THESIS

EVALUATING THE EFFECTS OF WILDFIRE IN PIÑON-JUNIPER WOODLANDS ON BIGHORN SHEEP HABITAT AND VEGETATION COMPOSITION

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ABSTRACT

EVALUATING THE EFFECTS OF WILDFIRE IN PIÑON-JUNIPER WOODLANDS ON BIGHORN SHEEP HABITAT AND VEGETATION COMPOSITION

I evaluated the efficacy of using woodland fire to alter vegetation composition in a manner that augments desert bighorn sheep (*Ovis canadensis nelsoni*) habitat in the Black Ridge Canyons Wilderness Area in western Colorado. I applied generalized linear mixed models to estimate pre-fire ewe habitat selection and then simulated a hypothetical widespread fire to spatially predict where fire would be most beneficial in expanding habitat. I found that ewes were avoiding habitats with high woodland canopy cover, the habitat most likely to be removed by fire. Given the removal of all woodlands, it is likely that habitat expansion would occur in areas near topographic escape terrain. Coupled with this analysis, I addressed concerns regarding potential negative effects of fire in this system by comparing vegetation composition of unburned habitats to burned habitats that were treated with a native seed mixture. I found that foliar cover in burned areas was on average two times greater than in unburned areas and that post-fire seeding efforts likely allowed for these differences to be proportionally similar between native and non-native grass species. My results provide an encompassing view on the effects of fire for a common management situation in which both land and wildlife values are of mutual interest.

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PREFACE

The Black Ridge Canyons Wilderness Area (BRCWA) is administered by the Bureau of Land Management (BLM) and is found within the McInnis Canyons National Conservation Area in western Colorado (Figure 1). The BLM has a strong collaborative relationship with the Colorado Parks and Wildlife (CPW) in which the agencies work together in managing the land, plants, and wildlife of the BRCWA. As stated in the Wilderness Act of 1964, a wilderness is an area "where earth and its community are untrammeled by man", where the landscape "retains its primeval character and influence", and holds "outstanding opportunities for solitude". Furthermore, a wilderness area is "managed to preserve its natural conditions" in a way which they are "affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable".

The Black Ridge Canyon Wilderness Area encompasses over 30,000 hectares of rugged mesas, vertical cliffs, and talus slopes. This scenic, isolated, and diverse habitat holds unique opportunities for a variety of plant and wildlife species to flourish. Specifically, the BRCWA is home to Colorado's largest population of desert bighorn sheep (*Ovis canadensis nelsoni*) which provides visitors with outstanding prospects to view this iconic game species in addition to recreational hunting opportunities. A large portion of the BRCWA is comprised of piñon pine and juniper woodlands. Wildfire in these woodlands has become more prevalent over the last 20 years likely due to combination of stand age, stand density, and accumulation of fine understory fuels. Managers view wildfire in the BRCWA as a natural ecosystem process but wildfire can alter ecosystems in both positive and negative ways. For example, wildfire greatly alters vegetation composition and often makes lands susceptible to invasion of non-native species, namely *Bromus tectorum* (i.e., cheatgrass, downy brome). On the contrary, wildfire can alter landscapes in positive ways by increasing wildlife habitat quantity and quality, especially for species which prefer open terrain like bighorn sheep.

Direct collaboration with both the BLM and the CPW has identified that further understanding the effects of wildfire on both bighorn sheep habitat and vegetation composition are management priorities for the wilderness area. The objective of this study was to address these management priorities by providing a detailed analysis on the effect of wildfire in the BRCWA in terms of both desert bighorn sheep habitat and vegetation composition. To do so, I conducted two separate but related observational field studies, the details of which can be found in chapters 1 and 2 and then synthesized in chapter 3

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SUMMARY

Historically, wildfires in the Black Ridge Canyons Wilderness Area of western Colorado have been immediately suppressed following ignition. However, natural wildfires may provide an opportunity to augment habitat for the state's largest population of desert bighorn sheep (*Ovis canadensis nelsoni*). Bighorn sheep are known to consistently select habitats with high visibility so they can detect predators from a distance and escape to steep sloped terrain. I used modern resource selection modelling methods to estimate current bighorn sheep habitat selection. I then predicted habitat selection following a hypothetical high intensity wildfire to describe the ability of wildfire to augment bighorn sheep habitat. I found that bighorn sheep in the Black Ridge Canyons Wilderness Area strongly select for habitats with rugged terrain and low piñon-juniper woodland canopy coverage. Removal of piñon-juniper canopy cover through fire was predicted to enhance desert bighorn sheep habitat, especially for locations within close proximity to high topographical relief. My results provide land and wildlife managers with a more complete picture regarding the potential benefits of wildfire in this sensitive ecosystem.

INTRODUCTION

Desert bighorn sheep (*Ovis canadensis nelsoni)* are thought to have been extirpated from a large majority of their historic range and multiple reintroduction efforts were conducted across the western United States starting in the mid-20th century. Buechner (1960) and Monson (1980) reported that western Colorado was once populated by desert bighorn sheep but that they were extirpated from the area due to disease, overhunting, and habitat degradation. Through a joint effort between the Bureau of Land Management (BLM) and the Colorado Parks and Wildlife (CPW), the current Black Ridge desert bighorn sheep population was established through three translocations during the mid and late 1900s. The population size was most recently estimated at 200 individuals (Banulis et al. 2011).

During the late 1980s and early 1990s, it was determined by CPW that the population's range did not extend to the western portion of the BRCWA (Creeden and Graham 1997). In 1999, a series of wildfires occurred in areas previously unoccupied by desert bighorn sheep. These wildfires burned over 1,200 hectares of piñon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) woodlands (PJ woodlands). In 2007, monitoring efforts began to reveal more about population size, recruitment, causes of mortality, and overall range through the use of VHF radio collars. Basic location data revealed that collared individuals on the western portion of the population's range were often found within or near a previously unoccupied burned expanse referred to as the Long Mesa Fire. However, bighorn sheep were not utilizing a separate but similar sized wildfire referred to as the Moore Canyon Fire, also within the population's range. This has lead managers to question the ability of wildfire to augment bighorn sheep habitat in the BRCWA.

Past research has shown that fire can indeed impact both bighorn sheep habitat quality and population distributions. For example, Peek et al. (1979) found that bighorn sheep chose burned habitats more often than adjacent unburned habitats, concluding that fire can be used to retard succession and improve forage palatability. Furthermore, Sawyer et al. (2009) found that bighorn sheep in the Laramie mountain range of Wyoming were preferentially selecting burned areas. It is suspected that the sheep may not be utilizing the Moore Canyon fire because the area may not hold the physical landscape attributes that are often associated with quality bighorn sheep habitat. Wildfire is still viewed as a potential means to augment bighorn sheep habitat but further investigation is required to better understand this spatial relationship.

Bighorn sheep are known to preferentially select habitats based on landscape characteristics that provide greater protection from predators. In Colorado, predators of bighorn sheep include mountain lions (*Puma concolor*), coyotes (*Canis latrans*), and black bears (*Ursus americanus*) with lambs also being susceptible to predation from bobcats (*Lynx rufus*), golden eagles (*Aquila chrysaetos*), and red foxes (*Vulpes vulpes*) (George et al. 2009). Several studies have shown that mountain lion predation is more likely to be a limiting factor for desert bighorn sheep than for Rocky Mountain bighorn sheep (Kamler et al. 2002, Rominger et al. 2004, Mckinney et al. 2006). Mountain lion predation is considered to be the primary limiting factor of the Black Ridge desert bighorn sheep population, due to its relatively small size and a lack of available alternate prey for the mountain lions (Creeden and Graham 1997).

The consistent distinguishing factor of bighorn sheep habitat is that it provides visibility and has access to steep escape terrain. Landscapes with steep slopes allow individuals to detect danger at a distance, visually communicate with other individuals, and escape to terrain where they can outmaneuver predators (Geist 1971, Risenhoover and Bailey 1980, George et al. 2009).

This is especially true for ewe and ewe-lamb groups, to the extent that they will sacrifice forage quality to obtain higher security (Bleich et al. 1997, Shackleton et al. 1999). Disease (most notably pneumonia) poses the greatest single threat to the persistence of any particular bighorn sheep population (Singer et al. 2001, George et al. 2008) but increasing the amount of available habitat should allow for larger populations and may ultimately safeguard against potential large-scale die-offs.

A single large scale fire or a series of smaller fires can increase the amount of suitable habitats for bighorn sheep (Holl et al. 2012). Fire also can have effects on forage quality and quantity, timing of green-up, and herbaceous species composition (Spowart et al. 1985, Greene 2010). Forage quantity is typically of concern in desert habitats due to low primary productivity, which can affect the health of individual sheep and the entire population. Most increases of herbaceous biomass resulting from fire in PJ woodlands are due to opening of the over story and increases in the amount of nutrients available to grasses, forbs, and shrubs (Whelan 1995). Fortunately, desert bighorn sheep are opportunistic and adaptable feeders (Leslie and Douglas 1979, Cunningham 1989, Krausman et al. 1989, Miller and Gaud 1989), allowing them to take advantage of general increases in forage availability.

My objective for this study was to evaluate how the hypothetical occurrence of high intensity wildfire in the BRCWA could affect the overall habitat use of female desert bighorn sheep. To do so, I estimated PJ woodland canopy coverage for the BRCWA in addition to all ewe home ranges. I then developed habitat selection models to predict habitat use relative to multiple landscape characteristics. Piñon-juniper woodland canopy cover was the only biotic landscape characteristic in addition to multiple geophysical landscape characteristics known to be preferentially selected for by bighorn sheep. Given the developed model, I predicted sheep use following removal of all PJ woodland canopy cover across the study area. My work provides an understanding of bighorn sheep habitat use and will be invaluable for conservation planning by identifying areas that wildfire would most improve sheep habitat.

Study Area

The Black Ridge desert bighorn sheep population resides almost entirely within the BRCWA and the adjacent Colorado National Monument (Figure 2). This area is located on the northern edge of the Colorado Plateau and ranges from 1300 meters to over 2100 meters in elevation. The landscape is dissected by steep rugged canyons running south to north and opening into the Colorado River as it flows west into Utah. To date, no individuals have been documented to cross the Colorado River to access habitats on the north side and only occasionally are they thought to venture past the southern border of the BRCWA. If range expansion were to occur, it is most likely to occur on the far eastern end beyond the Colorado National Monument, however high densities of PJ woodlands are thought to be deterring such movements.

The entire study area can be characterized as a semi-arid desert, annually receiving an average of 11.51inches (29.23 cm) of precipitation (National Oceanic and Atmospheric Administration 2013). I refer to the landscapes that lie between canyons as mesas landscapes which can range from rugged talus slopes to open sagebrush parks. Dense PJ woodlands often have sparse herbaceous understories due to competition for resources in this relatively unfavorable and arid environment. Mesa landscapes are mostly comprised of PJ woodlands of varying densities. Canyons typically have lower canopy cover with the exception of north facing slopes.

METHODS

Location data was collected from 26 VHF collared individuals from the spring of 2007 to the winter of 2011. Data included visual observations, fixed-wing locations, and ground triangulated locations. For this study, I limited analysis to visual observations and fixed-wing locations because the accuracy of ground triangulations is suspect due to inherent difficulties with VHF signals in canyon landscapes. I also limited my analysis to adult female locations because collared adult male bighorn sheep have exhibited exploratory movements for this herd which may lead to erroneous habitat use estimation. Fixed-wing locations were collected on a bi-weekly basis for most of the time during 2007-2011. Field technicians attempted to acquire visual observations for each individual on a weekly basis, but were not always successful due to the remoteness of the terrain. I pooled data across seasons for development of the habitat selection models that are the focus of this chapter. I also modeled seasonal habitat selection but results were considered uninformative in regards to management priorities because this herd is not seasonally migratory and in general, reports of seasonal home ranges are rare in desert bighorn sheep (McCarty and Bailey 1994). See Appendix A for details of seasonal habitat selection models.

In total, I analyzed 1,055 data points across 26 adult females. Group composition frequently changed on a day to day basis but each individual exhibited high fidelity to either the eastern, central, or western portion of population's overall range. To analyze the presence of potential meta-populations, I constructed localized convex hulls (LoCoH) using a fixed number of k points ($k_i = \sqrt{n}_i$) for each individual (Getz et al. 2007). Localized convex hull methods begin by constructing convex hulls (polygons) with each point location and its k-1 nearest neighbors, the union of which represents an individual's home range. Utilization distributions are then described by ordering hulls from smallest to largest where smaller hulls represent areas with higher density of use. Low density hulls of individuals from any of three regions were found to only slightly overlap with low density hulls of individuals from the adjacent region, thus each ewe could be spatially described as inhabiting one of three potential meta-populations (Figure 3). Such meta-population structure has also been exhibited by multiple populations of bighorn sheep (Bleich et al. 1990, Epps et al. 2007). Furthermore, individuals from within potential meta-populations may exhibit slightly different habitat selection processes due to different availability of resources in each. Therefore, I divided the study area into three portions and categorized each individual as belonging to one of the three potential meta-populations.

I analyzed habitat selection using resource selection functions (RSFs, Manly et al. 2002). Resource selection functions are tools that can facilitate understanding of the relative importance of individual habitat components necessary for making predictions by treating each habitat component as a variable that contributes to explaining overall variation in habitat use (Boyce et al. 2002, Johnson and Nielsen 2006). Further, investigators can then predict how changes in one or more of the components might influence an animal's habitat use (McDonald and McDonald 2002, Johnson et al. 2005, Bleich et al. 2010). I estimated RSF coefficients using logistic regression by generalized linear mixed-effect models (GLMM , logit link) with both fixed effects (landscape variables) and random intercepts for individual desert bighorn sheep nested within the three potential meta-populations. Including random effects assists in accounting for unbalanced sampling design and allows for more accurate conditional predictions (Gillies et al. 2006, Hebblewhite and Merrill 2008). Development and analysis of mixed models was conducted in the R statistical software version 3.02 (R Core Team 2013) using the lme4 package (version 1.0-5, http://cran.r-project.org/Ib/packages/lme4, accessed 1 Nov 2013).

The accuracy of coefficient estimates and the subsequent inference of RSFs is highly dependent on the way researchers develop such models. Some necessary considerations when structuring RSFs include choice of spatial scale (Boyce 2006, Beyer et al. 2010), assessing the impact of autocorrelation between both individuals and landscape variables (Fieberg et al. 2010), and the chosen ratio of used to available locations being analyzed (Northrup et al. 2013). This analysis was based on a 3rd order use-availability design with the available units being sampled from within individual home ranges described by 100 % minimum convex polygons (Johnson 1980). Coefficient estimates of RSFs have been shown to be sensitive to the ratio of used to available data points with the greater number of available points providing more accurate estimates (Northrup et al. 2013). However, as the number of available data locations increases so does the computational intensity of such models. To determine the level of available locations to be analyzed, I estimated coefficients of final models by generating available points numbering 5, 10, 25, 50, 75, 100, and 150 times the number of used points of each individual. I then identified the smallest number of available locations for which coefficient estimates stabilized. Coefficient estimates were found to stabilize when estimating models using 50 times more available locations than the number of known used locations (Figure 4).

I used ARC/INFO software (Version 10.1, Environmental Systems Research Institute, Redlands, California, USA) to develop and sample location data from 6 raster based landscape variables. Slope, aspect, an index of ruggedness, distance to escape terrain, and topographic wetness index were all derived using 30 meter x 30 meter resolution United States Geological Survey (USGS) digital elevation models. The ruggedness index was calculated using a 3x3 moving window as in Sappington et al. (2007). This index ranges from 0 to 1 with greater values representing areas with the greatest landscape roughness and was created specifically to describe terrain of bighorn sheep habitat in a way that minimizes correlation with slope. I defined escape terrain as areas with >60% slope and then measured distance in meters to the nearest escape terrain for each data point. A topographic wetness index (TWI) surface was calculated for each location similar to Bevin and Kirby (1979) where TWI is equal to the natural log of the upslope area divided by the slope for each location. Thus, greater TWI values represent basins and lower values represent peaks in the topography. PJ woodland canopy cover was estimated specifically for this study using supervised classification methods at a resolution of 30 meter x 30 meter (See Appendix B for details). It was estimated that 67% of the study area is comprised of PJ woodlands, most of which occurred on mesa landscapes between canyons with estimated densities ranging from 0% to 61% canopy cover . Water is thought to be widely available throughout the study area, although temporally variable. For example, pools of water collect in rocky areas and often last several weeks following rainfall. Therefore I do not include water sources in my modeling efforts.

Both used and available location data were first separated into two datasets to be used for model development and model validation. Data belonging to six individuals, two from each potential meta-population were randomly withheld to be used for model validation leaving data of 20 individuals for model development. Two separate mixed models were developed to assist in describing the habitat selection process which gave rise to the location data. First, I developed a full Population Model using all data not withheld for validation (n_{loc}=802, n_{ind}=20). The full Population Model is thought to best represent current habitat selection. However, it is likely that this model is highly dependent on variables correlated with canyon habitats (e.g., slope, distance to escape terrain, TWI) where the potential of wildfires is lowest and thus may improperly depict habitat use following wildfire where wildfire potential is greatest (i.e., mesas). So I developed a second model, referred to as the Landscape Model, using only location data that lies outside of canyon landscapes in addition to a 50m buffer (n_{loc}=261, n_{ind}=20). Canyon landscapes were defined as areas with >60% slope. The Landscape Model was intended to minimize unwanted canyon habitat correlations by identifying the explanatory value in geophysical variables for habitat use outside of canyons. In other words, the model addresses questions regarding where mesa habitats hold the geophysical attributes that best explain sheep use within mesa habitats, which may more appropriately estimate habitat selection within mesa habitats given the occurrence of a high intensity wildfire.

The initial full models included all singular variables in addition to interactions between all geophysical variables and PJ woodland canopy cover, resulting in a total of 11 fixed effects and two random effects (i.e., meta-population and individual). The full models were then backfit stepwise, removing fixed effects one at a time as a way to create competing RSF models. I then used Akaike's Information Criterion (AIC) as a metric to evaluate the likelihood of the competing models (Burnham and Anderson 2002) which is the currently preferred method for comparing competing models under the use-available data design (Boyce et al. 2002). Including individual sheep as a random effect was inherent in the study design and thus was not subject to removal. However, the potential meta-population which an individual belongs to was subject to removal if it did not improve the fit of the final model.

To quantify the accuracy of the final models I calculated Spearman-rank correlations between area-adjusted frequencies of presence only validation predictions and RSF bins as described by Boyce et al. (2002). Area-adjusted frequencies were defined as the number of predicted scores from presence only validation data within each bin divided by the area of that range of RSF scores available across the entire study area. I determined bin size by first dividing the predictions into 20 equal interval bins scaled between the maximum and minimum predictions and then simplified the scores into 8 bins approximately equal in size. Using this method, it is expected that a model with good predictive performance would exhibit strong positive correlation between area-adjusted use locations and RSF bins.

The final models were then used to spatially predict distribution across the study area under current conditions. Models were developed based only on data from within individual home ranges and thus predicting over the entire study area should be implemented with caution (Schooley 1994). However, the wilderness boundary does not greatly exceed home range boundaries and landscapes in such areas are similar in structure so all predictions are considered relevant. After predicting habitat use under current landscape conditions, I predicted use following a high intensity wildfire by setting PJ woodland canopy cover to zero in all areas. I acknowledge that simulating a widespread wildfire occurring across the entire study area is only hypothetical and not likely to actually occur given what is known about PJ woodland wildfire behavior but such predictions will provide an indication to managers as to where wildfire would be most beneficial in augmenting bighorn sheep habitat.

RESULTS

Population Model

The final Population Model revealed evidence for an effect of TWI, distance to escape terrain, ruggedness, slope, PJ woodland canopy cover, and the interaction of TWI and PJ woodland canopy cover on habitat use. See Table 1(a) for details of the selected model. See Table 2 (a) for details of the model selection process. In general the model suggests that ewes are selecting areas within or near canyons (i.e., steep slopes, greater TWI, and closer to escape terrain) which have lower PJ woodland canopy cover. With the influence of other variables held constant, predicted habitat use decreased by a factor of +0.97 for each 1% increase in PJ woodland canopy cover (see Figure 5 for graph of marginal predictions). Meta-population was included as a random effect in the final model because excluding it did not decrease AIC values by more than two integers. Although meta-populations may interbreed and thus not be technically defined as such, including meta-population as a random effect will allow more accurate and ultimately more useful conditional predictions across the study area (Breslow and Clayton 1993, Skrondal and Rabe-Hesketh 2004). Area adjusted frequencies exhibited significant positive rank values (Spearman-rank correlation) across relative use bins for the final Population Model (r_s=0.856, p-value=.0196), indicating good model performance.

Pre- and post-fire predictions were estimated by using the conditional meta-population random effects but marginalizing (i.e., averaging) individual random effects (Figure 6, left panels). As expected, predictions of use under current habitat conditions reveal a strong selection for canyon habitats along with select rugged areas outside of canyons. Predictions also spatially visualize how ewes are utilizing already burned habitats created by the Long Mesa Fire but not greatly selecting for habitats within the Moore Canyon Fire. Post-fire predictions revealed similar habitat use with the greatest difference in use occurring near escape terrain where PJ woodland canopy cover is currently greater. These results suggest that given a high intensity wildfire, sheep habitat may only be augmented in areas in close proximity to canyon rims or steep slopes. Wildfire behavior in areas with steep slopes is often unpredictable due to topographic dynamics and the lack of understory vegetation often found which may not provide the necessary fine fuels to support high intensity wildfires in PJ woodlands (Romme et al. 2009).

Landscape Model

When limiting analysis to locations found in mesas, the model with the lowest AIC value included only PJ woodland canopy cover and ruggedness variables. See Table 1 (b) for details of the selected model and see Table 2 (b) for details of the model selection process. Similarly to the Population Model, the Landscape Model benefited from the inclusion of both the individual and meta-population as random effects. Again with the influence of other variables held constant, predicted habitat use decreased by a factor of +0.97 for each 1% increase in PJ woodland canopy cover for the Landscape Model. This estimate of the influence of PJ woodland canopy cover is very similar to estimates of the population model. However, the influence of ruggedness was found to be greater than the final population model. Validation of the Landscape Model exhibited good model performance (r_s=0.781, p-value=.136) but lower model development sample size ($n_{loc=}$261, n_{ind}=20) likely influenced the degree and significance of the Spearman-rank correlation. Never the less, the model will be extremely useful to managers when addressing questions regarding habitat selection processes in areas of high wildfire potential.

Pre- and post-fire predictions were estimated in exactly the same manner as the final Population Model except predictions were not made for habitats classified as canyons. Spatial predictions of use under current habitat conditions revealed a strong selection for rugged areas with lower PJ woodland canopy cover (Figure 6, right panels). Post-fire Landscape Model predictions exhibited greater increases in use than the final Population Model due a lack of influence of escape terrain and steep slopes found within canyons. Post-fire predictions suggest that landscapes on the western portion of the population's range would benefit most from a high intensity wildfire. Fortunately, this area is where wildfires are also most likely to occur due to greater PJ woodland canopy cover. In fact, two large high intensity wildfires have occurred in this area during the last 3 years (BLM, unpublished report).

DISCUSSION

Wildfire can be used as a management tool to alter the composition and structure of habitats for multiple ungulate species. Wildfire has the ability to enhance bighorn sheep population persistence by both decreasing predation risk due to greater visibility in burned habitats and increasing carry capacity by allowing individuals to exploit previously unavailable habitats. Furthermore, wildfire in all areas regardless of geophysical attributes may create beneficial travel corridors allowing for better gene flow. PJ woodland fire return intervals are best measured on a scale of hundreds of years (Romme et al. 2009) so the effect of fire on bighorn sheep habitat is likely to be long term and the succession of burned habitat may be further delayed due to altered fire regimes.

Dense stands of piñon and juniper trees are commonly found in or near desert bighorn sheep habitats across the southwestern United States. In some areas PJ woodland canopy cover and distribution have increased substantially over the past 150 years (Romme et al. 2009), invading previously unoccupied adjacent grassland and shrub land communities (Miller and Rose 1995). The encroachment of PJ woodlands has been attributed to fire suppression, natural succession, and large scale environmental trends (Romme et al. 2009). Regardless of the cause, the loss of “openness” due to the encroachment of shrubs or trees is seen as limiting the amount of bighorn sheep habitat available and may be negatively impacting many sheep populations (Bleich et al. 2008, George et al. 2009). Fire in PJ woodlands is most often high intensity, killing most or all trees within the burned area regardless of tree size (Romme et al. 2009), thus dramatically increasing visibility and potentially increasing the amount of available bighorn sheep habitat.

I developed RSF models to assist in understanding of how wildfire may affect desert bighorn sheep habitat in the BRCWA. The RSF models provide a means to consider both the effects of wildfire and the specific habitat requirements of individuals within the population. In both RSF models, PJ woodland canopy cover was found to be significantly related to habitat use of female bighorn sheep. Geophysical landscape attributes such as ruggedness, slope, TWI, and distance to escape terrain were also strongly related to habitat use. The results suggest that ewes prefer landscape attributes that offer increased visibility which can be represented singularly or by a combination of steep slopes, rugged terrain, and lower PJ woodland canopy cover. Estimates regarding the influence of such variables on bighorn sheep habitat selection are similar to findings of multiple studies (Risenhoover and Bailey 1980, DeCesare and Pletscher 2006, Rubin et al. 2009, Sawyer et al. 2009).

Interpretation of RSF models must be carefully described. Keating and Cherry (2004) argued that using logistic regression to estimate RSFs does not guarantee that models will produce actual probabilities. Thus I am only able to make inference about the relative probability of use and do not make attempts to describe the amount of “unused” habitat converted to “used” habitat following wildfire. To further describe how the degree of use will change following fire, I hierarchically classified predicted landscape scores into four categories of use; Lowest, Low, Medium, and High. See Appendix C for details of hierarchical classification and predictions. Given that some landscapes may not hold the necessary geophysical attributes regardless of PJ woodland canopy cover, the utility of my models lies in the spatial description of where a high intensity wildfire is most likely to increase relative use. I spatially described the difference in use by subtracting pre-fire predictions from post-fire predictions.

When considering the implications of my models, I took into account both post-fire predictions and PJ woodland wildfire behavior. My predictions suggest that wildfire would be most beneficial in areas which are rugged, steep sloped, or in close proximity to escape terrain. For a wildfire in PJ woodlands to become intense and wide spread a combination factors is often required. Fine understory fuels such as *Bromus tectorum* (i.e., cheatgrass, downy brome) have been shown to greatly increase intensity of wildfire (Whisenant 1990, Knapp 1996). More specifically, *B. tectorum* has influenced the spread and intensity of both the Long Mesa Fire and the Moore Canyon Fire (Bureau of Land Management Environmental Assessment 1999). However, fine understory fuels are likely to be less prevalent in areas with steep slopes or along canyon rims due to a high degree of rocky ground cover. Therefore, I do not believe large high intensity wildfires will originate along canyon rims and recommend that managers do not suppress such fires due to the likelihood that the fire will not persist. Large high intensity wildfires are likely to occur within interior mesa habitats due to greater PJ woodland canopy cover and the prevalence of *B. tectorum*. The Landscape Model describes areas that would most benefit from wildfire as being located on the western portion of the population's range in addition to select rugged landscapes located between canyons throughout the wilderness area. Managers considering how to react to future wildfires can refer to post-fire habitat use predictions to assist in their decision. It is important to note that these habitats are also utilized by migratory elk (*Cervus canadensis*) and mule deer (*Odocoileus hemionus*) during the winter months and thus wildfire would also benefit these ungulate species.

Historically, the majority of wildfires in the BRCWA are suppressed immediately following ignition. The present BRCWA situation provides a unique opportunity to allow wildfire to burn naturally due to a lack of human structures present and multiple beneficial effects on wildlife habitat. However, other factors must be considered including post-fire non-native species prevalence, destruction of archeological artifacts, and the cost of rehabilitation. I suggest that managers consider the results of this study as one of the multiple factors when contemplating future wildfire suppression in the BRCWA.

Table 1. Estimates of random and fixed effects for the Population (a) and Landscape (b) Models developed using a portion of the entire dataset.

(a)

Population Mixed-Effects Model

Random Effects	**Variance**	**SD**			
Individual (20)	7.93E-13	8.90E-07			
Meta-population (3)	4.89E-13	6.99E-07			
Fixed Effects	**LOG Odds**	**St. Error**	**z-value**	**Pr (>|z|)**	
Intercept	-3.464539	0.114722	-30.2	2.00E-16	***
TWI	0.0282001	0.008536	3.304	0.00095	**
Escape Terrain	-0.003661	0.000369	-9.933	2.00E-16	***
Ruggedness	3.6789773	1.144681	3.214	0.00131	*
PJ Density	-0.033882	0.005467	-6.198	5.72E-10	***
Slope	0.0034481	0.001159	2.976	0.00292	***
TWI: PJ Density	-0.002538	0.000917	-2.77	0.00561	**

p-value sig. level: * < 0.05, ** < 0.01, *** < 0.001

(b)

Landscape Mixed-Effects Model

Random Effects	**Variance**	**SD**			
Individual (20)	1.18E-02	1.08E-01			
Meta-pop. (3)	2.30E-02	1.52E-01			
Fixed Effects	**LOG Odds**	**St. Error**	**z-value**	**Pr (>|z|)**	
Intercept	-4.490223	0.129733	-34.61	2.00E-16	***
Ruggedness	15.97933	2.55215	6.26	3.82E-10	***
PJ Density	-0.037223	0.005675	-6.56	5.42E-11	***

p-value sig. level: * < 0.05, ** < 0.01, *** < 0.001

Table 2. Stepwise backfitting processes for the Population Model (a) and Landscape Model (b) with associated AIC values used to choose the most parsimonious model from among all possible combinations of fixed and random effects. Backfitting was implemented using the LMERConvienceFunctions package in R Statistical software. The function tests all combinations of fixed effects and if it is determined that the fixed effect under consideration does not increase model fit based on AIC value, it is removed; otherwise it is kept. Higher order interactions are tested for exclusion first. Once all the highest order interactions are evaluated, the function evaluates all first order fixed effects. Then, random effects are evaluated. The model with the lowest AIC is chosen as the most parsimonious.

(a)

Population Model Stepwise Backfitting Process

Fixed	#	Random	#	AIC	Action
*ASP+ESC+PJ+RUG+SLP+TWI+(ASP*PJ)+(ESC*PJ)+(RUG*PJ)+(SLP*PJ)+(TWI*PJ)*	11	*IND + MTP*	2	9717	
*ASP+ESC+PJ+RUG+SLP+TWI+(ESC*PJ)+(RUG*PJ)+(SLP*PJ)+(TWI*PJ)*	10	*IND + MTP*	2	9716	Remove fixed effect interaction (ASP*PJ)
*ASP+ESC+PJ+RUG+SLP+TWI+(ESC*PJ)+(SLP*PJ)+(TWI*PJ)*	9	*IND + MTP*	2	9715	Remove fixed effect interaction (RUG*PJ)
*ASP+ESC+PJ+RUG+SLP+TWI+(SLP*PJ)+(TWI*PJ)*	8	*IND + MTP*	2	9721	Remove fixed effect interaction (ESC*PJ)
*ASP+ESC+PJ+RUG+SLP+TWI+(TWI*PJ)*	7	*IND + MTP*	2	9720	Remove fixed effect interaction (SLP*PJ)
ASP+ESC+PJ+RUG+SLP+TWI	6	*IND + MTP*	2	9721	Keep fixed effect interaction (TWI*PJ)
*ESC+PJ+RUG+SLP+TWI+(TWI*PJ)*	6	*IND + MTP*	2	9718	Remove fixed effect ASP
*‡ PJ+RUG+SLP+TWI+(TWI*PJ)*	5	*IND + MTP*	2	9713	Keep fixed effect ESC
*RUG+SLP+TWI+(TWI*PJ)*	4	*IND + MTP*	2	9721	Keep fixed effect PJ
*SLP+TWI+(TWI*PJ)*	3	*IND + MTP*	2	9720	Keep fixed effect RUG
Test Random Effects					
*PJ+RUG+SLP+TWI+(TWI*PJ)*	4	*IND*	1	9711	Keep both random effects, IND was not considered removable due to the inherent repeated measures for each individual. MTP was considered for removal but its was kept in the final model because removal did not decrease AIC value by greater than 2.
*PJ+RUG+SLP+TWI+(TWI*PJ)*	4	*MTP*	1	9711	

<u>Fixed Effects:</u> ASP=Aspect, ESC = Distance to Escape Terrain (m), PJ= PJ Woodland Canopy Cover (%), RUG= Ruggedness Index, SLP= Slope (%), TWI = Topographic Wetness Index, <u>Random Effects:</u> IND= Individual, MTP= Meta-population

‡ = final model used for predictions

(b)

Landscape Model Stepwise Backfitting Process					
Fixed	#	Random	#	AIC	Action
*ASP+ESC+PJ+RUG+SLP+TWI+(ASP*PJ)+(ESC*PJ)+(RUG*PJ)+(SLP*PJ)+(TWI*PJ)*	11	*IND + MTP*	2	3684	
*ASP+ESC+PJ+RUG+SLP+TWI+(ESC*PJ)+(RUG*PJ)+(SLP*PJ)+(TWI*PJ)*	10	*IND + MTP*	2	3682	Remove fixed effect interaction (ASP*PJ)
*ASP+ESC+PJ+RUG+SLP+TWI+(ESC*PJ)+(SLP*PJ)+(TWI*PJ)*	9	*IND + MTP*	2	3681	Remove fixed effect interaction (RUG*PJ)
*ASP+ESC+PJ+RUG+SLP+TWI+(SLP*PJ)+(TWI*PJ)*	8	*IND + MTP*	2	3680	Remove fixed effect interaction (ESC*PJ)
*ASP+ESC+PJ+RUG+SLP+TWI+(TWI*PJ)*	7	*IND + MTP*	2	3679	Remove fixed effect interaction (SLP*PJ)
ASP+ESC+PJ+RUG+SLP+TWI	6	*IND + MTP*	2	3679	Remove fixed effect interaction (TWI*PJ)
ESC+PJ+RUG+SLP+TWI	6	*IND + MTP*	2	3677	Remove fixed effect ASP
PJ+RUG+SLP+TWI	5	*IND + MTP*	2	3677	Remove fixed effect ESC
RUG+SLP+TWI	4	*IND + MTP*	2	3727	Keep fixed effect PJ
PJ+SLP+TWI		*IND + MTP*		3711	Keep fixed effect RUG
PJ+RUG+TWI		*IND + MTP*		3675	Remove fixed effect SLP
‡ PJ+RUG		*IND + MTP*		3672	Remove fixed effect TWI
Test Random Effects					
*PJ+RUG+(TWI*PJ)*	4	*IND*	1	3670	Keep both random effects, IND was not considered removable due to the blatent repeated measures for each individual. MTP was considered for removal but its was kept in the final model because removal did not decrease AIC value by greater than two intergers.
*PJ+RUG+(TWI*PJ)*	4	*MTP*	1	3670	

<u>Fixed Effects:</u> ASP=Aspect, ESC = Distance to Escape Terrain (m), PJ= PJ Woodland Canopy Cover (%), RUG= Ruggedness Index, SLP= Slope (%), TWI = Topographic Wetness Index, <u>Random Effects:</u> IND= Individual, MTP= Meta-population

‡ = final model used for predictions

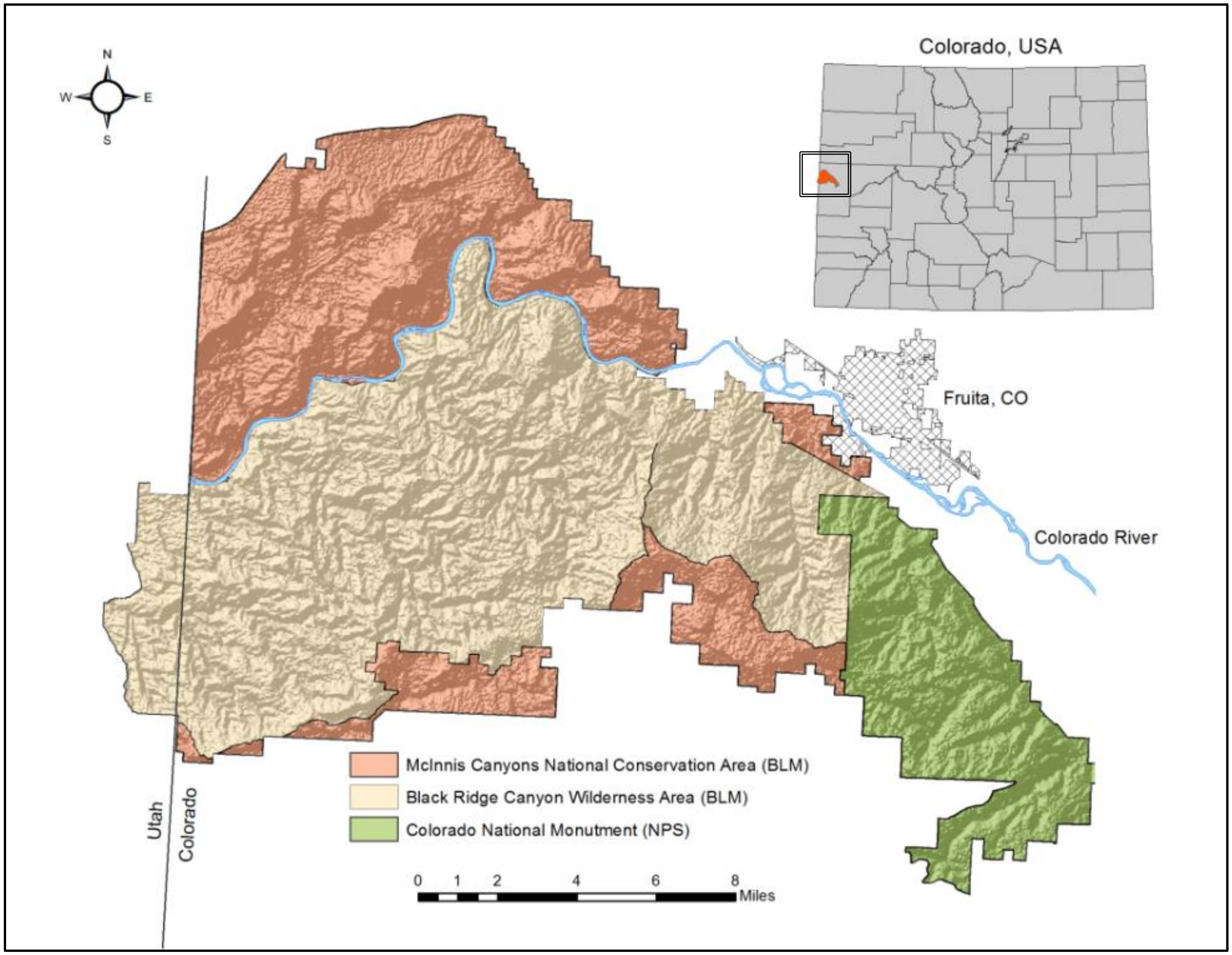


Figure 1. The study area includes the Black Ridge Canyons Wilderness Area which is part of the McInnis Canyons National Conservation Area and lies directly adjacent to the Colorado National Monument in western Colorado. The area is comprised of vertical red rock cliffs intersected by rugged mesas

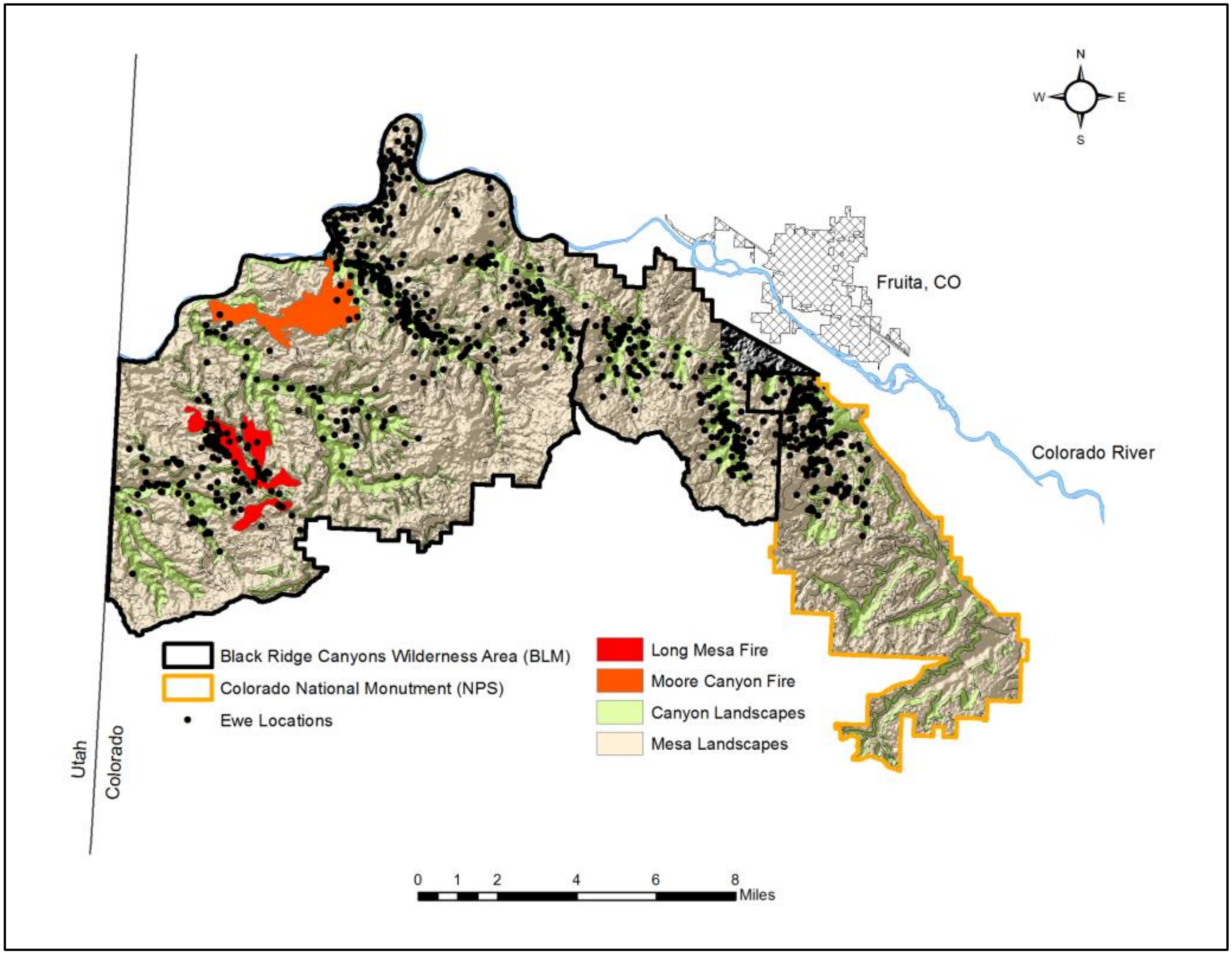


Figure 2. Study area includes the Black Ridge Canyons Wilderness and the adjacent Colorado National Monument. The two study fires can be seen on the western portion of the wilderness area. Ewe locations are clustered throughout the Black Ridge Canyons Wilderness area and on the western portion of the Colorado National Monument.

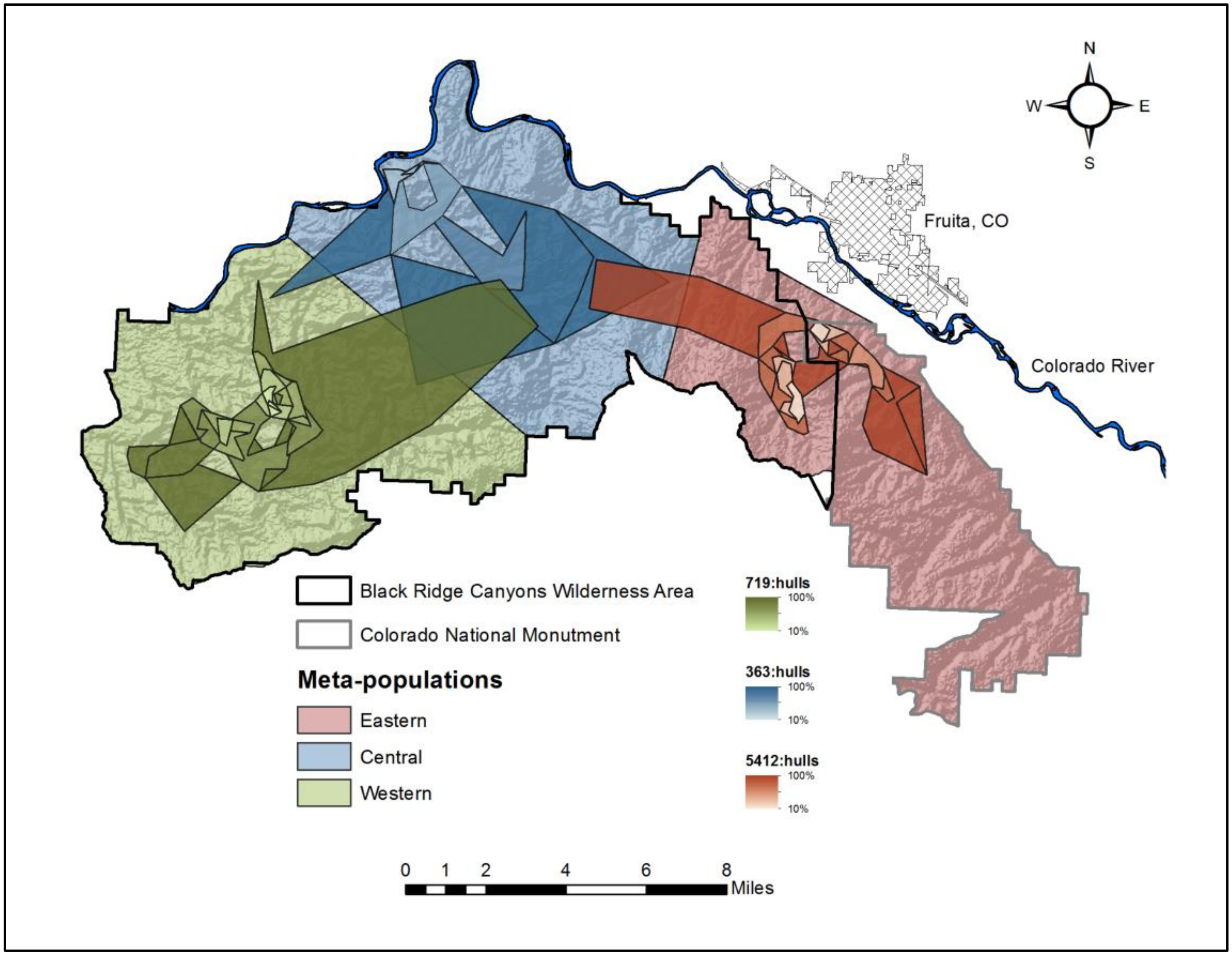
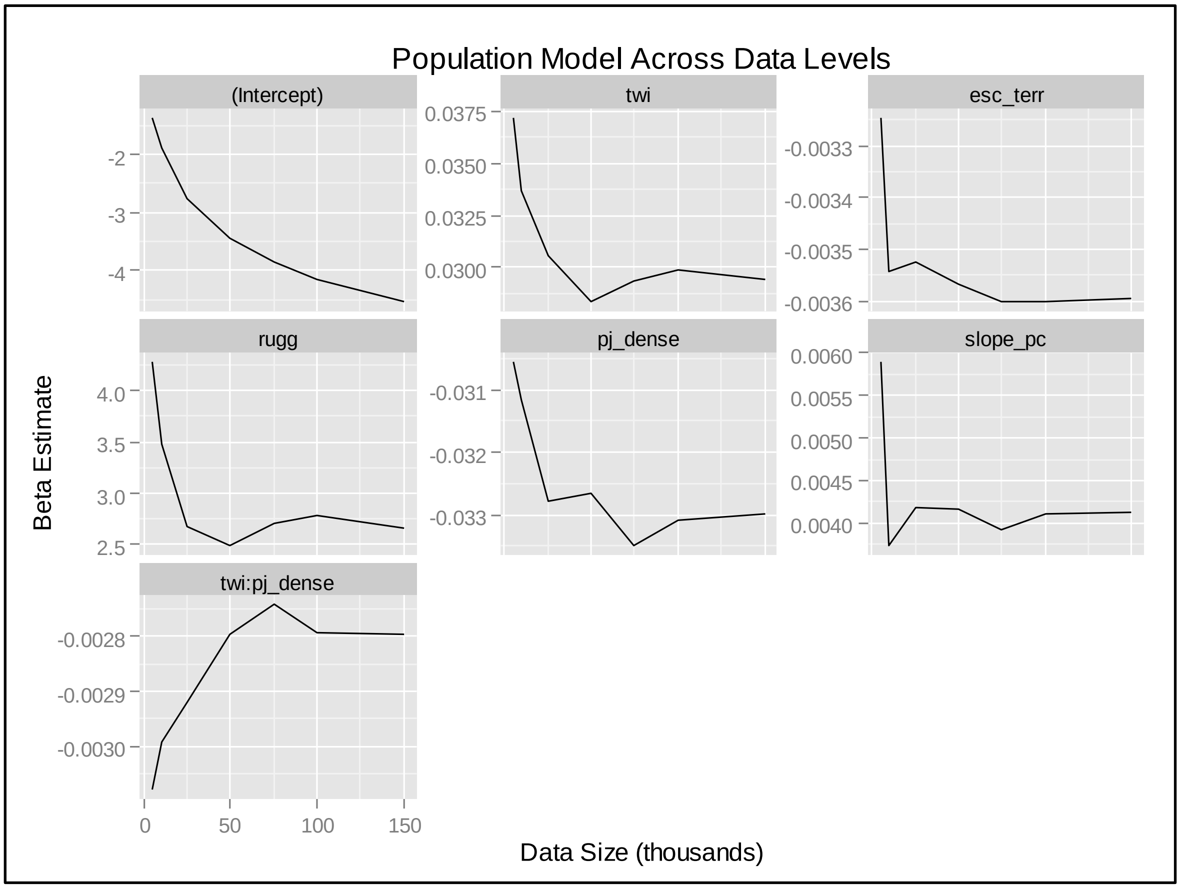


Figure 3. Localized convex hulls were used to spatially describe utilization distributions of individuals. To better visualize potential meta-populations, I selected one individual from each potential meta-population that exhibited the greatest overlap with the adjacent potential meta-population. Smaller hulls (10%) represent areas of greatest use and larger hulls (100%) represent areas of less common use. Hulls are labeled by the individual's identification code.



(b)

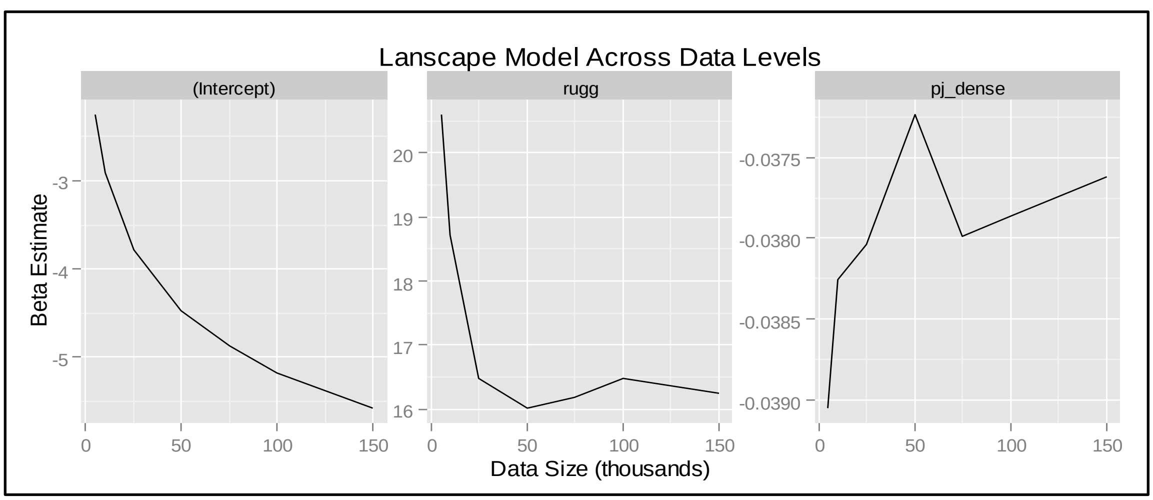


Figure 4. Graphs showing estimated fixed effects values for each the Population (a) and Landscape (b) final mixed-effects models using 5,10,25,50,75,100,and 150 times the number of random available points and the number of used points. I report and predict with estimates calculated using a 50:1 ratio of available:used points.

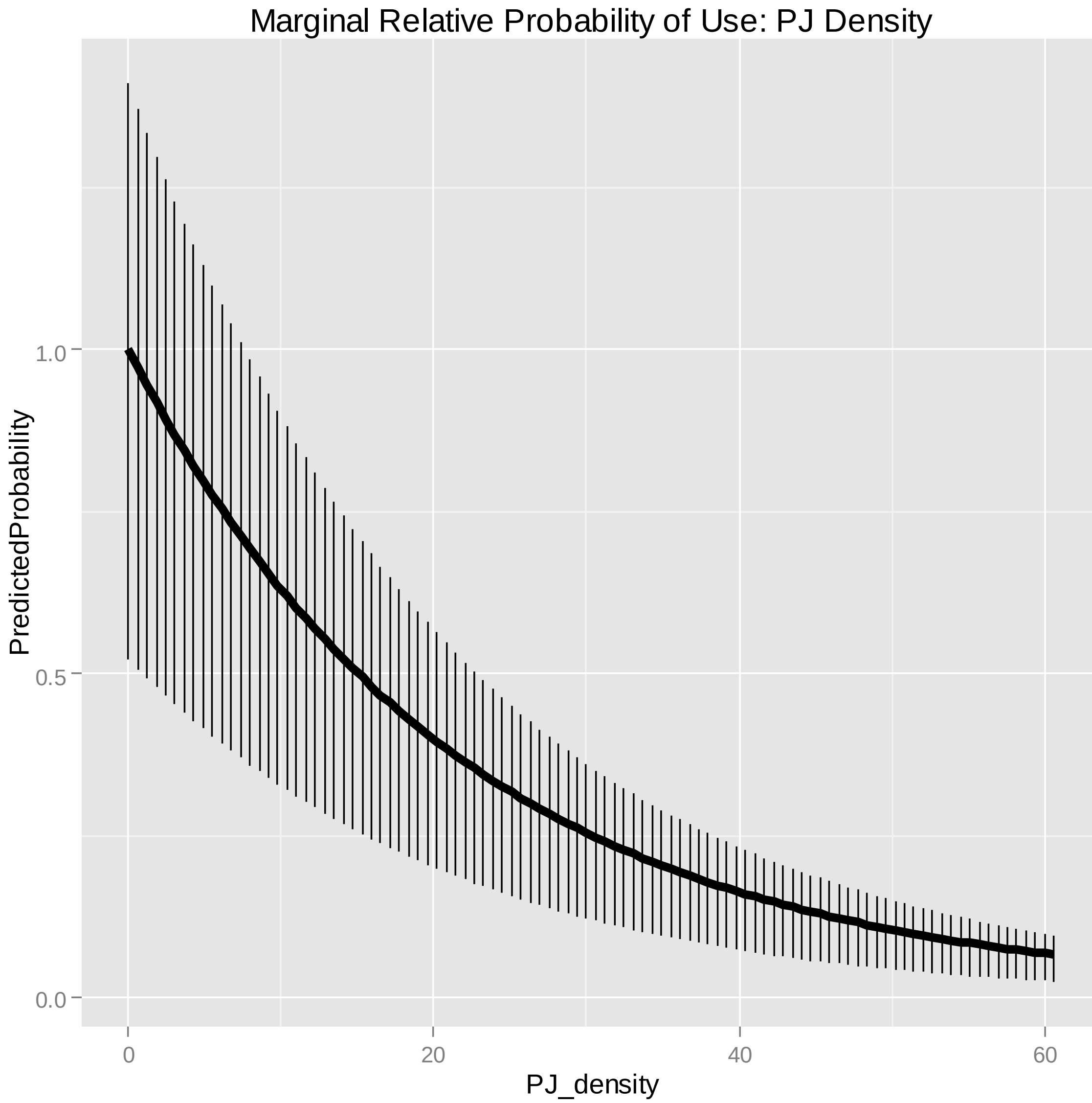


Figure 5. Population Model marginal relative probabilities of use in relation to PJ canopy cover were calculated by holding all other variables at their population means and varying PJ woodland canopy cover across its range of values. Probabilities were then scaled between 0 and 1. Vertical lines represent relative use with all other variables held at their 25th percentile and 75th percentile.

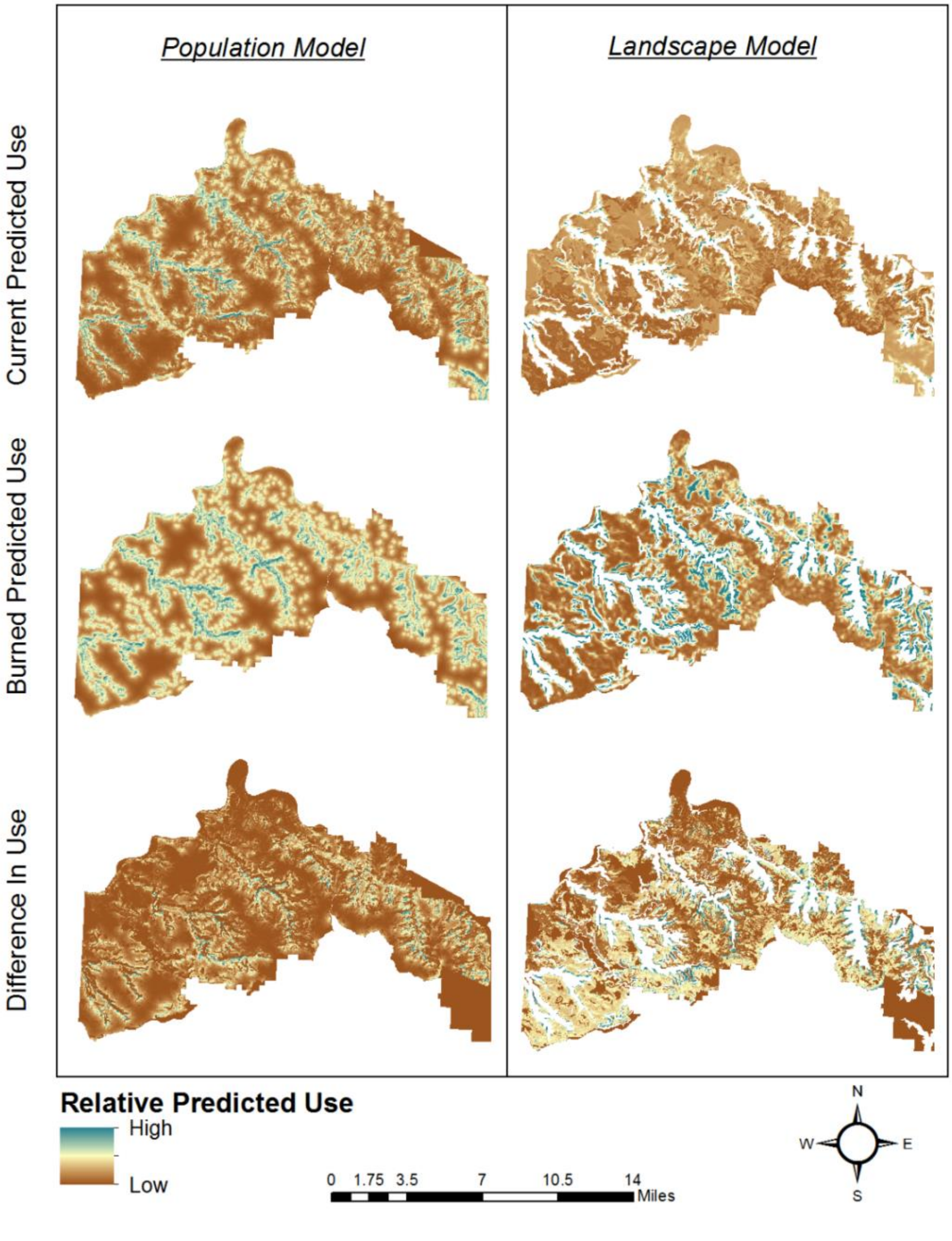


Figure 6**.** Spatial predictions of both the Population Model and the Landscape Model under current and burned conditions. The Landscape model was only used to predict in areas not considered canyons. "Difference In Use" spatial predictions were developed to better visualize change in habitat use by subtracting the current condition relative predictions from the burned relative predictions.

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CHAPTER 2: EVALUATING THE EFFECTS OF WILDFIRE AND POST-FIRE SEEDING EFFORTS IN PIÑON-JUNIPER WOODLANDS

SUMMARY

Wildfire and the subsequent effects of wildfire on vegetation composition are a growing concern in piñon and juniper woodlands across the western United States due to high stand density in some areas. I compared vegetation attributes of naturally burned piñon-juniper woodlands (12 years post-fire) to unburned piñon-juniper woodlands in the Black Ridge Canyons Wilderness of western Colorado. I also analyzed post-fire seeding efforts aimed at mitigating the prevalence of *Bromus tectorum*, a highly invasive annual grass which dominates millions of acres across the western United States. In general, burned habitats had significantly greater cover of both native and non-native grasses, greater native forbs cover, and less developed biological soil crusts. Significantly negative correlations between seed mixture species and *B. tectorum* indicate that post-fire seeding likely helped to mitigate the coverage of *B. tectorum*. I found additional evidence of post-fire seeding effectiveness by showing that the relative cover of native grass species to cover of *B. tectorum* increased proportionally between unburned and some burned habitats. My results provide managers a site specific assessment of past conservation efforts in addition to evidence that can be used for future conservation planning.

INTRODUCTION

Piñon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) woodlands (PJ woodlands) make up 67% of all habitats in the Black Ridge Canyons Wilderness Area (BRCWA), encompassing over 21,000 hectares (See Appendix B for details). The distribution, composition, and stand structure of PJ woodlands are generally considered both variable and dynamic. Natural variability is controlled by the productivity of soils in which they occur, disturbance events (e.g., wildfire), and the availability and timing of precipitation. Based on these factors, Romme et al. (2009) outlined three fundamentally different types of PJ woodlands; persistent piñon-juniper woodlands, piñon-juniper savannas, and wooded shrublands. Accordingly, stands within the BRCWA are considered persistent piñon-juniper woodlands where precipitation is bimodal (i.e., small peaks in both the summer and the winter) and high intensity fire is the dominant fire type.

Low-severity wildfires are not thought to greatly influence PJ woodland stand structure due to a lack of continuous fine understory fuels which, if present, allows fires to spread (Floyd et al. 2000, Baker and Shinneman 2004). Piñon pine and Utah juniper are relatively susceptible to fire because of their thin barks and low crown height. Therefore, fires are able to completely burn individual trees but the spread of fires is dependent on a combination of factors including stand density, understory composition, and weather conditions. Dense PJ woodlands often have sparse herbaceous understories due to competition for resources in a relatively unfavorable and arid environment. The soil surface typically features bare ground, biological soil crusts, and only patches of herbaceous plants with needle duff accumulating directly below trees. Huffman et al. (2012) found that total woody biomass in PJ woodlands of Arizona maximized around 250 years following stand replacing fire. However, increased prevalence of fine understory fuels in the form of *Bromus tectorum* (i.e., cheatgrass, downy brome) is thought to be altering fire regimes in some PJ woodlands.

Vulnerability of PJ woodlands to non-native plant invasions is a growing concern throughout the southwestern United States. Soil nutrients are inherently low in semi-arid systems but they can dramatically increase as a result of fire (Stubbs and Pyke 2005). *Bromus tectorum* can quickly invade following fire (Morrow and Stahlman 1984), while native perennial species may not fully take advantage of increased resources because of small seed banks, variable seed production and short lived seeds (Hassan and West 1986). Additionally, native grasses are generally poor competitors due to the ability of *B. tectorum* to germinate earlier in the fall and under colder winter conditions (Aguirre and Johnson 1991). Once *B. tectorum* has become established, natural ecosystem processes can be dramatically altered (Hobbs 2000). For example, *B. tectorum* has been shown to escalate fire frequency by increasing loads fine fuel loads necessary to carry fire (Whisenant 1990, Knapp 1996). Hull (1965) estimated that rangelands dominated by *B. tectorum* are 10-500 times more likely to burn than rangelands dominated by native bunchgrass species.

Fire can also result in negative and often irreversible effects on soil structure. In the intermountain west, high intensity short duration rainfall events often occur during and shortly after severe wildfires (Robichaud et al. 2000), putting watersheds in these areas vulnerable to erosion (Wagenbrenner 2006). Fire within PJ woodlands can substantially decrease cover of biological soil crusts which provide several beneficial ecosystem services in semi-arid environments (Evangelista et al. 2001). These include soil stability, water retention, and germination enhancement (Anderson et al. 1982, Belnap and Gardner 1993). Furthermore, soil crusts are seen as vital to occupying open spaces and may assist in limiting the establishment of *B. tectorum*.

The Bureau of Land Management makes every effort to rehabilitate burned landscapes to prevent soil erosion and the expansion of non-native species whenever it is economically and logistically feasible. Rehabilitation most often involves reseeding recently burned landscapes with native species but high costs and limited availability of native seeds often limits the composition of seed mixtures. The effects of seeding on native plant recovery can be strongly influenced by which species are seeded, post-fire precipitation, fire severity, and time since fire (Schoennagel & Waller, 1999; Barclay, Betancmyt, & Allen, 2004; Wagenbrenner, 2006). *Bromus tectorum* was already present in the burned and unburned portions of the BRCWA, making post-fire seeding of native species necessary to prevent further spread of this undesirable non-native species. Particular native species have been shown to compete with *B. tectorum*, though success can be variable. In a similar PJ woodland system, Getz and Baker (2008) found that both *Thinopyrum intermedium* (i.e., intermediate wheatgrass) and *Pleuraphis jamesii* (i.e., James' galleta) was able to restrict or limit *B. tectorum* success following a fire.

Currently, the canopy cover of PJ woodlands in the BRCWA is estimated to range from 0% - 61% (See Appendix B for details). Several large scale wildfires (>200 ha) have occurred in the area over the last 20 years, all of which are thought to have been ignited by lightning strikes. In particular, a series of wildfires occurred on July 2nd, 1999 burning over 1,200 ha of PJ woodlands in the western portion of the BRCWA. Post-fire reports by the Bureau of Land Management (1999) indicate that the fires were likely carried by high cover of *B. tectorum*. The fires were actively suppressed and a native seed mixture was aerially applied the following winter to limit erosion and the prevalence of *B. tectorum*. In 2012 and 2013, I conducted an assessment of the vegetation composition of the burned and unburned PJ woodland habitats in the BRCWA to determine the effects of wildfires and subsequent post-fire seeding efforts. I anticipate that my results will be valuable to managers in evaluating past restoration efforts and informing future management decisions; particularly in designated wilderness landscapes, where management goals are centered on preserving natural ecosystem processes. My analysis was focused on comparing vegetation species richness, grass and forb foliar cover, biological soil crust cover, and the spatial arrangement of invasive species between burned and unburned PJ woodlands.

METHODS

Data was collected from within two burned strata and two unburned strata (Figure 7). The burned strata are known as the Long Mesa Fire and the Moore Canyon Fire. Each fire was ignited by lightning strikes on July 2nd, 1999 and each received the same post-fire seed mixture. The Moore Canyon Fire is comprised of a single continuous burned area, while the Long Mesa fire is composed of three burned portions in close proximity to each other but separated by dramatic mesa cliffs. The two unburned strata were chosen to provide complete spatial coverage of all potential PJ woodlands within the BRCWA. All PJ woodlands occur within one of two soil types described for this area (National Resources Conservation System Soil Survey). I refer to areas with soils designated as gladel-bond rock outcrop complex as Lowland PJ and soils designated as zyme-rock outcrop-gladel complex as Upland PJ. Lowland PJ is dominated by sandy loam soils and Upland PJ is dominated by silty clay loam soils. The wildfires chosen for this study only occurred within Lowland PJ, therefore statistical comparisons will only be made between strata of the same soil type (i.e., Lowland PJ vs Long Mesa Fire and Lowland PJ vs. Moore Canyon Fire) and between the unburned strata (i.e., Lowland PJ vs Upland PJ).

Trade-offs between sampling extent, data specificity, and collection efficiency must be considered when selecting between vegetation sampling methods (Stohlgren 1994, Nusser and Goebel 1997). My study was focused on quantifying and comparing vegetation cover and species richness estimates during a two month summer sampling period. I chose to conduct vegetation sampling using intense modified-Whittaker plots (Barnett and Stohlgren 2003). The nested multi-scale plot design contains one centrally located 10 m^2 plot and four systematically located 1 m^2 subplots within a 100 m^2 outer plot (see Figure 8 for a diagram of plot design). The multi-scale design allowed for greater localized spatial coverage (Barnett and Stohlgren 2003) needed to describe the relatively sparse and often clustered vegetation found in the BRCWA. The non-overlapping placement of the 1 m^2 sub-plots reduces spatial autocorrelation which can be an issue with other commonly used quadrat and transect methods (Parker 1951, Daubenmire 1959). The smaller intensive version of the modified-Whittaker plot was chosen to increase sample size and spatial extent while maintaining the multi-scale features that allow for estimation of species cover and species richness (Barnett and Stohlgren 2003).

Plot sites were randomly allocated within each strata and constrained to landscapes with slopes <20% to avoid sampling in areas with high levels of erosion or exposed rock. A larger number of sampling sites were initially created to compensate for locations that were unreachable due to topography and rugged terrain. Plots at lower elevations were conducted first when possible and continued higher in elevation as the season progressed in attempt to correlate sampling effort with peak plant phenology. To achieve complete coverage of the habitats of interest, I allowed some randomly located plots to fall outside of the BRCWA within the adjacent Colorado National Monument. Plots were placed parallel to prominent vegetation gradients to capture maximum heterogeneity for each site. In each of the four 1 m^2 subplots, I recorded foliar cover by species and cover of litter, duff, rock, and bare ground. Overhanging shrub and tree cover was recorded separately. Cover estimates were taken for three developmental stages of biological soil crusts based on the six physical indices of development described by Belnap et al. (2008) . I further categorized these into three general developmental stages for my analyses (Figure 9). All coverage estimates ≥5% were quantified in 5% increments with estimates <5% quantified in 1% increments. The 10 m^2 center plot and the 100 m^2 exterior plot were exhaustively searched and unique species were recorded. Classification of species as to their invasive status was based on the Natural Resources Conservation Service PLANTS database (U.S. Department of Agriculture 2012). All intense modified-Whittaker plots were conducted during the months of June and July of 2012 and 2013 when peak phenology and biomass was expected for most species.

DATA ANALYSIS

Data analysis was conducted using R statistical software version 3.02 (R Core Team 2013). Data did not meet distributional assumptions required for parametric analysis. Various transformations of the data were tested, but also failed to meet parametric assumptions. Therefore, nonparametric Wilcoxon-Mann-Whitney tests were used to test for differences between means of vegetation parameters for the appropriate strata. Comparisons were made separately for each burn site with caution due to inherent pseudo-replication when sampling within the a single burn (Hurlbert 1984). I also separated data by season due to noticeable differences in precipitation during the months when the majority of vegetation growth occurs.

Specifically, from March 1st-June 31st 2012, only 22.6 mm (0.89 inches) of precipitation was received at the Colorado National Monument's weather station; but in 2013, 58.9 mm (2.32 inches) of precipitation was received during the same time period, an increase of over 2.5 times (National Oceanic and Atmospheric Administration 2013).

The vegetation parameters analyzed included species richness, cover of three functional groups (grass, forbs, and shrubs), and cover of biological soil crust. I also analyzed the proportion of native and non-native grasses in the burned and unburned strata. I report all Bonferroni adjusted p-values and considered estimates significantly different when found to be ≤ 0.05. Pearson correlation coefficients were calculated to statistically analyze if presence of a particular grass species may relate to decreased cover of *B. tectorum* within the same $1m^2$ subplot. Similarly, I tested for trends in *B. tectorum* cover directly beneath piñon pine and Utah juniper trees to further understand the spatial arrangement of *B. tectorum*.

RESULTS

I was able to conduct a balanced sample of 20 plots per strata during the 2013 sampling season. During the 2012 sampling season I established 36, 25, and 20 plots within Upland PJ, Lowland PJ and the Long Mesa Fire respectively. However, decreased amounts of snowmelt and spring rainfall in 2012 made access to the Moore Canyon Fire via the Colorado River almost impossible and I was only able to conduct 4 plots in this stratum. Estimates for 2012 from the Moore Canyon fire are reported, but no statistical tests were conducted due to limited sample size. Species accumulation curves were developed to assess the completeness of sampling for each sampling season by calculating the number of new unique species encountered with each additional plot conducted. In general, the species-accumulation curves began to level off for each of the four strata (Figure 10), indicating that the number of plots in each was adequate.

Throughout the study period a total of 86 unique species were identified from within vegetation plots (see Table 3 for complete list of species). The most prevalent species found during the study was *B. tectorum*, occurring in 84% (139/165) of plots. The species list is not thought to be exhaustive because sampling was only conducted during the months of June and July when most species are at their peak biomass. Less than 5% of species encountered were unidentified beyond the genus level. All unidentifiable plants were forbs, and in most cases, lacked the phenological features required for proper taxonomic classification. In these cases, the genus was recorded and the data was included in all cover estimates but excluded for the species richness analysis. Overall, each of the sampled strata held similar composition of species, with the exception of several *Eriogonum* species (e.g., Sand Buckwheat, *Eriogonum leptocladon*) only being found within burned strata.

Non-native species richness was found to be significantly greater in both burned areas than in unburned Lowland PJ (Table 4). Both burns had an average of ≈3 non-native species per plot while Lowland PJ and Upland PJ averaged ≈2 non-native species per plot. Most often the difference in non-native species richness could be attributed to *Sisymbrium loeselii* (i.e., Tumble Mustard). The number of native species per plot did not significantly differ between any of the strata tested. Both burned habitats averaged ≈10 native species while unburned Lowland PJ only had ≈9 native species. Upland PJ also averaged ≈10 native species per plot.

During both sampling seasons, native grass cover in Lowland PJ was found to be significantly less than native grass cover in both the Long Mesa Fire and the Moore Canyon Fire. Upland PJ exhibited greater native grass cover but estimates were not significantly different (p=0.074) from Lowland PJ. Non-native grass cover was also found to be less within Lowland PJ than both the Long Mesa Fire and the Moore Canyon Fire, although significant differences were only found during the 2013 sampling period. *Bromus tectorum* contributed more than 95% percent of all non-native grass cover in each of the four strata sampled. Native forb cover in Lowland PJ averaged ≈4% during both sampling seasons, with slightly less native forbs cover found during 2012. Both the Long Mesa Fire and the Moore Canyon Fire had significantly greater native forbs cover. Upland PJ also had greater native forbs cover, but significant differences were only detected during the 2013 sampling season. Non-native forb cover was found to be less than 3% for all strata. The Long Mesa Fire, the Moore Canyon Fire and Upland PJ all had greater non-native forbs cover than Lowland PJ but again differences were only significant during the 2013 sampling season.

I found no significant differences in 2012 or 2013 when analyzing if native and non-native grasses were proportionally different in burned compared to unburned habitats. Ratios of native grass cover to non-native grass cover for all strata did not meet parametric assumptions so I report both median and average estimates along with 95 % confidence intervals (Table 5). Lowland PJ, the Long Mesa Fire, and the Moore Canyon Fire all exhibited greater average ratios in 2012 when precipitation was dramatically less than that in 2013. The median of ratios in all seasons of these strata were extremely similar. However, average ratios and 95% confidence intervals were not as consistent, suggesting far greater variability within burned strata. In general, Upland PJ was found to have proportionally more native grass cover than non-native grass cover indicated by greater ratio values.

Cover of Level 3 (the most developed) biological soil crust was significantly greater within Lowland PJ than within both burned strata. (see Figure 11 for boxplot of 2013 biological soil crust cover estimates, see Table 6 for test results). Cover of biological soil crusts varied slightly by season likely due to contraction of crusts during drier conditions. Cover of Level 2 biological soil crust did not significantly differ between strata. Within Upland PJ, Level 2 biological soil crusts exhibited the greatest coverage. Both burned strata did exhibit significantly greater Level 1 biological soil crust cover than unburned strata.

Trees species other than *P. edulis* or *J. osteosperma* were rarely encountered (<2% of plots). Subplot ($1m^2$) cover was recorded for all tree species but it is likely that plots were not able to accurately quantify overall stand density on a broader scale due to asymmetric growth patterns and variable spatial arrangement of PJ woodlands. Stand density for the entire study area was estimated from aerial imagery in conjunction with the companion wildlife study referenced. Using this data, the average canopy cover of piñon pine and Utah juniper in Lowland PJ and Upland PJ was found to be 19.56% and 12.81%, respectively (See Appendix B for details).

Pearson's correlations between particular native grass species (some of which were aerially seeded following the fires) and *B. tectorum* all revealed negative relationships in cover within burned and unburned Lowland PJ soils (Table 7). It is important to note that a zero sum is not likely to exist between cover of different grass species and thus, correlations only assist in describing how native grass species are spatially related to *B. tectorum*. In the Long Mesa Fire, *Hesperostipa comata* (i.e., needle and thread grass) and *Plueraphis jamesii* (i.e., James' galleta grass) cover showed a significantly negative relationship with *B. tectorum* cover. In the Moore Canyon Fire, the negative relationship between *Poa secunda* (i.e., Sandberg bluegrass) and *B. tectorum* was significant. Correlations for the same species but within Upland PJ were found to actually be positive, although none were significant. Pearson's correlations were also calculated between overhanging piñon-pine or Utah juniper and *B. tectorum* within $1m^2$ subplots to further explain the spatial arrangement of *B. tectorum* on the unburned landscape. Significantly positive relationships do exist between overhanging tree species and *B. tectorum* within both the Lowland PJ ($\rho=0.37$, p-value<0.01) and Upland PJ ($\rho=0.73$, p-value<0.01).

DISCUSSION

Vegetation within burned habitats has had over 12 years to become established so it is likely that the current herbaceous composition is somewhat stable, and thus my study provides an appropriate retrospective view regarding the effects of wildfire and post-fire seeding efforts. Species richness of PJ woodland understories is influenced by stand density and canopy continuity with the greatest species richness occurring prior to stand establishment (Koniak and Everett 1982, Pieper 1990). I found that burned plots did not have significantly different native species richness but did have significantly greater non-native species richness (≈1 additional non-native species). Most non-native species encountered were those that have annual life strategies and high seed production (e.g., *B. tectorum*, *Sisymbrium loeselii*).

I also found increases in total grass and forbs cover in burned habitats which likely increases year round forage for ungulates such as for desert bighorn sheep (*Ovis canadensis nelsoni*) and seasonal forage for elk (*Cervus canadensis*) and mule deer (*Odocoileus hemionus*) that migrate through the area during the winter months. Most notably, burned areas had high cover of *H. comata*, a native drought tolerant, cool season bunchgrass that was included in the post-fire rehabilitation seed mixture. Although this species is identifiable by a sharp seed with a long twisted awn, it still provides quality forage for ungulates, especially in early spring (U.S. Department of Agriculture 2012). Burned areas also had greater cover of forb and shrub species than unburned habitats which is most likely to be taken advantage of by the local population of desert bighorn sheep due to their opportunistic and adaptable feeding behaviors (Leslie and Douglas 1979, Cunningham 1989, Krausman et al. 1989, Miller and Gaud 1989).

Differences in the cover of highly developed biological soil crusts between strata suggest that the burned habitats have yet to fully regenerate biological soil crusts 12+ years after the burn. Past research has shown that even nonliving soil crusts can stay intact following being burned and can continue to provide beneficial services (Belnap and Gardner 1993) which may assist in additional regeneration. My results are similar to the findings of Evangelista et al. (2001) who found that fire substantially decreased older and highly developed crusts. The remote nature of the burned habitats and current BRCWA management policies favors that post-fire seed mixtures be applied aerially which is much less destructive than mechanistic means that can further degrade biological soil crusts. Highly developed crusts are not as prevalent within Upland PJ which can likely be attributed to soil type and more rocky surfaces. Therefore, it is expected that biological crust regeneration following fire in Upland PJ would be additionally limited.

By analyzing the presence of seeded species in relation to *B. tectorum* managers can better grasp the success of seeding efforts (Table 8 for details of seed mixture). *Hesperostipa comata* and *Poa secunda* revealed significantly negative relationships with cover of *B. tectorum.* Interestingly, *Pascopyrum smithii* (i.e., western wheatgrass) was included in the seed mix but was not encountered within the burns. The lack of western wheatgrass in the burns could be due to poor seed germination or the environmental conditions following seeding. Regardless, the high cost of including western wheatgrass in the seed mix warrants hesitation when contemplating its inclusion in future post-fire rehabilitation seed mixtures. Alternatively, *P. jamesii* was not included in the seed mixture but was widely present in the burns and unburned areas. Furthermore, correlations between *P. jamesii* and *B. tectorum* were found to be significantly negative. The prevalence of *P. jamesii* in the burns is encouraging and demonstrates the ability of surrounding native species to colonize on their own.

I found significant correlations between *B. tectorum* and trees species. *Bromus. tectorum* is likely prevalent directly below trees due to the lack of biological soil crusts, litter accumulation, and increased available moisture. This correlation has direct implications for wildfire behavior because *B. tectorum* can provide the fine fuels required for fire in PJ woodlands to persist by allowing embers to ignite individual trees spatially separated from where the fire originated. However, this spatial arrangement of fine fuels directly below individual trees may only provide fire ignition of single tress and not fuel continuity between trees unless other environmental conditions such as wind and moisture levels are also supportive.

To further describe both the effects of fire and the effectiveness of seeding efforts, I compared the proportion of native grass cover to *B. tectorum* cover for the burned and unburned strata. These comparisons describe if grass cover in burned habitats is proportionally greater and provides an indication if the burn resulted in a general increase in grass cover or if either native grasses or *B. tectorum* increased disproportionally. Median ratios of native grass cover to *B. tectorum* cover revealed very similar results in both burned areas and Lowland PJ. But average ratios were greater in 2012 than in 2013, suggesting that prevalence of *B. tectorum* was affected greater by limited available moisture. This is likely because most native grass cover was in the form of bunchgrasses that remain present from year to year while *B. tectorum* is an annual species that depends on annual conditions for germination and growth. Furthermore, native grass cover was relatively consistent regardless of sampling season suggesting that *B. tectorum* cover was correlated to fluctuating levels of available moisture more than native grass cover. These observations are similar to Smith et al. (2008), who analyzed *B. tectorum* seed bank carryover and found higher prevalence of *B. tectorum* when moisture availability was greater. Interestingly, the same study also concluded that *B. tectorum* seeds rarely persist in the seed bank beyond two years in a semi-arid environment. However, it is likely that several years of drought conditions would do little to greatly hinder the long term success of *B. tectorum* due to its ability to potentially produce as many as 300 seeds per parent (Hulbert 1955) or thousands of seeds per square meter (Sheley and Larson 1994) when conditions improve. Higher variability of native to non-native grass cover ratios may indicate that seeding efforts were spatially inconsistent or site specific environmental variables may be playing a strong role in supporting *B. tectorum*. Overall, the similarity of median ratios suggests that coverage of both native and non-native grass species increases proportionally following fire which may be viewed optimistically as successfully mitigating the prevalence of *B. tectorum.*

In summary, future large scale wildfires in the BRCWA are likely to occur. Based on my results and the widespread prevalence of *B. tectorum* in the BRCWA, I suggest that post-fire aerially seeding of native grasses be conducted when possible. Seed mixtures should be carefully chosen with the results of this study in mind. Although some negative outcomes, such as increased non-native species richness and increased cover of *B. tectorum,* are likely to occur following a fire, proactive management strategies can also increase the establishment of native vegetation and help restore areas back to their natural states.

Table 3. Complete list of unique species encountered in plots, seperated by functional group.

Unique Species List: Black Ridge Canyon Vegetation Analysis		
Grasses	**Trees**	**Forbs (*cont'd*)**
Achnatherum hymenoides	*Fraxinus anomala*	*Lepidium densifolium*
Agropyron cristatum	*Juniperus osteosperma*	*Lepidium montanum*
Artistda purpurea	*Pinus edulis*	*Lepidium perfoliatum*
Bromus tectorum		*Linum lewisii*
Elymus elymoides	**Forbs**	*Lomatium grayi*
Hesperostipa comata	*Alyssum desertorum*	*Machaeranthera gracilis*
Leymus salinus	*Anntennaria dimorpha*	*Mirabilis glandulosa*
Pleuraphis jamesii	*Astagulus mollisimus*	*Opuntia polycantha*
Poa bulbosa	*Astragulus megacarpus*	*Penstemon strictus*
Poa secunda	*Astragulus oophorus*	*Penstemon watsonii*
Sporobolus cryptandrus	*Calochortus nuttallii*	*Petradoria pumila*
Elymus trachycaulus	*Castilleja integra*	*Phlox hoodii*
Pascopyrum smithii	*Chaetopappa ericoides*	*Phlox longifolia*
	Chamaesysce serpyllifolia	*Physaria acutifolia*
Shrubs	*Chorispora tenella*	*Salsola tragus*
Artimisia tridentata	*Cryptantha pterocarya*	*Schoenocrambe linifolia*
Atriplex canescens	*Cryptantha virgata*	*Sclerocactus whipplei*
Atriplex conferifolia	*Descurania pinnata*	*Sisymbrium loeselii*
Cercocarpus ledifolius	*Eriogonum cernuum*	*Sphaeralcea concinea*
Chrysothamnus viscidiflorus	*Eriogonum gordonii*	*Sphaeralcea leptophylla*
Ephedra viridis	*Eriogonum ovalifolium*	*Sphaeralcea parvifolia*
Eriogonum leptocladon	*Grindelia squarossa*	*Sphaerlacea fendleri*
Ericameria nauseosa	*Gutierrezia sarothrae*	*Stanleya pinnata*
Grayia Spinosa	*Halogeton glomeratus*	*Tetraneuris acaulis*
Krascheninnikovia lanata	*Hedysarium borreale*	*Tragapogon dubius*
Purshia stansburiana	*Heterotheca villosa*	*Vivia americana*
Sarcobatus vermiculatus	*Hymenopappus filifolius*	*Yucca baccata*
Quercus havardii	*Hymenoxis hoopesii*	
Ephedra torreyana	*Hymenoxys richardsonii*	

Table 4. Estimated means of vegetation composition attributes, including standard errors. Bonferonni adjusted p-values represent results of non-parametric Wilcoxon-Mann-Whitney tests for differences of means between Lowland PJ and specified strata.

Black Ridge Canyon Wilderness Area Vegetation Analysis			**Soil A**					**Soil B**	
			Unburned	*Burned*				*Unburned*	
			Lowland PJ	*Long Mesa Fire*		*Moore Canyon Fire*		*Upland PJ*	
			Mean (SE)	Mean (SE)	p (Bonf. adj.)	Mean (SE)	p (Bonf. adj.)	Mean (SE)	p (Bonf. adj.)
Species Richness	Native	2012	6.24 (0.40)	6.65 (0.56)	1	6.25 (0.45)	NA	4.25 (0.63)	1
		2013	5.45 (0.52)	6.7 (0.62)	0.453	7.05 (0.55)	0.109	7.75 (0.49)	0.018 **
	Non-Native	2012	0.84 (0.10)	1.3 (0.15)	0.106	1.5 (0.50)	NA	0.67 (0.15)	0.818
		2013	1.15 (0.17)	2.65 (0.25)	0.0001 ***	2.2 (0.28)	0.014 **	1.8 (0.12)	0.007 ***
Grass Cover	Native	2012	6.83 (1.11)	15.81 (1.64)	0.0004 ***	5.25 (4.43)	NA	11.98 (2.02)	0.074 *
		2013	5.58 (1.44)	13.38 (1.79)	0.006 ***	14.13 (1.06)	0.0002 ***	8.08 (1.29)	0.35
	Non-Native	2012	5.99 (1.08)	13.31 (3.84)	0.698	11.88 (6.78)	NA	5.42 (2.2)	0.506
		2013	6.30 (1.35)	20.21 (4.36)	0.003 ***	16.35 (2.70)	0.009 ***	4.93 (0.76)	1
Forb Cover	Native	2012	3.63 (0.95)	7.13 (1.11)	0.042 **	2.13 (0.81)	NA	4.72 (1.18)	0.14
		2013	4.58 (0.81)	10.66 (1.13)	0.00065 ***	7.51 (0.84)	0.052 *	13.60 (1.59)	0.00003 ***
	Non-Native	2012	0.01 (0.008)	0.15 (0.11)	1	1.69 (1.69)	NA	0.35 (0.26)	0.36
		2013	0.39 (0.16)	1.70 (0.39)	0.0014 ***	2.45 (0.56)	0.0018 ***	1.05 (0.22)	0.074 *
Shrub Cover	Native & Non-native	2012	10.44 (1.57)	3.43 (0.91)	0.018 **	4.25 (2.45)	NA	6.06 (1.39)	0.26
		2013	8.14 (1.81)	3.25 (0.94)	0.112	4.69 (0.83)	0.096 *	7.56 (2.15)	1

sig. levels of Bonferonni adjusted p-values: * < 0.10, ** < 0.05, *** < 0.01

Table 5. Estimated ratios of native to non-native grass cover based on averaged cover in intense modified-Whittaker plots during the 2012 and 2013 sampling seasons. I report median and average estimates in addition to 95% confidence intervals because of non-normal data distributions. 2012 ratios for the Moore Canyon fire are not likely accurate due to small sample size (n=4).

			Season	Median	Average	95% CI
Soil A	*Unburned*	*Lowland PJ*	2012	1.000	2.597	(.951 , 4.243)
			2013	1.000	1.502	(0.603 , 2.399)
	Burned	*Long Mesa Fire*	2012	1.090	8.090	(.067 , 8.869)
			2013	1.060	1.605	(0.765 , 5.444)
		Moore Canyon Fire	2012	0.305	2.000	(-1.672 , 5.625)
			2013	0.948	3.634	(0.843 , 6.4262)
Soil B	*Unburned*	*Upland PJ*	2012	2.175	2.175	(1.414 , 10.031)
			2013	1.955	1.955	(1.302 , 4.718)

Table 6. Non-parametric Wilcoxon-Mann-Whitney tests for difference between mean estimates of biological soil crust cover separated by development level. Bonferroni adjusted p-values are reported.

Black Ridge Canyon Wilderness Area Cover of Biological Crust			**Soil A**							**Soil B**		
			Unburned	*Burned*						*Unburned*		
			Lowland PJ	*Long Mesa Fire*			*Moore Canyon Fire*			*Upland PJ*		
			Mean (SE)	Mean (SE)	p (Bonf. adj.)		Mean (SE)	p (Bonf. adj.)		Mean (SE)	p (Bonf. adj.)	
Development Level	Level 1	2012	1.22 (0.63)	1.46 (0.55)	0.52		2.94 (1.90)	NA		2.78 (1.28)	0.29	
		2013	0.13 (0.09)	1.44 (0.51)	0.119		3.81 (0.64)	0.00001	***	1.75 (0.65)	0.109	
	Level 2	2012	4.78 (0.96)	5.08 (1.97)	0.695		7.50 (4.05)	NA		4.86 (1.53)	1	
		2013	3.06 (0.68)	3.69 (0.58)	1		2.50 (0.81)	0.83		7.33 (1.43)	0.138	
	Level 3	2012	10.20 (2.38)	0.44 (0.39)	0.0053	***	2.25 (4.14)	NA		3.47 (1.49)	0.264	
		2013	17.16 (3.19)	1.83 (0.87)	0.00057	***	0.81 (0.44)	6E-05	***	4.19 (1.70)	0.0039	***

Table 7. Pearson's correlation coefficients between selected grass species and *Bromus tectorum*.

Pearson's Correlations of Grass Species		**Soil A**					**Soil B**
Non-Native *B. tectorum* vs. Native Species		*Unburned*	*Burned*				*Unburned*
		Lowland PJ	*Long Mesa*		*Moore Canyon*		*Upland PJ*
Indian Ricegrass †	*Achnatherum hymenoides*	-0.02	-0.14		-0.17		0.17
Sandberg Bluegrass †	*Poa secunda*	-0.02	-0.04		-0.23	**	0.06
Needle and Thread †	*Hesperostipa commata*	-0.07	-0.31	***	-0.14		0.11
James Galleta	*Plueraphis jamesii*	-0.15	-0.23	**	-0.12		0.00

sig. level: * < 0.10, ** < 0.05, *** < 0.01 , † - species present in seed mixture applied to burned habitats

Table 8. Details of post-fire seed mixture applied to both the Long Mesa fire and the Moore Canyon fire during the winter of 1999-2000. pls# = pure live seed per pound.

Long Mesa and Moore Canyon Fire Seed Mixtures					
Species	$/pls#	#/ac	$/ac	Total Cost	Total pls/#
Indian Ricegrass	$12.00	1	$12.00	$11,640.00	970
Needle and Thread	$35.00	25	$8.75	$8,487.50	243
Sandberg Bluegrass	$112.00	1	$12.00	$11,640.00	970
Western Wheatgrass	$9.50	2	$19.00	$18,430.00	1940
Perrenial Forb Mix	$31.76	0.25	$7.94	$7,701.80	243
Totals		4.5	*$59.69*	$57,899.30	4366

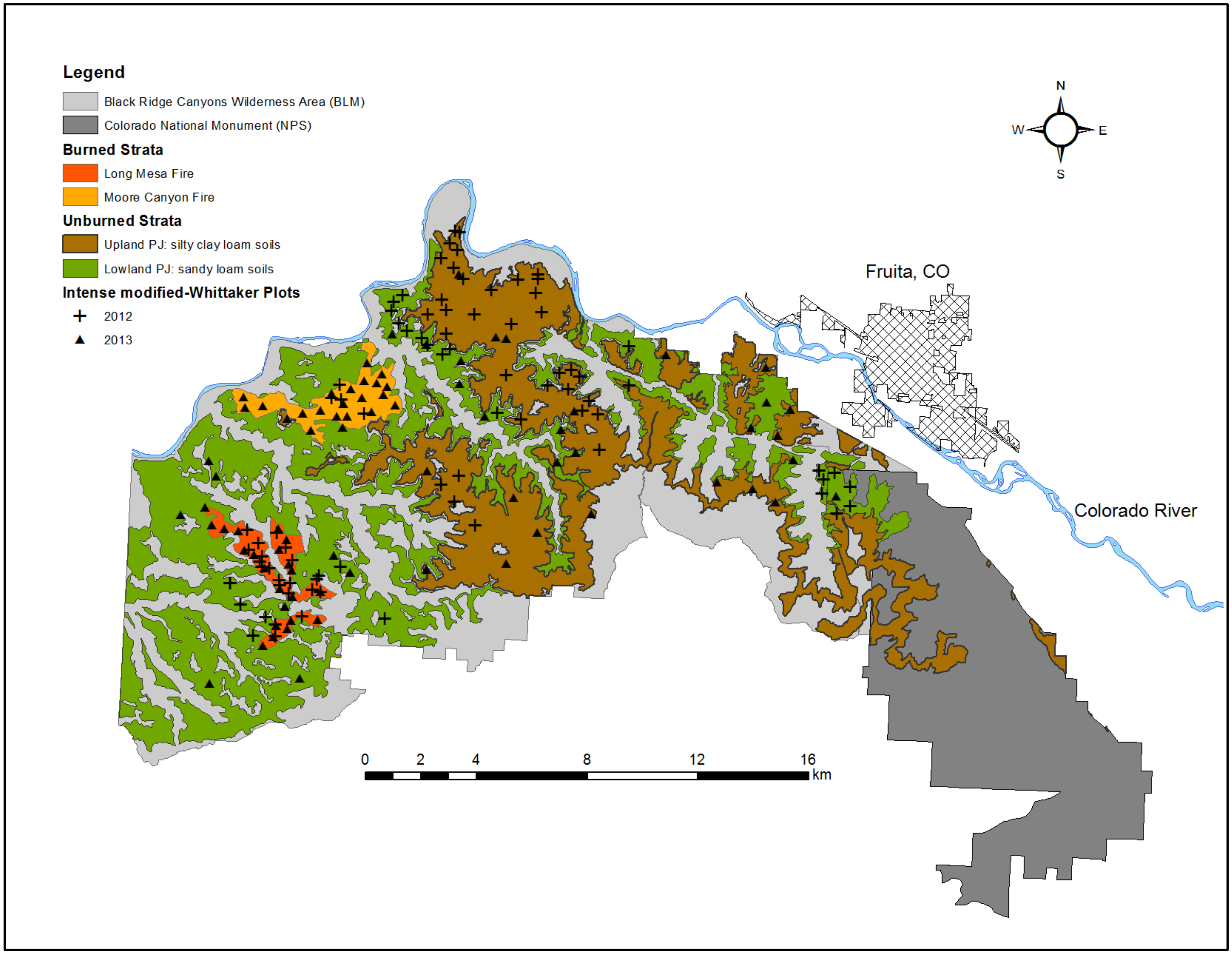

Figure 7. The locations of vegetation sampling plots within the burned and unburned strata for each sampling season.

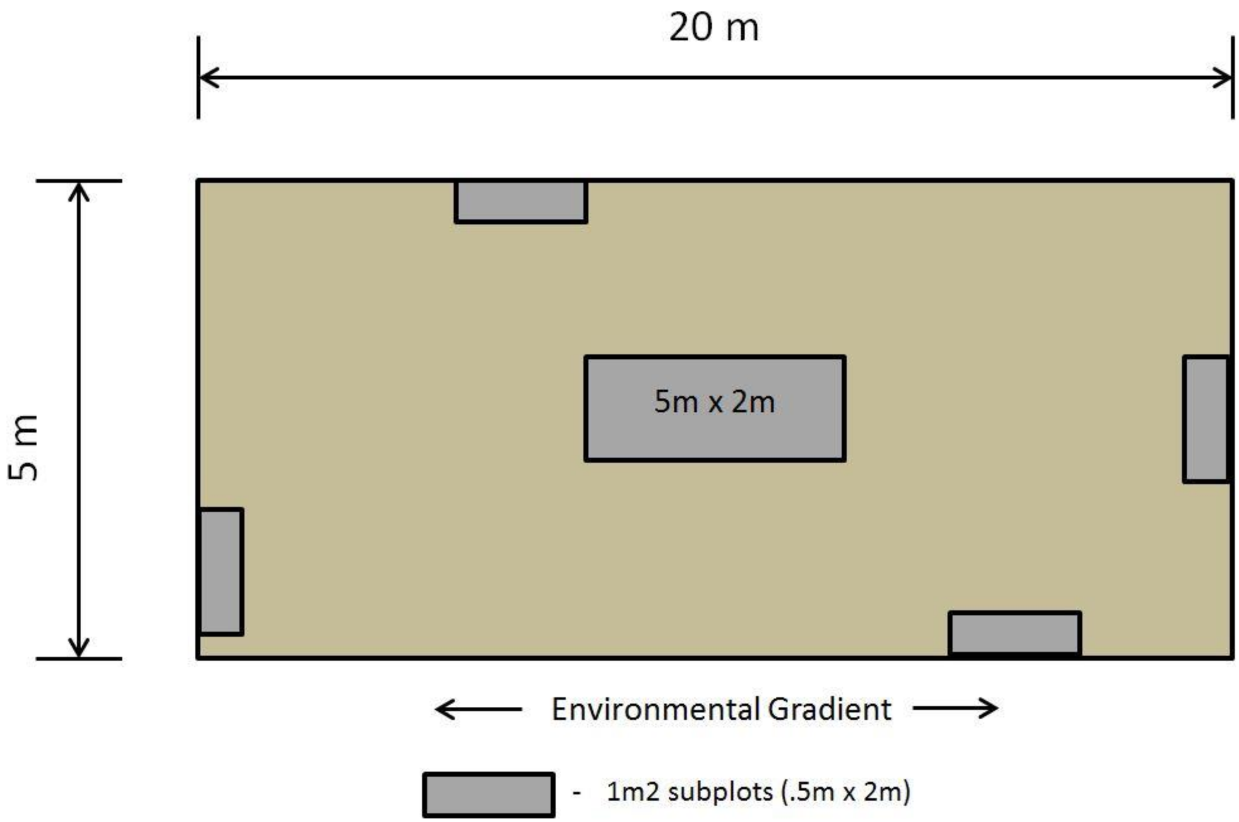

Figure 8. Intense modified-Whittaker plot design. Plots were placed parallel to the major environmental gradient of the vegetation type to assist in capturing maximum heterogeneity at that site. Both the 100 m^2 outer plot and the central 10 m^2 inner plots were exhaustively searched and unique species were recorded. I recorded foliar cover by species in each of the four 1 m^2 subplots as well as percent cover of litter, duff, rock, and bare ground. Overhanging shrub and tree cover was recorded separately.

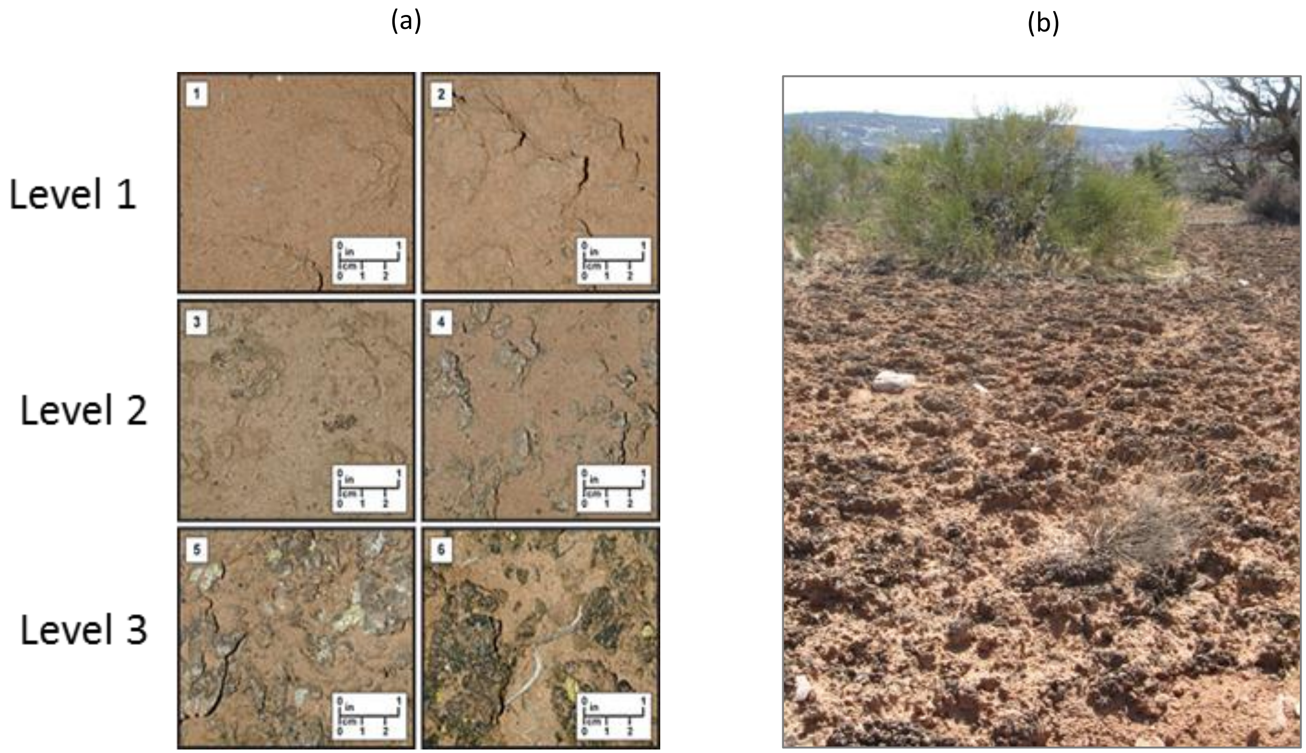

Figure 9. Biologicial soil crusts. (a) Visual aid used in the field to identify development stages of biological soil crusts. (b) Photograph of Level 3 biological soil crust encountered during sampling.

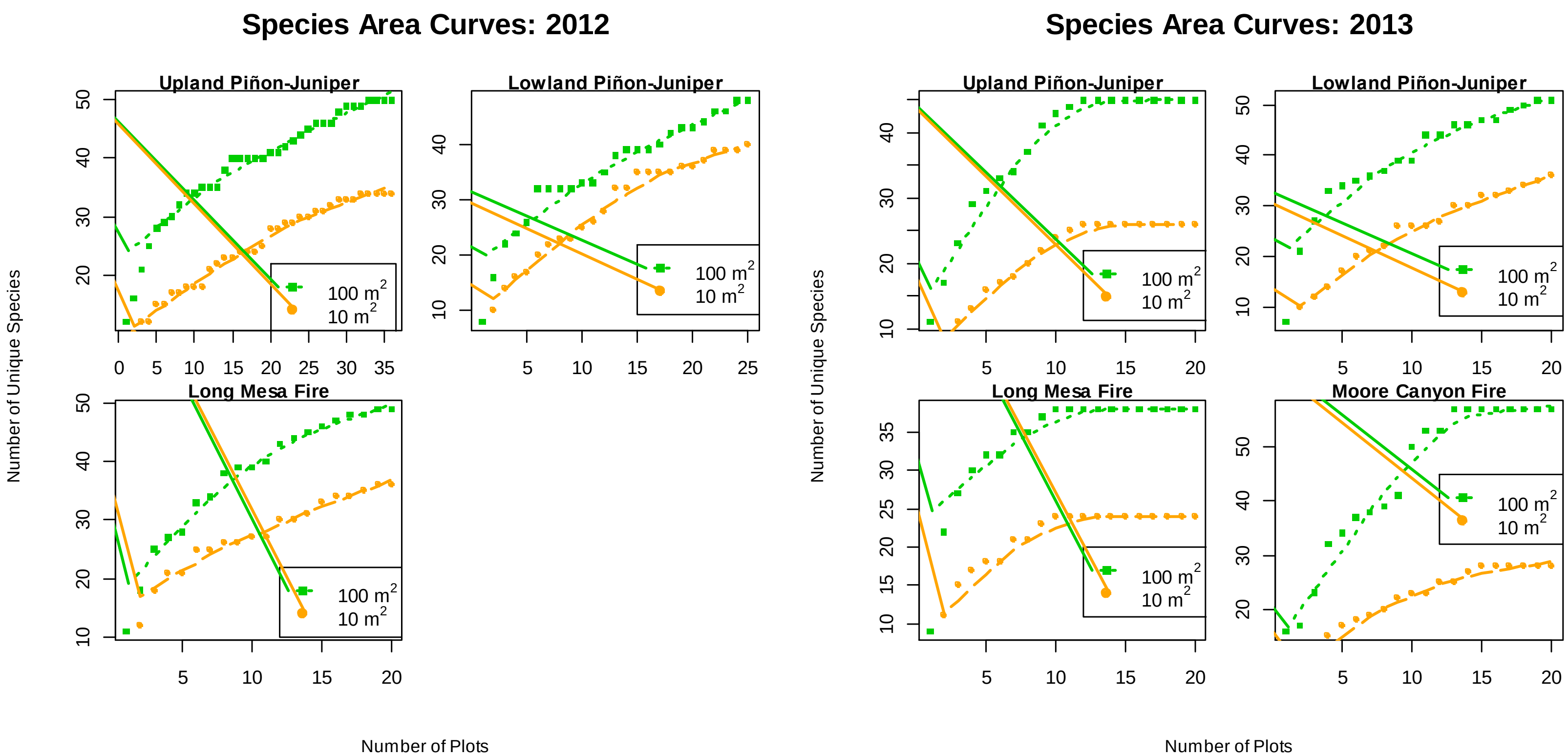

Figure 10. Species accumulation curves for the 2012 and 2013 vegetation sampling seasons.

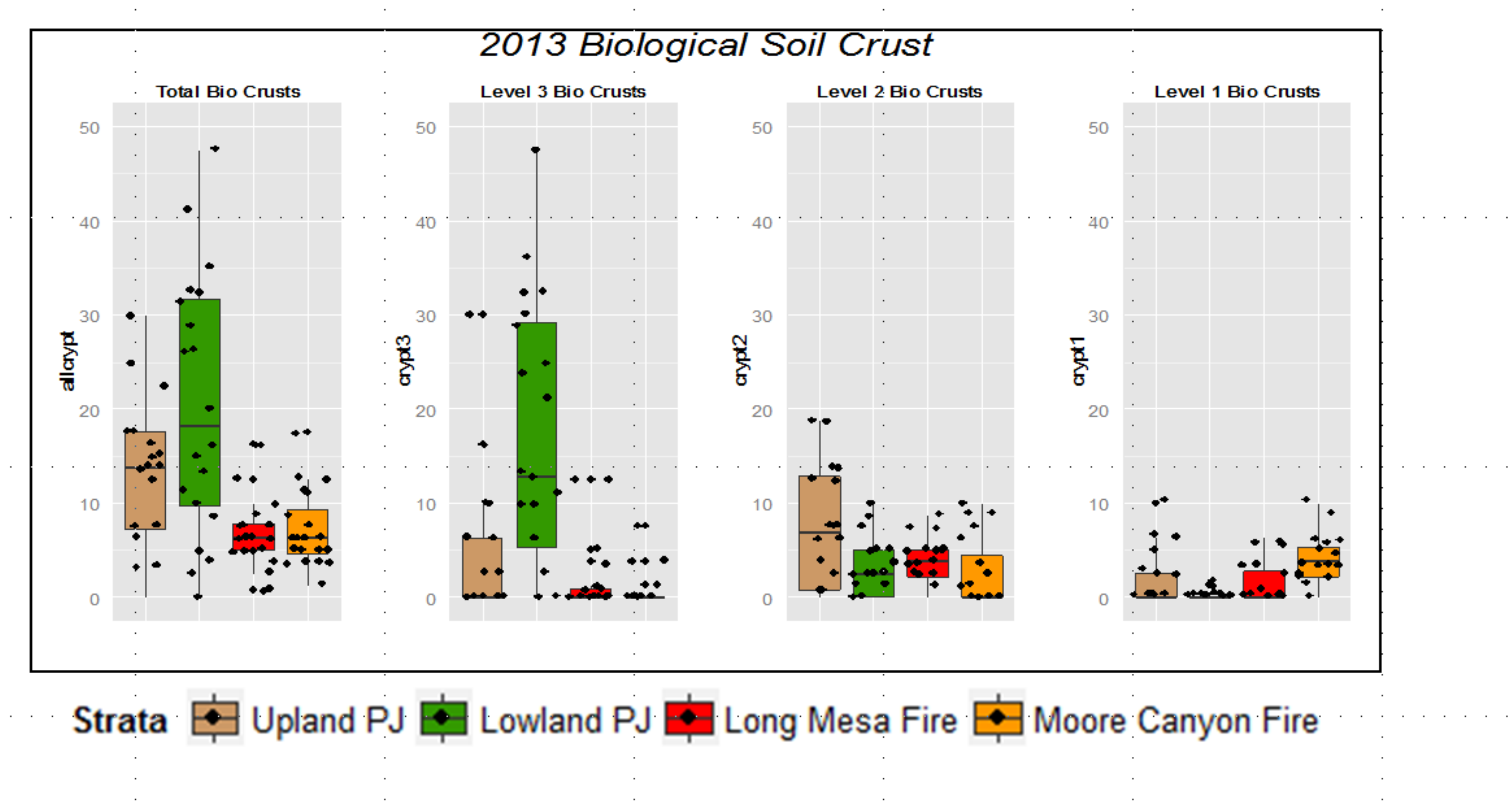

Figure 11. Boxplots of biological soil crust across sampled strata during the 2013 sampling season. Data points were randomly jittered to visualize overlapping data points.

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CHAPTER 3 PROJECT SYNTHESIS

SUMMARY

Piñon-juniper woodlands cover over 40 million ha (100 million acres)of the western United States (USGS National Gap Analysis Program 2004). These ecosystems are known to be variable and dynamic, thus requiring managers to consider site specific attributes when planning and conducting conservation efforts. The Black Ridge Canyons Wilderness Area in western Colorado annually experiences the threat of large scale wildfires in piñon-juniper woodlands. I addressed two land and wildlife management priorities regarding wildfire in the piñon-juniper woodlands of the Black Ridge Canyons Wilderness Area. I first analyzed the potential ability of wildfire to augment habitat for Colorado's largest population of desert bighorn sheep (*Ovis canadensis nelsoni*). Secondly, I compared rehabilitated burned woodlands to unburned woodlands to analyze the effect of wildfire on vegetation composition and the effectiveness of post-fire seeding efforts in terms of non-native species cover. I found that wildfire could greatly increase the amount of habitat available to the bighorn sheep population throughout the wilderness area and was able to identify where wildlife would be most benefited in doing so. I also found that foliar cover in burned areas was on average two times greater than in unburned areas and that post-fire seeding efforts likely allowed for these differences to be proportionally similar between native and non-native grass species. My results suggest that managers should strongly consider allowing future wildfires to burn when rehabilitation is feasible.

INTRODUCTION

It has become evident that ecological processes are often intertwined in complex ways, leading attentive natural resource managers to undertake an ecological perspective in which multiple and possibly competing values are mutually considered. High intensity wildfire is a natural ecosystem process for the piñon and juniper woodlands of the Black Ridge Canyons Wilderness Area (BRCWA) in western Colorado. Wildfire in piñon-juniper woodlands (PJ woodlands) is most often of high intensity but a continuous fuel structure is required for a fire to become widespread. Historically, wildfires in this area are immediately suppressed and a native seed mixture is aerially applied to burned landscapes in an effort to quickly establish native vegetation and reduce the establishment of non-native plant species.

Burned landscapes in the BRCWA typically have greater herbaceous cover of early succession plants (e.g., grasses) but may be more vulnerable to the establishment of non-native plant species, especially *Bromus tectorum* (e.g., cheatgrass, downy brome). Once *B. tectorum* has become established, natural ecosystem processes are extremely difficult to restore (Hobbs 2000). For example, *B. tectorum* has been shown to escalate the frequency of fires by increasing loads of the fine fuels necessary for the spread of fire resulting in an unnatural shortened cyclic fire regime (Whisenant 1990, Knapp 1996). *B. tectorum* is already present in both burned and unburned portions of the BRCWA, making post-fire seeding of native species necessary to prevent widespread coverage of this undesirable non-native species. Particular species have been shown to compete with *B. tectorum* but success is variable.

PJ woodlands make up an estimated 67% of all potential habitats for the desert bighorn sheep (*Ovis canadensis nelsoni*) population that inhabits the BRCWA. The large amount of PJ woodlands is thought to be limiting the quality and quantity of desert bighorn sheep habitat and may be limiting the overall success of the population. Research shows that fire has the ability to augment bighorn sheep habitat for multiple ecosystem types (Peek et al. 1979, Sawyer et al. 2009). Bighorn sheep are known to consistently select habitats with high visibility so they can detect predators from a distance and escape to steep sloped terrain. Fire in PJ woodlands is most often high intensity, killing most or all trees within the burned area regardless of tree size (Romme et al. 2009), thus dramatically increasing visibility and potentially increasing the amount of available bighorn sheep habitat. However, ewes within the BRCWA are known to be utilizing some burned habitats but not utilizing other burned habitats. So questions regarding the ability of fire to augment bighorn sheep habitat in the BRCWA still needed to be addressed.

From a management standpoint, wildfire can have both positive and negative effects on the ecology of a given landscape. I conducted two separate but related ecological analyses to describe the potential effects of wildfire in PJ woodlands for BRCWA. First, I evaluated the efficacy of using woodland fire to alter vegetation composition in a manner that augments bighorn sheep habitat. I applied generalized linear mixed models to estimate pre-fire ewe habitat selection and then simulated a hypothetical widespread fire to spatially predict where fire would be most beneficial in expanding habitat. Second, I addressed concerns regarding potential negative effects of fire in this system by comparing vegetation composition of unburned habitats to burned habitats that were treated with a native seed mixture.

IMPLICATIONS FOR DESERT BIGHORN SHEEP HABITAT

I estimated PJ woodland canopy cover for the BRCWA in addition to desert bighorn sheep home ranges that extended beyond the wilderness boundary. I then estimated the relative value of multiple landscape attributes that are associated with ewe habitat selection in the BRCWA, including PJ woodland canopy cover as the only biotic attribute. I found that utilized habitats can be described as locations which allow for increased visibility of surroundings. Visibility could be achieved by a combination of factors including proximity to steep slopes, rugged terrain, and low PJ woodland canopy cover. Predictions demonstrated that current habitat selection is highly related to canyon habitats or steep slopes outside of canyon habitats. Ewes have been observed to be utilizing some burned habitats but not others and predictions of current habitat use agreed with these field observations.

I then used the models to simulate a widespread high intensity wildfire by removing all PJ woodland canopy cover and then predicting post-fire ewe habitat selection. It is important to note that a widespread wildfire is only hypothetical but high intensity wildfires are expected as this is the dominant fire type for PJ woodlands in this area. Predictions of habitat use on the landscape indicated that wildfire would increase the relative probability of use in all areas and it is likely that habitat expansion would occur in areas near topographic escape terrain. Fire is also likely to greatly increase forage for desert bighorn sheep which are known to be adaptive feeders (Krausman et al. 1989, Miller and Gaud 1989). I focused on modeling but habitat selection following fire, but predictions can also be similarly interpreted in regards to mechanical removal of PJ woodlands.

VEGETATION COMPOSITION AND THE EFFICACY OF POST-FIRE REHABILITATION

I compared vegetation species composition between two burned and two unburned areas in the BRCWA. Sampling of vegetation cover and composition was conducted during June and July of 2012 and 2013 when peak herbaceous biomass was expected to occur. Sampling of the burned areas was conducted within two separate wildfires that were both ignited July 2nd, 1999 and both had a native seed mixture aerially applied the following the winter. Sampling of the two unburned areas was conducted across two soil types in which PJ woodlands are currently found. I compared species composition between burned and unburned areas in terms of species richness, foliar cover, and the development of biological soil crusts. Both burned areas occurred within one of two sampled soil types so it is important to note that statistical comparisons were only made between burned and unburned areas of the same soil type.

I found that foliar cover in burned habitats was on average 2 times greater than in unburned habitats and that post-fire seeding likely allowed for these differences to be proportionally similar between native and non-native grass species. *B. tectorum* was encountered throughout the study area (84% of all plots). Within the burned areas, *B. tectorum* was found to be negatively correlated with the cover of several native grass species that were included in the post-fire seed mixture, indicating that native grasses are mitigating the success of *B. tectorum*. Biological soil crusts have not become highly developed in burned but cover of less developed biological soil crusts similar to unburned areas.

DISCUSSION

Mutual competing values are common when prioritizing management strategies. I conducted an encompassing research project to analyze the potential effects of wildfire specifically in regards to two management priorities for BRCWA. I provided statistically rigorous predictive results that assist in understanding ecosystem processes for this unique and fragile landscape. I found that wildfire in PJ woodlands has resulted in dramatically altered vegetation composition and it is likely that fire will augment desert bighorn sheep habitat.

However, every statistical analysis is limited when interpreting the magnitude of results. In this section I summarize the entire project by offering guidance for interpretation of the results.

I found that ewes were avoiding habitats with high woodland canopy cover, the habitat most likely to be removed by fire. With the influence of other variables held constant, predicted habitat use decreased by a factor of +0.97 for each 1% increase in PJ woodland canopy cover. Given the removal of all woodlands, it is likely that habitat expansion would occur in areas near topographic escape terrain. Interpretation of these results must be strictly stated due to the nature of the habitat selection modeling process. Specifically, locations utilized by ewes can be known with certainty but it cannot be said with certainty that a particular location on the landscape in not used, making habitat selection predictions relative the composition of available habitats. Therefore, statistical limitations do not allow for habitat selection predictions on the landscape to be classified as either used or unused. Post-fire habitat selection predictions for both models are best interpreted as demonstrating the relative importance of PJ woodland canopy cover in regards to desert bighorn habitat use.

I described and compared the composition of vegetation in burned and unburned areas. Burned areas have had over 12 years to establish a more stable vegetative composition. It was found native grass species are likely to be mitigating the success of *B. tectorum* within the burned areas. Specifically, post-fire seeding efforts likely allowed for differences in herbaceous cover of burned and unburned areas to be proportionally similar between native and non-native grass species, indicating successful establishment of native grass species. However, *B. tectorum* is prevalent throughout the BRCWA in both burned and unburned habitats and its widespread presence will likely increase the risk of future wildfires. Although this project only sampled from two large burned areas, the landscape on which they occurred is similar and in close proximity to landscapes that currently pose the greatest risk of wildfire due to high PJ woodland canopy cover.

I focused on two management priorities in attempt to provide an encompassing but also detailed analysis. I acknowledge that other mutual priorities are rightfully valued by management. For example, fire would likely destroy archeological interests that are scattered throughout the area. Wildfires may also displace several wildlife species that consider PJ woodlands primary habitat. This research project strived to answer questions about the effects of wildfire on vegetation composition and desert bighorn sheep habitat selection. The results provide further ecological understanding and hold several implications for future conservation planning.

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APPENDICES

APPENDIX A: SEASONAL HABITAT SELECTION MODELS

SUMMARY

I developed resource selection models separately based on four biologically relevant seasons. The four seasons were defined in 3 month intervals with the Lambing season spanning from February 15th to May 15th, the Spring season spanning May 15th to August 15th, the Rut season spanning August 15th to November 15th, and the Winter season spanning from November 15th to February 15th. I used the same procedures described in chapter 1 to model and predict habitat use but separated data based on season that a location was recorded as used. Available locations were randomly sampled from within an individual's overall home range. Models were developed using the full data set with no effort to validate model accuracy. The goal of seasonal habitat selection was to further understand the habitat selection process but not to make ecological inference based on predictions. The following tables and figures describe the results of my seasonal modeling efforts. Please direct any questions to the author.

Table 9. Summary of seasonal resource selection models.

Summary Seasonal Mixed Models							
Odds Estimate Fixed Effects							
Models	Aspect	Esc Terrain	PJ Density	Ruggedness	Slope	TWI	n(y=1)
Lambing	–	0.994167	0.948731	331.9733	–	1.038142	302
Spring	–	0.9957038	0.961386	–	1.006842	–	248
Rut	1.001466	0.9981444	0.954447	–	1.005228	–	255
Winter	–	0.9945029	0.951137	–	–	1.035865	250

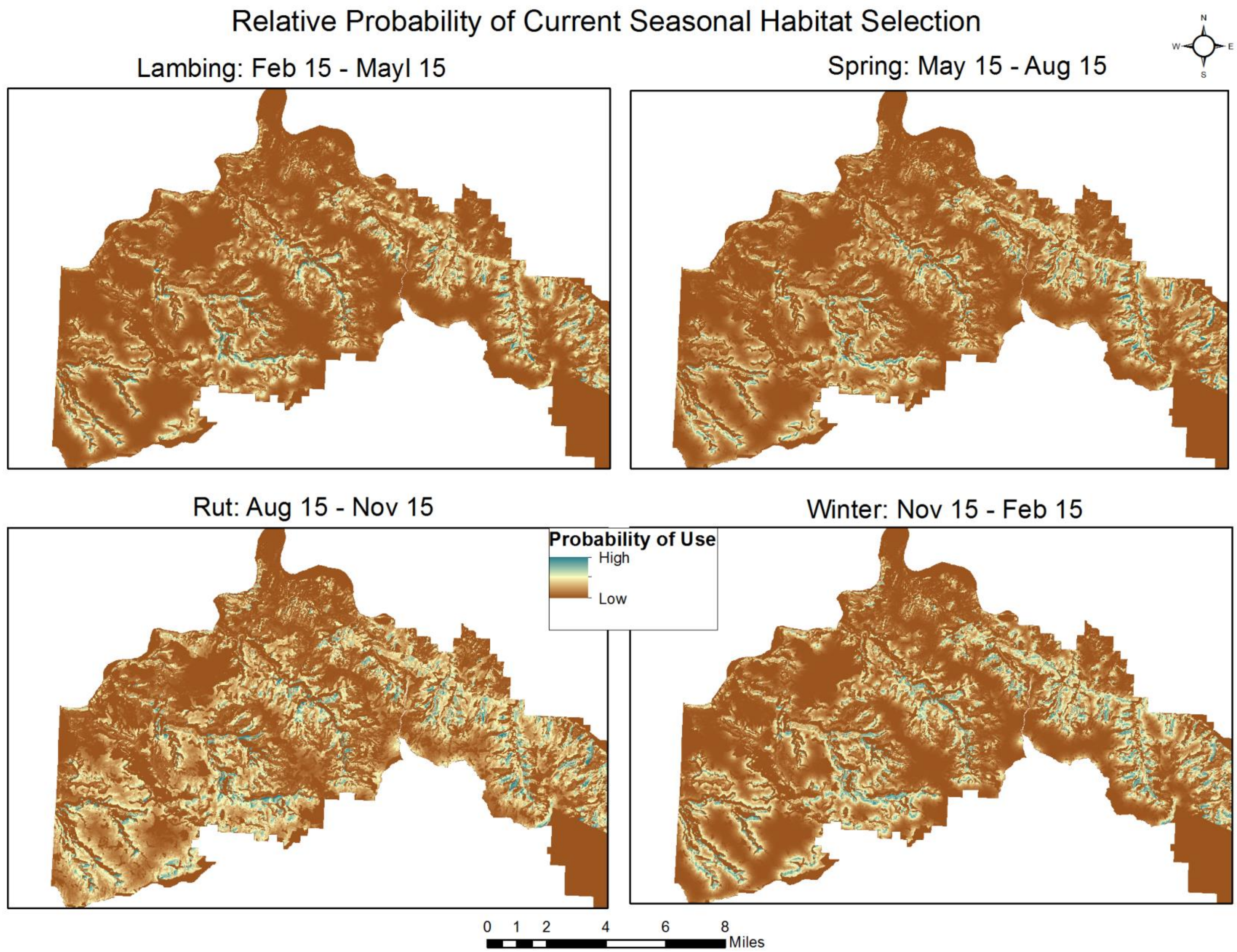

Figure 12. Seasonal relative probability of habitat use under current conditions.

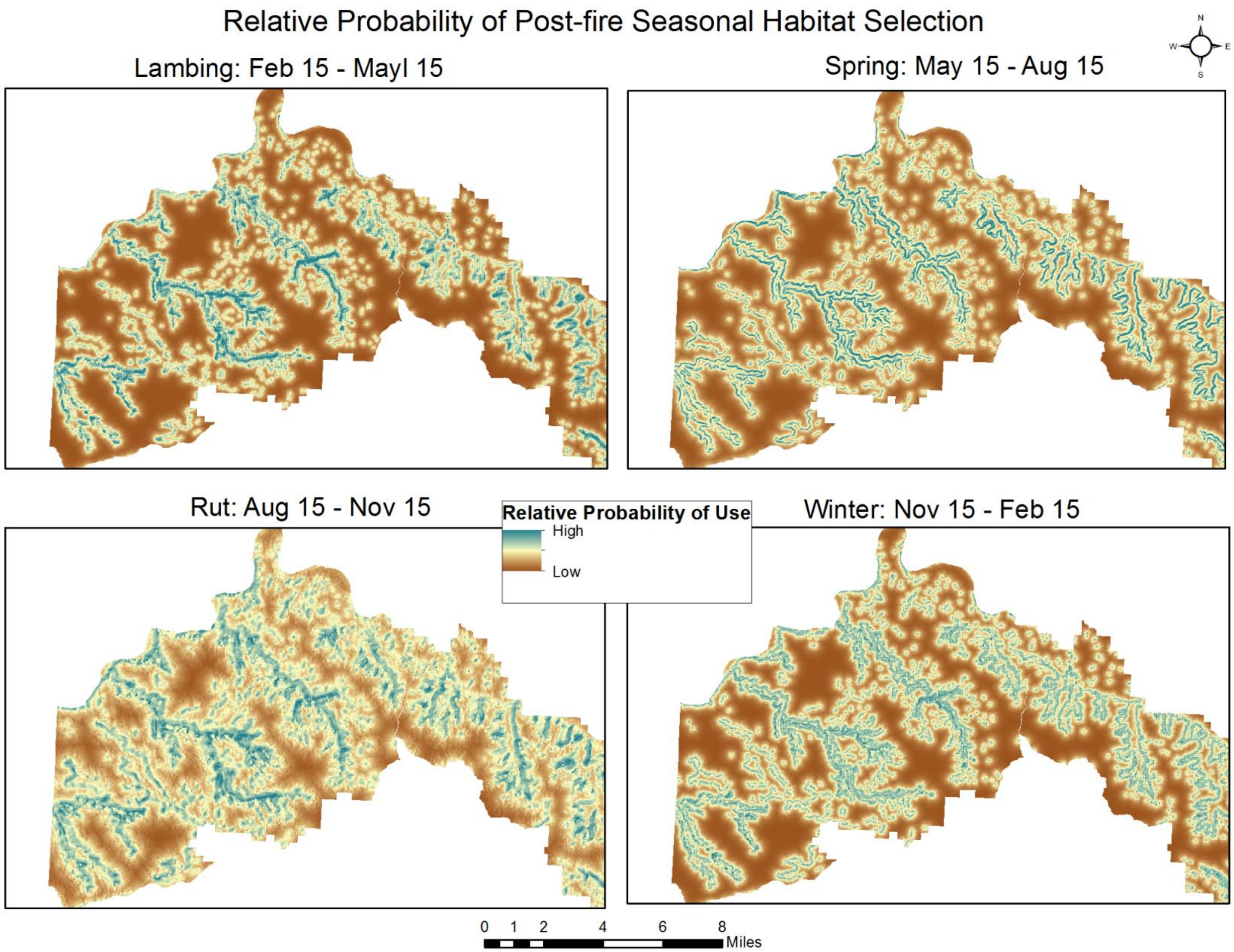

Figure 13. Seasonal relative probability of habitat use following simulated fire across the study area.

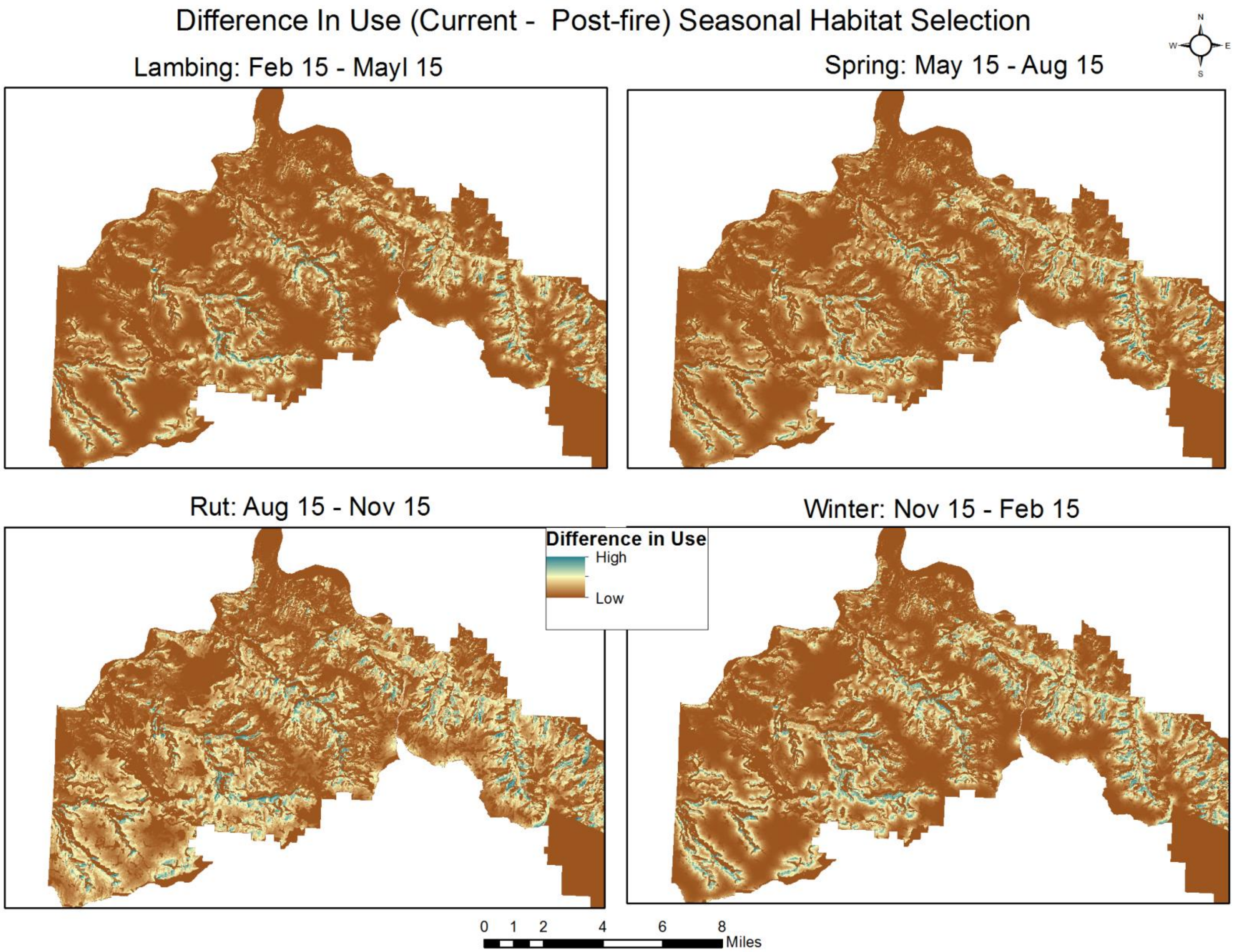

Figure 14. Difference in relative probability of habitat use calculated as the difference between current and post-fire predictions.

APPENDIX B. ESTIMATION OF PIÑON-JUNIPER WOODLAND CANOPY COVER

SUMMARY

I spatially estimated piñon-juniper woodland canopy coverage for the Black Ridge Canyons Wilderness Area in western Colorado to assist ongoing research in regards to the role of wildfire in this landscape. I used high resolution imagery to sample land cover class and piñon-juniper woodland canopy coverage for the entire study area. I then developed classification and regression tree models to perform a supervised land cover classification and estimate canopy cover. I estimated that 67% of the study area is comprised of piñon-juniper woodlands with canopy coverage ranging from 0%-62%.

INTRODUCTION

Piñon and juniper trees are the dominant tree species for over 21 million ha (50 million acres) of landscapes in the south western United States (USGS GAP Analysis). These landscapes are known to be variable and dynamic, thus requiring managers to consider site specific attributes when planning and conducting conservation efforts. Natural variability is controlled by the productivity of soils in which they occur, disturbance events (e.g., wildfire), and the availability and timing of precipitation. Based on these factors, Romme et al. (2009) outlined three fundamentally different types of PJ woodlands; persistent piñon-juniper woodlands, piñon-juniper savannas, and wooded shrublands. Accordingly, stands within the BRCWA are considered persistent PJ woodlands where precipitation is bimodal (i.e., small peaks in both the summer and the winter) and high intensity fire is the dominant fire type. Low severity wildfires are infrequent in persistent PJ woodlands due to a lack of fine understory fuels and relatively slow succession rates. When widespread wildfire does occur it is most often fueled by dense woodland structure resulting in a high intensity fire.

Wildfire is an annual concern for land and wildlife managers of the Black Ridge Canyons Wilderness Area. Historically, wildfires in this area are immediately suppressed and a native seed mixture is applied to burned landscapes in an effort to quickly establish native vegetation and limit the cover of non-native plant species. Managers are now contemplating the suppression of future wildfires and thus require an encompassing analysis on the potential effects of wildfire specific for this landscape. Basic land cover data is available for this area but no estimates of woodland canopy coverage exist. I used a supervised classification approach to map PJ woodland canopy coverage by applying classification and regression tree models.

STUDY AREA

The BRCWA is located in western Colorado on the south side of the Colorado River and the river carves its way into Utah. The landscape is comprised of over 30,000 ha (75,580 acres) of steep canyons and rugged mesas. The entire BRCWA can be characterized as a semi-arid desert ranging in elevation from 1300 meters to over 2100 meters. Within the BRCWA resides a non-migratory population of desert bighorn sheep (*Ovis canadensis nelsoni*) which is also of concern to managers. For this analysis the study area included the entire BRCWA in addition to known ewe bighorn sheep home ranges that extend beyond the boundaries of the BRCWA (Figure 15).

METHODS

Supervised land cover mapping is generally conducted by segmenting a landscape into relatively homogenous land cover classes based on digital imagery data. Classification and regression trees (CART) are a commonly used non-parametric method in ecological studies that are able to analyze both discrete and continuous variables to predict land cover classes across a landscape of interest (Parmenter and Hansen 2003, Sesnie et al. 2008). CART models recursively partition data into a branched tree structure where grouped patterns in the dataset are identified (Breiman et al. 1984). CART models work by first conducting an exhaustive search among potential splits in the dataset at each step in the partitioning process. Splits that generate the most homogenous subsamples with respect to the dependent variable are selected. Homogeneity is measured by deviance if the dependent variable is continuous resulting in a regression tree. If the dependent variable is discrete, homogeneity is most often measured by the Gini Index, resulting in a classification tree. I first estimated land cover classification for the entire study area and then estimated canopy coverage for all landscapes identified as PJ woodland.

Land cover class sampling was conducted using aerial imagery provided by Mesa County Department of Geographical Information Systems (Figure 16). Scenes were captured from fix winged aircraft during January of 2012 in the form of 3-bands (red, green, blue) and came in two different resolutions with greater resolution at lower elevations. Specifically, landscapes below ≈1800 meters in elevation were captured at a pixel height and width of 6.096 inches. But to reduce costs of collection, landscapes above ≈1800 meters in elevation were captured using a pixel height and width of 9.144 inches. A geodatabase of the aerial images was composed of several slightly overlapping tiles which I processed into a single mosaic to facilitate viewing.

Aerial imagery was used to sample habitat classes with ease but sampling woodland canopy cover may be more difficult due the irregular growth of piñon and juniper species. So I initially compared field sampling estimates to aerial sampling estimates to assess my ability to visually calculate percent canopy cover using the aerial imagery. The following procedures were conducted at five sites for each resolution level with attempts made to sample areas of low, medium, and high PJ woodland canopy cover. I first located and recorded the locations of identifiable landmarks from aerial imagery that were easily reachable from trails or roads such as boulders, unique tree assemblages, or fence posts. I then physically visited the exact landmark with the assistance of a field copy of the aerial imagery and location data. Upon arrival I used several measuring tapes to designate a 30 meter x30 meter grid centered on the landmark with the 2 sides of the grid running north and south and the remaining 2 sides running east and west. Canopy cover of each tree within the grid was then estimated by measuring each tree at its longest and shortest horizontal width (meters). Percent canopy cover for each tree was estimated by calculating the area of a circle with diameter equal to the average of the longest and shortest widths. Multiple trees were measured as one if the trees formed a continuous canopy. Flagging was then used to mark each tree to prevent resampling. Percent canopy cover was then summed within the entire grid and divided by $900m^2$ to achieve the total canopy coverage of the 30 meter x30 meter grid.

After completing the field sampling, I conducted very similar estimation of stand canopy cover using aerial imagery. Likewise, I first formed a 30 meter x 30 meter grid around the identifiable landmark. Additionally I included 5x5m sub-grids to assist in ocular estimation of tree canopy coverage. I then estimated and summed all tree canopy cover in 5% increments for entire 30 meter x 30 meter grid and divided by $900m^2$ to produce percent canopy cover for that grid. Aerial imagery estimates were then compared to field data estimates. Field estimates were found to differ from aerial estimates by an average of ±2% (standard deviation= .024, Table 10). I concluded that my ability to measure to canopy cover from aerial imagery was sufficiently accurate. To begin sampling PJ canopy cover and habitat classes I created a 30 meter x 30 meter grid for the entire study area and adjusted its position to align with predictor raster layers of the same resolution that will be used in the classification process. Again, I created 5 meter x 5 meter sub-grids within the 30 meter x 30 meter grids to assist in ocular estimation. Finally, I randomly placed 750 points and sampled from within the 30 meter x30 meter grids in which a randomly placed point lied within. When sampling I classified the habitat type in stages. The dominant and then sub-dominant habitat type was identified as rock, talus slopes, bare ground, riparian, grass, shrub, or coniferous. I then estimated the coniferous canopy cover regardless of habitat type. Upon completion it was found that my sampling did not adequately cover habitats dominated by bare rock, talus slopes, bare ground, or riparian vegetation due to their scattered and sparse presence. Therefore, I manually searched for and sampled from within these habitat classes making for a total of 963 sampled grids.

MODELLING

I used classification and regression tree (CART) analysis for mapping land cover in the Black Ridge Canyons Wilderness Area. I used 37 remotely sensed variables representing both biotic and topographic attributes of habitats as predictors in classification and regression tree (CART) models. Twenty biotic variables were derived from 30 meter x 30 meter Landsat 7 TM satellite cloud free scenes taken on two dates. First, a scene from June 19th, 2010 was chosen to represent habitats when primary production is at or near its greatest for most plant species. A second scene from October 28th, 2011 was chosen to represent habitats when primary production is lowest but before snowfall occurs. I extracted raw values from bands 1,2,3,4,5, and 7 to be included in the modeling process. For each date I also calculated the soil adjusted vegetation index (SAVI; Huete, 1988). SAVI is similar to the better known Normalized Difference Vegetation Index (NDVI) but is more appropriate in areas of low vegetation density that can exhibit high soil reflectance. I also generated three tassel cap transformations for each date known as wetness, greenness, and brightness (Kauth and Thomas 1976) which have been shown to have potential applications for identifying forest stand structure (Cohen et al. 1995).

Five geophysical variables were derived using 30 meter x 30 meter resolution United States Geological Survey digital elevation models. These included elevation, slope, aspect, and indices for landscape ruggedness and topographic wetness index. The ruggedness index was calculated using a 3x3 moving window as according Sappington et al. (2007). This index ranges from 0 to 1 with greater values representing areas with the greatest landscape roughness and was created specifically to describe terrain dynamics in a way that minimizes correlation with slope. Topographic wetness index (TWI) was calculated for each location similar to Bevin and Kirby (1979) where TWI is equal to the natural log of the upslope area divided by the slope for each location. Thus, greater TWI values represent basins and lower values represent peaks in the topography.

Next, I digitized the entire study area using the aerial imagery by manually drawing polygons (*n*= 5639) around land cover which were visually similar. I then converted the polygons to a single topology to relate all attributes and enforce shared geometry between adjacent polygons. Some additional data cleaning and manual editing was conducted to ensure a minimum mapping unit of 30 meter x 30 meter. I then calculated zonal mean values within each digitized polygons for all variables except elevation and the raw satellite band values. Final predictor variables consisted of 9 variables derived from digital elevation models (5 pixel-based, 4 polygon means) and 28 vegetation reflectance variables (20 pixel-based, 8 polygon means) across two dates.

All analysis was conducted in the R statistical software version 3.02 (R Core Team 2013) using the tree package (version 1.0, http://cran.r-project.org/web/packages/tree/tree.pdf, accessed September 3, 2013). I iteratively used classification and regression tree analysis to define each 30 meter x 30 meter pixel within the study area (Figure 17). First, I categorized and modelled each pixel as either having vegetation present or as bare ground. Next, pixels identified as vegetative were further modelled and classified as either dominated by grass, shrub, or coniferous forests. Pixels classified as non-vegetative were further modelled and classified as either dominated by bare soil/rock or talus broken rock. Pixels classified as coniferous forests were further modelled based on their dominant understory described as bare soil, grass, or shrub. Finally, percent canopy coverage of all coniferous forest pixels were estimated on a continuous scale regardless of understory classification. At each step, CART models were pruned by limiting the number of nodes within each developed tree as according to Breiman et al. (1984). Pruning assists in reducing complexity of the final classification rules to increase predictive accuracy and decrease over fitting.

MODEL VALIDATION

A field technician visually classified habitats at 97 locations in person from throughout the study area to evaluate the final classification models. The dominant and sub-dominant habitat classes at each location that was recorded. Additionally, photographs were taken facing north, east, south and west in case further classification was needed. I then calculated the classification error rate by comparing model results to evaluation data. I found that model correctly predicted the dominant habitat class 81% of the time (79/97). Similarly, I assessed the model's predictive ability in classifying forest understory. Coniferous forests were present in 26 of 97 validation locations that were physically visited. Understory type within coniferous forests were correctly classified 77% of the time (20/26). Five of the misclassified locations were due to discrepancies between grass and bare ground understories, and one was due to incorrectly predicting a grass understory as a shrub understory. Coniferous canopy cover was estimated for all pixels classified as coniferous. I first developed the regression tree model with the entire dataset. After which, 200 additional validation points were randomly sampled from aerial imagery to form an independent dataset. Model predictions at validation points had a root mean squared error of 6.39 (standard deviation=5.58).

DISCUSSION

I used statistical modeling methods to map land cover (Figure 18) and PJ woodland canopy cover (Figure 19). It was estimated that 67% of the study area is comprised of PJ woodlands (over 21,000 hectares). Models predicted canopy coverage ranging from 0%-61% although within sample canopy cover estimates were as high as 85%. Zonal means calculated from within digitized polygons representing distinct land cover types were found to be influential in the decision tree process. Therefore, final mapping products are considered accurate but are at a resolution coarser than the 30 meter x 30 meter independent data. The final map products produced from this analysis are the best currently available but should be used carefully as it is a generalized estimate of current land cover and PJ woodland canopy cover in the BRCWA.

Table 10. Estimate of PJ woodland canopy cover (%) from both field and aerial imagery sampling. Field estimates were found to differ from aerial estimate by an average of ±2% (standard deviation= 2.4%).

Sampling Technique Verification			
		Physical	Aerial Imagery
Resolution	≈6"	0.1333	0.15
		0.16322	0.15
		0.248977	0.25
		0.34567	0.30
		0.42587	0.40
	≈9"	0.9875	0.10
		0.18998	0.15
		0.31548	0.30
		0.34596	0.35
		0.57642	0.50

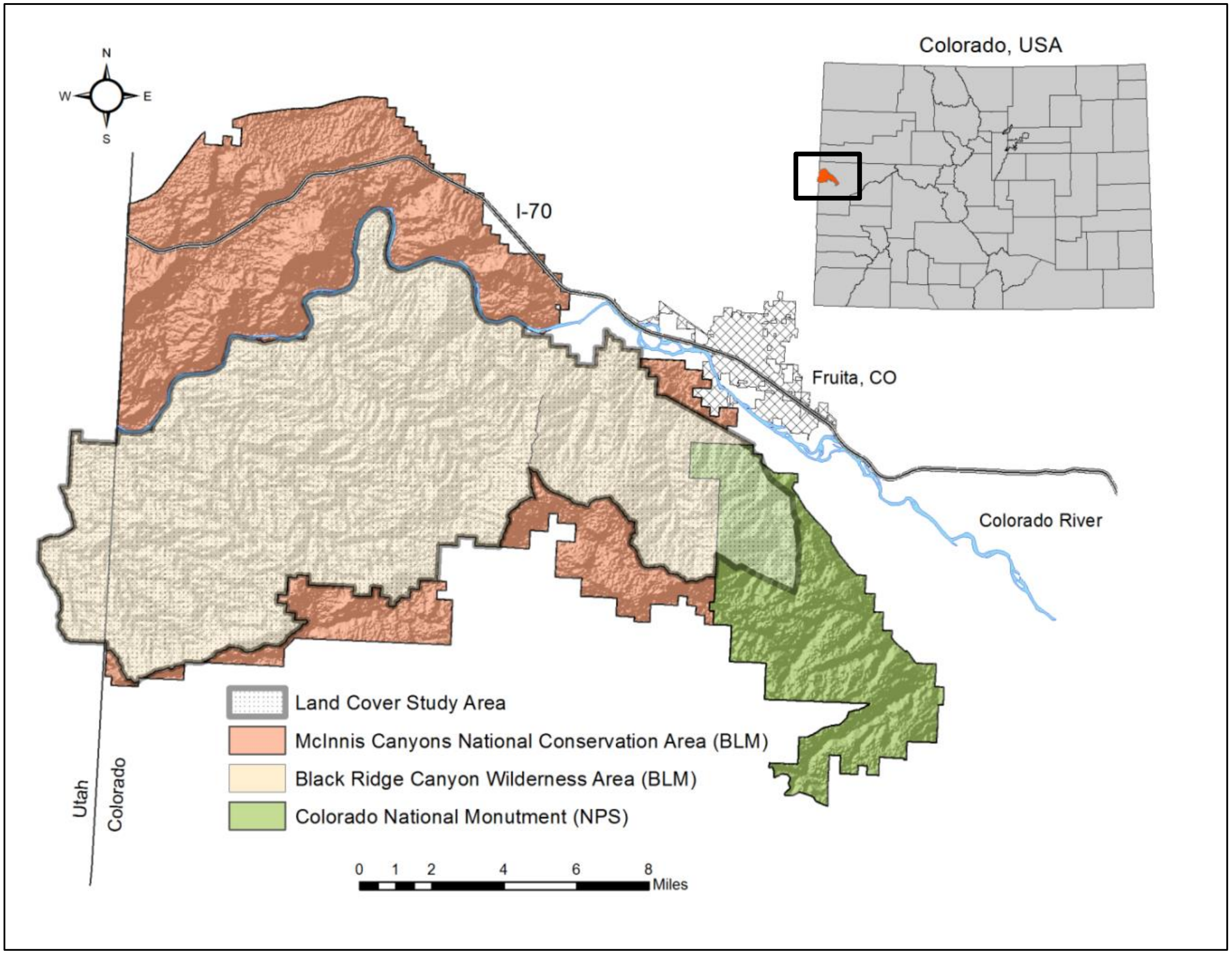

Figure 15. Study area for conducting supervised land cover classification and estimation of PJ woodland canopy cover. The study area extends eastward into the Colorado National Monument to include all female bighorn sheep home ranges beyond the Black Ridge Canyons Wilderness Area boundary.

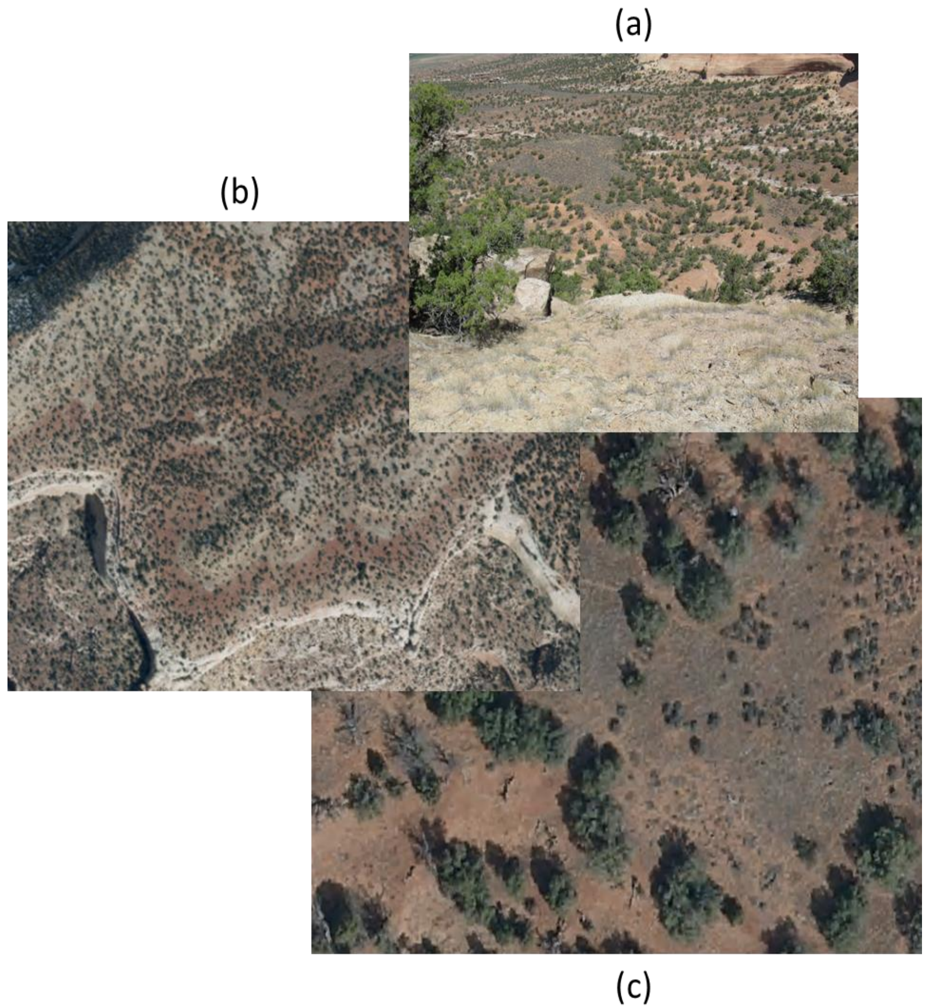

Figure 16. Examples of PJ woodlands in the Black Ridge Canyons Wilderness Area. (a) Photograph taken in person from a top a high point. (b) Aerial imagery snapshot viewed at a 1:4,000 scale. (c) Aerial imagery snapshot viewed at a 1:400 scale.

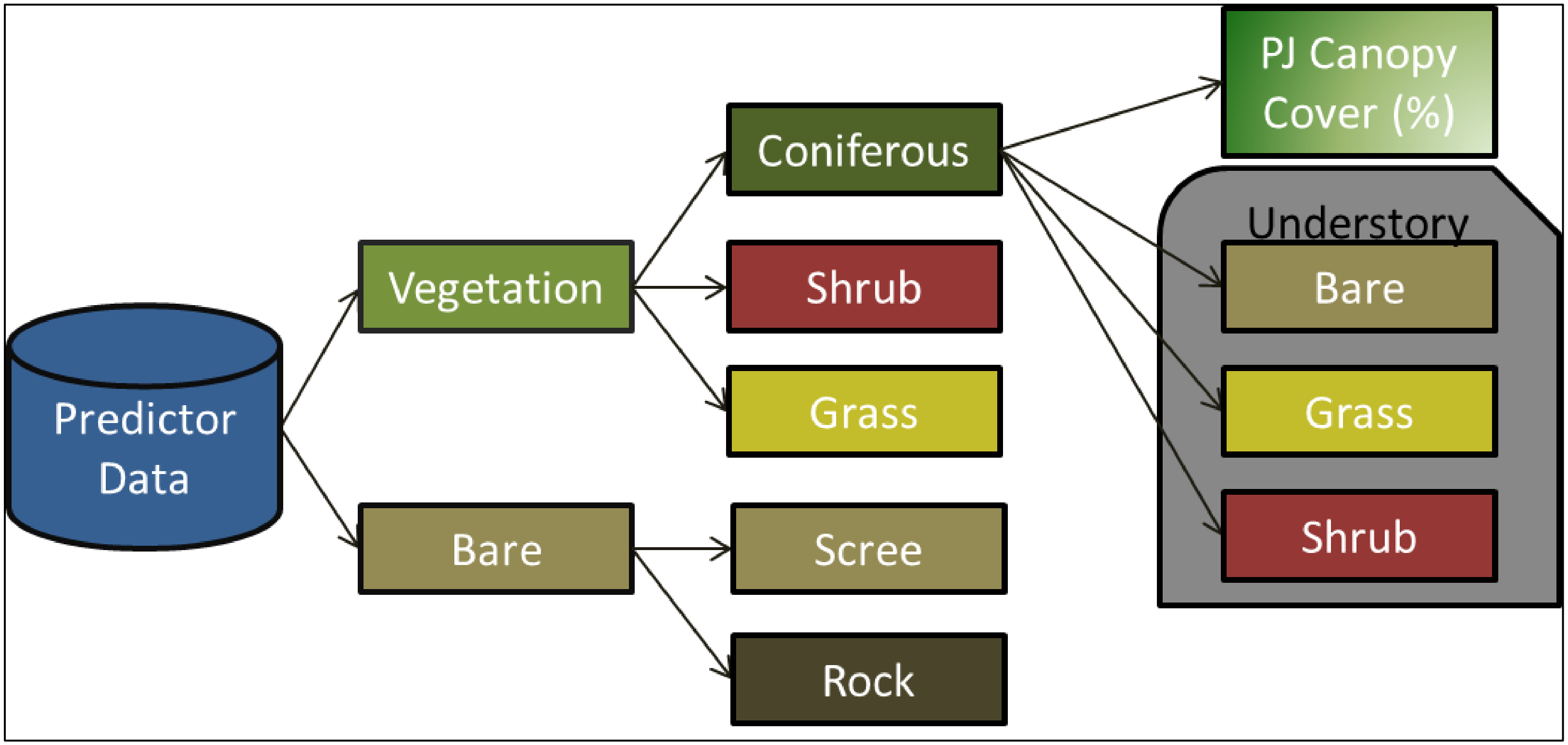

Figure 17. Iterative land cover classification and piñon-juniper canopy coverage estimation process using classification and regression tree analysis.

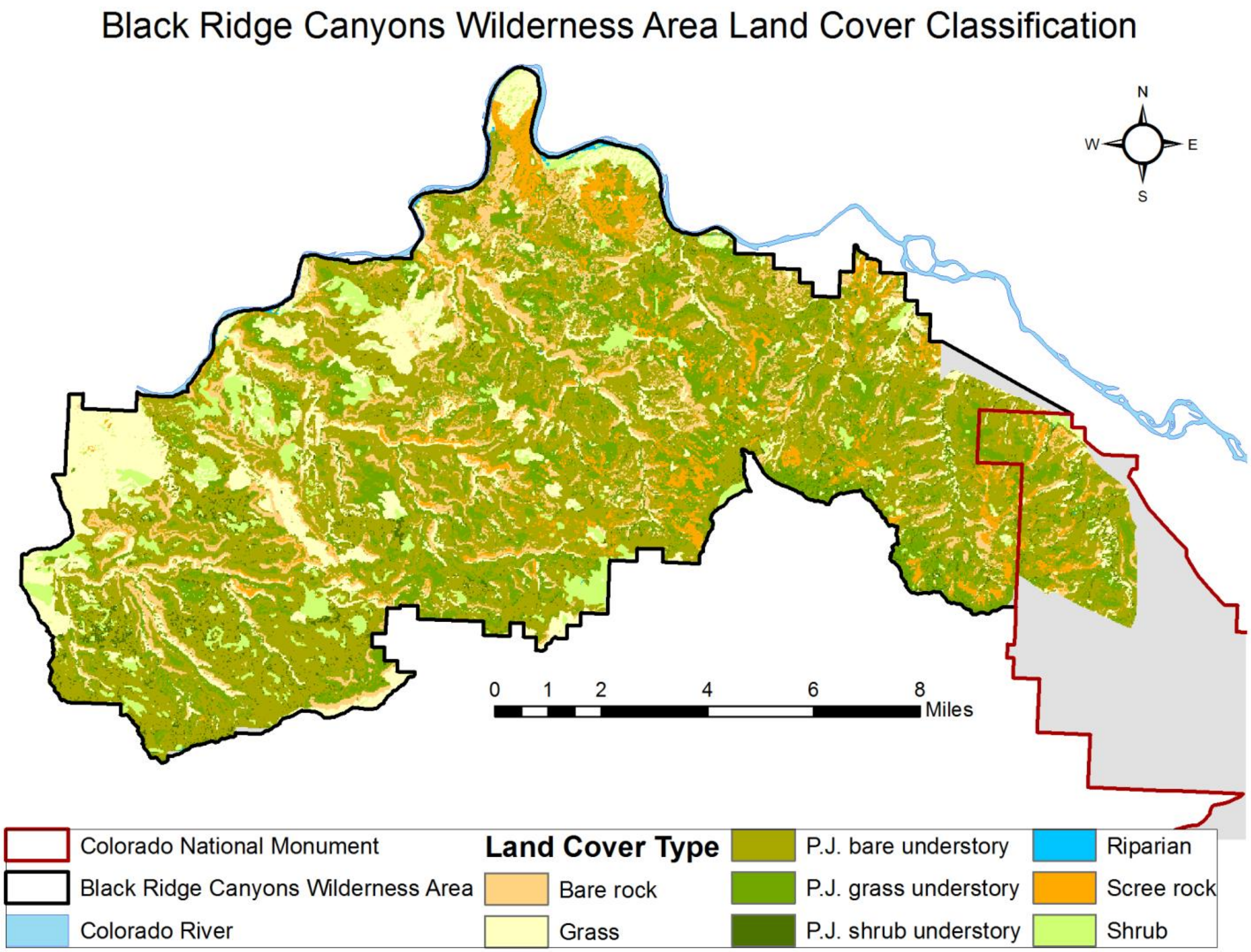

Figure 18. Land cover classification for the Black Ridge Canyons Wilderness and western portions of the Colorado National Monument.

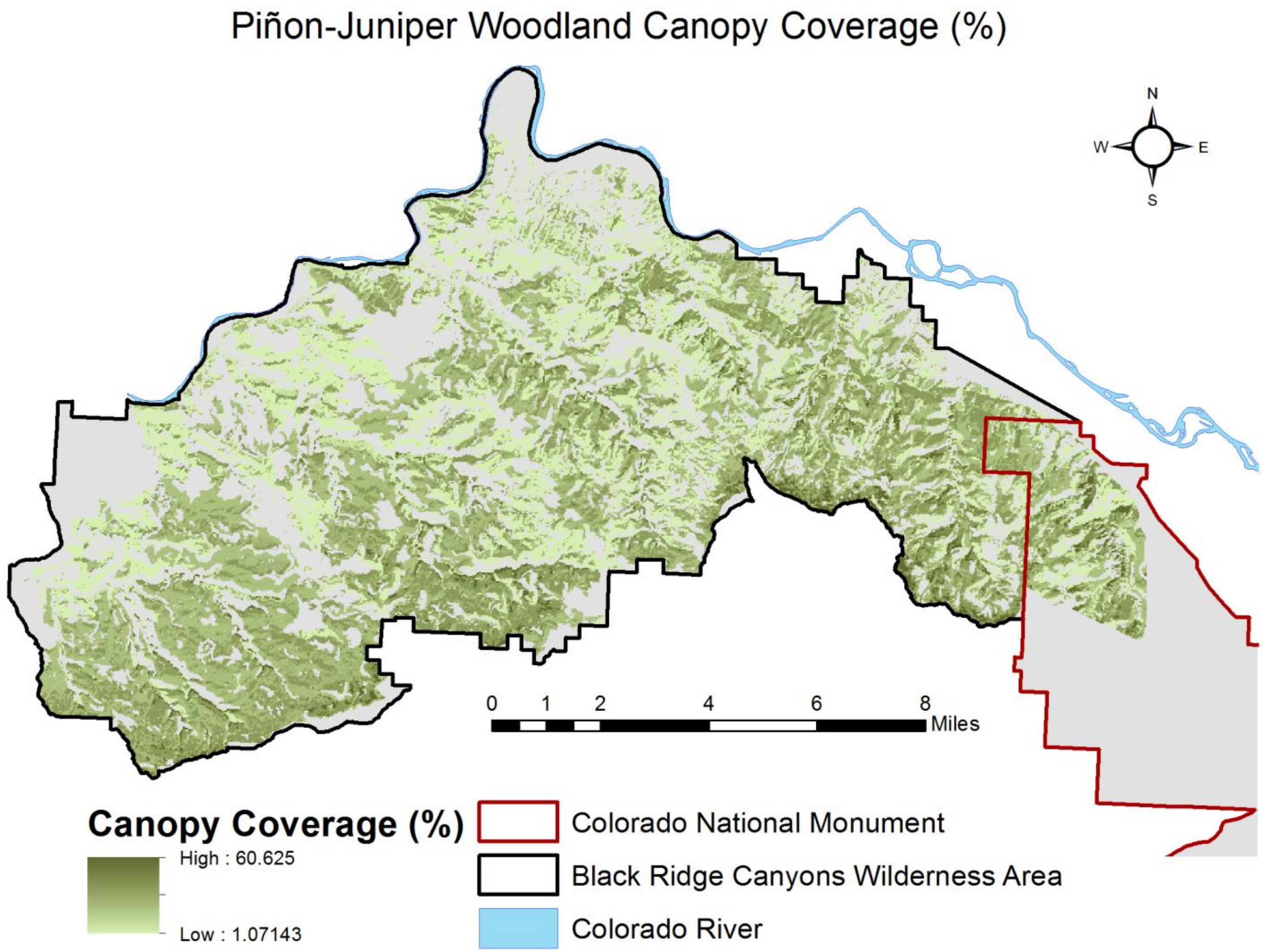

Figure 19. Estimated piñon-juniper woodland canopy cover for the Black Ridge Canyons Wilderness and western portions of the Colorado National Monument.

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APPENDIX C. HIERARCHICAL CLASSIFICATION OF PREDICTED HABITAT USE

SUMMARY

I classified relative pre- and post-fire habitat selection model predictions (Population and Landscape Models describe in Chapter 1) into levels of use to further understand what habitats wildfire would most augment and to what degree. I defined 4 levels using standard deviations (Slocum 1999), such that level 1 represented prediction scores from -2 to -1 standard deviations away from the mean predicted score. Additional levels followed sequentially with predicted scores greater than four standard deviations away from the mean being binned with level 4. I labeled levels 1,2,3,4 as Lowest Use, Low Use, Medium Use, and High Use. The following tables and charts describe the area (acres) each level of use accounted for in prediction for each habitat selection model for current conditions and following a hypothetical fire.

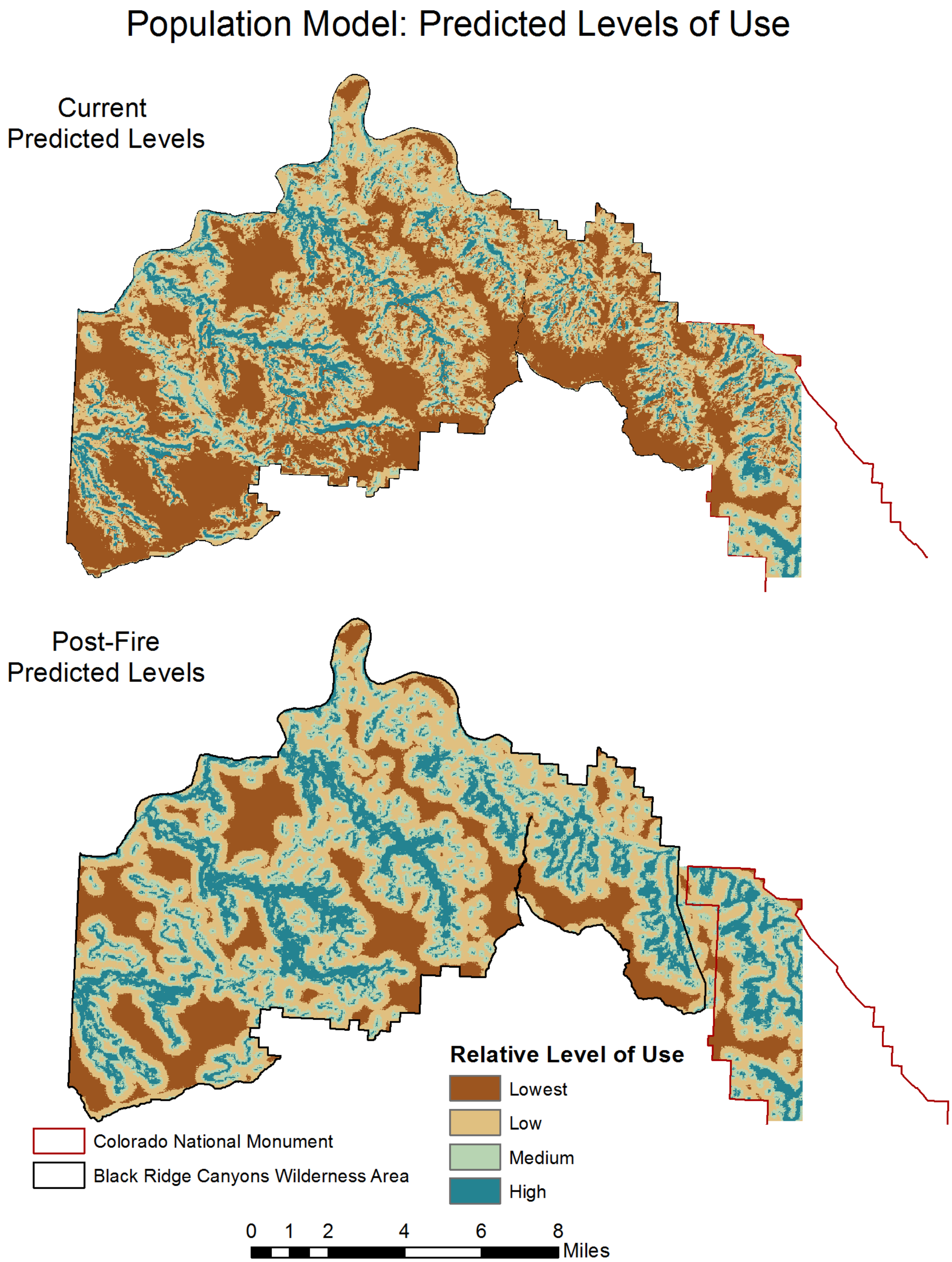

Figure 20. Current and post-fire relative probability of use predictions of the Population Model classified into four hierarchical levels.

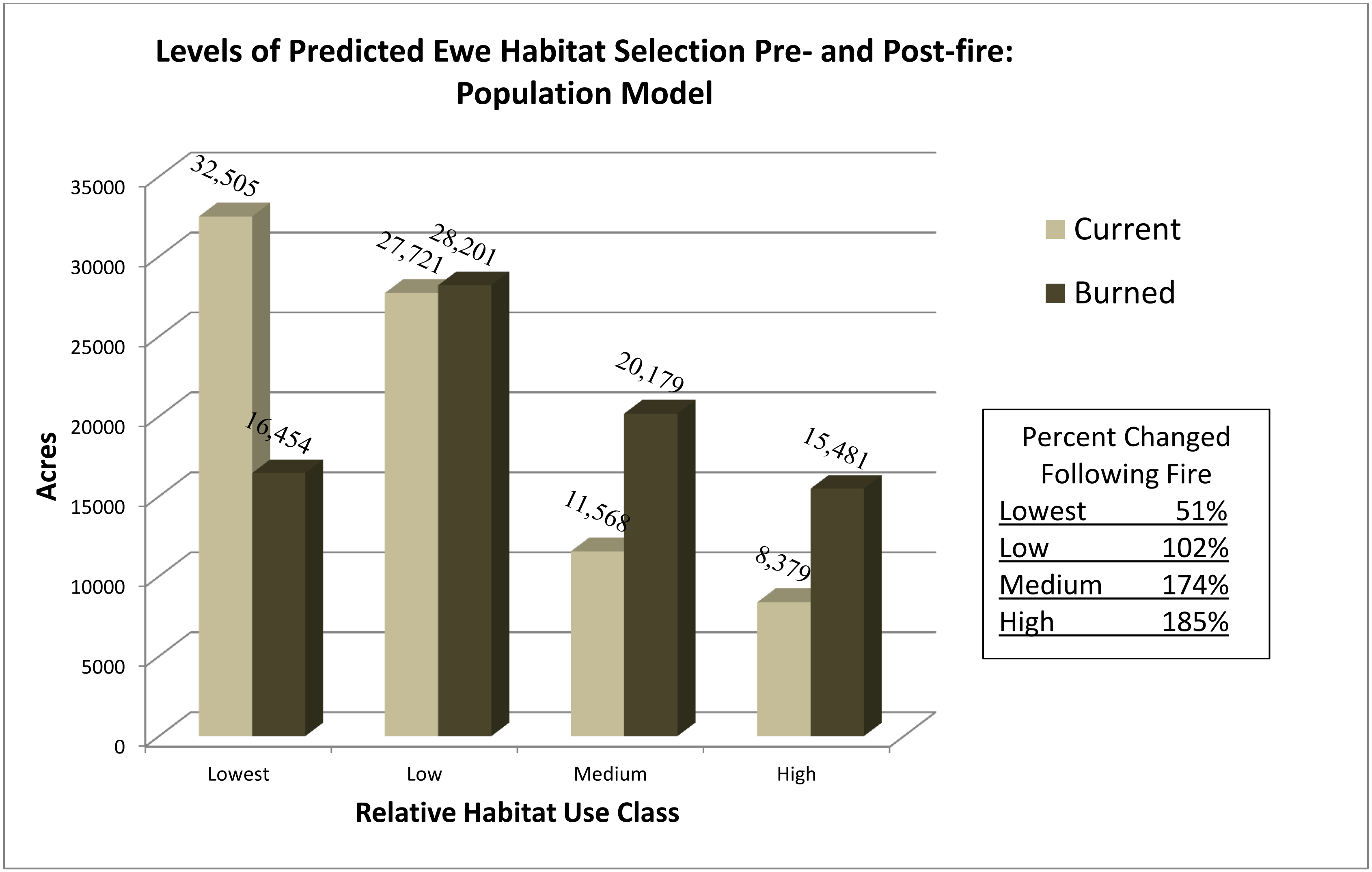

Figure 21. Population Model predicted scores pre-and post-fire. Levels were hierarchically binned based on distance from the mean predicted score in terms of standard deviation. Percent changed following fire is simply the proportional change of each level if under a hypothetical wildfire.

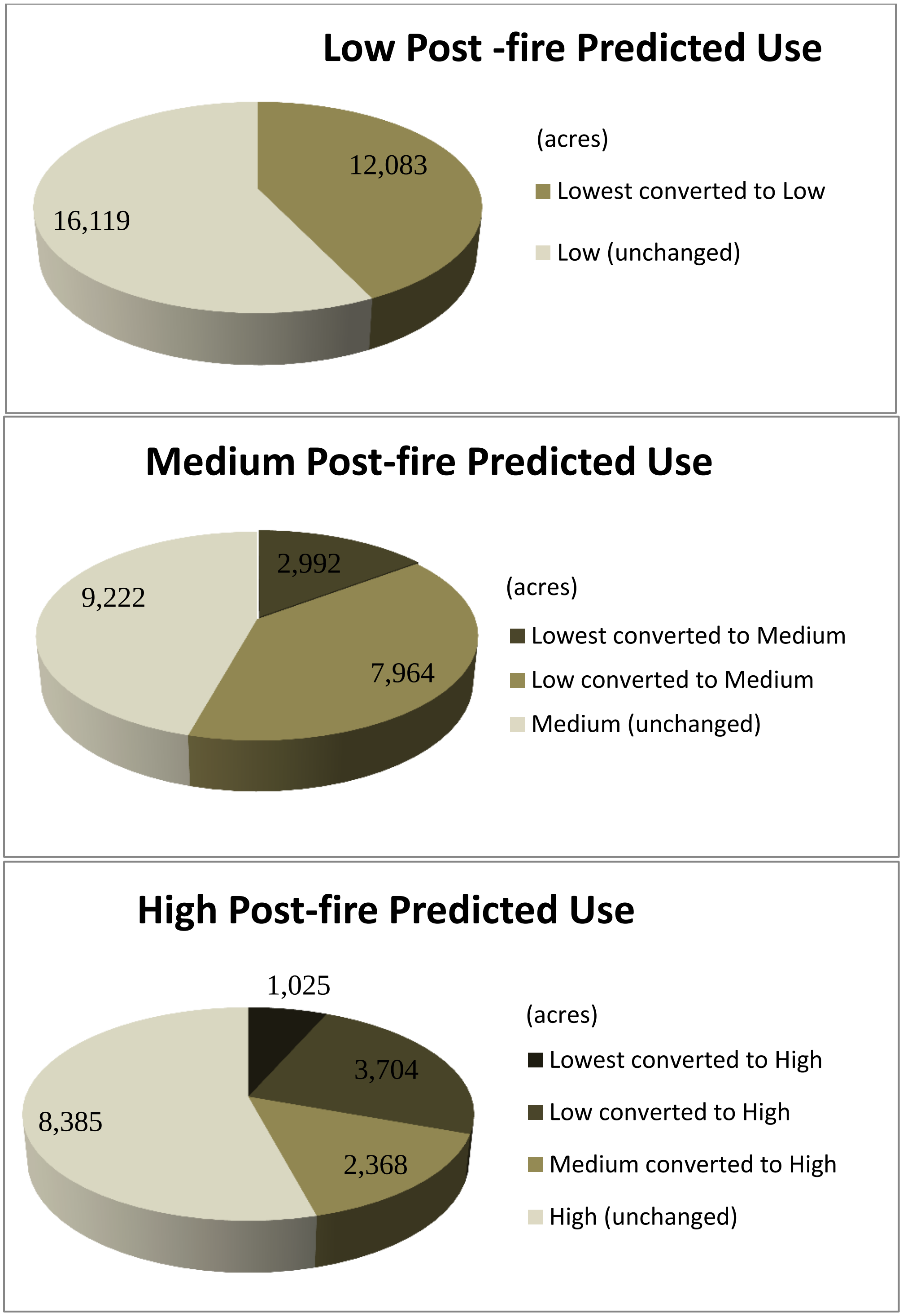

Figure 22. Population Model predicted habitat use level following simulated fire distinguished by current predicted level.

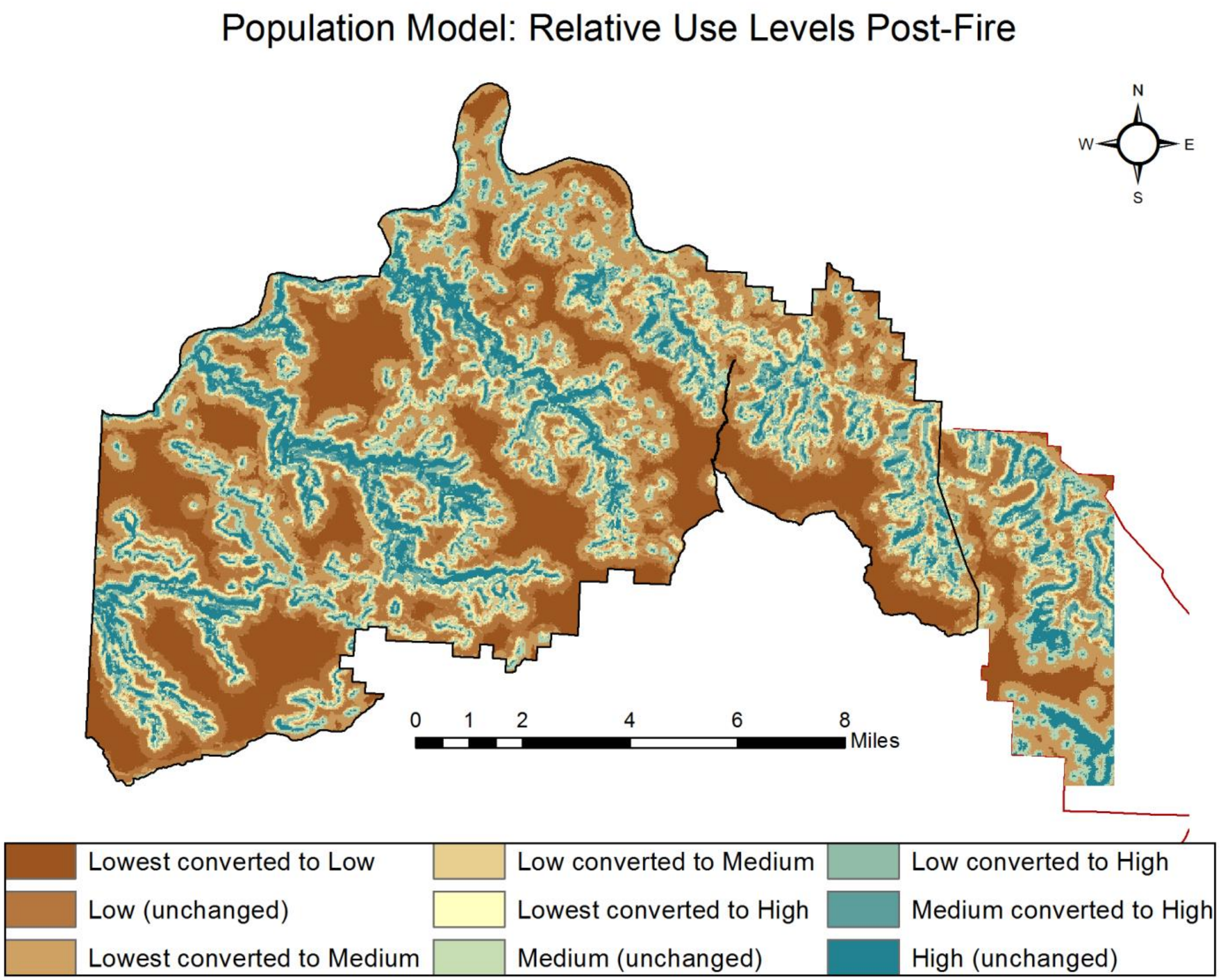

Figure 23. Population Model hierarchically classified post-fire predictions, distinguished by current predicted classification.

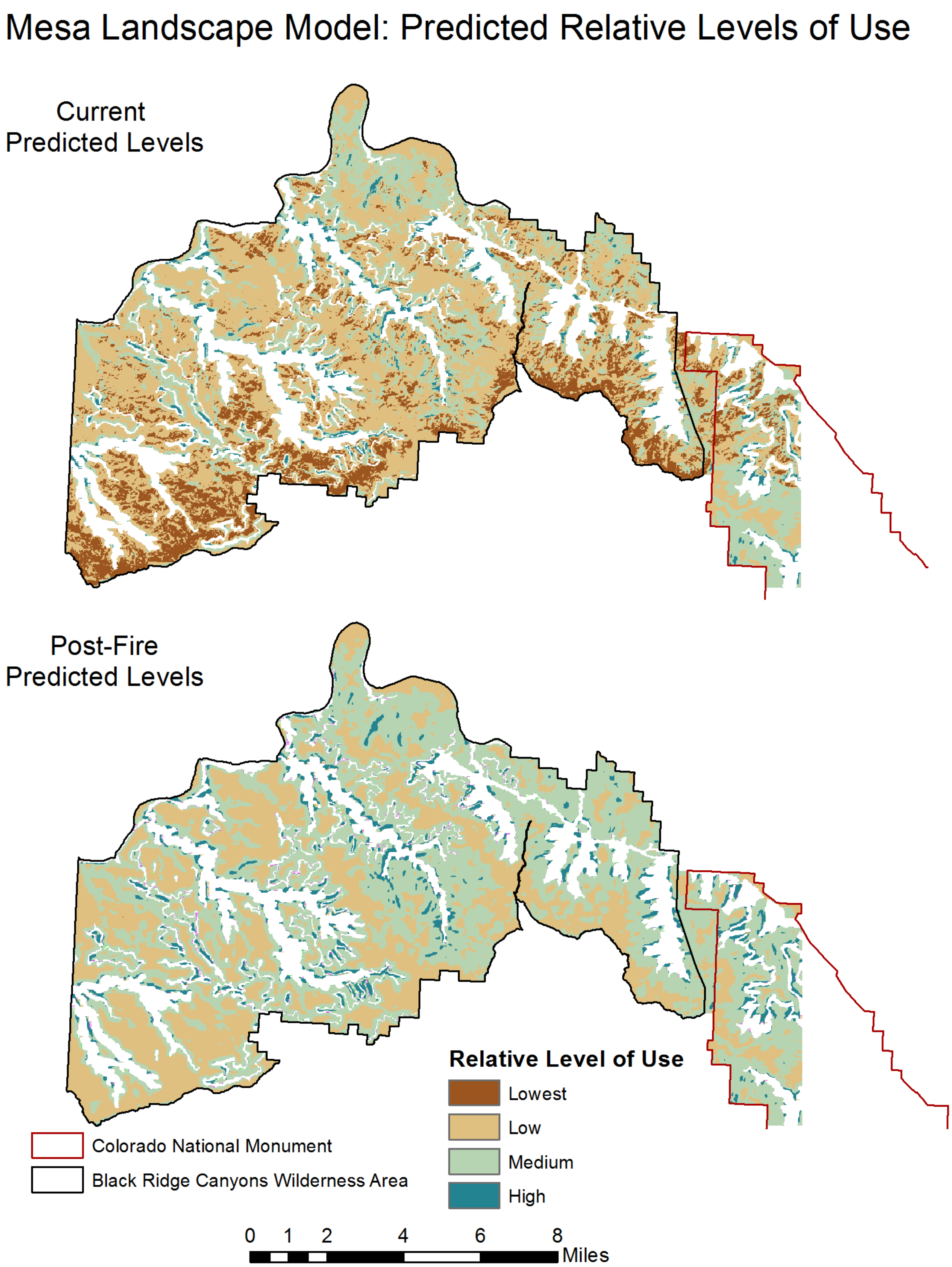

Figure 24. Current and post-fire relative probability of use predictions of the Landscape Model classified into four hierarchical levels.

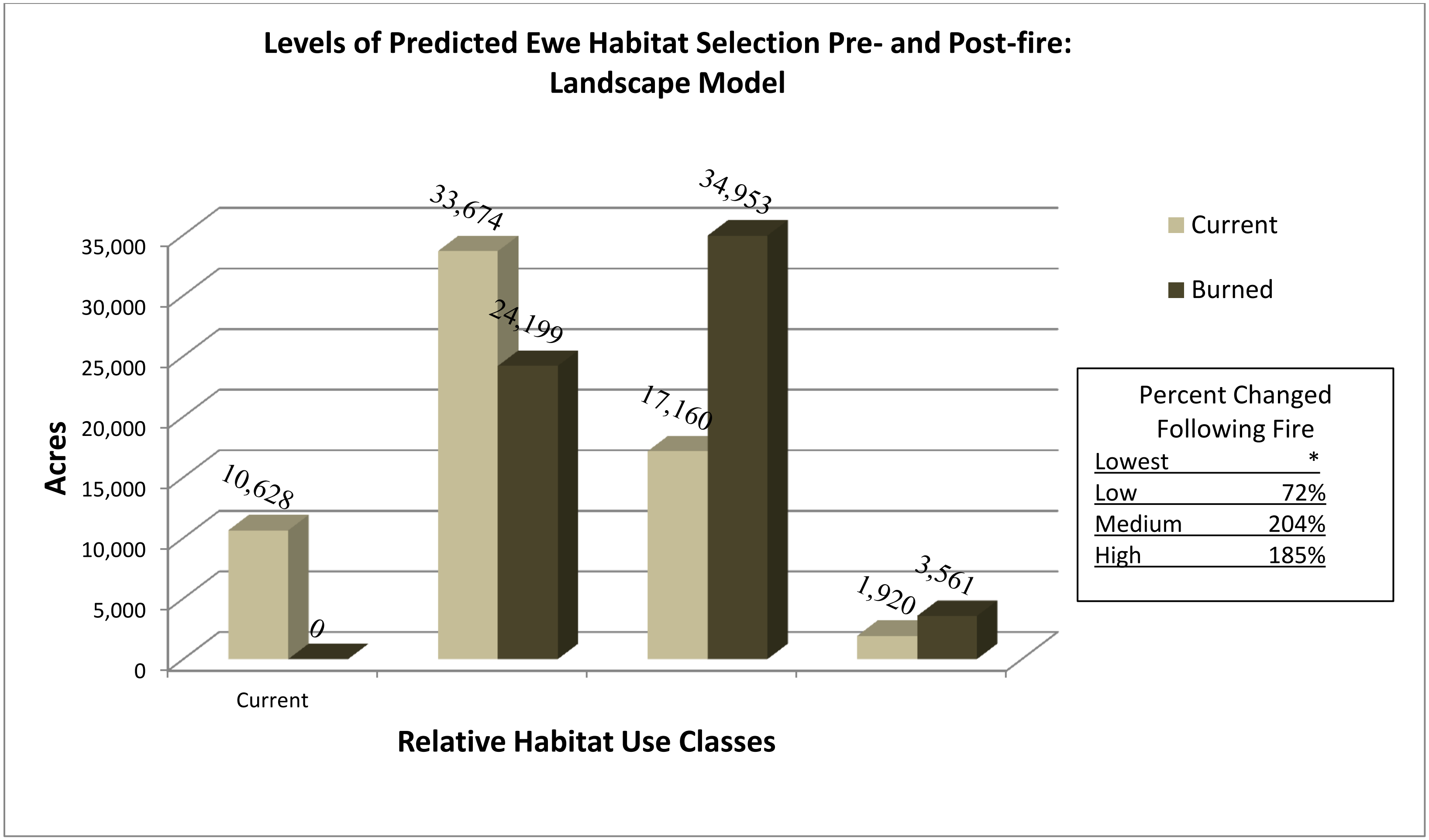

Figure 25. Landscape Model prediction scores pre-and post-fire. Levels were hierarchically binned based on distance from the mean predicted score in terms of standard deviation. Percent changed following fire is simply to the proportional change of each level if under a hypothetical wildfire.

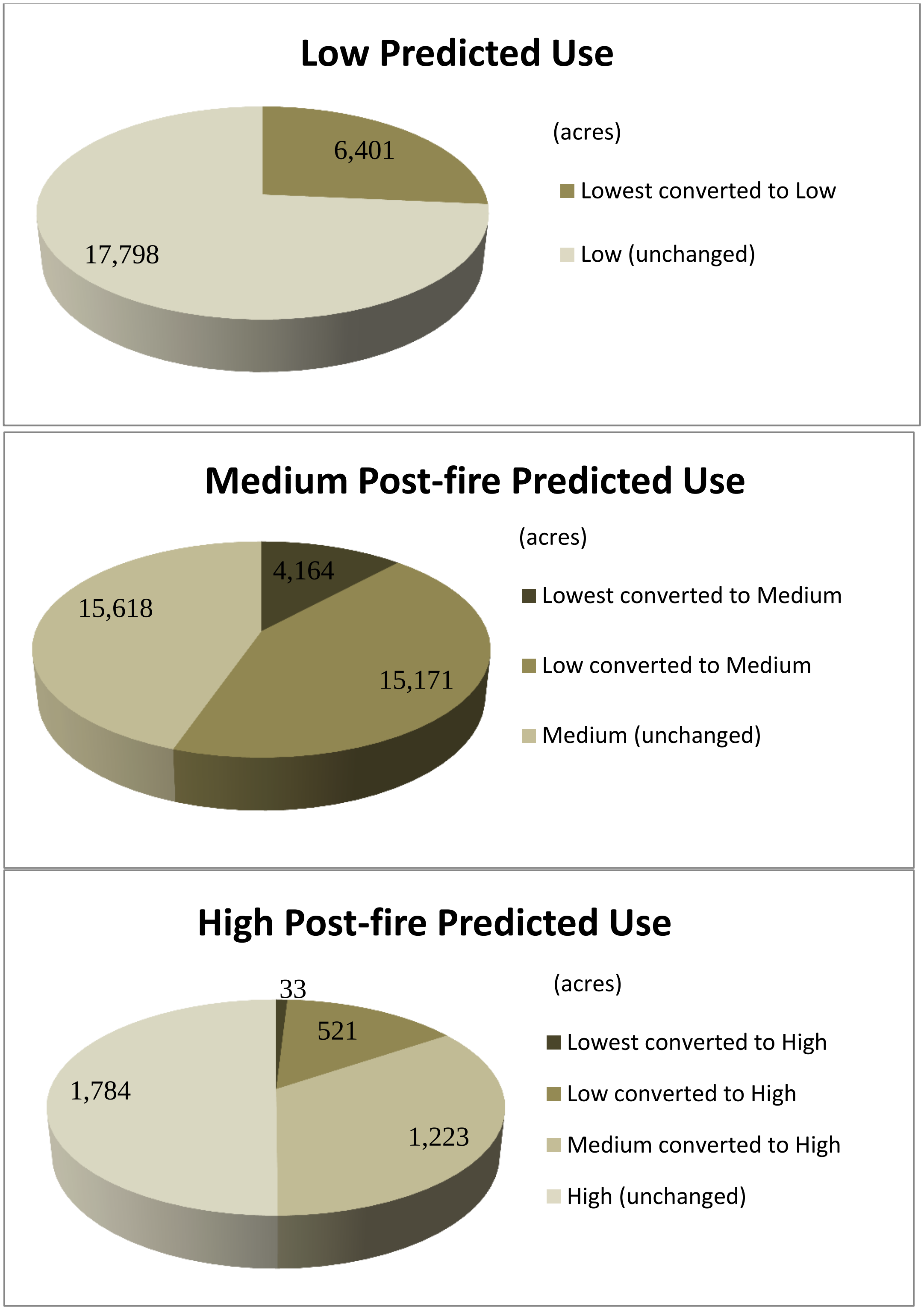

Figure 26. Lanscape Model predicted habitat use level following simulated fire distinguished by current predicted level.

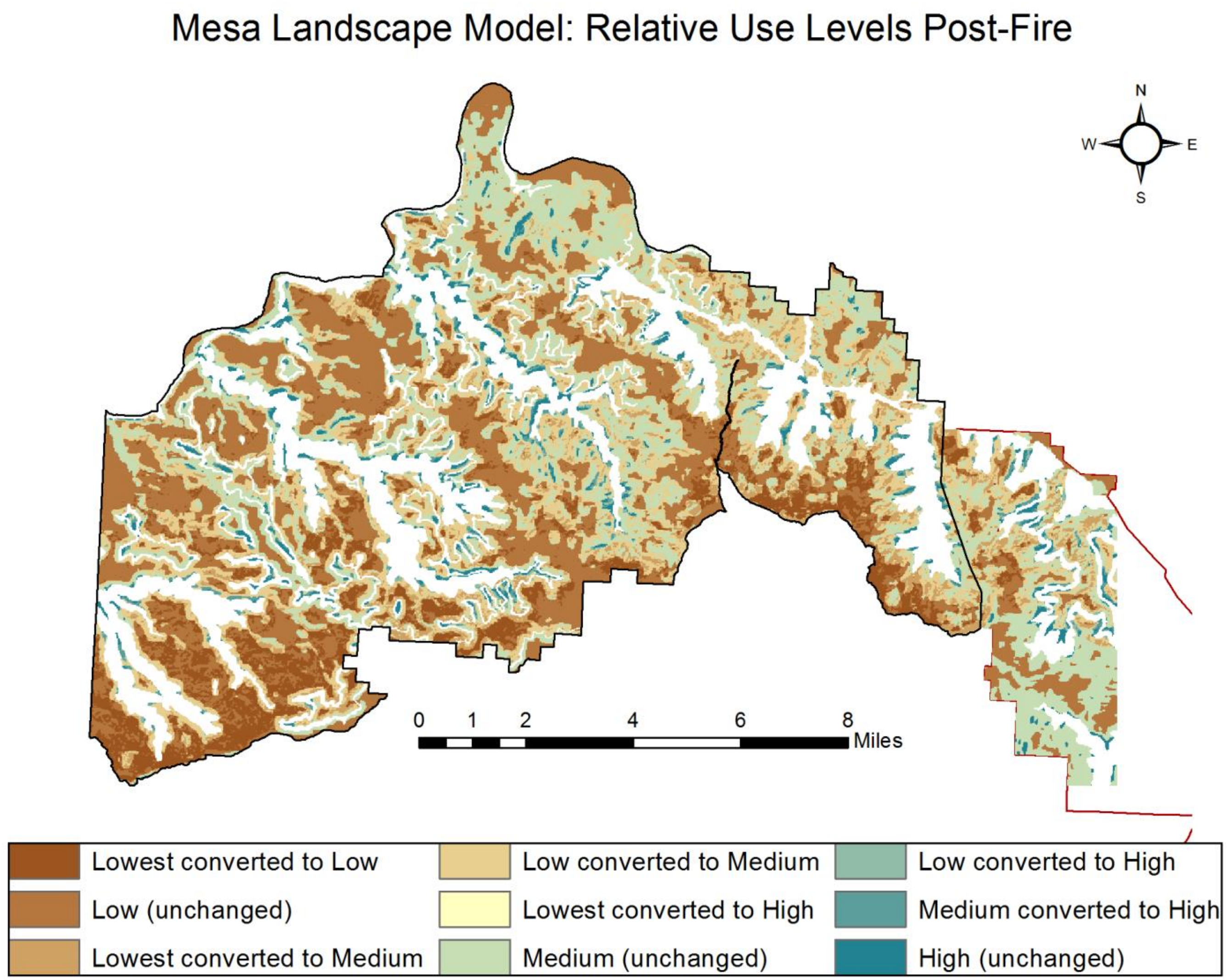

Figure 27. Landscape Model hierarchically classified post-fire predictions, distinguished by current predicted classification.