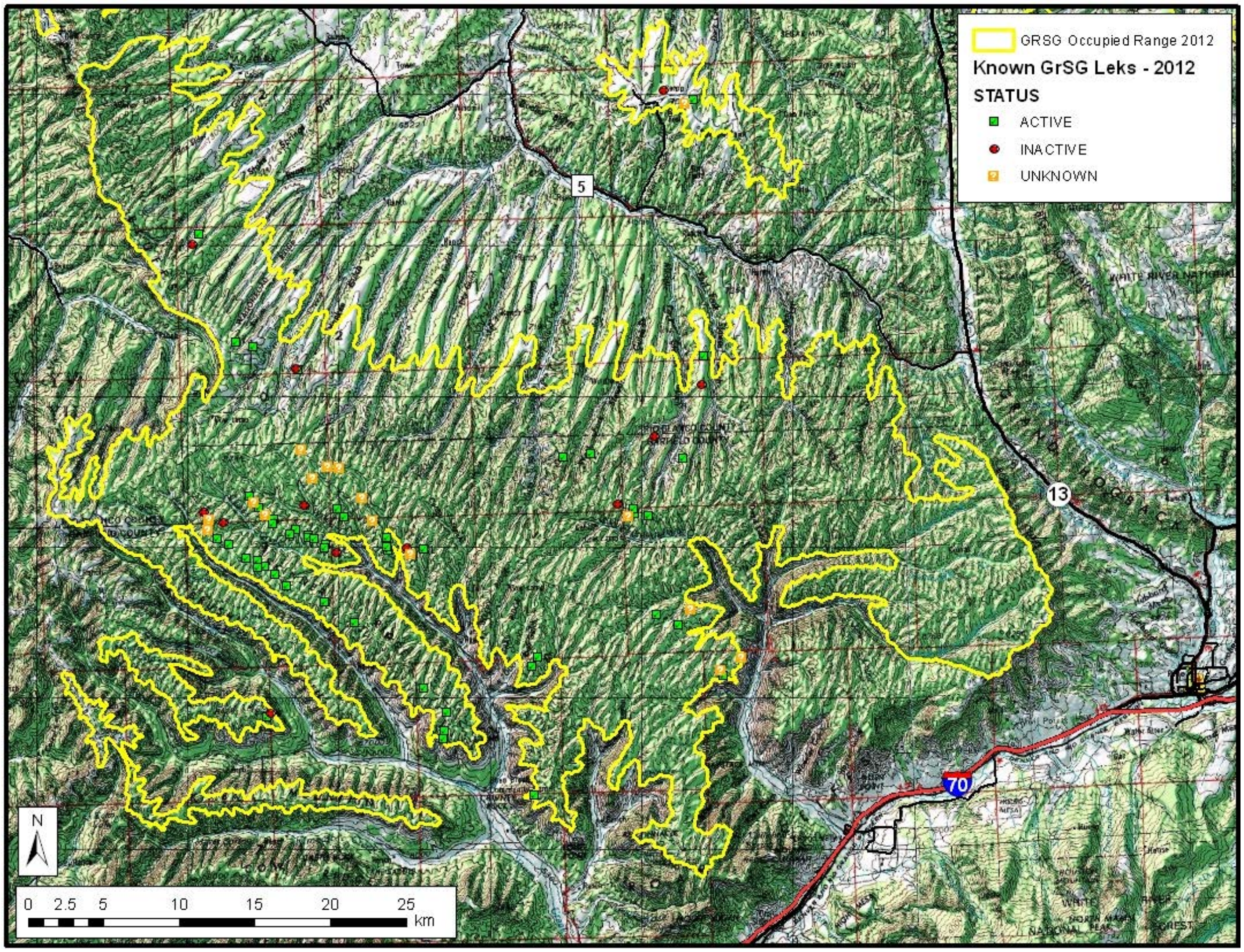

Figure 1. The Parachute-Piceance-Roan population study area showing known active greater sage-grouse leks (within the past five years), inactive leks, newly discovered leks (i.e., "unknown" status), and CPW's current occupied range boundary (as of February 2012). Newly discovered leks are assigned "unknown" status in CPW's state-wide database until they are confirmed active again in a subsequent year. Fifteen leks with strutting males were discovered in spring 2012 either during standard CPW lek-count flights by field crews on the ground, or on dual-frame sampling flights. Seven of the 15 newly discovered leks were found during dual-frame sampling flights. Fourteen of the 15 leks will be included in the list frame for sampling in spring 2013 (excluding the new lek in the Magnolia portion of the field).

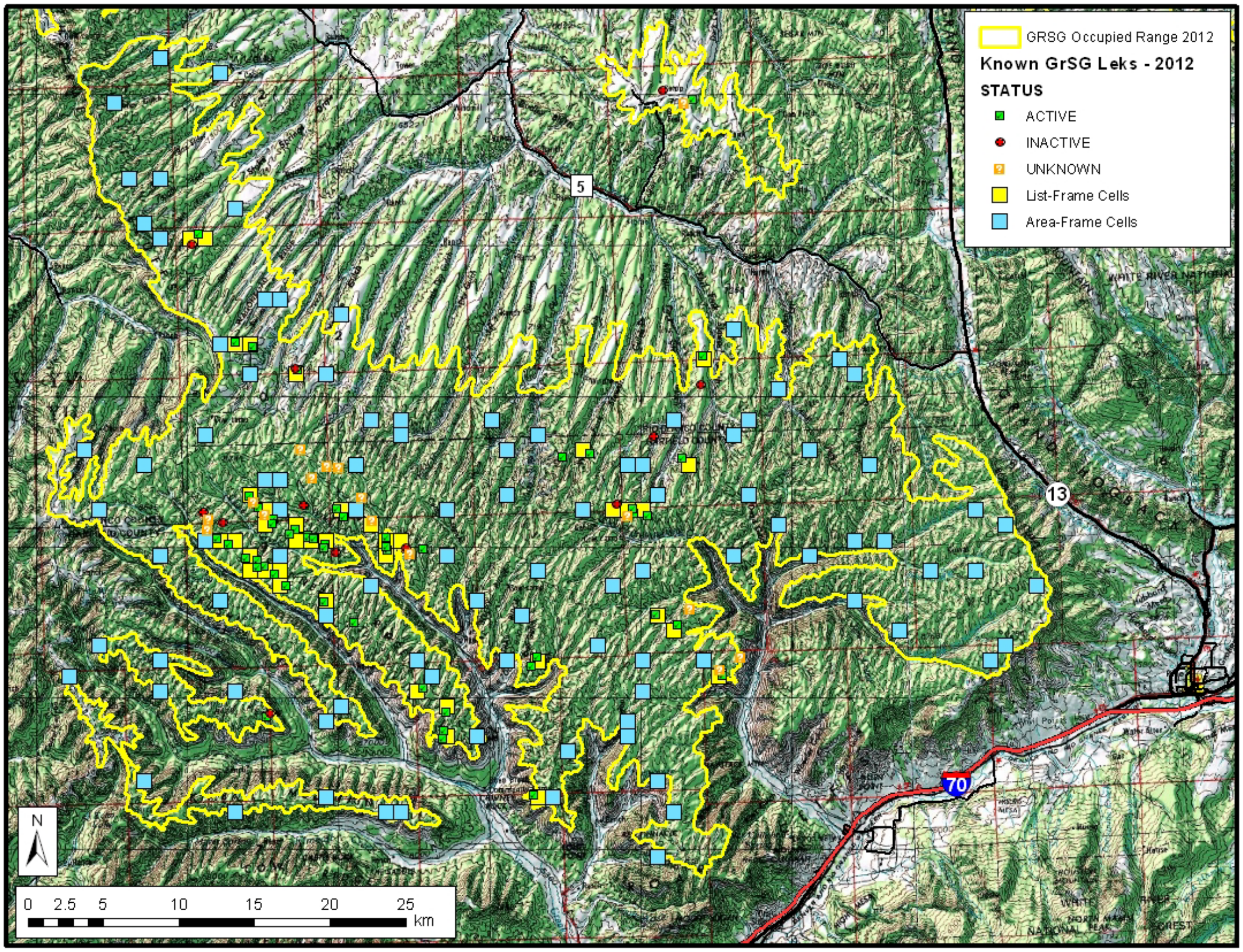

Figure 2. Parachute-Piceance-Roan study area showing list-frame (yellow squares) and area-frame (blue squares) 1-km^2 cells used in dual-frame sampling, April-May 2012.

Figure 3. Greater sage-grouse roost location in the snow with a roost pile and cecal droppings.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3420	:	Avian Research
Work Package:	0663	:	Terrestrial Species Conservation
Task No.:	N/A	:	Restoring energy fields for wildlife
Federal Aid Project No.	N/A		

Period Covered: September 1, 2011 – August 31, 2012

Author: D. B. Johnston

Personnel: B. deVergie, J.T. Romatzke, J.C. Rivale, and R. Velarde

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ABSTRACT

Restoring disturbed areas to wildlife habitat requires re-establishing a diverse mixture of perennial grasses, forbs and shrubs. Achieving this goal in Colorado oil and gas fields is often difficult because of the variety of impacted ecological zones and the threat of weed invasion. An area of particular concern is the Piceance Basin gas field because of its value to mule deer, sage-grouse and other wildlife. At elevations less than ~ 2100 m (7,000 ft), cheatgrass (*Bromus tectorum*) presents a major obstacle to reclamation. At higher elevations, reclamation is easier to achieve, but we lack reliable methods for restoring broadleaf forbs and shrubs. At elevation near 2100 m, the choice between minimizing the threat of weed invasion and maximizing the potential for plant community diversity can be difficult to make. In order to test techniques over their full range of potential usefulness, a series of five experiments was implemented in 2008 and 2009 on simulated well pads and pipelines that covered (?represented?) the wide range of precipitation and ecological conditions represented in the Piceance Basin gas field.

The Pipeline experiment began in 2008 on simulated pipeline disturbances at six lower elevation locations. It compares two approaches to controlling cheatgrass and promoting native plants: applying ™ herbicide (ammonium salt of imazapic, BASF corporation, *hereafter* Plateau) at 105 g ai/ha (6 oz/ac) just prior to seeding, and using soil tillage. The tillage treatments examined were disking, rolling, disking with rolling, and vibratory drum rolling. The tillage treatments were of interest because cheatgrass has been shown to be sensitive to seed burial and soil compaction. Vegetation response was quantified by assessing seedling density in 2009 and percent cover in 2010 and 2011. Three years post-application, Plateau plots had a seven-fold higher shrub cover, over two-fold lower cheatgrass cover, with similar perennial grass and forb cover compared to non-Plateau plots (no Plateau). Disking reduced initial cheatgrass density, but by the end of the experiment had little effect on cheatgrass cover. Disking slightly improved perennial grass cover, however other tillage treatments were ineffective. Initial cheatgrass density was greatly impacted by the pipeline disturbances, regardless of treatment. This is attributed to the timing of the disturbance, which maximized cheatgrass seed burial.

The Competition experiment began in 2009 on simulated well pad disturbances at two middle elevation sites. The goal of the Competition experiment is to examine novel factors which may affect the

competitive ability of native wheatgrasses versus cheatgrass. The density of both wheatgrass and cheatgrass seed was controlled. The treatments were: addition of a super-absorbent polymer called Luquasorb® (BASF Corporation); addition of a soil binding agent called DirtGlue® (DirtGlue® Enterprises); and rolling with a heavy lawn roller. In 2011, vegetation response was quantified by percent cover. Both super-absorbent polymer and binding agent reduced cheatgrass cover by almost half, although neither treatment was effective in the presence of the other. The super-absorbent polymer was effective where applied at 31 g/m^2 but ineffective where applied at 7 g/m^2. No treatments impacted 2011 perennial grass density.

The Gulley experiment began in 2009 on simulated well pad disturbances at four low elevation locations with very weedy surrounding landscapes. The Gulley experiment focuses on identifying which potential sources of weeds are important to control: those which originate from within the soil seed bank of the reclamation area, those which enter from the surrounding landscape, or both. The treatments were application of herbicide at 140 g ai/ha (8 oz/ac) just prior to seeding, fallowing for one year with the broad-spectrum pre-emergent herbicide Pendulum™ (pendamethilin, BASF Corporation), and surrounding plots with seed dispersal barriers composed of aluminum window screen secured to oak stakes. Unfallowed plots were seeded in 2009 and fallowed plots were seeded in 2010. In 2011 vegetation was assessed by percent cover. Fallowed plots had drastically lower perennial grass, shrub and cheatgrass cover, 60% lower forb cover, and higher annual forb cover than the unfallowed plots. Some of these differences can be attributed to the one-year lag in seeding time between fallowed and unfallowed plots, an effect which is expected to lessen in future years. Plateau application reduced perennial grass cover at one of four sites, increased shrub cover at three of four sites, reduced perennial forb cover at two of four sites, and successfully controlled cheatgrass. Barriers increased perennial grass cover by 20% across sites and reduced annual forb cover by about half in fallowed plots without Plateau at one site.

The Mountain Top experiment began in 2009 at four high elevation sites surrounded by desirable mixtures of grasses, forbs, and shrubs. In such situations, the best reclamation outcome would be to re-create the surrounding plant community. The Mountain Top experiment examines the degree to which current seeding practices help or hinder this outcome. Plots left unseeded are compared to plots seeded with a mixture containing a typical density of rhizomatous grass seeds, and these treatments are crossed with treatments designed to create favorable microsites for germination: a rough soil surface treatment consisting of mounds and holes, and a brush mulch treatment. In 2011 vegetation response was quantified by percent cover. Seeding increased perennial grass cover up to 10-fold, increased perennial forb cover up to four-fold, reduced annual forb cover up to 4-fold, and reduced shrub cover by 50%. The rough surface treatment increased perennial forb cover by 50% across sites when seeded, and controlled annual grasses at the one site with sufficient annual grass to permit analysis. The brush treatment increased perennial grass cover at two of four sites when seeded, and increased shrub cover by 70% across sites in flat surface plots.

The Strategy Choice experiment was implemented in 2009 on simulated well pad disturbances at 4 middle elevation sites with surrounding plant communities that contained both desirable and undesirable species. At sites such as these, the degree of threat from invasive weeds is often unclear. The Strategy Choice experiment combines some elements of the experiments conducted at lower and higher elevations in order to improve our understanding of optimal reclamation strategies. The treatments were: Plateau herbicide applied just prior to seeding at 140 g ai/ha (8 oz./ac), a rough soil surface with brush mulch versus a flat soil surface with straw mulch, and a high competition seed mix, including a typical density of rhizomatous grass seed, versus a low competition seed mix focused on forbs. Vegetation response in 2011 was assessed by percent cover. Plateau treatment successfully controlled annual grasses, but at a high price: perennial grass cover was four times lower, perennial forb cover was lower at one of two sites, shrub cover was almost three times lower, and annual forbs were two times higher in Plateau plots versus no Plateau plots. The rough surface treatment increased perennial grass cover at two of four sites, and brought about an over seven-fold decrease in annual grass cover in plots without Plateau at one site. The rough surface treatment also slightly reduced shrub cover. The seed mix focused on forbs produced 60% higher forb cover than the typical seed mix, with similar cover of weeds between seed mixes.

Results of Plateau application are mixed, generating beneficial results in one experiment, mixed results in another experiment, and largely detrimental results a third experiment. Successful use of this herbicide requires accurately applying a light rate and focusing on areas with cheatgrass cover prior to disturbance. In areas where cheatgrass is a threat, but is not evident prior to disturbance, using a roughened soil surface may provide adequate cheatgrass control, as was shown at two sites in two different experiments in this study. The rough soil surface may be effective because when cheatgrass seeds are few in number, they may be trapped within holes, where they experience a wetter microclimate in which they are less competitive. Both the super-absorbent polymer and the binding agent provided some cheatgrass control in 2011, although the super-absorbent polymer has performed more consistently and is less expensive to apply. The seed mix lacking rhizomatous grass seed performed well, producing high forb cover without allowing weed infestation. In the experiment where unseeded and seeded plots were compared, the expected initial result of higher annual forbs and lower perennial grasses and forbs in unseeded plots was found. However, unseeded plots had higher shrub cover than seeded plots, which may ultimately produce a more desirable plant community for wildlife.

Throughout the elevation range of this suite of experiments, treatments were found which improved the quality of post-reclamation wildlife habitat. Excellent recovery of wildlife habitat value should be the goal for (?repairing? restoring?) oil and gas disturbances. The Competition, Gulley, Mountaintop, and Strategy Choice experiments will continue to be monitored in 2012.

WILDLIFE RESEARCH REPORT

RESTORING ENERGY FIELDS FOR WILDLIFE

DANIELLE B. JOHNSTON

PROJECT OBJECTIVES

1. Develop reclamation techniques for big sagebrush (*Artemisia tridentata*) habitats impacted by oil and gas development in northwestern Colorado. Maximize wildlife habitat quality by promoting native, perennial plant communities containing a mixture of grasses, forbs and shrubs.
2. Determine which weed control techniques are effective in reclamation. Test techniques such as application of a selective herbicide, fallowing with a broad-spectrum herbicide, manipulation of soil density, and creation of barriers to weed seed dispersal. Determine where and how these weed control techniques should be applied.
3. Determine which techniques are effective at promoting plant community diversity in reclamation. Test techniques such as use of a low competition/high diversity seed mix, creation of a rough soil surface and use of brush mulch. Determine where and how these techniques should be applied.

SEGMENT OBJECTIVES

This project consists of five separate experiments with different objectives for this reporting year:

1. *Pipeline Experiment:* Assess vegetation response for three years following herbicide and tillage treatments by measuring plant cover in 10 plots at each of six research sites. Synthesize results over three years and prepare final report for publication.
2. *Competition Experiment:* Assess vegetation two years following soil additive and compaction treatments by measuring plant cover in 60 plots at each of two research sites. Assess soil moisture once in all plots.
3. *Gulley Experiment:* Assess vegetation one year post-implementation of a fallowing treatment, and two years post-treatment in non-fallowed plots. Measure plant cover in 24 plots at each of four research sites.
4. *Mountain Top Experiment:* Assess vegetation two years following seeding, soil surface roughening, and brush mulch treatments by measuring plant cover in 24 plots at each of four research sites.
5. *Strategy Choice Experiment:* Assess vegetation two years following herbicide, soil surface roughening, and seed mix treatments by measuring plant cover in 12-24 plots at each of four research sites.

INTRODUCTION

Preserving quality wildlife habitat in oil and gas fields requires effective restoration of impacted areas. Successful restoration entails preventing soil loss, overcoming the threat of weed invasion, and promoting natural plant successional processes so that a diverse mixture of perennial grasses, forbs and shrubs are established. A detailed knowledge of soils, climate, topography, land use history, and plant competition is needed to accomplish this goal, and optimal choices of reclamation techniques are site-specific. The need for site-specific knowledge often prompts local reclamation trials by organizations, such as coal mining companies, which cause large-scale disturbances. In oil and gas fields, however, local reclamation trials are difficult to implement due to the spatial pattern of disturbance.

In contrast to coal mines, which typically result in a small number of large disturbances, oil and gas fields result in a large number of smaller disturbances, each connected by a web of pipelines and access roads which may extend across hundreds of thousands of acres. The complexities of gathering knowledge at the appropriate scales, administering recommendations for the multitude of sites involved, and enforcing appropriate standards over such large areas often results in reclamation that falls short of the most basic standards (Avis 1997, Pilkington and Redente 2006).

Addressing these challenges is imperative, as the fragmented pattern of development means that wildlife and wildlife habitat are affected over a much larger area than that directly occupied by development activities. For instance, habitat use by greater sage-grouse (*Centrocercus urophasianus*) populations and mule deer (*Odocoileus hemionus*) may decline within large buffer areas surrounding development (Sawyer et al. 2006, Walker et al. 2007). Furthermore, the establishment of non-native species due to development (Bergquist et al. 2007) could reduce wildlife habitat quality over large areas if disturbances are allowed to provide vectors for weed invasion into otherwise undisturbed habitat (Trammell and Butler 1995). Because of this threat, preventing weed invasion through successful restoration of all impacted areas is a top management priority for wildlife. The goal of this study is to promote such restoration by replicating tests of promising techniques at the scale of an oil field.

The Piceance Basin in northwestern Colorado provides an ideal laboratory for conducting a large-scale study of restoration techniques. The area is currently experiencing an unprecedented level of natural gas development, it provides critical habitat for the largest migratory mule deer herd in the United States, and it has a complex topography which ensures that a wide range of precipitation, soil development, and plant community types are represented.

Because elevation is an important driver of precipitation, plant community composition and weed prevalence in the area, experiments were assigned according to elevation zone. Twelve study sites, ranging in elevation from 1561 to 2676 m, house five experiments, each repeated at two to six sites. Each experiment tests three to six treatments, and some treatments are tested in multiple experiments. Overlap of treatments allows the experiments to relate to one another in a way that will permit broad-scale conclusions, if appropriate, while the differences in the experiments permit tailoring of particular treatments to those portions of the landscape where they are potentially useful.

The three experiments conducted at lower elevations emphasize weed control, particularly that of cheatgrass, which presents a serious obstacle to effective reclamation in the study area (Pilkington and Redente 2006). The three lower elevation experiments are the Pipeline experiment (implemented at six sites ranging from 1561 to 2216 m in elevation), the Competition experiment (implemented at two sites of elevations 2004 and 2216 m), and the Gulley experiment (implemented at four sites ranging from 1561 to 2084 m in elevation). The remaining two experiments, conducted at high or middle elevations, emphasized maximizing plant community diversity. The Mountain Top experiment was implemented at the four highest elevation sites, ranging from 2342 to 2676 m. The Strategy Choice experiment was implemented at four moderate elevation sites ranging from 1662 to 2216 m.

The Pipeline experiment evaluates the effectiveness of tillage treatments versus an herbicide treatment at controlling cheatgrass and promoting establishment of a diverse, predominately perennial, native plant community. Oil and gas disturbances are amenable to tillage manipulations, as the ground is already disturbed and access routes for heavy equipment have already been created. In agricultural settings, combining lower levels of herbicide with tillage treatments, such as disk cultivation, has proven effective for controlling weeds (Mulugeta and Stoltenberg 1997, Mohler et al. 2006). Soil manipulations may be particularly effective for controlling cheatgrass because cheatgrass is sensitive to seed burial (Wicks 1997), does not germinate well in even slightly compacted soil surfaces (Thill et al. 1979), and is less competitive in denser soils (Kyle et al. 2007). Tillage manipulations examined include disking, rolling with a static roller, rolling with a vibratory drum roller, or disking plus compaction with a static roller. The herbicide investigated is Plateau ™ (ammonium salt of imazapic, BASF Corporation, Research Triangle Park, NC, *hereafter* Plateau), as it has been shown to reduce cheatgrass with little effect on some perennial grasses (Kyser et al. 2007). However, it also reduces the vigor and density of established forbs (Baker et al. 2007), and little is known about its effect on germination of desirable species.

The Competition experiment also examines compaction by rolling, but does so in conjunction with soil additives, in an environment where the density of cheatgrass seeds is controlled. Earlier work has shown that the density of weed seeds, or propagule pressure, has a large influence on the likelihood that a weed will become dominant when an ecosystem is disturbed (Thomsen et al. 2006). Therefore, variation in propagule pressure can confound attempts to study which reclamation techniques promote desirable species, particularly if the effects are subtle. Cheatgrass propagule pressure was controlled in the Competition experiment by adding a known quantity of cheatgrass seeds to areas that were previously free of cheatgrass, (and then surrounding the research area by physical and chemical barriers to prevent cheatgrass from leaving the area). The first soil additive examined is a super-absorbent polymer called Luquasorb® (cross-linked copolymer of Potassium acrylate and acrylic acid in granulated form, BASF Corporation, Ludwigshafen, Germany). When added to degraded soils, super-absorbent polymers absorb and then gradually release water, reducing the effects of water stress (Huttermann et al. 2009). This may hinder cheatgrass, as cheatgrass has been shown to be a more effective invader when soil moisture is more variable (Chambers et al. 2007). The second soil additive examined is a soil binding agent called DirtGlue® (DirtGlue® Enterprises, Amesbury, MA). Soil binding agents are commonly used to stabilize soil and facilitate binding of seed to the soil surface, but their effect on competitive interactions is unknown. DirtGlue® is used in this study because of its purported ability to bind soil particles while increasing water infiltration. The combination of using a soil binding agent with rolling was of interest because of the potential for creating a crust that might hinder cheatgrass emergence.

The Gulley experiment focuses on identifying which potential sources of weeds are important to control: those which originate from within the soil seed bank of the reclamation area, those which enter from the surrounding landscape, or both. Like the Pipeline experiment, the Gulley experiment includes a test of Plateau herbicide as a strategy to control certain species in the soil seed bank. A second herbicide is also tested: Pendulum® AquaCap™ (pendimethalin, BASF Corporation, Research Triangle Park, NC; *hereafter* Pendulum). Pendulum is a broad-spectrum pre-emergent herbicide, is effective for about six months, and is a drastic measure designed to eliminate as much of the existing seed bank as possible. To control seeds originating from areas surrounding the reclamation area, seed dispersal barriers were constructed of aluminum window screen, using a design that had been effective in a Utah seed bank study (Smith et al. 2008). This is of interest because a recent Colorado Parks and Wildlife (CPW) study demonstrated that a sufficient number of cheatgrass seeds may disperse from the edges of disturbance to compromise reclamation efforts (Johnston 2011a).

The Mountain Top experiment sites were surrounded by perennial, predominately native plant communities (Table 1); therefore weed control was not a great concern. At sites such as these, the goal of reclamation should be to re-create the desirable mixture of grasses, forbs and shrubs found in the undisturbed habitat. However, prior studies have shown that even after decades of recovery, reclamation areas may remain dominated by grasses (Newman and Redente 2001). Explanations for grass dominance include a loss of variability in soil resources when topsoil is redistributed, and a disproportionate influence of the grasses included in the reclamation seed mix (Redente et al. 1984). Creating treatments which re-establish resource heterogeneity, encourage native seed dispersal and avoid undue competition from seeded grasses may result in a plant community which better serves the needs of wildlife. In this study, we examine three such treatments: creating a rough soil surface of mounds and holes, spreading brush mulch, and foregoing seeding. A rough soil surface may be helpful because it creates variability in soil depth, creates microsites of higher moisture availability, and traps dispersing seeds (Chambers 2000). Similarly, brush mulch creates favorable germination conditions by causing snow to drift, creating shade, entrapping seeds (Kelrick 1991), and perhaps also providing a source of seed. These two treatments are applied with and without seeding in order to address the question: If the adjacent undisturbed area is desirable, how important is seeding versus creating soil heterogeneity and encouraging natural seed dispersal in order to establish a diverse plant community?

The Strategy Choice experiment was conducted at middle-elevation sites where the degree of threat from invasive weeds is ambiguous. Such situations raise the question: should one take a conservative strategy by seeding a highly competitive seed mix, using aggressive weed control measures, and avoiding contaminating the site with seed from the surrounding area? Such measures often come at a price of reduced plant diversity and forb establishment (Marlette and Anderson 1986, Chambers 2000, Krzic et al. 2000, Baker et al. 2007). Therefore, one might wish to adopt an optimistic strategy by seeding a low competition/high diversity seed mix with a minimal fraction of rhizomatous grasses, avoiding the use of herbicide, and entrapping seeds via brush mulch, holes, or other mechanisms. An optimistic strategy is the obvious choice when the surrounding plant community is desirable, and the risks of soil erosion and weed invasion are low. This study compares the results of these two strategies in situations where the risk of weed invasion is moderate and the surrounding plant community contains both desirable and undesirable species. The treatments examined include use of Plateau, creation of a rough soil surface with holes and brush mulch, and comparison of a high competition versus low competition/high diversity seed mix.

In all experiments, establishment of native, perennial plants was emphasized. Perennial plants are critical for wildlife because they provide nutritious forage for a longer portion of the growing season, their overall productivity is higher, and their productivity is less variable from year to year than that of annual plants (DiTomaso 2000). The experiments focus on large sagebrush communities, because of the need for better techniques for re-establishing these communities (Lysne 2005), their widespread distribution and their importance to wildlife (Davies et al. 2011).

STUDY AREA

The Piceance Basin study area is in Rio Blanco and Garfield Counties, Colorado (Figure 1). Elevation increases gradually from north to south as one travels from Piceance Creek (~1,800 m) to the top of the Roan Plateau (~2,500 m), then drops off sharply at the Book Cliffs to the Colorado River Valley (~1,500 m). Precipitation and temperature vary across the region with both elevation and latitude; more northerly sites are colder and receive less precipitation than southerly sites of similar elevation. Northernmost sites receive approximately 280 mm per year, 40% as snow. The southerly Colorado River Valley sites receive approximately 340 mm of precipitation per year, 25% as snow. The wettest, highest elevation sites are at the southern edge of the Roan Plateau, and receive approximately 500 mm per year, 60% as snow. Lower elevations are characterized by Wyoming big sagebrush, cheatgrass, Indian ricegrass (*Achnatherum hymenoides*), western wheatgrass (*Pascopyron smithii*), prairie junegrass (*Koeleria macrantha*), and globemallow (*Sphaeralcea coccinea*) in flatter areas with a mixture of pinyon pine (*Pinus* sp.), and Utah juniper (*Juniperous utahensis*) on steeper slopes and greasewood (*Sarcobatus vermiculatus*) in floodplains. Higher elevations are characterized by mountain big sagebrush, mountain brome (*Bromus marginatus*) and diverse forbs in flatter areas, serviceberry (*Amelanchier alnifolia*), snowberry (*Symphoricarpos rotundifolius*), Gambel's oak (*Quercus gambelii*) on slopes, and aspen (*Populus tremuloides*) mixed with Engleman spruce (*Picea engelmanii*) in the highest elevation, north-facing slopes.

Twelve research locations were chosen within the Piceance Basin in sagebrush habitats (Figure 1, Table 1). These 12 locations span most of the range of elevation, soil type and precipitation to be found in the area. The lowest elevation site, SK Holdings (SKH) lies at 1561 m (5,120 ft.), has alkaline, clayey soils and is characterized by high cheatgrass cover with interspersed Basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*). The highest elevation site, Square S (SQS), lies at 2676 m (8,777 ft.), has a sandy loam soil, and has a mixture of non-noxious forb, grass and mountain big sagebrush cover.

METHODS: DISTURBANCE CREATION

Two types of disturbances, simulated pipelines and simulated well pads, were created in order to provide templates for the experiments. Pipeline disturbances measured 11 m by 52 m and were simulated using a bulldozer and a backhoe. Vegetation was scraped and discarded, the top 20 cm of topsoil was scraped and stockpiled, and then a 1 m wide 1 m deep trench was dug. Trenches were left open for three weeks, and then the subsoil was replaced and the topsoil was spread evenly over the site. This work was completed in six locations in August and September of 2008. The Pipeline experiment was immediately implemented on these disturbances.

Well pads differ from pipelines in the length of time topsoil is stockpiled and in the degree to which subsoil disturbance occurs. Well pad disturbances measured 31 m by 52 m and were simulated using a bulldozer. Vegetation was cleared, the top 20 cm of topsoil was scraped and stockpiled in windrows less than 2 m in height, and then the subsoil was cut and filled to create a level surface. The initial work was completed in July and August of 2008, and the surface was kept weed-free for one year by repeated hand-spraying of emerging plants with 2% (v/v) glyphosate. In August of 2009, the subsoil was recontoured to approximate the original contour, and the stockpiled topsoil spread evenly across the surface of the site. Simulated well pads were created in 12 locations, each with slopes of 5% or less. The Gulley, Strategy Choice, Competition, and Mountain Top experiments were implemented on the well pad disturbances in 2009 and 2010.

All sites were fenced with 2.4 m (8-ft.) fencing after experiments were implemented. This eliminated variability from site to site in the degree of browsing and grazing pressure from wildlife and livestock.

PIPELINE EXPERIMENT

Overview

- Goal: Compare effectiveness of Plateau herbicide and tillage treatments for controlling cheatgrass and promoting perennial plants.
- Conducted at six sites: YC1, YC2, RYG, WRR, GVM and SKH (Figure 1, Table 1).
- Treatments:
  - Herbicide (two levels): Plateau applied (Plateau) or no Plateau applied (*hereafter* no Plateau).
  - Tillage (five levels): disking (D), compaction with a static roller (R), compaction with a vibratory drum roller (V), disking plus compaction with a static roller (DR), or control (C).
- Design: Factorial split-plot. Herbicide treatments were randomly assigned to whole plots, and tillage treatments were randomly assigned to subplots (Figure 2).
- Plot size: 11 m X 10 m.
- Responses measured: seedling density (2009) and plant cover (2010 and 2011). As this experiment is now mature, all three years of post-treatment data are synthesized in this report.

METHODS

Tillage treatments were implemented shortly after pipeline disturbances were created in the fall of 2008. In C plots, bulldozer and backhoe tracks were left in place. The soil surface varied from smooth to very rough. D plots were disked to four inches. R plots were rolled once with a static roller supplying a linear load of 20.8 lbs/in (36.5 N/cm). V plots received four passes with a vibratory drum roller (Wacker DH-12). DR plots were disked to four inches, then wetted to 1 cm using an ATV tow sprayer, and then rolled five times with a non-vibratory roller. The DR treatment was an effort to create slight soil compaction at the surface, while avoiding heavy compaction of the rooting zone, which can restrict root growth and compromise establishment of deeply-rooted perennial plants (Thompson et al. 1987). At the Yellow Creek sites, the V treatment was not implemented due to access constraints.

Herbicide was applied in October 2008. At the time of application, cheatgrass was at the one-leaf stage (~5 cm tall) at WRR. At RYG, cheatgrass had just begun emerging at the Yellow Creek sites, and had not emerged at SKH or GVM. Plateau was applied at 105 g ai/ha (420 g/ac or 6 oz./ac) with glyphosate at 210 g ai/ha (8 oz. *Roundup Pro*/ac) and methylated seed oil (2% v/v) using an ATV tow sprayer (Agri-Fab 45-0424). The rate of Plateau application was a compromise between the 700 g/ac rate, which has been shown to provide good brome control at the expense of strong negative effects on native forbs (Baker et al. 2007), and the 280 g/ac rate, which has been shown to avoid serious negative effects on most desirable species but provides only moderate brome control (Bekedam 2004). Glyphosate was added because cheatgrass had emerged at some sites at the time of application, and methylated seed soil was added to facilitate bonding of the herbicide to leaf surfaces.

Following herbicide application, sites were drill seeded using a Tye Pasture Pleaser rangeland drill, calibrated to plant seed approximately 1 cm deep in tilled soil. Drill rows were about 25 cm apart, and the drill produced a minimal amount of soil disturbance. All sites received the same seed mixture (Table 2). Grasses and shadscale species were mixed together, as were all forb species. Grass/shadscale and forb mixtures were seeded in separate rows by taping poster board dividers in the seed box, and placing seed mixes in alternating divisions. Rice hulls were added at 50% v/v in order to keep seeds of different sizes suspended evenly in the mixtures (St. John et al. 2005). Wyoming big sagebrush seed collected from Dry Creek Basin in Colorado, an area with similar temperature and moisture characteristics to the study area, was broadcast seeded onto snow in mid-January. Plots were seeded at a rate of 8.6 pounds pure live seed per acre. This low seeding rate was chosen because lower seeding rates facilitate establishment of mixed stands (Redente et al. 1984).

Soil bulk density samples were used to compare sites and to compare on-disturbance vs. off-disturbance locations. Jornada cone penetrometer measurements (Herrick and Jones 2002) were used to quantify within-site differences between the tillage plots. Penetrometer measurements are much more easily obtained, but because penetration resistance depends on soil moisture, penetration resistance is poor choice for comparing differences between sites (Miller et al. 2001). Bulk density samples were taken in September of 2008 using a 30.5 cm drop-hammer double cylinder core sampler fitted with six abutting 5.1 cm-long inner cylinders. Five cores were taken in undisturbed areas near each site, and six cores, three in each C soil treatment subplot, were taken within pipeline disturbances. A piece of metal flashing was inserted between adjoining cylinders to separate each core into six depth fractions. The dry weight of each fraction was divided by its volume to find bulk density. Five penetrometer measurements were taken in each subplot in May of 2009. The number of hammer drops required to move the penetrometer through the soil was recorded for each 5 cm depth increment from 4 cm to 29 cm, and the force required to penetrate the soil was calculated for each depth fraction.

Seedling counts were conducted in May and July of 2009 for nine mini-plots within each subplot, with one mini-plot in the center of the subplot, and the remaining mini-plots equidistant from the center mini-plot and either a subplot corner or the midpoint of a subplot edge. Because seedling density varied widely from site to site, the size of the sampled mini-plot was allowed to vary from 300 to 3000 cm^2 so that an area sufficient for sampling was obtained. In May, only cheatgrass seedlings were counted. In July, all seedlings were counted. Cheatgrass seedlings were also counted in nine mini-plots in undisturbed vegetation near the study sites in May and July of 2009.

Percent cover by species was quantified in July of 2010 and 2011. In 2010, nine 1 m^2 mini-plots were arrayed systematically per subplot as described above. In 2011, five 1 m^2 mini-plots were sampled per subplot, one in the center of the subplot, and the remaining mini-plots equidistant from the center mini-plot and a subplot corner. A grid containing 36 intersections was held over each mini-plot, and point-intercept hits were measured at each grid intersection using a laser point-intercept sampling device (Synergy Resource Solutions, Bozeman MT). All layers of vegetation were identified to species at each hit. When calculating percent cover of a given functional group (perennial grasses, perennial forbs, annual forbs, cheatgrass, or shrubs), overlapping hits of different species within a functional group (for instance, western wheatgrass overlying Sandberg bluegrass) were counted as a single instance of the functional group.

Cheatgrass propagule pressure was quantified using techniques outlined in Appendix 1. Seeds caught per square meter per Julian date was calculated and then averaged over years for each site.

Analysis of variance (ANOVA) in SAS PROC MIXED (SAS Institute Inc., Cary, NC) was used to analyze differences in responses to treatments. Site was considered a random effect. For bulk density, separate analyses were done for each depth fraction, and the fixed effect was a location variable (on or off pipeline). For penetration resistance, separate analyses were done for each depth fraction, and the fixed effects were the soil tillage treatments. Density and cover data were analyzed separately by functional group (cheatgrass, annual forbs, perennial grasses, perennial forbs, and shrubs), and a site* (?*?)Plateau random effect was used to account for the split-plot design. Biennial forbs were lumped with annual forbs. Cover data was transformed by an arcsine [square root (x)] transformation to improve normality. For cover data, fixed effects were treatments and up to 3-way interactions among them. For cheatgrass seedling density and cover, a repeated measures ANOVA was performed, with season or year as a fixed effect, and up to 3-way interactions involving season or year included in the initial model. The final model was determined using a backwards selection process with a cutoff value of $\alpha = 0.05$ for means and $\alpha = 0.10$ for interactions. Effect sizes are presented with 95% confidence intervals. Linear regression was used to examine the effect of soil penetration resistance on cheatgrass seedling density and cover, using only non-disked plots without the Plateau treatment.

RESULTS

The creation of the simulated pipeline disturbances increased soil bulk density by 0.13 ± 0.05 g/cm^3. The increase in bulk density was evident at all depth fractions except the 5-10 cm depth fraction ($p < 0.01$, Figure 3). Bulk density also varied across study sites with the discrepancy between the two most disparate sites, RYG and SKH, being 0.29 ± 0.08 g/cm^3. Off-pipeline, bulk density in the uppermost depth fraction varied from 0.64 g/cm^3 to 1.41 g/cm^3 with a mean of 1.06 g/cm^3. On-pipeline, bulk density in the uppermost depth fraction varied from 0.77 g/cm^3 to 1.52 g/cm^3 with a mean of 1.21 g/cm^3(Figure 3).

The soil tillage treatments significantly affected soil penetration resistance (Figure 4). For the 4-9 cm depth fraction, the soil had 99 ± 34 N greater resistance in the V treatment than in the control, 134 ± 29 N less resistance in the D treatment than in the control, and 74 ± 29 N less resistance in the DR treatment than in the control (Figure 4a). For the 9-14 cm depth fraction, the V treatment had 163 ± 64 N more resistance than the control, and the D treatment had 171 ± 56 N less resistance than the control (Figure 4b). For the 14-19 cm depth fraction, penetration resistance was 230 ± 107 N greater in the V treatment than in the control (Figure 4c). Differences were not evident for any treatment at depths greater than 19 cm, and the R treatment was not significantly different from the control at any depth.

In undisturbed, off-plot locations, 2009 cheatgrass seedling density was 506 ± 216 plants/m^2 in May and 139 ± 75 plants/m^2 in July. In treatment plots, 2009 cheatgrass seedling density was influenced by interactions between Plateau and season ($p < 0.0001$) and between Plateau and disking treatment ($p = 0.03$). Cheatgrass seedling density increased from 41 ± 20 in May to 201 ± 82 plants/m^2 in July in plots without Plateau ($p < 0.0001$), but there was no seasonal difference in plots with Plateau ($p = 0.12$; Figure 5). In May, Plateau reduced cheatgrass density from 53.3 plants/m^2 to 11.1 plants/m^2 in non-disked plots ($p = 0.03$), but had no detectable effect in disked plots ($p = 0.62$). In May, disking reduced cheatgrass seedling density from 53.3 to 23.3 plants/m^2 in the absence of Plateau ($p = 0.008$), but made no significant difference when applied with Plateau ($p = 0.83$; Figure 5). In July, Plateau reduced cheatgrass density from 242.5 to 44.0 plants/m^2 in undisked plots ($p = 0.03$), but had no detectable effect in disked plots ($p = 0.32$). In July, disking reduced cheatgrass seedling density from 242.5 to 138.6 plants/m^2 in plots without Plateau ($p = 0.002$), but had no detectable effect in plots with Plateau ($p = 0.87$; Figure 5). There were no detectable effects of any treatments on July seedling density of perennial grasses, perennial forbs, annual forbs, shrubs, or sagebrush. There was no significant relationship between soil penetration resistance and cheatgrass seedling density in May ($R^2 = 0.08$; $p = 0.27$) or July ($R^2 = 0.05$; $p = 0.43$).

Cheatgrass cover was influenced by year ($p = 0.03$), a strong main effect of Plateau ($p = 0.003$), and an interaction between disking and Plateau ($p = 0.02$). Cheatgrass cover increased from 32.9% in 2010 to 40.0% in 2011. Averaged across years and other treatments, cheatgrass cover was 52.1% in plots without Plateau and 21.6% in plots with Plateau. Disking tended to produce opposite effects depending on whether or not Plateau was applied, although individual contrasts of means were not significant. With Plateau, disking may have increased cheatgrass cover, from a mean of 17.6% to 25.7% ($p = 0.06$). In the absence of Plateau, disking may have decreased cheatgrass cover, from 54.6% to 47.8% ($p = 0.10$; Figure 6). There was no significant relationship between soil penetration resistance and cheatgrass cover in 2010 ($R^2 = 0.03$; $p = 0.37$) or 2011 ($R^2 = 0.02$; $p = 0.43$).

Perennial grass cover was influenced by disking ($p = 0.01$) and by year ($p = 0.002$). Perennial grass cover increased from 21.5% in 2010 to 27.5% in 2011. Averaged over years, perennial grass cover was 26.1% in disked plots and 22.9% in non-disked plots.

Perennial forb cover was not influenced by any factors ($p > 0.20$). Annual forb cover was not influenced by any treatments, but dropped from 26.0% in 2010 to 16.6% in 2011 (year effect $p = 0.0006$).

Shrub cover was influenced by an interaction between Plateau and year ($p = 0.001$) and a likely interaction between Plateau and rolling treatment ($p = 0.06$). In 2010, no Plateau effect was evident ($p = 0.24$), and shrub cover averaged 1.5%. In 2011, shrub cover depended on Plateau treatment ($p = 0.002$) with 9.1% shrub cover in Plateau plots and 1.2% shrub cover in no Plateau plots (Figures 7 and 8). Rolling had no apparent effect in the absence of Plateau ($p = 0.75$), but with Plateau, shrub cover dropped from 7.7% in not-rolled plots to 4.0% in rolled plots ($p = 0.02$; Figure 7).

Ambient cheatgrass propagule pressure in the study areas peaked between early June and mid-July, and then tapered off by early September (Figure 9). Peak values varied from 160 seeds/$m^2 \cdot$day at SKH to 1 seed/$m^2 \cdot$day at WRR (Figure 9).

DISCUSSION

The Plateau treatment reduced cheatgrass seedling density and cover, and the effects were still evident three years post-treatment. The Plateau treatment also greatly increased shrub cover three years post-treatment, and had no effect on forb or grass cover. These results contrast with some other studies in

Wyoming big sagebrush plant communities, in which cheatgrass cover in plots where Plateau was applied rebounded to levels as high (Owen et al. 2011) or higher (Morris et al. 2009) than that of control plots in two to three years, or in which Plateau negatively affected forbs (Baker et al. 2009, Owen et al. 2011) or grasses (Strategy Choice Experiment in this document, *see below*). Several factors likely contributed to the positive effect of Plateau in the Pipeline Experiment, including the timing of the pipeline disturbances, the application method and rate, and the timing of herbicide application relative to sagebrush seeding.

The timing of the pipeline disturbances may have reduced cheatgrass propagule pressure, acting additively with the herbicide to provide enough cheatgrass control for desirable perennial plants to establish. Prior work has shown that Plateau is more effective on annual grasses when applied after disturbances such as burning (Sheley et al. 2007, Davies and Sheley 2011). In a rate trial study, 70 g/ha of Plateau, when applied to bare soil, was as effective as 210 g/ha when applied without disturbance (Kyser et al. 2007). The reason commonly cited for the disturbance effect is that the herbicide reaches the soil surface more easily in the absence of thatch (DiTomaso 2000, Kyser et al. 2007, Sheley et al. 2007, Davies and Sheley 2011). However, the effect of disturbance at reducing propagule pressure is probably also an important factor, as burns can reduce cheatgrass seed density by 97% (Humphrey and Schupp 2001). In the spring following the pipeline disturbances, in the absence of Plateau, cheatgrass seedling density was five-fold lower in disturbed versus undisturbed locations. This may have been due to the timing of the disturbances the prior year. Cheatgrass seed distribution in the study areas peaks in June and continues until September (Figure 9), and the pipeline disturbances were completed in September. Topsoil removal, stockpiling and replacement likely buried the majority of cheatgrass seeds which had fallen on the soil surface during the 2008 growing season. The Plateau application may have been sufficient to provide continuing control for the portion of seeds that remained viable.

The Plateau rate used in this study, 105 g ai/ha (6 oz./acre), avoided substantial effects on forbs, which has been seen with rates of 132 g ai/ha (8 oz./acre) or higher (Baker et al. 2009, Owen et al. 2011), and was neutral with respect to grasses. Forbs, grasses, shadscale saltbush (*Atriplex confertifolia*), and fourwing saltbush (*Atriplex canescens*) were planted two to 20 days after herbicide application in this study. Another study has shown that the amount of time between Plateau application and seeding is important for western wheatgrass biomass, with a 90-day interval resulting in a positive impact of Plateau, and shorter intervals resulting in inconsistent or negative impacts (Sbatella et al. 2011). It is possible that a positive effect on grasses might have been realized in this study had the plant-back interval been longer. As big sagebrush was broadcast over snow, sagebrush plant-back interval was 75-90 days. This may explain why the Plateau treatment improved shrub cover to a greater degree than grass cover, as 77% of shrub cover was big sagebrush cover.

The effects of Plateau on shrub and cheatgrass cover are likely intertwined. In a study of competitive dynamics between cheatgrass and big sagebrush, sagebrush and cheatgrass competed for soil water and cheatgrass cover increased when sagebrush was removed (Prevey et al. 2010). In plots where Plateau was applied, sagebrush appears to have established well enough to limit cheatgrass cover, resulting in a long-term effect of Plateau on the composition of the plant community.

The disking treatment was moderately effective at reducing cheatgrass seedling density and improving perennial grass cover. This was probably in part due to directly killing germinating cheatgrass plants, as disking was applied near the time of cheatgrass emergence. Disking may have also helped improve perennial grass cover by relieving compaction of the rooting zone, which can restrict root growth and compromise establishment of deeply-rooted perennial plants (Thompson et al. 1987). Disking may also have buried a few cheatgrass seeds, which distributed over the disturbance in the three to four weeks between when the pipelines were reclaimed and the disking treatment was applied. However, the creation of the pipeline disturbances themselves likely achieved the majority of the benefit to be gained by this strategy. A recent study demonstrated that cheatgrass seeds distribute themselves very readily over bare soils (Johnston 2011a). The longer bare soil is exposed, the more important it may be to apply an additional disturbance to bury cheatgrass seeds, particularly if bare soil is exposed during the summer.

The goal of the rolling treatments was to increase soil bulk density to 1.2 -1.3 g/cm^3, as bulk densities in this range may reduce cheatgrass emergence by 52-60% (Thill et al. 1979). The pipeline disturbances increased bulk density from 1.1 to 1.2 g/cm^3, and the vibratory drum treatment further increased soil density. However, there was no effect of the vibratory drum treatment on cheatgrass, and no correlation between soil penetration resistance and cheatgrass seedling density or cover. In the field, whenever the soils presented a dense upper layer, cheatgrass was noted growing through cracks. Creating a layer of uniformly dense soil that would impede cheatgrass emergence may require the use of a soil binding agent, or may not be possible with soil types which shrink and swell, such as those found at the majority of sites in this study. Rolling also had a negative effect on shrub cover, therefore rolling is not recommended.

In summary, a moderate application rate of Plateau can be successful in limiting cheatgrass and promoting perennial plant establishment in pipeline restoration in Wyoming big sagebrush plant communities. The success of restoration may depend on using the disturbance itself to bury some cheatgrass seeds, and then applying herbicide for continuing control. If a reclaimed surface is left exposed during the time when cheatgrass seeds are distributing, it may also be helpful to disk the surface prior to planting. Rolling the surface to discourage cheatgrass emergence is not recommended. The initial, intensively monitored phase of the Pipeline Experiment is now considered to be complete. Monitoring of the Pipeline Experiment in the future will limited to every third year.

COMPETITION EXPERIMENT

Overview

- Goal: Test novel techniques for minimizing the competitive advantage of cheatgrass under a condition of controlled cheatgrass propagule pressure.
- Conducted at two sites: WRR and SGE (Figure 1, Table 1).
- Treatments:
    - Binding agent (three levels): a low level of binding agent applied (Low BA), a high level of binding agent applied (High BA), or no binding agent applied (No BA).
    - Super-absorbent polymer (2 levels): super-absorbent polymer applied (SAP) or no polymer applied (No SAP).
    - Rolling (2 levels): rolled with a static heavy roller (Rolled) or not rolled (Not rolled).
- Design: Factorial split-split plot, with completely randomized whole plots. The subplot factor was binding agent, the split plot factor was super-absorbent polymer and the whole plot factor was rolling (Figure 10).
- Plot size: 2.4 m X 2.4 m.
- Five replicates per site.
- Responses measured: Cover of perennial grasses and cheatgrass.

METHODS

Cheatgrass seed was collected using a lawnmower with a bagging attachment from monocultures or near-monocultures in four locations, each within 50 miles of the study sites. Collections were made in late June or early July 2009, when most or all of the cheatgrass in a location had fully ripened seed heads. Seed was allowed to dry and after-ripen in shallow containers in a dry, warm location for approximately three months. The density of apparently viable cheatgrass seeds was determined by gathering five 5 g subsamples from each collection, and then counting and weighing all of the fully developed, hard-coated cheatgrass seeds for each subsample. Equal quantities of seeds from each collection were mixed together, and then a volume of seed sufficient to supply 300 seeds/m^2 was prepared for each subplot. Seed was hand-broadcast in early October 2009, and then immediately lightly raked to incorporate seed into the soil. The 300 seeds/m^2 seeding rate is about 25% of the 2009 cheatgrass seed production at heavily cheatgrass-infested sites quantified for the Pipeline experiment, and therefore thought to be a reasonable seed density for a Piceance Basin site in the initial phases of invasion.

A mixture of native wheatgrasses (Table 3) was drill-seeded using a Plotmaster™ 400 (Tecomate Wildlife Systems, San Antonio, TX) in mid-October 2009. Seed was mixed 1:1 by volume with rice hulls to maintain suspension of the seed mixture. For SAP plots, granulated super-absorbent polymer was added to the seed/rice hull mixture. At SGE, 6.7 g/m^2 of polymer was added, and at WRR, 30.8 g/m^2 was added. These rates are near the lower and upper limits, respectively, of recommended application rates for agricultural purposes.

Next, whole plots receiving the rolling treatment were rolled 10 times with a static roller supplying a linear load of 36.5 N/cm (20.8 lbs/in). Binding agent subplots were then treated by sprinkling plots using hand watering cans. High BA plots received 4100 li/ha (440 gal/ac) of binding agent, diluted 6:1 with water. Low BA plots received 1600 li/ha (175 gal/ac) of binding agent, diluted 17:1 with water. No BA plots received 21,000 l/ha (3200 gal/ac) of plain water, an amount equivalent to the total amount of liquid applied to other plots.

Following implementation, the entire treatment area was surrounded by a barrier to prevent dispersal of cheatgrass seed out of the experiment area. A physical barrier of 0.6 m high aluminum window screen supported by oak stakes was constructed adjacent to the plots. Outside of this, we applied a chemical barrier of Pendulum, a broad spectrum pre-emergent herbicide, at 3200 g ai/ha (0.75 gal/ac) to a 1 m wide strip of bare ground.

To assess vegetation response, percent cover by species was quantified in late July and early August 2011 in a 1 m X 1 m subplot centered within each plot. A grid containing 36 intersections was held over the subplot, and point-intercept hits were measured at each grid intersection using a laser point-intercept sampling device (Synergy Resource Solutions, Bozeman, MT). All layers of vegetation were identified to species at each hit. When calculating percent cover of perennial grasses, overlapping hits of different species within that functional group (for instance, western wheatgrass overlying Sandberg bluegrass) were counted as a single instance of the functional group.

Three soil moisture readings were made in random locations within each plot on June 6, June 16, and July 19, 2011. Readings were taken to 12 cm using a Hydro Sense® Soil Water Measurement System (Campbell Scientific, Inc, Logan, Utah) and were averaged for each plot on each date. At SGE on June 16, 2011, only 15 of 60 plots were sampled due to equipment failure.

The cover of perennial grasses, cover of cheatgrass, and volumetric soil moisture in response to rolling, super-absorbent polymer, and binding agent treatments was analyzed in SAS PROC MIXED for a split-split plot structure with completely randomized whole plots. For cover data, site was included as a fixed effect, and interactions between treatments and site were also considered. Cover data was transformed by an arcsine [square root (x)] transformation to achieve normality. Soil moisture data was not transformed, and sites were analyzed separately. Due to the high proportion of missing data points at SGE on June 16, the June 16 date was not analyzed at SGE.

Main effects and all possible interactions were included as fixed effects in the initial model and a backwards model selection process was used to determine the final model. A significance level of $\alpha = 0.05$ was used to determine significantly different means, and a level of $\alpha = 0.10$ for interactions was used to determine which means to compare. For volumetric soil moisture, a repeated measures analysis with a compound symmetry autocorrelation coefficient was incorporated within the split-split plot design, and interactions between treatment effects and measurement date were included as fixed effects.

RESULTS

Perennial grass cover was influenced by site ($p = 0.0004$). Perennial grass cover averaged 45.7% at SGE and 58.5% at WRR. No treatments or interactions significantly affected perennial grass cover ($p > 0.15$).

Cheatgrass cover was influenced by site ($p = 0.0005$), a probable interaction between site and SAP, and an interaction between SAP and binding agent ($p = 0.05$). At SGE, cheatgrass cover averaged 11.2%, and no difference in SAP versus no SAP plots was evident ($p = 0.93$). At WRR, SAP reduced cheatgrass cover ($p = 0.02$) from a mean of 33.8% in plots without SAP to 22.4% in plots with SAP. Averaged across sites, in the absence of binding agent, SAP reduced cheatgrass cover ($p = 0.006$), but had no apparent effect with moderate or high levels of binding agent ($p > 0.26$). In the absence of SAP, the high binding agent treatment reduced cheatgrass cover from 26.8% to 16.9% relative to no binding agent treatment ($p = 0.01$). There was no significant difference between the moderate binding agent treatment and the no binding agent treatment, and there were no differences between any binding agent treatment levels in the presence of SAP ($p > .20$; Figure 11).

Volumetric soil moisture at WRR was influenced by many interacting effects, including three different three-way interactions: rolling with SAP and binding agent ($p = 0.04$), date with rolling with SAP ($p = 0.04$), and date with rolling and binding agent ($p = 0.02$); two different two-way interactions: rolling treatment with SAP ($p = 0.01$), and binding agent with date ($p = 0.005$), and two main effects: date ($p < 0.0001$) and binding agent ($p = 0.0009$). Soil moisture averaged 24.0% on June 3, 9.5% on June 16, and 11.4% on July 19 (Figure 12). On June 3, plots with low or high binding agent had 3.2 ± 2.1 % higher soil moisture than plots with no binding agent (Figure 12). On June 16, there was no difference due to binding agent ($p > 0.36$), and on July 19, there were no differences at the $\alpha = 0.05$ level ($p > 0.06$, Figure 12). The nature of the rolling by SAP interaction was such that SAP tended to increase soil moisture in the absence of rolling, but decrease soil moisture when applied with rolling (Figure 13). This two-way interaction was influenced by date and binding agent. On June 3, there was a probable three-way interaction between binding agent, SAP, and rolling ($p = 0.07$). With no or low binding agent, there were no clear effects of rolling, SAP, or their interaction ($p > 0.07$). With high binding agent, rolling and SAP interacted ($p = 0.008$) such that SAP increased soil moisture by 5.3 ± 4.7 % in the absence of rolling, but reduced soil moisture by 4.8 ± 4.6 % when applied with rolling (Figure 13). On June 16, there were no clear treatment effects ($p > 0.08$). On July 19, a three-way interaction occurred between rolling, SAP and binding agent ($p = 0.05$). With no binding agent, rolling and SAP interacted ($p = 0.05$) such that SAP may have increased soil moisture in the absence of rolling, but reduced soil moisture in the presence of rolling although individual contrasts were non-significant. With the low binding agent, there were no clear treatment effects ($p > 0.41$). With the high binding agent, rolling and SAP interacted ($p = 0.05$) such that SAP may have increased soil moisture in the absence of rolling, but reduced soil moisture in the presence of rolling,although individual contrasts were non-significant (Figure 13). At SGE, soil moisture was higher on June 3 than July 19, but there were no treatment effects ($p > 0.20$).

DISCUSSION

Under certain conditions, both binding agent and SAP reduced cheatgrass cover. Where SAP was not applied, the high binding agent treatment reduced cheatgrass cover, and where the binding agent treatment was not applied, the SAP treatment reduced cheatgrass cover. Averaged over binding agent treatments, SAP reduced cheatgrass cover from 34% to 22% at WRR, where it was applied at 30.8 g/m^2, but not at SGE, where SAP was applied at 6.7 g/m^2. In the absence of binding agent, SAP reduced cheatgrass cover from 37.5% to 19.7% at WRR.

Binding agent, SAP, and rolling interacted in complex ways to influence soil moisture. Binding agent increased soil moisture at WRR on one of three measurement dates in 2011, and increased soil moisture at both SGE and WRR on one of two measurement dates in 2010. These results confirm the manufacturer's claim that the product tested increases water infiltration, resulting in increased soil water at some points in time. At WRR in 2011, on certain dates and at certain levels of binding agent, there was an interaction by which SAP increased soil moisture in plots that were not rolled, but decreased soil moisture in plots that were rolled. This may be related to a pattern of perennial grass density which was evident in the 2010 data. In rolled plots at WRR, SAP promoted 2010 perennial grass density. Higher grass density would likely result in quicker utilization of soil moisture, resulting in the observed pattern in the soil moisture data.

It is unclear why neither SAP nor binding agent affected 2011 perennial grass cover, while effects of both of these products on cheatgrass cover are evident. It is possible that perennial grasses benefit from these products, but that the benefits are yet only realized belowground. The aboveground response of cheatgrass may be more sensitive to these treatments because cheatgrass fluctuates more widely in response to resource variability than do perennial plants (Bradley 2009). Another possibility is that there is some direct negative influence of SAP and binding agent on cheatgrass.

It is interesting to note that the effect of binding agent and SAP on cheatgrass cover seem to cancel one another. While both of these products reduced cheatgrass cover in 2011, neither was effective the presence of the other. The effect of each of these additives on soil moisture and competitive dynamics is complex. For restoration management purposes, the most important task is to determine which application or combination of applications may aid in cheatgrass control.

The effect of binding agent on cheatgrass has been inconsistent. In 2010, cheatgrass cover increased with the low level of binding agent at SGE. In 2011, lower cheatgrass cover was apparent in plots with the high binding agent treatment. This could be due to the stage of the experiment, and/or to different precipitation in different years. In 2010, early June soil moisture averaged 25%, while in 2011, it averaged 35%. The binding agent might have helped cheatgrass overcome a spring moisture limitation in 2010, while in 2011 the overriding factor may have been amplification of rain events critical for perennials.

Cheatgrass has adapted to complete its life cycle before the dry period of the summer in the intermountain west (Rice et al. 1992), making it an effective competitor in arid ecosystems with variable soil moisture (Chambers et al. 2007). By absorbing water and then gradually releasing it into the soil, SAPs can reduce the variability in soil moisture over time. With respect to cheatgrass control and perennial grass establishment, SAP addition had only beneficial or neutral effects. In addition to more reliably producing desirable results than the binding agent treatment, SAP addition is less expensive and requires less mobilization of machinery. In order to apply the binding agent in the manner tested in this study, 3200 gal/ac of water is needed, requiring a water truck. In contrast, granulated SAP can easily be applied through a drill seeder or fertilizer spreader.

If the results from 2012 corroborate those already observed, then addition of super-absorbent polymer at 30 g/m^2 may be a useful tool for promoting perennial grasses under competition from cheatgrass. Future work may involve investigating the best application method and rate of SAP, and testing the effectiveness of SAP addition in combination with other treatments that alter soil moisture.

GULLEY EXPERIMENT

Overview

- Goal*:* identify which potential sources of weeds are important to control: those that originate from within the soil seed bank of the reclamation area, those that enter from the surrounding landscape, or both.
- Conducted at four sites: RYG, SKH, YC1, and YC2 (Figure 1, Table 1).
- Treatments:
  - Fallowing (two levels): fallowed with Pendulum herbicide for one year prior to seeding (fallowed) or seeded immediately (Unfallowed).
  - Plateau application (two levels): Plateau applied (Plateau) or no Plateau applied (no Plateau).
  - Seed Barriers (two levels): surrounded by a seed dispersal barrier (barrier) or not surrounded (no barrier).
- Design: Factorial split-split plot, with completely randomized whole plots. The whole plot factor was fallowing, the subplot factor was seed barriers, and the sub-subplot factor was Plateau (Figure 14). Whole plots were completely randomized.
- Plot size: 9 m by 6 m.
- Three replicates per site.

METHODS

In late August and early September, 2009, fallowed plots were treated with Pendulum at 3200 g ai/ha (3 qt/ac), applied with a boom sprayer with 330 li/ha (35 gal/ac) of water. At the time of application, no germinated plants of any kind were evident at any of the sites. Once dry, the product was immediately incorporated into the soil with light disking to 5 cm (2 in) to prevent breakdown due to UV radiation. Next, the mixture of native grasses, forbs and shrubs in Table 4 (except big sagebrush) was hand-broadcast. Even seed distribution was ensured by preparing batches of the seed mix for each sub-subplot and seeding them individually. Seed was mixed 1:1 by volume with rice hulls to aid in even distribution of species. Seed was lightly raked to incorporate it into the soil after broadcasting. The same day as seeding, Plateau was applied at 140 g ai/ha (8 oz/ac) with 655 li/ha (70 gal./ac) of water using a backpack sprayer to unfallowed Plateau plots. Dye indicator was used to ensure even application.

To prevent wind and water erosion, DirtGlue soil binding agent was applied to all plots in September 2009. Soil binding agent was applied with a boom sprayer at 190 li/ha (50 gal/ac) diluted 10:1 with water. Next, barrier subplots were surrounded by aluminum window screen seed dispersal barriers. Barriers were 0.6 m high and were secured to oak stakes with staples. One meter wide buffer strips separated barrier subplots. Finally, locally collected big sagebrush seed was hand-broadcast on top of snow in unfallowed plots in December of 2009.

During the 2010 growing season, fallowed plots were maintained in a nearly unvegetated condition by applying glyphosate at 560 g/ac (8 oz./ac) in early June, and hand-pulling any plants nearing seed production in late June. In early September 2010, soil compaction was relieved in fallowed plots by ripping to 30 cm with a Plotmaster™ 400 (Tecomate Wildlife Systems, San Antonio, TX). This necessitated removing and then rebuilding the seed dispersal barriers in fallowed plots. Following ripping, Fallowed, Plateau plots were treated with Plateau at 140 g ai/ha (8 oz/ac) applied with 655 li/ha (70 gal/ac) of water with a backpack sprayer. Fallowed plots were seeded in late September using the same seed mixture and techniques as had been used in 2009 for unfallowed plots. Locally collected big sagebrush seed was hand-broadcast on top of snow in fallowed plots in December 2010.

Some cheatgrass seed that had been caught in the dispersal barriers in 2009 germinated and grew through the barrier. In order to fortify the barriers, we applied Plateau at 140 g ai/ha (8 oz/ac) in a 0.1 m strip between 9/14/10 and 9/28/10 at the base of the barrier.

A difficulty with constructing a fair test of the barriers is that subplots on the edge of the experiment area are likely to be subject to more seed blowing in from the surrounding landscape than are subplots in the interior. We moderated this effect by hand-broadcasting cheatgrass seed within the buffer strips separating subplots in 2009 and again in 2010. To determine how much seed to scatter, we used annual data on ambient cheatgrass seed rain known from our seed rain traps (Appendix 1). Because the traps were sticky and did not allow the seeds to redistribute, we scattered only half as much seed per unit area as these traps had caught. This compensated for the fact that under normal conditions, roughly half of cheatgrass seeds landing in a particular location move again (Kelrick 1991). The scattered cheatgrass seed had been collected from near-monocultures within 100 m of each site in June and July, when the seed was dry and nearly ready to fall. Seed was collected using a lawnmower with a bagging attachment. Viable cheatgrass seed content was estimated for each collection by gathering five 5g subsamples, and then counting and weighing all of the fully developed, hard-coated cheatgrass seeds from each subsample.

At two of the sites, RYG and SKH, the barriers were badly damaged by cows trampling the sites after the cheatgrass seed had been broadcast in 2009. The barriers were rebuilt, and secured with laths and wood screws, which were added to the oak stakes at all sites to better secure the window screen. The barrier treatments at RYG and SKH are best viewed as being functionally implemented in 2010, while those at YC1 and YC2 were effective for 2009 growing season. All of the sites were fenced to prevent further damage.

Vegetation was assessed by percent cover using five 1 m^2 mini-plots per sub-subplot. One mini-plot was located in the center of the sub sub-plot, and the remaining mini-plots were equidistant from the center mini-plot and a sub sub-plot corner. A grid containing 36 intersections was held over each mini-plot, and point-intercept hits were measured at each grid intersection using a laser point-intercept sampling device (Synergy Resource Solutions, Bozeman MT). All layers of vegetation were identified to species at each hit. When calculating percent cover of a given functional group, such as perennial grasses, overlapping hits of different species within a functional group (for instance, western wheatgrass overlying Sandberg bluegrass) were counted as a single instance of the functional group.

The cover of perennial grasses, perennial forbs, annual forbs, annual grasses, and shrubs in response to site, fallow treatment, Plateau treatment, and barrier treatment was analyzed using ANOVA in SAS PROC MIXED. All factors were considered fixed. Site and fallowing were considered between-subject effects for whole plots, and barriers and Plateau treatment (nested within barriers) were considered within-subject effects. Biennial forbs were lumped with annual forbs. Cover data was transformed by an arcsine [square root (x)] transformation to achieve normality. A full model including all possible interactions was first considered, and a backwards model selection process was used to determine the final model. A significance level of $\alpha = 0.05$ was used to determine significantly different means, and a level of $\alpha = 0.10$ for interactions was used to determine which means to compare. The percentage of native versus non-native species was calculated for all functional groups.

RESULTS

Perennial grass cover was influenced by site, fallowing treatment, and their interaction ($p < 0.0001$), barrier treatment ($p = 0.05$), and an interaction between site and Plateau treatment ($p = 0.03$). Fallowing reduced perennial grass cover from 19.6 to 3.8% at RYG, 45.7 to 3.1% at SKH, 63.5 to 7.6% at YC1, and 26.7 to 2.6% at YC2 ($p < 0.0008$; Figure 15a-d). Barriers increased perennial grass cover from an average of 19.5% in no barrier subplots to 23.7% in barrier subplots (Figure 16a). Plateau reduced perennial grass cover from 27.6 to 21.3% at SKH ($p = 0.01$), while Plateau effects at other sites were non-significant ($p > 0.06$). Native grasses were 94% of perennial grass cover.

Perennial forb cover was influenced by site, fallowing treatment, plateau treatment, and an interaction between site and Plateau treatment ($p < 0.0008$). Across sites, perennial forb cover averaged 8.1% in unfallowed plots and 5.4% in fallowed plots. Plateau treatment had an effect at YC1 and YC2 ($p < 0.0001$), but no apparent effect at SKH or RYG ($p > 0.39$). At YC1, perennial forb cover averaged 5.4% in Plateau plots and 23.3% in no Plateau plots. At YC2, perennial forb cover averaged 3.1% in Plateau plots and 10.5% in no Plateau plots (Figure 15e-h). The perennial forb cover was 99% native forbs.

Annual forb cover was influenced by many interacting effects, including a four-way interaction between site, fallowing treatment, barrier treatment, and Plateau treatment ($p = 0.004$). The analysis will be presented site-by-site. At RYG, Plateau treatment and fallowing treatment likely interacted ($p = 0.06$). Unfallowed, no Plateau plots averaged 37.0% annual forb cover, fallowed-only plots averaged 58.1%, Plateau-only plots averaged 6.9%, and plots both fallowed and treated with Plateau averaged 5.6% (Figure 15i). At SKH, Plateau and fallow treatment interacted ($p = 0.007$). Unfallowed, no Plateau plots averaged 48.0% annual forb cover, fallowed plots averaged 54.6%, Plateau plots averaged 4.9%, and plots both fallowed and treated with Plateau averaged 24.5% (Figure 15j). At YC1, only the fallowing treatment had an effect ($p = 0.02$). Fallowed plots averaged 24.4% annual forb cover, while unfallowed plots averaged 0.9% annual forb cover (Figure 15k). At YC2, a three-way interaction occurred between fallowing, Plateau, and barrier treatments ($p = 0.01$). Without fallowing, no other effects were significant ($p > 0.11$). With fallowing, Plateau and barrier treatment interacted ($p = 0.06$). Barriers were effective in the absence of Plateau ($p = 0.03$) but not in the presence of Plateau ($p = 0.75$). Annual forb cover averaged 72.4% in plots with neither Plateau nor barriers, 39.6% in plots with only barriers, 9.3% in plots with only Plateau, and 11.1% in plots with both Plateau and barriers (Figures 13l and 14b). The annual forb cover was 87% non-native.

Annual grass cover was influenced by Plateau treatment, fallow treatment and their interaction ($p < 0.0001$). Annual grass cover averaged 35.5% in unfallowed, no-Plateau plots, 1.7% in plots with only the fallow treatment, 3.8% in plots with only the Plateau treatment, and 1.0% in plots with both the fallow and Plateau treatments (Figure 15m-p). All - 100% - of annual grass cover was non-native.

Shrub cover was influenced by Plateau treatment, fallowing treatment, site, and all interactions between these three variables ($p < 0.0016$; Figure 15q-t). At RYG, SKH, and YC1, shrub cover was influenced by Plateau, fallowing and their interaction ($p < 0.003$). At RYG, shrub cover averaged 12.8% in unfallowed, no-Plateau plots, 0.8% in plots with only the fallow treatment, 38.9% in plots with only the Plateau treatment, and 0.9% in plots with both the fallow and Plateau treatments (Figure 15q). At SKH, shrub cover averaged 4.3% in unfallowed, no-Plateau plots, 0.1% in plots with only the fallow treatment, 12.0% in plots with only the Plateau treatment, and 0.1% in plots with both the fallow and Plateau treatments (Figure 15r). At YC1, shrub cover averaged 8.4% in unfallowed, no-Plateau plots, 0.3% in plots with only the fallow treatment, 26.9% in plots with only the Plateau treatment, and 0.2% in plots with both the fallow and Plateau treatments (Figure 15s). At YC2, only the fallowing treatment had an effect ($p = 0.02$). Shrub cover was 0.4% in fallowed plots and 4.5% in unfallowed plots (Figure 15t). All - 100% - of shrub cover was native.

DISCUSSION

One year after completion of treatments, the fallowing treatment had mostly undesirable results. Perennial grass, forb and shrub cover were reduced, but the annual forb cover was increased by the fallow treatment. In examining the effects of fallowing, it is important to recall that fallowed plots were seeded in fall 2010, while unfallowed plots were seeded in fall 2009. Cover of perennial grasses, perennial forbs, and shrubs can be expected to increase with time since seeding, and cover of annual forbs can be expected to decrease with time since seeding. A fair test of the fallowing treatment will only be realized after sufficient time has passed to moderate the effect of the one-year time lag between fallowed and unfallowed plots. The fallowing treatment did provide good control of annual grasses.

In combination with the fallow treatment, effects of the Plateau were usually not evident. But in unfallowed plots, the Plateau treatment resulted in lower cover of annual grasses, lower annual forbs at two of four sites, lower perennial forb cover at two of four sites, and lower perennial grass cover at one of four sites. Plateau also increased shrub cover at three of four sites. These results are similar to those shown in the pipeline experiment, which showed little effect of Plateau on perennial grasses, but a positive effect on shrubs. In this experiment, however, a negative effect on forbs was found, which corroborates earlier studies (Baker et al. 2007, Owen et al. 2011). The combination of Plateau and fallowing appears to be too heavy-handed, with very little cover of any functional groups in plots where both treatments were applied.

The barrier treatment improved perennial grass cover, and in one instance (fallowed, no Plateau plots at YC2), reduced cover of annual forbs. Unlike the Plateau and fallowing treatments, which controlled the seed bank within plots, the barrier treatment controlled seed input from without the plots. The application of the barriers, however, was far from perfect. Wind and cow trampling compromised the barriers at RYG and SKH during a critical time of cheatgrass dispersal in 2009. Design modifications improved the barriers over time, as we learned how to prevent weeds from passing beneath the barriers or growing through them. In spite of these imperfections, benefits of the technique were shown, and unlike herbicide application, the barriers have no undesirable effects on seeded species. Future work should focus on improving the design of barriers to increase their effectiveness.

The Gulley experiment will continue to be monitored for at least three additional growing seasons.

MOUNTAIN TOP EXPERIMENT

Overview

- Goal: Identify techniques to maximize plant diversity, shrub establishment and forb establishment in areas where the threat of weed invasion is low.
- Conducted at four sites: SCD, SPG, TGC and SQS (Figure 1, Table 1)
- Locations had predominately native and desirable surrounding plant communities, and varied in elevation from 2342 m (7681 ft) to 2676 m (8777 ft; Table 1).
- Treatments:
    - Seeding (two levels): Seeded or Unseeded.
    - Soil surface (two levels): roughened with holes and mounds (Rough) or left flat (Flat).
    - Brush mulch (two levels): mulched with brush (Brush) or not mulched with brush (No Brush).
- Design: Completely randomized factorial (Figure 17).
- Plot size: 9 m by 6 m.
- Three replications per site.

METHODS

Treatments were implemented in August and September of 2009. The rough surface treatment was created using a mini excavator to dig holes approximately 100 cm by 60 cm by 50 cm deep. Material removed was mounded next to each hole, and approximately 18 holes were dug per plot. This resulted in approximately 20% of the ground being allocated to holes, 30% to mounded soil and 50% to interspaces.

Seed (Table 5) was mixed 1:1 by volume with rice hulls to help ensure even distribution of species in seeded plots. In flat plots, seed was drilled approximately 1 cm deep using a Plotmaster™ 400 with a drill attachment. In rough plots, seed was broadcast and then lightly raked to incorporate the seed into the soil. Seeding rates were the same for both seeding methods.

The brush mulch treatment was achieved by distributing approximately 1.2 m^3 of stockpiled woody debris to each plot receiving the brush treatment. Because some topsoil was mixed with stockpiled brush, and this likely contained viable seed, an effort was made to distribute equal amounts of this topsoil. Approximately four liters of topsoil from brush stockpiles was scattered over each Brush plot.

Mountain big sagebrush seed was collected within 10 miles of each study site in November 2009 and broadcast seeded in November and December of 2009 in seeded plots.

Vegetation was assessed in 2011 by percent cover using five $1m^2$ mini-plots per plot. One mini-plot was located in the center of the plot, and the remaining mini-plots were equidistant from the center mini-plot and a plot corner. A grid containing 36 intersections was held over each mini-plot, and point-intercept hits were measured at each grid intersection using a laser point-intercept sampling device (Synergy Resource Solutions, Bozeman MT). All layers of vegetation were identified to species at each hit. When calculating percent cover of a given functional group, such as perennial grasses, overlapping hits of different species within a functional group (for instance, western wheatgrass overlying Sandberg bluegrass) were counted as a single instance of the functional group.

Analysis of variance in SAS PROC MIXED was used to analyze differences in responses to treatments. Site was included as a fixed effect. Cover data was analyzed separately by the following functional groups: perennial grasses, perennial forbs, annual grasses, annual forbs, and shrubs. Biennial forbs were lumped with annual forbs. Cover data was transformed by an arcsine [square root (x)] transformation to achieve normality. A full model including all possible interactions was first considered, and a backwards model selection process was used to determine the final model. A significance level of $\alpha = 0.05$ was used to determine significantly different means, and a level of $\alpha = 0.10$ for interactions was used to determine which means to compare. The percentage of native versus non-native species was calculated for all functional groups.

RESULTS

Perennial grass cover was influenced by site, the seeding treatment, and their interaction ($p < 0.0001$), as well as by a three-way interaction involving site, seeding treatment and brush treatment ($p = 0.02$). Seeding increased perennial grass cover from 14.3% to 44.8% at SCD, from 3.9% to 36.2% at SPG, from 6.8% to 20.2% at SQS, and from 5.6% to 37.8% at TGC. The three-way interaction occurred because there was an interaction between seeding and brush treatment at SQS ($p = 0.01$) and possibly at SPG ($p = 0.06$), but not at other sites ($p > 0.36$). At SQS, brush increased perennial grass cover from 14.7% to 26.7% when seeded ($p = 0.006$), but had no effect in the absence of seed ($p = 0.39$; Figure 18). At SPG, brush had a nearly significant effect in seeded plots ($p = 0.06$), but had no effect in unseeded plots ($p = 0.42$; Figure 18). The perennial grass cover was 99.6% native species.

Perennial forb cover was influenced by site, seeding treatment, a surface treatment by seeding interaction, a site by seeding treatment interaction, and a site by surface treatment interaction ($p < 0.033$). Across sites, surface treatment had an effect in seeded plots ($p = 0.01$), but not in unseeded plots ($p = 0.20$); forb cover averaged 7.2% in flat plots with seed and 10.8% in rough plots with seed. The seeding treatment had an effect at SCD, SPG, and SQS ($p < 0.05$), but not at TGC ($p = 0.78$). Seeding increased perennial forb cover from 2.4% to 8.9% at SCD, from 2.5% to 9.1% at SPG, and from 2.4% to 4.8% at SQS. A main effect of surface treatment was evident at SCD and TGC ($p < 0.006$), but not at other sites ($p > 0.22$). At SCD, the rough surface treatment increased forb cover from 2.7% to 8.6%, while at TGC, the rough surface treatment reduced forb cover from 18.2% to 9.3% (Figure 19). And, 97.4% of perennial forb cover was native.

Annual forb cover was influenced by site, seeding treatment, and their interaction ($p < 0.0001$). Seeding reduced annual forb cover from 49.0% to 14.35% at SCD, 58.6 to 21.1% at SPG, from 68.0% to 54.7% at SQS, and from 47.0 to 25.5% at TGC. And, 41.8% of annual forb cover was native.

Annual grass cover only occurred in sufficient levels for analysis at SCD, and even at this site, annual grass cover averaged only 0.4%. The seeding treatment ($p = 0.02$) and the surface treatment ($p = 0.03$) affected annual grass cover at SCD. Seeding reduced annual grass cover from 0.8% to 0.05%. The rough surface treatment reduced annual grass cover from 0.8% to 0.01% (Figure 20). All annual grass cover was non-native.

Shrub cover was influenced by site ($p < 0.0001$), brush treatment ($p = 0.04$), seeding ($p = 0.02$), and an interaction between brush treatment and surface treatment ($p = 0.05$). Shrub cover averaged 1.1% at MTN, 1.5% at SPG, 3.7% at SQS, and 1.3% at TGC. Across sites, seeding reduced shrub cover from 2.2% to 1.5%. The brush treatment increased shrub cover from 1.3% to 2.3% in flat surface plots ($p = 0.005$), but had no apparent effect in rough surface plots ($p = 0.97$). All shrub cover was native.

DISCUSSION

Seeding increased perennial grass and perennial forb cover, while reducing annual forb, annual grass and shrub cover two years post-treatment. Higher annual cover in unseeded plots was expected, as annual plants are typically prolific seed producers and tend to dominate following disturbances. The annual species most prevalent in the plots were the native Douglas' knotweed (*Polygonum douglasii*) and the non-native prostrate knotweed (*Polygonum aviculare*). Prostrate knotweed typically persists in highly compacted soils, which are not present at the study sites, and it is expected that prostrate knotweed cover will decline with time. Seeding also decreased shrub cover, even though shrubs were seeded. Higher shrub cover in unseeded plots was due to establishment of snowberry and green rabbitbrush (*Chrysothamnus vicidiflorus*), which were not included in the seed mix. Apparently, even at this early stage of the experiment, the lessened competition in unseeded plots is promoting establishment of shrubs from naturally occurring seed.

The rough soil surface treatment improved perennial forb cover, especially when applied with seed. The pattern of results TGC, however, differed from that of other sites. At TGC, forb cover was higher in flat surface plots when unseeded. This may be due to an imperfection in how the well pad disturbance was simulated. TGC was flatter than other sites and required very little cut-and-fill to create a level surface. The topsoil layer was deep and was not completely disturbed. The year after disturbance, mature silvery lupine (*Lupinus argentus*) and white locoweed (*Oxytropis sericea*) plants were noted at the site. In rough surface treatment plots, these plants were disrupted, but in flat surface plots, these deeply-rooted species may have survived treatment implementation, leading to very high forb cover in flat surface plots at TGC. Excluding this site from interpretation, we find a large positive effect of the rough surface treatment on forbs (Figure 19).

The rough soil surface treatment also reduced the prevalence of annual grasses, namely cheatgrass, at the one site with sufficient cheatgrass to permit analysis (Figure 20). This pattern is similar to that observed in the Strategy Choice Experiment at the MTN site. As is explained in the following section, this may be due to reduced dispersal of cheatgrass seeds in rough surface treatment plots.

The brush treatment appears to improve shrub cover, and also improved perennial grass cover when seeded. The addition of brush may have added shrub seed to the plots, and the brush may also improve the establishment of shrub seedlings.

The Mountain Top experiment contrasts extreme treatments: seeding with a high density of perennial grasses, shrubs and forbs, versus not seeding at all in order to gauge the ecological resiliency of higher-elevation sites. The results occurring in unseeded plots, over time, will provide a baseline of expectations for these sites when topsoil is managed well, micro-catchments for higher moisture availability are provided, and when the plots are mulched with native brush.

STRATEGY CHOICE EXPERIMENT

Overview

- Goal: compare two mutually exclusive reclamation strategies (one which maximizes plant diversity and one which minimizes weed invasion) in situations where the threat of weed invasion is ambiguous.
- Conducted at 4 sites: WRR, SGE, GVM, MTN (Figure 1, Table 1).
- Treatments include:
  - Seed mix (2 levels): seeded with a high competition seed mix (HC) or a low competition mix (LC).
  - Soil surface/mulch type (2 levels): flat with straw mulch (Flat/Straw) or rough surface with brush mulch (Rough/Brush).
  - Herbicide (2 levels): Plateau applied (Plateau) or no Plateau applied (No Plateau)
- Completely randomized factorial (Figure 21).
- Plot size: 9 m X 6 m.
- Three replications per site.
- The four locations had 0-15% non-native cover prior to the start of the experiment.

METHODS

At GVM and MTN, the full experiment with all three treatments was implemented. At WRR and SGE, space constraints mandated implementing an abbreviated form of the experiment, and the herbicide treatment was omitted. Treatments were implemented in October of 2009.

Seed mixes for the HC and LC plots are shown in Table 6. A key difference between the mixes is in the number and type of grass seeds used. In the high competition mix, 344 grass seeds/m^2 (32 seeds/ft^2) were used, and these were mostly rhizomatous wheatgrasses. In the low competition mix, 156 grass seeds/m^2 (15 seeds/ft^2) were used, and 90% of these were less competitive bunchgrass species.

In Rough/Brush plots, all species were hand-broadcast and raked, after creation of the holes but before the application of brush. On Flat/Straw plots, some seed was hand broadcast and then lightly raked, and the remainder was drill seeded approximately 1 cm deep using a Plotmaster™ 400 with a hunter grain drill attachment (Table 6). Seed was mixed 1:1 by volume with rice hulls to aid in an even distribution of species.

Certified weed-free straw was applied by hand at a rate of 4.0 Mg/ha (1.8 tons/ac) to Flat/Straw plots. Straw was crimped in place using a custom-built mini crimper pulled behind an ATV. Rough/Brush plots were treated using a 331 Bobcat® compact excavator to dig holes approximately 130 cm X 80 cm X 50 cm deep. Material removed was mounded next to each hole, and 18 holes were dug per plot. This resulted in approximately 1/3 of the ground being allocated to each of holes, mounds, and interspaces.

Plateau plots were sprayed with 140 g ai/ha of Plateau (8 oz /ac) applied with 655 li/ha of water (70 gal /ac) with a backpack sprayer. To hit the target rate, a quantity of liquid sufficient to treat 2 plots was mixed, and then that quantity was applied to the 2 plots with a dye indicator to ensure even application. In Plateau, Flat/Straw plots, the amount of water used in herbicide application was tripled to aid the Plateau in penetrating the straw mulch.

After Plateau application, brush that had been cleared and stockpiled next to each site was applied to Rough/Brush plots. Approximately 5 m^3 of brush was applied evenly to each plot.

Big sagebrush was hand-broadcast on top of snow in all plots in December of 2009.

Ambient cheatgrass propagule pressure was quantified at all 4 sites in 2011 using techniques outlined in Appendix 1. The seeds caught per square meter for the entire growing season were calculated for each site.

Vegetation was assessed in 2011 by percent cover using five $1m^2$ miniplots per plot. One miniplot was located in the center of the plot, and the remaining miniplots were equidistant from the center miniplot and a plot corner. A grid containing thirty-six intersections was held over each miniplot, and point-intercept hits were measured at each grid intersection using a laser point-intercept sampling device (Synergy Resource Solutions, Bozeman MT). All layers of vegetation were identified to species at each hit. When calculating percent cover of a given functional group, such as perennial grasses, overlapping hits of different species within a functional group (for instance, western wheatgrass overlying Sandberg bluegrass) were counted as a single instance of the functional group.

The cover of perennial grasses, perennial forbs, annual forbs, annual grasses, and shrubs in response to site, seed mix, surface/mulch type, and Plateau treatment was analyzed using ANOVA in SAS PROC MIXED. All factors were considered fixed. Biennial forbs were lumped with annual forbs. Cover data was transformed by an arcsine [square root (x)] transformation to achieve normality. A full model including all possible interactions was first considered, and a backwards model selection process was used to determine the final model. A significance level of $\alpha = 0.05$ was used to determine significantly different means, and a level of $\alpha = 0.10$ for interactions was used to determine which means to compare. Because the Plateau treatment was only conducted at 2 of the 4 sites (GVM and MTN), separate backwards model selection processes were conducted for models with the Plateau treatment versus those without. The SGE and WRR sites were excluded from models containing the Plateau treatment parameter (the “complete experiment” results, denoted hereafter as *CE*). Results for the surface and seed mix treatments include all 4 sites, but plots where Plateau was applied at GVM and MTN were excluded (the “all sites” results, denoted hereafter as *AS*). The percentage of native versus non-native species was calculated for all functional groups.

RESULTS

Ambient cheatgrass propagule pressure at the four study sites for the 2011 season were as follows: GVM, 1256 seeds/m^2; MTN, 1.3 seeds/m^2; SGE, 4.0 seeds/m^2; and WRR, 4.0 seeds/m^2.

Perennial grass cover was influenced site, Plateau treatment, and their interaction ($p < 0.0017$ *CE*) as well as by surface treatment ($p = 0.02$ *AS*), seed mix treatment ($p < 0.0001$ *AS*), and an interaction between site and surface treatment ($p = 0.003$ *AS*). At GVM, Plateau treatment reduced perennial grass cover from 28.3% in no-Plateau plots to 16.9% in Plateau plots (Figure 22a). At MTN, Plateau reduced perennial grass cover from 32.6% in no-Plateau plots to 7.0% in Plateau plots (Figure 22b). The seed mix treatment was consistent across sites, with perennial grass cover averaging 41.9% in High Competition plots and 29.9% in Low Competition plots (Figure 23). The surface treatment had significant effects at MTN ($p = 0.0005$) and SGE ($p = 0.05$) but not at other sites ($p > 0.09$). At MTN, perennial grass cover averaged 41.3% in rough surface plots and 23.8% in flat surface plots (Figure 24a). At SGE, perennial grass cover was 41.5% in rough surface plots and 31.8% in flat surface plots. One hundred percent of perennial grass cover was native.

Perennial forb cover was influenced by Plateau, site, and their interaction ($p < 0.05$ *CE*) as well as by seed mix treatment ($p < 0.0001$ *AS*). At MTN, perennial forb cover dropped from 28.6% in no-Plateau plots to 17.7% in Plateau plots ($p = 0.003$; Figure 22b). At GVM, there was no apparent effect of Plateau on perennial forb cover ($p = 0.69$; Figure 22a). Across sites, perennial forb cover averaged 25.7% in LC plots and 15.8% in HC plots (Figure 23). Perennial forb cover was 99% native.

Annual forb cover was influenced by a strong main effect of Plateau ($p = 0.0004$ *CE*) as well as by a probable 3-way interaction involving Plateau, surface treatment, and seed mix treatment ($p = 0.07$ *CE*). Averaged across other treatments, annual forb cover was 20.8% in Plateau plots and 9.2% in no-Plateau plots (Figure 22). With the High Competition mix, Plateau and surface treatment interacted ($p = 0.02$). Where Plateau was applied, annual forb cover was 31.7% with the flat surface treatment, and 13.1% with the rough surface treatment ($p = 0.02$ for comparison of means). Where Plateau was not applied, there was no apparent effect of surface treatment ($p = 0.29$). With the Low Competition mix, there was no interaction between Plateau and surface treatment ($p = 0.96$) and no significant difference between the flat and rough surface when applied either with or without Plateau ($p > 0.34$). Annual forb cover was 97% non-native.

Annual grass cover was influenced by a highly significant 3-way interaction between site, surface treatment, and Plateau ($p=0.0003$ *CE*). At GVM, no interaction between surface treatment and Plateau was evident ($p = 0.16$), but Plateau had an effect ($p = 0.001$), with 29.1% annual grass cover in no-Plateau plots, and 7.1% annual grass cover in Plateau plots (Figure 22a). At MTN, the surface treatment and the Plateau treatment interacted ($p < 0.0001$). In plots with Plateau, the surface treatment had no effect ($p = 0.56$), but in plots without Plateau, annual grass cover was 44.1% in flat surface plots and 5.9% in rough surface plots ($p < 0.0001$ for comparison of means; Figure 24b). Annual grass cover was 100% non-native and 97.6% cheatgrass.

Perennial shrub cover was influenced by a strong main effect of Plateau ($p < 0.0001$ *CE*) a 3-way interaction between site, surface treatment, and Plateau treatment ($p = 0.02$ *CE*) as well as by a main effect of the surface treatment ($p = 0.05$ *AS*). Averaged across other factors, perennial shrub cover was 2.9% in Plateau plots and 7.9% in no-Plateau plots (Figure 22). At GVM, no interaction between Plateau treatment and surface treatment was evident ($p = 0.74$). At MTN, the surface treatment and the Plateau treatment interacted ($p = 0.006$). In plots with Plateau, the surface treatment had an effect ($p = 0.01$), with 1.7% cover of perennial shrubs in flat surface plots, and 8.0% cover of perennial shrubs in rough surface plots. In plots without Plateau, no significant effect of the surface treatment at MTN was evident ($p = 0.14$; Figure 24c). In the cross-site analysis, however, surface treatment did have an effect, with 5.8% shrub cover in rough surface plots, and 8.5% shrub cover in flat surface plots. Perennial shrub cover was 100% native.

DISCUSSION

The Plateau treatment had mostly undesirable consequences in this experiment. Although Plateau did have the positive effect of reducing annual grass cover, it also reduced perennial grass cover, reduced perennial forb cover at 1 of 2 sites, reduced shrub cover, and increased annual forb cover. The detrimental effects of Plateau were more evident at the MTN site, where perennial grass cover was over 4 times lower in Plateau plots than in no-Plateau plots. Reduced competition with perennial grasses is a likely explanation for higher annual forb cover in Plateau plots. Although some detrimental effect of Plateau on non-target species is expected, especially in the first year post-application, the degree of injury seen in this experiment 2-years post-application is likely unacceptable to managers. More favorable results might result from a lighter application rate than the 8 oz./acre used here. Also, the method of application used, backpack spraying, is not recommended because the rate is difficult to control precisely. Finally, allowing 3 months or more between applying Plateau and seeding has been shown to reduce injury to some desirable rangeland species (Sbatella et al. 2011) and may promote more favorable results.

The Low Competition seed mix had favorable results for improving reclamation areas as wildlife habitat, because very forb cover, 25.7%, was achieved. Annual forb and annual grass cover were similar between the High Competition and Low Competition mixes, therefore higher forb cover did not come at the expense of increased weeds. The Low Competition seed mix differed from the High Competition mix in having a much lower density of rhizomatous grass (Figure 25) and by including three additional species of perennial forbs: hairy golden aster (*Heterotheca villosa*), multi-lobed groundsel (*Packera multilobata*) and showy fleabane (*Erigeron speciosus*). Hairy golden aster and multi-lobed groundsel established successfully where seeded, with hairy golden aster comprising 1.2% cover and multi-lobed groundsel comprising 3.5% cover in Low Competition plots. The difference in perennial forb cover between the two mixes was 9.9%. Therefore, a little less than half of the difference in forb cover between the two mixes is attributable to the forb species unique to the Low Competition mix. The rest of the difference is due to better establishment of species such as sulfur-flower buckwheat (*Eriogonum umbellatum*), Western yarrow (*Achillia millefolium*), Lewis flax (*Linum Lewisii*), and Utah sweetvetch (*Hedysarum boreale*), which were seeded at the same rate in both mixes (Table 6). Higher cover of these species is likely due to lessened competition with rhizomatous grasses. This highlights the increase in cost-effectiveness of seeding forbs when the density of rhizomatous grasses is greatly reduced. The density of rhizomatous grasses included in the High Competition mix was similar to a commonly-used mix recommended by the White River BLM office for the pinyon-juniper habitat type (Figure 25). This study suggests that a lower density of rhizomatous grass may result in more valuable wildlife habitat following disturbances.

The rough surface with brush treatment had many favorable results. Rough/Brush plots had higher perennial grass cover at two of four sites, lower annual forb cover when Plateau and the High Competition mix were also applied, higher shrub cover at MTN when Plateau was also applied, and lower annual grass cover at MTN when Plateau was not applied. One negative result of the rough surface treatment was lower shrub cover when averaged across sites in the absence of Plateau. Perhaps the most interesting result is the dramatic reduction in annual grass cover- from 44.1% to 5.9%- at MTN in the absence of Plateau. The MTN site was one of the three sites that had low ambient cheatgrass propagule pressure. At two other such sites, SGE and WRR, very little cheatgrass established regardless of treatment. At MTN, nearby disturbances with thick stands of cheatgrass indicated that the site was vulnerable to cheatgrass invasion. How did the rough surface treatment prevent cheatgrass from invading? Perennial grass cover was somewhat higher in Rough/Brush plots at MTN, so increased competition from grasses is one explanation. However, shrub cover was lower with the rough surface treatment, and shrubs such as sagebrush are known to compete with cheatgrass (Prevey et al. 2010), so the end result of competitive dynamics on cheatgrass is unclear. Another explanation is that the rough surface treatment impeded cheatgrass dispersal. Recent work has shown that cheatgrass seeds disperse much farther in the absence of obstructions than they do in intact ecosystems, leading to enhanced dispersal following disturbances such as well pad construction or fires (Johnston 2011a, Monty et al. in review). Cheatgrass seeds may have been trapped in the holes in the rough surface treatment, preventing them from dominating the Rough/Brush plots.

The benefit of the rough surface treatment when used in conjunction with Plateau indicates that this treatment may have broad applicability. Two of the detrimental effects of Plateau shown in this study, reduced shrub cover and increased annual forb cover, were ameliorated when the rough surface treatment was also applied. Furthermore, visual comparisons between flat and rough surface plots suggest that the rough surface treatment increases plant stature (Figure 26), which was not captured by the cover data.

Plans for the 2012 season include assessing biomass in this experiment in order to better quantify treatment responses.

The Strategy Choice experiment focuses on situations where a wide range of outcomes of reclamation are possible. The data from this experiment indicates that excellent restoration of wildlife habitat should be the goal in these cases. Elements of an optimistic strategy, including a seed mix focused on forbs, broadcasting seed over a roughened soil surface, and using brush mulch, resulted in a better outcome than the more common strategy of drill seeding grasses heavily over a flat surface with straw mulch. This is especially striking given that broadcast and drill seeded plots were seeded at the same rate, whereas the seeding rate for broadcast application is usually doubled. It should be noted that moisture in the 2010 and 2011 growing seasons was normal or above normal. Additional years of data including drought years will be assessed before a final recommendation is made.

CONCLUSION

Although the five experiments comprising this project vary in their degree of maturation (e.g. 1 year of post-treatment data for the Gulley Experiment, vs. 3 for the Pipeline Experiment), some broad-scale synthesis of results can be made at this time. Treatments which appear promising in improving the quality of reclaimed wildlife habitat include applying Plateau herbicide (with extreme caution), timing disturbances to maximize weed seed burial, creating a rough soil surface composed of mounds and holes, utilizing obstructions to prevent weed seed dispersal, treating soil with granulated super-absorbent polymer, and using a seed mix focused on perennial forbs.

This report contains results of three experiments in which Plateau herbicide was applied. Of these, one experiment demonstrated only positive effects of the herbicide, one demonstrated both positive and negative effects, and one demonstrated mainly negative effects. In all cases, the herbicide was applied in the fall just prior to seeding. In the Pipeline experiment, Plateau was applied with a boom sprayer at 105 g ai/ha (6 oz/ac) to low and mid-elevation sites, and results after 3 years are very favorable. The herbicide was neutral with respect to grasses and forbs, but greatly improved shrub cover and reduced annual grass cover. In the Gulley experiment, Plateau was applied with a backpack sprayer at 140 g ai/ha (8 oz/ac) to low-elevation, weedy sites, and results after 2 years are mixed: Plateau effectively controlled both annual grasses and forbs and positively affected shrubs, but negatively affected perennial forbs and grasses. In the Strategy Choice Experiment, Plateau was applied with a backpack sprayer at 140 g ai/ha (8 oz/ac) to mid-elevation sites, and results after 2 years are unfavorable: perennial grass and shrub cover are greatly reduced, and annual forb cover is increased where Plateau was applied. The differences cannot be entirely attributable to time since treatment, because the Pipeline Experiment showed favorable responses to Plateau application after only 2 years (Johnston 2011b). The lower rate used in the Pipeline Experiment partially explains these results. The difference between the Gulley experiment and the Strategy Choice experiment may be due to the difference in initial weediness of the treated sites. At the Gulley sites, cheatgrass was a major component of the plant community prior to disturbance, and the positive effects of cheatgrass control counteracted the direct negative effect of the herbicide. At the Strategy Choice sites, the rate was too high for the conditions. Using lower rates, matching the rate to the site, and increasing time between application and seeding are recommended. A rate of 6 oz./acre may be a good maximum for very weedy sites, with lower rates to be used at less weedy sites. Note that the 6 oz./acre rate has been shown to provide only fleeting and ultimately insufficient cheatgrass control in Wyoming sagebrush communities in a prior study (Morris et al. 2009). It appears that while a light rate of Plateau application may be beneficial in restoration, it cases of severe infestation, it should be coupled with other measures to control cheatgrass.

One such other measure is the judiciously-timed application of disturbance. Auxiliary data taken for the Pipeline and Gulley experiments shows the time course of cheatgrass seed dispersal in northwestern Colorado, with a peak in late June, and continued dispersal until mid-September (Figure X). At the weediest site measured, cheatgrass seed production peaks at 160 seeds/m^2·day. Since 40 cheatgrass seeds/m^2 is sufficient to hinder the growth of even the most competitive perennial grasses (Evans 1961), many times more cheatgrass seed is produced *in a single day* than is acceptable for establishing native plants on restoration sites. Furthermore, these will readily spread from the edge of disturbances into bare soil areas (Johnston 2011a). Given the timing of cheatgrass seed dispersal, the worst possible scenario is if a disturbance occurs before spring, and is left bare over the summer. If the disturbance occurs in the fall, however, and is planted immediately, then there is little opportunity for cheatgrass seeds to disperse before seeded species germinate. Because cheatgrass seeds are sensitive to burial (Wicks 1997), then a fall disturbance will partially control cheatgrass. This was the case in the Pipeline Experiment, where cheatgrass density was five times lower in the disturbed area than in the adjacent undisturbed area the spring following disturbance. If a disturbance must occur early in the year, then applying an additional disturbance such as disking prior to fall planting may limit the number of viable cheatgrass seeds on the soil surface. For very weedy areas, some strategy such as this is recommended to augment chemical control. Dense cheatgrass stands may produce 20,000 seeds/m^2, therefore even a 99% effective herbicide would leave 200 viable seeds/m^2, which is more than enough to compromise a seeding.

There were two experiments where a roughed soil surface of mounds and holes, coupled with broadcast seeding, was compared to a flat soil surface coupled with drill seeding. In the Mountain Top experiment, the rough soil surface treatment was crossed with a brush mulch treatment, while in the Strategy Choice experiment, the rough soil surface treatment was always applied with brush mulch, and the flat soil surface treatment was always applied with straw mulch. In both experiments the rough soil surface outperformed the flat soil surface in most respects. The rough soil surface produced higher perennial forb cover in the Mountain Top experiment, higher perennial grass cover at two of the Strategy Choice sites, and lower annual grass cover at one site in each experiment. For the Strategy Choice Experiment, the results differ from those reported for 2010. In 2010, density of most desirable functional groups was lower with the rough soil surface treatment. Apparently, improved growth of plants with the rough soil surface offset the effect of lower initial density. In both experiments, the seed was applied at the same rate in the rough surface plots, which were broadcast, as the flat surface plots, which were drill seeded. These results bring into question the common practice of doubling the seeding rate for broadcast seeding. If the seedbed is well-prepared, doubling the seeding rate may be wasteful. The machinery and mobilization costs for the two methods are comparable, therefore broadcast seeding over a rough soil surface appears to be a cost-effective alternative. These results confirm and extend those of Eldridge (2011) who found that a rough soil surface treatment improved the cover of native plants at low elevation sites in the Colorado River Valley (Eldridge et al. 2011).

The reduction in annual grass shown at two sites with a rough soil surface treatment, the SCD site in the Mountain Top experiment, and the MTN site in the Strategy Choice experiment, suggests that a rough soil surface may aid in cheatgrass control under certain conditions. Altered competitive dynamics is one explanation for these results, but altered seed dispersal is probably also important. In a study of many kinds of seeds, Chambers (2000) found that large holes capture more seeds than flat surfaces (Chambers 2000). Recent work done as part of this project has shown that cheatgrass seeds move farther in the absence of obstructions that they do in intact ecosystems (Johnston 2011a). At both the SCD and MTN sites, cheatgrass was not prevalent prior to disturbance. The rough soil surface probably prevented a few cheatgrass seeds introduced during disturbance from spreading, concentrating them in a higher-moisture microclimate, where they may have been less competitive. Another experiment, the Gulley experiment, looked explicitly at creating obstructions to seed dispersal, in the form of windowscreen barriers placed around plots. Although there were imperfections in the implementation of these barriers, there were still some benefits from the treatment- an increase in perennial grass cover and a decrease in annual forb cover in fallowed, no-Plateau plots at YC2. Creating barriers to seed dispersal, either though aboveground structures or large holes to act as seed traps, may improve reclamation of disturbed areas by limiting the movement of weed seeds. In less weedy areas, it is possible that this would alleviate the need for herbicide and associated injury to desirable species.

In the Competition experiment, we tested the effect of granulated super-absorbent polymer (SAP) on the competitive balance between perennial wheatgrasses and cheatgrass. In the absence of other treatments, SAP cut cheatgrass cover in half, 2 years post-treatment when applied at 31 g/m^2 and concentrated in drill-seeded rows. Further studies should investigate the optimal application rate and application method. Although it might be desirable to treat the entire soil volume for a broadcast seeding, such application would probably be prohibitively expensive. For instance, a 0.2% v/v application rate (Agaba et al.) to a depth of 10 cm would require 225 g/m^2 (2013 lbs/acre), at a current cost of $5,450/acre. Concentrating the product in either drill-seeded rows or the holes of a rough soil surface treatment are likely the only cost-effective application methods.

Two experiments examined the consequences of seed mix choices. In the Strategy Choice experiment, we compared a seed mix with almost 75% forbs by seed number and virtually no rhizomatous grass (the Low Competition mix) to a seed mix with fewer forbs and a typical, 4.4 kg/ha (3.9 PLS/acre) rate of rhizomatous grass (the High Competition mix). In the Mountain Top experiment, we compared a seed mix with 4.4 kg/ha rhizomatous grass to the extreme of not seeding at all. The Low Competition mix produced higher forb cover with similar weed cover to the High Competition mix. The unseeded plots in the Mountain Top experiment had more weeds than the seeded plots, but the weeds were not species thought to persist over time, and they also had higher shrub cover. Collectively, these studies suggest that post-reclamation wildlife habitat could be improved by altering the composition of seed mixes to focus on forbs, bunchgrasses, and shrubs. The idea that seed mixes should limit the proportion of rhizomatous grasses in order to promote a mixed plant stand was proposed nearly 30 years ago (Redente et al. 1984). However, most seed mixes continue to be dominated by competitive grasses, probably out of a fear of weed invasion, a lack of availability of appropriate forb seeds, and/or a need for an inexpensive seed mix. This study made use of several forb species provided by the Uncompahgre Partnership (http://www.upartnership.org/) that are either not yet commercially available or have no Colorado-specific variety available. Several of these species established well, including local cultivars of many-lobed groundsel, hairy golden aster, sulfur flower buckwheat, bluestem penstemon (*Penstemon cyanocaulis*), and Western yarrow. The results of this study highlight the importance of making species such as these available at a reasonable cost.

Treatments which do not appear promising include surface compaction, fallowing with Pendulum herbicide, and addition of a soil binding agent to the soil. Rolling to create slight soil surface compaction was attempted in 2two studies: the Pipeline experiment and the Competition experiment. The goal of this treatment in these experiments was to determine if creating a crust of compacted soil would benefit reclamation by preventing the emergence of cheatgrass. In the Pipeline experiment, compaction with both a static and vibratory roller was tested, and in the Competition experiment, the combination of a static roller with a soil binding agent was tested. In no case was cheatgrass emergence affected, and a negative effect on shrubs was found in the Pipeline experiment. Fallowing with Pendulum herbicide was attempted in the Gulley experiment, and although the 2011 data is only one year post-treatment, the results were so detrimental to perennial grasses and forbs that it appears unlikely that fallowing will be recommended. Soil binding agent was tested in the Competition experiment, and had mixed results, at times causing increased cheatgrass cover, and at times limiting it. Because of these inconsistencies and the cost of the treatment, it is unlikely to be recommended.

In summary, excellent restoration of wildlife habitat following oil and gas disturbances is possible over a wide range of elevations in northwestern Colorado. At lower elevations and in places with some cheatgrass cover prior to disturbance, then a combination of approaches to control cheatgrass and promote native plants should be used. This may include a light herbicide application, fall disking prior to planting, using a roughened soil surface, and amending soil with a super-absorbent polymer. At middle and higher elevations, using a roughened soil surface and using a seed mix primarily of forbs is recommended. Note that these results apply to slopes of less that 5% and areas protected from grazing. Steeper slopes and grazed areas may require using rhizomatous grasses to protect soil resources.

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Table 1. Study site information. Pie charts are baseline relative cover from undisturbed areas.

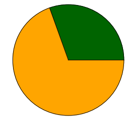
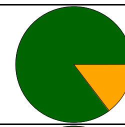
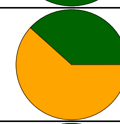
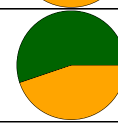
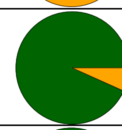
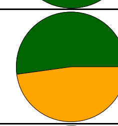
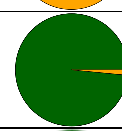
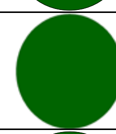
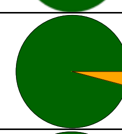
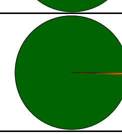
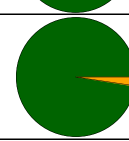
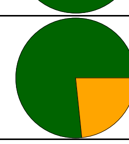
Code	Name	Landowner	Elev. m (ft)	Experiment(s) Conducted	Native NonNative
SKH	SK Holdings	Williams	1561 (5120)	Pipeline Gulley	
GVM	Grand Valley Mesa	Williams	1662 (5451)	Pipeline Strategy Choice	
YC2	Yellow Creek 2	DOW	1829 (5999)	Pipeline Gulley	
YC1	Yellow Creek 1	DOW	1905 (6248)	Pipeline Gulley	
SGE	Sagebrush	BLM	2004 (6573)	Strategy Choice Competition	
RYG	Ryan Gulch	Williams	2084 (6835)	Pipeline Gulley	
MTN	Mountain Shrub	BLM	2183 (7160)	Strategy Choice	
WRR	Wagon Road Ridge	Williams	2216 (7268)	Pipeline Strategy Choice Competition	
SCD	Scandard	BLM	2342 (7681)	Mountain Top	
SPG	Sprague	Conoco	2445 (8019)	Mountain Top	
TGC	The Girls' Claims	Encana	2527 (8288)	Mountain Top	
SQS	Square S	DOW	2676 (8777)	Mountain Top	

Table 2. Seed mix used in the Pipeline experiment

Scientific Name	Common Name	PLS (lbs/ ac)	Live seeds/ m^2
forbs			
Achillea millifolium	western yarrow	0.10	67
Erigonum umbellatum	sulfur flower buckwheat	1.03	53
Hedysarum boreale	Utah sweetvetch	0.44	11
Heterotheca villose	hairy golden aster	1.11	137
Linum lewisii	Lewis flax	0.38	28
Packera multilobata	multi-lobed groundsel	0.10	67
Penstemon strictus	Rocky Mountain penstemon	0.33	23
grasses			
Achnatherum lymenoides	Indian ricegrass (Nezpar)	0.83	33
Pascopyrum smithii	western wheatgrass	0.40	11
Pseudoroegneria spicata	bearded bluebunch wheatgrass P-7	0.38	11
Pseudoroegneria spicata	bearded bluebunch wheatgrass Secar	0.36	10
Elymus trachycaulus	slender wheatgrass	0.24	10
Elymus elymoides	bottlebrush squirreltail	0.45	21
Koeleria macrantha	prarie junegrass	0.18	105
Poa secunda	sandberg bluegrass	0.26	68
Stipa viridula	green needlegrass	0.68	22
shrubs			
Artemisia tridentata ssp. *Wyomingensis*	Wyoming big sagebrush	0.33	246
Atriplex canescens	fourwing saltbush	0.54	7
Atriplex confertifolia	shadscale saltbush	0.47	7
	TOTAL	**8.60**	

Table 3. Seed mix of grasses used in the Competition experiment. Cheatgrass (*Bromus tectorum*) was also seeded at 300 seeds/m2.

Scientific Name	Common Name	Variety	Seeds/ m2	PLS (kg/ha)	Seeds/ ft2	PLS (lbs/ac)
Elymus lanceolatus spp. lanceolatus	thickspike wheatgrass	Critana	150.7	4.5	14	4.0
Elymus trachycaulus spp. trachycaulus	slender wheatgrass	San Luis	150.7	5.1	14	4.5
Pascopyrum smithii	western wheatgrass	Rosana	150.7	5.8	14	5.2
		TOTAL	**452.1**	**15.3**	**42**	**13.7**

Table 4. Seed mix used in the Gulley experiment.

Scientific name	Common Name	Variety	Seeds/ m2	PLS (kg/ha)	Seeds/ ft2	PLS (lbs/ac)
forbs						
Achillia millefolium	western yarrow	VNS	183	0.3	17	0.3
Hedysarum boreale	Utah sweetvetch	Timp	22	2.1	2	1.9
Linum lewisii	lewis flax	Maple Gr.	54	0.8	5	0.7
grasses						
Achnatherum hymenoides	Indian ricegrass	Rimrock	108	3.0	10	2.7
Elymus elymoides	squirreltail	Toe Jam Ck.	108	2.5	10	2.3
Elymus lanceolatus spp. lanceolatus	thickspike wheatgrass	Critana	65	1.9	6	1.7
Elymus trachycaulus spp. trachycaulus	slender wheatgrass	San Luis	65	2.2	6	1.9
Leymus cinereus	basin wild rye	Trailhead	43	1.3	4	1.2
Pascopyrum smithii	western wheatgrass	Rosana	65	2.5	6	2.2
Pleuraphis jamesii	galleta grass	Viva	54	1.6	5	1.4
Poa fendleriana	muttongrass	VNS	323	0.7	30	0.7
Pseudoroegneria spicata spp. spicata	bluebunch wheatgrass	Anatone	108	3.9	10	3.5
shrubs						
Artemisia tridentat spp. Wyomingensis	Wyo. big sagebrush	VNS	250	0.6	23	0.5
Atriplex canescens	fourwing saltbush	VNS	32	3.3	3	3.0
Ericameria nauseosa	rubber rabbitbrush	VNS	22	0.2	2	0.2
Kraschenninnikovia lanata	winterfat	VNS	16	0.6	1.5	0.5
		TOTAL	**1514**	**28**	**141**	**25**

Table 5. Seed mix used in the Mountain Top experiment.

Scientific Name	Common Name	Variety	Seeds/ m^2	PLS (kg/ha)	Seeds/ ft^2	PLS (lbs/ac)
forbs						
Achillia millefolium	western yarrow	Eagle Mtn.	161	0.3	15	0.2
Hedysarum boreale	Utah sweetvetch	Timp	15	1.5	1	1.3
Penstemon palmeri	palmer penstemon	Cedar	215	1.7	20	1.5
Penstemon strictus	Rocky Mtn. penstemon	Bandera	108	1.7	10	1.5
grasses						
Bromus marginatus	mountain brome	Garnet	54	3.8	5	3.4
Elymus lanceolatus spp. lanceolatus	thickspike wheatgrass	Critana	22	0.6	2	0.6
Elymus trachycaulus spp. trachycaulus	slender wheatgrass	San Luis	65	2.2	6	1.9
Nassella viridula	green needlegrass	Lowdorm	43	1.2	4	1.0
Poa fendleriana	muttongrass	VNS	215	0.5	20	0.4
Pseudoroegneria spicata spp. spicata	bluebunch wheatgrass	Anatone	65	2.3	6	2.1
shrubs						
Artemisia cana	silver sage	VNS	323	1.3	30	1.2
*Artemisia tridentata spp. vaseyana**	mtn. big sagebrush	VNS	250	0.6	23	0.5
Ericameria nauseosa	rubber rabbitbrush	VNS	22	0.2	2	0.2
		TOTAL	**1556**	**17.8**	**145**	**15.9**

Table 6. Seed mixes used in the Strategy Choice experiment. Species noted as "drill seeded" were drill seeded in plots with a flat surface. In plots with a rough surface, all seed was broadcast.

				high comp.mix		low comp.mix	
	Scientific Name	**Common Name**	**Variety**	**seeds/ m^2**	**PLS (kg/ha)**	**seeds/ m^2**	**PLS (kg/ha)**
drill seeded	***forbs***						
	Hedysarum boreale	Utah sweetvetch	Timp	22	2.1	22	2.1
	grasses						
	Achnatherum hymenoides	Indian ricegrass	Rimrock	65	1.8	11	0.3
	Elymus lanceolatus spp. lanceolatus	thickspike wheatgrass	Critana	65	1.9		
	Elymus trachycaulus spp. trachycaulus	slender wheatgrass	San Luis	75	2.5	11	0.4
	Pascopyrum smithii	western wheatgrass	Rosana	65	2.5	5	0.2
	Pleuraphis jamesii	galleta grass	Viva	75	2.2		
	Poa fendleriana	muttongrass	VNS			54	0.1
	Pseudoroegneria spicata spp. spicata	bluebunch wheatgrass	Anatone			22	0.8
	shrubs						
	Atriplex canescens	fourwing saltbush	VNS CO	11	1.1	11	1.1
broadcast seeded	***forbs***						
	Achillia millefolium	western yarrow	VNS	129	0.2	129	0.2
	Erigeron speciosis	oregon daisy	VNS			323	0.9
	Eriogonum umbellatum	sulphur flower buckwheat	VNS	108	2.3	108	2.3
	Heterotheca villosa	hairy golden aster	VNS			215	1.3
	Linum lewisii	lewis flax	Maple Gr.	54	0.8	54	0.8
	Packera multilobata	many-lobed grounsel	VNS			215	1.3
	Penstemon cyanocaulis	bluestem penstemon	VNS	108	0.7	108	0.7
	grasses						
	Koeleria macrantha	prairie junegrass	VNS			54	0.1
	shrubs						
	Krascheninnikovia lanata	winterfat	VNS	22	0.8	22	0.8
	Artemisia tridentat spp. Wyomingensis	Wyoming big sagebrush	VNS	253	0.6	253	0.6
		GRASS TOTAL		***344***	***9.8***	***156***	***1.7***
		FORB TOTAL		420	5.6	1173	8.7
		SHRUB TOTAL		285	2.2	285	2.2
		OVERALL TOTAL		**1049**	**17.6**	**1614**	**12.6**

Table 7. Seed mixes used in the Strategy Choice experiment. Species noted as "drill seeded" were drill seeded in plots with a flat surface. In plots with a rough surface, all seed was broadcast.

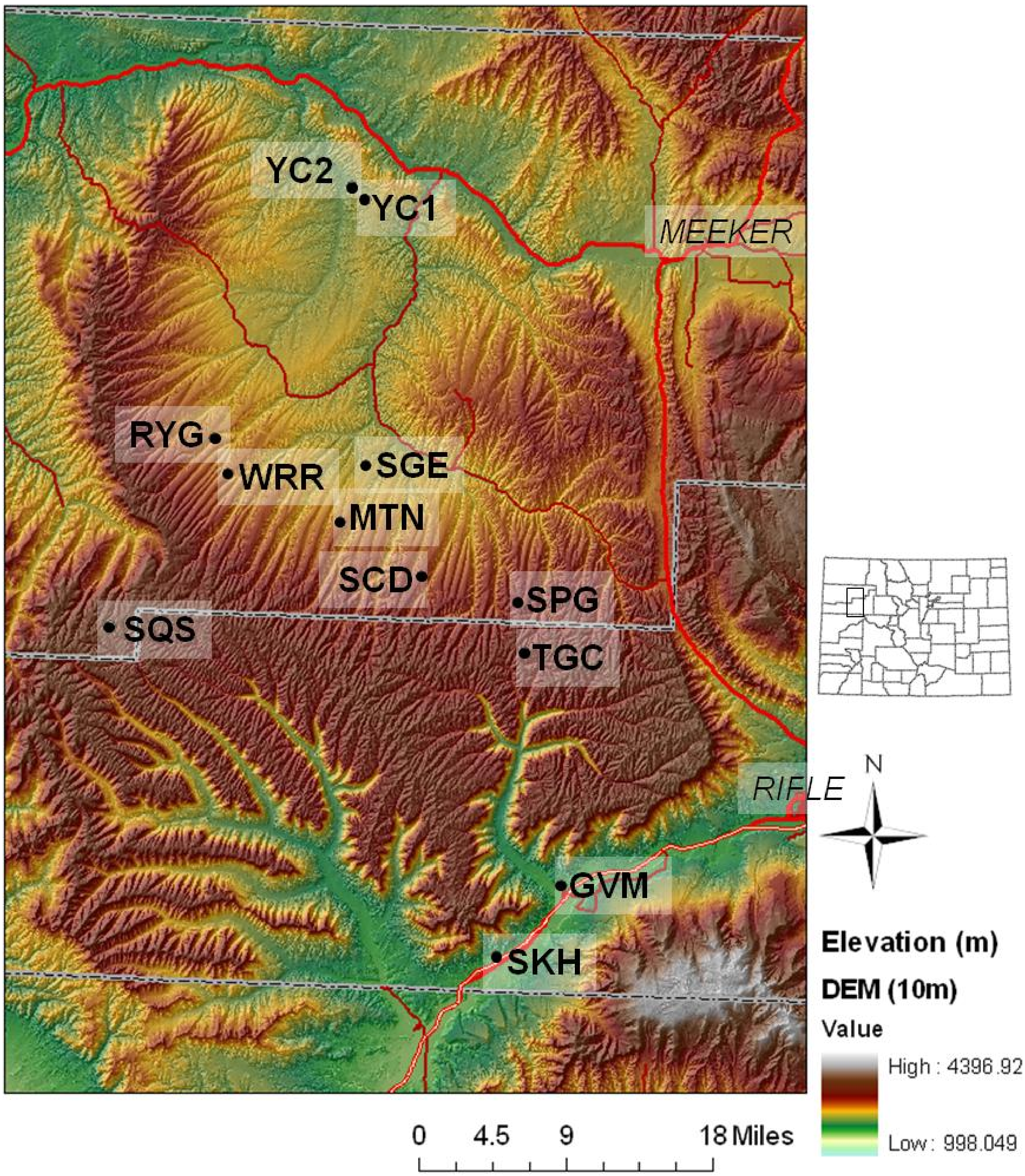


Figure 1. Locations of the 12 research sites in Rio Blanco and Garfield counties, Colorado.

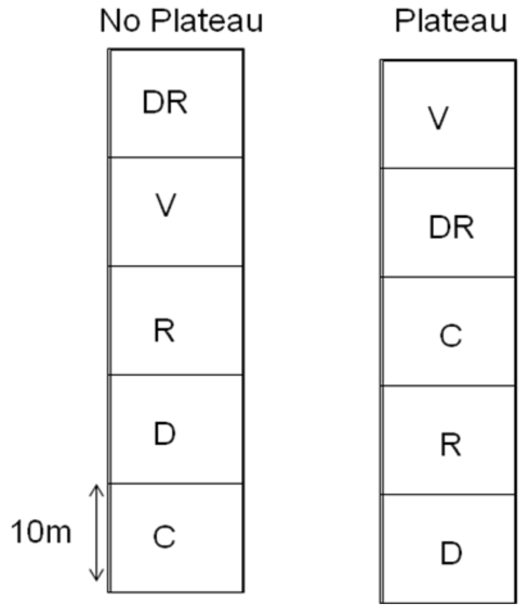


Figure 2. Layout of the Pipeline experiment at one of 6 sites. D = disked, R = rolled, DR = disked and rolled, V = rolled with a vibratory drum compactor, C = control.

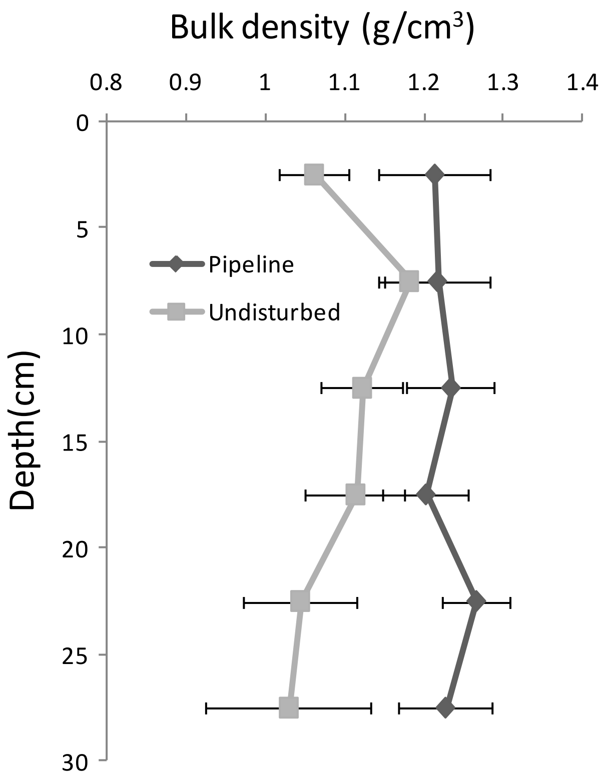


Figure 3. Bulk density of disturbed and undisturbed areas in the Pipeline experiment. Error bars = SE.

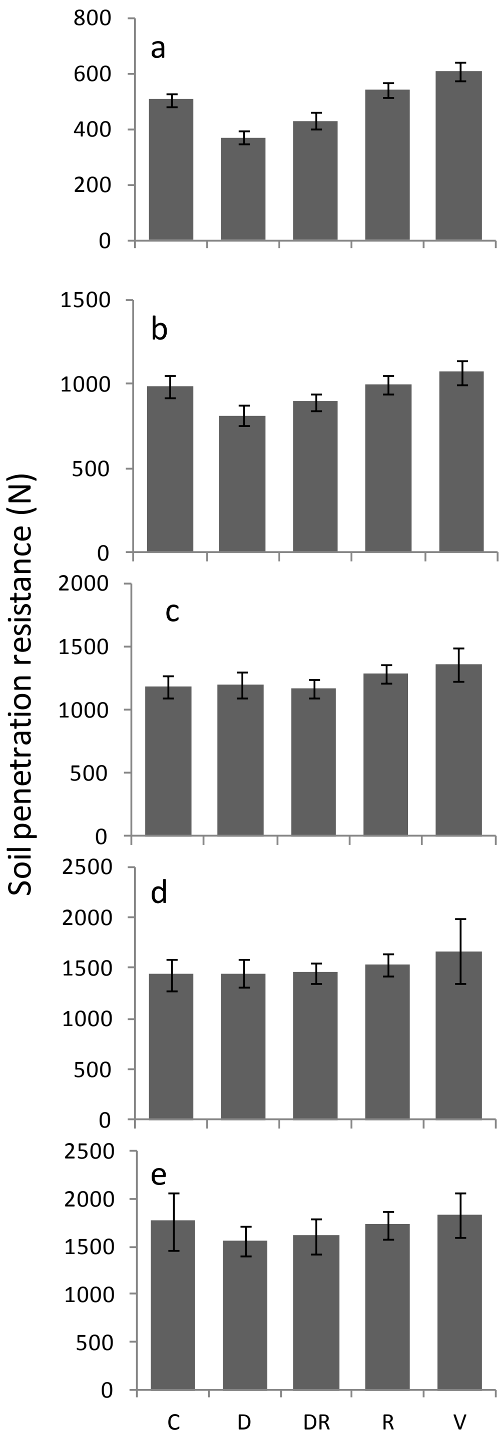


Figure 4. Pipeline experiment soil penetration resistance by soil tillage treatment at depths of: (a) 4- 9 cm; (b) 10- 14 cm; (c) 15- 19 cm; (d) 20- 24 cm and (e) 25-29 cm. C= control, D= disked, DR= disked and rolled, R= rolled, and V= rolled with vibratory drum compactor. Error bars= SE for 12 plots, 2 at each of 6 sites. Note differing y-axis scales.

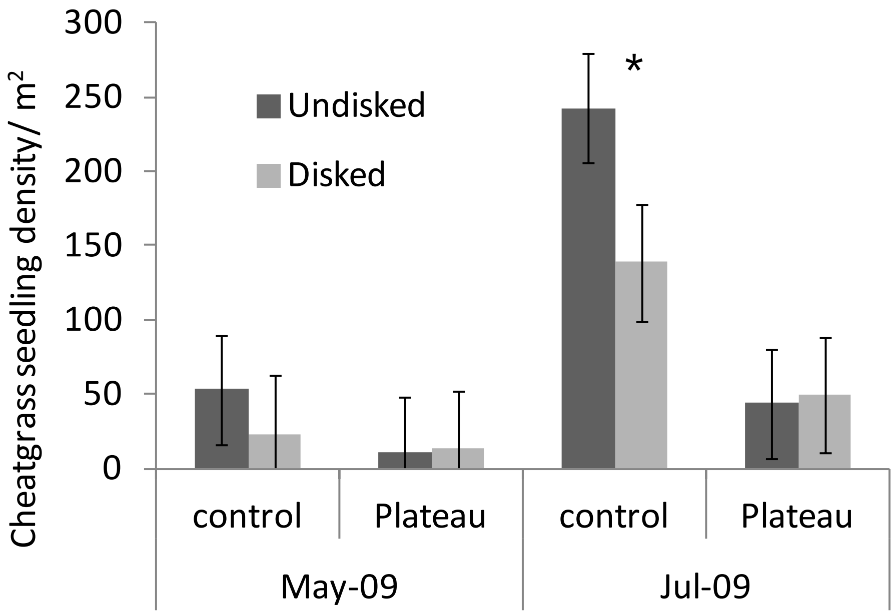


Figure 5. Density of cheatgrass seedlings in May and July of 2009 by disking and Plateau treatment in the Pipeline experiment. Bars represent least square means over sites and non-significant treatments. Error bars = SE.

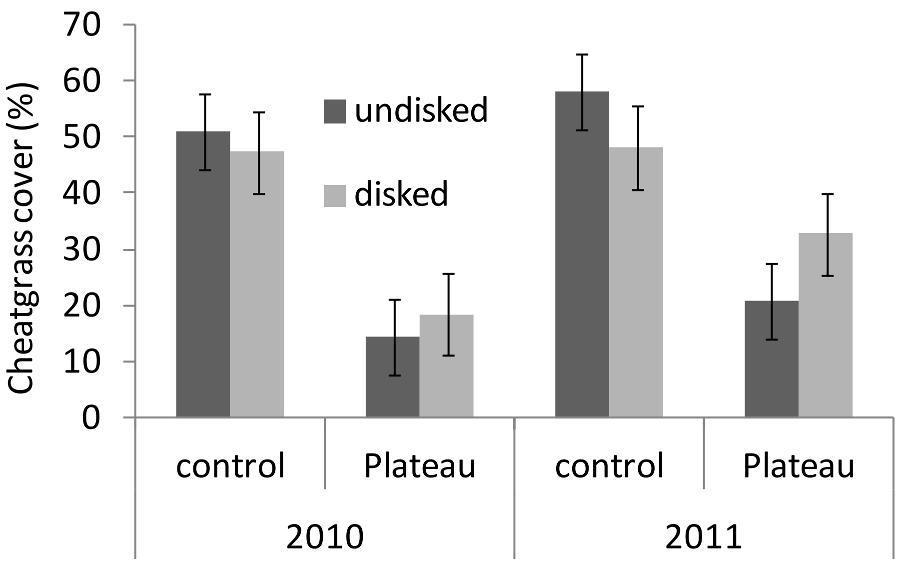


Figure 6. Cover of cheatgrass in 2010 and 2011 in response to disking and Plateau treatment in the Pipeline experiment. Bars represent least square means over sites and non-significant treatments. Error bars= SE.

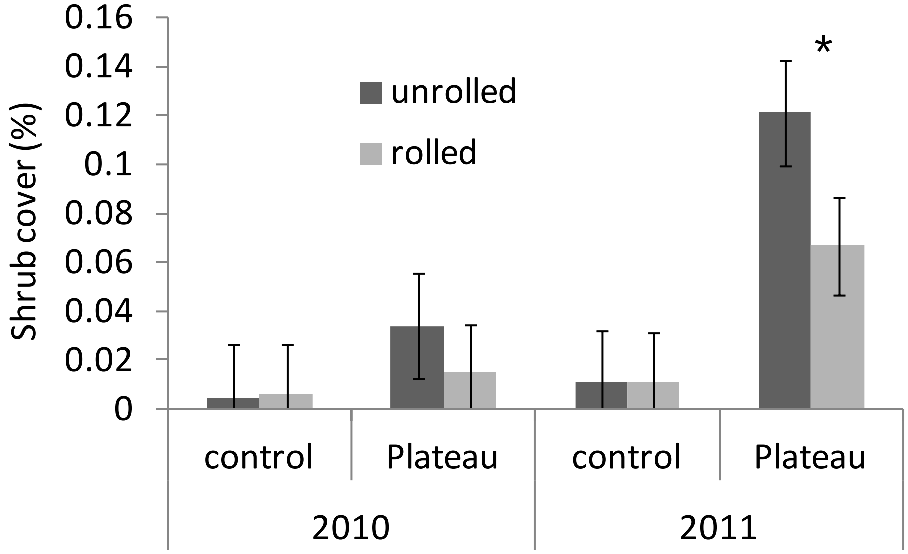


Figure 7. Cover of shrubs in 2010 and 2011 by rolling and Plateau treatment in the Pipeline experiment. Bars represent least square means over sites and non-significant treatments. Error bars= SE.

No Plateau

Plateau

Figure 8. Visual comparison of plots where Plateau was applied vs. no Plateau applied at the RYG site in the Pipeline experiment, three years post-application.

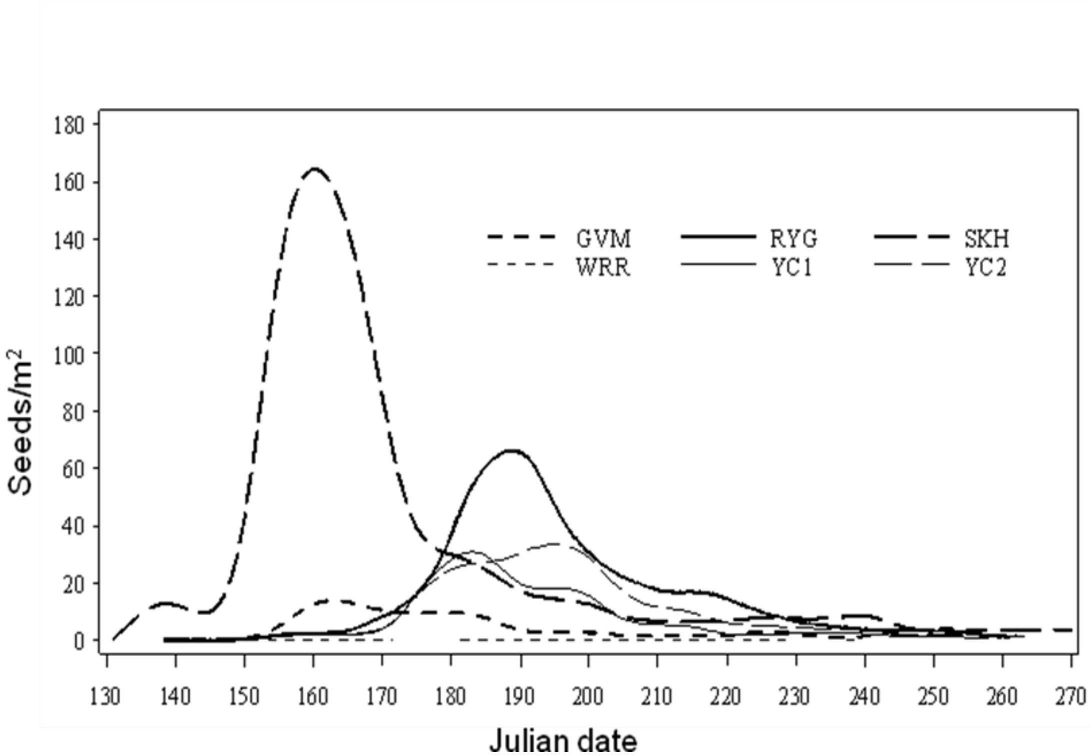


Figure 9. Prevalence of cheatgrass seeds between May and September in undisturbed locations near the 6 study sites of the Pipeline experiment. Data are averages over 3 years, 2009-11. The data was smoothed using a cubic spline with an nn value of 15 (Reinsch 1967).

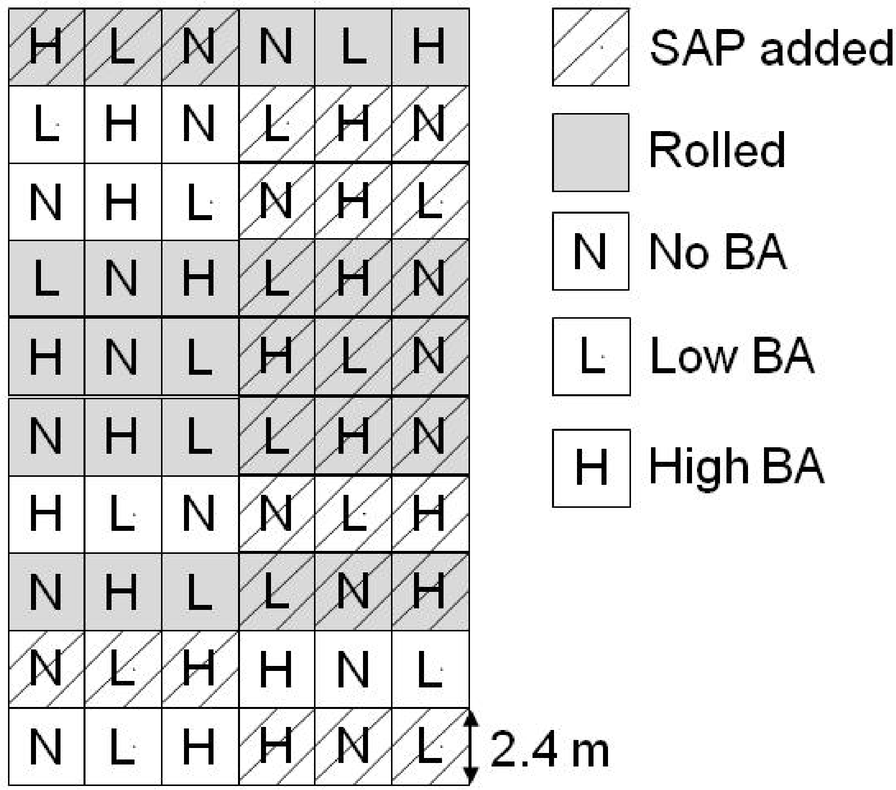


Figure 10. Layout of the Competition experiment at one of 2 research sites. SAP super- absorbent polymer. BA = soil binding agent.

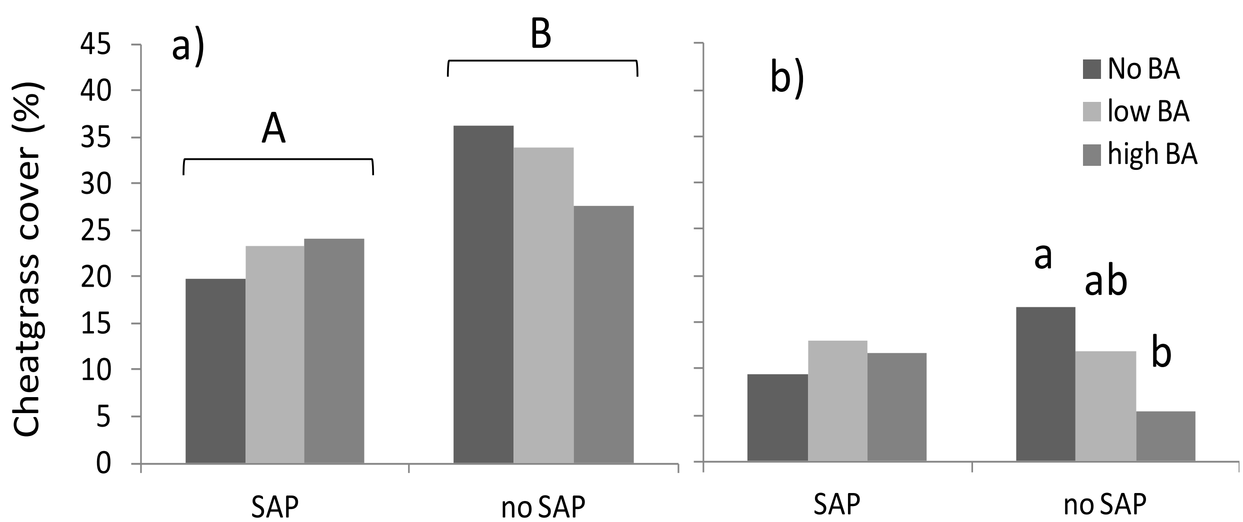


Figure 11. Cheatgrass cover in 2011 in response to 2009 addition of super-absorbent polymer (SAP) and binding agent (BA) at a) Wagon Road Ridge and b) Sagebrush 69 Road in the Competition experiment. Error bars = SE. Bars not sharing a letter denote significant differences within a site at α = 0.05.

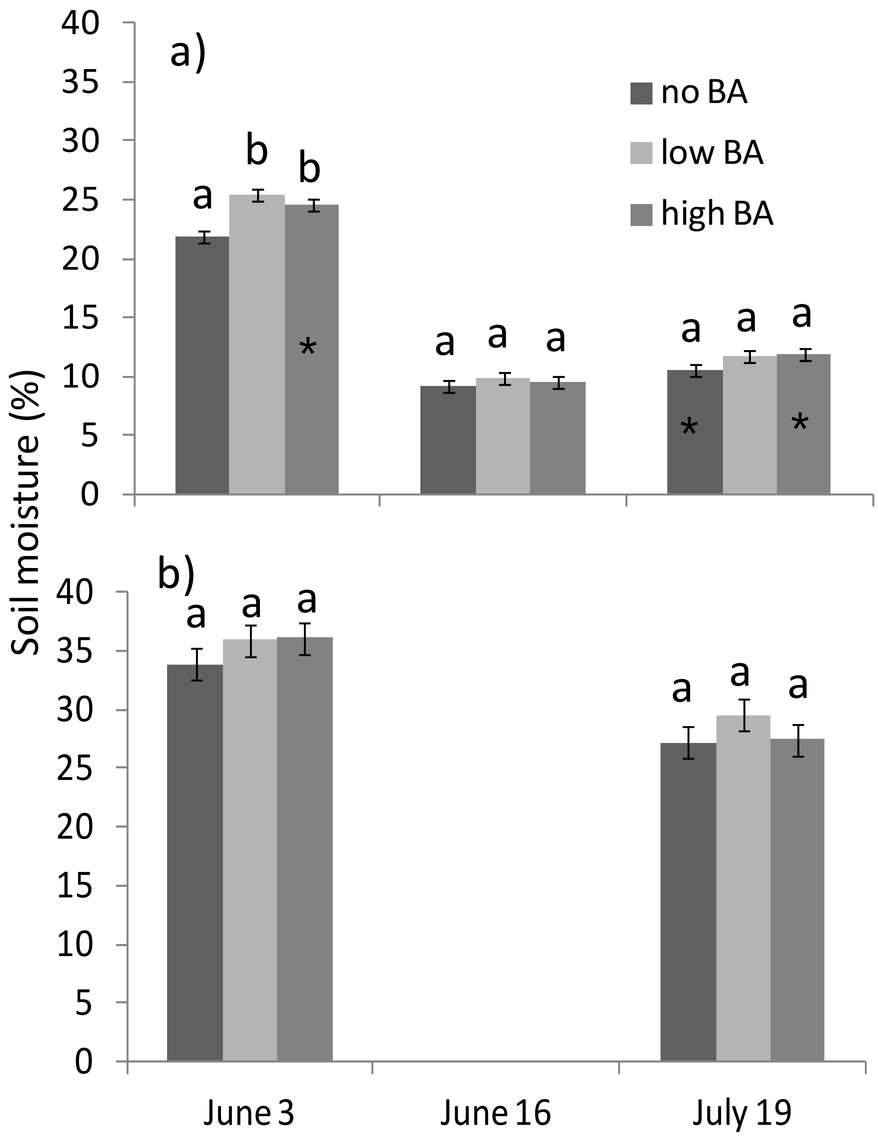


Figure 12. Effect of 3 levels of soil binding agent (none, low, high) on volumetric soil moisture at a) WRR and b) SGE study sites in 2011 in the Competition experiment. Bars not sharing a letter denote significant differences within a site and date at α = 0.05. Error bars = SE. Stars within bars indicate dates and BA levels for which an interaction between SAP and Rolling treatment occurred for soil moisture (Figure 11)

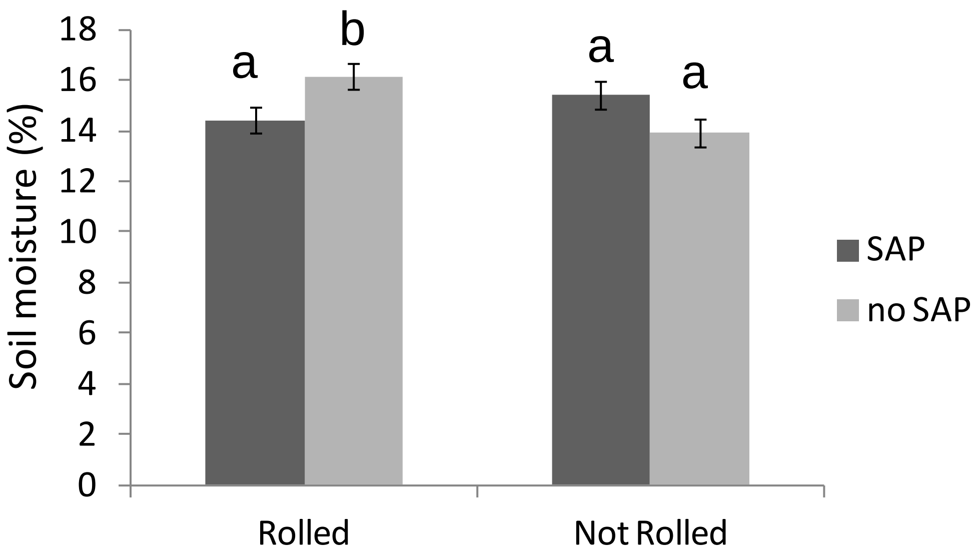


Figure 13. Effect of Rolling and super-absorbent polymer (SAP) addition on soil moisture at WRR, averaged over dates and binding agent treatments in the Competition experiment. Bars not sharing a letter denote significant differences within a site and date at α = 0.05. Error bars = SE.

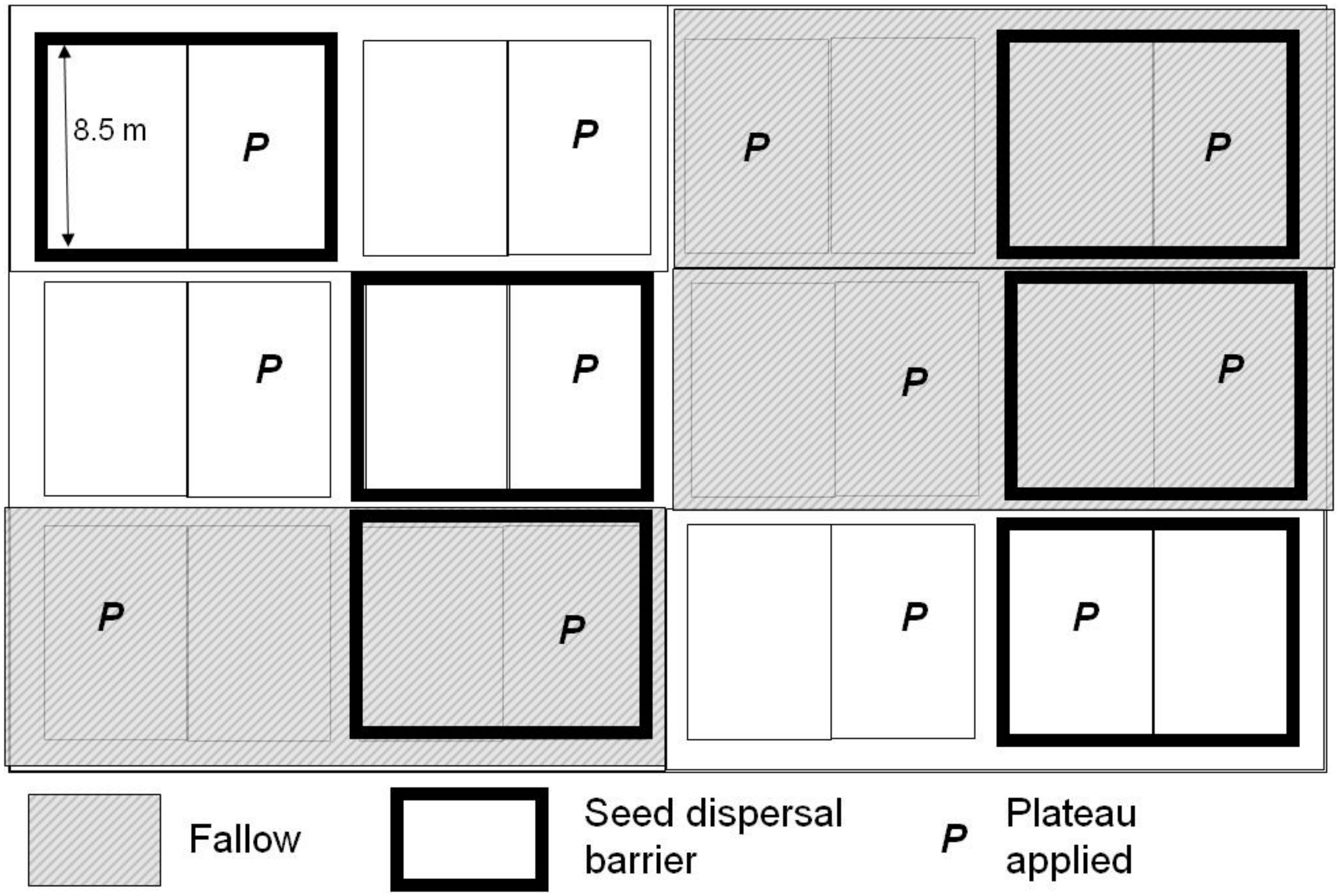


Figure 14. Layout of the Gulley experiment at one of 4 research sites.

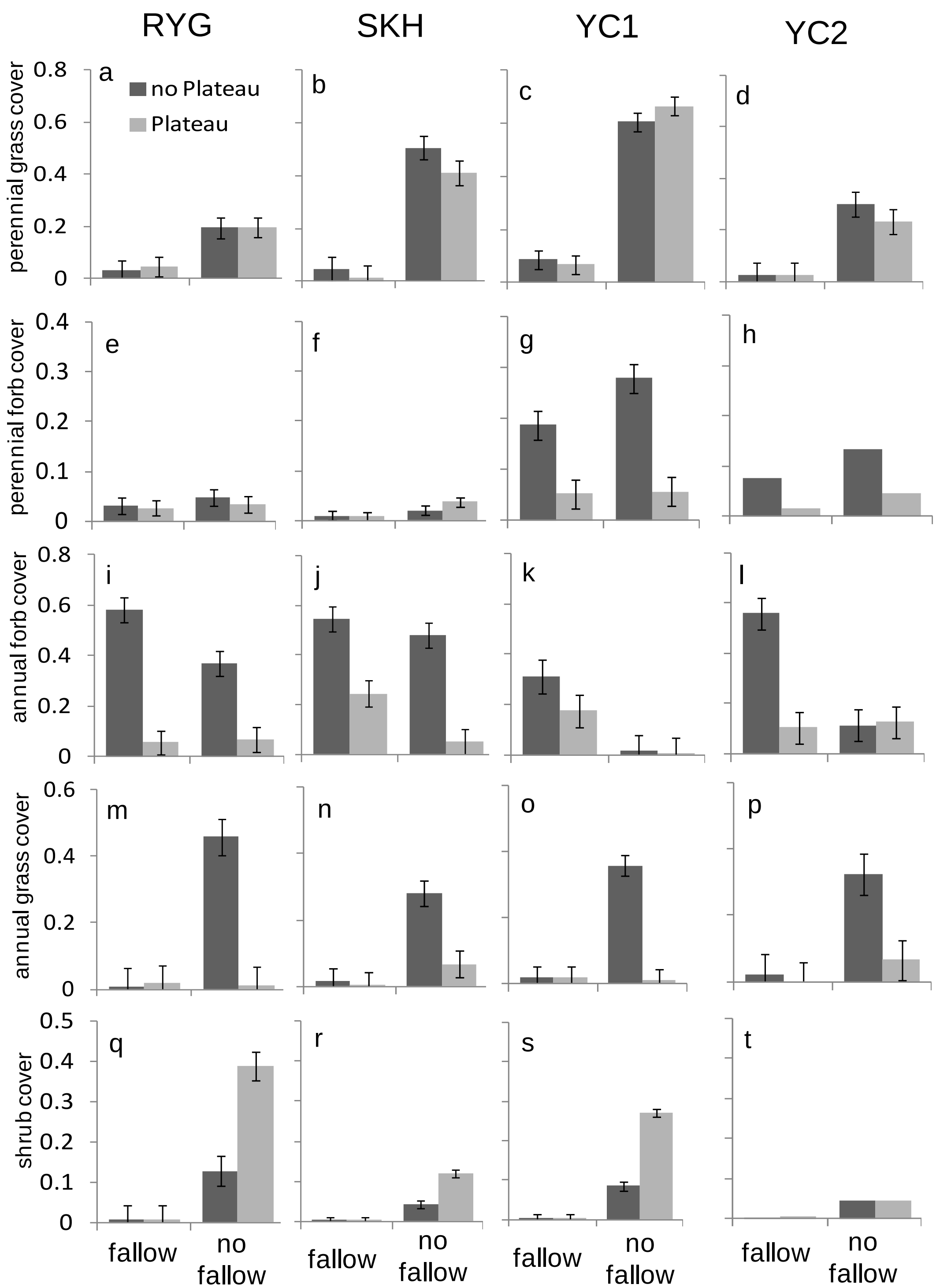


Figure 15. Effect of fallowing and Plateau treatment on cover of perennial grasses (a-d), perennial forbs (e-h), annual forbs (i-l), annual grasses (m-p), and shrubs (q-t) at four sites: RYG (a, i, m, and q), SKH (b, j, n, r), YC1 (c, k, o, s) and YC2 (q, r, s, t) in the Gulley experiment. In l, the effect of Plateau on annual forbs depended on barrier treatment; see Figure 16b. Error bars = SE of least square means taken over brush treatment.

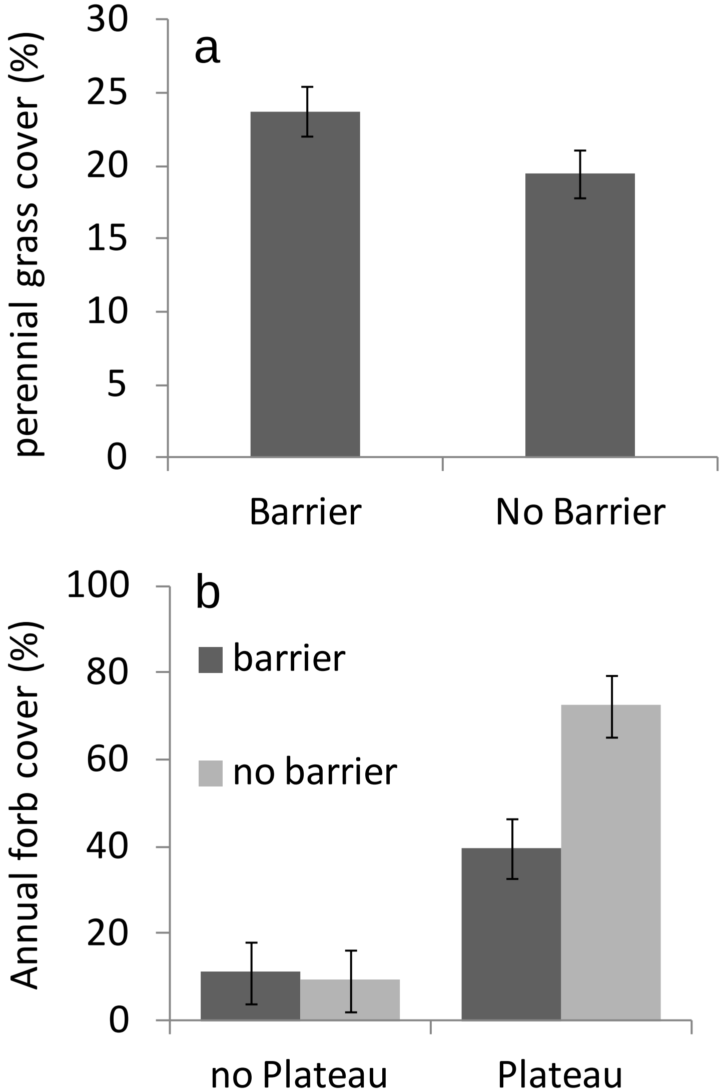


Figure 16. Effect of barrier treatment in the Gulley experiment on a) perennial grasses, across sites and other treatments and b) annual forbs in fallowed plots at the YC2 site. Error bars = SE of least square means.

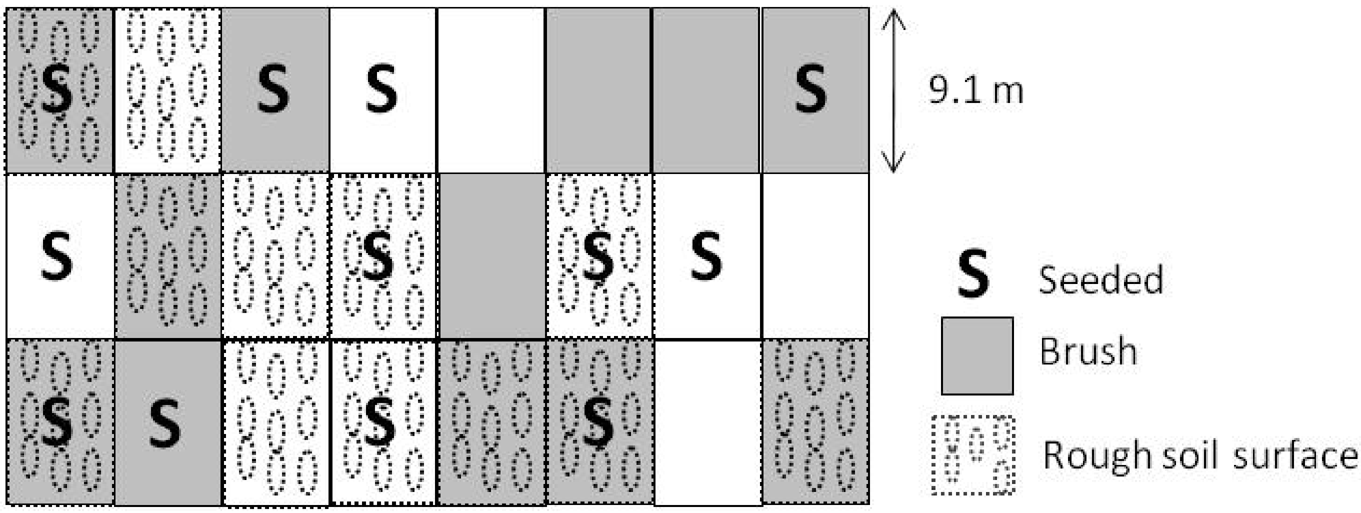


Figure 17. Layout of the Mountain Top experiment at one of four research sites.

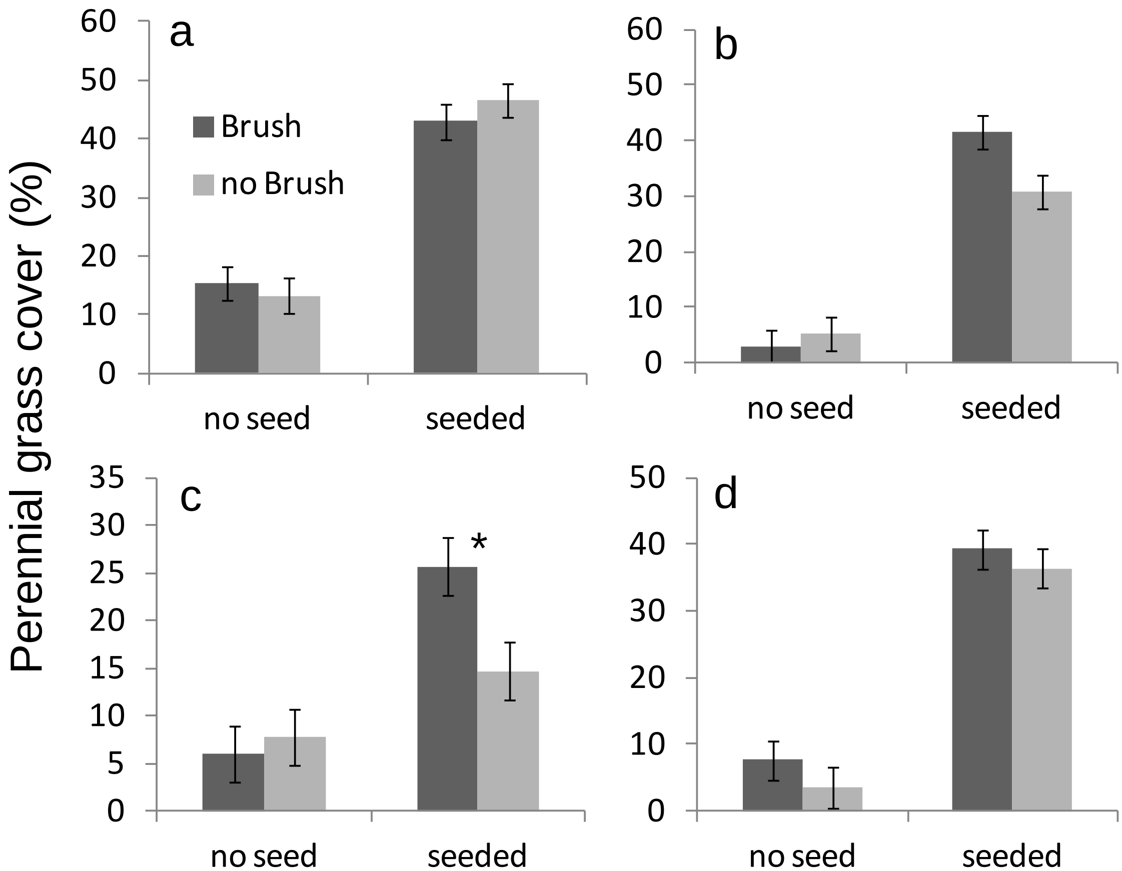


Figure 18. Effect of brush mulch and seeding on cover of perennial grasses in the Mountain Top experiment at 4 study sites: a) SCD, b) SPG, c) SQS, and d) TGC. Error bars = SE for least square means taken over soil surface treatment.

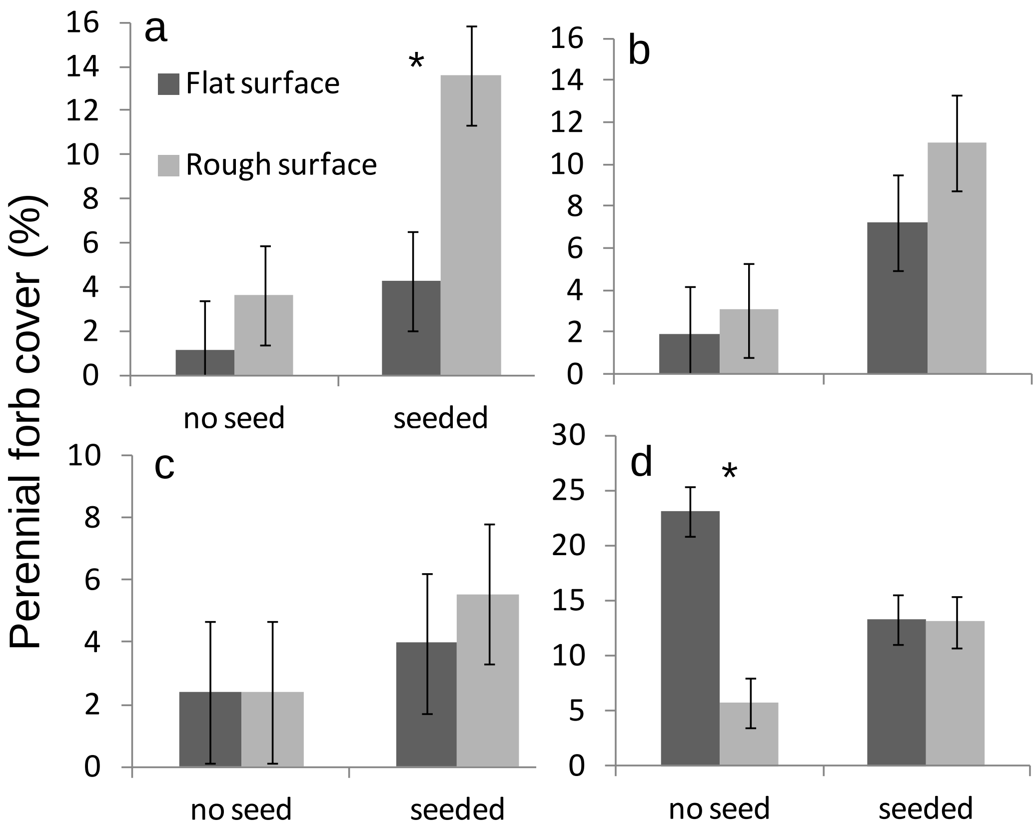


Figure 19. Effect of seeding and soil surface treatment on perennial forb cover in the Mountain Top experiment at four study sites: a) SCD, b) SPG, c) SQS, and d) TGC. Error bars = SE for least square means taken over brush treatment.

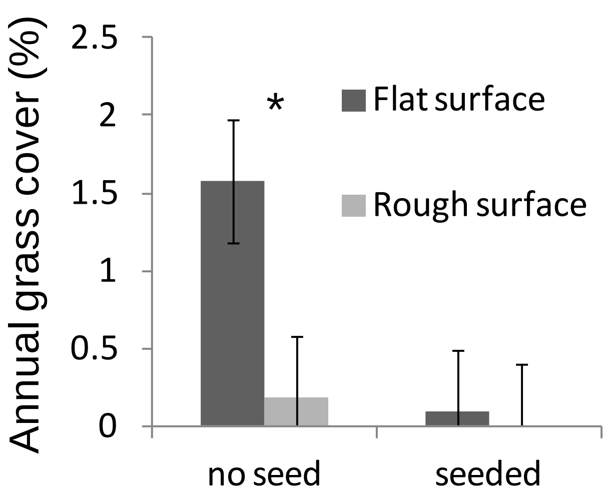


Figure 20. Effect of seeding and soil surface treatment on annual grass cover in the Mountain Top experiment at the SCD site. Error bars = SE for least square means taken over the brush treatment.

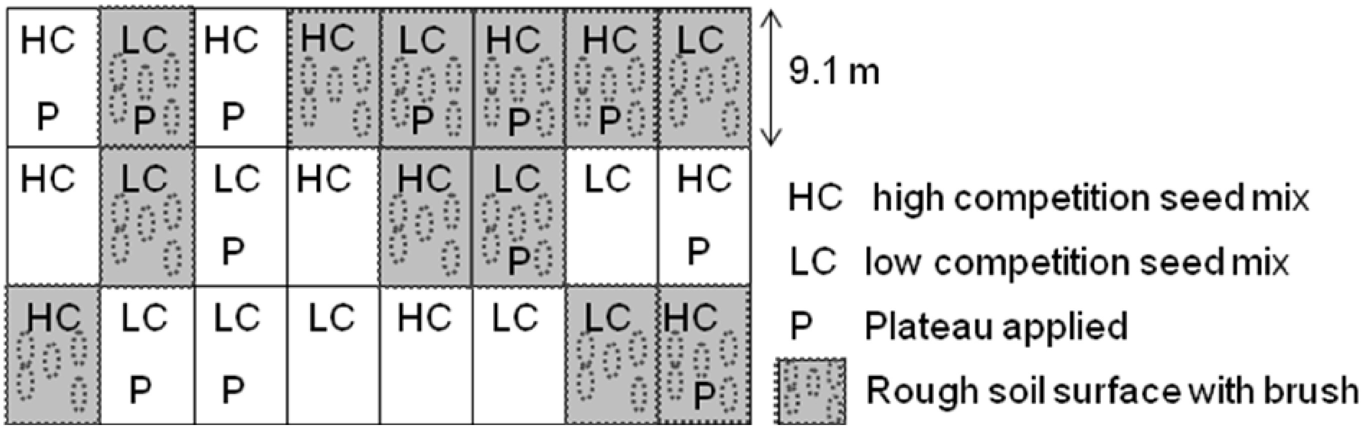


Figure 21. Layout of the Strategy Choice experiments at one of 2 sites where the full experiment was implemented. At 2 additional sites, a reduced form of the experiment lacking the Plateau treatment was implemented.

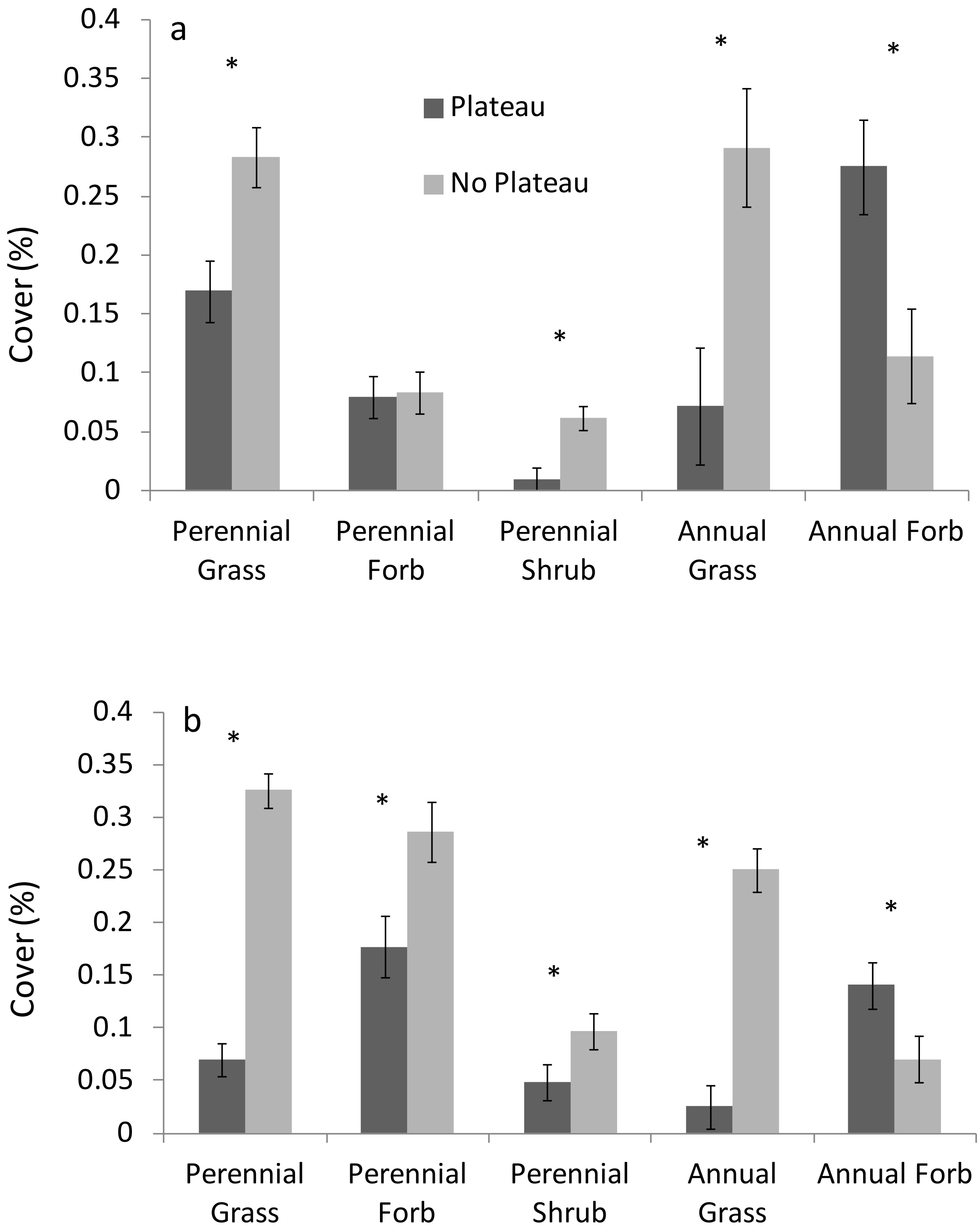


Figure 22. Effect of Plateau treatment on cover of functional groups at a) GVM and b) MTN in the Strategy Choice experiment. Error bars = SE of least square means taken over seed mix and soil surface treatments. Stars represent significantly different means at the $\alpha = 0.05$ level.

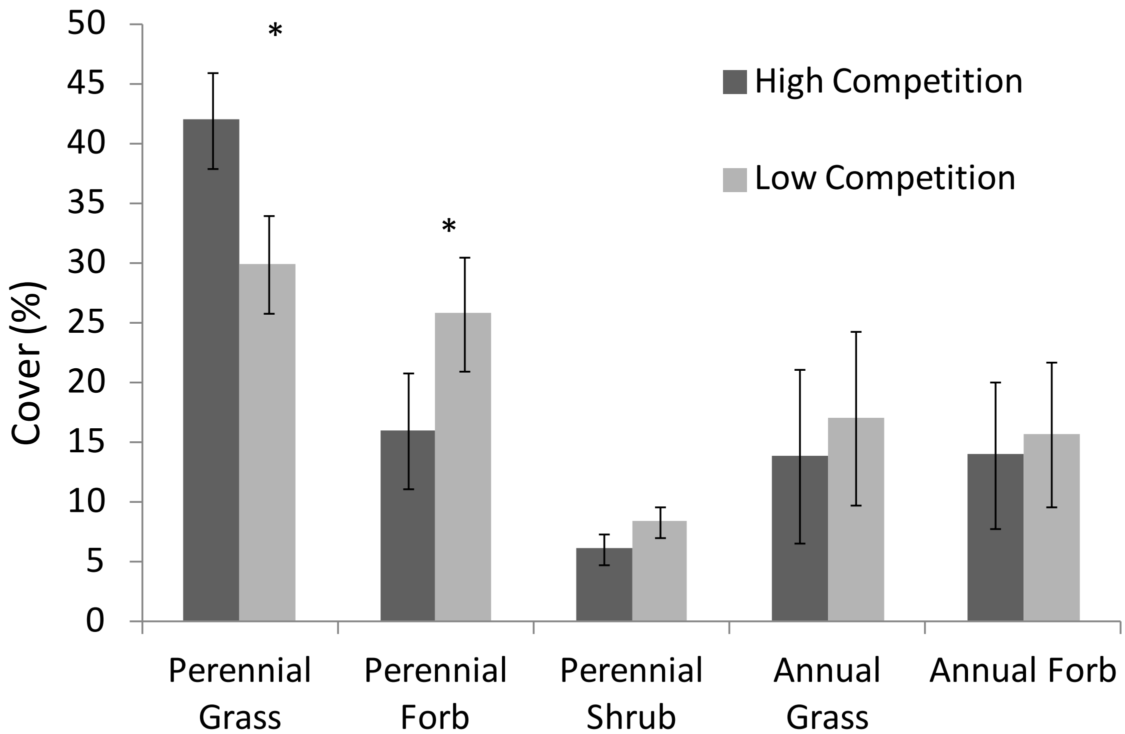


Figure 23. Effect of seed mix treatment on cover of functional groups in the Strategy Choice experiment (sites combined). Error bars = SE of least square means taken over soil surface treatments, with Plateau plots excluded. Stars represent significantly different means at the α = 0.05 level.

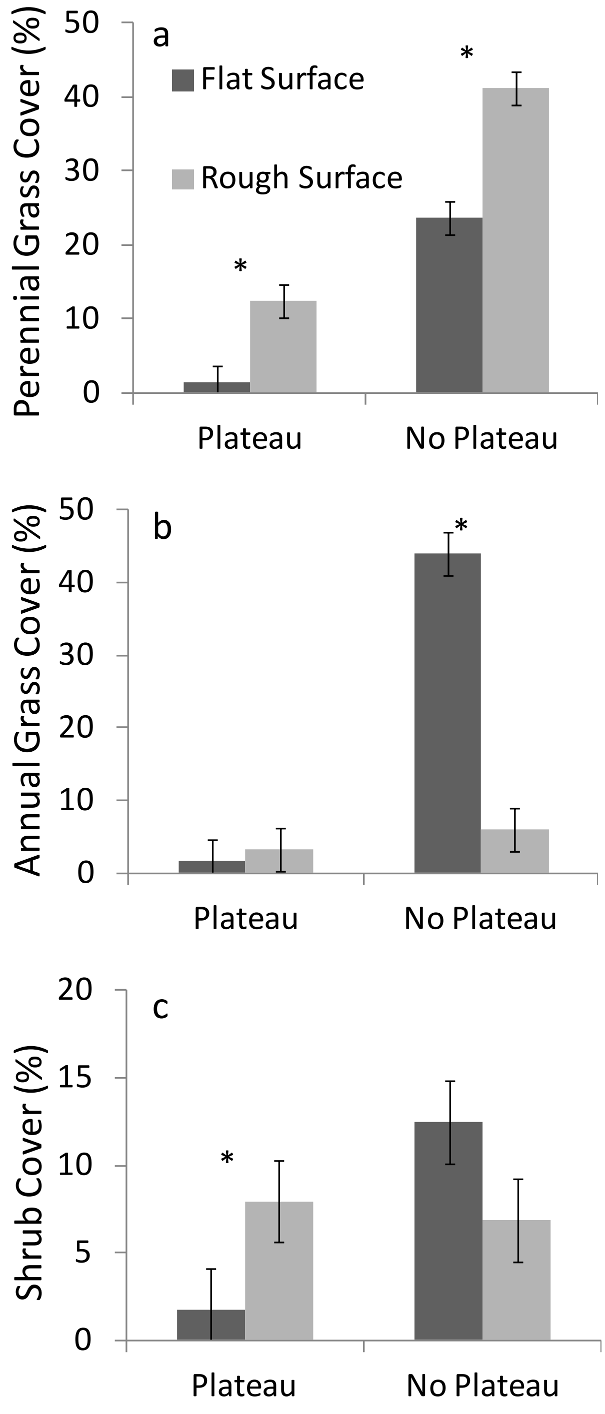


Figure 24. Effect of Plateau and soil surface treatment at the MTN site in the Strategy Choice experiment on a) perennial grass cover, b) annual grass cover, and c) shrub cover. Error bars = SE of least square means taken over seed mix treatment. Stars represent significantly different means at the $\alpha = 0.05$ level. For annual grass and shrub cover, an interaction between Plateau and surface treatment was significant. For perennial grass cover, only main effects of Plateau and surface treatment were significant.

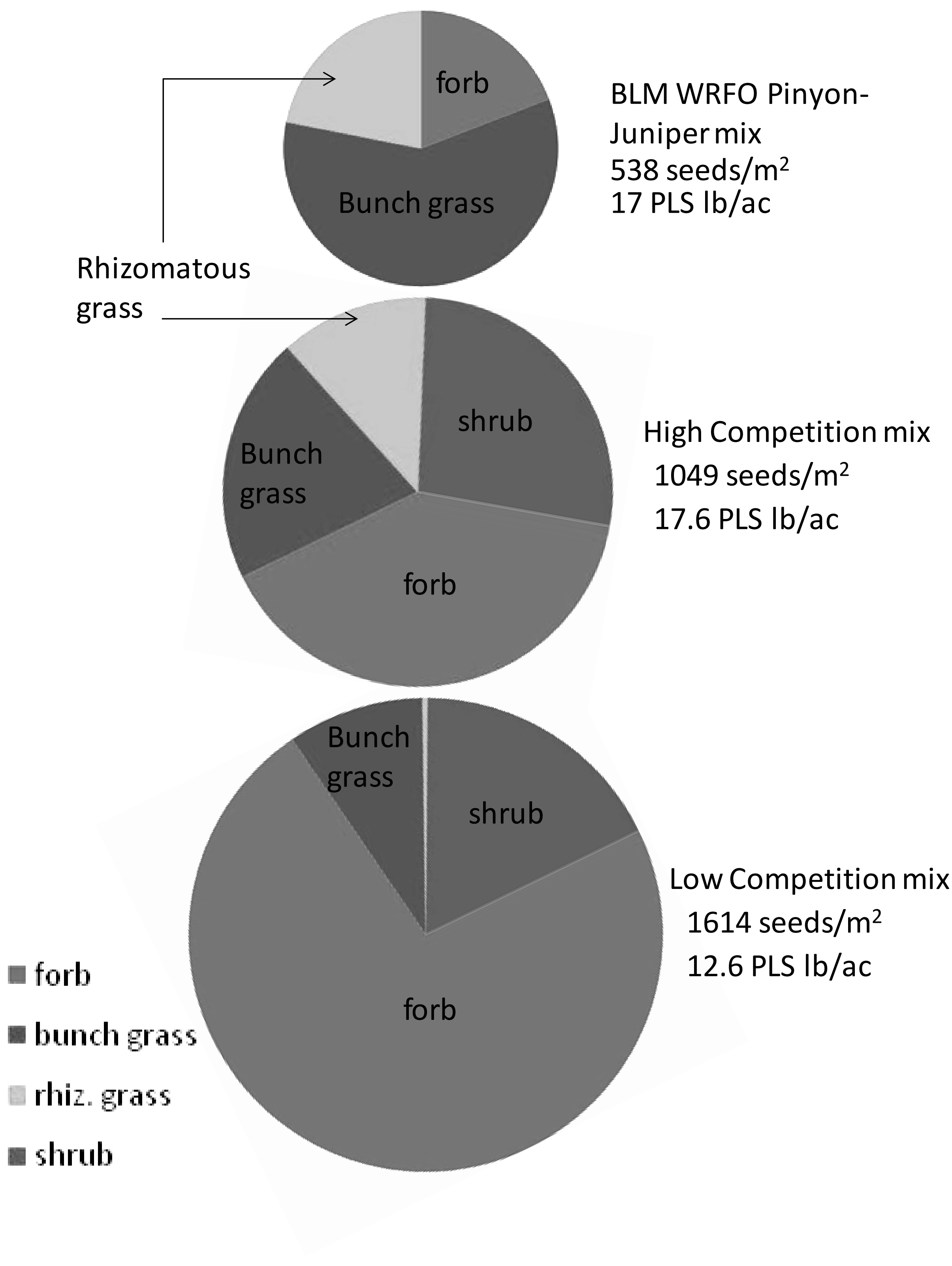


Figure 25. Comparison of the seed mixes used in the Strategy Choice experiment, with a commonly used Bureau of Land Management (BLM) mix for reference. The size of the pie charts is proportional to the number of seeds in the mix.

Flat

Rough

Figure 26. Visual comparison of flat surface versus rough surface plots in the Strategy Choice experiment at the MTN site. Both plots received the Low Competition seed mix and did not receive Plateau herbicide.

APPENDIX 1: CHEATGRASS PROPAGULE PRESSURE METHODS

The study sites chosen for these experiments had cheatgrass present in varying quantities. Prior work has shown that the quantity of weed seeds, or "propagule pressure", is important in understanding the outcome of revegetation (DiVittorio et al. 2007). Therefore, cheatgrass propagule pressure is an important covariate for the experiments. We quantified cheatgrass propagule pressure at the 8 sites where cheatgrass was present: SKH, GVM, RYG, YC1, YC2, WRR, SGE, and MTN.

We quantified cheatgrass propagule pressure at each study site using 0.1 m^2 seed rain traps constructed of posterboard covered with Tree Tanglefoot (The Tanglefoot Company, Grand Rapids, MI), a sticky resin (Figure A1-1). Eight traps were set in systematically chosen locations in undisturbed vegetation surrounding each site. Cheatgrass seeds were counted and removed from traps a mean of every 12 days from mid-May to late September, 2009- 2011. Tanglefoot was reapplied as necessary to ensure a sticky surface. Total growing season cheatgrass propagule pressure (seeds/m^2) was calculated by summing the seeds on each trap, and then taking an average for the site.

Figure A1-1. A seed trap.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	0663	: Terrestrial Species Conservation
Task No.:	N/A	: Examining the effectiveness of mechanical treatments as a restoration technique for mule deer habitat
Federal Aid Project No.	N/A	

Period Covered: September 1, 2011 – August 31, 2012

Author: D. B. Johnston

Personnel: M. Pastiche and G. Stephens, Colorado State University; L. Belmonte, Bureau of Land Management; B. deVergie and J.C. Rivale, CPW

ABSTRACT

The pinyon-juniper (PJ) habitat type has been expanding in the western United States and managers often seek methods of thinning or removing pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) trees in order to improve habitat for big game. Because prescribed fire is difficult or impossible to implement in many areas, mechanical tree removal has become common. Several methods of mechanical removal are available, including ship anchor chaining, roller-chopping, and hydro-axing. These differ in cost as well as in the type of woody debris produced. In order to compare the effectiveness of these three removal methods in PJ forests, a replicated field study was implemented in the Magnolia region of the Piceance Basin, Rio Blanco County, and Colo. Seven parcels were treated with each of the treatment types, and half of each parcel was seeded with a native seed mixture emphasizing palatable shrubs. Mechanical treatments were completed in the fall of 2011. Vegetation will monitored yearly for the next two to three growing seasons in order to determine which treatment produces the most desirable plant community. The effectiveness of seeding within each treatment type will be analyzed.

WILDLIFE RESEARCH REPORT

EXAMINING THE EFFECTIVENESS OF MECHANICAL TREATMENTS AS A RESTORATION TECHNIQUE FOR MULE DEER HABITAT

DANIELLE B. JOHNSTON

PROJECT OBJECTIVES

1. Assess vegetation response to removal of pinyon and juniper trees via three different mechanical treatments: ship anchor chaining (with two passes), hydro-axing, and roller chopping.
2. Assess response of desired shrubs to seeding within each mechanical treatment. Focal shrubs include chokecherry (*Prunus virginiana*), Saskatoon serviceberry (*Amelanchier alnifolia*), Utah serviceberry (*Amelanchier utahensis*), mountain mahogany (*Cercocarpus montanus*), bitterbrush (*Purshia tridentata*), and winterfat (*Kraschenninnikovia lanata*).
3. Compare cost-effectiveness of the three mechanical treatments.
4. Examine cost-effectiveness of seeding shrubs in the three mechanical treatments.

SEGMENT OBJECTIVES

1. Select study locations and obtain necessary permits for ground-disturbing activities.
2. Perform mechanical treatments.
3. Apply seed.

INTRODUCTION

Pinyon-juniper (PJ) woodlands play an important role in mule deer ecology. Pinyon pine (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*) and the associated understory shrub species such as mountain mahogany (*Cercocarpus montanus*), antelope bitterbrush (*Purshia tridentata*) and big sagebrush (*Artemisia tridentata*) are key to winter survival (Hansen and Dearden 1975, Heffelfinger 2006). Deer strongly select for this habitat type because of the escape and thermal cover provided by pinyon and juniper trees (Anderson et al.). However, pinyon-juniper habitats occasionally lack understory and may provide very little forage (Bender et al. 2007). It has been shown that increasing nutrition in poor quality pinyon-juniper winter range can increase deer populations in western Colorado (Bishop et al. 2009). Therefore, creating patches of habitat types with higher nutritional value within pinyon-juniper stands is a desirable management objective for mule deer.

The pinyon-juniper habitat type has increased in many parts of western North America over the past 100 years (Miller and Rose 1999, Schaffer et al. 2003, Bradley and Fleishman 2008). Disruption of natural fire regimes, overgrazing, and invasion by weedy species have led to a wide array of management problems. Of particular concern are overgrown stands of PJ that have allowed the overstory to shade out understory plant species. Because of the proximity to infrastructure and human activity, as well as the lack of continuous understory fuels, prescribed fire is often eliminated from consideration as a management tool. Alternatives to natural restoration do exist in the form of mechanically created disturbances, which can open up the canopy and reduce competition (Fairchild 1999). Over the past 50 years, it has been demonstrated in Utah that the mechanical modification of PJ, together with subsequent seeding of selected species, can be an effective restoration technique (Fairchild 1999). However, several mechanical removal methods exist, and little information is available to determine which method is most cost-effective.

Determining the most-effective approach to mechanical pinyon-juniper removal is imperative due to the current need for habitat improvements which may offset the impacts of oil and gas development. Recent studies have shown that oil and gas activities can influence deer populations due to both direct habitat loss and to indirect effects due to disturbance (Sawyer et al. 2006). In western North America, oil and gas development commonly coincides with pinyon-juniper habitats. A Colorado Parks and Wildlife research project is currently examining the degree to which oil and gas impacts on deer populations may be mitigated by removing pinyon and juniper trees (Anderson 2011). If mitigation proves successful, then removal of pinyon-juniper trees may be widely prescribed as a mitigation treatment for oil and gas impacts.

Mechanical treatments in PJ forests differ in the size of woody litter produced and in the degree of soil disturbance created. Chaining is a technique by which trees are removed by dragging a ship anchor chain between two bulldozers. Trees are uprooted and left intact and the action of uprooting creates a great degree of soil disturbance (Cain 1972). Roller chopping is a technique where a heavy rotating drum with protruding steel plates is pulled behind a bulldozer. The bulldozer knocks the trees over and the drum chops them into large pieces. The action of the roller chopper creates soil disturbance, though to a lesser degree than does chaining. Hydro-axing is a technique by which a rubber-tired tractor with a front-end mounted high powered blade mulches trees. Fine woody debris is produced, and there is little ground disturbance. Hydro-axing is a relatively new method which has gained favor because of the lower degree of ground disturbance, but only recently has any research been done to understand the effect of hydro-axing on plant communities (Battaglia et al. 2010). No studies have made head-to-head comparisons of older mechanical removal methods with hydro-axing.

Differences in the size of woody litter produced and the degree of soil disturbance may influence the germination and establishment of desirable understory species. For instance, the mulch layer produced by a hydro-axe treatment may have positive or negative effects on germination; germination may be inhibited by lower light availability at the soil surface, or it may be enhanced by higher soil moisture. In chaining and roller chopping, the higher degree of soil disturbance may provide an opportunity for seeded species to establish, or it may become a liability by allowing invasion by weedy species. Finally, in a chaining treatment, the tree skeletons may offer a few years of protection from herbivory, which could play an important role in allowing shrubs to establish. These differences may affect the success of seeding attempts following mechanical tree removal, but such differences have yet to be examined.

Our study has two goals: to compare the desirability of vegetation produced by three types of mechanical treatment (ship anchor chaining, roller chopping, and hydro-axing), and to determine the usefulness of seeding within each of these three treatments. Desirable vegetation in this context is native vegetation with a high proportion of ground cover consisting of broadleaf forbs and palatable shrubs.

STUDY AREA

The Piceance Creek Basin, located in northwestern Colorado, serves as winter range for one of North America's largest migratory mule deer (*Odocoileus hemionus*) populations (Lee 1984). The area, which spans across both Rio Blanco and Garfield counties, is also rich in oil shale and natural gas (Taylor 1987). Development of the basin for the extraction of these resources has gone on for decades and continues to grow. The Piceance Creek Basin ranges in elevation from 1706 meters to 2743 meters with the highest points near the edges (Tiedeman 1978). This basin encompasses nearly 4143 square kilometers and is bordered from the north by the White River, from the south by the Roan Plateau, from the east by the Grand Hogback and from the west by the Cathedral Bluffs (Taylor 1987). Terrain varies from rugged badlands, abrupt cliffs and sharp ridges to open valleys, parks and basins (Baker 1970). Its semiarid climate receives between 27 and 63 centimeters of annual precipitation, half coming in the form of snow during winter months (Tiedeman 1978). The basin is part of the Green River Geologic

Formation consisting of primarily sandstone, siltstone, mudstone, limestone, and shale (Campbell 1974). Soils range from deep sandy alluvial soils and heavy clay soils to entisols and dark mollisols, above 2346 meters (Campbell 1974, Tiedeman 1978).

Bottomland sagebrush and desert shrub dominate lower elevations (Tiedeman 1978). Middle elevations are dominated by upland sagebrush, mixed mountain shrub, and pinyon-juniper woodlands (Tiedeman 1978), grasslands and aspen (*Populus tremuloides*) and douglas-fir (*Pseudotsuga menziesii*) forests can be found at the highest elevations (Tiedeman 1978).

Historically, the land was sparsely populated and used primarily for agricultural and recreational purposes (Tiedeman 1978). In recent decades, natural resource extraction has dramatically altered the landscape. Today the oil and gas industry plays a prominent role in the basin with Rio Blanco and Garfield counties producing vast quantities of oil and natural gas (COGCC 2011). Development of this infrastructure has and continues to fragment suitable mule deer habitat through the construction of well pads, roads and compressor stations (Anderson 2011). Traffic, noise and increased human presence also contribute to adversely affect this important winter range (Anderson 2011).

METHODS

Site Selection

Study area selection was done in conjunction with Dr. Charles Anderson's larger-scale project to examine deer responses to PJ removal (Anderson 2011). First, several hundred PJ stands were delineated within the Magnolia area of Piceance Basin using aerial photography, excluding areas with slopes greater than 30%. Next, stands were visited and scored for suitability of treatment based on a scale of 1 to 3:

<u>Score 1 – most suitable acreage.</u> These parcels contained abundant younger trees growing in dense stands. Simultaneously, the understory of desired shrubs, grasses, and forbs appeared to be robust. Treatment of these areas should yield a strong growth response from that desired understory.

<u>Score 2 – highly suitable acreage.</u> These parcels contained a mix of younger and older trees that grew in less dense patches. The understory of desired shrubs was also less robust than a Score 1 site. Score 2 parcels were highly suitable for treatment, but will yield a lesser initial growth response from the desired understory than a Score 1 site.

<u>Score 3 – suitable acreage</u>. These parcels contained more mature PJ with less dense trees that possessed larger individual tree canopies. Diameter of tree trunks was larger than trees in Score 1 or Score 2 tracts. The understory of desired shrubs, grasses, and forbs was often lacking, and more bare ground was found here than Score 1 or 2 tracts.

Delineations and suitability scores were assigned by Todd Graham of Ranch Advisory Partners. A total of 203 tracts comprising 1,445 acres were deemed suitable for treatment. Next, two focal areas were selected based on the following criteria: at least 40 acres with the same suitability score were available, access routes for ground-disturbing equipment were available, and the cover of PJ trees within each area was as uniform as possible. These two focal areas, called North Magnolia (elevation 2194 m) and South Magnolia (elevation 1828 m), are shown in Figure 1. At the North Magnolia site, a contiguous parcel met the needed criteria. At South Magnolia, the study area was fragmented by gullies which were unsuitable for treatment. Oil and gas development is dense and can be found within a kilometer of the plots, although not within them.

Experimental Design and Setup

We implemented a split-plot design with four replications at the North Magnolia location, and three replications at the South Magnolia location. The whole plot was mechanically treated and the subplot was seeding (seeded or unseeded; Figure 1). Block divisions were designed to minimize variation within each block in PJ density, based on visual inspection of the aerial photography. Subplots were 0.40

ha (1 acre) in size and about three times as long as wide. The long axis of each subplot was arranged perpendicular to the slope. This is because mechanical treatments are typically applied across slopes, rather than up and down them, because it is safer and saves fuel to drive heavy machinery across the slope. Mechanical treatments were randomly assigned to plots within blocks. Each plot was further subdivided into two subplots, and seeding treatments were randomly assigned to subplots within plots.

Mechanical treatments

Mechanical treatments were applied between Oct. 23, 2011 and Nov. 28, 2011. Ship anchor chaining was done using two D8 bulldozers (Caterpiller, Inc., USA), each attached to one end of an 18 m (60-ft.) ship anchor chain with links weighing 40.8 kg (90 lbs.) each. Trees were pulled over by running the chain in one direction, and then killed more completely by running the chain back over the plots in the opposite direction (2-way chaining; Figure 2a-b). Roller chopping was accomplished by attaching a 3.7 m (12-ft.) long, 0.6 m (1.9-ft.) diameter roller chopper to a D8 dozer (Figure 2c). The drum weighed approximately 1100 kg (2,500 lbs.) when empty and held 8338 li (2,200 gal) of water. The drum was filled during operation for a total weight of approximately 9100 kg (20,000 lbs.). Roller chopper plates acted as blades to chop downed trees into pieces approximately 30 cm long (Figure 2c-d). Hydro-axing was accomplished used a 930 Barko industrial tractor with a drum-style FAE flail head which produced fine masticated material (Figure 2e-f). All vegetation was masticated to ground level (or as close as the equipment will allow; less than 30 cm). In the vicinity of former trees, masticated material was up to 40 cm deep. Equipment operators used handheld GPS units to ensure the correct areas were being thinned. Although the area of the seeded and unseeded subplots was only 0.4 ha, in some cases an area larger than this was mechanically treated. The estimated total area treated across all 21 thinned plots was 16.8 ha.

Seeding

All seeded plots received the same diverse native seed mix comprised of 10 shrub species, 14 forb species and 10 grass species. The mix emphasizes shrubs while incorporating light rates of forbs and grasses in order to fill resource niches and thereby reduce the likelihood of weed invasion (Table 1).

The method of seeding differed for each mechanical treatment. In seeded, hydro-axed subplots, all seed was broadcast using EarthWay® hand crank spreaders prior to treatment implementation. Because the seed mix contained seeds of varying sizes, seeds had to be grouped based on size (Table 1) in order for uniform seed dispersal to occur using the spreaders. Five evenly spaced passes, parallel to the long axis of the plot, were made through each seeded subplot using the hand spreaders. Two seeders followed one navigator using a handheld GPS unit to ensure dispersal occurred in the seeded subplot only. In seeded, roller chopped subplots, the majority of species were hand-broadcast prior to treatment implementation, but several large-seeded shrub species that benefit from deeper planting (D group in Table 1) were seeded using Hansen seed dribblers mounted to the tracks of the bulldozer (Figure 3). Chained subplots were seeded in the same manner as roller chopped plots.

FUTURE WORK AND EXPECTED BENEFITS

Percent cover and biomass by species will be evaluated in all plots in the growing seasons of 2012 and 2013. This project will produce an estimate of the quantity and quality of forage produced by three types of mechanical treatments in western Colorado pinyon-juniper forests. It will also produce an estimate of the establishment success of a suite of desirable shrub species when planted in a diverse seed mixture in conjunction with different types of mechanical treatments. The cost of each treatment and of seeding will be compiled. This information will aid managers in choosing a cost-effective treatment for similar sites in western Colorado.

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Table 1: Seed mix divided into seeding groups.

Seed Group	Type	Code	Common Name	Genus	Species
1	grass	ACHY	Indian ricegrass 'White River'	Achnatherum	hymenoides
1	grass	ELEL	Bottlebrush squirreltail 'Toe Jam Ck'	Elymus	elymoides
1	grass	ELTR7	Slender wheatgrass 'San Luis'	Elymus	trachycalus
1	grass	HECO26	Needle and thread	Hesperostipa	commata
1	grass	PASM	Western wheatgrass	Pascopyrum	smithii
1	forb	BASA3	Arrowleaf balsamroot	Balsamorhiza	sagittata
1	forb	CLSE	Rocky Mountain beeplant	Cleome	serrulata
1	forb	HEAN3	Common sunflower	Helianthus	annuus
1	forb	OECA10	Tufted evening primrose	Oenothera	caespitosa
1	forb	OEPA	Pale evening primrose	Oenothera	pallida
1	forb	PEST2	Rocky Mountain penstemon	Penstemon	strictus
1	forb	LILE3	Lewis flax- Maple Grove Var	Linum	lewsii
2	shrub	ARTRW8	Wyoming Sagebrush	Artemisia	tridentata
2	shrub	CHVI8	Yellow rabbitbrush	Chrysothamnus	viscidiflorus
2	shrub	ERNAN5	Rubber rabbitbrush	Chrysothamnus	nauseosus
2	grass	POSE	Sandberg Bluegrass	Poa	Secunda
2	grass	KOMA	Jrairie Junegrass	Koeleria	macrantha
2	grass	POFE	Muttongrass 'UP Ruin Canyon'	Poa	fendleriana
2	grass	VUOC	Six-weeks fescue	Vulpia	octoflora
2	forb	AMRE	Redroot amaranth	Amaranthus	retroflexus
2	forb	ARFR4	Fringed sagebrush	Artemisia	frigida
2	forb	ARLU	White sagebrush	Artemisia	ludoviciana
2	forb	CRAC2	Tufted hawksbeard	Crepis	acuminata
3	shrub	KRLA2	Winterfat	Krascheninnikovia	lanata
3	forb	ERUM	Sulfur-flower buckwheat	Eriogonum	umbellatum
3	forb	ERUM-B	Sulfur-flower buckwheat	Eriogonum	umbellatum
4	grass		Quick Guard	Triticum aestivum	x Secale cereale
4	shrub	PRVI	Chokecherry	Prunus	virginiana
D	shrub	AMAL2	Saskatoon serviceberry	Amelanchier	alnifolia
D	shrub	AMUT	Utah serviceberry	Amelnchier	utahensis
D	shrub	CEMO2	Mountain mahogany	Cercocarpus	montanus
D	shrub	PUTR2	Bitterbrush	Purshia	tridentata
D	shrub	RHTR	Skunkbush sumac	Rhus	trilobata
D	forb	HEBO	Utah sweetvetch	Hedysarum	boreale
D	forb	LUAR3	Silvery lupine	Lupinus	argenteus

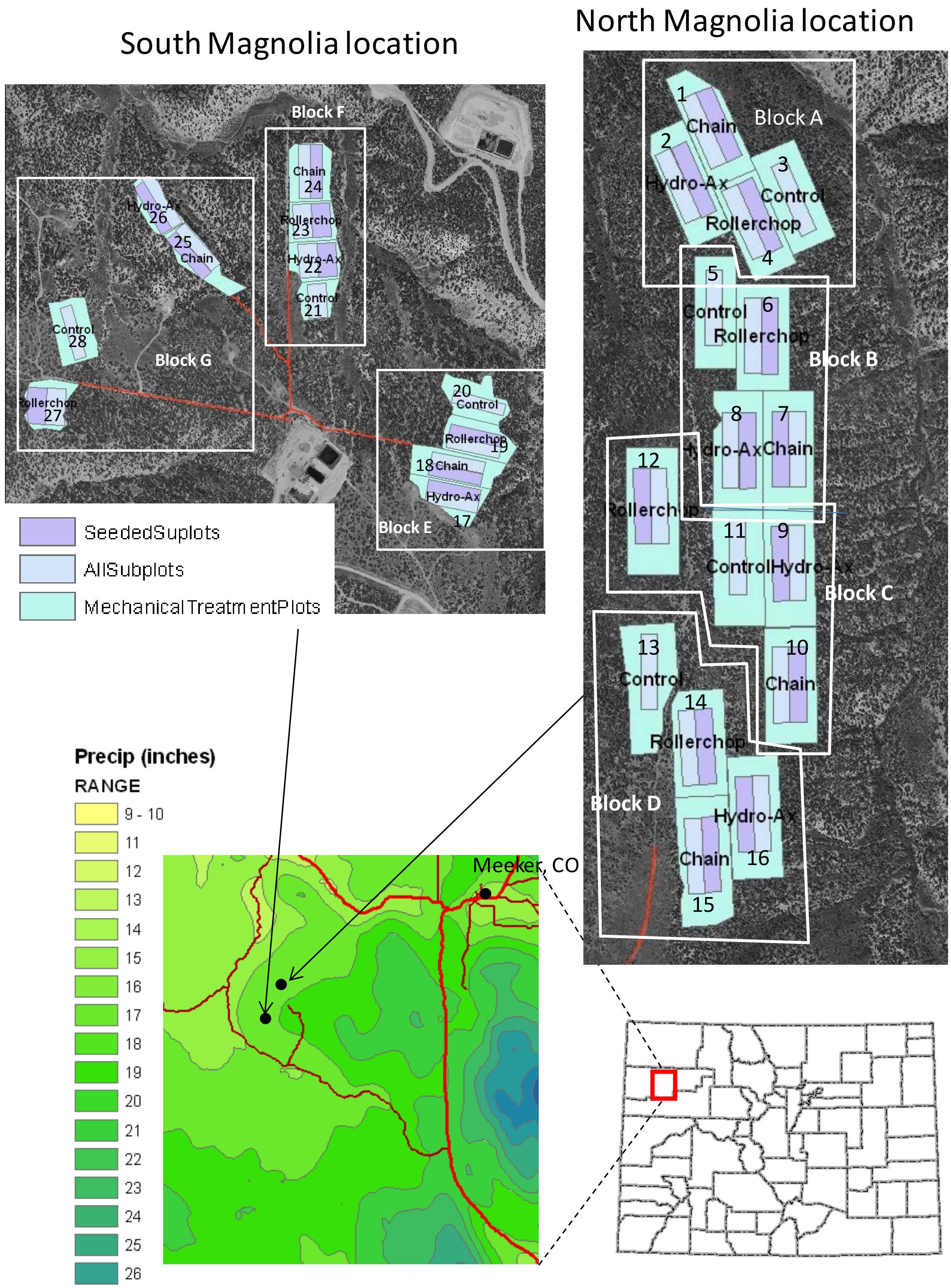

Figure 1. Layout of experiment within North and South Magnolia locations, Rio Blanco County, Colo.

Figure 2. Types of machinery used and woody debris produced: Ship anchor chaining (a) and tree skeletons left behind by chaining (b); roller chopper (c) and coarse debris left by roller-chopping (d); drum-style hydro-axe (e) with fine debris left behind by hydro-axing (f).

Figure 3. Hansen-style seed dribbler mounted to the track of bulldozer. Two such dribblers were mounted on each bulldozer.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	0665	: Mountain Plover Conservation
Task No.:	N/A	: Using rangewide information on chick survival in mountain plovers (Charadrius montanus) to inform management strategies

Federal Aid
Project No. N/A

Period Covered: January 1, 2011 – August 31, 2012

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ABSTRACT

The mountain plover (*Charadrius montanus*) is an upland shorebird that has experienced steep, constant declines in population size across its range since 1966. This precocial migratory species breeds throughout prairie ecosystems of the North American Great Plains, with primary breeding grounds located in eastern Colorado. The nesting ecology of mountain plovers has been well-studied across the species' breeding range, including areas in Colorado (Graul 1975, Knopf and Wunder 2006, Dreitz and Knopf 2007) and Montana (Knowles et al. 1982, Knowles and Knowles 1984, Dinsmore et al. 2002). Few studies to date have addressed chick survival during the post-hatching stage, the period from hatching to fledging. From a conservation perspective, information on the post-hatching stage is imperative because population dynamics often show great sensitivity to the survival of young (Anders et al. 1997; Colwell et al. 2007).

A stage-specific matrix model based on data obtained in Colorado and Montana suggested that conservation efforts restricted to breeding grounds should prioritize increasing chick survival (Dinsmore et al. 2010). Multiple factors may influence the survival of young birds. Young individuals lack experience with selective pressures such as predation, foraging efficiency, migration patterns, and extremes in environmental conditions which are correlated with habitat quality. Further, these selective pressures differ spatially and temporally across the species' range. The distribution of individuals among habitats reflects their ability to discriminate between habitat types and to assess habitat quality. Thus, the landscape configuration and the proximity of resources provided by different habitat types of the North American Great Plains may be critical to the reproductive output of mountain plovers.

The objective of this study is to investigate the factors influencing the survival of mountain plover chicks. Specifically, the study will: (1) investigate natural and anthropogenic factors influencing mountain plover chick survival across the species' range; (2) identify landscape characteristics across the species' breeding range that are positively correlated with the highest levels of chick survival; and (3) provide information to further develop conservation and management efforts for mountain plovers on public and private lands.

Here, we provide descriptive information for the 2011 and 2012 field seasons, years two and three of a three-year study. The 2011 field season in Colorado extended into early August and experienced similar weather conditions as the 2010 season, including a colder, wetter spring than in previous years. The 2012 field season experienced a much warmer, drier spring than in previous years, resulting in an earlier onset of breeding activity. In 2012, nesting began approximately two to three weeks earlier than in 2011 and concluded earlier, by early-to-mid July.

Our preliminary findings from 2011-2012 suggest that the causes of chick mortality differ among habitat types and years. Avian depredations occurred most often on prairie dog habitats during the 2011 field season, which is consistent with findings from the first year of the study (2010). In 2012, most avian-caused mortalities were identified on grasslands, likely because a higher proportion of avian predation was caused by hawks rather than burrowing owls (*Athene cunicularia*). Unlike in 2010, we were unable to confirm any mammalian depredations during 2011-2012, and thus cannot provide additional information on this mortality cause. As in 2010, weather events were responsible for some chick mortalities during the 2011 season, but these mortalities were not tied to a certain habitat type, as previously found. No weather-related chick mortalities were identified during the 2012 field season. Though the anticipated three years of field data collection for this chick mortality study have now been completed, additional years of field data collection would certainly strengthen the emerging patterns in chick mortality across different habitat types, especially given the differences in our findings across the last two years.

WILDLIFE RESEARCH REPORT

USING RANGEWIDE INFORMATION ON CHICK SURVIVAL IN MOUNTAIN PLOVERS (*CHARADRIUS MONTANUS*) TO INFORM MANAGEMENT STRATEGIES

VICTORIA J. DREITZ and LANI STINSON

PROJECT OBJECTIVES

The objective of this study is to investigate the factors influencing the survival of mountain plover chicks. The study: 1) Investigates natural and anthropogenic factors influencing mountain plover chick mortality across the species' range; 2) identifies the spatial areas across the species' breeding range that are positively correlated with the lowest levels of chick mortality; and 3) provides information to assist in developing management actions for mountain plovers on public and private lands. This is a comparative, range-wide study on mountain plovers including a 'highly fragmented' breeding locale in Colorado and a less fragmented breeding locale in Montana. Colorado Parks and Wildlife (CPW) oversaw field data collection in eastern Colorado. We report information pertaining to the Colorado effort.

SEGMENT OBJECTIVES

1. Investigate differences in mortality of mountain plover chicks between different habitat types – grassland with prairie dogs, grassland without prairie dogs, and agricultural fields.
2. Identify causes of mountain plover chick mortality on the three different habitats.
3. Summarize and analyze data, publish information as a Progress Report.

INTRODUCTION

The mountain plover (*Charadrius montanus*) is an upland shorebird found on the xeric tablelands from Mexico to northern Montana (Knopf and Wunder 2006). Steep, constant declines in population size have been reported for mountain plovers across their range since 1966. In 1999, the U.S. Fish and Wildlife Service (USFWS) petitioned for 'threatened' status of the mountain plover; the listing decision was found 'not warranted' in 2003 (USFWS 2003). The mountain plover was reconsidered for listing in 2010, and a decision to withdraw the proposed listing as threatened occurred on May 12, 2011 (USFWS 2011). Nevertheless, consistent population declines have prompted conservation agencies to assess the spatial extent and potential factors contributing to declines in this species.

Historically, mountain plovers were present across western prairies in areas of intensive grazing by bison (*Bison bison*) or prairie dogs (*Cynomys* spp.). Today, mountain plovers are still observed on areas grazed by prairie dogs, along with areas grazed by domestic cattle and sheep, and on agricultural fields (Knopf and Wunder 2006). The eastern plains of Colorado provide breeding habitat for more than half of the continental population of mountain plovers (Kuenning and Kingery 1998). Smaller, more isolated breeding areas occur throughout the western Great Plains region.

The nesting ecology of mountain plovers has been well-studied across the species' breeding range, including areas in Colorado (Graul 1975, Knopf and Wunder 2006, Dreitz and Knopf 2007) and Montana (Knowles et al. 1982, Knowles and Knowles 1984, Dinsmore et al. 2002). Detailed information on brood-rearing ecology has been conducted in both Colorado and Montana. Knopf and Rupert (1996) estimated daily chick survival on grassland habitat in northeastern Colorado at 10-day intervals ranging from 0.951-0.977. Lukacs et al. (2004) found that chick survival was lowest immediately after hatching and quickly increased within four days post-hatch on prairie dog colonies in Colorado. Knopf and Rupert (1996), Lukacs et al. (2004), and Dinsmore and Knopf (2005) indicated that daily survival rates increased with age of the chick. In eastern Colorado, Dreitz (2009) estimated chick survival from hatch to 30 days post-hatch to be higher on grassland with prairie dogs (0.75, CI = 0.54, 0.87), than grassland without prairie dogs (0.24, CI = 0.08, 0.45) and agricultural fields (0.23, CI = 0.14, 0.33) and the rate of brood movement off of prairie dog nest habitat was lower than grassland, but higher than agricultural fields for each year of the study. These patterns observed in chick survival and brood movements were not influenced by prey resources biomass or density (Dreitz 2009). None of the above studies determined causes of mortality in plover chicks, but Knopf and Rupert (1996) speculated that on grassland the main cause of chick mortality was predation by swift fox (*Vulpes velox*).

Multiple factors may influence the mortality of young birds. In general, young individuals lack experience with selective pressures such as predation, foraging efficiency, migration patterns, and extremes in environmental conditions which may be correlated with habitat quality. The distribution of individuals among habitats reflects their ability to discriminate between habitat types and to assess habitat quality. Thus, the landscape configuration and the proximity of resources provided by different habitat types of eastern Colorado may be critical to the reproductive output of mountain plovers. Information on the post-hatching stage is imperative for conservation efforts on mountain plovers because brood loss affects real reproductive output as well as the degree of subsequent recruitment and, in turn, the viability of the population.

Technological advances in radio transmitters have made it possible to determine the cause-specific mortality of mountain plover chicks (~10 g at hatching). Radios placed on the chicks ($\leq$ 0.38 g) follow established guidelines to not exceed 5% of body mass for small (<50 g) birds (Caccamise and Hedin 1985, Gaunt et al.1999). Various attachment methods have been evaluated in captivity, suggesting that a leg harness attachment is a suitable method with minimal to no observed impacts on survival, physiology, growth, and behavior (Dreitz 2007, 2008, Dreitz et al. 2011). Additionally, this attachment technique was tested in the field and proved to be a plausible method to track mountain plover chicks and evaluate cause specific mortality (Dreitz and Riordan 2009).

METHODS

Study Area

The study area was located exclusively on privately owned lands in Lincoln County, Colorado. Private landowners in this area have collaborated on previous studies on mountain plovers in Colorado (Dreitz and Knopf 2007, Dreitz unpubl. data) allowing continued access to >3000 km^2 of land. In eastern Colorado, mountain plovers primarily use the following habitats for breeding activity: grasslands occupied by black-tailed prairie dogs (*Cynomys ludovicianus*, hereafter simply ‘prairie dog’); native grassland without prairie dogs; and agricultural fields, predominantly dryland agriculture.

Placement of Transmitters on Chicks

This field study was conducted by monitoring radio transmittered mountain plover chicks from one day post-hatch to $\geq$ 30 days post-hatch. We identified ~1-day-old plover chicks by monitoring nests. We used egg flotation (Westerskov 1950) to age eggs and to estimate days until hatching. Mountain plover chicks are precocial and leave the nest shortly after hatching (Knopf and Wunder 2006). If the hatching date is missed, locating the adult and its brood is difficult. In an effort to avoid missing placement of transmitters on ~1-day-old chicks, we attached a 1.8 g radio transmitter to the nest-tending adult prior to hatching. Transmitters were affixed by applying a light coating of waterproof epoxy and sliding it under the upper back feathers. Care was taken to avoid exposing epoxy to the skin of the bird. This procedure enables the transmitters to drop off during molting (Dreitz et al. 2005; Dreitz 2009, 2010). In 2011, we experienced loss of many adult transmitters, likely caused by early molting. In response, we used an alternative method of transmitter attachment on adults involving a leg harness similar to that used with the chicks (see below), but sized to fit adult birds.

After hatching and when chicks were completely dry, we captured them by hand. Chicks were weighed and a small blood sample (<50 µL) was obtained by jugular or brachial venipuncture at this or subsequent captures (~15-days-old, see below). Average mass of mountain plover chicks at hatch is suggested to be 7-11 g (Graul 1975, Miller and Knopf 1993), however 1-to-2-day-old chicks in our study weighed 10-12 g. A 0.35 g (2011) or 0.38 g (2012) transmitter was placed on <1-day to ~16-day-old chicks. This size of transmitter falls within the suggested guidelines of transmitters not exceeding 5% of body mass for small birds (<50 g; Caccamise and Hedin 1985, Gaunt et al. 1999). Battery life of the 0.35 g and 0.38 g transmitters was ~18-20 days. To monitor radio transmittered plover chicks to fledging age (≥30 days), we recaptured transmittered chicks at ~15 days by catching them by hand. We replaced the 0.35 g or 0.38 g radio transmitter with a 0.53 g or 0.62 g radio transmitter. Data on body mass of mountain plover chicks is limited, but suggests ~15-day-old chicks are generally >20 g (Dreitz unpubl. data). At transmitter replacement, we obtained morphometric measurements, collected feather samples (if present), and collected blood samples (if not obtained at initial capture). Additionally, we captured all plover chicks we observed (i.e., chicks whose sibling had a transmitter or chicks from unknown nests) and processed them in a similar manner and banded all individual chicks. This additional banding of chicks is an attempt to increase the number of banded birds in our study area.

Captive studies have tested different transmitter attachment methods (e.g., glue, sutures) on a surrogate species, killdeer (*Charadrius vociferous*), and mountain plovers. A leg harness technique modified from one described by Rappole and Tipton (1991) has minimal to no impact on captive killdeer and mountain plover chicks from one to 42 days post-hatch (Dreitz 2007, 2008). The leg harness technique used in this study consists of a 100% polyurethane clear, flat elastic material (Stretchrite®) cut to ~1.5 mm in width. The material is assembled in a figure-8 design with two leg loops that expand to accommodate chick growth; the transmitter sits in the middle of the chick's back (Dreitz 2007, 2008; Dreitz et al. 2011). This transmitter attachment was assembled in advance with multiple loop sizes, so transmitter placement took <1 minute to accommodated chicks of varying sizes.

Monitoring Chicks from Hatch to Fledging

Once transmitters were deployed on chicks, they were monitored using radio telemetry at ≤ 24 hour intervals, depending on weather conditions. Chicks were monitored for ≥ 30 days post-hatch until fate could be determined as fledged or dead. We recorded information including status (alive/dead), number of chicks observed within the brood, habitat type, location coordinates, and observer for each location datum collected. Chicks without a transmitter were monitored indirectly by the transmittered adult and/or sibling(s). Initially the transmitters were located via all-terrain vehicle, then by foot. We conducted standardized telemetry transects to search for frequencies of missing chicks, focusing efforts on lands immediately surrounding the last known telemetry location, depending on access permission. If we could not locate transmitters for seven to 10 days on the ground, we expanded our search area by conducting aerial telemetry transects.

Suspected mortalities were located on foot and the area was thoroughly inspected for evidence suggesting mortality cause. We classified chick mortality cause as one of the following: avian predation, mammalian predation, reptilian (e.g., snake) predation, unknown predation, weather-induced (e.g., hail or rain), or unknown mortality cause. We determined avian predation based on evidence such as finding the transmitter near an avian plucking post, nest site, or in avian pellets. We identified mammalian predation when we found transmitters that were in mammalian dens, cached, scat was found near the transmitter or carcass, and/or other physical signs (e.g., teeth marks) were present on the carcass or transmitter. We determined predation to be reptilian when transmitters were detected from inside observed snakes or snake holes (e.g., holes too small to accommodate other potential predators, such as burrowing owls or swift foxes). We classified unknown predation to be the mortality cause when a transmitter was found with or without remains of a chick and no other evidence allowed for classification of the predator. Unknown predation also included scenarios where transmittered chicks were confirmed absent from a brood because they were not detected with the adult (through telemetry or visual observation) on multiple occasions. Mortalities were classified as weather-induced when the whole carcass of a chick was found with the transmitter and known extreme weather events (e.g., hail, flooding) had occurred 24 hours prior. These chicks were collected and evaluated by necropsy in an attempt to confirm a weather-induced mortality event. Mortalities were defined as unknown when there was not enough evidence to suggest one of the other mortality categories, but remains of the chick were found.

We collected chick carcasses that had not substantially decomposed. Dissections were performed <8 hours after collection including removal of the liver and brain which were then frozen. The remainder of the body was placed in formalin for preservation and later evaluation to confirm cause of death through necropsy and pathology diagnostics.

RESULTS

2011

We located and monitored 103 nests in three habitat types: native grassland (n = 43), grassland with prairie dogs (n = 29), and agricultural lands (n = 31). A total of 39 out of 103 nests successfully hatched ($\geq$ 1 egg hatched), which corresponds to an apparent nest success rate of 38%.

We placed 0.35 g radio transmitters on 93 1-5 day-old mountain plover chicks on three different habitat types in eastern Colorado: 24 chicks that hatched on prairie dog, 41 chicks that hatched on native grassland, and 12 chicks that hatched on agricultural fields (Table 1). There were 16 transmittered chicks that were not linked to a known nest habitat, including five chicks found crossing roads between habitat types. Beyond the 93 transmittered chicks, we also had the opportunity to monitor 38 chicks without transmitters (hereafter referenced as non-transmittered) through our monitoring of their transmittered siblings and/or parents (Table 1).

A total of 30 transmittered chicks survived to 16 days: four chicks were located on prairie dog at time of transmitter replacement, 20 chicks on native grassland, and six chicks on agricultural fields (Table 1). While the majority of the non-transmittered chicks were not detected again following hatching, we were able to confirm survival to 16 days for four of the non-transmittered chicks (Table 1). Of these, two were observed on grassland habitat and one was observed on prairie dog. The final non-transmittered chick was only known to have survived to 16 days because of a later observation at day 24; thus, the habitat at day 16 was unknown.

We confirmed fledging (survival to ≥30 d) of 10 transmittered chicks: two chicks were primarily located on prairie dog, seven chicks were primarily located on native grassland, and one chick was primarily locate on agricultural fields (Table 1). Additionally, we confirmed fledging in two non-transmittered chicks: one was primarily located on native grassland and another was primarily located on prairie dog (Table 1).

We confirmed mortalities of 36 transmittered chicks (39% of the 93 initially transmittered). Most mortalities (n=14, 39%) occurred due to avian predation, mainly by burrowing owls (*Athene cunicularia*) and prior to chicks reaching 15 days. In 2011, we were not able to confirm any mammalian predations, but we observed our first rattlesnake predation. Unknown predation occurred for three of the transmittered chicks. Unknown mortality, usually characterized by a chick found dead on the ground with no other evidence present, occurred in 13 transmittered chicks. Weather is suspected in the deaths of two transmittered chicks. Injury appeared to be the cause of two chick mortalities: one chick was found with a broken leg the day before it was found dead and another was found on the side of the road with a head injury the day before it died. Discussions with technicians involved in the injury cases resulted in the decision not to interfere beyond returning the following day to confirm mortality. Unfortunately, we had one accidental mortality by a research technician.

Of the 36 confirmed mortalities, 44% were found on prairie dog (n = 16), 39% were located on grassland (n = 14), and the remaining 17 % were found on agricultural fields (n = 6). Avian predation was confirmed more often on prairie dog habitat (n=13) than grassland (n=0) or agricultural fields (n=1), while mammalian predations could not be confirmed on any of the habitat types. The rattlesnake-caused mortality occurred on prairie dog habitat. Agricultural fields had the most unknown predations (n=2), compared to grassland (n=0) or prairie dog (n=1). Weather-induced mortality occurred only on grassland habitat (n=2). Unknown mortality occurred more often on grassland (n=7) than prairie dog (n=1) or agricultural fields (n=2). Samples from five of the 13 unknown mortalities were submitted for diagnostic laboratory examination. Diagnostic laboratory results for four of the five unknown mortalities suggested no definitive conclusion on the cause of mortality. The remaining chick was diagnosed with epicarditis and myocarditis and was found to have adequate stores of glycogen in the liver, suggesting rapid onset of a bacterial infection such as Chlamydophila. *? Chlamydophila*.

During the 2011 field season, we confirmed 11% of the mountain plover chicks survived from hatch to fledging (10 transmittered chicks out of 93 initially transmittered). In addition, we confirmed fledging of two untransmittered chicks. We were unable to determine survival or mortality in 50% of the transmittered chicks (47 of 93 initially transmittered chicks) due to sampling procedures (e.g., inability to access property), equipment malfunction (e.g., premature failure of radio transmitter battery life), potential predator taking remains out of study area, and other potential issues.

Most likely due to early molting of adult birds, a total of 11 adult transmitters (19 %; n = 58 total deployed) dropped prematurely (eight to 22 days after placement). We started recovering dropped transmitters in early June and continued finding them throughout the month of June. In late June, we modified our attachment method using the leg harness technique with the loop sizes adjusted to fit adult birds (~75 mm loops). We placed 14 transmitters on adults using this attachment method. Unfortunately, we were unable to evaluate the use of this attachment method on most adults because of high nest failure (n = 8). Five of the adults fitted with harness transmitters had successful nests (≥ 1 chick hatched). Of these five cases, three adult transmitters stopped working during the post-hatch stage and were never recovered. Another one of the transmittered adults was detected only for two days post-hatch when the entire brood was predated by a burrowing owl, and thus monitoring of that adult ceased. The remaining adult with a transmitter was observed with the transmitter intact until one chick fledged in late July (≥30 days post-hatch).

2012

We located and monitored 93 nests in three habitat types: native grassland (*n* = 37), grassland with prairie dogs (*n* = 37), and agricultural lands (*n* = 19). A total of 26 out of the 93 nests successfully hatched (≥ 1 egg hatched), which corresponds to 28% apparent nest success rate. Nesting activity began earlier than anticipated in 2012 and we observed many more chicks early in the season, without ever detecting their nests.

We placed 0.38 g radio transmitters on 44 1-5-day-old mountain plover chicks: 17 chicks that hatched on prairie dog, 10 chicks that hatched on native grassland, and 17 chicks that hatched on agricultural fields (Table 2). In addition to the chicks handled at nests, there were 17 chicks transmittered at age 1-5 days and six chicks transmittered at age > 5 days with unknown nest habitat. We had the opportunity to monitor two non-transmittered chicks in 2012.

Of the 44 chicks transmittered at hatching, a total of 17 chicks survived to 15 days: seven chicks were located on prairie dog at time of transmitter replacement, one chick on native grassland, and nine chicks on agricultural fields (Table 2). Of the 23 chicks that were not linked to known nest locations and were transmittered at various ages after leaving the nest, 17 were confirmed alive at 15 days. One of the two non-transmittered chicks we monitored lived to 15 days and was observed on prairie dog (Table 2).

We confirmed fledging (survival to ≥30 days) of nine transmittered chicks from known nests: three chicks were primarily located on prairie dog, two chicks were primarily located on native grassland, and four chicks were primarily located on agricultural fields (Table 2). We confirmed fledging in an additional 10 transmittered chicks that came from unknown nests: two chicks primarily located on prairie dog, three chicks primarily located on native grassland, and five chicks primarily located on agricultural fields.

We confirmed mortalities of 20 transmittered chicks (33% of the total 67 trasmittered chicks). Most mortalities (*n*=10, 50%) occurred due to unknown causes and were characterized by chicks found dead on the ground with no other evidence present. It is possible that heat exhaustion may have caused the death of several of these chicks, but we were unable to confirm that as a cause. Avian predation (*n*=6) was responsible for 30% of confirmed chick mortalities; however, unlike in 2011, the majority of avian predations were thought to be caused by hawks rather than burrowing owls (*Athene cunicularia*). We also confirmed three snake predations, either by detecting the transmitter from the location where a rattlesnake was observed, or from inside a snake hole. There was one additional predation that was unknown. As in 2011, we were again unable to confirm any mammalian predations in 2012. There were no chick mortalities this year where weather was determined as the cause of death.

Of the 20 confirmed chick moralities, 50% were found on grassland habitat (*n*=10), 30% were detected on prairie dog habitat (*n*=6), while 20% were located on agricultural lands (*n*=4). Avian predation was confirmed more often on grassland (*n*=4) than prairie dog (*n*=2) or agricultural fields (*n*=0), while mammalian predations could not be confirmed on any of the habitat types. The reptilian mortalities occurred on prairie dog habitat (*n*=2) and agricultural fields (*n*=1). The one mortality caused by an undetermined predator occurred on grassland. Unknown mortality occurred more often on grassland (*n*=5) than prairie dog (*n*=2) or agricultural fields (*n*=3). Samples from six of the 10 unknown cause mortalities were submitted for diagnostic laboratory examination, but results for only three chicks have been received to date. Diagnostic laboratory results for two of the unknown mortalities suggested no definitive conclusion on the cause of mortality. Histologic findings in the remaining chick, estimated age 23 days, suggested poor condition and energy store depletion, at least partially due to parasitism. The chick was diagnosed with two parasites: intraglandular nematode larvae in the proventriculus and intraluminal cestode parasite in the small intestine. The chick was also found to have moderate glycogen depletion in the liver and two fractures in the right leg (likely postmortem).

In 2012, we confirmed survival to fledging for 20% of chicks transmittered at hatching (nine out of 44 handled at nests), and 28% of all transmittered chicks (19 of 67 transmittered at various ages). We were unable to determine survival or mortality in 42% of the chicks due to sampling procedures (e.g., inability to access property), equipment malfunction (e.g., premature failure of radio transmitter battery life), potential predator taking remains out of study area, and other potential issues. We suspect that the majority of these chicks with unknown fates were predated such that their transmitters were no longer detectable by us, but we cannot classify these cases as confirmed mortalities.

We experienced premature dropping of adult transmitters in 2012, as we did during the 2011 field season. A total of nine adult transmitters (19 %; n = 46 total deployed) were recovered from the field after dropping prematurely off of adults (10-35 days after placement). We started recovering dropped transmitters as early as mid-May and continued finding them throughout May and June.

DISCUSSION

Chick Survival and Causes of Mortality

During the 2011 and 2012 field seasons, we confirmed survival to fledging in a total of 29 out of 160 radio-transmittered mountain plover chicks in eastern Colorado (18% of all chicks monitored across the two years). This fledging rate is about twice the percentage of transmittered chicks that were confirmed fledged in 2010 (nine of a total 93 transmittered). Of the 19 chicks that survived to fledge in 2012, nine were primarily located on agricultural fields, five were primarily located on prairie dog, and five were primarily located on grassland. In contrast, in 2011, only one fledged chick was primarily located on agricultural fields, two were primarily located on prairie dog, and seven were primarily located on grassland. Across all three years of the study 2010-12, fewer chicks fledged on prairie dog than on the other habitats. This trend may be related to a variety of factors, such as differences in resource availability or predation pressures on different habitat types.

The leading cause of chick mortality in 2011-2012 was predation, with 50% of all chick mortalities caused by raptors, snakes, or other unidentified predators. Of all depredated chicks, evidence suggested 71% were killed by avian predators, including burrowing owls and other raptors, including Swainson's hawks (*Buteo swainsoni*). While burrowing owls were the main avian predator in both 2010 and 2011, the 2012 data did not support this trend. Overall, there were fewer avian predations detected in 2012 as compared to previous years (2010: n=13, 2011: n=14, 2012: n=6,). Of those avian predations, a smaller proportion were thought to be caused by burrowing owls in 2012 (n=2) as compared to hawks (n=4). We speculate that the reduction in avian-caused predations may be related to the earlier onset of nesting experienced in 2012. Perhaps early-hatching plover chicks were able to avoid avian predators better because they could mature before the height of burrowing owl activity. It is possible that the reason why so many chicks were taken by burrowing owls in 2010-2011 was because the timing of plover chick hatching was more synchronized with the period of owlet rearing, and, thus, plover chicks were a more available food source at that time.

The vast majority of other mortalities (i.e. non-predations), comprising 41% of all confirmed chick mortalities in 2011-12, could not be linked to a certain cause. Other known causes such as weather events and injury were responsible for the remaining 9% of confirmed chick mortalities, though these causes were only identified in 2011. Unlike in 2010, we were not able to confirm any mammalian predations in either 2011 or 2012. We believe this may be due to the fact that we changed transmitter manufacturers between 2010 and 2011. The 2011 transmitter models were more destructible, likely not withstanding predator tampering or chewing, such that signals were not being emitted shortly after encounter with a mammalian predator. We had hoped to avoid this issue with the 2012 models, which featured a hard resin covering intended to make the transmitters more durable during predator encounters. However, the lack of any confirmed mammalian predations using the 2012 transmitters, together with the large number of transmitters that went missing, leads us to believe that despite improvements, the 2012 models were still not "tough" enough to withstand mammalian tampering. It is unfortunate that we were unable to document any mammalian predations in 2011 or 2012, especially because mammals were responsible for > 20% of chick mortalities during the first year (2010) of the study.

Habitat-Specific Mortality Cause

We detected differences in the causes of chick mortality across the three different habitat types; however findings from 2011 and 2012 were not consistent. The majority of chick mortalities were located on prairie dog habitat in 2011 (44%, n=16), while the majority of dead chicks were found on grassland in 2012 (50%, n=10). These differences could be related to changes in densities of predators in the different habitat types. In both years, agricultural fields comprised the smallest proportion of chick mortalities compared to the two other habitat types (2011: 14%, n = 5; 2012: 20%, n = 4). Dead chicks located on agricultural fields died mainly from undetermined causes.

In 2011, avian predation was the leading cause of mortality on prairie dog habitat, with burrowing owls as the main confirmed avian predator, especially of younger chicks (those <15 days old). In 2012, we did not find this same trend on prairie dog, but, rather, we confirmed equal numbers of mortalities from avian predators, snakes, and unknown causes in this habitat type. In both 2011 and 2012, the majority of dead chicks found on grassland (2011: n=9; 2012: n =5) could not be classified in terms of their cause of death, so it is difficult to draw many conclusions about grassland mortalities from these limited results. In 2011, there was only one predation (by an unknown predator) confirmed on grassland, whereas there was evidence of four hawk predations on grasslands in 2012. This shift from avian predations primarily caused by burrowing owls in 2011 to avian predations mainly caused by hawks in 2012 may be a result of a variety of interacting factors, including the timing of the mountain plover breeding season and varying densities of different types of avian predators during the two years. Interestingly, the majority of the mortalities located on grasslands in the first year of the study were caused by mammals. However, multiple hawk predations were also noted.

The results of this field study suggest that the causes of mountain plover chick mortality do differ among habitats but varying conditions in different years of the study made it difficult to tease out robust patterns in only a three-year period. Conservation and management efforts will be difficult and controversial given one of the main predators of mountain plover chicks, a species of conservation concern, is another species of conservation concern, the burrowing owl. Additional years of study are warranted to confirm or refute the 2010-2012 findings, especially pertaining to mammalian-caused chick mortalities, which we could not assess in either 2011 or

2010. We caution any interpretation or manipulation of these data beyond that contained in this report. The information in this report is preliminary and subject to further evaluation including more in-depth statistical analyses.

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Table 1. Summary of mountain plover (*Charadrius montanus*) chicks with and without transmitters monitored from hatch until fledging in the spring and summer of 2011.

	Prairie Dog Colonies	Native Grassland	Agricultural Fields	Total
1-5 Day Chicks				
Transmitter	24	41	12	93[a]
No Transmitter	28	8	2	38
15-18 Day Chicks				
Transmitter	4	20	6	30
No Transmitter	1	2	0	4[b]
≥30 Day Chicks				
Transmitter	2	7	1	10
No Transmitter	1	1	0	2

[a] There were 16 transmittered 1-5 day chicks with unknown habitat type at time of hatching.

[b] There was one untransmittered 15-18 day chick with unknown habitat type.

Table 2. Summary of mountain plover (*Charadrius montanus*) chicks with and without transmitters monitored until fledging in spring and summer of 2012.

	Prairie Dog Colonies	Native Grassland	Agricultural Fields	Total
1-5 Day Chicks				
Transmitter	21	23	17	61[a]
No Transmitter	1	1	0	2
15-18 Day Chicks				
Transmitter	12	10	12	34[b]
No Transmitter	1	0	0	1
≥30 Day Chicks				
Transmitter	5	5	9	19[c]
No Transmitter	0	0	0	0

[a] There were 17 transmittered chicks with unknown nest habitat. Habitat at time of handling (age 1-5 days) was used to classify these chicks..

[b] There were 17 transmittered chicks with unknown nest habitat confirmed to be alive at 15-18 days. Of these, 11 were transmittered at 1-5 days and six were transmittered at age > 5 days.

[c] There were 10 transmittered chicks with unknown nest habitat at the time of hatching, 6 of which were fitted with transmitters at 1-5 days and four of which were transmittered at > 5 days.

Colorado Division of Parks and Wildlife
September 2011-September 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	: Division of Parks and Wildlife
Cost Center:	3420	: Avian Research
Work Package:	3006	: Waterfowl and Other Small Game
Task No.:	N/A	: Evaluating the relationship between hunting regulations, habitat conditions, and duck hunting quality on State Wildlife Areas in northeastern Colorado
Federal Aid Project No.	N/A	

Period Covered: September 1, 2011 – August 31, 2012

Author: J. H. Gammonley and J. P. Runge

Personnel: J. Barron, M. Bishop, C. Caldwell, J. Coyle, C. Davison, B. Gipson, C. Gray, T. Kroening, M. McConnell, B. Smith

ABSTRACT

The lower South Platte River (SPR) corridor has historically supported the highest numbers of wintering ducks and highest hunter numbers and duck harvest of any region in Colorado. There is concern that harvest pressure has led to reduced numbers of wintering ducks and low harvest success, particularly on State Wildlife Areas (SWAs), which could in turn lead to lower hunter satisfaction and declining hunter recruitment and retention. The goal of this study is to determine the extent to which a set of more restrictive hunting regulations influence duck hunter success, hunter activity, hunter satisfaction, and duck distribution, compared to a set of less restrictive hunting regulations, on selected SWAs along the SPR corridor (Figure 1). We will also examine how the influence of regulations on these responses varies among SWAs with differing habitat conditions. The 2011-2012 regular duck season was the fourth field season of the project. We selected 3 pairs of SWAs representing different habitat conditions along the SPR corridor, and assigned 1 SWA in each pair a set of restrictive hunting regulations (hunting access permitted only on weekends, Wednesdays, and legal holidays; reservations required for a limited number of parties; and the property is closed to the public after 2 p.m.), with no restrictive regulations on the other SWA in each pair. We established check stations at each of the SWAs and required all waterfowl and small game hunters to check out during the regular duck season. We interviewed all waterfowl and small game hunters and recorded information on their hunting experience and methods, harvest success, and satisfaction. We also conducted monthly aerial counts of waterfowl along the SPR corridor. During the 2011-2012 duck season, we obtained information from 929 hunting parties on study SWAs, of which 711 were duck hunting parties. Jackson Lake SWA (unrestricted regulations) had the highest use, with 341 duck hunting parties and 737 duck hunter-days, and Overland Trail SWA (unrestricted regulations) had the lowest use, with 15 duck hunting parties and 35 duck hunter-days. From interview data, season-long

harvest success, measured as average ducks bagged per hunter per day, was greater at restricted areas than unrestricted areas of similar size and habitat type: season-long harvest success was 4 times greater at Atwood SWA than Overland trail SWA, 2.7 times greater at Jean K. Tool/Brush SWAs than Bravo SWA, and 2 times greater at Red Lion SWA than Jackson Lake SWA. Hunting parties' satisfaction with hunter crowding levels, habitat conditions, property-specific regulations, and their overall hunt experience averaged slightly satisfied or satisfied on all study SWAs; hunters tended to be dissatisfied or slightly satisfied with duck numbers. Numbers of migrating/wintering ducks in the SPR corridor were relatively high in October and November, and large numbers of ducks used open water in large reservoirs during January. This study is expected to continue for 2 additional years.

WILDLIFE RESEARCH REPORT

EVALUATING RELATIONSHIPS BETWEEN HUNTING REGULATIONS, HABITAT CONDITIONS, AND DUCK HUNTING QUALITY ON STATE WILDLIFE AREAS IN NORTHEASTERN COLORADO

JAMES H. GAMMONLEY and JONATHAN P. RUNGE

PROJECT OBJECTIVES

The goal of this study is to determine the extent to which a set of more restrictive hunting regulations influence duck hunter success, hunter activity, hunter satisfaction, and duck distribution, compared to a set of less restrictive hunting regulations, on selected state wildlife areas (SWAs) along the South Platte River (SPR) corridor. We will also examine how the influence of regulations on these responses varies among SWAs with differing habitat conditions. Specific objectives include:

1. Compare duck hunter success (ducks bagged per hunter) on selected SWAs with different hunting regulations and habitat conditions.
   Hypothesis 1: Average hunter success will be higher on properties with more restrictive hunting regulations than on properties with similar habitat conditions where hunting regulations are less restrictive.
   Hypothesis 2: Average hunter success will be lower on properties with more restrictive hunting regulations than on properties with similar habitat conditions where hunting regulations are less restrictive.
   Hypothesis 3: Differences between the two types of areas will be statistically indistinguishable.

2. Compare hunter activity (hunter use-days, party size, hunting methods, number of hours per day when hunters are present on the property) on selected SWAs with different hunting regulations and habitat conditions.
   Hypothesis 1: Properties with more restrictive hunting regulations will have less intensive use than properties with similar habitat conditions where hunting regulations are less restrictive.
   Hypothesis 2: Differences between the two types of areas will be statistically indistinguishable.

3. Compare self-reported indices of waterfowl hunter satisfaction on selected SWAs with different hunting regulations and habitat conditions.
   Hypothesis 1: Average indices of hunter satisfaction will be significantly higher on properties with more restrictive hunting regulations than on properties with similar habitat conditions where hunting regulations are less restrictive.
   Hypothesis 2: Average indices of hunter satisfaction will be lower on properties with more restrictive hunting regulations than on properties with similar habitat conditions where hunting regulations are less restrictive.
   Hypothesis 3: Differences between the two types of areas will be statistically indistinguishable.

4. Correlate overall duck numbers, climate data (temperature, precipitation), and indices of habitat conditions (river flows, percent of area flooded, percent of area frozen) with results from objectives 1-4.
   Prediction: These measures will explain a high proportion of the variation observed over space and time in the response variables for Objectives 1-4.

5. Based on results from objectives 1-4, develop recommendations for future duck hunting management of SWAs along the South Platte River corridor.

Because the purpose of restrictive regulations is to reduce disturbance to waterfowl on SWAs, it will also be necessary to restrict activities of other small game hunters. Although not the focus of this study, we will also measure the harvest, activity, and satisfaction of small game hunters on SWAs along the SPR.

SEGMENT OBJECTIVES

1. Collect information on hunting activities, harvest, and satisfaction levels from all waterfowl and small game hunting parties on 7 SWAs along the SPR corridor during the 2011-2012 regular duck hunting season.

2. Conduct periodic aerial surveys of waterfowl numbers and distribution along the SPR corridor throughout the 2011-2012 regular duck hunting season.

INTRODUCTION

About 50% of Colorado's annual statewide duck harvest occurs in five counties (Logan, Morgan, Sedgwick, Washington, and Weld) along the lower South Platte River (SPR) corridor in northeastern Colorado (U.S. Fish and Wildlife Service, unpublished harvest survey results). Over 60% of Colorado duck hunters hunt in this area, and a majority of these hunters hunt exclusively or regularly on public lands (Colorado Division of Wildlife 2006). There are 26 State Wildlife Areas (SWAs) located in the SPR corridor from Greeley to the state line, and duck hunting is a major activity and management emphasis on many of these areas. The Colorado Division of Parks and Wildlife (CPW) historically has managed to provide a range of duck hunting opportunities on SWAs along the SPR corridor. Some properties have no restrictions on hunting beyond the statewide regulations, and the management emphasis is on maximizing hunting opportunity. On other properties, CPW has attempted to address issues of hunting quality in part through property-specific restrictions in hunting regulations. Property-specific restrictions include requiring reservations for access, day closures (portions of the week when no hunting is allowed), and assigned areas. Use of hunting restrictions has been largely on an ad hoc, property-specific basis. No rigorous evaluation has been conducted on the effectiveness of restrictive hunting regulations on duck distribution or on hunter success, activity, or satisfaction.

Since the 1980s the annual midwinter index of ducks counted in the SPR corridor has averaged less than half the number counted during the 1970s. The possibility exists that detection probability decreased over those years, but it is unlikely that it decreased by 50%; thus winter abundance of ducks in the SPR has likely declined. Although overall duck harvest during 1999-2006 has been comparable to historic levels, in recent years there have been increasing concerns about the quality of duck hunting along the SPR corridor, particularly on SWAs. There is a desire to increase wintering populations of ducks, increase harvest success (i.e., average number of ducks bagged per hunter day), and recruit and retain more duck hunters. It has been suggested that disturbance from excessive hunting activity along the SPR corridor has led to decreased use of this area by ducks, poor harvest success, over-crowding and interference among hunters on public areas, and unsatisfactory experiences for duck hunters. This concern is supported by the results of a 2005 national duck hunter survey (National Flyway Council and Wildlife Management Institute 2006), in which 66% of Colorado duck hunters surveyed (n = 488) reported they believed hunting pressure had become worse compared to five years prior to the survey, 65% of hunters believed crowding was worse at hunting areas, 53% reported more interference from other hunters, and 50% believed ducks were more concentrated on fewer areas. Dissatisfaction with duck hunting could in turn result in declining duck hunter recruitment and retention. Concerns over the quality of duck hunting along the SPR have led to proposals to increase hunting restrictions in this area.

Recent monitoring of duck hunter activity and harvest on South Platte SWAs indicates that patterns of public use and duck harvests are variable among SWAs and on individual SWAs among years. Voluntary reporting data suggest that average duck harvest/hunter trip was similar between public areas with restrictive hunting regulations and areas without restrictive regulations in 2004-2005 and 2005-2006, but higher in unrestricted areas in 2006-2007. Patterns of hunter use and harvest success may vary among properties in relation to the property size and the habitat types present on the property (e.g., shallow marsh impoundments, river channels, warm-water sloughs). Harvest success, particularly on properties adjacent to the river channel, was weather-dependent: harvest success increased during colder, wetter duck seasons, and within a duck season harvest success was higher when temperatures were colder. Ducks use large reservoirs that act as refuge areas within the SPR corridor, and ducks often move to feeding areas after dark. Duck use of the river is limited until low temperatures cause reservoirs to freeze and the river provides the only available open water.

It is generally acknowledged that disturbance from hunting activity can influence the distribution of ducks at a variety of spatial scales (Baldassarre and Bolen 1994). Ducks quickly find refuge areas when hunting seasons begin, and alter their spatial and temporal activity patterns to avoid hunted areas (Cox and Afton 1998a, Fleskes 2002), although refuge size and habitat conditions may influence their use and value to waterfowl (Rave and Cordes 1993, Cox and Afton 1998b, Rave 1999, Cox and Afton 1999). Numerous studies have documented anthropogenic disturbance to waterfowl (Dahlgren and Korschgen 1992, Madsen 1995, Madsen and Fox 1995, Fox and Madsen 1997, Madsen 1998a, 1998b; Evans and Day 2001, 2002; Pease and Butler 2005). Most studies that examine hunting impacts compare bird use, usually measured by counts, on sanctuary or refuge areas (i.e., no hunting or other disturbance) to hunted areas, rather than comparing different levels or types of hunting disturbance. On a Danish wetland where hunting was permitted only once every one to three weeks, Bregnballe and Madsen (2004) determined the proportion of waterfowl occupying the wetland just prior to hunts that returned within one to days days after hunts, and found that response to hunting disturbance was variable among species and within species in relation to habitat conditions. Using a similar approach, Bregnballe et al. (2004) concluded that restricting hunting to the afternoon did not adequately reduce disturbance to maintain bird numbers and diversity. In addition, most studies focus exclusively on bird responses, but do not document changes in hunter activity, success, or satisfaction in relation to creation of refuges. Madsen (1998b) noted that following creation of refuge areas on two Danish wetlands, hunter numbers declined on hunted portions of one area, and numbers did not decline but were redistributed on the other wetland; hunter success was not reported. Hockin et al. (1992) and Hill et al. (1997) reviewed literature on studies investigating disturbance to birds from human activity and reported that most results were anecdotal, with only a small minority of studies having some sort of experimental design that compared control and treatment areas. They recommended increased use of manipulative studies to more rigorously assess impacts of disturbance or the effectiveness of controls on disturbance.

Relationships between federal frameworks for hunting (e.g., Flyway-specific season lengths and bag limits) and resulting duck harvests have been investigated at national and regional scales (Martin and Carney 1977), but few studies have been conducted to examine the influence of local-scale hunting regulations on hunter success or satisfaction. Hunting parties were assigned one of three alternative bag limit regulations (a two-bird limit, Flyway-specific regulations, or point system) and their performance and satisfaction were measured on a state game area during one season in Michigan (Mikula et al. 1972). However, this study did not examine impacts of regulations other than bag limit restrictions, and variation across years or among areas was not investigated. During 1963-1970, the CPW, in cooperation with the U.S. Fish and Wildlife Service, conducted intensive studies examining how local duck populations and duck hunters responded to various experimental duck hunting regulations in the San Luis Valley (Hopper et al. 1975). However, this study did not did not directly compare results to more restrictive regulatory approaches, and did not examine harvest success or hunter satisfaction in relation to hunting regulations at a more local scale.

Given the interest in reducing duck hunting pressure in the SPR corridor, there is a need to evaluate how more restrictive hunting regulations impact duck numbers and distribution, and hunter success and satisfaction, at local and regional scales. Here we summarize methods and results from the first three years of a management experiment in SWAs along the SPR corridor that examines this issue.

STUDY AREA AND METHODS

This study is being conducted in the SPR corridor between Greeley and the state line (Fig. 1). On seven non-randomly selected SWAs (see table below), we are using a quasi-experimental cross-over design to examine the influence of hunting restrictions on selected response variables. Properties were selected to represent the range of wetland habitat types on SWAs along the SPR, including areas off the river channel with shallow, seasonally-flooded wetland impoundments near large reservoirs; small properties on the river channel that have little other wetland habitat; and larger properties on the river channel that have more diverse wetland habitats. For each pair of properties with these habitat conditions, each member of the pair was assigned a different set of hunting regulations. On “Unrestricted” properties, no additional hunting restrictions are applied for waterfowl and small game hunting beyond the regulations that apply throughout eastern Colorado. A set of additional regulations are applied to “Restricted” properties, intended to limit hunting disturbance while still providing some hunting opportunity. These regulations include: (1) reservations are required for hunting access (a limited number of parties on the property, with no more than 4 hunters per party); (2) all parties must leave the property by 2 p.m.; (3) hunting is allowed only on Saturdays, Sundays, Wednesdays, and legal holidays; and (4) hunting parties are assigned to specific areas on the property. These restrictions apply to waterfowl and small game hunting during the regular duck hunting season, but not to deer and fall turkey hunting. The study design calls for Restricted (R) and Unrestricted (U) regulations will be applied to the selected properties for six years as described in the table below. A cross-over design is being used to account for site-specific influences on response variables for each pair of properties. Note that the crossover began with the 2011-12 duck season.

		Hunting Season Regulations (R = Restricted, U = Unrestricted)					
Type	**State Wildlife Area**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Off river channel	Jackson Lake	R	R	R	U	U	U
	Red Lion	U	U	U	R	R	R
On-channel small property	Overland Trail	R	R	R	U	U	U
	Atwood	U	U	U	R	R	R
On-channel large property	Bravo	R	R	R	U	U	U
	Jean K. Tool & Brush	U	U	U	R	R	R

Check stations were established at these 7 SWAs, and access to these areas was from designated parking areas only. During the regular duck hunting season, all waterfowl and small game hunters were required to check out at the check station before leaving the property. A check station attendant recorded information on the hunters, their harvest, hunting methods, and measures of satisfaction (Appendix A). Voluntary hunter check-out cards requesting the same information were also provided in case a check station attendant was not present when hunters checked out.

In past years, significant ice buildup was noted on the ponds at Jackson Lake and Red Lion SWAs by the end of November. For comparative purposes, data from before 23 November and on or after 23 November are summarized separately for these two SWAs. In 2011-12, the duck season dates were 8 October – 4 December and 23 December – 29 January, with the season closed during December

5-22. During 23 December 2011 – 29 January 29 2012, check station attendants were no longer assigned to Jackson Lake and Red Lion SWAs, and we relied on hunters filling out voluntary check-out cards.

Check station attendants recorded the license plate numbers of all vehicles at all study SWA parking lots daily throughout the season, and recorded license plate numbers gain as hunting parties checked out at check station. Whenever possible, attendants identified vehicles that belonged to people other than waterfowl and small game hunters (e.g., deer hunters). We used the proportion of total vehicles (excluding vehicles present for other uses) that checked out as an index to compliance with the requirement that all waterfowl and small game hunters check out during the regular duck station at study SWAs.

Aerial surveys of the SPR corridor from Greeley to the state line were conducted monthly during the regular duck hunting season (14 October, 16 November, 16 December, and 4 January) to provide an index to overall waterfowl numbers and distribution in the region. Observers recorded numbers and locations of ducks and geese on the river and associated sloughs, as well as ponds and reservoirs in the SPR corridor. Photographs were taken of a subset of areas counted where large concentrations of waterfowl occurred, and the number of waterfowl in each photograph was tallied to provide a more accurate count at these locations.

RESULTS AND DISCUSSION

During the 2011-12 waterfowl hunting season, we obtained harvest and satisfaction measures from 929 hunting parties. Of these, 711 (77%) were duck hunting parties. These totals are lower than during the three previous years of the study, which averaged 1,352 total hunting parties and 1,018 duck hunting parties per year. Part of the decline in hunting activity in 2011-12 might have been due to the changes in regulations that occurred this year. For example, Jean K. Tool/Brush SWAs were unrestricted and averaged 530 duck hunting parties per season during the previous three seasons, but were restricted during 2011-12 and only 150 duck hunting parties used these SWAs, a 72% decrease. These areas were paired with Bravo SWA, which averaged 80 duck hunting parties during the previous three years when restricted regulations were in place, but duck hunting parties actually decreased 15% to 68 in 2011-12 when unrestricted regulations were in place at Bravo SWA. At Jackson Lake SWA, where regulations switched from restricted during the previous three years to unrestricted in 2011-12, the number of duck hunting parties in 2011-12 increased 244% (361 parties) above the average during the previous three years (148 parties). Jackson Lake SWA is paired with Red Lion SWA, which averaged only 173 duck hunting parties per year over the previous 3 years when regulations were unrestricted; the number of duck hunting parties decreased 35% to 113 in 2011-12 when restricted regulations were put in place at Red Lion SWA. It appears that hunting activity is impacted differently by restricted versus unrestricted regulations at different study SWAs. In addition, although the overall length of the duck season was the same in 2011-12 as during the previous three years, the timing of the split (closed period) in the season was changed during 2011-12; this change may have had some (unmeasured) effect on the hunting activity of hunters who normally use the study SWAs. Finally, we are investigating whether the decline in hunter activity on study SWAs is biased toward hunters with certain characteristics (e.g., novice or casual hunters versus experienced or highly active hunters), to determine whether our research activities (i.e., check stations and hunter interviews) are introducing some unintended influence on the behavior of the pool of hunters who would normally use these SWAs.

During the 2011-12 season we interviewed 620 duck hunting parties, and 91 additional duck hunting parties left checkout cards at unmanned check stations. Jackson Lake SWA (unrestricted regulations) had the highest use, with 341 duck hunting parties and 737 duck hunter-days, and Overland Trail SWA (unrestricted regulations) had the lowest use, with 15 duck hunting parties and 35 duck hunter-days (Table 1).

Overall, 30% of duck hunters at the 7 study SWAs were in their first year of hunting the lower SPR corridor, 16% had hunted the area for two years, 10% for three years, 4% for four years, and 40% for five years or more. Most (83%) of the duck hunters surveyed hunted mainly public lands, 4% hunted mainly private lands, and 13% said they hunted both equally. The average duck hunting party size ranged from 1.7 on Bravo SWA to 2.3 on Atwood and Overland Trail SWAs (Table 1). Across all 7 SWAs, 85% of all duck hunting parties used standard decoys, 50% used spinning wing decoys, 42% used dogs, and 87% of used duck calls.

A total of 1,565 ducks was reported harvested on the seven study SWAs during the 2011-12 season (Table 2). The species was identified for 1,459 (93%) of the ducks harvested, and of the identified ducks, six species of dabbling ducks (mallard, northern shoveler, green-winged teal, gadwall, blue-winged teal, and American wigeon) comprised 83% of the duck harvest. Waterfowl hunters also harvested 80 geese at study SWAs, and small game hunters harvested a variety of other migratory game birds, upland game birds, and small mammals at these seven SWAs (Table 2).

Season-long duck harvest success was measured for each SWA as ducks bagged per hunter per party per day over the 2011-2012 regular duck season. From interview data, harvest success was greater at restricted areas than unrestricted areas of similar size and habitat type: season-long harvest success was four times greater at Atwood SWA than Overland trail SWA, 2.7 times greater at Jean K. Tool/Brush SWAs than Bravo SWA, and 2 times greater at Red Lion SWA than Jackson Lake SWA (Table 1).

As in previous years, frequency distributions of ducks shot per hunter per day showed that small on-channel properties (Atwood and Overland Trail SWAs) had the largest proportion of hunting parties with 0 ducks bagged and most successful hunting parties on these areas bagged <2 ducks per hunter (Figure 2). Hunters experienced slightly fewer 0 bag days on the large on-channel properties (Jean K. Tool/Brush and Bravo SWAs) and a greater percentage of hunting parties bagged >2 ducks per hunter (Figure 3). Hunters had greater success on off-channel properties (Red Lion and Jackson Lake SWAs), with fewer 0 bag days and more hunting parties with >2 ducks bagged per hunter (Figure 4). Across all study areas, average daily duck bag per hunter was variable over the course of the season, with generally higher success early in the season, and lower success late in the season (Figure 5a). Duck hunter activity was also variable across the season, with generally greater numbers of hunters per day on study SWAs early in the season, and fewer hunters in January (Figure 5b). The maximum number of duck hunters on the 7 study SWAs occurred on opening day of the regular duck season (October 8) at 34 hunters.

Satisfaction with number and proximity of other hunters (i.e., 'crowding') was high at all areas, ranging from an average rank of 4.0 (on a scale of 1-5) at Jackson Lake SWA (unrestricted) to an average of 4.8 at Bravo (unrestricted) and Red Lion (restricted) SWAs (Table 3). Among SWAs that switched from unrestricted regulations during the previous 3 seasons to restricted regulations in 2011-12, satisfaction with crowding decreased at Atwood SWA (average of 4.8 during the previous 3 seasons versus 4.3 during 2011-12) and increased at Jean K. Tool/Brush SWAs (4.4 versus 4.7) and Red Lion SWA (4.1 versus 4.8). Among SWAs that switched from restricted regulations during the previous 3 seasons to unrestricted regulations in 2011-12, satisfaction with crowding decreased slightly at Overland Trail SWA (4.5 versus 4.4) increased at Bravo SWA (4.6 versus 4.8) and decreased at Jackson Lake SWA (4.4 versus 4.0).

Satisfaction with property-specific hunting regulations was generally favorable at all areas, ranging from an average rank of 3.5 at Overland Trail SWA (unrestricted) to an average of 4.0 at Red Lion (restricted) and Jackson Lake (unrestricted) SWAs (Table 3). Compared to mean satisfaction rank of hunting regulations during the previous 3 seasons, mean satisfaction rank during 2011-12 increased at Bravo SWA (3.7 during the previous 3 seasons versus 3.9 during 2011-12), and decreased at all other study SWAs: 4.3 versus 3.8 at Atwood, 3.9 versus 3.5 at Overland Trail, 4.2 versus 3.9 at Jean K. Tool/Brush, 4.1 versus 4.0 at Red Lion and Jackson Lake.

Satisfaction with bird numbers was ranked lower than other measures at all SWAs, whereas satisfaction with habitat conditions was ranked favorably at all SWAs (Table 3). As in previous years, average satisfaction with bird numbers was higher at the off-channel properties (Red Lion and Jackson Lake SWAs) than other properties, with the exception in 2011-12 that Atwood SWA was ranked similarly to the off-channel properties. Additionally, habitat conditions were ranked highest at off-channel properties during previous seasons, but in 2011-12 Atwood SWA was ranked higher than off-channel properties.

Overall satisfaction with the day's hunt was generally ranked favorably at all study SWAs (Table 3). Overall satisfaction ranks were equal to or higher than the ranks during the previous three seasons at all SWAs except Jackson lake SWA, where overall satisfaction averaged 3.6 during 2011-12, compared to an average of 3.9 over the three previous seasons.

We estimated correlation coefficients between average ranks of satisfaction for crowding, hunting regulations, duck numbers seen, overall satisfaction, and average ducks shot per hunter per day on each study SWA (Table 4). Correlation coefficients provide a rough estimate of the effect these factors have upon one another. A correlation coefficient of 1.0 suggests a perfect positive correlation between two factors, and -1.0 suggests a perfect negative correlation between two factors. A correlation coefficient of 0.0 suggests no correlation between two factors. Although there were no strong correlations among various measures of satisfaction, satisfaction with bird numbers was most closely positively correlated with overall satisfaction with a day's hunt on all SWAs (range 0.42 – 0.47) except Overland Trail SWA, where there was a negative correlation (-0.37) between satisfaction with bird numbers and overall satisfaction (Table 4).

Our index to compliance with the requirement that all waterfowl and small game hunters check out at check stations was similar to that seen in previous years. Compliance was lowest (79%) at Brush and Bravo SWAs, but exceeded 90% at all other study SWAs (Table 5).

Indices of ducks during aerial surveys of the SPR corridor were relatively high during October (44,704) compared to previous years, increased substantially to 82,996 in November before declining to 63,482 in December, and increased again to 98,017 in January (Figure 6). Unusually warm weather occurred in late December 2011 and January 2012, and a large proportion of ducks were concentrated on open water in large reservoirs along the SPR corridor.

This study is expected to continue for 2 more years, with assignments of regulations to study SWAs the same as in the 2011-2012 hunting season. Data collection will resume in October 2012.

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Table 1. Statistics associated with duck hunting parties on selected State Wildlife Areas (SWAs) along the South Platte River corridor during 2010-2011. Percent statistics are the percent of parties that used standard decoys, spinning wing decoys, dogs, or duck calls.

SWA	Total parties	Total hunter days	Avg. hunters in party	Avg. total duck harvest	Avg. ducks /hunter /day	% Parties using decoys	% Using spinning wing	% Using dogs	% Using duck calls
Interviews									
Atwood (R)	22	51	2.3	0.9	0.4	77	45	45	82
Overland Trail (U)	14	32	2.3	0.1	0.1	64	36	43	71
Jean K Tool /Brush (R)	133	293	2.2	1.6	0.8	84	34	44	84
Bravo (U)	56	95	1.7	0.5	0.3	64	13	29	76
Red Lion (R)	89	196	2.2	4.7	2.2	96	58	58	94
Jackson Lake (U)	306	673	2.2	2.5	1.1	90	66	35	90
Check-out Cards									
Atwood (R)	2	3	1.5	0.0	0.0	0	0	100	0
Overland Trail (U)	1	3	3.0	0.0	0.0	0	0	100	100
Jean K Tool /Brush (R)	17	33	1.9	1.9	1.0	73	40	47	87
Bravo (U)	12	21	1.8	0.2	0.1	55	9	64	55
Red Lion (R)	4	9	2.3	3.0	1.4	75	75	25	67
Jackson Lake (U)	14	32	2.3	0.7	0.3	85	69	46	100
Interviews & Cards									
Atwood (R)	24	54	2.3	0.8	0.4	74	43	48	78
Overland Trail (U)	15	35	2.3	0.1	0.0	51	33	47	73
Jean K Tool /Brush (R)	150	326	2.2	1.6	0.8	53	34	44	85
Bravo (U)	68	116	1.7	0.4	0.3	62	12	35	73
Red Lion (R)	93	205	2.2	4.6	2.1	93	53	55	91
Jackson Lake (U)	320	705	2.2	2.5	1.1	90	64	37	91
After ice-up (Cards)									
Red Lion (R)	20	32	1.6	1.1	1.0	80	13	40	75
Jackson Lake (U)	21	32	1.5	1.6	1.2	81	43	57	86

Table 2. 2011-2012 harvest totals for all waterfowl and small game reported at the 7 study SWAs during the regular duck season.

Species	Atwood	Overland	JKT	Brush	Bravo	Red Lion	Jackson	**Total**
Ducks								
Mallard	18	1	101	11	20	137	109	**397**
Northern shoveler	0	0	4	0	0	49	164	**217**
Green-winged teal	1	0	17	1	1	44	120	**184**
Gadwall	0	1	29	5	0	43	89	**167**
Blue-winged teal	0	0	1	2	0	50	78	**131**
American wigeon	0	0	14	9	0	38	50	**111**
Redhead	0	0	0	0	0	21	60	**81**
Northern pintail	0	0	2	0	0	32	32	**66**
Wood duck	0	0	7	3	3	2	15	**30**
Mergansers	1	0	2	0	0	1	15	**19**
Ring-necked duck	0	0	0	0	0	1	14	**15**
Bufflehead	0	0	0	0	0	0	15	**15**
Scaup	0	0	0	0	0	2	9	**11**
Canvasback	0	0	0	0	0	4	4	**8**
Goldeneyes	0	0	1	0	2	0	3	**6**
Ruddy duck	0	0	0	0	0	0	1	**1**
Unspecified duck	0	0	27	5	3	29	42	**106**
Total Ducks	**20**	**2**	**205**	**36**	**29**	**453**	**820**	**1,565**
Geese								
Canada Goose	6	0	8	12	4	32	9	**71**
Snow Goose	0	0	0	0	4	4	1	**9**
Other small game								
Quail	20	3	3	16	62	0	0	**104**
Pheasant	4	0	1	1	3	0	10	**19**
American coot	0	0	0	0	0	5	12	**17**
Rabbit	1	0	1	3	4	0	0	**9**
Dove	1	0	0	0	3	0	0	**4**
Squirrel	0	0	2	0	0	0	1	**3**
Sandhill crane	0	0	0	0	0	0	3	**3**

Table 3. Average satisfaction measures of duck hunting parties on selected State Wildlife Areas (SWAs) along the South Platte River corridor during 2010-2011. Scale is 1 through 5, with 1 being the least favorable and 5 being the most favorable. SWAs are designated as Restricted (R) or Unrestricted (U) based on property regulations.

SWA	Total parties	Crowding	Bird numbers	Habitat conditions	Hunting regulations	Overall
Interviews						
Atwood (R)	22	4.3	3.1	4.5	3.9	4.0
Overland Trail (U)	14	4.4	2.1	4.1	3.5	3.5
Jean K Tool /Brush (R)	133	4.7	2.5	4.3	4.0	3.5
Bravo (U)	56	4.8	2.4	4.1	4.1	3.8
Red Lion (R)	89	4.8	3.4	4.1	4.0	4.0
Jackson Lake (U)	304	4.0	2.9	4.5	4.0	3.6
Check-out Cards						
Atwood (R)	2	4.5	1.0	5.0	2.0	5.0
Overland Trail (U)	0	-	-	-	-	-
Jean K Tool /Brush (R)	14	4.7	2.3	3.8	3.4	3.4
Bravo (U)	12	4.6	2.3	3.8	3.0	2.9
Red Lion (R)	4	5.0	2.8	3.3	3.3	3.3
Jackson Lake (U)	12	3.7	2.8	4.2	4.1	3.5
Interviews & Cards						
Atwood (R)	24	4.3	3.0	4.5	3.8	4.1
Overland Trail (U)	14	4.4	2.1	4.1	3.5	3.5
Jean K Tool /Brush (R)	147	4.7	2.5	4.2	3.9	3.5
Bravo (U)	67	4.8	2.4	4.1	3.9	3.6
Red Lion (R)	93	4.8	3.4	4.1	4.0	4.0
Jackson Lake (U)	316	4.0	2.9	4.4	4.0	3.6
After ice-up (Cards)						
Red Lion (R)	16	4.3	2.9	3.8	3.0	3.3
Jackson Lake (U)	20	4.2	3.5	4.4	4.3	3.7

Table 4. Correlation coefficients between some of the satisfaction measures from duck hunting parties at selected State Wildlife Areas (SWAs) along the South Platte River corridor during the 2011-2012 regular duck season.

SWA	Factor	Crowding	Bird numbers	Hunting regulations	Avg. ducks /hunter /day
Atwood (R)	Bird numbers	0.11			
	Hunting regulations	-0.01	-0.03		
	Avg. ducks /hunter /day	0.19	0.28	-0.31	
	Overall	0.37	0.45	0.32	0.19
Overland Trail (U)	Bird numbers	-0.30			
	Hunting regulations	0.21	-0.29		
	Avg. ducks /hunter /day	0.19	0.20	-0.15	
	Overall	0.44	-0.37	0.03	0.19
Jean K Tool / Brush (R)	Bird numbers	0.16			
	Hunting regulations	-0.05	-0.22		
	Avg. ducks /hunter /day	0.08	0.41	-0.02	
	Overall	0.09	0.44	0.29	0.31
Bravo (U)	Bird numbers	-0.07			
	Hunting regulations	-0.12	0.13		
	Avg. ducks /hunter /day	0.14	0.44	0.21	
	Overall	-0.08	0.47	0.08	0.22
Red Lion (R)	Bird numbers	-0.15			
	Hunting regulations	-0.03	0.08		
	Avg. ducks /hunter /day	0.00	0.33	0.04	
	Overall	0.09	0.45	0.30	0.41
Jackson Lake (U)	Bird numbers	0.05			
	Hunting regulations	0.19	0.16		
	Avg. ducks /hunter /day	0.02	0.44	-0.04	
	Overall	0.22	0.42	0.16	0.40

Table 5. Compliance index (number of vehicles present in parking lots, and number that checked out at check stations), and numbers of vehicles using study SWA parking lots for purposes other than waterfowl and small game hunting during 2011-2012 regular duck season.

	Vehicles that checked out			Vehicles present for other uses					
SWA	Yes	No	% compliance	Deer hunting	Fishing	Turkey hunting	Scouting	Walking	Other
Atwood (R)	39	4	91	4					
Overland Trail (U)	26	0	100	19	1		2		
Brush (R)	52	14	79	15	2				1
Jean K. Tool (R)	102	1	99	2	1	1			
Bravo (U)	129	35	79	38		1	3	7	4
Red Lion (R)	100	8	93	2			2		
Jackson (U)	355	21	94		4		2	1	3
Total	803	83	91	80	8	2	9	8	8

Figure 1. South Platte River corridor from Greeley to the state line, showing State Wildlife Areas included in the study.

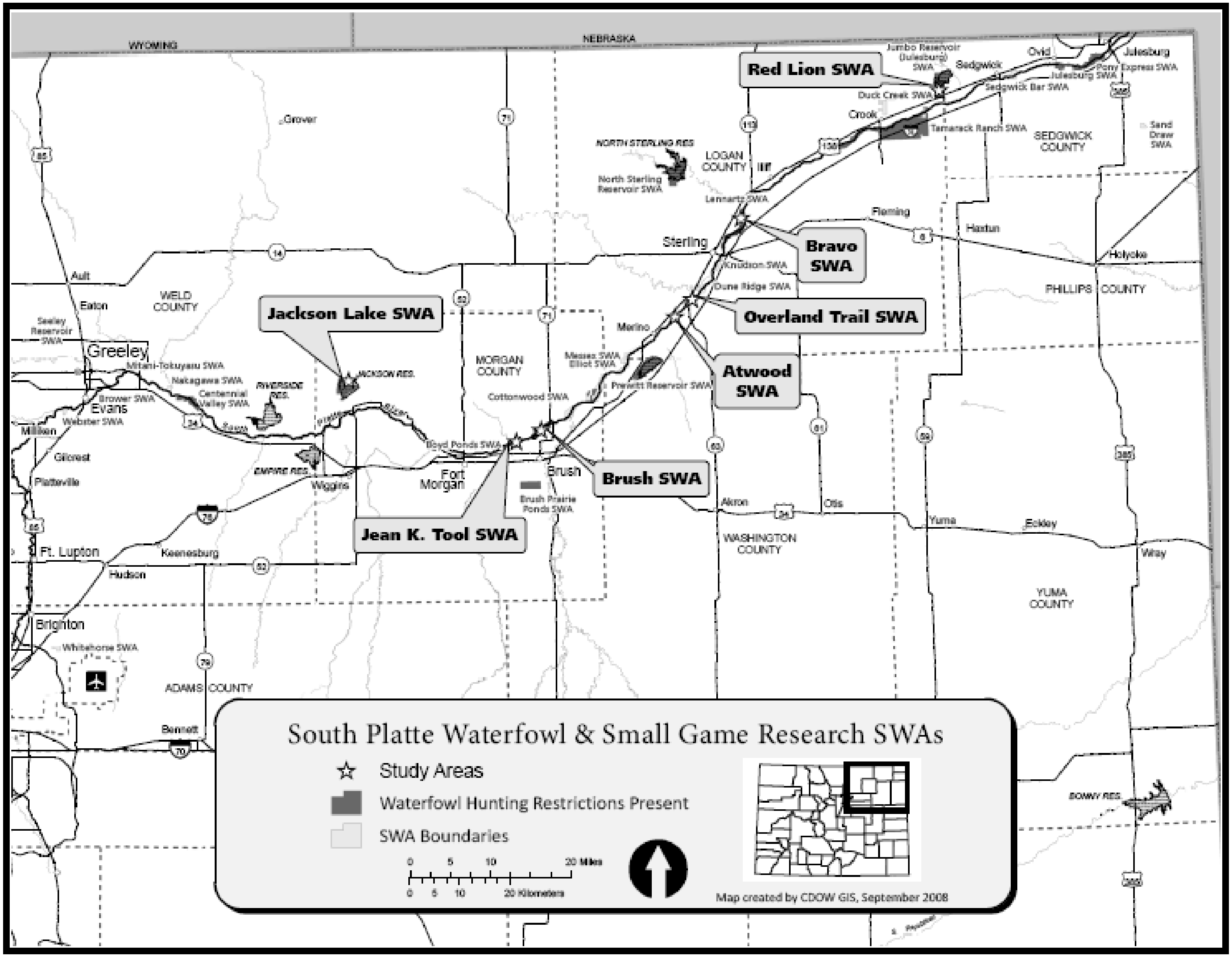


Figure 2. Distribution of average ducks harvested per hunter per day for parties hunting Atwood (Restricted) and Overland Trail (Unrestricted) SWAs during the 2011-2012 regular duck season.

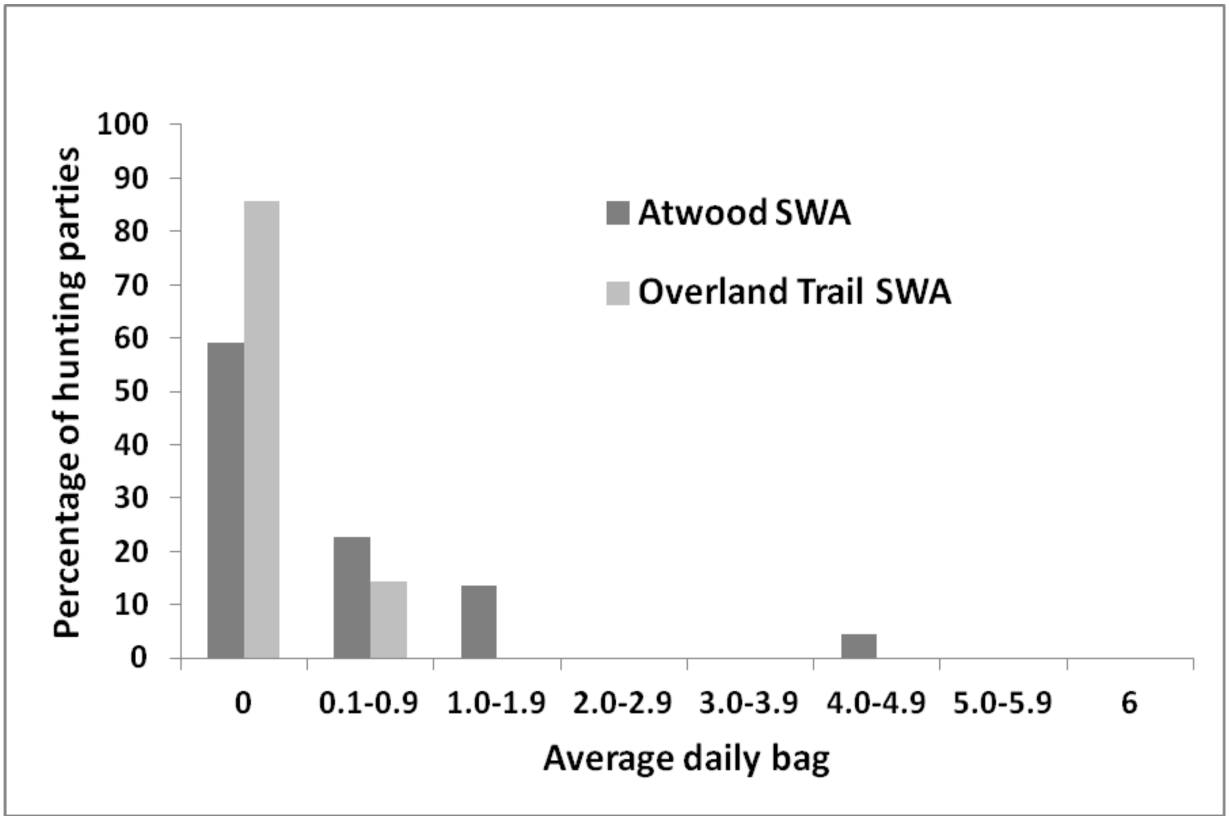


Figure 3. Distribution of average ducks harvested per hunter per day for parties hunting Jean K. Tool/Brush (Restricted) and Bravo (Unrestricted) SWAs during the 2011-2012 regular duck season.

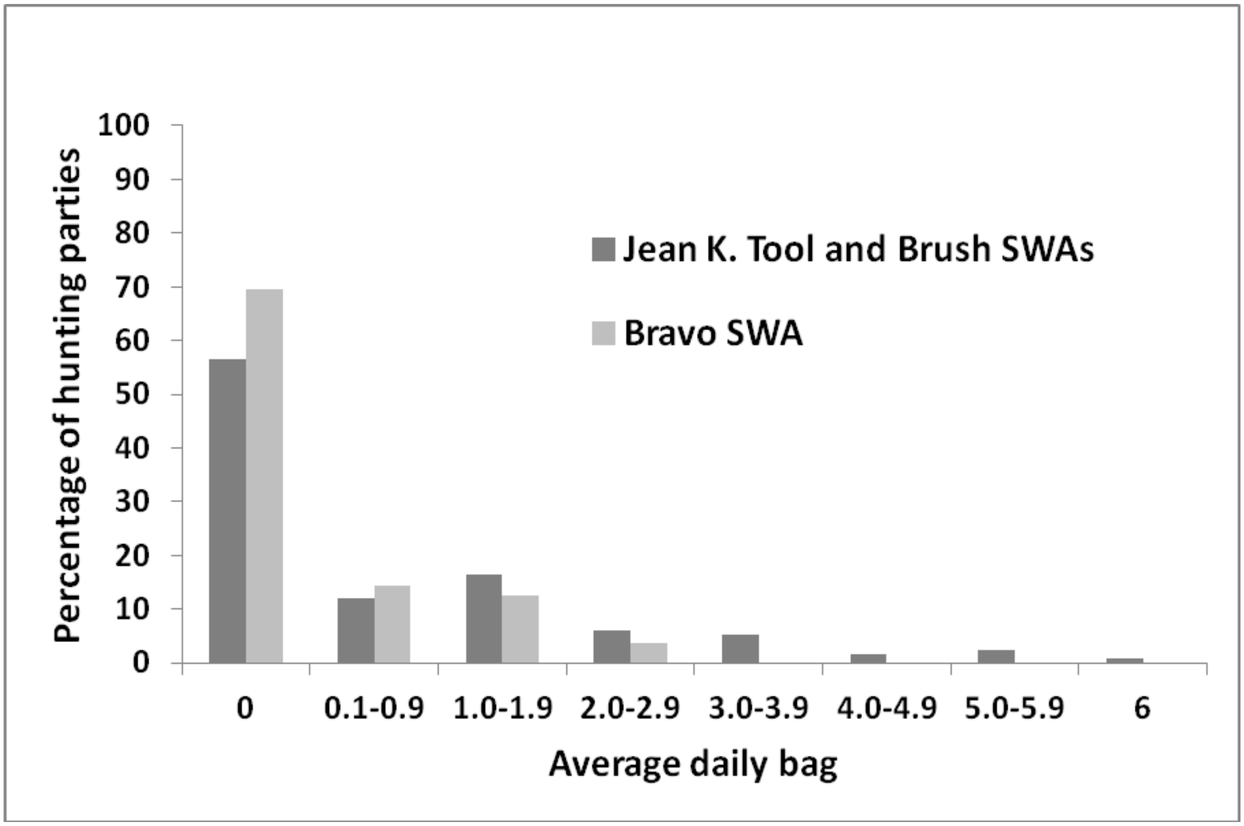


Figure 4. Distribution of average ducks harvested per hunter per day for parties hunting Red Lion (Restricted) and Jackson Lake (Unrestricted) SWAs during the 2011-2012 regular duck season.

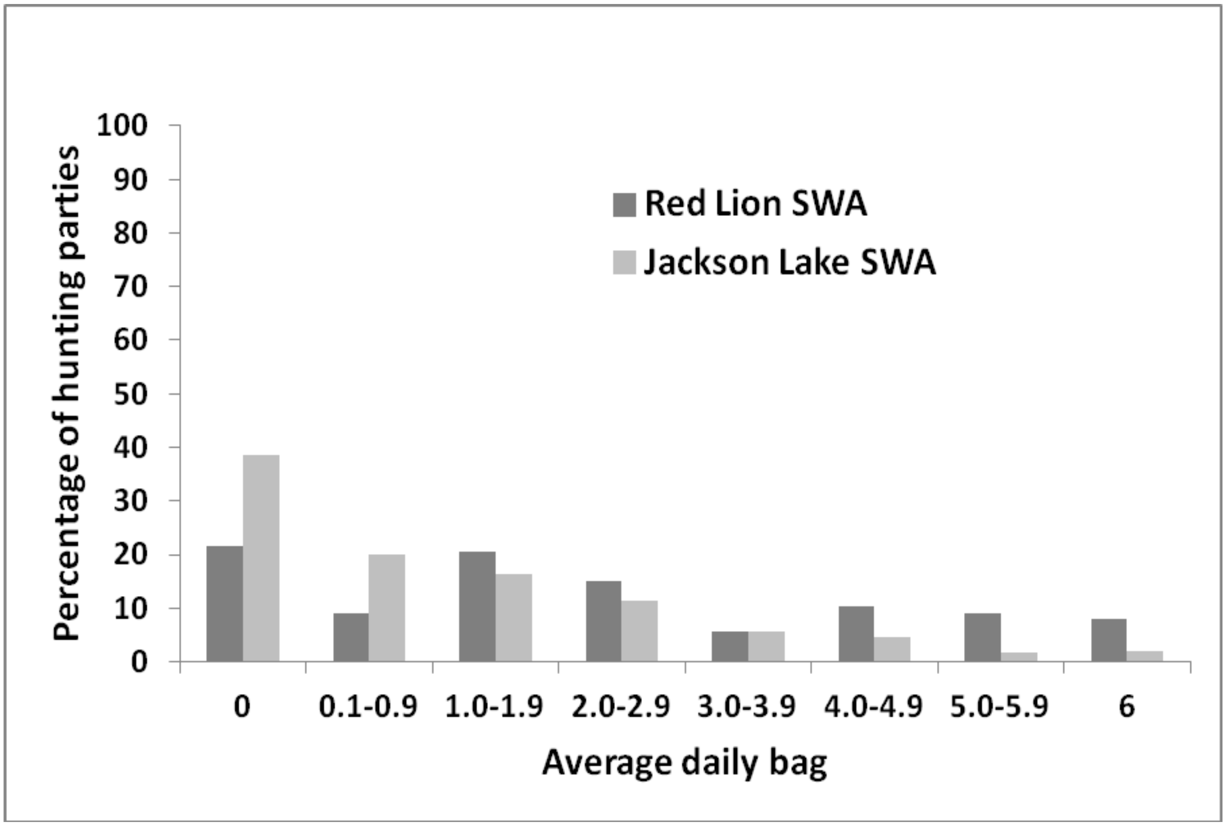
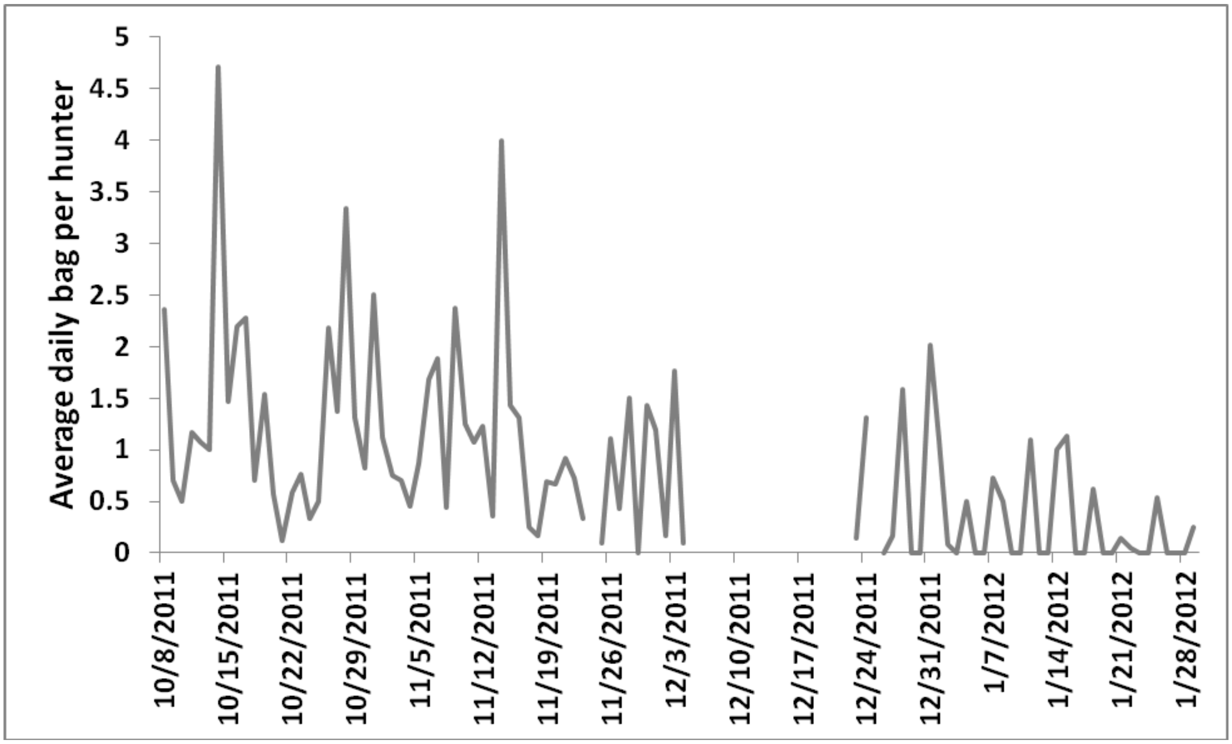


Figure 5. Daily duck hunting success (a) and (b) hunter activity on 7 SWAs along the South Platte River corridor during the 2011-2012 regular duck season. The season was closed during 5-22 December.

a.



b.

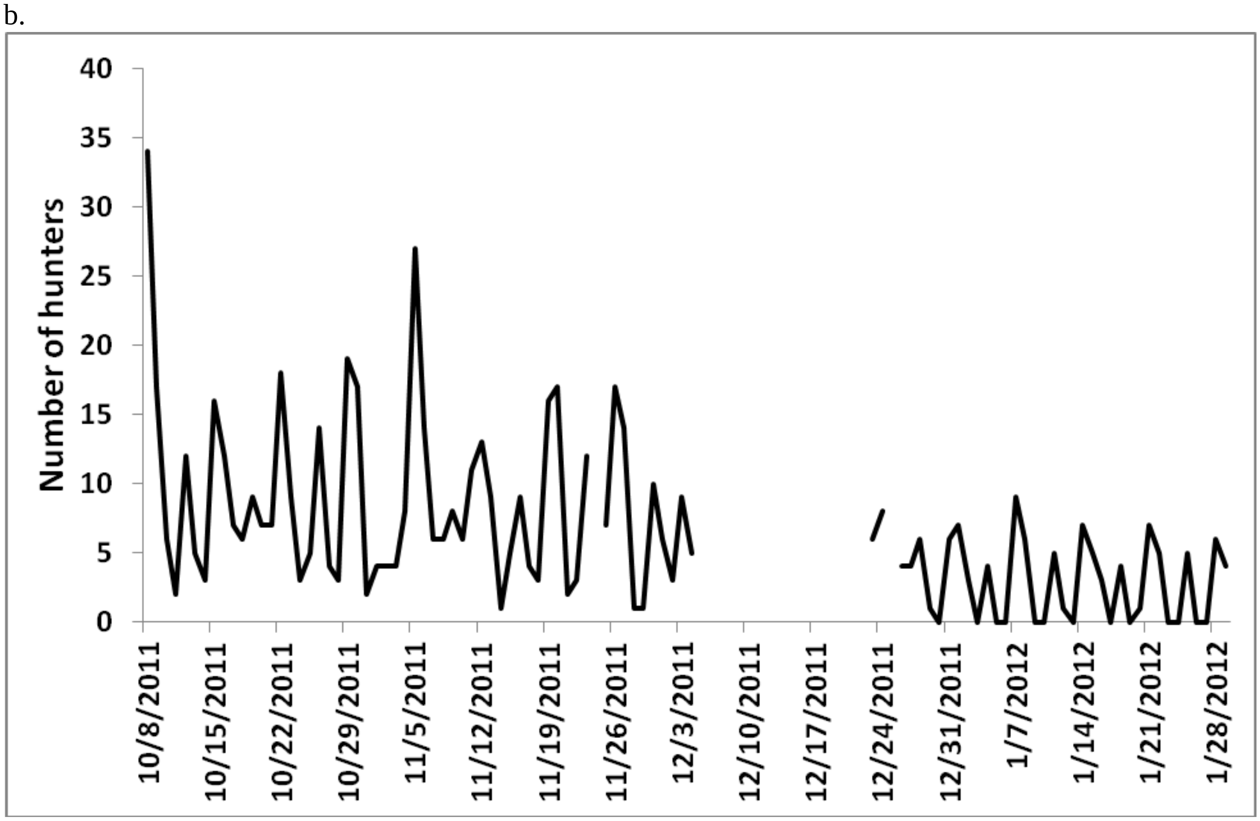
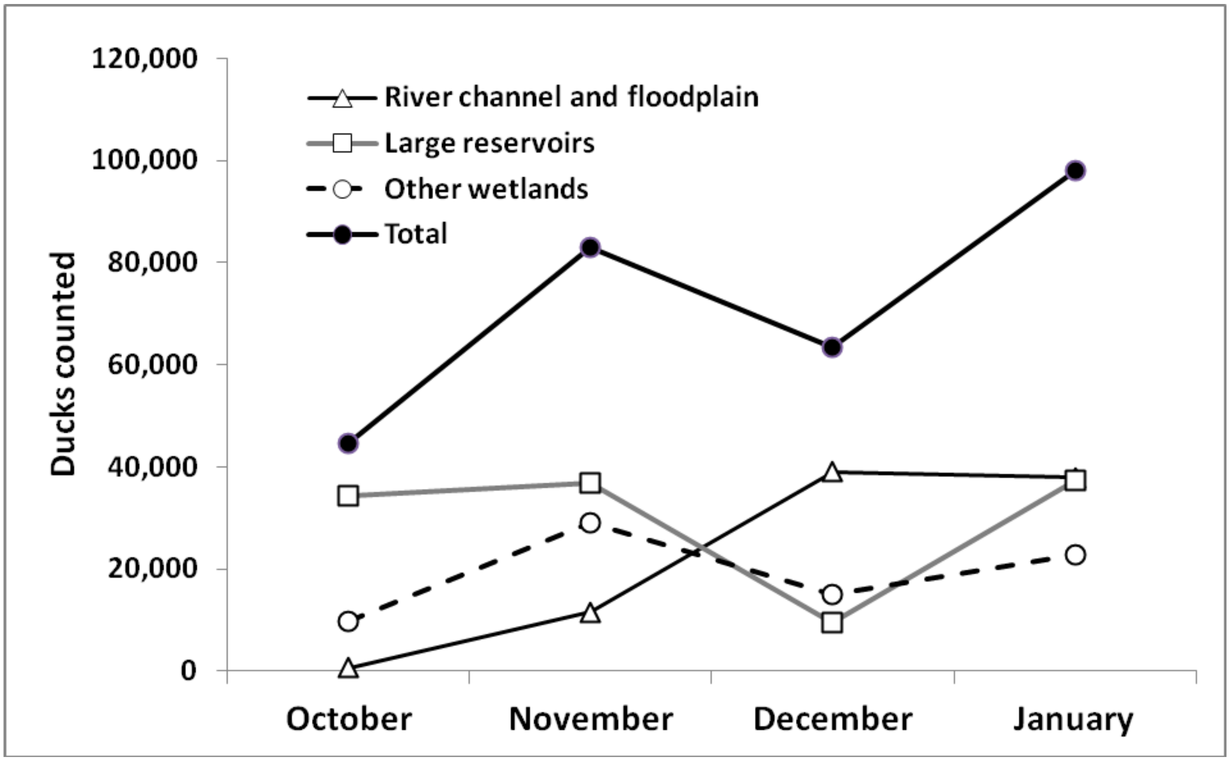


Figure 6. Indices of duck numbers in the South Platte River (SPR) corridor from October 2011 through January 2012.



Appendix A. Information collected from waterfowl and small game hunters on selected State Wildlife Areas along the South Platte River during the 2011-2012 regular duck hunting season.

South Platte River Corridor State Wildlife Area Hunting Study

State Wildlife Area ____________________________ Date __________________ Initials _________
Number in hunting party _____ Party arrival time ___________ Party departure time ____________
Parking Lot/Hunt Zone___________________ License plates __________________________________

CID number	Sex	Seasons out of last 5 hunted on SPR (counting this year, 1-5)?	Mostly public	Mostly private	Equal

Target Species (e.g., ducks, quail, squirrels, etc):

Harvest	Drake in plumage	Brown	Notes
Mallard			
Blue-winged teal			
American wigeon			
Gadwall			
Northern shoveler			
Wood duck			
Pheasant			
Bobwhite quail			

Decoys (# in dozens)? _____ Spinning-wing decoys (#)? ___ Dogs (#)? ______ Calls (Y/N)?_______ _______

Rank the following from 1 to 5 for today's hunt:

Crowding problems (1 = extreme crowding problems, 5 = no crowding problems) ________

Bird/game numbers seen (1 = no birds seen, 5 = abundant numbers of birds seen) ________

Habitat conditions on the area (1 = very poor, 3 = average, 5 = excellent conditions) ________

Current hunting regulations on the SWA (1 = very dissatisfied, 3 = neutral, 5 = very satisfied) ________

Overall satisfaction with the hunt (1 = very dissatisfied, 3 = neutral, 5 = very satisfied) ________